



Estimating changes in Scottish soil carbon stocks using ECOSSE. II. Application

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ABSTRACT: In order to predict the response of carbon (C)-rich soils to external change, models are needed that accurately reflect the conditions of these soils. Here we present an example application of the new Estimation of Carbon in Organic Soils – Sequestration and Emissions (ECOSSE) model to estimate net change in soil C in response to changes in land use in Scotland. The ECOSSE estimate of annual change in soil C stocks for Scotland between 2000 and 2009 is $-810 \pm 89 \text{ kt yr}^{-1}$, equivalent to $0.037 \pm 0.004 \% \text{ yr}^{-1}$. Increasing the area of land-use change from arable to grass has the greatest potential to sequester soil C, and reducing the area of change from grass to arable has the greatest potential to reduce losses of soil C. Across Scotland, simulated changes in soil C from C-rich soils (C content $>6\%$) between 1950 and 2009 is -63 Mt , compared with -35 Mt from non-C-rich mineral soils; losses from C-rich soils between 2000 and 2009 make up 64% of the total soil C losses. One mitigation option that could be used in upland soils to achieve zero net loss of C from Scottish soils is to stop conversion of semi-natural land to grassland and increase conversion of grassland to semi-natural land by 125% relative to the present rate. Mitigation options involving forestry are not included here because the data available to calculate losses of soil C do not account for losses of soil C on drainage of semi-natural land.

KEY WORDS: Organic soils · Dynamics simulation modelling · Changes in soil C stocks · Land-use change · Climate change

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1. INTRODUCTION

Accurate predictions of the effects of changes in climate and land use on greenhouse gas (GHG) emissions are vital for informing land-use policy. Peatlands, characterized as high carbon (C) soils, cover approximately 4 million km² globally, with a total C stock estimated in 2008 to be $\sim 450 \text{ Pg C}$ (Joosten 2009), and highly organic soils can make a significant contribution to national

GHG emissions. For example, $\sim 15\%$ of Scotland's total GHG emissions come from land-use changes on Scotland's C-rich soils (Smith et al. 2007a). Therefore, it is essential that national simulations of changes in soil C stocks include accurate simulation of highly organic soils. However, the majority of models that are currently used to predict differences in soil C and nitrogen (N) caused by climate or land-use changes have been derived from models originally developed for mineral

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soils (Smith 2001). None of these models is entirely satisfactory for describing what happens to organic soils following land-use change (Smith et al. 2007a).

The Estimation of Carbon in Organic Soils – Sequestration and Emissions (ECOSSE) model (Smith et al. 2010a) has been developed to simulate the impacts of land use and climate change on GHG emissions from both mineral and organic soils. It is driven by commonly available meteorological, land-use and soil data, and was developed from concepts originally derived for mineral soils in the RothC (Jenkinson & Rayner 1977, Jenkinson et al. 1987, Coleman & Jenkinson 1996) and SUNDIAL (Bradbury et al. 1993, Smith et al. 1996a) models. Following these established models, ECOSSE uses a pool-type approach, describing soil organic matter (SOM) as pools of inert organic matter, humus, biomass, resistant plant material and decomposable plant material. The structure and formulation of ECOSSE are described in detail by Smith et al. (2010a,b [2010 in this Special]).

ECOSSE differs from RothC and SUNDIAL in the addition of a number of processes and impacts that are not crucial in the mineral arable soils that these models were originally developed for. More importantly, ECOSSE differs from RothC and SUNDIAL in the way that it makes full use of the limited information that is available to run models at the national scale. In particular, measurements of soil C are used to interpolate the activity of the SOM and the plant inputs needed to achieve those measurements. If data are available describing soil water, plant inputs, nutrient applications and timing of management operations, these can be used to drive the model and so better apportion the factors determining the interpolated activity of the SOM. However, if any of this information is missing, the model can still provide adequate simulations of SOM turnover, although the impact of changes in conditions will be estimated with less accuracy due to the reduced detail of the inputs.

The processes of C and N turnover in the soil are included in the model using only simple equations driven by readily available input variables. This not only ensures that the model remains functional for data-limited runs, but also allows it to be applied at both field and national scales, without a great loss of accuracy. Because it is able to function at both field and national scales, if the model is driven by appropriate input data, field-scale evaluations can be used to determine uncertainty in national simulations. In an evaluation of uncertainty in simulations driven only by the limited data available at a national scale, Smith et al. (2010b) concluded that simulated values show a high degree of association with the measurements in both total C and change in C content of the soil. Over all sites where land-use change occurred, the average deviation be-

tween the simulated and the measured values of percentage change in soil C was less than the experimental error (11% simulation error, 53% measurement error), suggesting that the uncertainty in the national-scale simulations is ~11%. Only a small bias in the simulations compared to the measured values was observed, suggesting that a small underestimate of the change in soil C should be expected at the national scale (-4%).

Here we present an example application of ECOSSE, using the 11% simulation error obtained by Smith et al. (2010b). The model is applied to national-scale simulations of changes in soil C stocks in Scotland. Scotland is a relatively small country (~78 000 km²), and so provides an ideal example for developing the capability to do simulations for larger countries using a dynamic model. The soils in Scotland range in organic C content from <1% to almost 60% (Lilly et al. 2004), and so provide a good demonstration of the application of the model to simulate national emissions from soils with a range of C contents.

2. METHODS

2.1. Approach used in national simulations of Scotland

The ECOSSE model (Smith et al. 2010a,b) was used together with soils, land-use and weather data to simulate changes in soil C content across the whole of Scotland. As illustrated in Fig. 1, soil data were available on a 1 km² grid across Scotland (Lilly et al. 2004), whereas weather data were available on a 5 km² grid (http://www.ukcip.org.uk/index.php?id=161&option=com_content&task=view), and land-use and land-use change data were only available on a 20 km² grid (Mobbs & Dyson 2009, Thomson et al. unpubl. data). The same weather data were used for all twenty-five 1 km² cells in each 5 km² grid. The land-use and land-use change data were interpolated to a 1 km² grid by assuming the patterns were uniform across all four hundred 1 km² cells in the 20 km² grid. If the land use was not reported as occurring on a particular soil type according to the Scottish Soils Database (Lilly et al. 2004), the model set all land-use changes to or from that land use to zero and increased the area of the land-use change occurring on the other soil types within the 1 km² cell in proportion to the area of the soil type. If the land use was not reported on any of the other soil types in the cell, the land-use change was assumed to be absent from that 1 km² cell and was spread evenly across the other 1 km² cells in the 20 km² grid cell. This allowed results to be reported on a 1 km² grid, while avoiding unrealistic land uses on a given soil, such as arable crops on blanket peat.

