Climate change and soil carbon declines in England and Wales 1978-2003

Running title: *Climate change and soil carbon decline*

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Summary

It is not yet clear how soils are responding to a warming climate. A major study using the National Soil Inventory (NSI) of England and Wales reported large declines in soil carbon concentration across 11 land uses between 1978 and 2003 and concluded there was a link to climate change. But a second, almost contemporary study, recorded no significant changes, raising the possibility that the reported declines were caused by changes in land use and management rather than climate change. We have used "space for time" substitution on the data from the *initial* NSI study, combined with changes in rainfall and temperature over the survey period, to determine the extent to which the declines in soil carbon observed in the second study could be predicted from changes in climate. For organo-mineral and mineral soils, little of the observed decline in carbon concentration can be predicted from changes in climate; in contrast, 17- 40% of the change reported for organic soils in semi-natural habitats can. We found that carbon concentration in organic soils declines sharply as temperatures exceed ~7.5^{o}C, mirroring independent observations for the decline in bog and dense shrub moor vegetation as temperatures rise above 7^{o}C, and raising the possibility that climate change may influence soil carbon indirectly by changing vegetation cover and hence litter quality. Used with medium emission climate change projections, we estimate that soils in England and Wales could be losing an additional 0.77 – 2.18 Tg of carbon annually in response to climate change.

Introduction

Soils and vegetation store ~5% of global carbon but they contribute ~50% of the carbon dioxide flux to the atmosphere (IPCC, 1991; 2007). It is not yet clear how they are responding to a warming climate. The processes that determine soil carbon concentration – losses via soil respiration and gains through plant returns – are both expected to increase with temperature (Smith *et al.*, 2008; Smith & Fang, 2010); and for respiration, recent evidence appears to confirm an increase over time that is related to air temperatures (Bond-Lamberty & Thomson, 2010) What is less certain, but of crucial importance, is whether the balance between these opposing processes will change under a warming climate. If the loss of carbon through soil respiration increases more rapidly than carbon returns in plant debris, soils could provide a positive feedback to climate change (Cox *et al.*, 2000).

Measurements of changes in soil carbon concentration over time could, in principle, provide unequivocal evidence on the balance between losses and gains, but they are problematic. Detecting small changes against a large and variable background is challenging; disentangling land use effects from those due to climate change even more so. A major study using the National Soil Inventory of England and Wales (NSI) reported large declines in soil carbon between the periods 1978 – 83 and 1995 – 2003, called here the first and second NSI surveys (Bellamy *et al.*, 2005). The fact that the changes were irrespective of land use led the authors to conclude that there was a link to climate change. But data from the Countryside Survey (CS), collected over a similar period, showed no significant change in soil carbon concentration, suggesting that the declines in the NSI study were unrelated to climate change (Emmet *et al.*, 2010). More recently, two studies focussing primarily on the mineral soils in the NSI study, concluded that at most, 10-20% of the observed changes could be attributable to climate change. Reductions in cattle stocking densities on grazed land and the movement to new equilibrium conditions in arable land, were cited as more plausible mechanisms (Smith *et al.*, 2007; Kirk & Bellamy, 2010). It should be noted that 80% of the loss reported in the NSI study was for organic soils, a soil type not normally used for intensive cattle production in the UK. Other studies have reported no significant change in soil carbon over time (Kirby *et al.*, 2005; Tomlinson, 2005; Tomlinson & Milne, 2006); while studies in the Netherlands and Belgium concluded that the observed changes were explicable using changes in land use or management without the need to invoke climate change (Goidts & Van Wesemael, 2007; Reijneveld *et al.*, 2009). Indeed Bell *et al.*, (2012) have shown for the UK, that although an individual field under constant land use would have experienced declines in soil carbon concentration over the period, the pattern of land use change across the whole UK would have resulted in an increase in soil carbon nationally.

One alternative to studying soil carbon concentration over time is to use "space-for-time substitution" in which changes in carbon across climatic gradients at one time, are used to predict carbon under a future climate (Pickett, 1989; Fantappie *et al.*, 2011).

We have used the technique of "space for time substitution" on the soil carbon concentration from the *first* NSI survey (1978-83) to derive regression models between carbon concentration and mean annual temperature and rainfall for each of the 11 land uses reported by Bellamy *et al.*, (2005). The regression models were combined with the average temperatures and rainfall for organic or organo-mineral/mineral sites for each land use at the first and second NSI surveys, to model the change in soil carbon at the second survey that could be predicted *solely from changes in climate*. We have also used agricultural census data between the 1980s and 2000 to estimate the likely contribution to soil carbon declines from changes in animal stocking densities.

There is some evidence that vegetation in semi-natural habitats such as bogs, responds to increases in temperature in three stages. Up to a threshold temperature around 7^{o}C, little change is apparent; between 7 and 9 ^{o}C the probability of bog and dense shrub moor declines sharply, stabilising at values close to zero above 9^{o}C (Hossell *et al.*, 2000). We have examined the possibility that climate-related changes in vegetation cover, and hence substrate quality, could influence soil carbon concentration in semi-natural habitats by using two-stage regression models, and a logistic model (in effect a three stage regression model) similar to that used by Hossell *et al*., (2000).

In addition to predicting the changes in soil carbon between the first and second NSI surveys, we have combined the regressions with climate projections from the UK Climate Impacts Programme (Jenkins *et al*., 2009) to estimate climate-related changes in soil carbon stocks up to 2030.

Materials and Methods

Space for Time Substitution

Space–for-time substitution involves determining regression relationships across gradients at one time to predict future behaviour under conditions when one or more of the covariates has changed (Pickett,1989). It is attractive in that it allows long-term dynamics to be explored without the need for costly long-term experiments. For soil carbon it can also be realistic in that variations in soil carbon that could be related to climate are studied alongside other causal factors (Berg et al., 1993; Gholz et al., 2000; Meier & Leuschner, 2010;Fantappie et al., 2011). Fukami & Wardle concluded that space for time substitution using gradient studies are valuable “not necessarily because they represent typical ecosystems, but because there are few confounding factors that influence ecosystem processes, thus making it easier to infer causal relationships (Fukami & Wardle, 2012).

But there are limitations. Analysing studies in which space-for-time approaches were used to reconstruct vegetation succession chrono-sequences, Pickett concluded that the approach failed where “unrecognised effects in the past of a system were of large magnitude” so that differences in vegetation which actually stemmed from different starting states, were mistakenly interpreted as reflecting different stages in the chronosequence (Pickett, 1989). Because we are using observed data as the starting not the end point of our studies, this is not an issue here. Pickett also observed that the approach was compromised when the effects of other driving variables such as land use or cover, elevation and soil type were not minimised (Pickett, 1989). The NSI data are already stratified by land use; we have further stratified the data into those sites on organic or organo-mineral/mineral soils to reduce the effect of soil type and, indirectly, the influence of elevation (see later).

The reliability of space for time substitution to predict future behaviour depends on the stability of the regression relations over time. If the relation between the covariates disappears over time, the predictions will be unreliable. We have tested the robustness of the regressions based on the first NSI survey by comparing them with similar regressions based on the soil carbon and meteorological data for the second NSI survey, conducted approximately 20 years after the first.

Soil Carbon Data

The carbon concentration data from the first NSI survey (1978-1983), categorised into the same 11 land uses, were used in the regressions. We further stratified the data into those sites on organic soils and those on organo-mineral/mineral soils. This stabilised the variance of soil carbon in the regression analyses, and indirectly reduced the effect of soil type and elevation, one of the main problems in space for time substitutions (Picket, 1989). A carbon concentration of 150 g kg^{-1} was used to delineate organic from organo-mineral/mineral soils (Hodgson, 1997).

The second NSI survey (1995-2003) re-sampled a subset of sites sampled in the first survey (1978-1983). These data were also further stratified into those sites on organic soils and those on organo-mineral/mineral soils.

Changes in soil carbon between the first and second NSI surveys are as reported by Bellamy *et al.*, (2005); we refer to them as the NSI changes. The intervals between the first and second NSI surveys varied across and within the land uses. For this work, the initial survey was taken as centred on 1980. The resurveying of arable sites was centred on 1993; rotational grass sites on 1994; permanent grass on 1996 and for all other land uses on 2003. These intervals are important in estimating the changes in temperature and rainfall between the two surveys.

In addition to the NSI data, two other soil carbon data sets have accurate sample locations that allow space for time substitutions. These were used to check the regressions models derived using the NSI data. The Representative Soil Sampling scheme (RSS) ran from 1969 to 2000, sampling arable, rotational and permanent grass sites. Most samples were taken between 1978 and 1990, a period when mean temperature and rainfall was approximately constant (data not shown). The Woodland survey of 1971 provides soil carbon data for deciduous woodland at 103 identifiable locations (Kirby *et al.*, 2005).

