DEVELOPING A COST-EFFECTIVE FRAMEWORK FOR MONITORING SOIL EROSION IN ENGLAND AND WALES

Final report to Defra for project SP1303

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2. Introduction and project outline

There are two main objectives to the project:

1. To review all available literature on soil erosion monitoring and modelling in England and Wales, incorporating a summary of key findings from international soil erosion research: and

2. To design a scientifically robust yet cost-effective monitoring framework for soil erosion in England and Wales.

Thus, this report proposes a methodology to provide both accurate and reliable data which are fit for purpose at the correct spatial support and extent, whilst assessing the costs of delivering such information.

Objective 1. Review all available literature on soil erosion monitoring and modelling in England and Wales incorporating a summary of key findings from international soil erosion research

The review focusses on specific areas of scientific research in seven chapters (Appendix 1):

Chapter 1 - Uplands
Chapter 2 - Arable lands
Chapter 3 - Lowland grasslands
Chapter 4 - Mechanical erosion
Chapter 5 - Channel-bank and floodplain erosion
Chapter 6 - National erosion assessments
Chapter 7 - Soil erosion modelling

Within each of these areas a detailed literature review has been completed which synthesises:

(a) Monitoring data, providing a quantitative assessment of erosion rates by land use, soil type, pathway and spatial extent of erosion,
(b) Evidence (where applicable) for effects of erosion on soil properties (especially soil carbon storage),
(c) Modelling work, describing erosion predictions (with uncertainty where appropriate),
(d) Data requirements and future research needs to improve understanding
(e) Costs (financial and societal) and benefits (useability and applicability/relevance) of each approach.

This synthesis was informed by discussion with and presentations by ten erosion experts who attended the project workshop held at the University of Exeter, January 2011.

Objective 2. Design a scientifically robust yet cost-effective monitoring framework for soil erosion in England and Wales

This task was supported by the literature reviews detailed above and the expert workshop. The results are synthesised into three documents: (1) A national sampling strategy to monitor erosion in the UK, (2) An evaluation of cost-effective methods and a strategy for the cost-effective monitoring of soil erosion and (3) A proposed pilot-study of a national erosion-monitoring framework.

Finally, this report is concluded in a summary section.

3. A review of soil erosion monitoring and modelling in England and Wales incorporating a summary of key findings from international soil erosion research (summary)

This chapter incorporates an executive summary of the synthesis of erosion research. The accompanying figures and tables and extended text may be found in Appendix 1.

3.1 Upland Soil Erosion

The review conclusions were:

• Upland soils provide important ecosystem services which are significantly degraded by erosion.

• The UK uplands have undergone a significant phase of late Holocene erosion and there is a significant risk of negative impacts of climate change and land use change on upland erosion.

• The most significant upland erosion has occurred across upland peatlands leading to significant degradation of the UK's most significant terrestrial carbon store. In this context there is a clear need for systematic monitoring of the erosion status of upland soils.

• Most measurement of sediment yields (as a proxy for erosion) (both suspended sediment monitoring and lake sediment based estimates of sediment yield) has been carried out at catchment scale and for site specific purposes.

• There are sufficient data to characterise typical upland erosion rates from damaged or degraded environments (i.e. landscapes know to erode) but the piecemeal nature of measurements does not provide a robust basis for ongoing monitoring of change.

• Where sediment budgeting approaches have been developed these demonstrate the potential to link on-site erosion rates to catchment based measures of sediment yield and also provide understanding of erosion mechanisms critical to supporting land management decisions.

• Harmonised Monitoring Sites (HMS) provide valuable sediment yield data but because of their extent they are of limited value in understanding upland erosion rates.

• A cost effective approach to monitoring would maximise benefits from existing monitoring sites by developing sediment monitoring "type sites" for representative upland systems. There is potential for joined up monitoring encompassing, for example, Environmental Change Network sites, Acid Water Monitoring Network Sites, and CEH carbon catchment sites. All of these locations have regular headwater stream sampling programmes but do not all routinely measure suspended sediment concentrations.

• Systematic measurement of erosion extent has the potential to provide important information on broad erosion trends. Single surveys tell us most about the current extent of degraded systems, greater value comes with repeat surveys that allow the trajectory of erosion to be established and estimation of erosion rates (areas and, with additional data, volumes).

• Remote sensing technologies offer considerable scope for erosion monitoring in a national context. Of the available technologies, repeat survey LiDAR measurements offer the most useful scale-appropriate data for type-sites subject to intensive monitoring and can be used to provide estimates of volumetric change so that erosion rates can be calculated. It allows measurement of spatially distributed erosion rates at the landscape scale. In combination with vegetation data from high-resolution satellites or aerial photos there is the potential to assess erosion-re-vegetation dynamics. There are, however, some important caveats. First, detection of change is a function of the spatial and temporal scale of the erosion features and the sensitivity of the remote sensing instrument used. Detection of fine-scale gully erosion features requires spatial resolution below 2m (Evans and Lindsay 2010) and much of the national scale LiDAR monitoring is not undertaken at sub-2m resolution. Similarly, typical achievable vertical accuracies of 30-120 mm and horizontal accuracies of 50-200 mm mean that successful change detection requires relatively long periods between surveys (ca 10 years).

Key Consensus points from the workshop were:

- Uplands and their ecosystems services are at risk due to erosion.
- Erosion monitoring in uplands must account for their heterogeneity
- The visible nature of erosion impacts on uplands renders them amenable to remote sensing approaches.
- LiDAR is a potentially important technology and results are best understood when linked to field monitoring for ground validation of data.
- It is important to identify the spatially distributed nature of surface properties such as vegetation cover (which can be achieved from LiDAR or photo-interpretation/classification of optical RS data) since bare ground is an important proxy for soil loss. However, it is equally important to capture gullies, low resolution capture of bare ground alone is insufficient.
- In upland systems catchment monitoring is important since sediment delivery is complex, data are needed for explanation and mitigation. Laser based turbidity probe technology has been suggested as a cost effective solution, however, such data must be collected alongside measurements of extent of erosion, in order to quantify on-site erosion rates.
- There is a need to build on existing work including both previous survey work and ongoing catchment monitoring programmes to provide long-term understanding of upland erosion rates.
- There was unanimity amongst the three workshop sessions that for the uplands, a monitoring scheme should combine catchment monitoring at representative sites (providing process insight and fine temporal resolution) with wider extent survey based on LiDAR data and/or ground survey (providing a wider overview at longer timescales and also capturing diverse erosion pathways, including wind erosion).

In summary, the review and the workshop arrived at a reasonable consensus that, in line with the majority of work to date, monitoring of soil loss from upland systems should at least incorporate elements of a catchment-based approach. Maximum benefit at minimum cost would come from collaborating with existing catchment monitoring programmes to produce sediment flux estimates. Catchment-based work needs to be complemented by assessment of the changing extent of upland erosion. This can be achieved by re-survey of existing upland erosion survey points such as the McHugh work (see Appendix 1) and also by repeat remote sensing, in particular using LiDAR data. This resurvey could be made cost effective by moving towards relatively long resurvey times (perhaps 5 or even 10 years). Trends in erosion rates at shorter timescales should be captured by monitoring at the representative catchment monitoring sites.

3.2 Arable land erosion

3.2.1 Water erosion

There have been a number of studies examining the annual rates of water erosion on arable land in the UK and many are thoroughly reviewed in Brazier (2004). These studies can grouped into three categories: plot or hillside studies where eroded sediment is collected; volumetric surveys, where the volume of rills and/or gullies is estimated and sometimes combined with information from aerial photographs; and rate estimation using fall out radionuclides, principally ^{137}Cs. A number of other methodologies including rainfall simulation, sediment fingerprinting and tracing studies have been used to study soil erosion, but have not been used to develop annual soil erosion rates and are therefore discounted from this review.

The project workshop considered the advantages and disadvantages of different survey methodologies. The results obtained from three discussion groups are presented in Appendix 1 (Table 2.1). During the discussions, there was a lot of agreement between the invited experts, including healthy discussion of the merits of approaches such as ^{137}Cs tracing. The broad consensus from the workshop was that whilst ^{137}Cs tracing has some limitations, in that it cannot be used on its own to quantify water erosion rates, when combined with tillage erosion modelling, which can deconvolve the signatures of each it is a useful tool for estimating erosion rates.

3.2.2 Outdoor pigs

No papers or reports could be found which have tried to evaluate the erosion rates from outdoor pigs. However, they are anecdotally recognized as a serious problem resulting in leaflets describing measures which can be adopted to mitigate erosion (CSF, unknown; Defra, 2005).

3.2.3 Wind erosion

There has been no systematic monitoring of wind erosion in the UK. There are also relatively few papers quantifying local scale rates of wind erosion in the UK although it is a well-known problem on light and organic soils during dry conditions. Studies that have been undertaken are reviewed in Appendix 1.

3.2.4 New technologies

Two main areas of technological advance offer potential for application in erosion monitoring: the ground based acquisition of digital elevation data (particularly in areas where soil is bare) and across a range of landscapes, the use of tracers. Both ground-based laser scanning (a terrestrial version of the airborne LiDAR more commonly reported) and the use of ground based digital photography offer the potential to capture digital elevation data rapidly. In addition, they have the potential to allow better quantification of volumetric surveys of rills and gullies.

Tracers, such as rare Earth oxides (Deasy and Quinton, 2010), are increasingly being used to trace soil erosion and deposition within arable fields. They have the potential to be interpreted in the same way as ^{137}Cs data, but because several species of rare earth can be used within the same field, there is greater potential for tracing trajectories of soil movement.

3.3 Lowland grassland erosion

3.3.1 Introduction

Grasslands cover a large portion of the temperate landmass including significant areas of Europe, North America, New Zealand and Australia (Peeters, 2004). Despite their extensive coverage, there is very little understanding of the rates of erosion and yields of sediments and colloidal material (fine inorganic and organic particulate matter with a diameter of less than 62µm, collectively termed suspended solids) from these environments (Brazier et al., 2007). This knowledge is essential in understanding and controlling the various sources of sediment and sediment-related contaminants in catchment surface waters - a concern to those involved in meeting the demands of environmental legislation such as the EU Water Framework Directive (Neal and Jarvie, 2005). It is also critical to inform our understanding of soil health, as grasslands have historically been perceived as low erosion landscapes, where soil health is good, whereas this may not actually be the case (Brazier et al., 2007).

A recent study by Bilotta et al. (2008) – part of Defra PE0120 - which monitored rates of erosion and yields of sediments and phosphorus from 1 ha field-scale grassland lysimeters, suggested that contrary to the common perception, temperate grasslands can erode and can present a significant threat to soil and water quality. However, it is proposed that the propensity of grasslands to erode and yield sediment material is likely to be dependent upon land-management factors such as presence or absence of drainage and stocking density (SD)[1] as grazing animals carry out three important activities that could enhance rates of erosion and yields of sediments and colloidal material from grasslands. These activities are; (1) defoliation, (2) treading, and (3) excretion, and it has been argued that the magnitude of degradation increases with SD (Bilotta et al., 2007). Whilst there are sound logical reasons to support this suggestion and indeed there are discrete pieces of research into separate forms of degradation caused by grazing animals, (for example Mulholland and Fullen, 1991; Trimble and Mendel, 1995). there is very little scientific data to link the degradation of vegetation and soil health to the degradation of surface waters in a systematic manner.

The following sections synthesise understanding of grassland erosion under a number of headings, with an emphasis on work undertaken in the UK. Appendix 1, Table 3.1 summarises the range of approaches that have been used to monitor/model grassland erosion in the UK.

3.3.2 Rilling or concentrated erosion

Overland flow is common in intensively managed grasslands, most often due to saturation excess mechanisms, but increasingly, as a result of soil compaction, due to Hortonian mechanisms (rainfall in excess of infiltration - see discussion in 3.3.5). However, visible

[1] the number of grazing animals per unit area, typically measured in livestock units per hectare or LSU ha^{-1} (Carroll et al., 2004),

evidence of entrainment and erosion of soil is rarely documented in grasslands due to the good vegetation cover that is afforded by grass. Thus, the use of overflight or volumetric surveys to quantify grassland erosion (for example Evans, 1988) may not always be a pragmatic solution to capture diffuse erosion evidence. Notable exceptions include the studies of Bilotta et al. (2009), where significant rates of erosion were observed from intensively managed grasslands. Discussion of the techniques used to quantify these erosion rates is made in section 3.3.6.

3.3.3 Surface sheetwash

Though less erosive than the more concentrated flows described above, sheetwash can erode material in interrill areas, particularly if soils are bare, such as in areas where cattle congregate (gateways, tracks, feeding troughs etc), as raindrop impact is likely to entrain material from bare soils far more readily than in areas where a thick grass sward is evident. Recent studies have assessed field-scale contributions of sheetwash and near-surface flow to erosion of intensively-managed grasslands (Bilotta et al., 2008 for example) relying upon the use of 'traditional' field outlet flow and sediment monitoring[2]. Building on this work, Granger et al. (2010) applied a number of tracers, including artificial fluorescent beads, to represent two different particle size fractions (mineral and organic), as well as both C_3 and C_4 slurries to study the loss of particulate matter from field scales (up to 5 ha) in diffuse overland flow. The traditional monitoring of sediment yields illustrated that significant erosion events can occur, particularly in undrained grasslands, where soils have reached field capacity (i.e. they are saturated) and therefore promote rapid saturation excess, overland flow responses. The tracing work, using the artifical beads and slurries with different carbon content, demonstrated that understanding the source of eroded material in grasslands, must consider the carbon-rich amendments to grassland soils (slurries, manures direct excreta) as well as the erosion of the mineral soils.

3.3.4 Subsurface drainage

Very few studies of the loss of sediment from grasslands that are under drained have been carried out, despite the extensive areas of drained agricultural land, especially in the western parts of the UK. Those studies that are reported tend to focus on experimental scales, such as 1 ha field lysimeters (Bilotta et al. 2008) and have to date employed traditional techniques to quantify erosion. Such techniques are exportable to other environments, or other scales, but would require significant investment in hardware, due to the engineering required to isolate drainage in a lysimeter.

Other studies (Deasy et al. 2009, for example) employ 'end-of-pipe' sampling to monitor subsurface erosion, or unbounded plots to observe surface erosion (see Appendix 1, Table 2.1), again observing flows and calculating sediment loads from concentrations of pump-samples. Such monitoring is less costly and easier to carry out than the lysimeter approaches highlighted above and can be said to represent more 'real-world' scales of observation. However, both examples involve time-consuming fieldwork and, of course, only describe the erosion responses of the specific areas being monitored.

[2] Traditional techniques are defined as those which take direct samples of soil and water to quantify erosion rates, usually via use of pump samplers triggered by flow proportional monitoring of hydrology at-a-point in the landscape.

3.3.5 Poaching/Livestock induced erosion

Bilotta et al. (2007) review the impacts of livestock on the soils and vegetation of intensively managed grasslands, concluding that livestock can be considered as significant agents of geomorphic change and damaging to soil condition. Not only do livestock compact the soil, which may lead to reduced infiltration and increased runoff rates, but they also denude the soil of vegetation and disrupt the soil structure such that it is more likely to be entrained by erosive rainfall. Subsequently, either sheetflow, concentrated runoff or subsurface flow can transport sediment into surface waters.

Studies to quantify the effect of livestock on erosion are limited, though Bilotta (2008) reports on the use of rainfall simulation techniques (detailed in 3.3.7 below) that illustrate the effect of increased stocking densities, soil moisture conditions and slope gradients on the loss of soil from small plots. Broadly speaking, increases in stocking densities, soil moisture content and slope gradient all lead to increases in erosion rates, under constant rainfall intensities.

3.3.6 Monitoring studies (plot to catchment scale) and sediment flux data

The approaches that have been used to monitor erosion in grasslands do not differ greatly from those described in arable lands (Section 3.2), but are far less common due to the perception that grasslands are ‘low-risk’ environments in terms of erosion rates. Those studies that have been undertaken are reviewed and described at the plot or hillslope scale in Bilotta et al. (2008), field scale in Deasy et al. (2009), and catchment scale in Bilotta et al. (2009). All studies to date rely upon the direct sampling of flows to quantify erosion rates, rather than the use of sediment sensors (turbidity probes or similar) which are then rated to represent the ‘true’ sediment concentration and then load. As such, the datasets describe erosion rates at various scales in detail across storm events, with a high confidence in the outputs, though for short time scales, due to the intensive nature of field sampling. Arguably some improvement of such datasets would be possible by using sediment sensors, to extend description of sediment fluxes for longer time periods, though the drawbacks to these sensors (discussed in detail in Bilotta and Brazier 2008) are noted here (i.e. lack of consideration of characteristics of sediments, uncertain relationships between turbidity units and absolute sediment concentrations and the need for local calibration).

3.3.7 Rainfall simulation studies

The only studies that have been carried out (to our knowledge) on grasslands are those described by Bilotta (2008) at the Little Burrows site, in mid-Devon. These experiments simulated rainfall on small plots ($0.5m^2$), to understand the effects of stocking densities on land with different slope gradients and soil moisture content. The results showed significantly different erosion rates between treatments, with increases in stocking density from 0 through 2, 4 and 6 LSU ha^{-1} leading to increases in erosion rates. >80 plots were constructed for this study, in order to provide appropriate replication of each stocking density/slope/soil moisture combination. Therefore, although such results are highly useful in describing the potential mobilisation rates of soil loss from the Hallsworth soil series, the study would need to be replicated form other specific soil series, if results were to be extended more widely.

