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CROPS AND SOILS REVIEW

Issues and pressures facing the future of soil carbon stocks with particular emphasis on Scottish soils

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SUMMARY

Soil organic carbon (C) plays a critical role in supporting the productive capacity of soils and their ability to provide a wide range of ecologically important functions including the storage of atmospherically derived carbon dioxide (CO_2). The present paper collates available information on Scottish soil C stocks and C losses and reviews the potential pressures on terrestrial C, which may threaten future C stocks. Past, present and possible future land use, land management practices and land use changes (LUCs) including forestry, agriculture, nitrogen (N) additions, elevated CO_2 and climate change for Scotland are discussed and evaluated in relation to the anthropogenic pressures on soil C.

The review deduces that current available data show little suggestion of significant changes in C stocks of Scottish soils, although this may be due to a lack of long-term trend data. However, it can be concluded that there are many pressures, such as climate change, intensity of land use practices, scale of LUC, soil erosion and pollution, which may pose significant threats to the future of Scottish soil C if these factors are not taken into consideration in future land management decisions. In particular, this is due to the land area covered by vulnerable peats and highly organic soils in Scotland compared with other areas in the UK. It is therefore imperative that soil C stocks for different land use, management practices and LUCs are monitored in more detail to provide further insight into the potential changes in sequestered C and subsequent greenhouse gas emissions, as advised by the United Nations Framework Convention on Climate Change (UNFCCC).

INTRODUCTION

Terrestrial carbon (C) sequestration is the process whereby atmospheric carbon dioxide (CO_2) can be immobilized by the soil and held there in a relatively permanent form, i.e. the term 'sequestration' implies a combination of both capture and storage (Chapman 2010). Soils contain significantly more C than is present as CO_2 in the atmosphere so the stability of this soil store, particularly under changing temperature and other climatic factors, is a major source of uncertainty under future climate change predictions (Frogbrook *et al*. 2009). Globally, soils contain *c*. 1500 petagrams (Pg = 1 billion tonnes) C in the top metre (Smith *et al*. 2010), about three times the amount of C in vegetation and twice the amount in the atmosphere (IPCC 2000).

Carbon dioxide emissions increased by *c*. 80% between 1970 and 2004, and made up 0·80 of total global anthropogenic greenhouse gas (GHG) emissions in 2004 (IPCC 2007). Scotland plans to reduce emissions by 50% by 2030 and by 80% by 2050 (Glenk & Colombo 2011). In the UK, policies to control the increase in atmospheric CO_2 and consequent climate change have been defined by the UK Climate Change Bill (UK Government 2008) and Scottish Climate Change Act (Scottish Government 2009). There is an urgent requirement to improve our understanding of the processes contributing to C storage in soils. This arises from the need to sequester and/or conserve C to mitigate against emissions of CO_2 and effects of global climate change (Paustian *et al*. 1997) as well as improving soil quality, as we develop

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more sustainable land management practices (Carter 2002).

Increasing populations place further demands on the need to increase food production, conservation of biodiversity and soil/water/ecosystem quality. Maintaining and possibly enhancing soil organic matter (SOM) is an important component spanning local and global scales (Miller *et al.* 2009; Ostle *et al.* 2009; Rounsevell & Reay 2009; Crute & Muir 2011; Dobbie *et al.* 2011; Glenk & Colombo 2011). Soil organic matter plays a crucial role in supporting the productive capacity of soil. It does this by providing a store of nutrients, enhancing soil structural properties, contributing to an increased capacity to store water, and enhance soil microbial processes that are critical to supporting plant growth (Carter 2002).

In the present report, published data reporting Scottish soil C stocks are collated to determine whether evidence exists to support gains or losses of C in Scotland. Current and future pressures relating to C storage and soil C estimations are summarized. The report infers evidence from local to worldwide scale issues, with particular emphasis on Scotland.

SOIL CARBON STORES AND TRENDS

Carbon stocks reported for Scotland show some variation in estimated values. Milne & Brown (1997) reported soil C stocks for Great Britain (as opposed to the UK) to be 9838 Mt C with 2890 Mt C for soils in England and Wales, and 6948 Mt C in Scotland, of which 4523 Mt C was estimated to be in peat soils. These figures equate to 0·46 of soil C in Great Britain being stored in Scottish peatlands alone. Following a recalculation of peat C, Chapman *et al.* (2009) approximated the total soil C store in Scotland to be 2872 Mt C. This figure was compiled from C stocks reported for organomineral (754 Mt C) and mineral (498 Mt C) soils by Bradley *et al.* (2005) and peatland soils (1620 Mt C) from Chapman *et al.* (2009). Better estimates of soil bulk density are keys to understanding the differences in the soil C stock estimates. Chapman *et al.* (2009) explained that in the original peat surveys of Scotland, peat was defined as soil with a proportion of <0·2 ash and a depth of organic horizon of >30 cm. Later, the Soil Survey of Scotland (Soil Survey Staff 1984) defined peat as organic layer(s) exceeding 50 cm depth from the surface; however, in Scotland there are considerable areas covered by organomineral soils (surface organic horizon of <50 cm) as well as areas comprising mixtures of peat and organomineral soil. This makes soil classification and C store estimation of broad areas difficult.

Bellamy *et al.* (2005) reported an average C loss rate of 0·6% per year for topsoils (to 150 mm depth) between 1978 and 2003, utilizing National Soil Inventory data for England and Wales. A positive relationship was reported between total C loss rate and soil C content, with soils containing >100 g C/kg having loss rates of up to 2% per year. This could be a significant detrimental loss if applicable to Scottish soils. As this relationship was found to be irrespective of land use change (LUC), it was speculated that climate change was a controlling factor. However, subsequent analyses such as Thomson *et al.* (2008) indicated that the actual loss was much smaller at *c.* 160 g/m^2 or 3% between 1978 and 2003 (Rounsevell & Reay 2009). Smith *et al.* (2007*a*) noted that the results reported by Bellamy *et al.* (2005) contradict evidence that the UK and Europe, as a whole, are net CO_2 sinks as reported by Janssens *et al.* (2003). Recent estimates suggest that the UK soil C pool is slowly accumulating C at a rate of 0·22 Mt/year in 2000, as reported to UNFCCC (Dobbie *et al.* 2011).

More recently, Chamberlain *et al.* (2010) found there to be no significant difference in topsoil C concentrations between 1978 and 2007 using data from the Countryside Survey of Scotland, England and Wales. For Scotland, mean topsoil C concentrations were measured at 239, 258 and 242 g/kg in 1978, 1998 and 2007, respectively, and mean topsoil C stocks at 619, 623 and 628 Mt, again in 1978, 1998 and 2007. These data show little difference in C stores for topsoils in Scotland between 1978 and 2007. Soil samples sent by farmers (so, primarily agricultural topsoils) to the Scottish Agricultural College (SAC) analytical laboratory between 1996 and 2006 show no discernible change in SOM concentrations (12 700 routine samples; SAC, unpublished data); neither do data from the long-term rotational experiment at Craibstone (Aberdeen), which has occasional measurements of SOM dating back to 1922 (Walker *et al.* 2007; Dobbie *et al.* 2011). Smith *et al.* (2007*a*) discussed other environmental and analytical possibilities to explain declining trends in soil C reported by Bellamy *et al.* (2005). These included changes in land management, recovery from acidification, enhanced atmospheric nitrogen (N) deposition, burning practices, fertilization and/or liming in the uplands, the calculation of C stocks, depths chosen and bulk density measurement parameters.

There is also evidence of an increase in C loss by export of soluble C in UK drainage water, which may be indicative of losses from within soil C stores. Evans *et al*. (2005) stated that dissolved organic carbon (DOC) concentrations in UK upland waters have increased by an average of 91% during the last 15 years. Yallop *et al*. (2010) investigated fluvial export of the humic component of DOC and found actual C export via this pathway has doubled over the last three decades for three South Pennine catchments. McCartney *et al*. (2003) reported an increasing trend in concentrations of stream DOC for Loch Ard, west-central Scotland from 5 mg/l in the 1980s to 16 mg/l in 2003. Worrall & Burt (2007) suggest similar trends for DOC throughout the UK. A study of the C budget of a raised bog in south-east Scotland showed that losses of DOC represented 24% of net ecosystem exchange of C (Dinsmore *et al*. 2010).

Observed seasonal variation and rising long-term trends of DOC have been linked to temperature and climate change (Lumsdon *et al*. 2005; Bonnett *et al*. 2006), water fluxes (Buckingham *et al*. 2008) and recovery from acidification (Evans *et al*. 2006; Monteith *et al*. 2007; Clark *et al*. 2010). However, Evans *et al*. (2005) stressed the difficulty in isolating the driving factors contributing to increases in DOC with variability between sites being heavily influenced by the magnitude of spatial and temporal factors as discussed by Clark *et al*. (2010).

Large terrestrial C stocks in Scotland (in relation to other parts of the UK) and reports suggesting significant losses of soil C in some parts of the UK highlight Scotland as a crucial potential store or source of C. Measuring and understanding soil C trends are of vital importance in relation to Scottish soil quality and soil resources, and more widely in relation to efforts to reduce GHG emissions. Current and future anthropogenic and environmental pressures will bear heavily on attempts to conserve soil C in Scotland as well as on a global scale.

Difficulties in determining soil carbon stocks and trends

Variation in stock estimates for Scotland (and in other areas) stem, in general, from difficulties in accurately measuring and quantifying soil heterogeneity, soil depths (particularly for deep peats), estimating bulk densities and loss on ignition, which are discussed below. Scottish soils are highly variable due to gradients in climate, topography, parent material and land use. Complex interrelated processes of primary production, C inputs to the soil system, decomposition and mineralization processes, the adsorption, physical protection and stabilization of organic matter and C losses, all operating on a continuum of timescales and varying spatially, lead to considerable natural heterogeneity.

