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Optimising carbon sequestration in arid and semiarid rangelands

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ABSTRACT

Destocking degraded rangeland can potentially help climate change mitigation by re-sequestering emitted carbon. Broad-scale implementation has been limited by uncertainties in the magnitude, duration and location of sequestration and the profitability relative to the existing grazing land use. This paper employs a novel methodology to assess potential rangeland sequestration and its profitability, using 31 Mha of rangeland in New South Wales, Australia as a case-study. This approach combines remotely sensed data and modelled estimates of various components. Remotely sensed, synthetic aperture radar data were used to determine woody biomass of minimally degraded forest (benchmarks) and neighbouring more-degraded forest, followed by sequestration modelling using non-linear growth rates based on woody thickening and slow-growing plantations, scaled to the benchmarks. Livestock concentration and livestock-based farm profits were modelled. We compared sequestration and grazing net profits, for a carbon price of AUD\$10 Mg^{-1} CO_2-e, at different growth stages for different levels of forest attrition. We found that broad-scale destocking with subsequent C re-sequestration was initially unprofitable compared with grazing. However, after 50 years, with full costing of C emissions, the returns were similar for the two alternatives of continued grazing or re-sequestration, for areas with biomass below benchmark levels. Reforestation of recently deforested land represents the most profitable option with profitability increasing with growth rate. Emissions of soil organic carbon, set in motion by climate change over the next century, were calculated to be the largest of all sources. Emissions from biomass, induced by climate change, will be higher where vegetation cannot adapt. The secondary effects of climate change will reduce re-sequestration and grazing profits, possibly limiting the carbon stored by re-sequestration projects.

1. Introduction

Rangelands supporting commercial livestock grazing are contested ground in which production of meat and other animal products for the increasing human population, nature conservation and conserving or replenishing carbon stocks increasingly compete for space (e.g. Glenn et al., 1993; Schuman et al., 2002; Reid et al., 2004; Dutilly-Diane et al., 2006; Khan and Hanjra, 2009; Janzen, 2011). These rangelands have commonly experienced net vegetation and soil loss (e.g. Allen, 1983; Fanning, 1999; Zucca et al., 2010; Dotterweich, 2013) corresponding to net C (carbon) emission. Reversal of land degradation linked to that carbon emission process can theoretically replenish the lost C (Howden et al., 1991; McKeon et al., 1992; Glenn et al., 1993; Walker and Steffen, 1993; Henry et al., 2002). The refilling of that depleted carbon stock (henceforth termed re-sequestration) contrasts with sequestration projects storing C in a form or location different to its origin (e.g. afforestation or power-station carbon capture and storage). Uncertainties in the potential magnitude, duration, location and profitability of carbon re-sequestration projects, have limited their implementation.

Rangeland emissions can be lessened by reduced deforestation, protection and enhancement of soil organic carbon (SOC), and by reforestation (Henry et al., 2002). Reforestation can be intensively managed (e.g. plantings), or passive/'natural' (e.g. Rey Benayas et al., 2007; Grainger, 2009) by allowing woody thickening (i.e. infill, Rackham (1998)) and regrowth to mature. The passive type is considered here, though managed reforestation can be used if finances permit. SOC stocks are generally positively correlated with aboveground biomass (Jackson and Ash 1998; Harms et al., 2005; Young et al., 2005; Wynn et al., 2006), being primarily

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derived from root turnover and litterfall. This relationship accounts for some of the decline in SOC stocks with vegetation attrition (Dean et al., 2012a), the remainder being through erosion pathways (Dean et al., 2012b). Magnitudes of change in soil organic carbon (ΔSOC) have higher uncertainty than associated changes in rangeland biomass (Henry et al., 2002). Consequently the present work focuses on biomass but with discussion of linked ΔSOC.

Rangeland C re-sequestration opportunities coincide with overgrazing or deforestation. Localised benchmarks of potential C stocks can be derived from remnant ecosystems or spatially dependent environmental variables (Greve et al., 2013). This equates to determining 'carbon carrying capacity' (Roxburgh et al., 2006). The potential of plantations to replenish C on deforested semiarid to mesic arable land in southern Australia was estimated by Paterson and Bryan (2012). Our work is thematically similar, but we use a finer spatial scale, simulate natural (autonomous, unmanaged) regrowth, allow a longer duration, and to reflect the higher error margins in rangeland, we calculate at a coarser economic scale.

Remote-sensing calibrated by ground-truthing, or, more frequently, ground-based assessments alone, are employed in rangelands for regular land condition assessments and woody biomass monitoring. Adaptation of remote-sensing technology is slowly approaching a level suitable for routine operational usage over the large expanses for which it was originally intended (e.g. Graetz et al., 1976; Mackay and Zietsman, 1996; Ustin et al., 2009). Both LANDSAT and the Advanced Land Observing Satellite (ALOS), phased array L-band synthetic aperture radar (PALSAR) sensor have proven applicability for aboveground biomass assessment of arid and semiarid open woodland (Armston et al., 2010; Lucas et al., 2010) with radar more sensitive to woody biomass and LANDSAT more sensitive to vertical foliage distribution (Armston et al., 2009; Danaher et al., 2010). PALSAR has proven applicability for carbon flux assessment in complex situations, though the basic radar data are often integrated with other data types, such as LiDAR or LANDSAT, or undergo more complex processing (e.g. Carreiras et al., 2012; He et al., 2012; Sarker et al., 2012). Here we principally use pre-processed data from PALSAR, and compare results with those from the lower resolution NOAA-AVHRR sensor.

Managed reforestation, including rehabilitation of degraded rangeland to enable woody regrowth where there has been substantial top soil loss, may require financial inputs (Spooner et al., 2002; Sparrow et al., 2003; Mengistu et al., 2005; Neff et al., 2005; Polglase et al., 2013). No financial inputs would be necessary if a low C sequestration rate, similar to that for passive reforestation of degraded and grassy areas by natural regrowth and 'woody thickening', can be applied.

Our main aims in this paper are to determine the most lucrative places in rangeland for C re-sequestration, and to develop a method for determining the C re-sequestration potential and rates for those areas. We apply this to rangeland in New South Wales (NSW) Australia, which is used largely for the generation of profit from grazing domestic livestock (henceforth termed commercial rangeland). An understanding of the relevance of our findings to rangeland outside of the study area is facilitated by a global climate and biome comparison. We discuss options for avoidance of any carbon emissions leakage after destocking. We use a notional carbon price for comparative purposes, fully realising that there is a long way to go before markets for carbon and rules for accessing such markets gain widespread acceptance.

2. Methods

2.1. Terminology and definitions

The boundary of the Australian rangeland zone has been variously mapped (Donohue et al., 2005). The definition for rangeland that we adopt is areas where domestic livestock 'rove at large' (Chambers, 1908) in natural or semi-natural vegetation inhospitable to arable agriculture — a subset of the 661 Mha rangeland zone of Donohue et al. (2005). After exclusion of reserves and non-pastoral uses, the remaining 369 (±5) Mha is commercial rangeland (Dean et al., 2012b).

The definition of forest we use is that of the Australian Government (DCCEE, 2010): a stand of trees covering at least 0.2 ha, attaining at least 2 m high at maturity and with at least 20% projected canopy cover. A projected canopy cover of 20% corresponds to approximately 11% foliage projected cover — a threshold used to delineate forest cover by remote-sensing (Scarth

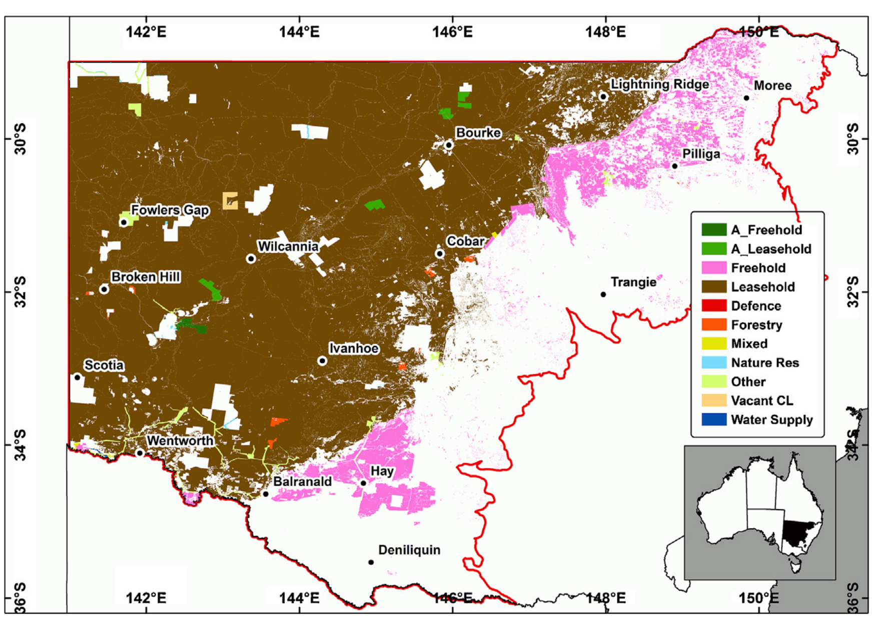

Fig. 1. Distribution of land tenure in the NSW commercially grazed rangelands. Rangeland zone boundary = red line. Abbreviations: 'A': Aboriginal, 'CL': crown land, 'Res': reserve, 'Mixed': multiple-use public land. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

et al., 2008). This definition extends the concept of forest to vegetation previously considered in Australia to be woodland, scrub or shrubland (Specht, 1972). We use the term 'forest degradation' to refer to both gradual attrition of a forest's woody biomass (whether through erosion from overgrazing, thinning, or reduced water infiltration from weeds or loss of debris etc.) or immediate deforestation, such as in 'land clearing' and 'fence-line clearing'.

Most spatial data on the weights of dry biomass are available for aboveground only. For total tree carbon in biomass we assume that the root:shoot ratio is 1:3, i.e. 25% is belowground, following Henry et al. (2002). This is a conservative value compared with other literature values for arid and semiarid ecosystems (Mokany et al., 2006).

The amount of carbon (C) per unit area (Mg ha^{-1}) is the carbon two-dimensional (vertically projected) 'density', or unit-area carbon stock. Change in carbon density is via a flux — mass movement per unit area per unit time. Of the three major organic C pools: vegetation biomass (above and belowground), SOC, and debris (e.g. coarse woody debris), we concentrated on vegetation as it is the easiest to measure by remote-sensing and the fastest to be influenced by land management.

