Chapter 16

Soil Organic Matter and Carbon Sequestration

Alan Richardson, Elizabeth Coonan, Clive Kirkby and Susan Orgill

Introduction

Organic matter is a fundamental component of soil that plays an important role in a wide range of physical, chemical and biological functions. Soil organic matter (SOM) is also central to the storage of carbon (C) in terrestrial ecosystems and is the major contributor to balancing the global C budget. Agricultural practices however, are a major disruptor to this balance and historically have resulted in large losses of SOM, particularly through intensive cultivation of soils. Consequently there is current interest world-wide to improve the management of SOM in agriculture that aim to 'build and retain' C in SOM to develop more sustainable systems that mitigate climate change. Broad-acre cropping systems play a significant role in this regard and conservation agriculture (CA) based on reduced or no-till (NT) systems are purported widely to be an effective management approach to redress this. Interestingly, the role of tillage in management of SOM received very scant attention in the original edition of '*Tillage*' (Cornish and Pratley1987) and there has since been much conjecture with respect to CA practices and SOM dynamics. Nonetheless it is evident that there is need for better understanding of the influences of crop management and tillage practices on SOM.

Importance of soil organic matter (SOM)

SOM contributes to soil function through a range of processes that are associated with improved soil structure through stabilisation of aggregates, enhanced water holding capacity and/or water infiltration and in supporting higher soil fertility (Figure 1, Petersen and Hoyle 2016). SOM is also the key energy substrate for microorganisms in soil that facilitate various soil biogeochemical processes and via mineralisation, provides nitrogen (N), phosphorus (P) and sulfur (S) to plants (Murphy 2015).

Soils are the most significant store of terrestrial C with an estimated global pool of 2,500 Pg of C, which is predominantly associated (~62%) directly with soil organic C (SOC) and thus SOM (Lal 2004). Collectively, the amount of C associated with SOC is some 3.3 and 4.5 times more than that contained in atmospheric CO_2 and living biomass, respectively. Whilst dependent on many geographical, climatic and pedological factors, soils contain SOC contents that typically range from ~45 to 140 Mg (or tonnes) C/ha (to 1 m depth, with extremes up to 725 Mg C/ha), which equates to 80 to 240 Mg of SOM/ha (Lal 2004). By equivalence, this represents between 0.5% to 1.6% SOM (by soil mass) when averaged to 1 m of soil depth (at soil bulk density of 1.5 g soil/cm^3) or ~1.8 to 5.3% SOM when averaged over the top 30 cm.

It is well recognised that anthropological influences (*e.g.* land-use change, deforestation, agricultural production, urbanisation) have caused a major decline in SOM content throughout the world. Some 50 to 75% of the antecedent total SOM content is estimated to have been lost due to agricultural practice with higher rates of loss occurring in recent times (Lal 2007, Sanderman *et al*. 2017). For broad-acre cropping systems in temperate climates this equates to typical losses of between 30 and 60 Mg C/ha. In Australian soils, Chan and McCoy (2010) have estimated that at least 50% of the original C stocks have been lost in intensively managed cropping systems. This is consistent also with reduced levels of organic N in Australian soils (which, as discussed below is closely linked to SOC content), where more than 50% of the organic N that is associated with SOM has been 'mined' from soil (*i.e.* mineralised from resident SOM). Reduced soil N is associated with both more intensive cropping, along with lesser dependence on biological N fixation through reduced legume rotation. Combined, this has led to a higher dependence on mineral-based N fertilisers to support crop growth in current practices (Angus and Grace 2017).

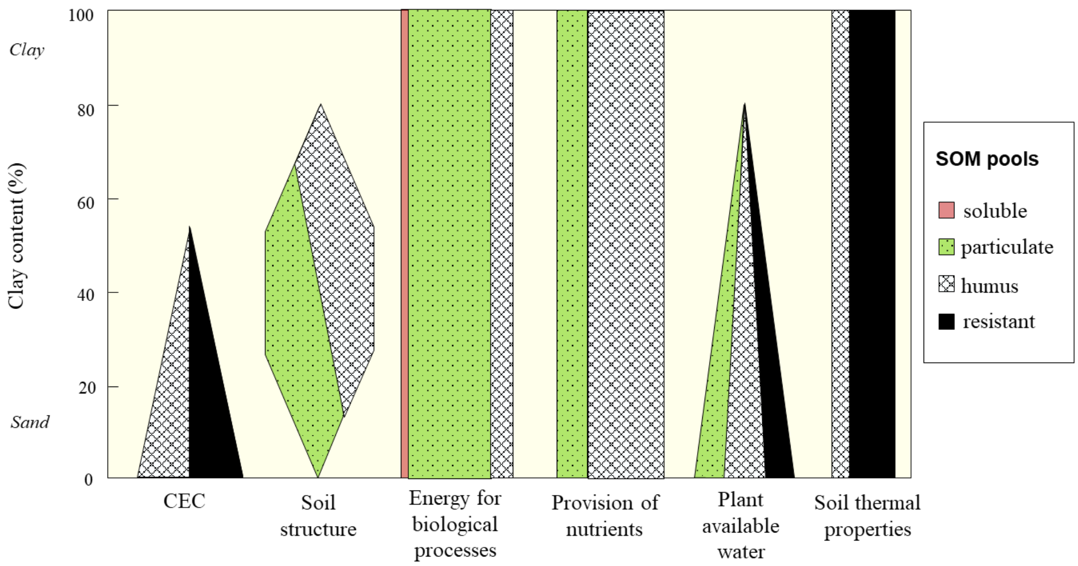

Figure 1. Conceptual role of SOM and pools of soil organic C (soluble, particulate, humus, resistant) on a range of soil functions. Object size indicates relative importance in relation to soil texture (sand to clay) (modified from Hoyle *et al.* 2011, courtesy of J. Baldock).

Rebuilding of 'lost soil C' and sequestration of soil C in managed ecosystems is therefore of considerable current interest to agricultural practitioners and environmental policymakers worldwide (Amudson and Biadeau 2018). Strategies proposed to restore soil C include; increased return of crop residues and bio-solids (manures, composts and other wastes) to soil, minimisation of soil disturbance, more continuous ground cover, a strengthening of nutrient recycling and a more positive nutrient balance, enhanced biodiversity and use of more diverse crop rotations, and a reduction in losses of water and nutrients from soils through erosion and leaching. These are associated with key farming principles in CA (Lal 2007, Giller *et al*. 2015) that promote:

- increased use of perennials;
- higher net primary productivity in agricultural systems (*i.e.* through crop choice and nutrient management); and
- greater adoption of NT farming practices with lesser soil disturbance.

Throughout Australia there has been a considerable increase in CA practices in recent decades which have led to increased return of crop residues to soil along with reduced cultivation (see Chapter 2). For example, in 2015 retained standing or surface stubble accounted for 56.8% (35.5% and 21.3%, respectively or 7.7 and 4.7 million ha) of crops, where broad-acre cereals accounted for a total area of 18 million ha (ABARES 2016). Associated with this is a significant reduction in burning of residues as a management tool. Despite this, there is conflicting evidence whether changed practice (*i.e.* increased adoption of NT and reduced burning) has led to an increase in C sequestration, with many studies finding little or no effect even when practised for a decade or longer.

Origins of SOM and pools of soil C

Development of cropping systems to effectively manage SOM requires understanding of SOM composition and the processes that drive the generation and degradation of SOM and the mechanisms associated with either its stabilisation or mineralisation in soils. Plant-derived C through photosynthesis is the primary source of C that contributes to SOM in both natural and agricultural systems. Plants contribute to SOM through rhizodeposition, root growth and return of above ground biomass (fresh organic matter; FOM) as litter, which includes crop residues (Figure 2). A large component of plant materials consumed by herbivores (*i.e.* grazing animals, insects and other macrofauna) is also returned to soil through excreta and death. Based on the annual C balance of a typical Australian wheat crop, ~40% of total annual C is allocated below ground in roots (20% to root growth and structure) and as rhizodeposits (10% as sloughed cells and root turnover), with ~10% of total photosynthate being released as root exudates. At a grain yield of 2.4 Mg/ha (and a typical harvest index of ~0.4) a wheat crop thus generates ~4.0 Mg/ha of stubble biomass per annum which equates to ~1.6 Mg C/ha (up to 4 Mg C/ha in high yielding crops), even without accounting for the contribution from roots and root exudates. Collectively, across Australia cereal crops thus generate around 16 to 20 million Mg C/year (40 to 50 million tonnes of stubble). In addition to its use for soil conservation purposes in NT systems or as a low-quality feed supplement, this C-rich residue has large potential to contribute to SOM. Similarly worldwide, crop residues, including cereal and rice stubbles and maize stover are a significant C resource.