Soil type and depth is an important parameter. Depending on the objectives of a study, different depths are often sampled. There is much spatial variation of total soil depths over landscapes, and biogeochemical processes can change with depth. The amount of soil organic C below 100 cm is not well known. However, by averaging across all deep blanket peat soil in Scotland it can be suggested that a maximum addition of 35 Mt C can be added to current Scottish peat soil estimates (Morison *et al*. 2010). Without knowing the total depth of soil, it is difficult to know whether the whole-profile depth has decreased due to destabilization and/or erosion processes resulting in a loss of total (whole-profile) C stocks. Owing to the dynamic nature of surface peat, relying on 'topsoil' measurements of peat for whole-profile assumptions of C content can be misleading if no quantification of deeper (more stable) peat C content is made.

There is some debate as to which bulk density values to use if not measured directly when estimating soil C stocks, as it can vary both spatially and temporally (Lee *et al*. 2009). Temporal variation is not taken into account in inventory reporting (Schrumpf *et al*. 2011) and can be related to land management, such as animal stocking rates and use of machinery as discussed in Sonneveld & van den Akker (2011). In a study comparing two moorland sites, Glensaugh (north-east Scotland) and Plynlimon (mid Wales), comprising organomineral and peat soils, Frogbrook *et al*. (2009) found bulk densities decreased with average depth down the organic layer, which indicates non-uniformity for soil bulk density. It may be argued that the measurement of bulk density is even more important in peat soils than in mineral soils because organic soils are more vulnerable to compaction, potentially resulting in high variation in bulk density. However, the risk of compaction is less than on more managed soils because peat is generally used for extensively grazed systems and consequently less exposed to compaction. Therefore soil management is

Table 1. *Estimates of Scottish land cover area (000 ha) for Scotland (S) and Great Britain (GB) for 1990, 1998 and 2007 reported by Countryside Survey (2007*a*) summary data (www.countrysidesurvey.org.uk/data-access). The proportion of GB land cover sited in Scotland is also presented*

	Area (000 ha)								
Land cover	S-1990	GB-1990	Scottish proportion of GB total 1990	S-1998	GB-1998	Scottish proportion of GB total 1998	S-2007	GB-2007	Scottish proportion of GB total 2007
Bog	1922	2050	0·94	2039	2222	0·92	2044	2232	0·92
Coniferous woodland	913	1239	0·74	1030	1386	0·74	956	1219	0·78
Improved grassland	815	4619	0·18	831	4251	0·20	907	4494	0·20
Dwarf shrub and heath	1007	1436	0·70	912	1299	0·70	894	1343	0·67
Arable and horticultural	593	5025	0·12	618	5067	0·12	534	4608	0·12
Neutral grassland	429	1669	0·26	430	2007	0·21	461	2176	0·21
Mixed broadleaf and yew woodland	284	1343	0·21	229	1328	0·17	251	1406	0·18
Fen, marsh, swamp	289	427	0·68	261	426	0·61	238	392	0·61
Built up and gardens	150	1266	0·12	153	1279	0·12	153	1323	0·12
Calcareous grassland	36	78	0·46	28	61	0·45	26	57	0·46

a key factor to consider in the variability of soil bulk density.

Smith *et al.* (2007*a*) showed that if the equations used by Bellamy *et al.* (2005) to predict bulk density were applied to Scottish and Irish peat, they consistently over-predicted bulk density. Smith *et al*. (2007*b*) and Milne & Brown (1997) also found that care needs to be taken when selecting an assumed bulk density (if not directly measured) as this can lead to large errors in estimating C stocks. Estimates of Scottish soil C pools in Smith *et al*. (2007*b*) increased by 30% when including the combination of organic material below 1 m depth and better estimates of bulk density.

An important consideration to be made when assessing the impacts of LUC upon soil C storage is time. Following changes to land use or management practices it may take decades or centuries for soils to reach a new equilibrium (if at all), resulting in long-term consequences. Powlson *et al.* (2011) highlights issues surrounding possible over- and underestimation of C stocks and trends over time due to the lack of long-term studies, i.e. misunderstanding the practices that lead to genuine C increases, an ignorance of the fundamental limitations of C sequestration and that the impacts of land management practices, which are beneficial for soil organic C, often overlook other GHG fluxes. The time frame required for a state of reasonable equilibrium to be achieved depends inevitably on the extent to which land use has been modified, the type of management practices applied and the sensitivity of the environment. The importance of these has been acknowledged by the IPCC with the introduction of the Good Practice Guide for Land Use, Land-use Change and Forestry (LULUCF) programme (IPCC 2006). Guidance is provided for estimating, monitoring and reporting C stocks through LULUCF, although these are somewhat controversial due to assumptions included (for example there being no change in long-term grassland C). In relation to Scottish soils, Table 1 suggests there have not been any considerable changes in dominant Scottish land cover for categories assessed by the Countryside Survey (2007*a*). However, there are still concerns regarding land management such as those relating to agricultural and forestry practices and development of windfarms (and any other soil-disturbing developments) as a result of increased demand for food and/or energy.

The quantification of changes in soil organic C stocks in grassland needs to take into account both organic C content and bulk density (Sonneveld & van den Akker 2011). Wellock *et al.* (2011) and Chapman *et al.* (2009) highlight the need for exhaustive data collection of bulk densities (and peat depths) to improve current estimates of C stocks in Scotland.

PRESSURES ON SOIL CARBON STOCKS

Both environmental and anthropogenic drivers influence C cycling and sequestration in soils. All processes relating to soil C dynamics are inextricably linked but can include climate, climate change, land use management, LUC, pollution, soil erosion and excavation (van Camp *et al*. 2004). Drivers of change can range in intensity as well as being spatially and temporally variable. Pressures upon Scottish soil C stocks are not restricted to Scotland alone and can be inferred from a variety of studies (Rees *et al.* 2011).

Land use and land use changes: implications for carbon loss and/or storage

Land management practices are intricately linked to soil processing and can determine whether soils become net sources or sinks (or maintain an equilibrium) of soil C and CO_2 emissions to the atmosphere. Effective management of forest and agricultural soils can benefit CO_2 removal and the sequestration of C to soils, which are allowable activities under Article 3·4 of the Kyoto Protocol (Smith 2004*a*). However, mechanisms currently reported to the Kyoto Protocol by the UK include forest management and soil C under bioenergy crops only (Smith 2004*a*), although this may be amended in the future.

Changes in land use add CO_2 to the atmosphere in two principal ways: (a) the release of C from biomass through burning or decomposition and (b) release of C following cultivation due to enhanced mineralization brought about by change in soil moisture, improved oxygen supply and temperature regimes and low rate of return of biomass to the soil (Lal *et al*. 1998). Under some land use types (e.g. forest and grasslands) SOM will tend to accumulate, but a significant proportion of this SOM can be quickly lost in certain circumstances such as cultivation (Foley *et al.* 2005) or the occurrence of wildfire (González-Pérez *et al.* 2004). Schimel *et al.* (2001) estimated that LUC emitted 1·6 Pg C/year globally during the 1990s. In a meta-analysis study, Guo & Gifford (2002) estimated the percentage loss of soil C through LUC on an international scale. The largest losses (59%) were calculated for the conversion of pasture to cropland and least destructive was the conversion of pasture to plantation (10%). Land use changes that result in gains of soil C were also noted, the most prevalent being the conversion of cropland to secondary forest (53%) and the least through changing native forest to pasture. Dawson & Smith (2007) show that the conversion of arable land to a ley-arable rotation could lead to a gain of 1·6 t C/ha/year, whereas cultivating peatlands could result in a net loss of 2·2–5·4 t C/ha/year.

The Scottish Soil Framework (Anonymous 2009) reports that an estimated 6·6 Mt CO_2-equivalents were emitted from land converted to cropland in Scotland, whereas the conversion of cropland to grassland removed 2·8 Mt CO_2-equivalents for 2006. A Countryside Survey (2007*b*) report on LUC in Scotland between 1998 and 2007 showed an increase in improved grassland (7%) and broadleaf/mixed woodland (9%) and decreases of arable/horticultural (14%) and conifer land area (7%). These changes present both positive and negative consequences for soil C stores.

The management of agricultural soils: pressure of crop yield *v.* carbon sequestration and how this relates to Scotland

In 2009, agriculture contributed 0·103 of GHG emissions in Europe (Eurostat 2011). Towers *et al.* (2006) reported that for Scotland, 0·12 of a total 64·7 Mt CO_2-eq originated from the agricultural sector in 2003, excluding removals from land-use, LUC and forestry (LULUCF) activities. This highlights agriculture as a key sector for evaluation in order to minimize C loss and increase C storage and sequestration of CO_2.

In a recent assessment of the European C cycle, Schulze *et al.* (2010) reported that forests, grasslands and sediment C sinks are offset by GHG emissions from croplands, peatlands and inland waters. Studies have suggested that the rate of C loss from European croplands is less than earlier reports suggest. A European assessment of C loss from cropland systems (Janssens *et al.* 2003) indicated that losses were occurring at a rate of 90 g C/m^2/year. However, more recently, Ciais *et al.* (2010) inferred from a compilation of inventories a mean loss of 17 g C/m^2/year for European soils. Within Europe (EU-15) it has been estimated that the C storage potential of cropland is *c*. 90–120 Mt C/year (Smith 2004*b*). For croplands in Great Britain, the mean GHG mitigation potentials for all cropland management practices range from 17 to 39 Mt CO_{2-eq} per 20 years (Fitton *et al.* 2011). Towers *et al.* (2006) reported that the National Soils Inventory for England and Wales showed a small decrease in C content in arable soils over recent years. This decrease, however, was not considered to be at a level to cause concern (Loveland & Webb 2003).

