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Managing temperate forests for carbon storage: impacts of logging versus forest protection on carbon stocks

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Abstract. Management of native forests offers opportunities to store more carbon in the land sector through two main activities. Emissions to the atmosphere can be avoided by ceasing logging. Removals of carbon dioxide from the atmosphere can be increased by allowing forests to continue growing. However, the relative benefits for carbon storage of managing native forests for wood production versus protection are contested. Additionally, the potential for carbon storage is impacted upon by disturbance events, such as wildfire, that alter the amount and longevity of carbon stocks.

Using a case study of montane ash forests in southeastern Australia, we demonstrated that the total biomass carbon stock in logged forest was 55% of the stock in old growth forest. Total biomass included above- and belowground, living and dead. Biomass carbon stock was calculated spatially as an average across the landscape, accounting for variation in environmental conditions and forest age distribution. Reduction in carbon stock in logged forest was due to 66% of the initial biomass being made into products with short lifetimes ($<$3 years), and to the lower average age of logged forest ($<$50 years compared with $>$100 years in old growth forest). Only 4% of the initial carbon stock in the native forest was converted to sawn timber products with lifetimes of 30–90 years.

Carbon stocks are depleted in a harvested forest system compared with an old growth forest, even when storage in wood products and landfill are included. We estimated that continued logging under current plans represented a loss of 5.56 Tg C over 5 years in the area logged (824 km^2), compared with a potential gain of 5.18–6.05 TgC over 5 years by allowing continued growth across the montane ash forest region (2326 km^2). Avoiding emissions by not logging native forests and allowing them to continue growing is therefore an important form of carbon sequestration. The mitigation value of forest management options of protection versus logging should be assessed in terms of the amount, longevity and resilience of the carbon stored in the forest, rather than the annual rate of carbon uptake.

Key words: carbon accounting; carbon emissions; carbon storage; disturbances; *Eucalyptus regnans*; forest management logging; montane ash forest; native forest; old growth forest.

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Introduction

With the current imperative to address the problem of climate change, it is widely recognized that maintaining and increasing stocks of carbon in forests is an important component of a comprehensive approach to mitigation (Nabuurs et al. 2007, Mackey et al. 2013). Forest management offers opportunities to store more carbon on the land than occurs currently. Protecting native forests avoids emissions due to logging and sequesters carbon through regrowth. This increasing stock of carbon in regrowth represents a replacement of the carbon previously emitted from deforestation and degradation (Houghton 2003, 2010, Olofsson and Hickler 2008). However, the type of forest management that promotes greatest carbon storage is subject to debate, with controversy between managing native forests for commercial wood production versus protecting forests (Harmon 2001, Kirschbaum 2003, Bellassen and Luyssaert 2014). Debate concerns the relative benefit for carbon storage of these management options as well as the different methods for quantifying these benefits. Evaluating the potential for carbon storage in different forest management systems must incorporate the impacts of emissions from natural disturbances such as wildfire, which occur in both protected and logged forests, as well as emissions from anthropogenic disturbances by logging. Such an evaluation identifies whether there are differences between these disturbance processes that should be included in carbon accounting systems. The processes involving stocks and fluxes of carbon must be estimated at appropriate temporal and spatial scales.

While the reduction in carbon stocks due to management activities is poorly quantified in most forest regions, cool moist or oceanic temperate forests (FAO 2001, Pan et al. 2013) warrant particular attention as they have high biomass carbon densities and are impacted by both natural disturbances (wildfire) and anthropogenic disturbances (logging) (Keith et al. 2009). Furthermore, these temperate forest ecosystems globally support a wide range of ecosystem services in addition to carbon storage, including regulating the supply and quality of fresh water, biodiversity conservation and landscape aesthetic values, as well as the production of timber and pulpwood, resulting in competing land use demands. Examples of these carbon-dense temperate forests include *Tsuga heterophylla, Picea sitchensis, Pseudotsuga menziesii* and *Abies amabilis* in the Pacific Northwest of North America (Fujimori et al. 1976, Grier and Logan 1977, Grier et al. 1981, Keyes and Grier 1981, Means et al. 1992, Smithwick et al. 2002, Leighty et al. 2006), *Eucalyptus regnans* and *E. obliqua* in Australia (Dean et al. 2003, Keith et al. 2009), *Agathis australis* in New Zealand (Silverster and Orchard 1999), and *Nothofagus dombeyi, Fitzroya cupressoides, Philgerodendron uviferum and Laureliopsis philippiana* in Chile (Battles et al. 2002, Carmona et al. 2002, Vann et al. 2002, Romero et al. 2007, Schlegel and Donoso 2008).

In the investigation we report here, we use the native montane ash forest in the Central Highlands of Victoria, southeastern Australia, as a case study for evaluating the impact on carbon stocks of logging compared with disturbance by wildfire that would occur periodically in protected forests. We investigated: (1) average carbon stocks in forests of different ages, (2) processes of carbon stock loss, and (3) the potential for increasing stocks in these forest types. We used results from our case study to evaluate alternative forest management strategies. Our objective was to provide quantitative information about the impacts of natural and anthropogenic disturbances on carbon stocks in native temperate forests.

Our study contributes new insights into the issue of quantifying carbon stocks by addressing the following key questions:

1) *What are the amounts and longevities of carbon stocks in a clearcut logging system?*

In the context of carbon accounting, logging represents transfers of carbon stocks within the forest and production system. Biomass carbon is removed off-site and a proportion is stored in wood products and landfill. Carbon is emitted through combustion where the slash is burnt, as well as from decomposition of dead biomass from the slash remaining after harvesting and waste during processing. Carbon is sequestered in the regenerating forest.

2) *How do carbon stock losses from logging compare with those from natural disturbance events such as wildfires?*

Purported similarities in the impacts of natural and anthropogenic disturbances on forest dynamics have been used as arguments to support continued logging in native forests (Attiwill and Adams 2013, Norris et al. 2010). Differences between these disturbance regimes have been demonstrated in terms of their impacts on vegetation structure, plant species composition and animal habitats (Lindenmayer and McCarthy 2002). We address this question in terms of the impacts of natural and anthropogenic disturbance regimes on the dynamics of carbon stocks.

3) How can the climate change mitigation value of forests be optimized?

From a climate change mitigation perspective, the key management issues are how best to avoid carbon emissions and maximize long-term carbon storage in forests (Nabuurs et al. 2007, Canadell and Raupach 2008, Mackey et al. 2013). In the investigation we report here, we used scenarios of forests managed under different strategies to predict the potential of forests to increase carbon stocks over time.

Methods

Study area

Location.—Our study region was the Central Highlands of Victoria in southeastern Australia (37°20′–37°55′ S and 145°30′–146°20′ E; Fig. 1). The region's native forests have a long history of management for multiple ecosystem services in addition to the industrial-scale logging for timber and fiber. These services include water supply, biodiversity conservation of threatened fauna, tourism, as well as carbon storage (Lindenmayer 2009, Viggers et al. 2013). Forests in this region occur mainly within State Forests that are used for logging (69% of the area within the region); and the remainder occur in conservation reserves. In our study, we mapped the Central Highlands region (the mapped square in Fig. 1), but constrained our analysis of logged areas and carbon stocks within the montane ash forest type (shown in Fig. 1 as the dominant species of *E. regnans*) (total area of 2326 km^2).

Montane ash forests are evergreen cool, moist temperate forests consisting predominantly of *Eucalyptus regnans* (F. Muell.), with some *E. delegatensis* (R.T. Baker) and *E. nitens* (Maiden). They are among the most carbon-dense forests in the world (Keith et al. 2009). The climate in the Central Highlands is characterized by cool, wet winters with occasional snow, and mild summers. This forest type occurs at altitudes between 400 and 900 m. The forests are multi-layered with an overstory of eucalypts, a mid-story tree stratum of rainforest species in moist, protected locations, and a tall shrub layer including tree ferns (2 m to 15 m). Dominant height of trees is 60–80 m and diameter (at 1.3 m height) can be in excess of 4 m (Lindenmayer et al. 2000). Old-growth forests are multi-aged and consist of a range of tree sizes related to differing histories of disturbance and regeneration, with different species occupying different size/height categories. Typical ages of old growth trees are 120–250 years, but a maximum age is considered to be up to 400–500 years (Wood et al. 2010).

Disturbance history.—Logging is the primary form of anthropogenic disturbance in montane ash forest and began in State Forests in the 19th century with selective logging but this was increasingly intensified during the 20th century. A wildfire in 1939 burnt most of the Central Highlands region and the forest was salvage logged for several decades (Noble 1977, Lindenmayer et al. 2008). Most of the unburnt old growth montane ash forest available in State Forests had been logged by about 1990. Logging of the 1939 regrowth commenced in the mid-1980s and is currently continuing. This history of logging is shown in Fig. 2 by decade of occurrence of cut blocks over the last century.

Water catchments that provide the water supply for the city of Melbourne have experienced minimal logging and other impacts of anthropogenic disturbance, except for the occurrence of wildfires (Viggers et al. 2013). The Maroondah, O'Shannassy and Upper Yarra catchments have been mostly closed for over 100 years and there is little evidence of Aboriginal use of these dense forests (Lindenmayer 2009). Other catchments have experienced small but increasing areas of logging, such as the Cement Creek and Armstrong catchments, or extensive logging over the last 50 years, such as Starvation Creek, McMahon's Creek, Thomson and Tarago catchments (Viggers et al. 2013). Twenty percent of the State Forest area is within these catchments.

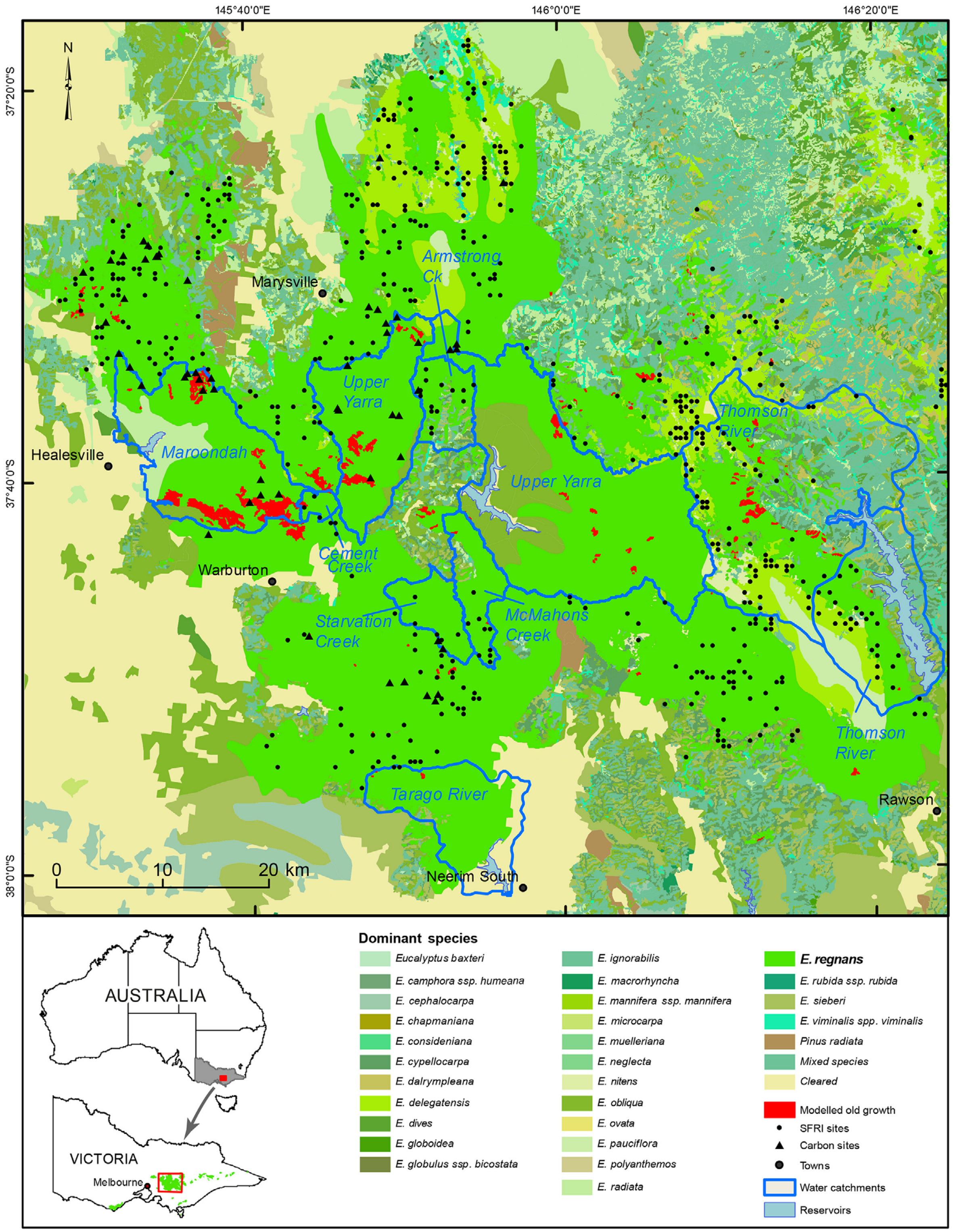

Fig. 1. Location and extent of montane ash forest type in the Central Highlands of Victoria, with the distribution of forest types, old growth forest, and location of the carbon and SFRI sites.

