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2. Climate and soil functions: impacts on Soil processes and properties and future implications in the UK – a review

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Summary

Current climate

- Soil erosion is an obvious problem in the UK, especially in England and Wales (High confidence).
- Nitrate leaching to the groundwater is high in some localities (High confidence).
- Ammonia emission in the UK is decreasing (Medium confidence).
- Land management reduce greenhouse gas emissions and sequester SOC (GHG) (Low confidence).

What could happen under climate change?

- Climate change will substantially impact many soil processes and consequently, influence a number of soil properties such as structure (Low confidence), trafficability/ workability (Medium confidence), pH (Medium confidence), fertility (Medium confidence) and biota (High confidence).
- Climate change will increase soil erosion (High confidence) and nitrate leaching (High confidence).
- The influence of climate change on soils depends on the size of future changes in climate and on the interaction between the influences of different parameters (Medium confidence).
- Climate change will generally increase GHG emissions (Low confidence) from soils but net primary productivity (NPP) is also expected to increase (Medium confidence).
- Future SOC stock will be determined by the balance between carbon (C) losses from decomposition and C gains from higher plant productivity (Medium confidence).
- SOC stock will significantly be affected by climate-induced changes in land use and management (Medium confidence).
- To increase C sequestration and NPP, and to decrease C flux under climate change, suitable land use and management practices that are resilient to new climatic regimes should be adopted (Medium confidence).

Introduction

Soils are the foundations on which all terrestrial ecosystems function. They influence water, vegetation and biogeochemical cycling (Rounsevell et al., 1999). Soil chemical and biological processes are controlled by a complex set of factors (Jenny, 1941) but most importantly by the balance between soil temperature and soil moisture. Temperature is key factor that can control many terrestrial biogeochemical processes, including litter (Hobbie, 1996) and soil organic matter (SOM) decomposition, N mineralization and nitrification (MacDonald et al., 1995), denitrification (Abdalla et al., 2009), respiration (Kirschbaum et al., 1995; Christensen et al., 1997; Davidson and Janssens, 2006), CH_4 emissions (Johnson et al., 1996) and plant nutrient uptake (BassiriRad, 2000).

Climate change, due to anthropogenic greenhouse gas (GHG) emissions (IPCC, 2013), is expected to influence soil functions and properties. For the UK, future climate change projections suggest that the mean temperature will increase by 2.4 to 4^{o}C by the year 2080, the maximum and minimum temperature for the winter and summer will increase, and extreme weather events will become more frequent and more severe. Further, summers are expected to be drier and the winter will be wetter, and sea level will rise (UKCP09; Murphy et al., 2009). These changes in temperature and precipitation, in addition to higher atmospheric CO_2 concentrations under climate change, will have substantial impacts on soil functions. Nutrient release from SOM by mineralisation is expected to increase due to higher temperatures (Fang et al., 2005).

Nevertheless, as atmospheric CO_2 concentration and air temperatures increase, soils could lose C to the atmosphere in the form of GHG emissions, resulting in positive feedback that could increase temperature further (Brevik, 2013). Likewise, rainfall affects a number of soil forming processes such as organic matter (OM) turnover, leaching (Stuart et al., 2007) and erosion (McHugh, 2007). Increased winter rainfall due to climate change will increase annual atmospheric N deposition which is currently range from 20 to 30 kg N ha^{-1} (Goulding, 1990), if nitrogen emission rates remain constant, but nutrient leaching will also be increased (Liu et al., 2008).

The effect of climate change on soils could occur directly, such as through the effects of temperature on SOM decomposition (Fang et al., 2005) or indirectly, such as through changes in soil moisture, due to changes in plant associated evapotranspiration (Naden and Watts, 2001). Soil moisture is influenced by direct climatic factors (precipitation and temperature), climate induced changes in vegetation, different plant growth rates and changed biogeochemical cycles, different rates of soil water extraction and the effect of enhanced CO_2 levels on plant transpiration (IPCC, 2013).

Soils, especially organic soils, are the largest carbon (C) store in terrestrial ecosystems in the UK (Thomson, 2008). Some important soil functions, like SOM decomposition and nutrient dynamics, are well coupled with plant roots and their related rhizosphere processes (Van Veen et al., 1991).The loss of SOM due to climate change would affect the stability of soil structure, topsoil water holding capacity, nutrient availability, soil erosion and land use, especially in the south of England. Drought will make mineral soils drier, and soils close to low-lying coastal areas could become inundated.

However, soils could have significant role in climate mitigation by sequestering C and reducing the concentration of CO_2 in the atmosphere (Smith, 2012). The sequestration of C in soils could be enhanced by increasing net primary productivity (NPP) relative to C mineralisation or by decreasing C loss from the ecosystem (Six et al., 2004). Improving soil management is important for reducing GHG emissions and for sequestering carbon into SOM (Lal, 2008).

A review of management techniques such as conservation tillage systems (e.g. no-till) in Europe by Holland (2004) found that these practices usually reduce CO_2 fluxes and increase carbon sequestration in the soil compared to conventional tillage systems (Six et al., 2004). West & Post (2002) analysed global data from long-term studies following conversion from conventional to no-tillage. They reported that no-till increased the amount of C sequestration by 0.57 ± 0.14 t C ha^{-1} y^{-1} compared with conventional tillage.