There are many reasons why arable systems may lead to an overall loss of soil C. These include C input limited mainly to growing seasons, and soil disturbance through management practices such as harvesting, residue removal and tillage (Smith 2008). There is a distinction between arable and grassland in terms of management practices applied and biogeochemical responses in terms of C gains and losses. Under similar conditions, permanent grasslands typically have higher soil organic C contents than arable crop rotations, because (i) they receive higher residue inputs, (ii) relatively more C is deposited belowground and (iii) decomposition is slower due to the absence of tillage-induced aeration and due to stronger soil aggregation (Paustian *et al.* 1997; Ammann *et al.* 2007). Also, grassland systems generally have permanent vegetation cover compared to most arable systems that have some windows of exposure. Rees *et al.* (2005) and Soussana *et al.* (2004) also report that soil respiration is increased in soils exposed to cultivation as a consequence of the accelerated oxidation of labile C.

It is widely reported that cultivation and tillage of cropland leads to increased CO_2 emissions and reduced soil C content (Loveland & Webb 2003; Smith 2004*b*; Rees *et al.* 2005; Chapman 2010). The disturbance of soil aggregates through tillage and ploughing promotes aeration of soils and exposes previously protected SOM to microbial breakdown, oxidation and weathering, as well as influencing soil structure, soil temperature and water regimes. Conant *et al.* (2007) estimates that 0·11 of C can be lost following a single tillage event, therefore having a detrimental effect upon C pools in these soil systems under annual tillage. In a recent review, Soane *et al.* (2012) suggest that soils with no-till applied for *c.* 5 years have an increased organic matter content and aggregate stability, especially near the surface. Smith (2004*b*) reports a realistic soil C sequestration potential for Europe (EU-15) of 2·4 Mt C/year (zero till) and <2·4 Mt C/year (reduced till) by 2012. A similar practice is conservation tillage, where plant residues are left on the soil surface to conserve water and reduce soil erosion (Chapman 2010). Piñeiro *et al.* (2010) suggest three mechanisms through which grazing can alter soil organic C: changes in net primary production, changes in N stocks and changes in organic matter decomposition rate. Reports of the impact of increasing grazing pressure on soil organic C varies, with Leifeld & Fuhrer (2010) reporting an increase and Medina-Roldán *et al.* (2012) reporting no change.

Intensive management and the enhancement of machinery and technologies have resulted in more efficient removal of agricultural residues from fields, resulting in less potential C input to the soil system. It has been estimated by Lal *et al.* (1999) that over 22 million t C can potentially be sequestered in US soils from crop residues. Set aside land and field margin management (grass margins, hedgerows, tree strips) can increase vegetative biomass by $2{\cdot}8 \times 10^3$ t C/ha (Falloon *et al.* 2004). Smith (2004*b*) summarizes C sequestration potential estimates for various agricultural practices (Table 2).

Fertilization of cropland can lead to increased yields and therefore potential C input through residues. In a global-scale meta-analysis of 257 studies, Lu *et al.* (2011) showed that an increase in N addition increased fresh organic C input (the litter pool) by 20·9%. Results by Lu *et al.* (2011) indicated that soil C did not change significantly in non-agricultural ecosystems in response to N addition, but increased by 3·5% in agriculture.

The rate of C accumulation is often higher in fertilized fields, but this carries a C 'cost' that is seldom assessed in the form of CO_2 emissions during the production and application of inorganic fertilizer (Schlesinger 2000). However, Khan *et al.* (2007) reported that long-term use of synthetic N fertilizer to promote yields has resulted in a hidden cost to soil resources in the form of net losses in SOC and residue C inputs. In a similar study, Mulvaney *et al.* (2009) found that fertilizer N depletes SOM by promoting microbial C utilization and N mineralization, and stressed that there is an immediate need for scientific and technological advances in input efficiencies. Khan *et al.* (2007) calls for a substantial reduction in fertilization beyond crop N requirements by shifting from yield- to soil-based N management, ideally implemented on a site-specific basis to limit or reverse organic matter loss in arable soils. As discussed by Mulvaney *et al.* (2009) options for reducing C loss may include matching the fertilizer input to crop requirements (quantity and synchronizing time when input is required), reducing reactive-fertilizer input and diversification through the use of legume-based crop rotations. Nevertheless, several reviews and meta-analyses (Hyvönen *et al.* 2007; Nave *et al.* 2009; Janssens *et al.* 2010) showed that N fertilization increased soil C storage slightly in forests with limited sample sizes (Lu *et al.* 2011).

Establishing a balance between optimal yield and minimal C loss is difficult and varies temporally and spatially. In a grassland study, Jones *et al.* (2006) applied three organic (sewage sludge, cattle slurry and poultry manure) and two mineral (NH_4NO_3 and urea) fertilizers for 2 years (south of Edinburgh, Scotland) and showed that organic fertilizers were able to

Table 2. *Estimated carbon (C) sequestration potentials for 2012, limited only by availability of land, biological resources and land-suitability. Data taken from Smith (2004b)*

Agricultural practice	Soil C sequestration potential (t C/ha/year)	Total soil C sequestration potential for EU15 (Mt C/year)	Realistic soil C sequestration potential for EU15 (Mt C/year) by 2012
Zero tillage	0·4	24·4	2·4
Reduced tillage	< 0·4	< 24·4	< 2·4
Set-aside	< 0·4	2·4	0
Permanent crops	0·6	0*	0*
Deep root crops	0·6	0*	0*
Animal manure	0·4	23·7	*
Cereal straw	0·7	5·5	*
Sewage sludge	0·3	2·1	*
Composting	0·4	3	3*
Improved rotations	> 0	0*	0*
Fertilization	0	0	0
Irrigation	0	0	0
Bioenergy crops	0·6	4·5	0·9
Extensification	0·5	11	*
Organic farming	0–0·5	3·9	3·9
Convert cropland to grassland	1·2–1·7	8·7–12·3	0
Convert cropland to woodland	0·6	4·5	4·5

* Refers to high uncertainty.

increase C storage and soil respiration after a 6-year period. When considering the global warming potentials it was concluded that C sequestration outweighed N_2O emissions in the study (Jones *et al.* 2006). Other aspects of management such as the use of rotational grass, effects of tillage and application of organic manures and slurries further obscure relationships between C sequestration and nutrient input that are observed in non-managed soil systems. It is worth noting that N inputs through fertilizer additions induce localized effects that are dependent on farm type and management applied, whereas atmospheric N deposition effects are more spatially widespread.

Organic farming is often considered to actively promote the natural build up of SOM through appropriate management strategies in order to maintain and maximize soil quality and crop growth. However, there are limited robust data on the impacts of organic farming on GHG emissions or C sequestration (and in connection to CO_2 reduction, non-C GHG emissions and C-sequestration simultaneously). A review by Watson *et al.* (2008) highlights that there is increasing recognition of the value of trans-disciplinary approaches in agricultural science which means that research in organic and conventional systems may become more similar in the future. A recent analysis of 68 published data sets comparing organic and conventional farming by Leifeld & Fuhrer (2010) suggests that there is not sufficient evidence to support claims that organic farming increases soil organic C. Results showed no consistent differences between organic and conventional farming in comparative experiments. It is vital that future comparative research is grounded in an improved understanding of the nature of farming systems themselves, allowing more valid comparisons to be made (Watson *et al.* 2008).

The Scottish Government (2011) June Agricultural Census shows 5·63 million ha of land on agricultural holdings, a large area of which is rough grazing (0·55) and grassland (0·24) with the remaining land constituting crop and fallow land (0·11) and land taken up by woodland (0·08) or 'other' (0·02 relating to roads, yards, scree, buildings, etc.). In Scotland there has been little annual variation in land area occupied by agricultural holdings over the last 10 years (Scottish Government (2011). However, of vital importance to Scotland's soil C store is the type of land cover and type of agriculture. Scottish Government (2011) and Countryside Survey (2007*a*) data (Table 1) show that in relation to Great Britain, Scotland has a large proportion of sensitive land areas such as peat and bog land under low intensity management. These land areas can potentially be soil C stores as opposed to

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Table 3. *Area (thousands of hectares) occupied by woodland area in Scotland, England, Wales, Northern Ireland and UK combined as reported by Forestry Commission statistics (2010). The calculated proportion of UK woodland situated in Scotland between 1921 and 2010 is also shown*

Year Total land area (1 000 000 ha)	Scotland 7·8	England 13·0	Wales 2·0	Northern Ireland 1·4	UK 24·3	Scottish proportion of UK total
1924	435	660	103	13	1211	0·36
1947	513	755	128	23	1419	0·36
1965	656	886	201	42	1784	0·37
1980	920	948	241	67	2175	0·42
1995–1999	1281	1097	287	81	2746	0·47
2010	1343	1130	284	88	2846	0·47

sources. However, this critically depends on future management practices and whether current soil C stores can be effectively conserved while minimising C loss (predominantly as CO_2) and maintaining the socio-economic value of the land.

Peatland restoration

The majority of the UK's peatland resource is damaged or deteriorating, through drainage, peat cutting, fire and the effect of livestock, releasing C as a result. Climate change is likely to cause further deterioration of damaged peatlands with increased erosion, C loss, floods and risk of wildfires (Bain *et al*. 2011). The restoration of degraded peat soils through afforestation, managed burning, drainage, drain-blocking, grazing removal and revegetation are options to consider. Worrall *et al*. (2009) estimated that afforestation, drain-blocking, revegetation, grazing removal and cessation of managed burning would bring a C benefit, whereas deforestation, managed burning and drainage would bring a disbenefit. However, as discussed previously, data availability for C content and losses in peatlands is limited. Under Kyoto Protocol reporting (IPCC 2006), unmanaged peat soils are not included. However, the second Kyoto Protocol commitment phase in GHG reporting from 2013 will potentially allow accounting of peatland rewetting and conservation (Joosten 2011).