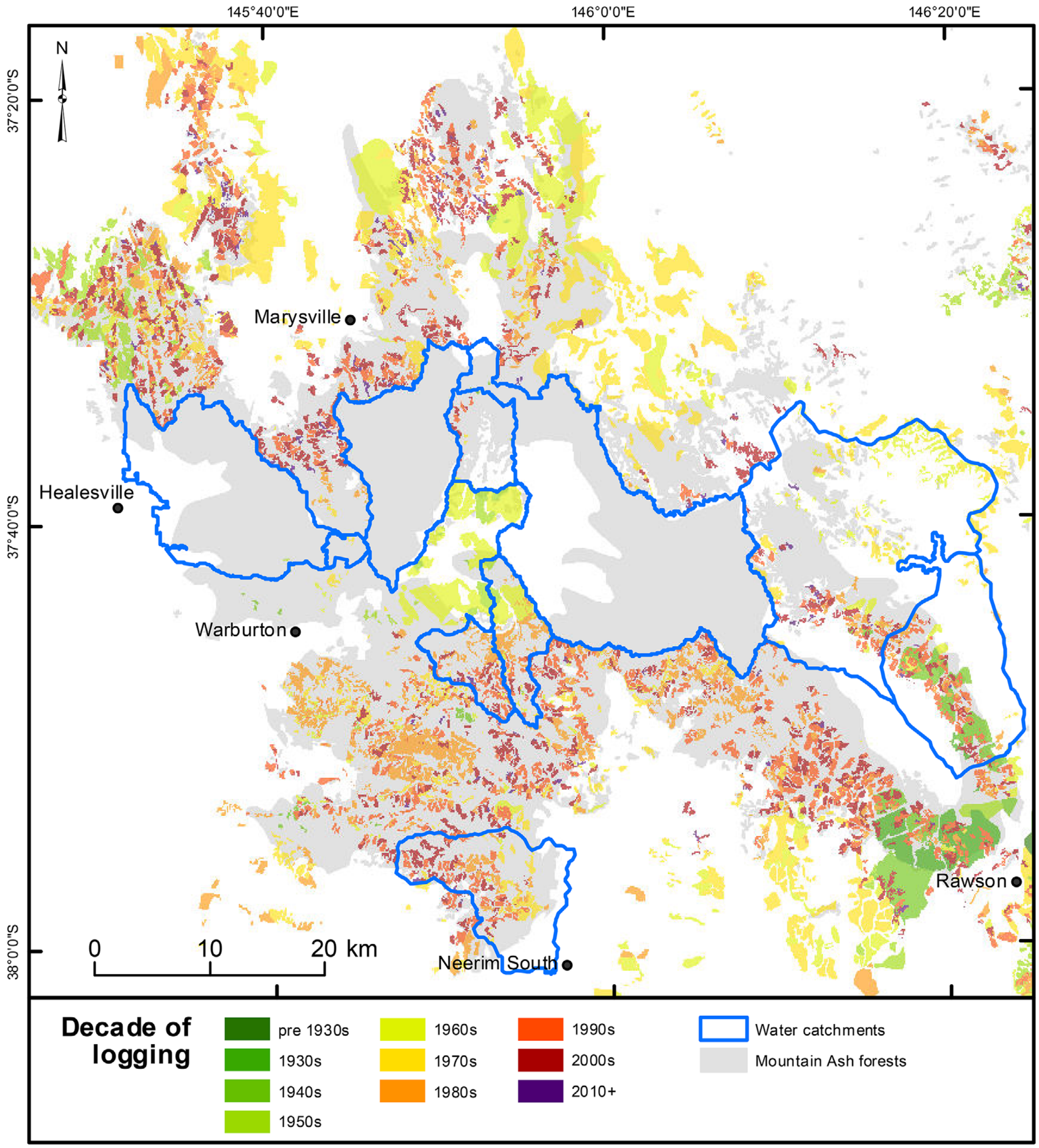

Fig. 2. Spatial distribution of historical logging is shown by decade over the last century in the montane ash forest region.

The natural disturbance regime in montane ash forest is infrequent, high severity wildfires (return interval of approximately two per century, McCarthy et al. 1999), although these fires increasingly have a human component (Mackey et al. 2002, Lindenmayer et al. 2011). *Eucalyptus regnans* and *E. delegatensis* are killed in severe fires if their canopies are fully scorched. They regenerate by shedding seed following the fire which germinate in the ash bed, thus forming even-aged forest stands (Ashton 1976). However, fire events are variable spatially in both extent and severity. In areas of less severe fire, the resulting partial tree death and subsequent regeneration alongside surviving trees leads to multi-aged stands (Smith and Woodgate 1985, McCarthy and Lindenmayer 1998, Mackey et al. 2002). Forest age distribution across the landscape, therefore, represents a mosaic of even-aged and multi-aged stands depending on fire severity. This variability in impacts of wildfires on tree survival has been difficult to assess from historical records where only the outer fire boundary was mapped. However, mapping of fire severity in recent fires has allowed identification of the high severity areas that result in regeneration (DSE 2009).

Currently, more than 98% of the montane ash forest in the Central Highlands is less than 73 years old due to mortality and salvage logging after wildfires in 1939, 1983 and 2009, and logging of older forests (Lindenmayer et al. 2012).

Logging systems.—Clearcutting and slash burning is the logging system commonly applied in montane ash forests in Victoria (Lutze et al. 1999, DSE 2007a, Flinn et al. 2007). The rotation length for clearcutting is prescribed as 50 years (DPI 2009) where even-aged 1939 regrowth is being logged, but has often been shorter. Clearcutting has been highly variable in the extent and proportion of biomass removed and success of subsequent regeneration.

Stemwood that has commercial value as wood products is removed off-site. Biomass remaining on-site consists of stumps, crowns, branches, bark, wood with defect, non-commercial species, fallen dead trees, litter and understory. This biomass remaining is burnt at high intensity to clear the ground surface of debris, release seed stored in capsules by heating the soil, and produce a layer of ash that improves conditions for seed germination (Slijepcevic 2001). The dead biomass that is not combusted, plus the belowground biomass, decompose over time and emit carbon dioxide to the atmosphere.

Harvested wood is used either for sawlogs or pulpwood with the products being sawn timber or paper. At each stage of processing of wood products, waste is produced that has short product lifetimes, such as woodchips, shavings and sawdust. Some of these materials are used as bioenergy during processing and others decompose rapidly. At the end of the life of each type of product, the carbon in the wood or paper may be combusted as waste or energy or added to landfill from which some decomposes and releases carbon dioxide and methane. Decomposition in landfill is highly variable depending on size of the product and the conditions at the site. Hence there is a high degree of uncertainty in estimated rates of carbon emissions (Skog and Nicholson 1998).

We used values reported in the literature for the proportions of biomass transferred in each of these stages in the wood products process. We tracked the carbon stock loss due to logging by the proportion of total biomass that was removed off-site, the products of harvested sawlog or pulpwood, and the proportion remaining on-site as slash.

Measurement of current carbon stock

We estimated current carbon stocks at two sets of sites: 54 long-term field research sites and 876 State Forests inventory sites. Total carbon stock of living and dead biomass, including above- and belowground, was estimated and related to dominant age of the forest at each site. We defined biomass as all intact organic components of the ecosystem, both living and dead, but excluding soil organic matter.

Field research sites.—We have an established network of long-term ecological research sites within the montane ash forest region of the Central Highlands of Victoria (Lindenmayer et al. 2003). From these, we selected 54 sites encompassing three forest ages by three disturbance histories to give nine categories, each with six sites (Fig. 1). These sites were categorized by: (1) forest age: young regrowth after clearcut logging or salvage logging after wildfire in 1983; regrowth after wildfire in 1939 and salvage logging; and old growth, and (2) fire severity in the 2009 wildfire: unburnt, low severity fire, high severity fire. The regrowth sites represented common age classes of montane ash forest that were usually considered even-aged. The old growth sites consisted of the oldest trees that had regenerated after a fire in approximately 1750 (Lindenmayer et al. 2000).

At our 54 sites, we estimated all biomass components in the forest (living biomass above- and belowground, dead standing biomass,

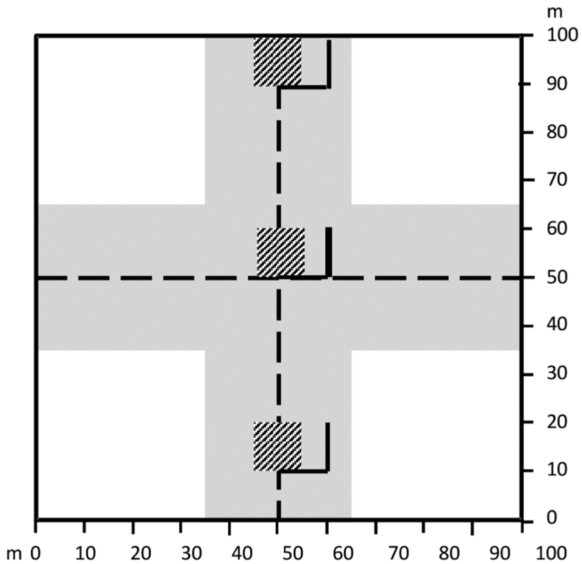

Fig. 3. Design of 1 ha (100 m × 100 m) experimental sites showing the central and perpendicular transects (dashed lines), 10 m × 10 m plots (hatched), 10m transects (black lines), and area sampled for large trees 2 × 30 m × 100 m = 0.51 ha (grey shaded area). Different biomass components were sampled in different parts of the site.

coarse dead and downed woody debris, and litter) to calculate the current carbon stock. We used different sampling strategies within the 1 ha sites for each biomass component to maximize accuracy and efficiency of measurements of different sizes, densities and distributions of components (Fig. 3, Table 1).

Stem and branch volumes were calculated using an allometric equation for *E. regnans* (Sillett et al. 2010) based on tree diameter (accounting for stem buttressing and internal wood decay), and multiplied by wood density and carbon concentration. An average root:shoot ratio for eucalypt forests was used to convert above-ground biomass to total biomass (Snowdon et al. 2000). Volume of coarse dead and downed woody debris was measured using the line intersect method, and converted to biomass using measured bulk density of different decay states of wood (McKenzie et al. 2001). Litter depth was measured along transects at all sites and related to litter biomass collected from quadrats on the forest floor. We describe the details of the sampling strategies, measurements and calculations for each component in Appendix A.

Inventory sites.—We used an additional 876 sites with existing forest inventory data (Statewide Forest Resource Inventory, SRFI, DSE 2007b) to calculate carbon stocks. The inventory data were derived from variable radius plots with measurements of tree diameter and probability of occurrence of a given sized tree (DSE 2007b). We calculated carbon stocks of living trees using tree diameter and an allometric equation (Sillett et al. 2010; detailed methods in Appendix A). We estimated the carbon stocks in dead biomass components from average values for the forest type and age (Woldendorp et al. 2002, DSE 2007b). These data provided less

Table 1. Methods for sampling and measuring biomass components within 1 ha sites.

Biomass component	Sampling strategy	Measurement
Trees (living and standing dead)		
<100 cm diameter	Three 10 m × 10 m plots (0.03 ha)	Estimated height and diameter in size categories
>100 cm diameter	Two perpendicular intersecting transects of 100 m × 30 m (0.51 ha)	Measured DBH and height
Understory (living and dead)	Three 10 m × 10 m plots (0.03 ha)	Estimated height and diameter in size categories
Coarse woody debris		
<60 cm diameter	Line intersect method, 6 × 10 m transects	Measured log diameter, decay class, hollows, charcoal and bulk density
>60 cm diameter	Line intersect method, 2 × 100 m transects	Measured log diameter, decay class, hollows, charcoal and bulk density
Litter layer† <2.5 cm diameter	25 points along transects per site; 30 quadrats randomly located at sites to cover range in litter depths	Measured litter depth and litter dry mass

† Litter layer refers to fine litter <2.5 cm diameter that constitutes intact pieces of biomass as distinct from undifferentiated organic material within the mineral soil, and it excludes roots.

precision of carbon stocks than at the 54 field research sites, but yielded greater spatial coverage across the montane ash forest region encompassing a range of environmental conditions and times since disturbance.