In our calculations dry biomass was considered to contain 50% carbon, by weight. The measured fraction can vary from 42% to 61%, depending on species, plant component and environment (Thomas and Martin, 2012) but 50% is a commonly used proportion (Gifford, 2001), and is a compromise between the IPCC's 47% (Aalde et al., 2006) and the effect of the likely contribution from volatile compounds lost during typical assay (Thomas and Martin, 2012). Our data (Dean, unpublished data) shows proportions of C close to 50% for semiarid *Acacia* species.

The value we used for the 'price of carbon' was AUD\$10 Mg^{-1} CO_2-e (CO_2 equivalent). It is close to the Climate Action Reserve's (USA) value of AUD\$11.50, the European Union's ETS value of AUD \$7.94, and China's Shenzhen value of AUD\$8.02 (prices as of January 2014). The value of AUD\$10 Mg^{-1} CO_2-e equates to a price for C of AUD\$36.641 Mg^{-1}. Note that the 'carbon price' can vary depending on the market in which it is traded, the source of the carbon, and over time in response to both policy decisions and supply and demand. Carbon masses used herein are in terms of C, not CO_2-e (unless otherwise specified).

2.2. Study region and data processing

All area calculations were performed in Albers Equal Area projection, as it provides minimal distortion over large areas (Steinward et al., 1995). The number of significant digits in tables in the present paper is rounded to represent error margins but double and floating point precision was used in calculations, to prevent accumulation of rounding errors.

Areas of commercial rangeland within NSW were determined from land-use data ('LanduseV1', Anni Blaxland Fuad, NSW Office of Environment and Heritage, personal communication, 2011) containing 150 land-use classes (Fig. 1, Supplementary information

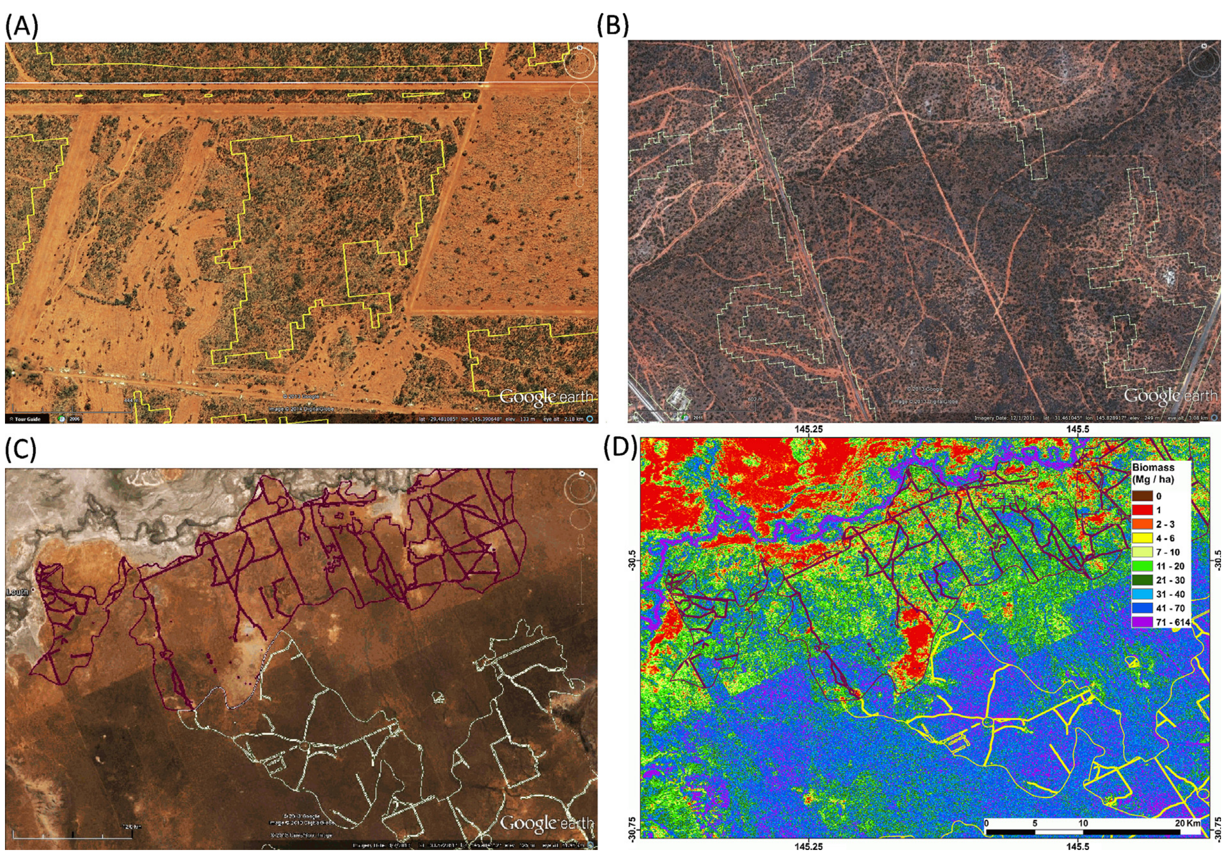

Fig. 2. Example degradation of remnant ecosystems (biomass benchmarks). (a) Location: ~86 km NW of Bourke, Lat/Long −29.523°/145.406°. Google Earth© Inc image of 23-Sep-2006, showing deforestation, livestock tracks, vehicular tracks, grazing land (deforested and non-deforested), roads and farm infrastructure, and typical forest attrition within the ecosystem remnant. (b) Higher level attrition of remnant ecosystem, centre: −31.460848°/145.829238°. (c) Centre: [−30.554402 S, 145.472336 E]. Nearly the whole ecosystem 'Mid-Darling Plains' (coloured burgundy), the closest to the river, has suffered substantial attrition. Parts of the neighbouring ecosystem 'Cobar Downs' (coloured yellow) were therefore incorrectly assigned higher potential biomass than the 'Mid-Darling Plains' to the north. (d) The ALOS data show that the fenceline contrasts are due to differences in woody biomass, which is confirmed in the Google Earth© closeup. (e) Centre: [−30.622615, 145.18126], closeup of left of (c). The fenceline contrast is due to a loss of mature trees to the NW of the fence-line. The Google Earth© finer-detail image shows the loss has been replaced by a multitude of smaller trees or bushes (i.e. 'woody thickening') (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

Table A.1). Other countries and other Australian states with similar ranges of annual rainfall and average temperature (i.e. potentially similar climate), and similar biomes, to our study area in NSW were identified using GIS. (see Supplementary information).

Within the greater rangeland zone for NSW, values for environmental attributes, ecosystem-type (from Mitchell Landscapes data, described in e.g. Gibbons et al. (2009)) and extant SOC to 0.3 m (Barson et al., 2002) were assigned to 62,371 polygons of conterminous land-use type (using ESRI ArcMAP GIS), ranging from 1 to 780,000 ha, with an average size of 490 ha. Polygons smaller than 1 ha were in total <1% of the total area and were discarded. Where single polygons overlapped a range of environmental values, then either an area-weighted average or majority was assigned, for numeric and thematic data, respectively. NSW rainfall and temperature data were mean annual averages from 1961 to 1990, with a pixel size of 0.025° (~25 km) (Bureau of Meteorology). Spatial data on deforestation, as determined from LANDSAT imagery with 25 m pixels, were from NSW Office of Environment and Heritage (Supplementary information Table B.5). The recently deforested land (from 1989 to 2007) was selected for viewing correlations between biomass and environmental variables because it avoided corruption of the correlation by interstitial grasslands. Data for deforestation prior to 1989 did not include dates and were grouped with other forms of forest degradation. Representative vegetation contrasts observed in remotely-sensed imagery were qualitatively examined on-ground in a subset of areas state-wide, between 1996 and 2006. The locations included long-term grazing exclosures, a carbon sampling experiment, and numerous roadside contrasts observed during journeys for such work.

Representative areas that theoretically had minimal attrition of vegetation — the benchmarks — were selected from 'remnant' ecosystems by a rigorous process of detecting and eliminating land that had been anthropogenically denuded of woody vegetation (Fig. 2, Supplementary information Fig. B.4, B.5). Infrastructure, urbanisation, and roads were subtracted, all with buffer zones due to the possibility of pixel misplacement and because neighbouring vegetation was usually depleted. Buffer distances were: deforestation 50 m; minor roads 20 m; major roads 50 m; homesteads and towns 500 m. There were 226 ecosystem types with remnant vegetation, covering 28 Mha of the greater rangeland zone in NSW.

Data on potential biomass were available in two forms: a (~5 km pixel) NOAA-AVHRR-derived layer from Berry and Roderick (2006) and a higher resolution layer ($0.0025 \times 0.0025°$, i.e.~250×250 m pixels) from the Commonwealth Department of Climate Change and Energy Efficiency, as used in the National Carbon Accounting System (NCAS) (Richards and Brack, 2004; Waterworth and Richards, 2008). Data on extant aboveground biomass were also available in two forms: a (~5 km pixel) NOAA-AVHRR-derived layer from Berry and Roderick (2006); and ALOS PALSAR radar data (acquired in 2009) pre-processed to yield extant, aboveground, woody biomass (50×50 m pixels) (Armston et al., 2010; Lucas et al., 2010) (from NSW Office of Environment and Heritage). The results from NOAA-AVHRR and PALSAR data were compared.

The ALOS (extant) and NCAS (potential) biomass values were compared for remnant ecosystems, i.e. where they should have been equal, and aberrations were interrogated. Areas of obvious incongruence, e.g. where canopy loss was >50% or infrastructure was present, were eliminated. The same buffer distances as described above for benchmark detection were applied. (Fig. 2, Supplementary information Fig. B.4, B.5.) Much of the fence-line clearing for 'infrastructure' was ~60 m wide (Fig. 2). Only about half such fence-line clearing was delineated as deforested in the LANDSAT-derived data. Ground-truthing showed that it was used for livestock grazing or mustering. It was delineated as rangeland grazing in the land-use data.

