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The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study

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ABSTRACT

Results from the UK were reviewed to quantify the impact on climate change mitigation of soil organic carbon (SOC) stocks as a result of (1) a change from conventional to less intensive tillage and (2) addition of organic materials including farm manures, digested biosolids, cereal straw, green manure and paper crumble. The average annual increase in SOC deriving from reduced tillage was 310 kg C ± 180 kg C ha^{-1} yr^{-1}. Even this accumulation of C is unlikely to be achieved in the UK and northwest Europe because farmers practice rotational tillage. N_2O emissions may increase under reduced tillage, counteracting increases in SOC. Addition of biosolids increased SOC (in kg C ha^{-1} yr^{-1} t^{-1} dry solids added) by on average 60 ± 20 (farm manures), 180 ± 24 (digested biosolids), 50 ± 15 (cereal straw), 60 ± 10 (green compost) and an estimated 60 (paper crumble). SOC accumulation declines in long-term experiments (>50 yr) with farm manure applications as a new equilibrium is approached. Biosolids are typically already applied to soil, so increases in SOC cannot be regarded as mitigation. Large increases in SOC were deduced for paper crumble (>6 t C ha^{-1} yr^{-1}) but outweighed by N_2O emissions deriving from additional fertiliser. Compost offers genuine potential for mitigation because application replaces disposal to landfill; it also decreases N_2O emission.

1. Introduction

There are two sets of reasons why governments, land managers and citizens should be concerned about maintaining or increasing the quantity of organic carbon (C) in agricultural soils.

1. Because organic C content has a profound influence on many soil properties and functions that affect both agricultural production and the roles that soils play in the wider environment through their biophysical and economic impacts on "ecosystem services", as recognised in the Millennium Ecosystem Assessment (http://www.millenniumassessment.org/en/Framework.aspx). This role of soil C is also emphasised in the UK National Ecosystem Assessment (UK National Ecosystem Assessment, 2011) and in discussion of soil functions in the UK Foresight study on Global Food and Farming (Foresight, 2011; Powlson et al., 2011b).
2. Because soils are a significant stock of C within the global C budget, (1500 Pg C in soils compared to 760 Pg C in the atmosphere, (e.g. Powlson et al., 2011c), changes in the soil C stock have implications for the mitigation or worsening of climate change (e.g. Smith et al., 2008).

The EU Commission has identified organic carbon (or organic matter) decline as one of the main threats to soils within Europe (Jones et al., 2004). In general, a soil with higher organic carbon content will have more stable structure than the same soil at lower organic carbon content, be less prone to runoff, erosion or surface capping and have a greater water infiltration rate, water retention and greater porosity (Snyder and Vazquez, 2005; Abiven et al., 2009; Bhogal et al., 2009; Johnston et al., 2009). It has been shown, at least in some situations, that even small increases in soil organic carbon (SOC) can markedly influence aggregate stability, water infiltration and the energy required for tillage (Blair et al., 2006; Watts et al., 2006).

Regarding the role of soils within the global carbon cycle, there has been much discussion of the possibility of mitigating climate change through C sequestration by increasing SOC stocks (e.g. Smith et al., 2008). Although the largest stocks of SOC on an area basis are often in non-agricultural situations (e.g. forests, wetlands), agricultural soils are amenable to alteration through management interventions. Also, in many regions of the world including Europe,

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agricultural soils cover very large areas so relatively small changes in C stock per unit area may translate into significant stock changes at the national or regional scale (Smith et al., 2000, 2008).

A net transfer of C from atmospheric CO_2 to land (either into soil or vegetation) is termed C sequestration and represents mitigation of climate change. It may be achieved through land-use change, such as afforestation, or though altered management of agricultural land. The latter can be accomplished in one of two ways: (1) application to soil of additional organic C that would otherwise have been returned more rapidly to the atmosphere (e.g. through burning or decomposition without incorporation into soil, such as disposal to landfill), or (2) use of practices that slow the rate of decomposition of organic C in soil. However, soil C sequestration does have well-known limitations in terms of climate change mitigation (Royal Society, 2001). First, the sink capacity for soil C is finite (Chung et al., 2010), moving towards a maximum after a period of decades or centuries as soil C content moves towards a new equilibrium value characteristic of the soil type and climate for the new management practice (Johnston et al., 2009). Second, the process is reversible, thus the new management practice leading to increased SOC must be continued indefinitely. The process is often misunderstood leading to an over-estimate of the climate change mitigation achievable; see Powlson et al. (2011c) for a discussion. On the positive side, the approach is one that can be started immediately, does not require development of new technologies and delivers a range of other benefits for soil quality and functioning as mentioned above. In this sense, the approach may be regarded as "win-win" or "no regrets". However, changes in SOC content and soil management can influence nitrous oxide (N_2O) emissions (Rochette, 2008; Sey et al., 2008) so that situations could arise where SOC is increased but the climate change benefits are counteracted by increased emission of N_2O which has a global warming potential (GWP) 298 times that of CO_2 on a 100 year timescale (Forster et al., 2007)).

This paper summarises findings from a review commissioned by the UK government to critically assess the evidence on the extent to which SOC stocks in agricultural soils could be increased through management changes (DEFRA, 2003). The full report of the review is available at: http://randd.defra.gov.uk/Document.aspx?Document=SP0561_6892_FRP.doc.

Two types of management practice were considered: (1) a change from conventional tillage to reduced/zero tillage and (2) increased application of organic materials (including greater quantities of "new" materials such as green compost and paper crumble) in addition to traditional farm manures and crop residues. The review specifically addressed England and Wales, taking account of the management practices applicable in the region and the types and amounts of organic materials available. However, the results will be more widely relevant to north-west Europe and many other temperate regions; and the reasoning adopted is applicable generically.

2. Reduced tillage

2.1. Definitions and current practice in the UK

Because the terminology used to describe different tillage systems varies in different parts of the world, the systems and terms used in the UK, and generally in Europe, are outlined here.

Most commonly, tillage crops are established in England and Wales by mouldboard ploughing to a depth of at least 20 cm (typically 20-25 cm), followed by secondary cultivations (e.g. harrowing, powered tillage, discing/tining) to provide a seedbed ('conventional tillage'). Cultivations are carried out in the autumn for all winter-sown and some spring-sown crops. 'Reduced tillage' is used to describe non-plough based cultivation practices. 'Conservation tillage', another term used in this context, has been defined as a tillage system that leaves at least 30% of the soil surface covered by crop residues (Alvarez, 2005). At the extreme, zero tillage or no-till, is where seed is drilled directly into an uncultivated soil ('direct drilling') or simply broadcast onto the soil surface. Where reduced tillage is used under UK conditions, seed is usually sown after shallow cultivation (discs or tines) working to 10-15 cm (or less), or even just following rotary-harrowing of the soil surface (i.e. combined harrow and drill techniques). In contrast to systems in many other regions, such as North and South America, reduced tillage practices in UK and northwest Europe are often carried out in conjunction with mouldboard ploughing every 3 to 4 years ('rotational ploughing') to relieve soil compaction and for weed control. In this paper we use the following terms: 'conventional tillage' (ploughing to at least 20 cm), 'zero tillage' (no cultivation, sometimes termed 'direct drilling') and 'reduced tillage' (cultivation of the surface soil to a depth of no more than 15 cm). In all tillage systems crop residues such as cereal straw may be either retained or removed; if retained they may either remain on the soil surface (if zero tillage is performed) or incorporated according to the tillage system practiced, although large amounts of cereal straw may need to be chopped before incorporation incurring a small consumption of energy.