Spatial estimation at landscape scales

We scaled up our site-based estimates of carbon stocks across the montane ash forest region by accounting for landscape variability at a resolution of 250 m using the statistical modelling approach of Keith et al. (2010). A multiple regression model was derived (Genstat v.14, Payne et al. 2012) relating site carbon stock (living and dead biomass above- and belowground) to site-specific values of environmental variables and disturbance history. The spatially explicit explanatory variables included: (1) climate parameters including precipitation, water availability index, temperature (minimum, mean, maximum) and radiation; (2) topographic parameters including topographic position, elevation, aspect and slope; (3) substrate parameters including soil parent material, lithology and soil organic carbon content; (4) gross primary productivity (GPP, derived from remote sensing); (5) forest type; (6) forest management area; and (7) forest age class distribution based on the disturbance history. Derivation of the spatial estimates of the environmental variables used in the model are described in Appendix B and were based on Mackey et al. (2002, 2008), Berry et al. (2007) and Keith et al. (2010).

Characteristics of disturbance history included disturbance event type (wildfire, or silvicultural system including clearcut, single tree selection, group selection and thinning), and time since disturbance for the most recent and second most recent events. We stratified our study region according to the most recent disturbance event types in each grid cell. Time since disturbance does not necessarily equate with age of a forest stand as not all disturbances kill all trees (McCarthy and Lindenmayer 1998). The relationship between biomass and time since disturbance represents the biomass (both living and dead) at a site with a given disturbance history, rather than the biomass of an even-aged stand derived from a single regeneration event. Forest age was determined from the year of the most recent disturbance event that resulted in regeneration; that is, high severity wildfire or clearcut logging. This represented the dominant age of the majority of trees. Additionally, other disturbance types that result in partial regeneration were included in the model as a factor of the second most recent disturbance event types. In the regression model, the factors for disturbance types produced different coefficients that accounted for the effect of disturbance type as well as age on the accumulated carbon stock of living and dead biomass. Models for each disturbance type accounted for 25–65% of the variance in the site data of carbon stocks.

Current carbon stock was then predicted for each grid cell by the statistical regression model, thus producing a spatial distribution of carbon density across the landscape. This model predicts current carbon stock as living and dead biomass at one point in time based on the empirical relationship with forest age, disturbance history and environmental conditions.

Comparison of disturbance types

Comparing impacts of the different disturbance regimes of logging and wildfire on carbon stock losses required quantification of the: (1) differences in the carbon density of forests pre- and post-impact, (2) area affected by each disturbance, (3) timeframe over which estimates are calculated, (4) initial condition or baseline for comparison, and (5) longevity of the carbon stock that is lost. Accounting was based on the long-term change in carbon stocks at the regional scale, which is a more robust estimate than the annual carbon flux from a site (McKinley et al. 2011, Ajani et al. 2013).

Potential maximum carbon stock at the landscape scale

Comparison of carbon stock change due to different disturbance types requires a baseline or initial condition. An appropriate baseline is the upper limit of carbon accumulation over time, or the potential maximum carbon stock. This maximum stock occurs where inputs and outputs of carbon in the growing forest are determined by natural environmental conditions and disturbance regimes at the landscape scale (Smithwick et al. 2002, Rhemtulla et al. 2009). This upper limit is referred to as the carbon carrying capacity (Gupta and Rao 1994, Mackey

Table 2. Current carbon stock in living and dead biomass components for different age classes of montane ash forest (mean ± SE; n = 6).

Forest age	Biomass carbon stock (tC/ha)			
	Living trees	Standing dead trees	Woody debris† + litter	Total
1983 regrowth	293 ± 43	34 ± 8	78 ± 15	405 ± 33
1939 regrowth	426 ± 64	89 ± 31	88 ± 25	603 ± 74
Old growth	930 ± 41	41 ± 25	65 ± 9	1039 ± 44

† Woody debris refers to dead and downed woody debris.

et al. 2008, Keith et al. 2009). We used the spatially modelled potential maximum carbon stock derived for native eucalypt forests across southeastern Australia, which have experienced minimal disturbance from human land use activities (Keith et al. 2009). This baseline carbon stock is a landscape-scale metric defined as a range that accounts for variability around the average condition due to climate variability and natural disturbance events. From this baseline, we estimated the carbon sequestration potential for continuing forest growth following disturbance, based on the difference between the current carbon stock and the potential maximum.

Carbon stock change over time

To assess changes over time in carbon stocks under scenarios of different disturbance regimes, we modelled each biomass component and the changes due to regeneration of living biomass, decomposition of dead biomass, and processing of wood products. We simulated changes in carbon stocks using functions derived from existing data and observations for montane ash forest to describe the following processes: regeneration, mortality, collapse of dead trees, and decomposition of dead and downed woody debris (Appendix D: Table D1). Initial carbon stock of the forest was 1040 tC/ha, which is the average value of total biomass carbon at 250 years calculated from the available field data (from Table 2). The landscape represents a mosaic of forest age classes due to wildfires, but it would be expected that the total carbon stock averaged over the area would be similar to the initial condition of an old growth forest.

We described regeneration as the accumulation of carbon stock over time in living biomass (above- and belowground) derived from the relationship between forest age and carbon stock. The relationship used the form of the Chapman-Richards equation (Richards 1959, Janisch and Harmon 2002): $Y = y_{\max}(1 - e^{-kx})^r$ (where $y_{\max}$ is the carbon stock at carrying capacity, k and r are empirically derived constants that determine the spread of the curve along the time axis and the shape of the curve, respectively). We derived the equation using data for carbon stocks of living biomass from sites where time since disturbance reflected predominant age of the trees. These sites included our field research sites ($n = 17$ sites unburnt in 2009), SFRI clearcut sites ($n = 94$) and some data from the literature ($n = 8$).

$$\text{living biomass} = 1200 \times (1 - \exp(-0.0045 \times \text{age}))^{0.7} \tag{1}$$

where living biomass is in tC/ha and age is in years ($r^2 = 0.72$, $n = 119$). In this equation, the three sources of data did not have significantly different relationships.

The maximum carbon stock in living biomass of a forest is required to define the asymptote of this equation. Maximum carbon stock of living biomass occurs in old growth forests, such as our research sites dominated by approximately 250-year-old trees. However, old growth forests of *E. regnans* and other eucalypts can have maximum ages up to 400–500 years (Gilbert 1959, Ogden 1978, Wellington and Noble 1985, Banks 1993, Looby 2007, Wood et al. 2010), and so the maximum stock could be higher than our site values (Stephenson et al. 2014). Defining this asymptote is hampered by limited data for old forests.

Mortality due to self-thinning of the regeneration was estimated from changes in stem density with age, based on data from Ashton (1976) and Watson and Vertessy (1996). Biomass from dead standing trees plus dead branches was transferred to the dead and downed woody debris pool based on a logistic function over a

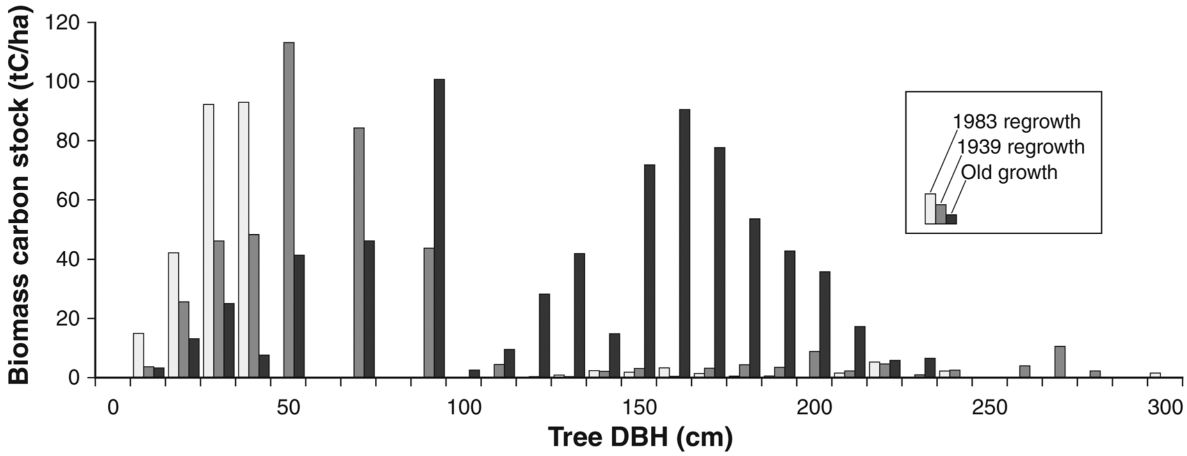

Fig. 4. Biomass carbon density in relation to tree size distribution for the three forest age classes of the field research sites. Biomass contribution per size class of woody stems >2 m height (living and dead) in each age category of unburnt sites ($n = 6$ sites per category). Data for carbon stock in trees were derived from the three 1 m × 1 m plots for stems <100 cm DBH (0.03 ha) and the two 100 m × 30 m transects for stems ≥100cm DBH (0.051 ha) in the 1-ha sites.

time period of up to 75 years (Lindenmayer et al. 2010, Lindenmayer and Wood 2010). Decomposition of woody debris was assumed to be at a rate of 0.07 per annum (default value used in Australian National Carbon Accounting System; Mackensen and Bauhus 1999).

In a harvested montane ash forest under a regime of clearcutting and slash burning on a 50-year rotation, we derived proportions of biomass carbon stock change at each stage of processing based on values in the literature (Appendix E: Table E1 where values for montane ash are shown in bold). Carbon stock lost from the site included 40% of total biomass as wood product removed (Appendix E; Raison and Squire 2007), 50% of the slash remaining at the site due to combustion (Appendix E; Gould and Cheney 2007), and the unburnt slash decomposed at a rate of 0.07 per annum (Mackensen and Bauhus 1999).

Results

Current carbon stocks

Current carbon stocks in living and dead biomass components plus the total biomass are shown for different forest age classes in Table 2, as a mean value from the six sites. Old growth forests had the highest total biomass carbon stock due to the contribution from large, living trees (≥100 cm diameter). Carbon stocks in standing dead trees, dead and downed woody debris, and litter were highest in 1939 regrowth stands. The size distribution of trees showed that the large trees contributed most to biomass carbon density (Fig. 4). These large trees (≥100 cm diameter) contributed 76% of the biomass in old growth sites, but only 43% of tree numbers. A few large trees also occurred in some 1983 and 1939 regrowth sites, having survived disturbances at these times, and they contributed large amounts of the biomass carbon density. Total carbon stocks were higher in sites containing large trees. The numbers of large trees were highly variable on regrowth sites, resulting in highly variable total carbon stocks (Fig. 4). A result of this variability was that the sizes of all trees on the regrowth sites did not exactly correspond to that expected if all trees had regenerated since the disturbance events.

Dead and downed woody debris biomass was highly variable among sites, ranging from 15 to 186 tC/ha, with a mean of 67 tC/ha. Dead and downed woody debris remained after both logging and wildfire events and on average was highest in 1939 regrowth stands. Litter biomass ranged from 6 tC/ha in 1983 regrowth to 9 tC/ha in 1939 regrowth and old growth forests (mean of six sites per age category).