SOM is the organic fraction of the soil (exclusive of un-decayed plant and animal residues) that is comprised of a continuum of materials in varying states of decay (Lehmann and Kleber 2015) and is comprised of both active and stable pools that collectively contribute to total soil C (or total SOM). Active pools are influenced more by management changes and typically account for up to 40% of SOM, and includes:

- soluble C (~2 to 5% of SOM) that is largely in soil solution and easily extracted in water;
- microbial C (~5 to 10% of SOM) that is the component in living microbial biomass; and
- particulate organic matter (POM; ~25% of SOM) which is derived from the immediate breakdown of residues most recently returned to soil.

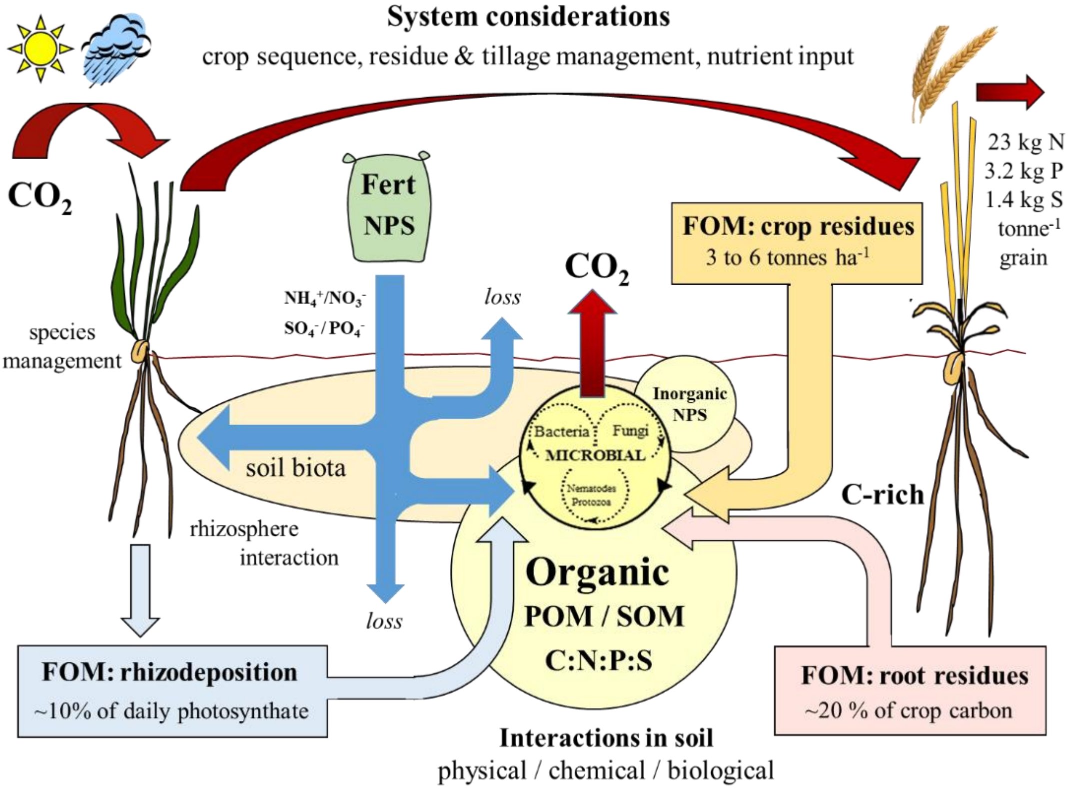

Figure 2. Schematic representation of carbon flow in plant-soil systems highlighting the interaction between inputs generated through net primary productivity and C pools in SOM

The soluble (or dissolved organic C) originates from root and microbial exudates and soluble compounds that are released through cellular degradation. Being predominantly in soil solution, soluble C (or soluble SOM) is readily metabolised by microorganisms or, through higher mobility, may be leached into deeper soil horizons (Strahm *et al*. 2009). Land-use impacts on soluble SOM are subject to seasonal fluctuations, but are enhanced by crop root growth (*e.g.* in spring) that contributes soluble C through exudate and turnover (Haynes 2005). Microbial biomass is the 'living component' of SOM and includes microflora (fungi, bacteria, archaea and smaller organisms such as phages) that collectively constitute ~90% of the biomass, with microfauna (10 to 100 µm, including protists and nematodes) making up the remainder (Glaser *et al.* 2004). Turnover of this pool of SOM is rapid (days to weeks) and is mediated by microbial growth and cell death that is strongly influenced by seasonal conditions; *viz* temperature and moisture (Hagerty *et al*. 2014) and management practices (*e.g.* use of agrochemicals and tillage), that in particular disrupt fungal hyphal networks (Grandy and Robertson 2006). Distribution of C between fungi and bacteria and their relative dominance in different agricultural systems is suggested to be a major mediator in SOM dynamics (Six *et al*. 2006).

Soil POM is formed initially from the breakdown of FOM inputs that included litter and excreta, green manure crops, organic fertilisers (manures and composts) and crop residues. POM is comprised of decomposing residues, fungal hyphae, fine plant roots and other associated biomass and is an intermediate stage between FOM and more stable SOM (Janzen *et al*. 1992). The POM (50 to 2000 µm fraction) and has a short turnover time (1-2 years) and is more readily mineralised in soil following system disturbance and cultivation (Hoyle *et al.* 2011). Thus, management practices that affect either the rate of residue input (*e.g.* fertilisation, crop rotation, crop yield, tillage practice, periods of fallow) or influence the rate of residue decomposition have major influence on changes in the size of the POM pool (Schmidt *et al*. 2011). Cultivated soils and low production systems, for instance, typically have less POM than undisturbed soils under native systems, forests and/or plantations and pastures. The efficiency of conversion of POM to SOM (*i.e.* net humification efficiency) is a key factor to understand the accumulation and stabilisation of soil C, with rates of conversion of FOM-C to SOM-C of between 10 to 30% typically reported (Kirkby *et al*. 2014).

The more stable and predominant fraction of SOM (60 to 80 % in most soils) consists of both humic (fulvic and humic acid fractions) and non-humic substances (including identifiable biopolymers and complex amines) that, compared with POM, are more resistant to degradation. The more stable component of SOM is commonly termed 'soil humus' which by definition is associated with the <53 µm soil fraction (Baldock *et al*. 2013, Hoyle *et al.* 2011). Various studies have shown that other fine fractions of soil (<400 µm sieved) are similarly associated with the more stable component of SOM (Kirkby *et al*. 2011, Magid and Kjærgaard 2001). This pool of more stabilised soil C has a slower turnover rate that is typically in the order of 2 to 3% per annum. Stable SOM is largely constituted by microbial detritus and to a lesser extent by lignified material from plant cell wells. As such, it represents a key target for effective sequestration of soil C.

The resistant fraction of SOM also includes charcoal, which depending on soil type and prior history, generally accounts for 0 to 10% of total SOM. Although not inert, charcoal in soil largely resists decomposition even after cultivation and has a half-life measured in centuries (Baldock *et al*. 2013).

Dynamics of SOM and stabilisation in soil

Degradation of FOM and POM involves the physical and biochemical decomposition of organic material followed by repeated processing (mineralisation) by soil microorganisms (Chen *et al*. 2014). During mineralisation, most of the C from the FOM/POM is returned (*i.e*. some 70 to 90%, depending on extent of humification efficiency) to the atmosphere as CO_2 (Stockmann *et al*. 2013). Through decomposition and mineralisation there is a concomitant release of inorganic nutrients (N, P, S) that are either re-utilised by microorganisms, are available for crop growth or undergo further interactions within the soil. This includes fixation reactions for P; adsorption and precipitation, denitrification and leaching for N, and depending on soil type, potential leaching of S and P.

SOM thus originates directly from FOM inputs that are added to soil or through POM that is either recalcitrant to mineralisation or, through microbial processing and generation of organic debris, becomes stabilised in soil. Historically, lignified plant material (which constitutes up to 20% of FOM) has been considered to be a major contributor to stable SOM, whereas more recent evidence indicates that microbial detritus is the more significant component. The processing of FOM and POM by microorganisms generates 'new' microbial biomass that when turned-over creates significant amounts of detritus. Accordingly, microbial detritus has been shown to make up to at least 50% of stabilised

SOM (Miltner *et al.* 2012, Ma *et al*. 2018). Comparatively, the microbial detritus contribution to SOM is some 40 times larger than the living microbial biomass. Management practices that provide large inputs of more labile C (*e.g.* application of nutrient-rich manures) that significantly increase microbial biomass in soil have been shown to contribute to increased accumulation of SOM (Engelking *et al*. 2008). Likewise, crop residues with appropriate management (especially through deliberate nutrient enrichment, as discussed below) similarly have potential to contribute to increased levels of SOM.

The concentration (or cumulative stock) of SOC in soil at any one time reflects the balance between FOM supply and SOM loss by decomposition and erosion. This net balance of SOC is the result of complex interactions between environmental factors and land management (Hoyle *et al.* 2011, Orgill *et al*. 2014, 2017a). Climate and soil type in particular explain a considerable proportion of the variability in SOC within a given land use, largely due to direct effects on net productivity (biomass production) and rate of decomposition. Once in the soil, the fate of FOM/SOM is mediated by soil temperature and moisture, the decomposer microbial community (see Chapter 15) and the degree of SOM protection against decomposition.

