Accounting for space and time in soil carbon dynamics in timbered rangelands

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ABSTRACT

Employing rangelands for climate change mitigation is hindered by conflicting reports on the direction and magnitude of change in soil organic carbon (ΔSOC) following changes in woody cover. Publications on woody thickening and deforestation, which had led to uncertainty in ΔSOC, were re-evaluated, and the dimensional-dependence of their data was determined. To model the fundamentals of SOC flux, linked SOC pools were simulated with first-order kinetics. Influences from forest development timelines and location of mature trees, with a potential for deep-set roots, were considered. We show that controversy or uncertainty has arisen when ΔSOC data were not measured along sufficient lengths of the three Cartesian axes and the time axis, i.e. in 4D. Thickening and deforestation experiments have particularly neglected factors affecting the time and depth axes, and sometimes neglected all four axes. Measurements of thickening must use time-spans beyond the calculable breakeven date – when thickening just recovers the SOC lost through land degradation: then all ecosystems are likely to incur net sequestration. The similarity between half-life of carbon pools, and the half-time required for sequestration, mandates that millennial time-spans must be considered in design of SOC experiments. Spatial and temporal averaging of ΔSOC data that accounted for environmentally dependent decomposition rates, revealed that deforestation to pasture incurred a higher and longer-term net emission than earlier reported. Published reports on thickening or deforestation appear no longer contradictory when one considers that they only presented views from lengths of the 4D axes that were too limited. Adoption of this understanding into carbon accounting will allow more precise estimates of carbon fluxes for emission trading schemes and national reports.

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1. Introduction

For many rangelands woody vegetation plays a major ecological role, through vegetation types such as: (a) shrubland, woodland, open-forest, riparian forests in grasslands, or savannah; (b) improved pastures after deforestation; or (c) extant grassland experiencing woody thickening; (e.g. Carnahan, 1977; Lund, 2007;

Ellis, 2011). The term 'timbered' rather than 'forested' is used here with respect to rangelands, to avoid confusion between different countries' definitions of forests, and to include the woody-thickened state. Soil organic carbon (SOC) is a substantial terrestrial carbon (C) pool, often positively correlated with plant biomass (Su and Zhao, 2003; Harms et al., 2005; Hughes et al., 2006; Wheeler et al., 2007). The correlation means that SOC in many rangelands can be influenced by: overgrazing and land rehabilitation; deforestation and regrowth; woody thickening; fire; and climate change (Batjes and Sombroek, 1997; Reeder, 2002; Smith and Johnson, 2004; Luo et al., 2007). Consequently, rangeland SOC can act as a C sink in support of greenhouse gas (GHG) mitigation projects (McKeon et al., 1992; Glenn et al., 1993; Walker and Steffen, 1993). However, the difficulty in measuring and forecasting rangeland SOC may have resulted in conflicting reports on the direction and magnitude of change in SOC (ΔSOC)

that accompanies land management. This is especially true where there's only tenuous evidence for causation between observed effects and anthropogenic activity. The magnitude and direction of ∆SOC, can differ between decadal and centennial time-scales for rangeland grazing (for example), being more-definitive in the longer term (Piñeiro et al., 2006). Similarly, shallow measurement of SOC (≤0.2 m) can both greatly underestimate the SOC pool and lead to incorrect conclusions on the direction of ∆SOC following management effects (Harrison et al., 2011). Uncertainty, which is higher for ∆SOC, stifles climate change mitigation projects in the rangelands (Harper et al., 2007; Brown et al., 2010) and it has tacitly prevented inclusion of some rangeland SOC fluxes in national GHG accounts, thereby detracting from accounting accuracy. Conceptualisation in four dimensions (4D, the three Cartesian axes plus time) has provided scientific insight and management options, in ecology and C dynamics (e.g. Ward, 1989; Dean et al., 2004; Dean and Roxburgh, 2006; Turnbull et al., 2008; Colyan and Ginzburg, 2010). Woody thickening in rangelands, has significant 4D diversity in SOC concentrations (Boutton et al., 1998). We hypothesize that conceptualising rangeland C dynamics in 4D can provide a coherent solution to dissonance over ∆SOC for significant rangeland processes. Two examples are selected for re-evaluation using this hypothesis, from reports where conclusions had important consequences for climate change mitigation.

The sampling span (extent, length) used in rangeland experiments is crucial. For example, a correlation between biomass and SOC has sometimes appeared ambivalent but was confirmed as positive when experiments were longer-term and when SOC was sampled at depths more related to root extent under woody vegetation (Groenenberg et al., 1998; García-Olivia and Masera, 2004; Liao et al., 2006a,b). Longer-duration experiments on change in biomass or ∆SOC are generally more definitive than shorter ones (Lugo and Scatena, 1996; Diochon et al., 2009). Conversely, short-duration (decadal) experiments may yield low-magnitude and conflicting results for ∆SOC (Lugo et al., 1986; Guo and Gifford, 2002; Liu et al., 2004; Marin-Spiotta et al., 2009). Although SOC is generally correlated with vegetation cover there can be short-term fluctuations in SOC when the vegetation cover changes, which can occlude the influence of interest. For example, such as from the priming effect (Fontaine et al., 2007), and early oscillations in SOC following deforestation, followed by long-term emission and an asymptote (García-Olivia et al., 1994; García-Olivia and Masera, 2004). SOC can take approximately two millennia to stabilize, such as for rangeland thickening (Hibbard et al., 2003) and for the decrease in SOC when forests were replaced by prairie due to fire during the Holocene (Baker et al., 1996).

Accounting for dimensions is also important during data processing. For example, when averaging samples, their equivalence with respect to the effect being studied is assumed, but that effect may change nonlinearly in a particular dimension. Therefore Jensen's inequality (Jensen, 1906; Welsh et al., 1988; Chesson, 1998; Ruel and Ayres, 1999) becomes relevant. It implies for a non-linear function that the average of the means is not necessarily equal to the means of the averages. For example, site location (environment), is a function of x–y position but SOC decomposition rates may vary between environments. SOC emission following deforestation is higher from higher SOC sites (Holmes et al., 2006) but proportionally higher from poorer soils (i.e. lower SOC sites) (Harms et al., 2005). Averaging over such complexity would blur effects and thereby limit accuracy of results. Here we consider the influences over 4D of deep-set SOC, and deep-rooted, long-lived plants and their decomposition products. The focus here is on the time and depth (z) axes as these have been the most neglected, but data from all four axes are integrated into our re-analyses.

Firstly, we provide an introduction to SOC distribution in rangelands and demonstrate the relative benefits of one-, two- and three-pool SOC models, then use one of those to emulate results from the CENTURY model (a three-SOC-pool system with feedbacks (Parton et al., 1987)). We did not run CENTURY directly because: (a) of the amount of site-specific data required (e.g. for climate, plant growth and decomposition), (b) the number of sites examined, and (c) we wanted to display the keystones of the dynamics transparently, using the simplest, applicable mathematics.

In the first example we address the influence of the time axis (t-axis) and z-axis (depth) in data sampling, processing and interpretation. Rangelands in the southwest USA were overgrazed in the 19th century, inducing degradation, with erosion, net C efflux, and reduction in ecosystem function, followed by recovery and thickening (Lund, 2007; Wheeler et al., 2007; Wilcox et al., 2008). Sequestration associated with such thickening in a Texan grassland was found to approach an asymptote with a nearly fourfold increase in SOC after ~2500 years (Hibbard et al., 2003). Conversely, in Jackson et al. (2002) where rainfall was above ~460 mm year^{-1}, thickening of mesquite (*Prosopis* spp.) was found to induce net SOC emission. This latter study was unique in that it tested for roots nearly as far down as a phreatophyte's lowest root (in this instance to 10 m), and proportional change in SOC with thickening, was determined to 3 m depth. Principally, its conclusions were that woody thickening incurs a net emission in wetter sites and, more generally that 'assessments relying on C stored from woody plant invasions to balance emissions may therefore be incorrect'. The findings, published in 'Nature', caused apprehension over whether or not the USA national C sink had been overestimated (Goodale and Davidson, 2002). While scepticism was voiced, there were no compelling explanations (Archer et al., 2004; Wheeler et al., 2007; Liao et al., 2008) and controversy remains.