However, a recent global review by Abdalla et al. (2013) found significant reductions in CO_2 flux but higher N_2O emissions compared to conventional tillage. Similarly, Powlson et al. (2014) reported that no-till is beneficial for soil quality and adaptation of agriculture to climate change, but its role in GHG mitigation may have been overstated due to possible change in N_2O emissions.

The impact of climate change on SOC is still a controversial topic. Two previously published reviews by CLIMSOIL (Schils et al., 2008) and Pacific Northwest National Laboratory (Qafoku, 2014) have found no concrete evidence of climate change effects on SOC. However, to adapt to changes in soil processes, an understanding of how climate and soils interact, and how changes in climate will lead to corresponding changes in soil, is required.

The aim of this review is to investigate the impacts of current and future climate on soil functions of managed agriculture and forestry in the UK. Section 2 focuses on the impacts of climate on soil processes. Section 3 reports on the impacts on soil properties. Section 4 briefly considers the impacts on soil key functions and Section 5 draws some conclusions, and highlights research needs concerning the impacts of climate change on soils in the UK.

Impacts on soils processes

Impacts on Nitrogen (N) mineralisation

Soil N mineralisation, by which organic matter is converted to inorganic N forms, depends on the amount and nature of the organic matter but also, among others, on temperature and rainfall (Rustad et al., 2001; Rowe et al., 2012). A study of 39 different soil types incubated under standard conditions, showed variable N mineralisation, although the proportion of N mineralised per week was similar (Stanford and Smith, 1972). In mineral and managed agricultural soils, mineralisation rate is correlated directly to soil organic matter content and temperature (Leirós et al., 1999; Abdalla et al., 2009) and soil moisture and rainfall (Emmett et al., 2004).The rate of mineralisation is usually small over the winter when temperature is < 4^oC. However, in organic soils, mineralisation results in a large amount of nitrate in the late spring and summer (Defra, 2011).

Changes in rainfall affect soil moisture and thereby, carbon inputs and nitrate uptake by vegetation (Ineson et al., 1998a; Ineson et al., 1998b). In a study covering 665 locations across Britain, Rowe et al. (2012) investigated N availability in relation to variations in climate and reactive N deposition, and found that N mineralisation increased with increasing mean annual temperature of the sites. However, soil characteristics affected this relationship and soil carbon content in particular was a major control on mineralisation rate. They observed that the stock of readily mineralisable N increased more with N deposition in organic compared to mineral soils. This suggested that increasing temperatures due to climate change are likely to increase the effects of N pollution on organic soils more than on mineral soils.

Increasing summer droughts, due to climate change, will reduce N mineralisation (Borken and Matzner, 2009) due to limited soil microorganism activities. However, in saturated soils, lack of oxygen limits N mineralization because only soil microorganisms that can survive under anaerobic conditions are active (Deenik, 2006). Organic rich histosols in agricultural production have higher mineralisation rates for added nitrogenous fertilisers than mineral soils and consequently, higher N_2O emissions (Skiba et al., 2012).

In a shrubland podzol soil in the UK, net N immobilisation was reported to occur frequently, most likely due to a combination of wetter and colder conditions and low availability of other nutrients (Emmet et al., 2004). Matzner and Borken (2008) reviewed the mechanisms causing the post-frost pulse and suggested that nitrate losses are more likely caused by reduced root uptake rather than by increased N net mineralisation. Net N mineralisation is a complex process and is the difference between gross mineralisation and immobilisation.

Nevertheless, considering all processes increasing or decreasing N production from mineralisation, Ducharne et al. (2007) expected a net increase of 20% in nitrate leaching and nitrate in streams, in France, if appropriate farming practices are not applied. A simulation study by Abdalla et al. (2010) using future climate scenarios in Ireland, predicted that climate change would increase N mineralisation in mineral soils.

Impacts on nitrification and denitrification

Soil nitrification and denitrification processes play an important role in regulating inorganic N concentration in soils, nitrate leaching and N_2O emissions. Nitrification is an aerobic process that is directly related to temperature and soil moisture (Emmett et al., 2004) whilst denitrification is an anaerobic process related to soil temperature, water-filled pore space and soil mineral N content (Conen et al., 2000; Sgouridis and Ullah, 2014). Soil ammonia-oxidizing archaea (AOA) are highly abundant and play an important role in the nitrogen cycle (Zhalnina et al., 2012). In acidic forest soil, where net nitrification was high, nitrification was driven by archaea however, addition of ammonium did not influence the nitrification rate (Stopnisek et al., 2010). In contrast, in nitrogen-rich grassland soils, nitrification was driven by bacteria, not archaea, and the rate of nitrification was related to the abundance of ammonia-oxidizing bacteria (AOB) rather than AOA (Di et al., 2009). Application of ammonium to agricultural alkaline soil increased the abundance of AOB during nitrification process (Jia and Conrad, 2009).

Soil moisture stimulates denitrification by reducing oxygen (Dobbie and Smith, 2001) and increasing solubility of organic carbon and nitrate in soils (Bowden and Bormann, 1986). Future higher atmospheric CO_2 concentration and higher temperature could have strong effects on soil moisture and soil biological activity (Rustad et al., 2001; Zak et al., 2000). Nitrous oxide emissions from soils are mainly regulated by mineral nitrogen, temperature, water content and labile organic compounds (Machefert et al., 2002). Higher mineralisation and denitrification due to increased temperature under climate change will result in higher N_2O emissions from soils (Abdalla et al., 2010).