3.3.8 Sediment fingerprinting and tracing studies

The most commonly used tracing approach in soil erosion studies across all landscapes, employs fallout radionuclides to quantify rates of gross or net erosion typically from agricultural land. However, one key assumption with this approach is that a baseline profile of ^{137}Cs or ^{210}Pb must be established in an undisturbed location, to enable quantification of the magnitude of disturbance that has occurred in areas where erosion (or deposition) has occurred. Many such baseline profiles are calculated in neighbouring level grassland fields, where it is know that land use, for example, has not changed since fallout. Clearly, the assumption is that erosion or deposition has not occurred in these fields, or that it has been negligible. Therefore, using fallout radionuclides in grasslands requires a rigorous methodology, which ensures that reference inventories are taken only from absolutely flat locations, where we can be confident that no erosion has occurred since fallout.

3.3.9 Modelling approaches

Due to the relatively low rates of erosion previously reported in grasslands, there has been very little attention on modelling such environments, in terms of soil erosion rates. Some recent studies have employed experimental modelling approaches to evaluate which (types of) models perform best in predicting grassland soil erosion (Krueger et al. 2009). Results showed that grassland systems are complex, responding to rainfall in very different ways depending upon rainfall dynamics and particularly the soil moisture status. Thus, it was shown to be difficult to develop a single 'model' of erosion behaviour from even simplified scales of observation such as the 1 ha plots, in Krueger et al.'s (2009) study. Such findings suggest that improved empirical knowledge of grassland erosion is required to develop understanding of the interactions between surface and subsurface flow and understand the impacts of the various land management strategies employed in grasslands before generic models of grassland erosion can be applied to the wider landscape.

3.3.10 Conclusions

- Grassland erosion studies are in their infancy, when compared with the study of soil loss in arable areas. The large spatial coverage of grasslands in England and Wales requires clear understanding of erosion rates and extent, due to the vast soil resource that is utilised to support grassland agriculture.

- The existing research suggests that most progress has been made using direct or traditional erosion monitoring approaches, though by their very nature, these techniques are limited to understanding processes occurring over limited spatial extents – i.e. up to the extent of the small catchment.

- Fallout radionuclide studies can be appropriate, where high confidence can be placed in the baseline inventory that is established, from local, level sites. Comparison of inventories between level, gently sloping and steeply sloping areas, where erosion is suspected will provide a multi-decadal perspective to set alongside the monitoring work.

- Other approaches such as volumetric or overflight surveys are less appropriate, as they fail to capture the detailed information required to explain the subtle nature of grassland erosion.

- Some progress has been made in modelling these environments, though only with very simple models, due to the lack of data support for anything more complex.

- At present grassland erosion is being overlooked (underpredicted) by the suite of existing models, as they are not parameterised (or well structured) to predict erosion under environments where near 100% vegetation cover predominates. However, grassland erosion is contributing to quite significant water quality problems in locations such as the Tamar Lakes and the Exe catchments.

- Improvements to tracing techniques are required before such approaches add value to grassland erosion understanding. Some progress has been made by Granger et al. (2009; 2010a,b), but these techniques tend to focus upon tracing amendments to the soil system (carbon in the form of slurries and manures for example) as opposed to material that is already part of the soil.

- Finally, further quantification (through field observations and tracing studies) of subsurface erosion, particularly through field drain systems is needed, as it clearly contributes to the overall erosion signal of grassland catchments, but is currently somewhat poorly understood.

3.4 Mechanical erosion

3.4.1 Tillage erosion

When soil is cultivated the cultivation layer is translocated (i.e. moved in the direction of tillage, or laterally, or both). If translocation distances were uniform and tillage directions were reversed each year no net translocation would take place. Although tillage directions are typically reversed in successive operations, translocation distances are not uniform and this results in net soil loss (erosion) where more soil is exported from a location than imported and net soil gain (aggradation) where the opposite condition pertains. Rates and patterns of net soil redistribution derived using fallout radionuclides suggest that net soil loss (thinning) and net soil gain (thickening) due to tillage erosion dominate on-site soil redistribution and the on-site impact of erosion on soil properties (Govers et al., 1996; Quine et al., 1997; Quine and Zhang, 2002). In areas of net soil erosion due to tillage (spurs and ridges; land adjacent to upslope field boundaries) either plough depth is maintained by incorporating subsoil into the plough layer, which is gradually depleted in SOC and nutrients, or there is a gradual reduction in soil depth. In contrast, in concavities, nutrient and SOC-rich former plough soil accumulates and is buried below the current plough layer. This uneven distribution of the resources most important for crop growth (including water-holding capacity) is likely to be deleterious to crop growth and there have been a small number of empirical studies that have provided evidence of this effect including one in the UK (Quine and Zhang, 2002).

Evidence base

There has been no systematic monitoring of rates of tillage erosion in the UK or elsewhere. Evidence for rates of tillage erosion has been derived from:

- Short-term, single or multiple pass ‘tillage erosion experiments’
- Long-term multiple pass ‘tillage erosion experiments’
- Numerical modelling of tillage erosion using relationships derived from experiments
- Deconvolution of the contribution of tillage erosion to net soil redistribution using tillage erosion models. The net soil redistribution data being obtained using: fallout radionuclides caesium-137 (^{137}Cs) and lead-210 (^{210}Pb); photogrammetric analysis of time series aerial photographs; and, serendipitous evidence for landscape change (e.g. land surfaces adjacent to hedges).

Tillage erosion experiments have been used to explore the topographic control on translocation distances and soil transport associated with tillage, and also to define a general relationship relating tillage translocation to slope that underpins the majority of tillage erosion models (Govers et al., 1994). In broad terms, the intensity of tillage erosion is determined by the topography and the soil transport coefficient (*ktil*). Single-pass experiments have demonstrated that the magnitude of *ktil* is determined by soil type and condition (moisture, antecedent treatment) and implement type, speed and depth of cultivation. Van Oost et al. (2006) synthesized the body of experimental studies to that date. They demonstrated that, for the experimental dataset, the soil transport coefficient (*ktil*) associated with mouldboard tillage can be predicted well (r^2 = 0.79) if soil bulk density and tillage speed, depth and direction (contour or up/down) are known' however, these data are not routinely available for agricultural land. The scarcity of data for other operations (e.g. chisel cultivation, secondary operations) prevents reliable development of equivalent predictive relationships. Therefore, in large-scale (national and larger) simulations to date (Quine et al., 2006; Van Oost et al., 2007, 2009;) 'reasonable' total soil transport coefficient (*ktil*) for the full suite of tillage operations (including no till, conventional tillage and conservation or minimum tillage) have been defined and used (respectively: 600; 600[3], 500 kg m^{-1} $year^{-1}$).

In summary, although a tillage erosion map for England and Wales exists (Quine et al., 2006) and is likely to be accurate within 1 order of magnitude (expert judgement) it relies on a relatively small database of experimental data that has for the greater part been gathered from sites located outside of the UK (1 of 34 mouldboard plough experiments cited by Van Oost et al. (2006) was a UK study, 14 of 34 were from Mediterranean environments; none of the chisel or secondary plough data were from the UK). Therefore, in the light of the magnitude of tillage erosion rates and their effect on soil properties within arable fields, there is a clear need to broaden the UK database for tillage erosion. Not only will this allow current tillage erosion rates to be estimated with less uncertainty, it will also allow identification of best practice to reduce tillage erosion and, will permit separation of the effect of tillage erosion on fallout radionuclide distributions and the derivation of multi-decadal water erosion rates.

3.4.2 Soil loss with crop harvest (SLCH)

Soil export with root crops has been a concern of the sugar industry in particular for several years as it is a by-product of their industry that requires removal from the crop and environmentally-responsible disposal. Soil export with vehicles occurs when soil adhering to vehicles in a source area is deposited (gravity or washing) elsewhere, thus contributing to net soil loss from the source area. The impact of SLCH on harvested fields is one of gradual topsoil and nutrient export. The impact on fields is uniform – unlike tillage erosion – and, therefore, the loss of nutrients can more readily be compensated in whole field treatment.

Evidence base

Despite the potential on-site and off-site costs associated with SLCH, it has received relatively little scientific attention. Ruysschaert *et al*. (2004) have synthesized European work on the subject and, in support of a Defra-funded scoping study (SP08007), Owens et al. (2006) reviewed the scientific and industrial literature and Quine et al. (2006) explored the potential to simulate SLCH rates at a national scale. No routine monitoring of SLCH is undertaken, however, two sources of data may be valuable is assessment of the scale of the problem: (i) isolated published studies; (ii) industry estimates of the magnitude of the soil tare associated with particular crops. Ruysschaert et al. (2004) include no UK examples but demonstrate the potential for significant losses (ca 1-10 t ha^{-1} $harvest^{-1}$) of soil associated with potatoes, carrots, chicory and sugar beet, results which could be tentatively extrapolated to England and Wales, for similar soil types. Quine et al. (2006) explore British Sugar (2002) data for total national soil removal with sugar beet, illustrating that

[3] Van Oost et al. (2007) used 600 kg m^{-1} a^{-1} for mechanized agriculture; 150 for non-mechanised and 300 where reduced tillage is widely employed.

they are consistent with a soil loss rate of 3 t ha^{-1} $harvest^{-1}$ (equivalent to a mean annual rate of 1 t ha^{-1} a^{-1} assuming a 3-year rotation and no soil export associated with the crops grown in the other 2 years of the rotation). The soil tare associated with sugar beet is estimated to have declined from >15% in 1987 to ca 6% in the mid 1990s. At today's yields the 1987 figure would equate to soil loss of 7.5 t ha^{-1} $harvest^{-1}$ and, therefore, some arable land may exhibit the legacy of significant past erosion associated with sugar beet cultivation. To date, equivalent data from other industries/crops have not been located.

Quine et al. (2006) explored the potential to use the synthesis of Ruysschaert et al. (2004) to develop a general modeling tool for SLCH for investigative application at the national scale. The sole consistent pattern was that soil moisture during the harvesting season was the major factor determining soil erosion rates and that rainfall depth during the harvest season was a good proxy for soil moisture. Therefore, Quine et al. (2006) used UKCIP rainfall data and a synthetic regression relationship (based on relationships derived for Belgium, France and Germany by Ruysschaert et al. 2005), with intercept scaled so that the average dirt tare for England and Wales equalled c. 6%, to explore potential patterns in SLCH. This approach showed promise, but would need to be validated against SLCH data at specific sites to demonstrate its real value.

In summary, although SLCH erosion modelling for England and Wales has been undertaken (Quine et al., 2006) the work should be seen as exploratory, given the limited evidence base and the resultant necessary simplicity of the modelling approach.

3.4.3 National-scale Monitoring of Mechanical Erosion: Approaches and Discussion Points

The challenge is to quantify soil loss that leaves no immediate visible trace. Approaches to *in situ* assessment of mechanical erosion will require the capacity to establish net change against a datum and options include: caesium-137 inventories; buried and relocated tracers; repeat LiDAR; measurement against a pre-existing local datum. All such approaches capture surface change due to all erosion processes (modelling is required to separate contributions). Short-term (single pass) tillage erosion experiments should be conducted to extend the database to include representative UK soil and tillage conditions. Assessment of total soil loss associated with root crop harvest could be based on measurement at processing plants (lacks spatial information).

3.5 Channel bank and floodplain erosion

3.5.1 Introduction

Erosion of channel banks occurs through a range of internal processes mediated by external forcings (e.g. Leopold et al., 1964). Internal processes differ in both style and magnitude as a function of both time-scale, spatial-scale and position within both the local and catchment river network (e.g. Couper and Maddock, 2001; Lawler, 1992, 1995, 1997; Thorne, 1991). Outer-banks constantly lose sediment through the secular entrainment of particles through shear-stress imposed on channel walls by fluid flow. Mass failures occur on a sporadic basis over short time-scales but more uniformly over longer time-scales. The rates and magnitudes of erosion and erosional events are dictated by material properties (e.g. Darby et al., 2000; Lawler, 1986; Thorne, 1991), antecedent conditions (e.g. Simon et al, 2000), local climate, water and sediment discharge (Parker, 1976, Howard, 1992; Mosley, 1975), riparian vegetation (e.g. Coulthard, 2005; De Baets et al., 2006; Tal and Paola, 2007; Wynn et al., 2004) and microbial activities (e.g. Gerbersdorf et al. 2009). The role of antecedent conditions is likely to be very important, as it leads to the likelihood of hysteresis and the simple but important consequence (for planners in particular) that the same forcing will generate different responses of channel bank erosion rates.

External forcings correspond to local and regional (catchment-scale) climate, land-cover and land-use management. Climate, in particular, is a complex forcing characterized by a number of important parameters (e.g. central tendencies, variance, frequency-magnitude distributions and clustering). Climates with the same mean value can be very different and have significantly different consequences when imposed on a specific environment.

In a dynamically steady-state reach, erosion of one part of a channel bank will be balanced by deposition on a downstream floodplain or onto the channel bed over the relevant time-scale. In this case, the statistical characteristics of the river network are likely to be constant over the relevant time-scale. At other time-scales, the geometrical characteristics of the river system is likely to fluctuate within set limits or where the system is out of equilibrium at all time-scales, erosion (or sedimentation) will by definition be unbalanced, and the geometrical and hydraulic characteristics of the river network will be non-stationary. The volume of sediment eroded from channel banks is rarely quantified at spatial and temporal scales that are useful for planning time-lines of 30-50 years. Where such data are known, however, the quantity of sediment derived from channel bank erosion is similar to the amount of sediment in transport (e.g. Dunne et al., 1998).

The following synopsis reviews the methodologies entailed in documenting UK channel bank (and floodplain) erosion, and summarizes the limited amount of extant data.

3.5.2 Livestock-induced bank erosion

Cattle are important agents in promoting channel bank erosion. Uncontrolled grazing was shown to cause about six times as much gross bank erosion as on a protected stretch in a small perennial stream in Tennessee (Trimble, 1993). Most of the difference was attributed to physical breakdown of the banks by trampling and consequential erosion, rather than scour as a consequence of the removal of vegetation.

Whilst there may not be an extensive literature on erosion rates associated with cattle grazing, in the UK, this process is considered in government guidance to good practice with respect to riparian vegetation management (SEPA, 2009) and protection against soil erosion (Defra, 2005). These documents draw attention to the potential for channel bank erosion associated with "poaching" of riverbanks by grazing animals. To minimise soil erosion both documents SEPA (2009) and Defra (2005) recommended that stocking rates are controlled and that overgrazing near river banks and uncontrolled access to watercourses is managed.

3.5.3 Sediment fingerprinting

A review of the development of techniques associated with sediment source tracing, or fingerprinting, in catchments and river systems to establish the relative contributions from potential sediment sources was presented by Walling (2005) and this forms a key source for the following summary.

Fingerprinting of suspended sediment can be applied to a range of applications:

- identifying sources of contaminants,
- sediment characterisation in urban catchments,
- relative contributions to erosion, e.g. relative contributions of bedrock, sheet and rill, channel, or gully erosion.

Ideally fingerprinting might comprise a single chemical or physical property that clearly differentiates potential source materials.

Walling (2005) presented a summary table of estimates of source type contributions for a selection of UK catchments obtained using fingerprinting. The % channel bank contribution is reproduced in Appendix 1, Table 5.1, which may be converted to volumetric contributions by going back to the source papers.

3.5.4 Monitoring studies and sediment flux data

Monitoring of bank and channel erosion has been undertaken using a number of techniques, including:

- Historic map and aerial photograph analysis
- Use of time series digital elevation models (DEMs)
- Erosion pin monitoring
- Photo-Electronic Erosion pin monitoring, which can be deployed with Thermal Consonance Timing (TCT, Lawler, 1996)
- Traditional surveying techniques; planimetric resurvey
- Repeated cross-profiling
- Automated video recording

- Electronic surveying techniques, including Lidar (airborne laser altimetry)
- Turbidity and associated colourimetric techniques
- Botanical evidence from floodplain surfaces and deposits

A review of the monitoring techniques was presented by Lawler (1993). This includes a comprehensive overview on the techniques used by researchers between 1863 and 1988 and the catchments to which they have been applied. The techniques are considered in the context of the temporal periods over which they are best applied, i.e. the long, intermediate and short timescales.

Additional techniques described by Lawler (1993): morphological evidence, local knowledge, hydrographic resurvey (of subaqueous bank recession), sediment traps (to collect material falling from above as a check on erosion pin monitoring), painted sections, or pebbles on river banks, erosion box for submersion of sediment in rivers to monitor erodibility, and the erosion frame: a box positioned on corner legs with penetrating holes through which the depth to the ground surface can be measured.

3.5.5 Historic map analysis

Historic maps and photographs can be particularly useful for assessing channel stability over longer time periods and for monitoring catchments in cohesionless deposits, which are less well suited to erosion pin monitoring (see Hooke, 2004; Hooke and Kain, 1982 for examples).

3.5.6 Use of successive digital elevation models (DEM) from aerial photographs

This is not yet a commonly applied technique for channel monitoring in the UK, although there have been some studies (Brasington et al., 2003) and remote sensing specialists have been developing automated photogrammetric algorithms for a range of applications of the analysis of oblique and vertical photographs, e.g. Chandler, 2001 and Mount et al., 2003. Algorithms developed by Mount et al. (2003) focused on image-to-image comparisons of the bankfull-width of the Afon Trannon, mid-Wales during the period 1948 to 1995.

Brassington et al. (2003) compared a photogrammetrically-derived DEM with GPS ground survey results in the uppermost braided section of the River Feshie, in the Cairngorm Mountains, Scotland. It was concluded that the remote methods provided qualitatively convincing, moderate precision DEMs of channel topography, but the error (+/- 0.21 m) renders volumetric change calculations unreliable.