SOM is stabilised through either resistance to mineralisation or by interaction within the soil matrix and subsequent protection within soil aggregates (Gupta and Germida 2015). Stabilisation involves interaction with soil mineral surfaces, particularly in soils with high clay content (Figure 1), which also reduces accessibility to microorganisms (von Lützow *et al*. 2006, Eldor 2016). Generally, SOC content is greater with increased precipitation and clay content, and decreases with an increase in temperature. Thus, climate largely regulates SOC in the surface layers, while clay content largely determines SOC in deeper soil layers. SOM in deeper soils is also considered to be more protected from mineralisation through its isolation from microorganisms that dominate in the surface layers. Generally, there is also a positive correlation between aggregation and SOM concentrations (Jastrow *et al*. 2007). Tillage is a key process that disrupts macro-aggregates in soil and has been shown to lead to subsequent losses of SOM (Six *et al*. 2004). Accordingly, increased frequency of cultivation increases the rate of loss of SOM, whereas reduced tillage is proposed widely as a means for the protection and subsequent accumulation of SOM.

Nutrient stoichiometry and its importance in SOM dynamics

The interaction of microorganisms in mediating SOM stabilisation in soil and directly contributing to SOM formation is further supported by the similarity in nutrient composition of SOM (*i.e*. stoichiometry of C:N:P:S) within the soil microbial biomass, and from the ^{13}C isotope enrichment 'signature' of the microbial biomass and that of stabilised SOM (Dijkstra *et al.* 2006, von Lützow *et al*. 2006). The ratio of C:N:P:S in SOM of agricultural soils across the globe is relatively constrained and is tightly linked across soils from both natural and agricultural ecosystems, with relatively little impact from management, soil type or climate (Cleveland and Liptzin 2007). This consistency in nutrient stoichiometry for SOM is even stronger when P is specifically considered in organic forms (Kirkby *et al.* 2011), and is consistent with the fact that most of the N and S in soil (>90%) occurs in organic form that (along with organic P) is intimately associated with SOM.

Importantly, the elemental nutrient ratio that occur in more stable SOM (*i.e*. typically 70:6:1:1 for C:N:P:S) is similar to the ratio that is found in the microbial biomass (60:7:1:1 for C:N:P:S), as compared with the wide range of ratios commonly found in plant residue inputs (263:5:0.5:1 C:N:P:S for wheat residue and 102:2:0.3:1 for canola residue). Likewise, the nutrient ratios in the POM fraction are considerably wider than in stable SOM and are closer to the ratios found in originating plant residues (Cleveland and Liptzin 2007, Kirkby *et al*. 2011, Richardson *et al.* 2014). Collectively, this indicates that the processing of FOM to SOM by microorganisms requires a 'concentration' (enrichment) of nutrients (*i.e.* narrowing of CNPS ratios) to reach that which is present in stabilised SOM (Tipping *et al*. 2016). This is particularly important for FOM inputs that have wide nutrient ratios (such as C-rich stubble) and means that the 'efficiency' of conversion of stubble to SOM is strongly mediated by the availability of nutrients (van Groenigen *et al.* 2006, Kirkby *et al.* 2014).

Microorganisms in soil primarily utilise C in FOM as an energy source for growth and production of microbial biomass. This growth requires a stoichiometric balance of nutrients (N, P, S) to meet microbial demand. Depending on inherent soil fertility and the quality of FOM (*i.e.* nutrient composition) that is returned to soil, these nutrients are either co-obtained from the decomposed FOM, or are directly acquired from soil or soil solution which, depending on the stage of crop development, may be in direct competition with plant demand. Thus, addition of C-rich crop residues to soil can in many cases 'induce' short-term nutrient deficiencies (especially N) that may limit crop growth (*e.g.* each tonne of stubble returned typically 'ties up' around 5 kg N/ha).

Alternatively, when deficient, microorganisms are also able to obtain nutrients to support their growth through re-utilisation (mineralisation) from previous generations of microbial biomass (*i.e.* fresh microbial residue) or effectively can directly 'mine' pre-existing SOM to obtain N, P or S. The balance of such processes is largely governed by the quality of C inputs and nutrient availability and is commonly referred to as the 'priming effect' (Kuzyakov 2010). This priming effect may lead to a direct increase in SOC formation (where SOM accumulates; known as a negative priming effect) or, more commonly with C-rich crop residues, results in increased rates of SOM mineralisation, where levels of SOC actually decline via a positive priming effect (Fontaine *et al*. 2004, Kuzyakov 2010, Stockmann *et al*. 2013).

The priming effect on SOM dynamics is driven by microbial demand for nutrients when provided with C-rich substrates. This means that retention of crop residues with wide C:N:P:S ratio, for example through conservation farming practices, may not necessarily increase SOM sequestration when nutrients are limiting (Kirkegaard *et al*. 2014, Baker *et al*. 2007). On the contrary, the provision and use of legume residues generally leads to an increase in the net sequestration of SOM as the legume residues have a higher nutrient quality (as compared with cereal stubble), thus providing more nutrients for microbial degradation of the plant residue with lesser dependence on mineralisation of pre-existing SOM (Drinkwater *et al*. 1998).

Recent studies under controlled laboratory conditions have shown that the addition of supplementary nutrients (based on nutrient ratios of stable SOM) alongside C-rich FOM inputs can lead to an increase in SOC accumulation. This increase in SOM was sufficient to overcome the mineralisation of 'native' SOM (*i.e.* SOM existing prior to the addition of FOM) arising from a positive priming effect (Kirkby *et al.* 2013, Orgill *et al*. 2017b). For example, in the incubation study by Kirkby *et al*. (2014), ^{13}C-labelled wheat straw was added to four contrasting soil types with markedly differing clay contents (8% to 60%). A three-fold increase in the conversion (gross humification efficiency) of stubble-C to SOM-C was observed where the straw was added with supplementary N,P and S. Importantly, it was demonstrated that addition of nutrients with the straw led to both a substantial increase in new SOC formed as well as a reduction in the loss of pre-existing SOC (SOM), resulting in an overall increase in net humification efficiency (*i.e.* from -17 to 10% without nutrients to 15% to 40% with nutrients across the soils). The supply of nutrients (N, P and S) with wheat straw reduced the need for soil microorganisms to mineralise pre-existing SOM to obtain nutrients, and thus can mediate either positive or negative change in SOC (Fontaine *et al*. 2004, Kirkby *et al*. 2014). Orgill *et al.* (2017b) demonstrated a similar concept in pasture soils using an incubation experiment to show that soils with high SOC concentrations were able to continue to accumulate SOC with increasing FOM inputs (*i.e.* C inputs), but only when additional nutrients were supplied. This mechanistic understanding of microbial mediated C dynamics in soil is important to interpret changes in the levels of soil SOM under field conditions (van Groenigen *et al*. 2006) in response to different soil types and crop and pasture management systems, including the adoption of different tillage practices.

Management of SOM and sequestration of C in agricultural systems

There is evidence that management can be adopted to build SOM in agricultural soils through a range of practices (Sanderman *et al*. 2010, Lal 2017, Poulton *et al*. 2018) including:

- crop and pasture sequence (especially through a pasture and/or legume phase);
- use of cover crops;

- amendment of soils with C-based materials (*e.g.* bio-solids, manures);
- nutrient inputs (fertiliser) to promote plant growth and/or the humification of crop residues;
- innovative tillage and residue management.

In particular, these strategies:

- increase the amount of above- and below-ground FOM inputs to soil with facilitation of microbial biomass generation and turnover;
- affect the location of FOM inputs within the soil profile; or
- influence the rate of FOM conversion to more stable forms of SOM.

Pastures and pasture leys in crop rotations

Under comparable pedo-climatic conditions managed grassland and pasture systems (like undisturbed natural systems) can support higher levels of SOM than intensively managed cropping soils. Conversion from cropping to pasture or use of a pasture phase in a cropping sequence can increase SOC content (Table 1). Reduced disturbance of soil under pasture and enhanced soil aggregation leads to increased protection of SOC (Six *et al*. 2004). Generally, there is also higher levels of biomass return in pasture systems via plant deposition and return of animal excreta, with less export of biomass.

