Soil Carbon Sequestration Potential:
A review for Australian agriculture

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Cover Photograph

Description: Soil core removed from wheat field near Williamstown, South Australia. Photographer: N Watkins © 2009 CSIRO

Executive Summary

Conversion of native land for agriculture has typically resulted in decreases in soil organic carbon (SOC) stocks on the order of 40 to 60% from pre-clearing levels. Globally, this loss of SOC has resulted in the emission of at least 150 Pentagrams (Pg) of carbon dioxide to the atmosphere (1 Pentagram = 1 Gigatonne = 10^{15} grams). Recapturing even a small fraction of these legacy emissions through improved land management would represent a significant greenhouse gas emissions reduction.

Currently, there is much uncertainty and debate, particularly within Australia, as to the total potential of agricultural soils to store additional carbon, the rate at which soils can accumulate carbon, the permanence of this sink, and how best to monitor changes in SOC stocks. To help clarify some of these issues, this report presents both a review of the mechanisms of carbon capture and storage in agricultural soils and an analysis of the published evidence for SOC stock changes resulting from shifts in agricultural management.

On average, improved management of cropland, whether enhanced rotation, adoption of no-till or stubble retention, has resulted in a relative gain of 0.2 – 0.3 Mg C ha^{-1} yr^{-1} compared to conventional management across a range of Australian soils. However, when time series data was available, even the improved management often showed significant absolute declines in SOC stocks, which, in many cases, was likely a direct result of these soils still responding to the initial cultivation of the native soil. Since the traditional management practice often lost SOC at a greater rate, when only comparing the two treatments at the end of the trial there was a relative SOC gain in the improved management treatment. This means that, for the more traditional agronomic systems tested in these trials, Australian soils may only be mitigating losses and not actually sequestering additional atmospheric carbon. Additionally, sequestration rates were found to diminish with increasing trial duration. The largest gains were generally found within the first 5 to 10 years with the rate of change diminishing to nearly 0 after 40 years.

While there was much less field evidence, the data at hand indicates that pasture improvements, including fertilisation, liming, irrigation and sowing of more productive grass varieties, generally have resulted in relative gains of 0.1 – 0.3 Mg C ha^{-1} yr^{-1}. Larger gains of 0.3 – 0.6 Mg C ha^{-1} yr^{-1} have been found for conversion of cultivated land to permanent pasture.

The majority of available field data comes from a fairly narrow range of management options for the dominant agricultural systems of Australia and little data exists on numerous management options which hold potential to sequester large quantities of SOC. Within an existing agricultural system, the greatest theoretical potential for C sequestration will likely come from large additions of organic materials (manure, green wastes, etc…), maximizing pasture phases in mixed cropping systems and shifting from annual to perennial species in permanent pastures. Perhaps the greatest gains can be expected from more radical management shifts such as conversion from cropping to permanent pasture and retirement and restoration of degraded land. These options are summarized in the accompanying table.

Many of these management options that may increase SOC tend to also increase overall farm productivity, profitability and sustainability, and as such are being rapidly adopted in various regions of Australia. However, numerous other management shifts (for example, converting from annual crops to pastures) which may have the

greatest positive impact on SOC stocks will likely need incentives, either in the form of direct government subsidies or credits from an emissions trading market, before wide-scale adoption is seen.

There is a strong theoretical basis partially supported by a limited number of field studies for significant SOC sequestration potential in several Australian agricultural sectors. However, a general lack of research in this area is currently preventing a more quantitative assessment of the carbon sequestration potential of agricultural soils.

Summary of major management options for sequestering carbon in agricultural soils

Management	SOC benefit[a]	Conf.[b]	Justification
1. Shifts within an existing cropping/mixed system			
a. Maximizing efficiencies - 1) water-use 2) nutrient-use	0/+	L	Yield and efficiency increases do not necessarily translate to increased C return to soil
b. Increased productivity - 1) irrigation 2) fertilization	0/+	L	Potential trade-off between increased C return to soil and increased decomposition rates
c. Stubble management – 1) Eliminate burning/grazing	+	M	Greater C return to the soil should increase SOC stocks
d. Tillage – 1) Reduced tillage 2) Direct drilling	0 0/+	M M	1) Reduced till has shown little SOC benefit; 2) Direct drill reduces erosion and destruction of soil structure thus slowing decomposition rates; however, surface residues decompose with only minor contribution to SOC pool
e. Rotation – 1) Eliminate fallow with cover crop 2) Inc. proportion of pasture to crops 3) Pasture cropping	+ +/++ ++	M H M	1) Losses continue during fallow without any new C inputs – cover crops mitigate this; 2) Pastures generally return more C to soil than crops; 3) Pasture cropping increases C return with the benefits of perennial grasses (listed below) but studies lacking
f. Organic matter and other offsite additions	++/+++	H	Direct input of C, often in a more stable form, into the soil; additional stimulation of plant productivity (see above)
2. Shifts within an existing pastoral system			
a. Increased productivity - 1) irrigation 2) fertilization	0/+	L	Potential trade-off between increased C return to soil and increased decomposition rates
b. Rotational grazing	+	L	Increased productivity, inc. root turnover and incorporation of residues by trampling but lacking field evidence
c. Shift to perennial species	++	M	Plants can utilize water throughout year, increased belowground allocation but few studies to date
3. Shift to different system			
a. Conventional to organic farming system	0/+/++	L	Likely highly variable depending on the specifics of the organic system (i.e. manuring, cover crops, etc…)
b. Cropping to pasture system	+/++	M	Generally greater C return to soil in pasture systems; will likely depend greatly upon the specifics of the switch
c. Retirement of land and restoration of degraded land	++ +++	H	Annual production, minus natural loss, is now returned to soil; active management to replant native species often results in large C gains

[a] Qualitative assessment of the SOC sequestration potential of a given management practice (0 = nil, + = low, ++ = moderate, +++ = high)
[b] Qualitative assessment of the confidence in this estimate of sequestration potential based on both theoretical and evidentiary lines (L = low, M = medium, H = high)

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1. Introduction

Policy makers in Australia, and many other nations, are currently debating how to design, implement and monitor carbon pollution reduction schemes (CPRS) as an important tool to reduce greenhouse gas emissions. Biospheric carbon offsets including soil carbon sequestration have the potential to be important components of any CPRS but numerous uncertainties still exist, especially within the agricultural sector, which are major barriers to effective policy implementation.

Soils, and managed agricultural soils in particular, represent a potentially significant low-to-no cost sink for greenhouse gases (GHGs) with multiple potential co-benefits to farm productivity and profitability (Lal 2004a; Pacala and Socolow 2004). The great majority of agronomists and soil scientists agree that most agricultural soils can store more carbon and even a modest increase in carbon stocks across the large land areas used for agriculture would represent a significant GHG mitigation. However, currently, there is much uncertainty and debate, particularly within Australia, as to the total potential of soils to store additional carbon, the rate at which soils can store carbon, the permanence of this carbon sink, and how best to monitor changes in soil carbon stocks.

Throughout this paper, we have primarily discussed the technical potential, defined by the biophysical conditions of the system, for agricultural land to store additional soil organic carbon (SOC) through improvements in management. It is very important to realize that this technical sequestration potential will likely never be fully realized due to a whole host of economic, social and political constraints (Section 5.5).

In this review, a "first-principles" approach is first used to address soil carbon sequestration potential by way of a review of the mechanisms of carbon capture and stabilisation in agricultural soils (Section 3). This is followed by a discussion of some of the difficulties in accurately measuring change in SOC stocks (Section 4). Finally, we present both a theoretical basis for potential sequestration due to agricultural management improvements and a summary of field evidence for stocks changes both in Australia and abroad (Section 5).

2. Background

2.1. Soil carbon

Soils contain large amounts of carbon in both organic and inorganic forms. Organic C is found in soils in the form of various organic compounds, collectively called soil organic matter (SOM). The amount of C found in SOM ranges from 40 to 60% by mass. Strictly speaking, SOM includes all living and non-living organic material in the soil (Baldock and Skjemstad 1999). The living component includes plants, soil fauna and microbial biomass. The non-living component, representing the bulk of SOM, includes a spectrum of material from fresh residues and simple monomeric compounds to highly condensed, irregular polymeric structures with residence times varying from days to millennia.

Globally, the top meter of soil stores approximately 1500 Pg as organic C and an additional 900-1700 Pg as inorganic C and exchanges 60 Pg C yr^{-1} with the atmosphere, which contains ~750 Pg C as carbon dioxide (CO_2) (Eswaran *et al.* 1993; Schlesinger 1997). The shear size of the soil carbon pool and the annual flux of carbon passing through the soil are two of the reasons that SOC can play a significant role in mitigating GHG emissions. Currently, there is not enough research on the topic of anthropogenic impacts on the formation and loss of soil inorganic C to be able to assess its sequestration potential and it is only the organic C pool that is under consideration for inclusion in various emission reduction schemes.

Historically, approximately 78 Pg C has been lost from the global soil pool due to land-use conversion for agriculture with approximately 26 Pg attributed to erosion and 52 Pg attributed to mineralization (Lal 2004b). Conversion of native forest and pasture to cropland has been found to reduce SOC stocks by an average of 42% and 59%, respectively (Guo and Gifford 2002). These large historic losses and the concomitant potential to return to pre-clearing SOC conditions are precisely the reason many researchers believe there is great potential for agricultural soils to sequester large amounts of atmospheric CO_2 relative to current SOC levels.

2.2. The Australian situation

Total soil organic carbon stocks in 1990 for the Australia continent were estimated at 19 Pg for the top 30 cm by Grace *et al.* (2006) with a natural flux of ~700 Tg CO_2 (cf Barrett 2002) exchanged between the soil and atmosphere every year. This natural exchange is ~12% larger than Australia's 2007 anthropogenic GHG emissions excluding land use and land-use change (available online at: http://www.climatechange.gov.au/inventory/index.html). At face value, a mere 0.8 % per annum increase in SOC stocks would effectively mitigate Australia's annual GHG emissions.

The agricultural sector covers approximately 60% of the land area of Australia, with >90% of that area (419 Mha) being used for low-density grazing of natural vegetation (Table 1). This leaves ~50 Mha of land that is actively managed, primarily for grazing of modified pastures and various cropping systems.

Table 1. National land use 2001/02 Summary (Bureau of Rural Sciences)

Land use type	Area (10^3 ha)	% of total
Grazing of natural vegetation	419,472	54.56
Grazing modified pastures	24,107	3.14
Dryland cereal cropping	19,214	2.50
Dryland legume cropping	2,348	0.31
Dryland oil seed cropping	1,522	0.20
Irrigated cereal cropping	636	0.08
Horticulture (mostly irrigated)	571	0.07
Sugarcane (~50:50 dry:irrigated)	559	0.07
Cotton (primarily irrigated)	482	0.06
Other crops (dry+irrigated)	180	0.02
Agriculture total	469,092	61.01
Managed ag total	49,620	6.45
Potentially tilled acreage	24,941	3.24
Total land area	768,850	

Actively managed lands can be divided into 5 broad agroecological zones:

1) **Summer Rainfall** (subtropical regions of New South Wales and Queensland). In this region, clay-rich vertisols are the dominant soil type for cropping, representing ~2.5 Mha. These soils have moderately high native SOC stocks (~70-100 Mg C ha^{-1}), but tend to lose that C rapidly upon cultivation. Summarizing much of the long-term field trial data in this region, Grace *et al.* (1998) concluded that changes in soil C inputs are the main driving factor for either depletion or sequestration of soil C.
2) **Mediterranean West** (south-western region of Western Australia). In this region, light textured soils with low %C, cation exchange capacity, nutrient retention, and water holding capacity dominate. The combination of these soils with the Mediterranean climate (hot, dry summers and cool, moderately wet winters) results in rapid and large losses of SOC upon cultivation (Dalal and Mayer 1986). Annual legume-based pastures mixed into wheat rotations with significant adoption of reduced and zero tillage cultivation have been the most effective management against SOC declines (Grace *et al.* 1998).
3) **Moist South East** (main wheat belt of Australia with rainfall >350 mm yr^{-1}). Soils are dominated by red earths and red-brown earths with generally good physical characteristics for OC storage, but nearly all soil types are found. Residue removal is a driver of SOC loss with fertilizer being effective for production but not cost-effective in mitigating SOC decline (Grace *et al.* 1998). In cash crops, summer fallows are very common and pasture rotations are typically well developed.
4) **Dry Marginal South East** (Mallee of Victoria, South Australia and New South Wales with rainfall 250-350 mm yr^{-1}). Grain yields in this region are typically low (<1.5 t ha^{-1}). Residue losses through stubble grazing combined with wind erosion in these naturally C-poor soils are primary drivers of C loss (Grace *et al.* 1998). Adoption of reduced and zero tillage systems to reduce erosion and protect SOC stocks has been significant in this region.
5) **High Rainfall Zone** (coastal NSW, VIC, SA and WA and most of Tasmania). In this region, a wide range of soils support a wide variety of often very productive cropping and pastoral systems with the dominant commonality being higher rainfall totals (typically >600 mm yr^{-1}).

Australian soils, especially in comparison to many northern hemisphere soils, are much older meaning many weathering-derived nutrients have been exhausted, salts have accumulated, and unique pedogenic features that impose substantial limitations to agriculture, such as hardpans, have had time to form. Nutrients, especially phosphorus and other micronutrients such as copper, zinc and molybdenum, are often deficient in Australian soils (Chen *et al.* 2009). The wheat cropping region is dominated by light-textured soils that often have sodicity and salinity problems leading to hardsetting and associated infiltration, crop emergence and erosion issues (Chan *et al.* 2003). These edaphic constraints, especially when combined with a generally less favourable climate, may act to reduce SOC sequestration potential in Australian soils relative to the northern hemisphere situation.

2.3. Soil organic matter in agriculture

In addition to the additional greenhouse gas burden on the atmosphere, the depletion of soil organic matter has numerous adverse ecological and economic consequences. Loss of SOM is accompanied by depletion of plant nutrients including potentially mineralisable N, P and S; increased soil bulk density; loss of aggregate structure; decreased water-holding capacity and hydraulic conductivity; decreased cation-exchange capacity; increased surface erosion; increased leaching of pesticides and heavy metals; a decline in soil biological activity and diversity; and ultimately declines in crop yields and quality (Amezketa 1999; Lal 2004a; Verrell and Obrien 1996; Whitbread *et al.* 1998). Referring to this multifaceted role of organic matter in soils of the United States, Albrecht (1938) observed that "soil organic matter is one of our most important national resources; its unwise exploitation has been devastating; and it must be given its proper rank in any conservation policy."

In contrast, improving SOM levels will help reverse the deleterious consequences outlined in the preceding paragraph, including improvements to soil structure, soil fertility (see Janzen (2006) for excellent discussion on this topic), nutrient retention, water holding capacity, and reduced soil erosion. Many studies have shown a strong correlation between increased SOM levels and improvements in soil physical properties such as aggregation, water infiltration, hydraulic conductivity and compaction (e.g. Blair *et al.* 2006a; Blair *et al.* 2006b; Whitbread *et al.* 2000b). These improvements generally translate to greater productivity and crop yields with reduced inputs of fertilizers, pesticides and water (Lal 2004b).

3. Carbon Cycling in Agroecosystems

Before discussing specific management impacts on soil organic carbon stocks (SOC), we feel it is critical to review the current scientific understanding of the factors that govern the accumulation and loss of SOC. In this section, we follow the flow of carbon (Figure 1) as atmospheric CO_2 is fixed during photosynthesis, transferred to the soil as organic carbon, and ultimately returned back to the atmosphere as CO_2 produced during mineralisation of SOC.

3.1. Inputs

Soil organic carbon is initially derived from photosynthetically captured carbon, thus plant productivity sets an absolute upper limit to potential carbon inputs to the soil system. In fact, over large regional areas, primary productivity appears to control the carbon balance of soils (Janssens *et al.* 2001). In this section, primary productivity is discussed in terms of the capture of light energy by plants and its conversion to chemical energy by photosynthesis, constraints to carbon capture imposed by light, water and nutrients, and carbon losses resulting from respiration. Then, continental-scale estimates of net primary productivity for Australia based on these principles are compared with more mechanistic plant growth modelling approaches. Finally, we briefly discuss agricultural management strategies to maximize carbon capture.

3.1.1. Upper limit of carbon capture

An upper limit for carbon inputs into a system can be defined by the potential conversion of incoming solar energy into chemical energy and metabolic intermediates by plant photosynthesis. Plants use carbon assimilated by photosynthesis for the synthesis of plant structural components and other metabolic requirements. Hence, net primary productivity (NPP) is defined as gross primary productivity (GPP or carbon assimilation) less autotrophic respiration. Estimation of primary productivity at regional scales requires an estimate of photosynthetically active radiation (PAR), the fraction of PAR absorbed by the vegetation (usually informed by a leaf area index), the light-use efficiency of the vegetation, and the amount of autotrophic respiration. Stress functions can be applied to estimates of potential NPP to take into account limitations imposed by nutrient and water availability that result in actual NPP being less than potential NPP. Depending on the approach being taken, these stress functions might be incorporated into the light-use efficiency and respiration terms, or applied separately.

3.1.2. Constraints to capturing carbon

Photosynthetically active radiation. The amount of incident radiation reaching the surface of the planet depends on earth-sun geometry and attenuation in the atmosphere. On average less than half of the solar radiation reaching the outer atmosphere reaches the earth's surface. The greatest insolation (incoming solar radiation, J m^{-2}) is at the equator and decreases towards the poles. Additionally, downward irradiance will be modified by cloud albedo and is therefore calculated hourly and integrated to give daily insolation totals (for Australia, see http://www.bom.gov.au/sat/solradinfo.shtml). Of the incident solar radiation reaching the canopy, plants can only utilise blue to red visible wavelengths (between approximately 400 and 700 nm) for photosynthesis which on average represents 45% of incoming energy from solar radiation.

Figure 1. Carbon flow in agroecosystems. Limitations on carbon capture given in white boxes. Carbon losses given in grey boxes with squiggly lines representing CO_2 losses. Inputs of C to the soil system can be partitioned between above- and belowground components. A fraction of NPP flows through roots to mycorrhizal fungi (AMF). Some C enters the soil as organic compounds excreted by roots and, if present, AMF fungi. The majority of C enters the soil as particulate organic C (POC). Aboveground C enters the soil as dissolved material in water leaching through residue layers or as POC mixed into the soil by soil fauna or tillage. A portion of the above- and belowground C entering the soil is slowly transformed to humus with the remainder being mineralised back to CO_2.
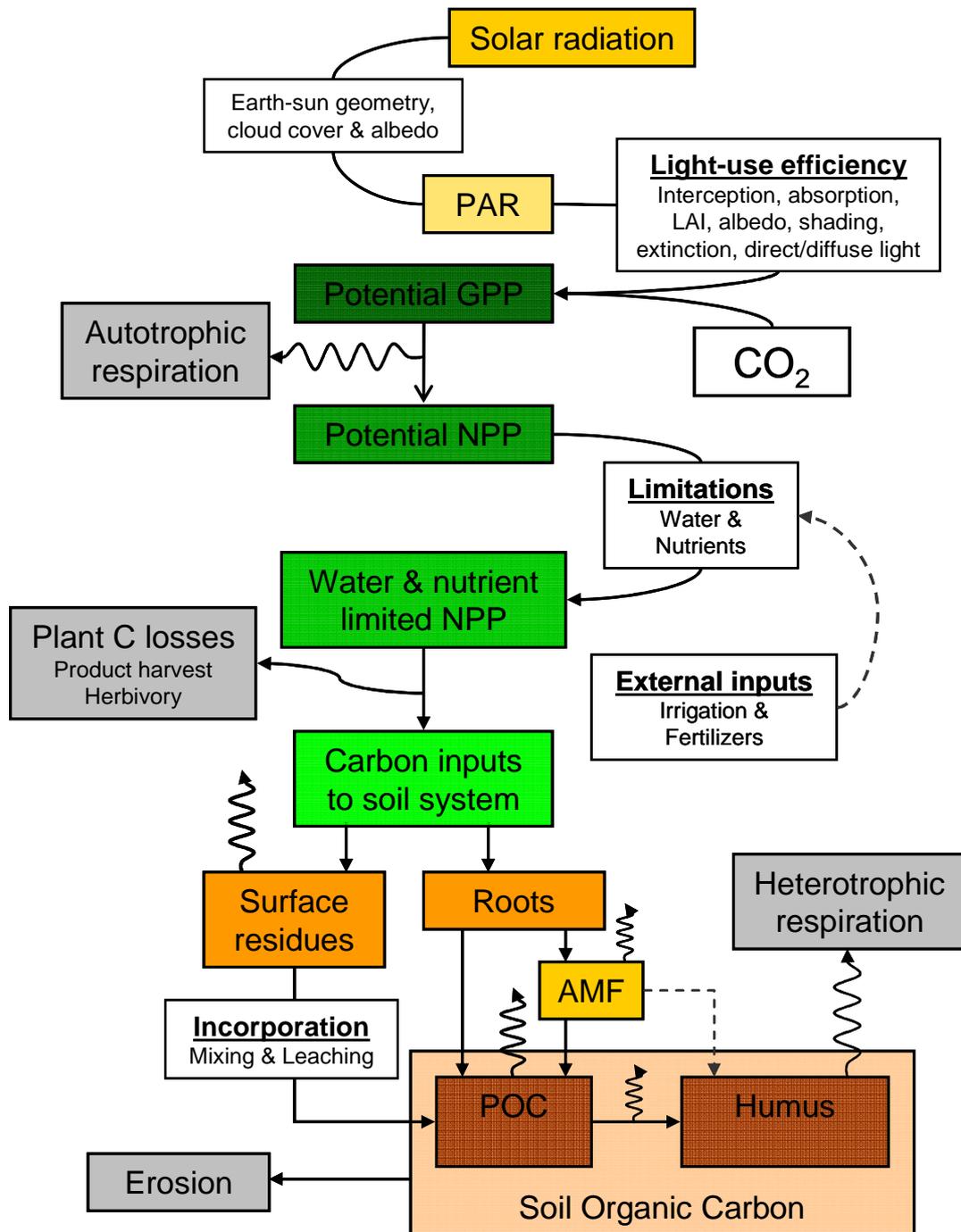
The main difficulty in estimating annual rates of photosynthesis at a regional level is in scaling up short term responses of leaves to canopies, and from sub-diurnal to annual temporal scales (Roxburgh *et al.* 2004). The amount of radiation absorbed by plant canopies depends on the incident radiation, canopy structure and the optical properties of the plants and the soil surface. Due to the spatial and temporal variability in vegetation coverage, the fraction of PAR absorbed by photosynthetic tissues needs to be accounted for. For regional modelling, remotely sensed data can be used to estimate the fraction of PAR that is absorbed (Gower *et al.* 1999).

Light-use efficiency. The ratio of primary productivity to absorbed PAR is termed the light-use efficiency (LUE). The concept of LUE is based on the positive relationship between NPP and PAR, first demonstrated by Monteith (1972). To simplify the estimation of NPP at regional or global scales, vegetative LUE is often assumed to be constant, however the importance of accounting for variation in LUE has been demonstrated, particularly for agroecosystems (Bradford *et al.* 2005; Lobell *et al.* 2002; Yan *et al.* 2009). Light-use efficiency has also been demonstrated to vary depending on cloudiness, phenology and biome specific parameters (Turner *et al.* 2003).

Light-use efficiency will also vary substantially between different photosynthetic pathways (Gower *et al.* 1999). In C3 plants, the RUBISCO enzyme responsible for photosynthesis is also the enzyme for uptake of CO_2. This photosynthetic pathway is more efficient than the C4 and CAM pathways in cool, moist conditions and normal light because fewer enzymes and no specialised anatomy are required. Photosynthesis occurs throughout the leaf in C3 plants and stomata are open only during the day. In hot, dry conditions stomata can close to prevent excess water loss; however, stomatal closure will lead to low internal CO_2 levels thus limiting photorespiration (Hurry *et al.* 2005) and decreasing the apparent LUE. In C4 plants (Furbank 1998) the stomata are also open during the day, but the PEP carboxylase enzyme allows CO_2 to be taken up by the plant quickly and delivered to RUBISCO for photosynthesis which occurs in inner cells and requires special leaf anatomy. C4 plants photosynthesise faster than C3 plants under high light intensity and temperatures and have better water-use efficiency because stomata do not need to remain open as long for the same uptake of CO_2. In CAM photosynthesis (Luttge 1998), stomata open at night and CO_2 is converted to an acid. During the day CO_2 is released to RUBISCO for photosynthesis by breaking down the acid. CAM plants have higher water-use efficiency than C3 (up to 6-fold) and C4 (up to 3-fold) plants in arid conditions (Borland *et al.* 2009) and are able to keep their stomata closed both day and night (e.g. CAM idle) in extreme aridity allowing quick recovery when water becomes available again.