Schneider et al. (2010) used multi-date DEMs from photogrammetrically derived elevation data together with bulk density data (from a 20 m x 20 m sampling grid) to generate 3D models of erosion and deposition in the software package GOCAD. The technique was applied to an artificially created catchment constructed as part of an open-cast coal remediation project: Chicken Creek, approximately 150 km to the south of Berlin, Germany. DEM-differencing gave a sediment-change volume of 122,608 m^3 with an accuracy of +/- 10%.

In New Zealand De Rose and Basher (2010) investigated river bank and cliff erosion over five decades along a 16- and 28-km stretch of the Waipaoa River, East Coast, North Island, New Zealand. Volumetric rates of bank-erosion were derived from a comparison of LIDAR-derived DEMs (from 2005) and those derived from photogrammetric analysis of historical aerial photography. The study showed that LIDAR derived DEMs can assist with the orthorectification and construction of surface elevation models from historical aerial photography with a greater level of accuracy than has previously been possible. This in turn enabled more accurate measurements of bank erosion and channel migration rates.

One difficulty with the use of DEMs in channel bank monitoring is that they do not include channel bed topography. Schäppi et al. (2010) make reference to the potential development of bathymetric LiDAR systems (e.g., Hilldale and Raff, 2008) and the use of multispectral imagery and aerial photography.

In summary, whilst there are clearly numerous techniques to monitor channel-bank erosion (reviewed in detail in Appendix 1), data from such research typically describe long-term erosion rates, at catchment scales. Consensus from the erosion workshop was that whilst such information are important, in understanding channel and catchment evolution, they are perhaps not so relevant to describe the impacts of erosion on the soil resource.

3.6 Erosion monitoring: National data sets and national erosion assessments

3.6.1 Introduction

The objective of this review was to look at all available literature on soil erosion monitoring and modelling at the national scale in England and Wales, where there are three categories of monitoring:

1. Monitoring the actual incidence of soil erosion (Appendix 1, Table 6.1)
2. Monitoring the indicators of erosion probability[4] (Appendix 1, Table 6.2)
3. Monitoring the off-site consequences of erosion (Appendix 1, Table 6.3)

Categories 2 and 3 provide estimates of actual erosion through proxy measures.

3.6.2 Category 1. Monitoring actual incidence of soil erosion at the national scale (Appendix 1, Table 6.1)

In order to monitor actual evidence of soil erosion, the wide range of literature reviewed (from all types of landscape reviewed above) it is suggested that techniques should monitor existing erosion features; degree of activity; and stage of development. Monitoring could then be static; dynamic; or sequential. Spatial variations in erosion intensity, location, nature and status could then be used to evaluate erosion prediction models, target areas for soil conservation and indicate future erosion probability.

3.6.2.1 Monitoring actual incidence of soil erosion with remote sensing techniques

The application of remote sensing techniques (satellite, airborne, ground-based) for monitoring soil erosion assumes:

- The erosion process being monitored produces a detectable change in the spatial or temporal pattern of electromagnetic radiation (EMR) recorded by the remote sensor (Milton at al., 1987; 1995).
- Changes in landscape features are separable from radiometric distortions arising from the sensor (Collins and Walling. 2004).
- Measurements are acquired at scales suited to erosion features (Kirkby, 2010).

For cost-effective monitoring an obvious source of data are the freely accessible, regional-scale satellite systems (e.g. NASA's Landsat and ASTER sensors). However, these deliver data at spatial resolutions coarser than the grain size of many erosion features. This scale issue can be addressed with "linear spectral un-mixing" algorithms, which enable the description of the percentage of each pixel devoted to each cover type (for example bare soil vs vegetation cover). For details see: (Hill et al., 2000; 105, Metternicht and Fermont., 1998; 64, Haboudane et al., 2002). However, these techniques do not provide information on the within-pixel spatial location of the different components (bare soil or vegetation cover, in this example), so retrieval of within-field scale variability is often difficult and the accuracy of this technique remains un-tested in soil erosion applications. On the other hand, the new generation of commercially-operated fine spatial resolution sensors (e.g. IKONOS, Quickbird and GeoEye) offer data at appropriate spatial resolutions for soil erosion investigations (pixel size <4 m), but data are expensive to buy[5].

[4] The term 'probability' is used rather than 'risk'. Risk is strictly defined as probability × damage

[5] The minimum order for a standard, natural-colour, pan-sharpened (highest resolution) Quickbird image from the archive is 25 km^2, at a cost of \$17/km^2, giving a total price of \$425. To commission a new acquisition is typically more costly than this quoted value.

At the other end of the scale, hyper-temporal sensors (e.g. Meteosat, NOAA AVHRR, and MERIS on Envisat) are typically capable of delivering at least an image per day, but this temporal revisit time is only made possible by compromising on spatial and spectral resolutions. This means that pixel sizes are very large relative to the scale of erosion features and thus, the ability to detect detailed changes in land surfaces are out of the question. Another issue to consider with Earth observation sensors is that with passive optical data (i.e. most of those sensors mentioned above), cloud cover limits data availability, reducing effective revisit time. The optimal way to address this challenge is to use both active (i.e. sensors producing their own radiation such as RADAR) and passive satellite data (multi-sensor approaches) to benefit from the positive aspects of both [Isaacs, and Srivastava., 1989; Moran et al., 2000; De Jong et al., 1999].

Airborne survey data can be acquired on demand when conditions are favourable, and at a scale-appropriate resolution (see Appendix 1, Table 6.4), but these carry higher data acquisition and processing costs, and the spatial coverage is reduced. Deploying airborne sensors over large extents can generate vast volumes of data, with high redundancy and high processing times. However, these data can be delivered at appropriate temporal periods assuming funds allow. Collection of airborne remote sensing data is also rapid (e.g. 6,000-8,000 ha can be acquired in a day). The erosion expert group discussed the use of remote sensing data and concluded that airborne surveys of erosion would be most cost-effective if applied to sites at the scale of the fluvial catchment or less. A more reasonably priced (but less responsive alternative) would be to utilise commercial satellite data from sensors such as IKONOS, because they can provide data at spatial resolutions appropriate for erosion studies (e.g. <4 m).

Some datasets (e.g. airborne LiDAR) will provide immediate visual assessment of larger erosion features, if collected at the appropriate spatial resolution. Monitoring is time specific, so that rare erosion events will be either over- or under-represented, unless regular monitoring takes place. The amount of remote sensing data needed for national coverage at sufficient spatial resolution would be prohibitively high and would generate a huge volume of data (with associated storage, archiving and processing costs). For this reason, remote sensing techniques are rarely used to monitor soil erosion directly over large spatial extents.

A recent advance in the field of remote sensing analysis has been developed by Google Earth, which offers a free-to-use GIS portal for global users. Through Google Earth users can access, view and perform basic manipulations of remote sensing data from the global to the local scale. Increasingly, fine spatial resolution datasets including national air photo surveys are being made available to users through Google Earth, a development that promises much for cost-effective monitoring of soil erosion. Although users cannot access the raw data, they are able to “zoom” to a very fine level of detail, potentially enabling basic site-specific assessment of soil erosion to be evaluated, recorded and disseminated.

3.6.2.2 Monitoring actual incidence of soil erosion using point sampling techniques

The National Soil Inventory (NSI) surveyed England and Wales, with 5660 sites (5km grid). The properties at each site such as soil type, land use, slope, aspect and the visual presence or absence of soil erosion or deposition were recorded. Sheet and rill erosion features were observed at 89 sites (Appendix 1, Figure 6.1; Table 6.5).

The National Soil Inventory Erosion Survey (1995-1998) (NT1014) used subsets of the NSI to derive erosion risk classes for arable land (270 sites), grassland (135) and uplands (339). Direct evidence of soil erosion and site characteristics were recorded at each site on a number of occasions (1995 – 1998). Where present, the volume of soil eroded and deposited was determined. A further survey of the 206 eroded upland sites (identified in 1997) was carried out (Defra SP0402). At each site the length, width and depth of erosion features were measured.

270 field sites located within arable agricultural areas across England and Wales and 139 upland sites were selected for the Arable and Upland NSI Erosion Resurvey (Defra SP0407). Each site was selected from the NSI sites used previously for assessment of arable erosion (1996-1998; NT1014; Harrod, 1998) and upland erosion in 1999 (SP0402). The average length, width and depth of erosion and deposition features were assessed. Most sites had to be surveyed using aerial stereoscopic images of each field site at 1:5000 in true colour (due to foot and mouth restrictions preventing access to the sites), showing erosion features, bare soil, crops and cultivations. All the upland sites were resurveyed using original survey techniques.

3.6.2.3 Monitoring actual incidence of soil erosion using questionnaires
Questionnaires have been used at the national level to canvas landowners' perceptions of soil erosion. In the 2007 Farm Practices Survey, 50% of farmers said their land suffered at least some erosion. This approach may give good coverage, but erosion may be over / underestimated, and in voluntary schemes, number of returns is usually low. However, the infrastructure for gathering this information already exists e.g. Cross compliance returns.

3.6.2.4 Conclusions to monitoring of actual incidence of erosion
Most monitoring of soil erosion features has occurred at the regional scale, often based on amalgamated field-scale data (Evans, 2002; Table 6.6). Evidence reviewed by Brazier (2004) suggests that soil erosion occurs on a wide range of soils and land uses throughout England and Wales. Soils most susceptible to water erosion are sandy soils in south west England (e.g. rates of 0.6 - 4.1 t ha^{-1} a^{-1};Walling et al., 2005); south east England (e.g. 0.65 - 6.5 t ha^{-1} yr^{-1} and 31 - 234 t ha^{-1} during one winter season; Boardman et al., 2003; 2009), East Anglia, the Midlands and South Wales; chalky soils on the Wolds and in East Anglia (Defra, 2005) and brown earths (Wood et al., 2006). Wind erosion has been recorded mainly on sandy and peaty soils in the eastern and middle counties of England and in the uplands of England and Wales, with mean rates estimated between 0.1 - 2 t ha^{-1} a^{-1} . In comparison to water erosion, the area affected is small (Chappell and Warren, 2003). Almost no data exist at the national scale of soil loss by tillage erosion or co-extraction on crops and vehicles (Owens et al., 2006a). The 'State of the Environment' (Environment Agency, 2005), estimated that only 17% of the land in England and Wales is affected by erosion. However, Evans (2002) estimated that 29% of farmland is affected at an average rate of 1.9 t ha^{-1} yr^{-1} (1982-1986; Table 6.6).

3.6.3. Category 2. Monitoring Indicators of erosion probability (Appendix 1, Table 6.2)

Several techniques measure and monitor indicators of erosion probability (e.g. rainfall/wind erosivity; soil erodibility; slope length, gradient and form; land use, cover and management). These are proxy data - they do not monitor soil erosion directly, but are assumed to correlate strongly with erosion incidence. These predictors should be verified by actual measurements of erosion, but these have been ad hoc in the UK (see other chapters). Some indicators are simple, robust, easily available and can be mapped to give an immediate visualisation of spatial differences in erosion probability. However, many of these parameters are dynamic in nature, (especially given the impact of climate change – see Defra SP1601) as such, accurate monitoring requires a high temporal resolution. Some data are difficult to derive (e.g. rainfall erosivity) or visualise (e.g. socio-economic factors).

Monitoring of single factors may lead to inaccurate assessment of erosion probability, as in reality combinations of factors interact to determine erosion probability. Methods of combining factors include thematic overlays (each factor given an equal weighting in determining erosion probability, with no interaction between factors) or using the data as input to erosion models (see Appendix 1, Chapter 7).

3.6.3.1 Monitoring erosion probability using remote sensing (RS) techniques

Remote sensing can be used to derive information used in the assessment of erosion probability, including land cover/use, DEMs, changes in surface roughness, land management, soil properties (moisture, roughness, texture, particle size distribution, SOC and sub-surface soil features; Appendix 1, Table 6.7). A spatial understanding of the distribution of these variables is very helpful in soil erosion monitoring. However, in the case of soil properties, conflicting “signals” in reflected or emitted radiation arise from different soil properties (moisture, roughness and SOC), which all have a similar effect on reflected radiation in visible (VIS; 400-700 nm), near infra-red (NIR; 700-1100 nm) and short-wave infra-red (SWIR; 1100-2500 nm) regions of the spectrum. An increase in near-surface moisture will cause a decline in the magnitude of the reflected energy (Appendix 1, Figure 6.2), while the same will occur if SOC increases, and if soil roughness increases. Decoupling the cause of changes in reflectance from each other requires ground validation data (e.g. from instrumented networks) to verify the data [Irons et al., 1989]. Similarly, it is computationally challenging to decouple the signals from surface moisture and roughness in RADAR data (Baghdadi et al., 2002).

A second challenge to the effective application of satellite observations to factors affecting soil erosion is the uncertainty in the relationship between chemical composition of soils and the spectral signature recorded by the sensor [Ben-Dor et al., 2002]. This issue is exacerbated if the number of sampling points in the electromagnetic spectrum are reduced (i.e. either as the effective spectral resolution or the number of spectral bands are reduced). The main issue here is that in addressing the spatial resolution of the data (i.e. by decreasing pixel size), the likelihood of spectral ambiguities increases. The new generation of satellite sensors with fine spatial resolution capabilities (IKONOS, Quickbird) illustrate the trade-off in spatial and spectral resolution. These sensors provide data at <5 m pixel resolution, but in doing so, they compromise the spectral domain by only supplying broadband data in four bands (R,G,B and NIR). Choice of data sources must consider the trade-off in spectral and spatial resolution.

3.6.3.2 Monitoring erosion probability using direct techniques

Environmental factors, either singularly or in combination (e.g. soil properties such as texture, carbon content; land cover, climate data) are used to map erosion probability. Examples include slope, soil and climate data (Wood et al., 2006; Appendix 1, Figure 6.3); rainfall erosivity and soil erodibility (Morgan, 1985 (Appendix 1, Figure 6.4); Boardman and Evans, 2006 (Appendix 1, Figure 6.5); and soil chemical composition to predict aeolian erosion potential (Appendix 1, Figure 6.6). The original LandIS erosion risk map (Appendix 1, Figure 6.7) is based on slope and texture, and classified into 'risk' (Y) or 'no risk' (N), assuming slopes greater than 7^o are likely to erode. When this map was validated against observed erosion features (NSI data; Appendix 1, Figure 6.1), 65% of eroded sites were correctly identified as having erosion risk, and 35% of eroded sites were incorrectly identified as having no erosion risk. By refining the input data on organic carbon, clay % and land use, the accuracy and reliability of this approach were improved with 96% of sites with observed erosion (NSI data) were correctly identified (Whitmore et al., 2004; Appendix 1, Figure 6.8; Figure 6.9).

3.6.3 Category 3. Monitoring off-site consequences of erosion (Table 6.3)

Since around 1970, the British Geological Survey has undertaken a baseline, headwater stream bed sediment sampling project (G-BASE) at 1 sample per 1.5 km^2; (Johnson et al., 2005; Appendix 1, Figure 6.10). The sediment samples are collected over the summer months and analysed for their geochemical composition. More recently, archived sediments have been analysed to determine physical properties (e.g. BET specific surface area (Rawlins et al., 2010)) and total organic carbon content to assess the spatial variation in carbon and phosphorus losses (Rawlins, submitted). Bed sediments have now been collected across around 70% of England and Wales, equating to around 85,000 sites. Appendix 1, Figure 6.11 shows the estimated distribution of sediment delivery for the 1 in 10 year storm event as modelled by McHugh et al. (2002). These studies are vital in monitoring the off-site impacts of erosion. However, they are spatially and temporally removed from on-site erosion events.

3.6.4 Conclusions

Direct monitoring of erosion at the national scale in England and Wales is limited. Large extent monitoring (e.g. using satellite imagery) is at odds with the scale at which erosion processes (and mitigation measures) operate, and cannot convey any local variations in erosion incidence/risk. Monitoring has been focused on arable and upland areas, focussing on erosion by water. Most studies have monitored the factors affecting erosion to produce maps or models of erosion probability. However, the relationships between these proxy variables and actual erosion rates may vary under changing climate/land use and management scenarios (see Defra SP1601) and there will also be uncertainties associated with the models used, that have not been considered.

Monitoring at the national scale will require calibration and validation at the field scale (from data summarised in Appendix 1, chapters 1-5). Most remote sensing data only provide proxy measurements, and these measurements must be validated on the ground. All observations could be better-integrated into erosion models, possibly within GIS. Remote sensing shows great potential for erosion monitoring, but image processing algorithms have to be better at considering pixel-to-pixel interactions. Geostatistical or contextural analysis may overcome this limitation (Isaacs, and Srivastava., 1989).

National scale monitoring with RS data, would be useful in targeting areas worthy of more detailed observations, using a wide range of techniques. This has partly been done already (e.g. Evans, 2002; Morgan and Nearing, 2010), identifying areas prone to erosion by water (but less so for other forms of erosion). Certain landscapes are likely to remain at very low erosion probability, however, it is suggested that RS monitoring must be undertaken in an unbiased manner across the UK, assessing both these sites and highly erosive sites, in order to establish the full spatial extent of erosion in the UK.

3.7 Modelling Soil Erosion

3.7.1 Introduction

There are a wide range of soil erosion models available (Appendix 1, Table 7.1) which broadly fit into two groups: empirical and process-based models. Volumes edited by Morgan and Nearing (2010) and Harmon and Doe III (2001) give more detail on soil erosion modelling, highlighting the strengths and weaknesses of each approach.