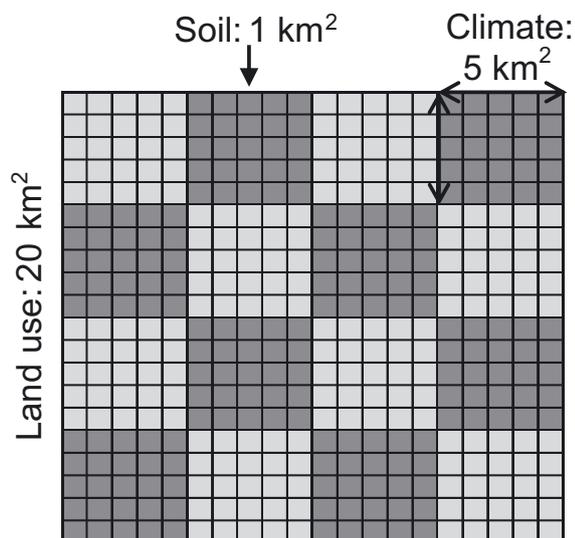


Fig. 1. Resolution of data available to run the Estimation of Carbon in Organic Soils—Sequestration and Emissions (ECOSSE) model at a national scale

Simulations were of 3 types: historical, future and simulations of the potential of mitigation options to reduce soil C losses. The historical simulations determined changes in soil C content for each decade since 1950, apportioning the losses to different soil types and changes in land use. These simulations quantified the C emissions from Scottish soils, and resolved which land-use changes are responsible for the changes in soil C and on which soil types most changes in soil C occur. The future simulations quantified the changes in soil C due to projected land-use and climate change over the next decade. The calculations of the different mitigation options determined which options have the greatest potential to reduce losses of soil C.

The national estimates of changes in soil C stocks not only determine the total change in soil C, but also apportion the observed change to specific changes in land use, management or climate. This is of crucial importance if we are to implement effective mitigation measures to protect vulnerable soil C sinks (Reay et al. 2007). Bellamy et al. (2005) suggested a possible link to climate change to explain the observed mean loss of soil organic C of $0.6\% \text{ yr}^{-1}$ between 1978 and 2003 in England and Wales. As discussed by Smith et al. (2007b), those findings contradict evidence that UK and European soils as a whole are a net CO_2 sink (Janssens et al. 2003), along with data from another long-term study of soil organic C in British woodlands (Kirkby et al. 2005), which suggest a small increase in soil organic matter over 30 yr ($0.094\% \text{ increase yr}^{-1}$). Other repeated sampling studies in Europe have shown contrasting results, with some showing loss of soil organic C (in Flemish croplands, Sleutel et al. 2003),

attributed to changing manure application practices, and others showing no loss of soil organic C (in Danish croplands, Heidmann et al. 2002, and in Austrian soils, Dersch & Bohm 1997). Our results provide modelling evidence that will allow the observed C changes in Scottish soils to be attributed to different drivers, and will provide an important contribution to this debate.

2.2. Input data used to drive the simulations

Land-use and land-use change data were obtained from the database held by the Centre for Ecology and Hydrology (CEH; Mobbs & Dyson 2009, Thomson et al. unpubl. data). The spatial distribution of the land-use change data across Scotland is shown in Figs. 8.3 & 8.4 of Smith et al. (2007a). The decadal changes in land use across Scotland since the 1950s are shown in Fig. 2. The land-use types specified were arable, grassland, forestry and semi-natural (moorland and other uncultivated land). Areas of human settlement and/or infrastructure and water bodies were excluded from the analysis. The data provided included land-use and land-use change data on a 20 km^2 grid, specifying historical decadal land-use change for 1950–1959, 1960–1969, 1970–1979, 1980–1989 and 1990–1999, and predicted decadal land-use change for 2000–2009 and 2010–2019 (average annual change in kha yr^{-1} allocated in each 20 km^2 cell).

Soil data were obtained from the Scottish Soils Knowledge and Information Base (SSKIB; Lilly et al. 2004). This database holds statistical summary data for a set of typical soil horizons for each soil taxonomic unit (series) delineated on the 1:250 000 soil map of Scotland. A digitized version of this map comprises around 85 000 georeferenced 1 km^2 grid cells each with the proportion and name of all the individual soil taxonomic units in that cell. This is further modified by the addition of land use that separates the uncultivated from the cultivated versions (phenotypes) of the same soil taxonomic unit. For the national-scale simulations, the 5 most extensive soils within each cell were used, giving a total of approximately 800 different soil and/or land-use types. This was later combined with climate data to provide a soil wetness class in each 1 km^2 cell based on soil and climate parameters (Lilly & Matthews 1994).

Typical horizon sequences and thicknesses were derived for each of the 800 soil phenotypes in SSKIB, except for those soils where rock normally occurred within 1 m (e.g. rankers and some upland soils), using information contained within the Scottish Soils Database (Lilly et al. 2004). The distinction between cultivated and uncultivated versions of the same soil series is important because both the soil chemistry and hori-

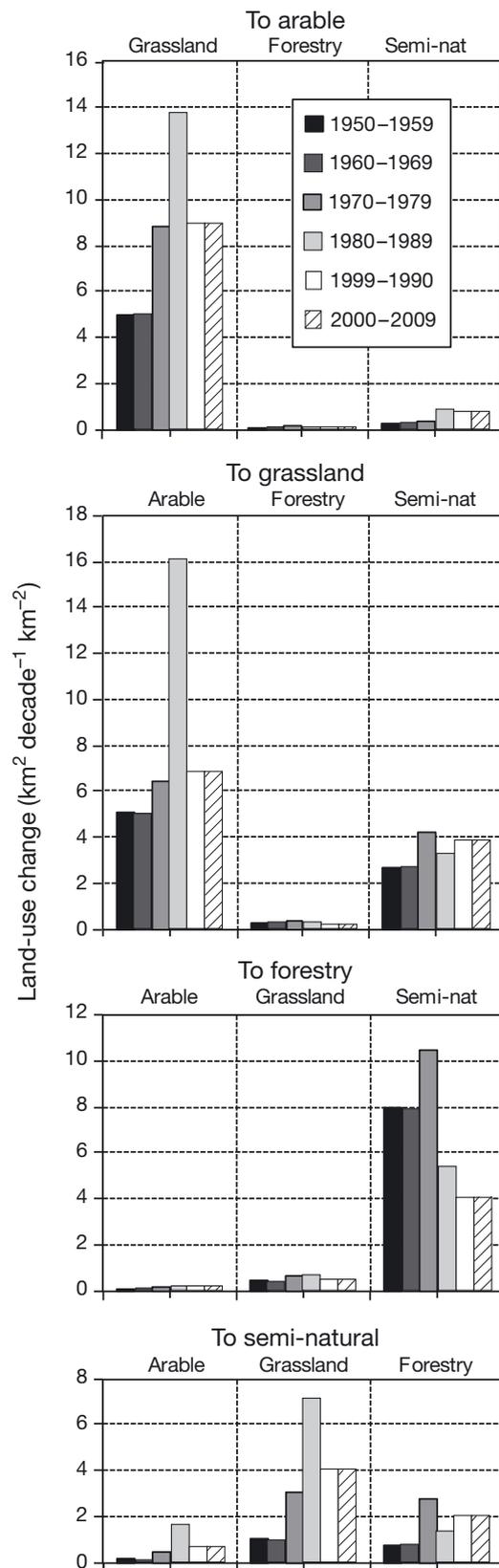


Fig. 2. Decadal changes in land use across Scotland since the 1950s (Mobbs & Dyson 2009, Thomson et al. unpubl. data)

zon sequences differ. Once the typical horizon sequence was determined, summary statistics were derived from the Scottish Soil Analytical Database (Lilly et al. 2004). Only soil horizons that matched those selected as being typical for that soil were used in the derivation of the summary statistics.