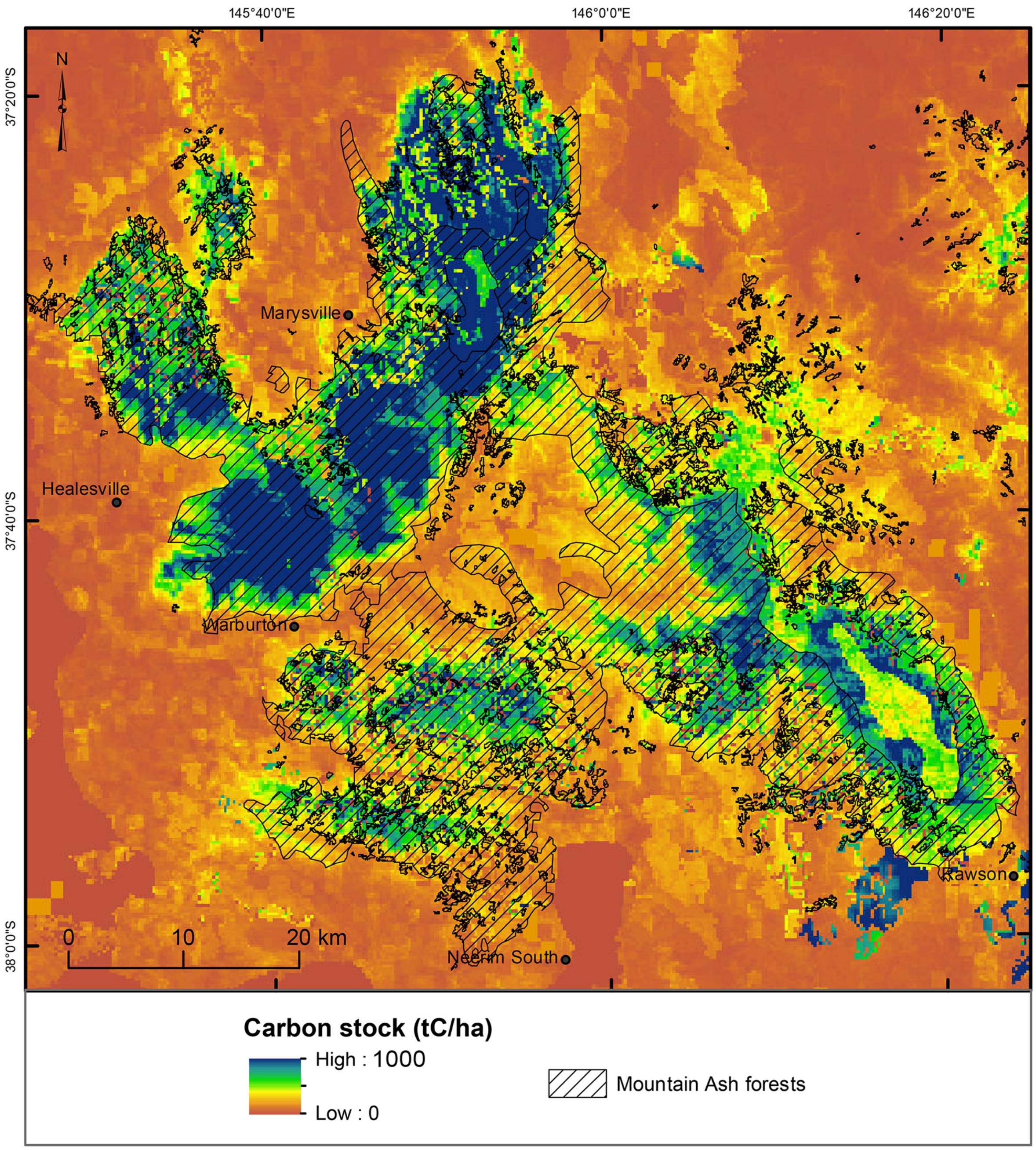

Fig. 5. Spatially modelled carbon density (tC/ha) across the montane ash forest in the study region, including total biomass of living and dead trees, above- and belowground, shrubs, litter and dead and downed woody debris.

Spatial distribution of carbon stock

The spatial distribution of carbon stocks in our study region of the Central Highlands of Victoria is shown in Fig. 5. The distribution of biomass carbon density by area is illustrated as a histogram (Fig. 6) with the mean distribution (green), lower 95% confidence limit (red) and upper 95% confidence limit (blue). The total biomass carbon stock in the montane ash forest region was estimated to be 112.8 Tg C in 2010.

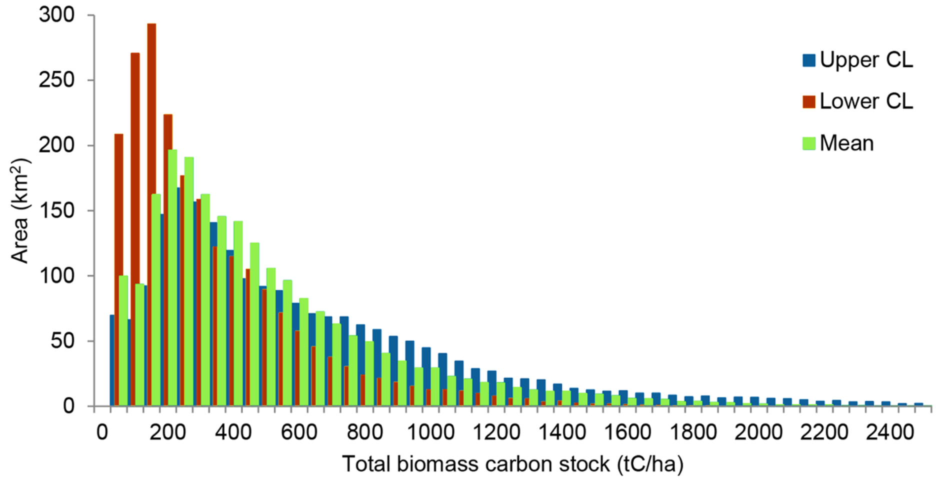

Fig. 6. Distribution of total biomass carbon density by area with a mean distribution (green), which is shown on the map in Fig. 5, and the lower 95% confidence limit (red) and the upper 95% confidence limit (blue).

Change in carbon stock of living biomass over time

The relationship describing carbon accumulation in living biomass is important for predicting carbon stock change over time. We compared our data with that in the literature (Fig. 7; Appendix C: Table C1) to assess the relative differences. Eq.1 predicted carbon stocks similar to, or higher than, stocks predicted by other equations reported for *E. regnans* (e.g., Borough et al. 1984, Grierson et al. 1992, West and Mattay 1996, Dean et al. 2003, Roxburgh et al. 2010, DCCEE 2010). The higher predictions from Eq.1 were mainly due to site data for old growth forests being included in the derivation of our equation, whereas the site data for most other equations were derived from forests of less than 100 years old.

An earlier equation that has been applied in the montane ash forests is that derived by Grierson et al. (1992) for aboveground living biomass. This equation had an original form of a polynomial and so could not be used to estimate carbon stocks in forests older than 76 years; the maximum range of the calibration data. To compare equations over the time periods of mature and old growth forests, we converted this equation to the same form of the Chapman-Richards equation as our Eq. 1, to derive Eq. 2. Aboveground living biomass was converted to living biomass (above- and belowground) using a standard expansion factor of 1.25 (Snowdon et al. 2000).

$$\text{living biomass} = 620 \times (1 - \exp(-0.0065 \times \text{age}))^{0.75} \times 1.25 \qquad (2)$$

where living biomass is in tC/ha and age is in years. Additionally, we compared our predictions of carbon stock increment with estimates of growth rates over specific tree age intervals reported in the literature (Table 3). We found that reports of early growth rates were highly variable, but predictions from Eq. 1 were within their range (see Table 3). Growth rates in forests older than 50 years were predicted to be higher by Eq. 1 than in other reports. However, data in the literature were limited from these older forests. The modelled equation from DCCEE (2010) showed very slow growth after 100 years and no growth after 200 years, and yet net growth of stands of *E. regnans* with mean dominant age exceeding 400 years has been reported (Banks 1993, Beale 2007, Wood et al. 2010). Growth rates used by the NGGI (1997) are averages that take into account variations in site quality, stocking, fire and management history. However, Flinn et al. (2007) considered these growth rates may be underestimates. Additionally, it was uncertain whether these estimates of gross volumes included all trees or only merchantable trees at the site (Raison and Squire 2007).

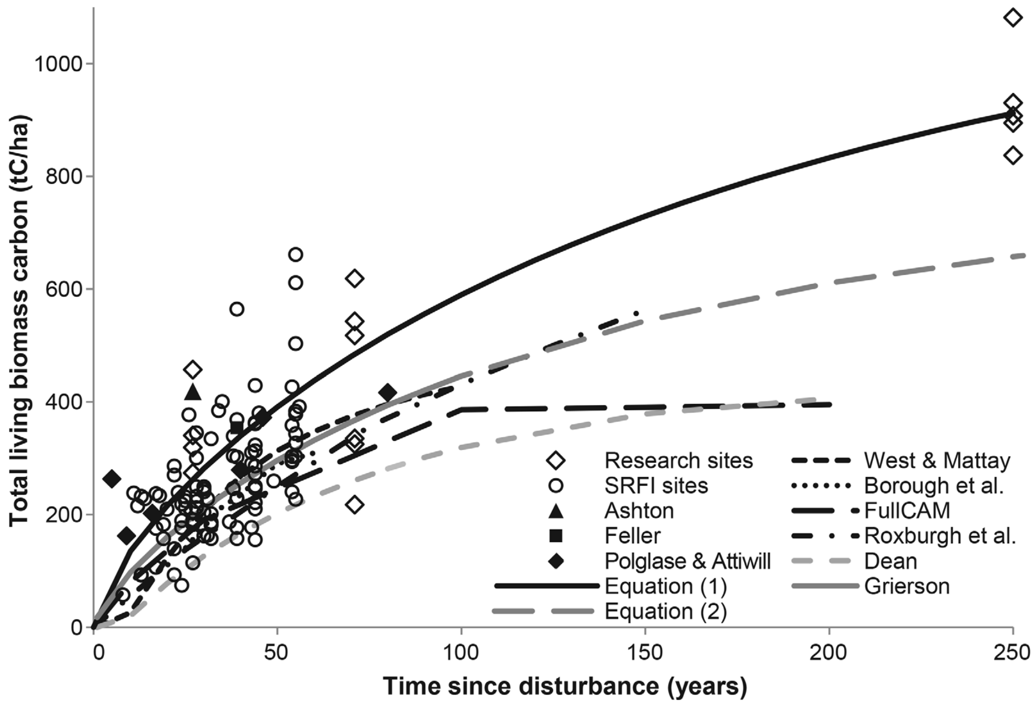

Fig. 7. Carbon accumulation in living biomass (above- and belowground) over time in *E. regnans* forest based on site data and equations from the literature and current study. Legend: Data from field research sites from the current study in forest unburnt in 2009, SFRI sites clearcut silvicultural treatment and sites from Ashton (1976), Feller (1980) and Polglase and Attiwill (1992) were combined to derive the carbon accumulation curve of Eq.1 (solid black line). Polynomial equation from Grierson et al. (1992) (solid grey line) and the extrapolation using an exponential model for Eq.2 (dashed grey line). Equations from the literature include West and Mattay (1996), Borough et al. (1984); FullCAM in DCCEE (2010); Roxburgh et al. (2010); Dean et al. (2003) and Dean and Roxburgh (2006). Equations for aboveground living biomass were converted to total living biomass using a standard expansion factor of 1.25 (Snowdon et al. 2000). Equations and sources are listed in Appendix C.

Amount and longevity of carbon stocks in a clearcut logging system

Transfers of carbon stocks during each stage of the harvesting process showed that 20% of the initial total biomass carbon stock resided in paper products and 4% in sawn timber (Fig. 8). Only the sawn timber products and dead and downed woody debris remaining on-site had mean residence times in the order of decades (Fig. 8).

At the landscape scale of the montane ash forest region, we calculated that 26.2% of the area had been logged up to 2011 (609 km^2) and 2.5% of the area had been logged more than once (59 km^2) (Fig. 9). We estimated a loss of 29 Tg C in carbon stock due to logging, derived from the difference between the modelled carbon carrying capacity (Keith et al. 2010) and the current carbon stock. This difference resulted from the younger ages of regrowth forests.

The area proposed for logging from 2011 to 2016 was 206 km^2 and we estimated the carbon stock in this area was 8.42 Tg C, based on the modelled carbon density in Fig. 5. The majority of this area would be clearcut or salvage logged after the 2009 wildfire. From this carbon stock proposed for logging, emissions to the atmosphere would be 5.56 Tg C (66% of total biomass). These emissions would be derived from slash burning (30% of total biomass), combustion or decomposition of waste during processing and short-term products within a few years (29% from pulp processing and paper products; 7% from sawlog processing) (percent-

Table 3. Increments in carbon stocks over specified time intervals, comparing values from reports in the literature with the equation for carbon accumulation of living biomass derived in this study (Eq.1).

Site characteristic	Volume increment ($m^3 \cdot ha^{-1} \cdot yr^{-1}$)	Biomass increment† ($tC \cdot ha^{-1} \cdot yr^{-1}$)	Reference
High quality regeneration, well-stocked	10–20, mean = 15	6.8	Borough et al. (1984), Attiwill (1979), West and Mattay (1996)
1–50 yrs	11	2.75	Feller (1980), Flinn et al. (2007)
50–80 yrs	5.5	1.37	Feller (1980), Flinn et al. (2007)
0–7 yrs		11	Attiwill (1992)
Average over the age of the stand	8	2	NGGI (1997)
1–10 yrs		6.44	DCCEE (2010)
11–30 yrs		4.41	DCCEE (2010)
31–100 yrs		2.23	DCCEE (2010)
100–200 yrs		0.74	DCCEE (2010)
>200 yrs		0	DCCEE (2010)
1–50 yrs		7.8	Eq. 1
51–100 yrs		4.0	Eq. 1
101–200 yrs		2.4	Eq. 1
>200 yrs		1.6	Eq. 1

† Biomass increment was calculated from volume increment using the default conversion factors listed in Appendix C to calculate total living biomass carbon stock.