3.1.3. Net Primary Productivity

Only a fraction of the total carbon capture, i.e. gross primary productivity (GPP), results in an accumulation of biomass carbon, since a large proportion (0.25 to 0.70) of assimilated carbon is respired by the plants (Lambers *et al.* 2005). Net primary productivity (NPP) is equal to plant photosynthesis (GPP) less plant respiration. Plant respiration is usually simplified in estimates of NPP, and potential NPP is commonly assumed to be approximately 45% of GPP (Gifford 2003; Landsberg and Waring 1997). This number is based on a convergence of the ratio of photosynthesis to autotrophic respiration and the assumption that GPP and plant respiration respond proportionately to environmental variables.

A diversity of approaches exists for estimating NPP depending on the scale of a particular study. These methods range from 'top down' approaches using remotely sensed data to

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'bottom up' approaches involving parameterisation of physiological plant growth models (Keating *et al.* 2003; Roxburgh *et al.* 2004). By necessity there are assumptions and simplifications in all approaches that contribute to the variation in NPP estimates. As part of the National Land and Water Resources Audit, Raupach *et al.* (2001a; 2001b) developed a model (BiosEquil) to assemble long-term average or steady state balances of water, carbon, nitrogen and phosphorous. As part of this work, a 'semi-mechanistic' expression to calculate NPP at the whole canopy level was developed using a simplified version of leaf scale photosynthetic models and empirical coefficients for the canopy scale. Light and water limited NPP was first calculated using transpiration and incident radiation via a water- and light-use efficiency (from climate data and satellite derived leaf area). Stress functions were then applied for nitrogen and phosphorous limitation to give nutrient limited NPP. Scenarios based on current (with agricultural inputs) and baseline (without agricultural inputs) states were simulated (see Raupach *et al*. 2001b for further details). The BiosEquil model was calibrated and tested against 184 NPP estimates in the VAST dataset (Barrett 2001). While the poor local skill of the model has been acknowledged, for continental scale NPP the BiosEquil model provided a reasonable indication of potential carbon capture by primary productivity across Australia, and hence inputs of carbon into agroecosystems (map #10 in Figure 2).

Roxburgh *et al.* (2004) reviewed 12 model estimates of NPP for the Australian continent representing a variety of approaches (Figure 2). The BiosEquil estimate of 0.94 Pg C yr^{-1} was at the lower end of the continental NPP estimates and could therefore be considered to be conservative. This may be related to the calibration of the model on the VAST dataset

Figure 2. Estimates of spatial distribution of NPP for the Australian continent from 12 different modelling exercises (from Roxburgh et al. 2004). Long-term average NPP (Pg C yr^{-1}) given in parentheses. Reproduced by permission of CSIRO Publishing.
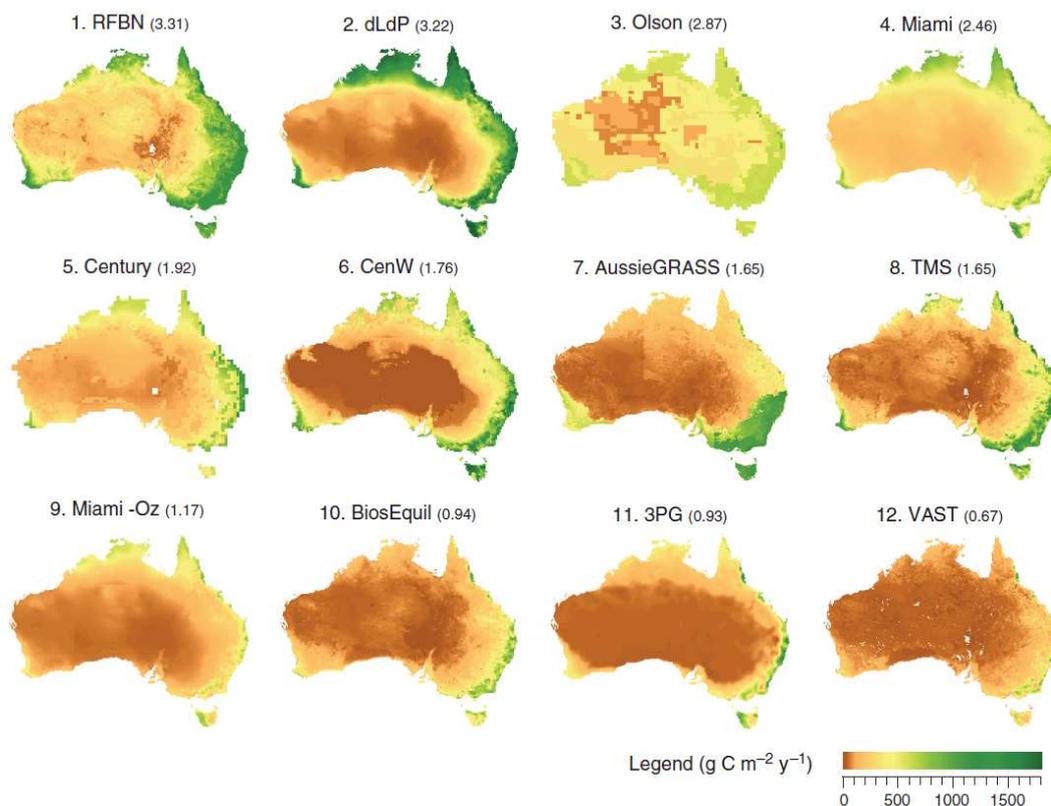
(Barrett 2001) which consisted of NPP observations from minimally disturbed systems assumed to be in steady state. Large variability exists in estimates of total and within continental NPP for Australia and there is a dearth of empirical NPP estimates for model development and validation (Roxburgh *et al.* 2004). Difficulties in obtaining comprehensive measures of NPP (Clark *et al.* 2001; Scurlock *et al.* 2002) contribute to the variability in NPP estimates. However, continental scale NPP estimates still provide a useful indication of the carbon yield across the landscape and hence potential carbon inputs into the soil.

3.1.4. Maximizing NPP

Agronomists have aimed to maximise the productivity of agricultural systems in the interests of profitability. This is usually coupled with the export either of harvestable plant products or livestock production. Whether the drive for profitability has concomitantly increased NPP alongside yield, or simply shifted the partitioning of total production towards harvestable products (by increasing harvest indices and decreasing the allocation of carbon to roots) will have broad implications for the carbon balance of agricultural systems and the ability to increase inputs of carbon to the soil through management.

The most direct way to maximise NPP is to alleviate the limitations to NPP. In most Australian agroecosystems water and nutrients represent the major limitations to NPP. Thus NPP in many agricultural systems is considered to be higher than that of native systems they replaced, in part because of inputs of fertiliser and irrigation (Raupach *et al.* 2001a). NPP can also be maximised by increasing the efficiency by which the available resources are used (see section 5.2.1).

One strategy that is rapidly gaining in popularity throughout many regions of Australia is to introduce perennial species into systems that are currently dominated by annuals to improve the use of available water in the summer months when light intensities, and thus potential photosynthesis rates, are greatest. Using the APSIM model (Keating *et al.* 2003), we found that there is theoretical potential to increase productivity by better utilisation of rain falling outside the typical annual cropping cycle by perennial grass. Modelling scenarios suggested that NPP could be doubled or even tripled under perennial grasses relative to winter cereal/summer fallow systems depending on rainfall distribution (see Appendix 1).

In summary, an understanding of the maximum carbon capture under different agricultural scenarios will set a theoretical upper limit to soil C inputs and thus soil sequestration potential. However, as highlighted earlier, there isn't necessarily a direct 1:1 relationship between gains in productivity and gains in C inputs to the soil, so predicting sequestration rates from productivity increases alone will likely not be possible.

3.2. Fate of captured carbon in managed systems

The fate of recently fixed carbon, whether it stays on the land or is exported as a farm commodity, and its mode of incorporation into the soil are two of the dominant factors controlling soil carbon stocks. On any given area of land, some quantity of carbon will be fixed from the atmosphere that will depend on the environmental and phenotypic factors outlined in the previous section. In agroecosystems, a substantial portion of this fixed C will be removed during harvest, with 30 – 50% of aboveground dry mass typically being removed for most modern cereal crops (Hay 1995; Johnson *et al.* 2006; Unkovich *et al.* 2009). Following harvest, the remainder of the annual fixed C will reside as aboveground residues, belowground residues (i.e. root biomass), or have already been transferred into the soil as root and mycorrhizal exudates (including turnover of short-lived fine roots and fungal hyphae). In perennial systems, such as pastures, there is a constant, although often

seasonally varying, turnover of both aboveground and belowground biomass with much of the aboveground biomass being consumed by grazers.

Traditionally, much of the remaining aboveground residues, i.e. stubble, were burned for fear of nurturing soil-borne diseases and to ease sowing of the next crop. This practice is now falling out of favour with an increased recognition of the benefits of stubble retention and/or incorporation. Retaining stubble will greatly reduce erosion and minimize water losses during fallow periods (Radford *et al.* 1992; Thomas *et al.* 2007b). However, stubble retained on the surface will make only a small contribution to longer-term soil C stocks and there appears to be a trade-off between the benefits of stubble retention and the benefits of incorporating the stubble in building SOM stocks with the balance depending on the mass of stubble produced (Kirkby *et al.* 2006).

For aboveground residues to persist for any significant period of time, this material generally needs to be incorporated into the mineral soil. The two dominant mechanisms of incorporation are physical mixing and solubilisation, transport and subsequent adsorption deeper in the soil profile. In many agroecosystems, the majority of mixing of surface residues into the mineral soil is accomplished during tillage. In natural and less heavily managed systems, there exists a myriad of soil fauna, especially earthworms and litter arthropods, which are very effective in fragmenting and mixing surface residues into the soil (Paoletti 1999; Seastedt 1984). Leaching of organic materials out of fresh surface residues into the soil may be responsible for upwards of 30% of initial mass loss (Sanderman and Amundson 2003). A fraction of this solubilised organic carbon will be rapidly lost to heterotrophic respiration (Wickland *et al.* 2007), with the remainder entering the mineral soil where it is thought to be an important humus precursor (Guggenberger and Zech 1994).

Belowground residues and root turnover represent direct inputs into the soil system, and as such have the potential to make major contributions to SOM stocks. The tight coupling between root distribution and SOC distribution with depth is often cited as evidence for the importance of root inputs in maintaining SOC stocks (Jackson *et al.* 1996; Jobbagy and Jackson 2000). In addition to the spatial location within the mineral soil, roots generally decay slower than aboveground residue (Rasse *et al.* 2005; Silver and Miya 2001) which has been attributed to both litter quality and environmental factors (Crow *et al.* 2009; Kogel-Knabner 2002).

Accurate measurement of the allocation of carbon belowground by plants is difficult (Bledsoe *et al.* 1999; Taylor 1986; Vogt *et al.* 1998) due to both the inaccessibility of roots and the high turnover rate of the finest roots. For these reasons, many carbon cycling studies rely on an allometric relationship, the root-to-shoot ratio, for estimating belowground C inputs. The measurement of the root-to-shoot ratio is typically a static measurement of total live biomass recovered in shoot and root components at a given point in time. While this ratio is helpful in partitioning assimilated C into aboveground and belowground parts and in back-calculating NPP from aboveground biomass measurements, it may produce a misleading picture where belowground C and total NPP are underestimated. A significant fraction of total root mass may be in short lived fine roots with turnover times on the order of days to weeks (Pritchard and Rogers 2000). Additionally, it is now widely recognized that 5 to 20% or more of assimilated C will be transferred to the soil via rhizodeposition and mycorrhizal fungal demand (see following sections). For example, Johnson *et al.* (2006) estimated the root-to-shoot ratio for wheat, maize and soybean, respectively, was 0.50, 0.33 and 0.37 when only considering root biomass, but the ratio rose to 0.82, 0.55 and 0.62 for the same crops when total belowground C allocation was considered.

Allocation of C to roots varies dramatically throughout the development of most cereal grains with root allocation typically highest during early growth and reduces to almost zero at flowering (Crawford *et al.* 2000; Hulugalle *et al.* 2009). For the crops, wheat, maize,

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soybean and lupin, Gregory (1994; 1996) found that root-to-shoot ratios decreased from about 0.4 to 0.6 during vegetative growth phases to as low as 0.1 at flowering. Low root-to-shoot ratios for cash crops are not surprising since these crops have been intensively bred to maximize grain yields.

Longer-lived species, such as perennial grasses, invest significantly more resources into maintaining a healthy root system resulting in substantially higher root-to-shoot ratios. In a typical New Zealand pasture, Stewart and Metherell (1999) found root-to-shoot ratios ranging from 1.1 to 2.3 depending on season and management. For mixed-grass prairies in the northern Great Plains of the United States with various grazing pressure, Schuman *et al.* (1999) found 5 to 11 times more biomass in roots than in all aboveground biomass, with the highest values reported for the most intensively grazed site. Root-to-shoot ratios were found to increase with decreasing water availability from a range of 3-5 to 18-25 as the grassland community shifted from tallgrass to shortgrass prairie along a west-to-east transect across the Great Plains (Derner *et al.* 2006; Sims *et al.* 1978), which Derner *et al.* (2006) attributed to enhanced root production in a water-limited environment.

In perennial systems, the reality of a standing stock of roots that persists year after year poses a challenge that is not faced in annual crop systems to modelling soil C inputs using the root-to-shoot ratio. Direct estimation of annual belowground NPP (BNPP) would be much more helpful. Gill *et al.* (2002) derived a simple algorithm for estimating BNPP that considered the proportion of live roots to total biomass and the root turnover rate. These authors found that BNPP was only 35 to 52% of measured belowground biomass for conditions typical of Australia. This calculation suggests that standing biomass in longer-lived species, both aboveground and belowground, may not be very representative of C inputs to the soil.

3.2.1. Rhizodeposition

Rhizodeposition describes all carbon loss from roots including lysates, sloughed cells, insoluble mucilages and mucigels, and water-soluble low-molecular weight exudates and secretions (Whipps and Lynch 1983). In total, these various rhizodeposits represent a significant C drain to plants, accounting for up to 40% of NPP under highly stressed conditions (Lynch and Whipps 1990), but more typical values range from 7 to 15% (Swinnen *et al.* 1995) with an average around 17% (Nguyen 2003). These numbers, however, should be viewed with some caution due to the difficulty in making these measurements (Rees *et al.* 2005).

There are numerous reasons for this high C flow from roots to the soil. A large portion of rhizodeposits are the indirect result of root growth and turnover, while a smaller, but ecologically significant, fraction are actively exuded and secreted by living cells. These exudates are now known to play active roles in plant-to-plant and plant-to-microbe signalling and in micronutrient acquisition via the stimulation of microbial communities (Bais *et al.* 2006).

Root exudates probably do not directly contribute much to soil C stocks. Boddy *et al.* (2007) found that these low-molecular weight exudates had half-lives of only 20 to 40 minutes in soil. If anything, root exudates contribute to the depletion of SOC stocks due to what has been termed the 'rhizosphere priming effect'. Rates of overall SOC decomposition have been shown to increase 3 to 5 fold in response to root exudates (Cheng *et al.* 1993; Dijkstra and Cheng 2007; Kuzyakov 2002). However, longer lived, more complex substrates derived from root turnover are likely to make significant contributions to SOC stocks due to both their

inherent biochemical resistance to decomposition and through indirect effects these compounds have on soil aggregation (Rees *et al.* 2005).

3.2.2. Mycorrhizal fungi and glomalin

Associations between plant roots and arbuscular mycorrhizal fungi (AMF) are ubiquitous in agroecosytems (Treseder and Cross 2006). Estimates of the amount of C allocated to fungal associates range from 4 to 20% of NPP (Graham 2000) with a large fraction of this C supporting the growth of new hyphae which have been estimated to have a turnover rate of days (Staddon *et al.* 2003) to months (Olsson and Johnson 2005). While the direct contributions to C stocks from hyphal turnover appears small (Zhu and Miller 2003), the indirect effects that hyphal growth has on soil structure and aggregate stability can have significant impacts on total SOC stocks (Miller and Jastrow 1990; Oades and Waters 1991) which will be discussed in subsequent sections.

Researchers have identified a glycoprotein-like substance, termed glomalin, which is produced by AMF hyphae (Driver *et al.* 2005; Wright and Upadhyaya 1996) and is found in significant quantities in most soils (Treseder and Turner 2007). The relatively high abundance of this glomalin-related soil protein (GRSP) is likely due to its slow decomposition rate (Rillig 2004; Rillig *et al.* 2001). The term, glomalin-related soil protein or GRSP, is used here because glomalin cannot be directly isolated and quantified in soil. Rather, GRSP is operationally defined by extraction procedures and detection by immunoreactivity assay (for detailed discussion of isolation and detection procedures see Rosier *et al.* 2006; Wright and Upadhyaya 1996; Treseder and Turner 2007) with different procedures producing differing quantities of GRSP especially in OM-rich soils where humic acids may be co-extracted (i.e. Rosier *et al.* 2006; Schindler *et al.* 2007).

Total GRSP has been reported to range from 4 to 8% of total organic C stocks across a range of soil and land-use types (Rillig *et al.* 2003; Rillig 2004), with one estimate as high as 22 to 27% (Nichols and Wright 2004). However, more conservative values in the range of 0.7 to 2.4% appear more common for agroecosystems (Table 1 of Treseder and Turner 2007). Given the current understanding of glomalin dynamics in soils, simple calculations suggest that values much above 2% of total C stocks are unreasonable based on the proportion of annual NPP that would be needed to support AMF hyphal growth (see Appendix 2). These calculations support the conclusions of Rosier *et al.* (2006) and Schindler *et al.* (2007) that certain GRSP extraction procedures are likely isolating more than just glomalin.

Regardless of the absolute accuracy of GRSP stock estimates, the majority of scientific evidence indicates that this substance is present in measurable but modest quantities in most soils and, more importantly, due to its biochemical properties, plays a substantial indirect role in C stabilization via soil aggregation (Wright *et al.* 1999; Wright and Upadhyaya 1996).

3.3. Losses and stabilisation

Ultimately, the ability for any given soil system to sequester carbon lies in the balance between net inputs, described previously, and net losses. Microbial decomposition and mineralisation to CO_2, is the dominant fate of organic carbon in the great majority of soils. In arid and semiarid systems, such as commonly found in Australia, photodegradation of surface litter also plays an important role (Austin and Vivanco 2006; Moorhead and Callaghan 1994). Austin and Vivanco (2006) found that blocking UV-light doubled the mean

residence time of surface litter in a semi-arid desert. Erosion has historically been a major loss mechanism for SOC from agroecosystems with estimates ranging from 20 to 50% of historic C losses (Lal 2004b). Eroded SOC can be a net sink for or a net source of CO_2 depending both on the frame of reference and on the fate of this eroded material (Stallard 1998; Yoo *et al.* 2005). In terms of C accounting on a given farm, erosional losses of SOC should be treated as a net liability since the fate of this C can no longer be accounted for within the boundary of the farm's area. More minor losses of carbon include gaseous losses as volatile organic compounds (Kesselmeier and Staudt 1999) and the leaching of soluble C from the soil to aquatic ecosystems. Schlesinger (1997) estimated that approximately 1% of NPP on a global average is lost to aquatic ecosystems as dissolved organic carbon (DOC), although DOC transport from the soil surface and adsorption to minerals in deeper horizons can be a significant sequestration mechanism (Sanderman and Amundson 2008).

On average, 1-2% of plant residues become stabilised as humified soil organic matter for significant periods of time (Schlesinger 1990). Yet this small net accrual over millennia has led to the large stores of C found in soils around the globe. In this section, we will define the most important concepts and then briefly highlight the dominant mechanisms of and controls on decomposition and stabilisation of soil organic matter in relation to carbon sequestration in agroecosystems.

3.3.1. Stable organic matter and humic substances

Strictly from a carbon accounting standpoint, the form of SOM that is added to a soil is not very important. Greatly increasing the inputs of a highly decomposable carbon source will increase SOC stocks, but if that new source of C inputs is removed this new SOC will be rapidly lost. However, if some of this new carbon becomes stabilised as humified SOM, then this sink has much greater permanence. Before discussing the mechanisms and factors that control the decomposition or stabilisation of SOM, it is important to define and describe this stable organic matter or humus.

The hard-to-characterize stable organic material, generically referred to as humus or humic substances (HS), is often described as "a series of relatively high molecular weight, brown to black coloured substances, formed by secondary synthesis reactions" (Stevenson 1982). Traditionally HS have been defined as the fraction of SOM that is not composed of recognizable biomolecules (MacCarthy, 2001), such as carbohydrates, proteinaceous materials, and lipids. Humic substances often represent 60 to 85% of total SOM, with the highest values often reported for agricultural soils (Haider and Guggenberger 2005), with the remainder being dominated by particulate OM which includes all plant residues.

The traditional concept of a highly condensed macromolecular structure for HS (e.g. Stevenson 1982) and even the exclusion of simple recognizable biomolecules from the HS pool have been thoroughly challenged in recent years (Kleber *et al.* 2007; Piccolo 2001; Sutton and Sposito 2005). With the aid of advanced molecular techniques, a new view of HS has emerged where HS are thought to be dynamic supramolecular associations (Kleber *et al.* 2007; Piccolo 2001) of diverse, relatively low molecular weight components. These components include recognizable but often partially-oxidized biomolecules (Nelson and Baldock 2005; Schnitzer 2000), stabilised by numerous mechanisms, with hydrophobic interactions and hydrogen bonding being of particular importance (Schnitzer 2000; Sutton and Sposito 2005). A major feature of this new view of HS is that stability is not imparted by the inherent recalcitrance the components; rather, the high degree of reactive (Schnitzer 2000) and hydrophobic (Kleber *et al.* 2007; Sutton and Sposito 2005) functional groups create an environment where enzymes are no longer effective in degrading the substrate. Of particular importance to SOC sequestration is the recognition that simple relatively-fresh

biomolecules can contribute directly to this stable OM pool and there does not necessarily have to be a long slow aging process to produce stable humus.

3.3.2. Decomposition and stabilisation

Decomposition is a key ecological process critical for maintaining a supply of most plant-essential nutrients. In natural ecosystems, nutrient recycling via decomposition often accounts for >90% of plant-available N and P and >70% for K and Ca (Chapin *et al.* 2002). Three concurrent processes contribute to decomposition: communition or fragmentation, leaching of soluble compounds, and microbial catabolism. Soil fauna will fragment and partially solubilise fresh plant residues, facilitating the establishment of saprotrophic microorganisms. Given an adequate water supply, a portion of the solubilised material will be transported deeper into the mineral soil or out into adjacent aquatic ecosystems. Following a cohort of residue through time, the initial microbial community will gradually shift as the most easily degraded compounds are used and the more recalcitrant materials accumulate. During aerobic metabolism, on average, 50% of a substrate will be oxidized to CO_2 in support of metabolic functions, while the other 50% is assimilated into microbial biomass (Prescott *et al.* 2004) which, in turn, becomes a significant and readily available secondary resource due to the rapid turnover of most microbial communities. Ultimately, given enough time and proper environmental conditions, most natural and xenobiotic compounds will be fully mineralized to inorganic forms (Marschner *et al.* 2008).

Stabilisation of SOM can be defined as any process which acts to slow the decomposition process. Some organic substrates, such as lignin and some paraffinic macromolecules, are energetically costly for microorganisms to degrade and these compounds often selectively accumulate, at least during the initial stages of decomposition (Kogel-Knabner *et al.* 2008), until more labile substrates are exhausted. It is now generally believed that lignin is only co-metabolized by a select group of fungi to gain access to cellulose and other more labile substrates located within the structural matrix of plant tissues (Prescott *et al.* 2004).

Depending on environmental conditions and biochemical characteristics, litter will take 10 years or more to fully decay (Harmon *et al.*, 2009). A growing body of evidence (Baldock *et al.* 1997; Ekschmitt *et al.* 2008; Marschner *et al.* 2008) suggests that beyond the decadal timeframe, selective preservation of relatively unaltered plant-derived compounds due to biochemical recalcitrance is not an important centennial-scale stabilisation mechanism. However, there are many soil processes that will act to slow down mineralisation which can be broadly grouped into two categories (Six *et al.* 2002; Sollins *et al.* 1996): 1) spatial inaccessibility of substrate to microbes and enzymes; and 2) interactions of organic matter with minerals, metal ions, and other organic substances. Both of these categories will impart a perceived recalcitrance due to the persistence of the stabilised material, but if that stabilisation process is interrupted, the substrate may be easily decomposed even if it is thousands of years old (Ewing *et al.* 2006).

Physical inaccessibility of an organic substrate to the decomposer, microbe or extracellular enzyme, can arise for numerous reasons. First, it is now evident that microorganisms and appropriate substrates are sparsely and heterogeneously distributed in the soil matrix, and many physical and diffusional barriers exist to microbial movement in the soil (Ekschmitt *et al.* 2008; Fry 2007; Young *et al.* 2008). Second, pores <0.2 μm in diameter are too small for most bacterial and fungal hyphae to enter, and enzymes can also be inhibited from entering as pore sizes decrease to <50 nm (Zimmerman *et al.* 2004). Third, the hydrophobicity of numerous substrates, especially partially-oxidized materials, will greatly reduce accessibility (Bachmann *et al.* 2008). Fourth, and perhaps most importantly in many topsoils, inaccessibility is caused by aggregation of the soil at multiple spatial scales (Six *et al.* 1999;

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Tisdall and Oades 1982) often creating strong diffusional limitations to enzyme and oxygen movement (Tokunaga *et al.* 2003).