Recent surveys in England and Wales (Anon, 2006) show that *c.*50% of primary tillage practices used mouldboard ploughing ('conventional tillage') and *c.*43% used reduced tillage methods (i.e. heavy discs, tines or powered cultivators), with direct drilling/broadcasting (i.e. no cultivation) occurring on only *c.*7% of the tillage area. The reason that zero tillage has been less popular in the UK and northwest Europe, compared to the Americas and Australia, has been the build-up of grass weeds, crop disease problems and soil compaction, all of which decrease crop yields and appear to be more prevalent in a moister climate. Also the larger crop yields achieved in northwest Europe (often 8-10 t grain ha^{-1}) leads to a larger quantity of straw which can cause problems of seedling emergence if left on the surface. The relatively small area that is under zero tillage in the UK is mainly calcareous clay soils that self-mulch as a result of wet-dry and freeze-thaw cycles, producing good tilth in a way not occurring on other soil types.

2.2. Generic issues regarding reduced tillage effects on soil organic carbon content

There are many claims in the literature that reduced and zero tillage lead to an increase in SOC, presumably through a slowing of organic C decomposition due to decreased disturbance and breaking of aggregates (e.g. West and Post, 2002). Reduced tillage was proposed in the Stern report on the economics of climate change as one option for climate change mitigation (Stern, 2006). Whilst, intuitively, slower decomposition of SOC under reduced tillage seems reasonable, several studies cast serious doubt on this as a general conclusion (Powlson and Jenkinson, 1981; Baker et al., 2007; Angers and Eriksen-Hamel, 2008; Blanco-Canqui and Lal, 2008; Luo et al., 2010).

In soil under reduced or zero tillage, SOC concentration is greatest near the surface and declines with increasing soil depth, reflecting root distribution and surface deposition of crop residues. This contrasts with soil under conventional tillage where SOC is evenly distributed within the cultivated layer (e.g. Machado et al., 2003). Consequently, for comparisons of SOC stock under different tillage systems, soils must be sampled to a depth of at least 30 cm (e.g. Poirier et al., 2009), depending on tillage depth in the conventionally cultivated comparison. Shallow sampling will inevitably show greater SOC in the reduced tillage treatment and incorrect sampling invalidates comparisons in many older studies (as discussed by Baker et al., 2007; Angers and Eriksen-Hamel, 2008; Powlson et al., 2011c). In addition, in a comparison of SOC in conventionally cultivated and reduced tillage treatments in 47 published studies (mainly from North America but including a few in South America and Europe), Angers and Eriksen-Hamel (2008) noted that an accumulation of SOC at or slightly below the depth of tillage in the treatments with full inversion tillage was common; this partially offset perceived increases in SOC near the surface in soil under reduced tillage. By contrast, Syswerda et al. (2011) found no evidence of this at a single site in North America. Despite the sub-surface accumulation noted by Angers and Eriksen-Hamel (2008) in their review, they did detect a slight trend for total SOC stock under zero tillage to become greater than under conventional ploughing as the period of the comparisons increased to 15 years or more.

There is a difficulty in sampling to depth in order to detect a possible increase in SOC stock resulting from reduced/zero tillage: spatial variation in SOC is generally greater in deeper soil layers than in plough-layer soil or its equivalent under zero-till making it more difficult to detect differences between treatments (Franzluebbers, 2009). Kravchenko and Robertson (2011) argue that this variability may be masking differences between tillage treatments in studies where deeper soil is sampled in order to obtain a "whole profile" SOC stock. VandenBygaart et al. (2010, 2011) discussed the issue of soil variability in relation to sampling depth for tillage comparisons and sampled 11 long-term sites across Canada. They detected a net increase in SOC stock under zero-till in some but not all sites, depending on the climatic region and depth of tillage in the conventional tillage comparison.

A related sampling issue is the difference in bulk density between soils under different tillage treatments - zero-till often leading to greater bulk density. Powlson and Jenkinson (1981) took account of this by sampling zero-till soils to a depth containing an equal mass of soil to that in a standardised depth of 25 cm in ploughed soil (a little greater than the usual plough depth): for four sites in the UK the average sampling depth for zero-till soils was 21.6 cm. See Powlson et al. (2011c) for a recent description of soil sampling on an "equivalent mass" basis. Such differences in soil bulk density between tillage systems draw attention to the need to express SOC data on a soil mass per area basis (i.e. as $t\,C\,ha^{-1}$ or $g\,C\,m^{-2}$) rather than as a concentration (i.e. as percentage or $mg\,C\,kg^{-1}$ soil) as normally expressed from laboratory soil analyses (Powlson et al., 2011c).

There is some evidence to suggest that increased SOC accumulation under zero tillage may be greater in tropical climates than temperate, though only if legume cover crops are included in the crop rotation (Boddey et al., 2009).

A key benefit of reduced tillage, especially zero-till, is water conservation. In regions of low or strongly seasonal rainfall and/or high evaporation this is often the main driver for adopting the system. In this situation, zero tillage often results in increased crop yields and thus greater organic inputs to soil from roots and crop residues. Angers and Eriksen-Hamel (2008) point out that this probably contributes to observations of increased SOC under zero till in semi-arid regions in both North and South America and Australia. This may well be relevant in southern Europe, but not in more humid regions such as northwest Europe where yields under different tillage systems are generally similar. A related benefit of zero tillage in dry regions is that fallow seasons, needed for conserving moisture for the following years' crop, can sometimes be eliminated. There are examples in North America of zero tillage permitting the rotation of spring wheat followed by a year of fallow to be replaced by continuous spring wheat (Paustian et al., 1997). Thus, zero tillage facilitates a cropping system that leads to increased crop residue returns to soil, and greater SOC accumulation. Again, this is not a factor in northwest Europe.

Table 1
Changes in SOC following zero tillage in the UK.

Site	Clay content (%)	Years	SOC change ($kg\,ha^{-1}\,yr^{-1}$)[a]	Reference
1. Rothamsted	20	5	−156	Powlson and Jenkinson (1981)
2. Boxworth	43	6	845	Powlson and Jenkinson (1981)
3. Headly Hall I	26	8	292	Powlson and Jenkinson (1981)
4. Penicuik I	13	10	−234	Powlson and Jenkinson (1981)
5. Jealotts Hill	n.d.	2	1365	Cannell and Finney (1973), Tomlinson (1974)
6. Headly Hall II	26	9	607	Chaney et al. (1985)
7. Penicuik II	13	23	509	Ball et al. (1994)
Mean[b] (se)			310 (180)	
95% CI			−140 to 760	

n.d.: no data; se: standard error; CI: confidence interval.
[a] SOC in zero till treatment minus that in conventional tillage; 0–30 cm depth, assuming a bulk density of $1.3\,g\,cm^{-3}$.
[b] Excluding Jealotts Hill.