Data for cultivated soils were then subdivided into 2 land-use groups, arable and grassland, and, for uncultivated soils, into forestry and semi-natural land covers. Rough grassland is included in the semi-natural category. This was achieved using correction factors derived from the Scottish Soils Analytical Database (Lilly et al. 2004). Because the summary statistics for cultivated soils includes data from soils with arable, grassland and rotational grassland land uses, correction factors were applied in order to estimate C and pH values for the arable and grassland subgroups. These correction factors were derived from over 2000 individual measurements of topsoil C and pH. In total, 0.2% was subtracted from all SSKIB cultivated topsoil mean C values to derive a value for arable soils, whereas 1.2% was added to derive a value for C contents of grassland soils. For pH, arable topsoils were increased by 0.04 pH units, whereas grassland topsoils were decreased by 0.46 (Smith et al. 2009). These values were used to initialize the soil C pools under grass and arable land uses, thus defining the activity of the SOM and the plant inputs as described by Smith et al. (2010b, Section 2.4).

There were only small differences in the organic C contents of organic layers under forestry compared to semi-natural soils, with some horizons having greater organic C contents in afforested compared with semi-natural soils, and other horizons showing the opposite trend (Smith et al. 2010b). Therefore, no adjustment was made to the C contents of the soil horizons of uncultivated soils to distinguish between afforested soils and those under semi-natural vegetation. However, it was assumed that soils under forestry were more likely to develop a deeper litter layer than those under semi-natural grasslands or moorland. The litter layer under forestry was determined from measurements. The average thickness of the partially decomposed organic debris (LF) horizon under forestry was determined along with the average C content and its standard deviation. This horizon was added to all uncultivated soil profiles to derive data for soils under forestry. The bulk density of the LF horizon was averaged from a number of measured LF horizons, whereas the bulk densities of the other soil horizons from both cultivated and uncultivated soils were derived by a series of pedotransfer functions. These pedotransfer functions are simple regression equations that relate the dry bulk

density of a soil horizon to the organic C and clay and/or silt contents and that were developed from data from a wide range of Scottish soils (Lilly 1999, 2000). Following the approach taken by King & Franzmeier (1981) and the Hydrology of Soil Types classification (Boorman et al. 1995), the soil horizons were grouped into topsoils and subsoils and according to the mode of deposition of the soil parent material. The soil characteristic summary data provided for each horizon and land-use type (arable, grassland, forestry and semi-natural) included the top and bottom depths of the horizon (cm), the C content ($\text{kt } 20 \text{ km}^{-2}$), pH, percent clay, silt and sand, and the estimated bulk density (g cm^{-3}).

The characteristics of each horizon in the soil series identified in each grid cell were used to drive the model. Total soil C (kg C ha^{-1}) was used to determine the organic inputs to the soil and the amount of C in each of the SOM pools at the start of the simulation, as described by Smith et al. (2010b, Section 2.4). This was done for soils defined under each land use identified in the cell, thus allowing the impact of land-use change on SOM turnover to be simulated by changing the organic inputs from those estimated for the original land use to the values obtained for the new land use. Soil pH was used to calculate rate-modifying factors for aerobic and anaerobic decomposition and to determine the stable C:N ratio. The percent clay was used to calculate the proportion of SOM retained in the soil on decomposition following the approach of Coleman & Jenkinson (1996) and, together with bulk density (g cm^{-3}) and silt content, to estimate the water content at field capacity, using the equation provided by Hall et al. (1977). This equation was originally derived to determine field capacity in soils from England and Wales, and thus will introduce some error when applied to Scottish soils, especially peats. However, because the activity of the SOM decomposition is initialized using the specified soil conditions, the impact of errors in estimated field capacity on changes in soil C will be small. If the focus of the simulations was the impact of changes in the soil water status on soil C, a more appropriate pedotransfer function would be required.

The moisture and temperature rate modifiers were calculated using weather data. The historical simulations use monthly long-term average data. Data were downloaded as a grid of 5 km^2 from the climate monitoring service of the MetOffice UK (http://www.ukcip.org.uk/index.php?id=161&option=com_content&task=view).

Monthly long-term average data are based on the period 1961–1990. Future weather data were ob-

Table 1. Relationship between UKCIP02 scenarios and IPCC SRES scenarios, showing mean atmospheric CO_2 concentration (ppm) and mean global annual surface temperature increase as simulated by the HadCM3 climate model. See Hulme et al. (2002) for more details

SRES emissions scenario	UKCIP02 climate change scenario	Mean global temp. increase ($^{\circ}\text{C}$)	Atmospheric CO_2 conc. (ppm)
A1F1	High emissions	3.9	810
A2	Medium-high emissions	3.3	715
B2	Medium-low emissions	2.3	562
B	Low emissions	2.0	525

tained from the UKCIP02 database (UKCIP 2002). UKCIP02 data were used in preference to the more recent UKCP09 because gridded spatial data were not available for UKCP09 at the time of the present study. The UKCIP02 data are derived from the HadCM3 model for 4 scenarios based on the predicted cumulative CO_2 emissions between 1990 and 2100, as shown in Table 1. The data include rainfall and temperature predictions for each scenario for 3 future time steps: 2020, 2050 and 2080. These data are in 5 km^2 grids covering the whole of Scotland.