In Scotland, peat restoration programmes already exist, for example in The Flow Country (Caithness and Sutherland, Scotland), an area that holds >0·10 of the UK's blanket peatland bog, storing >400 million t C (Cris *et al*. 2011). A peat restoration scheme has been implemented by the RSPB, the Nature Conservatory council and the UK Peatland Restoration Programme to address peatland damage and destruction due to drainage and forest plantation over the last 30 years (Cris *et al*. 2011). In addition, The Scottish Government (2012) recently announced (October 2012) that it will be supporting peatland restoration programmes in Scotland with plans to contribute £1·7 million funding. This may encourage future research and monitoring of peatlands that are vulnerable to C loss and assist in soil C conservation.

Forests: carbon sequestration of established woodlands and the implications for future plantations

In recent years, one of the most prominent LUCs in Britain has been the increase in afforestation, the majority of which has occurred in Scotland, as discussed by Mather & Murray (1988) and Forestry Commission (2010*a*). Scotland has the largest area of woodland (1343 000 ha) in the UK, occupying a greater relative proportion of the land area than in England, Wales or Northern Ireland (Table 3) (Forestry Commission 2010*a*). Cannell *et al*. (1999) describe UK forest trees and litter to be a C sink at a rate of 2·1 Mt C/year, forest products 0·5 Mt C/year and forest soils to sequester 0·1 Mt C/year. There are native woodland expansion schemes under way in Scotland (Chapman *et al*. 2003) with the Scottish Government planning to create 100 000 ha of new woodland over the period 2012–22 (McRobbie *et al*. 2012). It is estimated that this increase within the forestry sector will deliver annual C savings of 0·8 Mt C by 2015 and 1 Mt C by 2020.

Although growing trees sequester atmospheric CO_2, new afforestation necessitates a change of land use, which can impact upon soil and biomass C and may negate the rationale for afforestation. It has been shown that afforestation causes an initial loss of soil C, therefore impacting on GHG fluxes and DOC release (Zerva & Mencuccini 2005). Hargreaves *et al*. (2003)

found that newly drained peatland (2–4 years after ploughing) emitted 2–4 t C ha/year, but when ground vegetation re-colonized the peatland became a sink again, absorbing *c*. 3 t C ha/year 4–8 years after tree planting.

Grieve (2001) showed that significant decreases in pH and the quality and turnover of OM occurred in Scottish uplands following conifer afforestation of first generation plantations of the mid- to late 20th century. Wilson & Puri (2001) investigated soil properties of an ancient (8000 year old), semi-natural Scots pine woodland and a moorland site at Abernethy Forest, Scotland. Results showed thicker organic horizons and more C accumulation in the forest soils, with these soils having more capacity to sequester C and therefore potentially operating as C sinks. Conversely, Chapman *et al*. (2003) examined soil parameters, along three parallel transects across a moorland forest boundary at the southern edge of Abernethy Forest, Scotland. It was found that the soil C pool was lower (and more decomposed) under Scots pine forest plots in comparison to moorland. Chapman *et al*. (2003) concluded that at the Abernethy site, forest expansion may have resulted in some loss of soil C that would have been partly offset by increases in above ground C.

There is a distinct difference between woodlands managed for timber and those receiving little/no management. Felling for timber production is a direct loss of C in the form of above ground biomass and may result in soil disturbance and erosion, causing further loss of C. However, modern forestry practice standards aim to minimize soil disturbance, maintaining tree cover (as opposed to patch clear fell) and where possible the use of natural regeneration for successor tree crops, with the purpose of conserving above and below ground C stocks during plantation periods.

The Scottish Forestry Strategy (Scottish Executive 2006) describes actions undertaken in Scotland to support the minimizing effects of the timber industry while maximizing the economic potential of Scotland's timber resources through promoting the use of timber as a renewable, versatile raw material. The Land Use Strategy for Scotland (2011) notes that without additional plantings by 2020 the net amount of C sequestration by forestry will fall and that to sustain the contribution from forestry an increase of woodland creation is required at a rate of at least 10 000 ha/year. In addition, the Scottish Government has set a target of creating 100 000 ha of new woodland cover between 2012 and 2022 (McRobbie *et al*. 2012). The forestry sector of Scotland is committed to assisting soil C sequestration through delivering on a series of targets outlined by the Forestry Commission (2010*b*). This includes the creation of C sinks (through woodland creation and improved controls on permanent woodland removal), restoring and expanding lost habitats (native woodland) and helping to facilitate ecological adaptation.

Nitrogen additions and carbon storage

Additions of reactive N to soils have risen sharply over the past century (Erisman *et al*. 2008), principally as a result of N use to support intensive agriculture (Eggleston & Irwin 1995). Across Europe a potential relationship between N deposition and C sequestration can be seen (de Vries *et al*. 2006, 2009). Where N deposition is low in Northern Europe, the C sequestration is also small, whereas in Central and Eastern Europe both C sequestration and N deposition are high, reported by Jandl *et al*. (2007).

Recent studies have shown that increases in N addition to terrestrial environments are associated with increases in C sequestration, since biomass production is closely linked to N availability (Chiti *et al*. 2007; Jandl *et al*. 2007; Magnani *et al*. 2007; de Vries *et al*. 2009). Michel & Matzner (2002) and Magill & Aber (2000) showed that higher N contents (lower C:N ratios) slowed decomposition in the later stages, leading to OM stabilization. Hagedorn *et al*. (2003) reported that preservation of old and humified SOM under elevated N deposition might be a process that could lead to an increase in C storage in the long-term. A meta-analysis by Janssens *et al*. (2010) suggested that N deposition can impede OM decomposition, thus stimulating C sequestration in temperate forest soils (where N is not limiting microbial growth).

Elevated carbon dioxide, climate change and carbon storage

Elevated CO_2 has been shown to increase the ability of ecosystems to sequester C in some European forests and forested wetlands (Dawson & Smith 2007) and Canadian wetlands (Turunen *et al*. 2004). The evidence for elevated CO_2 effects is still ambiguous as an increased net primary productivity response may be offset by stimulated decomposition and soil respiration.

Temperature was determined as a controlling factor of soil C mineralization rates in a study by Dalias *et al*. (2001), with material produced from decomposition at higher temperatures being more recalcitrant than that at lower temperatures. The temperature sensitivity of SOM fractions was reviewed by von Lutzow & Kögel-Knabner (2009) and found conflicting relationships were found. For example, Trumbore *et al.* (1996) showed the decay rate of labile SOM to be very temperature-sensitive but this was not the case for stable SOM. In contrast, Fang *et al.* (2005) revealed similar responses to temperature changes for both labile and resistant pools of SOM. Plante *et al.* (2010) concluded that despite there being a general consensus that a warming climate will accelerate soil C mineralization, the debate over the relative temperature sensitivity of labile *v.* recalcitrant SOM has not been fully resolved.

Temperature and rainfall influences C input (photosynthesis), decomposition (microbial activity) and C loss (GHG emissions and DOC export). In areas of cool, moist climates, soil decomposition is retarded and typically promotes soil C accumulation. Scottish soil C is heavily influenced by east–west rainfall and north–south gradients in temperature, resulting in the highly organic soils (such as peats) found in the west and north regions of Scotland. Therefore climate is an important factor to consider when determining the soil's C equilibrium at present and in the future.

The pressure on soils as socio-economic resources and as carbon pools

Other land management practices that occur in Scotland and the UK in general and cause direct loss of above and/or below ground C stores are the excavation of peat resources and vegetation burning. Dawson & Smith (2007) state that localized management practices associated with peatlands show the export of CO_2, methane and DOC are usually higher from both peats and associated surface waters following disturbance from peat extraction or moorland burning. The use of soils as a raw material represents a loss of resource, which can be considered permanent in the timeframe of human life spans (Haygarth & Ritz 2009). Wind farm projects have also raised concerns for soil C stocks in Scotland where LUC and disturbance or drainage of peatland occurs (Nayak *et al.* 2008).

The conservation of peatland and productive arable land in Scotland is currently a challenge and will remain so in the future due to increased pressure on Scottish resources. This will encompass the protection of food security, the agricultural economy, energy resources and natural assets in future years through a changing environment and economic market. In terms of food security the management and productivity of arable soils is a primary topic outlined in current strategies such as the Scottish Soil Framework (Anonymous 2009) and the Scottish Forest Strategy (2006) and will no doubt continue to be in the future. Crute & Muir (2011) concluded that there is no simple overarching solution to the challenge of delivering increased productivity from terrestrial (or aquatic) food production systems. In particular, this is overlaid by the absolute necessity to deliver improved efficiency, both in terms of resource use and environmental impact.

Attaining a balance between GHG emissions, conserving C pools and maintaining economically viable agricultural output is a complicated but necessary objective for the future. However, there is some support from the Common Agricultural Policy (CAP), the new Rural Development Regulation for the programme period 2007–2013. Also, the 'Good Agricultural and Environmental Condition' (GAEC) operates within cross-compliance requirements and good practice guidance outlined by the ECOSSE project Smith *et al.* (2007*b*).

FUTURE OF SCOTTISH SOIL CARBON

Miller *et al.* (2009) highlighted potential issues to consider in relation to Scottish terrestrial resources. These include the possible conflict between biofuels and renewable energy proposals (such as the construction of windfarms) and agriculture as a result of the growing emphasis on both energy and food security. As Miller *et al.* (2009) discussed, biofuel policies provide incentives such as bioenergy consumption quotas and tax reductions for the conversion of agricultural land from food to bioenergy production. However, coinciding with these incentives are recent changes in agricultural policy such as the abolition of set-aside, which aims to increase land availability for food production, resulting in potential land use competition.