2.3. *Analysis of evidence on reduced and zero tillage impacts on soil organic carbon using data from UK*

There is a very limited number of publications giving results on the impact of reduced or zero tillage on soil C under the temperate humid climatic conditions of the UK or nearby regions of northwest Europe, as opposed to a large body of data from regions of continental climate in North America or tropical and sub-tropical regions in South America and elsewhere. Table 1 summarises published results from seven UK sites.

Data from these experiments, and others in Europe, was used by Smith et al. (1998) to estimate potential rates of SOC accumulation under zero tillage compared to conventional tillage. Only five of the UK sites in Table 1 show an increase in SOC due to zero tillage and they are mostly not statistically significant (e.g. Powlson and Jenkinson, 1981). The results from Headly Hall (I and II) were from soil samples taken from the same experiment; Chaney et al. (1985) sampled one year after Powlson and Jenkinson (1981). For Penicuik (I and II) Ball et al. (1994) sampled 13 years after Powlson and Jenkinson (1981). The results from Jealotts Hill (Cannell and Finney, 1973; Tomlinson, 1974) should be treated with caution as the experiment was sampled only 2 years after the treatments were established at a site previously in permanent pasture. Thus SOC content was declining rapidly in both treatments and it is highly questionable whether the result can be regarded as representative of the situation in long-term arable soils undergoing a change of tillage treatment.

Taking an average of the UK studies in Table 1, regardless of whether or not the changes in SOC were statistically significant and excluding the Jealotts Hill result (for the reason explained above), the average annual increase in SOC in zero tillage compared to conventional tillage was $310\,kg\,C\,ha^{-1}\,yr^{-1}$ (standard error: $180\,kg\,C\,ha^{-1}\,yr^{-1}$). Because of the wide variation in results between sites, this is not significantly different from zero, though it is consistent with the small trend towards slightly increased SOC stock under zero tillage noted by Angers and Eriksen-Hamel (2008) in long-term comparisons. van Groenigen et al. (2011) reported a mean annual increase in SOC in the 0–30 cm layer equivalent to >800 kg C ha^{-1} yr^{-1} over 9 years after converting from conventional to reduced tillage in an experiment in Ireland. This seems an unusually large value and is at variance with all other reported results.

2.4. *Influence of reduced/zero tillage on nitrous oxide (N_2O) emissions from soil*

There are contradictory reports in the literature regarding the influence of tillage on N_2O emissions from soil. Examples exist of emissions from zero tillage soil being increased, decreased or unchanged compared to conventional tillage. It is clear from a review of 25 comparisons (Rochette, 2008) that the state of soil aeration is a key factor determining the outcome, and this in turn is a result of a combination of soil type, drainage status and rainfall. Many situations in the UK and northwest Europe fall into the 'poor' soil aeration category of (Rochette, 2008) so are more likely to have greater N_2O emission under zero tillage. This is consistent with the small number of direct field measurements in the UK (e.g. Ball et al., 1999; Vinten et al., 2002a; Baggs et al., 2003), France (Oorts et al., 2007), Belgium (Beheydt et al., 2008) and Finland (Regina and Alakukku, 2010). Of course, increased N_2O emissions under zero tillage is not inevitable, but will depend on rainfall in a particular year and its timing with respect to fertilizer N applications, as increased emissions appear to result mainly from increased denitrification (Baggs et al., 2003). There is also evidence of interactions with crop residues: the presence of plant residues at or near the soil surface increases the likelihood of short-term poor aeration under zero tillage (Baggs et al., 2003; Ball et al., 2008). Ball et al. (2008), using soil cores taken from field plots with different tillage and residue treatments, found that N_2O emissions were greater in the presence of cover crop residues in zero tillage compared to conventional tillage even though there was no difference in bulk density. Under laboratory conditions, N_2O emissions from the zero tillage soils were between 1.5 and 35 times greater than from conventional tillage. Due to the large GWP of N_2O, even small increases in emission are significant. As pointed out by Rochette (2008), the global warming potential of 1 kg of emitted N_2O-N ha^{-1} is equivalent to a loss in soil C of approximately 125 kg ha^{-1}.

One might expect increased oxidation of atmospheric methane under reduced tillage in view of the decreased soil disturbance, but published results on this are variable. At six sites in the boreal region of Finland Regina and Alakukku (2010) found that methane fluxes were negligible and not affected by tillage practice. In irrigated soils in a semi-arid environment in North America, Alluvione et al. (2009) observed significantly increased methane emissions during the growing season of maize under zero-tillage compared to conventional tillage. By contrast, also in North America, Ussiri et al. (2009) found that zero till soil was a small sink for methane but conventionally tilled soils were a small source.

3. Organic material additions

3.1. *Generic issues on organic material additions.*

In assessing the potential of different organic materials for increasing SOC content, three separate issues are relevant:

1. The magnitude of the SOC increase obtained from a given application of each material. For example, composted materials that have already undergone a decomposition process are likely to increase SOC more than an equal mass of fresh plant material or manure.
2. Availability of different materials. Some, such as farm manures, will be available in large quantities and thus be a significant resource at national or regional scale. Other materials may be by-products of specific processes or industries and, even if they cause a large SOC increase per unit mass added, will only be of local significance because of the limited quantity available when assessed at national scale.
3. How much of each material is available in addition to that already applied to soil? If the majority of a material is already applied to soil there may be only limited possibilities for improving utilisation. By contrast, if a material is currently regarded as waste and, for example, disposed of to landfill or burned, then there is a possibility of significantly greater utilisation.

Table 2
Typical carbon content of selected organic materials. From: Defra Manure Analysis Database MANDE (DEFRA, 2003; Gibbs et al., 2005; Bhogal et al., 2006).

Manure type	Dry matter %	Organic C % ds
Cattle FYM	25	32
Dairy slurry	6	32
Broiler litter	60	32
Digested biosolids	25	35
Green compost	65	13
Paper crumble	40	37

In addition, as with reduced tillage, any impacts on net emissions of nitrous oxide (or methane) must be considered.

3.2. *Analysis of evidence on impacts or organic material additions on soil organic carbon using data from UK*

The organic materials potentially available for application to soil are extremely diverse in their properties (Table 2). In addition, each class of material is inherently variable; the values for organic C percentages (expressed on the basis of dry solids, ds) shown in Table 2 are typical, but a wide range can occur, especially for farm manures and composts.

Within the EU, the application rate of many organic materials is limited by their N content. Within Nitrate Vulnerable Zones (NVZs), which now cover over 60% of England and 100% of some EU Member States such as Denmark, farm manure total N applications are limited to a maximum of 170 kg ha^{-1} total N. Additionally, for England and Wales, there is a field N limit of 250 kg ha^{-1} total N (S.I., 2008). For the materials listed in Table 3 two values are given. First, the mean annual increase in SOC *per tonne of material applied*, averaged across all the experimental sites identified. Second, for farm manures, biosolids (treated sewage sludge) and green compost, the SOC increase potentially possible (compared to no organic application) *if applied at the maximum permissible rate in an NVZ* (i.e. 250 kg ha^{-1} total N in any 12 month period). For paper crumble and cereal straw, potential SOC increases are shown for typical application rates.