In the second example we scrutinise data interpretation on the t-axis and the x–y plane. Approximately 22 Mha of rangeland in Australia has been deforested for intensified, commercial grazing (Dean et al., submitted for publication). Curtailing the accompanying change in biomass was instrumental in Australia's adoption of the Kyoto Protocol, though the accompanying ∆SOC has not yet been sufficiently reconciled (Henry et al., 2002). Rates for the State of Queensland (QLD), Australia, have been the highest nationally for several decades, e.g. ~0.343 M ha year^{-1} in 2006, 59% of which was remnant native vegetation (DNRW, 2008). We analyse and reinterpret data from a comprehensive study of deforestation in QLD where ∆SOC following deforestation ranged from −5.4% to 1 m depth (Harms and Dalal, 2003; Harms et al., 2005), to −31% for 0.05 m depth (Dalal et al., 2005). The former measurement contributed to high uncertainty for grazing businesses in a study considering GHG benefits of land rehabilitation (Bray and Golden, 2008).

Lastly we discuss how identifying the components of events and their C fluxes can facilitate concord between existing reports on ∆SOC for rangelands.

2. Background to the 4D distribution of SOC in timbered rangelands

The likely distribution of C in 4D, when the SOC is supplied by woody vegetation, forms a foundation for our interpretation of ∆SOC. To portray key features, the biomass distribution of a semiarid thickening species (Fig. 1) was idealised from the literature (Heitschmidt et al., 1988; Gile et al., 1995, 1997; Dean et al., 2009). The time-dependent changes illustrated are inherent in the natural growth and decomposition of woody vegetation, of which self-thinning is a key component (discussed in the Appendix). From
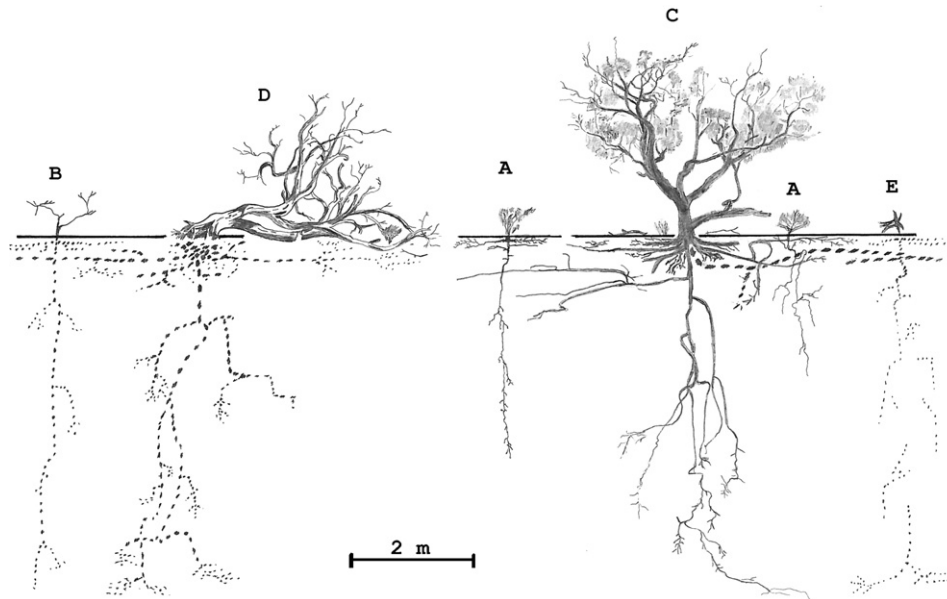
Fig. 1. Four-dimensional distribution of biomass, its decomposition residues and SOC from different growth stages of a woody thickening species in semiarid rangeland. The SOC portrayed is that arising from root residues, prior to any eluviation or soil turbation. Dotted lines indicate decomposing roots and resultant sites of concentrated SOC (Chabbi et al., 2009); fainter dots indicate more advanced decomposition. Tree growth stages: A, juvenile; B, juvenile dead; C, mature/senescent; D and E, dead and fallen; E, stump (dead). For clarity, the larger number of lateral, near-surface roots, have been omitted, as have the larger number of stages A and B for young or overgrazed stands. With self-thinning, the older and larger individuals will be further apart in the x–y plane, creating a 4D trend in SOC concentration. Biomass decomposition is necessary over several generations of shrubs (or trees) to form the maximum SOC stock (cf Figs. 2a and b and 4).

a SOC isotopic study of mesquite thickening in Texas (Boutton et al., 1998) self-thinning was found to be ongoing in the 1990s following land degradation in the previous century. Fig. 1 illustrates that the concentration of SOC derived from aboveground carbon and from roots (Chabbi et al., 2009), varies over the x–y plane and with depth (i.e. in 3D). The depth concentration of SOC is likely to vary with tree maturity, and due to self-thinning the distance between individual trees increases with age, and thus so will the 3D variation of SOC, but with a time lag while biomass decomposes. The distribution of biomass, and therefore of SOC, is perturbed by land-use such as overgrazing, deforestation and reforestation. In the form of a partial differential equation the SOC flux in 4D is:

$$\frac{\partial SOC}{\partial x \times \partial y \times \partial z \times \partial t} = \frac{\partial SOC}{\partial x \times \partial y \times \partial z} \times \frac{1}{\partial t} \tag{1}$$

i.e. the SOC flux is the magnitude of change in 3D SOC concentration multiplied by the rate of change. For the pedosphere however C flux is usually reported in change of mass per unit area per unit time (e.g. Mg ha^{-1} year^{-1}) which, although correct for an area-flux, has possibly helped engender complacency in measurement depth. The IPCC recommends that measurements be taken down to where significant change is expected to occur (IPCC, 2000). This condition of 'significant' is ambiguous, with deviations from the original intent facilitated by (a) the 0.3 m span common in analysis of arable lands, (b) faster change in the upper horizons (e.g. Dalal et al., 2005) perhaps being interpreted as the only change of significance, and (c) difficulty in sampling deeper or under lateral roots. Frequently these shortcomings are unaccounted for in data processing and in the conclusions drawn.

The relevance of sampling extent combined with the relevance of measuring in 4D, necessitates an appropriate sampling extent along each of the four axes. From a literature survey it appeared that to-date, ∆SOC in rangelands has only been measured in limited combinations of each of the four axes (within any particular experiment), e.g. in the x–y plane (the surface soil) at different scales (Reeder, 2002; Holmes et al., 2006; Wheeler et al., 2007; Throop and Archer, 2008), along the z-axis (Jackson et al., 2002; Fontaine et al., 2007), in x–y–z (Don et al., 2007; Chabbi et al., 2009), along the t-axis (i.e. temporally) (Hibbard et al., 2003; García-Olivia and Masera, 2004; Poussart et al., 2004; Chiti et al., 2009; Diochon et al., 2009; Schulp and Verburg, 2009), and in x–y–t (Wilson and Thompson, 2005). Details of appropriate sampling of SOC in the x–y plane have been previously described (e.g. Conant and Paustian, 2002; Poussart et al., 2004; Holmes et al., 2006; Allen et al., 2010). Temporal measurement of rangeland SOC has been discussed but only for seasonal affects (Allen et al., 2010), i.e. a short extent on t.

The SOC distribution along the z- and t-axes warrants comment. Belowground biomass is expected to provide the major source of long-term SOC and dominate SOC dynamics in semiarid areas through root turnover and tree mortality (Wang et al., 1996; Lorenz et al., 2007; Filley et al., 2008). Carbon from deeper roots, can contribute more to SOC than does aboveground biomass (Rasse et al., 2005). Woody plants in semiarid areas may have a dimorphic root structure and be phreatophytic, combining long, lateral, surface roots with well-developed tap or sinker roots to access ground water (Canadell et al., 1996; Gile et al., 1998; Scott et al., 2004; Grigg et al., 2010). Deeply penetrating roots can facilitate infiltration of water and dissolved organic carbon (DOC) to greater depths (Archer et al., 2002; Rumpel and Kögel-Knaber, 2011), and establish microbial communities and SOC fluxes there. Deep-set SOC can comprise at least half the total SOC (Harrison et al., 2011); it accumulates and is destabilized through a range of processes, and generally has a greater average age than that in surface horizons (<0.2 m), e.g. in the order of a thousand years (Rumpel and Kögel-Knaber, 2011). Therefore it is important to conceptualise its affect when designing rangeland SOC experiments. Few experiments have analysed the contribution to SOC from coarse roots (Rumpel and Kögel-Knaber,

2011) but there is increasing recognition of their contribution (e.g. Hancock et al., 2007; King et al., 2007; Kirschbaum et al., 2008; Nygren et al., 2009). Coarser woody-roots decompose more slowly (Ludovici et al., 2002; Kemp et al., 2003; Janisch et al., 2005). With the longer timelines involved in deep-set C and coarse woody-roots, measurement of SOC over a large extent of the t-axis is more critical, when there is change in woody vegetation.

