Research to develop the evidence base on soil erosion and water use in agriculture

Final Technical Report

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Executive Summary

Part I: Soil erosion

The aim of this research was to better understand the factors that influence the vulnerability of land to soil erosion, and how these factors (and hence erosion vulnerability) vary over space and time. Erosion risk factors at a particular site and at a particular time were identified as: the intensity, duration and timing of rainfall events (erosivity); the physical, biological and chemical properties of soils (erodibility); the length, gradient and form of slope; the type of vegetation/crop on the land and its stage of development; and the type and timing of singular or combined land management practices.

To check the sensitivity of erosion processes to these factors, the changes in risk factors should be compared with observed erosion at the national scale. However, these observations are very limited in spatial (and temporal) extent. Therefore, the risk factors were compared with soil erosion rates over space and over time, as generated by an erosion prediction model (the modified Morgan, Morgan and Finney model). The limitations of using a model without adequate validation with empirical observations are acknowledged. Also, some risk factors cannot be mapped at the national scale, such as land management practices and the socio-economic factors that affect erosion.

Due to the duration of the present project and accessibility of relevant meteorological and land use data, only 5 years were selected for analysis: 1969; 1979; 1988; 1997; and 2010. These years were specifically chosen because data relating to both rainfall (Met Office) and land use (Edina Agcensus) were readily available for these dates.

The correlation between rainfall (when expressed as annual precipitation or as a recognised index of erosivity) and the spatial distribution and rates of soil erosion is poor. Soil type will influence the spatial distribution of erosion risk, but a direct spatial correlation is complicated by other risk factors such as land use.

The total area of high erosion-risk crops in England did not change greatly among the individual years investigated. The greatest area under high risk crops occurred in 1969, which corresponds to the year with the highest percentage of land modelled as being at ‘high’ erosion risk. The evidence of a recent increase in land area under a potentially erosive crop (maize) is not reflected in the available datasets on land use which are only available up to 2010.

According to the modelled outputs, the proportion of Grade 1 land that is subject to ‘moderate’ or ‘high’ erosion rates does not vary greatly among the 5 time periods. The current analysis has identified areas in the country where relative soil erosion risk may be elevated due to the combination of risk factors. As suggested by Kibblewhite et al. (2014), this methodology could be further developed into a tiered approach to erosion risk assessment, where greater effort is focussed on areas of relative high erosion risk. In other words, specific areas can be identified where the combination of risk factors is likely to lead to unacceptable rates of soil erosion. This is the first stage of developing and implementing measures to control the irreversible loss of the soil resource through erosion.

Although unvalidated, the erosion prediction model, when run for each year separately, suggests that soil erosion rates have not changed markedly over the 5 time periods. However, a number of authors suggest that the extreme weather events and shifts in land use / cropping patterns associated with climate change will bring about accelerated rates of erosion in the future.

These conclusions are necessarily based on a number of assumptions. These have had to be made, given the limitations of the methodology that in turn are the consequence of the paucity of empirical observations of soil erosion over time at the national scale.

Part II: Water use in agriculture

The spatial distribution of 'hot-spots' for agricultural irrigation abstraction and livestock water use across England and Wales were mapped using irrigation or livestock water 'intensity' (m^3 per km^2) rather than volumetric (m^3 per catchment) indicators. Irrigation intensity conveys more accurately where there is high demand per unit of irrigated land use in water stressed sub catchments. The key conclusions for each sub-sector are briefly summarised below:

Irrigation water demand

A detailed temporal and spatial analysis of historical trends in irrigation water use was undertaken. The analysis shows how temporal patterns of irrigation abstraction have been closely linked to agroclimate variability (rainfall and evapotranspiration (ET)).

Irrigation demand for agricultural and horticultural cropping is mainly concentrated in eastern and south eastern England. A small number of key catchments have been identified where irrigation demand is concentrated and where water resources for irrigation abstraction are already under severe stress and hence where future resource problems are most likely to be encountered. These catchment areas include the Cam and Ely Ouse, East Suffolk, parts of Norfolk (Broadland Rivers), west Midlands (Worcestershire Middle Severn), north Essex, and the East Midlands around the Humber in north Lincolnshire.

An analysis of underlying historical trends in irrigation water use have shown that there has been a long period of strong growth (since the 1970s) followed by a more recent decline in both licensed and abstracted volumes. The change in the abstraction trend appears to have occurred earlier than the change in the licensed volume trend. Since the early 1990s, the volume abstracted appears to have been declining at an average rate of 2% to 3% (of the 2010 value) per annum. This, at least partly is attributed to the increasing yield and hence decreasing cropped areas needed (particularly for potatoes and some other major irrigated vegetable crops), together with increased efficiency. The increasing problems relating to reduced water availability and reliability, and hence a greater appreciation of its value, are also likely to have contributed to water conservation. It is noted, however, that these short-term trends do not yet reflect very recent changes due to higher food prices, particularly for cereals.

Livestock water demand

As there is no national data on water used for livestock, and it is difficult to separate water used on farm for livestock uses from other uses, livestock water demand has been estimated from livestock numbers and per capita water requirements. Given this, water demand for livestock in England has declined steadily until around 2000, and has since stabilised at around 120 Mm^3. Nearly three quarters (68%) is used for cattle (beef and dairy), with sheep (16%), poultry (9%) and pigs (7%) accounting for the remainder.

Most water for livestock drinking and washing comes from the public water supply (mains), therefore hot-spots for water use for livestock are those areas where the public water supply is seriously stressed and the intensity of livestock water use is high. In general, the intensity of livestock water use is greater in Water Company Areas that are not currently, or estimated in the short term, to be seriously water stressed. Therefore livestock water use is less vulnerable to water shortage than irrigation water use.

A detailed synthesis of literature, published case studies and evidence from key informants was used to understand the vulnerability of livestock production to reduced water availability, and to review what actions should be taken by the livestock sector to mitigate risks from future reduced water availability.

Intensity of water use in agriculture

The table below shows the total (irrigation + livestock) agricultural water use intensity in England and Wales by Environment Agency (EA) catchment. In all catchments with a total water intensity greater than 4,000 m^3/km^2, demand is dominated by water use for irrigation, whereas livestock water demand tends to dominate those catchments with a total intensity of less than 4,000 m^3/km^2.

Top 20 Environment Agency catchments for agricultural water intensity (m^3/km^2) and major (>67%) user.

EA catchment	*Water Intensity, m^3/km^2*			
	Irrigation	*Livestock*	*Total*	*Major user†*
Cam and Ely Ouse	7,186	414	7,600	Irrigation
North Norfolk	5,162	290	5,451	Irrigation
East Suffolk	4,770	574	5,344	Irrigation
Idle & Torne	4,169	349	4,518	Irrigation
Old Bedford	3,969	119	4,087	Irrigation
Shropshire Middle Severn	1,512	1,933	3,445	Both
Weaver and Dane	0	3,064	3,064	Livestock
Broadland Rivers	2,183	857	3,039	Irrigation
Dove	83	2,629	2,712	Livestock
Wyre	0	2,574	2,574	Livestock
Arun & Western Streams	1,796	569	2,365	Irrigation
Dee	49	2,311	2,360	Livestock
North West Norfolk	2,073	250	2,323	Irrigation
Otter, Sid, Axe and Lim	55	2,102	2,157	Livestock

EA catchment	*Water Intensity, m^3/km^2*			
	Irrigation	*Livestock*	*Total*	*Major user†*
Wye	687	1,466	2,153	Both
Worcestershire Middle Severn	1,462	654	2,116	Irrigation
Severn Uplands	122	1,977	2,099	Livestock
Lower Trent & Erewash	1,211	750	1,962	Both
Little Avon	0	1,956	1,956	Livestock
Torridge and Hartland Streams	1	1,944	1,945	Livestock

† Water use in a single sector >67% of total.

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About the authors

Dr Jerry Knox is a Reader in Agricultural Water Management with 22 years research and consultancy experience in the science, engineering and management of water resources for agriculture including the risks to crop productivity (yield) and sustainability of rainfed and irrigated production in the context of climate change and food security. He is a Director and Honorary Secretary of the UK Irrigation Association (UKIA).

Professor Jane Rickson (Chair in Soil Erosion and Conservation) has over 29 years' experience in land and water engineering and management, specialising in land degradation processes and their mitigation. Her work has focused on soil erosion risk assessment, erosion prediction modelling and experimentation in erosion research. She is a Chartered Environmentalist and a Fellow of the Institution of Agricultural Engineers.

Professor Keith Weatherhead is Professor of Water Resources and Climate Adaptation, with over 30 years' experience in soil and water engineering, irrigation, water resources and climate change impacts and adaptation. Current research includes water trading (EPSRC/LWEC) and water supply for agriculture (BBSRC, Defra). He is a member of the CIWEM Water Resources Panel and the NFU-led Water for Food group.

Dr Tim Hess is a Reader in Water Management with over 30 years' experience in the application of hydrology in managed rural environments, and a particular interest in the application of ecosystem services approaches to water management. His current research is linked to assessing the role of water in sustainable food supply chains, managing corporate sustainability and water foot printing.

Dr Andre Daccache was an Academic Fellow at Cranfield University for this study and until recently, and continues to work with staff there. He provides expertise in crop modelling, water resources and GIS analysis.

Dr Lynda Deeks (Research Fellow in Soil Science) has many years' experience in searching and reviewing literature related to soil erosion and carrying out semi-structured systematic reviews using a variety of sources. Her reviews have been part of reports to Defra (SP1601 Studies in support of the Soil Strategy, SP1611 Soil Quality Indicators (physical properties); SP1317 Soil depth and ecosystem goods and services; and SP1318 Scaling up the benefits of soil protection measures) and the HDC.

Tim Brewer (Senior Lecturer in Resource Survey) and Ian Truckell (Spatial Systems and GIS Analyst) have experience in soil erosion mapping and GIS analyses. They were involved in Defra SP1609 defining the priority area approach, which used a similar methodology as proposed in this project. Through their mapping and GIS expertise and skills, they have generated the statistics related to the area affected by erosion, and how this area is divided into the different ALC classes.

Caroline Keay (Senior Information Scientist) has experience working with the Agricultural Land Classification of England and Wales, and its relationship with soil properties and functions. She has assessed how this might be affected by future climate change, including increased frequency and magnitude of soil erosion, which both determines land capability and results from a reduction of land capability

Acknowledgement for front cover photo: William Stephens.

1. INTRODUCTION

The Adaptation Sub-Committee (ASC) of the Committee on Climate Change (CCC) has a statutory duty to report to Parliament with an independent assessment of the UK Government's progress in implementing its National Adaptation Programme (NAP). Previous ASC reports have identified the key components for assessing preparedness for climate change. The aim of this project is to improve the indicators of risk and adaptation action for both soil erosion and water use for agriculture.

The first Climate Change Risk Assessment (CCRA, 2012) discussed how predicted trends in future climate may impact on the risk of soil erosion in England and Wales. The intention of this research project is to better understand how soil erosion vulnerability is spatially distributed and how this compares to the spatial distribution of land use and the quality of agricultural land. It will provide an understanding of the factors that influence erosion vulnerability, how they vary geographically and how they have varied over time. This will provide the ASC with evidence on:

- The factors that influence erosion vulnerability
- Areas most at risk and how these have changed over time
- How these vary by grade of agricultural land

CCRA (2012) also identified increases in water demand for irrigation of crops and unsustainable water abstraction as risks from climate change to water availability for crop use. The ASC 2013 progress report estimated future imbalances between supply and demand of water for crop irrigation and assessed the location of demand at a River Basin District level. The purpose of the second part of the research is:

- To better understand the potential "hot-spots" of water availability due to abstraction demand for crop irrigation.
- To better understand the vulnerability of livestock production to reduced water availability.
- To better understand what actions are being taken to mitigate the risk from reduced water availability in the future with respect to livestock.

The focus of this study was on England; however, as some of the datasets cover England and Wales, and as the Environment Agency catchment boundaries do not coincide with national boundaries, some of the analysis covers Wales as well as England.

2. PART I: SOIL EROSION

2.1 Introduction

2.1.1 Setting the context

The first Climate Change Risk Assessment (CCRA, 2012) discussed how predicted trends in future climate may impact on the risk of soil erosion in England and Wales. These trends (as simulated by UKCIP02 scenarios) and their effects on erosion processes are shown in Table 1. In this context, 'risk' combines the probability, or frequency of soil erosion processes, and the magnitude of the consequences that will result when the soil is exposed to these processes. Risks to soil resources arise from potential hazards associated with different land uses that interact with the natural land system to initiate degradation, causing harm to soil functions (Kibblewhite et al., 2012).

Table 1 Impact of changing climate parameters on soil erosion.

Climatic trend	Likely impacts on erosion processes
Increasing summer temperatures	Drier soils that are more prone to wind erosion and hydrophobicity which can lead to increased runoff and associated erosion. (Evans, 1996). Change in land use / crop suitability e.g. more land under potentially erosive crops such as maize. More vigorous vegetation growth (if not limited by other factors such as water availability), which would offer greater protection from wind and rainfall.
Increasing winter temp	Change in land use / crop suitability More vigorous vegetation growth (if not limited by other factors such as water availability), which would offer greater protection from wind and rainfall. Warmer winters may extend growing periods, but the risk of increased soil degradation may rise due to higher predicted winter precipitation rates that will increase risks of working the land when wet. Later harvest may also increase the risk of loss of soil co-extracted on root vegetables and farm equipment if the soil moisture is high.
More extreme high temperature	Greater risk of unstable atmospheric conditions and high intensity thunderstorms. Also leading to drier soils – see above

Climatic trend	Likely impacts on erosion processes
Less extreme low temperature	Change in land use / crop suitability
Higher winter rainfall	Wetter soils that are more prone to aggregate breakdown, compaction, smearing and generation of surface flow. Risk higher on spring-planted crops such as maize and potatoes than on winter cereals.
Higher wind speeds	Greater wind speeds driven by higher atmospheric temperatures, combined with drier, more friable, soils in summer months will increase the potential for wind erosion. However, erosion by wind is not predicted to change under current predictions of wind speed changes.
Less summer rainfall	Drier soils (see above). Poorer crop canopy development, leading to more exposure of soil when rain does fall.
More intense downpours	Rainfall intensity is strongly and positively correlated with soil erosion rates (erosivity; Wischmeier and Smith, 1969). The more intense the storm, the greater the erosivity of the rainfall event and the greater the potential for erosion. Short duration, high intensity rainfall events may become the dominant mechanism of soil erosion in the future.
Sea level rise and increased coastal flood risk	Only very indirect impacts on land erosion through hydrological behaviour of the water table
More winter storms	Wetter soils, leading to shorter time to generation of runoff and greater volume of runoff, leading to increased erosion risk

Higher rates of soil erosion are associated with a loss of this virtually irreplaceable natural resource that delivers a number of vital goods and services that have been linked to individual health and well-being, which in turn determine the national economic wealth (Daily, 1997). Currently the costs of soil erosion to the UK economy have been estimated at £200 million per annum (Graves et al., 2011), but with predicted changes in erosion vulnerability, these costs are likely to rise. It is argued that soil erosion is an irreversible process, given that rates of soil formation are so low (Verheijen et al., 2009). One consequence of soil erosion is a reduction in the capability of land to support various end uses such as food production. This concept of land capability is expressed in the Agricultural

Land Classification (ALC; MAFF, 1988). Soil degradation, through erosion could cause an irreversible decline in the productive capacity of the land (ASC, 2013).

2.1.2 Aims and objectives

The aim of this research is to better understand the factors that influence the vulnerability of land to soil erosion by water and by wind, and how these factors (and hence erosion vulnerability) vary over space and time. In turn, spatial and/or temporal variations in erosion rates may impact on the quality of agricultural land (as indicated by the Agricultural Land Classification; MAFF, 1988). The following sections cover:

- The factors that influence the vulnerability of land to soil erosion, by water and by wind;
- How these factors are spatially distributed (and thus vary geographically);
- How these factors have varied over time;
- How the spatial distribution of these factors relates to (a) spatial distribution of land use and (b) the quality of agricultural land as expressed in the ALC.

Other types of soil erosion (including for example, loss of soil through co-extraction on agricultural machinery and harvested crops) are considered out of scope, but a review of these processes is given in Owens et al. (2006).

2.2 Literature review of the factors that affect soil erosion

The purpose of the literature review was to a) identify the factors that influence soil erosion vulnerability and b) give estimates of the effects of these factors on soil erosion.

2.2.1 Factors affecting soil erosion by water

Water-induced erosion is more widespread than wind erosion (Owens et al., 2006). It occurs most frequently on sloping land with bare soil or sparse crop cover where the soil is weakly structured and has a fine sandy or coarse silty texture. The risk is greatest during periods of heavy rainfall when the soil has become saturated and surface soil structure broken down by the impact of raindrops. Soil erosion can destroy crops in localised areas or bury them under deposited sediment downslope. Farm machinery may be hindered subsequently where gullies are wide and deep. Typical measured rates of erosion by water in the UK are given in Appendix 7.1. Details of the factors affecting erosion by water are given in Appendix 7.2.

Rainfall

Rainfall erosivity is the ability of rainfall to cause erosion (by detachment and subsequent transport of soil particles and small aggregates). Morgan et al (1984) suggest annual rainfall divided by the number of rain days can be a simple index of erosivity. Raindrops impacting on bare soil surfaces are the most important cause of detachment, which is the first phase of the erosion process. The effectiveness of rain splash in causing detachment depends on the size (mass) and velocity of the raindrop, which determines its erosive power and is related to the kinetic energy of the raindrop (Gilley and Finkner, 1985). These two characteristics are related: for example a 1 mm diameter raindrop has a terminal velocity of 4 m s^{-1} while a 5 mm diameter raindrop falls at 9 m s^{-1} (Morgan, 2005). The length of the storm also affects raindrop detachment rates. Significant soil erosion events in England are generally associated with rainfall intensities of > 4 mm hour and rainfall quantities >15 mm per day (Defra, 2005). The erosive power of rain is reduced if the soil surface is covered by a crop, stubble or mulch as these act to dissipate the energy of the raindrop before it hits the soil.

The transport phase of the erosion process is dominated by overland flow. Once rainfall intensity exceeds the soil's infiltration rate, surface ponding and runoff will occur. This occurs during heavy rainfall, after prolonged rainfall or as a result of reduced infiltration rate (e.g. due to surface capping or compaction).

Climate change is expected to affect the intensity, amount, frequency and type of rainfall, with an increasing trend for 'heavy' (greater than 10 mm d^{-1}) and 'extreme' (> 95th percentile) rainfall events (Simmons and Rickson, 2008). The Hadley Centre for Climate Prediction and Research (HCCPR) in the UK reported that over the last 40 year some parts of the UK have seen a two-fold increase in the magnitude of extreme rainfall events. In the future, the extremes of heavy precipitation are very likely to increase in both magnitude and frequency. However, the frequency of localised summer storms is difficult to predict at present.

Analysis in UKCP09 (Jenkins et al., 2009) suggests that annual mean precipitation over England and Wales has not changed significantly since records began in 1766. Between 1961 and 2006 there has been a slight increase in average annual precipitation throughout the UK, but this is not statistically significant for England. Seasonal rainfall is highly variable, but appears to have decreased in summer and increased in winter, although for the former, this trend is not statistically significant above background natural variation. The increase in average winter precipitation in all regions of the UK between 1961 and 2006 is only statistically significant above background natural variation in Northern England and Scotland where increases of 30 to 65% have been experienced. Within the winter precipitation, an increased contribution of heavy precipitation events has been observed, but this was not found for summer rainfall. The average number of rain days for any region of the UK has not changed significantly between 1961 and 2006.

In terms of projections, central estimates are for heavy rain days (rainfall greater than 25 mm) over most of the lowland UK to increase by a factor of between 2 and 3.5 in winter, and 1 to 2 in summer by the 2080s under the medium emissions scenario. This is likely to increase erosion risk if the heavy rainfall falls at higher intensity, with greater kinetic energy and erosivity. One consequence is the process of surface seal formation, where high intensity rainfall (or irrigation) causes physical breakdown of soil aggregates. The eroded material slumps on the soil surface such that it is reorganised into a near-continuous layer of structureless fines, with very low infiltration rates.

Soil properties

The susceptibility of soil to erosion is termed soil erodibility. Factors affecting erodibility include soil texture (particular the very fine sand, fine sand and silt content). Silt sized particles are the most vulnerable to detachment by raindrops (Poesen, 1985). Smaller particles (clays) are more resistant to detachment because of cohesion, and larger particles (sands) are resistant because of mass/weight. Clays are more cohesive and tend to form larger coarser structures that are less easily eroded (entrained and transported). However, they tend to be less permeable and are less able to soak up rainfall, which increases likelihood of overland flow. The runoff may be sufficient to scour and detach fine soil particles. Soils with a high proportion of silt or fine sand are inherently weakly structured, with low aggregate stability. These soils are prone to surface capping and slaking, especially if the top soils have low organic matter content (Valentine et al., 1992). Structural seals are formed in situ, by processes directly related to raindrop and/or irrigation sprinkler water-drop impacts and the associated rapid wetting of soil aggregates at the immediate soil surface West et al. (1992). Associated processes include purely physical disaggregation, physico-chemical dispersion (Agassi et al., 1985; Levy et al., 1986) and slaking (Le Bissonnais, 1990). Subsequent drying, dehydration and cementation of the structural seals results in the formation of surface crusts (Romkens, 1986).

The formation of surface seals and crusts has a profound influence on the erodibility of soils through a significant reduction in surface hydraulic conductivity and subsequent increase in runoff generation (Morin et al., 1981; Philip, 1998). This in turn has direct consequences not only to offsite diffuse pollution but also to soil moisture re-charge and increases the risk of crop water stress during critical plant growth stages with resultant impacts on yield and quality.

Over 36% of arable England is at moderate to very high risk of surface sealing and erosion, including much of the better-drained and more easily worked land, especially sandy soils (Evans, 1990). In the UK, the soil textural classes susceptible to structural sealing occupy some 15% of the arable/horticultural land area associated with high value crops, often coincident with areas associated with limited water available for supplementary irrigation. Optimising soil moisture re-charge and managing the response of the soil surface to rainfall is particularly pertinent in the light of UKCIP climate change predictions of wetter winter and drier summers with increased frequency of extreme high intensity rainfall events (Jenkins et al., 2007). In addition, mechanical impedance to seedling emergence caused by structural crusts is a limiting factor in stand establishment and has been shown for a wide variety of commercial crops to delay stand establishment and reduce plant populations (Whalley et al., 2004; Lehrsch et al., 2005). In some cases, final plant densities can be reduced to the point that replanting is necessary. Uniform emergence of seedlings, has a direct influence on final yields as well as the proportion of yield in high value size grades at harvest. Consequently, structural crusts can have a profound effect on the commercial viability of crop production systems.

UK climate change scenarios for the primary southern arable and horticultural areas specifically involve scenarios likely to lead to increased soil surface sealing (more intense rainfall events) and hydrophobicity (warmer, drier summers).

Other factors affecting erodibility include soil shear strength. As soil shear strength increases, detachment decreases exponentially. Soil moisture content, organic matter content, soil chemistry and the stability of aggregates also determine the susceptibility of soil to erosion. The proportion of water stable aggregates is highly correlated to soil erodibility (Bryan, 1969). As well as the other factors listed, aggregate stability is also linked to climatic conditions that may destabilise soil structure such as the wetting and drying of soil, freeze and thaw cycles or frost heave). Tillage operations can also break up soil aggregation and structure.

Of the 297 Soil Associations of the National Soil Inventory (Mackney et al., 1983), Evans (1990) identified those at risk from erosion (by water and wind) in England and Wales (Appendix 7.3). Typical rates of erosion for different Soil Associations are given in Table 2. Median values are given as it gives a better central tendency because of the skewed nature of erosion date, which has many low erosion rates and a few very large events.

Table 2. Rate of erosion at 17 monitored locations in England and Wales, 1982-1986 and the soil associations within them with more than 30 eroded fields (adapted from Boardman, 2013).

Locality	Median erosion (m^3 ha^{-1})	Soil association	Median erosion (m^3 ha^{-1})	Topsoil texture
Bedfordshire	0.31	411d Hanslope	0.29	Clayey
Cumbria	0.36			
Devon	1.22			
Dorset	0.83	41b Evesham 2	0.74	Clayey
Gwent	0.83	571b Bromyard	0.80	Fine silty
		541a Milford	0.79	Fine loamy
Hampshire	1.33	571i Harwell	1.30	Loamy
Herefordshire	0.68	571b Bromyard	0.67	Fine silty
Isle of Wight	1.52	571g Fyfield 4	1.62	Coarse loamy&sandy
Kent	3.58			
Norfolk East	0.76	551g Newport 4	0.50	Sandy
		541t Wick 3	0.38	Coarse loamy
Norfolk West	0.25	343g Newmarket 2	0.58	Coarse loamy&sandy
		581f Barrow	0.19	Coarse loamy
Nottinghamshire	0.71	551b Cuckney 1	0.72	Sandy&coarse loamy
Shropshire	0.90	572m Salwick	1.25	Fine loamy
		551d Newport 1	0.97	Sandy&coarse loamy
		551a Bridgnorth	0.96	Sandy&coarse loamy
Somerset	2.55	541m S. Petherton	2.10	Silty
		572i Curtisden	1.39	Silty
Staffordshire	0.82	551a Bridgnorth	1.06	Sandy&coarse loamy
		551g Newport 4	0.66	Sandy
Sussex East	0.32			
Sussex West	0.29	343h Andover 1	0.37	Silty

The extent of soils at different risks of erosion is given in Table 3. According to Evans (1990), 38.2% of surveyed land in England and Wales is at 'very small risk' of erosion (Table 3). The Soil Associations in this category (n = 108) cover 53,449 km^2. Land at 'small risk' of erosion represents 38.0% of the land (n = 109 Soil Associations). Land at 'moderate risk' covers

18.0% (25,157 km^2; n = 60 Soil Associations). Land at 'high risk' covers 4.4% (6,198 km^2; n = 15 Soil Associations). Only 1.5% of land is considered at 'very high' risk (n = 4 Soil Associations).

Table 3 Extent and land use of the different Soil Associations classified by erosion risk (adapted from Evans, 1990).

Risk of erosion*	% of surveyed land	No of soil associations (area covered by km^2)	Crop area (%)	Soil and landscapes
Very small risk	38.2	108 (53,449 km^2)	35.2% dominantly arable land use 52.0 % grass 0.8% forest 12% heath and moor	Soils are predominantly heavy textured and associations characterized by low relief (terraces, floodplains, costal marshes, clay vales and lowlands and level or gently sloping plateaux) 65% of associations are mostly slowly permeable, seasonally waterlogged or high ground water tables
Small risk	38.0	109	53.2%[1] arable 32.1%[1] grass 0.6%[1] wooded	Low relief, gently undulating till plains, drift-covered vales and lowlands, plateaux and terraces. Associations of variable soil textures
Moderate risk	18.0	60 (25,157 km^2)	77.7% arable	Soils are loamy, and peats and sands blow. Undulating or rolling relief (light textures), Strongly or steeply sloping sites
High risk	4.4	15 (6,198 km^2)	51.8% uplands 11 of the associations are mostly arable	Lowland soils are either sandy, light loams or light silts, some peat. Topography rolling or undulating. Uplands eroding slopes steeply or very steeply sloping
Very high risk	1.5	4 (2,046 km^2)	Mostly arable	Sandy or light loams

[1] % of Soil Associations not land area

*Key to Table 3.

Very small risk	Erosion occurs rarely or not at all
Small risk	Eroding fields or moorland cover ≤1% of the land each year
Moderate risk	Arable: 1 to 5% land eroding each year Upland: small areas overgrazed
High risk	Arable: >5% land eroding each year and greater mean and median volume eroded than lower risk categories. Uplands: extensively gullied and hagged, widespread overgrazing, extensive footpath erosion.
Very high risk	Lowlands: erosion rarely affect <5% fields each year. An average of 10% fields affected, and 2 years in 5 20-25% affected. Volume of soil eroded often greatest of any risk category.

Evans (1990) notes that land use and global climate change may change the numbers of Soil Associations at risk from soil erosion (Table 4). With global warming, the land at moderate to very high risk of erosion is likely to increase from 23.9% to 46.1% of the area of England and Wales, and 126 Soil Associations are likely to become more at risk. The % area of England and Wales at 'very small' and 'small' risk of erosion reduces from 217 Soil Associations to 148, and the percentage area of England and Wales considered to be at 'very high risk' more than doubles from 1.5% to 3.3%.

Table 4 Number and extent of Soil Associations by erosion risk category (after Evans, 1990).

	Erosion risk category				
	V. small risk (erosion occurs rarely or not at all)	Small risk (<1% of the association)	Moderate risk (1-5%)	High risk (5-10%)	V. high risk (>10%; 2 in 5 years 20-25%)
Actual risk					
No. of Soil Associations	108	109	60	15	4
% area of England & Wales	38.2	38	18.0	4.4	1.5
Potential risk					
No. of Soil Associations	79	69	96	38	14
% area of England & Wales	27.6	26.3	30.1	12.7	3.3

Slope

Slope gradient, form and length affect the volume and velocity of overland flow, which in turn affect flow erosivity to detach and transport eroded material. When flow velocity falls (e.g. due to a reduction in slope gradient), flow transport capacity is reduced, and deposition of eroded material may occur (Morgan, 2005). Defra (2005) point out that slope steepness in combination with soil texture affects the level of risk of soil erosion by overland flow (Table 5).

Table 5 Risk of erosion in relation to slope steepness and soil texture (Defra, 2005).

Soils	Steep slopes >7°	Moderate slopes 3° – 7°	Gentle slopes 2° – 3°	Level ground <2°
Sandy and light silty soils	Very high[1]	High[2]	Moderate[3]	Lower[4]
Medium and calcareous soils	High[2]	Moderate[3]	Lower[4]	Lower[4]
Heavy soils	Lower[4]	Lower[4]	Lower[4]	Lower[4]

[1]Very high risk: rills are likely to form in most years and gullies may develop in very wet periods. [2]High risk: Rills are likely to develop in most seasons during wet periods. [3]Moderate risk: Sediment may be seen running to roads, ditches or watercourses and rills may develop in some seasons during very wet periods. [4]Low risk: Sediment rarely seen to move but polluting runoff may enter ditches or water courses.

It is noted that very light soil with low organic matter content on gentle slopes in low rainfall areas can erode more severely than the risk classes indicate and may be as much as two risk classes higher (Defra, 2005).

Land use/crop type

Surface cover (e.g. vegetation, crop residues, erosion blankets) can act to reduce raindrop mass (by shattering raindrops on impact) and velocity (through raindrop interception in the cover). This reduces the erosivity of rainfall. Thus the soil is most vulnerable when soil surface cover is sparse, especially at times of the year when intensive rainfall events are expected. Plants shelter and fix soil with their roots. They reduce the energy of raindrops through canopy interception. Vegetation can form a physical barrier to flow, and the spatial distribution of plants along a slope can reduce sediment runoff and slope length (Zuazo, 2008). Crops can increase soil aggregate stability and cohesion, and improve water infiltration (Morgan and Rickson, 1995).

Some crop types/land uses are associated with higher rates of erosion than others (Table 6;

Table 7; Figure 1). Evans (2002) identified crop types most prone to soil erosion by water, stating most erosion occurred on winter and spring cereals, sugar beet, horticultural crops and potatoes. However, rates of erosion are twice as high in market garden crops, maize, ley grass and hops. Rates are significantly higher in sugar beet and potatoes (Table 8). Long term grass leys (>3 years) will help improve the organic matter content and aggregate stability of the topsoil, which improves structural resilience and reduces erosion risk when brought back into cultivation (Defra, 2005).

Table 6 Erosion risk categories for different crops / land uses (Defra, 2005).

Highly susceptible land use to be avoided	Moderately susceptible land use can be carried out with care	Less susceptible land use that may reduce erosion

Late sown winter cereals Potatoes Sugar beet Field vegetables Outdoor pigs Grass re-seeds Forage maize Out wintering stock Grazing forage crops in autumn or winter Horticultural crops Hops Soft fruit Orchards	Early sown winter cereals Oilseed rape – winter and spring sown Spring sown cereals Spring sown linseed Short rotation coppice/Miscanthus	Long grass leys Permanent grass Woodland (excluding short term coppice)

Table 7 Land use factors that affect erosion risk.

Crop / land use	Factors that influence erosion risk
Oil-seed rape	Planted close together and covers ground very quickly: low risk. Often grown on heavy, non-erodible land.
Maize	Planted about 0.8 m apart and takes up to 3 months to provide adequate crop cover: high risk Row crops grown up / down slope can increase concentrated flow and rill formation: High risk Late harvesting (October-November) and post-harvest compaction: high risk Forage maize, prepare seedbeds to avoid compaction and wheelings, establish winter cover crop, rough plough immediately after harvest to prevent runoff and erosion. 
Winter cereals	Drilling date, date of attainment of a sufficiently protective crop cover and the timing of rainfall are critical in assessing erosion risk (Boardman and Favis-Mortlock, 2014). Reach a crop cover value of about 30% in 2 months, which provides adequate protection against erosion. However, the 2 months prior to this can coincide with the wettest months (southern England): moderate risk  Newly planted winter cereal fields on the South Downs generally require about 30 mm of rainfall over a two-day period for runoff to occur (Boardman, 2013)
Potatoes	Ridging and row formation can channel surface runoff. High risk. Irrigated crops are particularly at risk of erosion if over-application of irrigation water occurs, or if an unexpected rainfall event follows soon after irrigation (Defra, 2005). High risk

Crop / land use	Factors that influence erosion risk
	Post harvest compaction: high risk 
Field vegetables e.g. asparagus	Fine seedbed preparation: High risk Long time to establishment following sowing: High risk Irrigation (when too much water is applied): High risk Harvest (particularly in winter when soil wet and vulnerable to compaction) can lead to runoff and rill formation: High risk Long periods of bare soil: High risk Favour light, sandy, erodible soils: High risk 
Crops grown in polytunnels, under plastic cloches etc.	Excessive rainfall concentrated at sides of covers – risk of concentrated flow paths, rilling. High risk. 
Fruit orchards	At highest risk of erosion when area between row crops is left bare and over autumn and winter when no tree canopy to intercept rainfall. Between rows is also often compacted and so infiltration rates are reduced leading to higher risk of surface runoff. Reduce risk by establishing grass or mulching between rows and orientating rows across slopes and try to avoid long row lengths. Avoid over irrigating. Grubbing out plants/trees can leave compacted, rutted (focus water) soils which are susceptible to erosion. 
Sugar beet	Establishment of the crop in spring can leave the land uncropped over the preceding winter: erosion risk moderate to high Tendency to grow sugar beet on lighter soils that are more susceptible to soil erosion by both wind and water. Roughening surface or applying mulch cover can reduce wind erosion risk. Fine seedbed preparation for precision seeding increases risk of surface sealing/capping, increasing runoff and subsequent risk of erosion. Relatively open nature of the crop during spring and summer period (April to June) leaving land exposed. Late harvests (late autumn or early winter) increase the risk of harvesting when soil conditions are inappropriately wet leading to soil compaction and increased risk of erosion from surface runoff. Bare rutted surfaces left after harvest also increases vulnerability to erosion. 

Crop / land use	Factors that influence erosion risk
Field beans	According to PGRO (2015), winter beans are suited to heavier land that is difficult to work in the spring. Land is often ploughed in the autumn to allow natural weathering over winter: but this leaves the soil exposed to erosive winter rainfall. Beans can tolerate a cloddy seedbed which is less susceptible to erosion than a fine tilth. Early drilling can lead to higher yields, but might incur compaction if the ground is still wet. Rolling is used to depress stones that could damage the combine during harvest, but this might lead to compaction and generation of runoff. Pulses leave a residue of up to 100 kg N ha^{-1} that protects the soil from rainfall as well as ensuring good establishment of cover in the following crop.
Peas	Soil compaction caused by number of passes across the land, weight of vehicles and accessing the land when conditions are too wet to support the weight of the vehicle: High risk Peas do not offer much protection against soil erosion early in the season. Planted April to late May in UK. Fields with excessive slope should be avoided as high risk of soil erosion because of lack of cover in the early stages. Can mean no crop cover in autumn.
Outdoor pigs	Removal of vegetation and compacted ground caused by animal trampling and tractor ruts can lead to generation of runoff on sloping ground, and detachment and transport (erosion) of soil (Evans, 2004).

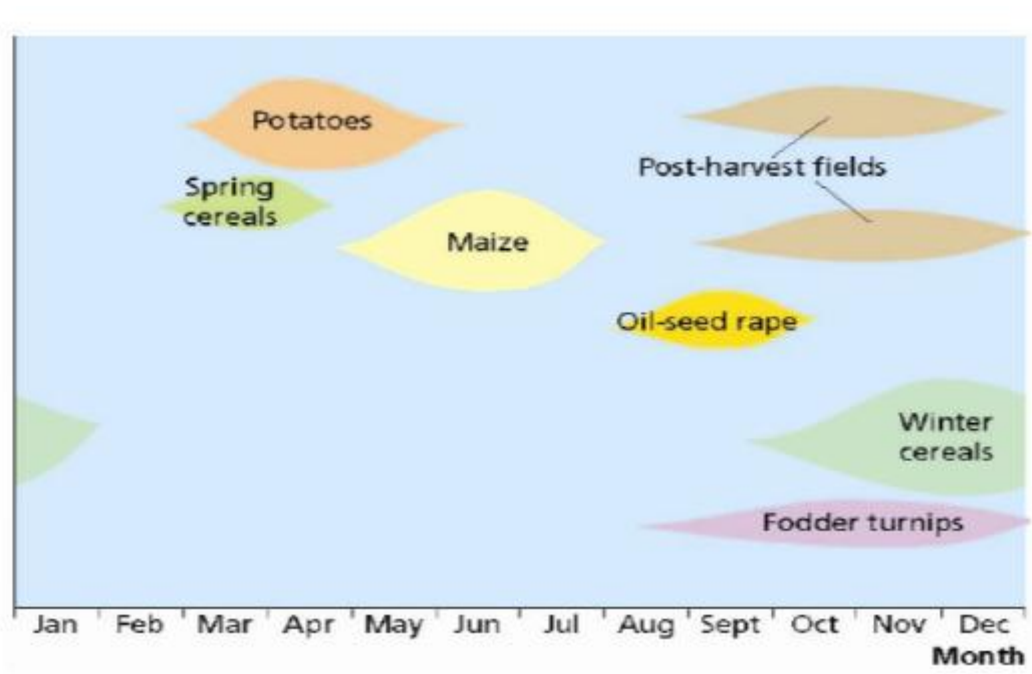

Figure 1 Relative erosion vulnerability for selected crops. Length of shape along x-axis represents the time period during which soils are vulnerable to erosion. The y-axis (and thickness of the shape) represents relative risk. Highest risks are associated with periods when the shapes are thickest (Boardman, 2013)

Table 8 Rates of erosion in soils drilled to different crops (from National Soil Monitoring Scheme, 1982-1986 (Evans 2002); adapted from Boardman, 2013).

Crop	% national crop area	% occurrence erosion	total % occurrence erosion	Mean erosion rate (Mg ha^{-1} yr^{-1})	Median erosion rate (Mg ha^{-1} yr^{-1})	Risk of erosion[1]
Winter cereals[2]	60.2%	43%	26%	1.85	0.68	1 field in 42
Spring cereals[3]	13.6%	12%	2%	1.75	0.71	1 field in 34
Oilseed rape	5.2%	2%	0%	1.92	0.30	1 field in 100
Temporary grass (Ley grass)	4.8%	0%	0%	4.09	1.14	1 field in 32
Sugar beet	4.4%	18%	1%	3.04	0.92	1 field in 7
Potatoes	3.2%	11%	0%	2.53	1.01	1 field in 10
Market garden	3.1%	6%	0%	5.08	1.47	1 field in 14
Peas	1.3%	1%	0%	1.21	0.91	1 field in 38
Bare soil/fallow[4]	1.1%	2%	0%	1.61	0.27	1 field in 21
Field beans	0.9%	0%	0%	0.47	0.22	1 field in 71
Kale	0.6%	1%	0%	2.10	1.41	1 field in 24
Maize	0.4%	2%	0%	4.48	1.00	1 field in 7
Hops	0.1%	1%	0%	3.92	1.01	1 field in 6
Other[5]	1.1%	3%	0%	2.67	1.07	1 field in 11

Notes: [1]Risk of erosion according to crop type (after Evans and Jaggard, 2003, in Evans, 2005); [2]Dominantly wheat, but also barley and to a lesser extent oats and triticale; [3]Predominantly spring barley; [4]Soil surface cultivated but not drilled or rough fallow; [5]Crops include soft fruit, root crops for stock feed, strawberries, orchards, and linseed.

The level of risk can be linked to:

- Rainfall received and subsequent soil moisture content (Figure 2b and 2d)
- timing and number of land preparation operations (Figure 2c)
- time of crop establishment (in terms of season and duration): Boardman and Favis-Mortlock (2014) point out the difficulties in timeliness of planting crops in the autumn to reduce erosion risk, when onset of rainfall is difficult to predict and cooler temperatures over winter limit the development of a protective crop cover. These authors model the 'window of opportunity' for erosion, comprising the relationship between drilling date, date of attainment of a sufficiently protective crop cover and the timing of rainfall. Of these three factors, only the date of drilling can be chosen by the farmer. The date of attaining a sufficiently protective crop cover can only be predicted approximately. The timing of rainfall cannot be predicted. Thus, erosion control advice to farmers, which is based on choice of date of drilling to minimize erosion during the 'window of opportunity', is both difficult to formulate and likely to be ineffective. Sites at risk of erosion need to have better thought-out mitigation measures in place, rather than relying on a fortuitous temporal pattern of autumn and winter rainfall to minimize the risk of erosion.
- canopy / crop cover (Figure 2a); For example, "Maize is susceptible to soil erosion since ground cover is slow to develop after sowing, and the soil surface can be poorly protected until mid-summer" (Defra, 2005).,
- harvest timing and techniques used e.g. spatial extent and weight of harvesting equipment creating compacted ground, susceptible to runoff generation and soil erosion on sloping fields.

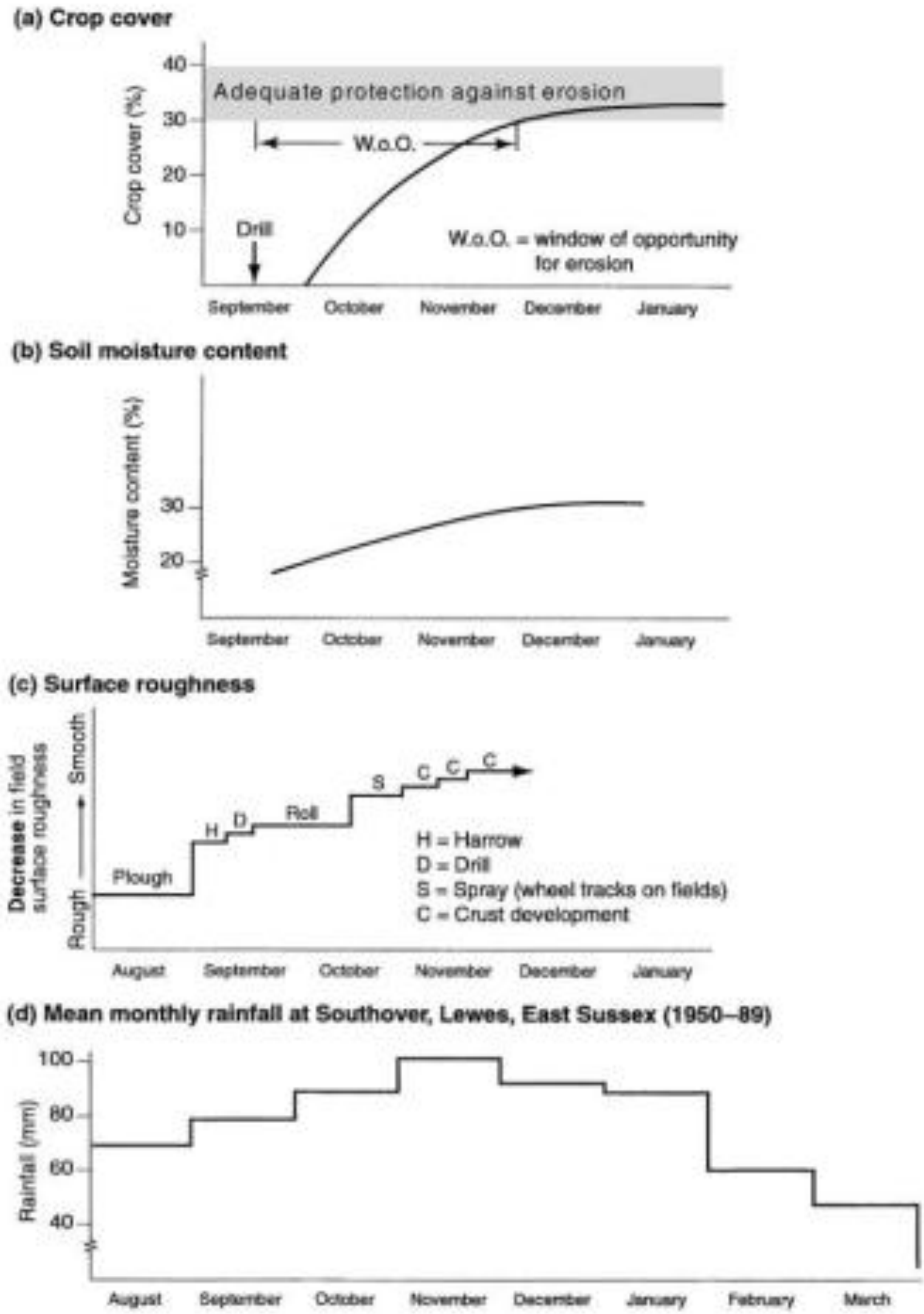

Figure 2 Effect of timeliness of cropping operations on erosion risk (from Boardman, 2003).

Given the number of variables involved, annual soil loss from crops on similar soil textures can be quite variable (Morgan, 1986; Table 9). For example, annual soil loss from cereals on sandy loam soils varied between 0.6 to 24 t ha^{-1}.

Table 9 Range of average annual soil losses in the Silsoe area, Bedfordshire (Morgan, 1986).

Land use and soil type	Range of average annual soil loss (t ha^{-1})
Bare sandy loam soil	10 - 45
Cereals on sandy loam soil	0.6 - 24
Cereals on chalky soil	0.6 - 21
Cereals on clay soil	0.3 - 0.7
Grass on sandy loam soil	0.1 - 3
Woodland on sandy loam soil	-0.01

Values are for slopes of 7 to 11 degrees, except for woodland where the slope was 20 degrees.

It is important to estimate the extent of crop hectarage too if estimates of erosion risk at the national scale are required. Although some crops can be associated with extremely high erosion risk (e.g. asparagus), their hectarage is limited at the national scale. According to Boardman (2013) the majority of eroded fields are in winter cereals because of the area that the crop covers (60% national cropped area).

Some concern has been raised anecdotally as to the increasing area under maize (considered to be an erosive crop; Figure 1; Boardman et al., 1996; Boardman and Favis-Mortlock, 1993), particularly in the wetter parts of the country to produce the biomass required as feedstock for newly built anaerobic digestion plants (http://www.theguardian.com/environment/georgemonbiot/2014/mar /14/uk-ban-maize-biogas). Dr. Stephen Marsh-Smith, executive director of the Wye and Usk Foundation has blamed the advent of anaerobic digestion plants and the increasing area of maize now being grown to feed AD plants. “We are not actually farming for food anymore, we are farming to fill AD plants and that’s causing run-off. Farmers are planting [maize] crops inappropriately on sloping ground and they are not taking the proper precautions to minimise soil erosion.” (http://www.fwi.co.uk/arable/welsh-farmers-warned-over-soil-erosion.htm).

Data from UK Agriculture (2015) and the Maize Growers Association (MGA) show an increasing trend in the area under maize (forage, grain and biogas; Figure 3). Limagrain (2015) estimated that in 1990, just 33,000 ha of maize was grown in the UK. The introduction of earlier maturing varieties meant the area more than tripled to 105,000ha by the year 2000. The arrival of even earlier maturing varieties ‘together with the recent expansion in use of maize for anaerobic digestion (AD) means that the area today has almost doubled again to around 200,000ha of maize grown in the UK’. Their website states that ‘large areas of maize are needed for an AD plant, for example a 500KW anaerobic digester requires approximately 220ha of maize silage’. They also acknowledge the potential threat to soil structure: ‘Varieties should be selected which will reach maturity early enough to allow harvesting before wet weather sets in to avoid soil structure damage.’

DEFRA statistics indicate that 195,801 ha of maize were grown throughout the UK during 2013 (https://www.gov.uk/government/uploads/system/uploads/attachment data/file/ 251222/ structure-jun2013prov-UK-17oct13a.pdf). This area represents a 24% increase on the 157,718 ha area grown in the previous year. The Maize Growers Association (pers.comm.) using Defra data estimate that in 2014, of the 196,000 ha in maize in the UK, 182,000 ha were grown in England. Of this, 135,000 ha were grown for forage, 29,000 ha for biogas and 7,000 ha for grain (Defra statistics). The MGA assume that the bulk of 2013’s increase in area is the result of maize being grown for biogas and as a replacement for mainstream arable crops, which were not established in the autumn of 2012 due to the poor weather.