The decomposition rate of added organic materials is greatly influenced by soil type in addition to environmental factors, with greater retention of added C in soils having a higher clay content; this is recognised in models such as RothC (Coleman and Jenkinson, 1996; Coleman et al., 1997) and CENTURY (Parton et al., 1993). However, in this review, results are averaged over all soil types represented in the experiments; for most materials these ranged from sandy loams with a clay content of 5% to clay loams with a clay content of 43%. This averaging contributes to the large confidence intervals in Table 3, but is considered a useful way of summarising the data.

3.2.1. *Farm manures*

Table 3 shows a mean annual rate of SOC increase in topsoils of 60 kg C ha^{-1} yr^{-1} t^{-1} ds, but with a 5-fold range of 20-100 kg in the 95% confidence interval. The wide range results from the

Table 3
Potential increases in SOC in topsoils following the application of a range of organic materials at 250 kg ha^{-1} total N.

Organic material	Number of sites	Application rate of dry solids[a]	SOC increase[b]		References
		t ha^{-1} yr^{-1}	t^{-1} dry solids	For maximum permitted application rate in UK Nitrate Vulnerable Zones[a] kg ha^{-1} yr^{-1}	
Farm manures	8	10.5	60 ± 20 (20–100)	630	Mattingly et al. (1975), Jenkinson and Johnston (1977), Jenkinson (1990), Bhogal et al. (2006, 2007)
Digested biosolids	10	8.3	180 ± 24 (130–230)	1500	Johnston (1975), Gibbs et al. (2006)
Raw sewage sludge	10		130 ± 20 (90–170)		Johnston (1975), Gibbs et al. (2006)
Green compost	4	23	60 ± 10 (40–80)	1400	Wallace (2005, 2007)
Paper crumble		30[c]	(60) [d]	1800	
Cereal straw	4	7.5[e]	50 ± 15[e] (20–80)	375	Nicholson et al. (1997), Smith et al. (2008), Johnston et al. (2009), Powlson et al. (2011a)

[a] Rate to supply 250 kg N ha^{-1}, maximum permitted under Nitrate Vulnerable Zone regulations in England and Wales (S.I., 2008).
[b] Mean ± standard error with 95% confidence interval in parenthesis.
[c] Expressed on fresh weight basis. Typical application rate of primary or secondary chemical/physically treated paper crumble = 75 t ha^{-1} fresh weight, supplying 150 kg ha^{-1} total N (Gibbs et al., 2005).
[d] Average SOC increase per tonne dry solids assumed to be the same as for farm manures.
[e] Expressed per t fresh weight of straw. Taken as typical application rate.

contrasting materials used in the experiments reviewed, the differing durations of the experiments, and the different soil types. The mean value includes data from three long-term experiments (>49 years of applications) with farmyard manure (FYM): in these the mean annual SOC increases were 10-22 kg C ha^{-1} yr^{-1} t^{-1} ds. It also includes a set of newer experiments (8-25 years) that included FYM, cattle slurry and broiler litter (Bhogal et al., 2006, 2007) with annual SOC increases in the range 30-200 kg C ha^{-1} yr^{-1} t^{-1} ds. The mean value in Table 3 of 60 kg C ha^{-1} yr^{-1} t^{-1} ds is rather higher than the value of 20 kg C ha^{-1} yr^{-1} t^{-1} ds derived from a set of European experiments reviewed by Freibauer et al. (2004) and King et al. (2004). This is almost certainly because the earlier reviews contained a larger proportion of longer-running experiments than in the current study and because rates of storage of organic carbon in relation to inputs decline as total SOC content approaches an equilibrium.

The trend for a decreasing rate of annual SOC increase over time is illustrated in Fig. 1, based on results from 15 of the 16 experiments with manures reviewed in the current study (Bhogal et al., 2006, 2007). The annual rate of SOC increase declined sharply after about 20 years. At two sites where manures (pig FYM or pig slurry) had been applied for only 8 years, the mean annual increase was almost 200 kg ha^{-1} yr^{-1} t^{-1} ds. At the other extreme, for the site where cattle FYM had been applied for 144 years, the annual rate of increase (averaged over the entire period) had declined to 10 kg ha^{-1} yr^{-1} t^{-1} ds. This is consistent with the concept of soils moving towards a new equilibrium SOC content as discussed by Johnston et al. (2009). The result from one treatment at one site was omitted from Fig. 1 because it appeared spurious: after 10 years application of cattle slurry SOC had apparently decreased, whilst application of cattle FYM at the same site led to an increase consistent with results from other sites.

The trend for SOC to move towards a new equilibrium value as organic materials are applied is also illustrated by the long-term data in Fig. 2. This shows the most recent data from the Broadbalk Experiment at Rothamsted, UK, showing data up to 2005. SOC is shown as C stock, expressed in t C ha^{-1}, not as % C. The data were adjusted for observed decreases in the bulk density of topsoils (0-23 cm) on plots given FYM, by including the appropriate amount of subsoil C directly below 23 cm, to ensure soil weights were comparable over time. For the treatment receiving farmyard manure (FYM) each year, starting in 1843, SOC content increased rapidly

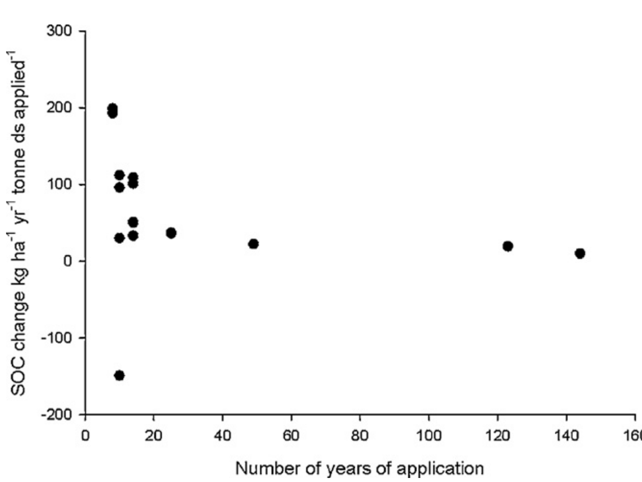

Fig. 1. Annual change in topsoil SOC per year as a result of adding the different amounts and kinds of organic matter shown in Table 3 and as a function of the time over which the additions were made.