The NSI data from the first survey, and the initial soil carbon concentrations, are summarised in Table 1.

Associating soil carbon concentration and meteorological data to develop regression models

Temperatures in Central England have risen by ~1.0^{o}C since the 1970s (Jenkins *et al.*, 2009). It is unlikely that changes in soil carbon concentration in response to annual changes in mean temperature and rainfall would be detectable, but the time lag over which a consistent change would be become detectable is not well defined. We derived mean temperatures and rainfall for each sample location in the first NSI survey, averaged over either 10 (1971-80) or 30 years (1951-80) prior to the sampling, by mapping each location to 5 x 5 km climate data supplied by the UK Meteorological Office. We also derived mean temperature and rainfall for each sample location, averaged over the period 1961-90, using 1 x 1 km climate data supplied by the UK Meteorological Office. The 1961-90 period is the baseline for climate change scenarios predictions. Regression coefficients derived using the 10 year average climate data were, with one exception, not significantly different from zero and are not reported here. The regression models using the 1951-80 and 1961-90 climate data were not significantly different; for clarity only the 1961-90 models will be presented here.

To estimate changes in soil carbon concentration between the first and second NSI surveys that could be attributed to climate, we combined the 1961-90 regression models with changes in the 30 year averaged rainfall and temperature between the first and second NSI surveys. This was done for each land use (and the subsets of organic or organo-mineral/mineral) in the first NSI survey. Average temperature and rainfall at the first and second NSI surveys are given in Table 1.

TABLE 1 HERE

We also combined the 1961-90 regression models with 25 x 25 km gridded climate predictions for 2010-2030 from UKCP09 (Jenkins *et al.*, 2009) to estimate changes in soil carbon under a warming climate. UKCP09 presents predictions for low, medium and high emission scenarios at probability levels of 10%, 50% and 90%. The 10%

probability level is interpreted as indicating that future change is unlikely to be less than the modelled scenarios; the 90% level indicates that change is unlikely to be more than the modelled scenarios. We have used climate predictions for the medium emission scenario for the period 2010-2030; predictions beyond 2030 yield temperatures outside the data used to derive the predictive regressions. All spatial analyses were performed using the Spatial Analyst module of ArcMap 9.3®.

We tested the stability of the regression models over time by deriving a second set of multiple regression models based on the soil carbon data from the second NSI survey combined with meteorological data averaged over the 30 years prior to the second sampling date. For arable, rotational and permanent grassland sites, meteorological data were averaged over the period 1965 – 1994; for other sites the averaging period was 1974 – 2003. These regression models based on data for the second NSI survey were then compared with the models derived from the first NSI survey.

R*egression analysis*

Preliminary analysis of the data sets indicated a small number of outlier data points (Studentised residual >2). These were removed for the main analyses.

We fitted multiple regression models as in equation (1) to all the data sets

$$Y = a + b*T + c*R \qquad (1)$$

Where Y is soil carbon concentration (g kg^{-1}), T and R are mean temperature (^{o}C) and rainfall (mm) respectively, and a, b and c are regression coefficients. Parabolic models (those including T^2 and P^2), or the inclusion of temperature-rainfall interaction provided no additional predictive power compared to a multiple regression on temperature and rainfall.

The NSI survey data was based on a 5 x 5 km regular grid. Ordinary Least Squares (OLS) to estimate regression coefficients is not appropriate for such sample designs as they assume the residuals are independent random variables and fail to account for any spatial correlation between samples (Lark & Cullis, 2004). We fitted multiple regression models based on equation (1) to all the data sets from the NSI first survey using Residual Maximum Likelihood (REML). For each data set we compared models with spherical and exponential spatial covariance error structures with models with no spatial structure. All REML analyses were carried out using the R package (Pinheiro & Bates, 2000).

Two-stage regression models

Equation (1) may not be the most appropriate model for organic soils where carbon concentrations are high. Carbon loss from high organic matter soils is not manifested by a reduction in carbon concentration because such soils have negligible mineral contents, and carbon loss results in a reduction in peat thickness rather than a decline in soil carbon concentration. There is no consensus on the carbon concentration above which dilution by a mineral fraction is negligible. Lucas (Lucas, 1982) quotes 520 g kg^{-1} as a typical carbon concentration for oligotrophic peats; Gorham (Gorham, 1991) uses a similar figure. Turunen (Turunen, 2008) estimated an average carbon concentration of 503 g kg^{-1} using data on 3670 Finnish peats. Lindsay (Lindsay, 2010) estimated an average carbon concentration for peats of 520 g kg^{-1}.

Preliminary analyses indicated that soil carbon in organic soils was negatively correlated with temperature but not correlated with rainfall. Accordingly, for the organic soils in the NSI data, in addition to models based on equation (1), we also fitted two-stage regression models to reflect the observation that at high soil carbon concentrations (i.e. low temperatures), dependence on temperature may not be apparent; such dependence only becoming apparent at higher temperatures (i.e. lower carbon concentration). Two stage regressions based on equations 1(a) and 1(b) were fitted using maximum likelihood estimation in the segmented package in R (Muggeo, 2003).

$$Y = a; \quad T \leq T_c \qquad (1a)$$

$$Y = a + b*(T - T_c) \quad T > T_c \qquad (1b)$$

Equations (1a) and (1b) imply that at temperatures below T_c, soil carbon concentration is constant; above T_c, soil carbon is related to temperature.

Logistic model

To test the hypothesis that organic soils in semi-natural habitats show similar temperature dependence to that observed for vegetation in such habitats, we also fitted a four parameter logistic model similar to that used by Hossell *et al*., (2000).

$$Y = Y_{\min} + \frac{(Y_{\max} - Y_{\min})}{1 + \left(\frac{T}{M}\right)^s} \qquad (2)$$

Where M is the mid point of the slope defined by s, and Y_{max} and Y_{min} are the maximum and minimum asymptotic soil carbon densities. Equation (2) was fitted to the pooled data for the four semi-natural land uses (rough grazing, upland grass, upland heath and bog) as these are likely to be the land uses in the study by Hossell *et al*., (2000).

As the data from the RSSS and the woodland survey were not based on a regular grid design, the regression coefficients in equation (1) were estimated using OLS.

Assessing model performance

Regression coefficients were tested for significance against the null hypothesis that they were zero; P values greater than 0.05 are reported as not significant.

The significance of spatial dependence in the NSI data sets was assessed by likelihood-ratio tests on models based on equation (1) with either spatial or exponential spatial error structures, compared to a null model with no spatial component (Pinheiro & Bates, 2000). Where spatial models were significantly different from the corresponding null model, model performance was assessed using Aikake information criterion (AIC) –the model with the lowest AIC considered the "best fit" – the range over which the spatial structure operated, and the proportion of the unexplained variation which was spatially structured. Adjusted r^2 was calculated for each model to give an estimate of the variability explained.

The performance of piecewise regression models, which were fitted using maximum likelihood, and the OLS regression models applied to the RSSS and woodland survey data were assessed using adjusted r^2.

Quantifying uncertainty

Uncertainties in the regression coefficients were propagated into predictions of soil carbon concentration either directly from the regression equations or using Monte Carlo simulations with the initial and final temperature and rainfall. All Monte Carlo simulations were carried out in @Risk®.

Estimates of Q_{10}

Applying Boltzmann's distribution law to the temperature response of biological processes has led to the Q_{10} concept. A Q_{10} of 2 implies the rate of a biologically mediated reaction doubles for a 10°C rise in temperature. It is not possible to derive Q_{10} values directly from the change in soil carbon across a geographical climatic gradient. For this work, Q_{10} was estimated using the single pool model of soil carbon following the approach used by Kirk and Bellamy (2010).

Equation 2 is the single-pool model of soil carbon dynamics most appropriate for the data in this study.

$$\frac{dC_t}{dt} = I - k.C(t) \tag{2}$$

Where C_t is soil carbon at time t, I is the annual rate of carbon input to the soil and k is the first order rate constant for soil carbon loss. The solution to equation (2) for constant input and first order rate constant is given in equation (3) where C_0 is soil carbon at t=0.

$$C_t = (C_o - I/k).e^{-kt} + I/k \tag{3}$$

Equation (3) indicates that as t increases, C_t tends to I/k, the equilibrium soil carbon for those conditions. Thus, for two sites at or near equilibrium (indicated by subscripts 1 and 2), which started with the same carbon concentration but were subject to different average temperatures and hence different rates of input and decomposition:

$$\frac{k_2}{k_1} = \frac{C_1}{C_2} \cdot \frac{I_2}{I_1} \qquad (4)$$

If $I_2 \approx I_1$

$$\frac{C_1}{C_2} \approx \frac{k_2}{k_1} \qquad (5)$$

Q_{10} is calculated as:

$$Q_{10} = \left(\frac{k_2}{k_1}\right)^{(10/\Delta T)} \qquad (6)$$

Where k_2 and k_1 are first order decomposition rates at two temperatures differing by Δt. Substituting equation 6 with equation (5) allows Q_{10} to be estimated directly from soil carbon concentration at two temperatures differing by ΔT as in equation (7).

$$Q_{10} = \left(\frac{C_1}{C_2}\right)^{(10/\Delta T)} \qquad (7)$$

The assumption that input rates do not change will tend to underestimate Q_{10} for decomposition. If the soils are not at equilibrium with respect to carbon, this approach will overestimate Q_{10} for soils rising to a new equilibrium and underestimate Q_{10} for soils declining to a new equilibrium soil carbon.