3. Methods

3.1. Formularising the temporal dependence of ΔSOC

Insights into the basic temporal dynamics of ΔSOC emerge from the mathematical correspondence established between some ecological and radioactive processes (Jenny et al., 1949; Olson, 1959, 1963). The decomposition sequence for SOC, where C pools are linked in series, is conceptually similar to chain radioactive decay with the SOC stock equating to the radiation background. Principles from the mathematics for radioactivity can be applied to SOC. In aerobic conditions SOC formation from plant residues can be represented by a three-stage process, linked to particle size and time-dependent chemical or physical stability (Parton et al., 1987; Liao et al., 2006a; Grandy and Neff, 2008) or approximated by a 2-pool system (Trumbore, 2000; Kirschbaum, 2004), or by a one-pool system with a time-dependent decomposition rate (Yang and Janseem, 2000; Feng, 2009). The mathematics for SOC changes in such pool systems have been presented by Bruun et al. (2004) and in this section we describe a system suitable for use in modelling the C dynamics of change in woody vegetation in rangelands.

Several principles arise for SOC from the comparison with chain radioactive decay:

(1) the half-time of sequestration (i.e. half the time to reach maximum stock) is similar to the half-life of the slowest (longest half-life) pool;
(2) the maximum stock is linearly proportional to the half-life of the slowest pool;
(3) the decomposition rate is similar to that of the slowest pool; and
(4) the time to accumulate 95% of maximum stock is triple the turnover time of the slowest pool (i.e. 3/decay constant) (Jenny et al., 1949).

These four principles apply directly to SOC sequestration under woody vegetation, with the following implications:

(1) because the SOC arising from woody vegetation is more recalcitrant than that from herbaceous vegetation (due to its higher concentration of aliphatic root suberin, waxes, glycerides, lignin and tannins) (Liao et al., 2006a; Lorenz et al., 2007; Filley et al., 2008), then together with its deeper roots the time to 95% sequestration is longer and the resultant SOC stock is higher; and
(2) because SOC longevity increases with depth (Dalal et al., 2005; Rumpel and Kögel-Knaber, 2011), the half-time for sequestration for the full soil profile is longer than that of the upper horizons alone.

Considering a one-pool model (Yang and Janseem, 2000; Feng, 2009), the amount of SOC remaining at time t (in years) is SOC_t given by:

$$SOC_t = SOC_0 \, \exp(-(kt)^\alpha) = SOC_0 \exp(-k't^\alpha) \tag{2}$$

where SOC_0 is the initial stock, k' is equal to k^α, with $k = 2.39$ and $\alpha = 0.22$, i.e. $k' = 1.21$. After a short period this system becomes equivalent to first-order decomposition with a half-life of ~35 years. (For example, SOC_{12} is essentially the same as with $\alpha = 1$ and $k' = 0.01959$.) Such rapid decomposition does not allow SOC to accumulate if shrubs or trees die more than a few decades apart. This absence of a "long tail" has been noted previously for a 1-pool model (Manzoni et al., 2009). The parameter values in Eq. (2) were derived from experiments mostly observing decomposition for less than 11 years (Yang and Janseem, 2000; Feng, 2009). Also, most samples were of the more-labile carbohydrates, or were derived from arable-crop, herbaceous species. Such short half-lives, and absence of more stable compounds, are inapplicable to comprehensive rangeland SOC dynamics, which include subsoil and woody vegetation.

Manzoni et al. (2009) found that a 2-pool, or more complex systems, may represent longer-term dynamics better than a one-pool model. Consequently, we chose parameters from rangeland studies that included woody vegetation and multiple pools.

Under woody thickening, a 3-pool system yielded SOC half-lives of 5, 300 and 393 years (Neff et al., 2009). Another thickening study (Liao et al., 2006a) yielded half-lives of 21, 42 and 250 years plus half-lives up to 770 years. The times to 95% sequestration (3/k of the slowest pool) for these two studies were 1700 and 1082 years, respectively. These times to 95% sequestration are of the same order of magnitude as the ~2000 years from CENTURY modelling of mesquite thickening (Hibbard et al., 2003), to 0.2 m depth (a depth constraint on that version of CENTURY). Both 3-pool studies were within the upper 0.15 m of soil, therefore the half-lives and mean residence times (MRTs, or turnover times = 1/k) for the full soil profile would be longer.

To encompass these two studies, we examined ΔSOC with: (a) a 3-pool system, with half-lives of 5, 150 and 500 years and (b) a 2-pool system, with half-lives of 2 and 700 years (Fig. 2a and b). The SOC pools were interconnected and without feedback. Input to the SOC pool was from biomass decomposition following (a) continuous (annual) and (b) episodic biomass death (namely once every 200 years – bicentennially).

Biomass decomposition was modelled as exponential decay, with half forming nascent SOC and half emitted to the atmosphere. Of the nascent SOC, 55% was delivered to the fast pool and 45% to the slower pool. Then 60% of the SOC from the fast pool went to the slow pool, along with 40% emission to the atmosphere (i.e. 'respiration'). This is similar to the 50:50 allocation for the 2-pool system in Manzoni et al. (2009). Similarly, in the 3-pool system, 60% went from the slow pool to the slowest pool, with 40% emission to the atmosphere. The slower-two pools in the 3-pool model were in series.

To achieve the same, long-term SOC stock of 52 Mg ha^{-1} for the two systems, inputs of C in dead biomass were 95 and 64 Mg bicentennially, for the 3- and 2-pool systems, respectively. The higher influx rate needed for the 3-pool system was due to extra respiration, being from the intermediate pool, giving faster SOC decomposition of each pulse from decomposed biomass.

The difference between continuous and episodic input to SOC from maturation of thickening was compared for both the 2- and 3-pool systems (Fig. 2). Episodic input, such as from mortality accompanying self-thinning, senescence or fire, produces undulating net SOC; whereas continuous (annual) input, e.g. representing litter fall, fine root turnover, exudates, or slow senescence, produces a smoother curve. The total SOC flux and maximum stock are approximately identical whether the input is bicentennial or annual, so long as the time-averaged input rate to the soil is the same. In reality, however, if the annual deposits are derived from fine roots, then they may contain substances
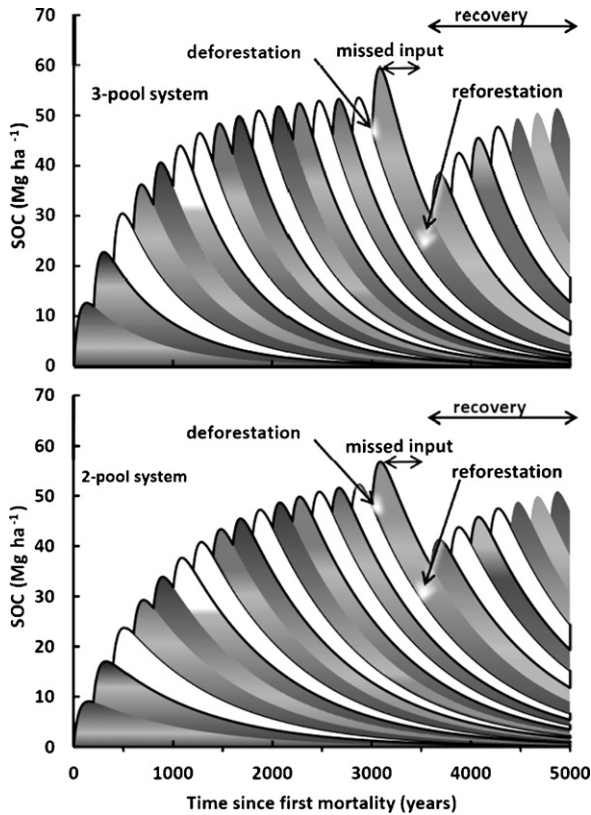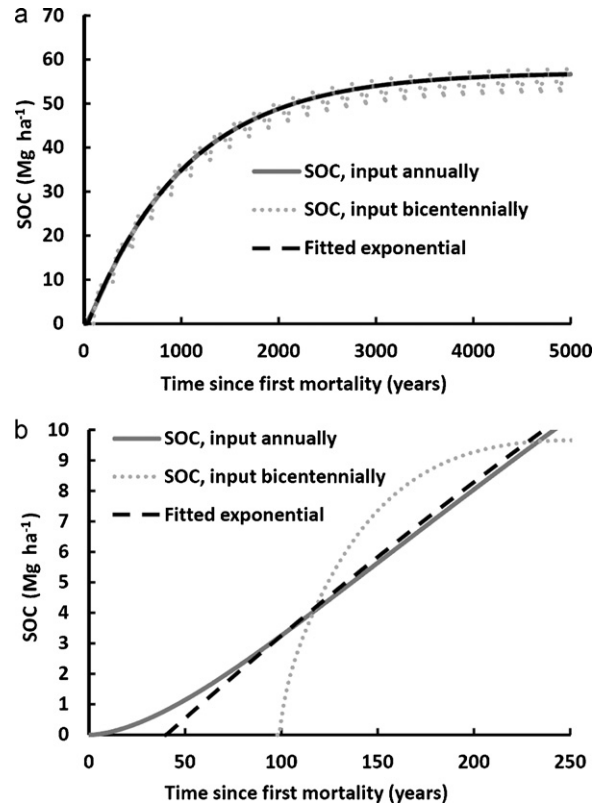
Fig. 2. SOC sequestration over the full soil profile, accompanying maturation and concomitant decomposition of woody thickening for 3- and 2-pool systems with bicentennial input to SOC from tree mortality. Shades distinguish different biomass death cohorts. The SOC decomposition and net sequestration profiles are steeper for the 3-pool system. Deforestation incurs an initial peak before a decline, then a long duration for recovery after reforestation. For clarity the SOC from grass has been omitted from the deforestation epoch (but is included in Fig. 4).