3.7.2 Scales of application

Most soil erosion models were designed for use at field or small catchment scale. Such models require numerous parameters that are not available in national data sets and have model structures, which are unsuited to large-scale application. National scale modelling therefore requires the use of models with more limited data requirements. These include PESERA (Kirkby et al., 2008; Wischmeir and Smith, 1978), MMF (Morgan, 2001; Morgan et al., 1984), the MIRSED approach (Brazier et al. 2001a;b) or the more empirical USLE (Wischmeir and Smith, 1978) and its subsequent modified versions. However, to date only the PESERA model has been applied to the UK at a national scale. This exercise was undertaken as part of an EU funded project which produced a pan European map of soil erosion rates (Appendix 1, Figure 7.1). The MMF model, MIRSED and the USLE have been applied at regional scales (see for example Lu et al., 2003; Brazier et al. 2001a,b; Scholz et al., 2008), all of these models could theoretically be applied at the national scale.

3.7.3 Model evaluation

Model evaluation is difficult at regional or national scales since there are few data available against which to test them. If any model testing is carried out at all, model evaluation is usually performed against plot or field scale data, as in Scholz et al., (2008) before the model is applied to wider areas. Although a pragmatic approach, due to the limited availability of data, this is unsatisfactory since there is a mismatch of scale between the evaluation exercise and the model application. Therefore, if as suggested at the project workshop, erosion models are to be used to interpolate the monitored data to areas where no monitoring was conducted it is essential that they are first evaluated within an uncertainty framework (see below in Figure 3.1).

3.7.4 Conclusions and ways forward

It is not practical to use complex, parameter intensive models to make predictions at national or regional scales. The information to parameterise such models does not exist and the computing requirements required to run the models would be too great. Therefore, we propose that less parameter intensive models, such as MMF, PESERA, or Minimum Information Requirement versions of complex models are used. We suggest adopting the process illustrated in Figure 3.1 below.

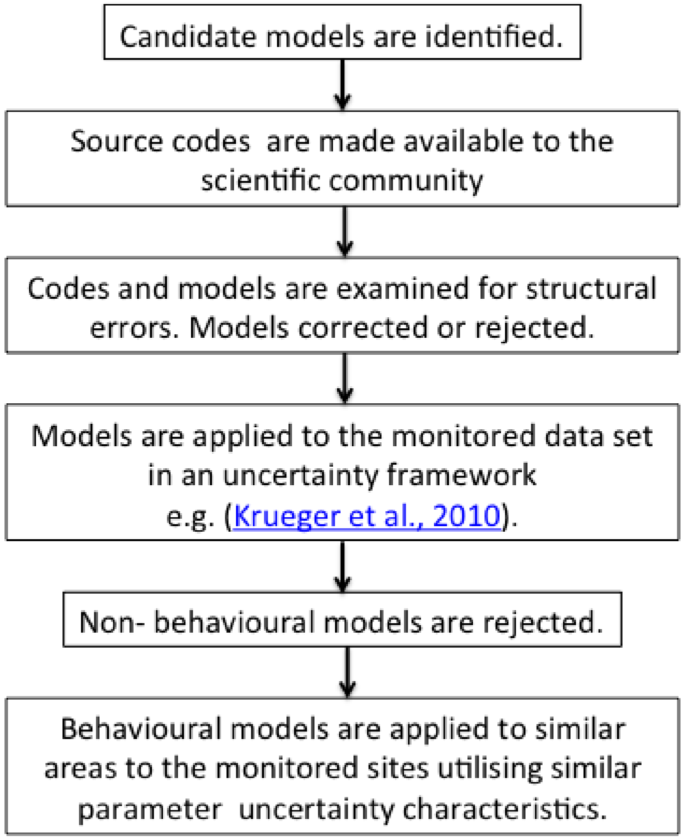

Figure 3.1 Proposed modelling framework

In this scheme, we do not propose the use of a single model; rather we propose the use of an ensemble of models, which are tested against the available data. This approach allows the rejection of those models that do not predict within the limits established by dialogue between modellers, monitors and policy colleagues. Behavioural models (those that pass the evaluation exercise) can then be used to extrapolate the monitored data. We suggest using more than one modelling approach since it is conceivable that different models will perform better in one environment than in others. For example, many of the models in Appendix 1, Table 7.1 were developed for arable agriculture and few have been evaluated for use in upland environments.

At the project workshop it became apparent that it may be necessary to zoom in on particular locations of concern. We suggest that the same strategy as presented above for the national scale modelling is adopted. However, it is likely that a different suite of models may be included in the ensemble, as different models (to the regional or national scale models discussed here) will be appropriate at finer spatial and temporal resolutions.

4. A national sampling strategy to monitor erosion in the UK

4.1 Introduction

In this section we propose an initial framework for monitoring soil erosion across England and Wales; specifically to monitor rates of actual soil erosion by all pathways. To design a statistically, scientifically robust and cost-effective framework, it is necessary to give broad consideration to various aspects of the sampling/monitoring design.

Prior to considering specific aspects of the monitoring network, it is necessary to decide whether it would be wise to estimate soil erosion rates across England and Wales as a whole, or within particular units of the landscape, perhaps defined by land use, soil type or climate, for example. Previous research on soil erosion across England and Wales suggests that land use or land cover type is likely to exert a dominant influence on erosion rates; for example, using ^{137}Cs as a tracer of soil erosion across 248 fields, it was shown that average erosion rates on cultivated land across England are substantially greater than for grassland (Defra, 2009). It is logical therefore to estimate mean erosion rates for broad land use categories across England and Wales. In the sections which follow we:

i) specify some of the features of the proposed monitoring framework according to a scheme established by de Gruijter et al. (2006),

ii) analyse existing data on the variation of soil erosion with time and consider its implications for design of the monitoring framework

iii) analyse existing data on the variation of soil erosion rate (cultivated soils) by soil texture class to assess whether soil texture should be included as a classification for identifying monitoring locations within each land use type

iv) undertake power analyses to estimate the number of monitoring locations required to detect a statistically significant change in soil erosion rate on cultivated land

v) summarise the outcomes from ii), iii) and iv) and propose how an initial set of soil erosion monitoring sites might be selected, and its flexibility

vi) consider the frequency with which sites should be monitored

4.2 Some features of the soil erosion monitoring framework

The basic design principles for natural resource monitoring are summarised well in the text *Sampling for Natural Resource Monitoring* by de Gruijter et al. (2006). We have selected from the authors' scheme 11 specific sections and subheadings which capture information relevant to the data acquisition, recording and processing which underpins the design of the monitoring network (see Box 4.1).

Box 4.1 – Eleven features of the proposed soil monitoring framework - adapted from de Gruijter et al. (2006).

1. **Detailed analysis and specification of the objective**

a) the target universe and the time over which it will be monitored: England and Wales; measured annually or bi-annually for at least 50 years
b) **the target domain:** the soil landscape for three land cover types: arable (A), improved grassland (IG) and upland plus semi natural (U+SN)
c) **target variable:** magnitude of soil erosion expressed as, for example, tonnes of soil ha^{-1} yr^{-1} for the three land cover types
d) **the target parameter:** mean and gross erosion rate by land cover class (short-term; first few years), changes over time intervals and over longer intervals (10 years), temporal trends in the erosion rate (more than ten years)
e) **target quantity:** i) mean (and gross) erosion rate, ii) change in the mean (and gross) rate, iii) trends in erosion rate, for A, IG and U+SN land cover types across England and Wales
f) **type of result:** quantitative estimate

2. **A quality measure of the monitoring –** the 95% confidence interval around the estimated mean change for each land use type

3. **Constraints –** budget for fieldwork, transport, and laboratory costs.

4. **Prior information to aid in the design of the survey:**
a) **sampling frame:** 1km grid of land cover classes (Land Cover Map 2000 or newer when available) and soil texture classes across England and Wales (Landis data ©Cranfield University). Digital Terrain Model (10 metre): derivation of landform indices (e.g. slope)
b) **existing measurements:** estimated quantities of soil erosion for seventeen transects (n=1227 field sites) across England and Wales based on a combination of air-photo interpretation and fieldwork (Evans, 2002). A subset of 93 sites had measurements in more than one calendar year.

5. **Sample support**[6] (s) – the sampling unit selected was either a field (because erosion processes relate to this land management unit) or 1 hectare in upland areas. In each case, erosion rates will be normalised to the hectare unit**.**

6. **Assessment method –** field and lab measurement procedures (see section 5 of the report and reviews in Appendix 1, chapters 1-5).

7. **Composite sampling?** Depends on the measurement technique (see section 5 of the report).

8. **Design-based or model-based inference:** Soil erosion is known to vary substantially over short spatial (and temporal) scales and so it is unlikely that the covariances between the observations based on the geographical coordinates of the sampling locations would be of practical use in model based (geostatistical)

[6] The size and shape (in three dimensions) of the volume of soil material which constitutes an individual specimen in a sample for which a single value is determined for any soil property. If referring to a composite sample in which individual samples have been combined (bulked), the support is defined by the description of the discrete samples (core shape, dimensions and depth), their number and the configuration over which they were collected (e.g. at the corners and centre of a square of specified dimensions).

inference. We therefore chose to pursue a design-based approach where inference is based on probability (random) sampling.

9. **Random sampling design type**: stratified random sampling – the strata are 3 aggregated land cover classes. We investigated whether the design should also include soil texture class (see below).

10. Identification of sampling sites: generated from maps of the stratified units for England and Wales, to include existing monitoring sites (Table 5.2)

11. **Method for statistical inference**: design-based estimate of mean change rate (domain and strata) single time point, design based estimate in differences between two times sets of observations separated by a time interval, design-based in space and model-based in time for detecting trends

By specifying the target domain as the soil landscape under three land cover types, we ensure that a wide range of soil landscapes will be monitored, not just those which might be considered most prone to erosion (coarser soil types under cultivation, for example). In the initial stages of erosion monitoring, it is possible that very low rates of erosion will be observed for the less erodible soil types and land use combinations, but it was our view that designating these areas *a priori* would be challenging and prone to error. It would also bias the national erosion strategy towards soils that were 'known' to erode. We therefore felt it was necessary to encompass the broad range of soil landscapes in the initial design, which could subsequently be modified based on the measurements in the early years of monitoring.

4.3 Implications of variation in soil erosion over time for the monitoring design

In undertaking soil monitoring, we need to consider whether there are advantages in sampling at fixed sites over two or more periods. For example, we can consider two variables (Z_1 and Z_2) as denoting measurements at time 1 and time 2 respectively. If the two variables are correlated, then there are likely to be advantages in sampling at fixed sites over the two periods, and determining the change directly at each sample site, then estimating the mean change from these observations (Lark, 2009). If, however, there is no correlation then the variables can be sampled independently with no disadvantage, and without the costs and logistic difficulties of maintaining and relocating fixed sites. In order to propose sampling procedures for erosion monitoring, we must therefore investigate whether measurements at particular locations appear to be correlated over time.

To address this question, we compared erosion estimates obtained on different dates from within the data described by Evans (2002). Although these data are drawn only from more erodible (cultivated) land use types, they were the only available data to investigate water erosion over time. N.B. as similar datasets do not exist for gross erosion from all pathways (water, wind, tillage and SLCH) we have not been able to perform this analysis for all erosion pathways (in the future should these datasets become available such analysis could be undertaken). Therefore, we assume that some element of repeat surveying will be proposed for non-visible erosion pathways, for example.

We identified 93 sites in the Evans (2002) dataset for which water erosion due to rilling or gullying had been measured on two or three occasions and generated a scatterplot showing the estimated rates on the first and last occasion for which erosion was estimated (see Figure 4.1). This shows no evidence for a relationship between erosion estimated on different dates. While we would expect inherent properties of the soil to influence erosion rate, and so to induce some correlation between measurement rates over time, the strong and non-linear dependence of erosion rate on particular weather events could plausibly produce such a relationship. This indicates that there is no expected advantage in sampling to monitor water erosion at fixed sites, and further analysis can assume independent sampling. However, one could still chose to sample at fixed sites, at least initially, so as to collect better data on the consistency of erosion rates within sites over time, for a wider range of soil and site conditions. Although there are no data for other land use types, we consider it is practical to assume that the same relationship holds for these classes too.

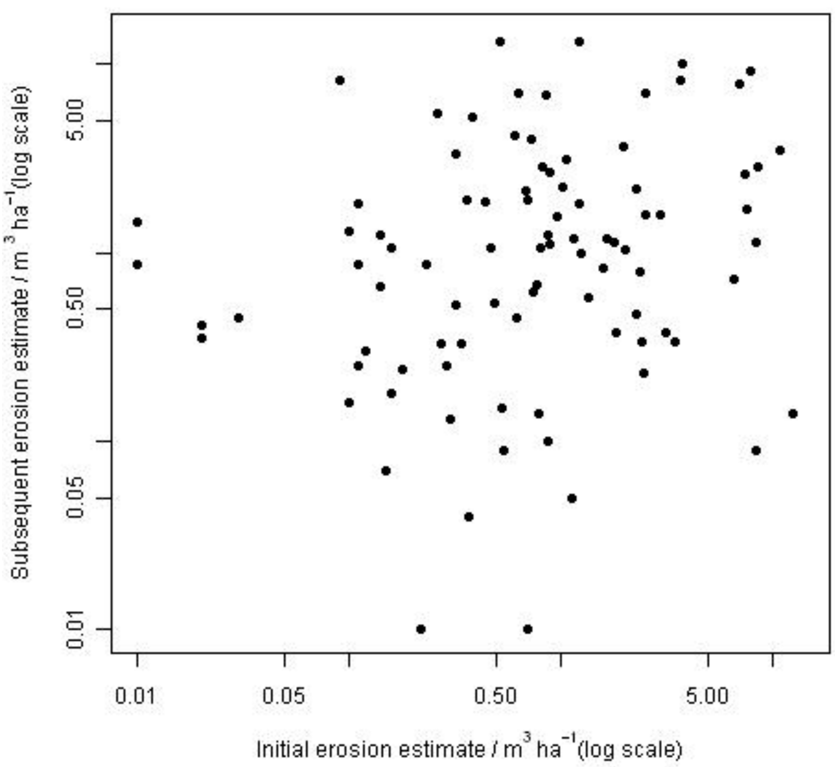

Figure 4.1 - Scatterplot of estimated erosion rates on different occasions (between 1982 and 1986) at the same sites (n=93) using data from Evans (2002)

4.4 Statistical analysis of the importance of soil texture for the monitoring framework

It is recognised that soil texture is a significant control on soil erosion by water. It may therefore be possible to improve the precision of estimates in the mean, or gross change in soil erosion rate by selecting sites within each land cover class based on stratifying them by soil texture class. To assess the evidence, we computed the mean of the estimated soil erosion rates for five different soil texture classes using data from Evans (2002) superimposed on the 1km soil texture data derived from the LandIs database (courtesy of Cranfield University). Table 4.1 shows that the sites surveyed in the study by Evans (2002) were biased towards the more erodible, coarser soil types of England and Wales. But there are sufficient data to make a comparison between the texture classes.

Table 4.1 – Proportion of soil texture classes for field sites surveyed in Evans (2002) and those across England and Wales (excluding organic soils).

Texture class (n sites)	Evans (2002): %	England and Wales (%)
Coarse	42	6
Medium	9	22
Medium-fine	38	54
Fine	11	18
Very fine	<0.05	1

Table 4.2 shows there are some differences in mean erosion rates between the different texture classes, but also that it may be possible to combine some classes with similar mean erosion rates for the purpose of investigating stratification for monitoring. For example, there is little difference between mean erosion rates in the coarse and medium texture classes, and there are so few data for very fine soils we considered it justified to combine this with the fine texture class. We propose a three-fold soil texture classification: CM (coarse and medium), MF (medium-fine) and FVF (fine and very fine).

Table 4.2 – Mean soil erosion rates by soil texture class using data from Evans (2002)

Texture class (n sites)	Mean erosion rate (m^3 ha^{-1})
Coarse (704)	2.9
Medium (150)	2.9
Medium-fine (633)	2.2
Fine (189)	1.4
Very fine (1)	0.5

The data from Evans (2002) could provide information on the variability of soil erosion rate, which is potentially useful for indicating likely sampling requirements for a national monitoring scheme. However, they are not obtained from a randomized sampling design, even for a domain of ‘erodible soils’, and the sampling on spatially-clustered transects will not provide unbiased estimates of, for example, the variance of erosion rate within soil texture classes. For this reason, we undertook a model-based analysis using the using the *likfit* procedure within the geoR library. R is a language and environment for statistical computing and graphics which is free to all end users and runs on any modern platform. The software was used to estimate a linear mixed model in which the fixed effects were (i) the overall mean value for the sampled domain or (ii) the mean values within the three texture classes defined above, within the sampled domain. The random effects comprised a correlated random variable and an uncorrelated error term. Variance parameters of the random effects were estimated using a Restricted Maximum Likelihood objective function (REML) using the *nlme* library in R. The *anova.lme* procedure was then used to test the null hypothesis that the mean erosion rates in the three texture classes were equal (given the model of correlated errors). The objective of this exercise was to ascertain what levels of variability exist within the best available dataset describing national soil erosion rates in England and Wales and what factors may control this variability.

The Evans (2002) data for soil erosion rate at individual sites (taking only the first observation from sites where multiple observations were made; n=1227) exhibit a strong positive skewness (skewness coefficient = 14.7). It is well known that estimates from strongly skewed variables are inefficient, and that the normality assumptions of REML and subsequent analyses require that such data are transformed. In this case transformation to natural logarithms reduced the overall skewness coefficient to -0.22. Table 4.3 shows the average values and sample variances for the transformed data, both considered as a whole and by the three broad texture classes.

