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Trajectory of change in land cover and carbon stocks following European settlement in Tasmania, Australia

Louise M. Romanin*, Lynda D. Prior, Grant J. Williamson, David M.J.S. Bowman

School of Biological Science, University of Tasmania, Private Bag 55, Hobart, Tasmania 7001, Australia

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ABSTRACT

The conversion of temperate biomes in the Americas, Australia and New Zealand by European colonists, creating 'neo-European landscapes', is emblematic of the global environmental change inherent in the Anthropocene concept. The Midlands of Tasmania is a valuable model system for studying changes to land cover and above ground biomass in neo-European landscapes. Europeans colonized this area in early 19th century and disrupted a hunter-gatherer economy that has persisted for over 30,000 years. Aerial imagery, historical reconstructions, field surveys and future climate projections provided tools to chart changes in tree canopy cover and carbon stores in the Northern Midlands for the period 1788–2070. In the ~160 years between 1788 and 1940s, large areas of open woodland were cleared but carbon loss was modest (−14 %). In the ~60 years between 1940s and 2010, carbon loss accelerated (a further −21%) as clearing shifted from woodlands to forests. An estimated ~28% of the study area would need to be replanted with eucalypt plantations to capture the carbon lost between 1788 and 2010. Three general circulation models (GCMs) representing climate predictions for 2070 suggest that carbon storage in the landscape would change by +13% to −13.2% of 2010 levels, without any restoration intervention.

1. Introduction

Clearing native vegetation at a broad scale is a potent driver of global loss of biodiversity (Vitousek, 1997; Sanderson et al., 2002; Monastersky, 2014) and a major anthropogenic source of atmospheric CO_2 (Le Quéré et al., 2009). Creating arable land is a prime motivation for clearing vegetation. This process has occurred since the development of agriculture in the mid to late Holocene. In the 18th–19th centuries, European colonisation created agricultural landscapes in North America, Australia, New Zealand and southern South America (Crosby, 1986). These 'neo-European' agricultural areas were typically previously modified by indigenous people, especially through the use of fire to clear and alter vegetation structure and composition (Jones, 1969; Weiser and Lepofsky, 2009; Lightfoot et al., 2013).

Neo-European landscapes present interesting case studies of the interplay of culture and environmental change that is at the root of the Anthropocene concept. We use the term 'Anthropocene' as a metaphor to frame human impacts on the Earth system in both the prehistoric and historic period rather than specifying a precise time frame. Such temporal ambiguity is justified because of the long history of human impacts on the Earth System, beginning with the hunting of megafauna and burning of landscapes by humans and our antecedents in the Pleistocene (Glikson, 2013; Foley et al., 2013; Bowman, 2014). The development of agriculture in the Holocene increased human impacts on the Earth System (Ruddiman, 2003; Ruddiman, 2005; Ruddiman, 2007). Undoubtedly, the Industrial Revolution, often used to demarcate the start of the Anthropocene, has triggered ongoing planetary-wide impacts associated with burning fossil biomass, rapid growth in human populations and land cover change. Since the Second World War (WWII) a step change occurred in impacts on the Earth system, a period Steffen et al. (2007) termed 'the great acceleration' due to rising resource consumption, human population growth and powerful technologies resulting in more intensive and widespread negative impacts on natural landscapes (Crutzen and Steffen, 2003; Ellis et al., 2010).

The Midlands of Tasmania provides an excellent case study to document the rapid transition from hunter-gatherer to modern temperate agricultural landscapes that is at the core the Anthropocene concept. The Midlands is the second oldest agricultural landscape in Australia, settled in the first decade of the 19th century (Fensham, 1989). Before European colonisation in 1802, Tasmanian Aboriginal hunter and gatherers existed on the island for over 35,000 years (Colhoun and Shimeld, 2012). Palaeoecological evidence points to ecological disruptions caused

* Corresponding author.
E-mail address: louise.romanin@utas.edu.au (L.M. Romanin).

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by Aboriginal colonisation in the late Pleistocene, including changed fire regimes and loss of marsupial megafauna (Turney et al., 2008). Despite these disruptions, the available evidence suggests that, throughout the Holocene, Aborigines had achieved ecological sustainability, coexisting with a rich biota, including numerous endemic plant and animal species that have become extinct or threatened with extinction following European colonisation (Bowman, 1998; Fletcher and Thomas, 2007; Fletcher and Thomas, 2010). Like other Australian Aboriginal cultures, a key aspect of the Tasmanian's economy was the use of fire (Jones, 1969). Aboriginal landscape burning is thought to have maintained the grasslands and open grassy woodlands, which were rich in game (Gammage, 2008; Bowman et al., 2013). Such open vegetation proved ideal for European sheep grazing. By 1825 (Fensham, 1989; Morgan, 1992), most of the grassy lowlands of the Midlands had been allocated to free settlers, and the Aborigines were expatriated (Ryan, 2012).