Many of the benchmark areas had reduced woody biomass, with numerous minor [unlisted] roads, vehicular tracks, intentional thinning, overgrazing and other forms of attrition (Fig. 2, Supplementary information Fig. B.4. B.5). These complications were too pervasive to subtract from representative remnant ecosystems and it was recognised they would affect calculation of potential biomass. The degree of biomass attrition varied between benchmarks. Benchmarks near major rivers were particularly depleted, as confirmed from ALOS data and Google Earth© imagery of fence-line contrasts (e.g. Fig. 2). Some were so uniformly and strongly below their potential, that their extant biomass (from ALOS data) appeared lower than their non-riparian neighbours, and thus caused the highest error margins in the calculations. Some highly degraded areas had some loss replaced by a large number of smaller trees or shrubs, i.e. 'woody thickening' (Fig. 2), which is often linked to overgrazing during droughts that are followed by heavy rains (Allen, 1983; McKeon and Hall, 2001; Laliberte et al., 2004; Svejcar et al., 2008). Variability within benchmarks added to error margins.

Ongoing annual emissions for the NSW rangeland were estimated from the annual emissions calculated in Dean et al. (2012b) for Australian commercial rangelands, apportioned on an area-basis for NSW. Firstly though the deforestation in QLD (20.47 Tg year^{-1}) and savannah burning (as deployed in tropical Australia) were subtracted. The estimate of (Dean et al., 2012b) did not include the NSW deforestation effects calculated here, which were added in. Also the efflux with general degradation of national Mulga Lands (Dean et al., 2012b) was apportioned to the 6.319 Mha of Mulga Lands in the NSW commercial rangelands. The difference in potential concentration of biomass between the QLD and the NSW Mulga Lands was calculated from the NCAS layer (QLD:NSW ratio of 1.29:1) and applied to the apportioning of Mulga Lands degradation for NSW.

SOC emission with deforestation for NSW were estimated on a pro-rata area basis from the QLD sums and the ratio of QLD:NSW SOC concentration (1.15) in Dean et al. (2012b), and compared to the −17% of SOC emitted over 30 years from Dean et al. (2012a) for QLD deforestation. That figure was applied to typical NSW deforestation rates over 50 years to get an annual SOC efflux.

The long-term effect of continued anthropogenic soil erosion and the benefit of land rehabilitation was modelled and graphed for Australian rangeland in Dean et al. (2012b) and those values can be approximated pro-rata, for NSW. Land rehabilitation does not

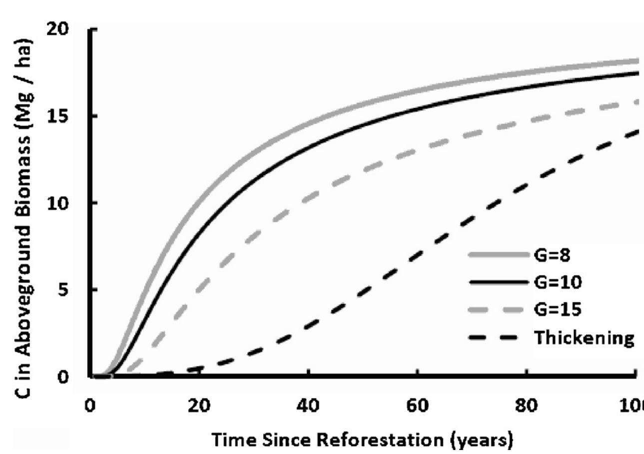

Fig. 3. Possible growth curves for the recently deforested 80,667 ha of commercial rangeland in NSW, showing effect of variation in year of maximum growth (G) for Eq. ((1)), and use of the lower rate 'Thickening' (Eq. (2)) modelled on uneven-aged stand development via woody-thickening. The three higher rates are based on growth curves for plantations in southern Australia (although scaled to local potential biomass).

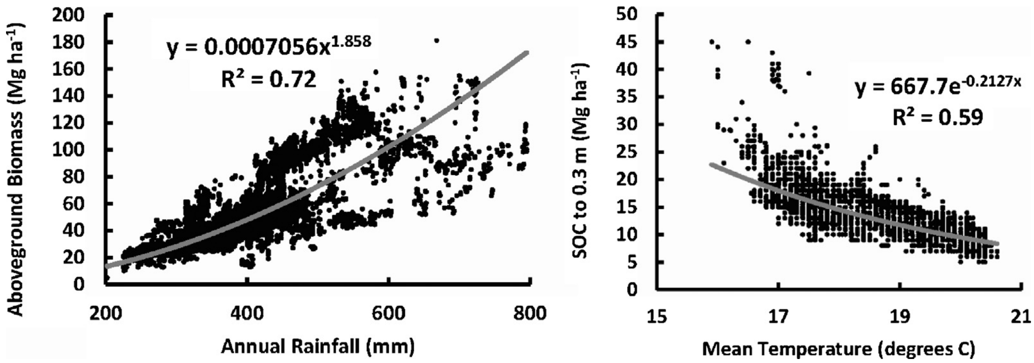

Fig. 4. Relationships between potential biomass, SOC and environmental variables, for land deforested from 1989–2007.

immediately stop soil erosion. If rehabilitation (most likely requiring financial input) was started in 2014, then the total SOC emission (to 0.3 m depth) due to commercial grazing to date would be 110 ($\pm$40) Tg, peaking in 2400, including 99 ($\pm$40) Tg after 2014. The true error margins for these estimates are high, possibly around 90%. The SOC emissions with rehabilitation beginning in 2014 and all anthropogenic erosion curtailed by 2053 were calculated and compared with emissions with business as usual (i.e. with current erosions rates).

2.3. Biomass growth modelling

Growth modelling of rangeland recovery suffers from insufficient data (Dwyer et al., 2009). Linear growth is often assumed, despite acknowledged variation with age; and only a single rate of sequestration is typically reported (e.g. Bray and Golden, 2008; Dwyer et al., 2009; Fensham and Guymer, 2009; Dean et al., 2012b). The non-linear growth formulation typically used in even-aged forestry (e.g. Waterworth et al., 2007) has been used to model reforestation by semiarid plantations (Paterson and Bryan, 2012). The function has two dependent variables (the potential biomass and the age of fastest growth, called 'G' — the inflection point of the growth curve). It could be used to model thickening and rangeland reforestation (slow, uneven-aged growth), with adjustment of G:

$$\text{currentC} = \text{potentialC} \times \exp\frac{-(2G - 1.25)}{t} \tag{1}$$

where currentC is the C (in Mg ha^{-1}) at time t (in years), potentialC is the maximum C attainable at the site (the benchmark in Mg ha^{-1}), and G is the inflection point of the curve — the age of maximum current annual increment for the stand (CAI) (in years). A range of values of G were tested here.

Fitting Eq. (1) to the thickening work of Hibbard et al. (2003) and Chew and Chew (1965) gave values for G of 29 and 89 years, respectively, even though G for individual trees in the latter study was 11 years (i.e. closer to that for plantations). The data of Kroeger et al. (2010) gave a value of G near 80 years for reforestation of oak savannah in Californian Mediterranean rangeland. Notably though, the function designed for high stocking rates and even-aged germination (i.e. Eq. (1)), did not fit the reality of thickening well. A better fit for stand growth with thickening was found using a more-flexible logistic function (Dean et al., 2012a) and that is applied here, in addition to Eq. (1), to determine sequestration with reforestation:

$$\text{currentC} = \text{potentialC} \times \left(1 - \frac{1}{1 + (\text{t}/\text{b})^{\text{c}}}\right) \tag{2}$$

where currentC is the C (in Mg ha^{-1}) at time t (in years), potentialC is the maximum C attainable at the site (the benchmark in Mg ha^{-1}), b is the time (in years) when half the potential is reached, and c is a dimensionless parameter that determines the steepness of the curve around the inflection point. Parameter values that fit the empirical and modelling work of Hibbard et al. (2003) and Chew and Chew (1965) (in Texas and Arizona respectively) on woody thickening are b = 75.2799 yr, and c = 2.8389 — those values were used herein (Fig. 3).

In both Eqs. (1) and (2) the biomass (as a function of age) is linearly proportional to the potential biomass. Consequently the spatial average flux, across different potentials, is linearly proportional to the spatial-average potential. That simplifies the calculations, with all sequestration rate curves, for differently sized areas, being linearly proportional. For calculation of re-sequestration the potential biomass of benchmarks was deduced from ALOS data only, except for areas recently deforested, where the average of ALOS and NCAS was used, as there the extant remnant vegetation was considered less representative.

2.4. Livestock modelling

In order to gauge opportunities from livestock stocking-level adjustment, spatial data on livestock carrying capacity and extant stocking levels were sought. The most recent data, supplied by David Burcell (NSW Livestock Health and Practices Authority (LHPA), personal communication, 2011) pertained to 21.2 Mha of the 30.57 Mha of NSW commercial rangeland, on one voluntary census night in 2010. Relationships were examined between notional (i.e. advised carrying capacities) and actual stocking rates and between these and environmental variables. Those relationships were spatially interpolated to provide stocking levels for areas without stocking data. The census data may have been subjective to some degree, and included such aberrations as stock from one large and frequently used paddock mustered into a smaller and less-frequently used paddock (or even shipped elsewhere at the time), which would for example weaken the correlation between biomass of a large paddock and its average stocking level.

NSW rangelands are currently mostly stocked with sheep, goats and cattle. Livestock census data were converted into 'dry sheep equivalents' (DSE) per hectare (with 1 DSE being one 50 kg wether or non-lactating ewe).

Data on farm [net] profits were obtained from Salahadin Khairo (Department of Primary Industries NSW, personal communication; 2011) and Khairo et al. (2008). These data were from three typical properties in different parts of the NSW rangelands. They included all typical running costs and incomes, apart from inputs from government grants, subsidies and allowances. Data for the three typical farms were averaged. The mean profit on an area basis was \$3.8 ($\pm$1.8) ha^{-1} year^{-1}, the mean profit per livestock unit was \$11.5 ($\pm$2.6) DSE^{-1} year^{-1} and the average stocking rate was 0.33 ($\pm$0.17) DSE ha^{-1}.