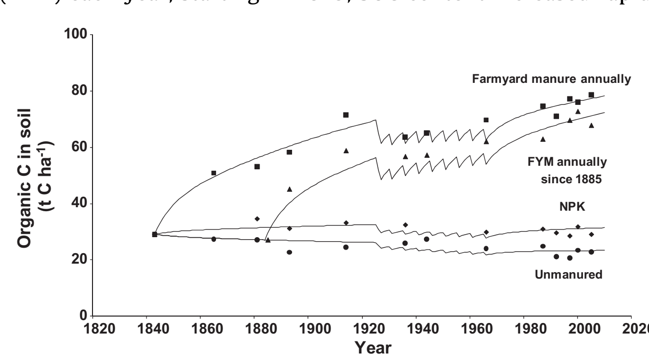

Fig. 2. Changes in SOC (0–23 cm) on Broadbalk, for plots given Farm Yard Manure (FYM) annually since 1843 (■) or 1885 (▲), and those given mineral fertilisers only (♦), including N, P and K at 144, 35 and 90 kg ha^{-1} respectively, and the unmanured control (●). Solid lines derived from RothC model.

until about 1920, though the most rapid increase was during the first 20 years. Taking output from the RothC simulation in Fig. 2, the average annual rate of SOC increase during the first 20 yr of FYM application was 1.0 t ha^{-1} yr^{-1}; during the final 20 yr the corresponding annual rate had slowed to 0.2 t ha^{-1} yr^{-1}. During the period 1926-1966 all plots on Broadbalk were bare-fallowed one year in five, to control weeds. This led to a small decline in SOC content in all treatments, but is most clearly seen in the treatment receiving FYM. After fallowing ceased (because weeds were controlled by herbicides) the increasing trend in the FYM treatment resumed but at a slower rate as the soil tended towards a new equilibrium value. If the soil had not had fallow years, the annual rate of SOC increase in latter years would almost certainly have been less. The same trend can be seen in the later FYM treatment that started in 1885. It is also clear in similar data from the treatment receiving FYM annually in the Hoosfield Experiment at Rothamsted where spring barley rather than winter wheat is grown each year, but with no fallowing (see Johnston et al., 2009). On Hoosfield the increase in SOC in the FYM treatment has continued, albeit at a diminishing rate, for about 130 years.

3.2.2. Biosolids

The mean annual rate of SOC increase from the application of digested biosolids, in the experiments reviewed here, is 180 kg C ha^{-1} yr^{-1} t^{-1} ds (Table 3). This is three times the corresponding value for farm manures, presumably reflecting the greater degree of decomposition that has already occurred during the digestion process: the organic C in biosolids, when applied to soil, is less decomposable than that in farm manures. The mean value for raw sewage sludge, which is no longer permitted to be applied to land in the UK (ADAS, 2001) but was included in some of the experiments, is slightly lower than for digested sludge (130 kg C ha^{-1} yr^{-1} t^{-1} dry solids; Table 3), reflecting the lesser degree of decomposition during treatment.

As with farm manures, the mean value for annual SOC increases masks a wide range of values, influenced by the length of time the experiments have continued, the soil types at the experimental sites and the rate of application. The mean includes a value of 80 kg C ha^{-1} yr^{-1} t^{-1} ds from the Woburn Market Garden Experiment (Johnston, 1975) where digested sewage sludge was applied to a sandy loam soil for 18 years. It also includes data after only 4 years of application in a network of nine newer experiments (Gibbs et al., 2006). In these, the annual rate of SOC increase ranges from 90 to 290 kg C ha^{-1} yr^{-1} t^{-1} ds, again reflecting the faster rate of increase in the early years and also the lower SOC accumulation in low clay soils: 90 kg C ha^{-1} yr^{-1} t^{-1} ds at a site with 8% clay compared to 280-290 kg C ha^{-1} yr^{-1} t^{-1} ds at sites with 23 or 30% clay.

The mean value for biosolids in Table 3 is larger than that of 90 kg C ha^{-1} yr^{-1} t^{-1} ds derived by King et al. (2004), probably a result of the greater predominance of longer-term data in their review. But it is not clear why our value is considerably lower than that of 300 kg C ha^{-1} yr^{-1} t^{-1} dry solids derived by Freibauer et al. (2004). The annual SOC increase achievable from biosolids applications within the maximum rate permitted in NVZs in the UK is 1500 kg C ha^{-1} yr^{-1} (Table 3). For comparison, this is slightly lower than the average annual rate of 1700 kg C ha^{-1} yr^{-1} measured over 34 years at non-experimental sites in North America (Tian et al., 2009). The higher rate over a long period in that study is not surprising because biosolids were being used for the reclamation of an area that had been subject to strip mining, so soil at the site started at a very low SOC content.

3.2.3. Green compost

This material is derived from composting grass cuttings, tree prunings and leaves from gardens and parks. The quantity produced has increased greatly over recent years as municipal authorities have sought to minimise the amount of organic material disposed of in landfill. In 2003-4 it was estimated that 480,000 t (fresh weight) of compost was recycled to agricultural land in the UK (Composting Association, 2005), this has risen to 1.3 M t of compost in 2007-8 (AfOR, 2009). The only data available on the impact of green compost on SOC in the UK is that of Wallace (2005, 2007). Results after 8 years of applications (1 site) and 5 years (3 sites) show a mean annual SOC increase of 60 kg C ha^{-1} yr^{-1} t^{-1} ds (Table 3). This is similar to the annual rate of increase of 70 kg C ha^{-1} yr^{-1} t^{-1} ds found in an experiment in Austria after 12 years (Ros et al., 2006). It is also similar to increases from farm manures on the basis of SOC increase per tonne applied. However the permissible application rate in NVZs is greater than for farm manures because of the lower N content of green compost. Thus, the increase in SOC that is practically achievable in many areas is more than twice that from farm manures (Table 3). The C retained in soil in these experiments is 23% of the organic C applied, about twice the corresponding value for farm manures, reflecting the additional decomposition occurring during treatment and production.

3.2.4. Paper crumble

Paper crumble is a waste product from the paper making industry, comprising fibres derived from the wood used for making pulp that cannot be used in paper manufacture. Around 700,000 tonnes fresh weight (FW) of paper crumble is applied to agricultural land in England and Wales (Gibbs et al., 2005). As yet, there is no data on the increases in SOC produced by this material in the UK, though two years data from experiments with paper mill residues in the USA showed very large SOC increases (Foley and Cooperband, 2002). Paper crumble and farm manures were found by Bhogal et al. (2007) to have similar contents of C in recalcitrant lignin-type material (20% and 30% of total C, respectively) and of readily decomposable C (80% and 70%, respectively). Thus, it seems likely that the increase in SOC per tonne material applied will be similar to farm manures: this has been assumed in Table 3. However, paper has a very low N content compared to farm manures. Accordingly, as for green compost, the actual increase in SOC that is practically achievable is larger on a per hectare basis. ‘

3.2.5. Cereal straw

The mean value for SOC increase in Table 3 is 50 kg C ha^{-1} yr^{-1} t^{-1} fresh weight of straw, similar to the value of 70 kg C ha^{-1} yr^{-1} t^{-1} derived by Smith et al. (1997). It represents data from four experimental sites with contrasting texture (6-24% clay) and straw application rates ranging from 4 to 18 t ha^{-1} fresh weight; measurements were taken at between 6 and 17 years since straw incorporation began (Nicholson et al., 1997; Johnston et al., 2009; Powlson et al., 2011a). As with the other organic material additions, the range of values reflects the range of conditions, especially soil type. At sites where more than one rate of straw was returned, the increase per tonne of straw was generally smaller for the higher rate. The mean annual increase in SOC (375 kg C ha^{-1} yr^{-1}; Table 3) for a typical straw application rate of 7.5 t fresh weight ha^{-1} yr^{-1} is slightly smaller than the annual rate of about 500 kg C ha^{-1} yr^{-1} observed over 8 years from straw incorporation at an experiment in Ireland under similar climatic conditions to the UK (van Groenigen et al., 2011). Although there is clearly a trend for cereal straw addition to increase SOC, the rate of change is usually small and differences between with and without straw treatments in experiments are often not significant: this was found in reviews of such experiments by Lemke et al. (2010) and Powlson et al. (2011a).