Estimating the contribution of changes in stocking density

Two papers analysing the original NSI results, speculated that much of the reported change in soil carbon concentration in grazed land on mineral soils might be attributable to changes in carbon returns resulting from changes in cattle stocking density (Smith *et al.*, 2007; Kirk & Bellamy, 2010). Equation (2) provides a simple estimate of the effect of reduced carbon inputs on equilibrium soil carbon (=I/k). Assuming k is constant, a reduction in carbon returns, I, translates into a proportionate reduction in equilibrium soil carbon. Between 1985 and 1994 (approximately the sampling period for rotational and permanent grass) cattle numbers (dairy, beef and calves) in England fell by 19% while sheep numbers (sheep and lambs) rose by 22% (Defra, 2009). Combined with estimated annual carbon returns per animal of 560 kg C for dairy, 505 kg C for beef, 100 kg C for calves, 60 kg C for sheep and 26 kg C for lambs (ADAS, 2011) this implies an average reduction in carbon returns of ~ 7% for permanent and rotational grass. For rough grazing, the calculation was performed for the period 1985 -2003 (the date of the second survey) and excluded dairy cattle which are unlikely to be present on rough grazing. This gave an estimated reduction in carbon returns of ~15%.

Results

Regression modelling

Of the 14 data sets in the first NSI survey (8 organo-mineral and 6 organic) for which a significant multiple linear regression was obtained, 5 showed no significant difference between spatial and null models, and for the remainder, up to 3% of the unexplained variation was spatially structured. Logistic models based on equation (2) including a spatial component were not significantly different from the corresponding null spatial model. Following Bellamy et al., (2005), we conclude that the inclusion of a spatial component in the regression models is unjustified.

Table 2 summarises the regression coefficients obtained for MLR applied to the organo-mineral/mineral soils in the NSI data, the 3 land uses in the RSS data set and the woodland survey 1971 data. Figures in brackets are the standard errors of the regression coefficients. Table 3 presents the regression coefficients for the two-stage regression models applied to the organic soils in the NSI data.

TABLES 2 AND 3 HERE

The regression coefficients derived from the RSS data for arable and permanent grass are within one standard error of those derived using the NSI data. The model for rotational grass using the RSS data differs from that based on the NSI data; the RSS model shows a significant negative correlation with temperature and explains 17% of the variability, while the NSI data shows no correlation with temperature and the model explains very little of the variability. Neither the NSI or the woodland survey regression models for deciduous woodland are significant. Overall, the models agree with other work indicating that in general, soil carbon concentration decreases with increasing temperature and increases with rainfall (Burke *et al.*, 1989; Miller *et al.*, 2004; Dai & Huang, 2006).

There is a clear separation between sites on organo-mineral/mineral and organic soils. Carbon concentration in organo-mineral/mineral soils is mainly weakly positively correlated with rainfall but insensitive to temperature. Only those sites under permanent grass (and rotational grass using RSS data) show a small negative correlation with temperature. Uniquely, carbon concentration in lowland heath sites (all organo-mineral/mineral) is positively correlated with both temperature and rainfall.

Little of the variation in soil carbon in organo-mineral/mineral soils under the 3 agricultural land uses in the NSI survey is explained by climate (Table 2). The apparent insensitivity to climate is plausible; climate-related signals in soil carbon concentration in agricultural soils are more likely to be masked by management intervention and disturbance. The regression models for semi-natural land uses, on both organic and organo-mineral/mineral soils, explain 14-50% of the variation in soil carbon concentration, possibly reflecting lower management intervention and a stronger climate-related signal.

Carbon concentration in all the sites on organic soils (Table 3) is strongly negatively correlated with temperature but not correlated with rainfall; and the regression coefficients for temperature are broadly similar across land uses. For the semi-natural land uses on organic soils, the temperature above which soil carbon concentration is related to temperature (T_c) ranges from 6.1 – 7.1 ^oC. Interestingly organic soils under permanent grass show no threshold temperature.

Our finding that the relation between soil carbon and temperature under some semi-natural habitats is best described by a two-stage regression with a threshold temperature ~ 6 - 7^{0}C, is similar to the relation between the occurrence of bog and dense shrub moor vegetation and temperature (Hossell *et al.,* 2000). Given the similarity in regression coefficients across land use, we pooled the data for carbon concentration from organic soils from the 4 semi-natural land uses most likely to be represented in Hossell's data (rough grazing, upland heath, upland grass and bog) to explore the extent to which the logistic model used by Hossell *et al*., (2000) could also describe the response of soil carbon concentration to temperature.

FIGURES 1 AND 2 HERE

Figure 1, redrawn from Hossell *et al*., (2000), shows a sharp decline in the occurrence of bog-type vegetation as mean annual temperatures rise above ~7^{o}C. Figure 2 shows the pooled data set for organic soils fitted to a logistic model similar to that used by Hossell *et al*., (2000). The dotted lines show the 95% confidence bands for the regression line. The logistic model explains 18% of the observed variation in soil carbon concentration, and suggests that average soil carbon concentration remains constant (~435 g kg^{-1}) at temperatures below ~7^{o}C, declines between 7 and 8.5^{o}C, and then remains approximately constant at 253 g kg^{-1} at temperatures >8.5^{o}C. This pattern is similar to that observed by Hossell *et al*., (2000).

Modelled climate-related changes in soil carbon over the NSI survey period

Figure 3 shows the modelled change in soil carbon concentration for the 11 land uses over the NSI survey period that can be predicted from changes in climate (grey diamonds). The changes reported by Bellamy *et al*., (2005) are shown as circles. The modelled changes are calculated from the regression equations, combined with the 30 year average temperature and rainfall at the beginning and end of the NSI survey period, either directly or from Monte Carlo simulations. For permanent grass, coniferous woodland and rough grazing, the modelled changes are derived from weighted averages for organic and organo-mineral/mineral sites. The error bars are 95% confidence limits.

FIGURE 3 HERE

With the exception of lowland heath, for which the regression models predict an increase in soil carbon in agreement with the NSI result, the regression models predict very little climate-related change for land uses on predominately organo-mineral/mineral soils. For arable soils, the regression models using NSI data, suggest that none of the NSI change can be predicted from changes in temperature and rainfall between the first and second NSI surveys. This agrees with an earlier study which, using a different approach to that here, concluded that the observed decline in carbon concentration is consistent with arable soils, originally under grass, moving to equilibrium, implying that land-use history rather than changes in climate was the main causal factor (Kirk & Bellamy, 2010)

For rotational grass, the regressions predict very small or no climate-related change in soil carbon between the first and second NSI surveys; for permanent grass, 7% of the change reported by Bellamy *et al.,* (2005) can be predicted from changes in climate. The implication is that for agricultural grasslands, most of the change in soil carbon reported by Bellamy *et al*., (2005) results from factors other than changes in climate. Table 4 summarises estimates of the contribution of changes in cattle stocking density to declines in soil carbon (Defra, 2009). For agricultural grassland, 42- 55% of the changes reported by Bellamy *et al.*, (2005) are consistent with changes in livestock numbers. This agrees with the conclusions reached by Smith *et al.*, (2007) and (Kirk & Bellamy, 2010) using a different analysis.

TABLE 4 HERE

For organic soils under rough grazing, upland grass, upland heath and bog, between 17-40% of the changes reported by Bellamy *et al*., (2005) can be predicted from changes in climate over the survey period. For rough grazing, reductions in cattle grazing could account for a further ~ 55% of the reported changes. This is likely to be upper estimates because we assume reductions in stocking density are uniform across organic and organo-mineral/mineral soils, whereas there is some evidence de-stocking was more marked in lowland areas where organo-mineral/mineral soils are more prevalent.

Estimates of Q10

Table 5 summarises the range of Q_{10} estimates for organic soils in each of the 5 land uses where soil carbon changes were related to temperature. For mineral soils under permanent grass, the range of Q_{10} is similar to other reported values (Kirschbaum, 1995). But the values for organic soils are considerably higher and suggest, either the regression analyses are over-estimating the extent of the soil carbon change attributable to temperature, or that the conventional interpretation of Q_{10} is not applicable. It should also be noted that estimates of Q_{10} involving temperatures changes less than 1^{o}C are very sensitive to small differences in the estimated temperature change.