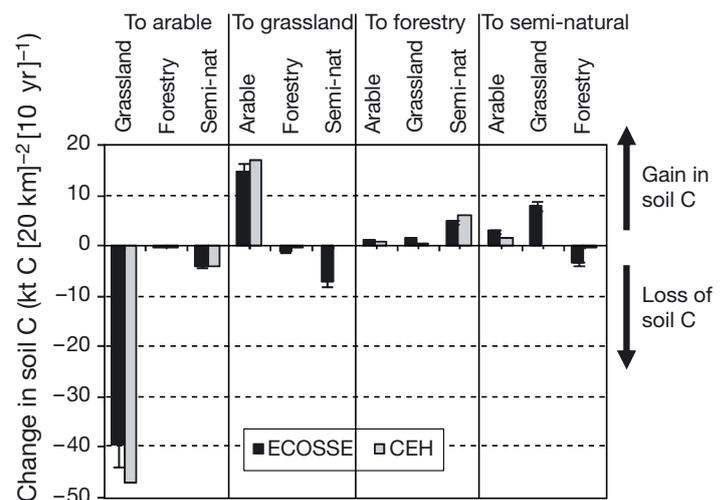


Fig. 3. Comparison of simulated changes in soil C content from 2000 to 2009 divided according to land-use change as estimated by ECOSSE and by the Centre for Ecology and Hydrology (CEH). Values are averaged across Scotland. Error bars indicate the uncertainty in the simulations calculated in Smith et al. (2010b) from the simulation of the National Soil Inventory of Scotland (NSIS) sites as the average deviation of the simulations from measurements (root mean squared deviation = 11%). Note the units used for change in soil C reflect the 20 km^2 grid cells and the decadal time slice used in the land-use change data

3. RESULTS

3.1. Change in soil C associated with different land-use changes

A comparison of the average changes in soil C content across Scotland as estimated by ECOSSE and by the CEH GHG inventory (Milne et al. 2004, summarised in Smith et al. 2007a, 2009) is shown for 2000–2009 in Fig. 3. The CEH approach estimates the change in soil C with change between 2 land uses from the equilibrium soil C content under the initial land use, the equilibrium soil C content under the final land use and a time constant for the change. The ECOSSE simulations use the SSKIB soil data and historical weather, land-use and land-use change data as described in Section 2.2. The results show the changes in soil C content calculated using a simulation started with 1950. Changes in soil C stocks due to land-use change occurring between 1950 and the end of the respective decade contribute to the changes in soil C stocks estimated in the given decade; for example, land-use changes occurring between 1950 and 2009 contribute to the changes in soil C stocks estimated during the decade 2000–2009. The effects on soil C of any land-use change that occurred before 1950 will be omitted from the results, but the effect of land-use change on soil C diminishes with time, so this is not anticipated to introduce a large error in the national estimates.

The correlation between the ECOSSE and CEH estimates is highly significant ($r^2 = 0.93$, $p < 0.001$; Smith & Smith 2007). A *t*-test also indicates that the mean values estimated by the 2 approaches are not significantly different, indicating that the 2 sets of estimates show similar trends. On average, the ECOSSE simulated results differ from the CEH estimates by $-0.2 \text{ kt C } 20 \text{ km}^{-2} 10 \text{ yr}^{-1}$, which is equivalent to a bias of -3% of the average size of change in soil C estimated by CEH (note the units used here reflect the 20 km^2 land-use change grid and the decadal time slice of land-use changes). This provides confidence in the simulations because these 2 very different approaches provide broadly similar results. The results do, however, differ in the detail, as evidenced by the average size of the absolute difference between the CEH and ECOSSE estimates, which is $2.9 \text{ kt C } 20 \text{ km}^{-2} 10 \text{ yr}^{-1}$, equivalent to 45% of the average size of change in soil C estimated by CEH.

Most land-use changes produce changes in soil C estimated by the 2 methods that are of a similar magnitude and sign. The ECOSSE simulations of changes in soil C due to land-use change from semi-natural to grassland show a larger loss of soil C than estimated by the CEH method. Similarly, ECOSSE simulates a

larger gain in soil C due to land-use change from grassland to semi-natural. Correspondingly, ECOSSE estimates a larger loss of soil C due to land-use change from forestry to semi-natural and a smaller gain in soil C for the land-use change from semi-natural to forestry. If these results were to be omitted from the analysis, the average size of the difference between the CEH and ECOSSE results would be reduced to 19% of the average size of change in soil C estimated by CEH. These differences occur due to processes that are included in ECOSSE associated with soil disturbance and reduced plant inputs when vegetation is immature, as described by Smith et al. (2010b, Section 2.5).

The greatest simulated losses in soil C are due to land-use change from grassland to arable, and the greatest simulated sequestration of C occurs due to land-use change from arable to grassland (Fig. 3). This result is consistent with the results reported by CEH (Milne et al. 2004) and Falloon et al. (2006). The distribution of changes in soil C simulated across Scotland for 2000–2009 due to all land-use change, conversion of grassland to arable and conversion of arable to grassland are shown in Fig. 4. The maps illustrate that the greatest losses due to conversion of grassland to arable occur in the east and south. Similarly, the greatest gains due to conversion of arable to grassland occur on the east coast. When the effects of all land-use changes are taken together, the losses of soil C outweigh the gains, resulting in a simulated change in soil C of $-810 \pm 89 \text{ kt yr}^{-1}$ across Scotland. Given a total C stock in the top 100 cm of Scottish soils of $2187 \times 10^3 \text{ kt}$ (Bradley et al. 2005), this is equivalent to a loss of soil C of $0.037 \pm 0.004 \text{ \% yr}^{-1}$ between 2000 and 2009.

3.2. Changes in soil C on different soil types

The ECOSSE simulated changes in soil C since 1950 across Scotland for C-rich soils and non-C-rich mineral soils are shown in Fig. 5. C-rich soils are defined here as soils with a C content of $>6\%$. This value for the C content of C-rich soils was selected following Smith et al. (2007a), but could equally have been set at another level if required. Between 2000 and 2009, losses from C-rich soils represent 64% of the total losses. Across Scotland, the simulated change in soil C on C-rich soils between 1950 and 2009 is -62.5 Mt , compared with -35.4 Mt on non-C-rich mineral soils. The distribution in the simulated changes in soil C stocks in C-rich soils is shown from 1950 to 2019 in Fig. 6. Note the high losses of C from C-rich soils up to 1980, when the losses decline as plant communities from the earlier land-use changes reach maturity. The losses between 2000 and 2009 from C-rich soils and non-C-rich mineral soils associated with different land-use changes are shown