The positive relationship between temperature and N_2O fluxes has been documented in the literature (Flessa and Ruser 2002; Dueri et al. 2007; Abdalla et al., 2014). The higher quantity and frequency of rainfall in late autumn and winter expected in the future will increase denitrification and thereby, N_2O fluxes from soils (Dobbie et al., 1999). Here, soil moisture increases the supply of C substrate for denitrification (De Catanzaro and Beauchamp, 1985). Nitrous oxide is also produced during nitrate ammonification in soils, but the contribution of this process to the emissions is not well quantified. In this process, NO_3^- is converted to NO_2^- and NH_4^+ (Mohan et al., 2004) and N_2O is produced when NO_2^- is reduced (Kelso et al., 1997).

Impacts on decomposition

Decomposition of litter plays an important role in C cycling in terrestrial ecosystems (Shiels, 2006). Soil respiration, i.e. the total CO_2 efflux at the soil surface, relates to litter decomposition and SOM and comprises autotrophic root respiration and heterotrophic respiration (Bernhardt et al., 2006). Prescott (2009) reported that to sequester more C in soil, more litter should be diverted into humus through microbial and chemical reactions rather than allowing it to decompose. The optimal strategy is to have litter transformed into humic substances and then chemically or physically protected in mineral soil. The addition of N by fertilizer or N-fixing plants is a reasonable way of stimulating humification. Climate change will alter the soil-plant system and impact on decomposition rates, and thereby the amount of SOM. Increasing temperature (Christensen et al., 1997) and atmospheric CO_2 concentration (Ball and Drake, 1998) will increase soil respiration. Elevated CO_2 concentrations will increase photosynthesis, plant growth, belowground C input and substrate and microbial activities (Zak et al., 2000; Anderson et al., 2001) if the soil N is not limited (Hungate et al., 2014).

Increased soil moisture content, due to reduced stomatal conductance and transpiration of plants under high atmospheric CO_2 concentrations, will enhance root and microbial activities and soil respiration (Morgan et al., 2004). However, a global review by Zhang et al (2008) found that a single factor such as climate, litter quality and geographic variable could not, by themselves, explain litter decomposition rates. The combination of litter quality and climatic factors, however, has a very important influence on litter decomposition rates.

Rhizosphere processes are important for the functioning of terrestrial ecosystems. They contribute about 50% of the global CO_2 loss from terrestrial ecosystems (Schimel, 1995), regulate almost all aspects of nutrient cycling (Smith and Read, 1997), and represent the primary gateway for plant water uptake (Jackson et al., 2000). Cheng (2009) found that the magnitude of the rhizosphere priming effect on SOM decomposition varied from 0 to 380% of the unplanted control, and was greatly influenced by plant species and phenology.

However, he suggested a possible decoupling of C cycling with N cycling in the rhizosphere because, the rhizosphere enhancement of soil carbon mineralization did not cause a proportional increase in net N mineralization.

Impacts on ammonia volatilisation

Agricultural soils account for around 82% of the total ammonia (NH_3) emissions in the UK. Ammonia emissions predominately arise from livestock manure and urine, though inorganic N fertilisers can also produce NH_3 as nitrogen reacts with compounds in the soil and air (Defra, 2013). This causes fertilizer loss, reduces air quality and causes eutrophication of ecosystems, leading to a loss of biodiversity (Fowler et al., 2013). Ammonia volatilisation is enhanced at higher temperatures and soil drying and therefore, future warmer and drier climates would be expected to increase NH_3 emissions from soils (Skjøth and Geels, 2013). Whitehead et al. (2006) reported higher NH_3 emissions from a groundwater-fed river in southeast England, mainly due to higher temperatures and enhanced microbial activity.

Ammonia can then be nitrified or denitrified further downstream, or re-deposited, increasing N_2O emissions. However, NH_3 emissions in the UK are decreasing. Projections using the 2010 model structure, in the UK, gave estimates of 306 and 216 Kt NH_3 emissions for the years 1990 and 2020, respectively. This inventory reports emission from livestock agriculture and from nitrogen fertilisers applied to agricultural land (Defra, 2012).

Impacts on nitrate leaching

Nitrate leaching to the groundwater in the UK is currently high in some localities (Rivett et al., 2007) depending on soil pH (i.e. this is the reason for presence of some nitrate vulnerable zones). Nitrate leaching comes from surplus inorganic N in arable soils during autumn / early winter when the soil is saturated with water (Shepherd et al., 2002) and plant demand for N is low, excessive livestock numbers, inappropriate use of manure and exposure of bare soil during the winter drainage period (Defra, 2000). More than 60% of nitrate leaching to rivers in England has been derived from agricultural land (Defra, 2009).

Nitrate leaching from mineral and organic soils would be almost the same if similar cropping systems and best practice are applied (i.e. about 50 kg N ha^{-1}; Stopes et al., 2002).The extent of the leaching depends on rainfall, water holding capacity, soil type, cropping, and the amount / timing of fertiliser or manure applications (White et al., 1983). Nitrate leaching to groundwater depends on the partitioning between run-off and infiltration. Changes in soil hydraulic properties (i.e. the ability of a soil to retain or transmit water and its dissolved constituents) due to higher rainfall intensity will lead to a change in partitioning between run off and recharge.