In a meta-analysis of the impact of land-use change on SOC concentrations, Guo and Gifford (2002) found that the SOC stocks increased by on average 19% after the transition from crop to pasture, with the length of time since conversion having no clear effect on the amount of C accumulated. The rate of SOC accumulation (sequestration) under pasture depends largely on soil type and climate, as compared with other management factors (Conyers *et al.* 2015, Rabbi *et al*. 2015, Sanderman *et al*. 2010). Nonetheless under long term pastures, SOM sequestration is increased particularly with the inclusion of legumes (with significant inputs of biological N) and with fertilisation (particularly P and S fertiliser in Australian soils) that increase net primary production (Haynes and Williams 1992, Orgill *et al.* 2017b). By contrast selection of grass species, including differences between annual and perennial species and/or introduced grasses verses native pasture plants had lesser impact (Schwenke *et al*. 2013, Conant *et al.* 2017). In the study by Coonan *et al.* (2019) a net difference in C accumulation of 12 Mg C/ha (19 % increase from 61.7 to 73.6 Mg C/ha) was observed over a 20 year period in a soil under a legume-based pasture fertilised with P and S and dependent on N from biological fixation. Whilst largest difference in C concentration was observed in the 0 to 10 cm layer of soil, soil C stocks were significantly increased to a depth of at least 60 cm. This is consistent with other studies that have shown typical annual rates of C accumulation in improved pasture soils under Australian conditions of around ~0.5 Mg C/ha/year (Table 1). Accumulation of SOC has similarly been observed in cropping soils that are either returned to pasture or during the pasture phase of a crop system, particularly when soils with initially low SOC concentrations were converted to pasture both in short or longer term rotations (Table 1).

Conversely, pasture management can have limited or no impact on SOC sequestration in cases where the growth potential of the pasture is limited by inadequate soil nutrition or where the composition of the pasture sward has low legume content (Badgery *et al.* 2014). The impact of management practices and land use change on SOC sequestration is indeed variable and measurements may also be limited by spatial and temporal variability in sampling. This includes, high pre-existing levels of SOM, poor sensitivity in measuring a change in soil C (against a large background), and lack of consideration of soil depth in calculating changes in either the concentration or stocks of SOC (Badgery *et al.* 2014, Robertson *et al.* 2016). Additionally, reported rates of soil C sequestration, such as those summarised in Table 1, are not necessarily linear and are likely to only be maintained for finite duration, being constrained in the longer term by soil characterstics and climatic factors (Conyers *et al.* 2015, Sanderman *et al*. 2010).

Table 1. Rate of soil C sequestration for pasture and the pasture phase of crop-pasture rotations in response to management factors (dns = data not specified).

Land use and location	Mean rate C sequestration (Mg C/ha/yr)	Management factors	Years	Soil depth (cm)	Reference
Permanent pasture					
Global meta-analysis	0.66 0.28 0.57	Legume pasture Grazing management Fertilisation	25 (0.7 to 200)	38 (2 to 800)	Conant *et al.* 2017
Brazil	0.61	Grazing management and fertilisation	8.8 (4 to 14)	30	Maia *et al*. 2009
Brazil	1.17 0.66	Legume pasture Non-legume pasture	10	30	Tarré *et al*. 2001
NSW, Australia	0.50	Annual and perennial pasture	13	30	Chan *et al*. 2011
ACT, Australia	0.60	Fertilisation and increased stocking rate	20	60	Coonan *et al.* 2019
Australian meta-analysis	0.1 to 0.3	Fertilisation, legumes and irrigation	dns	30+	Sanderman *et al*. 2010
Crop to permanent pasture					
Global meta-analysis	0.87	Cultivation to pasture	25 (0.7 to 200)	38 (2 to 800)	Conant *et al.* 2017
Global meta-analysis	0.33	Conversion from agricultural use to grassland	26 (6 to 81)	29.6 (5 to 300)	Post and Kwon 2000
Europe	0.56	Cropland to grassland	32 (16 to 41)	80	Poeplau and Don 2013
NSW, Australia.	0.48	Crop to perennial pasture	22	10	Young *et al*. 2009
Martinique	1.50	Cultivated to pasture	5	30	Chevallier *et al.* 2000
Australian meta-analysis	0.3 to 0.6	Fertilisation, legumes and irrigation	dns	30+	Sanderman *et al.* 2010
Crop to pasture in rotation (long-term)					
Denmark	2.5 (0.8 to 3.9)[1]	Converted from cereal crop to a 6-year arable pasture (clover) crop rotation	1 -5 years of ley phase	25	Müller-Stover *et al.* 2012
USA	1.9 (1.2 to 2.7)[2]	Wheat with ryegrass/clover leys of different ages	2-6 years of ley phase	23	Johnston *et al.* 1994
UK	0.7 (g C kg^{-1} $year^{-1}$)[3]	Grassland/clover leys following annual tillage	3 years of ley phase	12	Clement and Williams 1964
Denmark	1.1 (0.3 to 1.9)	Rotation with ley grass phase and barley crop phase	1-6 years of ley phase	20	Christensen *et al.* 2009
Crop to pasture in rotation (short-term)					
Sweden	0.36 to 0.59	Ley grassland/cereal crop rotation with 3 years of grassland and 1 year of crop	4 year rotation for 35 years	20	Börjesson *et al.* 2018
USA	0.15	Corn/wheat/clover rotation with fertilizer and manure	3 year rotation for 26 years	100	Buyanovsky and Wagner 1998
NSW, Australia	0.26	No till wheat with pasture (clover) rotation	2 year rotation for 25 years	30	Chan *et al*. 2011
NSW, Australia	0.23	Pasture in a crop rotation (33% to 67% pasture content)	2-6 year rotation for 18 years	30	Helyar *et al.* 1997
QLD, Australia	0.65	Cultivated with grass/legume leys	4 year rotation for 10 years	4	Dalal *et al*. 1995

[1] Sequestration calculated compared with a treatment with 0 years of pasture, determined using a bulk density (BD) of 1.55g soil cm^{-3}

[2] Calculated using a BD of 1.65 g soil cm^{-3}; C sequestration was compared with a treatment with 1 year of pasture

[3] Reported as an increased in % soil C calculated using appropriate adjustments for variation in soil volume; BD was not reported

Residue and tillage management in continuous crop systems

The trajectory of SOC levels (*i.e.* SOM) in farming systems is determined by the balance of organic materials returned to the soil (especially after harvest, Figure 2) and the amount of C lost by either microbial respiration or physical processes. Management practices that have major influence on this balance in continuous cropping systems are the amount and type (quality) of the residue inputs, tillage practice and level of fertilisation (van Wesemael *et al*. 2010, Sanderman *et al*. 2011).

Inputs Whilst return of C-rich crop residues to soil (inputs) would be expected to contribute to the build-up of SOM compared to removal or burning, the results from many studies are unclear (Table 2). In the meta-analysis conducted by Lehtinen *et al.* (2014), SOC levels tended to increase with residue retention with increasing response from duration of retention, and most notably was evident in soils with clay content >35%, which is consistent with other studies. In a review of various long-term experiments, Powlson *et al.* (2011) concluded that whilst most studies tended to report 'greater' SOC accumulation with residue retention, differences were statistically significant in a minority of studies only. In a long term experiment of our own (Harden, NSW, Australia), SOC levels from a NT system have been compared with a conventionally cultivated system (with stubble retained or burnt in both cases) with essentially no difference in SOC levels occurring after 25 years, although there were effects on SOM distribution within the uppermost layers of the soil profile (unpublished data).

By contrast to crop stubble, green manure cover crops or legume rotations consistently increase SOM. In a review from 37 trials, Poeplau and Don (2015a) found that soils under cover crops had significantly higher SOC stocks than associated reference crops, with a mean annual change of 0.32 Mg/ha/yr to a soil depth of at least 20 cm. While cover crops consistently increase SOC levels, their use in many parts of Australia is limited by water availability. Manures, composted recycled organics (RO) and other bio-solids similarly have potential to build SOM in agricultural soils (Gibson *et al*. 2002, Poulton *et al.* 2018). Typical increases in SOC are equivalent to 0.008 to 0.08 Mg C/ha per tonne of RO applied in the top 10 cm, with typical rates of manure application of 5 Mg ha^{-1} being applied. Whilst there is wide range in humification efficiency of RO products (5 to 50%), it compares favourably with the 4.6% efficiency reported for stubble retention under Australian conditions (Heenan *et al*. 1996), and the 15% average typically reported for plant residues (Lal 1997). The major existing barrier to the widespread use of RO products in the agricultural sector is the prohibitive cost of materials, transport and handling, and logistical factors associated with broad-acre application.

Tillage The effects of soil cultivation practices and intensity of tillage on mineralisation and/or sequestration of SOM has received wide attention, particularly in view of the increased adoption of NT. A predominant finding is that retention of crop residues in NT systems can lead to greater SOM levels in surface soil layers, but due to stratification within the soil profile can lead to lower SOC levels at depth (Baker *et al*. 2007). By contrast, incorporation of residues into soil has been shown to increase SOC deeper in the soil profile (Alcantara 2016).