Table 4.3 – Average and variances of $\log_e$ transformed soil erosion rates ($m^3\ ha^{-1}$) by all soils and for the three-fold soil texture classification.

Texture class	Average rate	Variance
All soils	− 0.31	2.65
CM	− 0.16	2.72
MF	− 0.37	2.73
FVF	− 0.68	1.98

The key parameters of interest to us are the *a priori* variance of the random effects under the two fixed effects models. This is the sum of the variances of the random effects (the sill variance in classical geostatistical terms), and the variance of a random observation (under the first model) or the within-stratum variance under the second model. The respective *a priori* variances were:

Model (i), quantity estimated σ^2 (*a priori* variance about the overall mean) estimate s^2=2.91. This is the expected variance of observations in a simple random sample.
Model (ii), quantity estimated σ_w^2 (*a priori* variance about the stratum mean) estimate

s_w^2=2.89. This is the expected within-stratum variance in stratified (by texture class) random sampling.

The null hypothesis that the three texture classes did not differ with respect to mean transformed erosion rate could not be rejected (p=0.454). For this reason we did not give further consideration to stratification by texture class, and subsequent power analyses are based on the first model fitted (with the overall mean the only fixed effect). Note that this variance is larger than the sample variances shown in Table 4.3. This is because the latter was biased downwards by the strong clustering in the data set. Note also that the reduction in the variance of the random effects by including texture class in model (ii) is very small. Even though the difference between the texture classes is statistically significant, they are very variable within the classes, and so are of little use as strata for purposes of sampling.

4.5 Power analyses to estimate the number of sites required to detect specified change in erosion rates

Power analysis is a statistical technique to answer the question of how large a sample size is needed for a particular task. Given the limited information available to us, we consider the task of estimating the mean difference in erosion rate between two dates, by independent, simple random sampling. Our data are transformed, so we define the task as detecting an underlying proportional change of κ in the geometric mean erosion rate, i.e. a change of $\log_e(1+\kappa)$ on the log-scale). We compute the power for a two-sample t-test specifying s^2 (simple random sampling) as the variance of the data. We specified a two-tail test (both increases and decreases in erosion are of interest), and a significance level of 0.05. The power is the probability that we would detect as significant (two-tailed test, significance level of 0.05) an underlying change of $\log_e(1+\kappa)$ on the log scale with a sample of specified size. The power analysis was undertaken with the STTEST procedure in GenStat (Payne, 2007).

More sophisticated power analyses (e.g. to detect trends from data in multiple years) should be computed once monitoring is underway, and used to refine the sampling design (perhaps by reducing the network size), but these initial results are indicative of the sampling effort that is likely to be needed.

Table 4.4 below shows the power results for the simple random sampling for different values of κ for a range of sample sizes. Figure 4.2 summarizes the results for simple random sampling.

Table 4.4 – Results of the power analyses to detect specified change; 0.1 is 10% and 1 is 100%.

	Power (p=0.05) to detect specified change				
	Simple Random Sampling				
Total Sample Size κ	**0.1**	**0.25**	**0.5**	**0.75**	**1**
50	0.05	0.1	0.22	0.38	0.53
100	0.06	0.15	0.4	0.65	0.83
150	0.07	0.21	0.55	0.82	0.94
200	0.08	0.26	0.67	0.91	0.98
250	0.09	0.32	0.77	0.96	1
300	0.1	0.37	0.84	0.98	1
350	0.11	0.42	0.89	0.99	1
400	0.12	0.46	0.93	1	1
500	0.14	0.55	0.97	1	1
600	0.16	0.63	0.99	1	1
700	0.18	0.7	0.99	1	1
800	0.2	0.75	1	1	1
900	0.22	0.8	1	1	1
1000	0.24	0.84	1	1	1

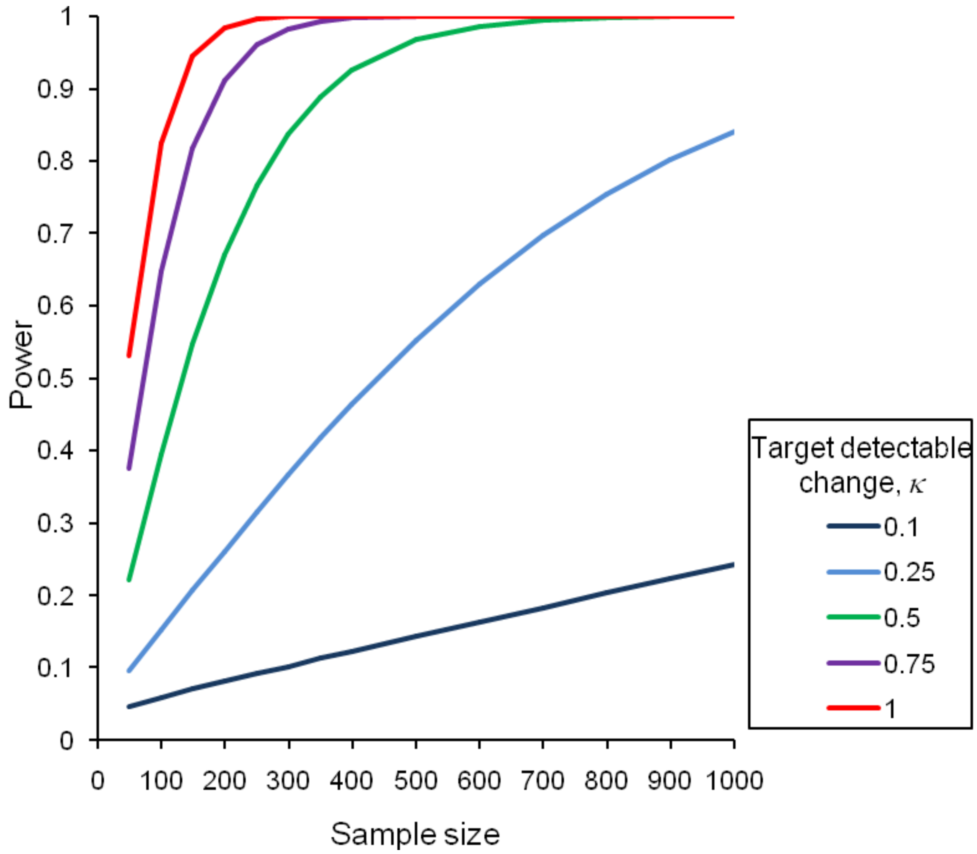

Figure 4.2 Power as a function of sample size for different values of κ□ simple random sampling.

A common target power is 80%. These results show that a total sample of 300 would be adequate to achieve this power to detect a 50% increase in erosion rate (two-date comparison), between 900 and 1000 samples would be needed to detect a 25% increase with the same power. A 10% increase is not detectable as a two-year comparison with 1000 samples, and clearly many more would be needed.

4.6 Summary of the proposed framework for a soil erosion-monitoring network and its potential flexibility

Our proposed monitoring framework is to estimate mean (or gross) soil erosion rates for three aggregate land cover types. The total area of these classes (dominant in each 1 km cell) shown in Figure 4.3 are: arable 61 177 km^2, improved grassland 43 218 km^2 and upland plus semi-natural 22 138 km^2 (total area: 126 533 km^2). In the first years of a national monitoring strategy, we suggest that 300 sites are chosen in each of these classes (900 sites in total). Quantitative data on both soil texture (from measurements) and relevant landscape indices (e.g. slope and log a/tanβ) should be used to in regression-based analysis of soil erosion rates (Brus, 2000). By including the latter data in the analyses it may be possible to discard some sites from the monitoring network without loss of precision.

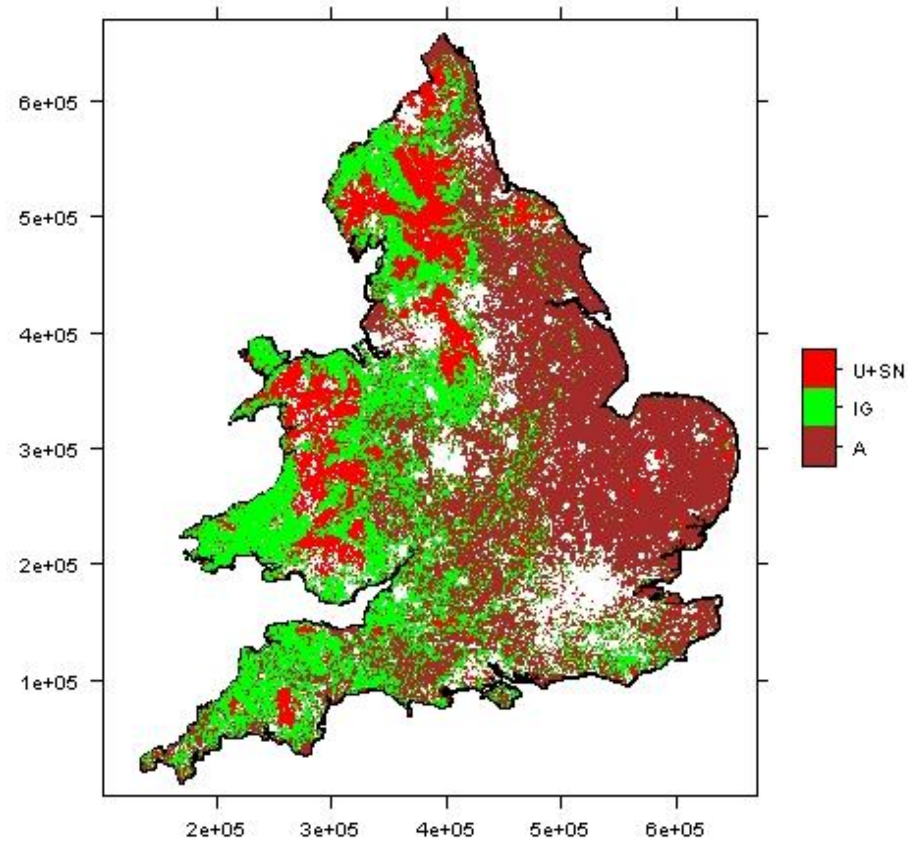

Figure 4.3 – Dominant aggregate land cover classes (A=arable, IG= improved grassland, U+SN= upland and semi-natural) on a 1km grid for England and Wales. Data from the Land Cover Map of Great Britain 2000, courtesy of the Centre for Ecology and Hydrology (© NERC).

4.7 Monitoring frequency

Climate – more specifically the combination of soil antecedent moisture conditions, rainfall intensity and duration – is a dominant driver of soil erosion by water. Its variation will lead to large differences in mean erosion rates between years for which measurements are made. There is currently insufficient information on the temporal variation of soil erosion to propose a sampling frequency; for example, annual or bi-annual. We suggest that data from the preliminary phase of an erosion monitoring programme – or data from a pilot study which is a precursor to such a programme – are used to determine an appropriate sampling frequency. It is likely that several years (at least five) of soil erosion monitoring – with associated climate measurements – will be needed to establish the typical range of erosive losses under prevailing climate conditions.

5. Evaluation of practical methods for monitoring soil erosion

The following section describes the proposed methods that might be used to monitor soil erosion at the national scale. Table 5.1 describes each potential method including details and limitations with an assessment of the costs for a range of spatial and temporal extents of monitoring and modelling. Imagery is costed at current rates, time is costed at approximate rates for technical staff involved on the basis of a FTE field technician cost of 25k per year, a FTA GIS/RS technician cost of 50k per year

5.1 Required attributes of a cost-effective monitoring scheme

A national monitoring framework (as per project scoping documentation), must deliver the following objectives:

1. Assess the spatial extent and severity of actual soil erosion across different erosion pathways, soil types and land uses at the 1 ha (≈100m × 100m) scale;
2. Identify how much soil is being eroded and where this is being deposited prior to entering fluvial or marine systems (and the implications for soil carbon losses and gains);
3. Be designed to facilitate routine cost-effective monitoring, combined with other techniques including modelling. The parameters affecting this needs to be explored in detail, and consideration should be given to the collection of data to populate any models;
4. Assess the implications of different frequencies of monitoring and issues of scaling up / interpolation / extrapolation and interpreting the results produced by the proposed framework/strategy;
5. Be designed to be flexible to be adapted to different financial constraints, whilst retaining scientific integrity, and to changes in the pattern and intensity of erosion as a result of climate change.

The following section describes how these objectives can be delivered. In addition, section 4 of this report has identified other characteristics that we propose as integral to a national scale monitoring capability.

6. Units = t ha^{-1} a^{-1}
7. Frequency = annually, or bi-annually
8. Sample size = 300 sites for each landcover type (i.e. Arable; Improved Grassland; Upland / semi-natural) = 900 sites
9. Locations – Random, stratified (by land cover; not soil type) sample. Monitoring (repeat sampling) at fixed sites preferred (but not essential)

Table 5.1. Cost effective methods of monitoring soil erosion at the national scale

Method		Proposed method(s), details and limitations	Costs for England and Wales
1. Remote sensing	**satellite data (e.g. Landsat, ASTER, IKONOS, Quickbird)**	Satellite imagery - Spectral signatures can be distorted (e.g. by cloud cover etc) and require calibration when used in time series analysis. - Spaceborne systems such as IKONOS and Quickbird can only deliver data when skies are cloud-free, which limits their effective response time to erosion events, though windows post-event can be used to assess volumetric changes. - Spatial / temporal resolution of satellite imagery is not always suited to soil erosion processes. - R&D still needed at present into some areas of image processing - Cost of buying imagery or acquiring data – can be £400 per scene for scale-appropriate spaceborne data (e.g. IKONOS) - Data processing of fine spatial resolution data provided by these systems requires the attention of a remote sensing expert, which carries further costs.	£360k for 900 sites per year £50k technical time per year
	airborne data, including aerial photograph y	- temporal & spatial resolution - revisit times can be user-defined - For most of the UK, data from air photo surveys are available at the finest zoom level (e.g. in Google Earth). - archive images may be available for sequential erosion monitoring (e.g. from commercial providers such as "Bluesky" or through the Environment Agency Geomatics Group or NERC) - costs may be £5K-£10K per acquisition for a commissioned capture of airborne data. Google Earth provides a GIS ready platform for delivery of data at a suitable spatial resolution for erosion presence / absence monitoring. There is potential to use this for national mapping of erosion features (rills, gullies, river bank erosion) - imagery is usually 2 dimensional (geometrically – n-dimensional if multiple wavelengths are considered as dimensions) areal extent of erosion, rather than volumetric measurement) - OK for progressive erosion, e.g. peat gullies, gradual channel migration; but not suited to arable landscapes where evidence is removed annually	- £0 for existing data via Google - £100k for 20 aerial photograph overflights covering 900 sites per year - £1M for LiDAR covering 900 sites per year

Method		Proposed method(s), details and limitations	Costs for England and Wales
	Ground based remote sensing (e.g. terrestrial laser scanning, digital cameras, differential GPS)	- temporal & spatial resolution - revisit times can be user-defined - R&D still needed at present. - costs of equipment, acquisition and processing are high – Terrestrial cloudscanner - large data sets generated, requiring substantial computational power for analysis	£75k one-off cost for cloud scanner or Diff GPS - £5k each Digital Camera £250 each £50k technical time for image analysis
	All remote sensing methods	- Significant technical leap needed to get from measurement of light to quantified erosion measurement - Optical methods alone are not likely to provide direct rates of soil loss (t ha^{-1} a^{-1}). - Training needs of operators / data processors / interpreters - Data costs can be high - Vegetation obscures erosion processes and features in some ecosystems - Spatial resolution and accuracy renders volumetric change calculations unreliable. - Trade off between resolution and spatial coverage (and handling of data sets generated) ✓ Terrestrial laser scanning at monitored sites which will offer the highest accuracy approach for detecting volumetric changes in soil as a result of erosion. ✓ LIDAR systems allow interpolation between different spatial scales (depending on ground vs air based system) e.g. plot to larger scales	

Method		Proposed method(s), details and limitations	Costs for England and Wales
2. Direct field survey methods	**Volumetric surveys (includes traditional surveying methods – tapes; Astroturf mats, etc.)**	✓ Estimates volume of rills and gullies and volume of readily identifiable deposits of sediment (water-borne alluvial fans and accumulated wind blown sediment) (t ha^{-1} yr^{-1}); river bank erosion ✓ Relatively quick, easy, cheap ✓ Provides a methodology capable of determining erosion across large spatial scales (Evans, 2002) ✓ Tried and tested technique (Harrod, 1998; McHugh, 2000; McHugh et al., 2002a) ✓ Could be undertaken by CSFOs in test catchments ✓Low equipment requirements. ✓ Could be supported by local, supplementary experiments e.g. rainfall simulation - No sheet or inter-rill erosion detected - No micro-rills routinely measured - No record of fragmented deposition (local deposition in furrows etc on slopes) - No tillage erosion detected - No record of nature of exported sediment unless source and deposits are characterised - Changes in volume of existing features only detectable after significant events. New features (rills, gullies) appear only after significant events – only detects features where significant soil movement has taken place. - Relatively high staff input, but no specialist knowledge / training needed. - the use of volumetric surveys to quantify grassland erosion is not appropriate (Brazier Appendix 1, Chapter 3), due to good vegetation cover obscuring entrainment and erosion).	900 person days per year for direct surveys £90k if dedicated staff making all measures
	^{137}Cs	✓ Proof of concept (Walling, 2008: SP0413) ✓ Data set already exists ✓ Quantifies soil loss that leaves no immediate visible trace ✓ includes all processes which actively redistribute soil; sheet, tillage, soil loss through crop harvest, wind, rill and gully erosion, as well as subsurface erosion. ✓ ^{137}Cs measurements across UK, combined with fallout map can provide national-scale erosion baseline that can be used to determine current soil status, inform soil 'lifespan' analysis and against which monitored rates can be assessed (process must be inferred from pattern or through modelling) (but with no indication of processes). ✓ Unlike other tracers, ^{137}Cs remains in the landscape ✓ Applicable to arable and pastoral landscapes on mineral soils; - Use in forward monitoring dependent on rates of erosion – repeat measurement after ca 5 years would be appropriate (as for other volumetric approaches used at field-scale) - Temporal average only– not very sensitive to soil losses over short time scales) - Caution needed if using in areas of high Chernobyl fallout (identifiable in advance by comparison with mapped fallout; or after the event by comparison of inventory with known bomb-fallout distribution) - Need suitable undisturbed reference site (baseline) close to the area of interest	£360k for analysis costs over all 900 sites (10 samples at each site, £40 per sample) £25k technician time for one year of field work