Fig. 3. SOC sequestration for woody thickening; 2-pool model. Biomass mortality supplying input to SOC and wood decomposition is simulated by two separate modes: episodic, 50 Mg ha^{-1} once every 200 years (undulating, grey, dotted line); continuous, 0.25 Mg ha^{-1} annually (solid grey line). Average, long-term sequestration trends are the same for both supply rates. The curve fitted by regression (black, dashed line) to the continuous input shows that long-term sequestration was approximately exponential (a), but a close-up of the first century shows it is actually logistic (b).
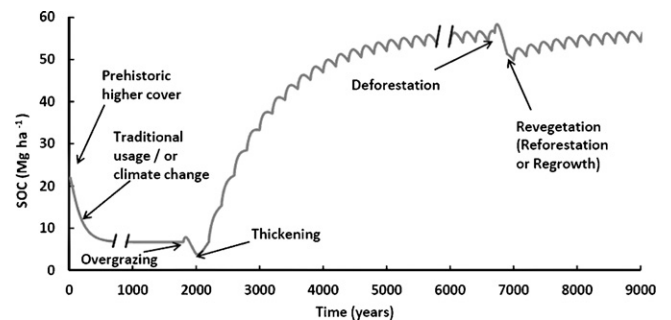
with different decomposition rates than from the more-episodic input from coarse-root death, and thereby produce different SOC stocks. There are few reports of relative contributions to the total, from continuous turnover and from episodic mortality, but one estimate (Chestnut et al., 1999) suggests they are approximately equal.

The annual input to SOC produced a logistic-shaped curve for ΔSOC, which could be approximated by exponential curves prior to, and after, the inflection point (Fig. 3). This concurs with other modelling of ΔSOC (Wynn and Bird, 2008), and mathematical approximations for logistic curves (Birch, 1999; Ford, 1999; Flynn and Ford, 2005). The use of logistic curves for modelling forest biomass growth is well-established (e.g. Bruce and Schumacher, 1942; Knuchel, 1953). The inflection points for the 2- and 3-pool models were 154 and 130 years, respectively. Prior to the inflection point on the logistic curve, is the early stage of thickening, where juvenile stands may self-thin, and where therefore input to SOC from biomass decomposition is likely to be more continuous than episodic. The long sections of the curves along the t-axis could be approximated by exponential functions, which allowed calculation of sequestration timelines assuming first-order kinetics.

The similarity in sequestration curves for the 2- and 3-pool systems (Fig. 2a and b) meant that a 2-pool system could be used for basic visualisation of changes in relative proportions of grass- and woody-cover. Long-term SOC stocks and fluxes, for grassland and thickening were adapted from CENTURY output from Hibbard et al. (2003). The 2-pool model was fitted to that data to show

Fig. 4. Time series of ΔSOC accompanying thickening, deforestation and reforestation, from a 2-pool model for SOC. The rippled line, corresponding to ΔSOC from woody growth, results from half-annual and half-bicentennial turnover. Chronologically, events are: traditional usage or climate change reduces vegetative cover to a savannah or grassland, subsequent overgrazing and land degradation reduces vegetation cover and loses topsoil, thickening incurs sequestration, deforestation (plus new grass growth) incurs C emission, and revegetation incurs sequestration. Note the long timelines involved after management affects, before substantial change occurs in SOC, e.g. for reforestation following an absence of woody vegetation for 200 years.

diagrammatically the effects of a sequence of major impacts on SOC: land degradation, thickening, deforestation and reforestation (Fig. 4). For woody growth the input to SOC was modelled as 50:50 annual and bicentennial; input from grass was annual. Deforestation was simulated simply by an extra pulse of dead biomass
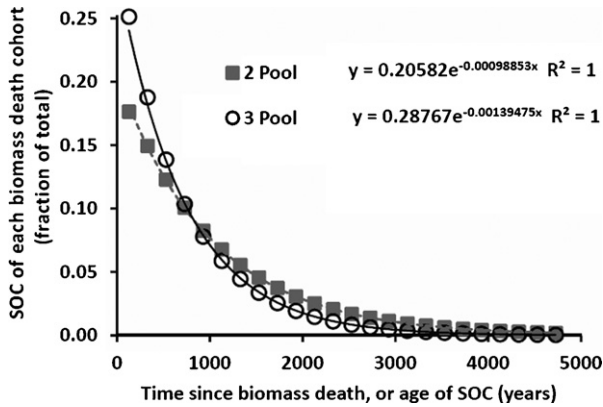
Fig. 5. Proportions of SOC present, as a function of when the contributing biomass died (i.e. age of SOC). Note that at any one instance in time there is a range of ages of SOC, with more of the younger SOC. For this simulation the average times since biomass died was 732 and 1005 years for the 3-pool and 2-pool systems, respectively; i.e. close to the turnover times of the slowest pools.

(corresponding to the freshly killed trees) followed by stopping input to SOC from woody vegetation for 200 years and meanwhile generating annual input to SOC from grass. Reforestation (with reduction in grass cover) was simulated by re-starting the input to SOC from woody thickening and stopping the input from grass.

To show how the 2- and 3-pool models represent the age demographics of SOC, the fraction of SOC present from each age cohort (i.e. from time since biomass death) was graphed as a function of time (Fig. 5). The creation date for SOC was selected as when nascent SOC production was highest from each tree-mortality episode. At any instance in time there was exponentially more SOC from the more recent cohorts – concurring with Bruun et al. (2004). The average SOC age for the 2- and 3-pool systems were: 1005 and 732 years, respectively. Error margin in these figures resulted from the gradual release of dead biomass to SOC. The resulting average ages for SOC are close to the MRTs for the slowest SOC pools (1010 and 721 years for the 2- and 3-pool systems, respectively), matching expectations from first-order kinetics (Manzoni et al., 2009). Thus the simple models adequately represent the major age demographics of SOC.

Following the logistic curve shapes (Fig. 5) for the net sequestration effects of SOC in the 2 and 3-pool models above, and for SOC in grassland degradation and woody thickening (Hibbard et al., 2003), it was appropriate to employ logistic curves to display the net effects of land degradation and thickening combined. The form of logistic function used to represent SOC_t from grassland and thickening was:

$$SOC_t = SOC_0 + flux\_mag \left(1 - \frac{1}{1 + (m \times t^k)}\right) \quad (3)$$

where t is time, SOC_0 is the initial stock, $flux\_mag$ is the maximum change in stock (>0 for sequestration and <0 for emission), and m and k are fitted parameters.

