Defra Project SP0571

Modelling the impact of climate change on soils using UK climate projections

Scientific report

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Executive Summary

This report summarises the outcomes of the 5 workpackages of Defra project SP0571 "Modelling the impact of climate change on soils using UK climate projections".

The soil is subject to a variety of threats, identified at a European level as organic matter decline, erosion, compaction, salinisation, landslides, sealing, contamination and declining biodiversity.. We start with the threats identified as Europe-wide, and seek to quantify the likely response of these aspects of soil functioning in England and Wales to climate change as expressed through UKCIP09 scenarios. This quantification is sought through modelling, while acknowledging that suitable models may not always exist. This activity was identified in the Defra First Soil Action Plan for England: 2004-2006 as a follow-up to the NSRI scoping study SP0538 funded by Defra.

The workpackages address the following:

WP1 Define criteria to be used in assessing models relating the soil threat response in England and Wales to climate change under UKCIP09 scenarios. In doing this, take account of the findings of the Defra scoping study SP0538, and consider identified specific threats. Identify candidate models for predicting the response of variables subject to priority threats.

WP2 Review candidate models with respect to criteria (WP2.1). Synthesise inferences about models to generate a selection of five (WP2.2).

WP3 Acquire models. If necessary, modify UKCIP08 scenarios for input to selected models, accounting for any scale differences. Obtain data sets needed to run models.

WP4 Run models with UKCIP08 scenario data, generating Monte Carlo sequences.

WP5 Assess model results (WP5.1). Present them in graphical form, both as maps using ArcGIS and plots and submit final report (WP5.2)

We list the criteria selected and for each soil threat and tabulate the suitability of a range of possible models. The summary suitability tables are accompanied by short model descriptions, tabulated comments and references by threat. On the basis of suitability when assessed against the criteria, we propose models to be taken forward for use with the UKCIP scenarios at England and Wales scale, and discuss modelling limitations for some specific threats. The models selected were ECOSSE for carbon, PESERA for water erosion, RWEQ for wind erosion, the Workable Days model for compaction, PSYCHIC for diffuse source phosphorus, VSD for acidification and SALTMED for salinisation. There were not considered to be suitable models for predicting the response of landslide frequency and magnitude to climate change, nor of the response of soil biodiversity. Of the sealing threats, only soil sealing was considered (so excluding sealing due to roads construction, urban development etc), and while models existed, projections of their driving variables were not available.

We describe the use of data from one member of the HadRM3 11-member climate projection ensemble as providing drivers for soil threat modelling. These were as a work-around following the unsuitability of the UKCIP09 data for national-scale modelling. The HadRM drivers were partially bias-corrected and down-scaled from 25 to 5km. Parameter sets needed for soil threat model running were acquired. The JULES land surface model driven by climate change scenarios was run to provide daily 5km estimates of variables not provided by HadRM3 but needed by some soil threat models.

The selected models were run, with a focus on the decades 2020s, 2030s, 2050s and 2090s. Given the drivers and models provided, the following general projections were made:

Carbon: results from the ECOSSE simulations suggest that climate change, based on the HadRM3Q0 ensemble run which assumes a medium emission scenario, will only be responsible for relatively small changes in soil C content in England and Wales over the period 2010 to 2080.

Water erosion: results suggest a projected increase in (spatially averaged) erosion in general to the end of the century due to increased winter rainfall. Increases are predicted to be greatest in upland areas of England and Wales.

Wind erosion: present climate change models do not predict major changes in wind speeds, and this is reflected in a lack of any significant effect of climate change on wind erosion. This may be a reflection of the limitations of climate change models in predicting wind speeds.

Compaction: based on 'Workable Days' defined by days drier than field capacity, soils will be slightly less vulnerable to compaction. In some regions, such as southwest England and Wales, localised risk of soil compaction will remain into the future because of very few winter workable days.

Phosphorus: phosphorus loss from the soil is strongly associated with runoff, and lower projected runoff in the 2030s and 2050s compared to the present day gives projected significant decreases in phosphorus loss from soils. However, concentrations of P in runoff and drainage from agricultural land do not change significantly.

Acidification: soils in England and Wales vulnerable to acidification are in the process of recovery from acidification in the recent past. Our results suggest increased weathering under projected climate change would tend to accelerate recovery, while reduced runoff would slow down recovery. The combined effect would be to make little difference to recovery. Only the temperature dependence of weathering was considered in our analysis. There may be additional effects of other soil processes whose temperature dependence was not considered.

Salinisation: we considered the effect of storm surges at a small number of coastal sites. The projected increased incidence of coastal flooding due to storm surges will be accompanied by short term salinisation. The ability of the soil to recover from inundation is charted.

Finally the dependence of the results on the selection of HadRM3 drivers and the model assumptions is discussed. It is suggested that the modelling protocols developed in the project be implemented with UKCIP09 data following the improvement of access to those data.

In addition to the main report, a fuller description of the work done appears in four Appendices.

Work Packages 1&2

WP1 "Define criteria to be used in assessing models relating the soil threat response in England and Wales to climate change under UKCIP08 scenarios. In doing this, take account of the findings of the Defra scoping study SP0538, and consider identified specific threats. Identify candidate models for predicting the response of variables subject to priority threats."
WP2 "Review candidate models with respect to criteria (WP2.1). Synthesise inferences about models to generate a selection of five (WP2.2)".

Climate change in England and Wales has the potential to affect the functioning of the soil. The nature of the effect is uncertain, but can be estimated through the use of models which approximate the cause and effect relationships between climatic drivers and the soil response. There are a number of potential changes in soil functions of particular concern, identified as "threats", namely changes in:

- Soil carbon
- Erosion by water and wind
- Contaminant transport
- Incidence and magnitude of landslides
- Soil compaction
- Salinity, either through non-marine accumulation or through ingress of marine water
- Surface sealing of agricultural soils through natural processes
- Biodiversity

Models of these threats are driven by climate variables, and UKCIP09 climate change scenario data for England and Wales were released during 2009, updating previous scenarios. We reviewed soil threat models which respond to climate change inputs and suggest a number to be run with the scenarios to estimate the soil threat response. Not all models are suitable for application, and ten basic criteria have been selected against which they are judged. These are shown in Table 1.

Table 1. The model criteria

No.	Criterion
1	Does it provide estimates of change in the soil threat as a function of climate change variables?
2	Are these at a space and time scale which can be used for national-scale decision making?
3	Does it cover all aspects of the soil threat, particularly those of greatest perceived importance - ecological, economic etc?
4	What is its track record in modelling historic data as a function of climate change variables? How well does it simulate response variables?
5	Does it require additional parameters or driving variables, and are these available into the future?
6	Is it responsive to other changes, particularly land use?
7	Where are the likely bottle-necks in applying the model?
8	If a model looks promising but requires a few changes, how readily can these be made? Can this be done within the scope of the project?
9	Cost implications
10	Compatibility with other models

UK specialists in each threat who form part of the project consortium have selected a small number of candidate models for each threat. These have been judged against the ten criteria. The models reviewed and the individuals selected are summarised, with a fuller analysis and references provided in Appendices A, B and C

Carbon

The main threat to the carbon content of soils is possible loss to the atmosphere, which in turn contributes to climate change. The **RothC** (Rothamsted Carbon) model (Coleman & Jenkinson, 1996) has been used to examine soil organic carbon (SOC) stock changes in the UK (Falloon et al., 2006) and Europe (Smith et al., 2005, 2006, 2007a). The **ECOSSE** (Estimating Carbon in Organic Soils - Sequestration and Emissions) model was developed by combining RothC with the Rothamsted N model (Sundial; Bradbury et al., 1993; Smith et al., 1996) and further developing it for the simulation of anaerobic conditions for use on organic soils (Smith et al., 2007b). **DayCent** (del Grosso et al., 2005) is the daily version of the CENTURY model (Parton et al., 1987), which along with RothC, is among the most widely tested and used SOC model in the world. Like ECOSSE, DayCent simulates carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions in addition to SOC turnover. It is being used by the University of Aberdeen team to examine N turnover in Europe as part of the EU-funded NitroEurope-IP project. **DNDC** (Denitrification-decomposition; Li et al., 1994) is a process based model of soil C & N turnover which also simulates CO_2, CH_4 and N_2O emissions in addition to SOC turnover. It is being used by the University of Aberdeen team to examine C and N turnover in Europe as part of the EU-funded CarboEurope-IP project (Wattenbach et al., 2009).