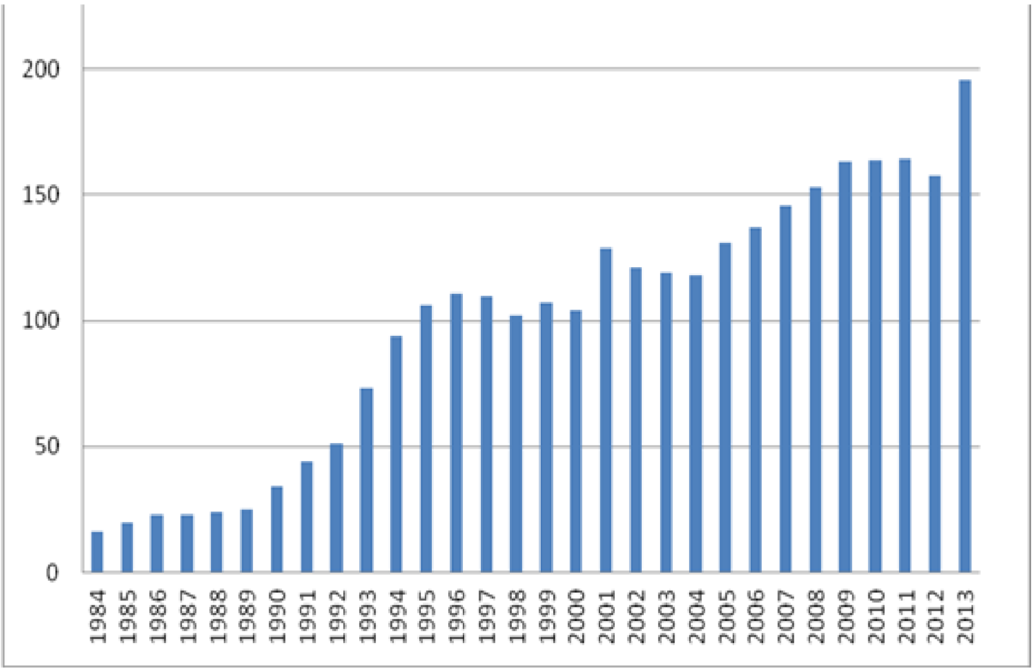

Figure 3 Area (000 ha) under maize (forage, biogas and grain) in the UK (Source: Maize Growers Association, *pers. comm*.)

According to climate projections (UKCP09; Jenkins et al., 2009), future risks are posed by increased winter precipitation (in the range of +10 – 30% over the majority of the country) and increases in precipitation on the wettest day of the winter. Wetter soils, with low crop cover over winter will therefore be at greater risk to erosion. Similarly, heavy rain days (rainfall greater than 25 mm) over most of the lowland UK are expected to increase by a factor of between 2 and 3.5 in winter, and 1 to 2 in summer by the 2080s under the medium emissions scenario. Other trends include the use of (rainfall-shedding) polytunnels to cultivate strawberries, potatoes and asparagus are increasing the risk of damaging runoff (Sarah Olney, CSF Officer, pers. comm.). Trends in global warming (as described by Jenkins et al., 2009) could encourage a change in crops with the potential to increase erosion risk (Mullan, 2013; Mullan et al., 2012; Boardman and Favis-Mortlock, 1993). This might include maize substituting for cereals, cereals grown where grass is currently and length of growing season may encourage farmers to overstock (Evans, 1990).

Land management

Land management practices directly affect soil erodibility and the degree of surface protection from the crop. Incorrect settings for applications of irrigation (e.g. large droplet size and excessive application rate) can cause soil surface sealing and capping (especially on fine sands and silty soils), as well as generation of potentially erosive runoff.

The timing of field operations will affect erosion risk, including seedbed preparation and harvesting (Figure 2; Boardman and Favis-Mortlock, 2014). Fields of the most vulnerable soil should be prepared after less 'risky' fields have been cultivated (Defra, 2005). This recommendation is based on the idea that if you leave preparing the most vulnerable land until last it give the land more time to warm up, so reducing the vulnerable time between seedbed preparation and establishment of a protective crop cover. Stubble and chopped straw can be left over winter to reduce erosion. Seedbed preparation should be timed to reduce erosion risks by leaving the soil surface protected as long as possible. Soils are most vulnerable to erosion when a fine seedbed has been prepared, but a crop cover has not yet developed. Ideally, a minimum ground cover of 25% is required by early winter (November) to be effective at reducing erosion risk. This generally means sowing not later than mid/late September (Figure 2a; Defra, 2005). Where spring sown cereals follow a late harvested root crop, it is recommended that tined cultivations are carried out as early as possible after harvest to minimise erosion from the bare rutted surface (Defra, 2005). Early-drilled spring crops are more at risk of erosion than later-drilled crops because of slow germination and emergence in colder temperature. Defra (2005) recommend to harvest the most vulnerable soil first, so that the land is accessed under optimal conditions. However, this is not always practical or economic if crops are not in optimum condition for harvest. Also, changes in weather patterns associated with climate change will affect the appropriate timing of field operations. Increased winter precipitation may make springtime operations even more problematic as soils remain wetter for longer, so restricting the window of opportunity when land can be accessed without incurring compaction. Soil left bare and rutted after root crop harvesting is particularly susceptible to erosion as ruts focus water into erosive channels. This land is best given a tined cultivation or rough ploughed as soon as practicable.

The degree of soil disturbance is determined by the tillage and cultivation implements used and the number of passes made. Deep ploughing should be avoided on erosion prone soils as it will bury organic matter at greater depths and increase the risk of causing a compacted layer at plough depth (plough pan). Shallow ploughing keeps organic matter near the surface, increasing surface aggregate stability. The use of conventional tillage practices, especially those used to produce fine seed beds for valuable crops such as winter wheat, potatoes and carrots, produce an abundant source of small, lighter, particles that are vulnerable to erosive forces. Redeposition of these fines on the soil surface can result in surface sealing and capping, which can reduce the infiltration capacity of a soil and increase the risk of surface runoff.

The area of reduced tillage (associated with less soil disturbance and the retention of crop residues and other surface covers) has increased since the mid-1990s (Knight et al., 2012; Figure 4), but this is in response to increasing fuel prices, rather than the need to reduce erosion rates associated with conventional tillage practices.

Tillage translocation of soil particles is a redistribution process that results from gravity and the disturbance of soil by farm implements. It generally results in soil loss in a down slope direction. Soils on convex slopes are often transferred to either a concave slope position or to the field boundary. Factors that control tillage erosion are shown in Figure 5 and include slope gradient and variations in slope curvature, tillage operation (including plough depth, direction and speed) and tillage implementation type (Owens et al., 2006).

Soil can also be lost from a field due to its co-extraction on crops during harvest, principally root vegetables such as potatoes, carrots, sugar beet and onions. Soil can also be removed when attached to agricultural farm vehicles and implements. Ruysschaert et al. (2004) provides an overview of factors that contribute to soil loss through co-extraction and on machinery (as reviewed in Owens et al., 2006). The most notable of these factors includes soil wetness and high clay content which increase the adhesion of soil to crops and farm equipment. Harvesting techniques and weather conditions also play a role in the amount of soil extracted with the crop. Drier summers forecast in climate change scenarios will lead to drier soils at harvest time with an associated reduction in the risk of co-extraction of soil on root vegetables.

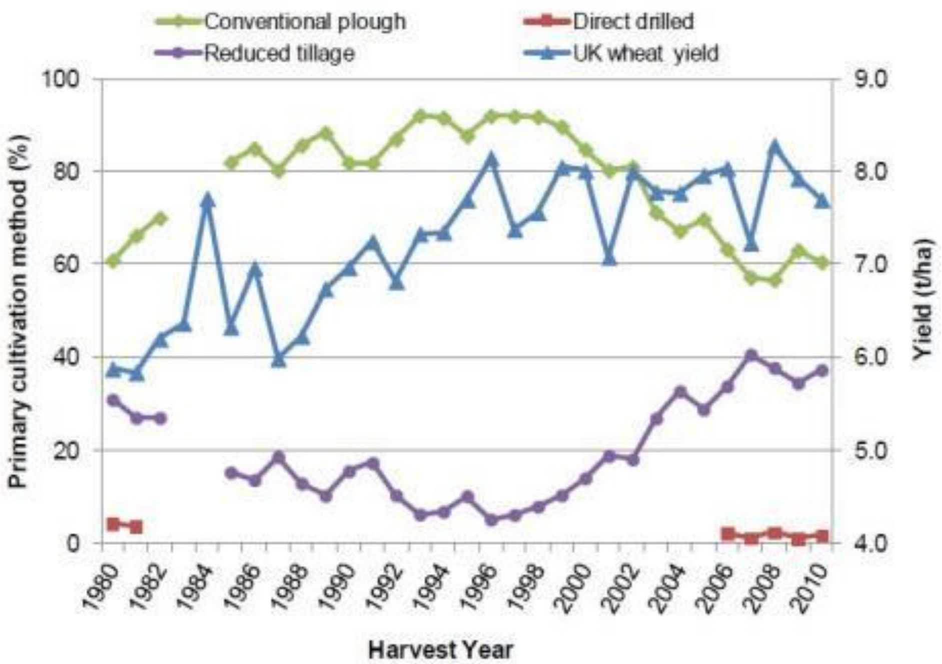

Figure 4 Proportion of winter wheat area established using various establishment methods in UK (Knight et al 2012).

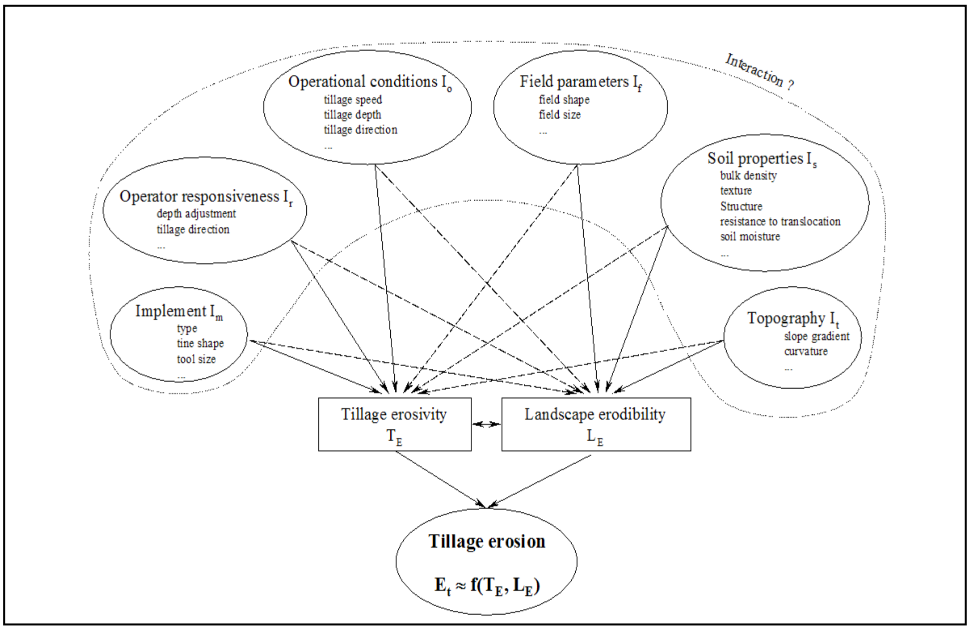

Figure 5 Factors affecting tillage erosion processes (from Van Oost et al., 2006).

While the wider use of four-wheel drive machines has enabled our ability to work more safely on steeper sloping land, where cultivation is involved there is an attendant risk of soil erosion particularly if the soil is weakly structured.

Livestock management factors such as intensity (number of animals per hectare) and duration on land also contribute to erosion risk. Evans (1990) argues that global warming may encourage farmers to increase animal numbers, leading to over-stocked land which can be poached / compacted by cattle. This is associated with reduced infiltration and generation of surface runoff. Despite the low risk of erosion from grass (see Table 6), there is some evidence of accelerated erosion rates on grassland through sheet erosion and wash (Bilotta et al., 2008; 2010; Brazier et al., 2007). Even on grasslands (associated with low risk of erosion) re-seeding is not appropriate on vulnerable land, because of the limited cover: surface pasture improvements should be considered instead. Compaction should also be removed before sowing the grass and precautions should be taken to avoid future compaction. Grazing of recently-re-seeded land can cause damage to both the soil and the crop and therefore increase erosion vulnerability (Defra, 2005). Wetter winters predicted by climate change scenarios will increase the risk of compaction as soils remain wetter for longer periods.

Maintaining soil organic matter content is important to reduce erodibility (Wischmeier and Mannering, 1969). Bellamy et al. (2005) showed a carbon loss of 0.6% between 1978-2003 based on all soils in England and Wales. Emmett et al (2010) confirmed a carbon loss from arable and horticultural field sites in Great Britain, suggesting that some of this loss was due to the increase in intensification of farming practice and/or deeper ploughing depths, diluting organic rich top soil with less organic rich subsoil. There is concern that losses of soil organic matter through decomposition are likely to exceed levels gained from increased plant growth, thus adding to atmospheric CO_2 levels and the greenhouse gas effect and to lower levels of soil organic matter (Defra, 2005). The impact of climate (in combination temperature, precipitation and evaporation) could cause significant losses of soil organic matter from most mineral soils in the UK, with the greatest losses expected in south east England, where rates of temperature increase are greatest (Defra, 2005). Cooper et al (2010) modelled the effects of climate change on carbon losses and concludes that there will be ‘small changes in soil C content in England and Wales over the period 2010 – 2080’.

Field drainage is also an important factor affecting erosion risk. Soils that drain quickly, either naturally or through artificial drainage systems are less likely to be affected by erosion from overland flow. However, rainfall (intensity and amount) will determine the generation of surface runoff, commonly as saturated overland flow or less frequently where rainfall intensity is in excess of infiltration rate (especially for soils prone to surface capping and crusting), even on gentle slope gradients. Given the predicted higher intensity rainfall events, the latter process may become more common.

Soil protection measures specifically used to control erosion such as buffer strips and reduced tillage are expected to reduce erosion risk (Rickson et al., 2010; Defra SP1601). However, much of the evidence is based on plot studies rather than field or catchment scale measurements. Maetens et al. (2012) reviewed 101 studies that considered the effectiveness of soil and water conservation techniques across Europe and the Mediterranean. Each study considered annual runoff and annual soil loss. The data represented 353 runoff plots from 103 plot-measuring stations. Comparisons of soil loss rates on cropland without and with the application of soil and water conservation techniques showed that these techniques lowered exceedance probability of tolerable soil loss rates by ca. 20%. Maetens et al. (2012) found that crop and vegetation management techniques (buffer strips, mulching and cover crops) and mechanical techniques (geotextiles, contour bunds and terraces) to be generally more effective than soil management techniques (no-tillage, reduced tillage and contour tillage). Soil and water conservation techniques were generally less effective in reducing runoff than in reducing soil loss. The data showed that no-tillage and conservation tillage became less effective in reducing runoff over time, but soil loss protection was retained. In the case of contour ploughing, this has limited applicability in the UK anyway, where slopes are often complex, and failure to follow the contour can result in catastrophic breakthrough and erosion (Defra, 2005).

Newell Price et al. (2011) reported on 27 mitigation measures that had an effect on water pollution by sediment but not necessarily soil erosion per se. SEPA (http:// apps.sepa.org.uk/bmp/Default.aspx) listed 61 soil erosion and sediment mitigation methods. Rickson (2014) combined these reviews and other information an identified over 73 erosion control mitigation methods with potential use in the UK. Quantified effectiveness data was found for only 43 of the 73 mitigation methods. The majority of reviewed studies on erosion mitigations in the UK relate to the effectiveness of reduced cultivation (23 studies); buffer strips – riparian (22 studies) and in-field (13 studies); and reduced; reduced grazing intensity (17 studies); and effect of managing organic matter content (2 studies), although their effectiveness and reliability is often uncertain (Rickson, 2014). The erosion control effectiveness of these different mitigation measures is shown in Figure 6.

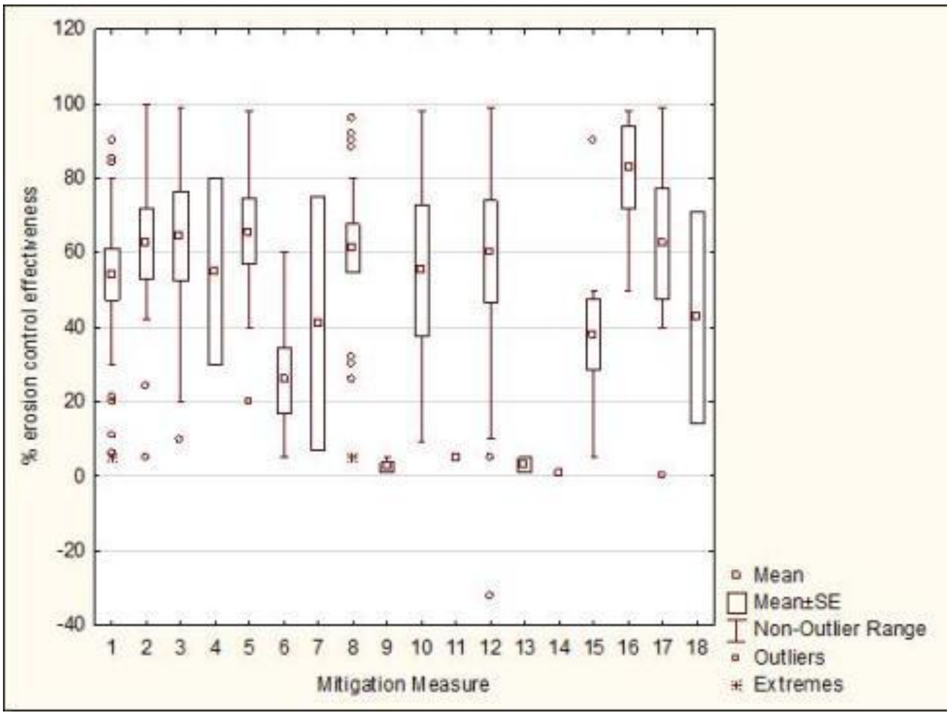

Figure 6 Effectiveness of selected soil erosion mitigation measures (expressed as %; Rickson, 2014).

Key (numbers in brackets are number of studies reviewed):

Riparian buffer strip / zone (20)	Cultivate and drill across the slope (6)
Establish in-field grass buffer strips (11)	Leave autumn seedbeds rough (1)
Establish edge-of-field buffer strips (8)	Tramline management (11)
Convert arable land to grassland (2)	Maintain and enhance soil organic matter levels (2)
Mulching / crop residue management (9)	Allow field drainage systems to deteriorate (1)
Cover cropping (7)	Reduce grazing intensity (13)
Strip cropping (2)	Constructed waterways (4)
Adopt minimal cultivation systems (21)	Infiltration / detention / retention basins, ponds and wetlands (6)
Cultivate compacted tillage soils (2)	Contour bund (2)

Maetens et al. (2012) suggests variability in effectiveness indicates the effect of environmental factors; however, lack of detailed information prevented environmental factors from being specifically identified. Thus the robust evidence of the true effectiveness, feasibility and uptake of these measures is uncertain and needs further research (ASC, 2013). This is being addressed in an on-going Defra research project (SP1318) on 'Scaling up the benefits of field scale soil protection measures to understand their impact at the landscape scale'. The aim is to demonstrate the level of effort required to control erosion to an acceptable degree (i.e. 50% of the current baseline). The project will also consider constraints on the uptake and effectiveness of measures. These will include:

- *Effectiveness Rules* which specify any physical constraints moderating effectiveness for individual fields
- *Evidence Rules* which limit the application of measures to environment ranges for which there is empirical evidence of a measure effect
- *Applicability Rules* where physical and system incentives or constraints affect the applicability of measures to individual fields or rotations
- *Measure Compatibility Rules* which reflect whether combinations of measures can be implemented together on individual fields
- *Implementation Limit Rules* which specify the maximum implementation of individual measures for individual fields

Socio-economic factors

Boardman (2013) suggested that it is important to distinguish between ultimate and underlying causes, and proximal and immediate causes of soil erosion. He defined the ultimate causes as 'socio-economic drivers of erosion', for example high value crops grown on inappropriate land (highly vulnerable to erosion), where the short-term economic return was seen as 'being worth the risk'. Boardman (2013) and others argue that erosion is largely driven not by any vagaries of the weather, but by the political and economic incentives which determine the way the farm is managed. This might include pressure from wholesalers/supermarkets to supply a given quantity and quality of crops on demand, so putting pressure on farmers/land managers to grow unsuitable crops and/or to access the land when conditions are less than ideal, so that their contract with their suppliers can be honoured.

Boardman and Evans (2006) argue that erosion in Britain is a function of agricultural and land use policy. One example of this is in 1952 farmers in East Anglia were encouraged by the Ministry of Agriculture to grow sugar beet. By 1989 the area under this crop reached 200,000 hectares (MAFF, 1989). However, land under sugar beet is more vulnerable to erosion than when under cereals. The introduction of this crop was associated with increased incidence and severity of erosion (Evans, 1990). The same occurred when large areas of grassland on the South Downs were ploughed up for food production as part of the ‘Dig for Victory’ campaign in the Second World War. Some of the highest erosion rates ever recorded in the UK were on these slopes due to steep slopes, vulnerable soils and erosion-vulnerable cropping practices (Boardman et al., 1994). In recent years maize and potatoes have been introduced into high rainfall areas of the country, which are not suitable to arable farming (Boardman, 2013). An increase in maize production and outdoor pig production has also contributed to an increase in erosion from this land. Changes in erosion risk in the future will be linked to land use and climate change. Land use changes are driven by market price and policies. More maize may be produced if we get warmer summers.

Intensification of agriculture over the past 50 years and its focus on highly mechanised food production has led to significant increases in soil erosion (Evans, 2013). In the 1980s and 1990s few measures were in place to control erosion and government attitudes were that it was a localised and on-farm problem (Boardman, 2013). Changes were brought about by a decline in crop price, criticism of uncontrolled farming subsidies and over-production of food crops, the growth of an effective environmental movement and an interest in the sustainable use of soils (Boardman, 2013). European funding moved away from subsidising over-production and moved towards funding to support environmentally-friendly approaches (e.g. reduced stocking densities, the introduction of set aside in 1992). Landscape protection, such as the Environmentally Sensitive Area scheme, was introduced in the late 1980s.

Increased awareness of the need to control off-farm impacts of agricultural erosion lead to the introduction of the European Water Framework Directive in 2000 with the target of all inland and coastal water bodies reaching good quality or “good ecological status” by 2015. Payment schemes such as the Single Payment Scheme, required consideration of risk of soil erosion and runoff to be made and prevented as a requisite for payment. Common Agricultural Policy reform (CAP) continues to recognise the importance of good soil management (Defra, 2014). Initiatives, such as Catchment Sensitive Farming Delivery Initiative (CSF), have been introduced to target diffuse pollution issues aimed at meeting the Water Framework Direct. Initially 40 priority catchments were identified under CSF, rising to 66 in 2013. This initiative provides free advice and training as well as providing a capital grant schemes to help farmers in priority areas manage their land more appropriately.

2.2.2 Factors affecting soil erosion by wind

In comparison to water erosion, the area of England affected by wind erosion is small (Chappell and Warren, 2003) and seems to have declined in recent years compared to 1950s and 1960s (Evans and McLaren, 1996; Boardman, 2013). However, locally, the rate of erosion can be greater by wind than by water. This is partly due to the fact that wind erosion is likely to impact on a whole field area while erosion by water is limited to where the water flow is concentrated (Table 10; Evans, 1996; Owens et al., 2006). Owens et al. (2006) identified wind erosion as a locally important hazard with rates in some areas ten times that of the mean wind erosion rate for England and Wales. Despite considerable anecdotal evidence there is a dearth of quantitative evidence to support wind erosion and deposition rates (Chappell and Thomas, 2002).

Table 10 Typical erosion rates for wind and water, for England and Wales (Owens et al 2006).

	Wind	Water
Typical erosion rate range (t ha^{-1} $year^{-1}$)	0.1 - 2.0	0.1 - 15.0
Land use affected	Arable, upland, some pasture	Arable, pasture, upland
Exported off field	yes	yes

Blowing can result in the loss of topsoil, seeds, seedlings and fertiliser and cause damage by abrasion to plants. Yields of re-sown crops are often reduced through late establishment and development. Evans (1996) reported that the value of the crop in wind eroded fields is often higher than that affected by water erosion, so the on-site cost of wind erosion is often greater (five times or more) than when fields suffer from water erosion. The national annual cost of agricultural inputs lost because of wind erosion in the mid-1980s was estimated at £210,000 and the loss of crop at £705,000: equivalent values for water erosion were £285,000 and £940,000 respectively, due to the larger area affected by water erosion (Quine et al. 2006).

Uplands are at risk of wind erosion, especially on bare soils and peat, where overgrazing can expose soil and peat (McHugh et al, 2002). Critically, erosion of peat will lead to a loss of carbon back into the atmosphere. Wind erosion can be a problem on arable farms in East Anglia. Farmers in this area expect moderate damage to crops from wind erosion once every three or four years and severe damage once in 10 years (Chappell and Thomas, 2002). Severe soil erosion by wind has also been reported in the Vale of York and Pickering (Radley and Sims, 1967), Lincolnshire (Robinson, 1968), the Fens (Pollard and Miller, 1968) and Nottinghamshire (Wilkinson et al., 1968). A series of major erosion events occurred in these areas in 1960s-1980s (Evans and McLaren, 1996). Such events are less common now due to expansion of winter cereals, loss of erodible peat soils and the adoption of mitigation measures (Boardman, 2013). Other areas observed to be affected by wind erosion are listed in Table 11.

Table 11 Incidence of wind erosion in England and Wales (and Scotland).

Area	Reference	Comments
East Anglia	Sneesby (unpub.); Spence (1957); Wilkinson *et al*. (1968); Pollard and Miller (1968); Chappell and Thomas (2002); Böhner *et al*. (2003); Chappell and Warren (2003); Look East (BBC TV; March 2006); Boardman and Evans (2006);	Fenland peat

Area	Reference	Comments
Suffolk	http://www.deere.com/en_GB/publications /the_furrows_aut04/page3_aut04.pdf	Peat and sandy soils
Lincolnshire	Robinson (1968); Wilkinson *et al*. (1968)	Areas mapped after erosion events in 1968
	Boardman and Evans (2006)	Former windblown deposits
West Midlands	Boardman and Evans (2006)	Fine sandy soils
East Midlands	Wilkinson *et al*. (1968)	Areas mapped after erosion events in 1968
	Boardman and Evans (2006)	Fine sandy soils
English Midlands	Quine and Walling (1991)	Soil loss by wind erosion inferred from ^{137}Cs measurements
East Yorkshire	Radley and Sims (1967)	
Vale of York	Boardman and Evans (2006); R. Palmer (pers. comm.)	Former windblown deposits
Tyneside	http://www.bbc.co.uk/tyne/content/articles/2005/07/01/ coast05walks_stage1_walk.shtml	Sand dune erosion
North Pennines	Warburton (2003)	Upland peat soil
Scotland – Buchan and Banff coastline, Murray Firth, the islands (on the "machair")	Boardman and Evans (2006)	Sandy soils, especially when exposed by ploughing
North Wales	Wiggs *et al*. (2002); Bailey and Bristow (2002)	Sand dune erosion / migration

Wind is the forcing agent and other factors (including physical and chemical soil properties, soil surface roughness/micro-topography, land cover) determine particle release potential. The seasonal cycle of crop development, agricultural activities and climate determine the seasonal cycle of wind erosion (Bärring et al., 2003). Vulnerable times are when the soil is dry and there is limited land cover, also following soil preparation that destroys any soil surface crust. Factors that affect vulnerability to wind erosion are detailed in Appendices 7.4 and 7.5.

According to UKCP09, severe windstorms around the UK have become more frequent in the past few decades, although not above that seen in the 1920s. There is considerable interest in possible trends in severe wind storms around the UK, but these are difficult to identify, due to low numbers of such storms, their decadal variability, and by the unreliability and lack of representativeness of direct wind speed observations (Jenkins et al., 2009).

Climatic factors

Wind erosivity (Wind velocity, duration of blow, length of fetch and direction): A wind speed of 30-40 km hr^{-1} is sufficient to dislodge particles from the soil and transport them either by saltation, deflation or surface creep. According to UKCP09, severe windstorms around the UK have become more frequent in the past few decades, although no worse than seen in the 1920s.

Rainfall: Drier summers may increase the risk from wind erosion as soils dry out and become friable (Bradley et al., 2005). Peat soils in particular will become more vulnerable to wind (and water) erosion, both through drier conditions (leading to vegetation loss and increased susceptibility to wind erosion), and from extreme rainfall events in the winter.

Soil properties

Wind erosion has mainly been recorded on sandy and peaty soils in the eastern and middle counties of England e.g. East Midlands and East Anglia, and parts of the uplands of England and Wales (MAFF, 1997; Chappell and Warren, 2003; Boardman and Evans, 2006). The Soil Associations at risk of wind erosion in England and Wales are given in Appendix 7.3 (based on data in Evans (1990) and Soil Associations from Mackney et al. (1983). Significant wind erosion is restricted to a relatively narrow range of susceptible soil types. The risk is greatest in spring or early summer on flat or gently sloping land where light textured, bare or sparsely vegetated soil is exposed to strong wind and the surface is dry. The soils most at risk are sands and loamy sands with a high fine sand content, organic sand, sandy and loamy peats and peats. The presence of stones reduces erosion risk to some extent.

Crop type / land use

Vegetation can decrease wind shear stress on an erodible surface by absorbing some of the wind's downward momentum, creating a barrier of slow moving air above the surface, and increasing the threshold friction velocity required to mobilise soil. Some crops are associated with high wind erosion risk. This is because a) they are usually grown on the soils that are susceptible to wind erosion (e.g. field vegetables); b) their cultivation requires creation of a fine erodible seed bed and long periods of exposure of bare soil; and c) their morphology / architecture does not reduce wind speeds close to the ground surface. Roughness imparted to wind flow by vegetation reduces wind speeds close to the ground surface. A plane of zero wind velocity occurs at a height that is equal to about 70% of plant height (Morgan, 2005). This is related to a parameter ‘Z_0’ which varies for different crops (Table 12).

Table 12 Increase in roughness length (Z_0) due to vegetation (from Morgan and Rickson, 1995).

Vegetation type	Z_0 (cm)
Grass	0.2-0.3
Sugar beet	0.4-1.6
Wheat	1.2-3.0
Planted straw strips	2.1
Onions	0.8
Peas	0.4
Potatoes	5.4
Coniferous forest	1.0
Deciduous forest	1.8

An increase in Spring-planted vegetable crops can increase wind erosion.

Landscape features

Increases in wind erosion have coincided with increasing size of fields and removal of wind-breaking vegetation (e.g. hedges, avenues) leaving a landscape more exposed to strong winds and more vulnerable to wind erosion (Bärring et al., 2003).

2.2.3 Estimates of the effects of these factors on soil erosion

To determine how these factors directly affect the incidence of soil erosion requires measurements of actual soil erosion rates in terms of magnitude of events, frequency of events (temporal occurrence) and extent of erosion (spatial occurrence). This will allow analysis of changes in the factors and resulting actual rates of erosion. In Britain, assessment of soil erosion has been principally based on a National Monitoring Scheme (specifically cultivated land in England and Wales) which ran from 1982 to 1986. The Scheme incorporated 17 locations, representing most major soil associations and about 700 km2 of farmland. The scheme used air photographs at 1:10,000 scale to identify eroding fields, which were then field checked, and rills, gullies and areas of deposition measured (wash erosion was not recorded). Subsequent monitoring was undertaken on smaller areas including the South Downs (1982-1991; Boardman 2003). Due to their limited spatial and temporal coverage, the resulting datasets are very limited at the national scale (Brazier et al., 2011; Defra, SP1303). This issue is currently being addressed in Defra funded project SP1311, “Piloting a Cost-Effective Framework for Monitoring Soil Erosion in England and Wales”, but this project is not due to report until 2016.

Dr Bob Evans’ (Anglia Ruskin University) work on approximately 1600 sites has been published and is probably the most extensive dataset available. However, most surveys do not record all the factors that may be contributing to erosion, so estimates of the effects of these factors cannot be made directly. Also, trends in the extent and severity of water erosion in lowland England and Wales cannot be identified with any certainty from the three national monitoring schemes. These were undertaken by the Soil Survey of England and Wales (1982-1986); Agricultural Development and Advisory Service (1989-1994); Soil Survey and Land Research Centre (1996-1998). The first of these surveys covered the largest areal extent and individual locations were much larger in size than any survey. The 3 surveys each identified a different percentage number of fields affected by erosion. The SSEW survey identified 340.4, the ADAS survey 29 and the SSLRC survey identified 44.7 in a single year. Erosion rates were estimated based on field area by SSEW and SSLRC and by net erosion by ADAS (Evans, 2005).

2.2.4 Changes to these relationships in the future

Recent climate change predictions suggest that the UK is likely to experience hotter drier summers, warmer wetter winters and an increased frequency of extreme weather conditions (e.g. heat waves, dry spells and more intensive rainfall events (Defra, 2009).

Precipitation

Projections from UKCP09 (Jenkins et al., 2009) suggest that although annual mean precipitation is unlikely to change by the 2080s, winter precipitation in most of the country might increase by 10 – 30%. Of this winter rainfall, more is projected to fall as heavy rainfall

(>25 mm) with associated increase in erosivity. Summer precipitation will decrease in some regions (e.g. SW England, with a 40% decrease) but this projection is less apparent going northwards, and it does depend on the probability levels used in the forecast.

Under a ''best guess'' rainfall scenario with a 10% increase in winter rainfall, Favis-Mortlock and Boardman (1995) predict increases in annual erosion of up to 150%. Erosion rates for individual years were shown to change in more complex nonlinear ways however, with decreases as well as increases occurring. These could be explained by the interaction of timing of rainfall with changes in the rate of crop growth.

Based on modelled outputs, Cooper et al (2010) suggest a projected increase in spatially averaged erosion by water due to increased winter rainfall, especially in the upland areas of England and Wales. These findings are supported by Mullan et al., 2012 and Defra SP0571 (Modelling the impact of climate change on soils using UK climate projections; 2005). However, this prediction is based on changes in climate only and does not consider how changes in land use or cropping might affect future rates of soil erosion.

Soils

The impact of climate change on soils is reported in Defra projects CC0301 (To investigate the impacts of climate change on soils), CC0375 (The development of a soil properties database for England and Wales for climate change impact studies), SP0538 (The impacts of climate change on soil functions), SP0571 (Modelling the impact of climate change on soils using UK Climate Projections) and SP1601 (Soil Functions, Quality and Degradation – Studies in Support of Implementation of Soil Policy). Under climate change, the anticipated changes in temperature and precipitation are expected to influence the structure and functioning of soils. Changes in physical structure may alter the hydrological regime, by, for example altering the water storage and transmission properties of the soil. This will impact on erosion likelihood. In addition to physical change, chemical and biological processes typically respond to changes in soil moisture and temperature. Of particular interest is the possible change in the soil carbon budget, for example by increased rates of net loss through enhanced breakdown of organic matter. In turn, lower organic matter increases soil erodibility.

Topography

Changes in land use / crop types due to climate change (see below) may bring about changes in field sizes, which in turn may increase or reduce soil erosion risk. A longer slope under larger field sizes will increase erosion risk all other factors being equal. Evidence of this can be seen where old hedgerows have been taken out to increase the size of agricultural the field unit, increasing slope length and thus the contributing area of runoff and accumulation of erosive flow downslope.

Land use/crop type changes

Favis-Mortlock et al. (1991) modelled the effect of increased CO2 levels associated with climate change on winter wheat development and yields, and soil erosion. Mullan (2013) points out the limitations of a number of studies that only consider the direct impacts (changed climate data) on erosion rates, and their failure to factor in the indirect impacts (changing land use and management). Using the Water Erosion Prediction Project (WEPP) model, as well as considering climatic changes, Mullan (2013) investigated the effects of changes in plant biomass and land use (resulting from climate change) on future soil erosion rates. Only the most extreme scenarios revealed the potential for on-site problems of soil erosion. Off-site impacts are likely to become a greater environmental issue in terms of water quality and muddy flooding under a wide range of future scenarios.

Evans (1990) notes that land use and global climate change may change the risk of soil erosion to one of a potential risk. With global warming, the land at moderate to very high risk of erosion is likely to increase from 23.9% to 46.1% of the area of England and Wales, and 126 soil associations are likely to become more at risk. However, 10 arable associations will become less at risk to wind erosion because much of the peat susceptible to erosion will have blown away.

Overall, climate change could both increase the area of land at risk of soil erosion and increase the severity of erosion (ASC, 2013). The risk of soil erosion as affected by changes in temperature, rainfall and wind is likely to increase (Table 1). Defra research SP0571 (Modelling the impact of climate change on soils using UK climate projections) reported on soil erosion risk from both water and wind, including the projected mean erosion rates across England and Wales from 2000 to 2090. Rates of erosion by water are predicted to increase by 0.1 t ha^{-1} yr^{-1} to an average of 0.55 t ha^{-1} yr^{-1} by 2080s (Cooper et al., 2010).

In terms of farmer perceptions, Robinson (1999) found that most farmers believe that the climate has changed in recent years, but impressions of how it has changed are inconsistent. Knowledge of potential future climate change is very limited. The majority of farmers in the two surveys consider erosion to be a minor annoyance to their farming operations which has minimal impact on their past, present or future agricultural land-use policy.

Further discussion of projected climate change and erosion relationships are found in Boardman and Favis Mortlock (1993); Boardman and Favis-Mortlock (2001); Bradley et al. (2005); Mullan (2013); Mullan et al. (2012); Nearing et al. (2004) and Robinson (1999).

2.3 Spatial mapping of erosion risk factors

The aim of this component of the work was to map the erosion risk factors identified in the literature review and whether changes in these factors lead to a change in soil erosion risk, where data are available for England. The maps would show the spatial variation of the factors, as well as how they change over time, by reporting on their values for a number of years. Due to the duration of the present project and accessibility of relevant meteorological and land use data, only 5 years were selected for this analysis: 1969; 1979; 1988; 1997; and 2010. These years were specifically chosen because data on both rainfall (Met Office data) and land use (Edina Agcensus data) could be accessed on these dates.

It should be noted that some factors affecting erosion risk cannot be mapped at the national scale. For example, land management practices will vary at smaller spatial scales (e.g. field by field) and over short time periods (e.g. practices associated with specific crops / time of year). Similarly, the socio-economic factors affecting erosion (Section 2.3.1.6) are difficult to map spatially at any scale.

To check the sensitivity of actual erosion occurrence to each risk factor, the outputs should be compared with the actual spatial distribution of observed erosion. However, national estimates of erosion rates are very limited in spatial (and temporal) extent (Evans, 2005; Brazier et al., 2011; Defra, SP1303). Therefore, results can be compared with existing erosion risk maps (Appendix 7.6). Alternatively, erosion prediction models can be run to predict a) the patterns in soil erosion risk over space and over time; and b) the sensitivity of predicted erosion occurrence to each risk factor. Because measurements of soil erosion at the national scale do not exist, we used an erosion prediction model, the modified Morgan, Morgan and Finney model (Morgan and Duzant, 2008) to estimate the mean annual rates of soil erosion (t ha^{-1} yr^{-1}) associated with the spatial distribution of the factors affecting erosion risk (Appendix 7.7). This approach has been used elsewhere (Scholz et al., 2008; Yang et al., 2003; Mullan (2012), but the limitations of using a model without adequate validation with empirical observations at the national scale are acknowledged by the authors.

For soil erosion by water, data on the factors affecting erosion risk are available and can be mapped:

- the Modified Fournier Index (MFI) of rainfall erosivity
- Dominant Cross-Compliance soil classes
- Mean slope angle (degree)
- Land use / land cover

However, the lack of data on the factors that affect erosion by wind meant it was not possible to produce meaningful maps or charts of changes in risk factors. Severe wind storms are difficult to identify, due to low numbers of such storms, their decadal variability, and by the unreliability and lack of representativeness of direct wind speed observations (Jenkins et al., 2009). Data on soil susceptibility to wind erosion are available, and have been mapped (Morgan, 2005; Appendix 7.6; Figures A1 and A2), but other risk factors are not included. Apart from the work by Quine et al (2006; Figure A5), no reliable / accurate annual index of wind erosivity in the UK was found. This reflects the findings of Kibblewhite et al. (2012; Defra SP1609 Exploring the Priority Area Approach, Final Report to Defra, 2012): "The current method for estimating risk from soil erosion by wind is only partly-developed and requires improved data and validation of modelled outputs" Therefore further spatial mapping and analysis of the factors affecting wind erosion was not undertaken, with agreement from the Project Management team (WebeX conference, 20/02/15).

2.3.1 Rainfall

Annual rainfall is poorly correlated with soil erosion rates (Morgan, 2005). This is mostly due to the relationship between rainfall amount and land use, with the wetter parts of the country supporting grassland systems, a land use associated with a low erosion risk (Table 6). Even so, Nearing et al. (2004) report that in the US, where rainfall amounts increase, "erosion and runoff will increase at an even greater rate: the ratio of erosion increase to annual rainfall increase is on the order of 1.7. Even in cases where annual rainfall would decrease, system feedbacks related to decreased biomass production could lead to greater susceptibility of the soil to erode".

A better indicator of erosion risk is rainfall erosivity, defined as the ability of rainfall to cause erosion. In most soil erosion studies, the calculation of rainfall erosivity is limited due to the lack of long-term time series rainfall data with high temporal resolution (< 60 min) (Panagos et al., 2015) and lack of correlation with measured soil losses (Gabriels, 2006). The most recognised erosivity index, the R factor of the Universal Soil Loss Equation (Wischmeier and Smith, 1978), would be difficult to estimate within the timescale of the present study as it requires the kinetic energy of rainfall from individual storm events in its calculation. In any case, this empirically derived index is poorly correlated with soil loss in regions outside the mid-West USA (east of the Rocky Mountains) where it was developed (i.e. in temperate, maritime climates such as in the UK). (Despite this, recent estimates of erosivity based on the R factor have been attempted (Panagos et al., 2015)). Other indices of rainfall erosivity such as the KE>25 mm hr^{-1} (Hudson, 1971) and the AI_m index (Lal, 1976) are not suitable for England as they were developed for tropical rainfall. As Gabriels (2006) points out there is no commonly accepted index of rainfall erosivity for the UK. Therefore, it was decided to use the Modified Fournier Index (MFI; Arnoldus, 1977) of rainfall erosivity for each year of interest: “the most commonly used index of rainfall aggressiveness” according to Morgan (2005; page 72). This index has been used throughout Europe, notably in the CORINE Soil Erosion Risk and Important Land Resources Project (CORINE, 1992). However, as with many estimates of rainfall erosivity, no validation with measured soil loss has been carried out (Gabriels, 2006). The equation for calculation of the MFI is:

Annual MFI (mm) = Σ (p^2 / P) for all months in the year

Where p^2 = mean monthly precipitation (mm); and P = annual rainfall (mm)

A classification of MFI values is given in Table 13.

Table 13 Classification of Modified Fournier Index values (after Gabriels, 2006).

MFI range	Description
< 60	Very low
60 – 90	Low
90 – 120	Moderate
– 160	High
>160	Very high

To obtain these parameters, a 5 km grid from the Met Office of annual rainfall for the years of interest (1969, 1979, 1988, 1997 and 2010) was used. The selection of these particular years was not to show some temporal trend in rainfall records, but to select representative dry, wet and average years to observe the relationship (if any) between received rainfall, the MFI and soil erosion risk. The grid was resampled to 2 km and then the mean annual and monthly rainfall value for each 2 km grid cell was extracted. Annual rainfall has been classified into 8 classes for ease of map representation, with 100 mm increments from 500 mm to 1200 mm (Figure 7). The process was repeated for all years of interest, and the proportion of each annual rainfall class was calculated for each year (Figure 8).

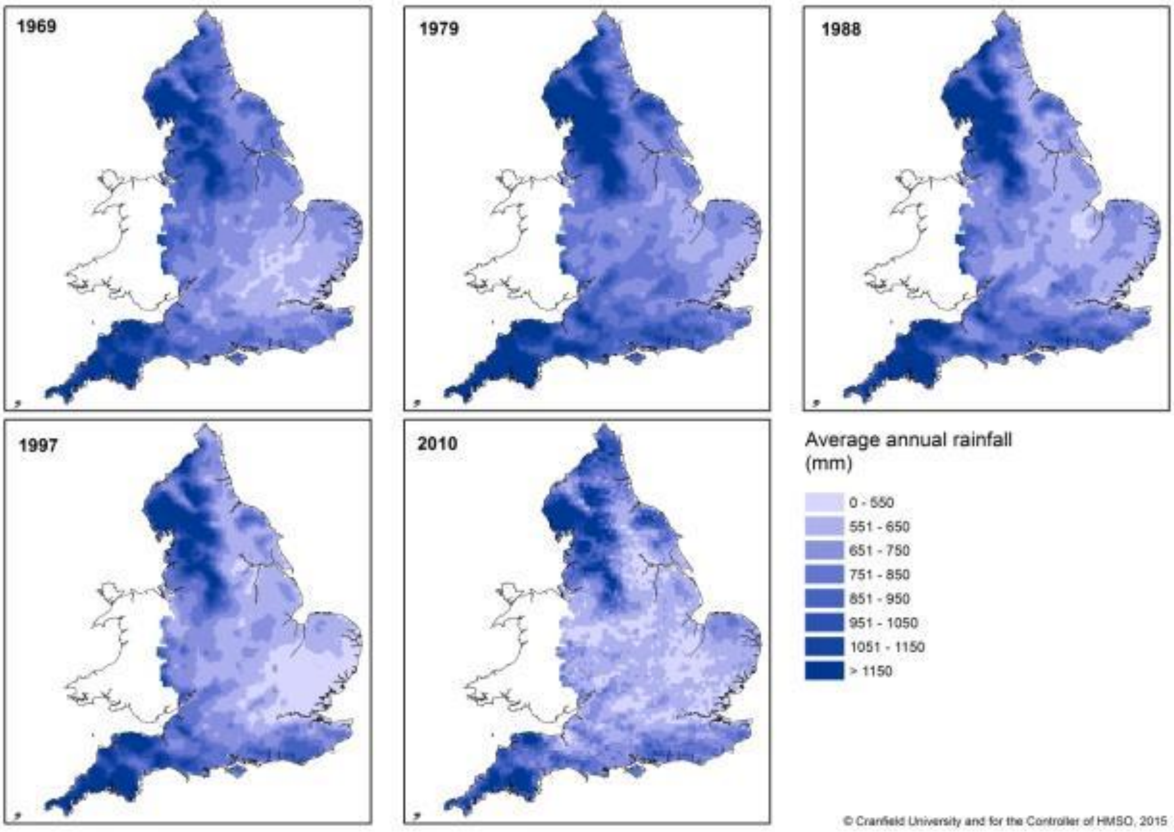

Figure 7 Annual rainfall (mm) for the selected 5 years of interest.

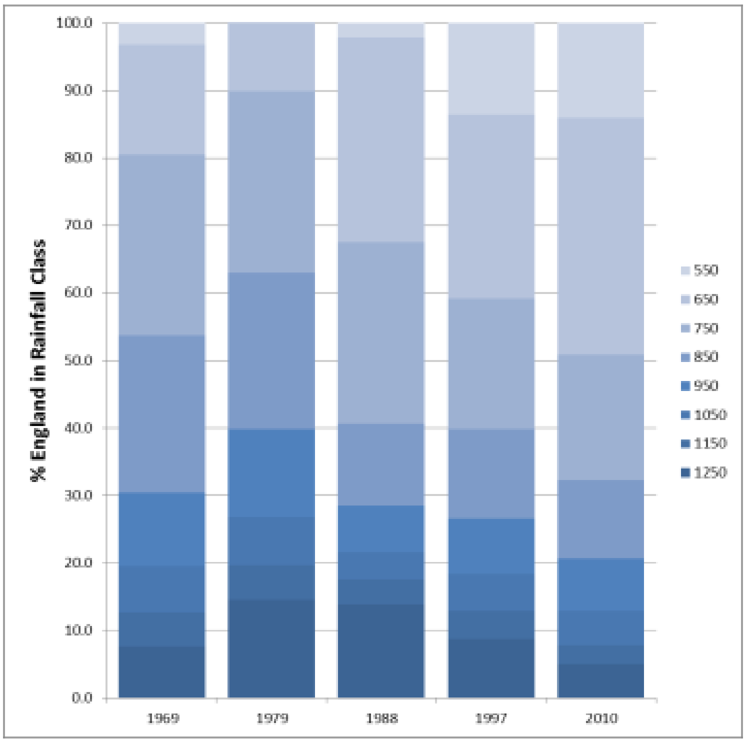

Figure 8 Proportion of rain falling in each rainfall class 1969 – 2010.

