Soil organic carbon in Wales. What is it and why is it important?

Seren
Sustainable Earth Solutions

Report prepared by: Andrew Detheridge, John Scullion, Gareth Griffith and Dylan Gwynn-Jones.

August 2014

ERDF
Llywodraeth Cymru
Welsh Government
Ewrop & Chymru: Buddsoddi yn eich dyfodol
Cronfa Datblygu Rhanbarthol Ewrop
Europe & Wales: Investing in your future
European Regional Development Fund

CARDIFF UNIVERSITY
PRIFYSGOL CAERDYDD

PRIFYSGOL ABERYSTWYTH UNIVERSITY

This report was co-funded through the Welsh Government Land Use and Climate Change Committee and the Seren programme. Seren is part funded by the European Regional Development Fund. Cardiff University and Aberystwyth University wish to acknowledge the support provided to the project by the Welsh European Union Funding Office (WEFO).

1 List of Abbreviations

µm	Micro (10^{-6}) metre
AMF	Arbuscular Mycorrhizal Fungi
C	Carbon
C:N	Carbon to Nitrogen ratio
CCW	Countryside Council for Wales
ER	Ecosystem Respiration
GHG	Greenhouse Gases
GJ	Giga (10^{9}) Joules
GPP	Gross Primary Productivity
Gt	Giga (10^{9}) Tons
ha	Hectare
Mg	Mega (10^{6}) Grams
MJ	Mega (10^{6}) Joules
Mt	Mega (10^{6}) Tons
N	Nitrogen
NECB	Net Ecosystem Carbon Budget
NEP	Net Ecosystem Productivity
ppmv	Parts per Million by Volume
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
t	Tons
UV	Ultra Violet
WAG	Welsh Assembly Government
yr	Year

1 Soil Organic Carbon

1.1 What is it and why is it important?

Soil organic carbon (SOC) is the carbon fraction of soil organic matter (SOM), which in practical terms is any material from biota in soils excluding living plant roots. This material may be as recognisable organic compounds such as lignin, polysaccharides, proteins, lipids etc. and other matter that has been modified by processes within the soil to be unrecognisable as a known biological compound.

The concentration of carbon dioxide (CO_2) in the atmosphere is currently approaching 400 ppmv, and is rising at approximately 20 ppmv per decade (Trans & Keeling, 2011). Increasing atmospheric CO_2 is of concern as the gas absorbs infra-red radiation and increases the warming effect of the atmosphere. This rapid rise in the concentration of atmospheric CO_2 is primarily due to the burning of fossil fuels. However land use change, where carbon is transferred from above ground biomass or soils to the atmosphere, is also a significant contributor (Bernstein *et al.*, 2007). Finding sinks other than the atmosphere for these carbon outputs can therefore help limit the potential warming.

In simplistic terms the Earth's carbon can be thought of as residing in four broad pools or sinks. These are the crust, the oceans, terrestrial ecosystems and the atmosphere. The terrestrial carbon pool can be further subdivided into the biotic (the carbon in living organic matter) and the soil pools. Total carbon in the pedologic pool, up to 2m depth, has been estimated by Batjes (1996) to be approximately 3000 Gt, mostly as SOC (2400 Gt). This is more than twice as large as the combined atmospheric (760 Gt) and above-ground terrestrial biotic pools (560 Gt) (Jobbagy & Jackson, 2000) and underlines the importance of soils in the global carbon cycle.

The primary flux to and from the pedologic pool is with the atmosphere as plants form the primary input into soils and the primary losses are due to biological oxidation of soil organic matter. Therefore, because the pedologic pool is so much larger than the atmospheric pool that small percentage net losses or gains of carbon can lead to much larger corresponding increases or reductions in atmospheric CO_2.

SOC is the primary constituent of organic matter in soils. On average 58% of organic matter by weight in soils is carbon (Post *et al.*, 2001), although this can vary depending on the composition of the source input material and the physiochemical nature of the soil (Howard *et al.*, 1998). As well as being a sink for carbon, soil organic matter is also an important driver for fertility as it reduces nutrient losses through leaching and promotes the formation of aggregates that improve soil physical properties including resistance to erosion (Lal, 2008).