The methane emissions from all types of livestock were converted to mass per DSE per unit time. A conservative estimate of the methane emission per DSE was 11.1 g DSE^{-1} day^{-1}, based on literature values (Leuning et al., 1999). With methane having a global warming potential of 23 times that of CO_2 over a 100 year time frame (reports citing 21 to 25 (IPCC, 2007; UNFCC, 2014)), the presence of one DSE year^{-1} equates to a C efflux of 0.025449 Mg DSE^{-1} year^{-1}. That conversion factor was used in estimates of avoided emissions upon destocking. A carbon price of AUD \$10 Mg^{-1} of CO_2-e equates to a methane emission cost of \$0.93 DSE^{-1} year^{-1}.

Our comparisons between carbon and grazing profits were not adjusted for discount rates as the sequestration mechanism requires minimal investment and the timing of potential income or fines for carbon flux is a matter for policy construction. Also, species that reforest without planting (passive reforestation) (we used woody-thickening growth rates) are more likely to be acclimatised to the local fire regime and will not be as easily killed by fire as are many plantations, thereby lowering the need for one of the discounts typically applied to plantation schemes.

3. Results

3.1. Climate and livestock

The main biomes in NSW are temperate (savannahs, grasslands, or shrublands) with lesser amounts of Mediterranean (forest, woodland and scrub) and xeric biomes (Supplementary information, Fig. D.24). Approximately 25% of the Australian commercial rangelands similar biomes and climate to those of NSW (Supplementary information Table D.7, Fig. D.24, D.25). The most widespread biome in NSW is the temperate grasslands, savannahs or shrublands. Significant areas of that biome are also in QLD (Australia) and Texas (USA). The areas of xeric and Mediterranean biomes in NSW are approximately equal. The ecosystem distribution data revealed that only ~1 (±0.3)% of NSW commercial rangelands are natural grassland and that even those have scattered trees or groves of trees or saltbush, which concurs with the historical qualitative analysis (Allen, 1983). In terms of absolute areas, after NSW the next biggest area of similar Mediterranean biome is in northern Africa. Mexico and southern Africa have the most area of similar xeric biome. Countries with significant portions of their rangeland similar to NSW are: Syria with temperate shrublands, grasslands or savannahs; Malta with Mediterranean forests, woodlands, or scrub; and Botswana with desert, or xeric shrublands.

For land in our study area deforested from 1989–2007, potential biomass and rainfall were closely related both increasing from west to east, and SOC and temperature were closely related, both increasing from north to south (Fig. 4, Supplementary information Fig. B.7). As the time since deforestation was short compared with SOC halflives, these correlations would have resulted from the woody vegetation state rather than the post-deforestation, grassland state.

Average annual rainfall, temperature, aboveground dry biomass, and SOC (to 0.3 m) (all area-weighted) for the NSW commercial rangeland are 320 mm year^{-1}, 19°C, 36 Mg ha^{-1}, and 10.3 Mg ha^{-1} respectively. Approximately half the area is arid (47%), the remainder semiarid (Supplementary information Fig. D24).

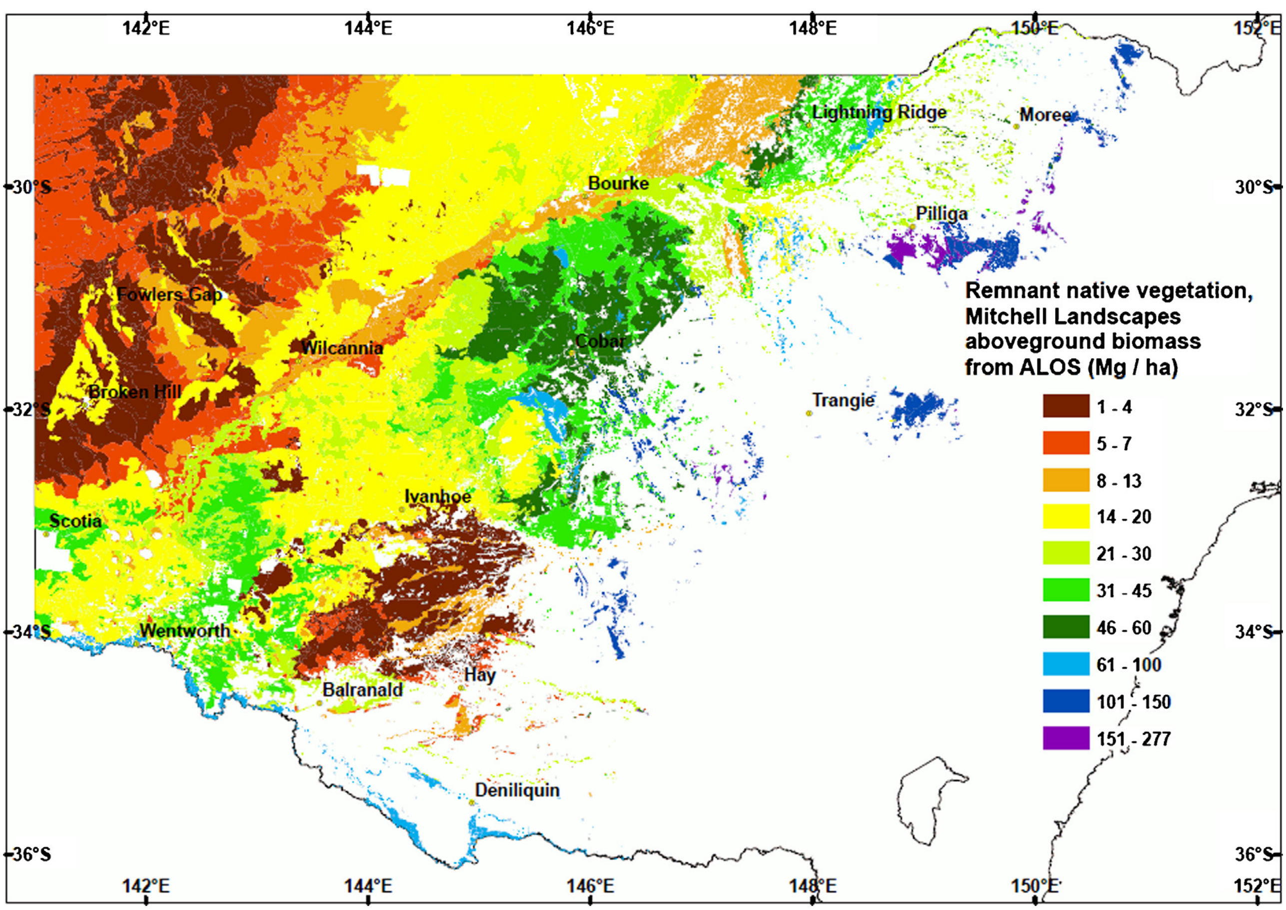

Fig. 5. Representative aboveground biomass from ALOS data, for the remnant ecosystems, grouped by ecosystem-type (Mitchell Landscape).

There was a positive relationship between notional carrying capacity and rainfall (Supplementary information, Fig. B.9). A lower correlation was found between actual stocking levels and rainfall. The notional carrying capacity formula is:

$$NCC = 5.992 \times 10^{-8} rain^{2.6808} r^2 = 0.66 \quad (3)$$

where NCC is in units of DSE ha^{-1} (dry sheep equivalent per hectare) and rain is the mean annual rainfall in mm. And the actual livestock stocking rate formula is:

$$DSEa = 6.338 \times 10^{-8} rain^{2.6005} r^2 = 0.31 \quad (4)$$

where DSEa is in units of DSE ha^{-1} (dry sheep equivalent per hectare) and rain is the annual rainfall in mm.

On average the actual stocking rates were 30% below notional capacity. Note however that only ~two thirds of livestock managers reported their actual livestock numbers, so the relationship could differ when including the whole State if those who did not report stocking had a particular management style.

The total number of DSE over the 30.57 Mha of NSW commercial rangelands is 7.1755 million if the land is stocked at the actual stocking level, or 10.895 million if the land is stocked at the notional-level (averaging 9 (±2) million). That corresponds to 0.2347 and 0.3564 DSE ha^{-1}, respectively for the actual and notional stocking levels per hectare, averaging 0.29 (±0.06) DSE ha^{-1}, which is close to the average of 0.3307 DSE ha^{-1} from the three regional examples in Khairo et al. (2008). The net profit from the livestock NSW-wide, assuming an average farm, is given by: 9.0 (±1.2) million DSE multiplied by \$11.5 (±2.6) DSE^{-1} $year^{-1}$ equalling \$100 (±30)M $year^{-1}$.

3.2. Biomass change

Over all land uses in NSW, the change in aboveground biomass, as determined using the ALOS data and remnant ecosystems, was −867.98 Tg, i.e. 434 Tg of C emitted. For commercial rangeland within that area the change was −51.695 Tg (i.e. −26 (±9) Tg of C (Figs. 5–7 a). The nominal 30% error margin is based on spatial variability within ecosystems. Using a root:shoot ratio of 1:3 (25%) the net emitted C in total biomass was then 35 (±10) Tg. When using the NOAA-AVHRR data in place of the ALOS data the net emitted C in total biomass was 35.50 Tg. These are similar results given that the error margin of current biomass stocks from Berry and Roderick (2006) is in the order of 15% nationwide but up to 50% in some locations (Dean et al., 2009).

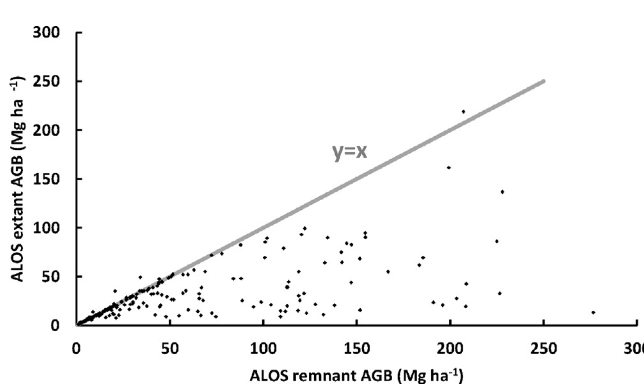

Fig. 6. Aboveground biomass (from ALOS) for the ecosystems: average-extant (present) versus remnant. As expected due to forest attrition with commercial grazing, the average-extant biomasses are mostly below the remnant values. A few ecosystems presently have higher biomass on average than in their remnants, mostly due to the purely 'riparian' land-use category not being classed as 'remnant' within the 'rangeland' land-use category (confirmed in Google Earth imagery).