The results show the spatial and temporal variation in annual rainfall amounts for England for the years selected. 1979 appears to be a particularly wet year (therefore it would be expected that erosion risk would be highest, all other risk factors being equal). Conversely, 1997 appears to be a relatively dry year, with large areas of East Anglia receiving less than 500 mm rainfall that year.

Using the Met Office data, the MFI for all years on record (1914 – 2011; Figure 9) was calculated. Although there seems to be some variation from year to year, no trend in rainfall erosivity can be seen. The MFI is particularly high in 1929, 1960 and 2000 (when relatively high erosion rates would be expected, all other erosion risk factors being equal), and particularly low in 1921, 1933, 1953, 1964, 1973, 1991, 1996, 2003 and 2005. For the 5 selected years of interest, the MFI values range from 77 to 94 for England (Table 14). The spatial distribution of MFI is shown in Figure 10. Whether these values correlate with (predicted) soil erosion rates will be analysed (Section 3.4.5.1).

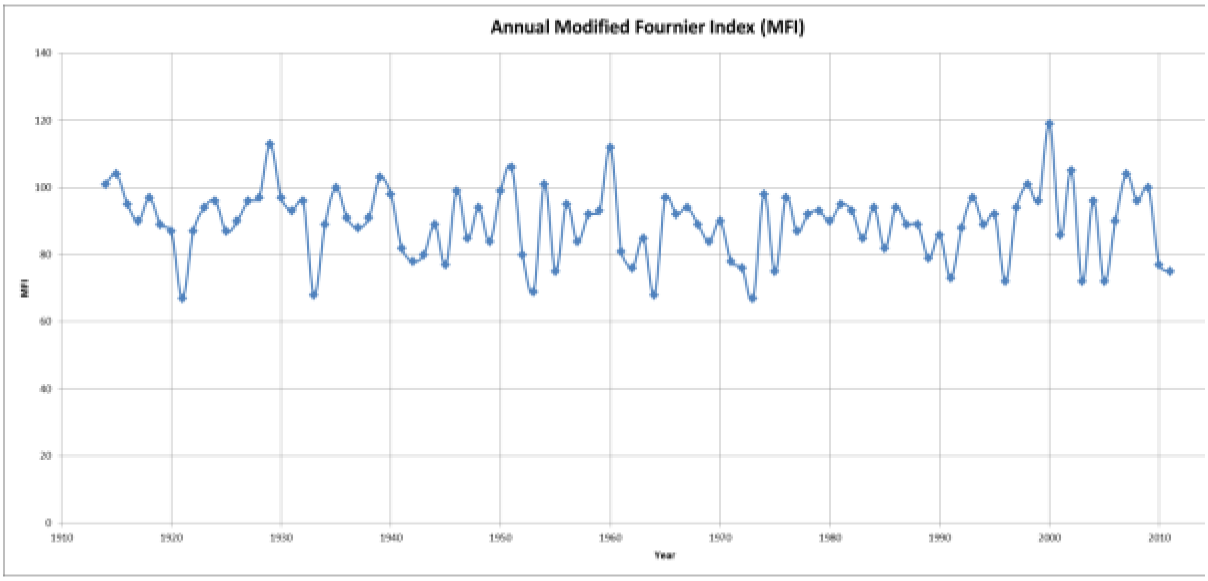

Figure 9 Modified Fournier Index of rainfall erosivity based on Met Office data (1914 – 2011).

Table 14 Modified Fournier Index of rainfall erosivity for selected years.

Year	Modified Fournier Index (MFI)
1969	84
1979	93
1988	89
1997	94
2010	77

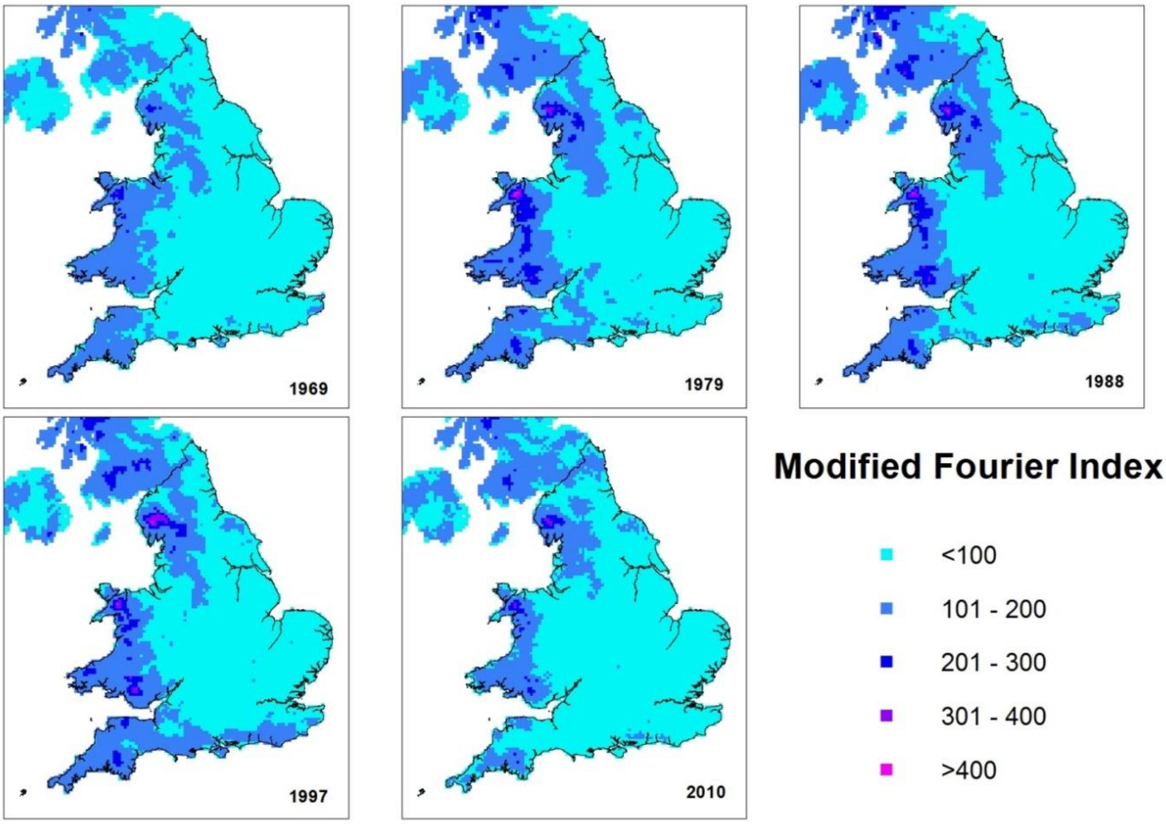

Figure 10 Modified Fournier Index (MFI).

2.3.2 Soil type

The National Soil Inventory database (Land Information Systems, LandIS) was used to map the soils susceptibility to erosion. The soil types in England (as surveyed from the 6127 sites between 1978 and 1983) were classified into Defra's Cross Compliance categories of Light, Medium, Heavy, and Chalk and Limestone at the 2 km resolution. These categories are considered appropriate for indicating erodibility (soil susceptibility to erosion) and sensible for map representation at the national scale. The Limestone/Chalk classes were further classified according to their textural characteristics (% clay content). The dominant soil class for each 2 km cell was then extracted (Figure 11).

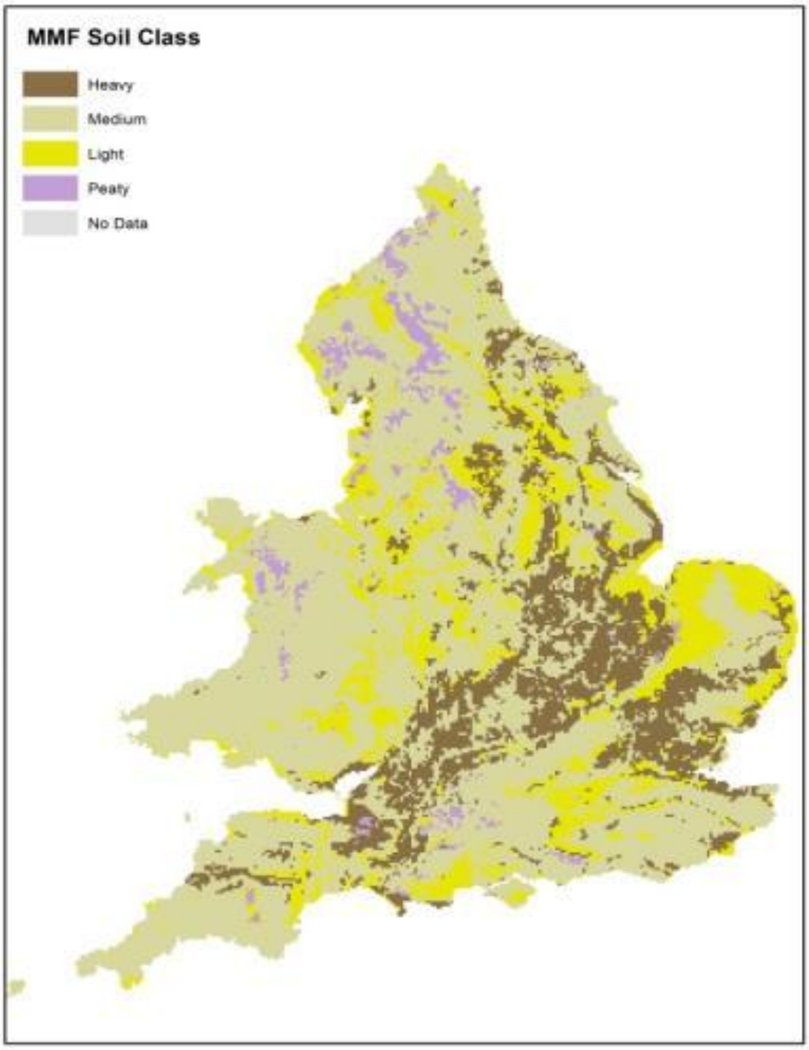

Figure 11 Cross Compliance soil types at the national scale, based on NSI survey points (1978-83).

It is unlikely that soil textural properties that affect erodibility (such as proportion of silt and very fine sand) will change within the period of interest. In any case, there has been only very limited resampling of the NSI points. In 1994/95, 842 of the 6127 sites were resampled on arable land; 745 sites on permanent managed grassland. In 2003, 562 sites were resampled. Therefore it is difficult to report changes (if any) in soil erodibility at the national scale over time.

2.3.3 Slope

The Ordnance Survey Terrain 50, a 50m Digital Terrain Model of Great Britain, was used to derive a slope product. The range of slope gradients ran from 0 – 15^{o} in 1^{o} increments. This is likely to capture all the agricultural land where soil erosion risk is highest (slopes much greater than 15^{o} are unlikely to be cultivated). The mean slope steepness value in each 2 km cell was calculated. The slope value was reclassified for ease of map representation (Table 15).

Table 15 Slope classification.

Slope range (degrees)	Reclass
0 -1	1
1.1 – 2	2
2.1 – 3	3
3.1 – 4	4
4.1 – 5	5
5.1 – 6	6
6.1 - 7	7
7.1 – 8	8
8.1 – 9	9
9.1 – 10	10
10.1 - 11	11
11.1 – 12	12
12.1 - 13	13
13.1-14	14
>14	15

The resultant map is shown in Figure 12. Because mean slope length over a 2 km grid is difficult to express, this parameter could not be mapped. In any case, both slope gradient and slope length are unlikely to change over time.

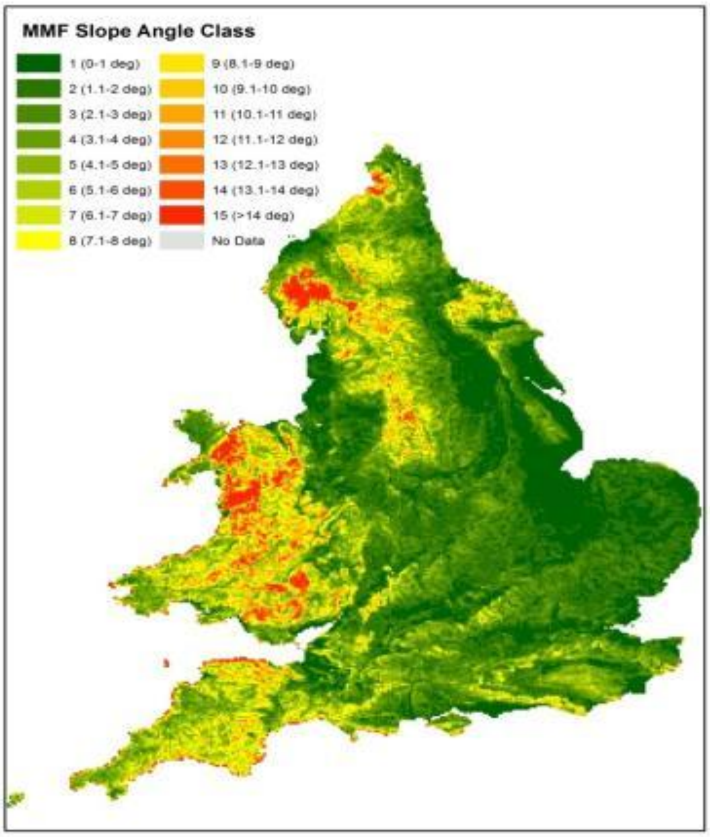

Figure 12 Mean slope gradient (aggregated at the 2 km resolution).

2.3.4 Land use/land cover/crop type

Agricultural land cover was derived from the June Agricultural Census (JAC) statistics. However, it was unrealistic to use the JAC data at spatial scales of individual field parcels for national mapping purposes. So we extracted the data at the 2 km resolution. For each 2 km grid square (= 400 ha), the proportion of each crop per total area of crop was calculated. This was repeated for all 5 years of interest: 1969, 1979, 1988, 1997 and 2010.

One flaw in the methodology of using these datasets is the discontinuities in the types of crops mapped at the different dates. The land use category 'Fruit' does not appear in the 1997 statistics. The area in oil seed rape is not reported for 1979. Spring cereals are not reported for 1969 and vegetables are not reported for 1997.

Maps of the changes in land use / land cover / crop type are given in Figure 13 - Figure 23 A graph of trends in land use change under each of the selected crops is given in Figure 24. Figure 25 shows the relative area under each of the crops separately and Figure 26 when grouped into low, moderate and high risk crops, according to Table 6 above (Defra, 2005).

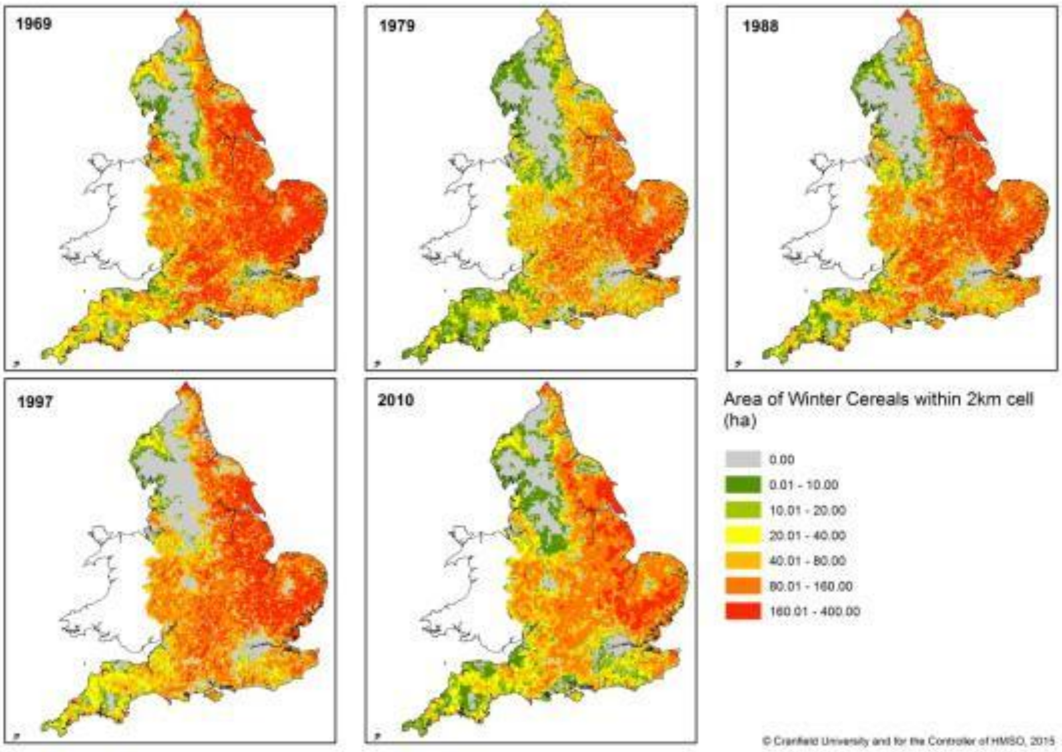

Figure 13 Change in land use under winter cereals.

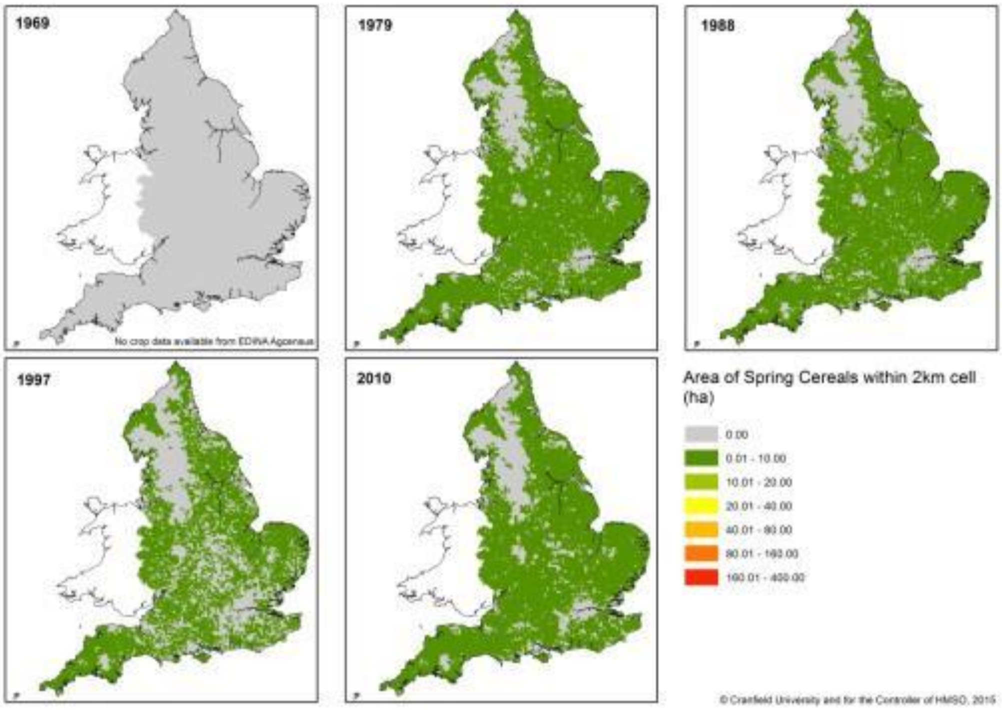

Figure 14 Change in land use under spring cereals.

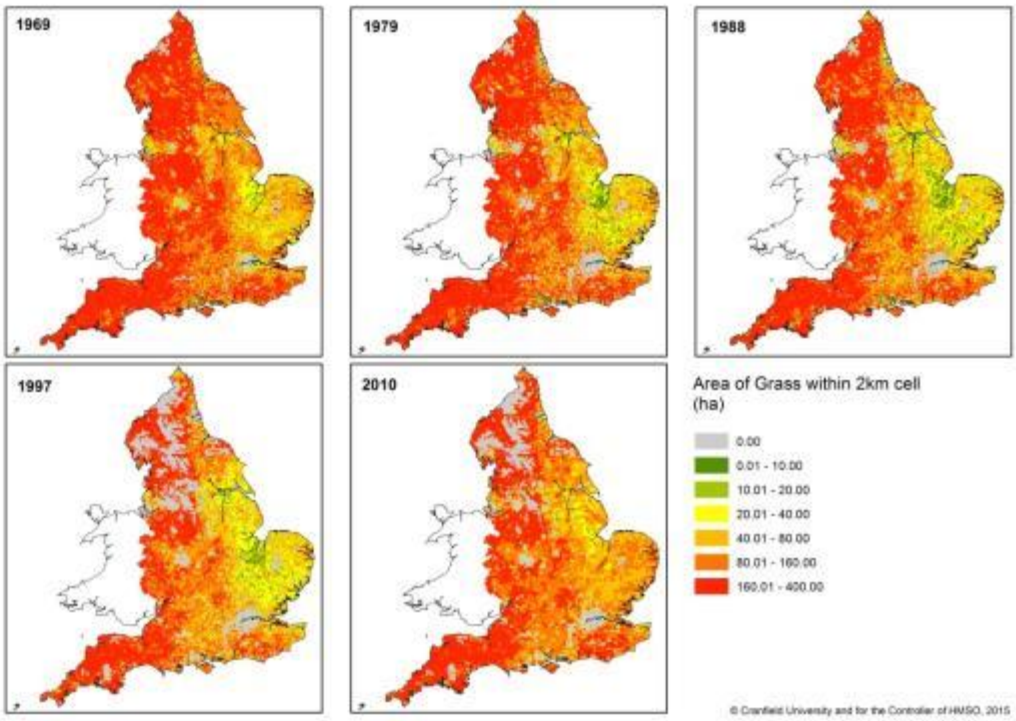

Figure 15 Change in land use under grassland.

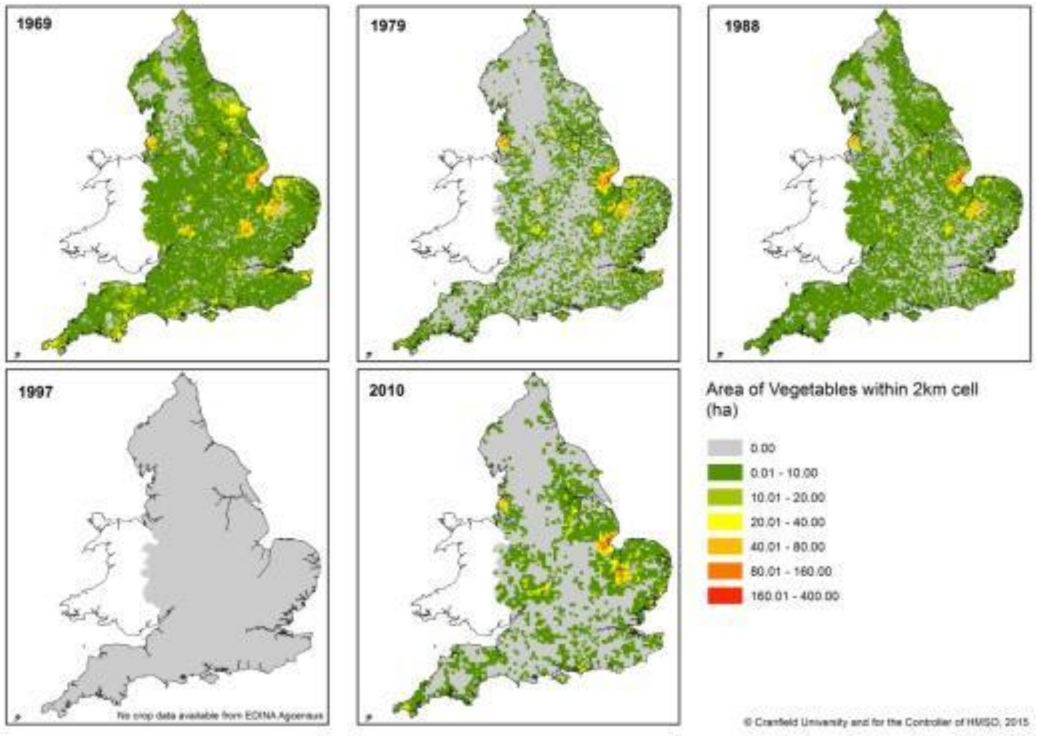

Figure 16 Change in land use under vegetables.

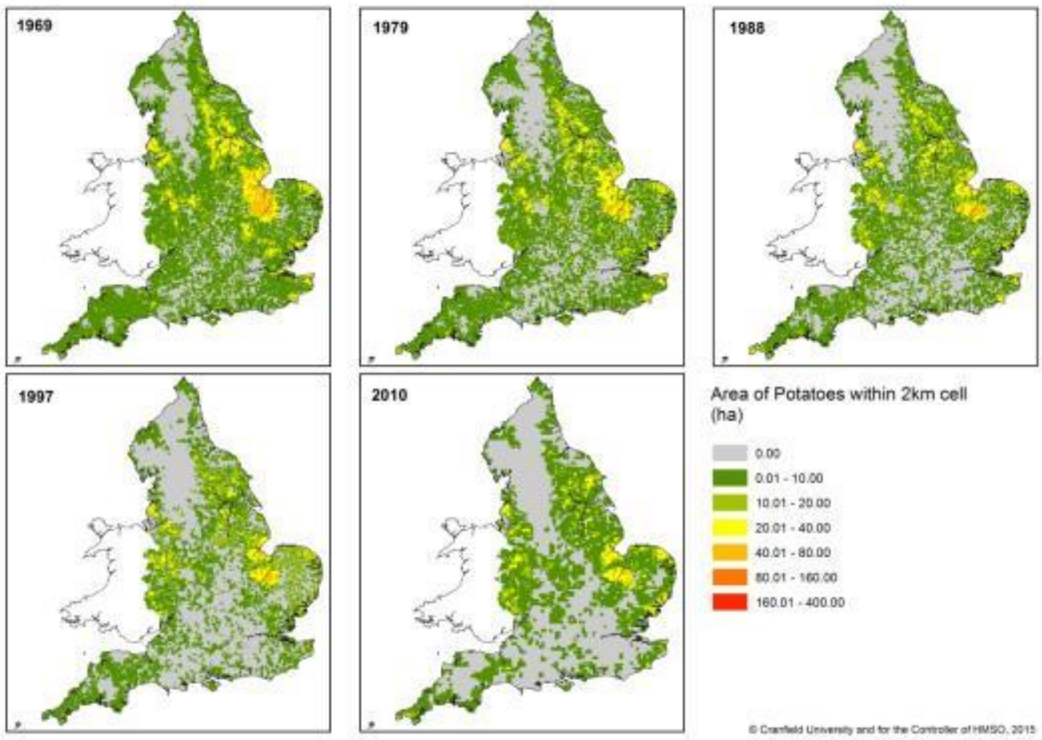

Figure 17 Change in land use under potatoes.

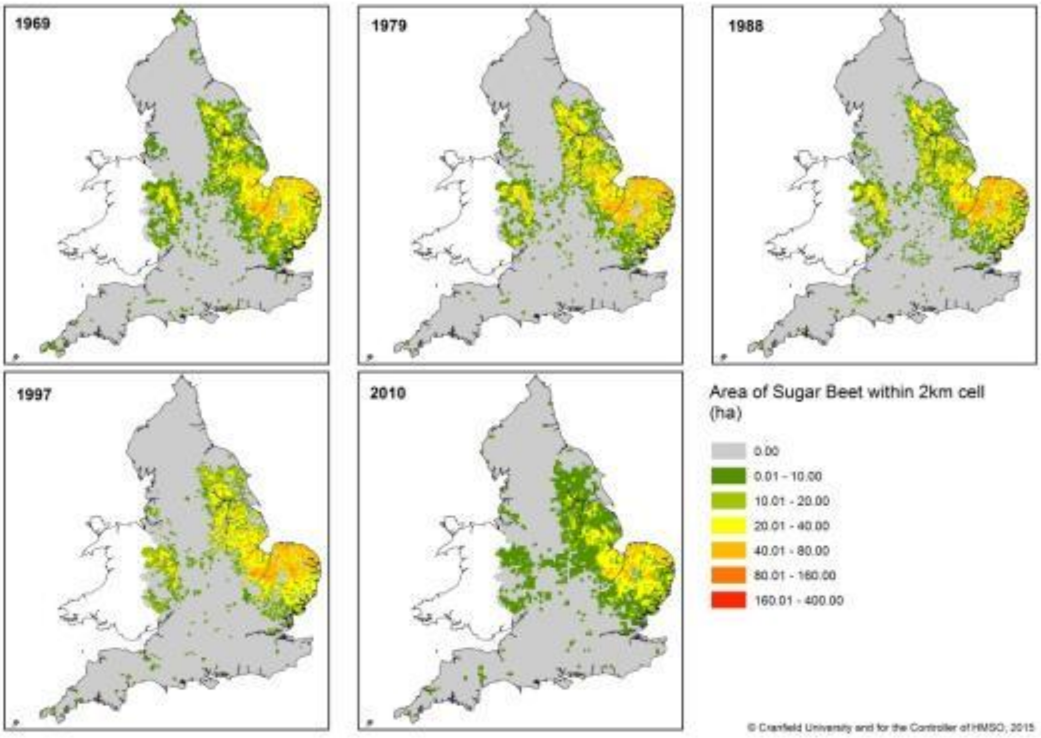

Figure 18 Change in land use under sugar beet.

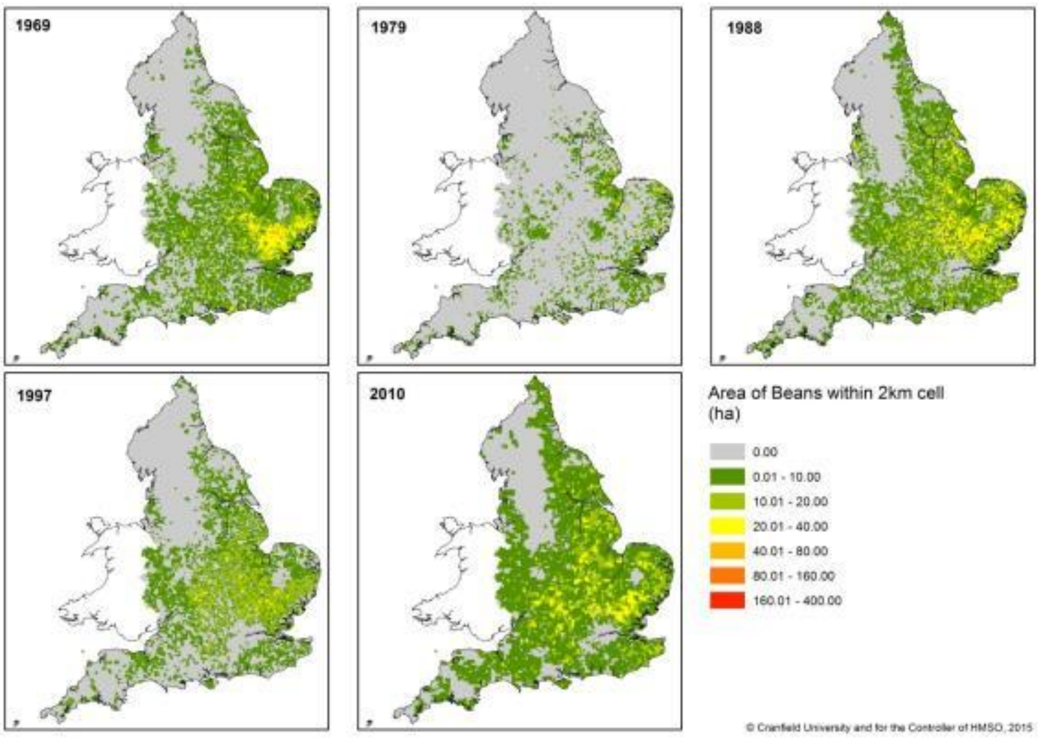

Figure 19 Change in land use under beans.

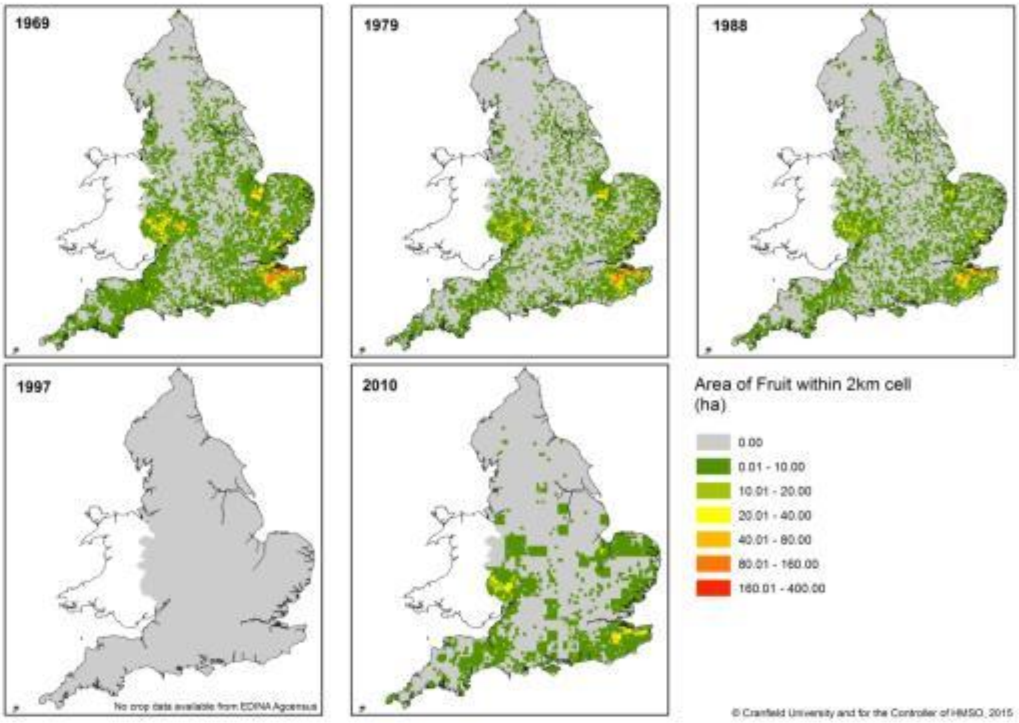

Figure 20 Change in land use under fruit.

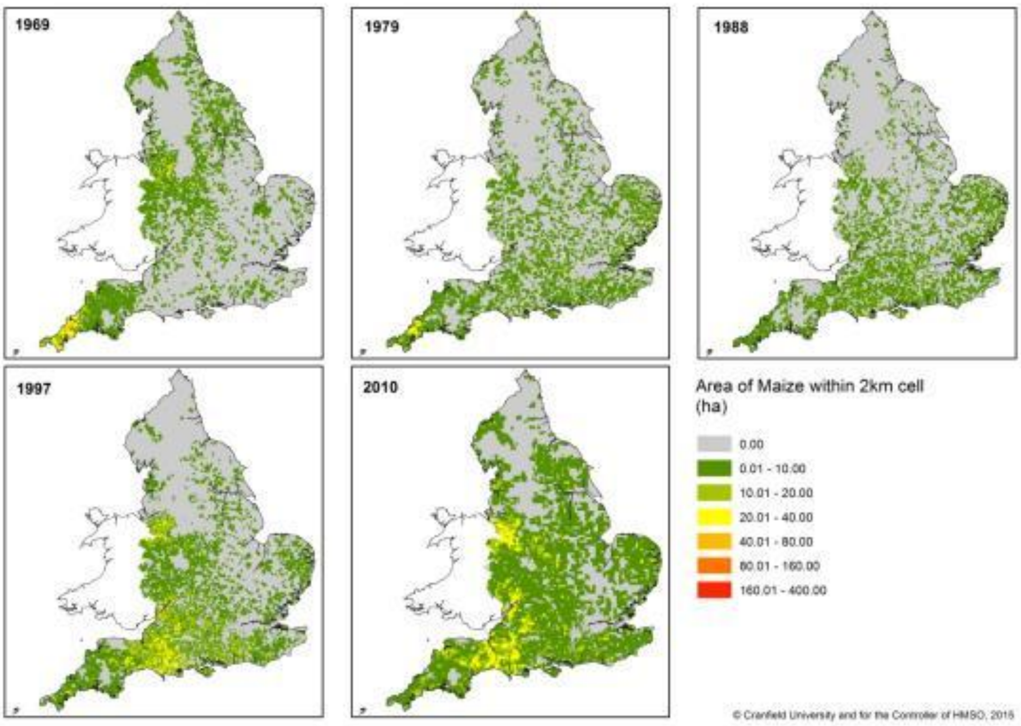

Figure 21 Change in land use under maize.

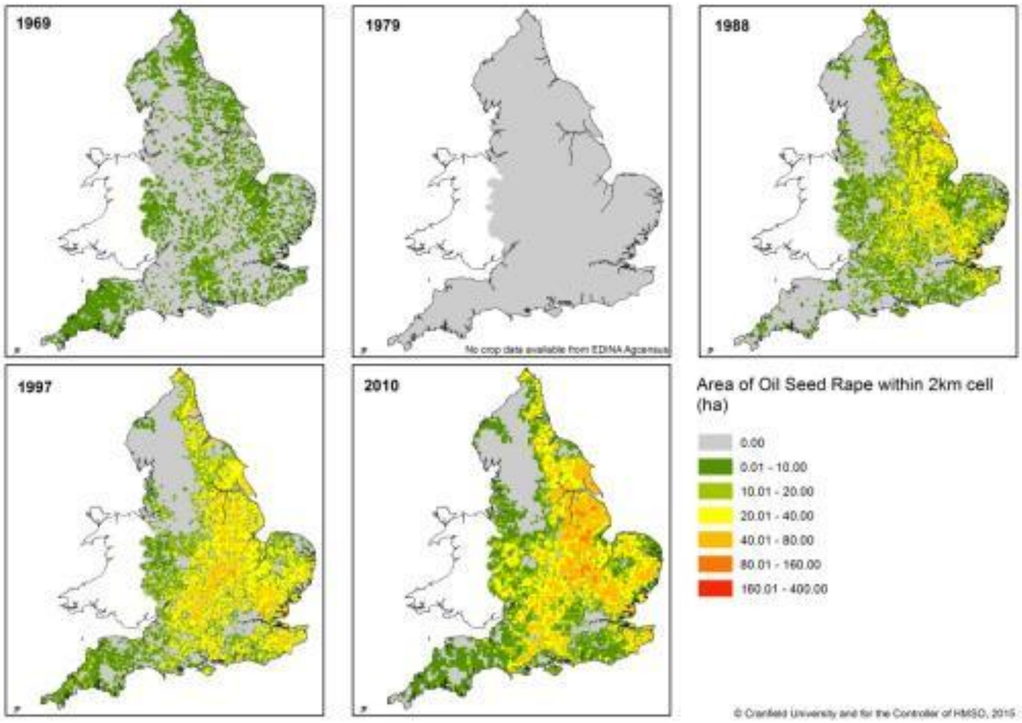

Figure 22 Change in land use under oil seed rape.

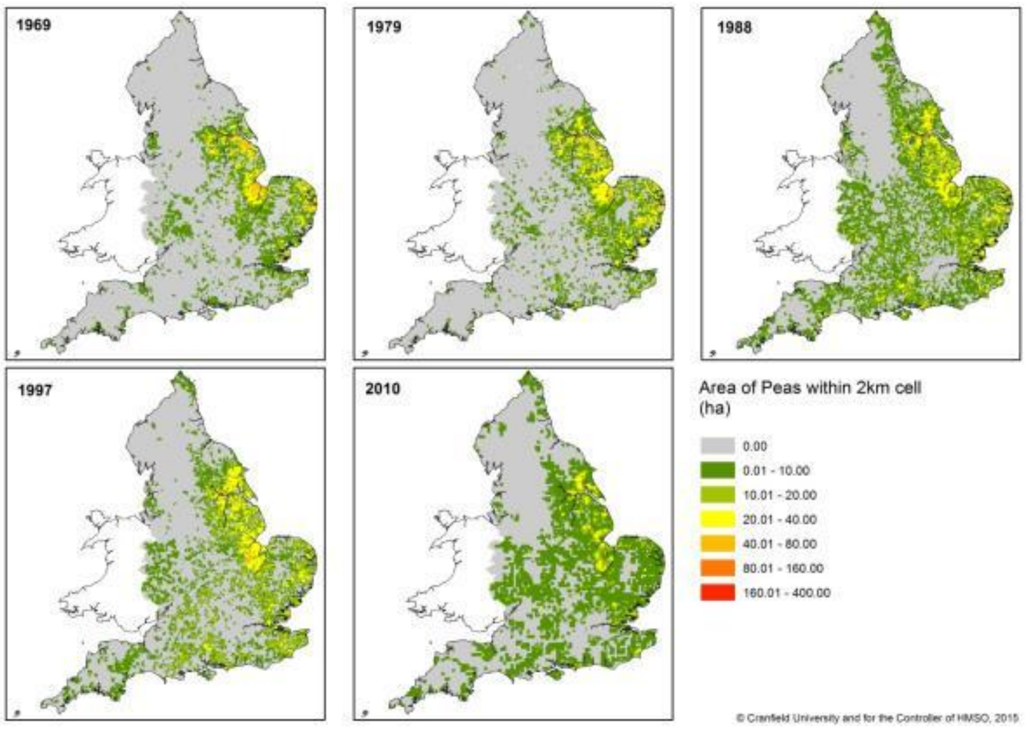

Figure 23 Change in land use under peas.

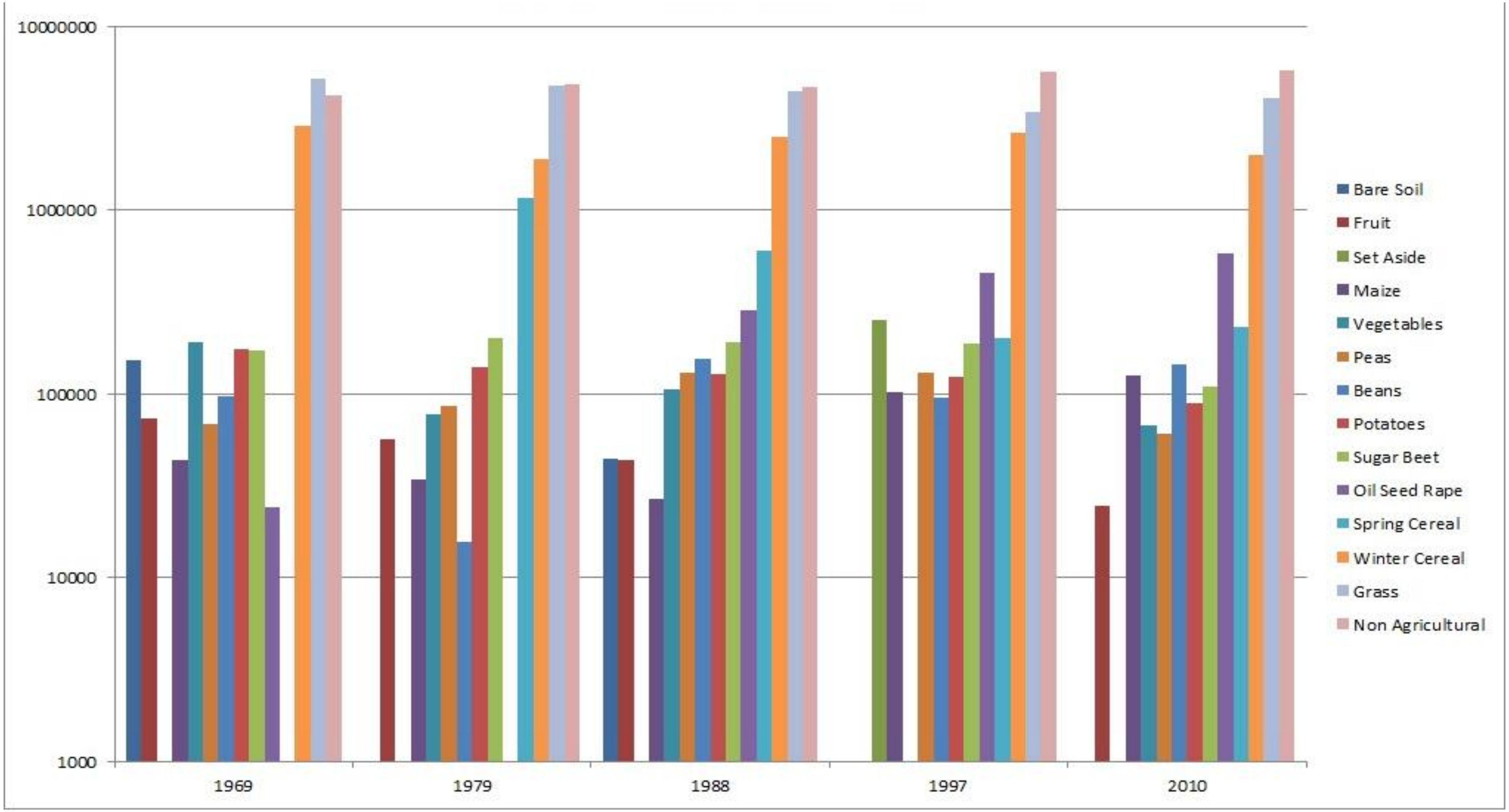

Figure 24 Trends in land use (AgCensus data) for the 5 years (1969, 1979, 1988, 1997 and 2010) (N.B. log scale on vertical axis).

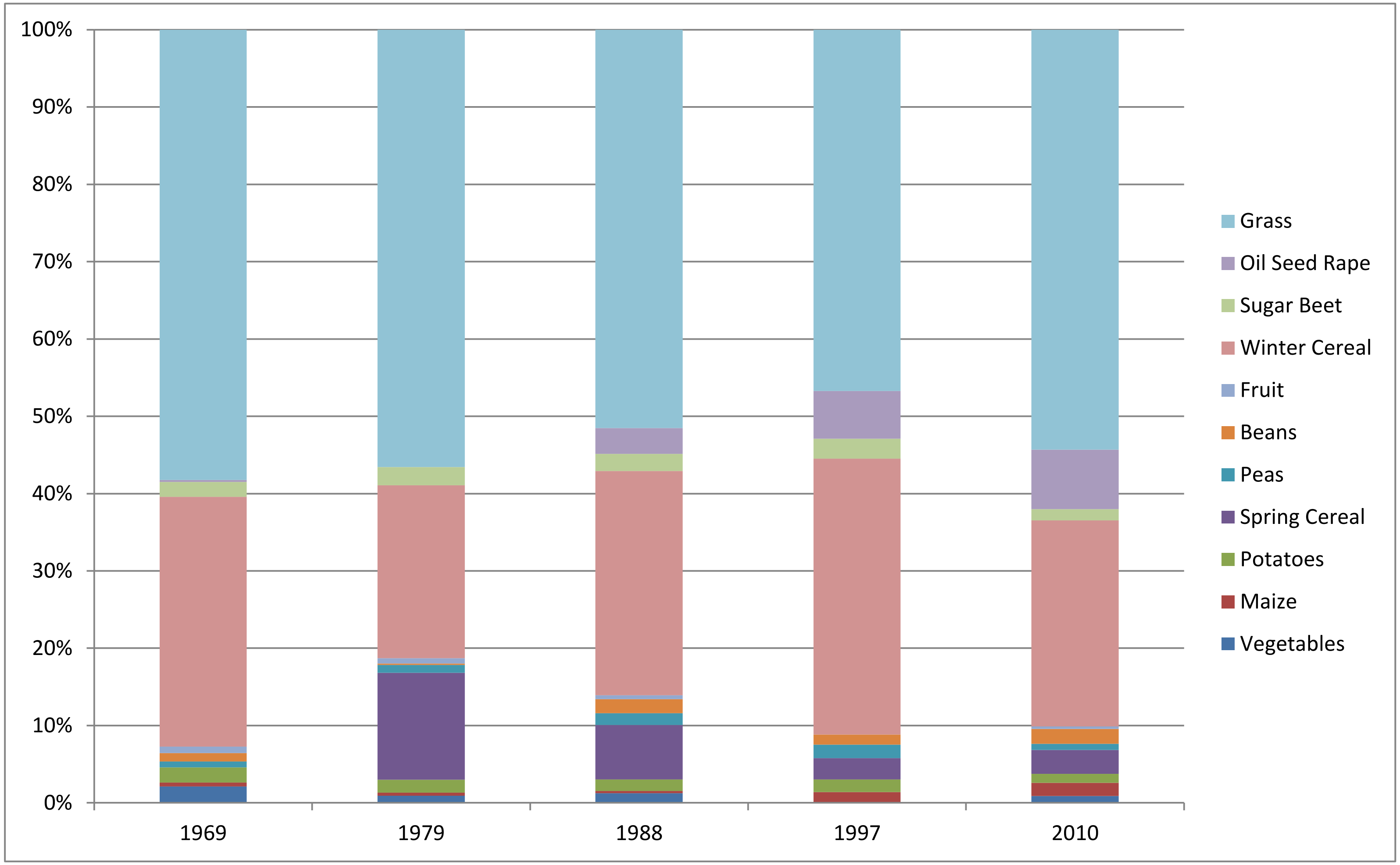

Figure 25 Relative agricultural area (%) under different crops over time (N.B. The category 'Spring Cereals' is not reported for 1969; 'Oil seed rape' is not reported for 1979; the categories 'Fruit' and 'Vegetables' are not reported for 1997).