In the UK, total SOC in the top 1m of soils has been estimated to be approximately 4500 Mt (Bradley *et al.*, 2005), this is likely to be an underestimate of total soil C stocks as peatland soils can be more than 2m deep. On the other hand, the amount of carbon stored in vegetation has been estimated to be approximately 114 Mt for mainland Britain (Milne & Brown, 1997) and 4 Mt in Northern Ireland (Cruickshank *et al.*, 2000). This indicates that the biotic pool of carbon in the UK is less than 4% of the pedologic pool, again emphasising the role of soils in sequestering carbon.

The majority of below ground carbon in the UK is found in Scottish soils (Bradley *et al.*, 2005). This is a function of climate as high rainfall and lower temperatures combine to reduce the decomposition rate of SOC. Bradley *et al*. (2005) estimated the Welsh soil carbon stock to be 340 Mt, a large proportion of which is sequestered in upland organic soils. A more recent estimate combining data from ECOSSE (organic soils of Scotland and Wales) (Smith *et al.*, 2007b) and the Bradley estimate has been derived in an analysis for the Welsh Assembly Government (WAG) (WAG, 2010). This report estimated the total soil carbon stock in Wales as 409 Mt. The difference between these two estimates is mainly due to variation in estimates of carbon below 15cm depth in upland organic soils.

2 What environmental factors control soil C stocks and whether C is sequestered?

Whether or not an ecosystem (above and below ground) accumulates or loses carbon is a function of inputs and outputs. Sequestered carbon can be defined as the difference between gross primary productivity (GPP) and ecosystem respiration (ER), which in turn is the sum of plant respiration and heterotrophic respiration of non-photosynthetic organisms. This has been termed net ecosystem productivity (NEP) (Chapin *et al.*, 2006). The final rate of accumulation or loss of carbon in a particular ecosystem (net ecosystem carbon budget (NECB)), in addition to NEP, will depend on external deposition of carbon (such as inputs of organic manures and dissolved carbon in rain water) and also loss through erosion, removal (harvesting) and non-biological oxidation through fire or ultraviolet (UV) radiation (Lovett *et al.*, 2006).

2.1.1 Input material composition

The rate of decomposition on the soil surface depends on the recalcitrance of the plant material. Woody tissue provides the greatest challenge to soil organisms, but much leaf and stem tissue is decomposed at the surface. What happens below ground is more complex, but most of the carbon inputs to soil are from root tissues and exudates, surface organic matter is incorporated into the soil via the action of soil animals and fungi (Abiven *et al.*, 2011; De Deyn *et al.*, 2008; Feng & Simpson, 2007; Leake *et al.*, 2006; Rasse *et al.*, 2005). Arbuscular mycorrhizal fungi (AMF) play an important part in carbon sequestration both through the stimulation of root exudates and also through their own exudation of a protein secretion that may stimulate soil aggregation and hence increase the physical protection of soil organic matter (Clemmensen *et al.*, 2013; Wilson *et al.*, 2009). Earthworms play an important role in moving litter from above ground to below ground and, through consuming soil particles, mix it with the mineral content of the soils enhancing adsorption (Scullion & Malik, 2000).

The fact that the pedologic carbon pool is significantly larger than the biotic pool indicates that soil carbon turns over more slowly than carbon above ground leading to accumulation until a steady state is reached specific to prevailing conditions. This implies that in soils, organic carbon is to some degree protected from oxidation. The soil properties that influence the degree of protection are therefore important in determining the potential level of soil carbon and these are discussed later.