TABLE 5 HERE

Predictions of changes in soil carbon using climate change scenarios from UKCP09

Table 6 presents estimates of soil carbon change derived by combining the 1961-90 regressions with estimates of future temperature and rainfall developed for the medium emission scenario at probability levels of 10% (change unlikely to be less than) to the 90% (change unlikely to be greater than) probability levels. The figures are calculated from the areas of each land use and soil bulk densities calculated using the pedotransfer function from the Countryside Survey 2007 (Emmet *et al.*, 2010). For the semi-natural land uses of upland grass, upland heath and bog, the results refer to organic sites.

The period over which these aggregated losses are calculated is 50 years (1980-2030) but there may be a time lag as soils move to new, lower carbon concentrations, effectively increasing the time over which the losses actually occur. We used the kinetic parameters estimated by Kirk & Bellamy (2010) for these data (Table 2 in Kirk & Bellamy (2010)) in equation (1) to estimate the time over which the changes in soil carbon presented in Table 7 will occur. For arable, rotational and permanent grass, the changes are accomplished in ~2 years. For the organic soils with larger declines, the time required is ~8 years. Note these are not new equilibrium carbon concentrations – reaching these would take between 40 and 100 years. Long-term average temperatures and rainfall changed little over the first ten years of the NSI survey (1980-1990); accordingly we have used a time period of 50 years to average the aggregated carbon losses presented in Table 6.

Table 7 presents estimates of the loss of soil carbon from the 0-15 cm soil layer at the UK scale, expressed as average annual loss in Tg C over the period 1980-2030.

Stability of the regression models over time

Table 8 compares the regression coefficients derived using soil carbon data from the first NSI survey and meteorological data averaged over 1961-90, with models derived using the soil carbon data from the second NSI survey and meteorological data averaged over the preceding 30 year period (either 1965-1994 or 1974-2003). Note the comparison uses MLR for both organo-mineral/mineral and organic soils as the low sample numbers for some land uses in the second NSI survey precludes two-stage regression models.

For organo-mineral/mineral soils, the regression coefficients for rainfall and intercept determined using the two NSI surveys agree to within one standard error for permanent grass, rotational grass, coniferous woodland and rough grazing, though not for arable. For organic soils, the regressions for upland heath, bog and coniferous woodland agree closely between the two surveys, although the regression model for bog based on the second NSI survey data explains less of the variability (17% compared to 33%). The regression models for rough grazing and upland grass based on the second NSI survey do not agree with the models derived from the first NSI survey. It is not clear whether this reflects a real difference or whether differences in the distributions of soil carbon in the second NSI survey are responsible. For both these land uses, the second NSI survey recorded a lower proportion of sites with soil carbon densities > 40 g kg^{-1} compared to the first survey.

Table 9 presents the logistic regressions based on the first and second NSI surveys using soil carbon data for organic soils under rough grazing, upland grass, upland heath and bog. The regression coefficients for C_{max}, C_{min} and the slope coefficient s agree to within one standard error, although the estimate of the slope using the second NSI data is associated with a high standard error. The estimated threshold temperatures, T_c, agree to within two standard errors.

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Discussion

With the exception of lowland heath sites, only a small proportion of the changes reported in the NSI study for sites on organo-mineral/mineral soils can be predicted from changes in climate over the survey period. Changes in land use and stocking density are more plausible explanations for part or all of the observed changes. This agrees with earlier work which came to the same conclusion using different approaches to those used here (Smith *et al.*, 2007; Kirk & Bellamy, 2010).

The regression results for lowland heath stand alone in that a) soil carbon is strongly positively correlated with both rainfall and temperature; and b) they predict a significant increase in soil carbon over the NSI survey period, in agreement with the observed (though statistically not significant) increase of 18.4 g kg^{-1}. Lowland heath is defined as lying below 300 m and usually found on poor, acid, well-drained soils with plant cover comprising heathers and gorses, bracken and Scots Pine and Birch. It encompasses both humid and dry habitats although the sites in the NSI survey appear to be mainly dry lowland heaths. The evidence of the effects of climate change on dry lowland heaths is sparse and contradictory, and it is difficult to separate changes due to management or climate change (Alonso, 2009). There is some evidence that from manipulation experiments that warming does increase productivity at temperature-limited sites (Penuelas *et al.*, 2004) – and it is plausible that increasing rainfall on well-drained sites could increase productivity - but it is less clear how plant species composition might be affected and this could have a marked influence on litter decomposition rates. Increased management and grazing of lowland heaths promotes the invasion of grass species at the expense of dwarf shrubs and mosses (Bonn *et al.*, 2011); but this would be expected to increase litter decomposition rates and thus reduce soil carbon concentration, in contrast to the results obtained in the NSI survey. Other work has suggested that increasing temperature alters the balance between bracken and heather in favour of bracken (Werkman & Callaghan, 2002). Mixtures of bracken and heather litter break down more rapidly than either component individually (Anderson & Hetherington, 1999), again in contrast to the results observed here. Thus, while the agreement between the regression results and those in the NSI study is persuasive, it is difficult to identify a credible mechanism.

The results for organic soils are very different. For all the land uses, carbon concentration is strongly negatively correlated with temperature, with broadly similar regression coefficients possibly indicating a common mechanism. Between 17 and 40% of the changes reported by Bellamy *et al.,* (2005) for land uses on predominately organic soils, are consistent with changes in climate over the NSI survey period, and for rough grazing, reductions in grazing intensity could explain a further ~55%. However, even attributing only part of the observed declines in carbon concentration to climate change implies unrealistically high estimates of Q_{10}. The concept of Q_{10} relates to the temperature response of a single reaction, or a cohort of closely related reactions. Q_{10} is problematic where a change in temperature results in the appearance of a cohort of new, unrelated processes that were absent at lower temperatures. In this case, there must be a plausible mechanism which does not involve a direct temperature effect on the decomposition rate of soil organic matter. We discuss two possible mechanisms below: the drying and oxidation of the upper layers of organic soils; and the influence of climate-related changes in vegetation and its effect on substrate quality.

The higher soil carbon densities in organic soils result, in part, from constraints on decomposition, such as anaerobic conditions and temperature, and in part from substrate quality. Anaerobic conditions constrain or suspend decomposition, rather than promoting the development of recalcitrant fractions of organic matter (Davidson & Janssens, 2006). Increasing temperatures could alleviate some of this constraint, particularly in the upper soil layers, exposing a new cohort of substrates for decomposition (Freeman *et al.*, 2001; Evans *et al.*, 2002). There is evidence that the export of dissolved organic carbon (DOC) has increased in recent years (Worrall *et al.*, 2003). Much of the increase derives from recently fixed plant material and also from the upper soil layers (Worrall *et al.*, 2003; Worrall *et al.*, 2006; Evans *et al.*, 2007); although there is no consensus on the extent to which this reflects increasing organic matter decomposition or rising dissolved organic carbon release as acid deposition declines (Evans *et al.*, 2006; Monteith *et al.*, 2007). Loss of carbon from peat soils also results from increased moorland burning and associated erosion; one study on 4 upland catchments estimated that around 80% of the increase in DOC export was attributable to increases in moorland burning, with changes in temperature and acid deposition accounting for 22-26% (Clutterbuck & Yallop, 2010).

Comment [F1]: This mechanism only works if the DOM has a higher C content than the soil layer it left. DOM has increased and DOM and DOM does have a higher C/N than peat.

A mechanism whereby plant debris becomes more available for decomposition, through the alleviation of anaerobic conditions at the soil surface, could result in apparent temperature sensitivities higher than the conventional Q_{10} approach suggests. However, the regression models for carbon concentration in organic soils indicate no response to rainfall, and it seems unlikely that the changes in surface moisture conditions following a rise in temperature of ~0.4^{0}C would be sufficient to cause marked declines in carbon concentration. We conclude that the data in this study are not consistent with surface warming and drying of organic coils being a significant factor in carbon concentration decline.

The similarity in response of soil carbon concentration to temperature and that observed for vegetation occurrence could suggest that temperature-related changes in vegetation cover, and hence substrate quality, are indirect drivers for soil carbon decline (Figures 1 and 2). Although temperature and rainfall for the organic sites are weakly correlated, there is no correlation between soil carbon and rainfall in organic sites of the 4 semi-natural habitats and we conclude that this is a temperature effect and that rainfall is not a confounding effect.