3.2. Example 1 – time and depth axes

To simulate $SOC(t)$ for grassland degradation and thickening we modelled the events described in the thickening study of Jackson et al. (2002) using Eq. (3). An approximate onset date for degradation was 1850 (Hibbard et al., 2003). The time since onset of thickening equals the stand age. To parameterize the SOC levels in Eq. (3) we fitted data from the output of CENTURY modelling by Hibbard et al. (2003), as there was insufficient information in Jackson et al. (2002) for parameterisation (e.g. no asymptotic
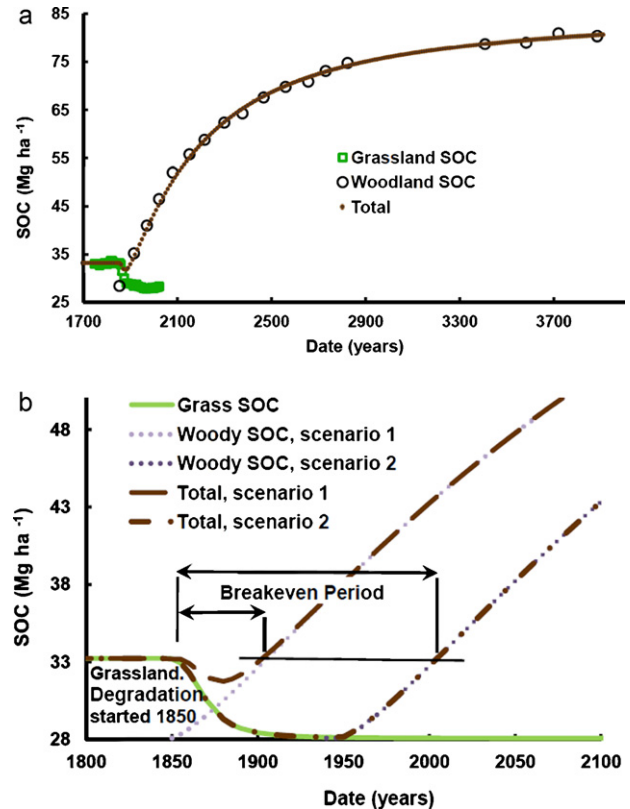
Fig. 6. (a) Modelling of SOC (to 0.2 m depth) accompanying grassland degradation and mesquite woody thickening. Data points are from published CENTURY modelling by Hibbard et al. (2003). Thickening starts simultaneously with land degradation, in 1850. (b) Close-up of (a) near the onset of thickening, but with two different scenarios for when thickening begins. Totals for two thickening scenarios: 1, thickening starts in 1850; 2, thickening starts in 1880. Straight lines show the breakeven period – the time taken to regain SOC to the pre-degraded grassland level. Measurement of ΔSOC before and after the breakeven date will record emission and sequestration, respectively.

stocks). Thickening was modelled as beginning at two possible dates: in 1850 (scenario 1, simultaneously with onset of land degradation), and in 1949 (scenario 2, 99 years after establishment of degradation) (Fig. 6b). The onset date for scenario 2 was chosen from the average stand age in Jackson et al. (2002), which concords with a SOC isotopic study of mesquite thickening in Texas (Boutton et al., 1998) which found that thickening began less than 100 years ago. That latter study also found that thickening started at different times on different topographic landscape components (which validates testing more than one date for onset of thickening). However, in their study there was the additional complication of SOC translocation together with different stand ages in different landscape positions, which prevented further interpretation of SOC age.

With time being crucial to the sign of ΔSOC we re-aggregated the original (Jackson et al., 2002) data as a function of stand age.

Each of the six data points in Jackson et al. (2002) consisted of a pair, with one member ostensibly corresponding to un-degraded grassland and the other member corresponding to a woody thickened site, with other factors being equal within each pair. Data were checked to see if they complied with that pairing condition. From Google-Earth Inc. imagery, it was observed that all study sites except Riesel were in extensive land-use zones, whereas the Riesel site was in an intensive, arable agriculture zone, containing regenerated prairie (Potter et al., 2004). Possible implications of this are: (a) the thickened member of the paired site may have been too

degraded for development as arable land (compared with neighbouring prairie) or (b) it had a locally unique substrate or (c) the grassland member of the pair had received remedial treatment (artificially incurring sequestration). This suggests a complex site history that could be grounds for discarding the Riesel site data point from the SOC trend investigation. Site history influences C distribution on the z- and t-axes, not just on the x–y plane. Also the Riesel site had a different trend with depth for $^{87}Sr/^{86}Sr$ isotope ratios to the other sites, and a second peak in SOC starting at 9 m depth (the z-axis), which was higher for the thickened area; but this extra C was not included in the regression with rainfall in the original study (Jackson et al., 2002). These two factors gave grounds to doubt the validity of Riesel as a paired site, and consequently we tested data for correlations with and without that site.

3.3. Example 2 – time and horizontal axes

In this example of deforestation for grazing, using studies from QLD, representation on the t-axis was approached by grouping $\triangle$SOC data as a function of time. Representation in the x–y plane was considered by: (a) calculating temporal trends for data from only environmentally equivalent sites and (b) spatial averaging of data.

In the original study (Harms et al., 2005) space was substituted for time by choosing sites deforested at different dates and by pairing those thought to have been environmentally equivalent prior to deforestation. Across 50 Mha there were 32 locations, represented by 47 paired sites. SOC was sampled to 1 m depth. Environmental differences (e.g. texture, including clay content; microbial species; moisture availability; temperature; and tannin and lignin contents of plant species) can strongly affect decomposition rates and amounts of biomass debris (Cornwell et al., 2008). To assist in controlling for such spatial heterogeneity, and to avoid bias from the early oscillations in fluxes (García-Olivia et al., 1994), we only used sites for which there were two or more distinct sampling dates after deforestation. One location, Kindon, had improperly paired sites owing to differences in clay content, cation exchange capacity and profile mixing (Harms et al., 2005) and was not used in the re-evaluation.

We graphed the percentage change (from the pre-deforestation SOC stock) as a function of time since deforestation. The original 32 locations became 9 trend lines. These lines formed vectors, which together, revealed a flux field (a separate change vector for each location). The average SOC efflux, was 0.58 (0.54)% year^{-1}. That average efflux rate was summed over the 5–35 year experimental time span to give the total SOC efflux for deforestation.

Insight into the time and environmental dependence of different spatial averaging methods can be provided by the [non-linear] formulation of SOC(t). Following deforestation, $\triangle$SOC can be represented by a logistic curve which, after the initial fluctuations and prior to the inflection point, can be approximated by an exponential function (see Section 3.1):

$$Ctot = C_g + ((C_w - C_g) \times \exp(-kt)) \qquad (4)$$

where t is the time since deforestation, $Ctot$ is the SOC after time t, C_w is the initial SOC under woody vegetation, C_g is the SOC under grass only, and k is the decomposition rate constant. Where deforestation incurs sequestration, k is negative.

Spatial averages for $\triangle$SOC with deforestation can be the average of all the percentage changes (the "disaggregated average", DA, giving $\triangle$SOC$_{DA}$), or the percent change after aggregation of SOC stocks (the "aggregated average", AA, giving $\triangle$SOC$_{AA}$). For the

ratio of $\triangle$SOC between the grassland and woody states, Jensen's inequality (see Section 1) states that $\triangle$SOC$_{DA} \neq \triangle$SOC$_{AA}$, i.e.:

$$\frac{1}{n}\sum_{j=1,n}^{j}\left(\frac{Ctot_j - C_{g_j}}{C_{w_j}}\right) \neq \frac{(1/n)\sum_{j=1,n}^{j}(Ctot_j - C_{g_j})}{(1/n)\sum_{j=1,n}^{j}(C_{w_j})} \qquad (5)$$

where n is the number of paired sites. Thus AA is mathematically incorrect when there is environmental variability, or where the values being measured are correlated to at least some degree and the function is non-linear, as is usual in the real world. However, if either C_g or C_w are constant across all sites (i.e. invariant in x–y), or if C_g is not correlated to C_w (i.e. if $\triangle$SOC is not related to the initial stocks) then the inequality may vanish.

Effects of the different methods of spatial averaging, along with different, speculative decay constants for two example sites were examined. The AA was used in Harms et al. (2005); therefore we re-processed the original data using DA.