Climate change will modify soil processes that underpin crop growth, and thereby could increase nitrate leaching in many places in the UK over the next decade (Stuart et al., 2007; Stuart et al., 2011). However, this occurs only under N saturated conditions (Schmidt et al., 2004). Olesen et al (2007) modelled nitrate leaching in the UK for the period 2071 to 2100 and found that nitrate leaching flux has a patchy increase, although this is not quantified. Future nitrate leaching could be mitigated by future changes in agricultural practices, such as planting of catch crops and use of improved crop rotations (Thomsen, 2005; Defra, 2009). However, it may also be influenced by economic responses to climate change (Stuart et al., 2011).

Impacts on soil erosion

Soil erosion is an obvious problem in the UK, especially in England and Wales, and has significantly increased in recent decades (McHugh, 2007). Soil erosion by water is more widespread than by wind. The UK loses about 2.2 million tonnes of topsoil per year and 17% of the arable land shows signs of erosion due to water driven erosion (EA, 2004). The rate of soil erosion by water has been estimated at 0.1-0.3 t ha^{-1} y^{-1}.Climate change is likely to affect soil erosion by water, through its effects on rainfall amount and intensity, soil erodibility, vegetative cover and patterns of land use (McHugh, 2007; Mullan et al., 2011). Climatic variations with future higher winter precipitation intensity (Hulme et al., 1993), and extreme weather events, will create ideal conditions for higher soil erosion. Simulation models predicted that a 10% increase in the winter rainfall in the UK could increase soil erosion by 150% in wet years but the long-term average show modest increase over the current condition (Favis-Mortlock and Boardman, 1995). For England and Wales, McHugh (2007) expected rainfall to increase soil erosion by an average of 0.6 t ha^{-1} y^{-1} by the 2080s.

For many areas, climate models predict seasonally more intense drying, coupled with increased quantities and intensity of precipitation at other times; conditions that could lead to a large increase in rates of erosion by water. Another potential effect of climate change on soil erosion relates to temperature and CO_2-driven changes in plant biomass, with increasing erosion rates possible owing to faster residue decomposition from increased soil microbial activity, i.e. reducing organic matter content and binding of particles (Nearing et al., 2005), or decreasing erosion rates possible with increasing soil surface canopy and biological ground cover (Rosenzweig and Hillel, 1998).

In addition, a more indirect effect of climate change on soil erosion could occur as a consequence of shifting land use and agricultural practice to accommodate the new climatic regime (Williams et al., 1996), i.e. changes in plant biomass and thereby changes in protection of the soil from erosion. Agricultural land will face flooding from rivers and seas. Therefore, some current areas will become unsuitable for agricultural activities due to salt water invading soils and ground water (Barclay, 2012). Generally, soil erosion and structural damage are significantly related to the clearance of natural vegetation for annual cropping and the use of unsuitable farming practices, heavy trampling of soil by sheep and cattle, poor forestry practices and run-off from urban land. Future modifications to planting and harvesting dates, and the implementation of new crops and land use changes are possible, all of which carry the potential to considerably alter rates and patterns of soil erosion (Nearing et al., 2005). However, the introduction of new crops that suit the warmer climate e.g. maize and sunflower which need longer time to provide adequate crop cover and clearance of forests would increase soil erosion problem.

Impacts on soil water and water uptake

Soil water is controlled by many factors including infiltration, percolation, drainage and run off, and the amount and distribution of rainfall or irrigation (Rounsevell et al., 1999). A decrease in SOM content due to future climate warming (Leirós et al., 1999) would affect soil hydraulic properties (Bowman et al., 2000). Higher future air temperatures will increase evaporation resulting in higher levels of atmospheric water vapour and a greater variability in the amount and intensity of precipitation (Houghton et al. 1992; Kattenberg et al. 1996).

Higher temperatures and accelerating evaporation through the spring lead to drying soils (Hough and Jones, 1997) and will have a great influence on many soil processes and land use. Burt and Shahgedanova (1998) calculated evaporation from 1815 to 1996 for Oxford and reported increases in potential evaporation (PE) but decreases in actual evaporation (AE) due to warmer temperature and less available water during the summer months. Kay et al. (2013) found some evidence of increasing PE throughout the UK since the 1960s. Temperature trends can be used to infer changes in PE as the two are correlated (Dai, 2011; Sheffield et al., 2012), though it is not clear that the relationship will remain constant in a changing climate.

As the result of soil drying and less water uptake by plants, transpiration will reduce under such conditions. However, higher CO_2 leads to lower stomatal conductance and higher leaf photosynthesis rate (Morgan et al., 2004). Thus, plants in the future will use water more efficiently to fix atmospheric C than at present. Nevertheless, water shortage, temperature, humidity, vapours pressure and soil nutrients could limit C fixation.

Impacts on seasonal processes
Climate change will have significant impact on seasonal processes e.g. gross primary productivity (GPP) and respiration, causing disturbances to the terrestrial-atmosphere C-flux balance (Falge et al., 2002). GPP is the main driver of land carbon sequestration. It plays a key role in the global carbon balance and partly offset anthropogenic CO_2 emissions (Janssens et al., 2003). Higher temperatures due to climate change have positive impacts on plant productivity by enhancing the photosynthesis when temperature is in a range of optimum level.