Several studies support that NT sequesters more SOC compared with CT (*e.g.* Syswerda *et al*. 2011, Varvel and Wilhel 2011). In most cases though, where sampling was >30 cm, increased levels of SOC at the surface in NT systems were generally offset by increased SOC levels at depth under conventional cultivation (Table 2). In addition, it is evident that surface retained SOM was generally less decomposed (*i.e.* higher POM component) and would be more prone to further decomposition, and thus readily lost from the system over time (Wander and Bidart 2000). Furthermore, it has been shown that incorporation of stubble-C into microbial biomass was greatly facilitated by cultivation and soil mixing as compared with surface retention (Helgason *et al.* 2014). SOM at depth in conventionally cultivated systems also appears to be more recalcitrant to mineralisation and is more aligned with stabilised SOM (Alcantara 2016). In our own long-term study (Harden, NSW), there was no difference in SOC levels after 25 years under residue retained systems with either NT or CT when measured to 1.6 m depth. Whilst SOC levels appeared higher in 0-30 cm layer with NT system compared with cultivation, the overall differences in SOC were not significant. Interestingly, even when residues (standing stubble) were burnt, the SOC levels in the (0-30 cm) layer were equal with both CT and NT, and again were not different over the whole soil profile.

In considering the adoption of tillage and management practices to promote C sequestration, it is important to recognise the impact of both the distribution of SOC throughout the profile and the potential of the material to contribute to stabilised SOM. Accordingly, the effectiveness of NT to sequester SOC compared with CT may be greatly reduced, negated or even reversed when the whole profile is considered (Baker *et al*. 2007, Angers and Eriksen-Hamel 2008, Luo *et al*. 2010, Chatterjee 2018, Powlson *et al.* 2014). Additionally, the effects of soil bulk density (BD) need to be considered to convert concentrations of SOC (% or mg C/kg soil) to C-stocks (Mg C/ha). Use of a fixed depth sampling to measure total SOC can introduce bias when compared with SOC stocks in soils that are subject to management induced changes in BD. This is particularly relevant for shallow sampling depths (<30 cm), where higher BDs generally occur under NT systems (Aguilera *et al*. 2013, Don *et al*. 2011, Palm *et al*. 2014). For example, as reviewed by Meurer *et al.* (2018), the positive benefits of NT observed in the 0-30 cm layer of soils from long-term trials were overestimated in more than 50% of cases when BD was not considered. Significant differences in SOC concentration observed between NT and CT were also negated when determined using an equivalent soil mass (Du *et al*. 2017).

The influence of gravel in soil (especially at depth) is of further importance, both for its effect on soil BD and recent findings that have shown that gravel may also contain significant amounts of SOM as a coating (*i.e.* up to 10% of total SOC in deep soil layers, Kirkby *et al.* unpublished), but is routinely excluded from soil analyses. Finally, the methodology used for C determination is important (*i.e.* such as simple oxidation methods) that generally over-estimate the POM fraction of SOM (*i.e.* relative to analytical-based techniques). This generates further bias between C in surface layers and C at depth by underestimating the amount of more stable SOM in surface soils under different tillage systems.

Fertilisation Based on evidence that higher quality nutrient-rich residues are more effective in contributing to SOM and the apparent nutrient requirements of the microbial biomass to process C-rich residues into SOM (*i.e.* stoichiometric interactions as discussed previously), over-fertilisation of crops or direct fertilisation of crop residues (nutrient enrichment) has been proposed as a useful management tool to promote C sequestration. Increased fertilisation has either increased SOC levels or had no effect, despite in most cases having a positive effects on net primary production (Table 2). Direct application of nutrients to crop residues is effective in enhancing the formation of stable SOC through increased humification efficiency under controlled laboratory incubations (Moran *et al*. 2005, Kirkby *et al*. 2013).

Extending this approach to the field, Jacinthe *et al*. (2002) showed that humification efficiency for wheat residue-C to SOC was increased over a 4 year period from 14 to 32% with deliberate application of nutrients to stubble. Kirkby *et al*. (2016) similarly demonstrated this in the field (Harden, NSW), where crop residues (primarily wheat; average input 9 Mg stubble/ha/year) were incorporated over a 5 year period into soil to ~15 cm depth as soon as possible after harvest, with or without supplementary nutrient addition (NPS).

Stocks of SOC were increased across the 5 year period by 5.5 Mg C/ha over 0 to 160 cm in the soil profile where supplementary nutrients were added, as compared with a decrease of 3.2 Mg/ha where wheat straw only was incorporated, with 90% of this loss (relative to initial levels) being in the 0 to 10 cm layer (Kirkby *et al*. 2016). Some 50% of the increase in SOC in response to nutrients occurred below 30 cm and was suggested to be a result of leaching of C from the surface layers as either soluble C, colloidal material including microbial detritus, or as a result of increased root growth at depth (Kirkby *et al*. 2016). The importance of nutrients (especially N) to promote C sequestration from residues has similar been demonstrated in other field-based studies (van Goenigen *et al.* 2006, Poeplau *et al*. 2017). On the other hand, recent 'on-farm' trials conducted across 8 sites throughout south-eastern Australia (Van Rees *et al*. 2017) found no clear benefit for SOC sequestration following nutrient supplementation on stubble over 3 years when applied to residues that were incorporated immediately prior to sowing of the subsequent crop. These trials however used relatively low stubble loads and overall there was low sensitivity in C measurements (to 30 cm depth) with no differences either for treatments that had stubble removed.

The opportunity to increase humification efficiency of stubble to increase soil C through stoichiometric balanced microbial processing requires further validation. For example, better understanding of timing

Table 2. Soil C sequestration in crop systems in response to management factors including; residue retention, tillage practice and fertilisation (dns = data not specified).

Management and location	C sequestration (+/=/-) change, or % increase)	Management factors	Soil depth (cm)	Reference
Residue inputs				
South Africa	(+) with retention	Various rotations of maize, fallow, soybean and wheat residue retained or removed	10	Gura and Mnkeni *20*19
Europe	mean 7% SOC increase	47 field trials with crop residue incorporation *cf* removal. Highest response at >35% clay content	dns	Lehtinen *et al*. 2014
Italy	mean 6.8% SOC increase	Field experiment *cf* residue removal or incorporation over 20 years. No further increase after 40 years	30	Poeplau *et al.* 2017
Europe	(+) with retention	14 field studies with different levels of crop residue *cf* with complete removal. Response only at > 4 t/ha/yr	30	Searle and Bitnere 2017
Sweden	(+) or (-) with retention	16 long-term trials (average 36 yrs) with retained *cf* removed stubble. Soils with clay >30% showed consistent response with retention	30	Poeplau *et al*. 2015b
Tillage practice and residue management				
Belgium	(=)	Reduced till *cf* conventional till with and without residue retention over 8 years	30	Hiel *et al*. 2018
USA	(=)	No-till and conventional till *cf* residue retention and 2 levels of removal (50 and 100%)	60	Guzman 2103
Kenya	(=)	Reduced till *cf* conventional till with and without residue retention over 11 years	30	Paul *et al*. 2013
China	(=)	Meta-analysis no-till *cf* conventional till from 57 sites. SOC with NT > CT in 0-20 cm layer; SOC with NT < CT below 20 cm. Overall NT > CT calculated using fixed depth, but NT=CT on equivalent soil mass	20+	Du *et al*. 2017
Australia	(=)	Reduced till *cf* conventional till with/without residue retention over 25 years No difference in SOC to 1 m depth	100	Kirkby *et al.* (unpublished)
Tillage practice				
Global	(+) upper layers (-) lower layers	Conventional till *cf* to no till in 69 experiments. NT increased SOC by 3.1 t/ha in 0 to10 cm layer, but decreased SOC by 3.3 t/ha below 20 cm	40	Lou *et al*. 2010
USA	(=)	No-till *cf* chisel till at 3 sites. NT at one site had higher SOC but only in 0-15 cm. No difference in whole profile SOC	90	Chatterjee2018
USA (mainly)	(+) upper layers (-) lower layers	Meta-analysis (24 long-term studies) with no-till *cf* conventional till. Generally SOC with NT > CT in upper layers (0-10 cm); SOC with NT<CT in layers >15 cm	dns	Angers and Eriksen-Hamel 2008
USA	(+)/(=)	Conventional till *cf* no-till on sites >12 years. Higher SOC under no-till 0 to 20 cm, no difference to 100 cm	100	Syswerda *et al.* 2011
USA	(+)	No-till *cf* with 5 tillage systems over 20 years. SOC in NT greater in 0 to 60 cm, no difference below 60 cm	150	Varvel and Wilhelm 2011
Fertilisation: crop and/or residue				
USA	(+) (=)	Corn rotations with varying N rates. Increased SOC in 0 to 7.6 cm; no difference over depth to 30.5 cm	30.5	Liebig *et al*. 2002
China	(+)	SOC assessed under different fertiliser regimes. Increase in SOC for NPK ~2 times cf with N alone	dns	Tian *et al*. 2015
China	(=)	Long-term fertilisation effects (23 years) on SOC. No difference between balanced inorganic fertilisers (NPK) and unfertilised control	100	Song *et al*. 2015
Australia	(+)	SOC increased by 5.5 t/ha over 5yr when supplementary nutrients were added to stubble prior to incorporation, SOC decreased by 3.2 t/ha with no nutrients added	160	Kirkby *et al*. 2016
India	(+)	After 9 years under five fertiliser treatments (N, NPK, FYM, FYM+N, FYM+NPK) SOC stocks increased by 2.6, 5.7, 4.1, 6.9 and 9.8 Mg/ha respectively cf unfertilised control	45	Bhattacharyya *et al*. 2009
USA	(+)	Wheat straw-C conversion to SOC averaged 32% when supplementary N was added to the straw retained in the field but only 14% when no supplementary N was added	10	Jacinthe *et al*. 2002

and form of nutrients applied to stubble and interaction with tillage practice (*e.g.* time of cultivation, tillage system, use of liquid nutrients applied directly to stubble) is needed.