Assessed against the evaluation criteria, the RothC/ECOSSE models appear to be the most appropriate for application in England and Wales for assessing climate impacts using UKCP09 data. **ECOSSE** was selected for modelling soil carbon

Erosion

A. Water

Loss of soil by water erosion potentially reduces agricultural productivity and other soil functions. It also affects downstream water quality. The **RUSLE** (Revised Universal Soil Loss Equation) computes average annual erosion from field slopes as a function of five factors and land management, specifically: a rainfall-runoff erosivity factor, a soil erodibility factor, a slope length factor, slope steepness, a cover management factor and conservation practice (Renard et al., 1997). RUSLE has been applied in the USA and in soil erosion studies worldwide (Merritt et al., 2003). The US **WEPP** (Watershed Erosion Prediction Project) developed a physically-based model (Laflen et al., 1991) containing sub-models which are parameterized by empirical relationships derived from plot studies. **EUROSEM** (Eurpoean Soil Erosion Model) was developed as a distributed, event-based hillslope model that predicts total runoff and soil loss, produces hydrographs and sediment graphs for each event. The modular structure represents rainfall, interception processes, infiltration, overland flow, sediment transport capacity and its deposition (Morgan et al., 1998). **PESERA** (Pan–European Soil Erosion Risk Assessment) model is a physically based, spatially distributed model developed to quantify soil erosion of environmentally sensitive areas relevant to a regional or European scale (Kirkby et al., 2008).

On the basis of the assessment criteria, the **PESERA** model ranks most highly and it has been selected in this study for the assessment of soil erosion by water. Its only potential weakness compared to the RUSLE model is the limited number of field-based validation studies against which it has been tested. However, the RUSLE has been available for several more years than PESERA and so the significance of this observation is limited. The PESERA model represents the processes of soil erosion and can be implemented at a scale or pixel resolution of 250 m^2 or 1000 m^2. It is well-suited to predicting average soil erosion rates in response to climate change data at the national-scale.

B. Wind

Wind erosion is a local phenomenon in England and Wales, but of sufficient potential importance to be worthy of study. The **RWEQ** (Revised Wind Erosion eQuation) model represents the latest developments in wind erosion prediction from research undertaken in the USA since the 1930s. The precursor to RWEQ was the original Wind Erosion Equation (WEQ). The RWEQ estimates soil eroded and transported by wind between the soil surface and a height of two meters (Fryrear et al., 1998). **WEELS** (Wind Erosion on European Lights Soils) wind erosion model was developed under a European Union (EU)-funded research project 'Wind Erosion on European Light Soils' (WEELS http://www2.geog.ucl.ac.uk/weels/report.htm). **WEPS** (Wind Erosion Prediction System) is a process-based, daily time-step model that predicts soil erosion via simulation of the fundamental processes controlling wind erosion. WEPS calculates soil movement, estimates plant damage, and predicts PM-10 emissions when wind speeds exceed an erosion threshold (Wagner et al., 2003). The basis of the empirical **WEQ** (Wind Erosion Equation) model is the soil erodibility factor (I) which is potential soil erosion in tonnes per hectare per annum from an unsheltered field with a bare, smooth surface. WEQ has been widely applied and validated (Fryrear et al., 2001; Buschiazzo and Zobeck, 2008).

The simplified **RWEQ** model ranks highest against the selection criteria and it has been selected in this study for the assessment of soil erosion by wind. Its areas of potential weakness are its limited response to land use change and representation of process complexity, but on all other assessment factors it scores highly. Its proxy measure for soil erosion (Q_{max} or maximum transport capacity) can be computed at the required spatial and temporal scales to provide an assessment of erosion in response to climatic variables. The RWEQ model has been widely applied and validated. To the best of our knowledge there are no models that have been designed to predict soil erosion by wind for organic-rich soils.

Contaminants

Soil contaminants include a very wide range of substances. Their potential effect is not only to degrade soil function, but to act as a source of downstream contamination in streams. In our review of contaminant models we consider 4 classes, these being (i) diffuse agricultural pollutants (e.g. NO_3^-, PO_4, N_2O), (ii) organics and pesticides, (iii) acidification and (iv) inorganic contaminants (e.g. PHE). The contaminants selected are based on those named in Defra Report SP0538 (Defra, 2005) and the EU Soil Thematic Strategy (Van Camp et al., 2004). A literature and web based review has been undertaken to identify candidate models. A brief discussion of the major models in each category is presented.

Numerous models have been produced capable of operating on a range of scales including plot, field, catchment and national scales to aid policy decisions. Many of the best plot and field scale models have been incorporated into national scale model frameworks along with other sub-models to describe processes such as leaching, plant uptake and evapotranspiration. In addition information from various national databases on soil properties such as the National Soil Inventory (NSI) soil databases, HOST (Boorman et al. 1995), agricultural census and climate are included. Therefore this effort has often resulted in one model evolving specifically for catchment or national scale that incorporates the best modelling approaches developed over a long period.

One such example is the MAGPIE framework managed by ADAS and the UK academic community (DEFRA, 2007a). It originally started as the framework for a nitrate leaching model (Lord and Anthony, 2000), but has since developed to include a number of nitrate and phosphorus pollution models that use the same databases. The MAGPIE model is based on several national databases and surveys along with sub-models linked with a GIS system and interpolated on a $1km^2$ grid. The sources of input data come from (i) the annual agricultural census data (crops and livestock), (ii) land use derived from satellite data and CEH's land cover map, (iii) climate information supplied by the MET office and (iv) soil type and property data (NSI).

A. Diffuse Agricultural Contaminant Models

The number of agricultural contaminants and their models is large, and it is not within the scope of this project to model each and every one. We therefore provide a review of models available for the major classes of contaminant, and identify two representative examples for modelling.

A.1 Nitrate models

The plot, field and catchment models that have been used to predict nitrate loss from soils include LEACHM (Sogbedji et al. (2001), GLEAMS (Webb et al. 2001) and SWAT (Neitsch et al. 2005). Two nitrate models run under the MAGPIE framework (NEAP-N and EveNFlow). These include NEAP-N that describes nitrate leaching to soils and is the basis for defining Nitrate Vulnerable Zones throughout England and Wales (DEFRA, 2008). EveNFlow (Euroharp, 2004) is a development on NEAP-N and can predict the nitrate leaching to rivers and water courses. NIRAMS is the Scottish equivalent (Dunn et al. 2004a, 2004b) and is very similar to MAGPIE but includes a more comprehensive methodology for hydrological transport and routing to predict stream water N concentrations. These models utilise national datasets with regard to land use and agricultural practice that are updated continuously. These models utilise a whole series of sub-models that have been developed to simulate the N cycle in soils and interactions between vegetation and hydrology in agricultural ecosystems. Of these models, **LEACHM** scored highest against the selected criteria.

A.2 N_2O models

A range of models exist for modelling N_2O that operate from site specific such as the DeNit model (Reth et al. 2005) to those capable of being run at national scale. The national scale models include DNDC (Institute for the study of Earth, Oceans & Space, 2007), CENTURY/DAYCENT (Del Grosso, 2007) and ECOSSE (Smith et al, 2007). **ECOSSE** was judged best for predicting N_2O loss from soils into the 21st century.