There is now evidence of long-term changes in the condition of mountain, moorland and heath habitats and, particularly relevant in this context, the expansion of grass species at the expense of moss and shrub-dominated communities (Bonn *et al.*, 2011). There are a number of potential causes, including changes in nitrogen and acid deposition, grazing pressure and changes in moorland management; but rises in temperature are also implicated (Hossell *et al.*, 2000). The similar temperature response of the two independent data sets – that for vegetation cover and that for soil carbon concentration – could suggest that soil carbon is responding indirectly to temperature through changes in vegetation cover and hence litter quality. This is a plausible (indirect) mechanism which could result in larger declines in soil carbon than a conventional Q_{10} approach, acting on the rate of organic matter decomposition would suggest. However, we cannot exclude the possibility that increased grazing pressure, with a consequent increase in more decomposable grasses, could be occurring disproportionately on warmer sites.

There is finally the issue of the extent to which the results for organic soils under semi-natural habitats in the NSI study are supported by the almost contemporaneous data from the Countryside Surveys (CS) of 1978 and 1998 (Emmet *et al.*, 2010). That study concluded there was no significant change in soil carbon across semi-natural habitats

Comparisons across all the semi-natural land uses, for which Bellamy *et al.*, (2005) reported large declines in soil carbon, are problematic because of differences in land use classification. However, both surveys include bog as a land use, but while NSI reported large declines in carbon concentration (~111g kg^{-1}), CS reported no significant change. The analysis here suggests that ~17% of the decline in soil carbon under bogs could be predicted from changes in climate, and it surprising that an effect of that magnitude is not discernible in the CS data, even though the Q_{10} analysis makes it unlikely this is a direct temperature effect on rates of organic matter decomposition. One aspect of the CS results is worth examining. The NSI survey, which dealt with England and Wales, recorded large declines where the initial soil carbon was 290 g kg^{-1} and above; CS data *for England* also recorded a decline in soil carbon between 1978 and 1998 of ~100 g kg^{-1} for soils with carbon contents of 300-600 g kg^{-1}, although the change was not statistically significant. There was a further (non-significant) decline of 24 g kg^{-1} between 1998 and 2007. This raises the possibility that some of the differences between the NSI and CS surveys are due to different land use/habitat classifications and differences in statistical power.

Future behaviour of soil carbon under a warming climate

The estimates in Tables 6 and 7 suggest that climate-related loss of soil carbon from organic soils in England and Wales over the period 2010 – 2030/40 is between 1.02 and 2.18 Tg yr^{-1}. If the contribution from rough grazing and upland grass are excluded, reflecting the uncertainty in the stability over time of the regression models, the estimated loss of soil carbon is 0.77 – 1.57 Tg yr^{-1}. This compares to the UK's commitment under the Climate Change Act of 2008 to reduce carbon emissions by 3 Tg C annually (Ostle *et al.*, 2009). These estimates should be viewed against others suggesting that land uses changes over the UK over the period 1925 – 2007 have increased soil carbon stocks by around 1.9 Tg C yr^{-1} annually (Bell *et al.*, 2012).

Conclusions

The regression models based on space for time substitution allow the changes in soil carbon concentration reported by Bellamy *et al.*, (2005) to be explicitly partitioned into that which could be predicted from climate change, and that which probably results from other factors. For those land uses predominately on organo-mineral/mineral soils, we conclude that, with the exception of lowland heath, little of the change reported by Bellamy *et al*., (2005) is consistent with changes in climate over the survey period, and that for agricultural soils, the reported changes are more plausibly linked to changes in land use and reductions in carbon returns from grazing animals. For organic soils under semi-natural habitats, we estimate that at most, between 17 and 40% of the reported change could be attributable to changes in climate, although the high estimates of Q_{10} indicate that, to be plausible, some indirect temperature-related mechanism must be involved. Our findings do not support the hypothesis that temperature-related drying of the upper layers of peat is a significant factor in increased carbon loss, but the similarity between the response to temperature of soil carbon concentration in organic soils and that of vegetation cover suggests that temperature-related changes in plant cover and hence the quality of litter returned to the soil are a more plausible indirect mechanism, although we cannot exclude the possibility that increased grazing pressure and the invasion of more decomposable grass species has been more prevalent at warmer sites.

We conclude that, based on the data used in this study, organo-mineral/mineral and organic soils under temperate conditions could show very different responses to changes in climate, with carbon concentration in mineral soils weakly positively correlated with rainfall, but insensitive to temperature, while carbon in organic soils is strongly negatively correlated with temperature. The sensitivity of organic soils to temperature, and the logistic analysis suggest that future monitoring should concentrate on those soils with soil carbon densities between 250 and 435 g kg^{-1}.

Combined with climate projections for the period 2010-2030, we predict an additional climate-related loss of soil carbon over England and Wales of between 1.02 and 2.18 Tg C annually over the period 1980-2030, or 0.77 – 1.57 Tg yr^{-1} if the contributions from rough grazing and upland grass are excluded. The lack of agreement between the NSI study and that of the Countryside Survey, together with the high Q_{10} values estimated for soils under some land uses, suggest caution. The clearest implication from this work is that future surveys of soil carbon should concentrate on the organic upland soils that, in this study, exhibited the highest sensitivity to changes in temperature, and that in the Countryside Survey study showed large but statistically non-significant declines in soil carbon.

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References

ADAS. 2011. The National Inventory and Map of Livestock Manure Loadings to Agricultural Land: MANURE-GIS. Defra, London.

Alonso, I. 2009. Managing Heathlands in the Face of Climate Change. In: *Proceedings of the 10th National Heathland Conference, 9th-11th September,* University of York. Natural England Commissioned Report Number 014. In: *10th National Heathlands Conference* (ed. Alonso, I.). Natural England, University of York.

Anderson, J.M. & Hetherington, S.L. 1999. Temperature, nitrogen availability and mixture effects on the decomposition of heather (*Calluna vulgaris* (L.) Hull) and bracken (*Pteridium aquilinum* (L.) Kuha) litters. *Functional Ecology*, **13,** 116-124.

Bell, M.J., Worrall, F., Smith, P., Bhogal, A., Black, H.I.J., Lilly, A., Barraclough, D. & Merrington, G. 2012. UK land-use change and its impact on SOC. *Global Biogeochemical Cycles*, **In Press**.

Bellamy, P.H., Loveland, P.J., Bradley, R.I., Lark, R.M. & Kirk, G.J.D. 2005. Carbon losses from all soils across England and Wales 1978-2003. *Nature*, **437,** 245-248.

Bond-Lamberty, B. & Thomson, A. 2010. Temperature-associated increases in the global soil respiration record. *Nature*, **doi:10.1038/nature08930,** 579-582.

Bonn, A., Monteith, D.T., Reed, M., Blackstock, K., Hanley, N., Thompson, D., Evans, M. & Alonso, I. 2011. UK National Ecosystem Assessment: Technical

Report. Chapter 5: Mountains, Moorlands and Heaths. In: *UK National Ecosystem Assessment* (eds. Watson, R. & Albon, S.). Defra
NERC, Cambridge.

Burke, I.C., Yonker, C.M., Parton, W.J., Cole, C.V., Flach, K. & Schimel, D.S. 1989. Texture, Climate and Cultivation Effects on Soil Organic Matter Content in US Grassland Soils. *Soil Science Society of America Journal*, **53,** 800-805.

Clutterbuck, B. & Yallop, A.R. 2010. Land management as a factor controlling dissolved organic carbon release from upland soils 2: Changes in DOC productivity over four decades. *Science of the Total Environment*, **408,** 6179-6191.

Cox, P.M., Betts, R.A., Jones, C.D., Spall, S.A. & Totterdell, I.J. 2000. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. *Nature*, **408,** 184-187.

Dai, W. & Huang, Y. 2006. Relation of soil organic matter concentration to climate and altitude in zonal soils of China. *Catena*, **65,** 87-94.

Davidson, E.A. & Janssens, I.A. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, **440,** 165-173.

Defra.2009.http://archive/defra.gov.uk/evidence/statistics/foodfarm/landuse/livestock/junesurvey. London. Accessed 10/08/2011.

Emmet, B.A., Reynolds, B., Chamberlain, P.M., Rowe, E., Spurgeon, D., Brittain, S.A., Frogbrook, Z., Hughes, S., Lawlor, S.J., Poskitt, J., Potter, E., Robinson, D.A., Scott, A., Wood, C. & Woods, C. 2010. Countryside Survey: Soils Report from 2007. Technical Report No. 9/07. 192pp. NERC/Centre for Ecology and Hydrology.

Evans, C.D., Chapman, C.J., Clark, J.M., Monteith, D.T. & Cresser, M.T. 2006. Alternative explanations for rising dissolved organic carbon export from organic soils. *Global Change Biology*, **12,** 2044 - 2053.

Evans, C.D., Freeman, C., Cork, L.G. & Thomas, D.N. 2007. Evidence against recent climate-induced destabilisation of soil carbon from ^{14}C analysis of riverine dissolved organic matter. *Geophysical Research Letters*, **34,** L077407.

Evans, C.D., Freeman, C. & Monteith, D.T. 2002. Climate change-Terrestrial export of organic carbon:communication arising. *Nature*, **415,** 862.