Pressures exerted on production in countries such as Asia, Latin America and Africa due to population growth and the effects of climate change have increased the demand for UK agricultural goods (Rounsevell & Reay 2009). This outlines potential pressures on future food security not just in Scotland but across the UK. As discussed, the proposed increase in afforestation, land for biofuel, agroforestry, biochar, windfarm projects and urbanization may result in high competition for land use in Scotland. Other mitigation options suggested as suitable for the agricultural sector in the UK include the development and use of more drought-resistant crop varieties, improved water storage and use efficiency, changes in soil management and behavioural changes such as altered sowing and harvest dates, double cropping and the avoidance of mechanized cultivation on waterlogged soils (Rounsevell & Reay 2009).

The Agriculture and Climate Change Stakeholder Group (ACCSG) (Scottish Government 2008) note in their recommendations that there is a need to pursue 'better integration between currently separate policy themes such as agriculture, forestry, deer management, flooding and biodiversity, all of which are linked to land use and require some degree of spatial co-ordination and co-operation across different parcels of land and therefore different farms' (Miller *et al.* 2009). The integration of hypotheses is also addressed by Foresight (2011) and Killham (2011) which introduce the main concepts behind integrated soil management, which aims to address the challenge of meeting the demands of the increasing world population, while maintaining sustainable agricultural systems, as judged from long-term soil fertility, environmental and socio-economic perspectives. These reports encompass many aspects of sustainable agriculture including scope for future technological development and the need for changes within current policies, governance and funding worldwide to conserve and manage the soil resource, with Killham (2011) focusing on integrated soil management and Foresight (2011) centred primarily on future food security, farming and global sustainability.

DISCUSSION AND CONCLUSION

Despite the evidence of existing pressures facing soil C stocks outlined in the present paper, the collated data indicates that there is little temporal change in Scottish soil C content. The potential for Scottish soils to be losing soil C may be inferred from increased DOC concentrations seen in McCartney *et al.* (2003) and other studies such as Evans *et al.* (2005) and Worrall & Burt (2007). If the soil is considered in terms of mass balance of C, then increased DOC concentrations in surface waters may be indicative of either (a) soil C loss through increased C input to the soil system or (b) loss of older, more stabilized C located deeper in the soil profile. Recent evidence from outside Scotland (Bellamy *et al.* 2005) suggests that loss of C from peaty soils could represent the most serious risk to Scottish soil C stocks (Dobbie *et al.* 2011). However, it is important to consider the depth to which soil C monitoring is conducted. As outlined by Dobbie *et al.* (2011), it is not known whether the total amount of organic C present in soil is changing because most previous studies do not consider the whole soil profile.

There is currently a reasonable understanding of the time scale over which flux terms might be expected to vary, and the environmental factors that influence these, but knowledge of the magnitude of these changes is still incomplete (Billett *et al.* 2010). Unfortunately there is insufficient temporal and spatial data to determine C stock trends with confidence and to relate these trends to C losses such as the source of DOC loss and to GHG emissions. Therefore C data available for Scottish soils may be somewhat misleading in terms of present and past total C stocks calculated and temporal trends that may be inferred from them.

It is evident that the knowledge gap requires addressing future monitoring and research as stated in the report by Dobbie *et al.* (2011). Monitoring is required in Scotland for two different objectives: (1) a surveillance soil monitoring network is required that can provide a general overview of the condition of Scotland's soil and (2) how it is changing through time. This should allow monitoring of existing threats on a national scale and potentially identify as yet unknown threats. This will also provide data for trend analysis (Dobbie *et al.* 2011).

The Scottish Soil Framework (Anonymous 2009) briefly outlines the importance of protecting and enhancing soil C stores where possible and optimising reductions in GHG emissions. Despite a significant body of policy relating to soils both directly and indirectly, the Scottish Soil Framework (Anonymous 2009) reports that no one legislative or policy tool has been developed specifically with the protection of soil in mind. The European Commission has adopted the Thematic Strategy for Soil Protection and a Framework Directive for the Protection of European Soil has been proposed, which aims to preserve soil function and to prevent and restore degraded areas.

How C stocks will be impacted in Scotland and the extent to which they may be affected in the future due to pressures such as climate change, pollution, erosion and alterations in land use will require new knowledge of biogeochemical processes, continuous long-term monitoring and predicting future changes through modelling. It is essential to improve understanding of soil processes, monitoring spatial and temporal changes, develop accurate audits of soil resources and to determine, with confidence, future pressures and threats, as discussed by Haygarth & Ritz (2009). It has been argued that land systems have to be viewed as coupled, multi-scale socio-ecological systems. This is to encompass different types of feedback that may exist between the different environmental, social and economic components, and the fact that policy interventions may have multiple and sometimes unpredictable results (Potschin 2009).

Scottish soils are in general more organic, more leached and wetter than most other European countries containing greater proportion of podzols, peats and gleys than Europe as a whole (Anon 2009). It is evident that Scottish soils are of national importance and a key contributor to UK terrestrial C stores. It is therefore essential to conserve these terrestrial systems for the purpose of maintaining soil quality and participating in mitigating climate change through terrestrial C sequestration.

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REFERENCES

AMMANN, C., FLECHARD, C. R., LEIFELD, J., NEFTEL, A. & FUHRER, N. J. (2007). The carbon budget of newly established temperate grassland depends on management intensity. *Agriculture, Ecosystems and Environment* **121**, 5–20.

Anon (2009). *The Scottish Soil Framework*. Edinburgh, UK: The Stationery Office. Available from: http://www.scotland.gov.uk/Publications/2009/05/20145602/0 (verified 18 April 2013).

BAIN, C. G., BONN, A., STONEMAN, R., CHAPMAN, S., COUPAR, A., EVANS, M., GEAREY, B., HOWAT, M., JOOSTEN, H., KEENLEYSIDE, C., LABADZ, J., LINDSAY, R., LITTLEWOOD, N., LUNT, P., MILLER, C. J., MOXEY, A., ORR, H., REED, M., SMITH, P., SWALES, V., THOMPSON, D. B. A., THOMPSON, P. S., VAN DE NOORT, R., WILSON, J. D. & WORRALL, F. (2011). *IUCN UK Commission of Inquiry on Peatlands*. Edinburgh, UK: IUCN UK Peatland Programme.

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.

BILLETT, M. F., CHARMAN, D. J., CLARK, J. M., EVANS, C. D., EVANS, M. G., OSTLE, N. J., WORRALL, F., BURDEN, A., DINSMORE, K. J., JONES, T., MCNAMARA, N. P., PARRY, L., ROWSON, J. G. & ROSE, R. (2010). Carbon balance of UK peatlands: current state of knowledge and future research challenges. *Climate Research* **45**, 13–29.

BONNETT, S. A. F., OSTLE, N. & FREEMAN, C. (2006). Seasonal variations in decomposition processes in a valley-bottom riparian peatland. *Science of the Total Environment* **370**, 561–573.

BRADLEY, R. I., BELL, J., GAULD, J. H., LILLY, A., JORDAN, C., HIGGINS, A. & MILNE, R. (2005). *UK Soil database for Modelling Soil Carbon Fluxes and Land Use for the National Carbon Dioxide Inventory*. Report to Defra Project SP0511. London: Defra.

BUCKINGHAM, S., TIPPING, E. & HAMILTON-TAYLOR, J. (2008). Concentration and fluxes of dissolved organic carbon in UK topsoils. *Science of the Total Environment* **407**, 460–470.

CANNELL, M. G. R., MILNE, R., HARGREAVES, K. J., BROWN, T. A. W., CRUICKSHANK, M. M., BRADLEY, R. I., SPENCER, T., HOPE, D., BILLET, M. F., ADGER, W. N. & SUBAK, S. (1999). National inventories of terrestrial carbon sources and sinks: the UK experience. *Climatic Change* **42**, 505–530.

CARTER, M. R. (2002). Soil quality for sustainable land management. *Agronomy Journal* **94**, 38–47.

CHAMBERLAIN, P. M., EMMETT, B. A., SCOTT, W. A., BLACK, H. I. J., HORNUNG, M. & FROGBROOK, Z. L. (2010). No change in topsoil carbon levels of Great Britain, 1978–2007. *Biogeosciences Discussions* **7**, 2267–2311.

CHAPMAN, S. J. (2010). Carbon sequestration in soils. In *Carbon Capture: Sequestration and Storage* (Eds R. E. Hester & R. M. Harrison), pp. 179–202. Issues in Environmental Science and Technology 29. Cambridge, UK: RSC Publishing.

CHAPMAN, S. J., CAMPBELL, C. D. & PURI, G. (2003). Native woodland expansion: soil chemical and microbiological indicators of change. *Soil Biology and Biochemistry* **35**, 753–764.

CHAPMAN, S. J., BELL, J., DONNELLY, D. & LILLY, A. (2009). Carbon stocks in Scottish peatlands. *Soil Use and Management* **25**, 105–112.

CHITI, T., CERTINI, G., PUGLISI, A., SANESI, G., CAPPERUCCI, A. & FORTE, C. (2007). Effects of associating a N-fixer species to monotypic oak plantations on the quantity and quality of organic matter in minesoils. *Geoderma* **138**, 162–169.

CIAIS, P., WATTENBACH, M., VUICHARD, N., SMITH, P., PIAO, S. L., DON, A., LUYSSAERT, S., JANSSENS, I. A., BONDEAU, A., DECHOW, R., LEIP, A., SMITH, P. C., BEER, C., VAN DER WERF, G. R., GERVOIS, S., VAN OOST, K., TOMELLERI, E., FREIBAUER, A., SCHULZE, E. D. & CarboEurope Synthesis Team (2010). The European carbon balance. Part 2: croplands. *Global Change Biology* **16**, 1409–1428.