A.3 Phosphorus models

Within the ADAS MAGPIE framework, two models exist that can be used to assess P loss from soils at a national scale. These are PIT and PYSCHIC (Davison et al. 2008; Heathwaite et al. 2003; Heathwaite et al, 2005; Liu et al. 2005). The evolution of these models is such that PIT has now been largely subsumed into PYSCHIC which is now capable of being run nationally (*pers. Comm*. Eunice Lord). LEACHM/GLEAMS and SWAT include phosphorus as well as nitrate loss in their nutrient loss components. We selected **PSYCHIC** for predicting phosphorus loss.

B. Pesticides and Organic Contaminants

Catchment scale models of the fate of pesticides and organic contaminants include SWATCATCH (Neitsch et al. 2005) and POPPIE (Brown et al. 2002). There are no UK national scale models actively being developed to predict leaching of pesticides. However, some previous attempts have been made to examine aspects of leaching on a larger scale. These include (i) the European wide EUROPEARL model (Tiktak *et al*. 2004) and (ii) an attempt to add a national scale groundwater component to the EA's POPPIE model which was developed by Holman *et al*. (2004).

C. Acidification

The SAFE soil acidification model works on profile scale (Alveteg et al., 1995; Sverdrup et al. 1995), whereas in the recent report on Critical Loads and Dynamic Modelling (CEH, 2007), two larger scale models were examined; the MAGIC (Cosby et al. 1985a & b; Hardekopf et al, 2008) and VSD (Posch et al., 2009; Reinds et al. 2008) models. The MAGIC model was used primarily for regional modelling of over 53 sites whilst VSD was used for UK wide modelling because of its greater simplicity (VSD has a single soil compartment while MAGIC allows up to 3 and process representation in VSD is slightly simpler). Whilst both models are similar and produce similar results if VSD is configured appropriately, the VSD model has advantages that it can be run from an Access database (a simpler calibration routine that enables large numbers of model runs) and that it has already been run for the UK. In view of this, **VSD** was chosen to model acidification.

D. Inorganic Contaminants

Inorganic contaminants include those elements described as ‘PHE’s (Potentially Harmful Elements) (lead, cadmium, arsenic, mercury, chromium, copper, nickel, zinc). Whilst, there are national soil inventories of heavy metal (lead, cadmium, arsenic, mercury, nickel, iron, zinc, manganese) concentrations in soils such as the Soil Geochemical Atlas of England & Wales (McGrath & Loveland, 1992) and the G-BASE geochemical survey (British Geological Survey), there are currently no available models for determining the effects that climate change may have on the mobility and speciation of PHE’s on a national scale. Typical models for predicting solubility and mineral phases of heavy metals typically require information that is site specific and of greater detail such as DOC, an estimate of labile metal, pore water concentrations, pH and organic carbon. Much of this detail is not held at national scale. Models such as WHAM (Tipping, 1994) are regularly used to undertake soil pore water speciation. Other models may include the Nica-Donnan model (Kinniburgh et al, 1996), and PhreeqC (Parkhurst & Appelo, 2000). In terms of the criteria selected there was little to choose between **WHAM** and **PHREEQC** depending on the element being studied**.**

Within the scope of the project it was judged not possible to model all potential contaminants. While models of all were reviewed, **only phosphorus and acidification were selected for modelling**. These represent first a contaminant derived from intensive agricultural management and secondly a form of contamination largely associated with fragile semi-natural conditions influenced by land use and atmospheric deposition We do not discuss in detail the assessment of models of other contaminants.

Compaction

The **Workable Days** model of Cooper et al. (1997) was the only model we found that has been used to predict changes in soil compaction risk associated with climate change. Cooper et al. (1997) used predictions from the Scottish Climate Change Group to model the influence on workable days. The model is based entirely on soil moisture, with a Workable Day defined as drier than a specific water potential, generally field capacity. Wetter conditions produce fewer Workable Days. In combination with the soil moisture model **JULES**, it was possible to gain a much more rigorous assessment of compaction risk for both topsoil (0-10 cm) and subsoil (10-35). With the **Workable Days** model even a few days difference will be detected, providing the level of resolution required to predict potential climate change impacts. Whereas Cooper et al. (1997) defined a workable day as drier than 110% water content at -5 kPa water potential, we used -10 kPa. The prediction of Workable Days has implications to the 'Wetness Class' in DEFRA's Agricultural Land Capability (ALC) assessment, although there is a minor difference in the shallowest depth considered between approaches. A future project could consider the combined implications of climate change predictions to temperature and soil moisture changes caused by climate change on ALC across England and Wales.

The **Workable Days** model only defines the vulnerability to compaction and does not consider susceptibility. The **Expert Model** presented by Jones et al. (2003) used England and Wales national soil survey data to categorise susceptibility based on soil texture, structure and packing density. However, the model produces only 4 classes of compaction risk, based on a combination of an initial 4 classes of soil susceptibility (texture etc.) and 5 classes of vulnerability (Workable Days). Most UK soils fall into 2 of the vulnerability classes so we felt the approach would not provide adequate resolution to detect differences in compaction risk without modification that is beyond the scope of this study. Moreover, the susceptibility classes are based on expert assessment rather than a mechanistic understanding of compaction. **Mechanistic models** exist and vary in complexity from deterministic models that require considerable data (Thu et al. 2007; Defossez et al. 2003) to qualitative models that can be applied across different soil types (Schafer et al. 1992; Horn and Fleige 2003; Horn and Fleige 2009). However, data on soil compression characteristics are not available in the UK to apply even the most qualitative mechanistic model.

On the basis of this assessment we recommend that the **Workable Days** model be used as the soil compaction threat model for use with UKCIP09 scenarios.

Landslides

The **Antecedent Water Status m**odel developed by Crozier (1999) and others has been applied to shallow, rainfall triggered landslides within the urban environment of Wellington, New Zealand. The model calculates an index of soil water over the preceding ten days before a storm event. Downscaling of General Circulation Models (GCM's) is motivated by evidence that climatic factors are proven to be among those that trigger instability, and GCM's are useful in providing future climate projections. GCM's provide a global coupled ocean-atmosphere model; however the patterns produced are large scale and have in the past been downscaled to provide information for regional/local studies (Dehn *et al*., 2000). Dixon and Brook (2007) point out that threshold types of studies are highly site, region and material specific and it is therefore not possible to use values taken from the literature to apply to landslides with different behaviour in different climates and geological material. **Threshold values** used can be derived from empirical, semi-empirical or physically based data (Crosta, 2003). The **Enhanced GeoSure** model utilises three datasets unique to the BGS – the GeoSure landslide susceptibility model, the BGS debris flow potential model and the BGS Quaternary Domain model http://www.bgs.ac.uk/products/geosure/ Reference will also be made to the BGS National Landslide Database and National geotechnical Database. Deterministic S**lope Stability** models, as used in ground engineering, are well-established (>70 years) and there are a variety of analytical methods available. The simplest form is the 'infinite slope analysis' (e.g. Schmidt & Dehn, 2000).

Our review indicates that at present there are **no landslide models** which could be used to quantitatively predict change in the incidence or severity of landslides in England and Wales in response to climate change.

Salinity

A. Terrestrial

While salinity is a serious soil threat elsewhere in Europe, it is not currently a major concern in England and Wales. On a global scale the problem is commonly due to the use of brackish irrigation water, linked to high evaporation and insufficient drainage to surface waters. The **SALTMOD** model was developed by Oosterbaan (2002) to simulates and predict the salinity of soil moisture, ground and drainage water (groundwater), the depth of the water table, drain discharge and leaching of salts in irrigated agricultural lands under different (geo) hydrologic conditions, varying water management options, including the (re) use of ground water for irrigation by pumping from wells (conjunctive use), and several crop rotation schedules. The **HYDRUS** model was developed by Simunek (2005), and has now reached version 4 (http://www.pc-progress.cz/Pg_Hydrus_1D.htm). The **SALTMED** model was developed by Ragab (2002) and Ragab *et al*. (2005), and also at http://www.ceh-wallingford.ac.uk/research/cairoworkshop.