Soil aggregation results from and is mediated by a combination of physicochemical and biological processes that result in the rearrangement, flocculation and cementation of soil particles (Bronick and Lal 2005; Six *et al.* 2004). Soil fauna, especially earthworms (Brown *et al.* 2000; Edwards and Bohlen 1996) when present, and flora are critical in creating and maintaining the stability of soil aggregates. Of particular importance, fine roots and fungal hyphae physically enmesh soil particles (Miller and Jastrow 1990), and various biological exudates, including polysaccharides, glycoproteins and mucigels, act as glues (Caesar-Tonthat 2002; Wright and Upadhyaya 1998). Numerous studies have found direct correlations between aggregate stability and SOC content and turnover rates (Jastrow *et al.* 1996; Six *et al.* 2000; Tisdall and Oades 1982). Much of the C that is lost during cultivation can be attributed to disruption of aggregates, decreased aggregate stability and increased turnover rates, especially of macroaggregates (Elliott 1986; Six *et al.* 1999; Six *et al.* 1998).

The second important class of stabilisation mechanisms involve various sorptive reactions with mineral surfaces, and complexation and precipitation reactions with polyvalent metals. Depending on the suite of soil minerals present and the solution chemistry, differing binding mechanisms with varying degrees of stability will operate (Kleber *et al.* 2005; Mikutta *et al.* 2007; Scheel *et al.* 2008). These complexes are thought to confer stability primarily by restricting enzyme and microbial access to the substrate (Vanloosdrecht *et al.* 1990). Concurrent with the increased acceptance of the importance of organo-mineral interactions in SOM stabilisation, there has been a proliferation of research into the movement, chemistry and stability of dissolved organic carbon (DOC) in soils (Kaiser and Guggenberger 2000; Michalzik *et al.* 2001; Neff and Asner 2001). For example, Sanderman and Amundson (2008) found that DOC transport from the litter and subsequent sorptive stabilisation in the mineral soil, while only representing a few percent of annual C inputs to the mineral soil, was responsible for 20 and 9% of total SOC stocks in a forest and grassland soil, respectively. Additionally, in soils that have lost a significant amount of structure due to years of cultivation, most of the remaining SOM is often found associated with mineral surfaces (Flessa *et al.* 2008).

So far, we have emphasized formation of humus through various physicochemical interactions; however, there is an important class of biochemically recalcitrant compounds, generically termed black carbon, formed as result of fire (Lehmann *et al.* 2008) that can constitute a significant fraction of SOC in most soils (Forbes *et al.* 2006; Skjemstad *et al.* 2002). The term black carbon (BC) refers to all residues of combustion with charcoal and soot dominating (Schmidt *et al.* 2001). While BC is not completely inert, research suggests that the majority of BC will turn over on centennial (Hammes *et al.* 2008) to millennial (Kuzyakov *et al.* 2009; Lehmann *et al.* 2008) time-scales. In fact, the FullCAM soil carbon accounting model has been successfully calibrated for Australian conditions by setting the inert SOM pool equal to the size of measured BC stocks (Skjemstad *et al.* 2004), recognizing that shifts in agricultural management are unlikely to impact this slow-cycling pool on relevant (decadal) timescales.

While inherent biochemical recalcitrance may slow degradation for a few years to decades, physical inaccessibility can retard degradation for decades to centuries, but organo-mineral and organo-metal interactions are now believed to be responsible for most of the highly stable (centuries to millennia) non-charred OM found in soils (Kogel-Knabner *et al.* 2008; Mikutta *et al.* 2006; Torn *et al.* 1997).

3.3.3. Controls on decomposition and stabilisation

With the exception of the abiotic process of photodegradation (e.g. Austin and Vivanco 2006) and the initial communition and mixing of residues by soil fauna (Hendrix *et al.* 1986; Seastedt 1984), each of which has its own set of environmental constraints, decomposition is primarily controlled by microbial activity. Any process or factor, such as aggregation and adsorption to mineral surfaces, that can act to constrain microbial activity, and ultimately enzymatic activity, will have an effect on overall decomposition rates. In a review of the decomposition literature, Lavelle *et al.* (1993) proposed a general model where decomposition is controlled by a hierarchy of factors which regulate microbial activity at a variety of spatial and temporal scales (Figure 3). Sanderman and Amundson (2003) recognized the utility of this model in integrating seemingly disparate results: 1) actual evapo-transpiration, a climatic metric combining temperature and moisture availability, could explain >90% of the variance in decay rates across wide geographic regions (Berg *et al.* 1993); 2) litter quality was found to explain 88% of the variance in decay rates between four forests in a similar region (Taylor *et al.* 1991); and 3) differences in the abundance and assemblage of soil fauna could lead to 3-fold increases in decay rates at a given site (Whitford *et al.* 1982). Given the overriding controls of climate and, secondarily, soil properties on decomposition and stabilisation, any study that attempts to detect management-induced changes in SOM stocks (e.g. Valzano *et al.* 2005) must carefully consider site selection.

Figure 3. A hierarchical model of the dominant factors controlling many soil processes in terrestrial ecosystems. Solid arrows represent direct regulation of biological processes and dashed arrows represent indirect controls. Not shown are the numerous biological feedbacks to the dominant factors. On right is an abbreviated list of anthropogenic factors that can significantly modify each of the main factors (after Lavelle et al., 1993).
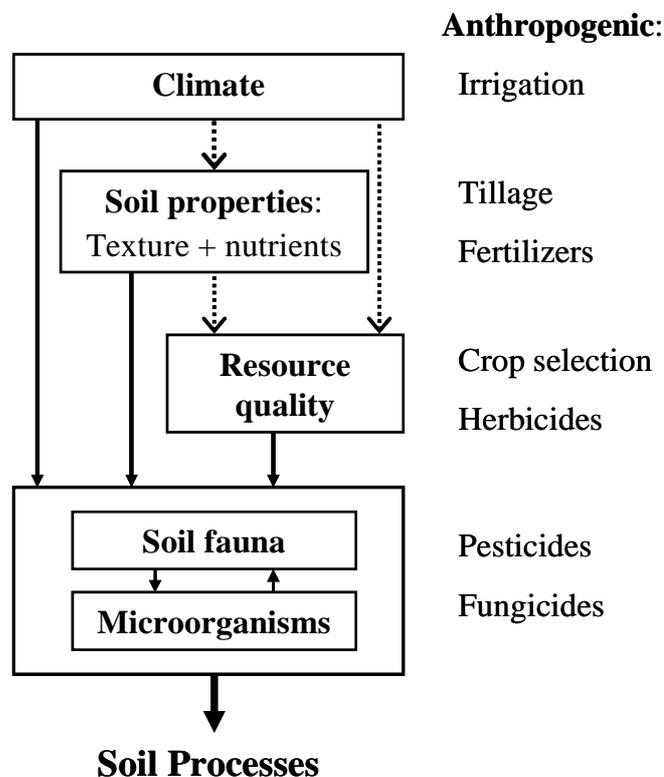
Over large geographic scales, climate, in the form of temperature and soil moisture, exert the strongest controls on C losses from soils. Soil respiration (Lloyd and Taylor 1994; Raich and Schlesinger 1992) and decomposition rates (Kirschbaum 1995; Sanderman *et al.* 2003; Trumbore *et al.* 1996) both increase exponentially with increasing temperature, often approximately doubling with each 10ºC increase in temperature. However, this temperature response will be moderated if moisture becomes limiting (Figure 4; Davidson *et al.* 1998; Xu and Qi 2001) and many seasonally dry ecosystems experience large pulses of respiration following intermittent rains. The sum of these rare pulse events can have a significant impact on the overall C budget of these ecosystems (Chou *et al.* 2008). Given the climatic extremes of much of Australia, year-to-year variability in temperature and especially soil moisture and their effects on decomposition rates may overwhelm any anticipated SOC response to a change in management.

Figure 4. Soil respiration versus temperature in an annual winter-rain dominated grassland when the soil is dry (volumetric water content (VWC) < 0.15) and wet (VWC > 0.35). Unpublished data from J. Sanderman.
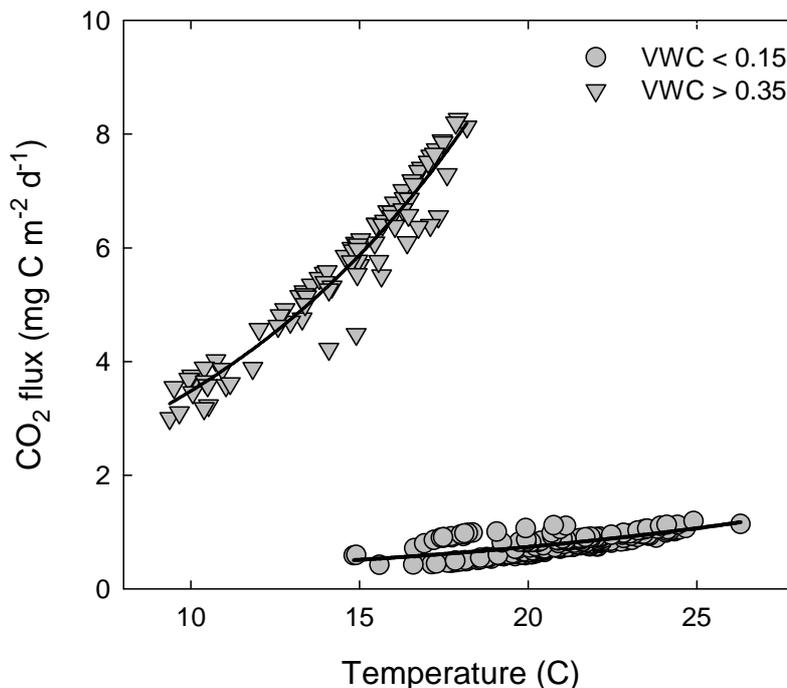
Soil physicochemical properties, such as texture, also exert strong controls on C losses primarily by reducing apparent decomposition rates through various stabilisation mechanisms, described above. Across Australia's cereal cropping belt, Dalal and Mayer (1986) found that C losses upon cultivation of virgin land occurred 20 times faster in sandy versus clayey soils (Figure 5). When respiration rates are normalized on a per unit substrate basis, researchers often find that decomposition rates of SOC slow with increasing clay content (i.e. Baldock and Skjemstad 2000; Schimel *et al.* 1985).

The hierarchical model presented in Figure 3 additionally suggests that the spatial location (e.g. surface, topsoil or subsoil) of decomposing material will have a large impact on decay rates and stabilisation processes. Surface residues, especially in semi-arid to arid regions, may be partially isolated, due to moisture limitations, from most soil fauna and flora resulting in decreased decay rates. However, in these same regions, photodegradation may be

sustaining moderately high decay rates (e.g. Austin and Vivanco 2006). The topsoil is often the most favourable decomposition environment with large and diverse faunal and microbial communities, adequate moisture, available nutrients, and good aeration. Conversely, the subsoil, especially in many of Australia's well weathered regions, presents the greatest potential for long-term stabilisation with smaller and less diverse microbial communities, limitations on oxygen diffusion, and an abundance of reactive mineral surfaces.

Figure 5. Drop in soil C upon cultivation of soils with varying texture. Specific plots were not followed through time, rather a chronosequence approach was used where similar fields under cultivation for differing amounts of time were sampled. Modelled losses using equation 2 are also shown with *k* values given in legend (redrawn from Dalal and Mayer, 1986).
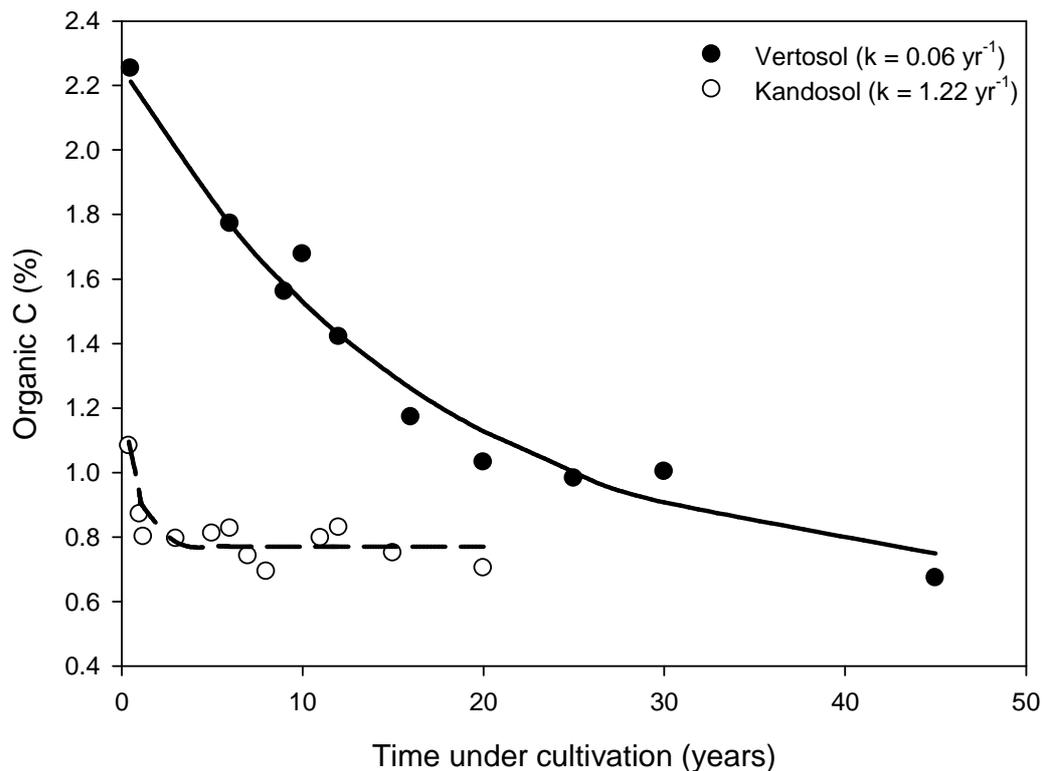
This division of the soil environment into surface, topsoil and subsoil, discounts the actual spatial heterogeneity found in the field. In reality, the region immediately surrounding actively growing roots, at least partially irrespective of depth, as well as preferential flow paths in well structured soils act as hot spots of biological activity (Bundt *et al.* 2001; Hinsinger *et al.* 2009) where both carbon inputs and decomposition are greatly accelerated.

In agroecosystems, nearly all management decisions will have either a direct or an indirect impact on microbial activity and thus decomposition rates (Figure 3). If carbon input levels do not change, management practices such as irrigation, especially in semi-arid regions, and tillage invariably increase decomposition rates by creating a more favourable environment for microbial activity and by disrupting the physical protection aggregates offer (Bronick and Lal 2005). Practices that purposely or inadvertently decrease faunal and microbial populations and diversity will likely decrease decomposition rates, but given the importance of fauna and flora in maintaining aggregate stability (Seastedt 1984), may indirectly increase the decomposition by way of reduced aggregate stabilisation (Six *et al.* 1999). A much fuller discussion of possible management impacts on decomposition rates is presented in Section 5.

3.4. Carbon balance

3.4.1. Equilibrium SOC values and multiple pools

Over long enough time periods, there must be an approximate balance between inputs and losses, otherwise soils would either be devoid of carbon or awash in the entire atmospheric carbon pool. Sixty years ago, Hans Jenny (Jenny *et al.* 1949) mathematically formalized this observation with a simple equation (Eq. 1) that has driven SOM research ever since. The change in carbon mass (C) over time is the difference between inputs (I) and losses, represented by first-order kinetics:

$$\frac{\partial C}{\partial t} = I - kC ,$$

where k is the fraction of C lost to heterotrophic respiration each year. This equation clearly illustrates that soil C sequestration can occur through two dominant pathways: increasing inputs (I) or decreasing losses (kC). If I is constant over time with an initial soil C content (C_0), the transient-state solution is:

$$C_t = \frac{1}{k}\left(I - Ie^{-kt}\right) + C_0 e^{-kt} .$$

From this second formulation, it is clear that as $t \rightarrow \infty$ carbon stocks will approach an equilibrium value dictated by the product of I and $1/k$ and the mean residence time (MRT) of the pool will be defined by C/I. While the assumption of a single homogenous pool is rarely if ever valid (Trumbore 2000), many interesting insights into regional and climatic differences in SOM cycling have been based on this equation (Amundson 2001; Raich and Schlesinger 1992; Sanderman *et al.* 2003).

In reality, given the diversity of organic substrates and various interactions with inorganic soil constituents, highlighted in the previous section, there exists a continuum of SOM pools each with differing input rates and residence times. This natural heterogeneity has been simplified in most current soil models (i.e. Parton *et al.* 1987; Richards *et al.* 2007; Skjemstad *et al.* 2004) by the introduction of three conceptual pools with distinct residence times: 1) an active pool that includes the majority of fresh plant residues with a residence time of, at most, a few years; 2) a slow pool with intermediate residence times, on the order of 10^1 to 10^2 years; and 3) a passive pool with residence times of 10^2 to 10^3 years.

An example of the importance of recognizing multiple pools with varying turnover times, τ, is given in Figure 6. In this hypothetical example, there is a step increase in C returns to the soil of 2 Mg C ha^{-1} yr^{-1}. Depending on the fate of this new C, there is a 3-fold difference in the amount of new C ultimately sequestered. If the new C remains as unprotected particulate C (POC), only 6 Mg C ha^{-1} are added to the soil with sequestration rates dropping to negligible levels after only 10 years. If some of this POC becomes incorporated into aggregates with decadal turnover times, then 8 Mg C ha^{-1} will be sequestered. If some of the POC and aggregate-protected C becomes adsorbed to mineral surfaces with a τ of 120 yrs, then 18 Mg C ha^{-1} will be added to the soil with sequestration rates >0.1 Mg C ha^{-1} yr^{-1} sustained for 30 years. An important collorary to this concept is that if the new inputs cease (Figure 6), the C stocks will return to their previous level rapidly if this new C has not become physically protected.

Figure 6. Total new soil C (A) and sequestration rates (B) following a 2 Mg C ha^{-1} yr^{-1} increase in inputs with a cessation of the new inputs after 60 years for 3 scenarios: #1 – all new C enters and stays in soil as unprotected POC (MRT = 3 yrs); #2 – all new C enters as POC but a fraction of the POC becomes protected within aggregates (MRT = 30 yrs); and #3 – as #2 with an additional fraction becoming stabilised by adsorption to minerals (MRT = 120 yrs). Model structure shown in (C), with the 3 pools shown as boxes (turnover time, τ, given for each pool), solid arrows represent transfers, and wavy arrows represent losses as CO_2. Steady state distribution of SOC amongst pools is ~25% as POC, ~15% as aggregate-protected and ~60% as sorbed C. This model is mathematically very similar to RothC and Century models, but the 3 pools are represented by specific stabilization processes.
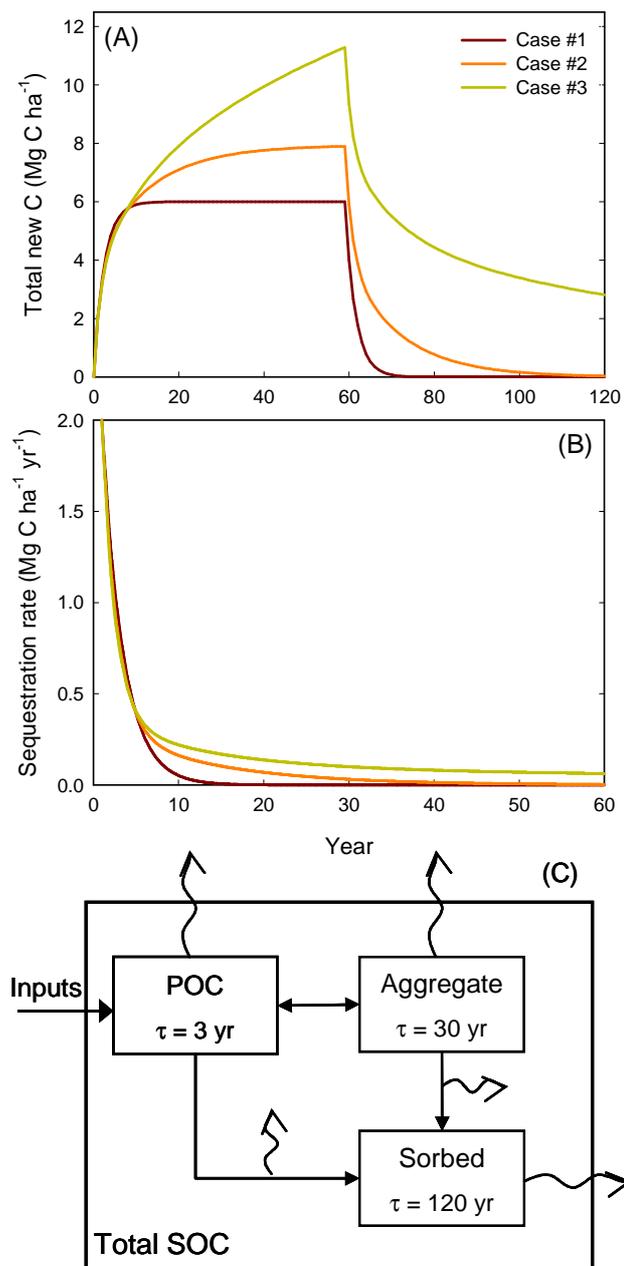
The concepts of equilibrium SOC levels and multiple pools leads to a second important insight for soil C management: the state of the soil system prior to a change in management may override any expected soil C gains (e.g. Persson *et al.* 2008). This is particularly true for soils that have only been cultivated for short periods of time with significant stocks of SOM derived from the native vegetation remaining. The initial drop in C input rates from the native state to the first cultivated state will likely take 50 to >100 years to be fully expressed as losses as the more stable SOM pools adjust to the new input levels. During this time period, a shift in management so that more residues, but still less than the amount prior to any cultivation, are returned to the soil may not result in increased C levels because the soil is still losing C as a result of the previous large drop in input rates (Figure 7a). However, if the soil had reached steady state with respect to the initial management, then the new management with increased residue returns would result in a significant gain in SOC (Figure 7b).

3.4.2. Concept of carbon saturation

Carbon saturation refers to the maximum level of C that a particular soil will be able to retain as stabilised SOM based on the physicochemical properties of the soil (Six *et al.* 2002; Stewart *et al.* 2007) with additional increases in inputs remaining as unprotected POM that will be rapidly cycled back to the atmosphere (Stewart *et al.* 2008). The reestablishment of equilibrium SOC levels following an increase in C inputs has been referred to as a form of carbon saturation by some researchers (e.g. West *et al.* 2004); however, we believe the term should only be used in the more narrow sense as defined above. While the idea of C saturation is conceptually and theoretically appealing, field data compiled from long-term agricultural experiments in support of the idea (i.e. Stewart *et al.* 2007) has been, in our opinion, less than convincing. For example, Blair *et al.* (2006) found that total C stocks, labile C, as defined as the portion oxidized by potassium permanganate, and non-labile C all increased linearly without showing any signs of saturating behaviour with manure input levels up to 200 Mg d.w. ha^{-1} yr^{-1} for 15 years. However, Stewart *et al.* (2008) did find evidence for saturation in several but not all mineral-associated C pools defined by a physical and chemical fractionation scheme (Six *et al.* 1998). Reactive mineral surface area is a finite resource, especially in topsoils, thus this is the stabilisation mechanism which would likely exhibit saturating behaviour.

If soils do indeed have a defined carbon saturation level, then similar management practices may result in positive sequestration in one soil that is far from its maximum C stabilisation level, while no change in another soil that is much closer to its saturation value (Six *et al.* 2002). Carbon saturation may be particularly important for understanding the long-term sequestration potential of input intensive practices such as manuring and other forms of organic matter amendment.

Figure 7. An illustration of the importance of the prior state of the soil system for determining the C sequestration response to a shift in management using the 3 pool model shown in Figure 5C.