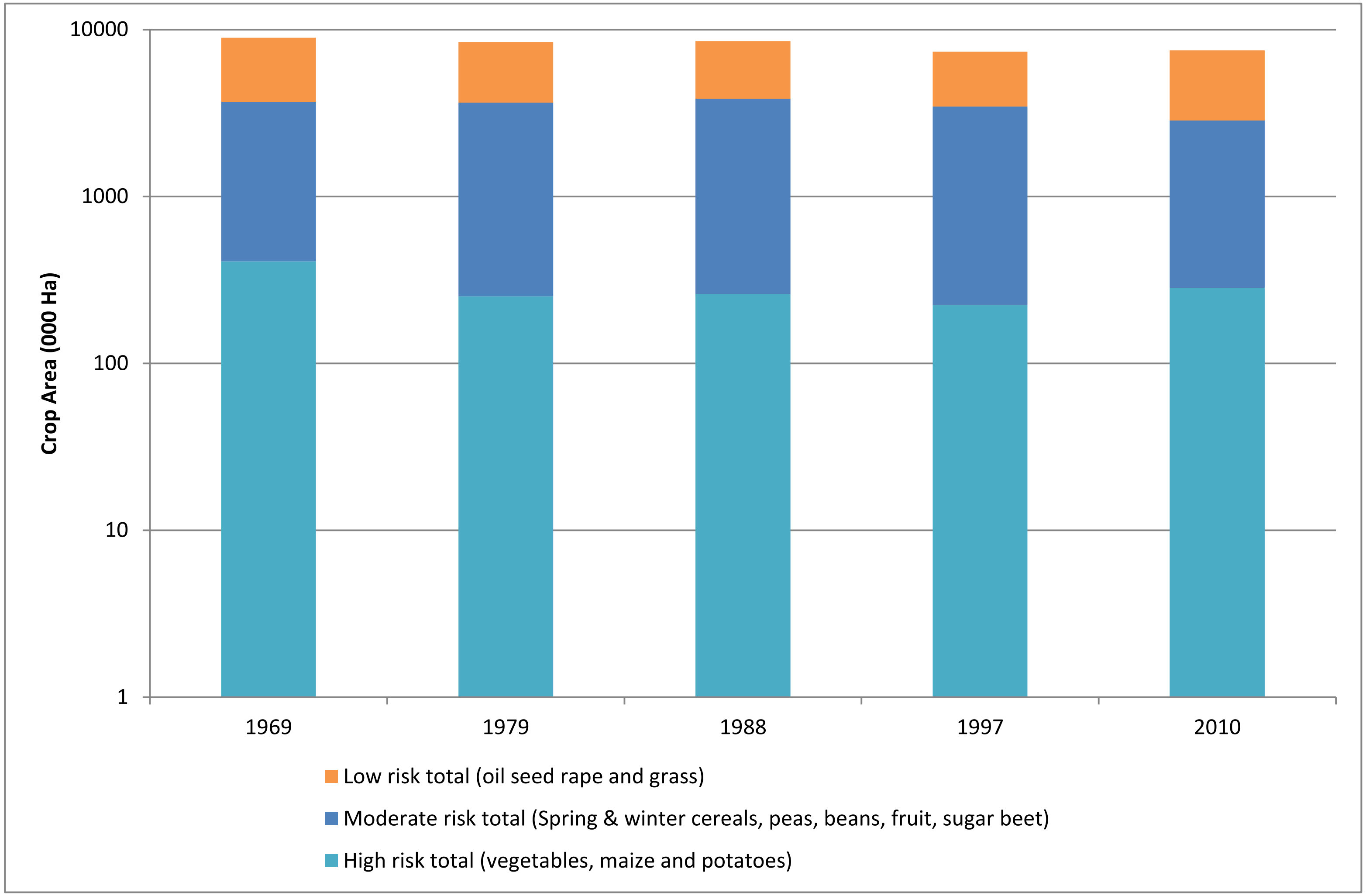

Figure 26 Area of low, moderate and high risk crops (grouped according to Table 6, Defra, 2005) N.B. Vertical axis is log scale.

Using the EDINA dataset and described methodology, there appear to be very few dramatic changes in land use from 1969 to 2010 (Figure 24). A large proportion of the country is dedicated to grass and winter cereal production (regarded as low to moderate erosion risk land uses). The area of spring cereals has fallen over time, and areas under sugar beet, potatoes, vegetables, peas and fruit (associated with moderate to high erosion risk) are slightly down in 2010 compared with 1969. The area of oil seed rape (low erosion risk) and maize (high erosion risk) has increased over this time period, as has the area under beans (moderate erosion risk), but to a lesser extent.

The lack of dramatic changes in land use might be explained by the fact that land use change may occur at spatial scales less than the 2 km (400 ha) resolution of the current study.

2.4 Mapping modelled soil erosion rates

Due to the limited number of actual observations of soil erosion frequency and magnitude at the national scale, an erosion prediction model was used to estimate spatial erosion risk and how this might change over time, as the risk factors affecting erosion change. The modified version of the Morgan, Morgan and Finney (MMF) model (Morgan and Duzant, 2008) had been run for a number of standardised parameters (risk factors) for the on-going Defra SP1318 project ("Scaling up the benefits of field scale soil protection measures to understand their impact at the landscape scale"). With the agreement of Defra, the output of this exercise was adopted in the present study to extract the modelled / predicted soil loss for each 2 km cell. In the present study, annual rainfall ranges have been reclassified into model input values (Table 16). Guide values for the input variables for the different Cross Compliance soil types shown in Figure 11 are taken from Morgan and Duzant (2008). Mean slope angle for each cell was classified as shown in Table 15.

Table 16 Classification of annual rainfall (mm)

Rainfall (mm)	Reclassified
0-550	550
551-650	650
651-750	750
751-850	850
851-950	950
951-1050	1010
1051-1150	1150
> 1151	1250

As each 2 km cell has more than one land cover, the proportion of the land cover observed in each cell has been used to predict a weighted soil loss value for each cell. As an example, if the cell had 40% wheat and 60% grass (as a percentage of the total agricultural area), the soil loss would be predicted by:

Cell predicted mean annual soil loss (t ha^{-1} yr^{-1})

= (0.4 * modelled soil loss under wheat) + (0.6 * modelled soil loss under grass)

The model was run for each 2 km cell, using its rainfall, soil, slope angle and land cover attributes as input parameters to the MMF model. This was repeated for each of the 5 years of interest. The result is shown in Figure 27. It should be noted that the predicted rates of erosion (t ha^{-1} yr^{-1}) presented in these maps represent a mean annual soil loss rate averaged over each 400 ha cell. Also, these values are unvalidated due to paucity of quantified observations of soil erosion rates. Due to the unvalidated nature of the output, the quantified results (t ha^{-1} yr^{-1}) were then classified into qualitative, relative classes of erosion rates (Table 17). This was based on the range of erosion rates collated from observed data in England and Wales (Rickson, 2014; Table A1) and the rates of "tolerable soil loss" given by Verheijen et al. (2009), ranging from ca. 0.3 to 1.4 t ha^{-1} yr^{-1}.

Table 17 Classification of erosion rates into qualitative classes.

Predicted mean annual soil loss rate (t ha^{-1} yr^{-1})	Class
0 - 0.25	Very Low
0.25 - 0.5	Low
0.5 - 1	Moderate
1 - 2	High
> 2	Very High

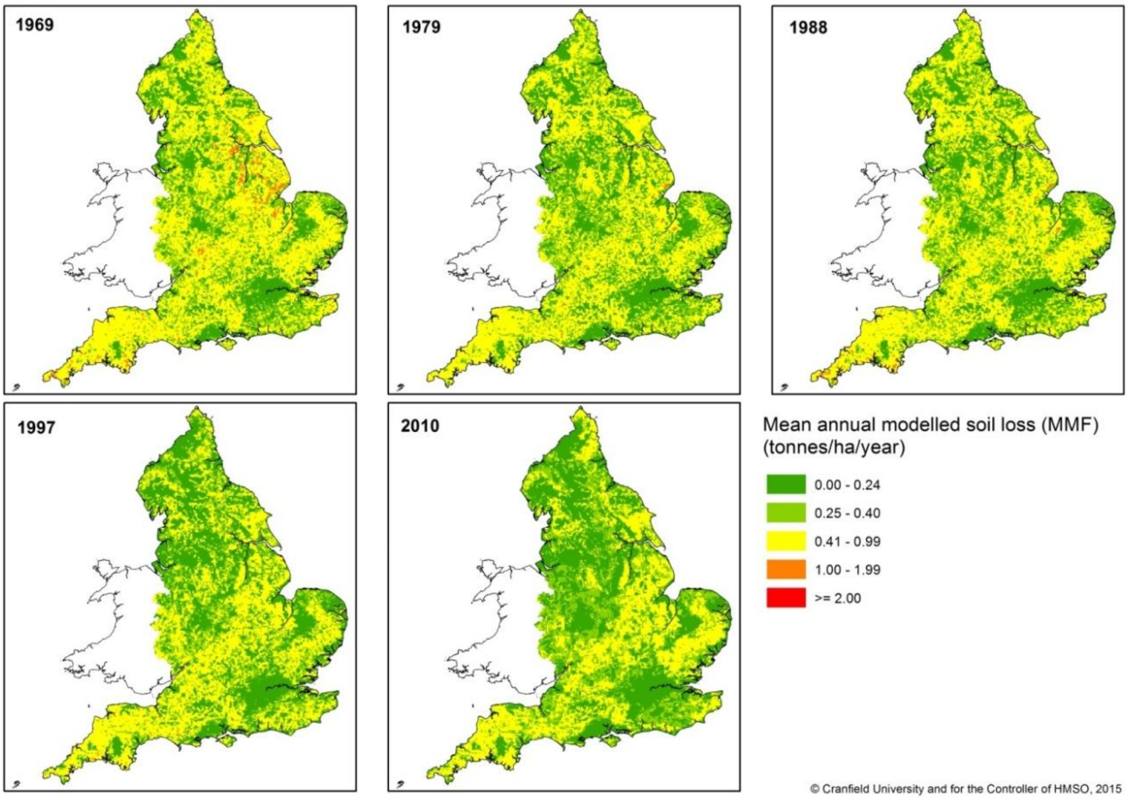

Figure 27. Predicted mean annual soil erosion rates (1969 – 2010), using the modified Morgan, Morgan and Finney model. N.B. Unvalidated. Also, results show mean annual soil erosion rate for each 2 km (= 400 ha) cell, so areas of higher erosion might occur within the 2 km / 400 ha cell. Similarly, high erosion rates predicted at 2 km resolution do not necessarily apply over the total 2 km (400 ha) area.

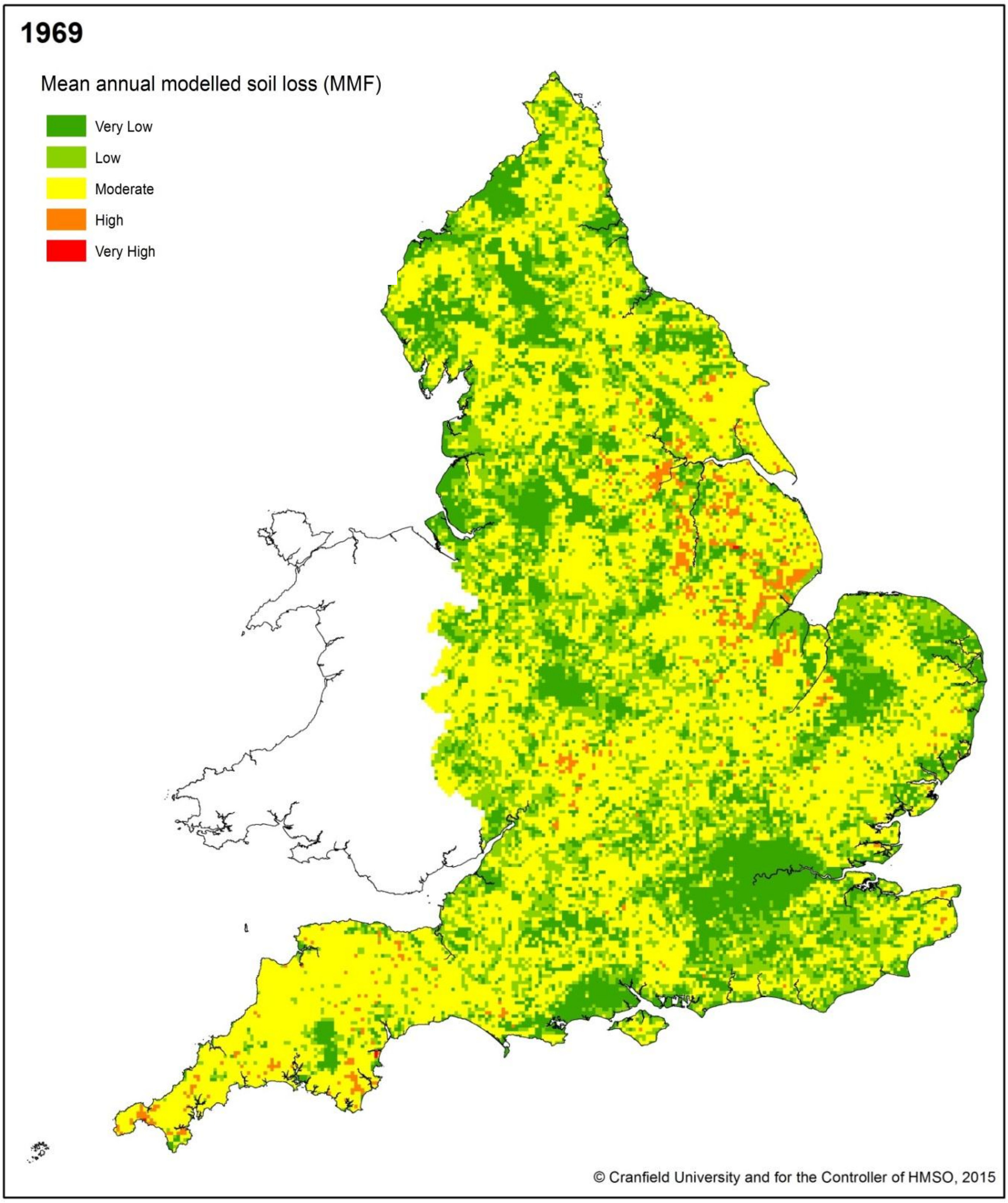

Figure 28a Qualitative classes of predicted soil erosion rates as predicted by the modified MMF model (1969).

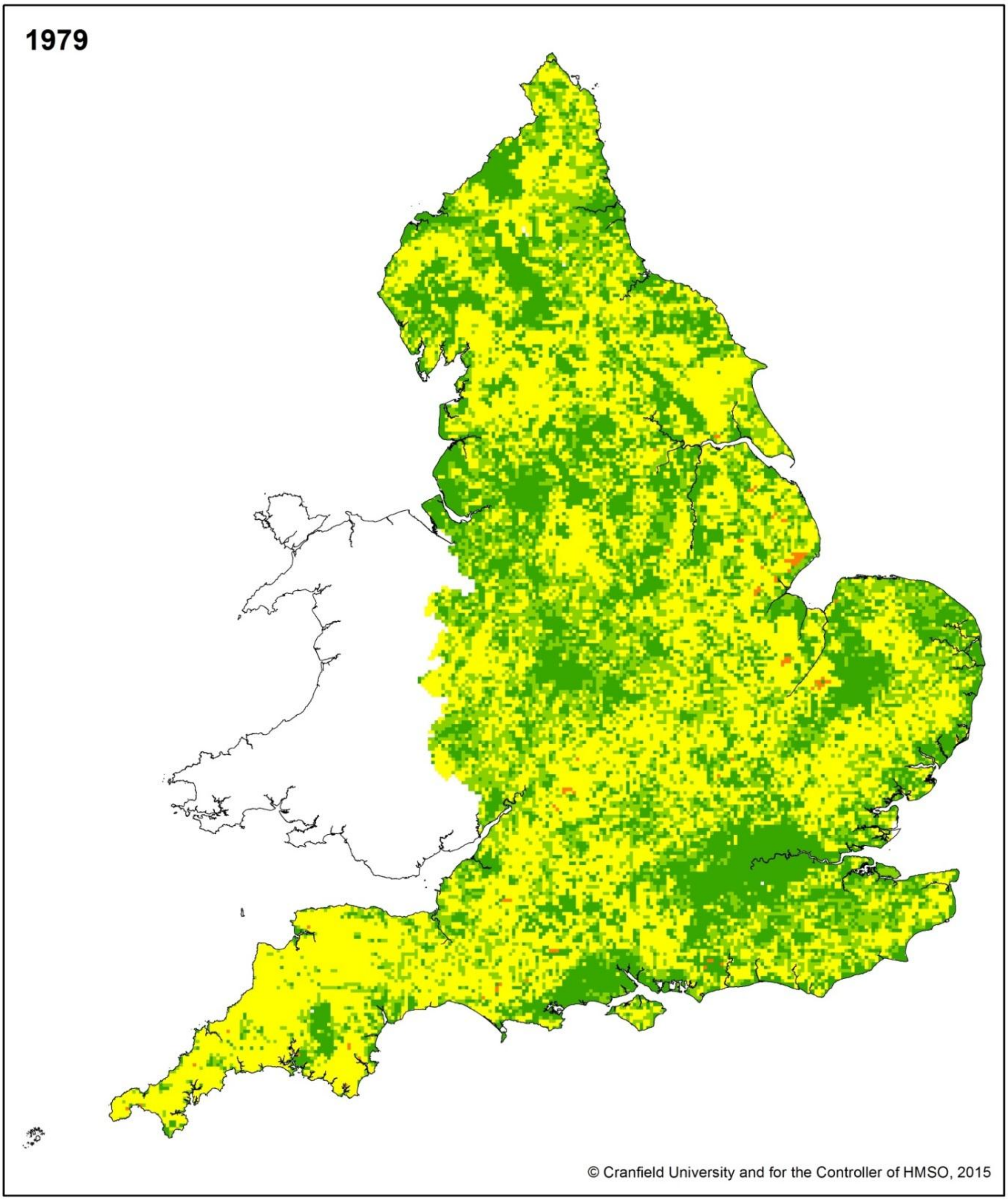

Figure 29b Qualitative classes of predicted soil erosion rates as predicted by the modified MMF model (1979).

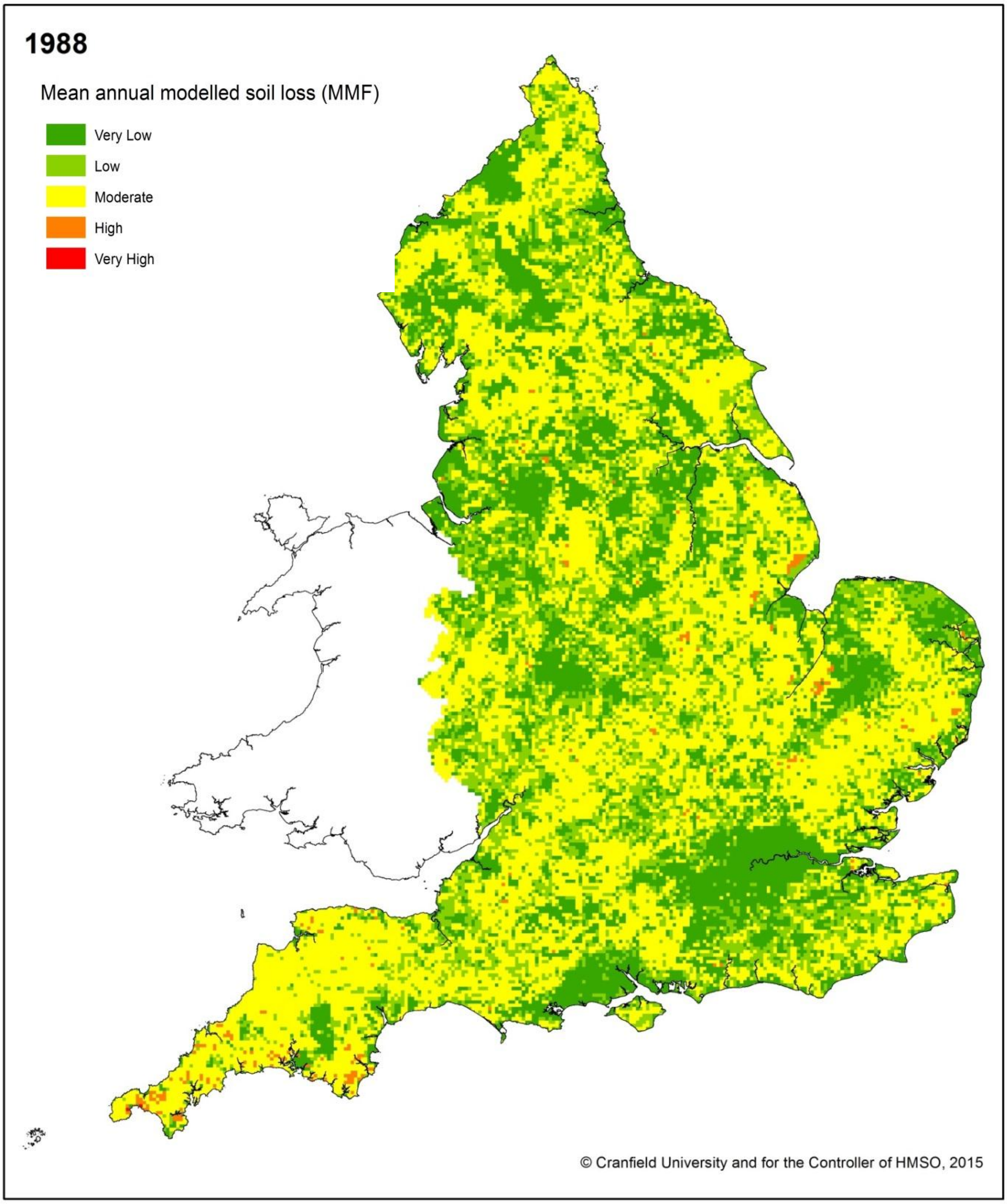

Figure 30c Qualitative classes of predicted soil erosion rates as predicted by the modified MMF model (1988).

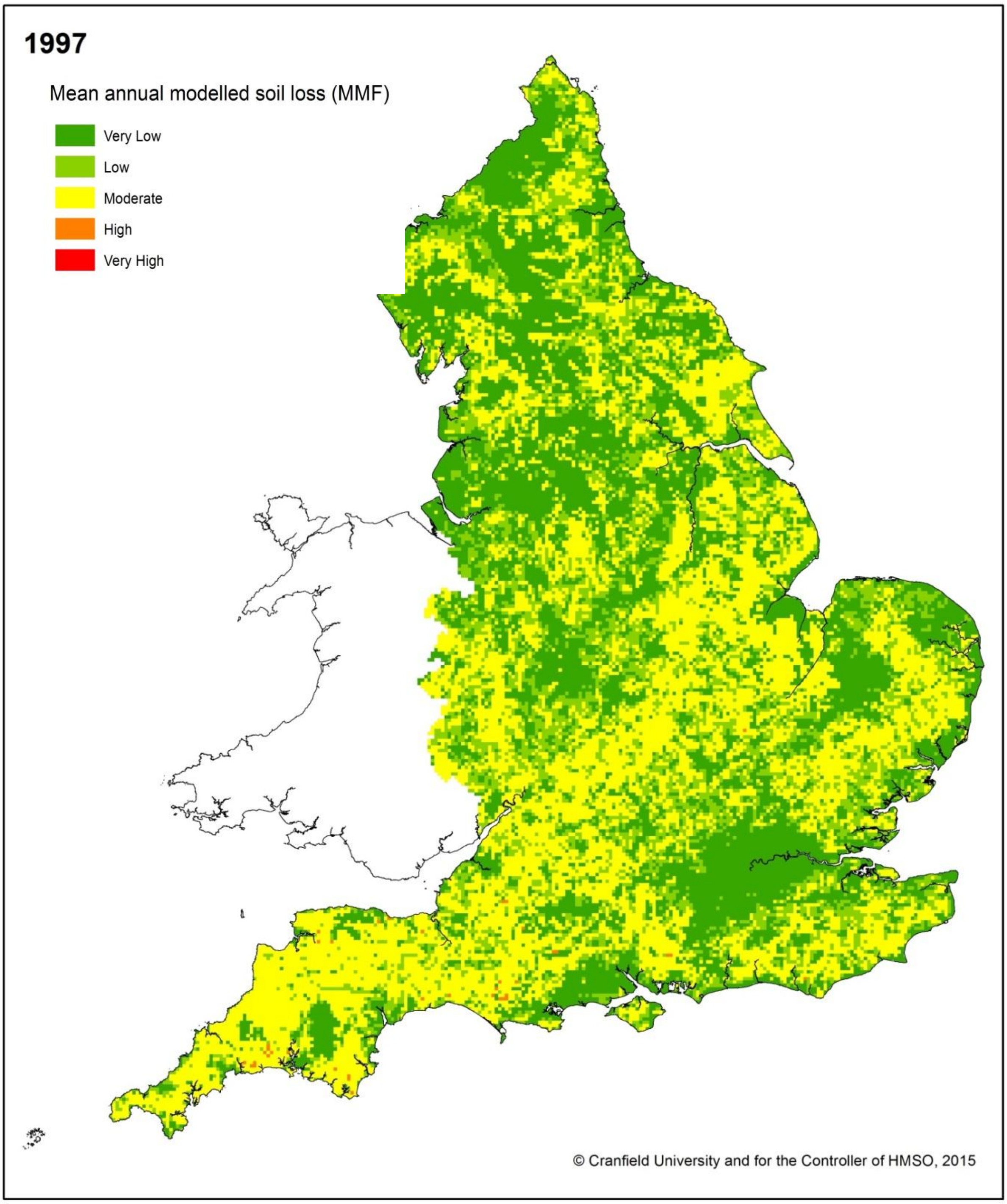

Figure 31d Qualitative classes of predicted soil erosion rates as predicted by the modified MMF model (1997).

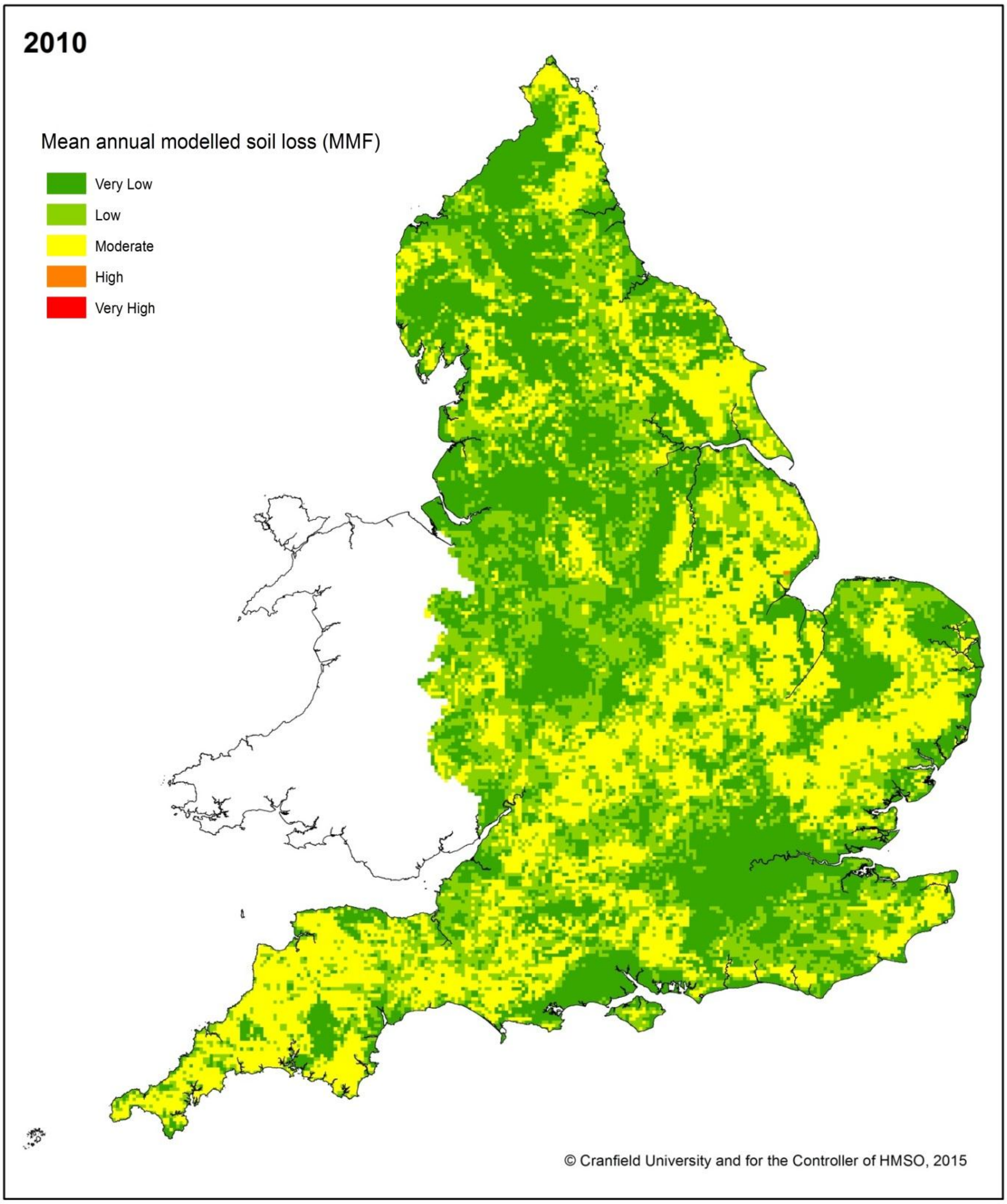

Figure 32e Qualitative classes of predicted soil erosion rates as predicted by the modified MMF model (2010).

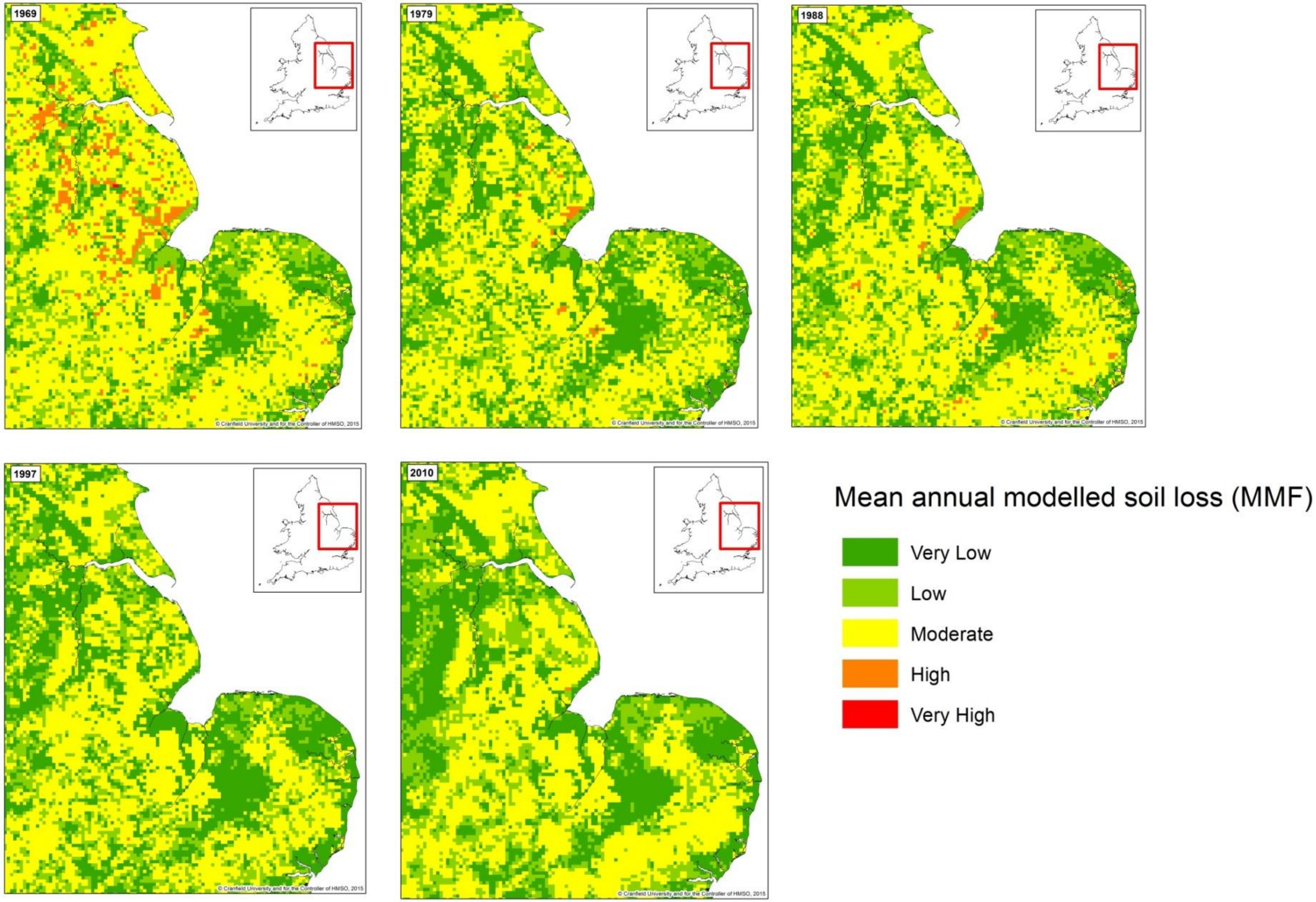

Figure 33 Excerpt showing the change in predicted erosion rates for the years of interest for eastern England.

The modelled outputs for the 5 years show highest predicted annual erosion rates were seen in 1969 with some areas classified as having ‘very high’ erosion rates, followed by 1988, 1979, 1997 and lowest predicted annual erosion in 2010 (Figure 32a to 32e). The proportion of land in each erosion rate class is shown in Figure 34. It should be noted that the modelled results from this exercise require validation from observed rates of soil erosion. Even so, the spatial and temporal patterns in predicted erosion rates (Figure 27) were then compared with the equivalent patterns in each of the risk factors.

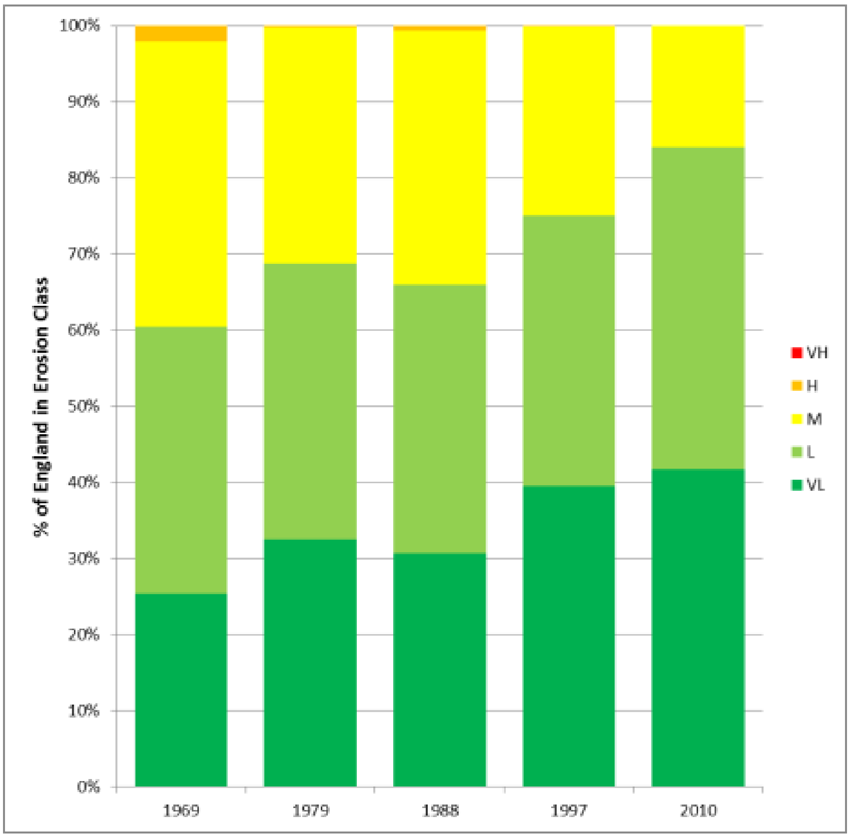

Figure 34 Proportion of land in each erosion rate class (1969 – 2010).

2.4.1 Spatial and temporal patterns in predicted erosion rates and rainfall

As expected, highest rainfall is received in the NW and SW of the country in all the years of interest (Figure 7). However, this pattern is not reflected in the highest predicted erosion rates, which tend to be located in the eastern regions (save some 2 km grid cells in the extreme SW of the country, especially in 1969 and 1988; Figure 32a to 32e). Similarly, lowest annual rainfall (e.g. in East Anglia) does not correspond with low erosion rates, with some grid cells in these low rainfall areas with predicted erosion rates above the defined ‘tolerable’ level of 1 t ha^{-1} yr^{-1} (e.g. in 1969, 1979 and 1988).

Over time, 1969 and 1979 were relatively wet years, and 1997 and 2010 are relatively dry years. Predicted erosion rates do appear higher for 1969, with a higher proportion of areas in high or very high erosion rate categories, and a lower proportion of land under ‘low’ or ‘very low’ erosion rates (Figure 34). This pattern is less obvious for 1979 however. In 2010 (a relatively dry year), there is a higher proportion of land in ‘low’ and ‘very low’ erosion categories (< 0.4 t ha^{-1} yr^{-1}) compared with the other years analysed. The conclusion is that changing rainfall patterns in space and time may not indicate changing trends in erosion status. This is not surprising, given the other risk factors affecting erosion (e.g. land use), and the fact that annual rainfall is a poor indicator of rainfall erosivity (Morgan, 2005).

The results of the MFI calculations give some range of erosivity values, which should be reflected to some degree in soil erosion values. The MFI results suggest that soil erosion should be highest in 1997 and lowest in 2010. However, this is not suggested by the predicted erosion rates (Figure 27 and 28). 1969 shows the highest predicted erosion rates, but the MFI calculated for this year is 84 (2nd lowest of the 5 years of interest). 1997 has the highest MFI (94), but there are very few areas of predicted 'high' or 'very high' erosion rates. However, in other years there is some agreement between MFI and erosion rates: in 2010 there appears to be the lowest erosion rates (Figure 28) and this corresponds with the lowest MFI (77) of all 5 years of interest.

In summary, this analysis shows a poor correlation between rainfall (when expressed in annual precipitation or as a recognised index of erosivity) and the spatial distribution and rates of soil erosion (although the estimates of erosion have been based on modelled (unvalidated) results, due to lack of a comprehensive national dataset of observed rates). It is likely that the characteristics of individual storms (intensity, duration, kinetic energy) as well as antecedent rainfall events (so affecting soil moisture conditions) will be better correlated with the frequency and magnitude of soil erosion. However, these highly dynamic characteristics are difficult to express for individual years and at the national scale. In any case, rainfall characteristics are often confounded by the other risk factors affecting erosion such as land use.

2.4.2 Spatial and temporal patterns in predicted erosion rates and soil types

The Soil Associations at risk of soil erosion (Appendix 7.3) include light textured soil types comprised predominantly of fine sands and silts. However, the distribution of these soils (Figure 11) does not correspond with areas of highest erosion rates as predicted by the MMF model (Figure 28). The MMF tends to predict low erosion rates for light soils (e.g. see coastal areas in East Anglia), probably because the model outputs are strongly sensitive to the amount of runoff generated by the water phase of the model (Appendix 7.7). Light soils can generate limited surface runoff (and therefore soil erosion) due to their high infiltration rates. The high erosion rates seen in the SW (e.g. around Exeter and the east Midlands, especially in 1969 and 1988) are not reflected in the soil map, which categorise these areas as 'medium' textured soils. This might suggest that a more detailed categorisation of soil types beyond the over-simplistic Cross Compliance classes might pick up these more localised (i.e. smaller spatial scale) patterns in soil type.

2.4.3 Spatial and temporal patterns in predicted erosion rates and slope gradients

No temporal analysis is possible as slope gradient at the national scale is unlikely to change at all over time. The spatial distribution of predicted erosion rates (Figure 28) does not appear to reflect slope gradient averaged over the 2 km cells (Figure 12), although the steep slopes around the coastal areas of SW of England do correspond with some of the highest predicted rates of erosion in 1969 and 1988. Other areas of high predicted erosion rates (for example, East Midlands; Figure 28) do not correspond with high average slope gradients.

2.4.4 Spatial and temporal patterns in predicted erosion rates and land use

From Figure 34 there is a general trend of increasing proportion of land in the 'very low' erosion class and a general reduction in the proportion of land in the 'moderate' erosion rate category. The increase in the 'very low' category might be explained by the increase in land under oilseed rape, defined as a low risk crop (Table 6; Figure 24). The trends in Figure 24 show a slight reduction in high risk crops (potatoes and vegetables) over time, corresponding to a reduction in predicted erosion rates. However, there is also a marked increase in maize area by 2010, associated with high risk of erosion (Section 2.1.4; Boardman and Favis-Mortlock, 1993). However, this is not reflected in a corresponding increase in the proportion of land in the 'high' or 'very high' erosion class for that year (Figure 34). In summary, using the current methodology, changes in land use and in modelled soil erosion (extent and rate) from 1969 to 2010 are not very evident.

2.5 Mapping agricultural land using ALS

The purpose of this component of the study was to ascertain how areas most at risk of soil erosion vary by grade of agricultural land (as expressed in the Agricultural Land Classification, ALC) over space and time. The maps of predicted soil erosion classes (based on the modified Morgan, Morgan and Finney model) were intersected with the provisional ALC layer (Figure 35). The area and proportion of erosion class in each ALC class for each year are shown in Figure 36 and Figure 37, respectively.

Due to the similarity of predicted erosion in all years of interest, the area of each ALC class affected by the different classes of erosion does not vary greatly over time (Figure 36). The proportion of Grade 1 and Grade 2 land within the 'high' erosion rate class is highest in 1969, compared with the other years sampled (Figure 37). Grade 5 land always has the highest proportion of land classed as having low and very low erosion, and this proportion tends to increase over time (Figure 37).

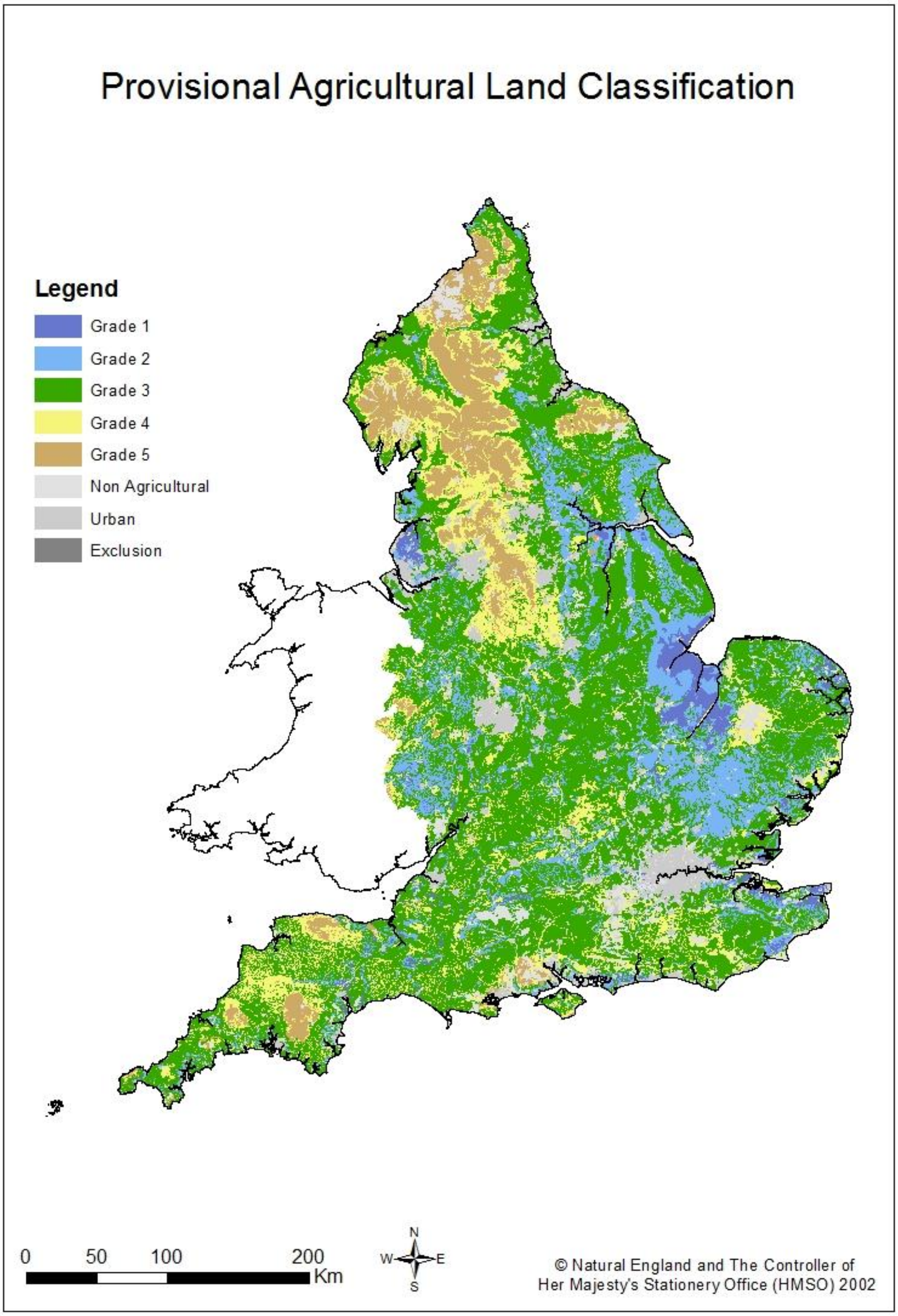

Figure 35 Provisional Agricultural Land Classification map for England.

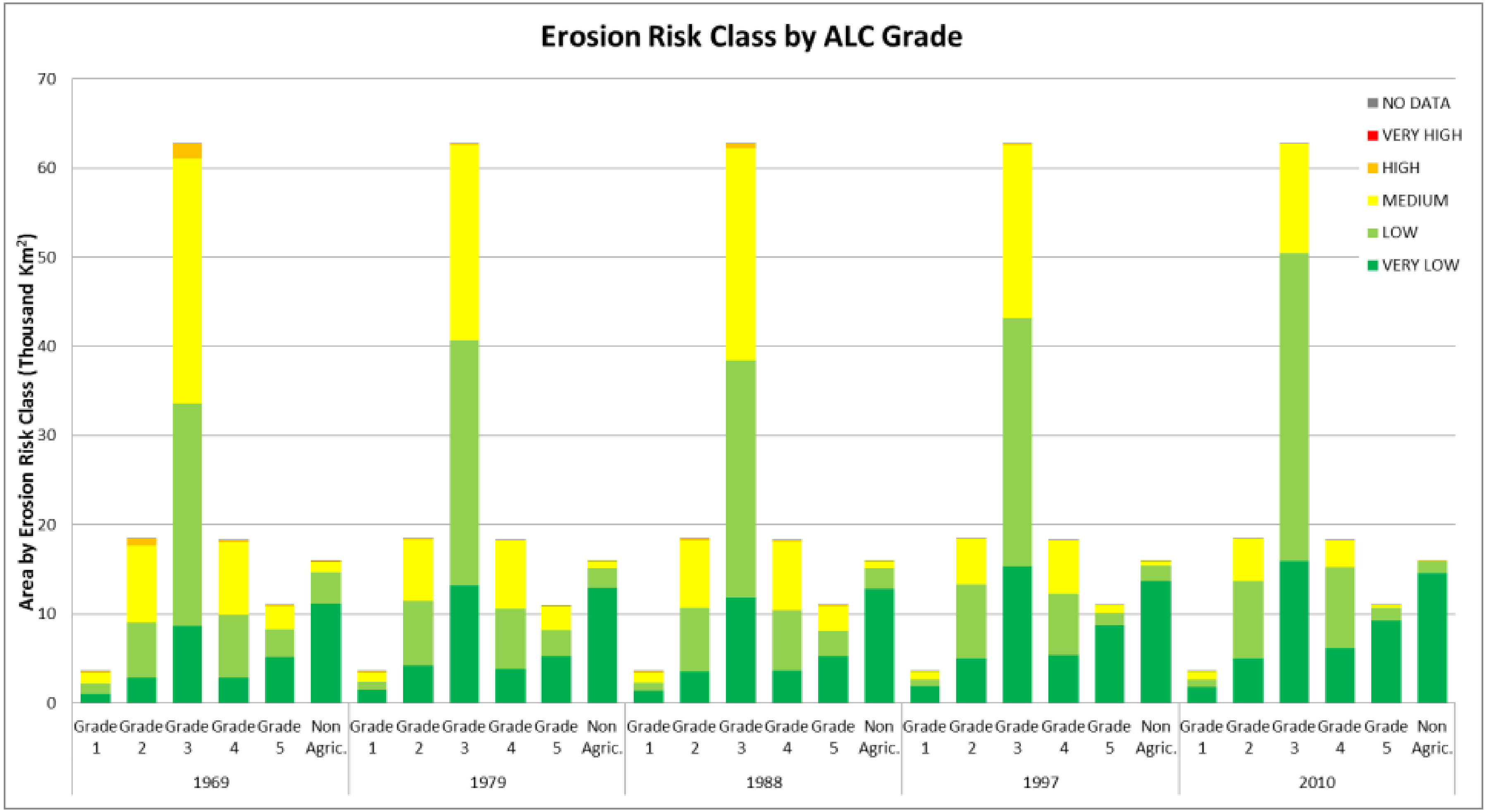

Figure 36 Area of ALC class affected by different predicted erosion rate class (as predicted by the modified Morgan, Morgan and Finney model.