Over the last 200 years, pastoralists cleared the Midlands to increase grass cover. This clearing targeted the most productive vegetation types, with a particularly intense period of tree removal following WWII, when artificial fertilisers, heavier machinery and the introduction of exotic grasses vastly improved the productivity of grazing lands (Kirkpatrick, 2007; Prior et al., 2013). In the past 20 years, large areas of Tasmania have been planted with fast growing eucalypts for woodchips. This region is too dry, however, for commercial forestry (Laffan, 2000). The contemporary landscape has less than 16% of the original native vegetation, which persists in highly fragmented, small and degraded patches (Fensham, 1989). Despite the small areas of remaining trees, land conversion has not ceased, with extensive irrigation programs under way. On the other hand, restoration schemes are designed to increase tree cover in this landscape (Bailey, 2013; Bryan, 2014; Greening Australia 2014a,b).

Using remotely sensed estimates of woody vegetation cover and above ground carbon stocks as response variables, we evaluate land cover changes since establishment of the agricultural landscape in the Midlands. We focus on four periods: (a) 1788, which provides a baseline under hunter-gatherer management, (b) 1800–1945 when agriculture was established, (c) 1945–2010, the period in which the current landscape was created, and (d) plausible future landscapes in 2070. We hypothesized that the first stage (1788–1945) of land cover conversion would have a minor effect on woody tree cover and carbon stocks compared to the second stage (1945–2010). This is because the hunter-gatherers had essentially opened up the landscape for pastoralism through targeted burning, whereas post World War II industrialised agriculture required far more extensive land conversion. By combining plans for intensive agriculture with three climate change scenarios we provide a reference point for current restoration schemes (Bailey, 2013). We estimate the amount of land that would need to be allocated for biosequestration to capture the carbon (a) lost between the 1940s and 2010 and (b) restore pre-agricultural (1788) levels.

2. Methods

2.1. Study region

We focus on land cover change in an area of 645 km^2 in the south of the Northern Midlands bioregion. This area is a broad floor of a north-south oriented graben (Fig. 1) (Fensham, 1989). The climate is classified as temperate with warm summer according to Köppen–Geiger classification (Peel et al., 2007). The region is located in a pronounced rain shadow, which is driest in Tasmania. The annual precipitation of $\sim$500 mm is evenly distributed throughout the year. The area has warm summers and cool

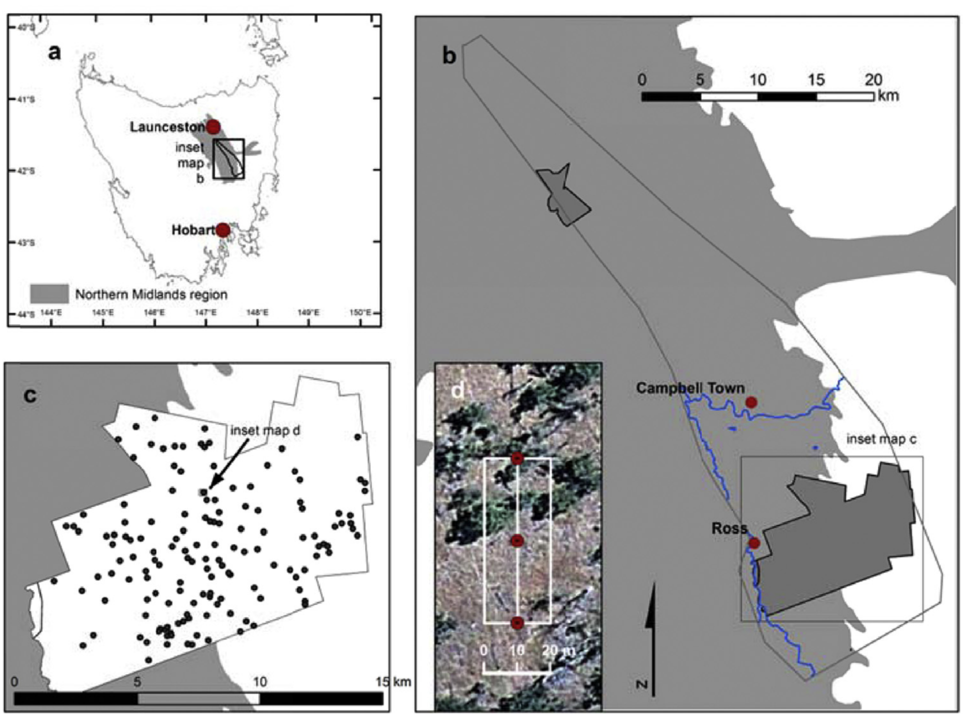

Fig. 1. (a) Map of Tasmania, showing the northern Midlands bioregion (shaded in grey), the cities of Launceston and Hobart and the study region. (b) Map of the study region (outlined), showing the two major towns major rivers in the region (blue) and the two field sites (dark grey), Tom Gibson Reserve in the north and a private property in the south. (c) Location within the private property of the field transects which were used in conjunction with aerial photography to estimate biomass across the study region. (d) Diagram of the field transect drawn to scale and superimposed on a satellite image. The white rectangle represents the belt transect inside which the diameters of all trees were measured. The three red circles indicate points at which tree cover was assessed using a densiometer.