4. Results

4.1. Example 1 – time and depth axes

The results for land degradation followed by thickening (Fig. 6) were:

1. Recovery of SOC to pre-degradation level (i.e. 'breakeven') was attained at dates 1903 and 2004, for scenarios 1 and 2, respectively. That is, the breakeven periods were 53 and 154 years, respectively, and the stand ages at the breakeven date were 47 and 55 years, respectively.
2. Emission was greater for scenario 2, i.e. when onset of thickening was delayed (15% (5.0 Mg ha^{-1}) compared with 4% (1.25 Mg ha^{-1}), in this example).
3. If measurements were taken either side of the breakeven date then they would provide a different sign for $\triangle$SOC, i.e. they would yield contrary conclusions. Measurements prior to breakeven would indicate emission; close to breakeven would yield $\triangle$SOC = 0; and after breakeven – sequestration. Only the Riesel site showed emission after the breakeven period, and there was ample reason to disregard that site. In Jackson et al. (2002) sites were sampled on both sides of the approximate breakeven date. More generally: short-term $\triangle$SOC experiments, with durations near the breakeven date can give contrary results.
4. If the minimum measurable $\triangle$SOC is 20% (e.g. Huntington et al., 1998; Yanai et al., 2003), then no significant change would be recorded until 1967 and 2065, for scenarios 1 and 2, respectively, i.e. ~110 years after onset of thickening. No stands in Jackson et al. (2002) were that old.
5. Breakeven would occur earlier if degradation ceased after onset of thickening.

The number of samples required to discern change is proportional to an inverse power of the time since change began (Poussart et al., 2004). Thus for a 90% chance of a discernable $\triangle$SOC, ~75× more samples are required if measured after 5 years rather than 50 years since change began (Poussart et al., 2004). Only half the data points in Jackson et al. (2002) had a stand age of 50 years or older – indicating that more samples may have been necessary when measuring the younger stands.

With elimination of the Riesel site there appeared a weak, positive correlation between $\triangle$SOC and stand age, R^2 = 0.39, p = 0.1, N = 5, which was absent when the Riesel data were included (Fig. 7). From the line of best fit it appeared that $\triangle$SOC = 0 when the stand age was 48 ($\pm$10) years. That value is similar to the stand ages
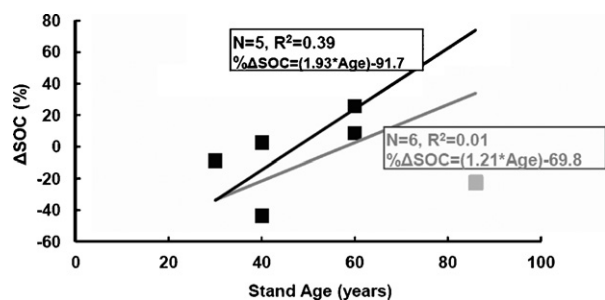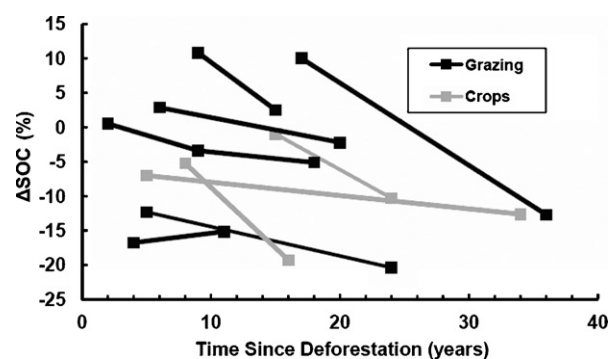
Fig. 7. Data from Jackson et al. (2002) re-aggregated as a function of stand age. For calculation of the straight line of best fit, the standardized major-axis analysis method (Warton et al., 2006) was used as uncertainty existed in both abscissa and ordinate. Lines of best fit: grey, all 6 data points; black, all data points minus that for the Riesel site (grey point). Note that a trend for $\Delta SOC(t)$ appears with elimination of the Riesel data.

Fig. 8. Chronosequence trends in SOC after deforestation, as a function of time since deforestation for sites in QLD (original data from Harms et al. (2005)). Percentage change is compared to the pre-deforestation SOC stock. Points joined by lines correspond to paired sites. After initial perturbation with deforestation, emission continues for at least several decades. The efflux rate is the averaged slope of the lines.
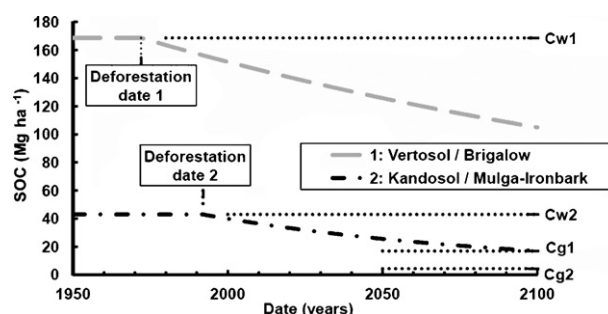
Fig. 9. Initial SOC stock and change in time for deforestation of two example sites in QLD. Percentage change for two sites, deforested at different times. 1: high SOC, Vertosol with brigalow or gidgee (*Acacia cambagei*); 2: low SOC, Sodosol with Ironbark. In site (1) deforestation occurred 40 years earlier. Corresponding data in Table 1.

at the breakeven dates for scenarios 1 and 2, and those reported elsewhere for thickening in southwest USA (Hughes et al., 2006; Wheeler et al., 2007; Throop and Archer, 2008). It is likely that a more definitive trend between ΔSOC and stand age could be achieved with appropriately purposeful experimentation.

The modelling suggests that sequestration may be significant for a few more centuries after the breakeven date. This indicates that much longer periods of measurement are required (i.e. a longer span of the *t*-axis) to make conclusions on the direction of ΔSOC with thickening, above the average stand age of 53 years used in Jackson et al. (2002). Also, this re-evaluation shows that site pairing is not just an issue in the *x–y* plane but site-based influences also arise from the *z*- and *t*-axes.

Note that elimination of the Riesel data did not significantly alter the relationship between ΔSOC and rainfall found in Jackson et al. (2002) – it was supported by the remaining five data points. Nor did our findings totally contradict the conclusion in Jackson et al. (2002) that GHG emission offsets may not be achieved through woody thickening – instead the findings confirmed that conclusion was true at the time of their experiment. For example, for scenario 2 – breakeven was achieved in 2004 but not measurably so until 2065 (both dates being later than the experiment by Jackson et al. (2002)). Furthermore, our parameterisation of the logistic curves was based on the 0.2 m depth work of Hibbard et al. (2003), and with deeper horizons having slower kinetics (see Section 3), i.e. requiring sampling of a comprehensive extent along the *z*-axis, then the breakeven date may be even later than that shown in Fig. 6b. Hence, the work here showed that the conclusion in Jackson et al. (2002) would require adjustment (examination of a longer period and sampling SOC to at least rooting depth) to account for the fact that although GHG emissions were not, at the time of measurement, offset by woody thickening in wetter areas, they may well be offset at some time in the future.

The 4D perspective indicates that the data in Jackson et al. (2002) fit ongoing sequestration, but it was not otherwise obvious because sampling was done over a small extent on the *t*-axis. The reanalysis indicates that with further time SOC under woody thickening is likely to increase, including in the higher rainfall regions. Consequently, the reanalysis portrays the data in Jackson et al. (2002)

as being in concord with the main body of work on thickening, i.e. that thickening engenders net C sequestration.

4.2. Example 2 – time and horizontal axes

The SOC efflux for deforestation over the 5–35 year experimental time span was 17%. That is higher than the 5.4% reported in Harms et al. (2005) where the spatially and temporally averaged absolute changes were used. Note that although we calculated ΔSOC as % year^{-1} (averaged over a specific time period), the rate is not constant and after a longer time period will approach an asymptote.

Additionally, Fig. 8 indicated that emission will most likely continue beyond the time period of the original study (Harms et al., 2005). This indicates that the ΔSOC corresponding to the asymptotic situation may be higher in magnitude than −17%.

Data for two example deforestation sites from Harms et al. (2005), re-processed using DA, are presented in Table 1 and the corresponding $SOC(t)$ is shown in Fig. 9. The DA for those sites varied more strongly with time since deforestation than did the AA, and generally DA showed a stronger influence on $\%\Delta SOC$ than did AA

Table 1
Data for two sites deforested for grazing in QLD. C_w is SOC of the woody state and C_g is SOC of the grassland state. k is the rate constant of exponential decomposition (Eq. (4)). Corresponding ΔSOC with time is in Fig. 9.