Based on our own experience it is evident that greater opportunity exists with balanced nutrient inputs (N, P, K, S) with high stubble loads (~10 Mg/ha) and in soils with initially low SOC levels (<2%) and high clay content. Moreover, high humification efficiencies appear to require thorough mixing of substrates with soil (be it stubble, manures or other organic amendments) with adequate moisture and temperature to ensure effective microbial processing in order to maximise the contribution to stabilised SOM. On the contrary, surface-retained stubble systems characteristically have slower rates of decomposition that are often constrained by nutrient stratification and sub-optimal conditions for microbial function (Kirkegaard *et al.* 2014) This is consistent with higher predominance of POM in surface soil and a lesser accumulation of stabilised SOM. As previously discussed, POM does not necessarily lead to long term changes in soil physicochemical properties or other benefits associated with SOM sequestration, as the POM is relatively unstable and is more readily lost from the soil (Wander and Bidart 2000, Baker *et al*. 2007).

Practical implications for C sequestration in farming systems

The adoption of any practice to facilitate C sequestration in cropping soils needs to be evaluated against a wide range of economic, environmental and practical criteria. Most important is that any rationale to 'build' soil SOC must be considered against other factors that drive the farm enterprise and thus needs to be effectively integrated within the 'whole of farm' system. In addition, the 'opportunity cost' of better utilising stubble to build SOM must be evaluated against other ecosystem services that surface-retained residues currently provide; for example, as ground cover for erosion control, water retention, weed suppression and low quality animal feeds (see Chapter 2).

Recent economic analysis of cereal crop systems in Western Australia, on sandy soils in rainfall and N limited environments, suggests a value of ~AU$8 Mg C/ha/year, with the benefit being derived predominantly (75%) from the sequestration value, as compared with 20% and 5%, respectively, for N replacement and productivity improvement (Petersen and Hoyle 2016). Even with extrapolation of this over multiple years, this benefit is somewhat modest relative to the cost of generating additional stored soil C. For instance, the inorganic nutrient requirement to generate SOM from C-rich wheat stubble based on stoichiometric composition has been reported as 73, 17 and 11 kg/ha of N, P and S, respectively (Richardson *et al*. 2014), per tonne of soil C (*i.e.* typical difference in units of nutrient per 1000 units of C for stubble compared with SOM). This represents a significant and real cost in terms of fertiliser equivalence (*i.e.* ~$150/ha based on current price) for nutrients that, although not lost from the soil system, would not immediately be available for plant growth as compared with fertiliser strategies that directly target crop growth only.

The longer-term availability of nutrients when 'stored' in SOM, and whether or not they provide a more synchronised supply to meet plant demands throughout the crop cycle (*i.e*. through more sustained rates of mineralisation as compared with a pulse of nutrient supply that is a common feature in current fertiliser practice) is an important consideration. Likewise, having higher retention of nutrients in crop soils may also increase risks associated with either greater leaching or other potential loss processes such as microbial-driven denitrification (Xia *et al.* 2018). This would in course offset any potential benefits. Nonetheless, it presents a 'paradigm shift' in thinking from 'fertilising the crop' to 'fertilising the system' (Richardson *et al*. 2014) and needs to be considered with regard to longer term trajectories for soil fertility. Indeed consideration of a more systems-based approach with greater emphasis on nutrient management and improved nutrient balance replacement has been proposed as an additional pillar to the basic principles of CA (see Chapter 14) that is currently structured around reduced tillage, permanent soil cover and diverse rotations (Giller *et al*. 2015).

Effective management of crop residues and tillage practice along with diverse rotations, organic amendments and fertiliser inputs remain key levers to manage SOM and its contribution to sustainable production. The objective to increase the sequestration of SOC is likely to require innovations around these practices. For example, strategic tillage (see Chapter 7) is proposed to address some of the emerging issues in NT systems including stratification of SOM and inorganic nutrients, herbicide resistant weeds and soil compaction, and may thus have an important role also in facilitating SOM dynamics and opportunities to sequester C in soils. This is particularly relevant to the management of deep soil C that has been shown to be a major contributor to whole soil C stocks (Rumpel and Köger-Knabner 2011), yet relative to surface soils, has received less attention especially in its role in contributing to stabilised SOM.

Further consideration of the long-term implications of 'not restoring' SOM also needs to be assessed from a range of perspectives including environmental, societal and economic. Key questions that remain to be addressed include:

- whether current levels of SOM or rates of depletion in crop-based systems are sustainable and what are the consequences if further depletion were to occur;
- whether it is practical or economically feasible (or even desirable) to return SOM to levels that existed in soil pre-agriculture, given that land-use change will inevitably have some impact on soils and ecosystem function; and, from that,
- what is the optimal level of SOM in agricultural soils for provision of benefits from SOM and long-term sustainability. Associated with this is understanding of SOM levels, where soils become 'dysfunctional' in terms of ecosystem services provided by SOM. These will depend on a range of geo-climatic factors and the production system being considered.

Lastly, the desire or need to increase C in agricultural soils through better utilisation of C-rich residues as a means to address climate change is an issue that lies beyond the responsibility of growers alone and requires commitment to act from wider society. Willingness to act may thus inevitably require implementation of policy directed at mitigation incentives, payment schemes or participation through industry-led C-trading programs.

References

Amudson R, Biadeau L (2018) Soil carbon sequestration is an elusive climate mitigation tool. *Proceedings National Academy Sciences* **11,** 11652-11656

Aguilera E, Lassaletta L, Gattingerd A, Gimenoe BS (2013) Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: a meta-analysis. *Agriculture, Ecosystems & Environment* **168**, 25-36

Alcantara V, Don A, Well R, Nieder R (2016) Deep ploughing increases agricultural soil organic matter stocks. *Global Change Biology* **22**, 2939-2956

Angers DA, Eriksen-Hamel NS (2008) Full-inversion tillage and organic carbon distribution in soil profiles: a meta-analysis. *Science Society of America Journal* **72**, 1370-1374

Angus JF, Grace PR (2017) Nitrogen balance in Australia and N-use efficiency on Australian farms. *Soil Research* **55**, 435-450

Badgery WB, Simmons AT, Murphy BW *et al*. (2014) The influence of land use and management on soil carbon levels for crop-pasture systems in central New South Wales, Australia *Agriculture, Ecosystems & Environment* **196**, 147-157

Baker JM, Ochsner TE, Ventera RT, Griffis TJ (2007) Tillage and soil carbon sequestration – what do we really know? *Agriculture, Ecosystems and Environment* **118,** 1-5

Baldock JA, Hawke B, Sanderman J, Macdonald LM (2013) Predicting contents of carbon and its component fractions in Australian soils from diffuse reflectance mid-infrared spectra. *Soil Research* **51,** 577-595

Bhattacharyya R, Prakash V, Kundu S *et al.* (2009) Effect of fertilisation on carbon sequestration in soybean-wheat rotation under two contrasting soils and management practices in the Indian Himalayas. *Australian Journal of Soil Research* **47**, 592-601

Börjesson G, Bolinder MA, Kirchmann H, Katterer T (2018) Organic carbon stocks in topsoil and subsoil in long-term ley and cereal monoculture rotations *Biology and Fertility of Soil* **54,** 549-558

Buyanovsky GA, Wagner GH (1998) Carbon cycling in cultivated land and its global significance. *Global Change Biology* **4,** 131-141

Chan K, McCoy D (2010) Soil carbon storage potential under perennial pastures in the mid-north coast of New South Wales, Australia. *Tropical Grasslands* **44**, 184-191

Chan KY, Conyers MK, Li GD *et al.* (2011) Soil carbon dynamics under different cropping and pasture management in temperate Australia: Results of three long-term experiments. *Soil Research* **49,** 320-328

Chatterjee A (2018) On-farm comparisons of soil organic carbon under no-tillage and chisel-plow systems. *Soil and Plant Science* **68,** 471-476

Chen R, Senbayram M, Blagodatsky S *et al*. (2014) Soil C and N availability determine the priming effect: microbial N mining and stoichiometric decomposition theories. *Global Change Biology* **20,** 2356-2367