winters; mean daily maximum temperature of hottest month (February) is 24.5 °C and minimum temperature of the coldest month (August) is 1.4 °C. Frost may occur during any month of the year and the average number of frost days across the study area is between 100 and 150 per year (Bureau of Meteorology, 2008). The original vegetation of the area was a mosaic of lowland grasslands and grassy eucalypt woodland (savanna) (Fensham, 1989).

Two sources of remotely sensed imagery were used to examine change in woody vegetation cover over the study region since 1945: aerial photography from the late 1940s (multiple years, black and white), and GeoEye-1 satellite imagery from 2010 to 2012. The aerial photos varied in scale: 1:15,840 for 1945 and 1947, 1:23,760 for 1950 and 1953. The GeoEye-1 imagery was 0.5 m resolution. The 1945 and 1947 photography was scanned at 600 DPI and the remaining at 2136 DPI. TASMAP provided the 1940s aerial imagery (www.tasmap.tas.gov.au), which was orthorectified using Landscape Mapper (Myriax Software, Hobart, Australia). Control points were taken from 2010 to 2012 Taswide Spot imagery, which has high ground-truthed accuracy. The 1788 vegetation patterns were adapted from Fensham (1989).

2.2. Remotely sensed image analysis

We assessed broad changes in woody cover using the 1940s and 2010s imagery. We drew polygons around all woody patches and split them into multiple parts as necessary to account for areas of different tree cover. We assigned each polygon a density score, adapted from the Specht (1970) vegetation classification with height of upper stratum ignored. We classified woody patches with 30–70% cover as open forest, 10–30% cover as woodland, and $<$10% cover classified as open woodland (Table 1). Areas with $\leq$1 tree per 10^4 m^2, including cropland, were defined as grasslands.

The map of 1788 vegetation was constructed by Fensham (1989) using historical sources and vegetation surveys. We digitised and orthorectified it in ArcGIS and then classified it according into the same four vegetation categories as those used for the remote sensed imagery. The scale of the map was coarser, and the precision of vegetation types lower, than in the remote sensing analysis.

2.3. Field assessment

We made field measurements of tree size and density for estimating biomass for the study area. We collected field data at 162 transects on a private property in the south of the study region and at Tom Gibson Nature Reserve (managed by Parks & Wildlife Service, Tasmania) in the north (Fig. 1). These two areas contain most of the vegetation types found in the study region. They also capture a cross section of different land management and conservation practices. We selected transects randomly from the most recent imagery and allocated them to the four vegetation classes.

At each field site, we established a belt-transect 50 m long and 20 m wide. These transects ran either north–south or east–west, randomly decided upon a toss of a coin. We made densiometer readings at three points along the transect and averaged them to assess canopy cover from the ground. We also measured the diameter at breast height (DBH; 130 cm above the ground) of all trees ($\geq$10 cm DBH), and saplings (>1.5 m tall and $<$10 cm DBH). We also assessed trees and saplings as alive or dead, and identified to species.

We measured woody debris in each transect using a modified line intersect sampling method (Marshall et al., 2000). We measured the diameter of all fine woody debris (FWD) (>1 cm–$<$10 cm) and coarse woody debris (CWD) (>10 cm) that crossed the midline of each transect. We calculated the volume per 10^4 m^2 of CWD and FWD for each transect using the formula in which all pieces are assumed to lie almost horizontal:

$$\mathrm{CWD} = \frac{\pi^2}{8L} \times \sum_{i=1}^{n} d_i^2$$

where CWD is the volume of debris (m^3 10^4 m^{-2}), L is transect length (m) and d_i is the diameter of the ith branch or log (cm) (Marshall et al., 2000).

2.4. Biomass calculations

We estimated the above ground biomass at each transect from the DBH measurements of all trees and saplings, using generic allometric equations (Paul et al., 2013). This equation applies to trees up to 100 cm DBH. Above this size, therefore, carbon may be overestimated. Reliable functions for larger trees are not yet available, however, so their development is an active field of research (Keith et al., 2009). Because only 0.9% of trees sampled were over 100 cm DBH, the error in our biomass estimates is small. We then estimated below ground biomass using a factor of 0.25 for live trees and 0.2 for dead trees (Keith et al., 2000). We then converted biomass to carbon using factors of 0.5 for live trees, and 0.45 for dead trees (Gifford, 2000).