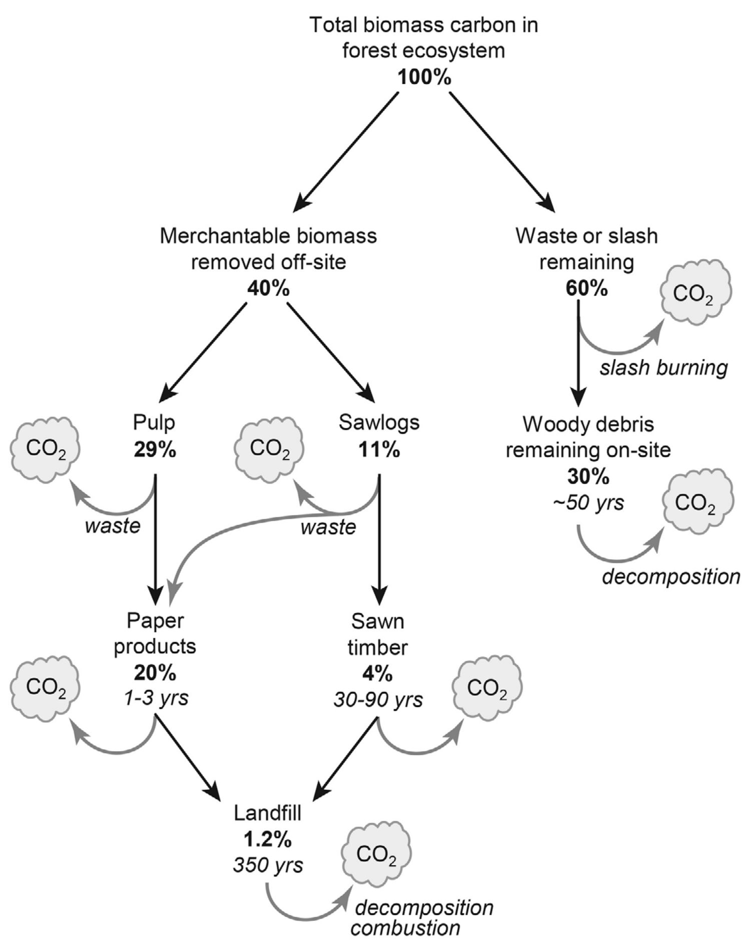

Fig. 8. Transfer of biomass carbon during harvesting and processing of wood products. Numbers in bold represent the proportion of the total biomass carbon in the forest that remains in each component. Numbers in italics are the average lifetime of the carbon pool (see data sources in Appendix E: Table E1).

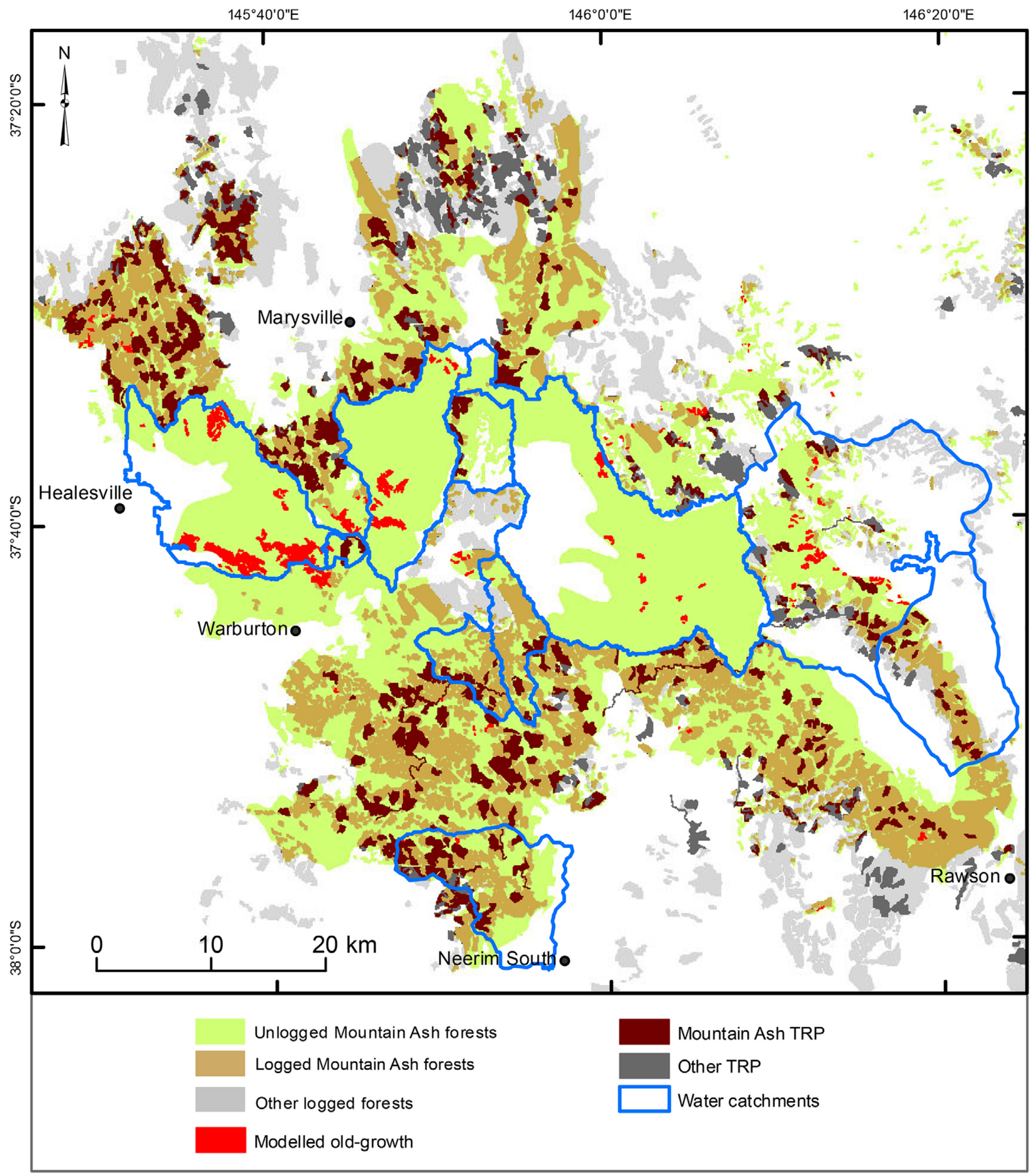

Fig. 9. Spatial distribution of areas that have been logged and are proposed for logging in the Timber Release Plans for 2011–2016, as well as the remaining areas of old growth forest.

age values from Fig. 8). The remaining carbon stock of 2.86 Tg C (34% of total biomass) would reside in timber products and dead and downed woody debris that decompose at slower rates, with mean residence time of about 50 years. Only 0.337 Tg C would be stored in long-term (30–90 years) timber products (4% of total biomass) (Appendix E).

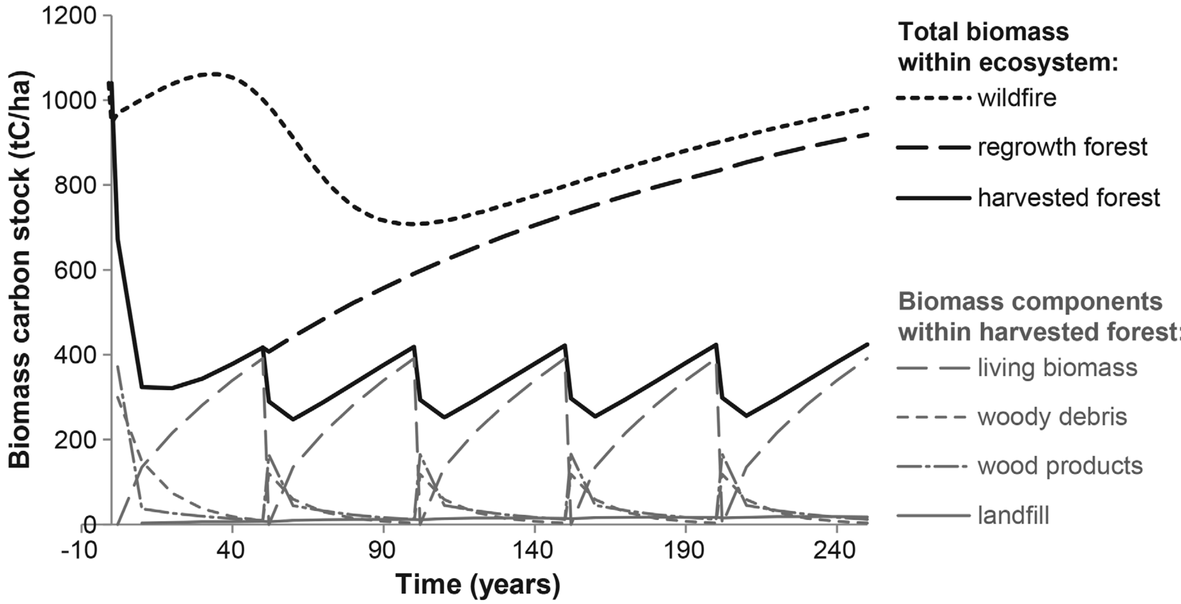

Fig. 10. Changes in total biomass carbon stock of the ecosystem over time under three scenarios (shown as black lines) from an initial stock of a native forest: (1) wildfire that occurred at time 0 years and then the forest regenerated and dead biomass decomposed over time, (2) regrowth forest after logging once and regeneration, and (3) harvested forest under a regime of repeated logging rotations consisting of clearcutting and slash burning on a 50 year cycle (Timber Industry Strategy State of Victoria, DPI 2009). The carbon stock within the harvested forest is separated into biomass components (shown as grey lines): (1) living biomass, (2) dead and downed woody debris, (3) wood products, and (4) landfill. These biomass components constitute part of the harvested forest system but are not all located at the same site; living biomass and dead and downed woody debris occur at the forest site, but wood products and landfill occur in different locations. The proportions allocated to each of these components and their rates of stock change are derived from the data in Fig. 8 and Appendix E: Table E1. Sustainable production is assumed over the course of the rotations.

Predicted carbon stock change due to different disturbance regimes

Our model of carbon stock change over time under different disturbance regimes predicted that a repeatedly logged forest site never regained its levels of initial carbon stock (Fig. 10). As a logging system averaged spatially across the landscape with areas at different times since logging, the average carbon stock was 37% of the initial stock. The maximum carbon stock at age 50 years was 44% of the initial stock. After a single logging event, accumulation of carbon took 250 years to regain the initial stock. The rate at which carbon continued to accumulate in older forests was not known and hence the initial stock was likely underestimated because trees can be older than 250 years (Stephenson et al. 2014).

In contrast to the carbon stock change due to logging, the modelled dynamics of carbon stock change due to wildfire showed gains and losses of less than 20% due to mortality, combustion, decomposition and regeneration (Fig. 10). Wildfire results in immediate losses of carbon during the fire, but rapid regeneration produces gains in stocks for several decades until dead trees collapse and start to decompose (Keith et al., *unpublished data*).