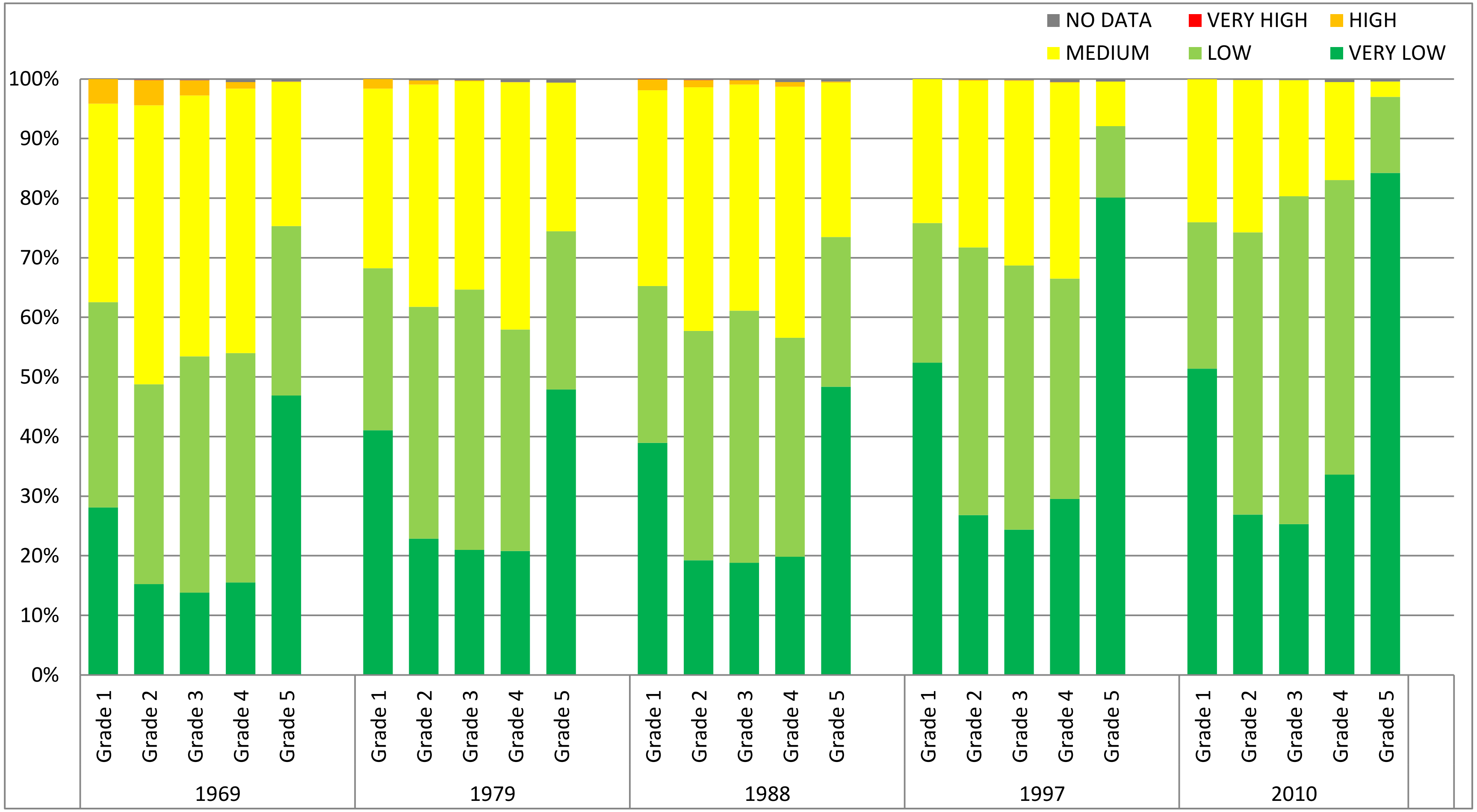

Figure 37 Proportion of ALC class affected by different predicted erosion rate class (as predicted by the modified Morgan, Morgan and Finney model).

2.6 Methodological limitations

Annual rainfall may not be the most appropriate index of erosivity (defined as the ability to cause erosion). Storm data such as intensity/duration relationships are likely to be better correlated with erosion risk, but these are difficult to map at the spatial (national) and temporal (annual) scales used in the present analysis. A simple erosivity index such as the Modified Fournier Index (MFI) should give more reliable results of changing erosivity over space and time, but this is an annual estimate which does not account for the event-driven nature of erosion processes. Also, data on soil erosion occurrence over space and time are not available to validate the MFI as a reliable and accurate indicator of rainfall erosivity. In any case, land use (itself a function of rainfall), slope and soil type are also important factors affecting erosion risk in addition to rainfall.

The analysis assumes that soil type and slope characteristics are constant over time. In reality, many soil properties (e.g. organic matter content, soil biota, bulk density etc.) may be highly dynamic in space and time (e.g. seasonal differences), but these are difficult to monitor at the spatial and temporal scales considered in this project. Also, slope length and form (concave, planar, convex or complex) were not included in the analysis as these are difficult to map at the spatial resolution used (2 km / 400 ha cells). Slope length can be important in determining critical thresholds for erosive flow and for sediment deposition processes. Again, this can change over time as a result of field boundary changes for example.

Caution should be taken when comparing land use changes over time, as land use categories used are not consistent between years, and in some years some crops are not recorded. Recent trends in changing land use are not captured at the spatial resolution of 2 km grid cells used in the present analysis. This includes the increased area under erosive crops such as maize, which is likely to increase erosion risk on a more local basis, compounded by changing weather patterns and more extreme weather events becoming the norm (Boardman and Favis-Mortlock, 1993).

Mapping individual factors affecting erosion risk separately can lead to poor correlations with erosion frequency and magnitude. This is because erosion occurs as a result of a combination of risk factors and the complex, often site- and time- specific interrelationships between them. Similarly observing trends in single factors over time and correlating these with erosion events is probably over-simplifying the processes operating.

The database of actual erosion occurrence at the national scale is limited, so modelled estimates of the spatial and temporal variation in erosion rates have been generated using the modified Morgan, Morgan and Finney model. The results need to be validated with actual field observations of soil erosion rates, which are notoriously unpredictable in space and time, and the geomorphological processes involved operate at sub-field scales.

The present analysis has shown that the modelled estimates of soil erosion rate at the national scale do not capture the site- and event- specific nature of erosion events. The 2 km resolution (= 400 ha) is too coarse for assessing the physical processes operating (i.e. soil detachment, transport and deposition) that take place at the field spatial scale (or finer). Local variations in rainfall intensity, duration and kinetic energy, soil conditions, slope and land use within this sampling unit will determine where and when erosion takes place (and at what rates). The modelled output gives mean annual rates of erosion; there will be considerable variation around this mean estimate, due to the event-driven nature of erosion processes that are highly variable in terms of magnitude and frequency within each year.

2.7 Conclusions: soil erosion analysis

The factors that influence the vulnerability of land to soil erosion have been reviewed extensively in the literature. Risk factors at a particular site at a particular time include the intensity, duration and timing of rainfall events (erosivity); the physical, biological and chemical properties of soils (erodibility); the length, gradient and form of slope; the type of vegetation / crop on the land and its stage of development; and the type and timing of singular or combined land management practices. Whilst most of these factors have been mapped separately in this study, the spatial distribution of each risk factor in isolation does not indicate areas of England with different erosion risk, due to the interdependence and interactions of the factors that influence the vulnerability of land to soil erosion.

The present analysis shows poor correlation between rainfall (when expressed as annual precipitation or as a recognised index of erosivity) and the spatial distribution and rates of soil erosion as estimated by an erosion prediction model for 5 years of interest (1969, 1979, 1988, 1997 and 2010). Soil type (defined by Soil Association) will influence the spatial distribution of erosion risk, but a direct spatial correlation is complicated by other risk factors such as land use.

According to the JAC/EDINA land use data (2 km resolution) used in this study, the total crop area in England occupied by high erosion risk crops such as vegetables and potatoes did not change greatly between the 5 years under investigation. The greatest area under high risk crops in England occurred in 1969, which corresponds to the year with the highest percentage of land modelled as being at 'high' erosion risk. The evidence of increasing land area under a potentially erosive crop (maize) is not reflected in the EDINA data when presented at the national scale.

The methodology used shows the proportion of Grade 1 land that is subject to 'moderate' or 'high' erosion rates does not vary greatly between the 5 years of interest. If the predicted rates are reliable, this implies that the quality of Grade 1 has not been degraded by soil erosion.

The current analysis has identified areas in the country where soil erosion risk may be elevated due to the combination of risk factors, even if this is on a relative rather than absolute basis, due to the unvalidated nature of the modelled output. As suggested by Kibblewhite et al. (2014) this methodology could be developed into a tiered approach to erosion risk assessment, where greater effort is focussed on areas of relative high erosion risk. In other words, specific areas can be identified where the combination of risk factors is likely to lead to unacceptable rates of soil erosion. This is the first stage of developing and implementing measures to control the irreversible loss of the soil resource through erosion.

Patterns or trends of soil erosion rates over time cannot be drawn from the analysis of 5 distinct years. Although the outputs of the erosion prediction model are unvalidated, when run for each year separately, the model suggests that soil erosion rates have not changed markedly for the 5 years under investigation. However, a number of authors suggest that the extreme weather events and shifts in land use / cropping patterns associated with climate change will bring about accelerated rates of erosion in the future.

These conclusions are based on a number of assumptions. These have had to be made, given the limitations of the methodology that in turn are the consequence of the paucity of empirical observations of soil erosion over time at the national scale.

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3. PART II: WATER USE IN AGRICULTURE

3.1 Introduction

UK agriculture accounts for about 75% of the total land area (Angus et al., 2009), including both cropping (arable, horticulture) and livestock (beef, dairying, pigs, poultry). It is strategically important for food provision, providing a substantial proportion of food consumed in the UK. Defra (2011) suggested that the total volume of water used in agriculture, through mains water use and direct abstraction was around 180 Mm^3 per year, with outdoor irrigation of field crops accounting for 42%, followed closely by drinking water for livestock at 40%. Although most crops in England are rainfed, supplemental irrigation could become much more economically important and nationally widespread if adequate water resources were available, not only for high-value field scale vegetable and potato cropping, but also on cereals where crop development is known to be impacted by extreme temperatures and drought stress (Knox et al., 2012). Recent research at Cranfield (El Chami et al., 2015) has re-evaluated the economic viability of irrigating wheat in the UK and highlighted the yield and water resource implications. A changing climate is likely to increase agricultural risks (Knox et al., 2012; 2010) and lead to greater uncertainty in future water demands, from both irrigated cropping and livestock production. Since over half of all irrigated production is located in catchments which are defined as ‘over-abstracted’ and/or ‘over-licensed’ (Hess et al., 2011) there are understandably major government, water regulatory and industry concerns regarding the environmental impact that any future increases in demand might have on water resources. A changing climate is likely to exacerbate the situation and make direct summer abstraction less reliable.

The Climate Change Risk Assessment (CCRA, 2012) identified a number of risks for cropping and livestock in agriculture; with reference to water use and availability, two risks were highlighted, (i) potentially large increases in water demand for supplemental irrigation of outdoor crops, and (ii) an increasing number of unsustainable water abstractions. These two risks have already received significant attention and detailed assessment by Cranfield staff as part of two recent projects for the Department for Environment, Food and Rural Affairs (Defra), including FFG1129 Assessing climate and land use impacts on water demand for agriculture and opportunities for adaptation: current and future demand (Phase I) (Knox et al., 2013) and FFG1112 Water for agriculture: collaborative approaches and on-farm storage (Weatherhead et al., 2014).

The objectives of FFG1129 (Phase I) were to model and map current water demand (for irrigated cropping and livestock) and to assess the impacts of climate and socio-economic change on future water demand (2030s and 2050s). Many of the core datasets and outputs generated in that work (for current conditions) are of direct relevance in this study, including for example, high resolution spatial datasets on land use, livestock stocking rates, water abstractions for agriculture and geospatial data on soils and agroclimate. FFG1129 also provided a first order estimate of irrigation abstraction ‘hot-spots’ in England and Wales based on a GIS analysis of theoretical irrigation demand and Environment Agency (EA) National Abstraction License Database (NALD) data. In a parallel study (FFG1112) Weatherhead et al. (2014) investigated opportunities for water resource collaboration and uptake of on-farm storage reservoirs, and correlated these ‘hot-spots’ against the location of abstraction licenses for irrigation storage reservoirs. In this study, an alternative improved approach to identify and map 'hot-spots' is presented.

One of the CCC objectives in this study was to better understand the potential hot-spots. The ASC therefore requested a re-run of this analysis at catchment scale, using the latest projections of current and future water availability in the Environment Agency's Case for Change analysis. Addressing this objective was achieved by drawing on the derived datasets from FFG1129 and reworking them using updated (2011 to 13) EA licensed and abstraction data. Assessing changes in historical water demand for livestock was based on estimated livestock numbers (by catchment) and unit water requirements based on our previous work with ADAS, and the pig and livestock levy boards (EBLEX and DairyCo). These outputs then provided the basis for providing a critique on the drivers of water demand.

It is stressed that significant effort was involved in collating, pre-processing and analysing datasets for FFG1129 and FFG1112 – given the short time frame for this project (2 months), the focus was on updating these datasets to provide new insights on underlying trends. This study also benefitted from an extensive inventory of data on rainfed and irrigated cropping, livestock statistics and water demand collated by Weatherhead et al. (2014) as part of their work to provide agricultural demand forecasts for the Environment Agency's most recent water resource strategy (2013). New outputs on mapping 'irrigation intensity' and correlating livestock demand to resource availability were also completed.

3.1.1 Study aim and objectives

The project aim was to provide new evidence and data to CCC/ASC to inform their understanding of the challenges in reconciling future water supply/demand imbalances in two agricultural sectors - irrigated crop production and livestock. Most of the datasets used relate to England and Wales (E&W), with outputs provided at national (E&W), regional (EA) and local (EA catchment) scales.

In our analyses, we have provided detailed evidence on the current 'baseline' situation (in most cases using 2010 as a reference baseline) and then provided analysis of trends for key variables as far as back as data allows, mostly back to the early 1970s. The study had three specific objectives:

1. To better understand the potential hot-spots for agricultural irrigation abstraction, corresponding to areas where the supply-demand balance is under greatest stress ;
2. To better understand the vulnerability of livestock production to reduced water availability, and;
3. To better understand what actions are being taken by the livestock sector to mitigate the risks from future reduced water availability.

3.2 Methodology

The following approach was adopted:

3.2.1 Develop a time series of total water demand for irrigated crop and livestock production

In this study, we have used water used for irrigation as a proxy for water used for irrigated crop production. Therefore, other on-farm water uses, such as water used for crop spraying, crop washing or sanitation have been excluded. A time series trend of crop irrigation demand for England were developed drawing on a wide range of sources including national and regional level data from the Ministry of Agriculture, Fisheries and Food (MAFF) and more recent Defra statistics (June Cropping Census data (as EDINA gridded 2 km^2 data) and Defra Irrigation Survey data). Cranfield were responsible for undertaking recent Irrigation Surveys for Defra so are well acquainted with the nuances and inclusions/exclusions inherent in these datasets. We have had access and wide experience in using long-term historical datasets held with the EA National Abstraction Licensing Dataset (NALD). We also have a number of derived long-term agrometeorological datasets (including 30 year daily time-step rainfall and Penman-Monteith reference evapotranspiration (ET) data) for a network of sites across England from which we have derived a suitable agroclimate indicator (Potential Soil Moisture Deficit, $PSMD_{max}$) to help interpret historical trends in crop water demand taking into account inter-annual weather variability. Note that using rainfall only to inform crop demand trends was too simplistic an approach for UK climatic conditions– a water balance model developed at Cranfield (Wasim) was therefore used to provide an historical 'theoretical' crop water demand trend for comparison against actual observed (reported) trends. This helps provide evidence to explain the drivers for demand and responses to weather variability.

Unfortunately, published data and records on mains-water use in agriculture is much more patchy – but we do have some limited, derived, datasets from previous studies for the Agriculture and Horticulture Development Board (AHDB) Horticultural Development Company (HDC) on mains-water use in horticulture (Knox and Hess, 2013) which will be re-analysed to assess the spatial and temporal distribution of mains-water dependent crop irrigation demand in England (within the protected cropping, soft fruit, and Hardy Nursery Stock. Some data is also available via the Defra Irrigation Surveys which was combined to provide broad estimates of mains-water use dependency for crop demand.

Water for livestock comes from a range of sources. Mains-water is frequently used for drinking water and yard wash-down. This may be supplemented by harvested rainwater, small borehole supplies and direct drinking from water bodies. There are no direct statistics on water use for livestock as water meter readings reflect a range of both farm and domestic water uses. We have therefore estimated total water use for livestock in England from livestock numbers (by type and age) from the gridded June Census data expressed by region and catchment combined with per head water for beef, lamb, dairy pigs and poultry. There is little observational data on water use by source but we drew on surveys from DairyCo, EBLEX and ADAS. In this study, we have only considered water used to directly support animal production (e.g. drinking and wash-down) and not water used in the processing of animal products. Water used for cooling in dairy parlours or meat processing, for example, has not been included.

3.2.2 Comment on the drivers of demand

The trend analyses generated for direct and mains abstraction for crop irrigation and livestock demand were then reviewed and assessed to identify the drivers of demand. We have drawn on extensive experience in understanding the nature and composition of irrigation demand in England and the drivers for their variability and published widely in the science literature (e.g. Knox et al., 2009; 2010; 2013; Hess and Knox, 2013), but it was nevertheless important to validate our commentary with industry opinion.

For this, we engaged with a number of key informants in the agriculture and horticulture sectors with whom we are closely connected to elicit their industry insight and feedback. The following stage (mapping hot-spots) also provided useful evidence to inform our assessment of the drivers of demand.

3.2.3 Map water demand spatially at catchment level against Environment Agency Case for Change categorisation of water resource status

The ASC (2013) progress report reported on future imbalances between supply and demand of water for crop irrigation drawing mainly on data from previous published research, and aggregated to the WFD river basin district level. In this study, a similar but updated analysis was undertaken to spatially map both irrigation and livestock water demand hot-spots at the EA CAMS catchment scale. This used spatial datasets and analyses described above combined with projections of current and future water availability from the EA published Case for Change analysis.

The GIS methodology to model and map irrigation hot-spots, which was originally developed in FFG1129, has also been modified to assess hot-spots based on a new indicator termed ‘irrigation intensity’ (m^3 per km^2) rather than solely on volumetric demand (m^3 per catchment). This conveys more accurately where there is high demand per unit of irrigated land use in water stressed sub catchments.

No equivalent dataset or mapping had previously been conducted for the livestock sector, so our hot-spot approach for irrigated cropping was extrapolated for livestock. New data and maps to support ASC publications on water demand for crop irrigation and livestock have been produced.

3.2.4 Assess uptake of actions to reduce water use in livestock production

In addition, mainly in response to comments received as part of a consultation on indicators, the ASC wished to better understand the vulnerability of livestock production to reduced water availability. Per-head water requirements for livestock are fairly conservative - being a function of animal weight, air temperature and farm hygiene and sanitation practices. Livestock farmers cannot compromise on drinking water, therefore, on farm water use per head is only affected by farm practices and efficiency. To a point, reduced water availability may drive innovation and investment in on-farm water efficiency (reducing leaks and wastage, water recycling and use of low water use coolers in dairy parlours), but beyond that the primary option is to change system or reduce livestock numbers. We reviewed the available evidence on regulatory and industry initiatives on water conservation.

3.3 Time series for water demand for irrigated cropping and livestock and drivers of demand

Historical trends in water demand for irrigated cropping and livestock production in England and Wales are summarised below. An assessment of current spatial demand together with an analysis of underlying trends and commentary on the drivers of demand are provided.

3.3.1 Trends in summer agroclimate

Irrigation demand in England and Wales is supplemental to rainfall and therefore varies from year to year depending on summer weather. In order to analyse past trends in annual irrigation water demand it is necessary to allow for the inter-annual differences in summer weather. In this study, the spatial and temporal variability in agroclimate was assessed using an aridity index termed maximum Potential Soil Moisture Deficit ($PSMD_{max}$). This is calculated from daily data for rainfall (P) and reference evapotranspiration (ETo), the driving variables that influence soil moisture, and hence the need for supplemental irrigation (See Appendix). The agroclimate index $PSMD_{max}$ has been widely used previously to assess irrigation requirements at national and regional scales in many different countries (e.g. Knox et al., 2010; De Silva et al., 2007; Rodríguez Díaz et al., 2007; Knox et al., 1997; Hess et al., 2015). It is also used by the regulatory authority (EA) in England and Wales for assessing the reasonable needs for setting abstraction licenses for spray irrigation (Rees et al., 2003).

The advantage of this index over others such as the Wetness Index (ratio of total annual rainfall and total annual evapotranspiration) is that the distribution of rainfall and ETo throughout the year is taken into account. Furthermore, in many countries where spatial information is sparse or non-existent, the $PSMD_{max}$ index is more appropriate than the Palmer Drought Severity Index (PDSI) which requires detailed spatial soils information (Narasimhan and Srinivasan, 2005). It is important to note that $PSMD_{max}$ is purely an agroclimatic indicator and does not take into account crop type or soil variability.

The Wasim model, a daily time-step water balance model developed at Cranfield University (Hess and Counsell, 2000) was used to estimate annual $PSMD_{max}$ for a site in central England (Silsoe, Bedfordshire) using a long-term (30 year) historical time series of weather data. Figure 37 shows, for example, how the PSMD develops over the growing season and can vary significantly from year to year depending on summer rainfall. Evapotranspiration (ET) is also an important driver of water demand during the summer but does not vary as much as rainfall.

For this site, the $PSMD_{max}$ in a dry year (2006) was 223 mm. In contrast, in 2008 (a wet year) it was 141 mm; for 2010 it was 217 mm. By identifying the $PSMD_{max}$ in each year, the inter-annual variability in agroclimate for a given site can be compared to identify periods when irrigation demand would have been low or high.

When analysing historical trends in water use for irrigated cropping, it is important to recognise the impact that the choice of time series for analysis can have a major impact. For the same site, the annual $PSMD_{max}$ for a long time series between 1962 and 2012 is shown in Figure 39.The historical drought periods of 1975-76, 1988-92, 1995-96 and 2003 which were widely reported for England and Wales as a whole (Marsh et al., 2007) can be seen clearly.

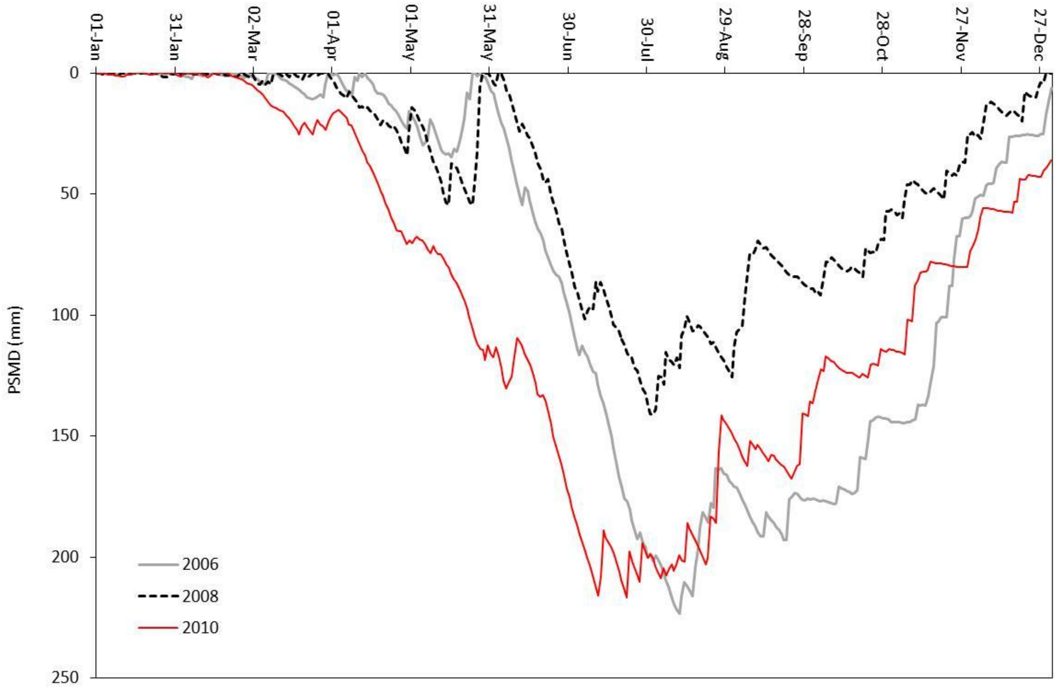

Figure 38 Potential soil moisture deficit (PSMD) on a daily time-step for selected years (2006, 2008 and 2010) at Silsoe (Bedfordshire). A relatively 'wet' year is represented by 2008 and a relatively 'dry' year in terms of aridity is represented by 2006. Data for a recent dry year (2010) is also shown for comparison.

The same data were then ranked (Figure 40) to highlight the range in agroclimate that exists for this site in central England. Similar trends for other sites can similarly be generated.

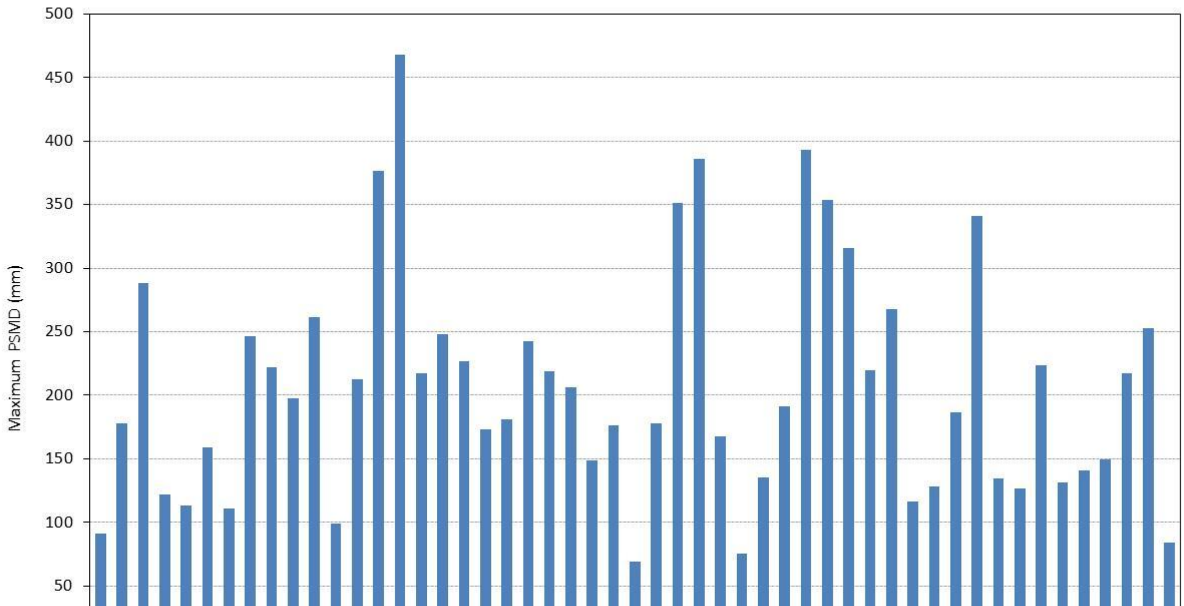

Figure 39 Modelled annual $PSMD_{max}$ at Silsoe (Bedfordshire) between 1962 and 2012.

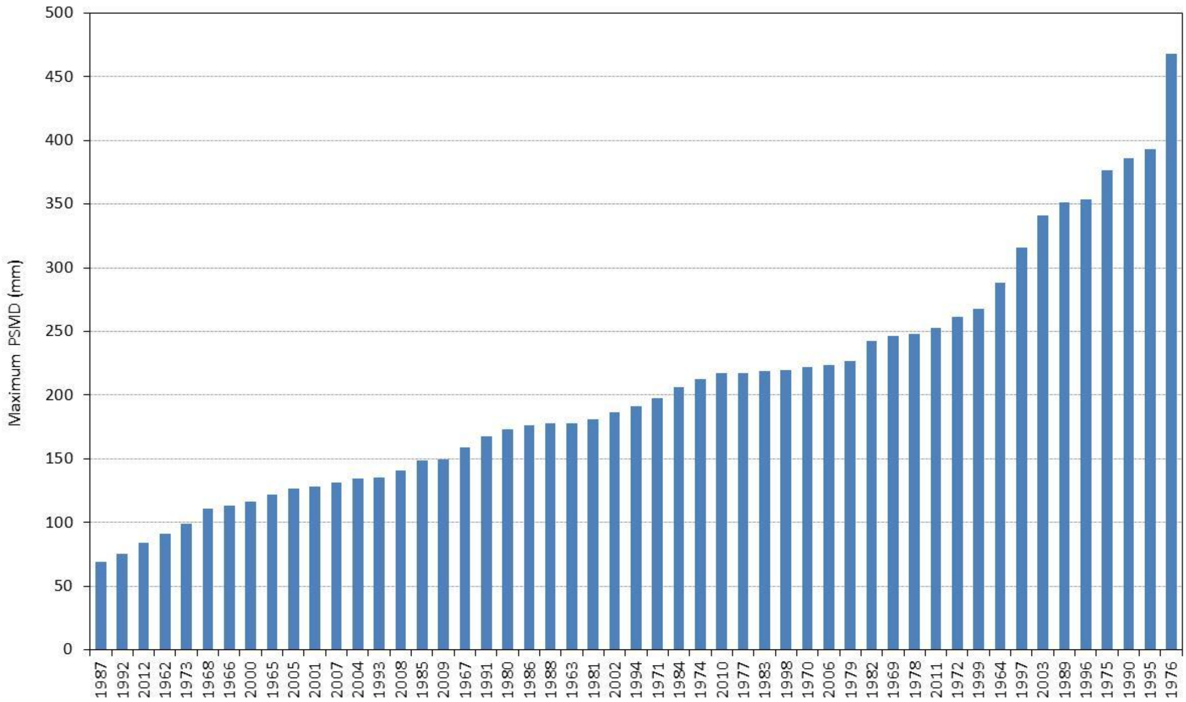

Figure 40 Modelled annual $PSMD_{max}$ at Silsoe (Bedfordshire) between 1962 and 2012 (ranked).

This analysis shows the significant inter-annual variability in $PSMD_{max}$ that can exist for a single site (Silsoe, Bedfordshire). But it also highlights the difficulty in drawing national inferences from a single site which is not located in the most sensitive agroclimatic region. To demonstrate the likely inter-annual variation in $PSMD_{max}$ that exists nationally, Knox et al (2013) conducted a regionally based assessment of PSMD variability (Appendix 4.10).

The spatial variation in agroclimate ($PSMD_{max}$) in England and Wales has also been modelled and mapped using a GIS and a $5km^2$ gridded climatology of mean monthly rainfall and ET, based on data for 1961-90 (Figure 41). The $PSMD_{max}$ values have been aggregated into agroclimatic zones. The agroclimatic zones with the highest $PSMD_{max}$ are located in the eastern and south eastern parts of the country, notably in Norfolk, Suffolk, Essex and Kent. These correspond to parts of the country where irrigation is most concentrated (Knox *et al.,* 1997; 2013) and where the reliability and availability of water resources for agriculture are under severe pressure (EA, 2010). In contrast, zones with the lowest $PSMD_{max}$ (< 75 mm) extend across much of Wales, the south west and north-west of England. The map is also useful for identifying regions or catchments where future water resource problems might arise. Individual agroclimate maps for 2005 to 2010 are shown in Figure 41, highlights how the agroclimate in England and Wales can vary markedly both spatially and inter-annually. For example, both 2007 and 2008 were considered relatively ‘wet’ years in terms of $PSMD_{max}$ at Silsoe (Bedfordshire) (Figure 39 and Figure 40); this is also reflected widely across the country in terms of low agroclimatic zones shown in Figure 41.

Conversely, the high $PSMD_{max}$ values for 2009 and 2010 (Figure 39 and Figure 40) are also reflected in Figure 41 when water resource constraints were experienced in eastern England, mainly due to a combination of low preceding winter rainfall and above average PSMD conditions.

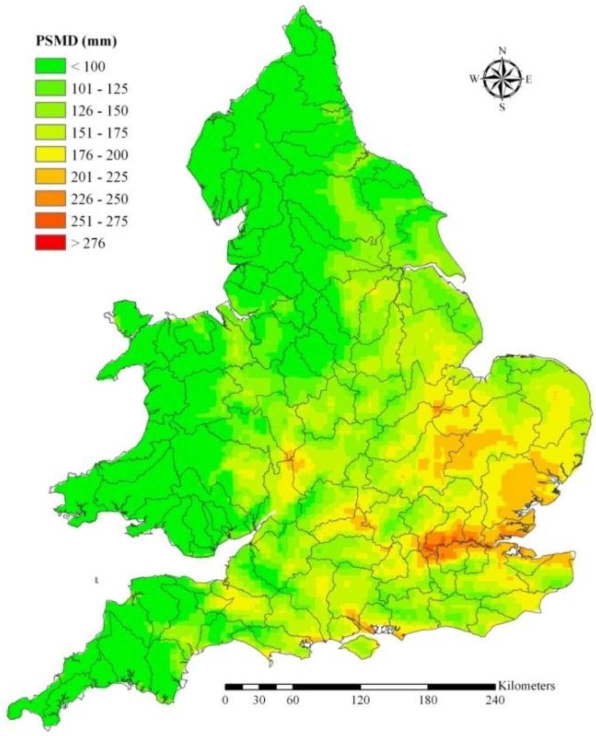

Figure 41 Spatial variation in average agroclimate ($PSMD_{max}$) based on 1961-90 for England and Wales. EA catchments are shown.

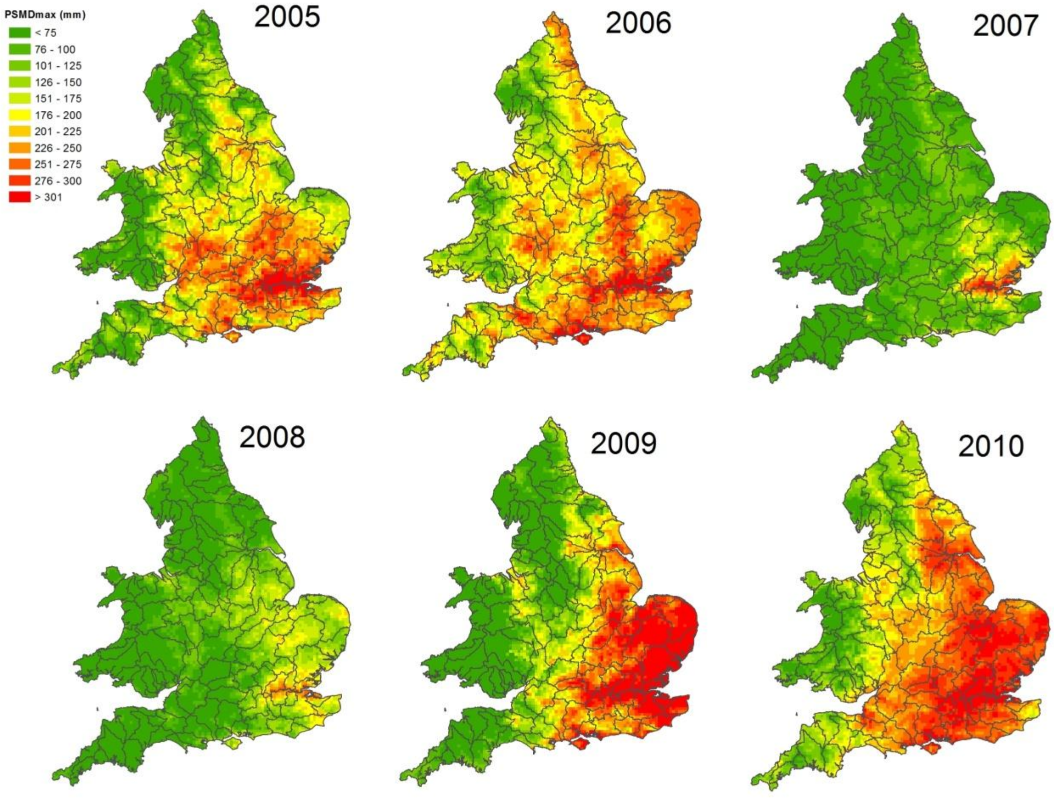

Figure 42 Spatial variation in agroclimate ($PSMD_{max}$) for England and Wales from 2005 to 2010. EA catchments are shown.

3.3.2 Trends in cropped area

The long-term trends in total cropped area (×000 ha), for selected crop categories, were derived from the annual Defra Agricultural and Horticultural Cropping Census data. Figure 43 to Figure 45 show these national trends at national (UK) level.

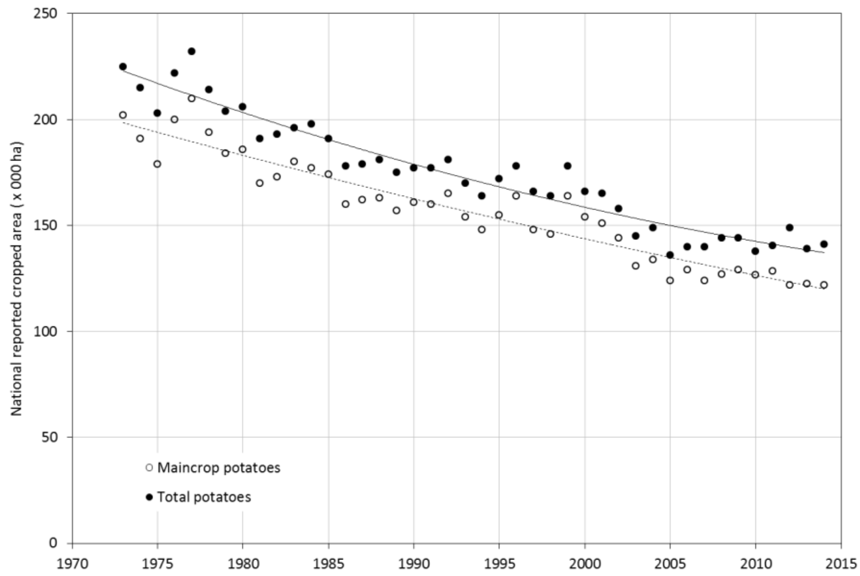

Figure 43 Cropped area (× 000 ha) trend for maincrop potatoes and total potatoes from 1973 to 2014 (Source: Defra (various) updated from Knox et al., 2013).

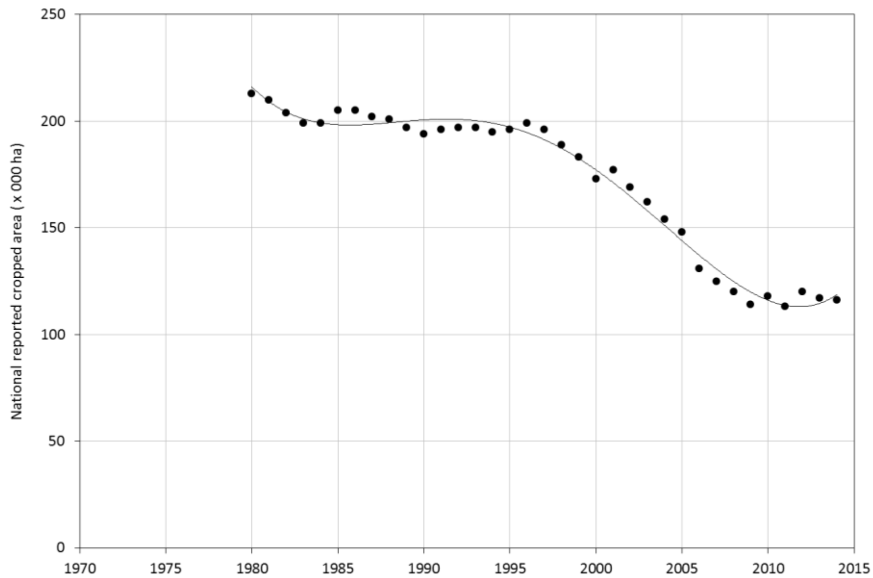

Figure 44 Cropped area (×000 ha) trend for sugar beet from 1980 to 2014 (Source: Defra (various) updated from Knox et al., 2013).

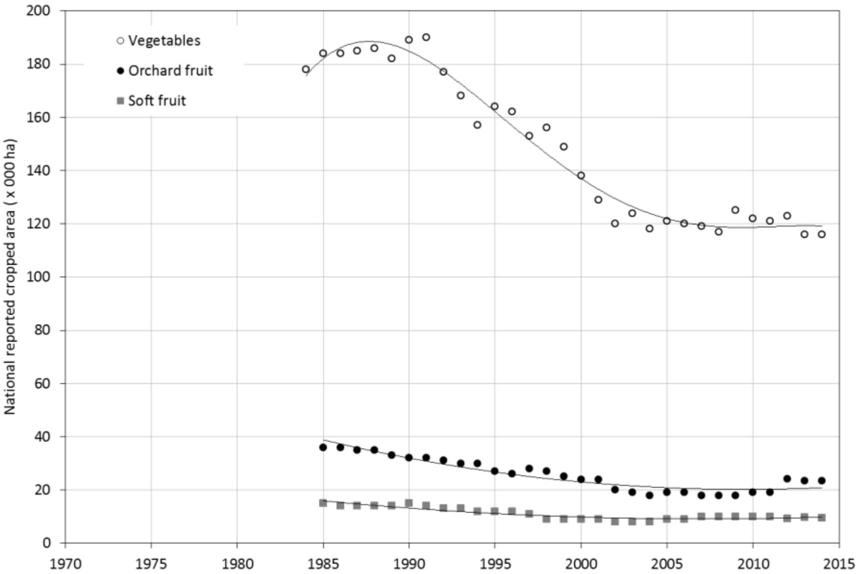

Figure 45 Cropped area (×000 ha) trend for field vegetables, soft and orchard fruit from 1985 to 2014 (Source: Defra (various) updated from Knox et al., 2013).

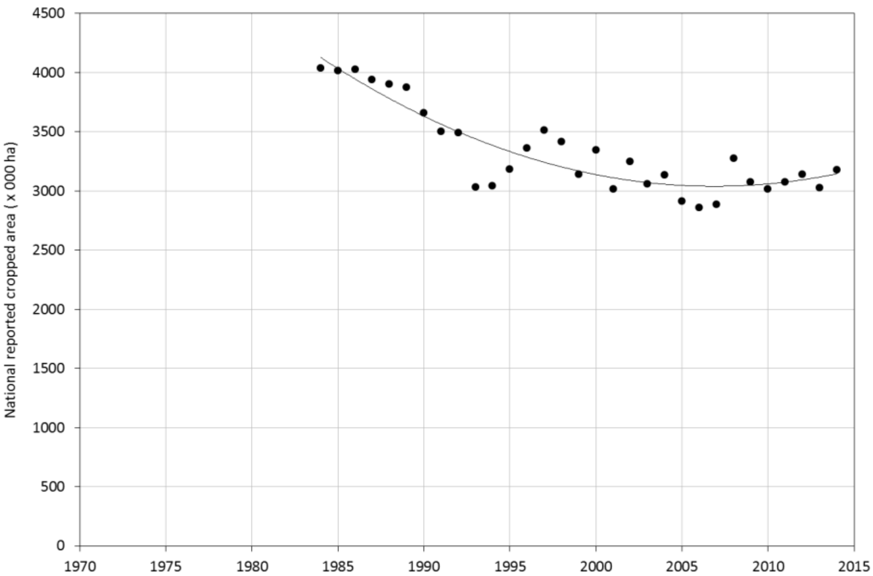

Figure 46 Cropped area (×000 ha) trend for cereals from 1984 to 2014 (Source: Defra (various) updated from Knox et al., 2013).

For all crop categories studied, a decline in cropped area is observed over the last 40 years. This is largely due to a combination of improvements in agronomic management, uptake and adoption of new technologies and better varietal choice and selection, leading to increased crop yields. Sustained or in most cases increased levels of productivity (t/ha) nationally can therefore be attained from a much reduced cropped area (ha). For potatoes, the decline has been steady since the early 1970s, whereas for vegetables and fruit, the cropped area seems to have plateaued since 2000. For cereals the decline mainly occurred before the mid-1990s, and for sugar beet, the decline has occurred since the mid-1990s.

Major restructuring in the UK agricultural sector and reforms in EU agro-economic policy have also impacted strongly on production trends, with crop sector consolidation and increases in farm size. A more detailed assessment of the combination of these factors can be seen within the UK potato sector. The viability of commercial potato production is influenced by spatial and temporal variability in soils and agroclimate, and the availability of water resources where supplemental irrigation is required. Soil characteristics and agroclimatic conditions greatly influence the cultivar choice, agronomic husbandry practices and the economics of production. In England and Wales, the potato industry has changed dramatically in recent decades, from a sector comprised of many small individual farms to one with far fewer but much larger agribusinesses, driven by the need to provide high quality product to the major processors and supermarkets (Knox *et al.,* 2010a).

An analysis of UK potato industry data shows that in 1960 there were ≈76,000 registered growers cultivating 280,000 ha potatoes. By 2013, the number of registered growers had dropped 97% to 2,300 and the cropped area had declined by 57% to 120,000 ha. Average yields, however, had more than doubled over the same period from 23 t ha^{-1} to 48 t ha^{-1} (AHDB Potato Council, 2013) (Figure 47). Note the low yield in 2012 is may be the result of the drought followed by waterlogging.

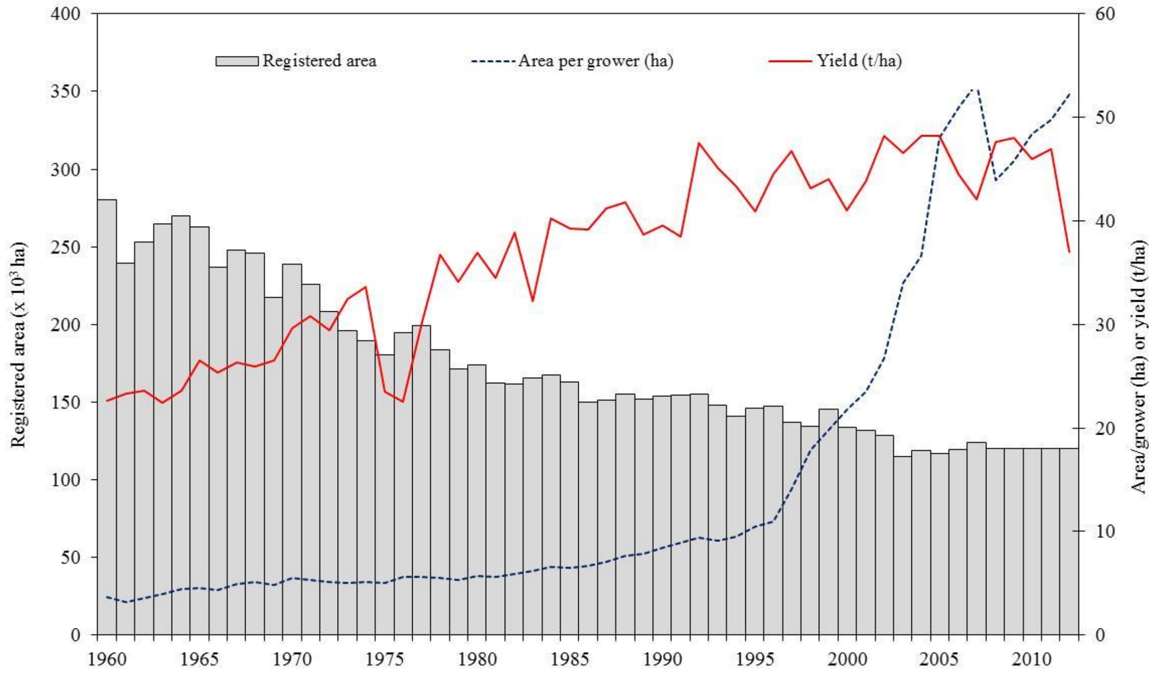

Figure 47 Reported total potato cropped area (ha), average cropped area per grower (ha) and average yield (t ha^{-1}) in the UK, 1960 to 2012 (Source: Potato Council, 2013).