We calculated the carbon in CWD and FWD at each transect using the formula:

$$\mathrm{C} = \text{Volume per} 10^4\text{m}^2 \times \text{Density Factor} \times \text{Density} \times \text{Carbon Factor}$$

We assumed a mean decay factor of 2 and density factor of 0.68 for all CWD. We calculated the mean density of wood before decay from the dominant eucalypts in the study region (*Eucalyptus amygdalina*, *Eucalyptus delegatensis*, *Eucalyptus pauciflora*, *Eucalyptus viminalis*, *Eucalyptus ovata*) as 567.7 kg m^3 (Ilic et al., 2000). The carbon factor was 0.49 (Gifford, 2000).

We determined the carbon density (total mass of carbon per m^2) at each transect by summing above ground biomass, below ground biomass and mass of woody debris. We then calculated average values for each vegetation category (Table 2). These values were multiplied by the ground area of each category for each period in the digital mapping to get a total carbon estimate for 1788, 1940s and 2010s.

2.5. Future tree cover in the Midlands

To estimate the tree cover in the study region into the future, we used Williamson et al.'s (2014) eucalypt cover projections for 2070.

Table 1
The vegetation classification categories of the various sources of data used in the study (Specht, 1970; Fensham, 1989).

Specht classification	Tree density	Fensham classification	Field transects (n)
Grassland	$\leq$1 tree 10^4 m^2	Treeless	38
Open woodland	$<$10% canopy cover	Open-woodland	28
Woodland	10–30% canopy cover	Woodland	24
Open forest	30–70% canopy cover	Forest	72

Table 2
Average carbon density estimates (kg m^{-2}) and standard error (S.E.) for woody debris, standing dead and live trees by vegetation category. The proportion of each component for each vegetation group is also presented. The mean density of stems per 10^4 m^2 and diameter at breast height (DBH), total and broken down by *Eucalyptus* and *Acacia*, for each vegetation category is included.

	Grassland (<1 tree 10^4 m^2)	Open Woodland (<10% cover)	Woodland (10–30% cover)	Open Forest (30–70% cover)
Total carbon (±S.E.)	1.29 (0.63)	5.0 (1.54)	9.75 (1.61)	17.73 (1.27)
Woody debris	4%	7%	8%	9%
Standing dead	33%	41%	28%	16%
Live trees	63%	53%	64%	75%
Stems 10^4 m^{-2}	10.3	36.4	146.7	730.4
Eucalyptus	59%	62%	28%	36%
Acacia	10%	35%	71%	59%
Mean DBH (±S.E.)				
Eucalyptus	54.0 (6.5)	54.8 (4.3)	40.5 (2.4)	26.2 (0.6)
Acacia	17.3 (2.8)	27.3 (2.8)	11.1 (0.9)	5.5 (0.1)

These projections are based on the relationship derived between eucalypt cover and current mean annual precipitation and annual water balance for southeast Australia. Three dynamically downscaled general circulation models (GCMs), run under a high greenhouse gas emission scenario (the A2 scenario from the IPCC Special Report on Emissions Scenarios (SRES), provided the basis for Williamson et al.'s (2014) climate projections (Nakicenovic and Swart, 2000)). The three GCMs were chosen to represent the most contrasting future predictions; (1) CSIRO-Mk3.5, (2) MIROC3.2 (medres), and (3) UKMO-HadCM3. Based on these three GCMs, Williamson et al. (2014) projected the following changes in tree cover for the northern Midlands: CSIRO-Mk3.5, −13.2%; MIROC3.2 (medres,), −9.8%; and UKMO-HadCM3, +13.0%. We used these values to predict carbon stores in 2070, based on the current values estimated in this study.

2.6. Planting to replace carbon stores

We estimated how much land area would need to be replanted with eucalypts to replace the carbon lost from the landscape since European settlement under the three future GCM scenarios. We created buffers of varying widths along all property boundaries, roads and rivers in currently treeless or lightly treed areas and modelled them as revegetation plantings. We omitted 102.76 km^2 of the study area proposed for an extensive irrigation system. We assumed restoration plantings with a carbon density of 13.2 kg m^{-2} under current climate conditions. This value was based on the average biomass density of mature dry eucalypt forests in Tasmania estimated by Moroni et al. (2010), with 9% added to the original 12.1 kg m^{-2} to account for CWD for forests (Table 2). We scaled the total area of restoration planting for each buffer by changes in the tree cover from Williamson et al. (2014), outlined above.

3. Results

3.1. Biomass and carbon

Comparing of the estimates of remotely sensed tree cover with ground-based densiometer measurements of cover at 164 transects validated our estimate of percentage canopy cover. The mean densiometer values correlated strongly (r^2 = 0.91) with, but were lower than, the remotely-sensed values of aerial cover.