CLARK, J. M., BOTTRELL, S. H., EVANS, C. D., MONTEITH, D. T., BARLETT, R., ROSE, R., NEWTON, R. J. & CHAPMAN, P. J. (2010). The importance of the relationship between scale and process in understanding long-term DOC dynamics. *Science of the Total Environment* **408**, 2768–2775.

CONANT, R. T., EASTER, M., PAUSTIAN, K., SWAN, A. & WILLIAMS, S. (2007). Impacts of periodic tillage on soil C stocks: a synthesis. *Soil and Tillage Research* **95**, 1–10.

Countryside Survey (2007*a*). *Countryside Survey Summary Data Access*. Countryside Survey Data owned by NERC – Centre for Ecology and Hydrology. Countryside Survey © Database Right/copyright NERC – Centre for Ecology and Hydrology. Wallingford, UK: CEH. Available from:

www.countrysidesurvey.org.uk/data-access (verified 29th November 2011).
Countryside Survey (2007*b*). *Countryside Survey 2007, Scotland Results from 2007 Executive Summary*. Countryside Survey © Database Right/Copyright NERC–Centre for Ecology & Hydrology. Wallingford, UK: CEH. Available from: http://www.countrysidesurvey.org.uk/outputs/scotland-results-2007 (verified 16 April 2013).
Cris R., Buckmaster S., Bain C. & Bonn A. (Eds) (2011). *UK Peatland Restoration — Demonstrating Success*. Edinburgh, UK: IUCN UK National Committee Peatland Programme.
Crute, I. R. & Muir, J. F. (2011). Editorial Improving the productivity and sustainability of terrestrial and aquatic food production systems: future perspectives. *Journal of Agricultural Science, Cambridge* **149** (Suppl. 1), 1–7.
Dalias, P., Anderson, J. M., Bottner, P. & Coûteaux, M.-M. (2001). Long-term effects of temperature on carbon mineralisation process. *Soil Biology and Biochemistry* **33**, 1049–1057.
Dawson, J. J. C. & Smith, P. (2007). Carbon losses from soil and its consequences for land-use management. *Science of the Total Environment* **382**, 165–190.
de Vries, W., Reinds, G. J., Gundersen, P. & Sterba, H. (2006). The impact of nitrogen deposition on carbon sequestration in European forests and forest soils. *Global Change Biology* **12**, 1151–1173.
de Vries, W., Solberg, S., Dobbertin, M., Sterba, H., Laubhann, D., van Oijen, M., Evans, C., Gundersen, P., Kros, J., Wamelink, G. W. W., Reinds, G. J. & Sutton, M. A. (2009). The impact of nitrogen deposition on carbon sequestration by European forests and heathlands. *Forest Ecology and Management* **258**, 1814–1823.
Dinsmore, K. J., Billett, M. F., Skiba, U. M., Rees, R. M., Drewer, J. & Helfter, C. (2010). Role of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland catchment. *Global Change Biology* **16**, 2750–2762.
Dobbie K. E., Bruneau P. M. C. & Towers W. (Eds). (2011). *The State of Scotland's Soil*. Edinburgh, UK: Natural Scotland, Scottish Government. Available from: www.sepa.org.uk/land/land_publications.aspx (verified 16 April 2013).
Eggleston, H. S. & Irwin, J. G. (1995). Trends in sulphur and nitrogen emissions. In *Acid Rain and its Impact: The Critical Loads Debate. Proceedings of a Conference held at the Environmental Change Research Centre, 17 September, 1993* (Ed. R. W. Battarbee), pp. 15–16. London: Ensis.
Erisman, J. W., Sutton, M. A., Galloway, J., Klimont, Z. & Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. *Nature Geoscience* **1**, 636–639.
Eurostat (2011). *Climate Change Statistics*. Brussels: European Commission. Available from: http://epp.eurostat.ec.europa.eu/statistics_explained/index.php/Climate_change_statistics (verified April 2012).
Evans, C. D., Monteith, D. T. & Cooper, D. M. (2005). Long-term increases in surface water dissolved organic carbon: observations, possible causes and environmental impacts. *Environmental Pollution* **137**, 55–71.
Evans, C. D., Chapman, P. J., Clark, J. M., Monteith, D. T. & Cresser, M. S. (2006). Alternative explanations for rising dissolved organic carbon export from organic soils. *Global Change Biology* **12**, 2044–2053.
Falloon, P., Powlson, D. & Smith, P. (2004). Managing field margins for biodiversity and carbon sequestration: a Great Britain case study. *Soil Use and Management* **20**, 240–247.
Fang, C., Smith, P. & Smith, J. U. (2005). Is resistant soil organic matter more sensitive to temperature than the labile organic matter? *Biogeosciences Discussion* **2**, 725–735.
Fitton, N., Ejerenwa, C. P., Bhogal, A., Edgington, P., Black, H., Lilly, A., Barraclough, D., Worrall, F., Hillier, J. & Smith, P. (2011). Greenhouse gas mitigation potential of agricultural land in Great Britain. *Soil Use and Management* **27**, 491–501.
Foley, J. A., De Fries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., Patz, J. A., Prentice, I. C., Ramankutty, N. & Snyder, P. K. (2005). Global consequences of land use. *Science* **309**, 570–574.
Foresight (2011). *Foresight. The Future of Food and Farming: Challenges and Choices for Global Sustainability. Final Project Report*. London: The Government Office for Science.
Forestry Commission (2010*a*). *Forestry Facts and Figures 2010: a Summary of Statistics about Woodland and Forestry*. London: The Stationery Office.
Forestry Commission (2010*b*). *The Right Tree in the Right Place: Planning for Forestry and Woodland*. Edinburgh: Forestry Commission Scotland.
Frogbrook, Z. L., Bell, J., Bradley, R. I., Evans, C., Lark, R. M., Reynolds, B., Smith, P. & Towers, W. (2009). Quantifying terrestrial carbon stocks: examining the spatial variation in two upland areas in the UK and a comparison to mapped estimates of soil carbon. *Soil Use and Management* **25**, 320–332.
Glenk, K. & Colombo, S. (2011). Designing policies to mitigate the agricultural contribution to climate change: an assessment of soil based carbon sequestration and its ancillary effects. *Climatic Change* **105**, 43–66.
González-Pérez, J. A., González-Vila, F. J., Almendros, G. & Knicker, H. (2004). The effect of fire on soil organic matter – a review. *Environment International* **30**, 855–870.
Grieve, I. C. (2001). Human impact on soil properties and their implications for the sensitivity of soil systems in Scotland. *Catena* **42**, 361–374.
Guo, L. B. & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology* **8**, 345–360.
Hagedorn, F., Spinnler, D. & Siegwolf, R. (2003). Increased N deposition retards mineralization of old soil organic matter. *Soil Biology and Biochemistry* **35**, 1683–1692.
Hargreaves, K. J., Milne, R. & Cannell, M. G. R. (2003). Carbon balance of afforested peatland in Scotland. *Forestry* **76**, 299–317.