Fantappie, M., L'Abate, G. & Costanini, E.A.C. 2011. The influence of climate change on the soil organic carbon content in Italy from 1961 to 2008. *Geomorphology*, **doi:10.1016/j.geomorph.2011.02.006**.

Freeman, C., Evans, C.D., Monteith, D.T. *et al*., 2001. Export of organic carbon from peat soils. *Nature*, **412,** 785.

Goidts, E. & Van Wesemael, B. 2007. Regional assessment of soil organic carbon changes under agriculture in Southern Belgium (1955-2005). *Geoderma*, **141,** 341-354.

Gorham, E. 1991. Northern peatlands: role in carbon cycling and probable responses to climate warming. *Ecological Applications*, **1,** 182-195.

Hodgson, J.M. 1997. *Soil Survey Field Handbook: Describing and Sampling Soil Profiles*. Soil Survey and Land Research Centre, Silsoe, England.

Hossell, J.E., Briggs, B. & Hepburn, I.R. 2000. Climate Change and UK Nature Conservation: A review of the impact of climate change on UK species and habitat conservation policy. p. 72. Department of the Environment, Transport and the Regions, London.

IPCC. 1991. Climate Change: The IPCC Scientific Assessment. Report prepared for IPCC by Working Group 1. (eds. Houghton, J.T., Jenkins, G.J. & Ephraums, J.J.). Meteorological Office, London.

IPCC. 2007. Contrinution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In: *IPCC 2007 Report* (eds. Solomon, S., Qin, D., Manning, M. & et al.), p. 996. Intergovernmental Panel on Climate Change, Cambridge.

Jenkins, G., Murphy, J., Sexton, D., Lowe, J., Jones, P. & Kilsby, C. 2009. UK Climate Projections: Briefing report. p. 56. Met Office Hadley Centre, Exeter.

Kirby, K.J., Smart, S.M., Black, H.I.J., Bunce, R.G.H., Corney, P.M. & Smithers, R.J. 2005. Long term ecological change in British woodland (1971-2001). English Nature Research Reports Number 653 p. 128, English Nature, Peterborough.

Kirk, G.J.D. & Bellamy, P.H. 2010. Analysis of changes in organic carbon in mineral soils across England and Wales using a simple single-pool model. *European Journal of Soil Science*, **61,** 406-411.

Kirschbaum, M.U.F. 1995. The Temperature Dependence of Soil Organic Matter Decomposition, and the Effect of Global Warming on Soil Organic C Storage. *Soil Biology Biochemistry*, **27,** 753-760.

Lark, R.M. & Cullis, B.R. 2004. Model-based analysis using REML for inference from systematically sampled data on soil. *European Journal of Soil Science*, **55**, 799-813.

Lindsay, R.A. 2010. Peat bogs and carbon: A critical synthesis. RSPB Scotland, University of east London, London.

Lucas, R.E. 1982. Organic Soils (Histosols). Formation, distribution and chemical properties and management for crop production. In: *Farm Science*. Michigan State University, Michigan.

Miller, A.J., Amundson, R., Burke, I.C. & Yonker, C. 2004. The effect of climate and cultivation on soil organic C and N. *Biogeochemistry*, **67,** 57-72.

Monteith, D.T., Stoddard, J.L. & Evans, C.D. 2007. Dissolved organic trends resulting from changes in atmospheric deposition. *Nature*, **450,** 537-540.

Muggeo, V.M.R. 2003. Estimating regression models with unknown break-points. *Statistics in Medecine*, **22**, 3055-3071.

Ostle, N.J., Levy, P.E., Evans, C.D. & Smith, P. 2009. UK land use and soil carbon sequestration. *Land Use Policy*, **26,** 274-283.

Penuelas, J., Gordon, C., Llorens, L., Nielsen, T., Tietema, A., Beier, C., Bruna, P., Emmet, B., Estiarte, M. & Gorissen, A. 2004. Nonintrusive field experiments show different plant responses to warming and drought among sites, seasons, and species in a north-south European gradient. *Ecosystems*, **7,** 598-612.

Pickett, S.A. 1989. Space-for-Time Substitution as an Alternative to Long-Term Studies. In: *Long-Term Studies in Ecology: Approaches and Alternatives* (ed. Likens, G.E.), pp. 110-135. Springer-Verlag, New York.

Pinheiro, J.C. & Bates, D.M. 2000. *Mixed Effects Models in S and S-Plus.* Springer, New York.

Reijneveld, A., van Wensum, J. & Oenema, O. 2009. Soil organic carbon contents of agricultural land in the Netherlands between 1984 and 2004. *Geoderma*, **152,** 231-238.

Smith, P., Chapman, S.J., Scott, W.A., Black, H.I.J., Wattenbach, M., Milne, R., Campbell, C.D., Lilly, A., Ostle, N., Levy, P.E., Lumsdon, D.G., Millard, P., Towers, W., Zaehle, S. & Smith, J.U. 2007. Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978-2003. *Global Change Biology*, **13,** 2605-2609.

Smith, P. & Fang, C. 2010. A warm response by soils. *Nature*, **464,** 499-500.

Smith, P., Fang, C., Dawson, J.J.C. & Moncreiff, J.B. 2008. Impact of global warming on soil carbon. *Advances in Agronomy*, **97,** 1-43.

Tomlinson, R.W. 2005. Soil carbon stocks and changes in the Republic of Ireland. *Journal of Environmental Management*, **76,** 77-93.

Tomlinson, R.W. & Milne, R.M. 2006. Soil carbon stocks and land cover in Northern ireland from 1939 to 2000. *Applied Geography*, **26,** 18-39.

Turunen, J. 2008. Development of Finnish peatland area and carbon storage. *Boreal Environment Research*, **13,** 319-334.

Werkman, B.R. & Callaghan, T.V. 2002. Response of bracken and heather to increased temperature and nitrogen addition, alone and in competition. *Basic and Applied Ecology*, **3,** 267-276.

Worrall, F., Burt, T.P. & Adamson, J. 2006. The rate of loss and controls upon DOC loss in a peat catchment. *Journal of Hydrology*, **321,** 311-325.

Worrall, F., Burt, T.P. & Sheddon, R. 2003. Long term records of riverine carbon flux. *Biogeochemistry*, **64,** 165-178.

Tables

Table 1. Details of the land uses in the NSRI survey

Table 2. Regression coefficients for organo-mineral and mineral soils in the initial NSI survey and for data from the RSS and woodland surveys. Figures in brackets are standard errors.

Coefficients significant at $p<0.05$ or better unless indicated (ns).

Table 3. Regression coefficients for organic soils in the initial NSI survey. Figures in brackets are standard errors.

Coefficients significant at $p<0.05$ or better unless indicated (ns).

Table 4. Changes in soil carbon for grazed land uses attributable to climate or reductions in stocking rate. Figures are expressed as a percentage of those reported by Bellamy *et al.,* (2005).

Table 5. Estimates of Q10 for 5 land uses.

Table 6. Estimated decline in soil carbon in g kg^{-1} under the medium emission scenario of UKCP09.

Table 7. Estimated annual loss of soil carbon in Tg between 1980-2030. Figures are for England and Wales.

Table 8. Regression coefficients for multiple linear regression, based on equation (1), using data from the 1st and 2nd NSI surveys. Figures in brackets are estimated standard errors. Coefficients are significant at $P<0.05$ unless non-significant (ns). Coefficients estimated using REML.

Table 9. Regression coefficients for logistic equation (2) using data from the 1st and 2nd NSI surveys. Data are pooled semi-natural land uses (rough grazing, upland grass, upland heath, bog). Figures in brackets are estimated standard errors. Coefficients estimated using REML.

Land use	n	Soil carbon at initial NSI survey /g kg^{-1}	Average temperature T /^{o}C and rainfall R /mm		Average temperature T /^{o}C and rainfall R /mm	
			30 year average (1951-80)		30 year average at second NSI survey	
			T	R	T	R
Arable						
o-mineral/mineral	1844	27.7	9.18	693	9.36	700
Rotational grass						
o-mineral/mineral	663	35.9	9.14	903	9.28	917
Permanent grass						
organic	32	245	8.77	1068	8.95	1087
o-mineral/mineral	1516	46.2	8.96	928	9.14	943
Deciduous woodland						
organic	17	233	9.07	833	9.5	849
o-mineral/mineral	252	48.8	9.14	889	9.57	908
Scrub						
organic	5	310	9.70	961	10.1	992
o-mineral/mineral	75	48.6	9.04	896	9.45	907
Lowland heath						
o-mineral/mineral	18	69.2	9.67	934	10.1	944
Coniferous woodland						
organic	34	339	7.59	1447	7.97	1511
o-mineral/mineral	171	52.9	8.62	1008	9.03	1037
Rough grazing						
organic	61	320	7.67	1480	8.03	1520
o-mineral/mineral	219	59.4	8.42	1124	8.81	1151
Upland grass						
organic	136	358	7.07	1656	7.44	1730
o-mineral/mineral	93	85.7	7.27	1553	7.63	1611
Upland heath						
organic	79	340	7.33	1317	7.71	1358
o-mineral/mineral	34	90.7	7.54	1218	7.93	1244
Bog						
organic	44	401	7.66	1502	8.03	1569

Table 1 Details of the land uses in the NSI survey. Soil carbon is in g kg^{-1}. Soil carbon at the initial NSI survey calculated from original data.