A monotonic increase in above ground carbon stocks amongst the four vegetation categories occurred, from around 1.3 kg m^{-2}in grassland to 17.7 kg m^{-2} in open forests (Table 2). Amongst all vegetation types, live trees provide storage for over half of the carbon. Woody debris stored less than 10%. We found marked differences in the importance of standing dead trees in storing carbon. This pool accounted for only 15% of the carbon in the open forest and over one third in the grassland and open woodland, reflecting the impact of the decline in paddock trees. For all vegetation types, the biomass measured was concentrated in eucalypts. These trees occur at lower density but are typically larger than acacias (Table 2).

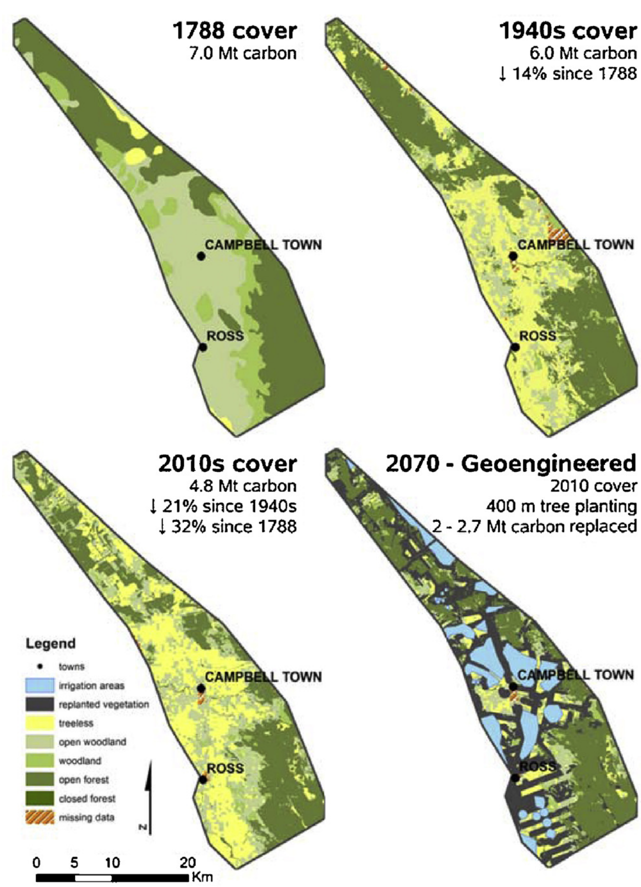

Fig. 2. A time series of woody cover change in the study area from 1788 to 2070. The 1788 map is based on a reconstruction by Fensham (1989), 1940s based on aerial photography and 2010s based on high resolution satellite imagery. The 2070 geoengineered scenario demonstrates the replacement of the 1788–2010s carbon debt and shows 2010s vegetation cover, proposed irrigation areas (blue) and 400 m wide strips of vegetation planted around all roads, property boundaries and rivers at a carbon mass of 13.2 kg m^{-2}.

3.2. Tree cover

From 1788 to 2010s, tree cover declined in the Midlands with a corresponding increase in grassland (Fig. 2). Between 1788 and 1940, one third of the woodlands was lost, as well as approximately one half of the open woodland. The area of open forest remained similar, however (Fig. 3). Between 1940 and 2010, a further decrease in woody cover occurred, with over half the study region converted to grassland. The clearing of open woodland and open forests largely drove this loss.

3.3. Carbon stores

Land clearance resulted in an estimated loss of about one third of the carbon (7–4.8 Mt) from the study area since European arrival (Fig. 4). Most of this loss occurred after WWII. In the ~160 years between 1788 and the late 1940s, approximately 1 Mt (1.56 kg m^{-2}) of carbon was removed from the landscape. In the ~60 years between 1940s and 2010s, about 1.24 Mt of carbon (1.92 kg m^{-2}) was lost from the study area. The clearing of open forests drove these larger losses after WWII, which totalled 19% from the estimated 4.64 Mt of carbon in 1788 (Fig. 3).

3.4. Replacing carbon

To restore the natural vegetation and replace the carbon lost through human activites from 1788s to 2010s, an estimated 178.42 km^2 would need to be planted at a carbon store value of 1.32 kg m^{-2} under current climate conditions (Fig. 5). The area of planting required represents nearly 28% of the entire study region. This area is equivalent to more than 400 m wide eucalypt plantings along every road, property boundary and river (Fig. 2). Under the UKMO-HadCM3 model, carbon storage potential is enhanced compared to both current conditions and the other climate models. Carbon storage in 400 m wide plantings, therefore, could be a total of 0.3 Mt more than if grown under current climate conditions. This storage would be 0.6 Mt more than in the least optimistic climate model, MIROC3.2(medres). The remaining two GCMs would lead to total carbon that was between 0.23 and 0.31 Mt short of replacing the 2.24 Mt carbon debt (Fig. 5). The post-WWII debt is almost replaced with 100 m wide buffers (~10% of study area) under current conditions and UKMO-HadCM3. Under all climate models and current climate, 40 m wide buffers (covering 3.9% of the study region) approximately restore the 1990 carbon debt. These buffers would contribute towards satisfying the requirements of the Kyoto protocol.