In (A), soil was at steady state with respect to native vegetation (I = 10 Mg C ha^{-1} yr^{-1}) when cultivation began in 1950 (new I = 6 Mg C ha^{-1} yr^{-1}, τ_{POC} decreased to 2 yr, aggregate turnover (i.e. formation/destruction of the physical structure) increased from 0.05 to 0.2), then a shift to greater stubble retention (I = 7 Mg C ha^{-1} yr^{-1}) and decreased tillage intensity (aggregate turnover halved to 0.1) occurred 50 years later. In (B), the shift to the improved management occurred after the soil had achieved new steady state with respect to the first management conditions. In the non-steady state case (A), small C gains are seen for only 5 yrs following the improved management and, after 50 yrs, there is a net loss of an additional 4 Mg C ha^{-1}. Whereas, if the system was at steady state (B), there would be a net gain of 5 Mg C ha^{-1} after 50 yrs.
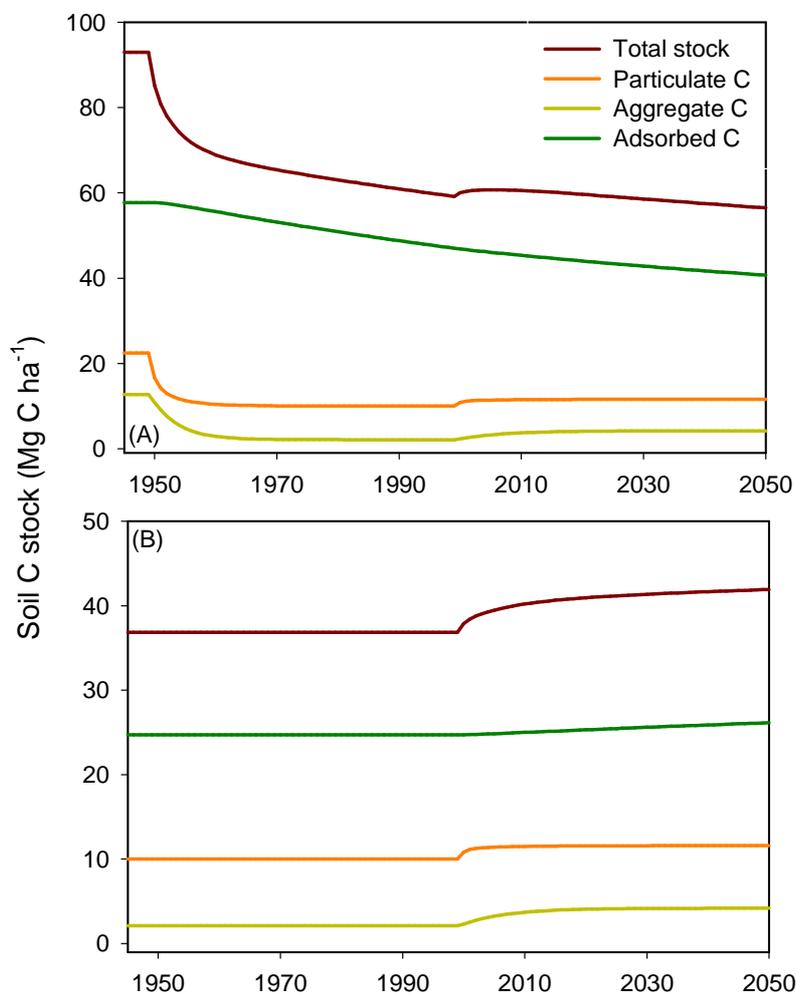
4. Measurement of Soil Carbon

While a full review of SOC measurement techniques is well outside the scope of this review, given that methodological difficulties and uncertainties are often assumed to be minor with what appears to be a straightforward measurement, we have highlighted some of the more pressing issues in this section. Stocks of organic C in soils are determined from two variables, namely SOC concentration and bulk density. Accurate measurement of C sequestration rates over time in soils based on typical repeated measure designs, necessitates the estimation of both variables each time (Don *et al.* 2007) and if a specified measurement depth is to be used, an adjustment to account for equivalent masses will be required if bulk density varies through time (Ellert *et al.* 2002; McKenzie *et al.* 2008).

Measurement of carbon concentration. The concentration of organic C in a soil sample is usually determined by converting a sample to CO_2 by either wet oxidation (i.e. Walkley and Black 1934) or dry combustion (i.e. Wang and Anderson 1998) and then quantifying the amount of CO_2 evolved. The wet oxidation method is known to underestimate the amount of organic C in most samples so a correction factor needs to be applied (Walkey and Black 1934). The magnitude of the correction factor is known to vary across soil types (Lowther *et al.* 1990; Navarro *et al.* 1993; Walkley and Black 1934; Wang *et al.* 1996) and by laboratory (Skjemstad *et al.* 2000). Figure 8 compares the C concentration results by wet oxidation (Walkley-Black method) and dry combustion (Leco CHN anaylzer) for a set of samples from various Queensland soils. For this data set, a correction factor of 1.21 seems fairly robust ($R^2 = 0.93$ with an intercept at the origin), however, the ratio can vary from 0.75 to 1.8 for any particular sample. Despite more accurate methods being available, the Walkely-Black technique is still used in some laboratories, particularly in Australia.

Figure 8. Plot of C concentration results obtained on the same samples by dry combustion versus wet oxidation methods (linear regression shown with 95% C.I.). Inset shows the actual sample to sample variability in the ratio of dry to wet methods with mean given as solid line. Data from Skjemstad et al. (2001).
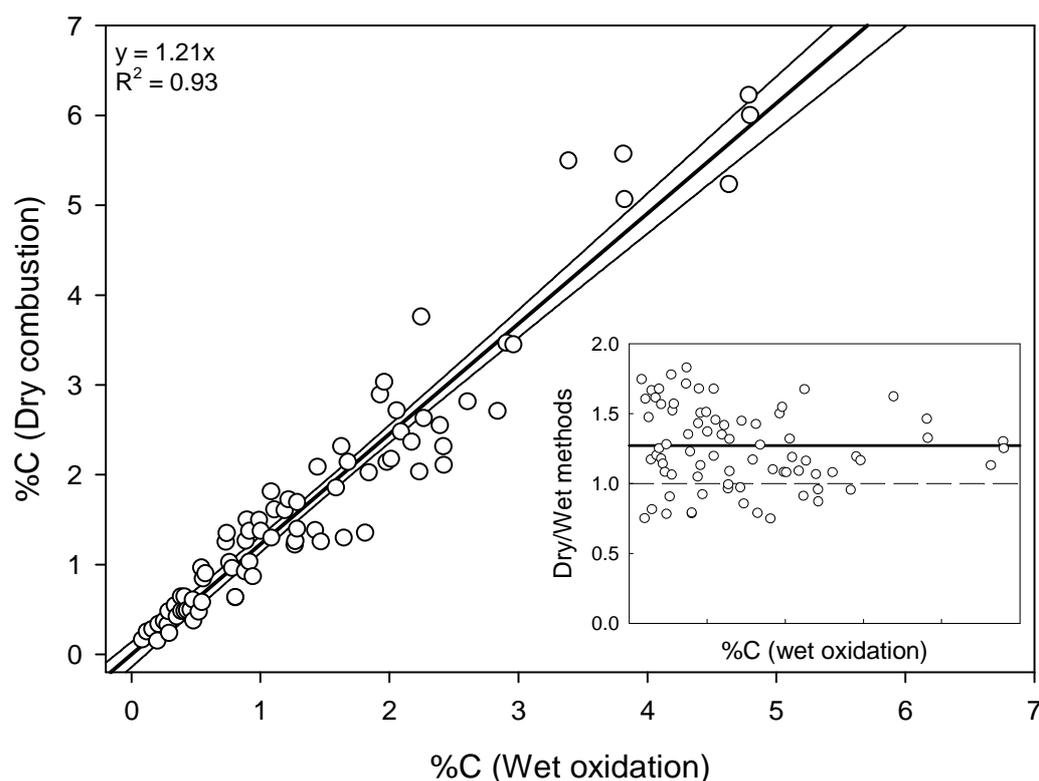
Measurement of bulk density. In many soils, accurate and precise determination of bulk density (ρ_b) is extremely difficult and time consuming due to the heterogeneous nature of texture, structure, compaction and gravel content. The most common measurement method involves driving small steel rings of a known volume into the soil area to be sampled. In stony soils, this method becomes impractical and researchers typically use either a water-replacement method or, if the soil is well aggregated, remove and analyse intact clods (see Cresswell and Hamilton (2002) for details on these methods). Recently, researchers have begun to test the applicability of gamma-ray attenuation probes commonly used in geotechnical applications for their utility across a range of nature soil conditions. Even small errors in ρ_b can translate into large absolute errors in SOC stocks. For example, in the upper 10 cm of a loamy grassland soil, ρ_b values were found to range from 1.12 to 1.59 Mg m^{-3} (J. Sanderman, *unpublished data*). With an average C concentration of 2%, SOC stocks in the 0 to 10 cm layer will vary by ~10 Mg C ha^{-1} across this range of ρ_b with this variance accounting for 30 to 50% of the SOC present. Even an overestimation of ρ_b by only 0.1 Mg m^{-3} will translate into a 2 Mg C ha^{-1} (or ~10% of the total) error in SOC for the soil in this example.

Equivalent mass. Despite numerous examples and appeals for sampling based on equivalent mass instead of similar depths (Dalal and Mayer 1986; Gifford and Roderick 2003; Wuest 2009), the great majority of studies and soil carbon accounting systems use a standard depth for calculating SOC stocks. The reason for preference for the equivalent mass approach is simple: all things being equal, a more compacted soil will contain more mass to a given depth, thus appearing to contain more carbon. In terms of SOC accounting, if similar depths are used as the baseline, management which tends to compact the soil may show erroneous gains in SOC while management which tends to inflate the soil may show erroneous decreases in SOC stocks. This issue underscores the need for accurate measurement of bulk density.

Coarse fragments. While the definition of soil organic matter *sensu strictu* includes living plant material, in practice a significant fraction of roots, living and dead, are generally excluded from a sample before C determination because analyses are typically performed on the < 2 mm fraction (Vaughan and Ord 1985). However, the amount of laboratory effort to either include or exclude roots varies significantly between studies introducing another source of uncertainty in comparing C estimates. Unfortunately, very few studies are clear in exactly what fraction of the organic matter found in soils is being included in their specific methodological definition of SOM; a complication that has hampered synthesis studies of soil C stocks and sequestration rates.

Natural heterogeneity. In all ecosystems, but particularly for those with patchy vegetation such as the majority of Australia's rangelands, there can be large spatial gradients in SOC stocks. For example, Jackson and Caldwell (1993) found a 5-fold variation in SOC content within a 120 m^2 area of sagebrush-steppe vegetation.

Figure 9. Changes in total SOC to 10 cm over time in an irrigated and fertilized *Kikuyu* grass trial at the Urrbrae Agricultural Centre (unpublished data courtesy Jan Skjemstad). Inset shows sequestration rates that would be measured over a two year period depending on which month of the year was chosen to sample. If different months were sampled in different years, then the results could have shown a net loss of C over time.
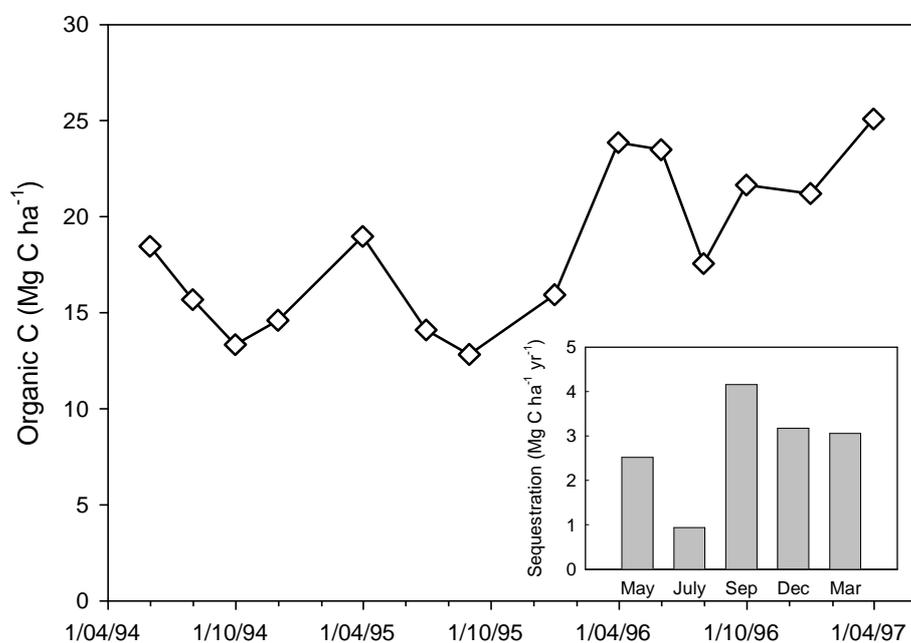
Topographic variation also often leads to significant spatial variation in SOC stocks (Gregorich *et al.* 1998), and paired-plot studies need to minimize differences in landscape position between treatments. In addition to spatial variability, soil C stocks in many ecosystems often display seasonal variations. This will be particularly true for annual cropping systems (Wang *et al.* 2004) that receive C inputs during a short growing season. In a wheat-fallow system, Wang *et al.* (2004) found that total soil carbon stocks in the top 10 cm decreased by 10% between early to mid-fallow (December and March sampling dates) and late fallow (July); however, replication was too low in this study to determine whether these temporal differences were simply due to spatial heterogeneity in sampling. Figure 9 highlights the importance of sampling date in determining sequestration rates during a *Kikuyu* grass trial.

In summarizing results from a 10 year mixed field trial, Peters *et al.* (1997) concluded that, "recording changes in total soil C and N may be useful for assessing the stability of a farming system after 50-100 years, but other physical and biological tests are needed for characterizing short term (1-10 yr) changes in the cycling dynamics of SOM." These researchers found clear and significant trends in water infiltration rates, percent water-stable aggregates, and several measures of microbial activity, all properties that are often positively correlated with SOM levels, but differences in C and N stocks were insignificantly higher due to large variances. Utilizing an intensive sampling scheme, Don *et al.* (2007) found minimum detectable differences (MDD) of 1.2 and 2.8 Mg C ha^{-1} for an Arenosl and a Vertisol, respectively, with a sample size of 20. Increasing sample size to 100 would only reduce the MDD to 0.4 and 0.8 Mg C ha^{-1}, for these two soils (Don *et al.* 2007). Similarly, Garten and Wullschleger (1999) found the MDD after 5 years of study was 1 Mg C ha^{-1} and that adequate statistical power (90% confidence) could only be achieved with a sample size

25

greater than 100. If sample size was reduced to 16 and the same statistical power was desired in this study, the MDD increased to 5 Mg C ha^{-1}.

Article 3.4 of the Kyoto Protocol states that "account uncertainties, transparency in reporting, and verifiability" should all be accounted for when monitoring carbon sink activities (Smith 2004b). Measuring and statistically verifying a small change in soil C stocks against a large and heterogeneous background is difficult at best, even at the farm scale, let alone at the regional or national level. While an individual scientific study may have the funds and resources to make accurate replicated measurements, a large-scale monitoring and verification system for accounting for changes in SOC stocks will depend on the level of stringency that a particular government finds acceptable and this level may likely be based on the financial trade-off between the value of the C credits and the cost of the monitoring program (Smith, 2004b). At the national scale, this system may take the form of robust modelling informed by detailed measurements in representative systems combined with verification of management practices and yields via reporting and remote sensing with some economic discounting to factor in verification uncertainty.

For all the reasons listed above, the Chicago Climate Exchange, one of a few markets that is currently including soil C sequestration in an emissions trading scheme relies on predicted changes in SOC stocks, ranging from 0.1 to 0.4 Mg C ha^{-1} yr^{-1}, for conservation tillage and either 0.3 or 0.7 Mg C ha^{-1} yr^{-1} for conversion to permanent grassland based on regional modelling and field trial results (available online at:
http://www.chicagoclimatex.com/content.jsf?id=781).

5. Management for Sequestration

In this section, we have attempted to present both a theoretical basis for potential sequestration due to management improvements and a summary of field evidence for stock changes both in Australia and abroad. We have attempted to cover the major agricultural practices as well as newer non-traditional practices. However, due to the shear volume of studies, a comprehensive summary of all available literature was not possible. In addition, there is currently a lack of published studies on many of the non-traditional management practices that theoretically hold potential to sequester carbon. Before discussing specific management practices, we begin this section with a brief review of the mechanisms responsible for the often observed large losses in SOC upon cultivating native land. We end with a brief discussion of large-scale synoptic studies and, finally, highlight additional uncertainties in predicting sequestration in light of potential climate change and economic uncertainty.

5.1. Why SOC stocks drop during cultivation

The reasons for the nearly universally observed reduction in SOC stocks that accompany clearing of native land for agriculture are thought to be well known. These reasons fall into two broad categories: 1) reduced inputs due to harvest and stubble burning; and 2) increased loss rates. Decreases in C inputs compared to the native state are well documented (Whitbread *et al.* 1998; Young *et al.* 2005) and the drop in soil C over time with cultivation has been successfully modelled by only decreasing input rates (Skjemstad *et al.* 2004). Cultivation may increase decomposition rates by exposing formerly occluded OM to microbial degradation due to both disruption of soil structure and changes in porosity that create a more favourable decomposition environment. Additionally, cultivation, especially when followed by fallow, creates favourable conditions for both wind and water erosion (Packer *et al.* 1992).

Interestingly, few studies have conclusively demonstrated that cultivation increases soil C losses (Dalal and Chan 2001). The majority of studies that have attempted to measure respiratory losses of CO_2 under differing management practices, often different tillage methods, have found either non-significant results (Drewitt *et al.* 2009), small transient effects (Dao 1998; Ellert and Janzen 1999; Reicosky *et al.* 1997; Roberts and Chan 1990), or unanticipated results where minimal tillage released more CO_2 than intensive tillage (Franzluebbers *et al.* 1995; Hendrix *et al.* 1988). On further thought, it is not surprising that short-term field measurements of soil respiration have failed to conclusively demonstrate that C losses increase under cultivation. Soil respiration is largely controlled by plant productivity (Janssens *et al.* 2001) and the size of the biologically active soil C pool (Amundson 2001), factors which are extremely difficult to control for in field trials. Additionally, measurement of respiration years after the initiation of cultivation may have missed much of the transient response of the more labile C fractions. However, studies that use more temporally integrated indirect methods, such as tracking isotopic shifts in specific SOC fractions, have been more conclusive in finding that cultivation, and tillage in particular, increases decomposition rates (Balesdent *et al.* 2000; Kisselle *et al.* 2001; Paustian *et al.* 2000). It is likely that cultivation does produce a small, but real, sustained increase in decomposition rates due to modification of the soil environment and disruption of soil aggregates, but there are too many confounding factors to easily measure this increase as a change in CO_2 losses.

5.2. Management shifts that can enhance SOC stocks

In general, any shift in management that increases inputs and/or reduces losses should build SOC stocks especially in soils that have lost significant quantities of SOC relative to pre-clearing conditions. Management shifts have been broadly classified into 7 categories: 1) management for increased yields; 2) tillage and stubble management; 3) crop rotation; 4) pasture and grazing management; 5) retirement of agricultural land; 6) offsite organic matter additions; and 7) alternative farming systems. Before delving into the details of each of these practices, we will highlight a few general concepts that require consideration in interpreting results regardless of management regime.

Given that climate and physicochemical characteristics of a particular soil exert such overriding controls on overall SOC dynamics, results from one field trial under a particular set of climate and soil conditions may not be applicable to another. This is especially true for Australia where we find both climatic and soil extremes. Additionally, the climatic conditions during the experiment period can mask any management effect. As an example, Sims and Bradford (2001) found that a semiarid moderately grazed rangeland, that was a net sink of up to 1.6 Mg C ha^{-1} yr^{-1} during a wetter than average year, was a net source of 0.5 Mg C ha^{-1} during a year that experienced a growing season drought. Moreover, positive management effects on SOC stocks have been found to decrease from moist to dry climates (Chan *et al.* 2003; Hamblin 1980; Ogle *et al.* 2005).

Only a small minority of studies have actually followed a change in management through time, the remainder have compared contrasting management styles after a number of years. Without the baseline data at the inception of a trial, it is impossible to determine if the difference between two treatments was due to one treatment increasing C stocks, the other treatment resulting in decreased C stocks, or some combination of the two. For example, Heenan *et al.* (1995) found that all trials (tillage × stubble treatments) at Wagga Wagga, NSW, resulted in decreases or no change in SOC content over the course of 14 years, but if only a single point measurement was made, the conservation practices would show significantly more SOC than the conventional practices (up to 31% as reported by Chan *et al.* 1992). In the following discussion of management effects, it is important to keep in mind that most of the presented sequestration values were actually derived from relative differences between two treatments. Section 5.3 develops and discusses the implications of this concept further.

The importance of long-term study has also been highlighted in several of the Australian trials. After 14 years at the Wagga Wagga trial there was no response to any of the conservation practices, but in years 14 – 21 (Heenan *et al.* 2004), the direct drilled/stubble mulched treatment started gaining SOC at a rate of 0.19 Mg C ha^{-1} yr^{-1}. Chan and Hullugalle (1999) and Hullugalle *et al.* (2002) offer a second example of the need for long-term monitoring. A large increase in SOC concentration was found under minimal tillage 3 years after converting from 15 years of continuous conventional cotton (Chan and Hullugalle 1999), but these gains disappeared and C stocks were lower than initial conditions by year 6 of study (Hullugalle *et al.* 2002). Hullugalle *et al.* (2002) attributed this delayed response to a transient effect of laser levelling of the fields at the start of the field trial. Without knowledge of the direction of the change in SOC stocks (i.e. an increase in stocks or just a reduction in losses) predicting future SOC response to changes in management will be difficult.

5.2.1. Management for increased yields

Historically agricultural systems have been managed to increase yields which has been widely believed to have corresponded with increased total plant production and hence inputs

of carbon into the soil. While true for some systems (Buyanovsky and Wagner 1998), improvements in yield have not necessarily corresponded with increases in soil carbon. Since yields (whether they be grain, forage, tubers) are essentially losses of assimilated carbon from agroecosystems, yields directly subtract from the amount of carbon that would otherwise contribute to soil C inputs. Therefore at the simplest level, yield increases present adverse outcomes for soil C inputs. Even when management has increased NPP concomitantly with yield improvements, those same management factors can result in higher rates of decomposition and declines in soil C (Khan *et al.* 2007).

Harvest indices describe the ratio of yield to total plant mass. Harvest index (HI) is usually conceptualised in terms of the partitioning of photosynthate between grain and other above ground biomass. Historically, shifts in HI have been responsible for much of the gain in yield for grains such as wheat, barley and rice (Sinclair 1998). For example, Perry and D'Antuono (1989) found that 80% of the yield increase in wheat was associated with increases in HI. In such cases, increases in yield have not necessitated increases in total plant production and, under such conditions, are unlikely to result in increased inputs of carbon into the soil.

Although widely assumed to be determined by the partitioning of photosynthate, over the longer term HI is perhaps more dependent on the availability of nitrogen and the ability of plants to accumulate it (Sinclair 1998). Accordingly the increases in yields in recent history have been associated with increased fertilisation and HI. Exceptions to this include maize which has had only a modest increase in HI, with most of the yield gain coming from increased total crop mass from fertilisation and breeding for cultivars that can tolerate tighter row-spacing (Sinclair 1998). Despite increased total productivity in maize, it has been proposed that impacts of fertilisation on residue quality, soil aggregates and decomposition of existing SOM can negate any gains in inputs due to increased net primary productivity (Fonte *et al.* 2009; Khan *et al.* 2007; Russell *et al.* 2009). In contrast, a review of nitrogen fertilisation and SOC found a positive relationship (Alvarez 2005) although interactions with other management practices such as tillage are also likely (Poirier *et al.* 2009).

Since the contribution of roots to soil C is greater than shoot derived C for many agricultural species (Rasse *et al.* 2005), shifts in root:shoot ratios will be of consequence to soil C sequestration. Root:shoot ratios are plastic, responding to the supply of assimilates and the availability of nutrients and water (Wilson 1988). Unfortunately, little information is available on the long term trends in root:shoot ratios of agricultural species that have accompanied increases in yields. Species with extensive root systems in combination with low input management should have the potential to increase inputs of C into the soil via partitioning more assimilate to the roots (Russell *et al.* 2009).

Much of Australian agricultural production is water limited. Increasing yields through breeding for increased water use efficiency (WUE) has been driven by plant traits that reduce transpiration and crop water use. However, maximizing WUE for crop production yields (i.e. grain) is not necessarily synonymous with maximizing productivity, leading Blum (2009) to argue that maximizing the effective use of water (i.e. minimizing non-stomatal transpiration and soil evaporation) is more important for increasing biomass than improvements in WUE.

In general, any practice that increases crop yields, as long as the harvest index and/or grazing pressure remains steady, should feedback into the soil system as increased C inputs (Denef *et al.* 2008; Gregorich *et al.* 1996; Kong *et al.* 2005). However, practices such as irrigation and fertilization will also stimulate microbial activity leading to increased decomposition rates (Calderon and Jackson 2002; Gillabel *et al.* 2007). The balance between increased inputs and increased decomposition will dictate how much, if any, C will be sequestered under these management practices. This trade-off between increased inputs and decomposition rates has been well illustrated in a study comparing centre-pivot

irrigated cereal crops to the dryland corners of the same fields (Denef *et al.* 2008; Gillabel *et al.* 2007). Inputs of C were ~ 3 Mg C ha^{-1} yr^{-1} higher in the irrigated versus the dryland portions of these fields, but after >20 years of irrigation, SOC stocks to 20 cm were only ~ 6 Mg C ha^{-1} greater under irrigation (Denef *et al.* 2008) due to concomitant increased decomposition rates (Gillabel *et al.* 2007).