For remnant ecosystems, the ALOS biomass values were mostly lower than the NCAS values, on average by 14% (Supplementary information Fig. B.14).

If instead of the net change in biomass from the ALOS analysis, only land below the remnant biomass values is assessed, namely 17.90 Mha the emitted C from aboveground biomass is 61 (±18) Tg (Table 1, Supplementary Table A.2). That value constitutes an amount that could be sequestered, by bringing that 17.901 Mha up to the remnant values for each ecosystem type. It is likely that somewhere between the net and the deficit alone could be sequestered in aboveground biomass, i.e. 26–61 Tg of C. If root biomass is included that range is 34–81 Tg, i.e. 58 (±24) Tg– similar to the value of 41 Tg derived from lower resolution data by Dean et al. (2009).

To allow revegetation, an estimated 4.49–6.82 MDSE would need to be removed (at actual and notional stocking rates respectively). This corresponds to a reduction in profit of \$65 (±34)M $year^{-1}$ at the average stocking rate (using the average farm profit of \$11.5 (±2.6) DSE^{-1} $year^{-1}$). Conversely, the income from re-sequestering the 58 (±24) Tg of aboveground C is \$2100 (±900)M (\$2800 (±1000)M including roots). This income is distributed over several decades, the timelines of which are discussed below.

An alternative re-sequestration goal could be partial rehabilitation/replenishment, enough to bring the poorest vegetation condition back up to the local, extant average for each ecosystem type. The carbon in aboveground biomass missing from the 12.85 Mha that are below the local, extant-averages (Fig. 7b), is 37 (±11) Tg (Table 1, Supplementary Table A.3), or 50 (±15) Tg if C including roots (equating to AUD\$190 (±60)M). To revegetate this area a total of 3.14–4.77 MDSE would need to be destocked (for actual and notional stocking rates respectively), equating to −\$46 (±12) M $year^{-1}$ in livestock profits. The avoided C effluxes from that livestock's methane are 1 (±0.3) Tg over ten years, and 10 (±3) Tg over 100 years. The latter is only just significant compared with sequestration in biomass, being of similar magnitude to the likely error margins. Thus, C re-sequestration does not recover forgone livestock income.

A total of 2.219 Mha of commercial rangeland (Fig. 7c) scored negatively when compared with both remnant vegetation (within each ecosystem type) and extant averages (for ecosystem types). Its carbon deficit of aboveground biomass was 3.7 (±1.1) Tg, 5.0 (±1.5) Tg including roots. This represents the absolute minimum of C that could be re-sequestered. Several, large regions consistently showed as negative (Fig. 7), across all three, NSW-wide analyses, the largest being 0.68 Mha east of the Cobar–Bourke road.

For the recent deforestation (1989–2007, 84,479 ha, Table 1), the emitted C from aboveground biomass is 1.59–1.96 (from ALOS and NCAS potential biomass respectively), (Table 1,Table 2, Supplementary information Table A.4). This represents what could be sequestered upon reforestation: an average of 1.8 (±0.2) Tg (2.3 (±0.3) Tg with roots), or 21 (±2) Mg ha^{-1}. A plausible error margin is likely to be at least twice the difference between the two estimates. If the emitted carbon as a result of land management activities is brought to account, the bill is \$87 (±26) M (including roots), or \$1000 (±300) ha^{-1}. If using farm (net) profits from livestock grazing, it would take 270 (±80) years to repay, either by the NSW government or the pastoralist on leasehold land, or by the owners of freehold land. (The area is 11% freehold, 89% leasehold.)

3.3. Re-sequestration rate for recently deforested land

Assuming that the same linear growth rate is applicable to the recently deforested land in our study area as was used for the state of QLD (Dean et al., 2012b), but scaled to local site potential (from the benchmarks), then the re-sequestration of the emitted C from aboveground biomass in our study area would be 13,934 Mg

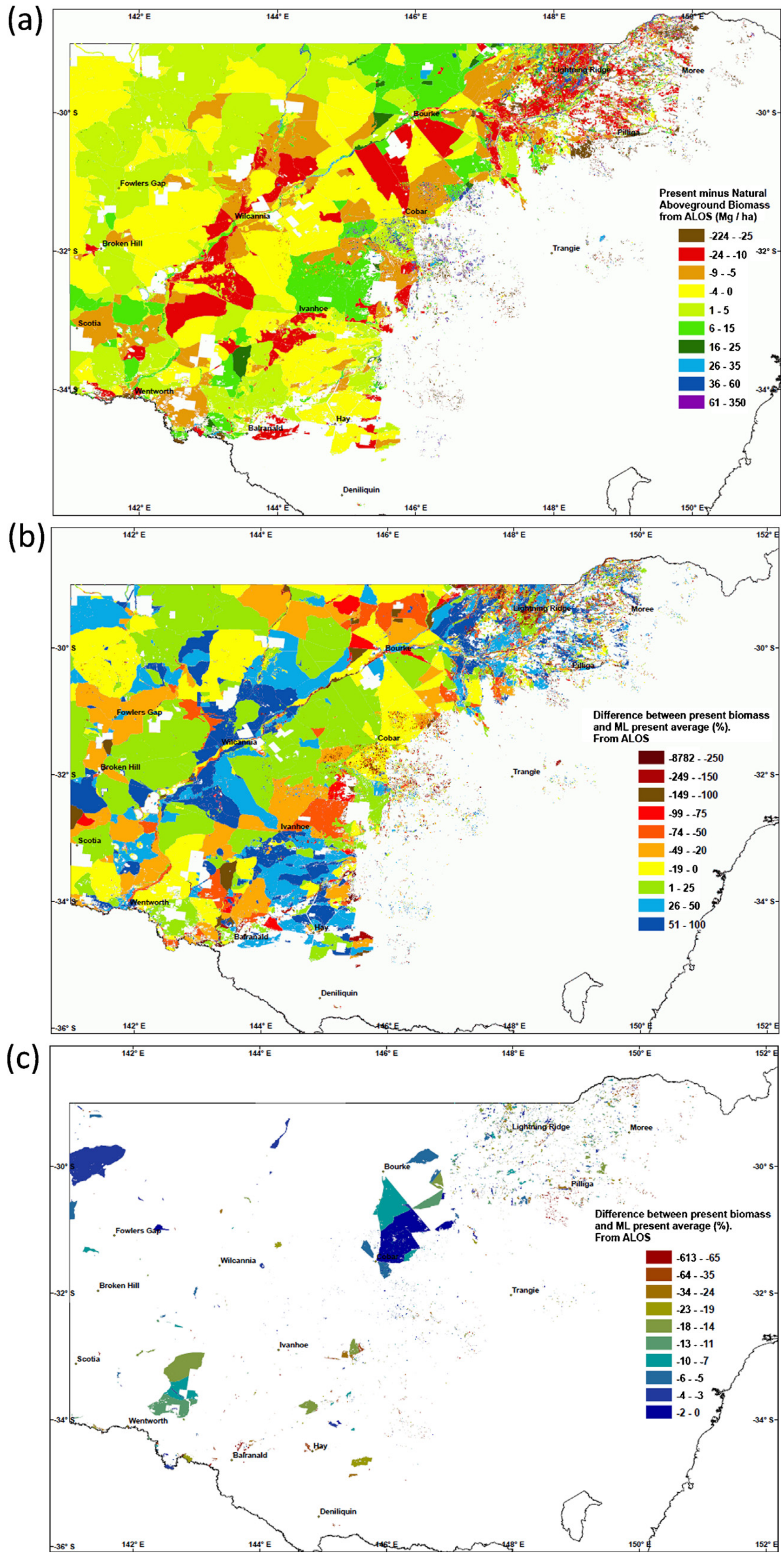

Fig. 7. Regions of biomass loss. Negative values indicate a deficit. (a) Extant minus remnant. The main negative area is a SW-NE band following the Darling River. (b) Extant minus average-extant, for each ecosystem-type. Compared with (a), this is a local, relative gauge. (c) Negative areas in both (a) and (b). The most consistently negative region is between Cobar and Bourke (upper centre).

Table 1
Levels of rangeland forest degradation targeted, and potential C re-sequestration. The C values listed are spatial averages: higher deficit (re-sequestration) is likely in the more denuded patches.

Commercial rangeland type	Area (Mha)	Deficit of C in aboveground biomass (Tg)	Deficit per hectare(Mg ha^{-1})
NSW, state-wide	30.57	26 (±9)	0.85 (±0.29)
Land below remnant value	17.90	61 (±18)	3.4 (±1.0)
Land below extant average	12.85	37 (±11)	2.9 (±0.9)
Land below remnant and extant average	2.219	3.7 (±1.1)	1.7 (±0.5)
Recent deforestation	0.08450	1.8 (±0.2)	21 (±6)

year^{-1}, or a sequestration flux of 0.173 Mg ha^{-1} year^{-1}, or including root carbon, 0.230 Mg ha^{-1} year^{-1}. As in Dean et al. (2012b) our rate for our study area assumes no detriment to growth from climate change effects, such as increases in water deficit, storm and fire effects.

For re-sequestration of aboveground C by non-linear growth, equivalent to woody-thickening or slow plantations (Fig. 3, Table 3), the value of potentialC used in Eqs. (1) and (2) was the average (NCAS and ALOS): 21.05 Mg ha^{-1}. Sequestration with reforestation would mostly be at 0.009 (±0.002)–0.16 (±0.04) Mg ha^{-1} year^{-1} for the first 10 years and 0.13 (±0.3)–0.32 (±0.08) Mg ha^{-1} year^{-1} for the first 50 years, and full recovery would take at least 200 years (Table 3). The 50 year rate for the average equation is thus close to that for linear growth.

The destocking necessary for re-sequestration, based on the rain-dependent stocking rate formulae (Eqs. (3) and (4)) is: 30,423 and 46,742 DSE for the actual stocking rate and the notional carrying capacity, respectively. The corresponding avoided emissions from enteric fermentation upon destocking the deforested land are 739 Mg and 1136 Mg respectively. After 50 year that corresponds to avoided C effluxes of 0.03871 and 0.05948 Tg, respectively. The carbon cost for the livestock enteric fermentation emissions is approximately a tenth of the deforestation cost; and a carbon price of \$10 Mg^{-1} of CO_2-e equates to only ~\$0.93 DSE^{-1} year^{-1}, which is small compared with the grazing profit per average sheep of \$11.54 DSE^{-1} year^{-1}. Therefore the cost of livestock methane emissions alone is unlikely to motivate destocking.