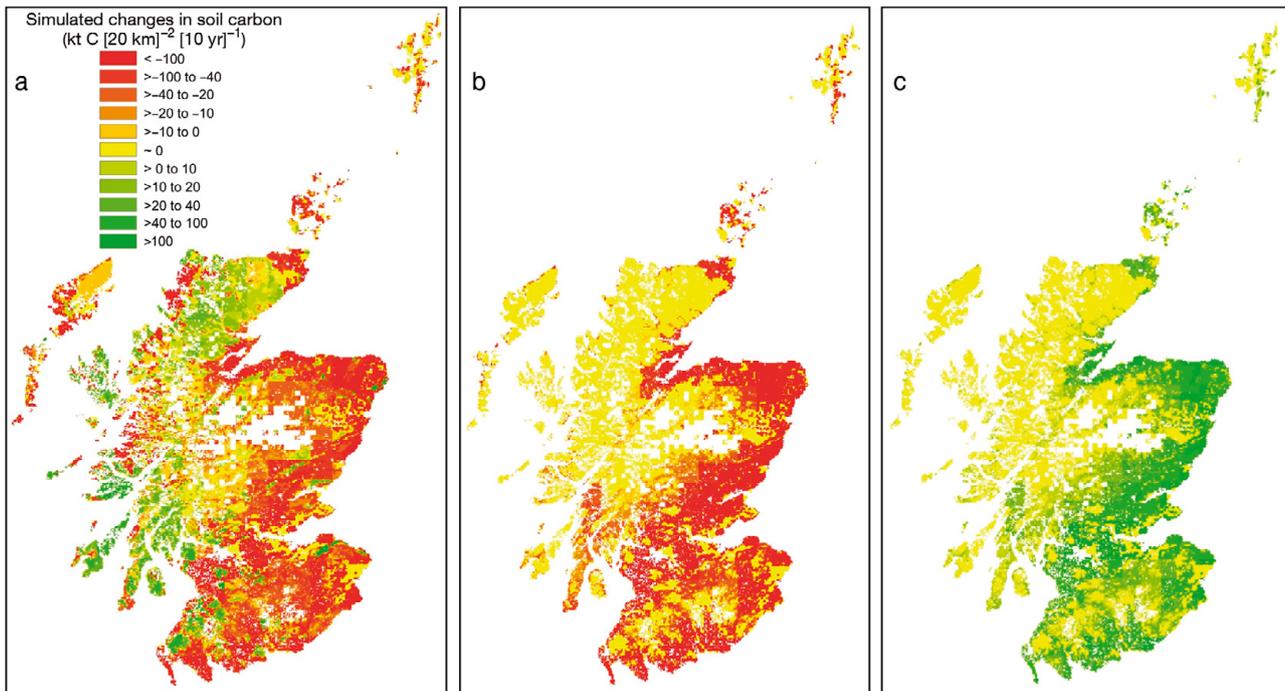


Fig. 4. Simulated change in soil C stocks due to all land-use changes occurring in Scotland 2000–2009. Change due to (a) all land-use changes, (b) conversion of grassland to arable and (c) conversion of arable to grassland. White areas: no land-use change

in Fig. 7. The results represent the emissions that occur between 2000 and 2009, but show the impact of land-use changes occurring between 1950 and 2009. The largest losses in soil C are associated with the conversion of non-C-rich mineral soil from grassland to arable, while the largest gains in soil C are associated with the conversion of non-C-rich mineral soil from arable to grassland. This reflects the higher proportion of these land-use changes occurring on non-C-rich mineral soils. Other large losses in soil C are observed due to conversion of C-rich soils from grassland to arable, and from semi-natural to grassland. Whereas the changes in soil C in non-C-rich mineral soils are largely compensatory (i.e. changes in soil C associated with one land-use change compensate for the changes in soil C associated with another), the changes in C-rich soils are not, resulting in an average loss of soil C from C-rich soils of $19 \text{ kt } 20 \text{ km}^{-2} 10 \text{ yr}^{-1}$, compared with an average loss from non-C-rich mineral soils of only $5.5 \text{ kt } 20 \text{ km}^{-2} 10 \text{ yr}^{-1}$. This emphasizes the importance of C-rich soils in any national estimates of GHG emissions. These results suggest that mitigation options to reduce losses of soil C might recommend different policies for land-use change on non-C-rich mineral and C-rich soils.

A gain in soil C is simulated due to the land-use change from semi-natural to forestry on C-rich soils (Fig. 7). However, this result may be an artifact of the approach used to derive the soil C contents used to

drive the model for afforested and semi-natural soils as described in Section 2.2, and so the results presented should be treated with caution. Because there was insufficient experimental evidence to differentiate the typical characteristics of soils under forestry or semi-natural land use, the typical soil parameters for uncul-

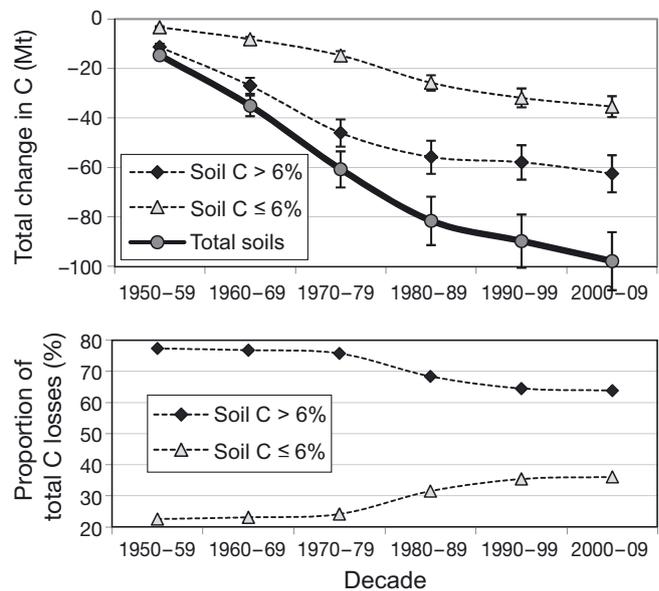


Fig. 5. Simulated total changes in C stocks across Scotland from 1950 to 2009 for different soil types as simulated by ECOSSE, assuming soil disturbance on afforestation