2.1.2 Land use changes

The SOC content for a soil of a given type is a function of net inputs of organic carbon to the soil and net outputs due to respiration and physical factors such as erosion. Changes in land use can alter either inputs or output of carbon or both. When forest or permanent grassland is changed to arable cropland then above ground inputs to the soil are significantly reduced as crops are removed when harvested and in addition any residues are more labile. Furthermore, tillage will break up soil aggregates and expose protected soil carbon to the oxidising atmosphere, resulting in increased outputs of carbon from the pool and potentially increasing erosion; whilst erosion may be viewed as a loss of SOC at a local level it has been suggested that the majority of the eroded carbon is deposited at other sites (Van Oost *et al.*, 2007). In a meta-analysis of studies of the effects of land use change on soil carbon, Guo *et al*. (2002) concluded that conversion from forestry or permanent grassland to arable cropland leads to marked losses of soil carbon with clear gains of carbon when this change is reversed. The losses/gains are less clear cut where land use changes from permanent grassland to forestry or vice versa. There is often a misconception that above ground vegetation is a good indicator of sequestered carbon but there are several examples where afforestation of permanent grassland leads to a loss of soil carbon (Chapela *et al.*, 2001).

Bradley *et al*. (2005) present a summary of soil carbon data from the UK. They characterised the carbon density (kg C per m^2) and national stocks (Mt C per land use type) under 5 different land uses, semi-natural vegetation (but not woodland) where there is little or no management, permanent managed grasslands, woodlands, arable land and gardens. The highest density and largest stock of soil carbon was found under semi-natural vegetation, which includes organic soils such as peats, most of which is in Scotland. Woodlands and permanent managed grasslands have similar overall carbon density in the first 0-30cm, but for woodland the carbon density at depth is higher than grasslands. Although the total C inputs for woodlands and grasslands are similar these inputs follow different patterns. Woodlands are deeper rooted and surface inputs such as leaf litter are very seasonal, although this applies more to broadleaved rather than coniferous systems; some of this litter C is incorporated to depth in the soil profile through the activity of earthworms, although much is broken down on the surface by fungi. The main difference in terms of ecosystem (as opposed to soil) carbon sequestration between permanent grassland and woodland is that more carbon is maintained in standing biomass in woodlands. For the UK, Milne and Brown (1997) estimated that the above ground carbon stock in woodland is approximately 90 Mt or 80% of the total carbon stock in vegetation in the UK although woodland only accounts for approximately 11% of the land area. The carbon stock in woodland soils is estimated to be 400 Mt indicating that for woodland approximately 18% of the total carbon stock is in the standing biomass whereas for the UK as a whole this figure is less than 4%.

2.1.3 Climate

Climate also plays a significant role in determining soil carbon content. High moisture contents can slow decomposition leading to high carbon contents. Cold conditions not only slow plant growth but also rates of decomposition and over time carbon can accumulate. In addition to climatic influence on carbon content, changes in climate also drive changes in fluxes of carbon from soils. If rainfall reduces then organic soils that have accumulated due to wet conditions can dry and the carbon will begin to oxidise; also drying peat soils are more prone to erosion. Alternatively, soils that have accumulated little carbon due to climate being unfavourable to plant growth (e.g. deserts) may begin to accumulate carbon if the climate becomes more conducive to growth.

Across England and Wales it has been suggested that losses in soil carbon between 1978 and 2003, based on the National Soil Inventory, were mainly due to change in climate (Bellamy *et al.*, 2005), with especially high losses in organic peat soils. The authors draw this conclusion as carbon losses occurred in the absence of land use change and hence are attributed to the average 0.5 °C rise in temperature across England and Wales over the study period (Hulme, 2002). These results are inconsistent with a comprehensive study of British Woodlands, which concluded that there have been no losses of soil carbon between 1971 and 2003 (Kirby *et al.*, 2005). In addition the methods used by Bellamy *et al.* to derive levels of carbon in peat soils have been criticised (Smith *et al.*, 2007a); bulk density measures of peat land soils are not taken in the National Soil Inventory, rather these were derived. However, small fluctuations in bulk density can lead to large apparent variations in soil carbon content and so accurate soil density measurement is crucial to an estimate of soil carbon This combined with the relatively few samples taken from organic soils leads Smith *et al.* to question the finding by Bellamy *et al*. of a 2% loss in peat carbon over the period.

2.3 What soil factors and processes limit soil C stocks and sequestration rates?

When discussing carbon sequestration in soils there are a number of concepts and theoretical models that underpin our understanding of what may happen to the total carbon stocks under different cropping and management regimes. This section introduces and defines the terminology in the review.