We calculated \$3.51 M and \$5.39 M profit (\$4.5 (±0.9) M) for ten year's livestock grazing and \$17.6 M and \$27.0 M (\$22 (±5) M) after 50 years (for actual and notional livestock capacities). These grazing profits are greater than the C sequestration profits if the reforestation growth rate is that of woody-thickening (Eq. (2)), but less than the C sequestration profits if growth is at the slow-plantation rate (Eq. (2) G = 15) (Table 3). If growth is half–way in between those rates then grazing profits are greater than those from C sequestration after the first ten years but less than after 50 years. The uncertainty in reforestation growth rate therefore makes it unknown whether grazing or C sequestration is more profitable after 50 years.

3.4. Other re-sequestration rates

The same curves used for modelling reforestation of the recent deforestation were used for modelling the rehabilitation of woody biomass for the other levels of rangeland modification (Table 4). The profit for carbon replenishment was less than that from commercial grazing (for both the slow plantation growth rate and the thickening growth rate). Note however that the rates are spatial averages, and that they would initially be higher for denuded but otherwise intact land than for land with trees and shrubs. Selection of locally denuded land alone would thus provide higher rates of return. Rates will also be higher in the higher-rainfall zones, and if it is found that the NCAS biomass values are more realistic for remnant vegetation than the ALOS values.

3.5. Cost of ongoing emissions

Apportioning of the Australian annual rangeland emission to NSW, with adjustment for the NSW environment, gave the average emission from deforestation over the last two years as 0.15036 Tg year^{-1} total C in biomass, and the carbon emission from Mulga Lands forest degradation was 0.228 Tg year^{-1} (Table 5). The ΔSOC for NSW deforestation estimated pro-rata from QLD data was estimated to be 0.0147 Tg year^{-1}, and on a flat −17% basis over 30 years it was 0.0184 Tg year^{-1} SOC efflux: giving an average of 0.0176 Tg year^{-1}.

The annual cost of maintaining the NSW rangelands under commercial grazing through current management methods is therefore, when accounting for C emissions, approximately \$37 M. The error margin for this figure is likely to be around 50 to 75%. After 50 years the cost of NSW commercial rangeland emissions is thus \$1800 (±1100) M.

Introducing land rehabilitation State-wide for NSW in 2014 to counteract the ongoing soil erosion associated with commercial grazing and cessation of all that particular source of erosion by 2053, could limit the SOC efflux since onset of rangeland commercial grazing to 12 (±011) Tg (a cost of \$434 M), and emit only 5.1 (±4.6) Tg from 2014–2053. Thus, a SOC emission of 99.4–5.1 = 94 (±41) Tg could be avoided by initiating rehabilitation in 2014 and ceasing anthropogenic erosion by 2053. That corresponds to an avoided cost of \$3400 M. The error margins for these estimates are high, possibly around 90%.

4. Discussion

4.1. Main findings and their wider applicability

We found that over the study area as a whole and under current climatic conditions, farming profits exceeded our notional return from carbon re-sequestration, even if ongoing emissions were also priced. Nevertheless, spatial variations suggest that some localised destocking of the more-C-depleted areas would be profitable.

Table 2
Summary statistics for commercial rangeland deforested from 1989 to 2007.

Tenure	Area deforested (ha)	C density in aboveground biomass (Mg ha^{-1}) (average of NCAS and ALOS)	Livestock density DSE ha^{-1}	Cost of total carbon emitted (AUD\$ million)
Freehold	9547	38 (±11)	0.792	18 (±5)
Leasehold	74,948	19 (±6)	0.305	69 (±21)
Total/ average	84,495	21 (±6)	0.360	87 (±20)

Table 3
C sequestration and C income for different growth rates for reforestation of recent deforestation. Sequestration values are for aboveground biomass and income is for total biomass. Error margins in monetary incomes reflect likely variability in growth rates.

Duration of flux and sequestration	C sequestration				C income for Eq. (1) with G = 15years(AUD$)	C income for Eq. (2) AUD$)	C income average Eq. (1)/G = 15 & Eq. (2) (AUD$)
	Eq. (1)			Eq. (2)			
	G (years)						
	8	10	15				
Average flux over 1st 10 years ($Mg\,ha^{-1}\,year^{-1}$)	0.48	0.32	0.12	6.8×10^{-3}	5.8 (±1.3) $ha^{-1}\,year^{-1}$	0.25 (±0.06) $ha^{-1}\,year^{-1}$	3.1 (±2.7) $ha^{-1}\,year^{-1}$
Average flux over 1st 50 years ($Mg\,ha^{-1}\,year^{-1}$)	0.31	0.29	0.24	0.10	12 (±5) $ha^{-1}\,year^{-1}$	3.7 (±0.9) $ha^{-1}\,year^{-1}$	8.2 (±3.3) $ha^{-1}\,year^{-1}$
Total after 10 years (Tg)	0.41	0.27	0.10	5.8×10^{-3}	4.9 (±1.1)M	0.21 (±0.05)M	2.6 (±2.3)M
Total after 50 years (Tg)	1.3	1.2	1.0	0.42	49 (±20)M	15 (±4)M	35 (±14)M

The complication of costs and carbon footprints of managed rehabilitation was circumvented in the present work by using 'natural' regeneration rates applicable to degraded rangeland, via modelling of woody thickening and slow-growing plantations. The approach focussed only on major effects and was robust, but the error margins were large. Finer-scale studies have reported sequestration profits, such as for ΔSOC with 'rotational grazing' (e.g. Gosnell et al., 2011) but they rarely include a full life-cycle C analysis of the intensive management, involving extra fencing, watering points and labour.

Our model of passive reforestation produced slower growth in the early stages compared with plantations (even-aged growth) (Fig. 3, Tables 3 and 4) because colonisation is likely to be patchy; infill may have to wait for seed production by the initial wave of colonisers (Chew and Chew, 1965), stand growth continues until the recruitment rate equals the mortality rate, and some later woody species may not germinate until the later stages of succession.

The linear sequestration rate assumed in most studies (e.g. Bray and Golden, 2008; Dwyer et al., 2009; Fensham and Guymer, 2009; Dean et al., 2012b) essentially equates to an average of Eq. (1) (with G = 15) and Eq. (2); faster than woody-thickening, but slower than plantations. Linear rates exaggerate sequestration in the early years. Our average rate for the first ten years (Table 3) is similar to that proposed for the Henbury property in the Northern Territory (Australia) (DSEWPC, 2011), which has similar rainfall to NSW (293 mm $year^{-1}$ and 320 mm $year^{-1}$, respectively).

Re-sequestration rates will be spatially variable. Areas with species with lignotubers (e.g. mallee eucalypts and brigalow) will regenerate faster (e.g. like Eq. (1) with G = 15 years), but sites that have been repeatedly deforested or undergone substantial soil erosion will be slower, similar to woody thickening (Eq. (2)), though after a century sequestration is similar. Rates will be higher in the less water-limited regions, but so too will livestock profits.

NSW rangelands contain three global biomes, which should allow the NSW re-sequestration rates to be applied to areas outside of Australia with similar biomes and climates (Fig. 8, Supplementary information, Appendix D) as initial estimates, although we strongly recommend that differences in biotic and abiotic conditions be taken into account if our methodology is applied in other countries. Australian rangelands differ from those in some other countries in that rather than being grassland they are predominantly shrubland and woodland with lesser amounts of scrub, heath and herbland (Carnahan, 1977; Luly, 1993).

4.2. Recent deforestation

Our calculations have assumed that the C is emitted spontaneously upon clearing, which is not so even under the common practice of heaping and burning debris (Dwyer et al., 2009), and with roots less likely to burn. If trees and shrubs are left to decompose after felling, C loss can take over a century. Positive feedback from climate change (Bonan, 2008) which can be nonlinear (Raupach et al., 2011; Raupach et al., 2014) means that

Table 4
C sequestration and income for different forest degradation levels. Sequestration values are for aboveground biomass and income is for total biomass. Error margins in monetary incomes reflect likely variability in growth rates.

Commercial rangeland targeted	Duration of flux and sequestration	C Sequestration		C income Eq. (1) G = 15 years (AUD$)	C income Eq. (2) (AUD$)
		Eq. (1) G = 15 years	Eq. (2)		
All	Average flux over 1st 10 years ($Mg\,ha^{-1}\,year^{-1}$)	4.7700×10^{-3}	2.7346×10^{-4}	0.23 (±0.06) $ha^{-1}\,year^{-1}$	0.010 (±0.003) $ha^{-1}\,year^{-1}$
	Average flux over 1st 50 years ($Mg\,ha^{-1}\,year^{-1}$)	9.5153×10^{-3}	4.0308×10^{-3}	0.46 (±0.12) $ha^{-1}\,year^{-1}$	0.15 (±0.04) $ha^{-1}\,year^{-1}$
	Total after 10 years (Tg)	1.4582	8.3601×10^{-2}	71 (±18)M	3.1 (±0.8)M
	Total after 50 years (Tg)	14.54456052	6.16123	710 (±180)M	230 (±60)M
Land below remnant value	Average flux over 1st 10 years ($Mg\,ha^{-1}\,year^{-1}$)	1.9228×10^{-2}	1.10237×10^{-3}	0.94 (±0.23) $ha^{-1}\,year^{-1}$	0.040 (±0.010) $ha^{-1}\,year^{-1}$
	Average flux over 1st 50 years ($Mg\,ha^{-1}\,year^{-1}$)	3.8357×10^{-2}	1.62486×10^{-2}	1.9 (±0.5) $ha^{-1}\,year^{-1}$	0.60 (±0.15) $ha^{-1}\,year^{-1}$
	Total after 10 years (Tg)	3.4420	0.197332	170 (±40)M	7 (±2)M
	Total after 50 years (Tg)	34.3312	14.5431	1700 (±420)M	530 (±130)M
Land below extant average	Average flux over 1st 10 years ($Mg\,ha^{-1}\,year^{-1}$)	1.6610×10^{-2}	9.5214×10^{-4}	0.81 (±0.20) $ha^{-1}\,year^{-1}$	0.035 (±0.009) $ha^{-1}\,year^{-1}$
	Average flux over 1st 50 years ($Mg\,ha^{-1}\,year^{-1}$)	3.3313×10^{-2}	1.4034×10^{-2}	1.6 (±0.4) $ha^{-1}\,year^{-1}$	0.51 (±0.12) $ha^{-1}\,year^{-1}$
	Total after 10 years (Tg)	2.1308	0.12216	100 (±30)M	4.5 (±1.1)M
	Total after 50 years (Tg)	21.25356	9.0032	1000 (±260)M	330 (±80)M

Table 5
Annual C emissions from NSW commercial rangelands. Where Australia-wide literature is cited, areas were determined in the present work and applied pro-rata to the national data.