As temperature exceeds the optimum level, it will increase the rate of respiration causing the net ecosystem productivity (NEP) to decline continuously because the increase in respiration is sharper than the increases in GPP (Grace and Zhang (200). A shift in seasonality will result in changing soil C-decomposition, gas fluxes and C sequestration. The dissolved organic carbon (DOC) output from soils results from decomposition of SOM during the dry summer months, which is removed following periods of high rainfall (Dawson et al., 2002). Nevertheless, Freeman et al. (2004) found that increased drying of soils leads to increases in CO_2 production and emissions rather than DOC.

Impacts on soil properties
Impacts on soil structure
Soil structure (aggregate stability and porosity) is considered to be an important soil property and a useful soil health indicator, due to its influential effects on water and gas movement in the soil (Allen et al., 2011). Aggregate stability, the resistance of soil aggregates to external energy such as high intensity rainfall and cultivation, determined by soil structure and a variety of chemical and biological properties and management practices (Moebius et al., 2007; Allen et al., 2011). Soil porosity, a measure of the void spaces in a material as a fraction, controls a range of soil physical indices including soil aeration capacity, plant available water capacity and relative field capacity (Reynolds et al., 2009). Recent studies to model soil water balance and ecosystem conditions under present-day and projected climatic scenarios use porosity as a model parameter (Porporato et al., 2005) because root development and soil enzyme activities are closely related to soil porosity and pore size distribution (Piglai and De Nobili, 1993).

Generally, soil structure is governed by inorganic and organic soil matters, tillage operations and some physical processes (water infiltration, bulk density, rooting depth, and soil surface cover) due to wetting / drying and freeze-thaw conditions (Rounsevell et al., 1999). Soil porosity and aeration status, beside other factors, govern CH_4 (Dalal et al., 2008) and N_2O (Dalal et al., 2003) gas emissions from soils. Soil structure can also be used to assess soil erosion (Rimal and Lal, 2009). It has a major influence on soil physical properties (Ball, 2013) which indirectly affects GHG production (Gregorich et al., 2006). Structure can override the influence of texture in regulating gas exchange, mainly because of its substantial influence on soil water content and pore continuity in soils of the same type. Soil structure affects roots ability to grow and supply leaves with water and nutrients (Passioura, 1991).

In adverse situation, it induces them to send hormonal signals that slow the growth of the shoot, even if they are able to take up sufficient water and nutrients (Passioura, 1991). These effects of soil structure on plant growth will also affect GHG emissions from soils. Climate change could influence soil structure by modifying soil physical processes and the amount of SOC available in soils (Carter and Stewart, 1996). Future low summer precipitation will lead to soil shrinkage and increased cracking, especially in clay-rich soils (Harrison et al., 2012). In a study in north Wales, Domınguez et al (2015) found that soil respiration in podzolic (organo-mineral) soils from wet shrublands was more vulnerable to recurrent drought than to warming, and that the drought impact did not diminish at decadal time scales. This stimulation of soil respiration was due to changes in soil structure which led to 54% reduction in water holding capacity. They suggested that changes in sub-dominant vegetation and soil physical properties may determine the climate change impacts on soil C dynamics.

Geris et al. (2015) examined short-term effects of extreme drought on storage dynamics and runoff response in hydropedological units in a headwater catchment in the Scottish highlands using isotopes experiments. They found incredible small storage changes in histosols compared with those in moorland and forested podzols. They suggested that during dry periods, large parts of the catchment are disconnected from the river network and run-off is generated mainly from the wet histosols however, during events, there was an intermittent connection of the hillslopes that recharged the wetland and stream. This participated to recovery and resilience of the catchment in its run-off response.

Impacts on soil pH
Soil pH is the degree of soil alkalinity or acidity and has a great impact on nutrient availability and micro-organism activity in soil. Changes in rainfall amount and intensity, and increased temperature due to climate change, will influence leaching intensity and soil mineral weathering, and thereby soil pH. In a European simulation study on forestry soils, Reinds et al. (2009) reported that climate change would lead to higher weathering rates and nitrogen uptake and limited positive effects on recovery from acidification compared to current climate.

Evans et al. (2005) used the MAGIC dynamic model (Model of Acidification of Groundwater In Catchments) and climate scenarios of rising temperature, and decreasing rainfall in the UK, to estimate impacts of climate change on soil and water recovery from acidification. They reported that high dissolved organic carbon (DOC) and elevated organic acidity, due to higher temperature, is expected to lower soil pH and increase leaching of basic cations into surface waters, bringing about recovery of these waters from acidity. However for agricultural soils, under climate change, lime could be used to control acidity. Soils with high clay and organic matter content are more able to resist a drop or rise in pH whilst sandy soils are more vulnerable to acidification.

Clay content cannot be modified, but organic matter content can be changed by following best land management practices. Salinization due to weather extremes e.g. drought and rainfall periods under climate change could inhibit biological N transformation (Curtin et al., 1999), N fixation capacity by legumes (Delgado et al., 1993) and decrease plant growth (i.e. less N use efficiency and higher N loss in the form of gases and leaching). Further, a combination of high pH and sodium has detrimental impacts on soil properties with implications for ecosystem function and services. This combination is associated with colloid dispersion, loss of organic carbon, decrease in soil permeability, and increase in run-off and erosion (Defra, 2011). However, salinity due to higher evaporation rates in relation to irrigation applied is not considered a likely problem in the UK, even under climate change (Defra, 2010).