3.3.3 Trends in irrigation water demand

There are various ways of defining irrigation water demand. In this study, two definitions are considered, termed *actual* demand and *theoretical* demand. Actual demand is based on the gross depths farmers have historically applied, as reported in the Environment Agency (EA) water abstraction returns and Defra Irrigation Survey data. It therefore reflects directly the irrigation practices that farmers found realistic, and includes the effects of equipment constraints, water shortages, scheduling errors, and the farmers' scheduling assumptions on irrigation losses. In contrast, theoretical demand (modelled above) is based on the calculated agronomic water requirements of the crops that are irrigated, assuming they are fully irrigated following industry scheduled recommendations.

Theoretical need

In order to understand and explain the historical pattern of agricultural irrigation demand it is first necessary to understand the relationship between crop water use (demand) and agroclimate (PSMD), both of which vary spatially and temporally. In this study, the water balance model Wasim was used to estimate the annual irrigation needs, using maincrop potatoes as a reference crop as irrigation of potatoes accounts for >50% of the water use for irrigation in E&W. The absolute requirement from other crops would vary but, as the model is driven by weather data, would be proportional to the figure for potatoes.

The model estimates the daily soil water balance for the selected crop and soil type, working from rainfall and reference evapotranspiration (ETo) data and pre-defined rules for irrigation. For each modelled year, outputs on crop water use, any irrigation applied and the proportional yield loss due to any water stress are generated. The Wasim model was run for a single station (Silsoe, Beds) and for a single crop/soil type permutation, using historical ETo and rainfall data for a 50 year period (1962-2011) (Figure 48).

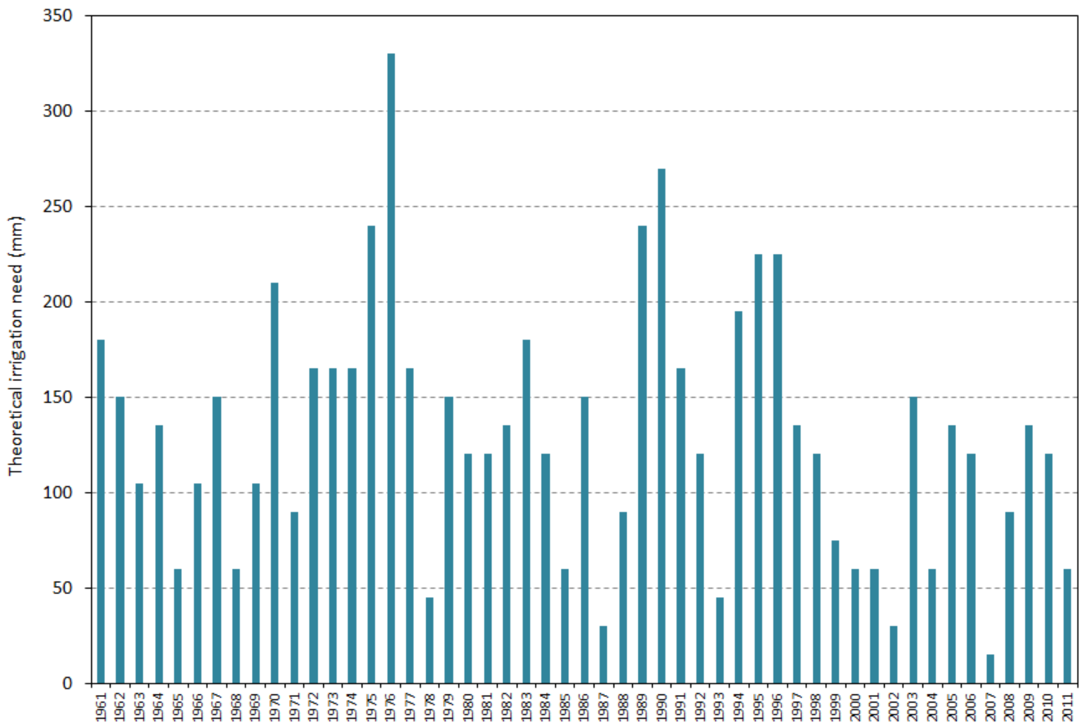

Figure 48 Modelled theoretical irrigation needs (mm) for maincrop potatoes at Silsoe (Bedfordshire) between 1961 and 2011.

It is apparent that the modelled or 'theoretical' irrigation needs follow a similar pattern to the annual agroclimate variability. The 'dry' years in irrigation terms correspond closely to those when irrigation needs are highest, and vice versa. Previous studies confirm there is a strong correlation between agroclimate ($PSMD_{max}$) variability and irrigation need. A correlation between the $PSMD_{max}$ (Figure 39) and modelled irrigation needs (Figure 48) for this site was derived using linear regression analysis (Figure 49).

This is useful for explaining how past changes in agroclimate impact on water demands for irrigated cropping. Figure 49 explains the relationship between agroclimate and irrigation water demand and shows how high $PSMD_{max}$ values lead to high irrigation demand. The following section draws on this relationship to explain how there is a correspondingly strong relationship between historical EA reported abstractions for spray irrigation and annual agroclimate variability.

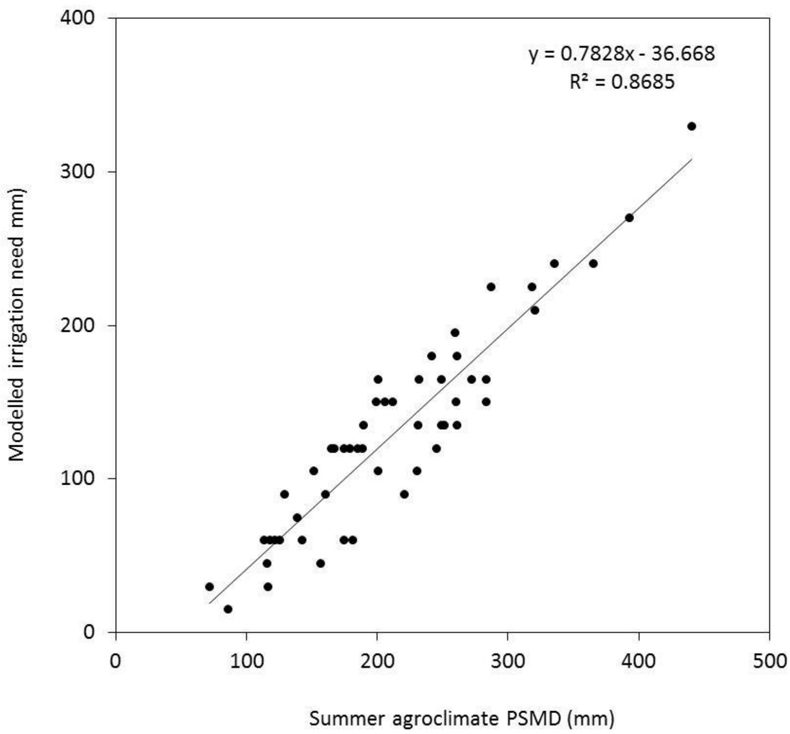

Figure 49 Example linear regression correlation between agroclimate ($PSMD_{max}$, mm) and theoretical annual irrigation need (mm) for maincrop potatoes on a sandy loam soil.

Licensed and abstracted volumes (from NALD/ABSTAT data)

Where irrigation is supplemental to rainfall, such as the UK, many crops are not irrigated, and even for crops that would benefit from irrigation, not all farmers irrigate. Furthermore, many farmers may apply less than agronomic demand, either because of equipment limitations, water limitations, or as a deliberate policy to maximise profit. In England, only a very small proportion of the total cropped area is actually irrigated. The extent of supplemental irrigation is a function of crop type, local soil and agroclimate conditions, water availability and cost, and the target market (e.g. processing or retail). The balance of these variables influence the economic viability of irrigation and the depths of water applied (mm). The volume of water abstracted (m^3) is thus a product of the crop area (ha), the proportion irrigated (%) and the total seasonal depth of water applied (mm).

Almost all irrigators abstracting >20 m^3d^{-1} are required to have an abstraction license and flow meter(s); those abstracting more than 100 m^3d^{-1} have to return data to the EA on their actual volumes abstracted. After statistical correction for non-returns and missing data, the aggregated results are published as ABSTAT data, available from the Defra website.

Historical data on the national licensed and actual volumes abstracted for irrigation purposes are published annually by the EA or processed from ABSTAT data published by Defra. Data on individual actual abstractions are confidential but can be made available for research purposes. The EA abstraction data excludes public mains supply and trickle irrigation, but can include indoor spray irrigation and spray irrigation for landscape and leisure, e.g. golf courses. Figure 50 shows the historical trend in national EA licensed and abstracted volumes for spray irrigation between 1973 and 2012 (data for 2013 will be published later in 2015).

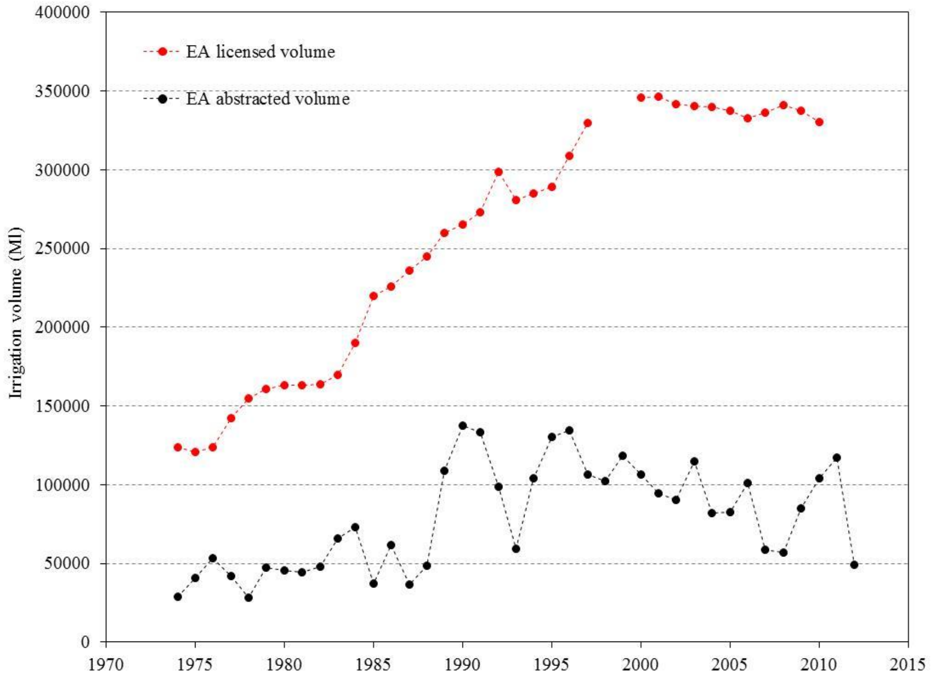

Figure 50 Historical trend in EA licensed and abstracted volumes of water for spray irrigation in England and Wales between 1973 and 2012 (updated analysis from Knox et al., 2013).

Until around 1998, the total volume licensed for spray irrigation was growing steadily at around 10,000 Ml/year, equivalent to around 3% of its 1998 value per annum.

Since late 1990s, however, the total licensed volume for irrigation has declined slowly, at about 0.3% per annum. This date roughly coincides with the recognition that many catchments in England were becoming over-abstracted leading to a moratorium on new licences being issued in many catchments and other significant changes in licensing policy and water regulation. Time-limited licences (particularly common in EA Anglian Region) have to meet increasingly stringent criteria before renewal. The current downward trend in licensed abstraction is expected to continue in response to meeting environmental objectives (e.g. in the EU Water Framework Directive) and major proposals for abstraction licensing reform (Defra, 2014).

This steady decline in the national licensed total does however disguise local and regional variation and significant ‘churn’ with some new licenses still being issued and others being reduced and/or relinquished.

In contrast, there is much more variation in the volumes reported by growers as actual abstraction. This is partly due to the weather (agroclimate) differences between individual years, as explained earlier. To allow for weather variation, the actual reported abstractions were correlated in a multiple regression analyses to the year and to each year’s $PSMD_{max}$ at a central site (Silsoe). The results show the underlying annual trend and an estimate of how much would have been abstracted each year if it had been a design dry year, taken as one with an 80% probability of non-exceedance (approximately the 16th driest year in 20, a standard widely used in irrigation design). This criterion is widely used in planning irrigation water resources, for example, for abstraction licensing and reservoir sizing; the farmer would then have adequate water for 80 years in 100, but would have to under-irrigate some crops in the other 20. This level of planned headroom is considerably less than for most other sectors.

The underlying decline in ‘dry’ year demand over the 1990 to 2010 period was estimated to be around 1.4% per annum (Figure 51). This probably partly reflects the tighter licence conditions and hence the increased cost of irrigation (e.g. on-farm storage reservoirs), but also higher yields and changes in crop demand.

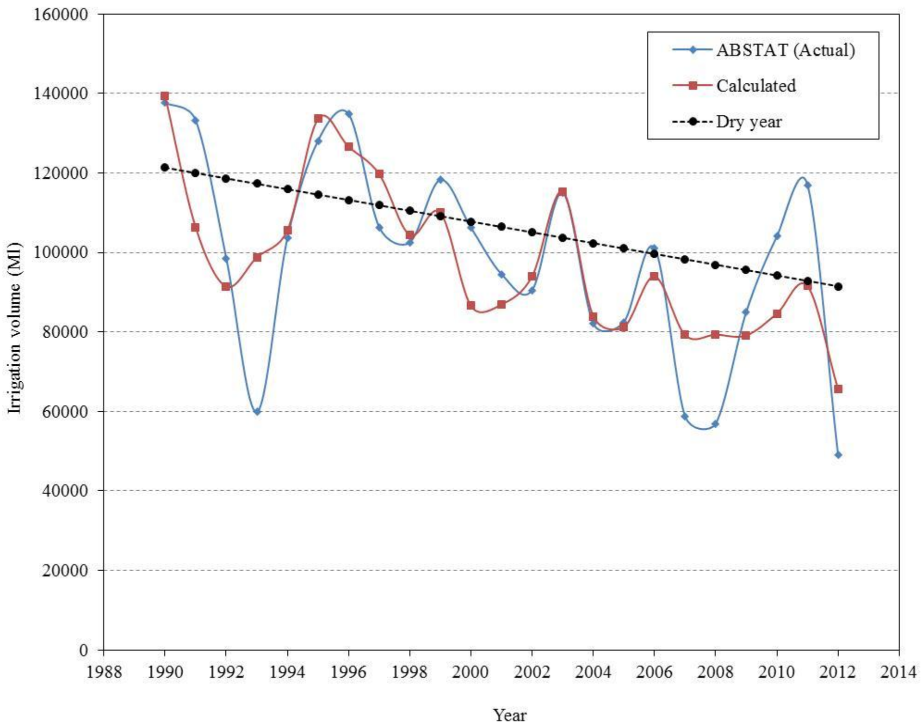

Figure 51 Underlying trend analysis of irrigation abstraction (Ml) between 1990 and 2012 with fitted curves showing calculated equivalent dry year abstraction (i.e. allowing for actual weather) and the underlying ‘dry year’ trend (updated analysis based on Knox et al., 2013).

Abstracted volumes (from Defra irrigation survey data)

More detailed data on irrigation use for individual crop types is available from the periodic Defra Surveys of Irrigation of Outdoor Crops. Since 1982 the main questions have been kept consistent, giving now nine sets of directly comparable data, for 1982, 1984, 1987, 1990, 1992, 1995 (all by MAFF), then 2001 and 2005 (for Defra by Cranfield University), and most recently in 2010 (by Defra). The data is broken down between 8 crop categories, by irrigated area and volume. Other data includes irrigation method, water source, and scheduling method. This data is much richer in content but is less complete, due to its intermittent nature and the lower return rate.

The address list for the irrigation survey is obtained from a trigger question in the annual Defra Agricultural and Horticultural Cropping census (June Census). However, that is only a full census in selected years (including 2010). Statistical corrections therefore have to be made for those not included in the June Census, non-returns to the June Census, and non-returns to the Defra Irrigation Survey. Corrections are applied at each stage, based on statistical analysis of replies from farms of similar sizes, based on the Defra Farm Business categories of Standard Labour Requirements (SLR). These corrections can be a major source of error for calculating national totals. In 2001, for example, the final returns were estimated to cover only about 40% of the total irrigated area. Returns also have to be cleaned for erroneous responses, e.g. using incorrect units. The survey also only refers to outdoor crops grown on registered agricultural holdings; it therefore excludes glasshouse crops, and landscape and other non-crop irrigation. It does include other water sources outside the NALD/ABSTAT dataset such as public mains supply, rainwater harvesting, water re-use, trickle irrigation and abstractions less than 20 m^3 day^{-1}, though these are relatively minor compared to direct abstraction from surface water or groundwater.

The total annual water use data from the Defra Irrigation Survey data shows a similar historical trend to the NALD/ABSTAT data, but they do not match entirely (Figure 52). This may be partly due to the different ranges of water sources covered and businesses surveyed, the different year end dates (affecting when reservoir refill is counted), and/or inaccurate returns from water users, but is probably mainly due to the difficulty of correcting for non-returns. Analyses based on the 1995 to 2010 Irrigation Surveys data appear to show very much faster rates of decline than the NALD/ABSTAT abstraction returns; however, their statistical reliability is lower, due to the limited number of surveys.

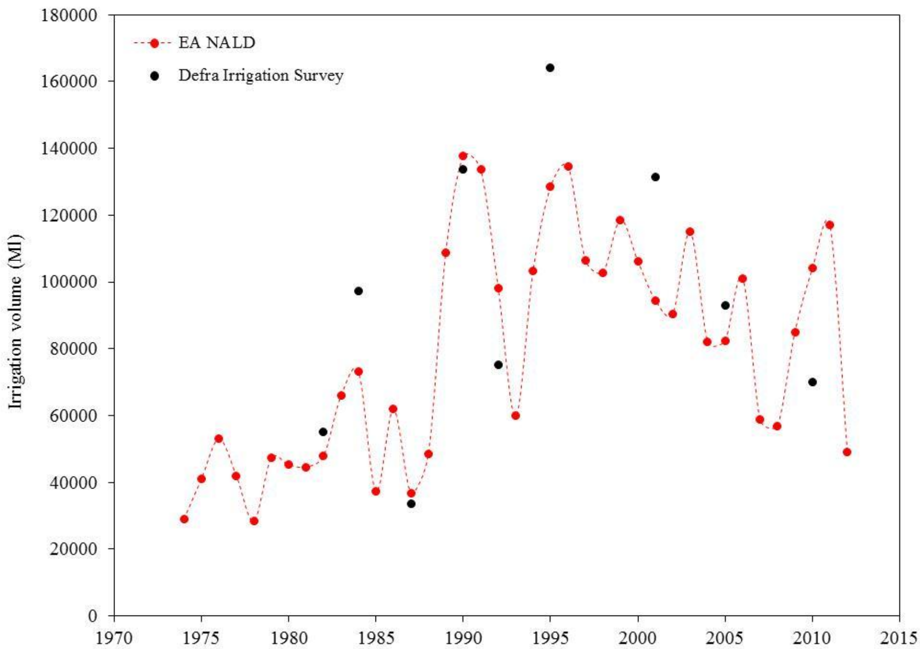

Figure 52 Comparison of the reported total volumes abstracted/applied annually between 1974 and 2012 based on NALD/ABSTAT and Defra Irrigation Survey datasets (updated analysis from Knox et al., 2013).

Synthesis

The NALD/ABSTAT data shows there has been a period of strong growth followed by a decline in both licensed and abstracted volumes. The change in the abstraction trend appears to have occurred earlier than the change in the licensed volume trend. Since around 1990, the volume abstracted appears to have been declining at an average rate of around 2% to 3% (of the 2010 value) per annum. The more detailed Irrigation Survey data suggests some variability between crops. The average depths applied (volume per unit area) have been falling slowly. There has been an underlying decline in both the area of potatoes irrigated and total volumes of water applied to potatoes. The picture is statistically less clear for vegetables, but tends to suggest an overall slow increase in both area and volumes, but possibly a short-term decline more recently. In contrast, irrigation of sugar beet, grass, cereals and orchard fruit all appear to have been in longer-term decline, but have perhaps seen recent increases. Irrigation of soft fruit (e.g. strawberries) shows a steady decline in both areas and total volumes, though this is one crop where the depths applied have increased (probably for quality assurance reasons).

The reversal in trend since the 1990s at least partly reflects the increasing yield and hence decreasing cropped areas needed (particularly for potatoes and some other major irrigated crops), together with increased efficiency and better scheduling. The increasing problems relating to reduced water availability and reliability, and hence a greater appreciation of its value, are also likely to have contributed to water conservation. It is noted however that these short-term trends do not yet reflect very recent changes due to higher food prices, particularly for cereals.

3.3.4 Stakeholder opinions on changing historical trends in irrigation demand

Recent projections of the downward underlying trend in irrigation water demand contrast starkly with previous analyses of agricultural water demand (Weatherhead et al., 1993; EA, 2008). In order to check whether these statistical observations concur with industry opinion, a number of key informants were identified via the UK Irrigation Association (UKIA) and asked "is irrigation water demand really going down?" They were also asked to critique the water demand trends (derived by Knox et al. (2013) and updated in this study) and to provide feedback based on their expert opinion on what other factors might explain why there has been such a change in trend. These practitioner insights are summarised anonymously below, aggregated into four main thematic areas to help identify the likely drivers for the changes in trend. The four thematic areas defined were (i) policy and markets, (ii) farm technology and production, and (iii) water and environment, and (iv) future expectations.

Policy and markets

- It is understandable that irrigation on potatoes has declined due to some consolidation in the industry, better water management, and use of more drought tolerant varieties. This trend may continue but it is difficult to anticipate how market forces will affect this trend over time – an assumption that the trend will continue regardless of other market and global condition would be dangerous;
- A change of attitude from buyer and consumers to the appearance of fresh produce would reduce irrigation demand;
- There is strong industry consensus that economics will preclude any [further] significant decline in the amount of irrigation applied unless: the cost of irrigation rises significantly; the efficiency of application improves significantly; legislation limits water availability; the value of produce declines. None of these seem likely at the moment or in the next 5+ years. Demand is likely to continue to increase at a modest rate;
- There is an ongoing move towards processing potatoes away from the fresh (prepack) market. As skin finish becomes less important, the total irrigation requirement becomes reduced. The opportunity for profiting from irrigating is less on contracted crop than free market crop in a dry year;
- There has been a significant reduction in the cropped area of Maris Piper, the dominant potato variety cropped in the UK. If this is a continuing trend it will be a significant factor due to the volume of water used for common scab control in Maris Piper;
- Market resistance to farm gate output price rises has been very high and will go some way towards explaining the contraction in cropping areas with producers at the margin giving up;
- The situation is driven by economics and markets and so could easily be reversed. We have exported a lot of food production to other parts of the world in recent years and are less self-sufficient in food than historically. It would not take much of a change in production cost economics or global food balances or even water shortages in other parts of the world to reverse this trend, and quickly;
- Considering the state of agriculture in the UK (which is not that good in the high cost crop sector), we should not be surprised by this decline and I suggest it will continue, plateau

and then begin to rise again. My guess on timescale is it will be 10 years before we see significant increase in agricultural water use. Reform of abstraction regulation will also have a major impact.

- Farming businesses involved with growing the main irrigated crops are adjusting to a very different market place, where the risks are far greater than in the past. Input inflation has increased the risk of growing these crops and it is not just a gamble on the weather.

Farm technology and production

- The area of cropping is probably shrinking because yields are improving and because we are reducing waste. Low profitability discourages over production;
- Even if we ignore climate change, demand for quality and maximum yield will demand adequate irrigation, but current usage might still decline further to reflect further yield and waste improvements. However we might find water used on new crops for the energy industry, such as maize and sugar beet;
- The drive to increase crop yield through plant breeding is likely to increase water use even if there is a desire to improve drought tolerance;
- Recent potato variety breeding has focused on water requirement as a key factor – most recent variety additions have been less thirsty;
- Recent wet years have meant that the overall profitability of irrigated farming in recent years has not been sufficient to support reinvestment in capital equipment such as reservoirs , underground main and pumps;
- The cost of energy has risen significantly in recent decades, up to the mid-2000s it was as little as 2p/kWh for electricity and 8p/l for diesel, it is now getting on for ten times that, this will have made many people think about the cost of irrigation and even the economics of growing the crop at all;
- With higher irrigating costs, some growers will have simply 'cut back' on irrigation; where in the past they might have given that extra application, maybe those same growers now say they will risk it;
- With higher costs, growers are also much more aware of efficiency of water use and are now scheduling much more carefully, so the figures might be reflecting the degree of over-irrigation in the past;
- The concentration of production of high cost crops into fewer hands tends to put a greater proportion of the crop into the care of more professional growers, increasing the proportion irrigated precisely and reducing the area where less care or even profligacy in water use might have been the case;
- The high marginal cost of irrigation might have influenced some to resist the temptation to irrigate low value crops such as cereals and sugar beet;

Water and environment

- Any changes in UK potato production and area will have a resulting large effect on total irrigation use;

- There has been a shift in ware potato production with more being grown in Scotland – this would reduce the cropped area in England and Wales;
- Water use on average is going down for a wide range of reasons – farmers are much better informed about crop water needs, water costs, crop responses, better weather information, and in general crop values have been static or downwards; so the marginal decision has been to use less. Farmers have also had a number of difficult harvests, with soil erosion problems, so that makes them more cautious at the end of the season. A couple of dry summers would change farmer behaviour in the opposite direction;
- It is believed that the irrigation of vegetable and salad crops are currently close to optimum (they receive what they need) so any future change in irrigation demand would be associated with a decline in cropped area (seems unlikely) or increased application efficiency (possible) or an associated decline in crop yield/quality (unlikely under current market conditions);
- Innovation and increased pressure from buyers/retailers to increase efficiency. For example, the PepsiCo challenge to reduce the water footprint of their products;
- The abstraction data appears to show a decline in water use since 1990 but an increase since 1975. The start and end points of the trend analysis are clearly important to the trend that you find, and starting with the highest value in a series will always show a decline (much like picking 1998 as the starting year for global temperature analysis). This trend will be further emphasised by the low values at the end of the series – presumably 2007 really was low because of the wet summer;
- Is the abstraction data plausible? I worry about the reliability of the 1989 – 1991 values simply because that's when the NRA was formed. Do we understand why abstraction might fall by 4-5000 Ml from one year to the next;
- It is possible that the decline in irrigation abstraction is explained by how much irrigation has switched from summer abstraction to winter storage reservoirs;

Future expectations

- There are major agronomic issues around overused potato land which has seen some land taken out of production in recent years; further constraints are expected especially linked to nematacide use. If this happens then clean or virgin land will be required. The majority of this land will need irrigation and if this replaces some of the traditional silt growing areas, then it will put an increased reliance on irrigation to produce the crop.
- Global commodity prices have risen significantly in recent years and there is every expectation that this trend will continue due to the global population growth. As commodity prices rise it seems likely that the financial returns from irrigation will also increase suggesting an overall increase in demand between now and the medium term (2030s);
- The global perspective with regards to water resources for agriculture is poor, again suggesting an increasing economic benefit from irrigated agriculture {for those that can irrigate};

- The irrigation of some crops which may have been of marginal economic value in the past may now become more widespread. Anecdotal evidence on the irrigation of cereals in recent droughts suggests this practice is increasing.

3.3.5 Trends in livestock water demand

In contrast to irrigated cropping, there are no nationally published statistics on water use for livestock in England and Wales by volume. Most water used for livestock drinking and yard washing is from public mains supply (King et al., 2006) and Defra (2010) estimated that most livestock farms (>75%) use mains water (Table 19). However, this is often supplemented by water abstracted from boreholes (often below the *de minimis* level requiring licensing), which has been identified as an increasing trend in the Dairy sector (DairyCo, 2011). Water from rivers, harvested rainwater and recycled water is also used on livestock farms. Drinking water may also be supplemented with water from ditches, ponds and rivers where the quality is appropriate (DairyCo, 2009). The water sources identified by Defra (2010) are similar to the survey by DairyCo (2011) that estimated 80% of UK dairy farms use mains water, 42% use boreholes, 32% use springs, and 16% use harvested roof water, in addition to other non-metered sources.

Table 18 Water sources (%) used by farm type (Defra (2010) cited in Gill et al. (2012).

Farm type	Mains water	Ground water	Reser-voir	River abstr-action	Other surface abstr-action	Harvested rainwater	Recycled water	Other storage facility
Dairy	75%	63%	10%	13%	0%	6%	5%	2%
Mixed Arable & Livestock	88%	37%	10%	10%	6%	10%	1%	1%
Pigs	90%	39%	8%	4%	2%	6%	2%	0%
Poultry	84%	33%	12%	4%	0%	11%	2%	0%

Note: Farms may use water from several sources.

Typically metered mains-water readings reflect a combination of various on-farm uses, farm office use and domestic (farmhouse) water use. Similarly, annual returns to the EA for direct abstraction for the ‘General Agriculture’ use category do not distinguish livestock as a specific usage sub-sector. Despite the lack of national statistics, a recent spatial assessment of livestock volumetric water demand was produced by Knox et al. (2013). Selected outputs from that work - including a map showing the spatial distribution of livestock water demand for England and Wales, and data aggregated by EA Region - are summarised in the Appendix. Knox et al. (2013) did not, however, conduct any assessment of historical trends in water use for livestock but rather focussed on a single reference year (2010).

We adopted a similar approach to that used by Knox et al. (2013) to estimate total water use for livestock working from gridded data on livestock numbers (type and age) available from the EDINA[1] (Defra) June Agricultural Census, combined with estimates of per head water requirements. Estimates of livestock drinking water consumption by category (cattle, pigs, sheep and poultry) were based on existing data published in recent Defra projects (WU0101, WU0132 and FFG1129 produced by the University of Warwick and ADAS (2006), ADAS (2012) and Knox et al., (2013), respectively (Table 19). These take into account the age and size of animals, the composition of their diets, production levels and ambient temperatures all of which are known to influence daily/annual water needs. A small survey of water use on beef and sheep farms by ADAS (2013) concluded that on-farm water consumption was, in some cases, very close to figures in the literature, but there was significant variation between farms due to local conditions.

Table 19 Estimated water requirements for livestock for drinking and washing, by type (Source: ADAS, 2010; Knox et al., 2013).

Livestock type	Livestock category	Cycle duration (days)	Drinking water per head per day (l)	Wash water per head per year (l)
Cattle	Dairy cow herd	365	90.61	29
	Beef cows & heifers	365	20	0
	Dairy & beef bulls	365	20	0
	Cattle <1yr	365	12.5	0
Poultry	Broilers	133	0.09	1.14
	Ducks, geese & other birds	56	0.2	2.71
	Turkeys	406	0.2	0.24
	Pullets	406	0.22	0.47
	Laying hens - caged	322	0.19	0.94
	Laying hens - non caged	63	1.22	4.13
	Broiler breeders, layer breeders, cocks	140	0.58	4.37
Sheep	Ewes	365	4.56	0.75
	Lambs	365	2.65	0.75
	Rams and other adult sheep	365	3.3	0.75
Pigs	Sows	365	13.73	453.22
	Maiden gilts	365	5.5	0
	Barren sows	365	5.5	0
	Weaners (20kg)	365	1.8	104.39
	Growers (50kg)	365	4.2	135.42
	Finishers	365	5.6	0
	Boars	365	10	0

[1] EDINA is the JISC-designated centre for digital expertise and online service delivery at the University of Edinburgh.

For the historical trend analysis, the total water demand for each livestock category was calculated by summing the estimated water requirement for each livestock category in each year. Historical data for the period 1983 to 2014 were obtained and a long term time series of demand generated (Figure 53). The analysis suggests that livestock water demand declined steadily until around 2000, and has since stabilised at around 120 Mm^3. Nearly three quarters (68%) is used for cattle (beef and dairy), with sheep (16%), poultry (9%) and pigs (7%) accounting for the remainder. Since we have assumed constant water use per head, the historical reduction in water demand is directly related to a reduction in livestock numbers.

Knox et al. (2013) estimated (2010 baseline) total water demand for livestock to be 147 Mm^3 with most (42%) reported to be used in dairying, and sheep (24%) and beef (20%) also being important sectors.

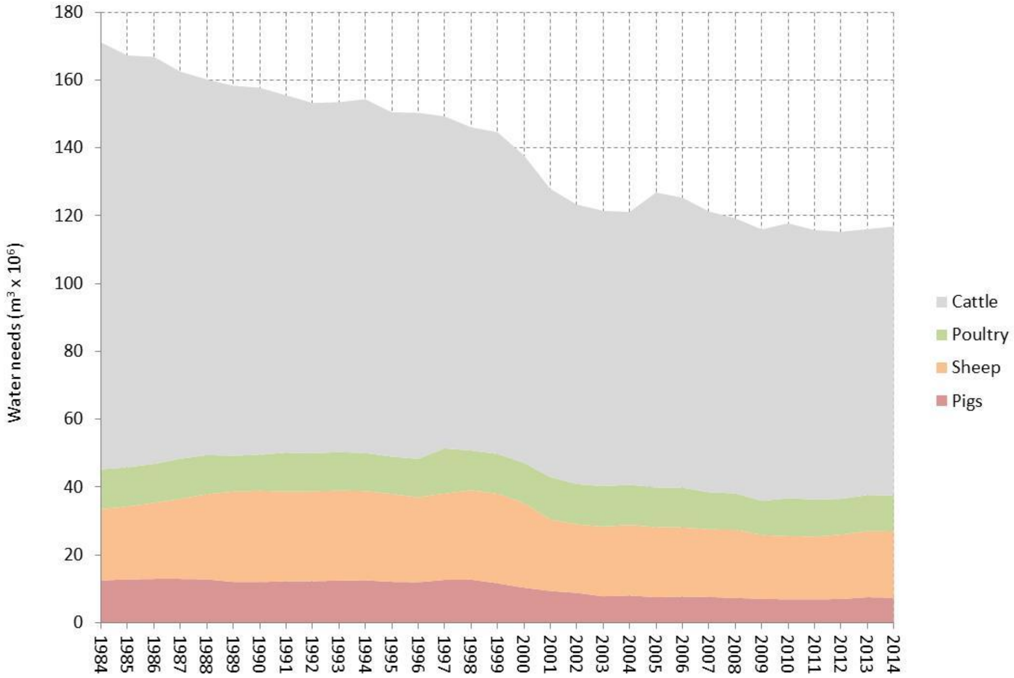

Figure 53 Underlying trend in livestock water use ($m^3 \times 10^6$) by sub-sector in England, between 1983 and 2014. Values are stacked for each sub-sector.

In contrast, Defra (2011) reported that livestock water use for drinking in 2010 was estimated to be 75 Mm^3. Other sectors where water use was important included spraying (4 Mm^3) washing down (13 Mm^3) and 'other' agricultural uses (18 Mm^3). Clearly, there are some major differences in figures published by Defra (2011) with more recent estimates by Knox et al. (2013) and others, reflecting the use of different datasets and methodologies and attempts to calculate theoretical livestock water use in the absence of robust actual consumption data.

3.4 Spatial distribution of demand

Future imbalances between water supply and demand for irrigated agriculture based on previous research and aggregated to WFD river basin level have recently been published by ASC (2013). By combining data on EA licensed abstractions for spray irrigation with EA data on water resource availability, Knox et al. (2013) developed a 'hot-spot' map at the EA CAMS catchment scale as part of Defra FFG1129. In this study, a similar but updated approach has been adopted to produce spatial water demand maps for irrigated cropping and livestock, and 'hot-spot' maps to inform future vulnerability assessments. A series of new outputs and maps have been generated to complement those produced previously by Knox et al. (2013) (selected outputs from that research are provided in the Appendix for comparison).

3.4.1 Irrigation water demand

There is already extensive published scientific research on modelling and mapping irrigation water demand in England and Wales (e.g. Knox et al., 1996; 1997; 2000; 2013) and internationally. All those assessments were based on mapping 'theoretical' (unconstrained) irrigation demand. In this study, a GIS was used to derive a series of maps showing the spatial distribution of 'actual' (i.e. reported) water demand for irrigated agriculture for a series of agroclimatic contrasting years (2005 to 2010) (Figure 54).

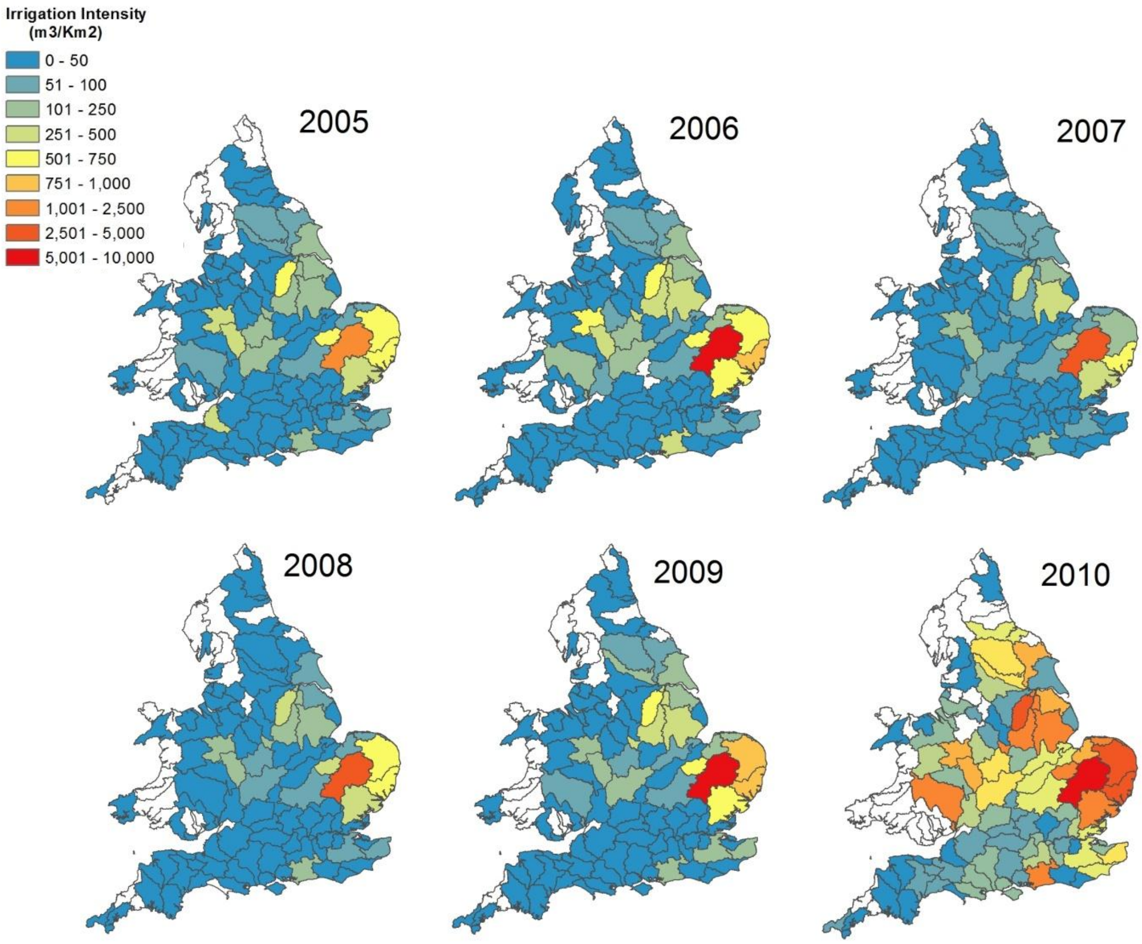

Figure 54 Reported irrigation water demand (m^3) in England and Wales, by EA catchment, based on abstraction data between 2005 and 2010.

The years match those used previously for mapping spatial and temporal changes in agroclimate ($PSMD_{max}$) (Figure 41). The water demand maps shown in Figure 54 were produced by aggregating individual EA abstraction returns on spray irrigation to catchment scale. They highlight significant temporal differences in demand driven by local agroclimatic conditions and the spatial concentration of water demand in a few selected catchments notably in eastern and central England. The Cam and Ely Ouse dominates the analysis given its large geographical extent and high concentration of high-value vegetable and potato cropping. Other important catchments for irrigation include East Suffolk, Broadland Rivers, Worcestershire Middle Severn, north Essex and around the Humber in north Lincolnshire.

However, the maps could be slightly misleading given that the size of the individual catchment dominates the extent to which high demand is evident. Figure 55 therefore shows the 'irrigation intensity' which is expressed as the volume of water demand per unit area (m^3/km^2). This provides a much more balanced assessment of irrigation water demand across England and Wales, and highlights a number of catchments particularly in eastern England where irrigation demand is concentrated. There are also a number of smaller catchments along the south coast (Hampshire) and in Nottinghamshire and the West Midlands which have high levels of irrigation intensity.

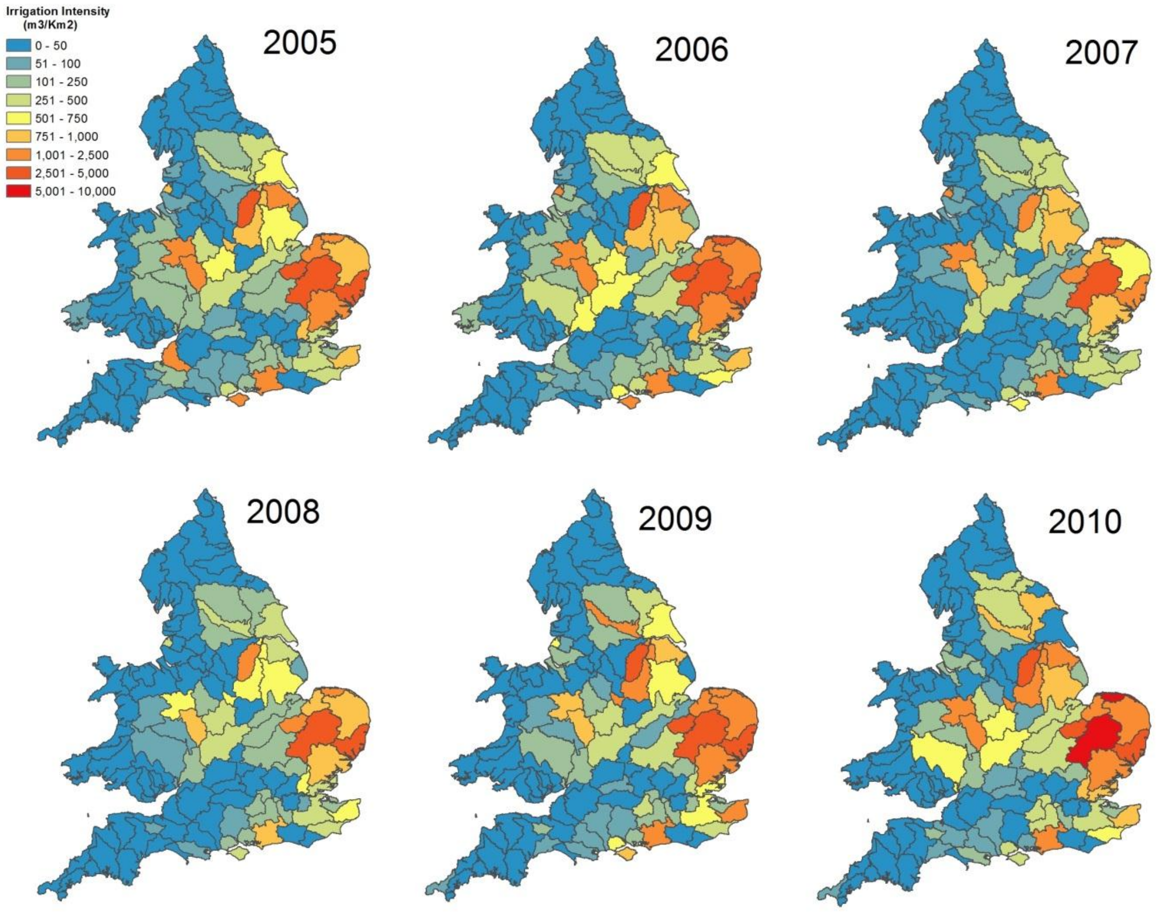

Figure 55 Irrigation intensity (m^3/km^2) in England and Wales based on EA abstraction data between 2005 and 2010.

These actual water demand maps for irrigated agriculture presented here can also be compared against the most recent theoretical water demand (Figure 66) map produced by Knox et al. (2013).

Our analysis based on actual patterns of abstraction in this study show that water demand for irrigated agriculture over the period 2005 to 2010 was on average 82 Mm^3, peaking at 110

Mm^3 in 2010. However, in 2010, there may have also been abstraction restrictions in force which supressed the volumes of actual abstraction reported in the EA NALD data. Knox et al. (2013) estimated a 'design' dry year (based on 2010 land use) theoretical water demand for irrigated agriculture to be 94 Mm^3. Their analysis excluded 'other crops grown in the open' which would typically add another 5% to the overall total, and protected cropping and irrigation for ornamental or nursery stock. Most irrigation demand was reported to be concentrated in EA Anglian Region (60%); EA Midlands and North East were also important, accounting for a further 20% of total demand.

3.4.2 Livestock water demand

Livestock production is concentrated in the wetter more upland catchments in England and Wales. Previous research by Knox et al. (2013) reported that EA Wales, South West and Midlands constituted the areas where water demand for livestock was greatest. In this study, the dataset previously derived by Knox et al. (2013) was used. The EDINA geo-spatial census data numbers were multiplied by published estimates of water demand per head of livestock to estimate spatial distribution of water demand. This represented a gridded dataset of mean water demand by livestock type and category for four individual survey years (2000, 2004, 2009 and 2010). Figure 56 shows the spatial distribution of livestock water need in England and Wales. It clearly shows the higher water demand in the north and west compared to the east, with a high concentration of demand in Cumbria, Lancashire and Cheshire, as well as localised areas of high demand in Somerset, Devon and Cornwall and Pembrokeshire.

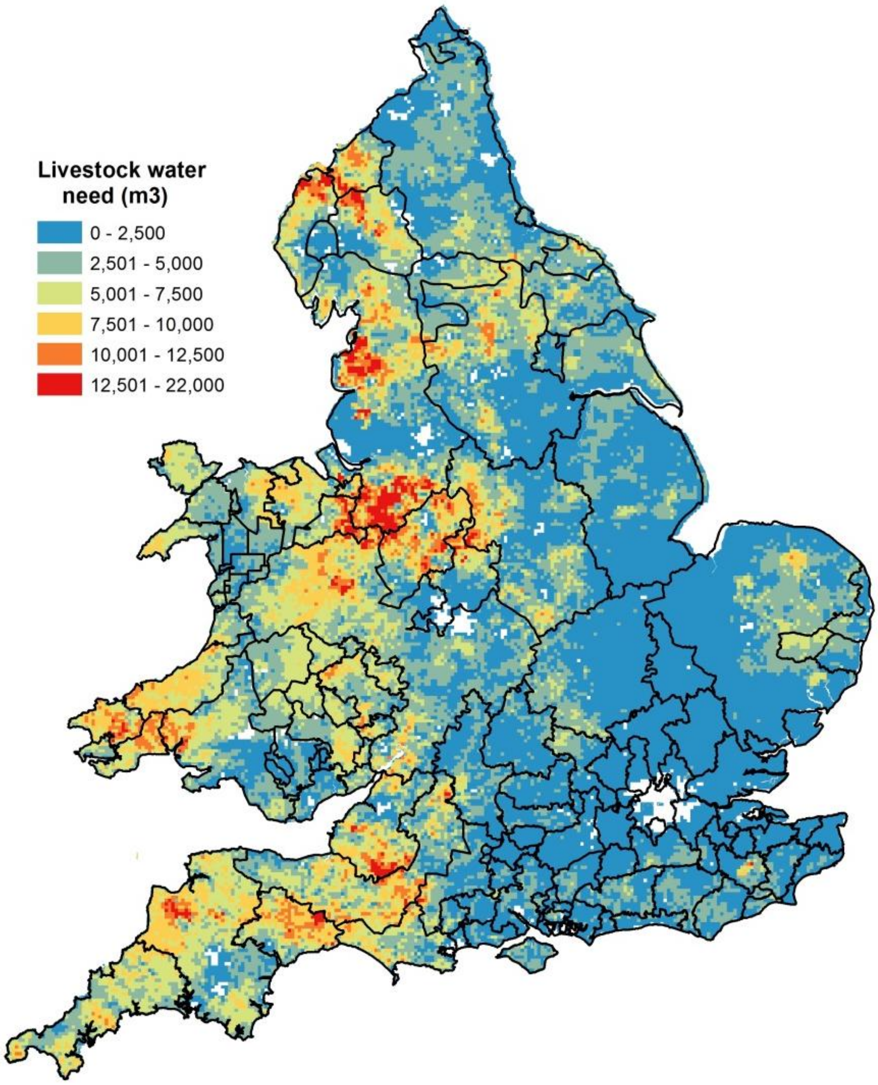

Figure 56 Spatial distribution of livestock water need in England and Wales by WRZ (m^3 per 2 km x 2 km grid square).

The spatial distribution of theoretical water demand for livestock was mapped by EA region (Figure 57). This shows the largest demand for water for livestock occurs in Wales (20%) and South West (20%) EA regions. The lowest concentration of water demand is in Anglian (9%), Thames (4%) and Southern (3%) EA regions.