4. Discussion

4.1. General issues regarding management impacts on soil organic carbon content

In summarising data from numerous experiments (Tables 1 and 3) we have expressed rates of increase in SOC on an annual basis, i.e. as kg C ha^{-1} yr^{-1}. This is convenient and permits comparisons between management practices, but the approach has clear limitations that must be taken into account. As stated above in Section 3.2.1, SOC does not increase or decrease at a linear rate in response to a change in management, but moves from one equilibrium level to another over a period of years. The actual equilibrium value is a function of soil type, climate, cropping system, organic matter inputs and a range of management factors, and is a balance between rates of organic inputs and decomposition. Thus, if a change in management such as reduced tillage or organic material addition causes an increase in SOC content, the rate of increase will be more rapid in the early years and then decrease as the new equilibrium value is approached, eventually leading to no further increase. This is clearly illustrated in Figs. 1 and 2 in relation to farm manure applications, but is equally valid for any management change as discussed by Johnston et al. (2009). It is essential that the finite nature of SOC increases is taken into account in any consideration of soil C sequestration.

4.2. Impacts of reduced/zero tillage on SOC and overall greenhouse gas emissions

Taking the value for SOC increases in Table 1 at face value, irrespective of whether or not they are significantly different from zero, gives an annual increase in SOC under zero tillage compared to conventional cultivation of 310 kg C ha^{-1} in the 0–30 cm layer, but with a large standard error (180 kg C ha^{-1}; see Table 1). This value is slightly lower than the "best estimate" suggested by Smith et al. (2005) of 400 kg C ha^{-1} yr^{-1}, but higher than the more recent estimate used in the Stern Report for cool-moist climates (mean 140 kg C ha^{-1} yr^{-1}, with a suggested range of between zero and 280 kg C ha^{-1} yr^{-1}; Stern, 2006; Smith et al., 2008). It is reasonably consistent with the tendency for a small SOC accumulation in the long term under zero tillage noted by Angers and Eriksen-Hamel (2008). This small increase, concentrated near the surface and combined with other changes caused by zero tillage, may be beneficial for soil quality and functioning on some, but not all, soil types.

According to Bradley et al. (2005) the mean SOC content (0–30 cm) of arable soils in England and Wales is 70 t C ha^{-1} so a mean annual increase of 310 kg C ha^{-1} would represent 0.4% of current values. If this increase could be achieved on all the arable land in England and Wales, and was continued for 10 years, the total SOC stock in these soils would increase from 361 to 375 Tg C (using the values for SOC stock given by Bradley et al., 2005). This increase of 1.4 Tg C yr^{-1} (equivalent to 5.1 Tg CO_2-eq yr^{-1}) is equivalent to 0.8% of annual UK emissions (635 Tg CO_2-eq yr^{-1}; see http://www.theccc.org.uk/topics/uk-and-regions), and so not negligible. However these are very optimistic assumptions as only 50% of arable land is currently under conventional cultivation with 43% already under reduced tillage (Anon, 2006).

If, as a first approximation, it is assumed that the annual SOC increase from converting reduced tillage to zero tillage is half the increase for conventional to zero tillage conversion (say 150 kg C^{-1} ha^{-1} yr^{-1}) the SOC increase over 10 years from a total conversion (i.e. both conventionally and reduced tilled arable land in England and Wales) to zero tillage would be 10.3 Tg instead of 14.4. But even this increase is unlikely to be realised in practice for two reasons. First, experience has shown that zero tillage is difficult to apply on many soil types in the UK and northwest Europe due a combination of soil type (especially poorly structured soils) and the large quantities of straw on the soil surface compared with the amount more common under drier conditions, with smaller crop yields, in North America. Second, even when reduced tillage is used in the UK, it is usual for farmers to plough every 3-4 years, to control weeds and relieve soil compaction. This periodic tillage is likely to cause a loss of at least part of the carbon accumulated during the period under zero tillage, but there are no direct measurements of the net result of this mix of tillage treatments under the soil and climatic conditions of northwest Europe. However, it seems that the rate of loss of SOC is generally greater than its rate of accumulation (Freibauer et al., 2004). A somewhat analogous situation is in rotations of arable crops with short periods of grass, such as in long-term ley-arable experiments on contrasting soil types in southeast England, where soil C increases under the periods of grass and then declines after ploughing for arable crops (Johnston et al., 2009). On a silty clay loam soil, after 24-27 years of ley-arable cropping (3 years grass/clover, 3 years arable), SOC (0-23 cm) was only increased by *c*.15% (from 15.7 g kg^{-1} under continuous arable cropping to 18.1 g kg^{-1} in the ley-arable system). On a loamy sand, SOC levels remained constant over 30 years of a ley-arable rotation (2 years arable, 1 year grass ley undersown in the preceding cereal). There is one study from a maize-soybean rotation in North America giving an indication that so-called "biennial tillage" (ploughing in alternate years) led to a larger SOC increase than zero-till compared to annual ploughing (Venterea et al., 2006). However, the authors concluded that this probably resulted from decreased C inputs under zero-till due to lower yields with this system. Reading across from these studies to the effect of rotational ploughing systems on SOC, it would not seem unreasonable to conclude that most of the extra C stored during years under zero tillage will be released to the atmosphere as CO_2 when the soil is ploughed. Thus we conclude that converting the area currently under conventional tillage to reduced or zero tillage, using agronomic practices that are appropriate for the climatic conditions in UK and northwest Europe (i.e. rotational ploughing), would lead to little additional sequestration of soil C.

An additional benefit of reduced tillage, of relevance to greenhouse gas emissions, is the decreased use of tractor fuel compared to conventional cultivation due to the decreased number of operations. Glendining et al. (2009, supplementary material 1) quote data suggesting that conventional seed-bed preparation requires 2393 MJ ha^{-1} and reduced and zero tillage 1868 and 418 MJ ha^{-1}, respectively; in turn these are equivalent to emissions of 166, 130 and 30 kg CO_2 ha^{-1}, respectively. Watts et al. (2006) found a reduction of 11% (10 kPa) in the specific draught (S) between plots containing 1.1% SOC compared with plots containing 0.89% SOC as a result of long-term management. Reductions in S on plots with 3% SOC resulting from long-term applications of FYM were better still, although not spectacularly so (18%). Similar modest reductions in diesel consumption and thus CO_2 emissions may be expected from practices that enhance SOC.