Impacts on traffic-ability/ workability
Trafficability is the capability of soil to permit movement of a vehicle over the land surface whilst workability the number of days in a given period suitable for field work (Reeve and Fausey, 1974).Climate change will affect both field traffic-ability and workability (Campbell and O'Sullivan, 1991). Soil moisture status is the most important influencing factor in determining the traffic-ability / workability of a land. Wet soils, during critical periods for management operations such as harvesting and ploughing will limit machinery access, and soils could be subject to compaction and structural damage (Earl, 1997). Reduction in

rainfall could improve soil workability in some wet and heavy soil areas in Scotland (MacDonald et al., 1994). Projected climate change scenarios showed a small increase in the number of workable days in the UK, due to lower soil water content at higher temperatures. However, localised threat of soil compaction will remain (Defra, 2010). A recent simulation study by Harding et al. (2015) suggests that future field operations across the UK will start earlier due to the dramatic decrease in frosts, especially in the highlands. Thus, growing seasons will be longer, which may increase crop productivity and carbon returns to the soil.

Impacts on soil biota

Soil is a haven for a huge diversity of bacteria, fungal species and animals. These organisms make up soil food web which is essential for the functioning of terrestrial ecosystems as its key role is to recycle unused organic matter derived from the above-ground (Bardgett 2005). Changes in future rainfall, including drought and flooding, will directly impact soil biota through changing soil water, and indirectly through changing the soil habitat e.g. shrinkage and swelling of clay rich soils (Harrison et al., 2012). Drought could result in less microbial biomass, less microbial activity, and the death of some larger soil organisms (Gordon et al., 2008; De Vries and Shade, 2013).

Climate change can also impact soil biota by increasing soil erosion, especially where extreme events increase and where climate-change induced changes in land use and management increase soil vulnerability to erosion (Nearing et al., 2004). A recent review by Blankinship et al. (2011) investigated the impacts of climate change on soil biota and reported that microbial biomass is significantly increased by elevated CO_2, but bacterial abundance is negatively affected by warming, and fungal biomass increased with increasing precipitation. Elevated CO_2 leads to higher plant photosynthesis and growth, and thereby increasing the rates of carbon input to soil, which in turn strongly modifies the growth and activity of soil biota (Phillips et al., 2011; Phillips et al., 2012).

Warming of the climate could stimulate the exudation of carbon from plant roots (Yin et al., 2013) and thus microbial feedback to climate change (Grayston et al., 1996); it increasing bacterial and fungal biomass, and affects the structure of food webs (De Vries and Bardgett, 2014). Nielsen et al. (2011) reported that changes number of functional groups present in soil and the diversity of these groups can result in significant effects on ecosystem functioning. Further, the effects of climate change on the composition and functioning of soil food webs could result in changing plant communities under the new condition (De Vries et al., 2014). However, changes in plant communities might be more important than direct effects of climate change for ecosystem functioning (Ward et al., 2013).

Impacts on soil fertility

The availability of soil nutrients has a great influence on plant growth and water use. Soil fertility is essentially related to SOM (Brevik, 2013). Climate change could cause soil degradation through erosion and losses of SOM. Low SOM content decreases soil fertility as a result of low nutrient and reduced water holding capacity, leading to a deterioration of soil structure (King et al., 2005). Higher mineralisation under climate change will also increase N loss by ammonia volatilisation (Rounsevell et al., 1996). Mineral and drained organic soils with sufficient amounts of OM are more productive than soils that have low organic matter.

Agriculture contributed 26% of the English total phosphorus (TP) load, 22% of the Scottish TP load, 57% of the Welsh TP load and about 47% of the TP load to inland and coastal waters of Northern Ireland (Defra, 2006). Agricultural soils are rich with P because of application of phosphate fertilizer for many years (Donnison, 2011). Intensively managed grassland could represent a source for reactive phosphate due to manure application and accumulation of P. The principle pathway for P to enter water is by erosion and overland flow. Enrichment of an ecosystem with N and P leads eutrophication that causes algae and higher forms of plant life in water to grow too fast and stimulate the growth of certain on land plants. This can disturbs the balance of organisms present in the water and the quality of the water concerned and make some on land plants dominant so that the natural diversity would be lost (Defra, 2011).

In a climate change simulation study for north England, Bouraoui et al. (2002) found higher nutrient uptake and high crop growth; however, nutrient (N and P) losses to surface water were increased due to climate accelerating soil processes such as mineralisation of SOM, and increasing water loss from the soil profile to rivers in case of extreme rainfall events. They expected great impacts on soils and crop management, and suggested a need for adjusting current management practices. To decrease N and P losses management of permanent pasture should be improved by reducing stocking density, nutrient input, run-off and erosion (Defra, 2009).