Method		Proposed method(s), details and limitations	Costs for England and Wales
	Artifically applied/installed tracers (e.g. rare earths)	✓ Quantifies soil loss that leaves no immediate visible trace ✓ Can be used to *monitor* soil loss due to erosion processes that leave no immediate visible trace (inter-rill erosion; tillage erosion; wind erosion) ✓ Can be used for repeat monitoring after initial installation ✓ Low-cost/moderate-skill sampling required after initial installation ✓ Applicable at any site where mineral soil is regularly perturbed - Cost of initial installation and subsequent analysis - Installation of tracers requires soil perturbation – inappropriate for undisturbed grassland/upland etc.	Analysis ca. £10 per sample (for rare earths), 900 x 10 samples = £90k
	Erosion pins	✓ Erosion pins are the most widely used approach to on-site erosion monitoring in the uplands (Evans and Warburton, 2007). ✓ measure erosion and depostiion ✓ Provide simple direct measurement of surface change ✓ Practical - arrays of erosion pins are cheaply and easily installed and quick to measure (one array of pins can be measured in 10 minutes). ✓Annual resurvey is usually required - only appropriate for semi natural sites where there is no agricultural ground disturbance. - although they provide good data on severity, they are point measurements which would require a careful sampling design to provide extent data. - Sensitive to changes in surface elevation that are not related to mass loss – e.g. change in volume with change in moisture content; surface super-elevation due to ice formation, etc – these likely to be most significant in uplands and semi-natural	Each array would cost ca. £100 to set-up Annual resurvey ca. £9k for all 900 sites
3. Instrumented catchments **e.g. turbidity sensors**		✓ Possibility of adding turbidity gauges to existing EA monitoring stations (presently ≈ 50 sites) - practical, but costly in instrumentation and data analysis. ✓ Could be used to corroborate on-site erosion data. ✓ allows extrapolation of chosen spatial survey scale (≈ 1ha) to catchment scale ✓ when combined with volumetric surveys, allows straightforward calculation of sediment delivery ratios i.e. the proportion of on-site erosion which is deposited before reaching water courses, If whole catchment is monitored for on-site erosion - Strong evidence that sediment delivery ratios (re total soil redistribution) are small – i.e. comparison between observed on-site erosion rates and sediment yields; magnitude of colluvial deposits – therefore, poor indicator of on-site erosion and extreme caution required in use for extrapolation - no direct monitoring of erosion rates on-site. - Large dataset to process if single events are monitored. - Costs of instrumentation (equipment), data analysis. Catchment monitoring is relatively expensive, because in contrast to annual survey, it requires perhaps monthly visits to maintain equipment.	New sites ca. £10k for set up, i.e. £900k for all 900 sites £0 cost for any existing sites

Method		Proposed method(s), details and limitations	Costs for England and Wales
4. Sediment-ological evidence e.g. reservoir surveys, flood plain sediment sequences		✓ Could be used to corroborate on-site erosion data, but only in the long term. - no direct monitoring of erosion rates on-site. - errors - trap efficiencies, - long term, average rates only – policy impacts not captured, although targets could be set for long term rates. - Reservoir survey is labour intensive	Not assessed as not relevant for many of the 900 sites

We propose a four-part strategy for cost-effective monitoring of soil erosion in England and Wales. The costs of each part of the strategy are detailed below, with reference to the costs included in Table 5.1. These options involve a range of sampling intensities and therefore describe soil erosion with differing levels of associated uncertainty. The sampling intensity will be controlled by the available budget to conduct the erosion monitoring. Reducing uncertainty of our national scale understanding of erosion will require high sampling intensities and therefore greater costs. This approach does not exclude the monitoring of any different erosion pathways, as we firmly believe that no erosion processes can be ignored until they have been quantified. Rather, the approach defines what needs to be monitored and outlines the different options for doing this.

5.2 Framework for cost-effective monitoring

We propose a framework for cost-effective monitoring of soil erosion rates that can be applied at the national-scale. It must be seen in conjunction with the sampling design outlined in section 4 and the modelling strategy described in section 6.

5.3 Part 1. Establish baseline spatial extent and severity of actual soil erosion

Future monitored rates of erosion will be more meaningful if placed in the context of current/recent erosion rates. This will certainly be necessary if monitored rates are to be used to evaluate the effect of policies/strategies introduced to control erosion. The baseline will be established through 3 procedures:

a. Assemble all existing observed data from across England and Wales, as reviewed in Appendix 1, and integrate in a GIS framework. These data will be useful both to provide a benchmark for erosion rates in England and Wales and also to support the initial modelling exercise described in section 5.6. **COST – One year of GIS technical time - £50k**

b. Undertake mapping based on existing aerial photographic imagery described in Table 5.1 (e.g. air photo archive data provided by BlueSky, or data available freely through Google Earth for example). 900 sample sites will be selected (as described in section 4) and archive imagery interrogated to obtain a snapshot of rill and gully erosion in the available imagery. This approach will provide a statistically-robust description of the presence / absence and spatial extent of rill and gully erosion in England and Wales. **COST = £0 for imagery, one year of RS technical time - £50k**

c. Undertake ^{137}Cs survey of 1/n of 900 sites to establish half-century erosion rates associated with all erosion processes (water, tillage, SLCH, wind). The magnitude of n is dependent on the level of investment available. These data will describe the severity of all erosion types in England and Wales. **COST (for all 900 sites, based on £400 per site, and ca. 1 person-day per 3 site) = £385k**

Total cost of Part 1, to establish baseline spatial extent and severity of actual soil erosion across 900 sites in England and Wales = <u>£485k based on ca. one year of work.</u>

5.4 Part 2. Conduct routine, cost-effective monitoring of frequency, spatial extent and rates of erosion

5.4.1 Annual monitoring

At 1/n of the 900 sites a framework for detailed on-site measurements of erosion will be established. The magnitude of n is dependent on the level of investment available. Costs modelled here for 900 sites.

a. Rill and Gully erosion associated with visible evidence:

   i. Using ground-based volumetric survey techniques described in Table 5.1 above. **COST = (0.5 person day per site i.e. 450 field days and 0.25 person days per site for analysis i.e. 225 days at £100/day) = £67,500 per year**

   ii. Terrestrial laser scanning and novel photogrammetric techniques will also be explored alongside these approaches for monitoring fine scale variations in soil structure and movement pre- and post-erosion. **(0.5 person day per site i.e. 450 field days and 0.25 person days per site for analysis i.e. 225 days at £150/day) = £101,250 per year**

   iii. Airborne surveys, including aerial photography and LiDAR will be used when data capture is coincident with ground surveys to build a picture of compatibility and information content of these data through time. **COST = £100k for aerial photography commissions, £1M for LiDAR overflights for all 900 sites = £1.1M per year**

b. Sediment yield: Where sites within the 900 locations are common to existing erosion monitoring at field and headwater catchment scales, construction of annual sediment budgets will be used to assess sediment yields. This scale of observation will provide a link between on-site erosion (quantified above) and larger-scale catchment monitoring. A list of potential sites that these observations could be made at are detailed in Table 5.2. **COST = £75k for 10 days work per year at existing 70 sites detailed in Table 5.2.**

c. Wind erosion traps, to quantify annual wind erosion rates. **COST = £100 for set-up i.e. £9k and 1 hour per site for annual rate measurement, i.e. 50 field technician days = £14k (assuming site visits are made for at least a(i) above.**

Table 5.2 Potential catchment monitoring sites for collaboration

Monitoring Scheme	Sites
ECN freshwater sites	29 sites
CEH carbon catchments	4 sites
Acid Water Monitoring Network Sites	11 stream sites
North Wyke Farm Platform	15 field-scale sites, 1-100ha
CSF enhanced monitoring catchments	10 sites
Demonstration Test Catchments	3 sites, numerous scales

Total cost of Part 2, annual monitoring across 900 sites in England and Wales = <u>£1,357,750 per year of monitoring.</u>

5.4.2 Part 3a. Monitoring on a 5-year cycle

Erosion pathways that leave no visible trace (Tillage erosion, SLCH) are omitted from (a) above and processes that do not lead to sediment export via the fluvial system are omitted from (a) and (b) above. There is, therefore, a need for separate monitoring of the magnitude of these processes, if all pathways of erosion are to be quantified. We argue herein, that this range of monitoring is critical, with recourse to Table 5.3, which describes significant erosion rates from all major soil erosion

Table 5.3 Comparison of soil loss for different erosion pathways in England and Wales (from Owens et al., 2006).

	Wind	**Tillage**	**Co-extraction with root vegetables and farm machinery**	**Water**
Typical erosion rate range (t ha^{-1} a^{-1})	0.1 – 2.0	0.1 – 10.0	0.1 – 5.0	0.1 – 15.0
Land use affected	Arable, upland, some pasture	Arable	Arable	Arable, pasture, upland
Exported off field	Yes	No	Yes	Yes

pathways identified in England and Wales.

The most appropriate methods to 'identify how much soil is being eroded and where this is being deposited prior to entering fluvial systems' for all erosion pathways are those that deliver data regarding net soil loss. This will be accomplished on a five year basis, in addition to the annual monitoring described above by:

d. Revisiting the sites sampled in Part 1 c above and re-measurement of ^{137}Cs

COST (for all 900 sites, based on £400 per site, and ca. 1 person-day per 3 site) = £385k

e. At the same sites, on first visit artificial tracers (rare earths or otherwise absent radionuclides) will be used to label plots at selected slope locations; on subsequent visits these will be re-sampled to derive rates of tracer loss and establish net rates of soil loss.

COST (for all 900 sites, based on 10 samples per site, £10 per sample) = £90k

f. Interpretation of data from a-e informed by annual monitoring to partition net loss between processes monitored annually and those that leave no visible trace (tillage erosion and SLCH).

COST across all sites is equivalent to a senior researcher FTE i.e. 70k per year for 5 years = £350k

Total cost of Part 3a, Monitoring across 900 sites in England and Wales on a 5 year cycle = £825k

5.5 Part 3b. Supplementary data collection to enhance modelling capacity at the national scale.

No matter what intensity of sampling is established it will be necessary to interpolate using models. Even where all 900 sampling locations are monitored using each applicable technique above, thus quantifying all erosion pathways at each site, there must still be efforts to 'fill in the gaps' of our spatio-temporal understanding of erosion. Indeed, the sampling strategy described in section 4 is designed specifically to allow this interpolation work and therefore deliver a national-scale assessment of erosion.

Datasets available for model evaluation and development should be collated in a national-scale GIS database as described in Part 1 a. However, our review of the data in Appendix 1 illustrates that data are most limited with respect to the non-visible pathways of erosion (tillage erosion and soil loss with crop harvest). Additional data are that will extend these datasets to permit reliable interpolation are:

Tillage Erosion: at 1/n (900 in this example) sites conduct tillage erosion measurements on a range of representative hillslopes using established methods (metal tracers buried and relocated). These permit derivation of soil transport coefficients for soil-implement combinations representative of UK arable land. These soil transport coefficients are essential for tillage erosion modelling (for interpolation and separation of process roles)

COST After initial installation (1 person day per site), additional site visit after 5 years i.e. 2 person-days per site or 1800 days = £125k field technician time

SLCH: at 1/n sites (900) monitor SLCH using established methods; analyse with respect to putative causal parameters, e.g. soil moisture, soil texture. Supplement these data with assembly of secondary data from root crop industry.

COST requires involvement on-site of farmers in liaison with field technician to quantify transfer of soil on vehicles or crops at times of harvest. 1 person day per site of field technician time, 900 days = £90k

Total cost of Part 3b, Monitoring SLCH across 900 sites in England and Wales on a 5 year cycle = £215k

5.6 Part 4. National modelling strategy

As outlined in section 4, a modelling strategy is most useful to interpolate between the 900 monitoring locations as it is not possible to monitor across England and Wales. The proposed modelling solution should employ a range of models, in an ensemble approach, to evaluate which models are best at modelling the range of different erosion pathways at each of the 900 sites. The modelling work will therefore be iterative, in dialogue with the national-scale monitoring and we propose that it will involve the following steps:

a. Collation of existing erosion models into an open source format, to enable all models to be run from the same platform. The models detailed in Appendix 1, Table 7.1 are all potential candidates for this exercise. No model will be favoured, until evaluation has been undertaken to assess which model(s) is(are) best. However, it is likely that the less parameter intensive models will be easier to apply with confidence at the national scale.

COST = 1 year of computer technician time = £50k

b. Model coding and debugging exercise against existing datasets collated through national GIS database (Part 1 a above)

COST = 0.5 year of computer technician time = £25k

c. Collation of national datasets (NSI, MAGPIE, Landcover 2000 maps) to support parameterisation of models

COST = 0.5 year of computer technician time = £25k

d. Initial model runs at the 1 ha scale, for all 900 sampling locations, evaluating model performance against:
    i. Spatial extent of visible erosion (from Part 1 b)
    ii. Total net erosion (from Part 1 c)

within an uncertainty framework. This initial modelling will identify which model(s) perform best across England and Wales and what the associated range of uncertainty around soil erosion predictions will be.

COST = 0.5 year of computer technician time = £25k

e. Behavioural (or best) models will then be retained to apply to data that are collected on an annual basis (Part 2), across the 900 sites, on an annual basis.

COST = 0.25 year of computer technician time = £12.5k

As annual monitoring data will also include; volumetric survey data (Part 2 a), sediment yield data (Part 2 b) and wind erosion data (Part 2 c), confidence will be high that model predictions of 'natural' erosion pathways are well captured.

f. The best, least uncertain models from each year will then be used to interpolate erosion rates and extents, for all visible pathways, at the 1 ha scale across England and Wales.
g. The non-visible (anthropogenic) pathways of erosion will be modelled initially against the data collected in Part 1 c, to evaluate whether the difference between net erosion (from ^{137}Cs data) and the visible erosion pathways data can be explained via modelling of Tillage erosion and SLCH. This modelling will also be undertaken using an ensemble of models, within an uncertainty framework
h. Those tillage erosion models (for example) that perform best, will be used to interpolate results across England and Wales.
i. As more data describing non-visible erosion become available (Part 2 d-f) on a five-year basis, further modelling will be undertaken to evaluate whether improved empirical knowledge has led to constraint of non-visible erosion prediction.

j. Again, the best models of tillage erosion and SLCH from this evaluation will be retained and applied to interpolate results across England and Wales.

Total cost of Part 4, National modelling strategy = <u>£137.5k for first year, £37.5k</u> per year thereafter

5.7 Conclusions

The modelling strategy outlined here is designed to be updated, as and when monitored data become available to improve model predictions. It can also be used to evaluate the quality of predictions when new models become available. If funds are limiting, this strategy will permit the prediction of erosion from all pathways across England and Wales, based on existing data, providing the low cost solution. However, it is noted that the addition of monitored data on an annual (suggested as the ideal scenario) or five-yearly basis (suggested as a reasonable compromise) will greatly enhance the predictive capability of erosion models, particularly if all erosion pathways, including non-visible erosion pathways are to be understood on a national scale.

6. A pilot project to support a national erosion-monitoring framework

A national-scale rollout of the proposed framework will be costly and will also carry associated risks with it. These risks include:

1. Use of remote-sensing data such as BlueSky imagery to define erosion extent has not been undertaken in an automated fashion for such large areas.

2. Combined surveys using RS imagery and ground-based survey have worked previously (Evans soil survey work), but new technologies such as terrestrial laser scanning, or photogrammetry have not been tested fully.

3. Combining observational evidence (such as ^{137}Cs data) of erosion pathways to describe total erosion, with re-surveys of such data has only been attempted at a limited number of research sites.

4. Model evaluation across such a large number of sites, using an ensemble approach which considers uncertainty, will be computationally demanding, so will require time to develop the platform, prior to use at the national scale.

Therefore, it is pragmatic to propose a pilot-study which trials the range of techniques proposed here on a limited number of sites, to evaluate the feasibility of a national-scale application of the suite of techniques, as well as to assess the true cost-effectiveness of the four-part strategy outlined above.

6.1 Pilot project structure

We propose a 1.5+3.5 year structure, which underpins the full, national scale work, with an intensive 18 months of research, to evaluate the full project feasibility. Ideally, the full project would then continue for 3.5 years to cover a 5-year cycle during which the full range of techniques described above would be delivered.