Improvements to the condition of the many degraded and inherently infertile soils in Australia have the potential to contribute to increased SOC levels. Lime and gypsum amendments have been shown to lead to increased SOC levels in a Sodosol, which was primarily attributed to increased plant production and associated grain yields (Valzano *et al.* 2001). Interestingly, the reduced tillage treatments generally had greater SOC levels than the no-till trials across a range of lime and gypsum application rates. In this trial, the tillage helped evenly incorporate the lime and/or gypsum leading to better soil conditions and improved grain yields. In another example, Willis *et al.* (1997) reported large increases in soybean yields and SOC levels in a hard-setting subnatric Sodosol with improved bed preparation and irrigation methods.

In summary, yield increases, which have averaged 3% yr^{-1} for many cash crops over the last several decades (ANRA, http://www.anra.gov.au/), have generally come at the expense of C being returned to the soil (Perry and D'Antuono 1989). However, management shifts that alleviate a major limitation on crop growth and result in greatly increased productivity will likely result in greater C return to the soil and hence have the potential for sequestering SOC. But any increase in C inputs to the soil may be cancelled out by increased decomposition rates especially when the productivity gains were the result of fertilization and irrigation (Khan *et al.* 2007).

5.2.1.1. Soil amendments

There exists a wide range of non-traditional soil amendments available to farmers ranging from common inoculants to soil humic extracts to various animal-derived products. For example, the use of *rhizobia* bacteria to promote N fixation and crop yields in association with leguminous plants is now widespread with nearly half of Australia's leguminous acreage receiving inoculants (Wakelin and Ryder 2004). Some of these non-traditional products, microbial inoculants in particular, have been purported to substantially increase soil C stocks in recent years.

Other microbial inoculants have shown positive results in increasing nutrient availability, especially for phosphorous, and controlling plant root pathogens in both laboratory and greenhouse studies; however, field trials have shown mixed results in terms of increased yields (Evans and Condon 2009; Okon and Hadar 1987; Saubidet *et al.* 2002; Tinker 1984). Additionally, many of these products work by accelerating mineralization of organic matter to increase the supply of inorganic nutrients to the crop, meaning that there would have to be large gains in plant C uptake and return to the soil to realize any gain in SOC. Other soil amendment products, such as soil humic acids, are purported to increase cation exchange capacity (CEC) thus binding polyvalent cations that interfere with N and P availability. However, as indicated by Crozier *et al.* (2009), these organic amendments applied in recommended quantities to a field would have a negligible impact on increasing CEC due to the shear mass of soil relative to the amendment even using precision dressing methods. Imbufe et al. (2005) found that application of humic acids, as potassium humate, promoted soil aggregation in both acidic and sodic soils; however, the application rates in this experiment were 25 to 5000 times greater than typically recommended. At the current time, inadequate scientific evidence exists to fully assess the potential impact on C sequestration of any of these amendments outside of the yield gains shown in some field trials.

5.2.2. Conservation tillage and stubble retention

Conservation tillage was initially promoted as a way to save on fuel in the 1970s (Pratley and Rowell 1987). Increases in cropping intensity due to greater flexibility in seeding operations and a reduced risk of soil degradation, particularly wind and water erosion, are now two of the leading reasons for the relatively large-scale adoption of conservation tillage (Chan and Pratley 1998; D'Emden *et al.* 2008). Conservation tillage is generally defined to include minimum tillage, direct drill and no-tillage sowing systems. Despite concerns about weed control and increased risks of herbicide resistance, most growers in the major grain regions of Australia are planning on adopting no-tillage management (D'Emden and Llewellyn 2006). Due to its large scale adoption over the past 2 decades (>45% of Australia's grain crops (ABARE, available online at: http://www.abareconomics.com)), the potential for conservation tillage to enhance SOC content has received the most attention in the scientific literature in regards to soil C sequestration.

As part of an overall conservation tillage plan, stubble retention appears critical in successfully maintaining or improving crop yields with the additional benefits of minimizing soil erosion, conserving soil water and adding organic matter to the soil (Chan and Pratley 1998). In Australia, adoption of stubble retention has been highly uneven with only 14% of crops sown into stubble in central NSW compared with 98% in some regions of Western Australia (Connell and Hooper 2002). The most common reason for low adoption rates is that there is often reduced seedling emergence and subsequent impact on final grain yields. However, these negative effects can be easily overcome with proper management and Whitbread *et al.* (2003; 2000a) have shown that effective stubble retention can mitigate nutrient and SOC losses in wheat-legume rotations.

It is generally believed that SOC levels should be higher in NT systems versus comparable CT systems for several reasons. First, tillage disrupts aggregate structure which should lead to increased decomposition rates of a formerly slow-turnover C pool (Balesdent *et al.* 2000; 1990). However, the formation of these stable aggregates may be much slower than their destruction, thus accumulation of SOC may not be as rapid as rates of decline (Balesdent *et al.* 2000). This concept of hysteresis was nicely demonstrated by Pankhurst *et al.* (2002) by switching management practices after 14 years of trial and then sampling 3 years later. Applying tillage to the previously NT plots resulted in large losses of C from the upper 10 cm of soil, however, applying NT to previously tilled plots resulted in non-significant changes (Pankhurst *et al.* 2002). Second, runoff and thus soil erosion is generally much higher in tilled versus untilled fields primarily due to the degradation of soil structure (Packer *et al.* 1992). For example, after 7 years under NT, Packer *et al.* (1992) found that for the same amount of total runoff, sediment losses were 75% lower under NT than CT. Third, productivity gains resulting from improved soil health in the NT system should result in greater return of C to the soil system.

It is important to note that reduced tillage, where there is generally just a reduction in the number of passes with the plow, generally results in similar C levels as CT (Chan *et al.* 1992; Grace *et al.* 1998; White 1990). For example, White (1990) found that, while all treatments resulted in a net loss of SOC, the rate of loss for the reduced tillage treatment was similar to the existing conventional tillage which was nearly double that of the direct drill with minimal tillage treatment.

In a global meta-analysis of long-term agricultural experiments, West and Post (2002) found that the mean relative sequestration rate for conversion of conventional tillage (CT) to no-till (NT) was 0.57 ± 0.14 Mg C ha^{-1} yr^{-1} and a new equilibrium was reached in 15-20 years, with

75% of the studies showing increased SOC stocks. However, Baker *et al.* (2007) argued that sampling depth protocol has biased these tillage trial results. These researchers found that 37 of 45 studies with sampling depth <30 cm reported positive results in NT trials, while 35 of 51 studies with depths >30 cm reported losses. Additionally, micrometeorological studies of net CO_2 exchange (e.g. Baker and Griffis 2005; Verma *et al.* 2005) have found NT systems to generally be small net sources of carbon.

The majority of SOC increase under NT has been found to be in the top 10 to 15cm with insignificant changes or even decreases relative to CT at depth (Angers and Eriksen-Hamel 2008; Balesdent *et al.* 2000; Blanco-Canqui and Lal 2008; Kern and Johnson 1993; Thomas *et al.* 2007a). Thus newly sequestered C is accumulating where it is most vulnerable to environmental and management pressures and the permanence of these increases can be questioned. In a comparison of NT and full-inversion tillage (FIT) trials, Angers and Eriksen-Hamel (2008) found that while there was a small but significant increase in total SOC stocks under NT (100.3 vs. 95.4 Mg C ha^{-1} for NT and FIT, respectively), all of this increase was found in the upper 10 cm and there was actually significantly more C at the bottom of the plough layer in the FIT treatments (Figure 10).

Figure 10. Relative change in SOC content under no-till (NT) compared with full-inversion tillage (FIT) for various depth increments. Geometric means for each depth given by filled squares (redrawn from Angers and Eriksen-Hamel, 2008).
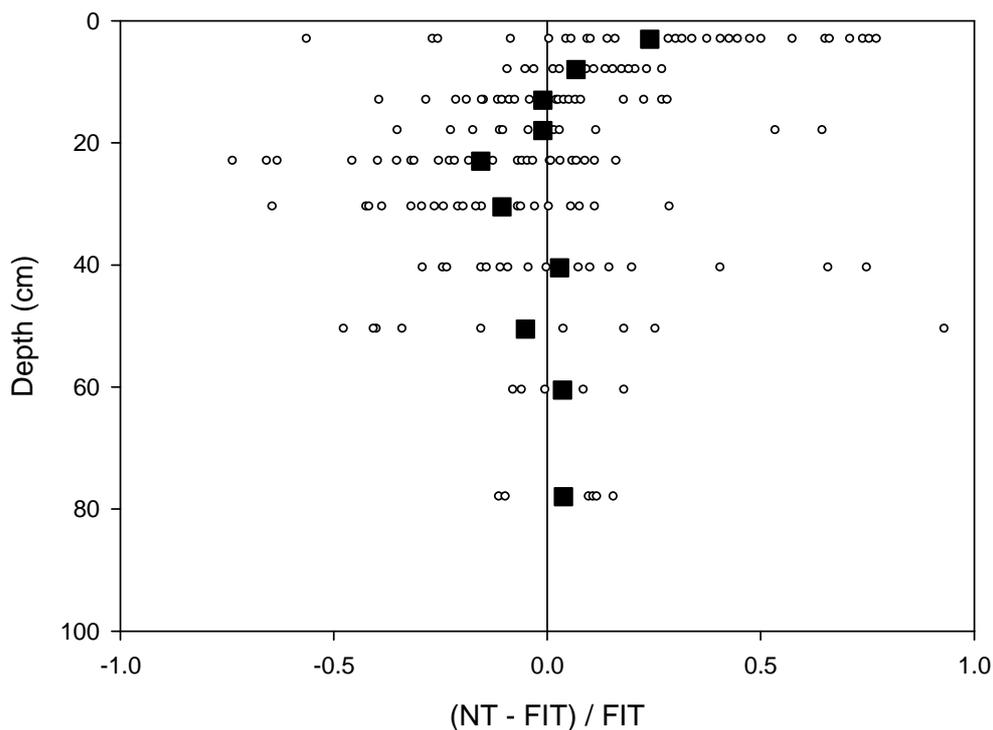
In dryland agroecosystems, there may be a trade-off between reduced tillage benefits and the benefits of additional organic matter incorporation into the mineral soil (Angers and Eriksen-Hamel 2008; Kushwaha *et al.* 2000; Olchin *et al.* 2008). For example, Olchin *et al.* (2008) found that with a plow depth of 15 cm, tillage-induced aggregate disruption had a greater influence on SOC stabilisation rates than residue incorporation into the profile resulting in net SOC gain under NT. However, when residues were distributed to a depth of 30 cm, the negative impact of aggregate disruption seemed to be offset by slower

decomposition of residues deeper in the soil profile resulting in no net SOC gain for NT and the possibility of a net loss of SOC over longer time periods.

The mixed results from tillage trials and differences in sequestration efficacy with different sampling depths are best understood by considering the competing stabilisation and destabilisation mechanisms at work. Tillage accelerates decomposition rates of older SOM (Balesdent *et al.* 2000; 1990) primarily by disrupting aggregate structure and, secondarily, by increasing soil temperature. Once a soil is tilled, smaller aggregates near the surface are exposed to more extreme wetting/drying cycles leading to further disruption of these aggregates (Olchin *et al.* 2008). As Balesdent *et al.* (2000) noted, the reconstruction of soil structure is a much slower process, taking years or decades, than the disruption process. However, tillage incorporates new OM deeper in the soil profile where decomposition rates are often lower than at the surface. In the upper 15 cm, soil temperatures decrease under NT, suppressing decay rates, but moisture levels generally increase, increasing decay rates. The balance of these competing factors will be dependent on climatic and soil physical properties (Balesdent *et al.* 2000), with the result likely being increased decomposition for drier soils.

Given the harsher climatic and soil conditions over much of Australia's grain region, results from more temperate northern hemisphere trials should be applied with caution, especially given that there has been little conclusive evidence of yield benefits in no-till systems in Australia (Kirkegaard 1994). Much of the available data on SOC stocks under different tillage practices for Australia has been summarized in a meta-analysis by Valzano *et al.* (2005). It is important to note that this analysis only compared C levels under different practices and did not explicitly examine temporal changes in C stocks in response to a change in management. In the executive summary, these authors concluded that while "there was a general tillage effect on soil carbon, with conservation tillage practices retaining up to 25% more carbon than conventional tillage practices, there were other factors that determine carbon levels in soil." This conclusion, which was subsequently misrepresented in the Garnaut Review (Garnaut 2008) to mean that a change from conventional to no-till will result in a 25% increase in C stocks, was not substantiated by the review of Valzano *et al.* (2005). There was only a small subset of studies in a specific climatic region where more SOC was found under NT than CT. In fact, among a large subset of sites in a moderately high rainfall zone, Valzano *et al.* (2005) found less C in NT than CT, which was attributed to the specific soil types, poorly structured sodosols and vertisols, in this grouping.

Sugar cane, primarily grown in the warm high-rainfall areas of coastal Queensland and northern NSW and covering approximately 550,000 ha (Table 1), has incredibly high productivity and significant quantities of residues are left behind annually after harvest. Traditionally, this "trash", representing 10 to 20 Mg d.m. ha^{-1} yr^{-1}, was burnt. The combination of large offsite removals, burning and cultivation has led to large decreases in SOC stocks (Blair *et al.* 1998). In the 1980s there was a shift away from burning to green cane trash blanketing, where residues were left as a mulch after harvesting. Wood (1991) noted numerous soil health benefits and increased sugar cane yields during initial field trials in Queensland. Despite this shift to a large annual influx of residues, field trials have only shown modest total SOC stock gains which have been confined to the uppermost soil layers (Ball-Coelho *et al.* 1993; Blair *et al.* 1998; Blair 2000; Robertson and Thorburn 2007), even in a 60-yr field trial in South Africa (Graham *et al.* 2002). These modest results are not surprising given that nearly 100 Mg d.m. ha^{-1} yr^{-1} are removed at harvest (ASMC 2005) and litter decomposition rates are extraordinarily fast given the warm, humid climate.

In general, it appears that in the drier regions of Australia, SOC stocks are less sensitive to changes in tillage and stubble management. Chan *et al.* (2003) reviewed tillage × stubble trials in light-textured Australian soils and found that management changes only resulted in significantly positive differences in regions of higher rainfall (>500 mm yr^{-1}) and thus higher

plant productivity. Hamblin (1980) after finding only small and variable differences across several Australian tillage trials in drier regions, concluded that "these results are somewhat disappointing viewed beside the improvements noted in wetter climates." An additional factor that may also be masking sequestration under NT in Australia is the prevalence of stubble grazing thus effectively removing this potential soil C input. As a stand-alone practice NT will likely not lead to increased SOC stocks, but if NT is integrated into a comprehensive management system that includes stubble retention and N fertilization, positive sequestration outcomes are more likely (Wang and Dalal 2006).

5.2.3. Cover crops and enhanced rotation

Fallowing in many cash crop systems has primarily been utilized to restore soil moisture and nitrate in rain fed semi-arid regions and to reduce weeds and disease (Sims 1977). However, bare fallows exacerbate SOC losses by eliminating soil C inputs for the fallow period, and by creating favourable conditions for enhanced decomposition (warmer, moister) and surface erosion. The importance of minimizing fallow is highlighted in Figure 11 where cumulative time under fallow was the dominant factor controlling long-term changes in SOC stocks under 6 different rotations (data from Holford *et al.* 1998).

Figure 11. Change in SOC stocks as a function of cumulative time under fallow. In a long-term trial at Tamworth, NSW, on two soil types, the average (error bars = 1 s.e.) change in SOC stocks (0–15 cm) for 6 different wheat rotations (long fallow, continuous wheat, subterranean clover, medic, lucerne, and chickpea) was found to be highly correlated to the cumulative time under fallow regardless of rotation. Stock changes were calculated from Holdford et al. (1998) for each measurement year 1987 to 1993 (n = 4) relative to initial measurements in 1966 for each treatment.
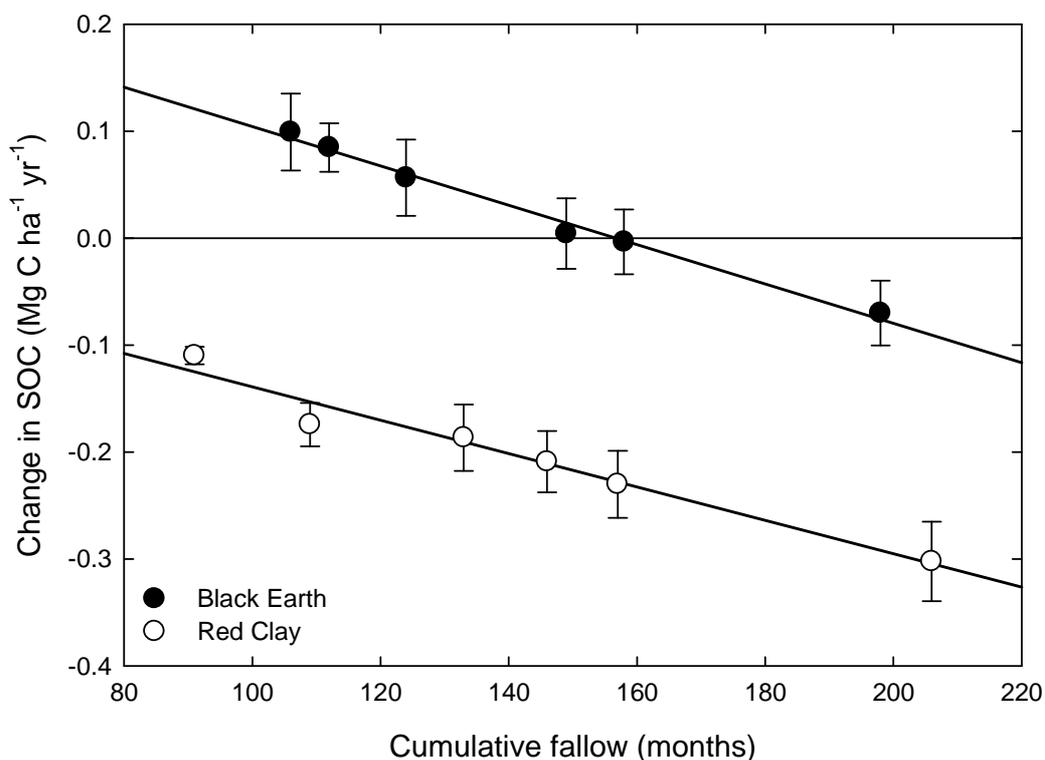
Cover cropping and planted fallows have the potential to reverse this SOC loss and help restore degraded land (Lal *et al.* 1979). For example, at the long-term Waite Permanent Rotation trials, SOC concentrations increased linearly with increasing frequency of pasture phase (Grace *et al.* 1995). Legume/cereal rotation, now a common practice in much of southern Australia, rotates N-fixing leguminous pastures, which are typically grazed by sheep or beef cattle, with cereal grains with an aim to improve soil nutrient levels and soil structure, as well as to provide a disease break and ultimately to improve the yield and quality of the subsequent cereal crop (Whitbread *et al.* 2000a). The importance of using legumes instead of annual grasses was highlighted by Whitbread and Clem (2006), who found large increases in soil nitrate levels and resultant sorghum yields following 2 years of legume pasture but not for grass-based pasture.

Sequestration results from Australian rotation trials have been mixed with positive SOC gains depending upon the length of pasture rotations and the initial state of the soil system. In a low-fertility Queensland Vertisol, Dalal *et al.* (1995) measured SOC increases of 0.65 Mg C ha^{-1} yr^{-1} for 4 years when a grass + legume pasture was introduced into a wheat cropping rotation. These authors attributed the increases in SOC to an increase in belowground C allocation (4.5 Mg C ha^{-1} yr^{-1} in grass vs. 0.9 Mg C ha^{-1} yr^{-1} under wheat) although the increases in SOC were confined to the top 2 cm of soil. When continuous wheat cropping resumed, SOC stocks declined, but were still higher than the continuous wheat fields after 4 years of cropping. Chan (1997) found that in these soils, particulate organic carbon (POC), defined as the >53 µm fraction, was preferentially lost during cultivation and was also the primary form of soil C gained upon return to pasture.

The type of legume planted during the pasture phase has been shown to impact resultant soil C gains. In a north-western NSW trial, Whitbread *et al.* (2000b) found significant SOC gains when a long fallow was replaced by lucerne but not when chickpea or medic was grown. These authors also found that the resulting decreases in C stocks and quality during the wheat phase partially depended on whether or not stubble was retained. However, highlighting the importance of the initial state of the soil system (see Section 3.4.1), Persson *et al.* (2008) found that incorporation of 2-yr perennial grass and mixed grass + legume leys into a 6-yr crop rotation only slowed the loss of SOC in the upper 30 cm over a 31 yr period.

In the sandy soils of Western Australia, results from 7 long-term rotation trials showed small but insignificant gains in SOC when various lengths of pasture rotations were incorporated into continuous wheat. Overall, the only significant increases in SOC stocks were found for continuous pasture trials (Figure 12a). Within the rotation treatments, trials that were under pasture <50% of the time lost SOC, while trials that were under pasture >50% of the time gained SOC (data not shown).

An innovative new management practice that can potentially offer increased farm profits, sustainability and high SOC sequestration rates is through the use of pasture cropping (Howden *et al.* 2005). Pasture cropping involves direct drilling of a cereal crop into existing perennial pasture grass thus maintaining year-round cover while producing an annual cereal crop and sustaining grazing for much of the remainder of the year. While there is anecdotal evidence (also, see Appendix 1 for a modelling perspective on productivity of pasture cropping) that pasture cropping can sequester significant quantities of C in wetter equiseasonal rainfall regions, to date, there has been a lack of well designed field trials to substantiate these claims. Recently in a paired-plot study in New South Wales, Chan *et al.* (2010) found no significant difference between pasture cropped land and paired pasture lands. Additionally, due to the high water demand by both crops, there will likely be geographic limitations to wide-scale adaptation of pasture cropping (Howden *et al.* 2005).

5.2.4. Pasture and rangeland management

The nature, frequency and intensity of disturbances are key drivers of the carbon balance in grassland ecosystems (Soussana *et al.* 2007b). In general, grazing at appropriate stocking levels will maintain or enhance soil C stocks (Conant *et al.* 2001) due to positive effects on vegetative growth (LeCain *et al.* 2000; McNaughton *et al.* 1996; Sims *et al.* 1978) and turnover of both aboveground shoots and belowground roots (Nyborg *et al.* 1999; Schuman *et al.* 1999; Sims and Singh 1978). However, many of the details are not well understood (Ingram *et al.* 2008; Parton *et al.* 2001) and mixed results are common (Derner *et al.* 2006; Pineiro *et al.* 2009). Schuman *et al.* (1999) speculated that animal traffic leads to enhanced physical breakdown, incorporation into the mineral soil, and decomposition of litter, all factors which should lead to enhanced SOC stocks. Additionally, the non-digestible fraction of consumed forage (25 to 40% of intake depending on digestibility) will be returned to the soil as already partially humified material (i.e. dung). Grazing also accelerates nitrogen cycling and promotes N losses through volatilization of ammonia from urine and dung patches (Pineiro *et al.* 2006) which, in nutrient limited systems, may constrain C inputs (Wedin and Tilman 1996) and humification rates. Pineiro *et al.* (2009) speculated that in semiarid grasslands that are constrained by water availability, mechanisms that effect C inputs will determine the net C balance of grazed lands, whereas in subhumid and humid grasslands, mechanisms that effect nutrient availability will determine the C balance.