Land use	n	Intercept /g kg^{-1}	T	R	Adj r^2
arable	1844	26.5 (9.2)	ns	+0.008 (0.003)	0.003
RSS	2965	21.5 (6.5)	ns	0.011 (0.002)	0.009
rotational grass	663	36.9 (10)	ns	+0.013 (0.003)	0.04
RSS	2475	19.1 (3.4)	-1.04 (0.4)	+0.022 (0.0009)	0.17
permanent grass	1516	53.7 (7.5)	-2.03 (0.74)	0.012 (0.002)	0.03
RSS	4441	45.6 (3.6)	-1.93 (0.36)	0.017 (0.0009)	0.09
deciduous woodland	252	ns	ns	0.01 (0.006)	-
woodland survey 1971	91	ns	ns	0.026 (0.0006)	-
scrub	75	ns	ns	ns	n/a
lowland heath	18	-289 (123)	+31.2 (11.2)	0.06 (0.02)	0.35
coniferous woodland	171	ns	ns	0.026 (0.006)	0.15
rough grazing	219	53.8 (19.2)	ns	0.022 (0.005)	0.14
upland grass	93	ns	ns	ns	-
upland heath	34	164 (62)	ns	ns	-

Table 2 Regression coefficients for organo-mineral/ mineral sites from the NSI initial survey and data from RSS and the Woodland Survey. Figures in brackets are standard error of coefficients. Regression coefficients significant at $P<0.05$ unless non - significant (ns).Regression coefficients for the NSI data fitted by REML; RSSS and woodland survey data fitted by OLS. See text for details.

Land use	n	T_c /°C	Intercept /g kg^{-1}	T	R	Adj r^2
Permanent grass	32	n/a	896 (162)	-74 (15)	ns	0.50
Coniferous woodland	34	6.1 (1.8)	410 (75)	-48 (23)	ns	0.12
Rough grazing	61	7.1 (0.4)	416 (20.6)	-76 (14)	ns	0.41
Upland grass	136	6.6 (0.7)	434 (15)	-81 (19)	ns	0.18
Upland heath	79	6.8 (0.3)	432 (21)	-88 (25)	ns	0.16
Bog	44	6.6 (0.4)	507 (15)	-50 (11)	ns	0.35

Table 3 Regression coefficients for organic sites from the initial NSI survey. Figures in brackets are standard error of coefficients. Regression coefficients significant at $P<0.05$ unless non-significant (ns). Coefficients fitted by ML. T_c is the inflection temperature; for details see text.

	Change in soil carbon attributable to		
Land use	Climate	Stocking density	Total
Rotational grass	0	55	55
Permanent grass	7	42	49
Rough grazing	18	55	73
Upland grass	27	-	27
Upland heath	40	-	40
Bog	17	-	17

Table 4 Changes in soil carbon concentration predicted from changes in climate and stocking density. Figures expressed as %age of change reported by Bellamy *et al.*, (2005)

Land use		Q_{10} estimated from modelled changes
Permanent grass	organic	29 - 1300
	organo-mineral/mineral	1.4 – 2.7
Rough grazing	organic	5.8 – 58
Upland heath	organic	5 –42
Upland grass	organic	7 – 82
Bog	organic	4 – 26

Table 5 Q_{10} estimated from changes in soil carbon attributable to temperature using equation 7.

Land use	Change in soil carbon in g kg^{-1} at probabilities of:		
	10%	50%	90%
Arable	-0.25 (0.20)	+0.016 (0.012)	+0.296 (0.20)
Rotational grass	-0.55 (0.24)	+0.026 (0.012)	+0.64 (0.29)
*Permanent grass	-4.03 (1.4)	-6.1 (2.5)	-8.5 (3.8)
Deciduous woodland		no significant change	
Scrub		no significant change	
Lowland heath	+47 (22)	+88 (38)	+135 (60)
*Coniferous woodland	-8.8 (7.2)	-13.8 (12.5)	-19.8 (19.3)
*Rough grazing	-16.3 (5.5)	-27.0 (9.7)	-39.8 (14.7)
Upland grass	-46 (22)	-81 (38)	-121 (56)
Upland heath	-57 (32)	-101 (56)	-153 (85)
Bog	-38 (17)	-75 (32)	-117 (52)

Table 6 Changes in soil carbon in g kg^{-1} over the period 1980 to 2030, estimated from regression analysis and the 10, 50 and 90% probability cases from the medium emission climate change scenarios from UKCP09. Figures in brackets are 95% confidence limits either calculated from regression equations or derived from Monte Carlo simulation. *indicates weighted average of organic and organo-mineral/mineral soils.

Land use	Area (E&W) ('000 ha)	Estimated annual change in soil carbon in Tg under UKCP09 medium emission scenarios for 2010-2030 at probability levels of:		
		10%	50%	90%
Arable	4609	-0.038 (0.027)	+0.0024 (0.0018)	+0.0452 (0.031)
Rotational grass	637	-0.009 (0.004)	+0.0004 (0.0002)	+0.011 (0.004)
Permanent grass	4999	-0.53 (0.19)	-0.79 (0.33)	-1.11 (0.50)
Rough grazing	2210	-0.21 (0.07)	-0.34 (0.12)	-0.50 (0.19)
Coniferous woodland	380	-0.05 (0.04)	-0.09 (0.09)	-0.12 (0.11)
Upland grass	177	-0.04 (0.02)	-0.07 (0.03)	-0.11 (0.05)
Upland heath	418	-0.12 (0.06)	-0.22 (0.12)	-0.33 (0.18)
Bog	100	-0.016 (0.007)	-0.03 (0.013)	-0.048 (0.02)
Total estimated annual loss 1980-2030		-1.02 (0.31)	1.55 (0.53)	2.18 (0.82)

Table 7 Annual loss of soil carbon over the period 1980-2030 estimated from regression models, UKCP09 medium scenarios and land use data for England and Wales. Figures in brackets are 95% confidence limits derived from Monte Carlo simulations.

Land use		n	Intercept /g kg^{-1}	Temperature	Rainfall
Organic soils					
Rough grazing	1st survey	61	72.3 (8.7)	-4.83 (0.87)	ns
	2nd survey	23	24.0 (9.4)	ns	ns
Upland grass	1st survey	136	76.8 (7.9)	-5.32 (0.93)	ns
	2nd survey	54	49.1 (11.5)	-2.64 (1.45)	ns
Upland heath	1st survey	79	69.7 (12.2)	-5.64 (1.51)	ns
	2nd survey	39	66.8 (11.8)	-4.19 (1.45)	ns
Bog	1st survey	44	74.4 (7.3)	-3.84 (0.82)	ns
	2nd survey	18	75.4 (18.7)	-4.91 (2.21)	ns
Coniferous woodland	1st survey	34	70.7 (17.8)	-4.71 (2.03)	ns
	2nd survey	16	50.1 (16.8)	-2.98 (2.16)	ns
Mineral soils					
Arable	1st survey	1844	26.5 (9.2)	ns	0.008 (0.0003)
	2nd survey	587	7.24 (15.2)	ns	0.001 (0.0005)
Permanent grass	1st survey	1516	53.7 (7.5)	-2.03 (0.74)	0.012 (0.002)
	2nd survey	534	47.7 (8.9)	-2.49 (0.80)	0.016 (0.002)
Rotational grass	1st survey	663	36.9 (10)	ns	0.013 (0.003)
	2nd survey	196	22.5 (15.8)	ns	0.016 (0.004)
Coniferous woodland	1st survey	171	46.6 (28.0)	ns	0.026 (0.006)
	2nd survey	92	22.6 (30.0)	ns	0.028 (0.007)
Rough grazing	1st survey	219	53.8 (19.2)	ns	0.022 (0.005)
	2nd survey	91	80.8 (29.8)	ns	0.024 (0.009)

Table 8 Regression coefficients for multiple linear regression (equation (1) determined from the 1st and 2nd NSI surveys by REML. Figures in brackets are estimated standard errors. Coefficients significant at $p<0.05$ unless non significant (ns).

	n	C_{max}	C_{min}	M	s
1st survey	339	435 (13)	253 (35)	7.9 (0.3)	-15.9 (6.6)
2nd survey	134	408 (46)	262 (26)	6.9 (0.4)	-13.8 (11.1)

Table 9 Regression coefficients for logistic equation (equation 2) determined from the 1st and 2nd NSI surveys by REML. Coefficients are as in equation (2). Data are pooled semi-natural land uses (rough grazing, upland grass, upland heat and bog). Figures in brackets are estimated standard errors.