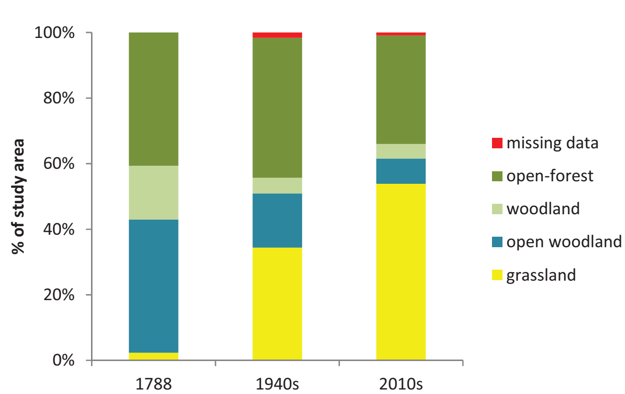

Fig. 3. Changes in the area occupied by different vegetation categories from 1788 to 2010s. Small areas of the remote sensed imagery were not available, resulting in areas of missing data.

If the status quo is maintained and no attempts are made to restore carbon into the landscape, the most optimistic climate model for 2070 used here, UKMO-HadCM3, predicted an increase of 0.62 Mt carbon on current stores (Fig. 4). Little difference was found between the remaining two climate models, which predicted decreases of 0.63 Mt (CSIRO-Mk3.5) and 0.47 Mt (MIROC3.2(medres)) compared with current stores.

4. Discussion

While considerable study has focused on changes in carbon stocks in various woodland vegetation types (Hughes et al., 2006) and changes in extent of temperate savanna (Brewer and Vankat, 2004), few estimates exist of loss of biomass that accompanied the transformation from indigenous management to modern agricultural systems. Houghton et al. (1999) calculated carbon fluxes across North America from 1700 to 1990 arising from changes in land use, including, cropping, wildfire and woody regrowth. To our knowledge, this work is the first attempt to account for carbon loss due to agricultural transformation in Australia.

4.1. Biomass calculations

Our calculated mean carbon density for open forests in the Midlands of 17.7 kg m^{-2} is well within the range of 15.0–25.0 kg m^{-2} reported for dry sclerophyll forests (Grierson et al., 1989; Roxburgh et al., 2006). This estimate is higher than the global value for cool temperate dry forests used by the IPCC, 12.1 kg m^{-2} (Keith et al., 2009). While carbon stores in grassy woodlands are not well documented, the 5.0 kg m^{-2} here in open woodlands are comparable to the range of 3.2–8.7 kg m^{-2} found in New South Wales (Young et al., 2005).

Because no allometric equations exist specifically for the species found in these Tasmanian woodlands and open forests, we used a generic equation (Paul et al., 2013). This equation was developed for trees up to 100 cm DBH, but some of our trees were larger, so we had to extrapolate outside of this range for the larger trees. We therefore potentially overestimated biomass because large trees are often damaged, senescent and contain hollows (Chambers et al., 2001; Roxburgh et al., 2006). A good relationship is unlikely between biomass and diameter for these large veteran trees.

4.2. Historical estimate of changes to land cover and carbon storage

The vegetation reconstruction by Fensham (1989) presents a landscape that was dominated by open woodland or savanna. The savannas allowed the creation of a 'neo-European' landscape by early colonists with relative ease despite clearing being undertaken manually. This rapid clearing of temperate savanna for agriculture is a pattern repeated in large swathes of North America (Conner et al., 2001; Brewer and Vankat, 2004). While we found the carbon loss in the 160-year period between hunter-gatherer management and WWII to be comparatively modest, the decrease in extent of woodlands and open woodlands was dramatic because the early settlers targeted these systems (Fig. 3). We acknowledge the uncertainty in using Fensham's (1989) reconstruction of 1788 vegetation patterns. Even if this reconstruction is near accurate, we could not predict the exact species composition, age- and size-classes and spatial distribution or patchiness of vegetation, which would affect carbon store estimates. For example, large trees contain and sequester more carbon than smaller ones (Stephenson et al., 2014). Many of the trees lost from the landscape were probably large, so we are likely to have underestimated early carbon stores.