In the simplest model of carbon sequestration the change in carbon over time is a function of input of carbon to the soil, from plant material or through additions to the soil, and outputs. These outputs or losses of carbon result from biological mineralisation or decomposition and physical losses through erosion etc. The outputs in turn can be thought of as a proportional function of the total carbon stock in the soil. In a soil with low levels of carbon and with enhanced inputs, the carbon stock increases with time. As the stock of carbon increases the outputs, as a function of the total carbon stock, also increase until they match the level of inputs and no further net carbon storage occurs. At this point the soil carbon stock is said to have reached equilibrium. If the level of input is then raised further via a regular application of manure say, then the soil will sequester more carbon until a new equilibrium is reached. This is termed the linear model of carbon sequestration and is used in many soil modelling applications (e.g. RothC, CENTURY). Some of the heterogeneity in soils and in the organic matter is captured by dividing the soil into separate pools with different decay rates - fast (active), intermediate and slow (recalcitrant) with turnover rates related to the natural recalcitrance of the input material (von Lützow *et al.*, 2006), also referred to as biochemical stabilisation.

The linear model of carbon sequestration implies that soil has an infinite capacity to sequester carbon and thus when the level of inputs is increased this will lead inevitably to a new higher equilibrium level of carbon stock. This model has been criticised as overstating a soil's capacity to store carbon, especially when soil carbon stocks are already high or when inputs are increasing. In addition, it does not accurately describe the variation in residence times of many of the different plant materials (Thevenot *et al.*, 2010).

In an alternative model of carbon sequestration (Six *et al.*, 2002), the physiochemical capacity (chemical stabilisation and physical protection) of the soil limits the carbon that can be stored. At some point all additional inputs are mineralised or lost through erosion and this is referred to as the saturation capacity of the soil. In this model turnover rates are not just limited by the recalcitrance of the input material but also by the ability of the soil to protect organic matter from decay. Some forms of carbon that are naturally labile may have long residence times in the soil if physically protected, whereas other more recalcitrant materials such as lignin may be more susceptible to decomposition if not protected.

Four pools of organic matter in soils can be thought to exist in this alternate model: chemically stabilised by clay particles (< 53 µm) (Hassink & Whitmore, 1997), the charged surfaces of these particles having a higher affinity for organic matter than the active sites of microbial enzymes; physically stabilised by micro aggregates (53 – 250 µm) (Six *et al.*, 2000), polysaccharides produced by soil organisms adsorb to charged soil particles and bind into aggregates rendering the organic matter inaccessible and limiting the diffusion of oxygen; biochemically stabilised due to the input material's natural recalcitrance; the unprotected pool which is the most active with rapid turnover rates consisting labile compounds not adsorbed or part of an aggregate.

These are important considerations when assessing carbon sequestration as it is not only the types of input that will drive sequestration but also soil clay contents and the extent of formation of micro aggregates facilitated by soil biota. This implies that the carbon saturation level of a soil/ecosystem depends upon both litter quality and the physiochemical properties of the soil. Carbon saturation gives the theoretical maximum level of carbon stocks in a soil. At a certain level of inputs the equilibrium level of soil carbon may be below this level but it provides an upper ceiling beyond which increasing inputs will not increase the organic carbon stocks.

Ingram and Fernandes (2001) sought to clarify some of the issues regarding soil carbon stocks and identified three levels of carbon sequestration. The maximum potential level is the soil carbon saturation level detailed above. If inputs were unlimited this would be the maximum carbon that the soil could retain. The attainable level is that which can be achieved given environmental constraints on the input levels of carbon to the system; without artificial inputs this is the net primary productivity (NPP) and is constrained by factors such as available nutrients, climate and photosynthetic capacity. This is the theoretical equilibrium level discussed above. The third level is the existing carbon stock achieved given the current management practices such as crop removal, tillage, drainage etc.