Figure 12. Results from Western Australia rotation trials. (A) Mean stock changes (0–30 cm) over time for continuous wheat (WW), permanent pasture (Pa), and rotations with varying proportion of pasture phase. Sequestration rates are calculated from time series data (black bars) and as relative treatment effects with WW as control (open bars). Significant differences between treatments for time series data are given as different letters above bars (Tukey-Kramer HSD, $\alpha = 0.05$). (B) Time series of SOC stock changes (0–30 cm) for the Merridian (light) trial comparing continuous wheat (WW) to 2 year pasture for every wheat crop (PPW) rotation with conventional tillage (CT) or direct drilling (DD) during wheat phase. The annual rate of change and R^2 values for linear regressions are given in parentheses next to symbols. Data from Skjemstad and Spouncer (2003).
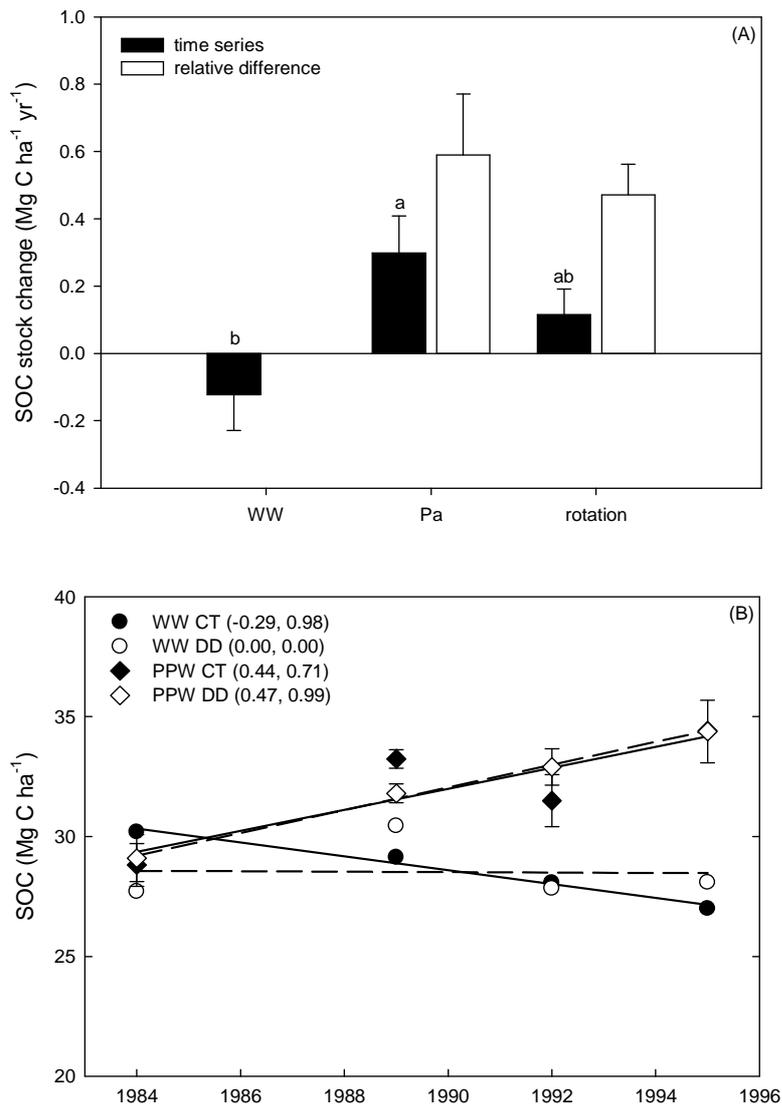
5.2.4.1. Improved Pastures

Conant *et al.* (2001), in a meta-analysis of grassland management effects on SOC storage, found that 75% of studies showed increases in SOC with a mean of 0.54 Mg C ha^{-1} yr^{-1} (n = 167). The range was 0.11 to 3.04 Mg C ha^{-1} yr^{-1}, although there were very few results with sequestration rates >1.0 Mg C ha^{-1} yr^{-1} and, of these studies, Conant *et al.* (2001) concluded that many may have "unique issues". Conversion from cultivation to pasture generally had the greatest sequestration response (mean = 1.01 Mg C ha^{-1} yr^{-1}, n = 23), improved grazing had a mean response of 0.35 Mg C ha^{-1} yr^{-1} (n = 45), and fertilization had a response of 0.30 Mg C ha^{-1} yr^{-1} (n = 42). Importantly, of the studies that examined different levels of grazing intensity, nearly 30% found lower SOC contents for moderately grazed as compared to heavily grazed treatments. In a series of studies on hayed native North American grassland, Nyborg *et al.* (1999; 1998) found that N + S fertilizer (no effect with N or S alone) increased C stocks at a rate of 0.5-1.0 Mg C ha^{-1} yr^{-1} over 5-13 years but that most of this increase came in the POC fraction. Micrometeorological studies of managed grasslands generally show that these systems are net sinks for CO_2 with average sequestration rates of 1.0 ± 0.7 Mg C ha^{-1} yr^{-1} (Soussana *et al.* 2007a), although once other greenhouse gases are considered, the carbon balance is no different from zero (Soussana *et al.* 2007a; 2007b).

Studies of improved pastures in Australia (Chan *et al.* 2010; Crocker and Holford 1991; Kohn *et al.* 1977; Ridley *et al.* 1990; Rixon 1966; Russell 1960; Williams and Donald 1957) have generally focused on pasture improvements, primarily fertilisation with phosphate and lime. Of the studies with suitable data, carbon stocks increased at a rate of 0.29 ± 0.17 Mg C ha^{-1} yr^{-1} (mean ± s.d., n = 15) with fertilisation and by 0.11 Mg C ha^{-1} yr^{-1} (n = 4) with other improvements, including irrigation and sowing of legumes. When pastures were established on previously cultivated land in both Western Australia and southern Queensland (Skjemstad and Spouncer 2003; Young *et al.* 2009), SOC stocks increased relative to cultivated controls at a rate of 0.5 – 0.6 Mg C ha^{-1} yr^{-1}, but only at an absolute rate of 0.33 ± 0.25 Mg C ha^{-1} yr^{-1} (n = 9).

Several non-traditional grazing practices, primarily involving short, intensive grazing periods with rotations through upward of 20 or more paddocks, have received a good deal of attention in Australia and abroad for proven increases in herbage production and stocking densities (Fulkerson *et al.* 1993; Waller *et al.* 2001; Warn *et al.* 2002; Watson *et al.* 2001). For example, Fulkerson *et al.* (1993) found that intensive rotational grazing, where paddocks are restocked based on grass growth-phase characteristics, increased aboveground dry matter production 25 and 35% over 4- and 2-week fixed interval rotations, respectively. Additionally, Warn *et al.* (2002) found that a similar intensive rotation allowed 17 to 21% more sheep stock to be carried without compromising wool production or quality in a sown pasture in Victoria. While increases in dry matter production, especially when herbage is maintained through dry summer months (i.e. Waller *et al.* 2001), and turnover of plant material due to increased grazing should translate into SOC stock gains, this remains to be conclusively demonstrated.

The introduction of perennial grasses, especially into marginal pastures, has the potential to greatly increase pasture productivity (Witham *et al.* 2007), mitigate drought and salinization risks, and build soil C stocks while minimizing surface erosion (Ward 2006). Perennial grasses, compared to annuals, generally allocate a greater fraction of productivity to the maintenance of a deeper and more extensive root system including associated mycorrhizal fungi. Increased belowground C allocation coupled with an increase in duration of C inputs and decreased surface erosion due to greater grass cover through dry months, especially when compared to a comparable annual grass pasture, should lead to increased soil C stocks (see Appendix 1 for a modelling perspective on potential productivity gains

associated with planting of perennial pastures). Additionally, respiratory losses of SOC may be reduced under perennial pastures because summer rains are effectively and rapidly utilized relative to typical summer fallow under annual grasses in winter-rain dominated climates (Paydar *et al.* 2005; Young *et al.* 2009) leading to lower soil water contents and reduced decomposition rates. However, the magnitude and permanence of an increase in SOC stocks under perennial pastures are currently unknown, but under study, and are likely to vary greatly across Australia's diverse soils. In a recently published study, Young *et al.* (2009) found significant increases in SOC stocks (0.15 – 0.35 Mg C ha^{-1} yr^{-1}) down to 20 cm over 6 years of repeated measurements when various perennial pastures were established on a formerly cultivated vertisol in NSW.

5.2.4.2. Rangelands

Sequestration estimates for more marginal and less-managed rangelands generally fall below 0.5 Mg C ha^{-1} yr^{-1} (Schuman *et al.* 2002; Smith *et al.* 2008; West and Post 2002). By adoption of U.S. National Resource Conservation Service (NRCS) recommended management practices, Schuman *et al.* (2002; 1999) estimated that the 183 Mha of U.S. rangelands could sequester C at a rate of 0.1 Mg C ha^{-1} yr^{-1} with an additional 0.2 to 0.3 Mg C ha^{-1} yr^{-1} in avoided emissions. Complicating efforts to generalize rangeland management is the marginal climate of much of the world's rangelands. For example, heavy grazing of a North American grassland lead to increased SOC stocks during a series of moderate to wet years (Schuman *et al.* 1999) but, in the 10-yr follow-up to this experiment, the heavily grazed treatment was found to have lost 30% of its SOC to 60 cm due a sustained drought that facilitated a shift in vegetative community (Ingram *et al.* 2008) leading to reduced C inputs to the soil system.

Unfortunately, much of the available scientific data on rangeland C management comes from the Americas where the soils generally contain significantly more organic matter (due to a combination of a more favourable climate and better soil properties) than typically found in Australian rangelands. The applicability of these results to Australia is, therefore, questionable. Currently, the most comprehensive effort to understand the interactions between grazing pressure and climatic variability on soil C stocks in Australia's rangelands is the Range-ASSESS model (Hill *et al.* 2003; Hill *et al.* 2006). While not intended to provide verifiable quantitative data, this modelling exercise (Hill *et al.* 2006) found that, due to the interactions between drought, grazing pressure (both livestock and feral grazers), fire and vegetation cover, the most conservative outcome of current stocking levels would be a continued decrease in C stocks at 100 to 200 Tg C per 5 yr period across all of Australia's rangeland (approx. 0.04 – 0.08 Mg C ha^{-1} yr^{-1}) with large regional differences. Additionally, only a small number of scenarios were estimated to increase in soil C stocks and these primarily involved reducing stocking levels to 20% or less of 1997 levels (Table 2).

The introduction or re-establishment of certain salt- and drought-tolerant palatable plants, such as saltbush, tagasaste and other perennial shrubs into rangeland, is being promoted to improve the soil health of marginal and degraded lands while allowing for stocking densities to remain the same or even increase (Lauder 2008; Masters *et al.* 2006; Thomas *et al.* 2009). In general, perennial desert shrubs act as "islands of fertility" and have significantly greater soil C and nutrient levels beneath the shrubs than in the inter-shrub spaces (Ewing *et al.* 2007; Schlesinger *et al.* 1996). While replicated and controlled studies are currently lacking, the introduction or re-introduction of deep rooted perennial shrubs into Australian pastures currently dominated by annual grasses should increase SOM levels at least in the vicinity of these shrubs.

Given the potential for more active management on improved pasture lands there appears to be a much greater sequestration potential on these lands relative to Australia's rangelands.

However, given the sheer extent of the rangelands, even a very modest increase in SOC stocks will result in a large offset for Australian GHG emissions.

Table 2. Five year change in soil C density (Mg C ha^{-1} yr^{-1}) for differing stocking intensities (% of 1997 levels) derived using the Range-ASSESS model. See Hill et al. (2003) for definition of rangeland regions. Statistics were derived from running the model 100 times using different years of a 100 year climate record as the starting point for 5 year scenarios to create a range of climatic conditions.

Region	Area (000 ha)	Current SOC (Mg C ha^{-1})	Change in SOC (Mg C ha^{-1} yr^{-1}) with given stocking density			
			200%	100%	50%	0%
S-A Wood	377	44.2	-1.58 (0.1)	-1.43 (0.1)	-0.68 (0.2)	-0.02 (0.1)
Chenopod	566	24.4	-6.58 (7.4)	-4.24 (5.5)	-0.62 (2.9)	1.87 (1.1)
Mallee	82	41.6	-0.41 (0.5)	-0.23 (0.3)	0.02 (0.1)	0.02 (0.0)
Mitchell	432	38.1	-1.37 (1.6)	-0.37 (0.8)	0.18 (0.4)	0.39 (0.2)
NTallgrass	559	50.0	-0.87 (0.2)	-1.04 (0.1)	-1.02 (0.1)	-1.61 (0.1)
HumWood	783	55.5	0.02 (0.0)	0.02 (0.0)	0.01 (0.0)	0.00 (0.0)
AridMulga	593	22.7	-0.46 (1.8)	-0.04 (1.6)	0.21 (1.5)	1.17 (0.6)
ETallrass	217	93.0	0.20 (0.6)	0.27 (0.5)	0.58 (0.2)	0.58 (0.2)
Midgrass	278	57.9	0.11 (0.7)	0.41 (0.4)	0.53 (0.2)	0.53 (0.2)
Cracking	81	118.8	-0.70 (1.0)	-0.18 (0.2)	-0.02 (0.1)	0.01 (0.1)
Total	5542	40.0	-0.77 (0.8)	-0.49 (0.5)	-0.10 (0.3)	0.10 (0.1)

5.2.5. Retirement of agricultural land

Retirement of degraded and marginal land from agriculture by planting permanent vegetation through the conservation reserve program (CRP) in the United States was initiated in response to severe soil erosion, but in more recent years these lands have been seen as potential large carbon sinks with sequestration rates ranging from nil to 1.1 Mg C ha^{-1} yr^{-1} (Baer *et al.* 2002; Gebhart *et al.* 1994; Potter *et al.* 1999; Reeder *et al.* 1998). For example, in cultivated clay soils in central Texas, USA, using a chronosequence approach, Potter *et al.* (1999) found that retirement of agricultural land that was under long-term cultivation through the CRP could sequester 0.45 Mg C ha^{-1} yr^{-1} to 60 cm over a 60 year period.

There are direct and indirect costs involved in retiring agricultural land and the CRP program has been successful in the United States because the federal government pays the landowner in 10 year contracts. Currently, there would be little incentive in Australia for farmers to voluntarily retire cultivated land. However, if an emissions trading market developed that included credits for SOC sequestration, large amounts of marginally productive land are potentially available for retirement. In an economic scenario analysis, Harper *et al.* (2007) estimated that large-scale destocking of rangeland and reforestation of cleared farmland in Western Australia would be economically viable to landowners when carbon prices reached AU$15 t CO$_{2\text{-eq}}$$^{-1}$.

5.2.6. Offsite additions

Offsite additions refer to a range of organic matter amendments, including manure, composts, biosolids from waste management and biochar produced from various feedstocks. The fact that these additions represent direct inputs of organic C to the soil system and that these materials are often in forms that are much more stable than fresh plant residues should, in the absence of additional constraints, translate into moderate to

high sequestration rates. Actual rates of SOC sequestration will be defined by the degradability of the material added, the amount of material added and the frequency of addition. With the exception of farmyard manure (discussed below), there is currently little comprehensive research on into the long-term C sequestration potential of many of these materials. Additionally, since most of these materials will be produced elsewhere, a full life-cycle analysis will be critical to determine the net greenhouse benefit of these management options.

Adding farmyard manure back to the fields has been a common agricultural practice for maintaining productivity for centuries. There have been several long-term agricultural experiments designed to examine the impacts of differing manure application rates (Blair *et al.* 2006a; 2006b; Gulde *et al.* 2008; Stewart *et al.* 2007). After 150 years of manure application (35 Mg ha^{-1} yr^{-1}) at the Broadbalk experiment in the UK, total C levels (top 10 cm) were 2.5 times greater than the non-manured cultivated control, but only one half the level of the non-cultivated pasture (Blair *et al.* 2006a). However, at another long-term experimental station in Germany with 15 yrs of extreme levels of manure application (200 Mg ha^{-1} yr^{-1}), total C levels were greater than those found in an adjacent non-cultivated grassland (Blair *et al.* 2006b). In both of these experiments, new labile C levels, defined as the difference between labile C in the manure and no-manure treatment, represented only 30% of the new total C levels indicating that, of the fraction of the manure that was retained in the soil, a significant fraction had been incorporated into more stable SOM fractions. It is also important to note here, that only 3 to 6% of the total C added as manure was retained in the soil (Blair *et al.* 2006a, 2006b), but 11 to 18% was retained in another trial on more productive soils in Lethbridge, Canada (Gulde *et al.* 2008).

The application of manure derived from Australian feedlots, which often contain high levels of sodium, could result in long-term detrimental effects to soil health by saturating the subsurface soil horizons with exchangeable sodium. If the application of these manure products is prolonged at any single site over time, the risk of subsurface soil constraints to plant growth may need to be weighed against productivity gains due to increased SOC and nutrient levels (Slattery *et al.* 2002).

Summarizing the evidence from these field trials, it appears that there is significant potential for sequestering C in soils through manure and other offsite organic applications, but that the proportion stabilised will likely depend on local climatic and soil edaphic conditions as well as the degradability of the material being added. In general, we would expect sequestration rates to be somewhat lower in Australia due to the generally greater decomposition rates in a climate that is much warmer than where most of these long-term trials have been conducted.

5.2.7. Alternative farming systems

So far in this section we have only considered management shifts within a narrow range of dominant traditional farming practices. There is evidence that large-scale changes in farming practices to organic and so-called "biological" farming systems which rely heavily on alternative soil amendments (Section 5.2.1.1) and manure applications can decrease net GHG emissions (Robertson *et al.* 2000; Stolze *et al.* 2000) and increase SOC stocks (Wells *et al.* 2000) while maintaining or increasing farm profitability (Lotter 2003) especially during extreme climatic years (Lotter *et al.* 2003). For example, Wells *et al.* (2000) found that in intensive vegetable farming, both organic and hybrid organic-best management practice systems significantly increased SOC levels over more traditional management systems. After 8 years SOC stocks to 15 cm were found to increase by 0.5 and 0.3 Mg C ha^{-1} yr^{-1} for organic and low-input crop rotation systems, respectively, in the central valley of California,

USA, while the conventional treatment showed no change over the same period (Clark *et al.* 1998).

At the Rodale Institute Farming Systems Trial in Pennsylvania, USA, 22 years of manure-based and legume-based organic farming system accumulated SOC at a rate 0.7 and 0.3 Mg C ha^{-1} yr^{-1} higher, respectively, than their conventional counterpart (Pimentel *et al.* 2005). However, a more detailed investigation of the SOM in these treatments indicated that the new SOC primarily accumulated in biologically active fractions (Wander and Traina 1996; Wander *et al.* 1994) that may have little permanence if the new inputs ceased.

Currently, most organic farming systems rely on multiple tillage operations to control weeds and incorporate manure and cover crop residues. Due to this tillage requirement, it is unclear whether organic farming systems will improve SOC levels more than conventional no-tillage systems. In one of the few studies that directly addressed this issue, Teasdale *et al.* (2007) found in a corn-soybean-wheat rotation that, despite depressed corn yields, the organic systems had increased SOC levels ~20% over 9 years when compared to the conventional no-tillage system.

Overall, anecdotal evidence is strong that these systems can capture and store more SOC than their traditional counterparts; however, with a few notable exceptions (i.e. Clark *et al.* 1998; Pimentel *et al.* 2005; Wells *et al.* 2000), detailed and replicated studies are currently lacking. Another potentially troubling, but also scientifically intriguing, issue is that there is no readily apparent mechanistic explanation for some of the very large SOC gains being anecdotally reported.

5.3. Summary of field trial evidence

From the published literature, it appears that globally most soils are responsive to various management changes that can increase SOC stocks. Sequestration rates generally range between 0.05 and 0.8 Mg C ha^{-1} yr^{-1}, with the greatest increases being noted for retirement and restoration of degraded agricultural lands and manure/bio-solid applications (Hutchinson *et al.* 2007; Ogle *et al.* 2005; Smith *et al.* 2008). Hutchinson *et al.* (2007) summarized sequestration potential for numerous management practices and found that on average improvements to cropland resulted in gains of 0.24 ± 0.20 Mg C ha^{-1} yr^{-1} (mean ± s.d.), while improvements to pasture lands resulted in increases of 0.44 ± 0.20 Mg C ha^{-1} yr^{-1}, and conversion from cropland to pasture gained 0.82 ± 0.28 Mg C ha^{-1} yr^{-1}. Further inspection of these results (i.e. Smith *et al.* 2008) indicates that nearly every management practice also has the potential to be a net source of carbon. While most studies have shown increased sequestration rates, a full 25% of the studies in reviews of tillage practices (West and Post 2002) and grassland management (Conant *et al.* 2001) had nil or negative effects for the treatment designed to increase SOC stocks. A number of these non-results may be due to poor experimental design, but it is also likely that local factors such as climate or soil properties exert an overriding control that may mask any management effect (e.g. Valzano *et al.* 2005).

Table 3. Summary of Australian field trial data (see Appendix 3 for full details). Soil carbon stock changes were calculated in two ways: 1) rate of change for improved management = slope of linear regression with time when time series data was available; and 2) relative change = (improved management – traditional management) / duration of trial.   For mean annual precipitation, clay content, sampling depth and duration, all of the trial data have been binned into 5 categories with the mean value of each bin given in "bin mean" column.

	Bin mean	Rate of Change[a] (Mg C ha^{-1} yr^{-1})				Relative Change[a] (Mg C ha^{-1} yr^{-1})			
		mean	s.e.	n	HSD[b]	mean	s.e.	n	HSD[b]
A. Trial treatment	Rotation	-0.10	0.05	19	a	0.20	0.04	28	a
	Stubble	-0.30	0.10	12	a	0.19	0.08	32	a
	Tillage	-0.26	0.09	17	a	0.34	0.06	34	a
B. State	NSW	0.00	0.07	26	a	0.20	0.04	46	a
	QLD	-0.16	0.08	12	a	0.13	0.05	23	a
	SA	-0.13	---	1	a	0.24	0.04	14	a
	VIC	---	---	0		0.02	0.03	2	a
	WA	-0.42	0.15	10	b	0.21	0.04	10	a
C. Mean annual precipitation (mm)	325	-0.49	0.09	9	b	0.24	0.05	9	a
	473	-0.27	0.08	3	ab	0.31	0.09	16	a
	552	-0.08	0.09	10	a	0.28	0.06	25	a
	666	-0.15	0.06	26	a	0.16	0.03	37	a
	2158	---	---	0		0.43	0.40	7	a
D. Clay content (%)	13	-0.36	0.09	8	b	0.39	0.08	23	a
	24	-0.21	0.09	16	bc	0.23	0.06	38	abc
	41	-0.80	0.06	4	d	0.38	0.18	11	ac
	50	-0.05	0.04	11	ac	0.05	0.06	12	b
	65	0.01	0.03	9	a	0.04	0.03	10	bc
E. Sampling depth (cm)	4.5	-0.17	0.16	4	ab	0.64	0.26	10	a
	10	-0.33	0.07	23	b	0.26	0.04	50	b
	15	-0.04	0.04	10	a	0.14	0.03	11	bc
	21	-0.11	0.12	9	ab	0.08	0.04	14	c
	44	-0.05	0.08	2	ab	0.11	0.02	9	bc
F. Trial duration (years)	4	-0.11	---	1	a	0.33	0.16	15	ab
	8	-0.43	0.07	21	b	0.35	0.06	40	a
	17	---	---	0		0.16	0.07	8	abc
	24	-0.05	0.04	18	a	0.14	0.02	18	bc
	42	0.01	0.01	8	a	0.05	0.02	13	c

[a] Rate of change and relative difference data have been normalized to a depth of 15 cm to avoid bias from thicker samples.

[b] For each summary category, different letters represent significant differences within a column (Tukey-Kramer honestly significant difference test ($\alpha = 0.05$))

Focusing on the Australian continent, results of field trials of standard best-management practices (see Appendix 3) have been less impressive than those from more temperate northern hemisphere locations. For example, the mean relative difference between paired till and no-till plots across all cropping systems in Australia was found to be 0.34 Mg C ha^{-1} yr^{-1} (Table 3), but this value was closer to 0.6 Mg C ha^{-1} yr^{-1} for a dataset (West and Post 2002) dominated by northern hemisphere trials.

A major conclusion that can be drawn from this compilation of Australian field trial data (Table 3) is that when SOC stocks were followed through time, even the improved management often showed significant declines, which, in many cases, was likely a direct result of these soils still responding to the initial cultivation of the native soil (e.g. Figure 7). However, since the traditional management practice often lost SOC at a greater rate, when only comparing the two treatments at the end of the trial there was a relative SOC gain in the improved management treatment. This means that, at least for the more traditional agronomic systems tested in these trials, Australian soils will generally only be mitigating losses and not actually sequestering additional atmospheric CO_2 (Sanderman and Baldock 2010).

A further examination of the Australian field trial data (Table 3) reveals that there are no significant differences amongst trial types or between states, with the exception that the Western Australia trials lost SOC at a greater rate than other states. Relative differences between treatments decrease with increasing sampling depth and trial duration (Figure 13) which is in accord with results from outside of Australia (Angers and Eriksen-Hamel 2008; Balesdent *et al.* 2000; Kern and Johnson 1993). Importantly, trends in the relative difference are due to a greater decrease in the rate of loss with depth and time for the traditional management practice than the improved management (Figure 13). This meta-analysis also found that the absolute rate of SOC loss was greatest at the driest locations (Table 3). Additionally, light-textured soils showed the greatest relative increases in SOC stocks (Table 3); however these results may be confounded by climatic differences in the regions where the different soil types are found.

In many of the studies that found both relative and absolute increases in SOC stocks with time, the majority of the new carbon was found to have accumulated in the light or particulate fraction (Chan 1997; 2001; Nyborg *et al.* 1998; Wander *et al.* 1994). For example, Chan (2001) found that 69 to 94% of changes in total SOC stocks occurred in the particulate fraction for several diverse management shifts. This is the SOC fraction that is most sensitive to perturbation and could be easily lost with only minor management or environmental change, thus the permanence of these sinks may be questionable.