On the basis of the above considerations it is unrealistic to expect significant climate change mitigation from reduced tillage in the UK or northwest Europe. The main factors leading to this conclusion are: (a) very small accumulation of SOC under the agronomic and climatic conditions of the region; (b) the small savings in emissions from fuel used for tillage operations; (c) the risk of increased N_2O emissions under reduced/zero tillage, at least in some years, as described in Section 2.4. It would be appropriate to make decisions on the application of reduced/zero tillage practices on the basis of other considerations (e.g. reduced risk of soil erosion, other soil quality issues, labour and cost factors) and to assume no major impact on climate change, either positive or negative.

4.3. Impacts of organic material additions on SOC and overall greenhouse gas emissions

Table 3 shows the increases in SOC, on a per hectare basis, for a range of organic material additions. Superficially these values appear large and might be taken as indicating a considerable potential for C sequestration. However, to assess the extent to which they genuinely contribute to climate change mitigation, a range of other factors must be considered. In particular: (1) the quantities of each material available at national scale, (2) the extent to which they are currently utilised, and (3) impacts on fluxes of greenhouse gases other than CO_2, in particular N_2O through effects on N fertilizer requirements and N losses.

In the case of farm manures, large quantities are available: Williams et al. (2000) estimated that 90 Mt (fresh weight) of farm manures are produced in the UK. However, virtually all of the manure produced is already applied to soil. Thus, the increases in SOC shown in Table 3 cannot be regarded as *additional* C storage to be utilised in a new strategy to mitigate climate change—this accumulation of SOC has already occurred or is currently occurring and contributing to the maintenance of the present national SOC stock. One exception is the ca. 580,000 t of poultry litter currently burned for electricity generation (less than 1% of total manure production). Another likely exception is manure currently applied to, or deposited on, grassland soils that are already high in SOC. It is likely that part of the organic C in manure deposited on the surface of grassland soil will decompose to CO_2, leaving little residue in the soil. Hopkins et al. (2009) provide some evidence for this from old grassland soils in the Palace Leas and Park Grass Experiments in UK, both of which have continued for over 100 years. SOC content in the treatments receiving farmyard manure for long periods was only slightly greater than (Palace Leas) or no different from (Park Grass) that in the controls not receiving manure. Thus total national SOC storage could probably be increased if the manure now spread on grassland soils were transported to areas of arable land that are currently low in SOC. However, there are obvious logistical and cost limitations to such a strategy.

Similarly, it is estimated that 1.4 Mt biosolids ds is currently produced in the UK (Water UK, 2010). It is estimated that ca. 1 Mt (77%) is applied to agricultural land, a further 16% is incinerated (mostly with energy recovery). Thus there is little scope to increase applications to soil and increase the contribution of biosolids to increasing the UK SOC stock. Annual straw production in England and Wales is estimated at about 11.4 Mt (Copeland and Turley, 2008). About 50% of this is returned directly to soil, with the vast majority of the remainder being used as animal bedding and subsequently in production of farmyard manure, and then returned to soil. Of course, significant decomposition of straw C will occur during the production of farmyard manure, so less straw-derived C will be returned through this route than if straw was directly incorporated. But the practical benefits from using straw as animal bedding, and then as a component of manure, are very strong. Copeland and Turley (2008) estimate that less than 2% of straw is currently not utilised in agriculture: a combination of mushroom production (40,000 t) and 200,000 t currently burned for electricity generation. The proportion could rise to about 10% if plans for new power generation schemes are realised. Thus, with both biosolids and straw, it appears that the vast majority of these materials are currently applied to soil, in some form, leaving very little scope for additional utilisation as a strategy to increase SOC stocks. The potentially negative impacts on soil quality of removing larger quantities of straw for any major expansion of its use in bioenergy were discussed by Powlson et al. (2011a).

Two materials that are relatively new and are beginning to be utilised for soil application are green compost (estimated that 480,000 t fresh weight was recycled to agricultural land in 2003–4, Composting Association (2005); this has risen to 1.3 Mt of compost in 2007–8, AfOR report (2009)) and paper crumble, a waste product from the paper-making industry of which 700,000 t (fresh weight) was recently estimated to be applied to agricultural land (Gibbs et al., 2005). Although the use of paper crumble is limited to relatively small areas close to paper-making factories, green compost is a material that is likely to become available in increasing quantities throughout Europe as part of attempts to decrease the quantity of organic material disposed to landfill. The application of these materials to soil could be regarded as a genuine new contribution to increasing SOC storage, and thus mitigating climate change within the finite limits possible from soil C sequestration.

A detailed assessment of C losses and stabilisation from land fill compared to aerobic composting followed by land application would be valuable, but is beyond the scope of this paper. With landfill disposal at least part of the decomposition process will occur under anaerobic conditions, leading to methane emission. Any comparison of the climate change impacts of the two processes would have to take account of the extent to which methane from landfill is captured and utilised.

One “new” material that we have omitted from this review is biochar. This is because, as yet, there are no studies under the temperate climate and soil types of the UK or northwest Europe on which to base any assessment of its potential to increase soil C stocks, whether through accumulation of C in a very stable form in biochar itself or through a mechanism of conferring enhanced stability on other forms of organic C. Powlson et al. (2011c) discuss the conflicting evidence on biochar impacts on soil C.

The second major issue to be considered is the impact of organic material additions on the N cycle, through partial replacement of inorganic N fertilizers and either decreases or increases in direct and indirect emissions of N_2O. Table 4 brings together estimates of these changes for a range of different organic materials applied either at a rate containing 250 kg N ha^{-1} (the maximum permissible in an individual field under UK NVZ regulations) or (in the case of paper crumble) a typical application rate.

If increased SOC storage is included, all organic materials appear to be highly beneficial in leading to a CO_2 emission saving by increasing the SOC stocks (Table 4, column F). However, as discussed above, most of the materials are already fully utilised, apart from green compost and paper crumble. So, apart from these two materials, it is incorrect to include SOC increases when assessing the potential for new contributions to climate change mitigation. Thus the final column in Table 4, that excludes SOC change (column G), is more realistic—and an order of magnitude smaller.

With the three farm manure examples (cattle FYM, dairy slurry, broiler litter) which all supply crop-available N, the saving in N fertilizer, and hence the emissions from N fertilizer manufacture, are substantial (180–300 kg CO_2-eq ha^{-1} yr^{-1}; Table 4 column C). Dairy slurry and broiler litter supply the most crop available N, leading to fertilizer N savings of about 80 kg N ha^{-1} yr^{-1}; 270–300 kg CO_2-eq ha^{-1} yr^{-1}). But these materials are also expected to give the largest losses of N through ammonia volatilisation and leaching, as well as the largest resultant emissions of N_2O (direct and indirect). These manure-associated emissions partially counteract the benefit from replacing N fertilizer—hence the negative values in column D of Table 4, indicating a net *increase* in N_2O emissions. With FYM, where N is less readily available and less subject to loss, the net increase in N_2O emission is much lower and the overall benefit compared to no manure use is greater (almost 300 kg CO_2-eq ha^{-1} yr^{-1}), compared to about 100 kg CO_2-eq ha^{-1} yr^{-1} for dairy slurry and broiler litter (Table 4, column F). Digested biosolids and green compost both provide somewhat less crop available N than dairy slurry and broiler litter, and thus less saving of emissions from replaced inorganic N fertilizer use (Table 4, column C). But they also are subject to smaller N losses, and thus smaller N_2O emissions, so

Table 4
Net changes in greenhouse gas emissions resulting from application of various organic materials.