Ingram and Fernandes also point out that differences in time scales over which carbon is measured can lead to different conclusions as to which planting and management regime may be the most effective for sequestering carbon. In a hypothetical example the same soil type is planted with birch or oak and carbon levels measured every 5 years. In the first 30 years soil carbon levels under birch rise more rapidly due to faster growth rates and higher inputs. However, birch litter is more labile than that of oak and so as the carbon stock increases rises, outputs due to decomposition increase more rapidly than under oak. The continued growth of oak would then lead to higher rates of carbon sequestration than in the soil planted with birch until in the longer term (100 years) the oak soils achieve a higher equilibrium carbon stock. This illustrates the importance of time frames when discussing policies aimed at increasing C stocks. In the short term the level of inputs are likely to have an important impact, but if the saturation model is correct then edaphic conditions and decomposition rates will play a more significant role in the longer term.

Non-mineral soils such as peats are different in that the organic matter does not decompose because waterlogging, low pH and temperature slows the rate of decomposition. Peatland soils are the most important for carbon sequestration in the UK accounting for 50 – 60% of the total carbon stock (Ostle *et al.*, 2009). However, creating new wetland will not significantly alter the carbon budget in the short term due to the slow rate of accumulation.

Organic matter in soils can have very long residence times (Jobbagy & Jackson, 2000). However, organic inputs are thermodynamically unstable and hence should decompose more rapidly than their residence time in soil suggests; which poses the question, why should this be the case (Hedges *et al.*, 2000)? Recent research has led to a new consensus that persistence is not necessarily due to variation in the recalcitrance of different types of organic matter but is due rather to the physical, chemical and biological nature of the soils (Marschner *et al.*, 2008; Schmidt *et al.*, 2011). This emerging consensus supports the saturation model of Six *et al*. (2002) and has important implications for how we view and model soils as a carbon sink. There is a limit to the carbon storage capacity of any soil.

The majority of carbon in soils is derived from roots. Deeper rooting plants may therefore increase the potential of the deep soil to store more carbon (Kell, 2011). Deep soil carbon (below 30 cm) tends to be at lower concentrations than carbon at shallower depths; but, due to increases in bulk density at depth and the extended depth of some soils, it is estimated to account for at least 50% of global soil stocks (Jobbagy & Jackson, 2000). In addition sub-soil organic matter is characterised by very slow turnover rates (Rumpel & Kogel-Knabner, 2011). It would seem, therefore, that deeper rooting plants may increase carbon storage. However, recent work suggests that there may be a "priming" effect of new root organic matter in the deep soil horizons (Fontaine *et al.*, 2007). In this scenario the stability of organic carbon at depth is dependent upon the lack of labile organic carbon to act as an energy and nutrient source for the microbial community. If deeper rooting plants were to supply this fresh carbon then Fontaine *et al.* showed that the result is not only decomposition of the new organic matter but also some decomposition of the indigenous stable organic carbon.

In summary carbon sequestration/stock is a function of carbon input, which, without applications of a carbon-based amendment, is related to plant growth. It is also a function of the chemical nature of these inputs especially when they remain unprotected in soils. Finally and perhaps most importantly, it is a function of soil composition-structure, as this protects otherwise labile organic matter from decay; the extent of this protection is reduced markedly by disturbance, such as tillage or more aggressive forms of disturbance such as mining.

When estimating the potential of a soil to sequester carbon there are two main factors that need to be considered. The starting level of carbon and the saturation level of carbon for that particular soil.

If carbon stocks are well below the saturation potential of the soil there is larger scope for increasing carbon levels in the soil. If stocks are close to saturation not only is there little scope for increasing carbon stock, but the rates of sequestration are likely to be low. Disturbed soils have a carbon stock well below their potential, very heavily disturbed soils, such as mine spoil, may be considered as having a zero carbon stock and so offer great potential for carbon sequestration on an area basis; this will be discussed in more detail in section 4.

Changes in the nature and amount of C inputs to soils also have the potential to promote C sequestration and increase soil stocks. Crops (including bioenergy) or grasses with greater allocation of C to belowground structures, deeper rooting traits, with greater root turnover and/or higher contents of more recalcitrant compounds compared to current equivalents may lead to C sequestration and accumulation.