Of the three candidate models, **SALTMED** is judged the most suitable for modelling the terrestrial scale soil salinity threat. One possible disadvantage for national-scale use is that the model is conceptualised at plot scale.

B. Salt water intrusion

Salt water intrusion is the process by which saline water infiltrates groundwater bodies, and takes the place of freshwater abstracted for irrigation. The **SWI** (Sea Water Intrusion) model (Bakker & Schaars, 2003) is a variation of the MODFLOW model with two water densities but runs at a smaller spatial scale than MODFLOW (hundreds of metres or less). **SEAWAT** (v.4; Langevin *et al*. 2008) simulates three-dimensional variable-density groundwater flow. It is based on MODFLOW (v 2000)/MT3DMS to simulate groundwater flow coupled with multi-species solute and heat transport. **SHARP** (Essaid, 1990) is a quasi three dimensional finite difference model to simulate a fresh water and saltwater flow separated by a sharp interface in layered coastal aquifers when the transition zone is small relative to thickness of the aquifer. **SUTRA** (v. 2.1; Voss and Provost, 2008) is a two or three dimensional saturated-unsaturated, variable density groundwater flow with solute or energy transport. The model allows irregular meshes and accepts input data from separate files

The sea water intrusion models are aquifer specific and run for each aquifer separately. They are suitable to investigate the aquifer threat rather than soil threat. Since this project is focused on soil threats, **sea water intrusion models are not considered relevant.**

Sealing

In the context of pan-European soil threats, soil sealing is interpreted as covering the soil with impermeable material such as buildings, paving, concrete, asphalt, etc, usually associated with urban development and not influenced by climate change in any mechanistic fashion. However, we consider soil sealing only as the development of the natural impermeable surface layer which is found in some soils under some weather conditions. **Assouline and Mualem** (1997) developed a dynamic model that relates the formation of a seal at the surface of a bare soil exposed to water drops impacts to the initial soil mechanical and hydraulic properties as well as the physical characteristics of the regional rainfall or the applied irrigation. **Darcy type** models are based on reduced flux at the surface (Darcy (1856), Maidment (1993)). **Horton type** models are based on infiltration and conductivity reduction (Kostiakov (1932), Horton (1940) and SCS curve number method (USDA, 1972)). **Conceptual models** based on saturated seal layer of fixed thickness. A further model class is based on **Richards equation** (Richards, 1931; Marshal and Holmes, 1979), and finally there are **Approximation based** models (Green & Ampt (1911), Philips (1957)).

While there are several models which can model changes in infiltration following change in surface soil characteristics leading to sealing, only the model of Assouline and Mualem simulates changes in soil sealing under changed climatic conditions. This requires rainfall intensity variables which are not available from either UKCIP09 or the soil moisture model JULES, so that **soil sealing was not modelled**.

Biodiversity

Soil biodiversity depends on the nature of the soil environment, and any change in this is likely to affect the species balance of the soil due to changes in competitive advantage. The **Gbmove** model has been developed to allow prediction of the impacts of changing soil and climatic conditions on plant species composition and biomass in UK Priority Habitats (Smart et al., submitted). **Soil biodiversity** model and other **food web** models are available that simulate carbon flow through the soil community (http://www.nerc.ac.uk/publications/other/documents/soilbio_soilmodel.pdf). These utilise data on food web composition and energy flow to calculate the carbon flows between groups. **PERPEST** (van den Brink et al., 2006) is a model that Predicts the Ecological Risks of PESTicides in freshwater ecosystems. In particular, taxa specific effects of pesticide residues are considered. The system predicts the effects of a particular concentration of a pesticide on various (community) endpoints, based on empirical data extracted from the literature. **SOILPACS** uses a traditional approach to community characterisation is using morphological keys and classic biodiversity-based comparisons. A working example of this for UK rivers is the UK River Invertebrate Prediction and Classification System (RIVPACS) (Clarke *et al.* 2003).

Our review indicates that at present there are **no models** which could be used to quantitatively predict change in soil biodiversity in England and Wales in response to climate change.

Work Packages 3-5

WP3 "Acquire models. If necessary, modify UKCIP08 scenarios for input to selected models, accounting for any scale differences. Obtain data sets needed to run models".
WP4 "Run models with UKCIP08 scenario data, generating Monte Carlo sequences".
WP5 "Assess model results (WP5.1). Present them in graphical form, both as maps using ArcGIS and plots and submit final report (WP5.2)"

Soil threat models are driven by climate variables, and it was in anticipation of the publication of new climate change scenarios for England and Wales that this project was initiated. UK climate projections (UKCIP09; http://ukclimateprojections.defra.gov.uk/) were duly released in summer 2009, providing simulations of a range of climate variables from the present to 2100 on a 25km model grid. Access to the data is through a web-based user interface. While the original intention was to use the UKCIP data as climate drivers for models of soil threats, in the event this presented a number of difficulties. Access to daily projections was point-wise through the point-and-click user interface, generating large amounts of data from a set of randomisations of model runs. This was impractical for use at national scale. Monthly data on a 25km grid were more readily available from the UKCIP09 website, but data at this temporal scale were not suitable for use with some models. In addition, a number of variables needed for models were absent from UKCIP09, notably short and long wave radiation.

The UKCIP09 projections were derived from the HadRM3 downscaled GCM projections (http://badc.nerc.ac.uk/data/hadrm3-ppe-uk/) on the same 25km grid. The daily HadRM3 data comprise outputs from an 11-member ensemble of runs of the Hadley Centre Regional Model. The data are readily accessible and downloadable in computer-readable form from the British Atmospheric Data Centre (BADC; http://badc.nerc.ac.uk/home/). The wide range of variables generated by the HadRM3 model includes most of those required for soil threat modelling which are absent from the UKCIP09 data set. To overcome the difficulties of using the UKCIP09 data we used data from a single ensemble run of the HadRM3 data, HadRM3Q0. We downloaded daily HadRM3Q0 data to 2100 from the BADC website. The HadRM3 run used to generate these data assumes a medium emission scenario (A1B). Computational limitations restricted the modelling to the use of a single ensemble member, and this has the disadvantage of not representing the full range of variability which the HadRM model is capable of producing. The HadRM data are also not bias-corrected for the UK. These are two limitations do not apply to the UKCIP data since these, despite their other limitations, account for the variability in the 11-member ensemble and are bias-corrected against historic data.

The trajectory of the projections given by the single HadRM3Q0 ensemble member deviates from the central values of the UKCIP projections. To assess the extent of deviation, we aggregated HadRM3 daily projections of temperature and rainfall to coincide spatially and temporally with the UKCIP09 data. This comparison has been made for monthly data for each England and Wales 25 km square. UKCIP09 data are provided by decade, and the decadal values are computed over a 30 year period centred on the decade of interest. In order to make a comparison, we therefore aggregated HadRM data over the 30 year periods corresponding to the 2020s and the 2080s. We then regressed the UKCIP09 values on the HadRM aggregate values using location (by 25km grid square) and month as explanatory variables to provide a simple predictive equation for removing bias with respect to UKCIP09 data from the HadRM3Q0 projections. For both the 2020s and the 2080s the regression coefficients associated with location and month were similar and on the basis of this, the same regression equation was used for the remaining decades of interest. This procedure provided a monthly and a site bias removal factor which was applied to all daily HadRM rainfall (multiplicative factors) and temperature (additive factors).