We calculated the carbon stock loss due to combustion of biomass during the wildfire in 2009, accounting for the area burned at different severities, the biomass carbon density, and the burning efficiency of each biomass component under different fire severities. Within the montane ash forest region, 485 km^2 was burnt at low severity, 283 km^2 was burnt at high severity, and 1558 km^2 was unburnt. Emissions from combusted biomass averaged 40 tC/ha in low severity fires, and 58 tC/ha in high severity fires (Keith et

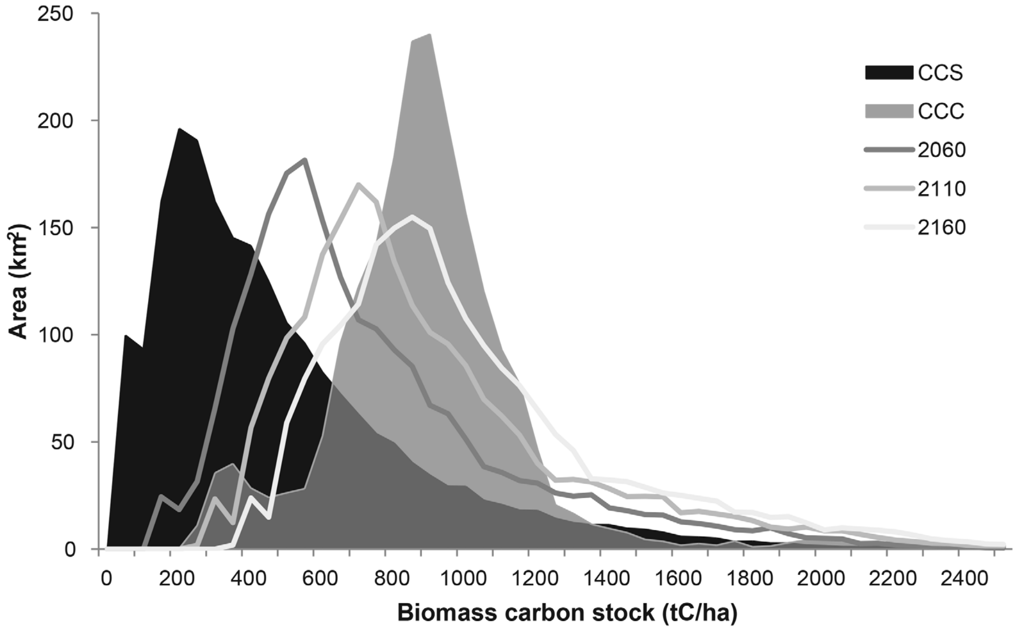

Fig. 11. Frequency distribution of carbon stocks across the study area based on carbon accumulation calculated from Eq.1, with scenarios for different time periods of current carbon stock (CCS) in 2010, + 50 years in 2060, + 100 years in 2100, and + 150 years in 2160, and carbon carrying capacity (CCC).

al., *unpublished data*). Total emissions of 3.88 Tg C were calculated spatially based on forest age and fire severity. These emissions represented 8.5% of the carbon stock in the area burnt, but only 3.4% of the carbon stock in the montane ash forest region as mapped in Fig. 5, which included burnt and unburnt areas.

Potential to increase carbon stocks

We calculated the potential to increase carbon stocks under scenarios of changed forest management such that emissions from logging were avoided. The frequency distribution of carbon density across the montane ash forest region showed the shift in carbon stocks over time for each scenario (Fig. 11). The current carbon stock represented an average across the landscape that accounted for the current age distribution of the forest. The age cohorts of these trees continued to grow and accumulated carbon over time for 50, 100 and 150 years, and hence had similar shaped distributions but with increasing carbon density. The frequency distribution for carbon carrying capacity differed from the current carbon stock and scenarios because it was estimated for native eucalypt forest with specified site conditions for each grid cell, and was not based on the existing age distribution of the forest. The current carbon stock was 51–58% of the predicted stock in 150 years (range depending on the carbon accumulation derived from Eqs. 1 or 2), and 55% of the predicted stock at carbon carrying capacity (Table 4). Interestingly, predictions from both equations after 150 years were a similar magnitude to the carbon carrying capacity, which was estimated independently (Keith et al. 2010). The difference in carbon stocks predicted by Eqs. 1 and 2 reflected the differences in biomass of the calibration sites for each equation and particularly the maximum age of forests. This range in predicted carbon stocks provided bounds of uncertainty in potential stocks.

We compared this potential accumulation of carbon over time by the regrowing forest with the carbon loss due to logging. Continued logging as planned for 2011–2016 would represent a loss of carbon stock off-site of 5.56 Tg C over 5 years across the area that would be logged. Continued growth of the current montane ash forest across the region would represent a gain in carbon stock of 6.05 Tg C over 5 years using Eq. 1 or 5.18 Tg C using Eq. 2.

Table 4. Projected biomass carbon stocks in the montane ash forest study area (2326 km^2) estimated from the current carbon stock (CCS) in 2010; predictions for +20 years (2030), +50 years (2060), +100 years (2110) and +150 years (2160); and the carbon carrying capacity (CCC).

	Total biomass carbon stock (Mt C)‡					
Carbon accumulation method†	CCS	2030	2060	2110	2160	CCC
Eq. 1	113	133	162	196	221	204
Eq. 2	113	130	152	177	194	204

† Calculations of CCS and predictions used the equations for carbon accumulation in living biomass from either Eq. 1 or Eq. 2.

‡ Total biomass carbon stock was calculated as the living biomass predicted for each time period by Eqs. 1 or 2, plus the same estimate of dead biomass for each time period.

Discussion

Issues about forest management for carbon accounting

We assessed management practices to maximize carbon storage using our empirical data of carbon stocks in forests subject to different disturbance types. Our case study of carbon stock changes in the montane ash forests in Victoria provided an example of the type of data required for a carbon accounting system that allows the role of land management practices to be quantified within the global carbon cycle. The methods we used for measuring carbon stocks at the site scale, and accounting for spatial variability in relation to environmental conditions and disturbance history at the regional scale, are likely applicable for other temperate forests and many other forest types. A benefit of a comprehensive system is that biomass carbon can be accounted for inclusive of transfers between stocks of different qualities, in terms of their amount, longevity and resilience (Ajani et al. 2013). This approach is necessary in a forest logging system where emissions occur from both on- and off-site and over short- and long-time frames.

Assessment of carbon stock changes within a comprehensive accounting system at appropriate spatial and temporal scales enables decisions to be made about the relative benefits of different management practices. Our results about carbon stock changes help inform debates about forest management, as well as improve the conceptual framework for carbon accounting systems. The key findings were:

1) Large trees contributed a high proportion to forest biomass carbon stocks, particularly in old growth forests, but also as residual trees in regrowth forests. Hence, sampling strategies appropriate for their measurement at field sites is critical. In old-growth montane ash forest, trees >100 cm diameter contributed an average of 76% to the biomass carbon stock. In other forest types, large trees may have a different size threshold.
2) The current carbon stock in logged forests across the montane ash forest region (accounting for the age distribution of the forest) was 55% of the stock at carbon carrying capacity. Conversely, this difference represented the carbon sequestration potential if the forest was allowed to grow to its maximum.
3) Current carbon stock of living and dead biomass in forests represented an average stock based on the history of disturbance, which involved mortality, survival and regeneration of trees, rather than the accumulated carbon over a given time since a disturbance event.
4) Longevity of carbon stocks was lower in a logged forest system compared with an old growth forest. Of the initial biomass carbon stock in a forest that was logged, 66% was converted to products with short lifetimes (<3 years), and the average age of the forest was less (<50 years) compared with >100 years in old growth forest.

We discuss below the significance of these key findings in relation to the three questions posed in the Introduction concerning the impacts of natural and anthropogenic disturbances on carbon stocks in forests at regional scales.

Table 5. Average carbon stock in logged forest, averaged spatially across the landscape, compared with the stock in old growth forest.

Forest type	Percentage change	Reference
Wet eucalypt forest, Victoria, Australia	55	This study
Cool, moist temperate forest, US Pacific Northwest	57	Krankina and Harmon (2006)
Wet eucalypt forest, Tasmania, Australia	50	Dean and Wardell-Johnson (2010), Dean et al. (2012)
Temperate coniferous forest, US Pacific Northwest	50	Janisch and Harmon (2002)

Amount and longevity of carbon stocks in a clearcut logging system

Our result that logged montane ash forests contained 55% of the carbon stock in old growth forests was within the range of 50–60% reported by empirical studies from a variety of forest types (Table 5). Modelling studies of carbon accounts of entire forest and wood products systems also have demonstrated similar reductions in the stocks of logged forests (Harmon et al. 1990, 1996, Thornley and Cannell 2000, Schulze 2000). Although montane ash forests have higher biomass carbon stock, greater longevity of tree species, and a higher proportion of logged biomass that is transferred to short-term products and waste than many other temperate forests, the proportion of overall carbon stock loss is similar.

This reduction in carbon stock in a logged forest system, inclusive of products and landfill, was due to biomass components being changed to shorter lifetimes (<3 years); namely slash combusted, processing waste, and pulp products. These short-term products represented 66% of the initial biomass in montane ash forest. Components with longer lifetimes included sawn timber which represented 4%, and dead and downed woody debris which represented 30% of the initial biomass in the forest. In a regime of logged native forest, the amount of carbon stored in products and landfill for longer than the rotation length is very small; as is any accumulation of carbon under this system. The maximum age of temperate native forests generally is two to ten times greater than harvest rotation lengths (Krankina and Harmon 2006, Wood et al. 2010).

The magnitude of the reduction in the carbon stock of a logged forest varies depending on the proportions of biomass transferred to different pools and their longevities. We compared these proportions in montane ash forest with other forest types (Appendix E: Table E1). Removal of biomass off-site and the proportions of sawlog and pulp from the products depend on the silvicultural system, tree species, form and age of trees, amount of wood degrade, market demand, economics of transporting to processing facilities, and supply quota agreements. These factors influence the amount of biomass used as commercial product and the waste left on-site from branchwood, stumps, stem defect, non-commercial species and understory. Consequently, the biological attributes of the forest type contribute only partially to the resulting carbon stock loss. Variability in carbon dynamics of different forests and logging regimes results from the integration of all these factors.

Comparison of carbon stock loss from natural and anthropogenic disturbances

The two main differences between the impacts of logging and wildfire on carbon stocks are the longevity of the components lost and the average stock in the forest. Emissions of carbon by combustion in wildfire are derived mostly from short-lived biomass components in leaves, twigs, litter, shrubs and small dead woody debris with mean residence times of a few years to a decade, plus some hollow standing dead trees. Longer-term changes in carbon stocks post-fire due to mortality and decomposition are restored over time if the fire regime remains stable. In forest burnt by wildfire, most trees remain with some transferred to the dead biomass pool but in many areas they remain alive (of the montane ash forest area burnt in the 2009 wildfire, more than half was burnt at low to moderate severity where trees survived). Even this dead biomass pool is long-lived, where large dead trees remain standing for up to 75 years before falling to become dead and downed woody debris, and beginning to decompose (Lindenmayer and Wood 2010).

Emissions from logging the original old growth forest are only partly balanced by removals because the carbon stock in the regenerating forest at the end of the rotation does not regain that of the old growth forest. Rotations of logging and regeneration result in a stand structure of younger trees (Leighty et al. 2006, Gough et al. 2007). Emissions from logging are derived from long-lived carbon stocks in living tree stems that take decades to centuries to recover. In the montane ash forest, only 4% of the biomass was converted to long-lived sawn timber products. The difference in the longevity of the carbon stock emitted is important for evaluating the long-term impact of the disturbance on the carbon cycle.

In the montane ash forest region, we estimated that the total amount of carbon stock loss due to previous logging over about a century was 29 Tg C within 26% of our mapped region in the Central Highlands. We estimated the carbon stock loss due to the 2009 wildfire that burnt 33% of the mapped region was 3.85 Tg C. The mean return interval of wildfires in montane ash forest was estimated to be approximately 75–150 years. Data on tree mortality suggested that approximately half the trees survive wildfire, thus the mean interval between all fires is about 37–75 years (McCarthy et al. 1999); hence approximately two fires per century. This estimate is supported by the spatial distribution of wildfires within the mapped region that shows that most areas have been subject to one or two fires over the last century, but only very small areas have been subject to three fires (Keith et al., *unpublished data*). Thus, an approximation of emissions from wildfires is 7.7 Tg C over a century. Similarly, emissions from logging temperate forest in Oregon were three times higher than those from the largest fire on record (Campbell et al. 2007). The spatial extent of wildfires is greater than that of logging, but the proportion of the total carbon stock lost is much lower.

The frequency of 50 years may be similar for both disturbance types, but the temporal pattern is different. Logging occurs at a regular frequency, whereas fire frequency is highly variable, with a maximum of 350–400 years (McCarthy et al. 1999). Additionally, fire intensity is variable so that not all areas burnt result in even-aged regeneration. Even if average frequencies of logging and wildfire disturbances are similar, random disturbance intervals allow patches of forest that are older than the average to persist in the landscape, which results in an overall higher carbon stock (Smithwick et al. 2007).

Mitigation value of protecting carbon stocks in forests

Avoiding emissions from forest degradation and allowing logged forests to regrow naturally are important activities for climate change mitigation. The former prevents further increases, and the latter helps reduce atmospheric concentrations of carbon dioxide. This kind of rapid response over the next few decades is important to allow time for technological advances in renewable energy sources that will hopefully eliminate the need for fossil fuel use (Houghton 2012).