2.4 How are soil C stocks distributed in Wales and what is the potential for further sequestration?

Approximately half of the Welsh soil C stock is found in the 23.4% of land surface classed as uplands, emphasising the importance of these organic soils as a carbon store. The importance of the uplands and of upland subsoils for total C stocks is further underlined by C distribution patterns in Wales (see Figures 1a and b). The highest densities of C (>1000 t ha^{-1}) are found only in a small area of upland deeper peat soils. Data in Figure 1 also emphasise the major contribution of soil layers below 15 cms depths to overall soil C stocks and the varying extent of this contribution between soil types.

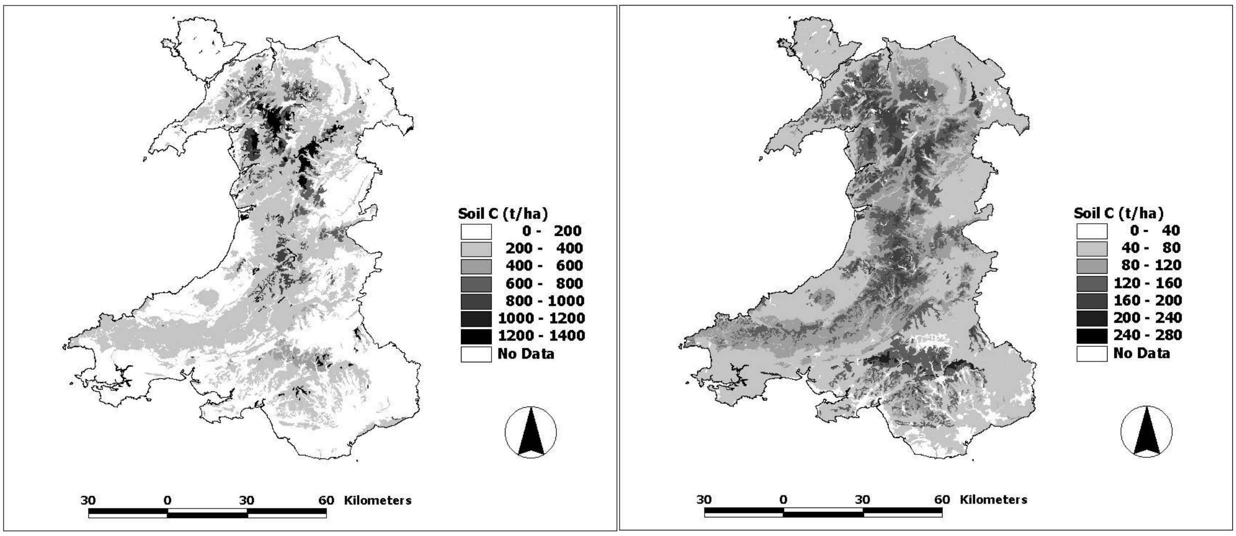

Fig 1a

Fig 1b

Figure 1. Distribution of soil C stocks in Wales to a) 100 cm and b) 15 cm depths (source Ibn Malik (2006))

The potential of a particular crop to sequester or lose soil carbon on a landscape scale will depend on the area of land of suitable habitat. Here the areas of land in Wales under particular habitats are taken from the CCW phase 1 habitat survey (Howe *et al.*, 2005) and are summarised in table 1.

Table 1. Areas of land in Wales for broad habitat types (Howe *et al.*, 2005)

Habitat Type	**Area ha**	**% Cover**
Woodland and Scrub	289,216	13.9%
Improved Grassland	1,002,236	48.2%
Semi or Unimproved Grassland	270,006	13.0%
Bracken	64,805	3.1%
Heathland	106,447	5.1%
Mire & bog	63,701	3.1%
Open Water	15,713	0.8%
Coastal edge	56,452	2.7%
Quarry/Spoil/Mine	10,264	0.5%
Other	199,060	9.6%

Almost two thirds of the land area of Wales consists of grassland of some classification, so changes in land use or species composition of these grasslands may have a significant impact on overall soil C stocks in Wales. Grassland soils, especially those in the semi or unimproved class, and land classified as bracken/heathland are often associated with the uplands. These soils have high C stocks already (see Figure 1) so their capacity for further C sequestration in topsoils may be limited; only the C deficient subsoil layers may offer the capacity for significant additions to soil C stocks. Representing ~ 14% of land cover, conversion from woodland to other more disturbed land uses might also have a marked effect on a national scale. Mire and bog cover a relatively small area but make a disproportionate contribution to total soil C stocks due to the very high densities of C on an area basis to greater depths (see Figure 1). Of the categories listed above Quarry/Spoil/Mine probably has the greatest potential for C accumulation on an area basis in that these 'soils' are deficient in C and therefore will act as a C sink on restoration; however this habitat class represents a relatively small proportion of land cover in Wales.