5.4. Synoptic studies

There have been several large scale efforts to calculate the soil C sequestration potential for the agricultural sector at national (Bruce *et al.* 1999; Lal *et al.* 1999; Smith *et al.* 2000; Sperow *et al.* 2003) and global scales (Lal 2004b; Smith *et al.* 2008). These studies have generally taken one of two approaches: 1) scaled conservative sequestration estimates (Mg ha^{-1} yr^{-1}) by the aggregated acreage of each practice; or 2) used a spatially explicit inventory-based approach recommended by the IPCC. It is important to note here that most of these studies have relied on similar data sets of field trials to derive SOC sequestration estimates that are generally biased towards temperate northern hemisphere locations. Additionally, the estimated sequestration rates and stock change factors in the IPCC accounting approach (Penman *et al.* 2003) have generally been derived from single point-in-time measurements, not from time series data. See introduction to Section 5.2 for the potential implications of this type of estimation.

Figure 13. Rate of change, measured as both the relative difference (diamonds) between treatments and as the actual change through time (circles), in soil C stocks plotted as a function of (A) sampling depth and (B) trial duration. The relative difference between treatments decreased as (A) sampling depth and (B) trial duration increased across all studies in Appendix 3. The decrease in the relative difference was due to a larger shift in the rate of change of the traditional management with depth and time than in the actual stock change of the improved management. Error bars = 1 s.e. (see Table 3 for statistical summary).
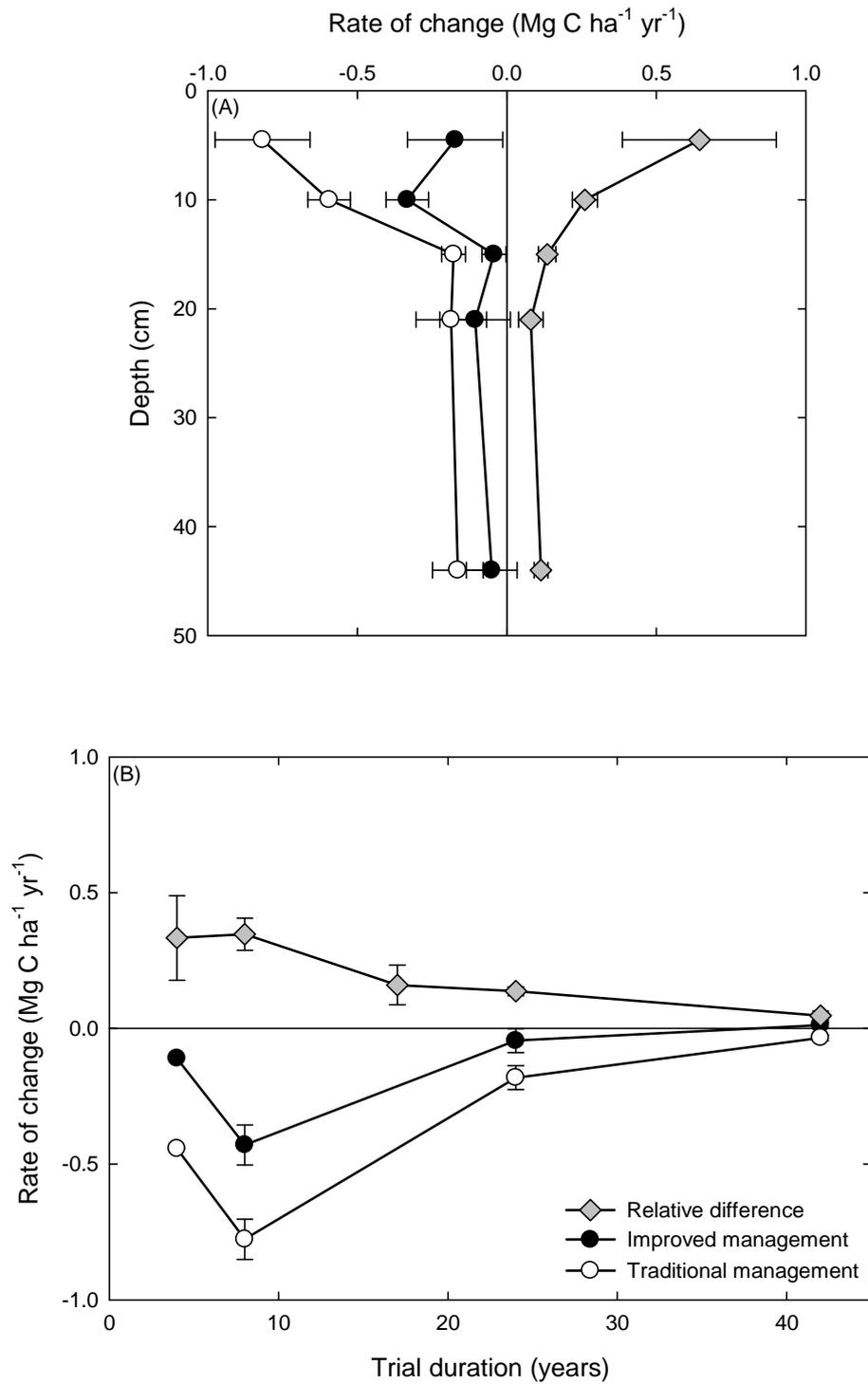
Globally, Lal (2004a) estimated that 78 Pg of carbon have been lost due to agricultural practices, with 26 Pg attributed to erosion and 52 Pg attributed to mineralization. If best-management practices, including reduced or no-till farming, cover crops, nutrient management, manuring, efficient irrigation, and setting aside marginal lands, were adopted on all agricultural land, Lal (2004a) estimates that 50-66% of historic losses could be sequestered over the next 50 years. Globally, this equates to 30 to 60 Pg C at a rate of 0.9 ± 0.3 Pg C yr^{-1} over 25 to 50 years. This sequestration rate is equivalent to approximately 10% of current total GHG emissions due to fossil fuel burning and land-use change (IPCC 2007) or 25 to 33% of the annual increase in atmospheric CO_2 levels. Using a more detailed inventory approach, Smith *et al.* (2008) estimated that the global technical mitigation potential, considering all greenhouse gases, from agriculture by 2030 is 5.5 to 6.0 Pg CO_{2-eq} yr^{-1} (or ~ 1.5 Pg C yr^{-1}). It is extremely important to keep in mind that these types of estimates assume adoption of best management practices across every acre of land under production.

Extracting data for Australia from the analysis of Smith *et al.* (2008), it is found that the sequestration potential, including avoided emissions, for Australia's agricultural land is approximately 100 Tg CO_{2-eq} yr^{-1} for the land area considered in this study, which would equate to nearly 15% of Australia's current annual emissions. Given the differences that we found in average sequestration rates between Australian specific data (Table 3) and the global data sets, this value is likely an overestimation of the actual potential. Additionally, this broad analysis does not consider the impact of drought years that are known to have severe impacts on productivity which should translate into SOC losses for the drought period.

5.5. Future uncertainty in light of climate change and economic factors

Climatic uncertainty. In the absence of active management, most soils are believed to become net sources of CO_2 to atmosphere primarily due to rising temperatures (Cox *et al.* 2000; Torn and Harte 2006). The degree to which SOC is lost with rising temperatures will depend on a host of factors (reviewed in section 3) and likely vary with agricultural management, but, at a minimum, this will add additional uncertainty into projected soil carbon storage potentials and in the permanence of sequestered soil C. Predicted shifts in regional precipitation patterns (Chiew *et al.* 1995; Groisman *et al.* 1999), while less certain than projected temperature changes, will create even more uncertainty in predicting sequestration rates out into the future.

In Australia's vast rangelands, the combined effect of changes in temperature, precipitation and CO_2 levels will likely drive shifts in the vegetative community with some regions becoming drier and supporting less productive ecosystems and other regions becoming wetter and perhaps supporting more productive ecosystems (McAlpine *et al.* 2009; McKeon *et al.* 2009; Smith *et al.* 2007). Any shift in plant productivity will feedback directly into the soil carbon balance.

Figure 14. Constraints on agricultural carbon sequestration potential (adapted from Smith et al., 2005). The maximum feasible sequestration potential at any given location or set of soil properties will likely never be realized due to a series of biological, physical, economic, social, and political constraints.
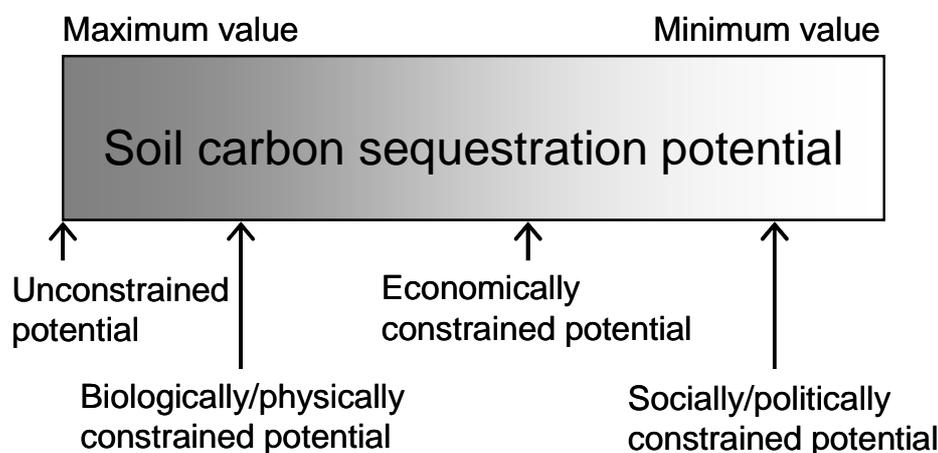
Economic uncertainty. At its heart, agriculture is a free market enterprise and if a farmer decides that the current management practices are not maximizing profits or commodity prices significantly change, then s/he may very well change management which may have significant deleterious effects on newly stored carbon.

In section 5.4, the economic and socio-political feasibility of adoption of proposed management changes was not considered (Figure 14). When economic feasibility was addressed (i.e. Smith *et al.*, 2008), the global sequestration potential dropped to 0.4 Pg C yr^{-1} for carbon prices up to 20 US$ Mg CO_{2-eq}^{-1} but only to 1.1 Pg C yr^{-1} for carbon prices up to 100 US$ Mg CO_{2-eq}^{-1}. Even this economically-constrained potential may not be achieved due to other socio-political constraints (Smith *et al.* 2005; Smith *et al.* 2007). Smith *et al.* (2005) demonstrated that actual sequestration was small or negative for the period 1990 to 2000 for the European Union and that this trend should continue through 2010. These authors concluded that "without incentives for carbon sequestration in the future, cropland carbon sequestration under Article 3.4 of the Kyoto Protocol will not be an option."

The emerging biofuels industry may exert additional pressures on soil C sequestration. Large scale stubble removal for cellulosic ethanol may likely have negative impacts on SOC and soil fertility levels (Blanco-Canqui and Lal 2009). However, conversion from relatively low-yielding cereal crops to high productivity perennial grasses, such as Miscanthus and switchgrass, may greatly enhance SOC sequestration rates (Hansen *et al.* 2004), provided enough residue remains after harvest.

In this review, we have primarily focused on the biophysical potential of soils to sequester atmospheric CO_2 but, as is evident from this brief discussion of climatic and economic uncertainties, a whole host of additional factors (Figure 14) need to be considered in order to make an educated estimation of realistic soil carbon sequestration potentials.

6. Conclusions

From a theoretical standpoint, any management practice which results in greater C return to the soil, increased stabilisation of soil C, or a reduction in losses should lead to positive SOC sequestration rates. Indeed, a majority of the field trial evidence summarized in this review indicated that there was a relative gain in SOC between conventional and improved-management treatments (Table 3). However, when SOC stocks were followed through time, the majority of studies indicated that there was an actual decrease in the quantity of C stored in the soil. These seemingly contradictory results suggest that much of Australia's agricultural soils may still be responding to initial land clearing and that many management improvements are just slowing the rate of loss SOC (i.e. Figure 7). While this reduction in rate of loss of SOC represents a real GHG abatement in the form of avoided emissions, it may be extremely difficult to project these findings out into the future where the soil C condition is unknown (Sanderman and Baldock 2010). Further complicating predictive efforts is the fact that many of these results may be a heavily influenced by the local climatic and edaphic conditions.

Additionally, the majority of this evidence comes from field trials that were designed to test best management practices in terms of typical agronomic properties (i.e. crop yields, nutrient- or water-use efficiency, etc…) not soil carbon levels. It just so happens that many of these practices have indirect benefits to SOC stocks. Relatively minor shifts in production practices will, in most cases, have only minor impacts on SOC levels. However, large scale shifts from, for example, a fallow crop system to a permanent pasture where significantly more annual production is returned to the soil will result in greater SOC gains. Retirement of marginal agricultural land will likely have the greatest positive benefit to SOC stocks because nearly all of the annual NPP will be returned to the soil. Large, but not unlimited (see Section 3.4.2), SOC gains will also be likely in systems where significant quantities of organic residues (manure, compost, biosolids, biochar, etc…) are added annually. In Table 4, we have produced qualitative estimates of the biophysical sequestration potential for the various management options discussed in this review.

Nearly 90% of Australia's agricultural land is devoted to low-to-medium intensity grazing of natural vegetation (Table 1). These lands are generally comprised of soil and/or climate conditions that are not suitable for more intensive agricultural practices and given these constraints are not likely to be able to store large quantities of SOC. Although given the areal extent of these grazing lands, even a small relative improvement (for example, 0.04 Mg C ha^{-1} yr^{-1}) across this acreage would translate into large net sequestration rates (60 Tg CO$_{2\text{-eq}}$ yr^{-1} or 10% of 2007 emissions).

Accurate monitoring and verification of soil C stock changes, due to the large and heterogeneous background levels are difficult and often prohibitively expensive (see Section 4). A large-scale monitoring and verification system for estimating SOC stock changes will depend on the level of stringency that a particular government or emissions trading scheme finds acceptable and this level may likely be based on the financial trade-off between the value of the C credits and the cost of the monitoring program (Smith 2004b). At the national scale, this system may take the form of robust modelling informed by detailed measurements in representative systems combined with verification of management practices and yields via reporting and remote sensing with some economic discounting to factor in verification uncertainty.

Table 4. Summary of major management options for sequestering carbon in agricultural soils.

Management	SOC benefit[a]	Conf.[b]	Justification
1. Shifts within an existing cropping/mixed system			
a. Maximizing efficiencies - 1) water-use 2) nutrient-use	0/+	L	Yield and efficiency increases do not necessarily translate to increased C return to soil
b. Increased productivity - 1) irrigation 2) fertilization	0/+	L	Potential trade-off between increased C return to soil and increased decomposition rates
c. Stubble management – 1) Elimination of burning and grazing	+	M	Greater C return to the soil should increase SOC stocks
d. Tillage – 1) Reduced tillage 2) Direct drilling	0 0/+	M M	1) Reduced till has shown little SOC benefit; 2) Direct drill reduces erosion and destruction of soil structure thus slowing decomposition rates; however, surface residues decompose with only minor contribution to SOC pool
e. Rotation – 1) Eliminate fallow with cover crop 2) Inc. proportion of pasture to crops 3) Pasture cropping	+ +/++ ++	M H M	1) Losses continue during fallow without any new C inputs – cover crops mitigate this; 2) Pastures generally return more C to soil than crops; 3) Pasture cropping increases C return with the benefits of perennial grasses (listed below) but studies lacking
f. Organic matter and other offsite additions	++/+++	H	Direct input of C, often in a more stable form, into the soil; additional stimulation of plant productivity (see above)
2. Shifts within an existing pastoral system			
a. Increased productivity - 1) irrigation 2) fertilization	0/+	L	Potential trade-off between increased C return to soil and increased decomposition rates
b. Rotational grazing	+	L	Increased productivity, inc. root turnover and incorporation of residues by trampling but lacking field evidence
c. Shift to perennial species	++	M	Plants can utilize water throughout year, increased belowground allocation but few studies to date
3. Shift to different system			
a. Conventional to organic farming system	0/+/++	L	Likely highly variable depending on the specifics of the organic system (i.e. manuring, cover crops, etc…)
b. Cropping to pasture system	+/++	M	Generally greater C return to soil in pasture systems; will likely depend greatly upon the specifics of the switch
c. Retirement of land and restoration of degraded land	++ +++	H	Annual production, minus natural loss, is now returned to soil; active management to replant native species often results in large C gains

[a] Qualitative assessment of the SOC sequestration potential of a given management practice (0 = nil, + = low, ++ = moderate, +++ = high)

[b] Qualitative assessment of the confidence in this estimate of sequestration potential based on both theoretical and evidentiary lines (L = low, M = medium, H = high)

If the agricultural sector is to be issued credits for sequestering C in the soil as SOC, then a full accounting of all GHG emissions should be required. Some of the recommended management changes will likely greatly reduce an individual farm's GHG burden (Smith 2004a; Smith *et al.* 2008). For example, a management shift from conventional preparation to direct drilling with stubble retention will greatly reduce fossil fuel usage during tillage operations (West and Marland 2002). However, other changes, such as fertilizer and manure application will likely increase other GHG emissions (Schlesinger 2009). A full life-cycle analysis will be needed for management practices that involve significant use of offsite additions.

Many of the management options discussed in this review tend to increase overall profitability within existing agricultural systems and as such are being rapidly adopted in various regions of Australia. However, numerous other management shifts (for example, converting from annual crops to pastures) which may have the greatest positive impact on SOC stocks will likely need incentives before wide-scale adoption is seen (Smith *et al.* 2007). As a society, we will have to assess whether or not it is acceptable to decrease production of certain crops because they are not profitable from a C trading point of view.

Overall, this review suggests that stemming the loss of SOC from current agricultural practices and at a minimum recapturing some fraction of the carbon lost from soils since initial land clearing is possible from a biophysical perspective. However, due to the complex web of factors that governs the C balance of any particular soil, quantitative predictions of SOC sequestration rates will likely always entail a large degree of uncertainty. Given that many mitigation options in the agricultural sector have numerous co-benefits in terms of food security, environmental sustainability and farm profitability, we believe that governmental policies that promote adoption of these best management practices should be pursued regardless of the final status of agricultural soils in any carbon pollution reduction scheme.

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Appendix I. Modelling NPP in perennial agroecosystems.

The approach for estimating continental NPP for the National Land and Water Resources Audit (Raupach *et al.* 2001a; 2001b) was described in section 3.1. Roxburgh *et al.* (2004) compared a range of model estimates of NPP for the Australian continent. Individual estimates of NPP, especially at the continental scale (i.e. Raupach *et al.* 2001a; Roxburgh *et al.* 2004), necessitate a range of assumptions, are better suited to certain ecosystems and are reliant on input data which may have considerable uncertainty. This was exemplified by the five fold range in continental NPP estimates in the Roxburgh *et al.* (2004) overview. To provide more specific estimates of NPP in agricultural systems as well as to explore the importance of rainfall distribution and vegetation type in maximising carbon capture, estimates of NPP under different cropping scenarios were obtained using the APSIM model (Keating *et al.* 2003). APSIM was developed to simulate biophysical processes in agricultural systems. It provides a more mechanistic approach to estimating plant growth by detailed modelling of plant phenology, water balance, nutrient availability and management controls. Here we use APSIM (version 7.0) to model NPP under annual and perennial vegetation in agricultural systems using climate data representative of winter dominant rainfall, equiseasonal rainfall and summer dominant rainfall regions.

Methods.

Patched point climate files downloaded from SILO (Jeffrey *et al.* 2001) for Mintaro SA (winter dominant rainfall), Parkes NSW (equiseasonal rainfall) and Goondiwindi QLD (summer dominant rainfall) and generic APSIM soil files for sand (plant available water capacity (PAWC) 97.5mm to 2.5m depth) and clay (PAWC 134.7 mm to 2.4 m) were used. Two soil fertility scenarios were examined: 1) 120 kg urea-N yr^{-1} to alleviate any potential nutrient limitations on NPP; and 2) simulations without N resets or fertiliser as the nutrient-limited case. Simulations were run continuously from 1975 to 2005 with annual surface organic matter resets on the 1st of March. Initial water was set to zero and was not reset thereafter.

Table A1.1. Sites included in modelling study

Site	Latitude	Longitude	Rainfall distribution	Average annual rainfall (1975-2005)	NLWRA baseline NPP (Mg C ha^{-1} yr^{-1})	NLWRA NPP (Mg C ha^{-1} yr^{-1}) with agricultural inputs
Mintaro	-33.91	138.72	Winter dominant	564	3.06	3.77
Parkes	-33.14	148.16	Equiseasonal	630	2.49	3.16
Goondiwindi	-28.52	150.33	Summer dominant	642	2.34	2.58

Vegetation types were wheat-fallow representing an annual system, lucerne representing a leguminous perennial system, bambatsi representing a pure C4 perennial system and wheat-bambatsi intercropping representing a mixed annual-perennial system. Existing APSIM plant modules were used (available online at: http://www.apsim.info/apsim/releases/Apsim.asp). As the only annual species, wheat (Wyalkatchem) was sown on the 1st of May each year. Sowing density was 100 plants per m^2, depth of 50 mm and row spacing of 180 mm and wheat was harvested when ripe. Trifecta lucerne was sown on the 1st of April in 1975. Lucerne was sown at 25 mm depth and a plant density of 60 per m^2. Lucerne was cut at flowering to a height of 40 mm, removing 60% of the biomass. *Panicum coloratum* cv bambatsi was established on the 1st of August, 1975, at 40,000 plants per hectare. Bambatsi productivity was maximized by cutting, maintaining the grass in its actively growing phase, when above ground biomass

reached 3 tonnes and 90% of the biomass was removed. The bambatsi-wheat intercropping was managed as for bambatsi alone and wheat alone.

Simulations were set up as a 3×4×2×2 factorial of climate (winter dominant, equiseasonal, summer dominant), vegetation (bambatsi, bambatsi-wheat, wheat, lucerne), soil (sand, clay) and N fertility (120 kg fertiliser, no fertiliser or reset). Outputs from APSIM were cumulative daily root and shoot dry matter growth. To convert from dry matter to carbon, a carbon content of 45% was assumed for all plant tissues. Only 25 years of simulations from 1980 to 2004 were used for calculation of NPP, which is reported as average of the two soil types, to avoid anomalies due to establishment of the perennials in the first few years of simulation.

Table A1.2. Results from APSIM simulations. Annual NPP given as mean (all values are averaged across the two soil types) with 1 s.d. in parentheses.

Climate	Vegetation	Nitrogen fertilization	NPP (Mg C ha^{-1} yr^{-1})	% C to roots
Summer dominant	C4 perennial	120	12.5 (3.8)	33
		0	5.8 (0.6)	39
	Mixed system	120	10.5 (3.7)	32
		0	5.6 (1.2)	38
	Lucerne	120	5.3 (0.7)	27
		0	5.3 (0.6)	28
	Wheat	120	3.9 (1.3)	17
		0	2.5 (1.9)	16
	BiosEquil		2.6	
Equiseasonal	C4 perennial	120	11.0 (2.9)	31
		0	4.8 (0.7)	38
	Mixed system	120	10.4 (2.9)	27
		0	4.9 (0.8)	37
	Lucerne	120	5.6 (0.5)	27
		0	5.6 (0.5)	27
	Wheat	120	4.3 (0.9)	14
		0	2.3 (1.8)	15
	BiosEquil		3.2	
Winter dominant	C4 perennial	120	8.7 (2.2)	29
		0	3.7 (0.5)	38
	Mixed system	120	9.7 (1.6)	23
		0	3.7 (0.8)	37
	Lucerne	120	5.9 (0.7)	27
		0	5.9 (0.7)	27
	Wheat	120	2.0 (0.8)	13
		0	3.3 (1.7)	14
	BiosEquil		3.8	

Results.

Climate, vegetation type, N fertility and soil type (not presented) all affected the NPP estimates in APSIM. The average annual NPP estimates for the annual system were comparable to NPP for current agricultural inputs estimated by BiosEquil and presented in the NLWRA NPP map (see Figure 2). Where alternative systems including perennials were

simulated and nutrient limitations were alleviated by fertilisation, NPP simulated by APSIM was significantly greater than that simulated by BiosEquil. This modelling exercise suggests that the potential exists for perennial species to assimilate more carbon than an annual crop-fallow system.

Bambatsi assimilated the most carbon under all climate regimes where nitrogen was not limiting. The relative increase in NPP of the perennial C4 grass to the annual wheat with summer fallow was greatest under the summer dominant rainfall climate regime followed by the equiseasonal rainfall and had the least improvement under the winter dominant rainfall climate. The increases were greater with higher soil fertility, and may be contributed to by the ability of bambatsi to access nutrients and water at depth. As a N-fixing perennial, lucerne was not influenced by limiting amounts of nitrogen. Lucerne assimilated more carbon than wheat under all climate regimes, and more than bambatsi under the winter dominant rainfall climate with limiting nitrogen.

While vegetation and N fertility were important drivers, NPP fluctuated from year to year under all fertilisation and vegetation types reflecting the importance of rainfall as a major driver of NPP. Soil type was important in some circumstances due to the different ability to store and supply water and nitrogen.