The critical factors in assessing the mitigation value of forest management for protection versus logging are the amount, longevity and resilience of carbon stored in forests, not the rate of carbon removal from the atmosphere assessed on an annual basis (Schulze et al. 2002, Ajani et al. 2013). Protection of carbon stocks in native forests, and particularly old growth forests, is the best strategy in the montane ash forest to maximize mitigation. This will also be true in other forest systems with large carbon stocks, long-lived tree biomass stocks, and attributes which make them resilient to impacts of natural disturbances. Carbon stock is reduced when the mean residence time of products is less than the age of the trees, and some of the biomass is converted to components with short residence times, such as slash, waste and paper products. This recommendation has been advocated for other forest types globally (Harmon et al. 1990, Schulze et al. 2002).

Montane ash forest classified as old growth (DSE 1996) occurs in only 29.9 km^2 or 1.3% of our study region in the Central Highlands. The area of old growth has been reduced rapidly from estimates of 30% prior to the wildfire in 1939 (Lindenmayer and McCarthy 2002), and 15% of the region 25 years ago (Kuczera 1987). Maintenance of high carbon stocks in old growth forests, both in the Central Highlands and other temperate forests, depends on their capacity for self-regeneration. The aggregated spatial pattern of large trees, typical of old growth forests (Lutz et al. 2012), suggests regeneration is dependent on random disturbance events that result in clustered establishment, such as occurs following spatially variable severity of fires. In addition to storing large carbon stocks, old growth forests continue to remove atmospheric carbon dioxide as shown by estimates of net ecosystem productivity across global sites (including temperature forest sites) in the FluxNet with average carbon sequestration of 0.75 ± 0.2 $tC \cdot ha^{-1} \cdot yr^{-1}$ to 2.4 ± 0.8 $tC \cdot ha^{-1} \cdot yr^{-1}$ in forests older than 200 years (Luyssaert et al. 2008, 2010, Bellassen and Luyssaert 2014).

Carbon stocks in logged forests can be increased by lengthening rotations and increasing mean residence times of wood products (Krankina and Harmon 2006). For example, wood products from the highly productive temperate forests in the Central Highlands are used 72% for pulp and 28% (or less) for sawlogs (DSE 2009), whereas products from the US Pacific Northwest consist of 10% for pulp and 90% for sawlogs (Harmon et al. 1996). However, Dean et al. (2012) suggested that emissions from logging a wet temperate eucalypt forest would not be recovered by sequestration in wood products unless their half-lives were an order of magnitude more than contemporary values, thus requiring 200–1000 years for recovery.

Potential future loss of carbon from forests due to natural disturbances, such as fire or pests, has been identified as a risk to long-term storage (Balshi et al. 2009). In some cases this has been used as an argument to support continued logging of forests as it is considered that high carbon storage could not be maintained (Kurz and Apps 1999, Attiwill and Adams 2008, Kurz et al. 2008, Norris et al. 2010). However, our results for the emissions of carbon from wildfire showed that forests maintain carbon stocks within a relatively small range of the old growth state compared with the impacts of logging. The capacity of native forest ecosystems for resilience to and self-regeneration from natural disturbances comes from the adapted functional traits of their characteristic biodiversity that in turn provides for stable biomass carbon stocks (Thompson et al. 2009).

Factors contributing to uncertainty

Improved methods and data for carbon accounting in forests will reduce the uncertainty in predictions of the impacts of different management activities on carbon dynamics. This is an important issue at several scales; namely, international reporting of national inventories and compliance with emissions reduction targets, regional and project scale accounting for carbon trading schemes, and natural resource management. We identified the following key factors that contribute to uncertainty in the prediction of carbon stocks at both site and landscape scales, and hence require further research:

1) The distribution of large trees needs to be assessed at landscape scales, particularly the occurrence of biological legacies remaining after disturbance events (Franklin et al. 2000, Lindenmayer et al. 2012).
2) Characteristics of disturbance events that affect carbon stocks need improved quantification, in particular, the magnitude or intensity, exact boundaries, spatial variability within the boundary, and impact on the vegetation.
3) Impacts of disturbance events on tree size distribution need to be quantified, including the number and sizes of tree mortalities, and the effect on rates of regeneration and growth.
4) Additional site data that relate carbon stocks to forest stands of known age are required for calibration of growth functions for different forest types and regions, as this relationship is critical for predicting carbon stock change over time. These data rely on chronosequences of forest sites of known age and the assumption of space-for-time substitution. Obtaining adequate site data for old forests is particularly difficult.
5) Site data for tree dimensions, density and biomass are required to quantify maximum carbon stocks in old growth forests as this defines the asymptote of the accumulation curve. Currently, data are limited in older forests, particularly beyond 100 years. However, given the relative scarcity of old growth temperate forests, non-destructive techniques are needed.
6) Published data on the amounts and trans-

fers of carbon through the dead biomass pool need to be complemented by additional field data, including the amount of slash remaining on-site after harvesting, the proportion combusted, and rates of decomposition of dead and downed woody debris, belowground biomass and wood products in landfill.

7) Disturbance events, such as logging and wildfire, affect soil carbon stocks through combustion, erosion, and changes in rates of soil respiration and litter inputs. Soil data for forests globally are limited (Rab 1994, 1999, 2004, Gough et al. 2007, Diochon et al. 2009).

We recommend that estimating carbon stock changes based on stock longevity is the most robust method of accounting (Nabuurs and Sikkema 2001, Ajani et al. 2013). This method of accounting provides the information needed for policies that provide incentives to protect carbon stocks in forests and to use wood in long-lived products. The effects of disturbance regimes and forest management activities on carbon stocks are best assessed at the landscape scale as this incorporates the mosaic of forest patches of different ages.

Conclusions

Changing forest management policy to avoid emissions from logging contributes to the global objective of reducing atmospheric carbon dioxide emissions and to national targets for reducing emissions. A forest managed on a typical commercial logging rotation can never regain the initial carbon stock of the old growth forest, even including the stock in wood products and landfill. This conclusion demonstrates the importance of accounting for changes in stocks of carbon over an appropriate timeframe, rather than only measuring rates of annual carbon uptake.

Results from this study contribute to addressing issues of international reporting of greenhouse gas emissions, national methodologies for carbon accounting, and regional debates about land resource management. These issues are currently important because policies and related methodologies are still under development and need to be informed by the best available scientific evidence. Reporting on emissions and removals from forest management and disturbance events is mandatory for United Nations Framework Convention on Climate Change (UNFCCC) national inventory reports and for Annex 1 signatory nations under the Kyoto Protocol (Article 3.4) second commitment period (2013 to 2020) (UNFCCC 1998, 2011). The introduction of mandatory reporting of forest management in the Kyoto Protocol means there is an urgent need to develop methodologies and obtain data to quantify the effects of forest management on carbon stocks.

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SUPPLEMENTAL MATERIAL

APPENDIX A

Methods for Measuring Biomass Components

Trees.—All woody stems (living and dead) greater than 2 m height and less than 100 cm diameter were assessed in height and diameter categories for each species within three 10 m × 10 m plots (0.03 ha). Trees greater than 100 cm were measured in two perpendicular and intersecting 100-m transects by 30 m width (0.51 ha). In large trees that form buttresses with fluted stems (Ashton 1975), there is a cross-sectional area deficit that was accounted for by converting to a 'functional' diameter (Sillett et al. 2010). The allometric equation for *E. regnans* derived by Sillett et al. (2010) was used to calculate stem and branch volume, and multiplied by wood density and carbon concentration to derive aboveground biomass carbon content. Average stem wood density was 0.520 g cm^{-3} (Chafe 1985, Mackensen and Bauhus 1999, Illic et al. 2000, Bootle 2005) and branch wood density was 0.677 g cm^{-3} (Sillett et al. 2010). A carbon concentration of 0.5 gC/g was used for all biomass components (Gifford 2000, Keith et al. 2009). Allometric equations for other vegetation components included *Acacia* spp as mid-story trees (Feller 1980), a general rainforest equation for the range of other mid- and understory species (Keith et al. 2000), and treeferns were calculated as a cylindrical volume and measured wood density. Root mass in *E. regnans* forests has not been measured, hence an average root : shoot ratio for eucalypt forests of 0.25 (Snowdon et al. 2000) was used to convert aboveground biomass to total tree biomass for living and dead trees.

Internal decay or hollows in stems need to be taken into account when biomass is derived from stem volume, rather than weighing the tree (Mackowski 1987, Gibbons and Lindenmayer 2002). Heartwood decay is common in large old eucalypts (typically more than 120 years) and is influenced by age of the tree, physical damage, species, environmental stresses to growing conditions, and presence of decay organisms (Ambrose 1982, Wilkes 1982, Mackowski 1987, Looby 2007). Equations to predict occurrence and volume of decay related to tree size were derived from a subset of inventory data (734 trees) (DSE 2007*a*) where dimensions of defect in sawlogs had been measured.

Dead and downed woody debris.—All dead woody material ≥25 mm diameter on the ground was measured using the line intersect method (van Wagner 1968, McKenzie et al. 2001). Logs less than 60 cm diameter were measured along 6 × 10 m transects within the site and logs greater than 60 cm diameter were measured along 2 × 100 m perpendicular transects. Each piece of woody material was assessed for hollows and degree of decay in three categories. Bulk density was measured for each category of decay to determine biomass.

Litter.—The litter layer consists of organic material less than 25 mm diameter including leaves, twigs, insect detritus, animal scats, and comminuted material that is recognizable as organic material. Litter depth was measured at 25 points per site on a compressed litter pack to standardize the quantity of loose material. Depth was converted to biomass using a relationship derived from 30 quadrats sampled for dry weight of litter.

APPENDIX B

Description of Spatial Estimates of Environmental Variables

Table B1. Description of spatial data layers used for model development and extrapolation.

Symbol	Layer description	Resolution	Source
Climate			
T	Mean annual near surface air temperature, °C	1 km	ANUCLIM; Hutchinson (2005)
P	Annual precipitation, mm/yr	1 km	ANUCLIM; Hutchinson (2005)
Q_s	Solar radiation received at the surface, $MJ \cdot m^{-2} \cdot yr^{-1}$	1 km	ANUCLIM; Hutchinson (2005)
W	$W = P - \frac{Q_s}{\rho L}$, mm/yr where ρ is the density of liquid water (~1000 kg/m^3) and L is the latent heat of vaporization of water (~2.45 $\times 10^6$ J/kg H_2O); W is a negative number	1 km	Berry and Roderick (2002)
W_{pos}	$W_{pos} = 1380 - W$, mm/yr; W_{pos} allows calculation of the logarithm	1 km	Berry and Roderick (2002)
Vegetation			
MVG	National Vegetation Information System; Australia—Present Major Vegetation Groups (MVG)—NVIS Stage 1, Version 3.0 Classes: MVG 2 (eucalypt tall open forests; trees taller than 30 m); MVG 3 (eucalypt open forests; trees 10–30 m tall)	100 m	NVIS (2008)
NDVI	MODIS 16-Day L3 Global 250-m (MOD13Q1) satellite imagery	250 m	LPDAAC (2008); MODIS data products processed/re-formatted by CSIRO using the MODIS Reprojection Tool ('mrtmosaic', 'resample')
GPP	Gross primary productivity ($mol\ CO_2 \cdot m^{-2} \cdot yr^{-1}$)	250 m	Berry et al. (2007)
Topography			
Elev	Ground-level elevation; Digital Elevation Model Version 3	9 second	Hutchinson et al. (2008)
Topo	Topographic position index, calculated from the 250-m digital elevation model; Classes: i ridge top; ii, upper slope; iii, mid slope; iv, lower slope; v, valley	250 m	Gallant and Dowling (2003)
Edaphic			
Geol	Classes: i, intrusive igneous; ii, extrusive igneous; iii, metamorphic; iv, sedimentary. Classes were derived from the surface lithology descriptions in the tables associated with the digitized map layers.	1:1,000,000	Liu et al. (2005*a,b,c*)
%OC	Soil carbon concentration of A and B horizons (%C)	1 km	McKenzie et al. (2005)
Sd_A, Sd_B	Soil depth of A and B horizons (m)	1 km	McKenzie et al. (2005)
BD_A, BD_B	Soil bulk density of A and B horizons (Mg/m^3)	1 km	McKenzie et al. (2005)
Sd_{AB}	$Sd_{AB} = Sd_A + Sd_B$ (m)	1 km	
SOC_A	$SOC_A = Sd_A BD_A \frac{\%\ OC}{100}$ (kg C/m^2 of ground surface)	1 km	
SOC_B	$SOC_B = Sd_B BD_B \frac{\%\ OC}{100}$ (kg C/m^2 of ground surface)	1 km	

Appendix C

Carbon Accumulation Over Time

Site data and growth functions describing the rate of carbon accumulation in *E. regnans* forests, derived from the current study and the literature, were collated (Table C1) and illustrated in Fig. 3.