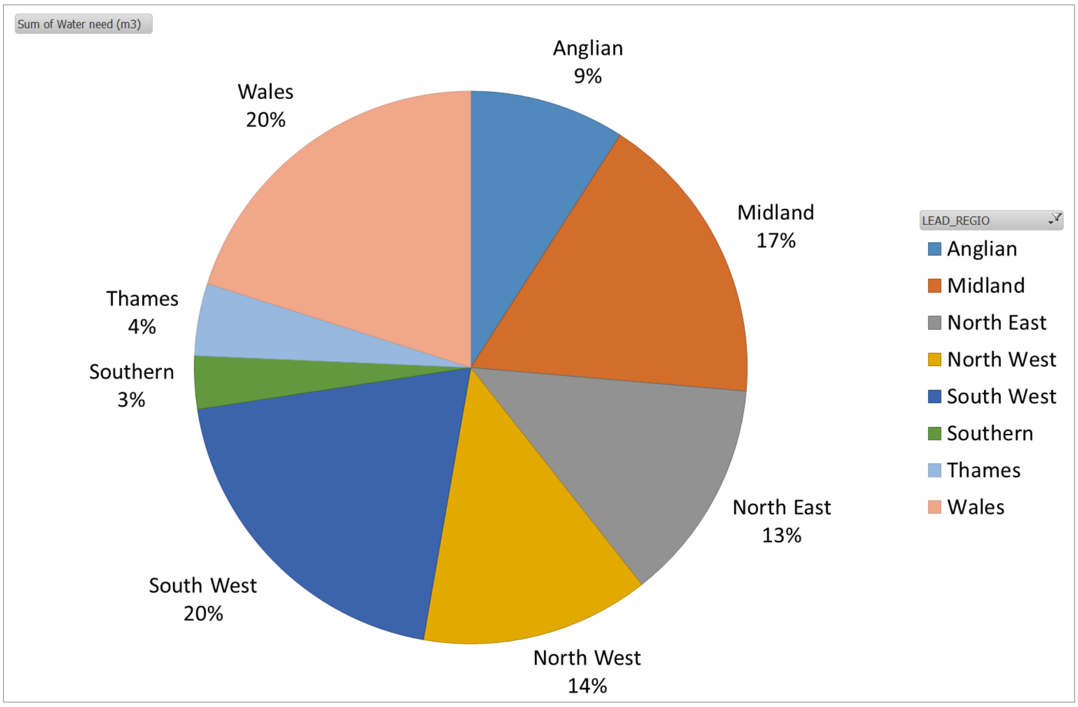

Figure 57 Theoretical water demand (m^3 $year^{-1}$) for livestock mapped by EA region.

3.4.3 Agricultural water demand 'hot-spots'

Irrigation

Identifying irrigation hot-spots can help inform future vulnerability assessments and the implementation of strategic measures to mitigate water scarcity on agricultural production. The demand maps and data described above were therefore combined with EA spray irrigation abstraction data from National Abstraction Licensing Database (NALD) and with the EA's latest estimates of resource availability to identify irrigation abstraction 'hot-spots' (Figure 58).

Irrigation abstraction 'hot-spots' were defined as water resource management units (WRMUs) which are most constrained in terms of resource availability (i.e. classified as Over Abstracted (EA, 2002)) which also have a high intensity (m^3/km^2) of reported abstraction for spray irrigation. Irrigation intensity was categorised as;

- Very High (>10,000 m^3/km^2)
- High (5,000 - 10,000 m^3/km^2)
- Medium (1,000 - 5,000 m^3/km^2)

These highlight areas where competition for water and pressures on the sustainability of irrigation are most likely to emerge.

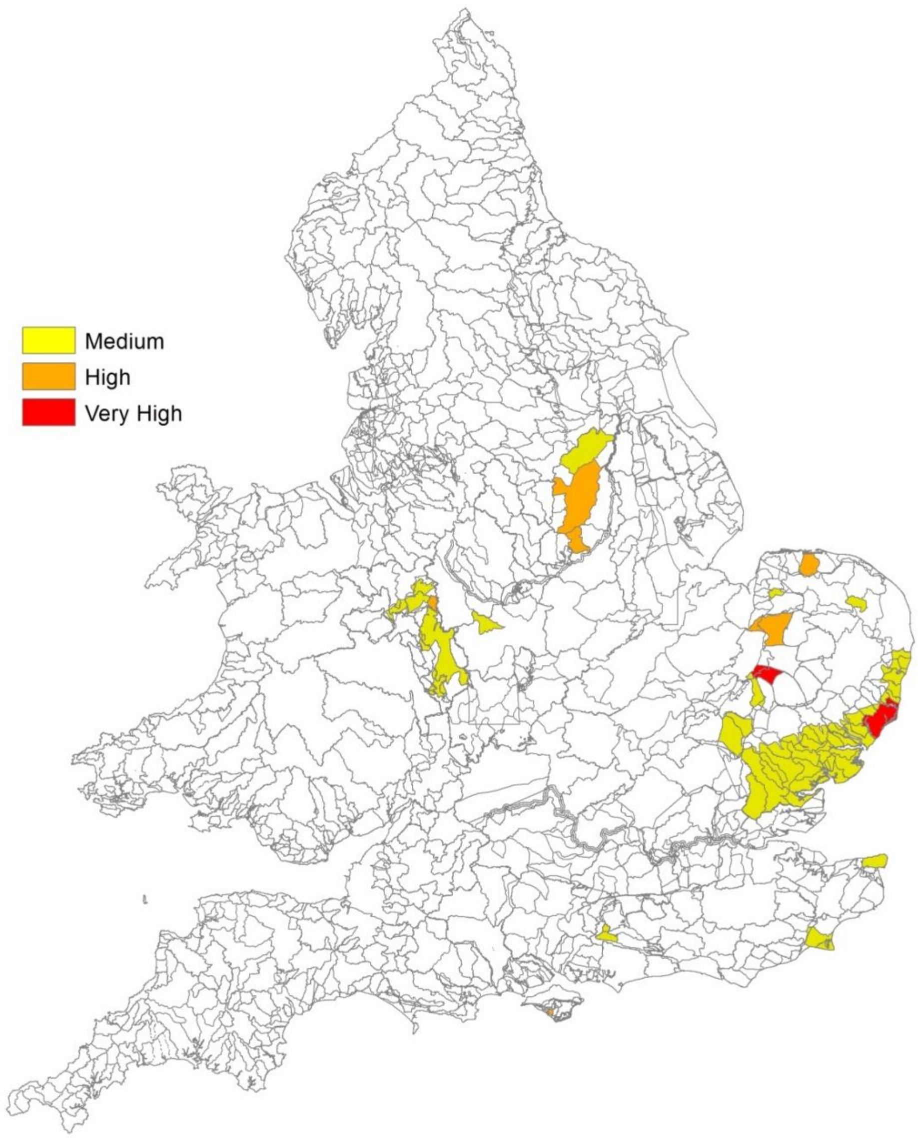

Figure 58 Irrigation abstraction 'hot-spots' in England and Wales based on 2010 abstraction data, classified by EA WRMU.

The hot-spots identified in this study based on historical patterns of actual abstraction are quite different to those derived previously by Knox et al. (2013) based on 'theoretical' irrigation demand (Appendix 4.11, Figure 66). This is due to working from abstracted volume rather than licensed volume data.

The most important hot-spots, which represent the top 30% of national licensed and actual abstraction for irrigation in water stressed (over-abstracted) catchments are summarised in Figure 59. The data shows, for example, that 6.5% of national irrigation abstraction is located in East Suffolk which is an over-abstracted catchment. Together with the Cam and Ely Ouse, Idle and Torne and North Essex, these four catchments collectively account for a quarter (25%) of total national irrigation abstraction, with all having a significant proportion of their WRMUs being defined as being over-abstracted.

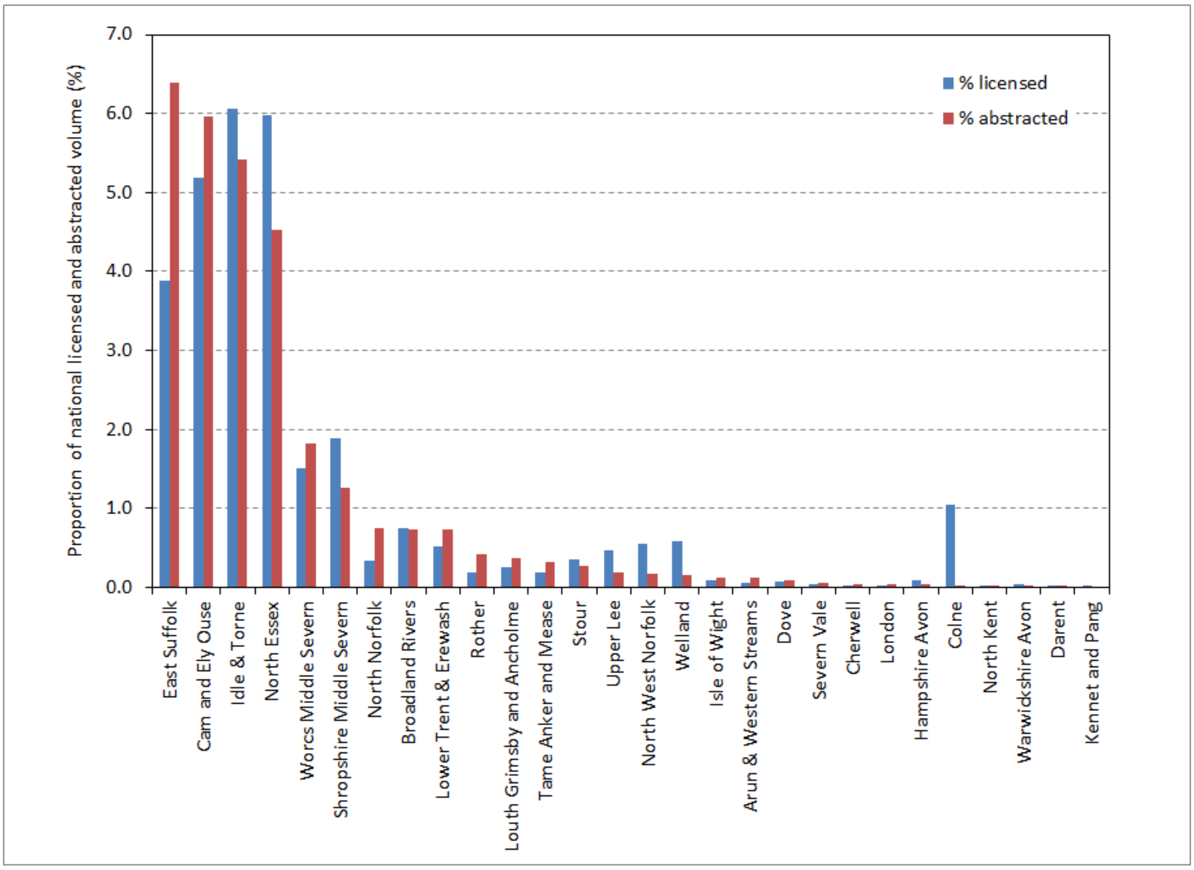

Figure 59 Hot-spot catchments with the highest proportions of licensed and actual abstraction for irrigation, based on EA data for 2010. Note that all catchments shown are defined as being 'over-abstracted'.

Livestock

Most water for livestock production is drawn from the public mains water supply (PWS). Using a GIS, the proportion of livestock water demand within each PWS management unit (termed water resource zone, WRZ) has been mapped. Livestock water demand is concentrated mainly within three water company areas (United Utilities, South West Water and Severn Trent Water).

The availability of water in water company areas depends on local water resources, but also the degree to which water is stored and transferred between catchments. EA & NRW (2013) have classified water companies in England and Wales according to water stress. This was based on the degree to which the resources within the area are exploited by water companies, businesses and farmers.

Water company areas were classified as seriously water stressed if,

a) The current household demand for water is a high proportion of the current effective rainfall which is available to meet that demand; or
b) The future household demand for water is likely to be a high proportion of the effective rainfall available to meet that demand.

Livestock water intensity (defined as total livestock water use, m^3, divided by area served by a water company, km^2) was calculated for each Water Company Area and Water Resource Zone.

The average livestock water intensity in Seriously Stressed Water Company Areas (484 m^3/km^2) is significantly lower than for Not Seriously Stressed Water Company Areas (1,209 m^3/km^2) and the top ten Water Company Areas, in terms of livestock water use intensity, are all Not Seriously Stressed. The livestock water use intensity for the Seriously Stressed Water Company Areas is shown in Table 20. This suggests that Essex & Suffolk Water has the highest demand for water for livestock of the seriously stressed Water Company Areas.

Table 20 Livestock water use intensity, m^3/km^2, for seriously stressed Water Company Areas in England and Wales.

Water Company Area	Livestock water use intensity, (m^3/km^2)
Essex & Suffolk Water	590
Thames Water	542
Affinity Water (formerly Veolia Water South East)	526
South East Water	521
Anglian Water	504
Sutton & East Surrey Water	499
Southern Water	402
Affinity Water (formerly Veolia Water East)	254
Affinity Water (formerly Veolia Water Central)	208

Figure 60 shows the average livestock water intensity for each Water Resource Zone, with those in seriously water stressed Water Company Areas shaded. It is clear that the Water Resource Zones with the highest livestock water intensity are all in Water Company Areas that are not seriously stressed. The highest livestock water intensity occurs in the Dee Valley Water, Dwr Cymru - Welsh Water, Severn Trent Water and United Utilities Water Company Areas (Table 21). However, a particular hot-spot for livestock water demand is the Hartismere Water Resource Zone (Essex and Suffolk Water) where demand for water for livestock is high (>1,500 m^3/km^2) and the Water Company Area is classified as "seriously stressed".

Table 21 Water resource zones with livestock water intensity >2,000 m^3/km^2 in England and Wales.

Water resource zone	Water Company Area	Livestock water intensity (m^3/km^2)
Chester	Dee Valley Water	4,180
Wrexham	Dee Valley Water	3,238
Pilleth	Dwr Cymru - Welsh Water	2,986
Abergynolwyn	Dwr Cymru - Welsh Water	2,684
Staffs and Telford	Severn Trent Water	2,472
Clwyd Coastal	Dwr Cymru - Welsh Water	2,361
Whitbourne	Dwr Cymru - Welsh Water	2,320
South Pembrokeshire	Dwr Cymru - Welsh Water	2,172
Llyswen	Dwr Cymru - Welsh Water	2,140

Carlisle	United Utilities	2,085

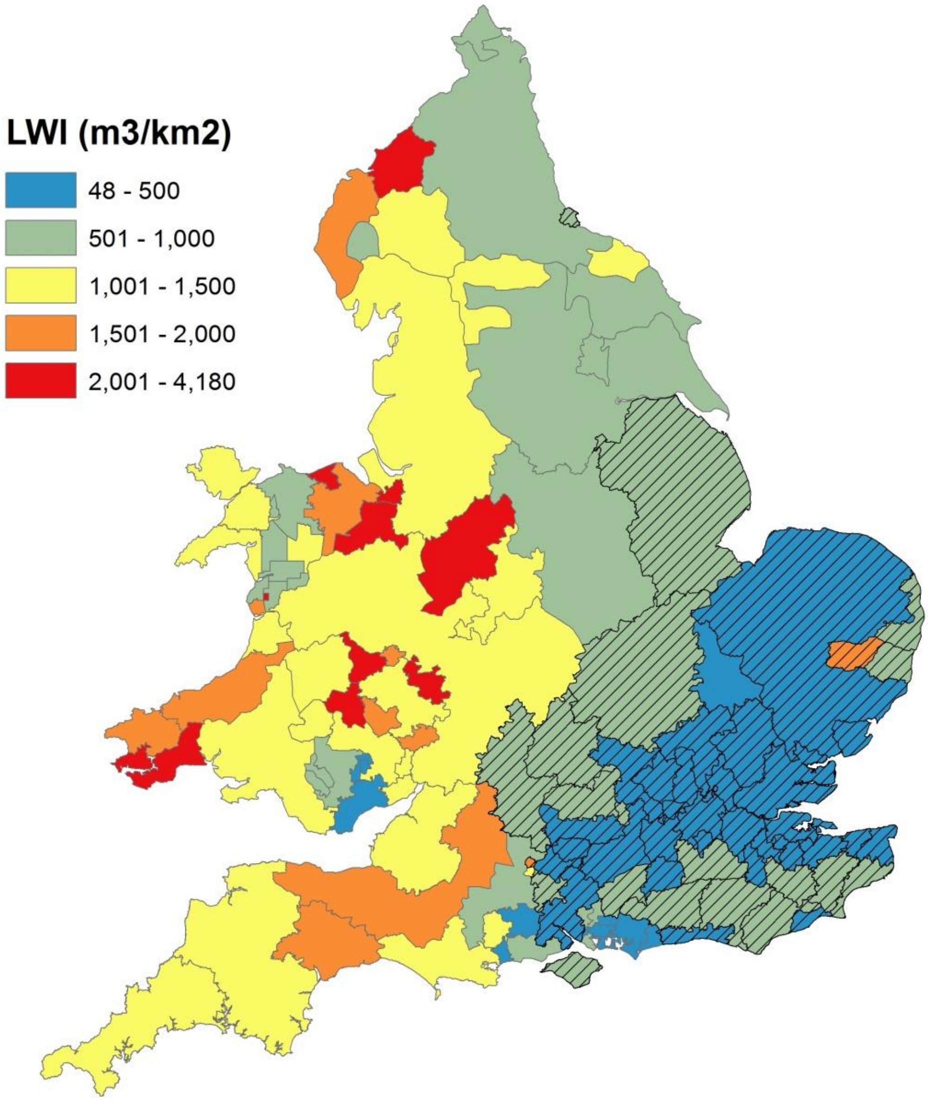

Figure 60 Average livestock water intensity (m^3/km^2) by public water supply (PWS) water resource zone (WRZ), based on 2010 data. Shaded areas are Water Company Areas classified as "Seriously Stressed"[2].

By analysing the spatial distribution of total water demand for livestock at the EA CAMS sub-catchment (WRMU) level, an assessment of potential livestock water demand hot-spots has been derived. Half (50%) of water demand for livestock is within 25 EA catchments, with seven accounting for a fifth of national livestock water demand (Figure 61). However, when compared to water resource availability, over half (53%) of livestock water demand is in catchments defined as having 'no more water available'; 11% is in over-abstracted, 10% in over-licensed and 26% in catchments with 'water available'. Within 'over-abstracted' catchments, the Dove, Broadland Rivers, Hampshire Avon, Eden and Usk and North Essex

[2] Hartlepool Water (N E England) is shown as "Seriously Stressed" as it is owned by Anglian Water and the analysis by EA & NRW (2013) was by water company.

catchments are shown to have the largest water demand for livestock, recognising that overall their proportion nationally is still very small (c5%) (Figure 62).

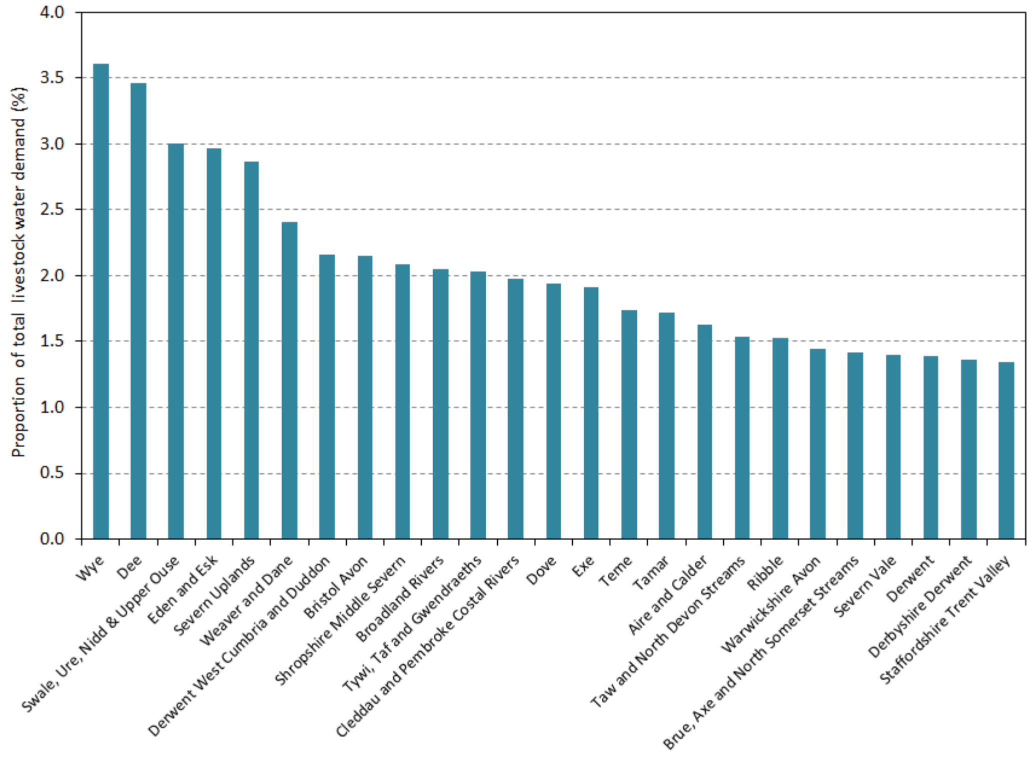

Figure 61 Catchments estimated to have the highest proportion of water demand (%) for livestock.

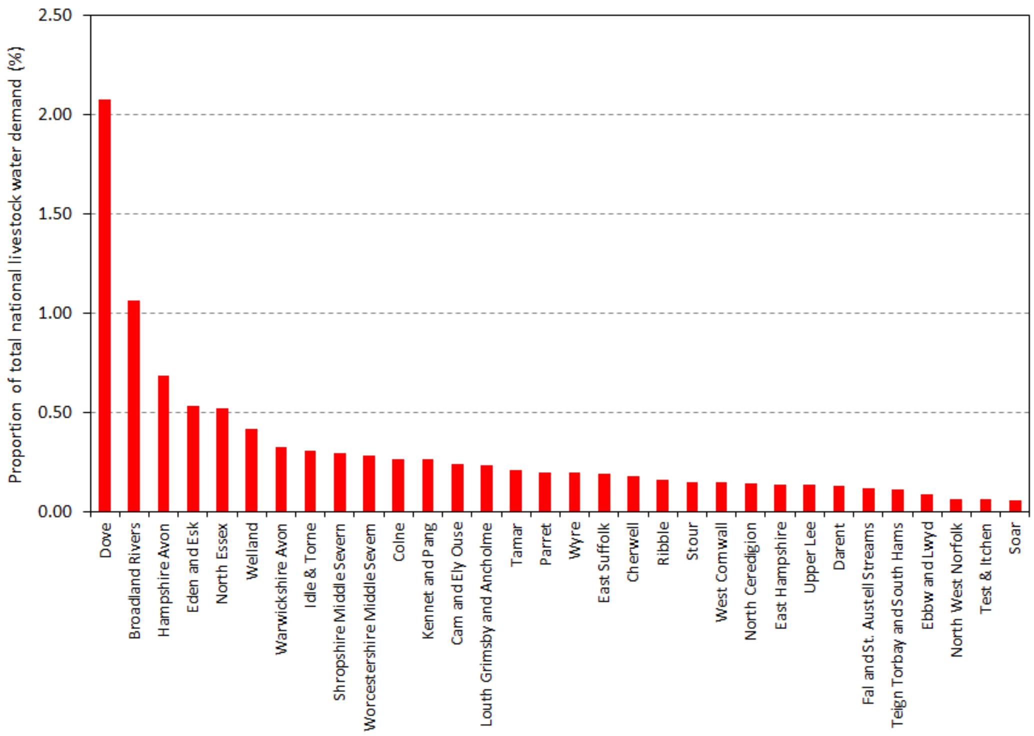

Figure 62 Livestock water demand showing the proportion (%) of total national livestock demand in each catchment. Note all catchments shown are defined as being 'over-abstracted'.

3.5 The impacts of drought in the agricultural sector

Drought has an impact on agriculture in three main ways:

1. Reduced rainfall affects the production of rain-fed crops, forage and grazing;
2. Reduced water resource availability affects abstraction from surface and groundwater resources. This is especially important for the irrigated cropping sub-sector as this relies to a large extent on direct on-farm water abstraction, however, it also impacts on the livestock sub-sector;
3. Reduced public water supply availability has an impact on all farm businesses, but particularly livestock farming where mains water is used for animal drinking and sanitation and for protected cropping, such as soft fruit, where mains water is often used for irrigation.

Like all businesses, some of the impacts of drought on agriculture may be acute, that is, where lack of water has an impact for the period during which water availability is reduced, but after a short period of recovery, production returns to normal. However, most of the impacts on agricultural systems are chronic and cumulative. For example, there is a clear distinction between acute impacts of water shortage on housed livestock (e.g. poultry) and the cumulative impacts on cropping. Dry weather in the spring, or early restrictions on abstraction for irrigation, would affect plant germination and crop development; adequate rainfall or water later in the season will not compensate loss of yield potential. The economic impacts on crop quality could therefore be more significant than any yield penalty. In this way, the effects of drought on agriculture may extend beyond the end of the drought. The impacts are also sensitive to the timing of the drought in relation to the cropping calendar-dry weather in the winter may have negligible impact on crops whereas dry weather at the time of germination may result in crop failure and the need to re-seed when the rain has returned.

The financial impacts on agriculture are also sensitive to the geographical extent of a drought (locally) and global market conditions. If drought is confined to a region of the country, it is likely that crops and animal feeds can be sourced from elsewhere. However, as happened in 2011-12, when the drought extended across most of Western Europe, alternative sources were similarly affected and, as a consequence, the prices of crop and animal products rose.

3.5.1 Impacts of drought on rainfed cropping

Rain-fed cropping is not reliant on abstraction and uses modest amounts of mains water for general on-farm use. The main impacts would be from low rainfall in the spring and summer affecting crop yields and quality, but soil degradation and increased pollution could also have resulted from the inefficient uptake of nutrients. There is some evidence that spring cereals on light land were affected by early drought, but the wet summer conditions had a bigger impact on final crop performance.

3.5.2 Drought impacts on livestock

There are two main impacts of drought on livestock production: Direct impacts of reduced rainfall on farm water supply and; indirect impacts of dry weather on forage and feed.

In the event of a severe drought, public supplies for use on livestock farms would presumably be subject to restrictions before domestic customers, and would have a major impact. More likely, some farmers who have private boreholes, may find supplies restricted. Most small farm boreholes fall below the minimum abstraction rate for licensing, so would not be 'cut-off', but in a drought, lower water tables could result in private supplies running dry.

Restrictions in water supply would result in less water available for animal drinking and cleaning of animals and housing, with a potential biosecurity implication. It is difficult for livestock farmers to respond to short-term restrictions as alternative water supplies, for example, farm ponds, are also likely to be unavailable. In an extreme situation, farmers would need to reduce herd size by slaughtering or sale of stock to farmers in regions of the country with more water available.

Dry weather in the growing season leads to reduced grass growth, reducing the amount of grass for grazing and conservation. The first silage cut in the spring is very important as it tends to have the highest quantity and quality, so reduced grass growth would lead to a reduction in the availability of winter feeds and increased costs of providing alternative feeds. The impact of drought on cereal production would also lead to increased supplementary feed costs. Other impacts include: lack of good quality grazing at the time of livestock breeding, when animals benefit from 'nutritional flushing', which can lead to reduced animal fertility; reduced growth rates of growing cattle and lambs and a potential increase in animal diseases and parasites, heat stress and sunstroke.

The livestock sector is increasingly aware of its vulnerability to drought. The sector is particularly trying to reduce its vulnerability to fluctuating feed costs by increasing the use and flexibility of home-grown feeds. There is increasing interest in alternative forage species that are more drought resistant, such as chicory, red clover or cereals that have the flexibility to be sold as a grain crop or used for whole crop silage. Farmers are increasingly aware of the need for water efficiency in all areas, particularly with regard to reducing the costs of water and fuel.

3.5.3 Experience from the 2011-12 winter drought

The 2011-12 drought resulted from dry and warm weather in 2010 and intensified through the winter of 2011/12 as parts of England recorded their lowest 18-month rainfall in at least 100 years (Kendon et al., 2013). As a result, around 300 irrigators were subject to restrictions to filling winter storage reservoirs and those with summer abstraction licenses were unsure if they would be able to abstract. Others agreed a self-imposed voluntary restriction on abstraction. Some growers responded by reducing their planted area of irrigated crops or switching to more drought tolerant varieties, but many were inflexible due to long planning times and contractual commitments. Some with secure water supplies increased the planted area in the hope of high crop prices.

Although there were no restrictions on public water supplies, some private boreholes dried up and livestock farmers were increasingly reliant on public mains water. In the event, the impact of the water resources drought on livestock sector was minimal; however, had there been restrictions on mains water supplies or pressure reductions, then there would have been a major impact.

In 2012, the low spring rainfall resulted in reduced grass growth, limiting the amount of feed for spring grazing or conservation for winter (hay and silage), although the onset of the rains mitigated the impact. Had the drought continued, grazing and forage yields would have fallen. Combined with reduced yields and higher processing cost of grains, it would have significantly increased winter feeds costs.

The impact of the 2011-12 drought on agriculture was complicated by an unusually wet summer. The summer rainfall in England was the highest since 1912 and wet conditions persisted through the autumn. The waterlogged conditions exacerbated many of the crop husbandry problems that had been triggered by the drought; delayed harvest by up to four weeks; and harvesting during wet conditions has resulted in soil damage. In some regions, some crops (such as potatoes) were harvested and some were abandoned. As a consequence, crop yields and quality were affected by both early drought and subsequent waterlogging and it has been difficult to isolate the impacts of drought from the impacts of waterlogging, although the latter has probably had the greatest impact on harvested yields and crop quality. For these reasons, the material impacts of the 2011-12 drought were difficult to quantify.

3.6 Actions to reduce water use in livestock production

This section aims to highlight areas where the vulnerability of the livestock sector to extremes in weather (most notably drought) and exposure to increasing costs of water can be reduced and the actions to mitigate their impact.

Most water used on livestock farms is from the public water supply (mains), however, mains water is expensive and in some areas, public water supply is seriously stressed. The average cost of mains water is £1.17/m^3 (Defra, 2011) with variations across river basin catchment areas ranging from £0.95/m^3 (South East and Northumbria) to £1.50/m^3 (South West). The cost of mains water has increased by about 38% over the last 20 years - a reflection of the investment that has been made in upgrading infrastructure to improve water quality and increase the reliability of mains water supply. Those directly abstracting water will also have experienced an increase in charges to reflect the cost of managing water resources effectively and restoring sustainable abstraction.

Water use and costs for livestock production can be reduced by improving the efficiency (i.e. water use per head) with which water is used. Most water used in livestock farming is for animal drinking and there is little scope for savings apart from changing the animal's diet or the ambient temperature of animal housing. However, Defra (2009) identified a number of drinking devices, for pigs in particular, which may reduce water use by as much as 40%.

Water troughs need maintenance to repair leaks or blockages and regular cleaning to maintain good quality water which in itself can use large quantities of water. Water troughs can be replaced by water bowls, bite type drinkers, nipple drinkers or animal operated valves which can be effective in reducing spillage, and reducing the potential for leaks. The float on ball-valves allowing feed water into troughs can be lowered to reduce the risk of overflow, and troughs can be isolated when not in use to avoid frost damage leading to leaks. Fitting drinkers with catch basins and a guide rail retains overflow and prevents water seeping from the side of the animal's mouth when drinking (Defra, 2009).

Using water from alternative sources can save money and reduce vulnerability to water shortages, however, water quality is very important for drinking water and using streams to supply drinking water is discouraged. It also increases risks of bank erosion, sedimentation and pollution of the stream. Using water from private boreholes can save costs and reduce reliance on the public water supply. DairyCo carried out annual water surveys between 2009 and 2011 (DairyCo, 2011) and found an increase in the use of boreholes to supplement mains supply on dairy farms.

Although less water is used for farm washing than for drinking, it accounts for 20% of the water used for dairy cows (Thompson et al., 2007). Relatively simple changes in management practice lead to significant water savings in wash-down water (Defra, 2009);

- Pre-soaking parlours, yards and housing to loosen dirt before washing
- Scraping yards to remove dirt before washing
- Using high-pressure bulk tank washing systems to save water
- Using enzyme based methods to clean milking machines

Simple actions can help reduce water losses from piped systems, including identifying and repairing leaks quickly, insulating pipework to avoid frost damage and reading and recording water meter readings regularly to detect anomalies and understand patterns of water consumption on-farm. It has been estimated that a 1 litre leak per minute raises water consumption by the same amount as increasing a dairy herd by 25 animals. DairyCo (2011) found that 97% of the dairy farmers surveyed were checking regularly for leaks.

Rainwater harvesting is a relatively straightforward but expensive option to incorporate into new and existing buildings. Rainwater harvested from roofs of farm buildings can be used, with the right treatment, instead of mains water for livestock, for yard and equipment washing, or to top up a farm reservoir. Rainwater harvesting can help to significantly reduce mains water consumption on farm, but it does require investment. A system based on a new above ground 30 m^3 tank is estimated to cost around £5,500 or £0.80/m^3 (Defra, 2012) when written off over a period of 10 years. Systems vary greatly depending on whether a second hand or new tank is used and whether it is placed above or below ground and filtration requirements. Siting the tank is important for the storage period and risk of pipe freezing. As mains water costs are expected to continue rising, rainwater harvesting technologies will become increasingly cost-effective. DairyCo (2011) estimated that 16% of dairy farms use some form of roof water harvesting and Defra (2010) shows that rainwater harvesting is used in all livestock sectors (Table 18).

Water can be saved by recycling water after it has been used for another process. However, the opportunities for recycling depend on the quality of the water after the first use. For example, in dairy parlours milk cooling water can be re-used for animal drinking or washing.

Water audits provide a simple way of working out where, when and how much water is being used on-farm. This data can then be used as a benchmark to assess performance relative to other farm businesses. Guidelines on water auditing for the livestock sector have been produced by the EA (Waterwise on the Farm) as well as from ADAS and the dairy levy board (AHDB DairyCo).

There are no national statistics on the uptake of water saving actions in the livestock sector, however, Defra (2009, Annex 1) provide a summary of a number of case studies of the uptake of water saving on livestock farms. These show that the savings can be considerable in terms of the volume of water and the costs saved (Table 22). It is worth noting that reducing the water used for yard wash-down also reduces the volume of slurry that has to be stored and disposed of, resulting in considerable further cost savings.

Table 22 Summary of livestock water saving case studies (after Defra, 2009).

Farm type	Size, ha	Action	Water saving	
			m^3/year	£/year
Dairy	600	Rainwater harvesting Recycling plate cooling water	≈33%	
Dairy	100	Recycling plate cooling water		£786
Mixed	800	Dew ponds for livestock watering	1,626 (≈10%)	
Dairy	175	Rainwater harvesting	4,392	> £10,000
Dairy & Sheep	100	Rainwater harvesting		£574
Poultry	6.5	Rainwater harvesting		£120,000
Mixed	325	Leak detection and general water efficiency	1,000	

3.7 Data limitations

The trend analyses and GIS mapping techniques used in this study were heavily dependent on having research access to detailed, high resolution and good quality historical datasets, relating to agroclimate, land use and water abstraction. A number of sources were used including government cropping census data, information from the water regulatory authority (EA) on abstraction and published agrometeorological data. However, gaining access and assessing the quality and provenance of these datasets coupled with the licensing challenges regarding the use of such statistics is a recurrent frustration and challenge for those undertaking important agricultural systems or environmental assessments and options appraisals.

Such challenges faced in this project regarding data acquisition are known to have been replicated in other studies and will inevitably recur in future projects. It is therefore requested that greater efforts be targeted to support those responsible for data collection, archiving and provision including government agencies and others to improve their data management policies and practices. Future studies investigating climate related risks to the environment and society will only become more dependent on such data.

3.8 Conclusions: water use in agriculture

New evidence and data to support CCC policy objectives to understand the underlying historical trends and future challenges in agricultural water demand, focussing on two key sectors - irrigated crop production and livestock has been provided.

A GIS based methodology was developed to model and map the spatial distribution of 'hot-spots' for agricultural irrigation abstraction and livestock water use across England and Wales. This involved deriving a new water stress indicator termed 'irrigation intensity' (m^3 per km^2) rather than volumetric water demand (m^3 per catchment) as used in previous research. Irrigation intensity conveys more accurately where there is high demand per unit of irrigated land use in water stressed sub catchments. A similar approach was developed for mapping hotspots for livestock water demand. In addition to the GIS modelling, a detailed synthesis of literature, published case studies and evidence from key informants was used to understand the vulnerability of livestock production to reduced water availability, and to review what actions should be taken by the livestock sector to mitigate risks from future reduced water availability. The key conclusions for each sub-sector are briefly summarised below:

Irrigation water demand

A detailed temporal and spatial analysis of historical trends in irrigation water use was undertaken, by combining agrometeorological information with data from the EA on reported abstractions for spray irrigation. Using an aridity index termed potential soil moisture deficit (PSMD) the analysis shows how temporal patterns of irrigation abstraction have been closely linked to agroclimate variability (rainfall and evapotranspiration, ET).

GIS maps showing the spatial variation in aridity by EA catchment and for a sequence of agroclimatically contrasting years, clearly highlight why irrigation demand is concentrated in eastern and south eastern England. Maps and data have similarly been generated to show the spatial variation in EA reported volumetric water demand and irrigation intensity. These identify a small number of key catchments where irrigation demand is concentrated and where water resources for irrigation abstraction are already under severe stress and hence where future resource problems are most likely to be encountered. These catchment areas include the Cam and Ely Ouse, East Suffolk, parts of Norfolk (Broadland Rivers), west Midlands (Worcestershire Middle Severn), north Essex, and the East Midlands around the Humber in north Lincolnshire.

An analysis of underlying historical trends in irrigation water use have shown that there has been a long period of strong growth (since the 1970s) followed by a more recent decline in both licensed and abstracted volumes. The change in the abstraction trend appears to have occurred earlier than the change in the licensed volume trend. Since around 1990, the volume abstracted appears to have been declining at an average rate of around 2% to 3% (of the 2010 value) per annum.

The reversal in trend since the 1990s at least partly reflects the increasing yield and hence decreasing cropped areas needed (particularly for potatoes and some other major irrigated crops), together with increased efficiency and better scheduling. The increasing problems relating to reduced water availability and reliability, and hence a greater appreciation of its value, are also likely to have contributed to water conservation. It is noted, however, that these short-term trends do not yet reflect very recent changes due to higher food prices, particularly for cereals.

Livestock water demand

As there is no national data on water used for livestock, and it is difficult to separate water used on farm for livestock uses from other uses, livestock water demand has been estimated from livestock numbers and per capita water requirements. Given this, water demand for livestock has declined steadily until around 2000, and has since stabilised at around 120 Mm^3. Nearly three quarters (68%) is used for cattle (beef and dairy), with sheep (16%), poultry (9%) and pigs (7%) accounting for the remainder.

Most water for livestock drinking and washing comes from the public water supply (mains), therefore hot-spots for water use for livestock are those areas where the public water supply is seriously stressed and the intensity of livestock water use is high. In general, the intensity of livestock water use (m^3/km^2) is greater in Water Company Areas that are not currently, or estimated in the short term, to be seriously water stressed. Therefore livestock water use is less vulnerable than irrigation water use.

Intensity of water use in agriculture

A combined analysis of total agricultural water use intensity in England and Wales both sub-sectors (irrigation + livestock) by catchment was then completed. The results are shown below in Table 23. In all catchments with a total agricultural water intensity greater than 4,000 m^3/km^2, demand is dominated by water use for irrigation, whereas livestock water demand tends to dominate those catchments with a total intensity of less than 4,000 m^3/km^2.

Table 23 EA catchments for agricultural water intensity (m^3/km^2) and major (>67%) user.

EA CAMS catchment	*Water Intensity, m^3/km^2*			
	Irrigation	*Livestock*	*Total*	*Major user†*
Cam and Ely Ouse	7,186	414	7,600	Irrigation
North Norfolk	5,162	290	5,451	Irrigation
East Suffolk	4,770	574	5,344	Irrigation
Idle & Torne	4,169	349	4,518	Irrigation
Old Bedford	3,969	119	4,087	Irrigation
Shropshire Middle Severn	1,512	1,933	3,445	Both
Weaver and Dane	0	3,064	3,064	Livestock
Broadland Rivers	2,183	857	3,039	Irrigation
Dove	83	2,629	2,712	Livestock
Wyre	0	2,574	2,574	Livestock
Arun & Western Streams	1,796	569	2,365	Irrigation
Dee	49	2,311	2,360	Livestock
North West Norfolk	2,073	250	2,323	Irrigation
Otter, Sid, Axe and Lim	55	2,102	2,157	Livestock
Wye	687	1,466	2,153	Both
Worcestershire Middle Severn	1,462	654	2,116	Irrigation
Severn Uplands	122	1,977	2,099	Livestock
Lower Trent & Erewash	1,211	750	1,962	Both
Little Avon	0	1,956	1,956	Livestock
Torridge and Hartland Streams	1	1,944	1,945	Livestock

† Water use in a single sector >67% of total.

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4. APPENDICES

4.1 Measured erosion rates for different UK soil/land use combinations

Land use	Typical erosion rates* (t ha^{-1} yr^{-1}) reported in different soilscapes			
	Clay	Silt	Sand	Peat
Arable	1.92 a 0.23 – 0.36 b 0.9 c < 2 d 0.10 – 5.56 e	22.1 g 22.7 h 3.2 c 4.5 f 11.2 i 0.33 – 7.44 e	5.08 a 16 j 10.8 - 10.9 b 0.3 to 44.4g 22.7 h 0.4 c 0.43 b 0.41 to 1.7 k 1.48 f 3.47 d 11.2 i	
Grassland / pasture	0.36 f 0.02 – 3.54 e 1.29 f	4.09 a 4.89 f 2.82 – 4.92 e 2.07 f	0.2 – 0.98 b 4.09 a 0.22 f	
Forestry/ Woodland			0.05 b	29.76 l

a Evans, 2002; b Morgan, 1980; c Deasy et al., 2008; 2009; d Cooper, 2006 e Walling et al., 2002; f Brazier, 2004; g Morgan et al., 1987; h Robinson and Boardman, 1988; I Fullen, 1992, j Reed, 1983; 1986; k Quinton and Catt, 2004; l Carling et al., 2001

Notes: Data from Evans (2002) do not specify soil type, but do specify crop / land use. Reasoned assumptions have been made as to which soil type is used for various crops (e.g. oilseed rape on heavy (clay) soil). Data from Brazier (2004) derives from Evans (1988, 1993) and Skinner and Chambers (1996). These erosion rates relate to soil types only: No land use data are given. Reasoned assumptions have been made regarding likely land use for different soil types and resulting erosion rates as presented in Brazier (2004). Data from Walling et al (2002) are net erosion rates, based on 137Cs techniques

4.2 Factors affecting soil erosion by water

(High risk: Erosion features seen in most years during wet periods; Moderate risk: Erosion features seen in some years during wet periods and in most years during very wet periods; Low risk: Erosion features seen in some years during very wet periods.

	H	M	L
Soil texture (top soil)	Soils with a high sand or silt content Sandy soils erode most frequently as they often carry a wide range of crops, but soils rich in silt and fine sand erode most severely (Evans, 1990). Red sandstone soils (East and West Midlands and south Devon; Boardman, 2013). Silty and fine sandy soils in Somerset (Bridport Sands around Yeovil; Boardman, 2013)	High clay content (>18%) are more stable soil crumbs and aggregates. However, less permeable therefore risk of overland flow.	
Organic matter	Low OM content due to continuous ploughing or reinstated profile. Deeper ploughing depths.		High OM content help bind aggregates
Rainfall	Rainfall exceeds infiltration capacity Intensity >4mm hr; quantity >15 mm day	10 mm falling on saturated soils (Boardman, 2013)	
Cultivation	Continuous arable cropping (reduces OM and breakdown soil aggregates) Use of power-harrows to produce fine tilth. Fine seedbeds provide source of fine sediment to be eroded and tent to be vulnerable to capping. Bare land after root crop harvesting. Disturbed soil with no cover. Rolling in autumn on vulnerable land, especially if the soil is wet. Contour ploughing on complex slopes can	Rotations involving medium to long-term leys Rough ploughed/cultivated land On gentle slopes, working across the slope can reduce erosion in combination with a reversible plough to throw the soil upslope	Rotations involving long-term leys Use of subsoilers to break up pans and compacted layers Cereal stubble, land with good crop/vegetation cover Conservation tillage techniques (providing compaction is prevented), using tines, discs or shallow one pass systems. These leave crop residue on the surface and can increase OM in the long term. Increased surface indentations (dikes) to

	H	M	L
	lead to breakthrough and erosion Compacted tramlines Soil compaction in winter cereals (caused by previous crop harvest or wheelings/tramlines) Ploughing and cultivation carried out well in advance Early-spring colder therefore slower germination and growth. Also wetter therefore higher risk of compaction		increase water retention and encourage infiltration Tramlines eliminated or avoided until spring Where land has been protected with cover crop of stubble leave ploughing and cultivation until just before sowing. Late spring sowing is generally warmer and dryer.
Irrigation	Large droplet size and excessive application rate		Boom irrigators are more consistent, produce smaller drops and apply them close to the ground Trickle (drip) irrigation
Cropping (Defra, 2005)	Late sown autumn/winter crops Potatoes (most at risk April to June) Sugar beet (most vulnerable April to June) Field vegetables Outdoor pigs Grass re-seeds (especially late drilled) Forage maize Out wintering stock Grazing forage crops in autumn or winter Little or no vegetation cover Winter cereal seedbed prepared after late harvested sugar beet Linseed Soft fruit and orchards with bare ground between cropped rows. Lower Greensand soils of Bedfordshire, West Sussex and the Isle of Wight, which are intensively farmed with crops such as	Early sown winter cereals Oilseed rape – winter and spring sown Spring sown cereals Spring sown linseed Short rotation coppice/Miscanthus Early harvesting root crops and maize Spring re-seeded grass Early maturing varieties of forage maize A minimum ground cover of 25% is required by early winter to be effective against erosion (Defra, 2005) Fruit and orchards with grassed areas between rows	Long grass leys Permanent grass Woodland (excluding short term coppice) Leave stubble and chopped cereal stubble Undersowing of cover crop

	H	M	L
	potatoes, maize, vegetables and some cereals.		
Stocking	High stocking rates. Poaching damage from livestock especially in heavily trampled areas such as around feeders and water troughs and gateways. Pigs on sandy soils. Exposing soil by poaching Animals accessing rivers causing bank and bed erosion.	Reduce stocking density. Remove animals when the soil is wet.	Move gateways from lower edge of field. Move feeders and water troughs. Restrict access to rivers.
Soil water storage capacity	Restricted or reduced storage capacity due to reduction in soil depth		Maximum storage capacity achieved with good poor size distribution and minimal soil loss/reduction in soil depth.
Infiltration	Infiltration-excess overland flow. Capped surface reduces infiltration and increases risk of overland flow. Topsoil compaction restricts infiltration into the soil promoting overland flow. Poor subsurface drainage limits capacity of soil to absorb water and restricts percolation. Lead to rapid build-up of water near the soil surface which reduces infiltration and promotes overland flow.	Subsoil compaction. Capacity to absorb rainfall/irrigation but capacity is limited by depth to compaction.	No compaction, good connectivity through poor space, course rough seed bed, subsurface drainage to maintain a lower water table.
Slope length	Long (>critical length) increased potential to accumulate larger volumes of run-off water		Short (<critical length), reduced upslope accumulation risk. Contour strips to break up slope (strips 5-15m wide positioned every 50-150 m down slope) Use of soil walls across slope in potato furrows (tied ridges)
Field size	Large blocks of the same crop give rise to		Smaller blocks of different crops varying

	H	M	L
	bare ground at the same time. Chalklands of southern England, because under large areas of winter cereals, large fields and compaction along wheelings (Boardman, 2013)		area left bare at any time
Slope gradient	Steep slopes and valley bottoms susceptible to flooding or where concentrated runoff from upslope areas occurs. >7^o (all soils[1], Defra, 2005) In pastures and upland areas on slopes >20^o (Evans, 1990)	3^o to 7^o (all soils[2], Defra, 2005)	Terracing or bunded to reduce the effect of gradient <3^o (all soils[3], Defra, 2005)
Water table	High with minimal additional storage capacity to accommodate new rainfall		Low, maximum storage capacity to accommodate new rainfall
Soil moisture	High (saturation-excess overland flow), limits infiltration and promotes runoff	Drier but with limited storage capacity and very dry (possibly hydrophobic: peat soils)	Dry (with available moisture)
Morphology	Undulating terrain is most vulnerable to water erosion (Evans, 1980). Convex slope. Valley floors, with water flowing from both sides of a valley wheeling, furrows Smooth surface (capped)	Concave Reduced surface roughness (secondary ploughing for fine seed bed)	Long straight slopes least risk (Evans, 1980) Very rough surface
Flooding	Land that floods at least 1 year in 3 (Defra, 2005)		Land that does not flood
Compaction	Surface compaction restricting infiltration and wheelings that focus water flow	Subsurface compaction limiting soil water storage capacity	No compaction

4.3 Soil associations at risk of wind and water erosion in England and Wales

Based on data in Evans (1990) and associations from Mackney *et al*. (1983) (Table 3 defines meaning of risk category)

Erosion type	**Very small risk**	**Small risk**	**Moderate risk**	**High risk**	**Very high risk**
Water	22, 342d, 346, 411a, 411c^{+}, 421a, 421b, 511h, 511i, 512a, 512b, 512c, 512d, 512e, 512f, 532a, 532b, 541a^{+}, 541i^{+}, 541v, 541w, 541B, 561b, 561b’, 561c’, 572t, 573a, 581a, 581b, 581c, 581d, 612a^{+}, 641b, 643a^{+}, 643c, 711b^{+}, 711c, 711d, 711f, 711g, 711h, 711k, 711m, 711p^{+}, 711r, 711s, 711t^{+}, 712a, 712b^{+}, 712c^{+}, 712d, 712e, 712f, 712g, 712h, 712i, 713b^{+}, 713c, 713d^{+}, 713e, 713f^{+}, 713g^{+}, 714a, 714b^{+}, 714c, 714d, 811a^{+}’, 811b’, 811c’, 811d, 811e, 812a^{+}’, 812c, 813a,	313b, 341, 342a, 342c, 343c, 343e, 343i, 411b^{+}, 411d^{+}, 431^{+}, 511a, 511c, 511d, 511f^{+}, 511j, 541d, 541f^{+}, 541g, 541h, 541j, 541k, 541l^{+}, 541n^{+}, 541o, 541q, 541u, 541x, 541y, 541z. 541C, 541D, 542, 543, 517a, 571g^{+}, 571l, 571m, 571n, 571p, 571r, 571s, 571t, 571u, 571v, 571w, 571z^{+}, 517A, 572a, 572b, 572d, 572f, 572g, 572h, 572i$^{+§}$, 572j, 572l, 572n, 572o, 572q, 572r, 581e^{+}, 581g, 582a, 582b, 582c, 582d, 611b, 631c, 634, 643b, 711a, 711e, 711i, 711j,	342b, 343a, 343b, 343d, 343h^{+}, 511b, 511e, 511g, 513, 541c^{-}, 541e, 544, 561a’, 561d’, 571b, 571c, 571h, 571i, 571j, 571k, 571o, 571q, 571x, 571y, 572c, 572e, 572k, 572m, 572p, 572s, 573b, 582e, 631d, 92c	451b^{-}, 541m^{-}, 541s, 571d,	541A,

Erosion type	Very small risk	Small risk	Moderate risk	High risk	Very high risk
	813b$^{+\prime}$, 813c$^{+\prime}$, 813d, 813e, 813f, 813g, 813h, 814a’, 814b, 814c, 831a, 831c, 832, 841b, 841c, 841d, 871b, 871c, 92a,	711l, 711n^{+}, 711q, 711u, 711v, 711w, 713a^{-}, 841a, 841e, 92b			
Water but also at risk of wind	643d^{+}, 812b^{+}, 1011a	555, 581f^{+}, 631b^{+}, 631e^{+}, 631f^{+}, 711o	343g, 541r, 541t, 551g, 552b, 571f, 641a	551c, 551e, 554a, 571e	551a, 551b, 551d^{-}
Wind	521^{+}, 554b^{+}, 861a, 1021^{+}, 1022b^{+}, 1024c, 1025^{+}	343f, 815, 821b^{+}, 831b, 851b^{+}, 872a, 872b, 873	372, 551f, 641c, 851a^{-}, 851c, 861b^{-}, 1022a^{-}, 1024b^{-}	361, 552a, 821a^{-}, 1024a^{-}	
Upland (includes by wind)	721a^{+}, 721b^{+}, 721c^{+}, 721d^{+}, 721e^{+}, 871a	311d, 611a^{+}, 611c^{+}, 611d, 612b, 631a^{+}, 633^{+}, 651b, 654b^{+}, 654c	311b, 311c, 313a, 313c, 611e^{+}, 651a, 651c, 652^{-}, 654^{-},1013^{+}, 1013b^{+}	311a, 311e, 1011b^{+}	

+Locally risk of erosion is greater
-Locally risk of erosion is less
§Marginal to higher category
Also at risk of upland erosion
‘Overbank erosion of flood plain

Key to Appendix 7.3.