HAYGARTH, P. M. & RITZ, K. (2009). The future of soils and land use in the UK: soil systems for the provision of land-based ecosystem services. *Land Use Policy* **26** (Suppl. 1), S187–S197.
HYVÖNEN, R., AGREN, G. I., LINDER, S., PERSSON, T., COTRUFO, M. N., EKBLAD, A., FREEMAN, M., GRELLE, A., JANSSENS, I. A., JARVIS, P. G., KELLOMAKI, S., LINDROTH, A., LOUSTAU, D., LUNDMARK, T., NORBY, R. J., OREN, R., PILEGAARD, K., RYAN, M. G., SIGURDSSON, B. D., STROMGREN, M., VAN OIJEN, M. & WALLIN, G. (2007). The likely impact of elevated [CO_2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. *New Phytologist* **173**, 463–480.
IPCC (2000). *Special Report on Land Use, Land Use Change, and Forestry* (Eds R. T. Watson, I. R. Noble, B. Bodin, N. H. Ravindranath, D. J. Verardo & D. J. Dokken). Cambridge, UK: Cambridge University Press.
IPCC (2006). *IPCC Guidelines for National Greenhouse Gas Inventories; Prepared by the National Greenhouse Gas Inventories Programme*. Japan: IPCC.
IPCC (2007). Summary for policy makers. In *Climate Change 2007: Mitigation of Climate Change. Working Group III Contribution to the International Panel on Climate Change Fourth Assessment Report*. (Eds B. Metz, O. R. Davidson, P. R. Bosch, R. Dave & L. A. Meyer), pp. 3–23. Cambridge, UK: Cambridge University Press.
JANDL, R., LINDNER, M., VESTERDAL, L., BAUWENS, B., BARITZ, R., HAGEDORN, F., JOHNSON, D. W., MINKKINEN, K. & BYRNE, K. A. (2007). How strongly can forest management influence soil carbon sequestration? *Geoderma* **137**, 253–268.
JANSSENS, I. A., FREIBAUER, A., CIAIS, P., SMITH, P., NABUURS, G.-J., FOLBERTH, G., SCHLAMADINGER, B., HUTJES, R. W. A., CEULEMANS, R., SCHULZE, E-D., VALENTINI, R. & DOLMAN, A. J. (2003). Europe's terrestrial biosphere absorbs 7 to 12% of Europe's anthropogenic CO_2 emissions. *Science* **300**, 1538–1542.
JANSSENS, I. A., DIELEMAN, W., LUYSSAERT, S., SUBKE, J.-A., REICHSTEIN, M., CEULEMANS, R., CIAIS, P., DOLMAN, A. J., GRACE, J., MATTEUCCI, G., PAPALE, D., PIAO, S. L., SCHULZE, E.-D., TANG, J. & LAW, B. E. (2010). Reduction of forest soil respiration in response to nitrogen deposition. *Nature Geoscience* **3**, 315–322.
JONES, S. K., REES, R. M., KOSMAS, D., BALL, B. C. & SKIBA, U. M. (2006). Carbon sequestration in a temperate grassland: management and climatic controls. *Soil Use and Management* **22**, 132–142.
JOOSTEN, H. (2011). *Peatlands, Policies and Markets*. Edinburgh, UK: IUCN UK Peatland Programme.
KHAN, S. A., MULVANEY, R. L., ELLSWORTH, T. R. & BOAST, C. W. (2007). The myth of nitrogen fertilization for soil carbon sequestration. *Journal of Environmental Quality* **36**, 1821–1832.
KILLHAM, K. (2011). Integrated soil management – moving towards globally sustainable agriculture. *Journal of Agricultural Science, Cambridge* **149** (Suppl. 1), 29–36.
LAL, R., KIMBLE, J. M., FOLLETT, R. F. & COLE, C. V. (1998). *The Potential of US Cropland to Sequester Carbon and Mitigate Greenhouse Effect*. Boca Raton, FL, USA: CRC Press.
LAL, R., FOLLETT, R. F., KIMBLE, J. & COLE, C. V. (1999). Managing US cropland to sequester carbon in soil. *Journal of Soil and Water Conservation* **54**, 374–381.
Land Use Strategy for Scotland (2011). *Getting the Best from our Land. A Land Use Strategy for Scotland*. Edinburgh: The Stationery Office.
LEE, J., HOPMANS, J. W., ROLSTON, D. E., BAER, S. G. & SIX, J. (2009). Determining soil carbon stock changes: Simple bulk density corrections fail. *Agriculture, Ecosystems and Environment* **134**, 251–256.
LEIFELD, J. & FUHRER, J. (2010). Organic farming and soil carbon sequestration: what do we really know about the benefits? *AMBIO: A Journal of the Human Environment* **39**, 585–599.
LOVELAND, P. & WEBB, J. (2003). Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. *Soil and Tillage Research* **70**, 1–18.
LU, M., ZHOU, X., LUO, Y., YANG, Y., FANG, C., CHEN, J. & LI, B. (2011). Minor stimulation of soil carbon storage by nitrogen addition: a meta-analysis. *Agriculture, Ecosystems and Environment* **140**, 234–244.
LUMSDON, D. G., STUTTER, M. I., COOPER, R. J. & MANSON, J. R. (2005). Model assessment of biogeochemical controls on dissolved organic carbon partitioning in an acid organic soil. *Environmental Science Technology* **39**, 8057–8063.
MAGILL, A. H. & ABER, J. D. (2000). Dissolved organic carbon and nitrogen relationships in forest litter as affected by nitrogen deposition. *Soil Biology and Biochemistry* **32**, 603–613.
MAGNANI, F., MENCUCCINI, M., BORGHETTI, M., BERBIGIER, P., BERNINGER, F., DELZON, S., GRELLE, A., HARI, P., JARVIS, P. G., KOLARI, P., KOWALSKI, A. S., LANKREIJER, H., LAW, B. E., LINDROTH, A., LOUSTAU, D., MANCA, G., MONCRIEFF, J. B., RAYMENT, M., TEDESCHI, V., VALENTINI, R. & GRACE, J. (2007). The human footprint in the carbon cycle of temperate and boreal forests. *Nature* **447**, 849–851.
MATHER, A. S. & MURRAY, N. C. (1988). The dynamics of rural land use change. *Land Use Policy* **5**, 103–120.
MCCARTNEY, A. G., HARRIMAN, R., WATT, A. W., MOORE, D. W., TAYLOR, E. M., COLLEN, P. & KEAY, E. J. (2003). Long-term trends in pH, aluminium and dissolved organic carbon in Scottish fresh waters; implications for brown trout (*Salmo trutta*) survival. *Science of the Total Environment* **310**, 133–141.
MCROBBIE, G., MCCALL, A., MIDGLEY, A., MILLER, N., HALL, J., MILNE, G., BARNES, D., MAXWELL, J., BARBOUR, A., MCINTOSH, B., DAVIES, S., O'HARA, J., MACLEOD, H., AITKEN, M., GOODALL, S., SWALES, V., PEPPER, S., ROSS, I. & RITCHIE, B. (2012). *Report of the Woodland Expansion Advisory Group to the Cabinet Secretary for Rural Affairs and Environment, Richard Lochhead MSP*. Edinburgh, UK: WEAG. Available from: http://www.forestry.gov.uk/weag (verified 12 December 2012).
MEDINA-ROLDÁN, E., PAZ-FERREIRO, J. & BARDGETT, R. D. (2012). Grazing exclusion affects soil and plant communities, but has no impact on soil carbon storage in an upland grassland. *Agriculture, Ecosystems and Environment* **149**, 118–123.
MICHEL, K. & MATZNER, E. (2002). Nitrogen content of forest floor Oa layers affects carbon pathways and nitrogen

mineralization. *Soil Biology and Biochemistry* **34**, 1807–1813.

Miller, D., Schwarz, G., Sutherland, L.-A., Morrice, J., Aspinall, R., Barnes, A., Blackstock, K., Buchan, K., Donnelly, D., Hawes, C., McCrum, G., McKenzie, B., Matthews, K., Miller, D., Renwick, A., Smith, M., Squire, G. & Toma, L. (2009). Changing land-use in rural Scotland – Drivers and decision-making. Rural land use study project 1. *Scottish Government 2009*.

Milne, R. & Brown, T. A. (1997). Carbon in the vegetation and soils of Great Britain. *Journal of Environmental Management* **49**, 413–433.

Monteith, D. T., Stoddard, J. L., Evans, C. D., de Wit, H. A., Forsius, M., Høgåsen, T., Wildander, A., Skjelkvåle, B. L., Jeffries, D. S., Vuorenmaa, J., Keller, B., Kopácek, J. & Vesely, J. (2007). Dissolved organic carbon trends resulting from changes in atmospheric deposition chemistry. *Nature* **450**, 537–540.

Morison, J., Vanguelova, E., Broadmeadow, S., Perks, M., Yamulki, S. & Randle, T. (2010). *Understanding the GHG Implications of Forestry on Peat Soils in Scotland*. Report by Forest Research for Forestry Commission Scotland. Edinburgh, UK: The Stationery Office.

Mulvaney, R. L., Khan, S. A. & Ellsworth, T. R. (2009). Synthetic nitrogen fertilizers deplete soil nitrogen: a global dilemma for sustainable cereal production. *Journal of Environmental Quality* **38**, 2295–2314.

Nave, E., Vance, E. D., Swanston, C. W. & Curtis, P. S. (2009). Impacts of elevated N inputs on north temperate forest soil C storage, C/N, and net N-mineralization. *Geoderma* **153**, 231–240.

Nayak, D. N., Miller, D., Nolan, A., Smith, P. & Smith, J. (2008). *Calculating Carbon Savings from Wind Farms on Scottish Peat Lands – a New Approach*. Final report to the Rural and Environment Research and Analysis Directorate of the Scottish Government, Science Policy and Co-ordination Division. Edinburhg, UK: The Stationery Office. Available from: http://www.scotland.gov.uk/Publications/2008/06/25114657/18 (verified 17 April 2013).

Ostle, N. J., Levy, P. E., Evans, C. D. & Smith, P. (2009). UK land use and soil carbon sequestration. *Land Use Policy* **26** (Suppl. 1), S274–S283.

Paustian, K., Andren, O., Janzen, H. H., Lal, R., Smith, P., Tian, G., Tiessen, H., Van Noordwijk, M. & Woomer, P. L. (1997). Agricultural soils as a sink to mitigate CO_2 emissions. *Soil Use and Management* **13** (Suppl. s4), 230–244.

Piñeiro, G., Paruelo, J. M., Oesterheld, M. & Jobbágy, E. G. (2010). Pathways of grazing effects on soil organic carbon and nitrogen. *Rangeland Ecology and Management* **63**, 109–119.

Plante, A. F., Conant, R. T., Carlson, J., Greenwood, R., Shulman, J. M., Haddix, M. L. & Paul, E. A. (2010). Decomposition temperature sensitivity of isolated soil organic matter fractions. *Soil Biology and Biochemistry* **42**, 1991–1996.

Potschin, M. (2009). Land use and the state of the natural environment. *Land Use Policy* **26** (Suppl. 1), S170–S177.

Powlson, D. S., Whitmore, A. P. & Goulding, K. W. T. (2011). Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science* **62**, 42–55.

Rees, R. M., Bingham, I. J., Baddeley, J. A. & Watson, C. A. (2005). The role of plants and land management in sequestering soil carbon in temperate arable and grassland ecosystems. *Geoderma* **128**, 130–154.

Rees, R. M., Black, H., Chapman, S. J., Clayden, H., Edwards, A. C. & Waldron, S. (2011). Loss of soil organic matter. In *The State of Scotland's Soil* (Eds K. E. Dobbie, P. M. C. Bruneau & W. Towers), pp. 23–33. Edinburgh, UK: Natural Scotland. Available from: www.sepa.org.uk/land/land_publications.aspx (verified 18 April 2013).

Rounsevell, M. D. A. & Reay, D. S. (2009). Land use and climate change in the UK. *Land Use Policy* **26** (Suppl. 1), S160–S169.