Symbols are: A = age (years); V_p = stem volume under bark (m^3/ha) at the plot scale; V_t = stem volume under bark (m^3) for an individual tree; AGB = aboveground living biomass carbon (tC/ha); LB = living biomass carbon above- and belowground (tC/ha); S = stocking (stem/ha); SI = site index (mean dominant height at 20 years) (m).

Default factors used in calculations are: Mean Site Index of 30 m (average value given in West and Mattay [1996], value used in Borough et al. [1984], average value calculated for the young regrowth carbon sites in the current study).

Expansion factor for ABG : stemwood = 1.18 (mean of 1.17 [Ashton 1976], 1.22 [Feller 1980], 1.14 [Keith and Sillett, *unpublished data*]).

Expansion factor for Total biomass : ABG = 1.25 (Snowdon et al. 2000); wood density = 0.5 t/m^3 (Illic et al. 2000); carbon concentration = 0.5 g/g (Gifford et al. 2000).

Table C1. Equations to describe carbon accumulation of living biomass over time for *E. regnans*.

Equation	Data description	Max. age (yrs)	Sample no. (n)	r^2	Source
$LB = 1200 \times (1 - \exp(-0.0045 \times A))^{0.7}$	Plot data for forest unburnt in 2009, of known age and regrowth from clearfell	250	119	0.72	1
$AGB = 620 \times (1 - \exp(-0.0065 \times A))^{0.75}$	Extrapolation of the Grierson data using the Chapman-Richards form of the equation that increases monotonically				2
$AGB = 4.99 + 12.83 \times A + (-0.0654 \times A^2)$	Plot data for standing timber volumes, wood density, expansion factor.	76	45	0.95	3
$V_p = 3.93 - 31.2/A + 0.1146 \times SI$ $SI = H\{[1 - \exp(-0.03 \times 20)]/[1 - \exp(-0.03 \times A)]\}^{0.902}$	Plot data, fully stocked, monospecific forests, regrowth and plantation	85	215	0.78	4
	Data in the form of yield tables of stem volume for fully stocked, unthinned forest with SI = 30m				5
6.44 (1–10 yrs), 4.41 (11–30 yrs), 2.23 (31–100 yrs), 0.74 (100–200 yrs), 0 (>200 yrs) $tC \cdot ha^{-1} \cdot yr^{-1}$	Default values of growth increments ($tC \cdot ha^{-1} \cdot yr^{-1}$) as constants for age classes of tall dense eucalypt forest				6
$DBH(m) = 3.53 - (3.59 \times e^{-0.00323 \times A})$ $V_t = 1100 \times \{1 - [1 + (DBH(m)/9.2)^2]^{-1}\}$ $S = \exp(10.6 - (1.27 \times \ln(A)))$	Data for DBH and age from Ashton (1976*b*) in Victoria, coupe data and measured plots in Tasmania. Even-aged stands with no thinning. Data for individual trees and stocking per ha related to age.	450	2184 2	0.82 0.92	7
$\ln DBH(cm) = 1.206 + 0.719 \ln(A - 5.04)$	Relationship between mean DBH and age for even-aged stands	230	17	0.99	8
Graph only, equation not given	Plot data for biomass in Victoria with a fitted growth curve	150			9

Notes: Sources are: 1, Eq. 1; 2, Eq. 2; 3, Grierson et al. (1992); 4, West and Mattay (1996); 5, Borough et al. (1984); 6, FullCAM DCCEE (2010); 7, Dean et al. (2003) and Dean and Roxburgh (2006); 8, Watson and Vertessy (1996); 9, Roxburgh et al. (2010). Abbreviations are: LB, living biomass; AGB, aboveground biomass; A, age; S, stocking, trees/ha; H, tree height; V_p, stem volume per plot; V_t, stem volume per tree; DBH, diameter at breast height of 1.3 m; SI, site index; exp, exponential; ln, natural logarithm.

Appendix D

Modelling Change in Carbon Stocks

Table D1. Input data and equations to predict carbon stock (tC/ha) change as functions of time (t is time since disturbance). Initial conditions include total biomass carbon stock in an old growth forest of 1040 tC/ha.

Process	Function	Data	Source
Regeneration (living biomass) (=gain in carbon)	$B_{\text{regen}}(t) = 1200 \times (1 - \exp(-0.0045 \times t))^{0.7}$	Inventory data from 99 sites	1, 2
Stem density	$\text{Log}_{10}(\text{SD}_t) = 1.28 + 3.16 \times 0.913^t + 1.9 \times 0.99^t$ $\text{Log}_{10}(\text{SD}_t) = 11.61 - 1.624 \log_{10}(t)$ Average from the two functions used.	Exponential function fitted to chronosequence site data for *E. regnans*.	3, 4
Mortality of regeneration (=input to dead trees)	$B_{\text{dead\_trees}}(t) = [(\text{SD}(t-1) - \text{SD}(t))/\text{SD}(t)] \times B_{\text{regen}}(t)$	Empirically derived from stem density changes	1
Branch fall (=input to woody debris)	$B_{\text{branch}}(t) = B_{\text{regen}}(t) \times 0.005$	Rate constant derived empirically to produce CWD biomass in the range observed (3 − 255 tC/ha)	1, 5
Mortality of surviving trees (=input to dead trees)	$B_{\text{dead\_trees}}(t) = 135 \exp(-0.015 \times t)$	Rate of mortality estimated from site data.	6
Collapse of dead trees (=input to woody debris)	$B_{\text{debris\_in}}(t) = 687/(1 + \exp(0.1 \times t - 5)$	Logistic function derived to fit observations that dead trees remain standing for 10 to 75 years.	7, 8
Decomposition of woody debris (=loss of carbon)	$B_{\text{debris\_loss}}(t) = 124 \exp(-0.07 \times t)$	Rate constant; modelled range in CWD consistent with site data mean (51 tC/ha) and maximum (255 tC/ha)	1, 5, 9

Notes: Sources are: 1, current study; 2, DSE (2007*b*); 3, Ashton (1976); 4, Watson and Vertessy (1996); 5, Lindenmayer et al. (1999); 6, Lindenmayer et al. (2012); 7, Lindenmayer et al. (2010); 8, Lindenmayer and Wood (2010); 9, Mackensen and Bauhus (1999). Abbreviations are: t, time; B_{regen}, regenerated living biomass; $B_{\text{dead\_trees}}$, biomass in dead trees; B_{branch}, biomass in branches; $B_{\text{debris\_in}}$, biomass of woody debris input; $B_{\text{debris\_loss}}$, loss of biomass from woody debris; SD, stem density (trees/ha).

Appendix E

Transfer of Biomass Carbon Stocks during the Logging Process

Table E1. Proportion of the carbon stock changed in each biomass component during transfer processes in the forest logging system. Values in bold were used in Fig. 8.

Biomass component	Proportion changed	Longevity $t_{0.95}$ (yrs)	Derivation of estimate	Source
Removed off-site	**40% of total biomass**		Default value for moist, high quality forest under integrated harvesting for sawlog and pulpwood used in NCAS	1
	20–80% of total biomass		Range for eucalypt forests used in NCAS	1
	44% of total biomass		Site data from ash forests. Range in biomass remaining as slash of 42–102 tC/ha but up to 275 tC/ha.	2
	45% of total biomass		Estimated average for eucalypt forests in Victoria	3
	26% of aboveground; 21% of living biomass		Site data for *E. regnans* with temperate rainforest understory in Tasmania (root:shoot ratio of 0.25 assumed [Snowdon et al. 2000])	4

Table E1. Continued.

Biomass component	Proportion changed	Longevity $t_{0.95}$ (yrs)	Derivation of estimate	Source
	58% of aboveground; 46% of living biomass		Site data for tall, wet regrowth forest of *Corymbia maculata* on the south coast of NSW (root:shoot ratio of 0.25 assumed [Snowdon et al. 2000])	5, 6, 7
	70% of aboveground; 56% of living biomass		Site data for tall, wet regrowth forest of *E. obliqua* on the south coast of NSW (root:shoot ratio of 0.25 assumed [Snowdon et al. 2000])	5, 6, 7
	45% of aboveground; 36% of living biomass		Site data for tall, wet regrowth forest of *E. pilularis* on the south coast of NSW (root:shoot ratio of 0.25 assumed [Snowdon et al. 2000])	5, 6, 7
	37% of total biomass		Average biomass carbon density in areas proposed for logging = 409 tC/ha; Wood product harvested from clearfelling = 600 m^3/ha = 150 tC/ha (where wood density = 0.5 t/m^3, carbon concentration = 0.5 g/g)	8, 9
	50–77% of aboveground; 40–62% of living biomass		Estimated range in expansion factors for proportion of aboveground biomass used for wood products. (root:shoot ratio of 0.25 assumed [Snowdon et al. 2000])	10
Combusted in slash burning	**50% of slash; 30% of total biomass**		Average burning efficiency including all available fuel present (slash, understory, fallen dead trees, woody debris, litter) over the total area combusted.	11
	50% of slash		50% for woody debris (>100 mm log sizes having 80% loss over 70% of coupe area for 90% of coupes), and 70% for fine fuels (<100 mm log sizes having 100% loss over 80% of coupe area for 90% of coupes).	2
	58–63% of slash		Site data from tall, wet *E. obliqua* forest in Tasmania	12
	31–89% of slash		Site data from tall, wet *E. diversicolor* forest in Western Australia.	13
Woody debris remaining on-site	**50% of slash; 30% of total biomass**	**43**	Woody debris from *E. regnans* for logs 10–30 cm diameter of the ground surface	14, 15
		92	Average for Australian timber species	14
		38	Average decomposition of woody debris in temperate deciduous forests	16
		28–91	Chronosequence and time series site data for boreal forests	17
		10	IPCC default value for decomposition of all litter	18
Dead coarse roots		214	Estimated half-life ($t_{0.5}$) of 50 years in moist eucalypt forest	19
Sawlogs	**28% of roundwood; 11% of total biomass**		Harvesting reports of products from merchantable roundwood for the forest management area	9
	50% of roundwood		Average value in a sawlog-driven industry	20, 21, 22

Table E1. Continued.

Biomass component	Proportion changed	Longevity $t_{0.95}$ (yrs)	Derivation of estimate	Source
Pulpwood	**72% of roundwood; 29% of total biomass**		Harvesting reports of products from merchantable roundwood for the forest management area	9
Sawn timber products	**33% of sawlogs; 4% of total biomass**	**30–90**	Timber products including sawn timber, plywood and veneer	23, 24, 25, 26, 27, 28
		30–300	Range of lifetimes for sawn timber products used in the USA	29
		100	Global carbon budget default value for lifetime for timber products	30
		152	IPCC default value for lifetime for timber products	31
		169	Timber products from the Pacific Northwest	20
Paper products	**66% of pulpwood; 20% of total biomass**	**1–3**	Paper products including 60% in packaging, cardboard, newsprint, household and sanitary paper and 40% in printing and writing paper with lifetimes of 1–3 years	24, 25, 26, 32
		10	Global carbon budget default value for lifetime for pulp and paper products	30
		9	IPCC default value for lifetime for paper products	31
Landfill organic carbon	**1.2% of total biomass**			
Decomposable	**23% of wood**	67	Default value in Australian inventory; Experimental data: half-life of wood in landfill of 350 years	26, 33, 34
(Residual)	**77% of wood**	1500		

Note: Sources are: 1, Raison and Squire (2007); 2, Flinn et al. (2007); 3, Norris et al. (2010); 4, Green (2002); 5, Ximenes et al. (2004); 6, Ximenes et al. (2008*b*); 7, Ximenes and Gardner (2005); 8, Current study; 9, DSE (2009); 10, Snowdon et al. (2000); 11, Gould and Cheney (2007); 12, Slijepcevic (2001); 13, McCaw et al. (1997); 14, Mackensen et al. (2003); 15, Mackensen and Bauhus (2003); 16, Harmon (2002); 17, Harmon et al. (2000); 18, Eggleston et al. (2006); 19, Ximenes and Gardner (2005); 20, Harmon et al. (1996); 21, Harmon (2001); 22, Perez-Garcia et al. (2005); 23, Skog and Nicholson (1998); 24, Jaako Pöyry (1999); 25, Jaako Pöyry (2000); 26, Richards et al. (2007); 27, Ximenes et al. (2005); 28, Ximenes et al. (2008*a*); 29, Krankina and Harmon (2006); 30, Le Quéré et al. (2012); 31, Eggleston et al. (2006); 32, Australian Paper Industry (2004); 33, DCCEE (2010); 34, Ximenes et al. (2008*c*).