	Soil	Woody species	Date deforested	C_w (Mg ha^{-1})	k (year^{-1})	C_g (Mg ha^{-1})
Site 1	Vertosol	*Acacia harpophylla* (brigalow)	1972	168.6	0.004301	16.86
Site 2	Kandosol	*A. aneura* (mulga) and *Eucalyptus melanophloia* (ironbark)	1992	42.94	0.01031	4.294
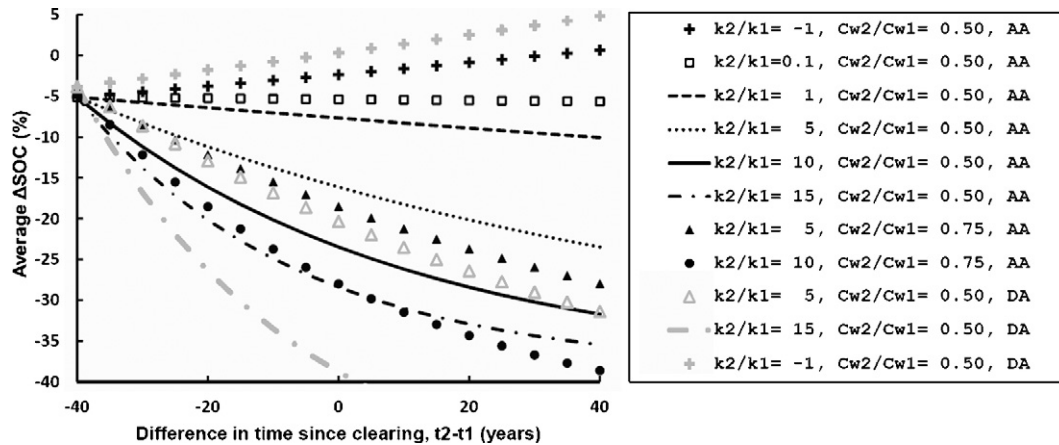
Fig. 10. Simulated examples of ΔSOC (%) with deforestation, from different spatial averaging methods, different levels of woody-phase SOC, and different decomposition rates ($k1$ and $k2$) (grey: disaggregated averaging, DA; black: aggregated averaging, AA). Note that there are strong effects on the spatial averages of ΔSOC data resulting from: (a) the relative initial SOC stocks, (b) the time since deforestation, and (c) different decomposition rates; presenting implications for data collection and analysis.

(Fig. 10), which must therefore be accommodated in data sampling and analysis.

The effect of appropriate representation in the x–y plane, through DA rather than AA, was significant. With the full data set: ΔSOC$_{DA}$ = −3.5% and ΔSOC$_{AA}$ = −5.4%. Harms et al. (2005) only reported ΔSOC$_{AA}$ – the latter figure. From our chronosequence subset, but without taking into account the time since deforestation: ΔSOC$_{DA}$ = −6.8% and ΔSOC$_{AA}$ = −5.5%. Taking into account both the t-axis and the x–y plane for the chronosequence subset: ΔSOC$_{DA}$ = −17%. That figure is closer to the 35% emission over 25 years reported for the upper soil surface, where decomposition rates are higher (Dalal et al., 2005).

5. Discussion

5.1. Implications of time dependency

The similarity between half-life of the slowest SOC pool, and the half-time for sequestration of SOC, reaffirms that long time spans are necessary for ΔSOC experiments. The 4D conceptualisation of biomass distribution and decomposition (Fig. 1) demonstrated that the representative SOC over 3D, at any one instance in time, is dependent on the historical and spatial distribution of tree mortality, and combined with the temporal conceptualisation (Fig. 2), reveals how SOC approaches its asymptote after multiple stand generations.

Apart from the timelines in the mathematical modelling presented here, other factors suggesting that deforestation studies require long extents on the t-axis are: (a) the short-term peak in SOC due to decomposition of the nascent root and surface debris (Figs. 2 and 4); (b) the short-term direction of SOC flux varies with location (the x–y plane) (Harms et al., 2005); (c) for native pastures the positive impact of deforestation on productivity may be quickly followed by reductions in ecologically beneficial soil properties (Kaur et al., 2007), which would lead to C emission in the long-term; and (d) extra emission may occur through the 'priming effect' (Fontaine et al., 2007) – when substrate limitations on microbial mineralisation of the more passive SOC are released by a fresh supply of more-easily assimilated root compounds (from nascent root litter). Deforestation studies also require long extents on the z-axis, especially where woody vegetation may access deepset ground water.

Different models can yield different trends for age distributions of SOC (Bruun et al., 2004) – with more complex models forecasting greater longevity of some SOC pools (Manzoni et al., 2009). Our results showed some slight differences between the 2- and 3-pool systems in the shape of their sequestration and emission curves and SOC age demographics. The logistic functions used here (e.g. to determine the breakeven period) modelled the output from a complex model (CENTURY, which includes for example, feedback between pools), and therefore our modelling of thickening (example 1) inherently included effects from such complexity. Possible long-term fluxes, beyond the time-span of measurement, were provided diagrammatically by the 2- and 3-pool models but modelling of those sites with the latest version of CENTURY could yield a more accurate forecast of long-term SOC flux.

5.2. Importance of 4D experimental integration

Both examples studied here showed the importance of controlling for multiple dependencies, which can operate on more than one axis and require extensive data collection. The use of mathematical models to determine the time span needed for ΔSOC experiments relies on existing information about the system. There is therefore a bootstrapping process (or 'power analysis') to measurement of ΔSOC.

Attention to representative sampling and data processing is applicable to all four axes. For example, both DA and AA are non-linear (and thus not interchangeable) with respect to the rate constant k (i.e. environmental variation in the x–y plane and also on the z-axis (Dalal et al., 2005)) and time t (Eq. (4)).

Affects of site history on SOC are interconnected in 4D, which must be accommodated, e.g.:

1. farmers deforesting the more productive (i.e. probably higher SOC) land first (Holmes et al., 2006) means that comparing C stocks in long-deforested land with those in more-recently deforested land (vestigial, less desirable), is inequitable;
2. although effects of thickening and land degradation may both be apparent simultaneously on the t-axis (e.g. Fig. 11), the site history (which often provides clues to degradation) must be considered; and

Fig. 11. Degradation with concomitant thickening (southern ironwood, *Acacia estrophiolata* F. Muel., central Australia), showing that multiple effects can be concurrent, although not necessarily started simultaneously. The site has a long history of overgrazing; thickening germinated after heavy rains following drought in the late 1960s.

Photo: C. Dean 2005.

3. substituting space-for-time can be inferior to long-term experiments, due to spatial heterogeneity, the likelihood of improperly paired sites (Harms and Dalal, 2003), and interconnections for SOC in 4D.

Longevity of SOC usually increases with depth due to, for example, apparent increased SOC recalcitrance, which may be due to substrate limitation, reduced decomposition activity, or specific mineral associations; rather than inherent differences in chemical structure of source material (Fontaine et al., 2007; Marschner et al., 2008). Also, in semiarid rangeland there is an effect from rainfall penetration with different pluvial duration – longer events (which are less frequent) allow more emission at greater depth whereas light rain only induces microbe-mediated emission at shallow depths (McLain et al., 2008). Longevity of SOC pools derived from roots varies not only with depth but between woody species and environments (Pansu et al., 2009). In some soils, bioturbation may mix the age cohorts of some forms of SOC, even to 2 m depth (Pressenda et al., 1996). At wider scales the C budget can be affected by transport, during erosion and groundwater transit for example, and then by dynamics at the deposition site (Greene et al., 1994; Lal et al., 2004; Hoffman et al., 2008).

Due to physical difficulties of measuring SOC under tree trunks or coarse lateral roots, sometimes bulk excavation must be employed rather than coring, for both coarse root and SOC sampling (Archer et al., 2002; Resh et al., 2003; King et al., 2007). It is unclear if near-stem regions were measured by Jackson et al. (2002) but coarse root biomass was not measured – if the reason for its omission was physical difficulty then that may have precluded measurement of the spatially representative SOC of thickening. The meta-analysis of thickening data in Jackson et al. (2002) also showed a negative correlation for ∆SOC with rainfall. However, the measurement depth there was only 1 m, whereas thickening species can manifest SOC to at least 2 m depth (as demonstrated in the empirical section of that study) – a *z*-axis dependence. Furthermore, a broader meta-analysis showed that SOC was consistently increased through woody thickening (Eldridge et al., 2011).