Impacts on the soil key functions

Impacts on SOC stock

The relationship between soil functions and SOC is strong (Tate, 1992). Changes in SOC take place over a long period of time and are governed by many factors such as climatic (e.g. temperature and precipitation) and edaphic (e.g. soil parent material, clay content, cation exchange capacity) factors (Dawson et al., 2007). Chapman et al. (2013) reported no change in the overall total soil C stock at 100 cm depth, for 25 years period, across the Scotland. However, they found C stock for soils under woodland (mainly coniferous plantation) significantly increased. By recalculating the C stock to a depth of 15 cm the overall C stock (when deep peat sites were excluded) was significantly increased. Though both improved grassland soils and those initially under arable cultivation showed a significant decrease in C content. Another study by Reynolds et al. (2013) covering a 30 years' period (1978-2007) also reported no change in soil C across Great Britain for the depth of 15cm. Further, Hopkins et al. (2009) found no consistently significant changes in SOC stocks due to climate over a long-term (>100 years) experiment on grassland soil.

Bellamy et al. (2005) reported that climate change led to SOC loss from England and Wales at a rate of 0.6% per year over the period between 1978 and 2003. However, Smith et al. (2007) found that the main driver for C loss is land use change and climate change was responsible for only 10-20% of the amount of SOC loss reported by Bellamy et al. (2005). Using isotope experiments, Evans et al. (2007) also reported these negative impacts of agriculture intensification in the UK on the loss of SOC. By reanalysing the data of England and Wales (1978-2003), Kirk and Bellamy (2010) suggested that modifications in land use (conversion of natural vegetation and grassland to arable land) and improved management practices (drainage, mineral fertilizers and stocking rates) were the most significant drivers for the C loss than climate change.

More recently, the analysis of Barraclough et al. (2015) suggested that only 0-5% of soil C change in mineral soil could be attributed to climate change, though 9-22% of soil C change in organic soils could be due to climate change However, although there is a conflict between the evidences regarding the impacts of climate change on SOC in forests and grasslands, consistent evidences show reduction of SOC in arable agriculture due to climate change (Goidts and Van Wesemael, 2007; Chapman et al., 2013). Ciais et al. (2010) reported that management and land use will have more significant impacts on future SOC than the climatic factors. Rees et al. (2005) found that the increases of crop yield under climate change did not guarantee an increase in SOC. They observed an increase of 4.26 t ha^{-1} in wheat yield between 1948 and 2001 but this did not prevent SOM loss (Jastrow et al., 2007). With climate change, the NPP and litter input will increase and could compensate for the SOC loss (Smith et al., 2006; Gottschalk et al., 2012). Thus, the amount of SOC depends on the balance between the impacts of future increased temperature and decreased soil moisture on decomposition rates and C losses from decomposition and C gains from higher plant productivity (Smith, 2012).

Fluvial C fluxes also represent an important part of the C budget; they constitute an important pathway for carbon loss from organic soils. Soil C losses increase through losses of DOC and particulate organic carbon (POC). DOC is the carbon included within SOM in solution. Schulze and Freibauer (2005) reported that DOC loss had originated from the SOM, therefore contributing to the overall decrease in SOC. The loss of DOC is controlled by rainfall but also, among others, is associated with solar radiation and temperature (Harrison et al., 2008).

Long-term increase in DOC has a severe negative impact on soil biota, water industry (i.e. increase the cost) and C stock (Evans et al., 2005). McCartney et al. (2003) found a clear rising trend in DOC concentrations in a stream draining a mature forest catchment in Scotland from around 5 mg l^{-1} in the early 1980s to around 16 mg l^{-1} in 2003. Climate change is expected to influence the loss of DOC from soils to rivers and lakes in the UK. Freeman et al. (2001) reported an increase of 65% in DOC concentration in freshwaters draining from 20 sites, of different soils, land uses and locations in the UK, and concluded that rising temperatures increased DOC loss from soils. Evans et al. (2005) reported an increase in DOC by 91% in the UK water over 15 years period. However, in a long-term study, Worrall et al. (2003) found that the increases in DOC concentrations in rivers coincided with increases in mean summer temperatures. Worrall et al. (2004) suggested that DOC is most likely driven by temperature and the frequency of severe droughts. Worrall and Burt (2004) confirmed that climate, especially severe drought, is a main driver for DOC loss.

Pawson et al (2008) reported that POC accounted for 80% of a very large aquatic C flux of 93g C m^{-2} y^{-1} in the south Pennine catchment. Oxidation of this POC amount will create a great C loss in the form of CO_2 (Schlesinger, 1995). However, Smith et al. (2001) argue that only a small fraction of the eroded POC is decomposed and released as CO_2 to the atmosphere. Changes in rainfall due to climate change will influence runoff, which drives POC (Dawson et al., 2002). Furthermore, Chaplot and Cooper (2015) noted a tendency for clayey soils, which were fully covered by grass, to present stable aggregates and thus to yield greater CO_2 emissions, but lower POC and DOC outputs, than degraded sandy soils of low aggregate stability.

Impacts on greenhouse gas emissions

Production and release of GHGs are essentially due to biological processes; however, soil physical condition can impact biology by affecting the soil physical environment (Gregorich et al., 2006). Changes in temperature (Fiscus et al., 1997) and precipitation (Izaurralde et al., 2003; Mearns, 2003) due to climate change will influence mineralization and denitrification, and thereby GHG production. The loss of SOM under high temperature will reduce soil fertility and consequently deteriorate soil structure. Higher temperatures influence microbial activity in soils and increase GHG emissions to the atmosphere (Ball, 2013). Climate change would reduce porosity and soil aeration resulting in reductions in rates of CH_4 oxidation and CO_2 release from decomposition, but increasing rates of denitrification and N_2O release. Smith et al (2003) found that gas diffusivity was very important for CH_4 oxidation rate. Well-aerated, moist soil is suitable for CH_4 oxidation and CO_2 release (Ball, 2013).