6.1.1 Pilot project workpackages (WP's)

Work package 1.1. Part 1 a of the strategy (described above in section 5), which seeks to collate all information describing soil erosion in England and Wales into a GIS database is a one year task for a FTE member of staff with the relevant GIS/RS skills and soil erosion understanding. **COST = £50k.**

Work package 1.2. Whilst sampling 900 locations for evidence of erosion is a significant task, it would be feasible to take a 10% sample across the three identified land uses, to trial Part 1b of the proposed strategy. This work could run parallel to Part 1 a, and would require specific expertise in the manipulation of archived, remotely-sensed data to build the first unbiased 'snapshot' of erosion extent in England and Wales. This work would also take three months to deliver for a FTE member of staff. **COST = £12.5k**

Work package 1.3. Part 1 c could also be applied to the 90 sampling locations. The risks associated with this work are low, as the approach has been applied previously, though not in an un-biased strategy as suggested here. N.B. Previous data from Defra SP0413 might be incorporated into this pilot-scheme, provided they were sampled to represent the required absence of bias. This work would take one year for a FTE, plus laboratory and field technical support. **COST = £50k**

Annual monitoring will be possible for the first year, during the pilot project period at the 90 sites identified. Ideally, these sites would correspond with as many existing monitoring sites in England and Wales as possible (retaining unbiased sampling strategy), to support the collection of 1 ha scale data with continuous monitoring of sediment yields at the headwater catchment scale. **COST = £20k**

Work package 2.1. Part 2 a i, would be the minimum necessary for "on the ground" surveying of the 90 locations, to correspond with the air photo image analysis described in WP1.2. This work would enable the quality of the data to be assessed in terms of its ability to quantify true volumetric erosion, as well as providing an actual measurement of erosion. Such work would involve site visits, likely to involve one day at each site (i.e. 90 days work for a FTE field technician). Additional funds would permit the use of techniques described in Part 2 a ii and iii, namely, terrestrial laser scanning and novel photogrammetric techniques, as well as directed LiDAR overflights to the 90 sites. Whilst these approaches would have a cost implication in terms of data (see Table 5.1 for airborne LiDAR costs) and processing time, there would be no extra cost in terms of person-hours at each of the field locations. The extension of simple on-the-ground surveys to include these more novel techniques is therefore suggested, as it would lead to more automated approaches for erosion monitoring in the future.

COST = ca. £100k for LiDAR overflights and ca. £20k for technical support for data processing

Work package 2.2. For those sites that corresponded with existing erosion monitoring experiments, data could be worked up for the pilot-project first year, describing sediment yields at the headwater catchment scale, to add context to the 1 ha data collected in the previous work packages. Such work is necessary, particularly in upland areas, where erosion rates are difficult to determine using the previously discussed techniques. These data would further add to the GIS database of erosion collated in WP1.1. As most existing monitoring sites collect information describing sediment yields already, this WP is largely seen as a desk-based exercise to collate existing information.

COST = £0 if existing sites are used.

If existing sites do not routinely monitor sediment yields, we would propose the addition of equipment to facilitate this monitoring, at a number of these sites, in close collaboration with researchers at each site. This work would take one year (of monitoring), but would essentially involve three months of data collation after the first year of the pilot project for a FTE.

COST = £10k for each additional site that is set up.

Work package 2.3. The installation of wind erosion traps (Part 2 c) to capture annual wind erosion, could be undertaken at the same time as WP2.1, at each of the 90 sites. Again after one year, data would need to be worked up, as in WP2.2, to synthesise the response of each site in terms of wind blown erosion/deposition (One month for a FTE).

COST = £2k assuming set up and one year of monitoring

Though desirable, the longer-term work proposed in Parts 2 d and f, is not feasible within a pilot project. However, it would be feasible to carry out the installation of tracers, Part 2 e, to support work, which improved understanding of net soil erosion rates for the national-scale project, over the full 5 year period, at the 90 sites (ca. 90 days work for a FTE).

Work Package 3.1 Tillage erosion and SLCH monitoring

Appendix 1, section 4.2.1 describes the single-pass tillage erosion experiments that have been used to establish soil translocation rates via tillage erosion. During the pilot-project period, whilst it would not be feasible to establish long-term, multiple-pass tillage erosion rates at all arable sites (within the 90 locations). It would be possible to use tracers to estimate tillage erosion rates over the first year of the project. Such information would not only prove the concept of this approach, but would also underpin the longer-term monitoring as well as provide better parameterisation data for the tillage erosion models that would be developed.

For each arable field, or field where harvest led to mechanical erosion, total soil loss via the SLCH pathway could also be assessed during the first project year. These data would again prove the concept of SLCH monitoring for a sub-sample of the 90 field locations, as is discussed in Appendix 1, section 4.3.1, but would also provide better data for parameterisation of any national modelling approaches, as outlined in Appendix 1, section 4.3.2.

COST = £21.5k

Work package 4.1 Model and data assimilation

Significant work is needed to bring existing erosion models into an open-source format, to run on existing national scale datasets. The Exeter workshop highlighted that this work, however, is essential in moving the hitherto piecemeal approaches to modelling erosion in England and Wales forward. This WP would focus on the assimilation of all available erosion models, including those developed in previous Defra R and D projects, 'off-the-shelf' models and research-based models into a common format. Parallel with this work, existing national datasets that could be used to parameterise these models would be drawn together in the same national, GIS database that held the existing erosion monitoring data (from WP 1.1). This WP would therefore deliver the objectives outlined in Part 4 a-c, over the first year of the project (1FTE).

COST = £100k

Work package 4.2 Initial model runs

Initial model runs at the 1 ha scale, could be carried out for the 90 sampling locations, evaluating model performance against:

- Spatial extent of visible erosion (from WP1.2)
- Total net erosion (from WP1.3)

This work would be carried out during the last six months of the pilot-project, to take advantage of the previous data collection described above and to allow conclusions to be drawn about how the modelling work would need to progress for the remaining 3.5 years of the project.

As the modelling would be carried out in an uncertainty framework that allowed quantification of which models were 'best' at predicting spatial extent of erosion and total net erosion, it would then be possible to execute a second set of model runs that could be evaluated against the first year of annual monitoring data. Whilst this work would not actually test models at the national scale, it would provide a framework to support this, once the full data collection was feasible across all 900 sites.

COST = £5k

<u>**Total cost of 1.5 year pilot project = £391k**</u>

7. Summary

The original research requirements for this project were to:

- Review available literature on soil erosion monitoring and modelling
    - Focus of this review should be on England and Wales but can include relevant international literature where appropriate;
    - Analysis of the literature should present the information into categories to include (but not necessarily limited to):
        - Erosion pathways
        - Soil Types
        - Land Use;
    - Review differing monitoring methodology, including the use of Earth Observation techniques, and assess their cost-effectiveness (e.g. Cost Benefit Analysis); and
    - Review modelling techniques and how well the outputs compare to actual measurements of erosion.

Section 3 summarises the findings of Appendix 1, which details all erosion pathways in the UK and reviews all techniques that have previously been used to monitor and model erosion. The Appendix is divided into:

- Upland soil erosion
- Arable land soil erosion
- Lowland grassland soil erosion
- Mechanical erosion
- Channel-bank and floodplain erosion
- National erosion assessments
- Soil erosion modelling

Each chapter illustrates which techniques have been used to assess the extent and rates of erosion from different pathways and is intended to serve as a detailed reference document, in support of the proposed strategies.

The second part of the project was tailored to:

- Design a scientifically robust cost-effective monitoring framework for England and Wales:
    - To assess the spatial extent and severity of *actual* soil erosion across different erosion pathways, soil types and land uses;
    - To identify how much soil is being eroded and where this is being deposited prior to entering fluvial or marine systems (and the implications for soil carbon losses and gains);
    - The framework/strategy needs to be designed to facilitate routine cost-effective monitoring and could be made up of a combination of different techniques including modelling. The parameters affecting this needs to be

explored in detail, and consideration should be given to the availability of data to populate any models;

- o Assess the implications of different frequencies of monitoring and issues of scaling up and interpreting the results produced by the proposed framework/strategy; and
- o This framework will need to be designed to be flexible to be adapted to different financial constraints, whilst retaining scientific integrity, and to changes in the pattern and intensity of erosion as a result of climate change.

A four-part strategy is proposed to deliver a cost-effective framework to monitor (and model) soil erosion in England and Wales. Section 4 details a national sampling strategy to monitor erosion in the UK, with an emphasis on constructing a statistically sound approach to locating sampling sites for soil erosion monitoring. It is well noted here, that whilst there has been a great deal of erosion monitoring in the UK historically, it has not been placed within an unbiased sampling framework. We suggest that the proposed sampling strategy is used to underpin a national erosion monitoring capability, whilst also making best use of existing sites that either monitor, or could be adapted to monitor, erosion in the future.

Section 5 brings together the wide variety of erosion monitoring and modelling techniques that are available. We assess cost of each technique and provide an assessment of the pros and cons of each technique. We then outline a suggested strategy for the cost-effective monitoring (and modelling) of soil erosion:

1. Establish baseline spatial extent and severity of actual soil erosion

2. Conduct routine, cost-effective monitoring of frequency, spatial extent and rates of erosion at both annual and five yearly timesteps

3. Supplementary data collection to enhance modelling capacity at the national scale.

4. National modelling strategy

The level of monitoring or modelling that is included within each part of the national strategy is dependant upon funding available. Therefore, Table 7.1 below describes a range of suggested options, from a basic baseline characterisation of erosion extent and severity for England and Wales, through to a full monitoring and modelling strategy.

Table 7.1 A summary of cost options for a national soil erosion monitoring framework for England and Wales

Option	**Cost (900 sites)**	**Cost (600 sites)**	**Cost (300 sites)**
1. Baseline erosion survey (1a and 1b)	£100k	£84k	£67k
2. Baseline erosion survey (1a, 1b and 1c)	£485k	£341k	£196k
3. Annual erosion monitoring (2a i)	£67.5k	£45k	£23k
4. Annual erosion monitoring (2a i and ii)	£168.75k	£112.5k	£56.4k
5. Annual erosion	£1,268k	£845.3k	£423k

monitoring (2a i-iii)			
6. Annual erosion monitoring (2a and b)*	£1,358k	£898.9k	£455k
7. Annual erosion monitoring (2a,b and c)	£1,372k	£908.2k	£459.7k
8. Five year SLCH monitoring (3d)	£385k	£256.7k	£128.4k
9. Five year SLCH monitoring (3d and e)	£475k	£316.7k	£158.4k
10. Five year SLCH monitoring (3d, e and f)	£825k	£550k	£275k
11. Supplementary 5 yearly SLCH data collection in support of modelling	£215k	£143.4k	£71.7k
12. National erosion modelling against year one monitoring data (4a-d)	£125k	£100k	£75k
13. Annual repeat modelling	£12.5k	£8.4k	£4.2k

*assumes that 70, 50 and 30 sites are monitored for sediment yield in collaboration with existing projects.

It is clear that establishing a baseline is critical to underpin a national erosion capability and as such the basic cost of between £196 and £485k (300 – 900 sites) represents a minimum cost for this understanding. Subsequent routine monitoring is also fundamental, at the appropriate number of locations (again 900 being statistically sound), but the range of techniques applied to each location can be modified depending on either funds available, or the specific pathway of erosion that is of interest. Therefore, costs are estimated to range from £67.5 to £1,372k per year. Supplementary data collection is important to improve parameterisation of models, particularly of non-visible erosion, though is not vital to the delivery of national water erosion estimates. The costs here are therefore the most flexible, (£385 – £825k for 900 sites) Finally, the national modelling strategy is essential to provide interpolation between monitoring locations. Minimum suggested costs here are £75 – £125k, for one year of modelling work. It is also critical to support predictions of how erosion may change, in light of climate or land use changes. As the quality of model predictions are directly related to the quality of empirical understanding, it is highly likely that models, evaluated against the proposed national erosion data, will improve through time. The application of models to the evolving dataset is therefore seen as prudent strategy for the future, with annual repeat modelling against observed data providing a cost-effective option (£12.5k for 900 sites).

Finally, in order to demonstrate the feasibility of the proposed approach(es), we suggest a pilot project structure. This pilot will apply the strategy at a limited number of sites (ca. 10% of the national locations), over a 1.5 year period. Within this period, conclusions could be drawn as to how (cost) effective the approach is, prior to the proposed implementation of a further 3.5 years of monitoring/modelling, to deliver a medium-term understanding of national-scale soil erosion in England and Wales. The cost of the pilot project is estimated at £391k.

8. References

Baghdadi, N., S. Gaultier, and C. King, 2002. Retrieving surface roughness and soil moisture from synthetic aperture radar (SAR) data using neural networks. Canadian Journal of Remote Sensing, **28**(5): p. 701-711.

Ben-Dor, E., et al., 2002. Mapping of several soil properties using DAIS-7915 hyperspectral scanner data - a case study over clayey soils in Israel. International Journal of Remote Sensing, **23**(6): p. 1043-1062.

Bilotta, G.S. 2008. The mobilisation and transport of sediments colloids andphosphorus from intensively-managed, temperate grasslands. Unpublished PhD thesis, Department of Geography, University of Exeter.

Bilotta, G.S., Brazier, R.E. 2008. Understanding the Influence of Suspended Solids on Water Quality and Aquatic Biota. Water Research doi:10.1016/j.watres.2008.03.018

Bilotta, G.S., Brazier, R.E., Haygarth, P. M., Macleod, C. J. A., Butler, P., Granger, S., Krueger T., Freer, J. and Quinton, J. 2008. Rethinking the Contribution of Drained and Undrained Grasslands to Sediment Related Water Quality Problems. Journal of Environmental Quality 37, 906-914. Doi:10.2134/jeq2007.0457

Bilotta, G.S., Brazier, R.E. and Haygarth, P.M. 2007. The impact of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. Advances in Agronomy, 94.

Bilotta G.S., Krueger T., Brazier R.E., Butler P., Freer J., Hawkins J., Macleod C.J.A., Haygarth, P.M., Quinton J. 2009. Assessing Catchment-Scale Erosion and Yields of Suspended Solids from Improved Temperate Grassland. Journal of Environmental Monitoring. DOI:10.1039/b921584k

Brasington, J., Langham, J. and Rumsby, B. 2003. Methodological sensitivity of morphometric estimates of coarse fluvial sediment transport. Geomorphology, **53**, 299 – 316.

Brazier, R.E., Rowan, J.S., Quinn, P. and Anthony, S. 2001a. Towards an MIR (Minimum Information Requirement) approach to modelling on-site soil loss at the National scale. CATENA, 42 pp59-79

Brazier, R.E., Beven, K.J., Anthony, S., Rowan, J.S. and Quinn, P. 2001b. Implications of complex model uncertainty for the mapping of hillslope scale soil erosion predictions. Earth Surface Processes and Landforms. 26 pp1333-1352

Brazier, R.E. 2004. Quantifying soil erosion by water in the UK: a review of monitoring and modelling approaches. Progress in Physical Geography, 28, 340-365.340-365

Brazier, R.E., Bilotta, G.S. and 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 Surface Processes and Landforms, 32 pp964-967 doi:10.1002/esp.1484

British Sugar. 2002. The Environmental Impact of Growing Sugar Beet – British Sugar's submission to DEFRA. http://www.britishsugar.co.uk/bsweb/bsgroup/Defra_impact.htm.

Brus, D. J. 2000. Using regression models in design-based estimation of spatial means of soil properties. *European Journal of Soil Science,* **51,** 159-172.

Carroll, Z.L., Reynolds, B., Emmett, B.A., Sinclair, F.L., Ruiz de Ona, C., Williams, P. 2004. The effect of stocking density on soil in upland Wales. Centre for Ecology and Hydrology, Bangor.

Chandler, J.H. 2001. Terrain measurements using automated digital photogrammetry. Geological Society, London, Engineering Geology Special Publications, 2001, **18**, 13-18.

Collins, A.L. and D.E. Walling, 2004. Documenting catchment suspended sediment sources: problems, approaches and prospects. Progress in Physical Geography, **28**(2): p. 159-196.

Coulthard, T.J. 2005. Effects of vegetation on braided stream pattern dymanics. Water Resources Research, 41, W04003, doi:10.1029/2004WR003201.

Couper, P.R. and Maddock, I.P. 2001. Subaerial river bank erosion processes and their interaction with other bank erosion mechanisms on the River Arrow, Warwickshire, UK. Earth Surface Processes and Landforms, **26**, 631 – 646.

Couper, P., Stott, T. and Maddock, I. 2002. Insights into river bank erosion processes derived from analysis of negative erosion-pin recordings: observations from three recent UK studies. Earth Surface Processes and Landforms, **27**, 59-79.

Darby, S.E. Gessler, D. and Thorne, C.R. 2000. Computer program for stability analysis of steep, cohesive riverbanks. Earth Surface Processes and Landforms **25**, 175-190.

De Rose, R.C. and Basher, L.R. 2010. Measurement of river bank and cliff erosion from sequential LIDAR and historical aerial photography. Geomorphology, doi: 10.1016/j.geomorph.2010.10.037.

Deasy, C. Brazier, R.E., Heathwaite, A.L. and Hodgkinson, R. (2009) Pathways of runoff and sediment transfer in small agricultural catchments. Hydrological Processes, 23 (9). pp. 1349-1358.

De Baets, S., Poesen, J., Gyssels, G. and Knapen, A. 2006. Effects of grass roots on the erodibility of topsoils during concentrated flow. Geomorphology, **76**, 54-67.

de Gruijter, J., Brus, D., Bierkens, M. & Knotters, M. 2006. *Sampling for Natural Resource Monitoring,* Springer, Berlin.

De Jong, S.M., et al., Regional assessment of soil erosion using the distributed model SEMMED and remotely sensed data. Catena, 1999. **37**(3-4): p. 291-308.