Activity producing emission	Fundamental datasource + present work	Tg year^{-1}	Cost (AUD$ million)	% of total 1	% of total 2
Mulga Lands degradation	Moore et al. (2001)	0.1768	6.48	32	18
Diesel consumption	Rolfe (2002)	0.01140	0.418	2.1	1.1
Soil erosion	Dean et al. (2012b)	0.1406	5.15	26	14
Deforestation–biomass C	Present work	0.1504	5.51	28	15
Deforestation–SOC	Dean et al. (2012b)	0.01755	0.643	3.2	1.8
Total 1		0.5475	20.0	100	55
Livestock methane	Robertson (2003)	0.6783	24.9		68
Livestock methane	Present work	0.2299	8.43		23
Total 2		1.001	36.7		100

the timing of emissions may contribute to their effect. The black carbon and methane from burning may produce and effect additional to the CO_2 emission alone (M. Raupach, personal communication, 2014) which under full costing, would incur a higher penalty than aerobic decomposition. After felling, decomposition emissions can be reduced by sealed deep burial (e.g. Zeng, 2008) as can fire hazard. The balance between the two post-felling activities will also be influenced by residual charcoal, how much tree carbon becomes SOC, how much decomposition is aerobic (as opposed to anaerobic [methane] emissions), and the ecological functioning of the surface debris. Such processes are lengthy and so it would be more feasible to recover emission costs at deforestation time: for example by applying fees for lost biomass, associated ΔSOC and any positive feedback.

Both present and future livestock densities and carbon prices are important when considering land use alternatives. With actual and notional stocking rates differing by ~29% and the sequestration profitability determined by the choice of regrowth formula, it is obvious that the real stocking rate has a strong affect on outcomes. A complication in assessment is that stocking rates can usually be maintained high for a couple of decades after deforestation, while the nutrients from decomposing biomass and the legacy nutrients are available, after which stocking rate must drop (Sangha et al., 2005; Kaur et al., 2006; Radford et al., 2007; Kirschbaum et al., 2008).

The potential reforestation C flux of 0.13 ($\pm$0.3)–0.32 ($\pm$0.08) Mg ha^{-1} year^{-1} (including roots) is similar to the 0.22 ($\pm$0.02) Mg ha^{-1} year^{-1} (excluding roots) of Witt et al. (2011) for degraded Mulga forest in QLD, but less than the 0.36 ($\pm$18) Mg ha^{-1} year^{-1} for QLD (Dean et al., 2012b) but the QLD rangeland included more high-productivity land and the remnant ecosystems may have been less degraded.

The methane emissions avoided by destocking were comparable with the sequestration in biomass for the slowest growth rates. However, there are other C benefits, such as avoiding typical SOC emissions resulting from grazing. For stronger climate change mitigation, a key part of carbon management is conservation of carbon stocks, that is to forgo the deforestation, but the current market and policy implementations are inadequate (e.g. Krause et al., 2013). One study found that objections to deforestation controls were economic, though a carbon pollution cost for deforestation was not included (Davidson et al., 2006).

4.3. Major biomass change for different levels of forest degradation

The major re-sequestration options calculated here vary considerably in their C totals and hence potential financial dividends (Table 1). Our estimates of 34.5 and 35.5 Tg from ALOS and NOAA-AVHRR processed data respectively for NSW-wide re-sequestration can be compared with 42.3 Tg from Dean et al. (2012b). Our values used refined rangeland boundaries and should therefore be more spatially representative. An apparent paradox is that, on average, the entire NSW rangelands can re-sequester 35 ($\pm$10) Tg whereas tallying only the land below-remnant-ecosystem-biomass can re-sequester 61 Tg. This arises from using two different summation methods. In the former, land that was already above potential was tallied together with land that was below potential (over wide expanses), i.e. allowing pluses and minuses to partially cancel each other. Although not shown in Table 1, if only land below-potential was considered NSW-wide then 81 Tg could be re-sequestered.

The reasons for merging above- and below-potential land were: (a) to smooth over noisy data, and (b) to remain conservative in the calculation of monetary dividends for re-sequestration and thus

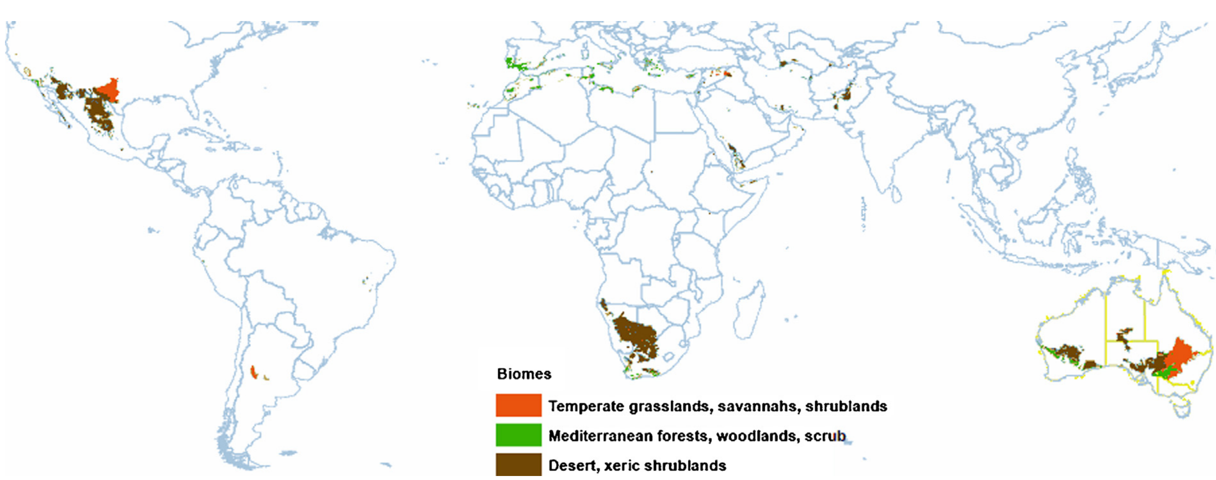

Fig. 8. Global rangelands with the same biomes and similar climate (rainfall and average temperature) to those of NSW's rangelands. Close-up maps of major regions are in the Supplementary information Fig. D.27.

avoid disappointing speculation. The 35 (±10) Tg NSW-wide is also conservative because reliance on depauperate remnants as indicators of potential gives a false indication that more rangeland is already above its potential carbon stock than it really is — these areas then mathematically cancel out the more severe C deficits upon averaging, to give an overall low re-sequestration forecast.

The low biomass values from ALOS data for ecosystem remnants (compared with NCAS or NOAA-AVHRR values) (Supplementary information Fig. B.14) can be attributed to biomass attrition with commercialisation (Allen, 1983) and to the NCAS formulation not accounting for native grassland (Supplementary information Fig. B.6). Some entire remnant ecosystems had substantial biomass attrition (Fig. 2), especially those neighbouring riparian zones (e.g. 'Mid-Darling Plains' between Cobar and Bourke, alongside the Darling River, Fig. 7). The concentration of attrition in riparian forests in rangelands is Australia-wide (Dean et al., 2012b) and reveals the need for more-adequate benchmarks for broad-scale carbon assessment schemes.

Conversely, ALOS values were higher than NCAS values in the few high-rainfall areas, which have taller forest (Supplementary information Fig. B.14). That may be due to the tendency of NCAS to underestimate biomass for tall native forests (Keith et al., 2010) or for radar to saturate in some higher-biomass woodlands (Lucas et al., 2000).

4.4. Climate change effects

The effects of climate change, and accompanying changes in atmospheric chemistry, upon vegetation and land condition (and thus on rehabilitation), depend on a range of climate attributes and biotic responses, and on feedbacks between these. In summary, the positive influences of CO_2 and N fertilisation are overshadowed by the multitude of secondary negative effects such as increased fire and seasonal stressors, as discussed in Dean et al. (2012b). Arguably the best estimate of change in biomass due to climate change over the coming century comes from an ecologically based calculation of ΔSOC. The relationship is reflected in the correlations found here between biomass and rainfall, SOC and temperature, and livestock carrying capacity and rainfall (Eqs. (3), and (4)). Forecast ΔSOC to 0.3 m depth, from 2000 to 2100 for NSW rangeland, for a range of climate change scenarios, averaged −27 (±11)% or a SOC emission of 163 (±66) Tg (Dean et al., 2012b), equating to AUD$6000 (±2400) M. That modelling was based on the relationship between rainfall, temperature, biomass and SOC, and inherently includes tributary characteristics such as drought and fire, which are expected to have an increased influence with climate change (Gonzalez et al., 2010). Our 58 (±24) Tg possible C re-sequestration (for biomass) for NSW was calculated in the absence of climate change and may thus need to be reduced by up to ~27%. The effect of climate on ΔSOC is unlikely to be expressed within one century — it will take longer and be largely in response to change in biomass over that period, i.e. ΔSOC will be delayed compared with the change in biomass.

The temperature and rainfall extremes of climate change are direct threats, influencing net growth (e.g. Wallace et al., 2009; Piao et al., 2011). If ecosystems cannot adapt fast enough to the new climatic extremes then biomass will decrease more than forecast by change in annual rainfall and temperature averages alone. A study of climate change impacts for Namibia (which has a similar xeric biome to that of NSW (Supplementary information, Appendix D)) found that with decreasing rainfall and increasing temperature (as are forecast for most of NSW), there was a significant decrease in tree biomass when change in fire regimes was included (Sarker et al., 2012). Most climate-change-impact studies neglect change in fire regimes. Many also neglect the increase in decomposition rates that has been forecast to offset CO_2 fertilisation (Trumbore, 1997). A climate-change study for Morocco (Schilling et al., 2012) which has a significant proportion of rangeland similar to the Mediterranean-biome in NSW, showed similar climate-change effects to those for NSW. The study highlighted a forecast reduction in agricultural production and the need for conservation of productive assets. The positive correlation between livestock carrying capacity and rainfall found by us for NSW suggest a similar decrease in productivity.