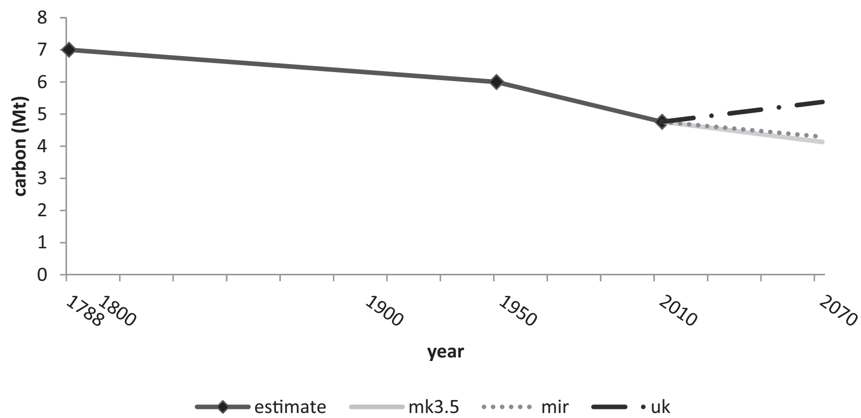

Fig. 4. Changes to total above ground carbon stocks in the study region over time. Less carbon was lost in the 160 years between 1788 and 1940s than during the 60 years between 1940s and 2010s, reflecting the commencement of clearing open forests after WWII. Carbon in the landscape in 2070 was predicted using three climate change scenarios and assuming no attempts were made to restore native vegetation. The most optimistic model was UKMO-HadCM3, which predicted more carbon by 2070 than is currently in the landscape because of greater eucalypt cover. There was little difference between the remaining two models, both predicting stores lower than in the current climate. Climate model abbreviations are UK = UKMO-HadCM3, mk35 = CSIRO-Mk3.5 and mir = MIROC3.2(medres).

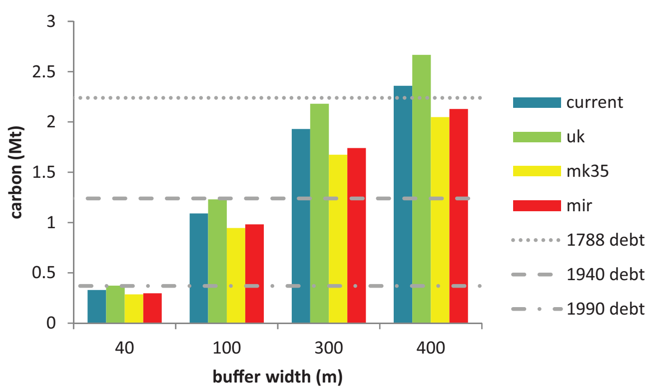

Fig. 5. Revegetation plantings at the carbon density equivalent to a mature dry eucalypt forest (13.2 kg m^{-2}) estimated under different GCMs. More than 100 m buffers around all property boundaries, roads and rivers would be needed to make up the carbon debt accrued from 1940s to 2010s, and approximately 400 m to replace the 1788 debt. The most optimistic model (UKMO-HadCM3) predicts far more carbon in all planting configurations. The debt from 1990 to 2010s would require a little more than 40 m wide plantings to be replaced. The additional loss of carbon that would occur from 2010s to 2070 has not been accounted for here. Climate model abbreviations are UK = UKMO-HadCM3, mir = MIROC3.2(medres) and mk35 = CSIRO-Mk3.5.

The aerial imagery used to examine changes to the landscape between WWII and the present time allowed a greater precision in vegetation mapping and carbon estimates. More carbon was removed from the landscape in the 60 years post-1940s than in the 160 years before (Fig. 4). The greater carbon loss was due to the intensification of agriculture in the region, supporting the notion of 'the great acceleration' in the latter stages of the Anthropocene (Steffen et al., 2007). Although the area of open woodlands continued to decrease appreciably, most of the carbon lost in this period was from open forests, which were targeted as more open, productive areas became scarce (Fensham and Kirkpatrick, 1989). Deliberate removal caused much of the tree loss in the region, especially since the 1970s including 'secondary clearing' such as removal of single trees that stood in the way of increasing larger agricultural machinery and pivot irrigation infrastructure (Prior et al., 2013). The remaining patches of native vegetation are mostly located in areas unsuitable for agriculture, so they are relatively safe from deliberate clearance into the future. However, rural tree decline, a syndrome of premature dieback of trees due to a constellation of stressors including agricultural fertilisers, insect and mammalian herbivores, drought and heat stress, is also a major contributor to tree cover loss (Neyland, 1999; Close and Davidson, 2004; Davidson et al., 2007).

In south eastern Australia and other 'neo-European' regions, biomass has increased in remnant forests and woodlands (Birdsey et al., 1993; Asner et al., 2003; Gibbons et al., 2008; Geddes et al., 2011; Prior et al., 2013). This trend was not evident in our study, but we did detect high density of *Acacia* species with small diameters in the understory of open forests. Although *Acacia* contributed only a small amount (8%) to the estimated biomass of the forests, the abundance of these early-successional plants signals forests recovering from past disturbances (Onans and Parsons, 1980; Grant and Loneragan, 2001; May and Attiwill, 2003). In this area, disturbances were most probably frequent fires set by grazers to increase the abundance of forage for stock and low intensity logging for fencing and firewood.