Defra. 2005. Controlling soil erosion. 42pp.

Defra, 2005. Controlling soil erosion. Incorporating former advisory leaflets on grazing livestock, wind, outdoor pigs and the uplands (Ed Defra), pp. 41. Defra, London.

Defra, 2009. Documenting soil erosion rates on agricultural land in England and Wales - Part 2. SP0413. Defra, London.

Dunne, T., Mertes, L. A. K., Meade, R. H., Richey, J. E., and Forsberg, B. R., 1998. Exchanges of sediment between the flood plain and channel of the Amazon River in Brazil, Geol. Soc. Am., Bulletin, **110**, 450-467.

Evans, R. 2002. An alternative way to assess water erosion of cultivated land - field-based measurements: and analysis of some results. *Applied Geography,* **22,** 187-207.

Evans, M., and Lindsay, J. 2010. High-resolution quantification of gully erosion in upland peatlands at the landscape scale. Earth Surface Processes and Landforms 35 (8), 876-886.

Evans, R. 1988. Water erosion in England and Wales 1982–1984. Survey and Land Research Centre, Silsoe: Cranfield University.

Gerbersdor, S.U., Bittner, R., Lubarsky, H., Manz, W and Paterson, D.M. 2009. Microbial assemblages as ecosystem engineers of sediment stability. Journal of Soils Sediments, **9**, 640-652. Doi. 10.1007/s1 1368-009-0142-5.

Govers, G., Vandaele, K., Desmet, P., Poesen, J. and Bunte, K. 1994. The role of tillage in soil redistribution on hillslopes. European Journal of Soil Science, **45**, 469-478.

Govers, G., Quine, T.A., Desmet, P.J.J. and Walling, D.E. 1996. The relative contribution of soil tillage and overland flow erosion to soil redistribution on agricultural land. Earth Surface Processes and Landforms, **21**, 929-946.

Granger, S.J. Hawkins, J.M.B., Bol, R., White, S.M., Naden, P., Old, G., Bilotta, G.S., Brazier, R.E., Macleod, C.J.A. and Haygarth, P.M. 2009. High temporal resolution monitoring of multiple pollutant responses in drainage from an intensively managed grassland catchment caused by a summer storm. Water, Air and Soil Pollution, 205:1-4:377-393

Granger, S.J. Bol, R., Dixon, L., Naden, P., Old, G., Marsh, J., Bilotta, G.S., Brazier,R.E., White, S. and. Haygarth, P.M. 2010a. Assessing multiple novel tracers to improve the understanding of the contribution of agricultural farm waste to diffuse water pollution. Journal of Environmental Monitoring. DOI:10.1039/b915929k

Granger, S.J., Bol, R., Hawkins, J.M.B., White, S.M., Naden, P., Old, G., Marsh, J.K. Bilotta, G.S. Brazier, R.E. Macleod, C.J.A and Haygarth, P.M. 2010b. Using artificial fluorescent particles as tracers of livestock wastes within an agricultural catchment. Science of the Total Environment. 10.1016/j.scitotenv.2010.12.005

Haboudane, D., et al., Land degradation and erosion risk mapping by fusion of spectrally-based information and digital geomorphometric attributes. International Journal of Remote Sensing, 2002. **23**: p. 3795-3820.

Hagerty, D.J. 1991. Piping/sapping erosion I: Basic considerations. Journal of Hydraulic Engineering **117**, 991-1008.

Harmon, R. S., and W. W. Doe III, editors. 2001. Landscape erosion and evolution modelling. Kluwer Academic, New York. Catena 37 (3-4) 1999. GTCE comparison of erosion models.

Hill, J. and B. Schütt, 2000. Mapping complex patterns of erosion and stability in dry mediterranean ecosystems. Remote Sensing of Environment, **74**(3): p. 557-569.

Hilldale, R.C. andd Raff, D. 2008. Assessing the ability of airborne LIDAR to map river bathymetry. Earth Surface Processes and Landforms, **33**, 5, 773-783.

Hooke, J. M. 2004. Cutoffs galore! occurrence and causes of multiple cutoffs on a meandering river. Geomorphology vol 61 pp 225-238

Hooke, J.M. Kain, R.J.P. 1982. Historical change in the physical environment : a guide to sources and techniques. Butterworth Scientific

Howard, A. D., 1992. Modelling channel migration and floodplain development in meandering streams, in: P. A. Carling and G. E. Petts, (eds), Lowland Floodplain Rivers, Chichester, Wiley, pp 1-42.

Irons, J.R., R.A. Weismiller, and G.W. Petersen, 1989. Soil reflectance, in Theory and applications of optical remote sensing, G. Asrar, Editor. John Wiley and Sons: New York, USA. p. 66-106.

Isaacs, E.H. and R.M. Srivastava, An introduction to applied geostatistics. 1989. New York: Oxford Universtiy Press.

Kirkby, M. J., Irvine, B. J., Jones, R. J. A., Govers, G., and team, P., 2008. The PESERA coarse scale erosion model for Europe. I. – Model rationale and implementation: European Journal of Soil Science, v. 59, no. 6, p. 1293-1306.

Kirkby, M.I. 2010. Distance, time and scale in soil erosion processes. Earth Surface Processes and Landforms, **35**(13): p. 1621-1623.

Krueger, T., Quinton, J.N., Freer, J., Macleod, C.J.A., Bilotta, G.S. Brazier, R.E., Butler, P. and P.M. Haygarth. 2009. Uncertainties in data and models to describe event dynamics of agricultural sediment and phosphorus transfer. Journal of Environmental Quality 38(2), doi:10.2134/jeq2008.0179.

Krueger T, Freer J, Quinton J, Macleod CJA, Bilotta GS, Brazier RE, Hawkins, J. and Haygarth PM. (in review) Learning about catchment sediment and phosphorus dynamics through model hypothesis-testing under data uncertainty. *European Journal of Soil Science*.

Krueger, T., J. Freer, J. N. Quinton, C. J. A. Macleod, G. S. Bilotta, R. E. Brazier, P Butler and P. M. Haygarth. 2010. Ensemble evaluation of hydrological model hypotheses, *Water Resour. Res*., 46, W07516, doi:10.1029/2009WR007845.

Lark, R.M. 2009. Estimating the regional mean status and change of soil properties: two distinct objectives for soil survey. *European Journal of Soil Science,* **60**, 748–756.

Lawler, D.M. 2005. The importance of high-resolution monitoring in erosion and deposition dynamics studies: examples from estuarine and fluvial systems. Geomorphology, **64**, 1-23.

Lawler, D.M. 1995. The impact of scale on the processes of channel-side sediment supply: a conceptual model. Effects of Scale on Interpretation and Management of Sediment and Water Quality (Proceedings of a Boulder Symposium, July 1995. IAHS Publ. No. 226, 175 – 184.

Lawler, D.M. 1993. The measurement of river bank erosion and lateral channel change: a review. Earth Surface Processes and Landforms, **18**, 777-821.

Lawler, D.M. 1992. Process dominance in bank erosion systems. In: Carling, P.A and Petts, G.E Lowland Floodplain Rivers: Geomorphological processes. Wiley, Chichester, UK, 117-143.

Lawler, D.M. 1991. A new technique for the automatic monitoring of erosion and deposition rates. Water Resources Research, **27**, 2125 – 2128.

Lawler, D.M. 1986. River bank erosion and the influence of frost: a statistical examination. Transactions of the Institute of British Geographers **11**, 227- 242.

Lawler, D.M., West, J.R., Couperthwaite, J.S. and Mitchell, S.B. 2001. Application of a Novel Automatic Erosion and Deposition Monitoring System at a Channel Bank Site on the Tidal River Trent, U.K. Estuarine, Coastal and Shelf Science, **53**, 237 – 247.

Lawler, D.M., Grove, J.R., Couperthwaite, J.S. and Leeks, G.J.L. 1999. Downstream change in river bank erosion rates in the Swale-Ouse system, northern England. Hydrological Processes, **13**, 7, 977-992.

Lawler, D.M., Couperthwaite, J., Bull, L.J. and Harris, N.M. 1987. Bank erosion events and processes in the Upper Severn basin. Hydrology and Earth System Sciences, **1**, 2, 523-534.

Lu, H., Prosser, I. P., Moran, C. J., Gallant, J. C., Priestley, G., and Stevenson, J. G., 2003. Predicting sheetwash and rill erosion over the Australian continent: Soil Research, v. 41, no. 6, p. 1037-1062.

Metternicht, G.I. and A. Fermont. 1998. Estimating erosion surface features by linear mixture modelling. Remote Sensing of Environment. **64**(3): p. 254-265.

Milton, E.J. and J.P. Webb. 1987. Ground radiometry and airborne multispectral survey of bare soils. International Journal of Remote Sensing, **8**(1): p. 3-14.

Milton, E.J., D.J. Gilvear, and I.D. Hooper. 1995. Investigating change in fluvial systems using remotely sensed data, in Changing river channels, A.M. Gurnell and G. Petts, Editors. Wiley: Chichester. p. 277-301.

Moran, M.S., et al. 2000. Soil moisture evaluation using multi-temporal synthetic aperture radar (SAR) in semiarid rangeland. Agricultural and Forest Meteorology, **105**(1-3): p. 69-80.

Morgan, R. P. C., and Quinton, J. N. 2001. Erosion modelling, in Harmon, R. S., and Doe III, W. W., eds., Landscape erosion and evolution modelling: New York, Kluwer Academic/Plenum Publishers, p. 117-143.

Morgan, R. P. C., Morgan, D. D. V., and Finney, H. J. 1984. A predictive model for the assessment of soil erosion risk: Journal of Agricultural Engineering Research, v. 30, p. 245-253.

Morgan, R.P.C. and Nearing, M.A. 2010. Handbook of soil erosion modelling. Wiley Blackwell.

Mosley, M.P. 1975. Channel changes on the River Bollin, Cheshire 1872-1973. East Midland Geographer, **6**, 185-199.

Mount, N.J., Louis, J., Teeuw, R.M., Zukowskyj, P.M. and Stott, T. 2003. Estimation of error in bankfull width comparisons from temporally sequenced raw and corrected aerial photographs. Geomorphology, **56**, 65 – 77.

Mukundan, R, Radcliffe, D.E. and Risse, L.M. 2010. Spatial resolution of soil data and channel erosion effects on SWAT model predictions of flow and sediment. Journal of Soil and Water Conservation, **6**, 2, 92 – 104.

Mulholland, B., and Fullen, M. A. (1991). Cattle trampling and soil compaction on loamy sands. Soil Use and Management. 7, 189–193.

Murray, A. B. and Paola, C. 1994. A cellular model of braided rivers, Nature, **371**, 54-57.

Neal, C., and Jarvie, H.P. 2005. Agriculture, community, river eutrophication and theWater Framework Directive. Hydrological Processes 19:1895-1901.

Owens, P.N., Rickson, R.J., Clarke, M.A., Dresser, M., Deeks, L.K., Jones, R.J.A., Woods, G.A., Van Oost, K. and Quine, T.A. 2006. Review of the existing knowledge base on magnitude, extent, causes and implications of soil loss due to wind, tillage and co-extraction with root vegetables in England and Wales, and recommendations for research priorities. National Soil Resources Institute (NSRI) Report to DEFRA, Project SP08007, NSRI, Cranfield University, UK.

Parker, G. 1976. On the causes and characteristic scales of meandering and braiding in rivers, J. Fluid Mechanics, **76**, 457-480.

Payne, R.W. 2007. STTEST. In: *GenStat Release 10 Reference Manual Part 3: Procedure Library PL18* (ed. R.W. Payne), pp. 714–716. VSN International, Hemel Hempstead.

Peeters, A. 2004. Wild and sown grasses. Profiles of a temperate species selection; Ecology, biodiversity and use. Blackwell, London.

Quine, T.A., Govers, G., Walling, D.E., Zhang, X.B., Desmet, P.J.J., Zhang, Y.S. and

Vandaele, K. 1997. Erosion processes and landform evolution on agricultural land - New perspectives from caesium-137 measurements and topographic-based erosion modelling. Earth Surface Processes and Landforms, **22**, 799-816.

Quine, T.A. Zhang, Y. 2002. An investigation of spatial variation in soil erosion, soil properties, and crop production within an agricultural field in Devon, United Kingdom. Journal of Soil and Water Conservation, **57**, 55-65.

Quine, T.A., Van Oost, K., Walling, D.E. and Owens, P.N. 2006. Development and application of GIS-based models to estimate national rates of soil erosion by tillage, wind and root crop harvest. University of Exeter Report to DEFRA, Project SP08007, University of Exeter, UK.

Quinton , J.N., Krueger, T., Freer, J. and Bilotta, G. S., Brazier, R.E. 2010). EUROSEM: An Evaluation of the dynamic capability of the EUROSEM model using GLUE. In: R.P.C. Morgan and M.A. Nearing (Editors), Handbook of Erosion Modelling. Blackwell.

Ruysschaert, G., Poesen, J., Verstraeten, G. and Govers, G. 2004. Soil loss due to crop harvesting: significance and determining factors. Progress in Physical Geography, **28**, 467-501.

Ruysschaert, G., Poesen, J., Verstraeten, G. and Govers, G. 2005. Interannual variation of soil losses due to sugar beet harvesting in West Europe. Agriculture, Ecosystems and Environment, **107**, 317-329.

Schäppi, B., Perona, P., Schnieder, P. and Burlando, P. 2002. Integrating river cross section measurements with digital terrain models for improved flow modelling applicatons. Computers and Geosciences, **36**, 707-716.

Scholz, G., Quinton, J. N., and Strauss, P. 2008. Soil erosion from sugar beet in Central Europe in response to climate change induced seasonal precipitation variations: Catena, v. 72, no. 1, p. 91-105.

Schneider, A., Gerke, H.H. and Maurer, T. 3D initial sediment distribution and quantification of mass balances of an artificially-created hydrological catchment based on DEMs from aerial photographs using GOCAD. Journal of Physics and Chemistry of the Earth. Doi: 10.1016/j.pce.2010.03.023.

Scottish Environment Protection Agency (SEPA). 2009. Engineering in the Water Environment Good Practice Guide Riparian Vegetation Management. 47pp.

Simon, A., Curini, A., Darby, S.E., Langendoen, E.J. 2000. Bank and near-bank processes in an incised channel. Geomorphology **35**, 193-217.

Stott, T., 1997. A comparison of stream bank erosion processes on forested and moorland streams in the Balquihidder catchments, Central Scotland. Earth Surface Processes and Landforms, **23**, 383 – 399.

Tal, M. and Paola, C. 2007. Dynamic single-thread channels maintained by the interaction of flow and vegetation, Geology, **35**, 347-350.

Thoma, D.P., Gupta, S.C. and Bauer, M.E. 2001. Quantifying river bank erosion with scanning laser altimetry. International Archives of Photogrammetry and Remote Sensing, XXX1V-3/W4, Annapolis,MD, 22-24 October 2001.

Thorne, C.R. 1991. Bank erosion and meander migration of the Red and Mississippi rivers, USA. Hydrology for the Water Management of Large River Basins (Proceedings of the Vienna Symposium, August 1991). IAHS Publication No. **201**. 301 – 313.

Trimble, S.W. 1994. Erosional effects of cattle on streambanks in Tennessee, USA. Earth Processses and Landforms, **19**, 451-464.

Trimble, T. W., and Mendel, A. C. (1995). The cow as a geomorphic agent—A critical review. Geomorphology 13, 233–253.

Van Oost, K., Govers, G. and Desmet, P. 2000. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. Landscape Ecology, **15**, 577-589.

Van Oost, K., Govers, G., Quine, T.A., Heckrath, G., Olesen, J.E., De Cryze, S. and Merckx, R. 2005. Landscape-scale modelling of carbon cycling under the impact of soil redistribution: the role of tillage erosion. Global Biogeochemical Cycles, **19**, Article GB4014.

Van Oost, K., Govers, G., de Alba, S. and Quine, T.A. 2006. Tillage erosion: a review of controlling factors and implications for soil quality. Progress in Physical Geography, **30**, 443-466.

Van Oost, K., Quine, T. A., Govers, G., De Gryze, S., Six, J., Harden, J. W., Ritchie, J. C., McCarty, G. W., Heckrath, G, Kosmas, C., Giraldez, J. V., da Silva, J. R., Merckx, R. 2007. The impact of agricultural soil erosion on the global carbon cycle, Science, 318, 626-629.

Van Oost, K., Cerdan, O., Quine, T.A. 2009. Accelerated sediment fluxes by water and tillage erosion on European agricultural land, Earth Surf. Proc. & Landforms, 34, 1625-1634

Walling, D.E. 2005. Tracing suspended sediment sources in catchments and river systems. Science of the Total Environment, **344**, 159 – 184.

Walling, D.E. 2008. Documenting soil erosion rates on agricultural land in England and Wales: Phase 2. SP0413, Defra, London.

Warburton, J., Danks, M. and Wishart, D. 2002. Stability of an upland gravel-bed stream, Swinhope Burn, Northern England. Catena, **49**, 309 – 329.

Wischmeir, W. H., and Smith, D. D. 1978. Predicting rainfall erosion losses: a guide to conservation planning.: USDA agricultural handbook, 537.

Wynn, T.M., Mostaghimi, S. and Alphin, E.F. 2004. The effects of vegetation on stream bank erosion. ASAE/CSAE Annual International Meeting, Fairmont Chateau Laurier, The Westin, Government Centre, Ottawa, Ontario, Canada. 1-4 August 2004. Paper 042226. 15pp.