The selected spatial scale for soil threat modelling was 5km, this being a compromise between the 1km scale at which relatively stable land characteristic data were available, and the 25km scale of the climate projections. Nevertheless there is some doubt as to the suitability of climate projections at a scale finer than 25km (or even at a scale as fine as 25km) as drivers for environmental models. The use of projections at this spatial scale are best regarded as a feasibility exercise aiding in the provision of general understanding of the regional consequences of climate changes in a particular direction (Nature, 463, 7283, p849 (editorial))

After bias-correcting precipitation and temperature data, the HadRM3 data were downscaled from the 25km to the 5km grid using methods appropriate to each variable used in modelling. Our modelling focuses on the decades 2000s, 2020s, 2030s, 2050s and 2090s. For some models a continuous run of data was required through the 21st century to provide continuity and appropriate initial conditions between decades.

Downscaled HadRM3 variables were sufficient as drivers for most of the soil threat models. However, ECOSSE requires projected net primary productivity (NPP) values, and the Working Days model (compaction) requires projected water content through the soil profile. These variables are provided by the JULES land surface water and energy budget model (Blyth *et al*., 2010), itself using HadRM3 outputs as drivers. JULES was therefore run at a 5km grid scale on a daily time step to 2100 with England and Wales soil and vegetation properties, along with the appropriate HadRM3 drivers.

Soil and land cover parameters are required as parameters of both JULES and the soil threat models. The key soil parameters are those which are associated with the amount of water the soil can hold, and the rate at which water can be transmitted through it. These hydraulic parameters have been empirically derived for each England and Wales soil class, and for this project were provided under licence by Cranfield University from its Land Information System (LandIS). The key datasets which were used to generate soil property information at the 1 km grid scale were: i) the National Soil Map (1:250 000 scale) which includes information on the spatial distribution of all Soil Associations, ii) the proportions of the different Soil Series within the Associations, and iii) the property information within individual soil horizons of the Soil Series. These datasets are linked in a GIS using identifiers which can be used to generate estimates of a variety of topsoil properties across England and Wales including soil organic carbon content (%), texture (proportions of sand, silt and clay particles), bulk density (g cm^{-3}), parameters of the van Genuchten soil water release curve, saturated hydraulic condiuctivity, water content at certain fixed water potentials. These hydraulic data are provided for different horizons within the soil profile.

The chemical and biological properties of the soil (and possibly bedrock) are important for some threats, notably acidification. The sources of these more model-specific parameters are provided under the descriptions of the individual models.

In the 5km square grid-based modelling used here, the soil parameters used the parameters of the dominant soil class in that grid square. In reality, local soil properties may vary greatly within a 5km square, and in this case the useof the dominant class may be a significant approximation. Some of these properties, such as soil texture, vary over short scales (100-200 metres) and so the 1km grid data provide typical values for each grid square. The JULES model distributes water and energy through 4 (idealised) soil layers of depths of 0.1, 0.25, 0.65 and 2.0m. Soil hydraulic parameters are estimated for each soil depth.

Vegetation influences evapotranspiration and therefore water accumulation in the soil following rainfall, and thus indirectly affects soil threats. Vegetation can also provide protection from erosion, and a sink for nutrients which are otherwise potential contaminants. Depending on its form, vegetation may have different effects. Vegetation classes ("plant functional types" or PFTs) used in JULES are broad-leaved trees, needle-leaved trees, crops, grass and shrubs. Non-vegetated land classes are urban, lake and soil (ie bare soil). The finer classification of the LCM2000 (http://www.ceh.ac.uk/sci_programmes/BioGeoChem/LandCoverMap2000.html) is the basis for defining the five PFTs. The NPP of each PFT is used in the ECOSSE model to estimate the carbon budget of the soil. Vegetation classes are identified at a 1km grid square scale, and the dominant class is used for 5km scale modelling using JULES. Each vegetation class is assigned a set of global-scale parameters associated with its physiological response to climate and season. It is likely that the use of global parameters introduced bias in the vegetation response in England and Wales.

The salinity model SALTMED uses crop data taken from the annual Defra survey 2km square estimates available to researchers from UKBORDERS (http://edina.ac.uk/ukborders/). It also uses storm surge data rather than HadRM3 or JULES data directly as its climate change driver.

One further variable required for interpolation is elevation, obtained from the IHDTM (Morris & Flavin, 1990). Elevation is used in the interpolation of rainfall and temperature from the 25km to the 5km grid.

Carbon

Climate change has been implicated as the cause of an observed mean loss of soil organic C of 0.6% yr^{-1} between 1978 and 2003 in England and Wales based on data from the National Soil Inventory (Bellamy et al. 2005). As discussed by Smith et al. (2007), those findings contradict evidence that UK and European soils as a whole are a net CO_2 sink (Janssens et al. 2003). The Countryside Survey reported that carbon concentration in the top layer of the soil (0-15 cm) increased in Great Britain between 1978 and 1998, and decreased between 1998 and 2007 (Emmett et al. 2007). The Countryside Survey found no overall change in carbon concentration in Great Britain between 1978 and 2007 and could not confirm the loss reported by the National Soil Inventory. Data from another long term study of soil organic C in British woodlands (Kirkby et al. 2005) suggested a small increase in soil organic matter over 30 years (0.094% increase yr^{-1}). Our results provide modelling evidence that will allow the threat of climate change on the soil organic C stocks of England and Wales to be quantified by incorporating direct climate impacts on the soil and indirect effects via temperature and precipitation driven changes in net primary production (NPP).

The ECOSSE (Estimation of Carbon in Organic Soils – Sequestration and Emissions) model was used to assess the threat to soil carbon stocks from climate change. ECOSSE was chosen for this task because a) it simulates changes in soil C and N in response to climate in highly organic as well as mineral soils and b) it allows simulations of soil C and N turnover using only the limited meteorological, land use and soil data that is available at the national scale.

In order to estimate the effects of climate change on soil C stocks ECOSSE was run twice: once with climate change and once without climate change. Land use was held fixed for both scenarios. The differences in predicted C stocks between the two simulations were used to infer the climate change induced impacts on soil C content. The climate change simulation utilised predicted temperature and precipitation values derived from the HadRM3 model for the simulation period 2010-2100. The no-climate change simulation used repeated long term average temperature and precipitation calculated from the HadRM3 output for 1980-2009. In both simulations NPP was estimated from the temperature and precipitation data used for the simulation.

The distribution of soil C stocks at the start of the simulation followed the expected pattern with high C contents in peat areas such as the Pennines, the Somerset Levels and The Fens. It is important to be aware that these C stock values are not intended to represent the actual C stock at the start of the simulation, i.e. in 2010, since these data are not available. Instead, they provide a recent, empirically derived start point from which to estimate proportional changes in C content resulting from climate change. The overall projected changes in total soil C content for England and Wales are -0.24 % by 2020, -0.13 % by 2050 and -1.4% by 2080. These equate to changes of -0.02 %, -0.003% and -0.02% yr^{-1} respectively, which are comparable to those reported for England and Wales between 1978 and 2003 by Smith et al. (2007) using the Roth C soil carbon model with changes in climate and NPP included.

The simulated annual rates of change are small compared to the observed +0.4 % yr^{-1} between 1978 and 1998 and -0.72 % yr^{-1} between 1998 and 2007 in Great Britain reported by the Countryside Survey (Emmett et al. 2007) and the -0.6 % yr^{-1} between 1978 and 2003 for England and Wales reported by Bellamy et al (2005). The observed rates of change include the effects of drivers other than climate, such as changes in land use and management. As discussed by Smith et al. (2007), changes in climate can only be responsible for 10-20% of the observed changes in soil C content between 1978 and 2003.

The results from the ECOSSE simulations suggest that climate change, based on the HadRM3Q0 ensemble run which assumes a medium emission scenario, will only be responsible for relatively small changes in soil C content in England and Wales over the period 2010 to 2080. Based on the recent observed rates of soil C change it is likely that other drivers (e.g. land use/management) have a greater potential to affect soil C stocks than climate change.