Very small risk	Erosion occurs rarely or not at all
Small risk	Eroding fields or moorland cover ≤1% of the land each year
Moderate risk	Arable: 1 to 5% land eroding each year Upland: small areas overgrazed
High risk	Arable: >5% land eroding each year and greater mean and median volume eroded than lower risk categories. Uplands: extensively gullied and hagged, widespread overgrazing, extensive footpath erosion.
Very high risk	Lowlands: erosion rarely affect <5% fields each year. An average of 10% fields affected, and 2 years in 5 20-25% affected. Volume of soil eroded often greatest of any risk category.

4.4 Factors affecting erosion by wind

Factor		
Soil type	Mobility coefficient Bulk density Particle size distribution Cohesiveness (tensile strength) Organic matter content Stoniness Clod size Soil moisture content	An increase in magnitude of these will require more energy for soil entrainment Soil moisture influences soil erodibility and is related to aggregation and cohesiveness (electrostatic forces) and tensile strength. Aggregation increases surface roughness and the drag coefficient Drainage and conversion to arable of peat and sandy land. Leads to oxidation of organic matter.
	Sedimentology Texture mineralogy	Affects threshold wind shear velocities
	Clay content	Controls soil cohesiveness Aggregation (% aggregates >0.85 mm diam related to aggregate stability and erodibility (Cornish, 1983) Formation of surface crusts Resistance to deflation
	Surface crusting	Barrier to abrasive winds
	Surface roughness	Increased roughness increases turbulence and eddying. Reduces energy
	Wind velocity	Static or fluid threshold – wind velocity at which grain movement takes place. A wind speed of 30-40 km hr^{-1} is sufficient to dislodge particles from the soil and transport them either by saltation, deflation or surface creep Bärring et al. (2003) identified four scales of at which wind strength varied: 1) Synoptic weather patterns which affect wind speed over large-scale down to 100 km, 2) Regional wind variations 1-100km, dependent on the landscape and major land-use, 3) local scale of 100 m to 10 km, affected by local topography, land-use and obstacles to wind flow e.g. groves, rows of trees, 4) within-field, less than 100 m wind speed is affected by furrow height, crop selection and phenology, weed control measures, row orientation and stand structure. Dry winds are more erosive than cold, humid winds.
	Fetch Air pressure gradient Surface cover	

Factor		
	(roughness effects) Duration Predominance of wind direction	
Rainfall	Soil moisture content Vegetation cover Structure and density Evaporation Formation of surface crusts (protects from wind erosion)	
Vegetation cover	Cover Height Spatial distribution or geometry in a landscape Roughness length (see Table 12) Timing	Vegetation can decrease wind shear stress on an erodible surface by absorbing some of the wind's downward momentum, creating a barrier of slow moving air above the surface, and increasing the threshold friction velocity required to mobilise soil. Increase in spring-planted vegetable crops has increased wind erosion
Grazing	Overgrazing reducing vegetation cover especially of vulnerable upland areas	High number of sheep as the result of subsidy payments led to an unsustainable number. Reducing the number of sheep reduces erosion (Evans, 1977)
Synthetic stabilizers	PVA PAM	On sands not peats
Mechanisation	Depletion and oxidation of organic matter Plough and press may reduce wind erosion on sandy soils	
Field size		

4.5 Factors affecting risk of wind erosion

Factors	**High**	**Medium**	**Low**
Soil texture	Sandy soils (especially fine sand) and peat soils (Evans, 1990) Red sandstone soils (East and West Midlands and south Devon; Boardman, 2013).		Clay rich soils (unless they aggregate and form peds that act like sand grains)
Organic matter content			
Soil moisture content			
Land use and cover	Sugar beet Potatoes Carrots Onions Horticultural crops Poor vegetation cover Sheep grazing on upland areas		Winter rye Winter barley Mustard Field boundaries (trees and hedges)
Wind erosivity (wind speed and fetch)			
Time	March to June		
Geographical location	East Midlands, Yorkshire and East Anglia Upland peat areas		
Soil depth	In uplands shallow soils and deep peats		

4.6 Existing maps of erosion risk

Figure A1 illustrates locations assessed as vulnerable to erosion (by water, wind, and wind and water) based on identifying those soils (including peat soils) which the Soil Survey of England and Wales had identified as vulnerable in their descriptions in the legend to the 1:250,000 national soil map (Soil Series level; Table A3.

Figure A1 was updated by Evans (1990) and Boardman and Evans (2006) based on a field erosion survey run between 1982-1986, and ad hoc information for a series of soil associations (Figure A2). Criteria were based on land use, soil type (i.e. texture) and landform (i.e. slope type), rainfall amount and intensity was not used to assign risk.

Other estimates of erosion rates at the national scale include Wood et al. (2006; Figure A3) and McHugh et al. (2006; Figure A4). Kibblewhite et al (2014) used the outputs of the Pan European Soil Erosion Risk Assessment (PESERA) model to estimate priority areas for soil protection, using different levels of erosion rates and probability of occurence. There are very few estimates of wind erosion at the national scale (Owens et al., 2006), although Figure A6 shows a predicted annual wind erosion hazard for mineral soils (SOM<5%) (Quine *et al*., 2006).

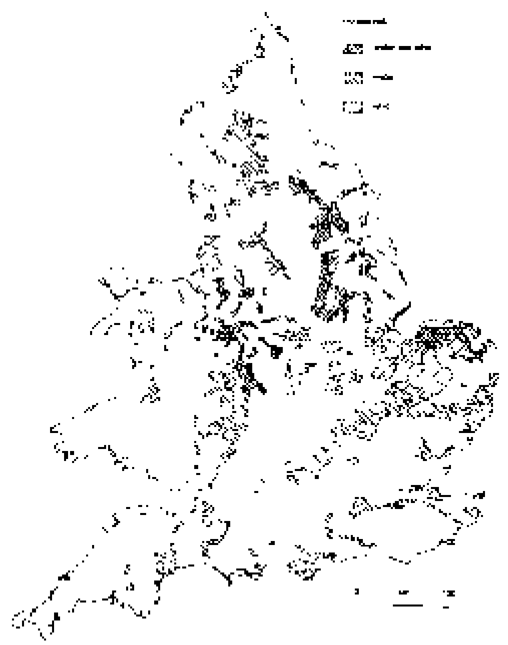


Figure A1. Areas of soil erosion risk in England and Wales (from Morgan, 1985)

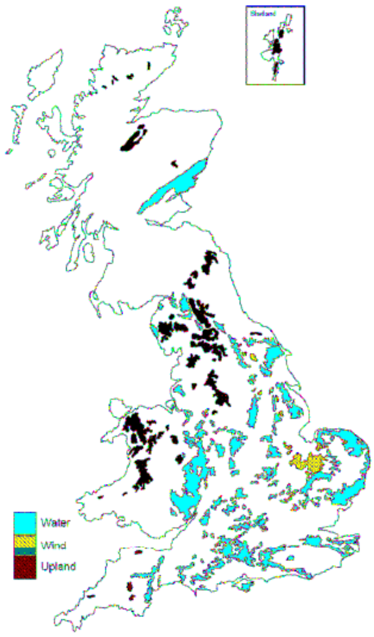


Figure A2. Areas of soil erosion risk in England, Wales and Scotland (from Boardman and Evans, 2006

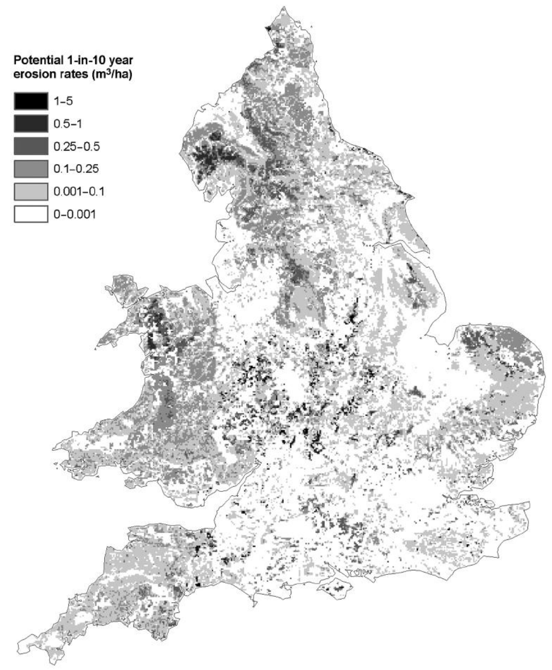


Figure A3: Potential 1-in-10 year soil erosion rates in England and Wales (Wood *et al*., 2006

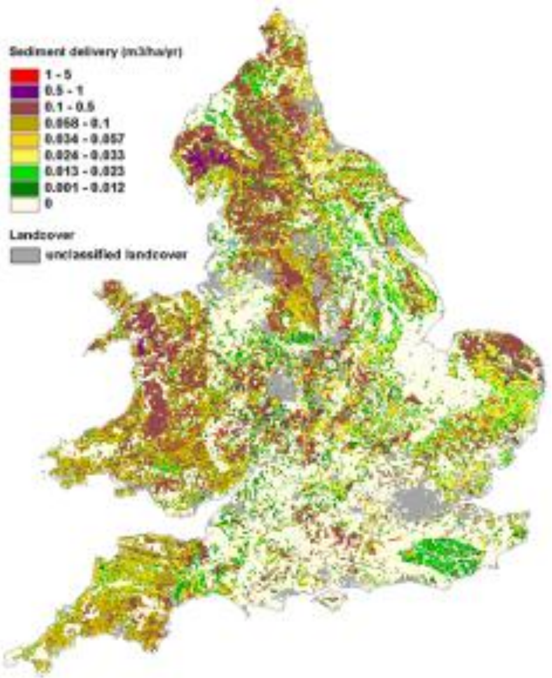


Figure A4: Distribution of estimated 1-in-10 year sediment delivery to watercourses in England and Wales expected to occur annually (from McHugh *et al*., 2002)

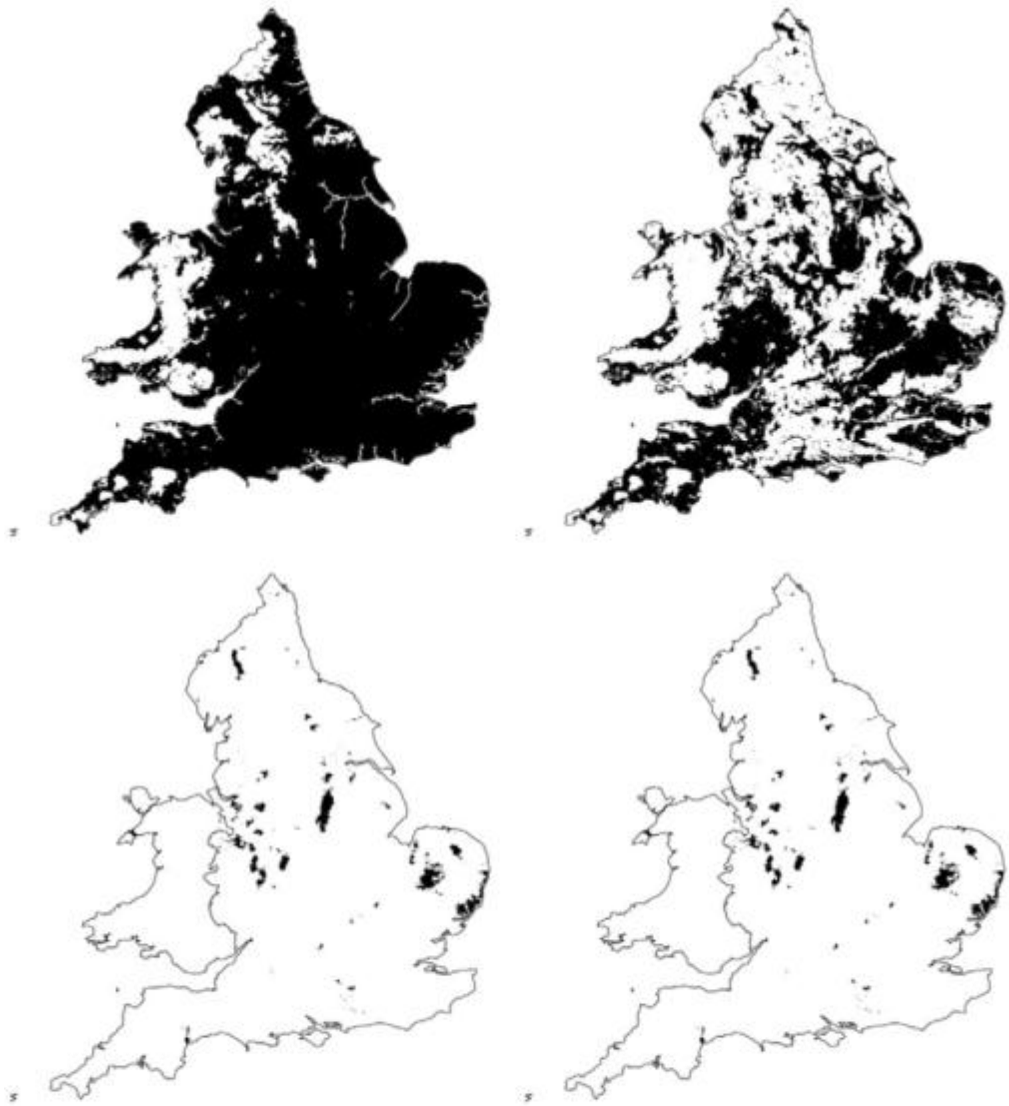


Figure A5 Areas of England and Wales estimated to exceed a rate of soil erosion of 1, 2, 4 and 8 t ha^{-1} yr^{-1} respectively (probability = 1 in 20). Based on PESERA modelled estimates (unvalidated) (Kibblewhite et al., 2014).

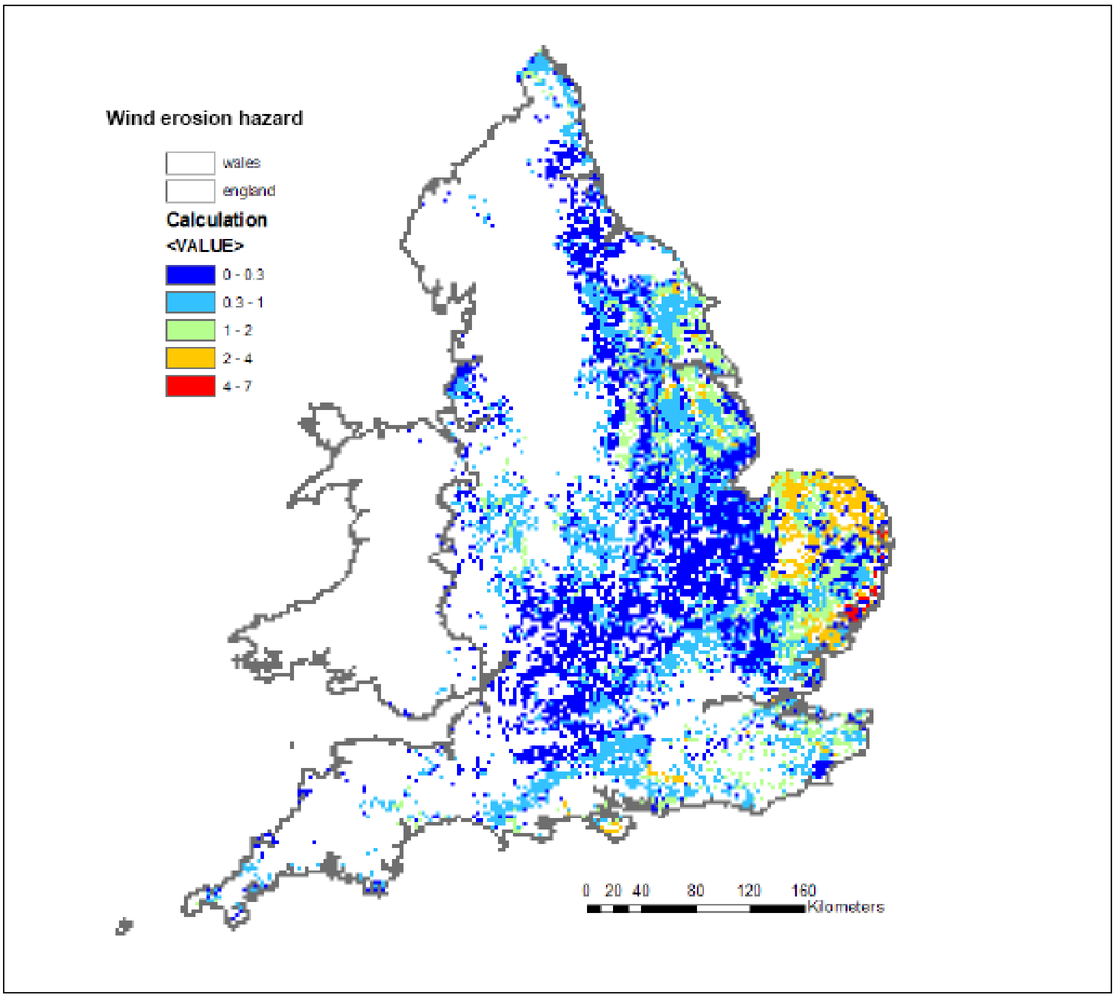
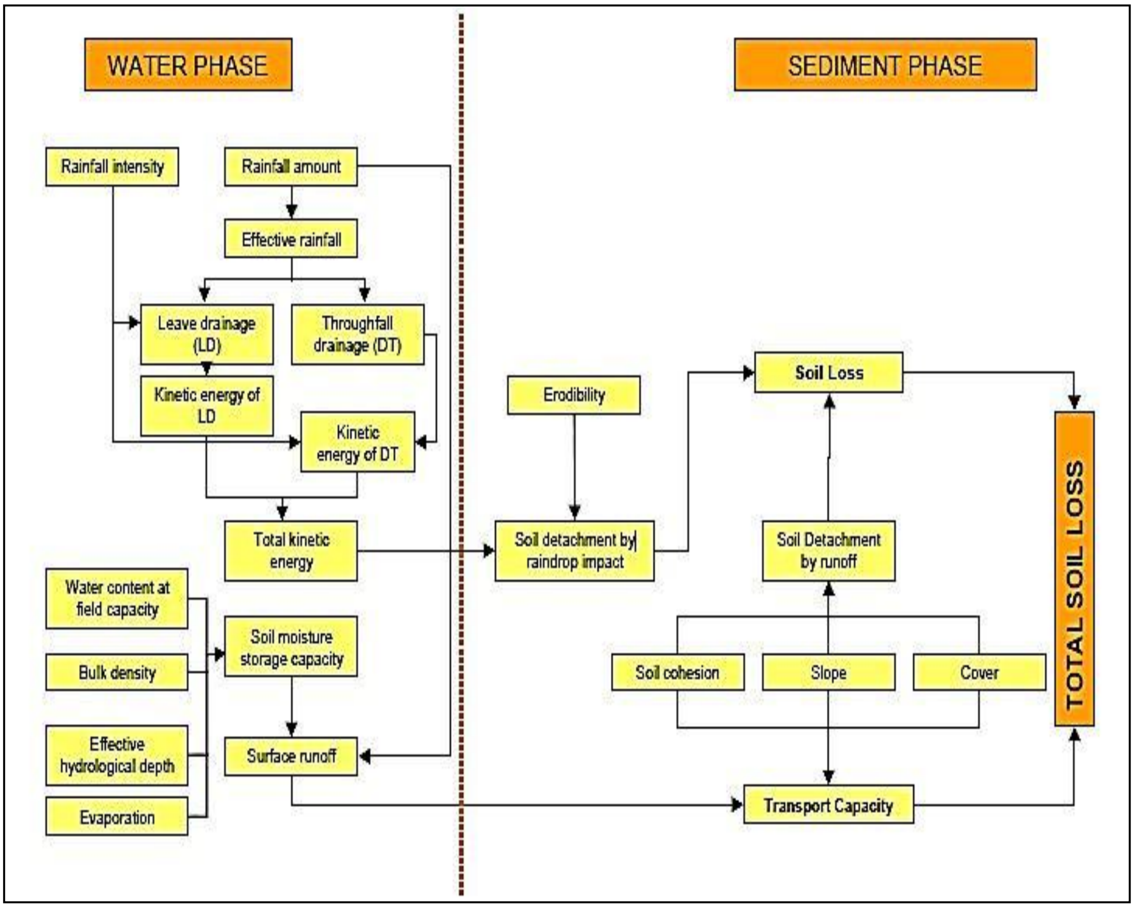


Figure A6 Annual wind erosion hazard for mineral soils (SOM<5%) derived by summation of monthly wind erosion potential based on the RWEQ Qmx parameter. The scale is a quantitative relative scale. Source data: UKCIP 1961-1990 scenario (simulated by HadRM3 with SRES A2 Medium –High emissions scenario); MetOffice 5 km gridded data; and Digital Soil Information from NSRI: NATMAP, SOILSERIES and HORIZON © Cranfield University (NSRI) 2006. (Taken from Quine *et al.*, 2006)

4.7 Morgan, Morgan and Finney (MMF) erosion prediction model

The MMF model was first developed in 1984 by Morgan, Morgan and Finney, revised in 2001 by Morgan and modified by Morgan and Duzant in 2008.The model divides the erosion process into two phases – a water phase in which the volume and kinetic energies of rainfall, drainage and runoff are computed, and a sediment phase in which the computed water variables are used to calculate erosion. The model describes soil erosion in terms of the relationship between rainfall, surface runoff, soil detachment, flow transport capacity and finally deposition of soil particles. It is suitable for predicting the mean annual rate of soil loss on hill slopes from field-sized areas and in areas of moderate erosion rates. It has been used in different countries under varying conditions (Morgan, 1995). The original model computes the rate of soil detachment by rain splash and the transport capacity of the overland flow. The smaller value is taken to be the predicted annual rate of soil loss. The model has been validated in countries throughout Europe.



Flow chart showing the phases and processes in the MMF model (Source: Morgan, 2001).

Table 24 Input parameters used in the MMF model.

Factor	Parameter	Definition	Unit
Rainfall	R	Mean annual or mean rainfall	mm
	R_n	Number of days of rain in a year	-
	I	Erosive rain intensity, with typical value of 10, 25 and 30 respectively for Temperate, tropical and strongly seasonal climates e.g. the Mediterranean and monsoon types	mm/h
Soil	MS	Moisture content of soil at field capacity or 1/3 bar tension	(% w/w)
	BD	Bulk density of the soil top layer	Mg/m^3
	RD	Rooting depth of the top soil, defined as soil depth from the surface to a stony or impermeable layer; to, the dominant root base (the base of the A horizon), or to 1m depth, whichever is the shallowest.	m
	EHD	Soil effective hydrological depth, which depends on crop cover,vegetation,presence/absence of surface crust, presence of impermeable layer within 0.15 m depth of the surface	m
	K	Detachability index of soil, defined as the weight of soil that is detached from the soil mass per unit energy of rainfall	g/J
	COH	Soil surface cohesion, measured with a torvane under saturated soil conditions	kPa
Landform	S	Steepness of slope	$(^0)$
Land cover	A	The rainfall intercepted by vegetation or crop cover, proportioned between 0 and 1	-
	E_t/E_0	The ratio of actual to potential evapotranspiration ET	-
	C	Crop cover management factor; it combines the *C* and P factors of the USLE	-
	CC	Percentage of canopy cover, which is expressed as a proportion between 0 and 1	-
	GC	Percentage of ground cover,which is expressed as a proportion between 0 and 1	-
	PH	Height of plant, representing the height from which raindrops fall from the vegetation or crop cover to the surface of the ground	m

Obtained from Morgan (1995; 2005)

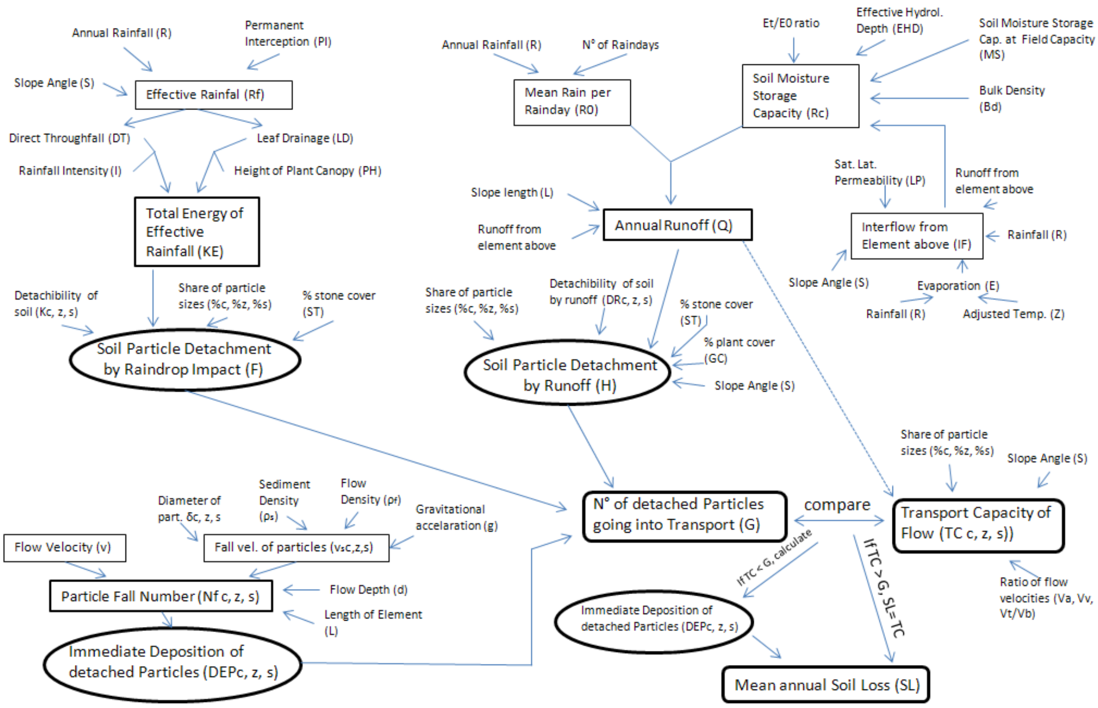


Figure 63 Structure of the modified MMF erosion prediction model (after Schepp, 2014)

4.8 Cropped areas (Source: EDINA)

	Crop area (ha)													
Year	Beans	Bare Soil	Fruit	Grass	Maize	Oil Seed Rape	Peas	Potatoes	Set Aside	Sugar Beet	Vegetables	Winter Cereal	Spring Cereal	Non Agricultural
1969	97107	152727	73883	5189922	43754	24300	68933	174390	0	170697	189716	2879071	0	4195100
1979	15694	0	56466	4758832	33954	0	85358	140289	0	199662	77355	1881144	1164469	4834777
1988	153942	44481	43794	4393962	26590	284380	130017	127604	0	189511	105743	2475338	599012	4685227
1997	95315	0	0	3436320	101498	455091	129546	122438	250677	188576	0	2624213	201056	5654869
2010	144856	0	24427	4075270	126426	578080	60448	89041	0	108825	66968	1998824	230330	5771705

4.9 Calculation of maximum potential soil moisture deficit

Potential soil moisture deficit on day *i* ($PSMD_i$), mm, is calculated from:

$$PSMD_i = PSMD_{i-1} + ETo_i - P_i$$

Where:

ETo_i : reference evapotranspiration for day *i*, mm

P_i : rainfall on day *i*, mm

On days where $P_i > (PSMD_{i-1} + ET_i)$, no soil moisture deficit is assumed to occur and $PSMD_i$ is zero.

In the UK, soil moisture deficits typically start to build up in early spring as evapotranspiration, (ET) is greater than precipitation (P) peak in mid-summer (July-August) and then decline through autumn and winter as P > ET. Therefore in the UK, the estimation of PSMD can start with January as month $i = 1$. The maximum PSMD ($PSMD_{max}$) is the maximum $PSMD_i$ value reached in the 12 months of each year.

Reference evapotranspiration, ETo, approximates to the potential evapotranspiration of short green grass and is estimated from the Penman-Monteith equaltion (Allen et al., 1998).

$$ETo = \frac{\Delta}{\Delta + \gamma^*} \frac{(R_n - G)}{\lambda} + \frac{86.4}{\lambda} \frac{1}{\Delta + \gamma^*} \frac{\rho cp}{ra} (ea - ed)$$

Where Δ is the slope of vapour pressure curve (kPa °C^{-1}), λ is the latent heat of vapourisation (MJ kg^{-1}), ρ is the atmospheric density (kg m^{-3}), cp is the specific heat of moist air 1.013 kJ kg^{-1} °C^{-1}, ea is the mean saturation vapour pressure (kPa) and ed the actual vapour pressure (kPa). Net radiation, R_n (MJ m^{-2} d^{-1}), is estimated from measured shortwave (solar) radiation, assuming a fixed albedo of 0.23 for the reference surface, and longwave radiation estimated from sunshine fraction, air temperature and vapour pressure (Allen et al., 1998). Soil heat-flux, G (MJ m^{-2} d^{-1}), is estimated from change in daily mean temperature (Wright & Jensen, 1972). Aerodynamic resistance, ra (s m^{-1}), is estimated from wind speed using fixed roughness parameters for heat, water vapour and momentum for the reference surface (Allen et al., 1998).

γ^* is the modified psychrometric constant,

$$\gamma^* = \gamma \left(1 + \frac{rc}{ra} \right)$$

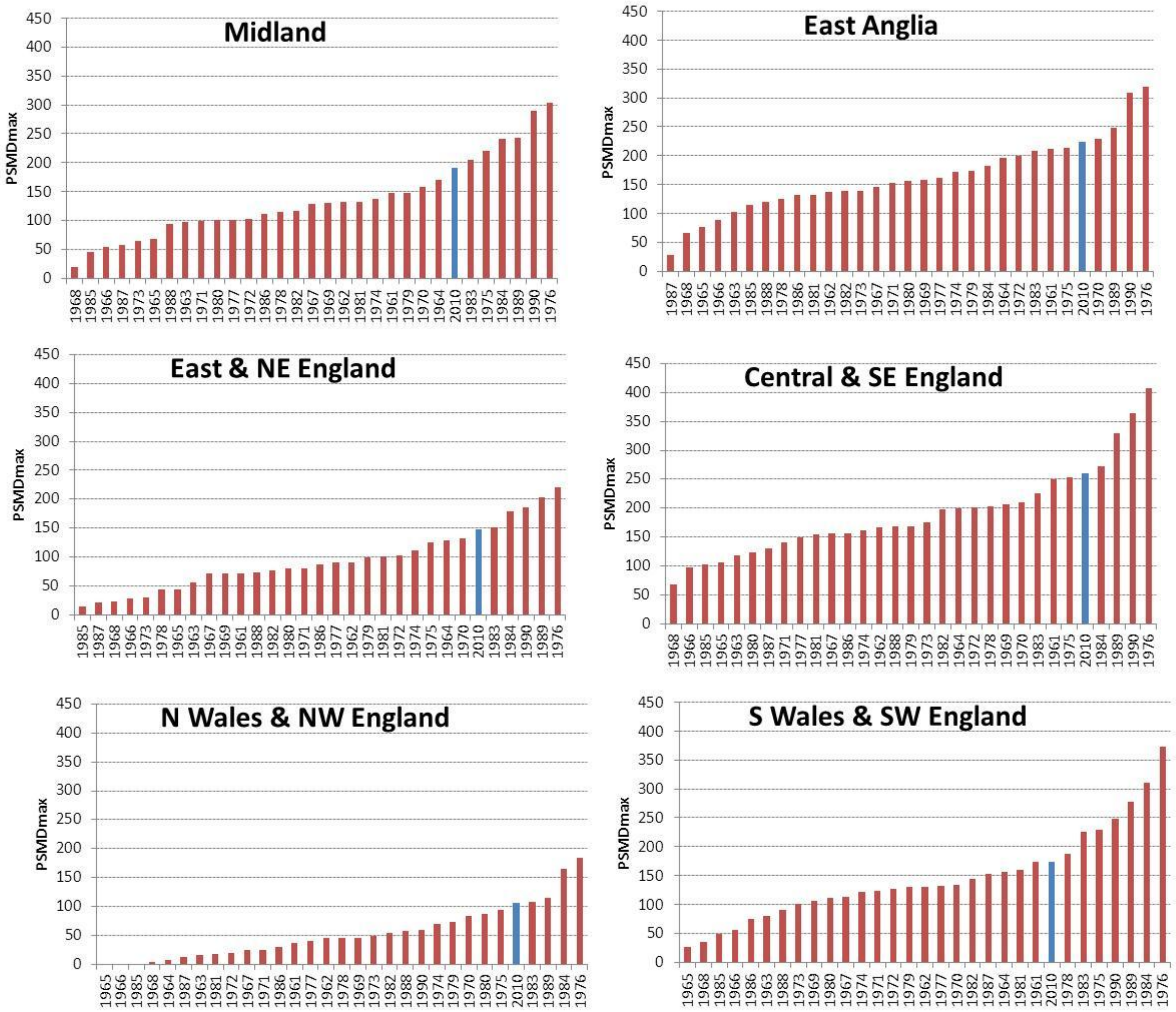
Where γ is the psychrometric constant (kPa °C^{-1}) and rc is the bulk surface resistance for the reference surface (70 s m^{-1}).

The required meteorological data are daily maximum and minimum air temperature, daily mean relative humidity, and daily solar radiation and mean daily wind speed. Where daily solar radiation is not available it can be estimated from the sunshine fraction.

4.10 Regional variations in agroclimate (PSMDmax)

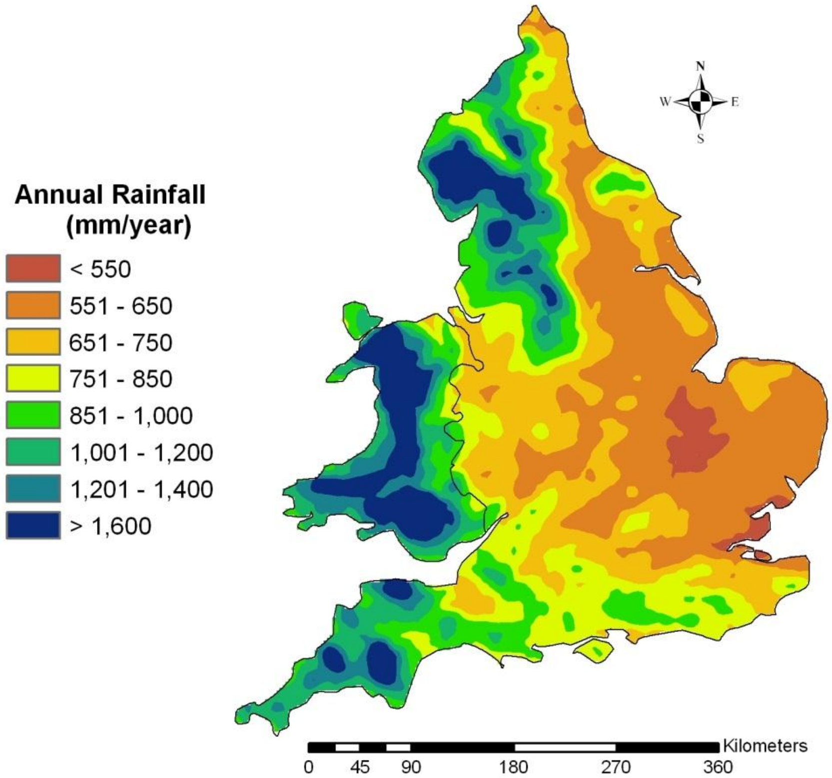
The analysis in Section 3.3.1 highlighted the inter-annual variability in $PSMD_{max}$ for a single site (Silsoe, Bedfordshire). A similar analysis has previously been conducted by Knox et al (2013) at regional level, to show the inter-annual variability in $PSMD_{max}$, and in particular to highlight how representative 2010 was as a 'design' dry year (Figure 61).

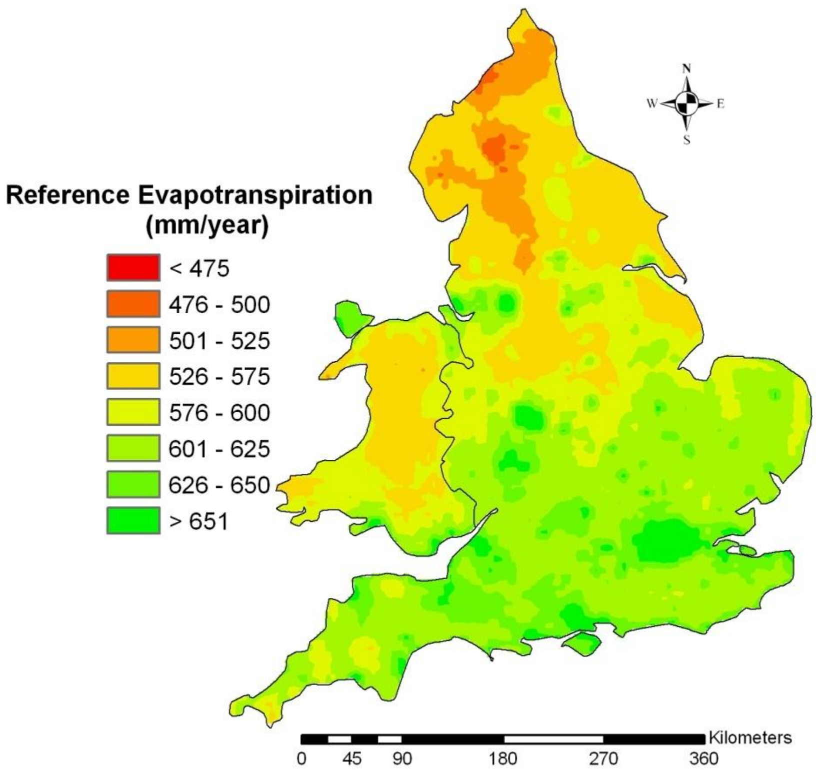
Figure 64 Defining a baseline for demand forecasting using $PSMD_{max}$.



4.11 Annual rainfall, ETo and irrigation need

Figure 65 Spatial variation in average annual rainfall and evapotranspiration (ETo, mm) in England and Wales based on 1961-90 (UKCP09 baseline climatology).





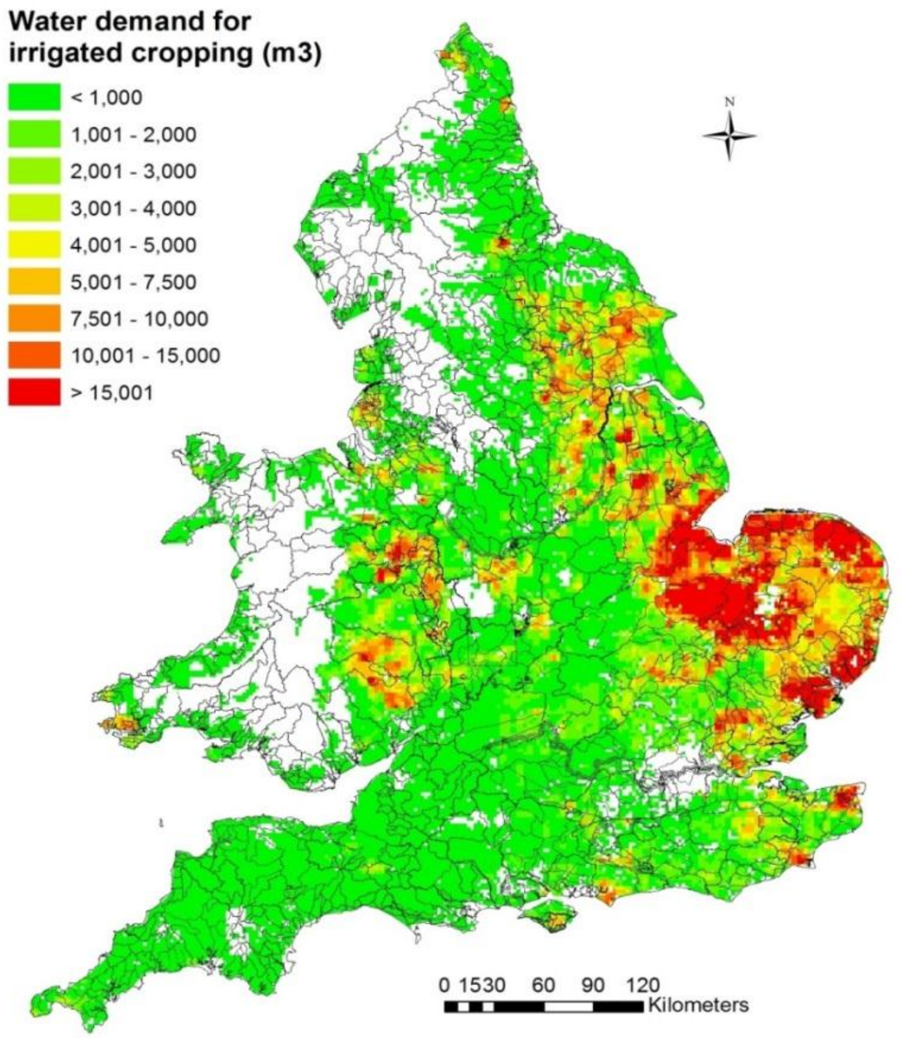


Figure 66 Theoretical water demand in a 'design' dry year for outdoor irrigated cropping, based on 2010 land use (Source Knox et al., 2013).

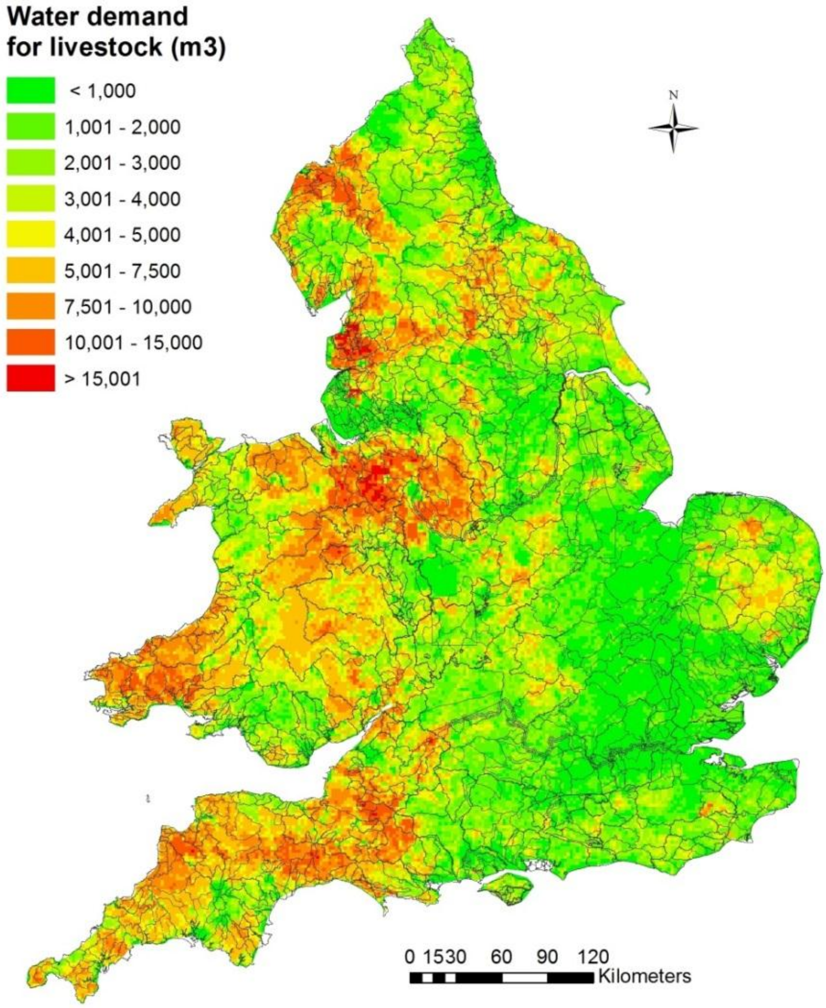


Figure 67 Theoretical water demand for livestock, based on 2010 data (Source: Knox et al., 2013).

Table 25 Summary of theoretical volumetric livestock demand (× 000 m^3), by sub-sector, by EA Region.

Sub sector	Anglian	EA Wales	Midlands	North East	North West	South West	Southern	Thames	Total
Sheep	1499	12254	5474	5922	4151	3864	1220	781	35166
Poultry	4727	1537	1821	1245	992	1457	629	438	12847
Pigs	2703	150	906	2382	312	845	213	400	7911
Dairy	1981	10928	11921	4944	11348	16652	2311	1882	61967
Beef	2368	5705	5057	4317	3109	6012	1142	1194	28904
Total	13279	30574	25179	18810	19913	28829	5516	4694	146795