In addition to annual NPP being higher under the perennial scenarios, a greater fraction of that fixed C was allocated belowground, thus reducing the proportion of assimilated carbon that could be lost to herbivory, fire, erosion and decomposition on the soil surface. Approximately 35% of plant dry matter (and carbon if assuming a constant %C for the entire plant) was allocated to roots in bambatsi, compared with 27% for lucerne and 14% for wheat. Consistent with ecophysiological theory (Derner *et al.* 2006), the allocation of C to roots in bambatsi increased in the low fertility scenario.

Comparing NPP on a monthly basis demonstrated the importance of matching rainfall with plant phenology to maximise NPP (Figure A1.1). The mixed annual-perennial system allowed productivity to continue into the summer months when an annual system would be in fallow. The wheat-bambatsi system was predicted to have lower productivity than wheat alone over the winter months in the summer dominant and to a lesser extent the equiseasonal rainfall climate, probably due less stored water that would otherwise exist in a wheat-fallow system. However, these modelling results suggest that there is potential to greatly increase annual NPP, and carbon inputs to the soil system, by incorporating perennials under all three climate regimes.

Figure A1.1. Simulation of mean monthly NPP for winter wheat (grey bars) and mixed wheat-bambatsi system (stacked bars with bambatsi in black and wheat in white) with non-limiting source of nutrients (120 kg N fertilizer yr^{-1}) for 3 locations with differing rainfall distributions. Monthly distributions of mean rainfall are shown in inserts.
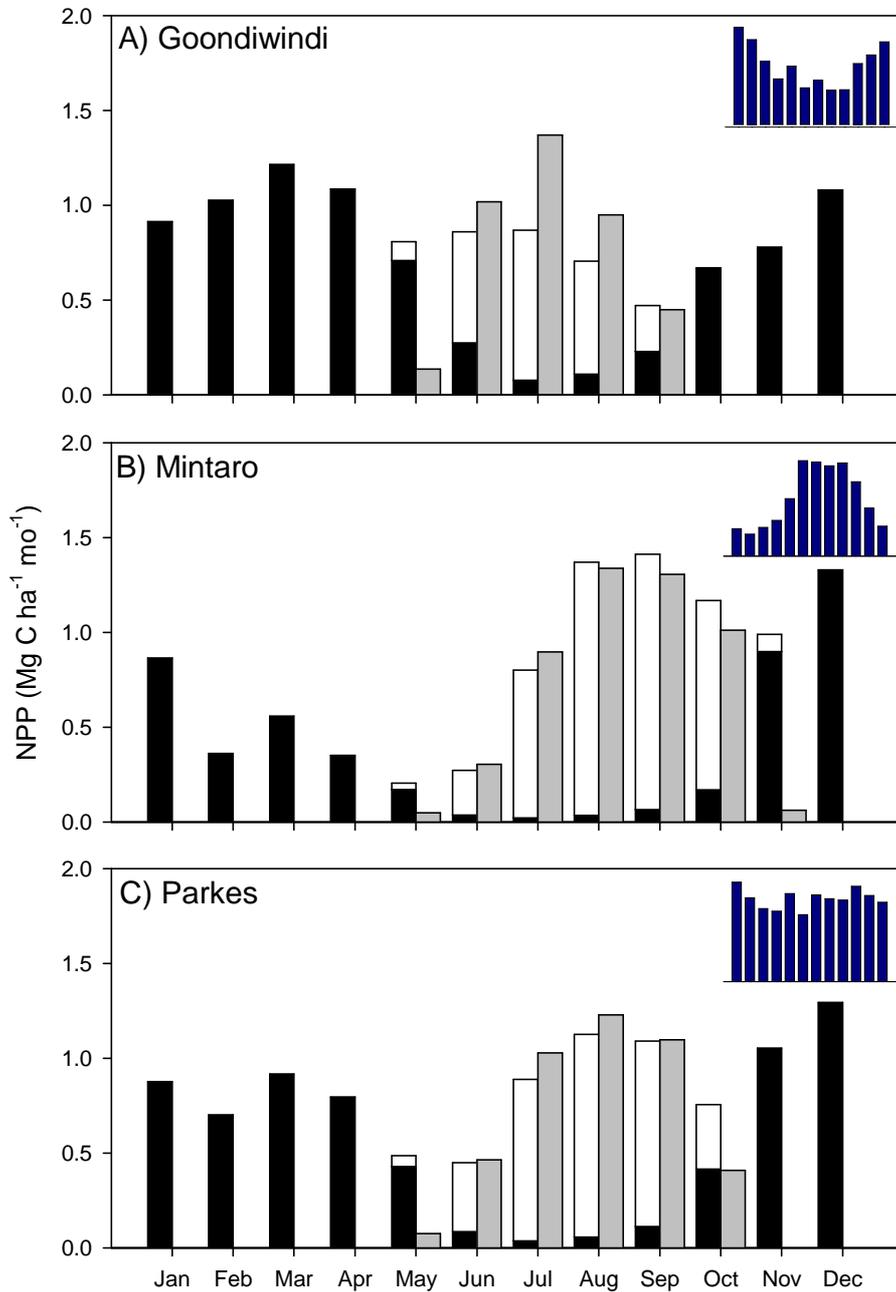
Appendix II. Are estimates of glomalin stocks reasonable?

Ultimately, all glomalin-C found in soil is derived from plant photosynthates. Here, we have taken two simple approaches to examine these stock estimates using the following information: 1) Annual NPP allocation to AMF for most agroecosystems ranges from 4 to 20% (Graham 2000; Tinker *et al.* 1994); 2) Glomalin enters the soil when AMF hyphae die and decompose (Driver *et al.* 2005; Treseder and Allen 2000); 3) Hyphal standing stocks have been estimated to be on the order of 5 to 90 g C m^{-2} (Zhu and Miller 2003) with turnover times of days to months (Staddon *et al.* 2003; Zhu and Miller 2003); 4) Glomalin constitutes 0.4 to 6% of AMF hyphae biomass (Lovelock *et al.* 2004; Wright and Upadhyaya 1996) with a C content of approximately 35% (Nichols and Wright 2005); and 5) The mean residence time of glomalin in the soil system is 6 to 42 yrs (Rillig *et al.* 2001).

In a top-down approach, we have estimated the size of a steady-state glomalin-C stock for a hypothetical soil, containing 60 Mg C ha^{-1} in the upper 30cm with annual NPP of 10 Mg C ha^{-1} yr^{-1}, as follows:

Glomalin stock = (NPP allocation) × (glomalin in hyphae) × (glomalin MRT in soil)

We have eliminated concerns about variations in hyphal stocks and turnover rates by invoking steady-state conditions in this calculation. Additionally, we have assumed that the C content of glomalin and the fungal hyphae are similar. Given the large range in input parameters, we simulated glomalin stocks (% of total soil C stock) for 36 combinations of *glomalin in hyphae* and *glomalin MRT* for differing NPP allocations (Table A2.1). These calculations indicate that even at 20% of annual NPP allocation to AMF, glomalin-C would only average 2.2 ± 2.2% of total SOC stocks.

Table A2.1. Glomalin stock estimates (% of total soil C stock) based on differing levels of NPP allocation to AM fungi.

	% of annual NPP allocation					
	4%	8%	12%	16%	20%	50%
Mean ± 1 sd	0.5±0.4	0.9±0.9	1.3±1.3	1.8±1.7	2.2±2.2	5.6±5.4
Median	0.31	0.61	0.92	1.23	1.53	3.83
Range	0.01-1.6	0.03-3.2	0.04-4.8	0.05-6.4	0.07-8.0	0.2-20

In the bottom-up approach, we have estimated the percent of annual NPP allocation for the same hypothetical soil using the same set of assumptions as:

NPP allocation = (Glomalin stock) ÷ (glomalin MRT in soil) ÷ (glomalin in hyphae)

Here, we simulated NPP allocation for the same combinations of variables as earlier for a range of glomalin stock sizes and found that as the proportion of glomalin-C of total SOC increases beyond 1%, the annual NPP allocation becomes unreasonably high (Table A2.2).

Table A2.2. NPP allocation (% of annual NPP) to AM fungi needed to produce given glomalin-C stocks.

	Glomalin-C stock (% of SOC)					
	0.1%	0.5%	1%	4%	8%	27%
Mean ± 1 sd	3.7±6.4	18±32	37±64	150±260	290±510	990±1700
Median	1.3	6.5	13	52	100	350
Range	0.3-30	1.3-150	3-300	10-1200	20-2400	68-8100

Given the results of this simple exercise, glomalin-C stocks much over 1-2% of the total soil C pool seem highly unlikely, unless much more NPP is allocated belowground to AM fungi than is currently understood.

Appendix III. Australian field trial results.

Changes in soil organic carbon stocks reported from Australian field trials. In trials with multiple treatments, each effect was considered independently whenever possible. The rate of change (Mg C ha^{-1} yr^{-1}) is the slope of the linear regression of SOC for the improved management, in terms of SOC sequestration potential (listed first in relevant comparison column), versus years calculated for trials with time series data. The relative change is the SOC difference between the improved and traditional management treatments divided by number of years under trial (Mg C ha^{-1} yr^{-1}). For studies that did not report bulk density (listed in footnote A), a fixed value of 1.4 Mg m^{-3} was assumed. Multiple publications existed for many of the trials. In these cases, the most recent publication or the publication with time series data was used.

1. ROTATION

Site	State	Ref[a]	MAP	Crop rotation[b]	Tillage[c]	Stubble[d]	N Fert[e]	clay %	Depth (cm)	# of years	Rate of change[f]		Relative change[f]	
Auscott-Warren	NSW	16	479	CoCo v CoF	MT	SR	NF	52	60	4	-0.42	n.s.	0.00	n.s.
McMaster	NSW	28	642	LW v WF	CT	SR	NF	25	5	7	0.10	n.s.	0.39	**
McMaster	NSW	28	642	CW v WF	CT	SR	NF	25	5	7	-0.09	n.s.	0.10	**
McMaster	NSW	28	642	MW v WF	CT	SR	NF	25	5	7	-0.11	n.s.	0.10	**
Merredin	WA	19	312	WW v WF	CT	SI	NFm	20	10	7	-0.20	n.s.	0.35	**
Merredin	WA	19	312	WW v WF	CT	SB	NFm	20	10	7	-0.37	***	0.22	*
Narrabri	NSW	17	616	CoCo v CoWF	MT	SR	NF	60	60	8			0.52	n.s.
Tamworth (BE)	NSW	15	671	WC v WF	CT	SG	NFm	54	15	27	0.10	**	0.16	**
Tamworth (BE)	NSW	15	671	WL v WF	CT	SG	NFm	54	15	27	0.08	**	0.14	**
Tamworth (BE)	NSW	15	671	WW v WF	CT	SG	NFm	54	15	27	0.00	n.s.	0.06	n.s.
Tamworth (BE)	NSW	15	671	WC v WW	CT	SG	NFm	54	15	27	0.10	**	0.10	**
Tamworth (BE)	NSW	15	671	WL v WW	CT	SG	NFm	54	15	27	0.08	**	0.09	**
Tamworth (RC)	NSW	15	671	WC v WF	CT	SG	NFm	46	15	27	-0.11	**	0.18	**
Tamworth (RC)	NSW	15	671	WL v WF	CT	SG	NFm	46	15	27	-0.17	**	0.12	**
Tamworth (RC)	NSW	15	671	WW v WF	CT	SG	NFm	46	15	27	-0.23	**	0.07	n.s.
Tamworth (RC)	NSW	15	671	WC v WW	CT	SG	NFm	46	15	27	-0.11	**	0.11	**
Tamworth (RC)	NSW	15	671	WL v WW	CT	SG	NFm	46	15	27	-0.17	**	0.05	n.s.
Tarlee	SA	18	500	WP v WW	CT	SB	NFm	14	10	10			0.15	***

Site	State	Ref	MAP	Crop rotation	Tillage	Stubble	N Fert	clay %	Depth (cm)	# of years	Rate of change		Relative difference	
Tarlee	SA	18	500	WP v WW	CT	SI	NFm	14	10	10			0.20	***
Tarlee	SA	18	500	WP v WW	RT	SR	NFm	14	10	10			0.33	***
Wagga Wagga	NSW	14	550	LW v WW	CT	SB	N0	27	10	21	-0.28	***	0.05	
Wagga Wagga	NSW	14	550	SW v LW	CT	SR	N0	27	10	21	0.00	n.s.	0.05	
Wagga Wagga	NSW	14	550	SW v LW	DD	SR	N0	27	10	21	0.18	***	0.07	
Waite	SA	1	626	WW v WF	CT	SG or SRm	N0	18	40	72			0.19	n.s.
Waite	SA	1	626	WC v WF	CT	SG or SRm	N0	18	40	72			0.26	*
Waite	SA	1	626	PPFW v WF	CT	SG or SRm	N0	18	40	47			0.30	**
Waite	SA	1	626	PPPP v WF	CT	SG or SRm	N0	18	40	47			0.51	***
Waite	SA	1	626	PPPP v PPFW	CT	SG or SRm	N0	18	40	47			0.21	**
										mean (different from 0[9])	**-0.09**	**	**0.18**	***
										s.d.	0.17		0.14	

2. STUBBLE

Site	State	Ref	MAP	Crop rotation	Tillage	Stubble	N Fert	clay %	Depth (cm)	# of years	Rate of change		Relative difference	
Ayr	QLD	3	950	Sc		GCTB v SB	N?	40	25	7			-0.05	n.s.
Biloela	QLD	22	698	SgSg	CT	SR v SRm	N0	41	10	6			0.21	**
Biloela	QLD	22	698	SgSg	DD	SR v SRm	N0	41	10	6			0.23	**
Biloela	QLD	24	698	SgSg	CT	SR v SRm	N0	41	10	7	-0.63	**	0.06	n.s.
Biloela	QLD	24	698	SgSg	DD	SR v SRm	N0	41	10	7	-0.46	***	0.10	n.s.
Condobolin	NSW	11	430	WW	DD	SR v SRm	NFm	25	10	14			0.03	n.s.
Condobolin	NSW	11	430	WW	CT	SR v SRm	NFm	25	10	14			0.01	n.s.
Harwood	NSW	21	1021	Sc		GCTB v SB	NF	48	5	2			-0.18	n.s.
Hermitage	QLD	27	685	WW	CT	SR v SB	N0	65	20	33	0.03	n.s.	0.00	n.s.
Hermitage	QLD	27	685	WW	CT	SR v SB	NF	65	20	33	-0.01	n.s.	-0.03	n.s.
Hermitage	QLD	27	685	WW	DD	SR v SB	N0	65	20	33	-0.01	n.s.	0.01	n.s.
Hermitage	QLD	27	685	WW	DD	SR v SB	NF	65	20	33	0.07	**	0.03	n.s.
Mackay	QLD	2	1668	Sc	CT/DD	GCTB v SB	NF	18	10	5			-0.56	**
Mackay	QLD	21	1668	Sc		GCTB v SB	NF	18	5	5			0.42	**
Merredin	WA	19	312	WW	CT	SI v SB	NFm	7	10	7	-0.20	n.s.	0.14	n.s.
Merredin	WA	19	312	WF	CT	SI v SB	NFm	7	10	7	-0.52	**	0.02	n.s.
Nabawa	WA	19	457	WW	CT	SI v SB	NFm	10	10	10	-0.19	**	0.13	**

Site	State	Ref	MAP	Crop rotation	Tillage	Stubble	N Fert	clay %	Depth (cm)	# of years	Rate of change		Relative difference	
Narrabri	NSW	10	616	CoCo	T?	SI v SB	NFm	60	30	3	1.47	n.s.	0.90	*
Rutherglen	VIC	5	593	WW	DD	SR v SB	Ndp	22	2.5	10			-0.01	n.s.
Tarlee	SA	18	500	WW	CT	SI v SB	NFm	14	10	10			0.14	**
Tarlee	SA	18	500	WP	CT	SI v SB	NFm	14	10	10			0.20	**
Tarlee	SA	18	500	WW	CT	SR v SB	NFm	14	10	10			0.06	**
Tarlee	SA	18	500	WP	CT	SR v SB	NFm	14	10	10			0.24	**
Temora	NSW	7	535	LW	DD	SR v SB	N0	15	10	5			-0.14	n.s.
Temora	NSW	7	535	LW	CT	SR v SB	N0	15	10	5			-0.08	n.s.
Tully	QLD	3	4067	Sc		GCTB v SB	N?	39	25	4			0.21	n.s.
Tully	QLD	21	4067	Sc		GCTB v SB	NF	39	5	6			0.70	**
Wagga Wagga	NSW	7	550	LW	DD	SR v SB	N0	27	10	19			0.12	**
Wagga Wagga	NSW	7	550	LW	CT	SR v SB	N0	27	10	19			0.07	**
Wagga Wagga	NSW	9	550	LW	CT/DD	SR v SB	NF	27	20	10			0.51	**
Wagga Wagga	NSW	14	550	LW	DD	SR v SB	N0	27	10	21	0.00	n.s.	0.13	
Wagga Wagga	NSW	14	550	LW	CT	SR v SB	N0	27	10	21	-0.20	***	0.19	
Wongan Hills	WA	19	347	WW	CT	SI v SB	NFm	10	10	6	-0.20	n.s.	0.48	**
							mean (different from 0^9)				**-0.07**	n.s.	**0.13**	**
							s.d.				0.51		0.26	

3. TILLAGE

Site	State	Ref	MAP	Crop rotation	Tillage	Stubble	N Fert	clay %	Depth (cm)	# of years	Rate of change		Relative difference	
Avondale	WA	29	389	WW	DD v CT	SG...SB	NFm	16	10	9	-0.26	**	0.19	
Biloela	QLD	22	698	SgSg	DD v CT	SR	N0	41	10	6			0.21	**
Biloela	QLD	22	698	SgSg	DD v CT	SRm	N0	41	10	6			0.19	**
Biloela	QLD	24	698	SgSg	DD v CT	SR	N0	41	10	7	-0.46	**	0.17	n.s.
Biloela	QLD	24	698	SgSg	DD v CT	SRm	N0	41	10	7	-0.57	***	0.14	n.s.
Condobolin	NSW	11	430	WW	DD v CT	SB	NFm	25	10	14			-0.02	n.s.
Condobolin	NSW	11	430	WW	DD v CT	SR	NFm	25	10	14			0.00	n.s.
Cowra	NSW	8	564	WW	DD v CT	SG	N?	13	15	3			0.40	n.s.
Cowra	NSW	20	564	WW	DD v CT	SG	N?	13	10	7	0.09	n.s.	0.77	
Forbes	NSW	6	527	WW	DD v CT	S?	N?	15	10	3			0.49	**

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Location	State	Ref	Rain	Crop	Comparison	Rotation	N							
Goondiwindi	QLD	25	620	WW	DD v CT	SR/SRm	NFm	30	30	9			0.22	n.s.
Grenfell	NSW	20	564	WW	DD v CT	SG	N?	14	10	7	-0.39	**	0.21	
Hermitage	QLD	27	685	WW	DD v CT	SB	N0	65	20	33	-0.02	n.s.	-0.02	n.s.
Hermitage	QLD	27	685	WW	DD v CT	SB	NF	65	20	33	0.04	n.s.	0.00	n.s.
Hermitage	QLD	27	685	WW	DD v CT	SR	N0	65	20	33	-0.01	n.s.	0.01	n.s.
Hermitage	QLD	27	685	WW	DD v CT	SR	NF	65	20	33	0.07	**	0.07	**
Hudson	NSW	30	684	WF	DD[h]	SRm	NF	75	40	7	0.03	n.s.		
Kapunda	SA	12	492	LW	DD v CT	SG	NF	11	5	9	-0.13	*	0.07	
Kapunda	SA	23	492	WW/WP/WL	DD v CT	SG	NF	11	5	5			0.49	***
Lockhart	NSW	4	430	WW	DD v CT	SG...SB	N?	15	10	3			0.35	n.s.
Mackay	QLD	2	1668	Sc	DD v CT	GCTB/SB	NF	18	10	5			0.63	**
Merredin	WA	13	287	WW	DD v CT	SB	N?	20	25	6	-1.75	n.s.	0.35	n.s.
Merredin	WA	29	307	WW	DD v CT	SG...SB	NFm	20	10	9	-0.23	*	0.11	
Narrabri	NSW	17	616	CoCo	MT v CT	SR	NF	60	60	8			0.95	n.s.
Peak Hill	NSW	26	550	WW	DD v CT	SR	NF	28	20	3			-0.08	n.s.
Rutherglen	VIC	5	593	WW	DD v CT	SB	NF	22	2.5	10			0.04	*
Temora	NSW	7	535	LW	DD v CT	SR	N0	15	10	5			0.36	**
Temora	NSW	7	535	LW	DD v CT	SB	N0	15	10	5			0.42	**
Wagga Wagga	NSW	7	550	LW	DD v CT	SR	N0	27	10	19			0.29	**
Wagga Wagga	NSW	7	550	LW	DD v CT	SB	N0	27	10	19			0.34	**
Wagga Wagga	NSW	9	550	LW	DD v CT	SR/SB	NF	27	20	10			0.57	**
Wagga Wagga	NSW	14	550	LW	DD v CT	SR	N0	27	10	21	0.00	n.s.	0.14	
Wagga Wagga	NSW	14	550	LW	DD v CT	SB	N0	27	10	21	-0.14	**	0.12	
Wagga Wagga	NSW	14	550	SW	DD v CT	SR	N0	27	10	21	0.19	***	0.17	
Wongan Hills	WA	29	345	WW	DD v CT	SG...SB	NFm	10	10	9	-0.25	**	0.14	
									mean (different from 0[g])		**-0.22**	*	**0.25**	***
									s.d.		0.45		0.24	

[a] 1 = J Baldock, unpublished; 2* = Blair 2000; 3* = Blair et al. 1998; 4* = Burch et al. 1986; 5* = Carter and Mele 1992; 6 = Cavanagh et al. 1991; 7 = Chan and Heenan 2005; 8 = Chan and Mead 1988; 9 = Chan et al. 1992; 10* = Conteh et al. 1998; 11* = Fettell and Gill 1995; 12* = Grace et al. 1998; 13* = Hamblin 1984; 14 = Heenan et al. 2004; 15 = Holford et al. 1998; 16 = Hullugalle et al. 1998; 17* = Hullugalle and Entwistle 1997; 18 = Ladd et al. 1994; 19* = Mason 1992; 20 = Packer et al. 1992; 21* = Robertson and Thorburn 2007; 22 = Saffigna et al. 1989; 23* = Smettem et al. 1992; 24 = Standley et al. 1990; 25 = Thomas et al. 2007; 26* = Valzano et al. 2001; 27 = Wang and Dalal 2006; 28 = Whitbread et al. 2003; 29* = White 1990; 30 = Young et al. 2009 (* following number = no bulk density reported)

[b] CoCo = continuous cotton; CoF = cotton/fallow; CoWF = cotton/wheat/fallow; Sc = sugarcane; SgSg = continuous sorghum; WW = continuous wheat; WF = wheat/fallow; WL = wheat/lupine; WP = wheat/pasture; WS = wheat/subterranean clover; WC = wheat/chickpea; PPFW = pasture/pasture/fallow/wheat; PPPP = permanent pasture

[c] DD = direct drilled (i.e. no-till); CT = conventional tillage, usually district practice which may be reduced tillage; MT = minimum tillage, no-till expect for bed prep for cotton crops; RT = reduced tillage

[d] SB = burned; SRm = removed; SG = grazed; SI = incorporated; SR = retained; GCTB = green cane trash blanketing; _/_ = mean of treatments; SG...SB = grazing then burning

[e] N0 = no fertilizer; NF = fertilized, usually at district practice rate; Nfm = mean of multiple N treatments; N? = not reported

[f] Statistical significance – for rate of change, is the slope different from 0; for relative difference, are values different from each other: : * = significant at $\alpha < 0.01$; ** < 0.05; *** < 0.005; blank = could not be determined

[g] 2-tail t-test: * = significant at $\alpha < 0.01$; ** < 0.05; *** < 0.005

[h] This study imposed DD cultivation on fields that were previously under CT, sown pasture and DD. Rate of change is mean of treatments.
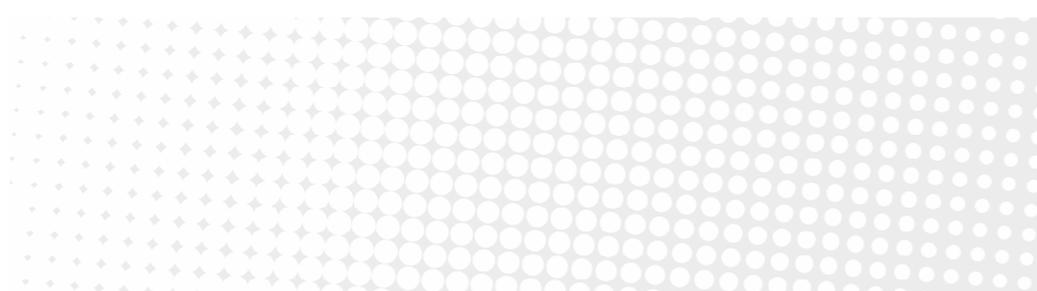
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