Water erosion

Projected average water erosion rates estimated by PESERA and driven by HadRM3 climate variables are presented for the five decades of interest. Erosion is projected to increase through the coming century by 0.1 tonnes ha^{-1} yr^{-1} to an average of 0.55 tonnes ha^{-1} yr^{-1}. Localised areas of arable and pasture areas can be identified as high risk and require further consideration with finer detail of local management practice. Since erosion rates vary seasonally and spatially, the overall mean is not a good indicator of active response/mitigation. However, projected erosion rates are summarised for each decade and major land use in Table 2 and Table 3 respectively. The projected proportional increase in erosion between 2000 and 2090 is greatest for arable land.

Table 2. Estimated average erosion rate per decade (tonnes ha^{-1} yr^{-1})

Decade	mean	std dev	max
2000	0.44	0.82	17.1
2020	0.45	0.87	16.9
2030	0.45	0.88	17.7
2050	0.46	0.85	16.8
2090	0.55	1.01	23.4

Table 3. Estimated average erosion rates by major land use class (tonnes ha^{-1} yr^{-1})

Decade	Land use	area (km^2)	mean	std dev
2000	arable	45584	0.3	1.11
	pasture	73523	0.38	0.97
	forest/moorland	12885	0.53	1.19
2090	arable	45584	0.4	1.62
	pasture	73523	0.47	1.12
	forest/moorland	12885	0.57	1.36

Considering seasonal changes, erosion in autumn and winter is significantly influenced by changes in rainfall while erosion in summer may be more influenced by the cropping calendar and/or reduced cover due to limited water availability (AET / PET >> 1)

The coefficient of variation of rain per rain day for the present day HadRM3 data appear to be higher than estimated from direct measurements. This may drive a higher rate of projected erosion than that actually observed. It could therefore be concluded that the rate of change of erosion (both annual and seasonal) considered as a change in erosion risk are more reliable/realistic output than considering absolute values**.** Baseline conditions derived from observed rainfall network would improve confidence in results.

Although average estimates tend towards an erosion rate not dissimilar to the present, areas of both arable and pasture can be identified as being at high erosion risk and require further consideration with finer detail of local management practice and cropping calendar. As erosion rates vary seasonally and spatially the mean value is a poor indicator of active response/mitigation.

The statistical properties of the key climate drivers are at variance with those of measured data. This means we are reluctant to draw conclusions about potential changes in water erosion.

Wind erosion

Monthly values of wind erosion potential were calculated using the soil factor data and monthly weather factor data based on the RWEQ equation. Monthly values were calculated in each of five years spanning five-year periods: 2000-2004, 2020-2024, 2030-2034, 2050-2054 and 2090-2094. The calculations were originally made on the 1 km soil grid and subsequently upscaled to the 5km grid by taking the mean of the finer resolution data. In local areas where soils have organic matter contents both above and below 5%, estimates of wind erosion potential are only available for the latter, so the mean values are biased towards the soils with smaller organic matter contents.

There is no overall trend in projected wind erosion potential between 2000 and 2090 by month; the time-steps of monthly erosion potential values show considerable variation by month. It is useful to consider in more detail the months of September and February where wind erosion is likely to be most severe because soil is exposed on arable land before winter and spring crops, respectively, are sown. The overall distribution of mean projected wind erosion potential for September shows a small decline between 2000-2004 and 2090-2094. In the case of mean monthly wind erosion potential in February, there is no clear pattern of change; the period with the smallest projected mean values is 2050-2054.

The most severe wind erosion events are caused by strong winds blowing across dry, bare soil surfaces and so it is also important to consider the maximum monthly wind erosion potential for the same periods. In common with the distributions for projected mean wind erosion potential, there is no clear pattern of change through time from 2000-2004 and 2090-2094. The impact of variations in monthly climate has a greater impact in wind erosion potential than the variation over the periods of climate projections. The months of September and February when arable soils are most susceptible to erosion show the pattern as that observed for the mean figures; a small decline over the projected time periods during September and large variations in February but no clear temporal pattern with time.

There are two main types of uncertainty and limitation associated with the predictions of wind erosion potential: i) the uncertainties inherent in the climate variable predictions and, ii) the limitation of the RWEQ model. We deal with these separately below.

Much of the uncertainty associated with the climate variable data has been discussed in earlier sections of this report. While soil moisture content has a minor influence on wind erosion, wind speed is the dominant driver. In our modelling we therefore focus on climate projection of wind speed to estimate future wind erosion. No wind speed projections were provided as part of the UKCIP09 probabilistic climate projections because they were considered to be subject to too much uncertainty (UKCIP09, 2010). There is very poor agreement between different global and regional climate models on wind speed prediction. In a study of prediction of extreme near-surface wind speeds, Rockel and Woth (2007) concluded that, "an ensemble of models is essential in assessing the future change in extreme wind speeds: only in a few regions and for a few months did all models even agree on the sign of the change". This is clearly of great significance for estimating wind erosion potential and raises important questions over the confidence that should be placed on the modelled outputs presented above. We can state that there was no discernible change in the predicted decadal wind factor values between 2000-2004 and 2090-2094. An analysis of UKCIP wind speed projections for 2080 have suggested a small (0.5%) overall increase in annual mean wind speeds averaged across the UK, but with more significant seasonal trends of 15% increases in winter wind speed in southern England (Harrison et al., 2008). If climate change does lead to significant changes in wind speeds, then the RWEQ model outputs cannot be relied upon. However, if we assume there is no substantial change in wind speeds over the period considered here, the RWEQ model can still inform us whether projected changes in soil moisture (rainfall and evapotranspiration) will affect wind erosion potential.

In a previous report assessing wind erosion potential across England and Wales (Defra, 2006), a preliminary assessment of the wind erosion potential across England and Wales had been undertaken using climate data for the period between 1961 and 1990. As we might expect the overall patterns of wind erosion potential reported in that study and the current study are similar. The previous study compared the RWEQ wind factor for the period 1961-1990 and climate projection data for 2050 using HadRM3 with the SRES A2 emissions scenario. The authors concluded that their preliminary analysis did not indicate an increase in wind erosion potential over this period, there was some evidence for a small decline in the wind factor. Wind speeds across England and

Wales are known to be seasonal; the highest wind speeds occur during the winter. An analysis of UKCIP wind speed projections for 2080 have suggested a small (0.5%) overall increase in annual mean wind speeds averaged across the UK, but with more significant seasonal trends of 15% increases in winter wind speed in southern England (Harrison et al., 2008). On the basis of HadRM3 climate projections and assuming no substantial changes in land use, **we conclude that there is unlikely to be a substantial change in the magnitude of wind erosion of mineral soils (<5% organic matter content) across England and Wales.** If land use patterns change towards increasing the proportion of arable land, the area where wind erosion may increase substantially is the Cheshire plain and some parts of the West Midlands where land use is currently grassland.

Phosphorus

Following acquisition of required data (see Appendix C) the PSYCHIC model was run for three scenarios, using climate realisations for 2000, 2030 and 2050 from the HadRM3 model. With the exception of climate, all other inputs to the PSYCHIC model were constant across each of the simulations. Using constant values of land use, land management and soil condition in the scenario runs allows the direct effects of the changes in climate to be evaluated. However, it should be noted that climatic change may also cause indirect effects on the other inputs that will not be accounted for.

Phosphorus loads from agriculture to rivers for the 2000 climate ranged from under 1 kg P per km^2 to over 340 kg P per km^2. The total national loss of P from agriculture loss was 5.3 kt P per year, the highest losses being in areas of high rainfall resulting in high mobilisation rates or where there is underdrainage (commonly located on heaver soils) resulting in high levels of delivery. Hotspots can also be seen in intensive agricultural areas where manure applications are high (typically associated with extensive pig and poultry farming), such as in Nottinghamshire, Suffolk and Norfolk.