Chevallier T, Voltz M, Blanchart E *et al*. (2000) Spatial and temporal changes of soil C after establishment of a pasture on a long-term cultivated vertisol. *Geoderma* **94,** 43-58

Christensen BT, Rasmussen J, Eriksen J, Hansen EM (2009) Soil carbon storage and yields of spring barley following grass leys of different age. *European Journal of Agronomy* **31,** 29-35

Clement CR, Williams TE (1964) Leys and soil organic matter. I. Accumulation of organic carbon under different leys. *Journal of Agricultural Science* **63,** 377-383

Cleveland CC, Liptzin D (2007) C:N:P stoichiometry in soil: is there a “Redfield ratio” for the microbial biomass? *Biogeochemistry* **85,** 235-252

Conant RT, Cerri CEP, Osborne BB, Paustian K (2017) Grassland management impacts on soil carbon stocks: a new synthesis. *Ecological Applications* **27,** 662-668

Conyers M, Liu DL, Kirkegaard J *et al*. (2015) A review of organic carbon accumulation in soils within the agricultural context of southern New South Wales, Australia. *Field Crops Research* **184,** 177-182

Coonan EC, Richardson AE, Kirkby CA *et al*. (2019) Soil carbon sequestration to depth in response to long-term phosphorus fertilization of grazed pasture *Geoderma* **338,** 226-235

Cornish PS, Pratley JE (Eds.) (1987) “Tillage: New Directions in Australian Agriculture”. (Inkata Press:Melbourne)

Dalal R, Strong W, Weston E *et al*. (1995) Sustaining productivity of a Vertisol at Warra, Queensland, with fertilizers, no-tillage, or legumes. 1. Organic matter status. *Australian Journal of Experimental Agriculture* **35,** 903-913

Dijkstra P, Ishizu A, Doucett R *et al.* (2006) ^{13}C and ^{15}N natural abundance of the soil microbial biomass. *Soil Biology & Biochemistry* **38,** 3257-3266

Don A, Schumacher J, Freibauer A (2011) Impact of tropical land-use change on soil organic carbon stocks; a meta-analysis. *Global Change Biology* **17,** 1658-1670

Drinkwater LE, Wagoner P, Sarrantonio M (1998) Legume-based cropping systems have reduced carbon and nitrogen losses. *Nature* **396,** 262-265

Du Z, Angers DA, Ren T *et al*. (2017) The effect of no-till on organic C storage in Chinese soils should not be overemphasized: A meta-analysis. *Agriculture, Ecosystems & Environment* **236,** 1-11

Eldor P (2016) The nature and dynamics of soil organic matter: Plant inputs, microbial transformations, and organic matter stabilization. *Soil Biology & Biochemistry* **98,** 109-126

Engelking B, Flessa H, Joergensen RG (2008) Formation and use of microbial residues after adding sugarcane sucrose to a heated soil devoid of soil organic matter. *Soil Biology & Biochemistry* **40,** 97-105

Fontaine S, Bardoux G, Abbadie L, Mariotti A (2004) Carbon input to soil may decrease soil carbon content *Ecology Letters* **7,** 314-320

Giller KE, Andersson J, Corbeels M *et al*. (2015) Beyond conservation agriculture. *Frontiers in Plant Science* **6,** 870

Gibson TS, Chan KY, Sharma G, Shearman R (2002) Soil carbon sequestration utilising recycled organics. A review of the scientific literature. *The Organic Waste Recycling Unit, NSW Agriculture*, 95 pp

Glaser B, Turrión Ma-B, Alef K (2004) Amino sugars and muramic acid – biomarkers for soil microbial community structure analysis. *Soil Biology & Biochemistry* **36,** 399-407

Grandy S, Robertson G (2006) Aggregation and organic matter protection following tillage of a previously uncultivated soil. *Soil Science Society of America Journal* **70,** 1398-1406

Guo L, Gifford R (2002) Soil carbon stocks and land use change: a meta-analysis. *Global Change Biology* **8,** 345-360

Gupta VVSR, Germida JJ (2015) Soil aggregation: Influence on microbial biomass and implications for biological processes. *Soil Biology & Biochemistry* **80,** A3-A9

Guzman JG, Al-Kaisi MM (2014) Residue removal and management practices effects on soil environment and carbon budget. *Soil Science Society of America Journal* **78,** 609-623

Hagerty SB, Van Groenigen KJ, Allison SD *et al*. (2014) Accelerated microbial turnover but constant growth efficiency with warming in soil. *Nature Climate Change* **4,** 903-906

Haynes RJ, Williams PH (1992) Accumulation of soil organic matter and the forms, mineralization potential and plant-availability of accumulated organic sulphur: Effects of pasture improvement and intensive cultivation. *Soil Biology & Biochemistry* **24,** 209-217

Haynes RJ (2005) Labile organic matter fractions as central components of the quality of agricultural soils: an overview. *Advances in Agronomy* **85,** 221-268

Heenan DP, McGhie WJ, Thompson F, Chan KY (1996) Decline in soil organic carbon and total nitrogen in relation to tillage, stubble management and rotation. *Australian Journal of Experimental Agriculture* **35,** 877-884

Helgason BL, Gregorich EG, Janzen HH *et al.* (2014) Long-term microbial retention of residue C is site-specific and depends on residue placement *Soil Biology & Biochemistry* **68,** 231-240

Helyar KR, Cullis BR, Furniss K *et al*. (1997) Changes in the acidity and fertility of a red earth soil under wheat-annual pasture rotations. *Australian Journal of Agricultural Research* **48,** 561-586

Hiel M, Barbieux S, Pierreux J *et al*. (2018) Impact of crop residue management on crop production and soil chemistry after seven years of crop rotation in temperate climate, loamy soils. *Peer J* **6,** e4836

Hoyle FC, Baldock JA, Murphy DV (2011) Soil organic carbon – role in rainfed farming systems with particular reference to Australian conditions. In (Eds. P Tow, I Cooper, I Partridge) "Rainfed Farming Systems" pp 339-361 (Springer International)

Jacinthe PA, Lal R, Kimble JM (2002) Effects of wheat residue fertilization on accumulation and biochemical attributes of organic carbon in a Central Ohio Luvisol. Soil Science **167**, 750-758

Janzen HH, Campbell CA, Brandt SA *et al*. (1992) Light-fraction organic matter in soils from long-term crop rotations. *Soil Science Society of America Journal* **56,** 1799-1806

Jastrow J, Amonette J, Bailey V (2007) Mechanisms controlling soil carbon turnover and their potential application for enhancing carbon sequestration. *Climatic Change* **80,** 5-23

Johnston AE, McEwen J, Lane PW *et al.* (1994) Effect of one to 6 year-old ryegrass clover leys on soil nitrogen and on the subsequent yields and fertilizer nitrogen requirements of the arable sequence winter-wheat, potatoes, winter-wheat, winter beans grown on a sandy loam soil. *Journal of Agricultural Science* **122,** 73-89

Kirkby CA, Richardson AE, Wade LJ *et al*. (2014) Nutrient availability limits carbon sequestration in arable soils. *Soil Biology & Biochemistry* **68,** 402-409

Kirkby CA, Kirkegaard JA, Richardson AE *et al.* (2011) Stable soil organic matter: a comparison of C:N:P:S ratios in Australian and other world soils. *Geoderma* **163,** 197-208

Kirkby CA, Richardson AE, Wade LJ *et al*. (2016) Inorganic nutrients increase humification efficiency and C-sequestration in an annually cropped soil. *PLoS ONE* **11,** e0153698

Kirkby CA, Richardson AE, Wade LJ *et al.* (2013) Carbon-nutrient stoichiometry to increase soil carbon sequestration. *Soil Biology and Biochemistry* **60,** 77-86

Kirkegaard JA, Conyers MK, Hunt JR *et al*. (2014) Sense and nonsense in conservation agriculture: principles, pragmatism and productivity in Australian mixed farming systems. *Agriculture, Ecosystems & Environment* **187,** 133-145

Kuzyakov Y (2010) Priming effects: Interactions between living and dead organic matter. *Soil Biology & Biochemistry* **42,** 1363-1371

Lal R (1997) Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO_2-enrichment. *Soil and Tillage Research* **43,** 81-107

Lal R (2004) Soil carbon sequestration impacts on global climate change and food security. *Science* **304,** 1623-1627

Lal R (2007) Carbon management in agricultural soils. *Mitigation and Adaptation Strategies for Global Change* **12,** 303-322

Lehmann J, Kleber M (2015) The contentious nature of soil organic matter. *Nature* **528,** 60-68

Lehtinen T, Schlatter N, Baumgarten A *et al*. (2014) Effect of crop residue incorporation on soil organic carbon and greenhouse gas emissions in European agricultural soils. *Soil Use and Management* **30,** 524-538

Liebig M A, Varvel GE, Doran JW, Wienhold BJ (2002) Crop sequence and nitrogen fertilization effects on soil properties in the western corn-belt. *Soil Science Society of America Journal* **66,** 596-601