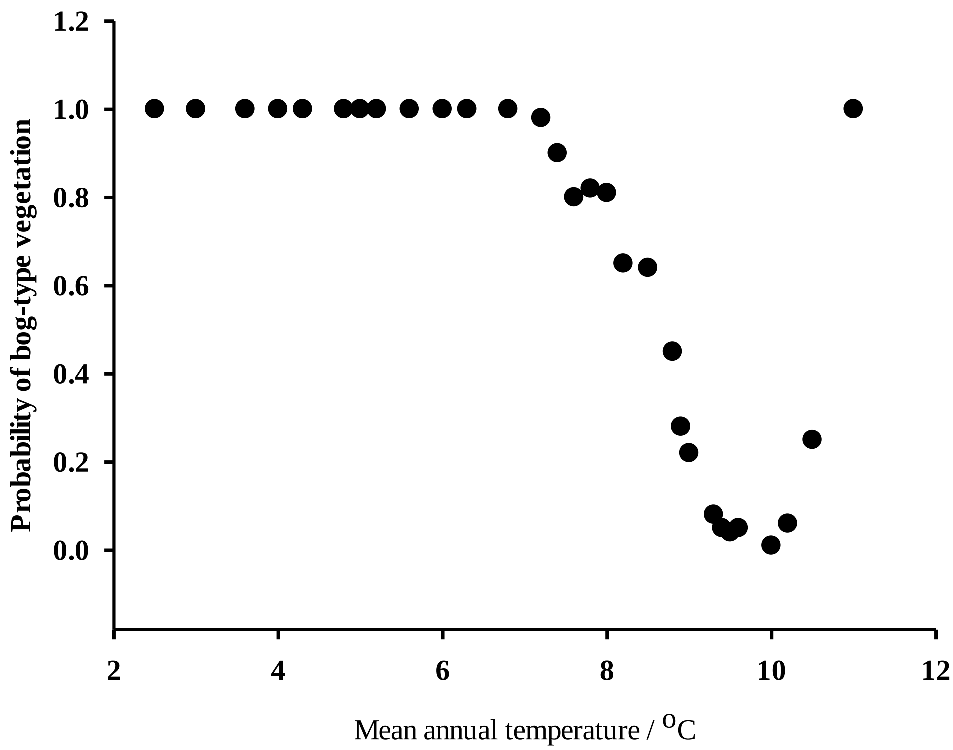
Figures

Figure 1. Probability of occurrence of bog and dense shrub moor vegetation as a function of mean annual temperature. Redrawn from Hossell *et al*., (2000).

Figure 2. Pooled soil carbon concentration data for organic soils under rough grazing, upland grass, upland heath and bog, as a function of mean annual temperature 1961-1990. Regression line is the logistic equation (2) with C_{max}=435 g kg^{-1}, C_{min}=253 g kg^{-1}, M=7,9 ^{0}C, s=-15.9; adjusted r^2=0.18; P<0.0001. Dotted lines are the 95% confidence limits for the regression.

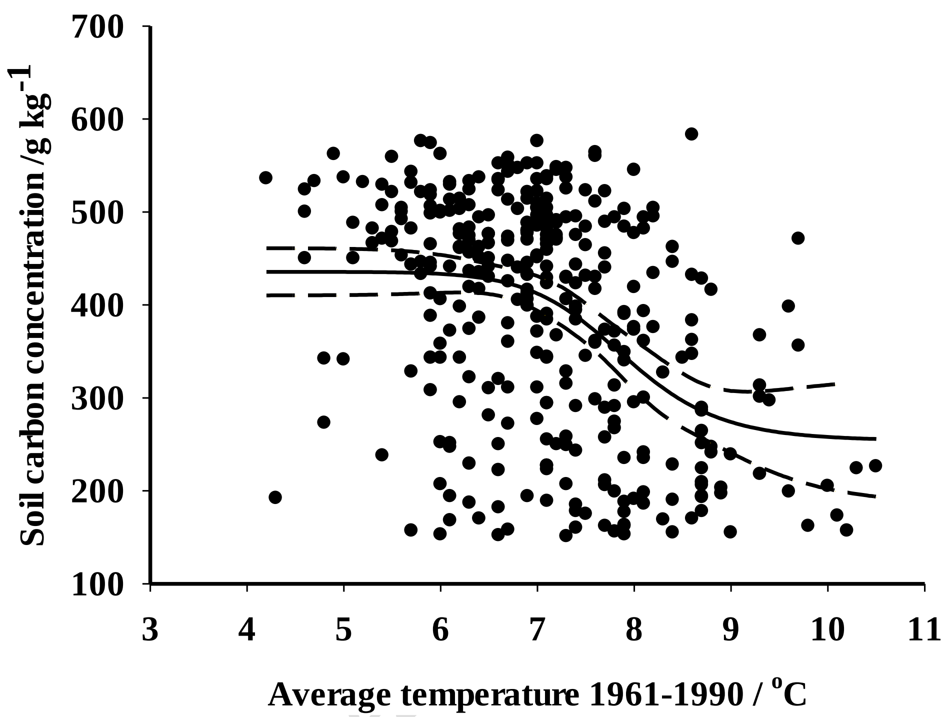
Figure 3. Changes in soil carbon calculated from 30 year regression models (diamonds) compared to those reported by Bellamy *et al.*, (2005) (circles). Error bars are 95% confidence limits

Figure 1.



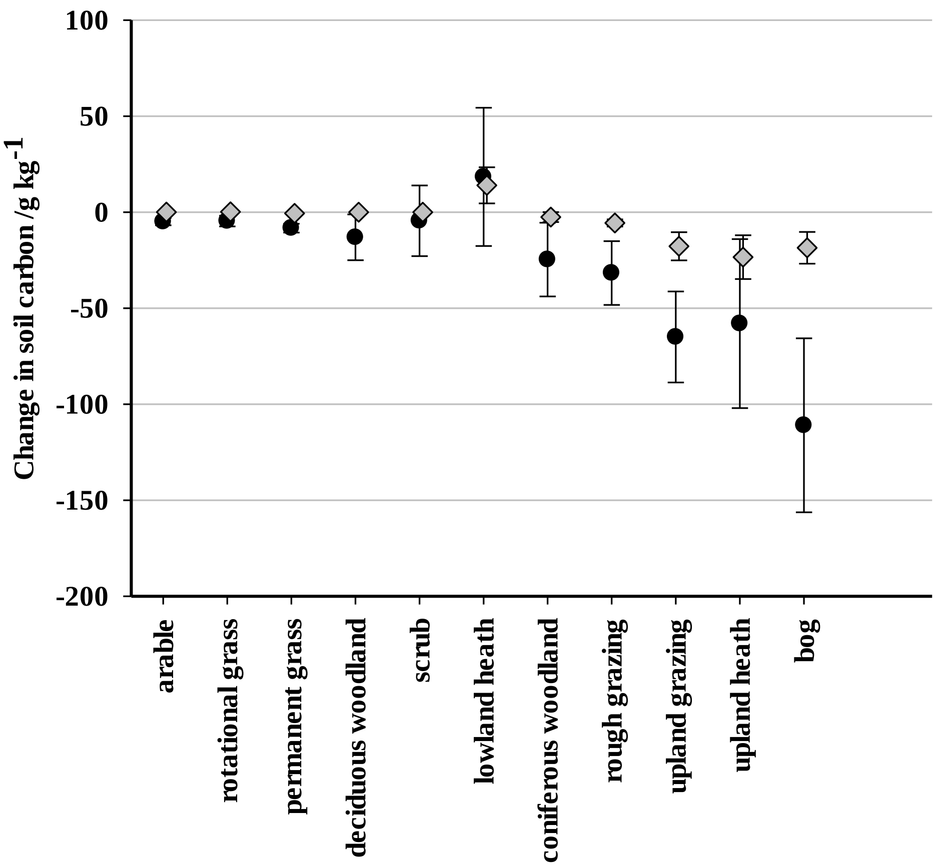
1.2
1.0
0.8
0.6
0.4
0.2
0.0
Probability of bog-type vegetation
2
4
6
8
10
12
Mean annual temperature / ^{o}C

Figure 2.



700
600
500
400
300
200
100
Soil carbon concentration /g kg-1
3
4
5
6
7
8
9
10
11
Average temperature 1961-1990 / oC

Figure 3.



100
50
0
-50
-100
-150
-200
Change in soil carbon /g kg-1
arable
rotational grass
permanent grass
deciduous woodland
scrub
lowland heath
coniferous woodland
rough grazing
upland grazing
upland heath
bog