Between the 2000s and the 2030s, projected annual losses decline across 80% of England and Wales. The spatial distribution of these changes is approximately in accord with the projected annual mean rainfall changes. Where rainfall increases, the majority of this additional rainfall ends up as drainage, and thus there is a comparable increase in P loss. However, where rainfall decreases (and particularly where rainfall is already low), there will be a larger proportional decrease in drainage, as the proportional change in rainfall minus evaporative demands will be greater. Despite projected maximum reductions in P loss per km^2 of over 20%, the net national result is only a projected 4% reduction (Table 1, Appendix C). This is a result of the greatest percentage reductions in projected losses generally being in areas where the magnitudes of P losses are smallest. By the 2050s, losses are simulated to have decreased by a further 16%, although the similarity to the changes in rainfall distribution is less apparent. These reductions in P loss are a result of the combined effect of differences in projected rainfall with differences in the responsiveness of different soil types to P loss. For example, the greatest reductions in projected rainfall occur in areas such as the chalk catchments of the Hampshire Avon, but projected reductions in P loss are not as extreme here due to there being less underdrained soils than in other parts of the country (Figure 1, Appendix C).

The major cause of the reduction in predicted P losses to rivers from agriculture is the reduction in rainfall predicted in the climatic conditions in the 2030s and 2050s. Any temperature increases would be expected to amplify the reduced rainfall effect, with increasing evapotranspiration and hence higher soil moisture deficits leading to reduced surface runoff and drainage.

The projected changes in annual P loss and sediment are summarised in Table 4. Projected reductions in particulate P are directly proportional to reductions in the sediment load. Simulated decreases in sediment and associated particulate P show greater reductions by 2050 than 2030, compared to dissolved P. This may be attributed to both changes in the spatial distribution of rainfall (predicted annual rainfall in areas most susceptible to soil erosion such as the Pennines, fall in 2050 relative to 2030) as well as projected changes in the temporal distribution of rainfall (through complex interactions between temperature and rainfall, and thus drainage, and how this corresponds to the timings of manure applications and amount of crop cover). However, as dissolved P accounts for around 60% of the total P entering the river in all 3 climate scenarios, the change in projected total P is more closely aligned to that of dissolved P.

Table 4. Changes in projected annual phosphorus and sediment losses delivered to water courses for the three climate scenarios. Brackets indicate the percentage change from the loss value in climate year 2000.

Year	Phosphorus Loss (kT P yr^{-1})			Sediment Loss (kT yr^{-1})
	Total P	Dissolved P	Particulate P	
2000	5.3	3.1	2.2	1850
2030	5.1(-4%)	2.9(-6%)	2.2(-2%)	1810 (-2%)
2050	4.4(-16%)	2.6(-14%)	1.8(-19%)	1510 (-19%)

The results presented in this report indicate that under the assumed climate projections for 2030 and 2050, national annual losses of phosphorus from agriculture would decline by 4% and 16% respectively. This is a direct consequence of national projected reductions in rainfall by 2030 and 2050, and the consequent reductions in both drainage and surface runoff, particularly the latter as it is the main mechanism by which P and sediment is exported from most catchments. Locally, projected reductions of P reached 30%, associated with simulated changes in rainfall of up to 20%. The impact of the change in rainfall would be further amplified by projected increases in temperature and changes in wind speed and solar radiation, which would affect evapotranspiration and hence the soil moisture deficit. Projected changes in the number of rain days by 2030 and 2050 would also lead to differences in the mobilisation and delivery of sediment and associated particulate P loss, due to changes in the assumed rainfall intensity and the resulting erosive potential and volume of surface runoff generated.

With the exception of climate, all other inputs to the PSYCHIC model were held constant across each of the simulations. However, there are likely to be significant changes in both land use and land management and changes in livestock numbers and their diets. The uptake of mitigation methods will increase, due to requirements for improved water quality, reducing greenhouse gas emissions, increasing biodiversity etc. The erosive potential of rainfall may change over the coming decades and there may also be changes in the soil P status.

Under the climate realisations used in the simulations, the magnitude of P loss from agriculture was seen to decline in the 2030's and 2050's comparative to the 2000's, with little change in the relative proportions of dissolved and particulate fractions. However, concentrations of P in drainage from agricultural land do not change significantly, and concentration is the key driver for water quality. If inputs from sewage treatment works and other sources were to remain constant into the future, then the relative contribution from agriculture to phosphorus pollution pressures would decrease.

Acidification

One source of soil contamination which has been of concern in recent decades is acidification due to atmospheric deposition of oxides of sulphur and nitrogen. The mechanisms involved are described by Reuss *et al*. (1987). Following reductions in emissions of these oxides to the atmosphere there is widespread evidence of the soil and fresh water recovery predicted by modelling, although this recovery is generally still in progress (Evans et *al*., 2001). Temperature effects are not explicitly included in the standard version of VSD, the chosen acidification threat model, and it has been extended to allow temperature-dependence of weathering. The method used follows Posch et al. (2002), providing the functional form of a temperature dependent multiplicative term to modify weathering rates, and we have included this in VSD. The cation exchange processes, aluminium and bicarbonate equilibrium and nitrogen cycle processes may also be temperature dependent, but this has not been accounted for in the modelling. These should be considered in a fuller modelling exercise. However, there are major approximations to soil processes whose uncertainty in field application has not been fully understood. This needs to be taken into consideration in assessing the value of including a climate change response in the model. The model is commonly run with equilibrium runoff. We have allowed the runoff to vary annually following the HadRM3 projections of rainfall, adjusted for estimated evaporation. Posch also adjusts NPP, which influences the N cycle component of VSD. We have not made this adjustment.

The results we present are for changes in soil pH. A more refined analysis would consider UK changes in critical load in response to climate change. The acidification status of soils in England and Wales is not at equilibrium but is recovering from past high atmospheric inputs. Further recovery from acidification is anticipated regardless of climate change. The VSD model has therefore also been run with drivers which assume no change in climate and standard emission scenarios. The pH differences computed between these two scenarios are presented, and show small differences in pH, both negative (slower recovery from acidification) and positive. In general, there is faster projected recovery under climate change in southern and eastern England, and slower projected recovery in Wales. Projected recovery in the Pennines is variable. The suggested changes in the rate of recovery of soil pH given the projected climate change data and the assumed process response are not major. In terms of mechanisms, the temperature effect is to increase base cation weathering, which has a net effect of raising the projections of pH generated by VSD. The effect of changes in the water budget is to reduce pH through increased evaporation.

The conclusion is **that recovery from acidification is not likely to be significantly affected by climate change.** The minor effect of projected increased weathering in raising pH is offset by projected increased evaporation in raising ionic concentrations generally, including the protons responsible for lowering pH. However, in our application of VSD the inclusion of temperature dependence of processes has been limited to weathering. In addition, the VSD model is under continual development and the relative importance and parameters of some processes may not be well-represented.

Compaction

Typically in England and Wales, compaction is a risk when agricultural machinery is taken onto the land when the soil is too wet. The Workable Days model works with a water content associated with a threshold soil moisture potential to define a “workable day”. We have used the water content when the soil moisture potential is -10kPa, a value which is estimated from knowledge of the shape of the water release curve of the soil. This has been determined using the van Genuchten parameters of the each soil class, obtained from the LandIS database. The daily soil moisture estimates for four soil layers (0.1, 0.25, 0.65 and 2.0m) were obtained by running the JULES model, driven by the HadRM3 data. Average annual workable days across all of England and Wales for a given decade were estimated to increase in future decades (Table 5). By 2090, there was a predicted 10% increase in topsoil and 8% increase in subsoil workable days compared to 2000. These predictions were done separately for topsoil and subsoil, so they do not account for days when one depth may be workable but the other depth is not workable. In practice, both depths would need to be workable to avoid damage from soil compaction. Regional changes (correspond to the annual precipitation data, where the north of England is shown to have increased precipitation in 2020-2039 and the south of England and east Wales have the greatest reduction in precipitation in 2050-2059.