Luo ZK, Wang EL, Sun OJ (2010). Soil carbon change and its responses to agricultural practices in Australian agro-ecosystems: a review and synthesis. *Geoderma* **155,** 211-223

Ma T, Zhu S, Wang Z *et al.* (2018) Divergent accumulation of microbial necromass and plant lignin components in grassland soils. *Nature Communications* **9,** 3480

Magid J, Kjærgaard C (2001) Recovering decomposing plant residues from the particulate soil organic matter fraction: size versus density separation. *Biology and Fertility of Soils* **33,** 252-257

Maia SMF, Ogle SM, Cerri CEP, Cerri CC (2009) Effect of grassland management on soil carbon sequestration in Rondônia and Mato Grosso states, Brazil. *Geoderma* **149,** 84-91

Meurer KHE, Haddaway NR, Bolinder MA, Kätterer, T (2018) Tillage intensity affects total SOC stocks in boreo-temperate regions only in the topsoil: A systematic review using an ESM approach. *Earth Sciences Review* **177,** 613-622

Miltner A, Bombach P, Schmidt-Brücken B, Kästner M (2012) SOM genesis: microbial biomass as a significant source. *Biogeochemistry* **111,** 41-55

Moran KK, Six J, Horwath WR, van Kessel (2005) Role of mineral-nitrogen in residue decomposition and stable soil organic matter formation. *Soil Science Society of America Journal* **69,** 1730-1736

Müller-Stover D, Hauggaard-Nielsen H, Eriksen J *et al*. (2012) Microbial biomass, microbial diversity, soil carbon storage, and stability after incubation of soil from grass-clover pastures of different age. *Biology and fertility of soils* **48,** 371-383

Murphy BW (2015) Impact of soil organic matter on soil properties—a review with emphasis on Australian soils. *Soil Research* **53,** 605-635

Orgill SE, Condon JR, Conyers MK *et al*. (2014) Sensitivity of soil carbon to management and environmental factors within Australian perennial pasture systems. *Geoderma* **214,** 70-79

Orgill SE, Condon JR, Conyers MK *et al*. (2017a) Parent material and climate affect soil organic carbon fractions under pastures in south-eastern Australia. *Soil Research* **55,** 799-808

Orgill SE, Condon JR, Kirkby CA *et al*. (2017b) Soil with high organic carbon concentration continues to sequester carbon with increasing carbon inputs. *Geoderma* **285,** 151-163

Palm C, Blanco-Canqui H, DeClerck F *et al*. (2014) Conservation agriculture and ecosystem services: an overview. *Agriculture Ecosystems and Environment* **187,** 87-105

Paul BK, Vanlauwe B, Ayuke F *et al.* (2013) Medium-term impact of tillage and residue management on soil aggregate stability, soil carbon and crop productivity. *Agriculture Ecosystems and Environment* **164,** 14-22

Petersen EH, Hoyle FC (2016) Estimating the economic value of soil organic carbon for grains cropping systems in Western Australia. *Soil Research* **54***,* 383-396

Poeplau C, Don A (2013) Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe. *Geoderma* **192,** 189-201

Poeplau C, Don A (2015a) Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. *Agriculture, Ecosystems and Environment* **200,** 33-41

Poeplau C, Kätterer T, Bolinder MA *et al*. (2015b) Low stabilization of aboveground crop residue carbon in sandy soils of Swedish long-term experiments. *Geoderma* **237,** 246-255

Poeplau C, Reiter L, Berti A, Kätterer T (2017) Qualitative and quantitative response of soil organic C to 40 years of crop residue incorporation under contrasting N fertilisation regimes. *Soil Research* **55,** 1-9

Poulton P, Johnston J, Macdonald A *et al*. (2018) Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom. *Global Change Biology* **24,** 2563-2584

Powlson DS, Glendining MJ, Coleman K, Whitmore AP (2011) Implications for soil properties of removing cereal straw: results from long-term studies. *Agronomy Journal* **103,** 279-287

Powlson DS, Stirling CM, Jat ML *et al*. (2014) Limited potential of no-till agriculture for climate change mitigation. *Nature Climate Change* **4,** 678-683

Rabbi SMF, Tighe M, Delgado-Baquerizo M *et al*. (2015) Climate and soil properties limit the positive effects of land use reversion on carbon storage in Eastern Australia. *Scientific Reports* **5,** 17866

Richardson AE, Kirkby, CA, Banerjee, S, Kirkegaard, JA (2014) The inorganic nutrient cost of building soil carbon. *Carbon Management* **5,** 265-268

Robertson F, Crawford D, Partington D *et al.* (2016) Soil organic carbon in cropping and pasture systems of Victoria, Australia. *Soil Research* ***54,*** 64-77

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

Sanderman J, Farquharson R, Baldock J (2010) Soil carbon sequestration potential: a review for Australian agriculture. CSIRO, Adelaide, Australia. 81 pp

Sanderman J, Hengl T, Fiske GJ (2017) Soil carbon debt of 12,000 years of human land use. *Proceedings National Academy Sciences* **114,** 9575-9580

Schmidt MWI, Torn MS, Abiven S *et al*. (2011) Persistence of soil organic matter as an ecosystem property. *Nature* **478,** 49-56

Schwenke GD, McLeod MK, Murphy SR *et al*. (2013) The potential for sown tropical perennial grass pastures to improve soil organic carbon in the North-West Slopes and Plains of New South Wales. *Soil Research* **51,** 726-737

Searle S, Bitnere K (2017) Review of the impact of crop residue management on soil organic carbon in Europe. International Council on Clean Transportation, Working Paper 2017-15

Six J, Bossuyt H, Degryze S, Denef K (2004) A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. *Soil and Tillage Research* **79,** 7-31

Six J, Frey S, Thiet R, Batten K (2006) Bacterial and fungal contributions to carbon sequestration in agroecosystems. *Soil Science Society of America Journal* **70,** 555-569

Song ZW, Zhu P, Gao HJ *et al*. (2015) Effects of long-term fertilization on soil organic carbon content and aggregate composition under continuous maize cropping in Northeast China. *Journal of Agricultural Science* **153,** 236-244

Stockmann U, Adams MA, Crawford JW *et al*. (2013) The knowns, known unknowns and unknowns of sequestration of soil organic carbon. *Agriculture, Ecosystems & Environment* **164,** 80-99

Strahm BD, Harrison RB, Terry TA *et al*. (2009) Changes in dissolved organic matter with depth suggest the potential for postharvest organic matter retention to increase subsurface soil carbon pools. *Forest Ecology and Management* **258,** 2347-2352

Syswerda SP, Corbin AT, Mokma DL *et al.* (2011) Agricultural Management and Soil CarbonStorage in Surface vs. Deep Layers. *Soil Science Society of America Journal* **75,** 92-101

Tarré R, Macedo R, Cantarutti RB *et al*. (2001) The effect of the presence of a forage legume on nitrogen and carbon levels in soils under *Brachiaria* pastures in the Atlantic forest region of the south of Bahia, Brazil. *Plant Soil* **234,** 15-26

Tian K, Zhao Y, Xu X *et al*. (2015) Effects of long-term fertilization and residue management on soil organic carbon changes in paddy soils of China: A meta-analysis. *Agriculture Ecosystems & Environment* **204,** 40-50

Tipping E, Somerville CJ, Luster J (2016) The C:N:P:S stoichiometry of soil organic matter. *Biogeochemistry* **130,** 117-131

van Groenigen KJ, Six J, Hungate BA *et al*. (2006) Element interactions limit soil carbon storage. *Proceedings National Academy Sciences* **103,** 6571-6574

van Rees H, Jackman A, Beveridge M, Baldock J (2017) Can soil organic matter be increased in a continuous cropping system in the low to high rainfall zone? *Southern Farming Systems Trial Results* 134-137

van Wesemael B, Paustian K, Meersmans J *et al*. (2010) Agricultural management explains historic changes in regional soil carbon stocks. *Proceedings National Academy Sciences USA* **107,** 14926-14930

Varvel GE, Wilhelm WW (2011) No-tillage increases soil profile carbon and nitrogen under long-term rain-fed cropping systems. *Soil and Tillage Research* **114,** 28-36

von Lützow, M, Kögel-Knabner I, Ekschmitt K *et al*. (2006) Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions: a review. *European Journal of Soil Science* **57,** 426-445

Wander MM, Bidart MG (2000) Tillage practice influences on the physical protection, bioavailability and composition of particulate organic matter. *Biology and Fertility of Soils* **32,** 360-367

Xia L, Lam SK, Wolf B *et al*. (2018) Trade-offs between soil carbon sequestration and reactive nitrogen losses under straw return in global agroecosystems. *Global Change Biology* **24,** 5919-5932

Young RR, Wilson B, Harden S, Bernardi A (2009) Accumulation of soil carbon under zero tillage cropping and perennial vegetation on the Liverpool Plains, eastern Australia. *Australian Journal of Soil Research* **47,** 273-285