The largest single threat to Australian rangeland C sequestration is fire, which is forecast to increase with climate change and with population. Fires that are too frequent reduce site-quality and long-term carbon storage, and may increase erosion (e.g. Janzen, 2011; Shi et al., 2013). Conversely, decreased fire frequency is correlated with decreased grass density and conversion of savannah and woodland to forest (Anella and Wright, 2004; Dutilly-Diane et al., 2006; Fosse et al., 2008). Grass fires can offset increase in woody biomass following good rains, eliminate much coarse woody debris and cause increases in fire-tolerant invasive species such as buffel grass (*Cenchrus ciliaris* L. (syn. *Pennisetum ciliare* (L.) Link)). The negative impact of buffel grass on woody biomass has been discussed previously (Yates et al., 2000; Butler and Fairfax, 2003; Dwyer et al., 2009; Dean et al., 2012b). It will become more of a problem for NSW with climate change because it will follow the wave of suitable climatic conditions from the continental interior southwards and south-easterly (Steel et al., 2008; Martin et al., 2012). Biological control methods for buffel grass might be effective in reducing its threat.

4.5. Cost of ongoing emissions

With a carbon price of AUD$10 Mg^{-1} CO_2-e, after 50 years the cost of NSW commercial rangeland emissions is $1800 (±1100)M. That is of similar magnitude to the income of $1200 (±480) M after 50 years from rehabilitating the 17.90 Mha of land that has carbon stocks below the 'remnant' values. These are opposite management alternatives, but if sequestration is chosen over grazing then there is both an avoided cost (of ongoing emissions) and a monetary dividend (from re-sequestration in vegetation), making a difference of $3027 M, if the same market place is used for both transactions. That type of balance would also apply at the property-level. Alternatively, the annual NSW emissions cost of $36.7 M, is ~35% of the NSW farm net profits of $104.3 M. Thus, on average, the emission costs of rangeland farmers could be recovered by a fee of 35% of their net profits, and used for managed reforestation of abandoned land.

Our estimates of emissions from livestock were significantly lower than those from previous work (Robertson, 2003). Our livestock numbers for NSW are a possibly more-direct estimate (relying on NSW State data) rather than the result of dividing the nationwide estimate of methane emission according to land area, pro-rata, and our methane emission per DSE was conservative. Slightly counteracting this conservatism we have assumed that destocking would not substantially increase feral goat populations, with an assumption of decommissioning artificial waters and some continuance of goat harvesting.

NSW-wide, the two-highest singular annual emissions were forest degradation in the Mulga Lands and methane from livestock. Conversely the flux of SOC emissions accompanying deforestation is low but continues for one or two millennia, due to a longer average halflife of SOC. Additionally one must note that deforestation produces a positive spike in SOC stock for up to 15 years, which experimentation and any C-trade assessment must accommodate (Dean et al., 2012a).

The SOC efflux for NSW due to erosion induced by commercialisation (derived pro-rata from the Australia-wide estimate of

Dean et al. (2012b)) is 7.1 (±1.8) Tg. SOC takes an order of magnitude longer to sequester than biomass carbon, and land rehabilitation is an additional time requirement; therefore the 7.1 Tg is likely to take several centuries to re-sequester. Despite the financial irrelevance of such a timeline in current markets, rehabilitation also avoids the C effluxes of continued grazing. The NSW-wide annual costs for current SOC losses from erosion and deforestation are \$5.2 M and \$0.62 M, respectively (Table 5). Measuring ΔSOC appropriately is paramount, though very few experiments have sampled the dynamics of SOC over the whole soil profile, or over sufficient time, and accounted for the lag of ΔSOC behind change in woody biomass (Dean et al., 2012a; Shi et al., 2013). Thus the monetary value of ΔSOC is unlikely to be realised by financially focussed management and thus change in current practices would require alternative motivations.

4.6. Increasing certainty

Defining the rangelands by a single boundary is an administrative aid, whereas in reality land uses are variegated either side of that line, as for most agricultural boundaries (Whittlesey, 1936). Classification based on land use may also be ambiguous. For example, where the native woody vegetation has been kept cleared such that it cannot regenerate autonomously, and the grasses in its place are exotics; then the land could be considered intensive agriculture (e.g. Supplementary information Figs. C.18, C.19). Overall, 'rangeland-ness' needs to be better-determined for rangeland carbon accounting.

Potential biomass defined by remnant vegetation was most probably underestimated, due to almost ubiquitous forest degradation and because farmers usually deforest the most-productive land first (Pressey et al., 1996; Holmes et al., 2006). The strongest effects of this selective process were near major rivers, and in the arid west where there was woody biomass attrition without broad-scale deforestation. Also, rivers with high biomass from overhanging *Eucalypts camaldulensis*, were included in the extant 'average' but were absent from the 'potential' because they were not flagged as native ecosystems [as their remotely sensed pixels had the possibility of being water-dominated]. Better, though more time-consuming representation of potential could possibly be obtained by detailed modelling of ecosystems, with interpolation over deforested and other highly degraded areas using software such as ANUClim (Xu and Hutchinson, 2014) along with ground-truthing and use of historical aerial photography.

Ideally, stocking levels would be available in the form of annual averages over several years, because land condition responds cumulatively to long-term effects. This would allow determination of a more robust correlation between stocking levels and environmental variables. Nevertheless, possibly due to the large size of NSW, a degree of correlation was noted. The paucity of paddock-level stocking data for Australia, and apparent covertness or passive-non-compliance by farmers and some government departments, has been noted previously (Bartel and Barclay, 2011; Dean et al., 2012b) and remains a hurdle to productive environmental management and scientific progress.

The north–south gradation in temperature in NSW contrasts with the east–west gradation in rainfall (Supplementary information, Fig. B.7, B.8). This produces a greater range of ecosystem types than if the two gradients were parallel, and thus a greater variety of knowledge is required to develop C trading scheme 'methodologies'. Areas found to be depleted in the present study are guides only — on-ground verification is advised prior to investment.

The high error margins for re-sequestration were partly due to the use of broad-scale averages and because of spatial variability in forest degradation level. Most individual land-use polygons (the area unit of calculation) spanned a mixture of barren areas, degraded forest and vestigial trees or shrubs. Sequestration will primarily be by infill. Within the barren patches sequestration rates will be similar to those for forest-stand growth, but for the larger polygons it will necessarily be less. Thus, in practical terms for greater certainty in financial sequestration projects, fine-scale vegetation mapping is required and different growth rates assigned to canopies and canopy gaps. Also, the high error margins noted for sequestration over the first decade cannot be reduced without more knowledge about the dynamics of autonomous reforestation.

4.7. Wider implications

Our models indicate that there may be some circumstances in which destocking for carbon gives a better return than maintaining stock grazing. There are environmental, social, cultural and other economic consequences of any resultant shifts in land use. Multiplier effects from carbon payments may result in different outcomes than those from farming activities. While we expect that destocking will result in greater availability for commercial harvesting of feral and native animals (even when artificial watering points are decommissioned), thereby mitigating the impact on re-sequestration of any wild animal population increases, the employment types resulting from carbon management and altered protein sourcing are likely to be very different from those of the present, with likely positive and negative social and cultural consequences. In many areas where soil and the native seedbank have been severely degraded (Brown and Potter, 1971; Condon, 1986; Sparrow et al., 2003; Hunt et al., 2014), investment in rehabilitation is likely to be necessary to achieve our re-sequestration forecasts. The achievement of maximum carbon re-sequestration in the landscape is not necessarily the best nature conservation outcome in the short term because maturation of woody-thickening (accompanied by self-thinning) may take well over a century (Hibbard et al., 2003; Dean et al., 2012a), and meanwhile woody-thickening is regarded as degradation in the NSW regulatory processes for land clearance.

In a much broader context, our results also raise the need for the substitution of protein derived from any destocked rangelands by protein from other sources (to avoid carbon emission leakage by mere transferral of activity, Tamminen (2011)). At a global level, options include the increased use of vegetable sources and a reduction in food wastage (Wirsenius et al., 2010; Ripple et al., 2014). In the longer term, tissue-cultured protein may provide a solution (e.g. Edelman et al., 2005; Hopkins and Dacey, 2008).

5. Conclusion

While the present study has shown how information and certainty on the spatial concentration and fluxes of carbon in rangelands can be increased, and has indicated the potential for use of some land for carbon re-sequestration, there is insufficient detail for carbon trading. For financial sequestration projects, vegetation mapping would be required at a scale where individual tree canopies and bare ground are identifiable. Knowledge of long-term ΔSOC would also greatly improve accounts of national emissions. To increase precision in both emissions and sequestration values, experiments are needed to increase knowledge of:

(a) allometrics for trees and shrubs including deep-set roots (which access ground water),
(b) natural regeneration growth rates,
(c) the timelines involved in SOC formation, mobilisation and decomposition for different ecosystems, and
(d) changes in tree carbon stocks under grazing

(e) the locations of representative remnant ecosystems, with at least 90% of pre-commercialisation biomass present.

Despite the low returns from C sequestration calculated from NSW-wide averages, the accumulated debts from ongoing emissions are high, even at the middling carbon price of AUD \$10 Mg^{-1} of CO_2-e. At that carbon price, and a hypothetical situation of "polluter pays", the current annual net emissions from the Australian rangelands equate to AUD\$0.74 ($\pm$0.4) billion, or AUD\$1.1 ($\pm$0.5) billion including methane from livestock and savannah burning. Additionally, SOC emissions following deforestation continue for several centuries.

With global climate change the emissions from rangelands will increase, with the best regional estimate coming from the change in soil carbon (down to 0.3 m). For the Australian forested rangelands SOC emissions initiated by climate change to 2100 (though completed over several centuries) will be 1130 Tg (Dean et al., 2012b), equating to AUD\$41 billion, which is the largest emission calculated in the present study. A similar debt will be associated with carbon in lost biomass, though that is likely to occur sooner.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2014.09.125.

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