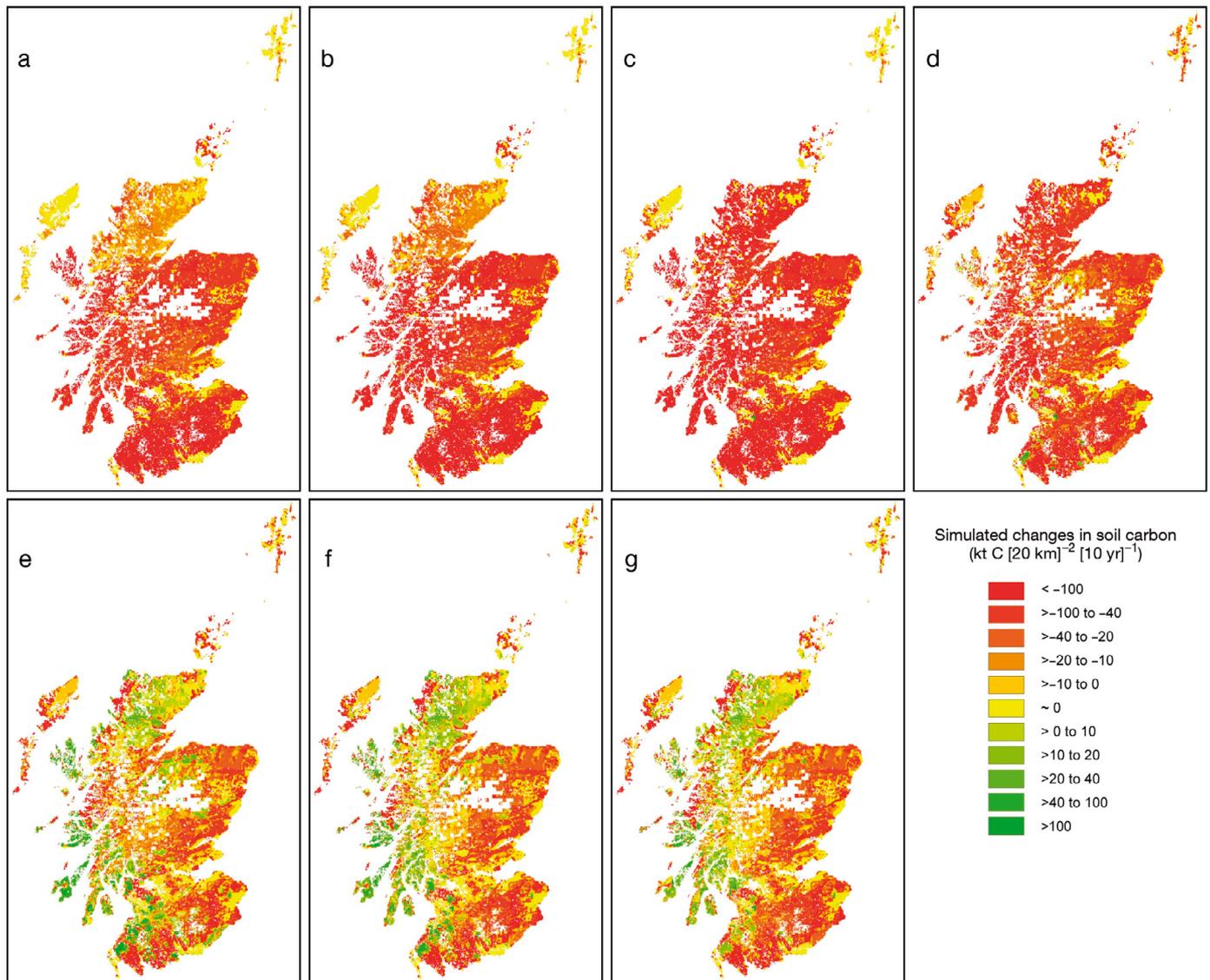


Fig. 6. ECOSSE simulations of changes in soil C stocks on C-rich soils (>6% soil C) across Scotland due to all land-use changes occurring between 1950 and 2010. Simulations assume soil disturbance on afforestation in the (a) 1950s, (b) 1960s, (c) 1970s, (d) 1980s, (e) 1990s, (f) 2000s and (g) 2010s

tivated soil types were used for soil horizons under both forestry and semi-natural land use. This neglects any impact of drainage of saturated semi-natural soils prior to afforestation. Increased respiration and release of CO₂ has been observed on draining peats in both laboratory and field experiments (Freeman et al. 1993, Moore & Dalva 1993, Funk et al. 1994, Martikainen et al. 1995). Data on litter layers were added to the surface of the forested soils, resulting in a profile that holds more total C than the semi-natural soil. Therefore, despite initial losses in soil C due to soil disturbance and reduced plant inputs on land-use change, over the long term, the land-use change from semi-natural to forestry will result in an apparent gain in soil

C. This result may or may not be correct, and more data on the C contents of forested soils are needed to confirm this result.

3.3. Predicted changes in soil C due to projected land-use and climate change over the next decade

The predicted changes in soil C due to projected land-use and climate change over the next decade are very similar to the changes simulated for 2000–2009. The overall trend is towards increased emissions, with a total additional loss of 135 kt C yr⁻¹ from the soil due to the projected scenarios of land-use change in Scot-

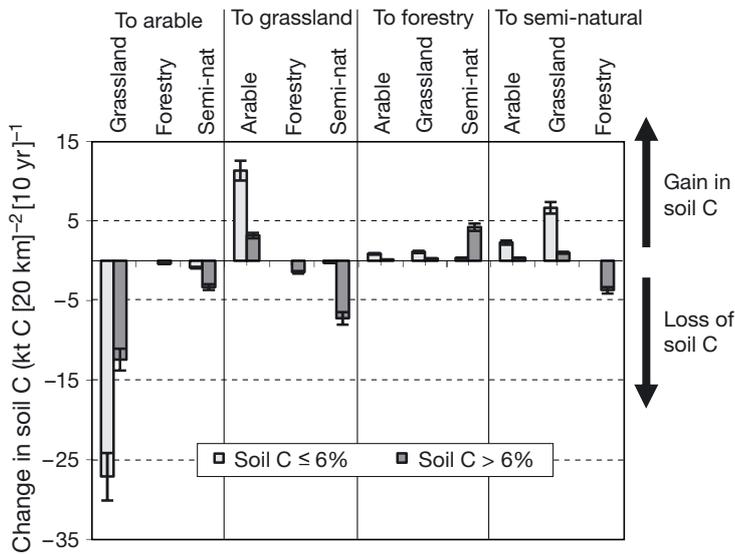


Fig. 7. Comparison of simulated changes in soil C content from 2000 to 2009 divided according to land-use change on organic and mineral soils as estimated by ECOSSE. Values are averaged across Scotland. Error bars indicate the uncertainty in the simulations calculated from the simulation of the National Soil Inventory of Scotland (NSIS) sites as the average deviation of the simulations from measurements (root mean squared deviation = 11%). Note the units used for change in soil C reflect the 20 km² grid cells and the decadal time slice used in the land-use change data

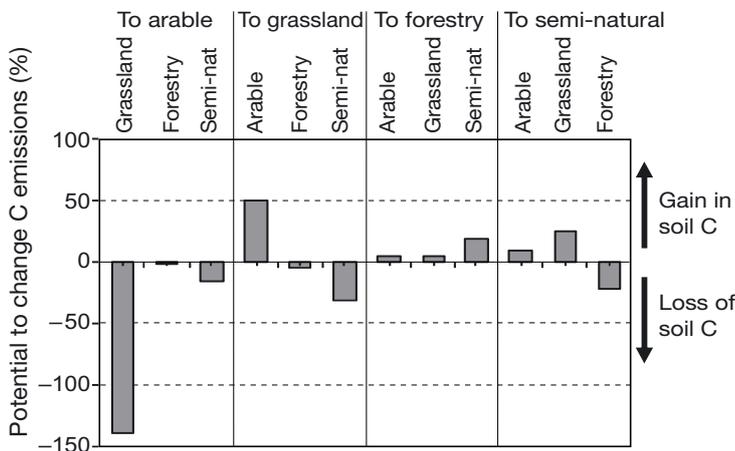


Fig. 8. The impact of land-use change on the total C emissions of Scotland over the next decade as estimated by ECOSSE

land. This is equivalent to a total projected loss of soil C of $0.043 \pm 0.005 \text{ \% yr}^{-1}$ between 2010 and 2019.

3.4. Mitigation options to reduce losses of soil C

Fig. 8 shows the potential of each land-use change to affect the C emissions of Scotland. The potential to affect C emissions is calculated from the simulated change in total C emissions that occur when the given land-use change is reduced to zero, and thus accounts

for both the area of predicted land-use change and the size of the impact of the land-use change on soil C stocks.