5.3. Wider implications

Lowering the carbon footprint of rangeland activities represents ecological engineering, for climate change mitigation. From the long timescales needed for ∆SOC determination there are wide implications for monitoring and verification of ∆SOC in rangeland rehabilitation projects. The assessment interval of 5 years in Kyoto verification is known to be unmanageable for ∆SOC determination for mesic, temperate land uses (Smith, 2004); and the present work shows that for some common events in rangelands, 30–90 years is too brief. This concurs with Piñeiro et al. (2006) who found grazing impact on ∆SOC to be definitive over several centuries but different in magnitude and direction at a decadal time-scale. This implies the need for proxies such as modelling or substituting space-for-time, although well-paired sites (with equal capacity for production) are often hard to find.

Arid land rehabilitation, can incur measurable SOC sequestration to 0.2 m after three decades under irrigated woody cover (e.g. Su et al., 2007). The carbon footprint of that activity is possibly borderline in the short term but after longer durations and deeper sampling it may well represent climate change mitigation. Tropical rainforest has been degraded to 'derived savannah', with ongoing degradation and biomass loss, but rehabilitation of such agricultural zones is aided by reforestation (Badejo, 1998). Modelling, such as that used here, would be necessary for speculative carbon credits which could add impetus to such projects.

Using a similar approach to that in the examples above, the discord in the literature over the sign of ∆SOC with rangeland deforestation for pasture can be re-examined. In reviews, the 4D breadth of the individual experiments is often unstated. One particular meta-review of ∆SOC with land-use change – Guo and Gifford (2002) – is often-cited (e.g. Goodale and Davidson, 2002; Grünzweig et al., 2007; Kirschbaum et al., 2008; Marin-Spiotta et al., 2009). One of its conclusions was that ∆SOC is negligible, or that sequestration is minimal, upon conversion from forest to pasture. Other reviews (McGrath et al., 2001; Murty et al., 2002) found ∆SOC = 0. This has serious implications for climate change mitigation when environmental management decisions (such as conversion of forest to pasture) hinge on finances of carbon balance (e.g. Harper et al., 2007). For example the lower figure derived by Harms et al. (2005) (than derived here) contributed to high uncertainty in a study of GHG budgets for grazing businesses with land rehabilitation (Bray and Golden, 2008). In Guo and Gifford (2002) a net emission of only 8% of SOC upon deforestation was found by combining 170 observations from 19 publications. We noted several factors contributing to uncertainty in that value:

(1) it used several reviews with an unknown number of samples, therefore, the relative weights applied to different "observations" may have been inequitable;
(2) several input data were from one Amazonian region where sampling was ≤0.3 m, and where the chronosequence most likely included preferential, earlier deforestation of land with higher initial SOC;
(3) their Fig. 2 showed that the trend of finding sequestration with deforestation was stronger if measured <0.3 m rather than for the full profile, nevertheless they averaged of all values;
(4) several of the sites had earlier undergone selective-logging or a crop phase (with fertilisation and accompanying SOC fluxes) before conversion to pasture;
(5) many pastures had sowing of exotic species, and intensive management, which makes comparison inequitable (as the forest had not received the same attention to productivity) and thus

extends the carbon accounting to a much wider geographical area.

Thus, there were a wide variety of previous land-use histories, all with unaccounted-for but most likely ongoing influences on SOC (due to long timelines for ∆SOC). Such influences can lead to unwarranted conclusions about ∆SOC if they are not accommodated in experimental design and analysis. Only close re-examination of past experiments, or carefully taking into account 4D factors in future experiments, will produce results suitable for reconcilable climate change mitigation. This is pertinent to reforestation, where SOC takes ∼2 millennia to recover after woody vegetation was absent for only ∼200 years (Figs. 2 and 4).

Both overgrazing and deforestation incur an initial peak in SOC followed by a long emission, through their effect on biomass (e.g. Irvine et al., 2007; Kirschbaum et al., 2008). Specifically, overgrazing decreases root volume (Villamil et al., 2001), giving a pulse of dead biomass to the soil, but decreased inputs in the long-term, when biomass ceases to be replenished. Overgrazing of woody vegetation (e.g. erosion and gradual decline and loss of riparian vegetation) is slower and patchier than broad-scale deforestation, and land rehabilitation after overgrazing can be arduous (Christian and Perry, 1969). Therefore, the extent of the t-axis used in experiments studying ∆SOC with overgrazing (and land rehabilitation) should be longer than that used for studying more spontaneous deforestation and reforestation.

6. Conclusion

The process of applying comprehensive 4D sampling and analysis, for quantifying rangeland SOC fluxes, revealed important implications for presently or previously timbered rangelands; such as: higher magnitude fluxes and longer time-spans for unambiguous measurement. Specifically, through fundamental mathematics we showed that: (a) although thickening will not necessarily increase total SOC stocks in all situations, there is as yet, insufficient evidence to suggest that SOC will decrease in the long-term in higher-rainfall zones with thickening and (b) the SOC efflux with deforestation is likely to be higher and continue for longer than previously estimated. Earlier findings on thickening or deforestation appear no longer contradictory when one considers that they only presented views from different sections of the 4D axes, some being too small in span on at least one axis. The necessary experimental and analysis methods described here, and their implications for C budgets, can be applied to property-level, regional and national carbon accounting. Due to higher uncertainty in ∆SOC in the more-extreme rainfall zones, the acceptable annual rainfall range for carbon trading with the Chicago Climate Exchange is currently 330–1016 mm year^{-1} (CCX, 2009). With appropriate sampling in 4D it is possible that uncertainty in ∆SOC with change in land management may lessen in the even more-xeric rangelands. Application of the suggested approach should generally decrease uncertainty and so attenuate the impasse on ∆SOC forecasts for timbered rangelands, subsequently engendering financial viability in rangeland C trading and more reliable national GHG accounts, and in turn provide for more-serviceable climate change mitigation.

Acknowledgements

Appendix: self-thinning in woody thickening.

When discussing the timelines of woody thickening and the accompanying ∆SOC it is necessary to consider underlying ecosystem dynamics. After the onset of thickening, in situations where the environment has supported a high, initial stand density of seedlings and saplings, then stand density gradually decreases, due to competition for resources. The juvenile trees (or shrubs) (Fig. 1) that are outcompeted die and survivors become larger – the biomass of both the survivors and the entire stand increases. Stand density usually decreases logarithmically. This process of stand development, known as 'self-thinning' forms a foundation of the mathematics and ecology common to forest science. Self-thinning is not often considered in carbon accounting or ecological studies in the rangelands but it has been observed in woody thickening in the rangelands (e.g. Felker et al., 1990; Boutton et al., 1998; Fensham and Fairfax, 2005). Additionally, size-age distributions, which fit the dynamics of self-thinning (although not identified as such) appear in several reports on rangeland shrubs and trees (e.g. Adams, 1967; Crisp and Lange, 1976; Eldridge, 1988; Watson and Holm, 1994; Zerihun et al., 2006; Krull et al., 2007).

Drought, overgrazing and poor site quality can all reduce net growth and thereby stymie self-thinning (Horne and Robinson, 1987; Florence, 1996; Svejcar et al., 2008). In these cases the self-thinning process may stall or appear to be absent, and a strong perturbation may be necessary to initiate faster change in stand density (e.g. destocking, flood or mechanical damage (although some such perturbations may also precipitate a second thickening germination event)).

Self-thinning is more obvious where germination events are infrequent. This is because repeated germination events prior to stand maturity of the oldest age cohort will create (a) multiple self-thinning stands that overlap in both time and space and (b) competition between different age cohorts. With temporal overlap, a mixture of mature, dead and early-stage-thickening may prevail (e.g. Grice et al., 1994; Fensham and Fairfax, 2005).

Three major effects of self-thinning that must be considered when measuring ∆SOC are: (a) SOC will be more concentrated where the roots of the thinned individuals once were (e.g. Chabbi et al., 2009); (b) higher concentrations of SOC will accrue after the larger individuals have senesced; and (b) it may take several cycles of germination and senescence for the maximum, stand-wide distribution of SOC to accumulate.

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