4.3. Future change to land cover and carbon storage

The main factors likely to affect future tree cover in the Midlands is a new round of intensification of agriculture associated with planned large irrigation schemes, and the effects of climate change on remnant tree populations (Benayas et al., 2008; Plantinga and Wu, 2014). The predictions of future climate change in the Midlands showed considerable variability, primarily because MAP differs under the different climate predictions. The most optimistic model, UKMO-HadCM3, predicted increased rainfall in the Northern Midlands, leading to an increase in the cover of eucalypts. This increase is possible because of enhanced tree growth and survival, and greater recruitment of juveniles. Two of the GCMs predicted hotter and generally drier conditions, and therefore decreased eucalypt cover. This decrease would be a result of both increased mortality and decreased recruitment, which have already been observed in the region (Neyland 1999; Close and Davidson 2004). Thus, although available climate projections show variable trends, tree cover and carbon storage will likely decline due to reduced water balance (Williamson et al., 2014).

Limited water availability and decreasing rainfall since the 1980s have, until now, restricted the extent of cropping in the Midlands (Close and Davidson, 2004). The implementation of an extensive irrigation scheme, however, will encourage a change from grazing to cropping enterprises. Our estimates based on current plans indicate that a 2.4% loss of biomass will occur through clearing for irrigated crops. Additional irrigation water in the landscape could prove detrimental to tree cover through exacerbating dryland salinity (Bastick and Walker, 2000;

Bastick and Lynch, 2003; Davies and Barker, 2005). Given that water balance is crucial in determining eucalypt cover (Williamson et al., 2014), the interactive effects of irrigation and climate change on the persistence or reestablishment of trees in this landscape are worthy of further investigation.

4.4. Replacing the carbon debt with plantations

Establishing restoration plantations has been promoted as a practical way of mitigating increasing CO_2 (Houghton et al., 1999; Bailey, 2013; Plantinga and Wu, 2014). The Midlands plantations are unlikely to repay the current carbon debt incurred because of clearing since European colonisation. To replace the 1788 debt, we suggest that mature plantings at the carbon density of mature open forests would need to cover almost 28% of the study region. These plantings should be located in currently cleared, agricultural land and subject to climate conditions no worse than the current climate. A number of factors make this proposition unreasonable. The density of these plantings is unattainable by 2070. Because of time lags between planting and maturity, temperate sclerophyll forests make take more than 150 years to attain 90% of their carbon carrying capacity and more than 100 years to develop structural complexity (Roxburgh et al., 2006; Vesk et al., 2008). Additionally, densely planted revegetation has less natural recruitment and individuals have slower DBH growth (Vesk et al., 2008). Climate change may increase drought-induced mortality (Allen et al., 2010), increase the risk of destructive fires (Flannigan et al., 2013), and make insect or pathogen outbreaks more frequent (Nelson et al., 2013), thus threatening the persistence of plantations. Although CO_2 fertilisation increases water use efficiency, concurrent increases in tree growth rates have not been observed (Silva et al., 2010; Peñuelas et al., 2011), Therefore, it is unlikely that growth rates will be higher than those observed today. The usual carbon accounting procedure pays a unit price per tonne of CO_2 offset or captured annually (Cacho et al., 2003). Despite this, it is highly unlikely that almost 20% of an agricultural landscape would be surrendered to carbon storage even with financial incentives. It is much more feasible to repay the carbon debt with tree plantings using the 1990 baseline employed by the Kyoto Protocol (Van Kooten, 2000; Cacho et al., 2003). Our calculations suggest that this would entail restoring approximately 4% of the study area with high density plantations. These plantations, while young and with high stem density, would have high carbon sequestration rates per area despite lower per stem accumulation (Xue et al., 2011; Paul et al., 2015). Expecting agricultural regions such as the Tasmanian Midlands to be carbon neutral is unrealistic, so carbon sequestration programs are constrained and best thought of as a subsidiary rather than primary objective. A sensible way to maximise carbon capture for climate change mitigation in Tasmania would be to protect established forests while increasing the forest estate with areas of high-density restoration plantings in underutilised agricultural regions.

Even if restoring carbon to pre-European levels is an unattainable goal, trees must be re-established in this landscape. Biodiverse plantings can remediate damaged agricultural land and restore some of the ecosystem services that have been lost with land-use change (Jackson et al., 2005). Co-benefits of carbon sequestration can include reducing nitrogen runoff, erosion control and salinity control (George et al., 1999; Vesk et al., 2008; Plantinga and Wu, 2014). Salinity is already a risk in the drier parts of the Midlands (Davies and Barker, 2005). A transition to irrigated agriculture is likely to exacerbate this risk (Kokkoris, 2003). Greening Australia's newly established plantings in the region are located on floodplains and along rivers. These areas are sensibly chosen to be beneficial for re-establishing connectivity for wildlife (Bailey, 2013). They are more likely to support faster tree growth than less productive sites (Vesk and Mac Nally, 2006).

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