Reducing land-use change from grassland to arable has the greatest potential to reduce soil C losses. If the area of land converted from grassland to arable was reduced to 28% of its present rate of conversion, the soil C losses across Scotland would be reduced to zero. In other words, if, in a given grid square, the conversion of arable to grassland from 2010 to 2019 was 100 ha, reducing the area of land converted from arable to grassland to 28 ha would reduce the losses of soil C such that the net losses of soil C across Scotland would be zero. However, given the current agricultural market, such a mitigation option may be unrealistic.

Other significant losses of soil C occur due to the conversion of semi-natural land to arable or grassland. The results suggest that if policies were designed to reduce conversion of semi-natural land to arable or grassland, net losses of soil C could be reduced to 53% of the present emissions. If this were coupled with an increase in the conversion of grassland to semi-natural land by 125% of the present rate of conversion (i.e. increase 100 ha conversion of grassland to semi-natural to 225 ha), net losses of soil C would be reduced to zero. Alternatively, a 63% increase in the present rate of conversion of arable to grassland (i.e. increase 100 ha conversion of arable to grassland to 163 ha) would also result in zero net losses of soil C when coupled with the reduced conversion of semi-natural land to arable or grassland. This could equally be achieved by decreasing the current rate of conversion of grassland to arable to 77% of its present rate (i.e. decrease 100 ha conversion of grassland to arable to 77 ha).

The results also suggest that conversion of semi-natural land to forestry would sequester additional soil C. However, the national soils data used to drive the model does not presently account for the detail of changes in soil C likely to occur due to drainage and disturbance of semi-natural soils following afforestation. Therefore, in the present study, conversion of semi-natural land to forestry was not included as a potential method to sequester soil C. Further work on the C content of forested soils and the changes occurring in soil C on afforestation is urgently needed.

Similarly, the results suggest that conversion of forestry to semi-natural land results in a loss of soil C, and so this land-use change should be reduced. While a short-term loss of soil C is consistent with a reduction

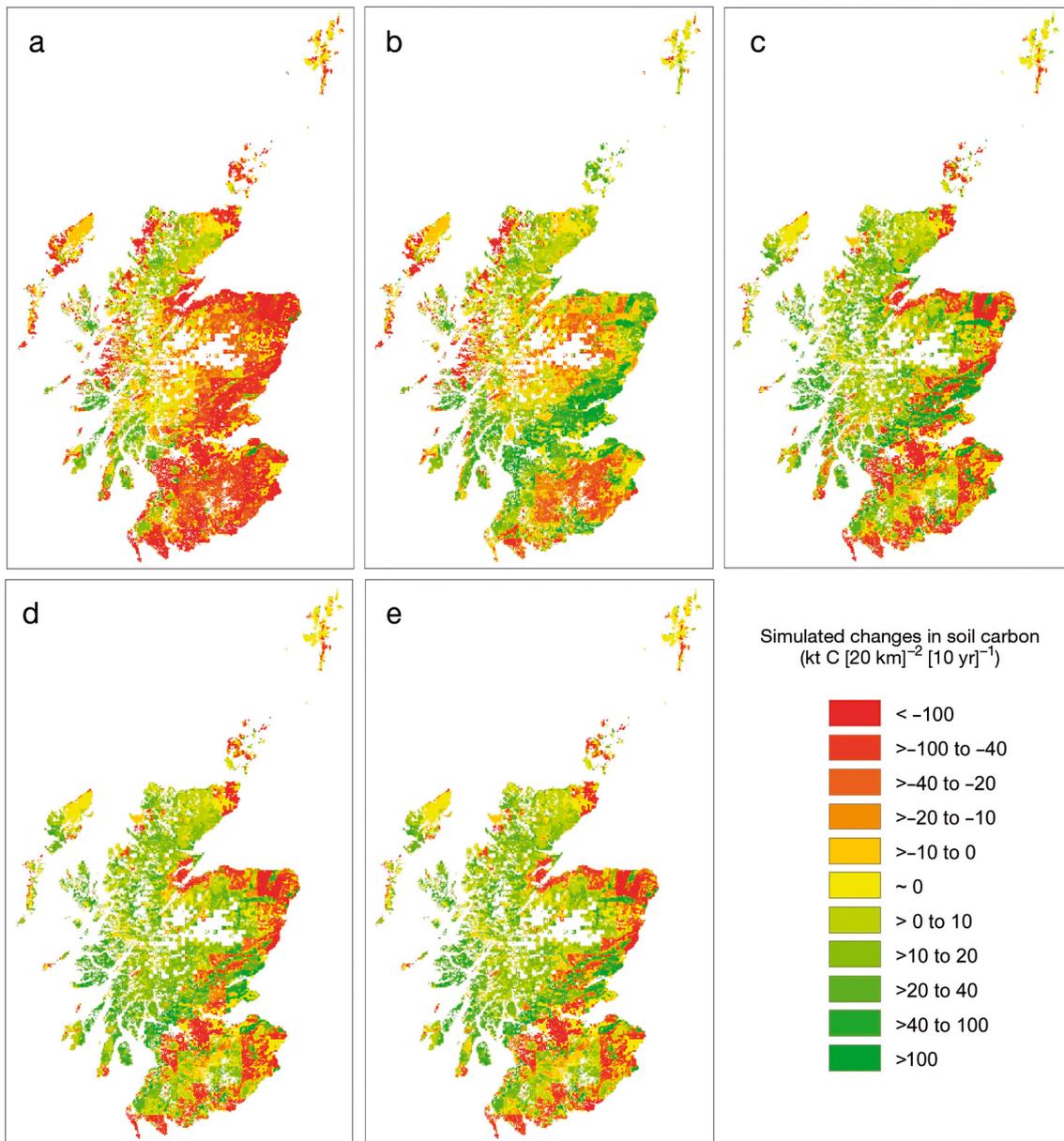


Fig. 9. Simulated changes in soil C stocks in 2010 to 2019 as predicted by ECOSSE. (a) No mitigation options applied; (b) rate of conversion of grassland to arable decreased to 28% of the present rate; (c) conversion of semi-natural land to arable or grassland stopped and conversion of grassland to semi-natural increased by 125% of the present rate; (d) conversion of semi-natural land to arable or grassland stopped and conversion of arable to grassland increased by 63% of the present rate; and (e) conversion of semi-natural land to arable or grassland stopped and conversion of grassland to arable decreased to 77% of the present rate

in soil C on disturbance of the soil, as discussed in the previous sections, this result could be an artifact of the method used to derive the data for afforested and semi-natural soils. No account is taken of the impact of possible changes in soil hydrology if drains are blocked on restoration of the land to its semi-natural state. Therefore, this result should not be used to suggest reduced conversion of forestry to semi-natural land as a mitigation measure. Again, further work is

urgently needed on the C content of afforested soils and the changes in soil C occurring on conversion of forested to semi-natural land.