Table 5. Estimated average annual Workable Days per decade

Decade	Topsoil (0-10 cm)			Subsoil (10-35 cm)		
	mean	std dev	min	mean	std dev	min
2000	252	64	27	262	70	10
2020	265	62	27	274	68	9
2030	267	64	30	274	70	4
2050	283	62	41	290	68	7
2090	278	56	54	282	62	22

Changes are seasonal, with parts of northern England predicted to have fewer Autumn Workable Days in 2030 and Spring Workable Days in 2020 (Figure C4 and C5). By 2050 these differences diminish. Wales and southwest England, the regions with the fewest Workable Days, have increases in workable Days in 2090-2099 limited to Autumn and Summer, with some areas still having very few winter workable days. The current seasonal and localised risk of soil compaction in these regions will remain into the future.

The conclusion is that under the projected climate change scenarios used, there would be a small increase in the number of workable days generally, due to lower soil water content due to increased evaporation at higher temperatures. In terms of soil compaction, this suggests decreased risk of machinery on land during wet conditions when compaction may occur. However, limitations in modelling soil water dynamics were apparent from the greater number of subsoil workable days predicted.

Salinity

While we have carried out no formal investigation, salinisation associated with irrigation, excess evaporation and insufficient drainage is not thought to be a risk in England and Wales over the coming century. There is sufficient winter rainfall, combined with low evaporation, to ensure that any possible accumulation of salt during summer months is washed through as drainage. This situation is thought unlikely to change. However, occasional coastal inundation by storm surges is considered a risk, although in restricted areas. Such inundation will result in a degree of salinisation, the severity of which will depend on the soil itself and the time and depth of inundation. We have modelled this inundation process.

We have identified seven low–lying coastal areas where inundation is a risk, with a single 5km square being identified within each region for modelling. Estimated storm surge water levels provided by the Met Office indicated that three sites out of the seven were at higher risk. Subsequently, these three sites were considered in this study. They are 5km squares in the Parrett estuary, the Fens and the Norfolk Broads.

The elevations of these sites were obtained from the IHDTM. Heights are interpolated between high water mark (for which an elevation of 3m has been assumed) and the contour lines on OS 1:50,000 maps, which are at a vertical interval of 10m and are therefore only approximate. The elevations in the Wash and Norfolk Broads are less than 3m because in these areas there is a 0 m contour.

The risk of inundation at the sites has been based on Met Office estimates of the distribution of storm surges at the identified risk locations. The surges were estimated from an ensemble of the Hadley Centre model HadCM3 (Gordon et al. 2000), by perturbing the physical parameters within the model (the ensemble also contained the unperturbed member). These global models are too coarse to provide the necessary boundary conditions for the surge model, so it was down-scaled with the regional climate model HadRM3, in such as way as to maintain a consistent regional ensemble. This provided the wind and surface pressure fields required to drive the surge model POLCS3 (http://www.pol.ac.uk/ntslf/model.html, used operationally at the Met Office). The surge model was run for the complete 150 years (from 1950 to 2100). From this time series of surface elevation was extracted for the seven locations specified. The scenario is that of the unperturbed member of the ensemble, under the A1B scenario. The exact offset that used for the elevation of each site is expected to introduce uncertainty into the system.

The model has been run assuming a breach at the start of each decade, without any repair being made. This means any subsequent surge of sufficient height will inundate through the breach. For two of the sites, the Parrett and the Fens, this means inundation is so frequent that there is effectively no recovery from salinisation, through washing through with rainwater infiltration. At the Parrett estuary during the first month of 2020, the salinity in the top 2m reached over 40 dS/m, by March 2020 the salinity front reached 4m and by November the salinity of the 10 meters soil profile was close to 45dS/m. Salinity stayed close to that level over the decade as rainfall was not sufficient to leach out the salt from the deeper layers. A similar trend was seen for 2030-2039, 2050-2059, 2090-2099 but with earlier dates, i.e. soil salinity progressed relatively faster with time from 2020 to 2099. This reflects the higher storm surges during the later decades.

In contrast to the Parrett and the Fens, the site on the Norfolk Broads would be less susceptible to salinity under projected climate since inundation would be less frequent. Modelling showed periods of recovery as salt was washed from the soil profile by rainfall. By the end of each decade the modelled salinity profile did not show complete replacement of fresh by salt water. The precise pattern of salinity down the profile depends largely on the previous history of rainfall and inundation, partly influenced by crop cover. Under the same atmospheric inputs and for the same soil conditions there is some projected difference in salinity between different crops. This difference is believed to be a reflection of different water uptake patterns.

The conclusion is that the risk of inundation would be greater later in the century due to higher projected storm surges. Saline water entering the soil during short-term inundation could be diluted to a lower and possibly un-harmful salinity level to soil and plants if followed by a significant rainfall shortly after. However, this is very much dependant on initial soil salinity, duration and intensity of the rainfall that follows the short-term inundation.

Discussion

The objectives of Workpackages 1 and 2 were achieved as reported in full in Appendix A. Workpackages 3 to 5 objectives (Appendices B and C) were not fully achieved because of the unsuitability of the UKCIP09 data for the national scale modelling of the sort envisaged. The UKCIP09 data would be the preferred drivers had they been suitable. The work-around using a single member of the HadRM3 ensemble, while allowing modelling to progress, has meant using data which do not fully account for bias or variability. In this respect they are inferior to the UKCIP09 data. Nevertheless, all models selected were run satisfactorily using the HadRM3 data as drivers, all the necessary parameters having been sourced. In any future modelling work, it is assumed that access to the UKCIP09 data would be resolved, and research would progress using these. Insofar as this is the case, any inferences made using the HadRM3 work-around would be superseded. In anticipation of this, it was not though useful to go to great lengths to generate the best possible derived dataset from HadRM3, but rather ensure that the modelling process at the daily 5km scale from climate scenario to soil threat model was functioning and in place for replacement of HadRM3 drivers by UKCIP09 drivers.

The projections made are not those which would be made using UKCIP09, and, in particular with those sensitive to rainfall changes (which is most), are likely to deviate. The HadRM3 ensemble member chosen drifts downward mid-decade and this influences many of the projections which depend on the water balance. There is also uncertainty over the best approach to estimating evapotranspiration, which is also a key variable in the hydrological cycle.

The risk of using projections for policy making has recently been highlighted (Nature 2010) and the reservations expressed there apply to our work. The main achievements of the project are:

1. Successful model linkage from climate change scenarios to soil threat models, following downscaling from 25km to 5km for England and Wales;
2. Linked model application to 2100 for 7 selected threats, also including processing by the intermediate land-surface model JULES;
3. Identification of sensitivities of soil threats to changes in climate variables; and
4. Identification of weaknesses in existing models and data

We have not provided a statistical analysis of the projections made because we believe they would be misleading. The uncertainty estimates provided by UKCIP09 would help in providing uncertainty estimates for soil threats. However, even those uncertainty estimates are not standard errors in the true statistical sense, depending as they do on model assumptions in the GCMs. The uncertainty introduced within JULES and the soil threat models is not addressed, and is not a feature of the models themselves. In many cases, statistical validation of the models against field measurements has not been carried out in any systematic way, and is an unknown quantity.

We do not suggest that modelling using the single ensemble HadRM3 member be continued. It would be preferable to await the availability of more suitable UKCIP09 data, and substitute these data through the modelling protocols developed. This would also allow the uncertainty estimates included in UKCIP09 to be included in the modelling.

All references in this report are provided in the associated Appendices.