Schimel, D. S., House, J. I., Hibbard, K. A., Bousquet, P., Ciais, P., Peylin, P., Braswell, B. H., Apps, M. J., Baker, D., Bondeau, A., Canadell, J., Churkina, G., Cramer, W., Denning, A. S., Field, C. B., Friedlingstein, P., Goodale, C., Heimann, M., Houghton, R. A., Melillo, J. M., Moore, B., Murdiyarso, D., Noble, I., Pacala, S. W., Prentice, I. C., Raupach, M. R., Rayner, P. J., Scholes, R. J., Steffen, W. L. & Wirth, C. (2001). Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. *Nature* **414**, 169–172.

Schlesinger, W. H. (2000). Carbon sequestration in soils: some cautions and optimism. *Agriculture, Ecosystems and Environment* **82**, 121–127.

Schrumpf, M., Schulze, E. D., Kaiser, K. & Schumacher, J. (2011). How accurately can soil organic carbon stocks and stock changes be quantified by soil inventories? *Biogeosciences* **8**, 1193–1212.

Schulze, E. D., Ciais, P., Luyssaert, S., Schrumpf, M., Janssens, I. A., Thiruchittampalam, B., Theloke, J., Saurat, M., Bringezu, S., Lelieveld, J., Lohila, A., Rebmann, C., Jung, M., Bastviken, D., Abril, G., Grassi, G., Leip, A., Freibauer, A., Kutsch, W., Don, A., Nieschulze, J., Börner, A., Gash, J. H. & Dolman, A. J. (2010). The European carbon balance. Part 4: integration of carbon and other trace-gas fluxes. *Global Change Biology* **16**, 1451–1469.

Scottish Executive (2006). *The Scottish Forestry Strategy, SE/2006/155*. Forestry Commission Scotland. Edinburgh, UK: The Stationery Office.

Scottish Government (2008). *Climate Change and Scottish Agriculture: Report and Recommendations of the Agriculture and Climate Change Stakeholder Group*. Victoria Quay, Edinburgh, UK. Available from: http://www.scotland.gov.uk/Publications/2008/05/15115150/0 (verified 30 April 2013).

Scottish Government (2009). *Climate Change (Scotland) Act 2009 asp12*. Edinburgh, UK: The Stationery Office. Available from: http://www.legislation.gov.uk/asp/2009/12/contents (verified 18 April 2013).

Scottish Government (2011). *Final Results from 2011 June Agricultural Census, 27th September 2011. The Scottish Government 2011*. Statistical Publication, Agriculture Series. Edinburgh, UK: The Stationery Office. Available from: http://www.scotland.gov.uk/Publications/2011/09/27083355/0 (verified 18 April 2013).

Scottish Government (2012). *Funding for Peatland Restoration*. Edinburgh, UK: The Stationery Office. Available from: http://www.scotland.gov.uk/News/Releases/2012/10/Peatlands22102012 (verified 5th November 2012).

Smith, P. (2004*a*). Monitoring and verification of soil carbon changes under Article 3·4 of the Kyoto Protocol. *Soil Use and Management* **20**, 264–270.

Smith, P. (2004*b*). Carbon sequestration in croplands: the potential in Europe and the global context. *European Journal of Agronomy* **20**, 229–236.

Smith, P. (2008). Land use change and soil organic carbon dynamics. *Nutrient Cycling in Agroecosystems* **81**, 169–178.

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*a*). 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., Smith, J., Flynn, H., Killham, K., Rangel-Castro, I., Foereid, B., Aitkenhead, M., Chapman, S., Towers, W., Bell, J., Lumsdon, D., Milne, R., Thomson, A., Simmons, I., Skiba, U., Reynolds, B., Evans, C., Frogbrook, Z., Bradley, I., Whitmore, A. & Falloon, P. (2007*b*). *ECOSSE*: *Estimating Carbon in Organic Soils Sequestration and Emissions*. Scottish Executive Environmental and Rural Affairs Department, Environmental Research. Edinburgh, UK: The Stationery Office.

Smith, P., Bhogal, A., Edgington, P., Black, H., Lilly, A., Barraclough, D., Worral, F., Hillier, J. & Merrington, G. (2010). Consequences of feasible future agricultural land-use change on soil organic carbon socks and greenhouse gas emissions in Great Britain. *Soil Use and Management* **26**, 381–398.

Soane, B. D., Ball, B. C., Arvidsson, J., Basch, G., Moreno, F. & Roger-Estrade, J. (2012). No-till in northern, western and south-western Europe: a review of problems and opportunities for crop production and the environment. *Soil and Tillage Research* **118**, 66–87.

Soil Survey Staff (1984). *Organisation and Methods of the 1 : 250 000 Soil Survey of Scotland*. Handbook 8. Aberdeen, UK: Macaulay Institute for Soil Research.

Sonneveld, M. P. W. & Van den Akker, J. J. H. (2011). Quantification of C and N stocks in grassland topsoils in a Dutch region dominated by dairy farming. *Journal of Agricultural Science, Cambridge* **149**, 63–71.

Soussana, J. F., Pilegaard, K., Ambus, P., Berbigier, P., Ceschia, E., Clifton-Brown, J., Czobel, Sz., de Groot, T., Fuhrer, J., Horvagh, L., Hensen, A., Jones, M., Kasper, J., Martin, C., Milford, C., Nagy, Z., Neftel, A., Raschi, A., Rees, R. M., Skiba, U., Stefani, P., Saletes, S., Sutton, M. A., Tuba, S. & Weidinger, T. (2004). Annual greenhouse gas balance of European grasslands. First results from the GreenGrass project. In *Greenhouse Gas Emissions from Agriculture: Mitigation Options and Strategies. Conference Proceedings* (Ed. A. Weiske), pp. 25–30. Leipzig, Germany: Institute for Energy and Environment.

Thomson, A. M., Mobbs, D. C., Milne, R., Skiba, U., Clark, A., Levy, P. E., Jones, S. K., Billett, M. F., Van Oijen, M., Ostle, N., Foereid, B., Fung, W. S., Smith, P., Matthews, R. W., Mackie, E., Bellamy, P., Rivas-Casado, M., Grace, J., Higgins, A., Jordan, C. & Tomlinson, R. W. (2008). *Inventory and Projections of UK Emissions by Sources and Removals by Sinks due to Land Use, Land Use Change and Forestry. Annual Report to the UK Department for the Environment, Food and Rural Affairs (DEFRA), July 2008*. DEFRA Contract no. GA01088. Edinburgh, UK: CEH/Defera.

Towers, W., Grieve, I. C., Hudson, G., Campbell, C. D., Lilly, A., Davidson, D. A., Bacon, J. R., Langan, S. J. & Hopkins, D. A. (2006). *Scotland's Soil Resource – Current State and Threats*. Edinburgh, UK: The Stationery Office. Available from: http://www.scotland.gov.uk/Publications/2006/09/21115639/0 (verified 18 April 2013).

Trumbore, S. E., Chadwick, O. A. & Amundson, R. (1996). Rapid exchange between soil carbon and atmospheric carbon dioxide driven by temperature change. *Science* **272**, 393–396.

Turunen, J., Roulet, N. T., Moore, T. R. & Richard, P. J. H. (2004). Nitrogen deposition and increased carbon accumulation in ombrotrophic peatlands in eastern Canada. *Global Biogeochemical Cycles* **18**, GB3002. doi:10.1029/2003GB002154

UK Government (2008). *Climate Change Act 2008*. Chapter 27. London: The Stationery Office. Available from: http://www.legislation.gov.uk/ukpga/2008/27/contents (verified 18 April 2013).

van Camp, L., Bujarrabal, B., Gentile, A. R., Jones, R. J. A., Montanarella, L., Olazabal, C. & Selvaradjou, S. -K. (2004). *Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. Volume III, Organic Matter*. EUR 21319 EN/3 European Commission. Luxembourg: Office for Official Publications of the European Communities.

von Lutzow, M. & Kögel-Knabner, I. (2009). Temperature sensitivity of soil organic matter decomposition – what do we know? *Biology and Fertility of Soils* **46**, 1–15.

Walker, R. L., Topp, K. & Watson, C. A. (2007). SAC crop rotation trials – history, present and future. In *Measuring Food Quality: Concepts, Methods and Challenges. International Training and Exchange Workshop, 12–14 February 2007, Driebergen, Netherlands* (Ed. L. P. L. van de Vijver), pp. 33–34. Driebergen, Netherlands: Lois Bolk Institute.

Watson, C. A., Walker, R. L. & Stockdale, E. A. (2008). Research in organic production systems – past, present and future. *Journal of Agricultural Science, Cambridge* **146**, 1–19.

Wellock, M. L., Reidy, B., Laperle, C. M., Bolger, T. & Keily, G. (2011). Soil organic carbon stocks of afforested peatlands in Ireland. *Forestry* **84**, 441–451.

Wilson, B. & Puri, G. (2001). A comparison of pinewood and moorland soils in the Abernethy Forest

Reserve, Scotland. *Global Ecology and Biogeography* **10**, 291–303.

Worrall, F. & Burt, T. P. (2007). Trends in DOC concentration in Great Britain. *Journal of Hydrology* **346**, 81–92.

Worrall, F., Bell, M. J. & Bhogal, A. (2009). Assessing the probability of carbon and greenhouse gas benefit from the management of peat soils. *Science of the Total Environment* **408**, 2657–2666.

Yallop, A. R., Clutterbuck, B. & Thacker, J. (2010). Increases in humic dissolved organic carbon export from upland peat catchments: the role of temperature, declining sulphur deposition and changes in land management. *Climate Research* **45**, 43–56.

Zerva, A. & Mencuccini, M. (2005). Short-term effects of clearfelling on soil CO_2, CH_4 and N_2O fluxes in a Sitka spruce plantation. *Soil Biology and Biochemistry* **37**, 2025–2036.