These results suggest at least 4 mitigation options that would be effective tools for achieving zero net losses of C from Scottish soils: (1) reduce the predicted rate of conversion of grassland to arable to 28% of the present rate, or stop conversion of semi-natural land to arable or grassland and either (2) increase the conver-

sion of grassland to semi-natural land by 125% of the present rate, (3) increase the conversion of arable to grassland by 63% of the present rate or (4) reduce the conversion of grassland to arable land to 77% of the present rate.

The mitigation options were applied as a multiplication factor to the predicted land-use changes for 2010–2019. The impacts on the distribution of changes in soil C stocks of applying these mitigation options are shown in Fig. 9. Reducing conversion of grassland to arable land results in a reduction in the loss of soil C across the east coast and central belt of Scotland, whereas the losses on the west coast remain relatively unchanged. Stopping conversion of semi-natural land to arable or grassland and either increasing conversion of grassland to semi-natural or arable to grassland, or reducing conversion of grassland to arable land, all result in reduced loss of soil C across the west coast and smaller reductions on the east coast and central belt. These mitigation options represent simple changes to the amount of land-use change occurring. More subtle changes in land use, such as changes in the distribution of crops, management of forestry and grazing of animals, could be applied in a similar way through modification of the land-use types selected.

4. IMPLICATIONS FOR UPLAND ORGANIC SOILS

The ECOSSE model has been applied to provide national simulations of soil C turnover in organic as well as mineral soils from principles established in existing models. These models were originally developed on mineral soils but, by adding descriptions of processes and impacts that are important in organic soils, the same model can be used to simulate a wider range of soil types. This is an important facility, because under the influence of land-use and climate change, upland organic soils may become more aerobic, thus changing their basic functioning. In order to design approaches to manage such changes, it is necessary to simulate the transition states of soils undergoing land-use and climate change.

The land uses that dominate in upland organic soils are forestry and extensive grazing of semi-natural vegetation (Mobbs & Dyson 2009). However, there are insufficient data available in the SSKIB database to determine typical horizon sequences for soils under forestry and to determine their C contents. Statistical analysis of the data suggests that the C contents of soil horizons in uncultivated soils provide a good estimate of soil C contents of both forested and semi-natural soils, while higher inputs of more recalcitrant plant material (Coleman et al. 1997) lead to a deeper litter layer under forestry. This approach leads to a

higher total C content in the soil profile and litter layer of forested soils compared to semi-natural soils. However, in upland organic soils, this may not be the case.

Upland organic soils under semi-natural land use tend to be highly saturated and often require drainage during preparation for afforestation, potentially leading to large losses of soil C due to an increased rate of aerobic decomposition following drainage (Freeman et al. 1993, Moore & Dalva 1993, Funk et al. 1994, Martikainen et al. 1995). Experimental evidence into the changes in soil C contents following afforestation can be contradictory and highly site specific. Further detailed research is urgently needed to provide data on soil C contents for semi-natural and forested soils, particularly in upland areas.

If the forested soil C had been overestimated in upland organic soils compared to semi-natural soils, the simulations would tend towards a steady state with a greater soil C content under forestry than under semi-natural land use. This would then suggest that, in the long-term, the conversion of semi-natural land to forestry would sequester more soil C; if the semi-natural land use was a saturated highly organic soil, this is unlikely to be the correct result as large losses of soil C are likely to occur on drainage. An alternative simulation approach combines a dynamic model of changes in forest biomass stocks to generate soil inputs with a description of initial losses due to soil disturbance (Hargreaves et al. 2003). However, inaccuracy in the estimation of plant inputs and incomplete knowledge of the soil conditions at a national scale is likely to result in greater errors than are introduced through the use of the soil C data to initialize the model.

Land-use changes, such as from semi-natural to forestry—which is a change from a long-term uncultivated land use to a land use with slowly establishing vegetation—result in losses of soil C due to soil disturbance and reduced plant inputs when the new vegetation is immature. The parameters describing release of physically protected SOM on soil disturbance and reduced plant inputs are currently set according to the normal forestry practices used in the UK. If forestry practices were to move to minimal soil disturbance, these parameter values might change, resulting in a smaller release of soil C on afforestation of long-term uncultivated soil than is currently assumed, and a smaller short-term loss associated with afforestation. Differentiation of management practices used in upland and lowland sites might further improve the accuracy of the simulated impacts of soil disturbance and reduced plant inputs following afforestation. Research into the physical protection of SOM in organic soils and the release of SOM following different types of soil disturbance is needed to inform parameteriza-

tion of the model and to design forestry practices to reduce C losses.

Across Scotland, the simulated change in soil C from C-rich soils between 1950 and 2009 is -62.5 Mt, compared with -35.4 Mt from non-C-rich mineral soils; losses from C-rich soils between 2000 and 2009 comprise 64% of the total soil C losses. This emphasizes the importance of C-rich soils in any national estimates of GHG emissions, and the need to avoid land-use changes that result in C losses on these soils. These results also suggest that mitigation options to reduce losses of soil C might recommend different policies for land-use change on non-C-rich mineral soils versus C-rich soils.

One mitigation option that could be used in upland soils to achieve zero net loss of C from Scottish soils is to stop conversion of semi-natural land to managed grassland and increase conversion of managed grassland to semi-natural land by 125% of the present rate. Possible ways of encouraging practices to reduce soil-C losses from the agricultural sector may come about through improving implementation of existing good agricultural and environmental condition (GAEC) standards for soil protection by strengthening rural development policy. In Scotland, this could be through the Scotland Rural Development Programme (Scottish Government 2010), redesign of the European Union Common Agricultural Policy to encourage the maintenance of existing C stocks (Smith et al. 2007c), or voluntary C markets for improved C sequestration in soils (Stern 2007, Blyth et al. 2009). Reducing land-use change on upland organic soils is likely to encourage a compensatory increase in land-use change on non-C-rich mineral soils to meet the demands of the market. Without a change in consumer demand, these potential mitigation options represent an oversimplified view of reality. More realistic mitigation options should be derived by coupling these simulations to an economic analysis of the response of the market to changes in supply.

Another mitigation option that could be used on highly organic soils in Scotland is peatland restoration. Estimates of the area and severity of peatland degradation vary and estimates of the mitigation potential of restoration are also uncertain, but the International Union for the Conservation of Nature has proposed that Scotland could increase C stocks by 2.76 Mt CO_2 equiv. yr^{-1} , assuming a, possibly optimistic, accrual rate of 4.6 t CO_2 equiv. yr^{-1} (Byrne et al. 2004) and the biodiversity target of 600 000 ha restoration by 2015 (IUCN UK Committee Peatland Programme 2010). Models such as ECOSSE could have a significant role in better quantifying the potential of these mitigation targets, and in reducing the uncertainty associated with the estimates.

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