3 References

Abiven, S., Heim, A. & Schmidt, M. W. I. (2011). Lignin content and chemical characteristics in maize and wheat vary between plant organs and growth stages: consequences for assessing lignin dynamics in soil. *Plant Soil* **343**, 369-378.

Batjes, N. H. (1996). Total carbon and nitrogen in the soils of the world. *Eur J Soil Sci* **47**, 151-163.

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.

Bernstein, L., Bosch, P., Canziani, O., Chen, Z., Christ, R. & Davidson, O. (2007). Climate change 2007: synthesis report. Summary for policymakers. *Geneva; IPCC, 2007, 22 p mapas, ilus*.

Bradley, R. I., Milne, R., Bell, J., Lilly, A., Jordan, C. & Higgins, A. (2005). A soil carbon and land use database for the United Kingdom. *Soil Use Manage* **21**, 363-369.

Chapela, I. H., Osher, L. J., Horton, T. R. & Henn, M. R. (2001). Ectomycorrhizal fungi introduced with exotic pine plantations induce soil carbon depletion. *Soil Biology and Biochemistry* **33**, 1733-1740.

Chapin, F. S., Woodwell, G. M., Randerson, J. T. & other authors (2006). Reconciling carbon-cycle concepts, terminology, and methods. *Ecosystems* **9**, 1041-1050.

Clemmensen, K. E., Bahr, A., Ovaskainen, O. & other authors (2013). Roots and Associated Fungi Drive Long-Term Carbon Sequestration in Boreal Forest. *Science* **339**, 1615-1618.

Cruickshank, M. M., Tomlinson, R. W. & Trew, S. (2000). Application of CORINE land-cover mapping to estimate carbon stored in the vegetation of Ireland. *J Environ Manage* **58**, 269-287.

De Deyn, G. B., Cornelissen, J. H. C. & Bardgett, R. D. (2008). Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecol Lett* **11**, 516-531.

Feng, X. J. & Simpson, M. J. (2007). The distribution and degradation of biomarkers in Alberta grassland soil profiles. *Org Geochem* **38**, 1558-1570.

Fontaine, S., Barot, S., Barré, P., Bdioui, N., Mary, B. & Rumpel, C. (2007). Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature* **450**, 277-280.

Guo, L. B. & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta analysis. *Glob Change Biol* **8**, 345-360.

Hassink, J. & Whitmore, A. P. (1997). A model of the physical protection of organic matter in soils. *Soil Sci Soc Am J* **61**, 131-139.

Hedges, J. I., Eglinton, G., Hatcher, P. G. & other authors (2000). The molecularly-uncharacterized component of nonliving organic matter in natural environments. *Org Geochem* **31**, 945-958.

Howard, P. J. A., Howard, D. M. & Lowe, L. E. (1998). Effects of tree species and soil physico-chemical conditions on the nature of soil organic matter. *Soil Biology & Biochemistry* **30**, 285-297.

Howe, L., Blackstock, T., Burrows, C. & Stevens, J. (2005). The habitat survey of Wales. *British Wildlife* **16**, 153-162.

Hulme, M. (2002). Climate change scenarios for the United Kingdom: the UKCIP02 scientific report. Norwich: Tyndall Centre for Climate Change Research.

Ibn Malik, A. (2006) Terrestrial carbon in Wales *PhD. Thesis* Bangor

Ingram, J. S. I. & Fernandes, E. C. M. (2001). Managing carbon sequestration in soils: concepts and terminology. *Agric Ecosyst Environ* **87**, 111-117.