Organic material	Application rate[a]	Crop available N[b]	CO_2-eq saving from fertilizer manufacture[c]	Net CO_2-eq changes in N_2O emissions[d]	Net CO_2-eq change due to SOC increase[e]	Net CO_2-eq saving[f]	
						Including[h] SOC change	Excluding SOC change
	A	B	C	D	E	F	G
	t or m^3 ha^{-1} fresh weight	kg ha^{-1}	CO_2-eq, kg ha^{-1} yr^{-1}				
Cattle FYM (Fresh)	42	50	180	110	2310	2600	290
Dairy slurry	83	85	305	−185	1100	1220	120
Broiler litter	8	75	270	−185	1065	1150	85
Digested biosolids	33	35	125	55	5500	5680	180
Green compost	36	15	55	55	5130	5240	110
Paper crumble	75	−60[g]	−215	−370	6600	6015	−585

[a] Quantity containing 250 kg N ha^{-1} (i.e. maximum permissible individual field rate under UK NVZ regulations). For paper crumble, typical application rate.
[b] Quantity of N estimated to become available to a crop in the year of application, and hence amount by which inorganic N fertilizer application may be decreased (using data in DEFRA, 2010).
[c] Saving in emissions from N fertilizer manufacture from decrease in N fertilizer use shown in column B. Mainly CO_2 from the energy used in manufacture, but a contribution from N_2O emitted from manufacture of ammonium nitrate (Flynn and Smith, 2010).
[d] Net result of (a) decreased emissions from decreased rate of N fertilizer and (b) emissions associated with organic materials. Based on emissions factors from IPCC (2006), but for all materials used emission factor of 1.25% instead of currently recommended value of 1% plus indirect emissions. This makes some allowance for indirect emissions but it is unrealistic to attempt more precise estimates in view of wide range of material considered.
[e] Using SOC values from Table 3.
[f] Net result of emissions changes in earlier columns either including (column F) or excluding (column G) changes in SOC content.
[g] Negative value because additional N fertilizer at 60 kg N ha^{-1} typically applied with paper crumble.
[h] Not correct to include except for green compost and paper crumble—see text.

their overall impacts (excluding SOC changes) are of the same order as farm manures (Table 4, column F).

As was argued for SOC changes, farm manures and biosolids are already being used, so the emissions savings from these materials associated with changes in the N cycle are already occurring. Thus they cannot be counted as new savings, but show the benefits already accruing through the utilisation of these organic materials. Thus even the annual savings of about 80–300 kg CO_2-eq from farm manures and biosolids resulting from their impacts on the N cycle (Table 4, column G) do not represent *additional* climate change mitigation. Only with green compost is there a real potential for additional mitigation because of the small quantities currently used and the scope for a substantial increase.

Paper crumble is a special case compared to the other materials. It causes the largest increase in SOC when applied at a typical rate but, because it contains very little N, it is customary to apply additional N fertilizer to counteract the expected immobilisation of soil-derived mineral N. The additional N fertilizer requires us to attribute a large increase in total greenhouse gas emissions (N_2O) to this application. Thus, if SOC changes are excluded, the overall impact is an *increase* in greenhouse gas emissions of >500 kg CO_2-eq ha^{-1} yr^{-1} (Table 4). On the other hand, autumn applied paper crumble could potentially be used as a means of immobilising soil mineral N and decreasing over-winter nitrate leaching losses and resulting indirect N_2O emissions. Additional studies should be conducted to test whether the current additions of N fertilizer that accompany paper crumble are really necessary. Vinten et al. (2002b) found between 1 and 7 mg N immobilised g^{-1} cellulose-C and glucose-C respectively added to laboratory soils. Most of this N was remobilised within 6 weeks. At 1 mg N g^{-1} C, a typical application of paper crumble (Tables 3 and 4), if similar to cellulose, would appear able to lock up 11 kg N ha^{-1} for 6 weeks at laboratory temperatures. This is substantially less than the 60 kg N ha^{-1} typically applied (footnote to Table 4). More work is needed here because, as Whitmore and Groot (1997) have shown, remobilisation of nitrogen depends on soil type. If further utilisation of paper crumble or similar materials were possible, the increases in SOC achieved could be regarded as additional climate change mitigation giving a net positive impact—depending on the alternative fate of the material if it were not applied to land.

5. Conclusions

Overall we conclude that there is very limited scope for additional climate change mitigation in England and Wales by increasing carbon stocks in agricultural soils, either through greater application of reduced or zero tillage or through increased applications of organic materials to soil.

In the case of reduced tillage this is because the evidence for increased SOC stock (as opposed to an increased SOC concentration near the soil surface) is highly questionable, at least in the climate of this moist temperate region and with the agronomic practices so far found appropriate. A specific issue is the need to combine reduced tillage with periodic inversion tillage: even if there is a small increase in SOC stock during the period of reduced or zero tillage, it seems virtually certain that this will be lost as a result of inversion tillage in the "rotational ploughing" system found necessary in the region. In addition, as 43% of arable land in England and Wales is already under reduced tillage (with 7% under zero tillage), and practical experience has shown which soils appear to be unsuitable for these practices, the scope for further expansion in this region is limited.

In the case of additions of organic material the situation is more complex. A major factor is that the materials available in largest quantities (farm manures, cereal straw, biosolids) are already utilised to a very large extent. Their existing addition to soil is already maintaining SOC stocks and there is very little scope for increased utilisation. A few exceptions have been discussed earlier in the paper. In addition to impacts on SOC, utilisation of these materials is beneficial from the viewpoint of decreasing requirements for N fertilizer, thus saving fertilizer-associated GHG emissions (provided the offsetting impact of N_2O from the organic materials themselves is taken into account). But, again, as these materials are already largely utilised, the N-related impacts (positive and negative) are already part of the national budget and cannot be counted again. However, there is much evidence that the

N derived from organic materials could be utilised more efficiently with significant benefits for N-related emissions. We conclude that a key exception to this negative result is the utilisation of green compost derived from household and municipal sources. Much of the material used to generate this type of compost was previously disposed to landfill, so any applications to land contribute to a genuine increase in SOC stock and thus climate change mitigation through additional C sequestration. It is clearly appropriate to encourage developments of this nature, of course taking measures to minimise any negative impacts on soil quality or public health through inclusion of undesirable materials in the composts. The carbon cost of transport of moving compost is unlikely to outweigh the mitigation potential (Smith and Smith, 2000).

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