However, higher temperature alone could reduce emissions of CH_4 by changing the hydrology of the soil. Temperature increases evapotranspiration and thereby lowers the water table leading to decreased CH_4 emissions, since aerobic conditions enhance CO_2 release relative to CH_4 (Moore et al., 1998). Moreover, changes in rainfall could lead to changes in the water table, and thus CH_4 emissions from soils (Mojeremane et al., 2010). Nitrous oxide production and emissions in soils is stimulated by temperature and water filled pore space (WFPS; an indicator for soil aeration status) (Smith et al., 2003). Dobbie and

Smith (2001) found a linear relationship between WFPS and mean N_2O fluxes for a grassland soil in Scotland.

Land management with climate change (i.e. climate-induced changes in land management) can have substantial impacts on soils. Graves and Morris (2013) found interaction between the projected land use management scenarios and climate change resulted in soil degradation. Schils et al. (2008) estimated the loss of SOC in Europe due to land use change especially drainage, by 20-40 tonnes of CO_2 $ha^{-1}y^{-1}$. Lowering water table in organic soils by drainage increases CO_2 loss but decreases CH_4 emissions to the atmosphere (Smith et al., 2003). However, restoration could reduce C fluxes, but may increase CH_4 emissions (e.g. Waddington and Day, 2007). Precision application of N (Pierce and Nowak, 1999) and management to reduce the fallow period and/ or minimise soil disturbances reduce N_2O losses and thereby, GHG emissions (Osborne et al., 2010). Webb et al. (2001) reported that the loss of mixed farming will result in soil C loss.

This is because the association between crop and animal outputs can give suitable inputs of nutrients to soils and protect the crop. The decrease in SOC content in the English arable soils is due to replacing grassland in mixed farming rotations by permanent arable cropping (King et al., 2005). In a literature review, Post and Kwon (2000) reported that land use change from arable cropping to grassland would increase soil C by 33 g C m^{-2} y^{-1} however, rainfall intensity and the species sown in the new pasture can considerably affect the rate. Tillage practices and climate variations affect the release of GHGs from soils. No-tillage increases N_2O emissions, but decreases CO_2 emissions (Ball et al., 1999; Abdalla et al., 2013). Changes in vegetation cover could alter runoff and nutrient losses as well as SOM content through C input to soils. Soil ploughing increases CO_2-evolution (Reicosky and Archer, 2007) because ploughing increases soil disturbance, crop residues distribution (Grigera et al., 2007) and microclimate i.e. increases soil temperature (Vinther and Dahlmann-Hansen, 2005). However, an application of combined management to reduce N and C emissions could be a useful approach to prevent trade off and swapping of emissions between the GHG gases CO_2, CH_4 and N_2O (Schils et al., 2008).

Holman et al. (2005) predicted a possible future reduction in the arable land area in East Anglia and conversion of upland grass to arable in the Northwest as a result of economic pressure under future climate scenarios. They also predict the introduction of new crops such as sunflower, grain and forage maize and the altering of current rotations. However, soil erosion and loss of SOM have the potential to limit the ability of farmers to take advantage of future opportunities to increase agricultural production (MLCC, 2013). Preserving upland vegetation cover is a key win-win management strategy that will reduce erosion and loss of soil carbon, and protect a variety of services, such as the continued delivery of a high quality water resource.

Further, the growing of energy crops such as willow and *Miscanthus* which increase SOM and mitigate GHG emissions (Hiller et al., 2009) could be encouraged under future climatic condition (Evans et al., 1995). Land use change will also be moderated by potential policy goals that seek to reduce GHG emissions from land and / or increase the size of land-based sinks. This will include strategies to reduce C and N fluxes through increased efficiency, afforestation and biomass production.

Conclusions

We have reviewed the impacts of anthropogenic climate and atmospheric CO_2 concentration change on soil functions in the UK. Currently, soil erosion is high especially in England and Wales, nitrate leaching is high in some localities but ammonia emission is decreasing. Also, arable agriculture is currently losing SOC. Climate change will accelerate soil processes, and could lead to more rapid decomposition of SOM, increased microbiological activity, quicker release of nutrients, increased rates of nitrification and denitrification, increased nitrate leaching and soil erosion. These changes in soil processes will influence soil structure, pH, trafficability / workability and fertility. However, the extent of this influence depends on the size of future changes in climate and on the interaction between the influences of different parameters. Climate change would reduce CH_4 oxidation and CO_2 release from decomposition, but increase rates of denitrification and N_2O release. Under climate change, the NPP and litter input will increase and could compensate for potential SOC losses due to increased decomposition rates.

The net SOC stock will therefore, be determined by the balance between the C losses from decomposition and C gains from the high crop productivity. However, this will significantly be affected by the climate-induced changes in land use and management. Most of the available evidences show that management and land use systems and not climate change will have the greater impacts on future SOC. However, the evidences from experimental and simulation researches on soils and impacts of future climate change on SOC in the UK are inadequate. To fill this key knowledge gap we suggest proposing new strategies that emphasize additional necessary researches.

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