Jobbagy, E. G. & Jackson, R. B. (2000). The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* **10**, 423-436.

Kell, D. B. (2011). Breeding crop plants with deep roots: their role in sustainable carbon, nutrient and water sequestration. *Annals of Botany*.

Kirby, K. J., Smart, S. M., Black, H. I. J., Bunce, R. G. H., Corney, P. M. & Smithers, R. J. (2005). *Long-term Ecological Change in British Woodlands (1971-2001)*: English Nature.

Lal, R. (2008). Carbon sequestration. *Philosophical Transactions of the Royal Society B: Biological Sciences* **363**, 815-830.

Leake, J. R., Ostle, N. J., Rangel-Castro, J. I. & Johnson, D. (2006). Carbon fluxes from plants through soil organisms determined by field (CO2)-C-13 pulse-labelling in an upland grassland. *Applied Soil Ecology* **33**, 152-175.

Lovett, G., Cole, J. & Pace, M. (2006). Is net ecosystem production equal to ecosystem carbon accumulation? *Ecosystems* **9**, 152-155.

Marschner, B., Brodowski, S., Dreves, A. & other authors (2008). How relevant is recalcitrance for the stabilization of organic matter in soils? *J Plant Nutr Soil Sci-Z Pflanzenernahr Bodenkd* **171**, 91-110.

Milne, R. & Brown, T. A. (1997). Carbon in the vegetation and soils of Great Britain. *J Environ Manage* **49**, 413-433.

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

Post, W. M., Izaurralde, R. C., Mann, L. K. & Bliss, N. (2001). Monitoring and verifying changes of organic carbon in soil. *Climatic Change* **51**, 73-99.

Rasse, D. P., Rumpel, C. & Dignac, M. F. (2005). Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant Soil* **269**, 341-356.

Rumpel, C. & Kogel-Knabner, I. (2011). Deep soil organic matter-a key but poorly understood component of terrestrial C cycle. *Plant Soil* **338**, 143-158.

Schmidt, M. W. I., Torn, M. S., Abiven, S. & other authors (2011). Persistence of soil organic matter as an ecosystem property. *Nature* **478**, 49-56.

Scullion, J. & Malik, A. (2000). Earthworm activity affecting organic matter, aggregation and microbial activity in soils restored after opencast mining for coal. *Soil Biology & Biochemistry* **32**, 119-126.

Six, J., Paustian, K., Elliott, E. T. & Combrink, C. (2000). Soil structure and organic matter: I. Distribution of aggregate-size classes and aggregate-associated carbon. *Soil Sci Soc Am J* **64**, 681-689.

Six, J., Conant, R. T., Paul, E. A. & Paustian, K. (2002). Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant Soil* **241**, 155-176.

Smith, P., Chapman, S. J., Scott, W. A. & other authors (2007a). Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978-2003. *Glob Change Biol* **13**, 2605-2609.

Smith, P., Smith, J., Flynn, H. & other authors (2007b). ECOSSE: Estimating Carbon in Organic Soils-Sequestration and Emissions: Final Report.

Thevenot, M., Dignac, M. F. & Rumpel, C. (2010). Fate of lignins in soils: A review. *Soil Biology & Biochemistry* **42**, 1200-1211.

Trans, P. & Keeling, R. (2011) Trends in Carbon Dioxide (NOAA)
http://www.esrl.noaa.gov/gmd/ccgg/trends/ Last Accessed -

Van Oost, K., Quine, T., Govers, G. & other authors (2007). The impact of agricultural soil erosion on the global carbon cycle. *Science* **318**, 626-629.

von Lützow, M., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B. & Flessa, H. (2006). Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions - a review. *Eur J Soil Sci* **57**, 426-445.

WAG (2010). Land Use & Climate Change. Cardiff: Welsh Assembly Government.

Wilson, G. W. T., Rice, C. W., Rillig, M. C., Springer, A. & Hartnett, D. C. (2009). Soil aggregation and carbon sequestration are tightly correlated with the abundance of arbuscular mycorrhizal fungi: results from long-term field experiments. *Ecol Lett* **12**, 452-461.