Report to Melbourne Water

Carbon stocks and impacts of disturbance in native eucalypt forest ecosystems in the Central Highlands catchments supplying water to Melbourne

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Executive summary

Forests provide the ecosystem service of storage and sequestration of carbon. The stock of carbon is stored in natural, self-regenerating ecosystems that can be considered permanent, in the sense that a carbon stock will always exist in the natural ecosystem although the actual carbon atoms will cycle through the biosphere. The climate mitigation value of the forest carbon stock is derived from (i) avoided emissions by not harvesting timber and pulpwood, and (ii) sequestration by allowing the forest to continuing growing.

Carbon stocks were quantified in the forest ecosystems in the catchments supplying water to Melbourne (Maroondah, O'Shannassy, Upper Yarra, Cement Creek, Armstrong, Starvation Creek, McMahon's Creek, Thompson stages 1, 2, and 3, Tarago) within the Central Highlands of Victoria. The objective was to evaluate impacts of management activities and disturbance regimes. The forest consists mainly of *Eucalyptus regnans* (Mountain Ash) and *E. delegatensis* (Alpine Ash), as well as minor eucalypt species, and is referred to as montane ash forest. Carbon stocks were measured in all ecosystem components: living and dead biomass, coarse woody debris and litter. Soil carbon stocks (excluding belowground biomass) were estimated from the Australian soils map (Mckenzie et al. 2005). Data from two sets of sites were used in the analysis to provide both intensive and extensive information across the study region. These sets of sites were: (i) 54 sites from the ANU network of monitoring sites in different stand age and fire severity categories, and (ii) 876 inventory sites from the Statewide Forest Resource Inventory (DSE 2007b).

Information required to calculate biomass of montane ash forests were collated, analysed and tested. This information included tree diameter / height relationship, functional diameter of buttressed stems, allometric equations to predict volume and biomass, wood density, expansion factors from stem to aboveground biomass and total biomass, and reduction in stem volume due to pipes or areas of decayed wood and hollows. Individual tree biomass can be greatly reduced by internal decay, by up to 70%, but as an average across all trees and plots, the reduction was 2% and across the old growth carbon sites was 9%. Trees up to 400 cm DBH were included in the site data but only 5% of trees were greater than 150 cm DBH in the SRFI inventory and hence only a small proportion were likely to be affected by internal wood decay and hollows. Synthesis of this information about the reduction in stem volume due to internal decay will allow more robust and consistent estimation of biomass carbon stocks in eucalypt forests.

Average biomass carbon stocks in forest stands regenerated at different times include: 400±33 tC ha^{-1} in 1983 regrowth, 600 ± 74 tC ha^{-1} in 1939 regrowth and 920 ± 122 tC ha^{-1} in old growth (n = 6 sites for each age and fire category). Accounting for both living and dead biomass is important in the calculation of ecosystem carbon stocks. Approximately a quarter of the total carbon stock is dead biomass, although this varies with stand age and disturbance history. The variability in carbon stocks among sites within age / fire categories is indicative of the heterogeneous nature of the landscape and forest cover.

Time since disturbance, including clearfelling and wildfire, does not determine the age of all trees in a stand. Residual large trees can remain after all disturbance types, in various numbers depending on the intensity of the disturbance. These residual trees can contribute large amounts to the carbon stock and their distribution results in high spatial variability in carbon density. Old growth stands, in particular, have a highly variable distribution of trees from multiple age cohorts that were likely derived from several disturbance events. Hence, carbon stocks in forest stands do not represent the accumulated carbon over a given time since disturbance, but rather an average carbon stock for a stand with a given history of disturbance that involves regeneration and survival of trees.

Biomass carbon density was estimated spatially across landscapes using an empirical model relating measured site-level carbon stocks with environmental variables that influence growing conditions, and natural and human-induced disturbance history that influences age structure of the forest. Spatial modelling of biomass carbon stocks at landscape scales requires a large number of calibration sites across a wide range of environmental conditions and temporal range in time since disturbance. Time since disturbance and type of disturbance event were derived from spatial data layers of fire and logging history for the Central Highlands. The total current carbon stock in the biomass within the catchment areas was 64 M tC. The mean biomass carbon density varied spatially with the highest densities of 935 tC ha^{-1} and 845 tC ha^{-1} in the O'Shannassy and Maroondah catchments, respectively.

Loss of carbon from forest ecosystems due to wildfire is an important issue for quantifying emissions in national and regional carbon accounts and for assessing the differences in emissions from natural and human-induced disturbance events. Carbon emissions from the 2009 wildfire were quantified using site measurements of biomass components combusted; including litter, coarse woody debris, canopy leaves and twigs, shrubs, decorticating bark and rough bark, and large old trees. In low severity fires 6 – 7% of the biomass carbon was combusted, which equated to approximately 30 – 40 tC ha^{-1}. In high severity fires 9 – 14% of the biomass was combusted, or approximately 46 – 58 tC ha^{-1}. The proportion of biomass combusted was scaled up to a landscape scale based on the areas burnt by low and high severity fire. The total carbon stock in the catchment areas was 63.76 MtC pre-2009 and an estimated 2.42 MtC, or 4%, was lost by emissions from biomass combustion during the 2009 wildfire. The main impact of wildfire on the ecosystem was a redistribution of biomass from living to dead carbon pools. In areas subject to high severity fire, most plants were killed, but in areas of low severity fire most of the trees, even ash species, survived. The large pool of dead biomass slowly decays over many decades, but at the same time regeneration produces new living biomass.

The total carbon stock in the area available for logging (Timber Release Plans 2011 – 2016) within the catchments was estimated to be 2.31 MtC in an area of 5100 ha, and the agreed limit on the area actually harvested over the 5 year period was 1500 ha with a carbon stock of 0.67 MtC. Approximately 40% of the biomass carbon at a site is removed by clearfell harvesting, which represented 0.27 MtC. In most harvested areas the logging slash is burnt, which represented emissions of 0.20 MtC (assuming a burning efficiency of 50%). A total of 0.42 MtC over the 5 year period would be emitted to the atmosphere within a few years after harvesting due to slash burning, combustion or decomposition of waste during processing, and decomposition of paper products. The remaining carbon stock of 0.25 MtC in timber products and coarse woody debris remaining on the ground would decompose at a slow rate. Thinning operations remove approximately half the living biomass of trees from a site, there is no slash burning, but some of the remaining trees are damaged and may die. Thinning is only a small proportion of the area proposed for logging and would account for approximately 8,000 tC.

The estimated carbon credits for the catchment areas that were proposed for logging are 84,420 t C yr^{-1} or 309,540 t CO_2 yr^{-1} of emissions from 295 ha harvested per year, which is equivalent to a carbon emissions density of 286 tC ha^{-1}. By comparison, the estimated total emissions from the 2009 wildfire, were 2,420,000 tC over the 138,500 ha that were burnt, which is equivalent to a carbon emissions density of 17 tC ha^{-1}. Hence, an equivalent emission to that created by the wildfire would result from 29 years of logging over about 6% of the area that was burnt.

Carbon sequestration potential was estimated over the time periods of 20, 50 and 150 years under a range of management scenarios. Carbon sequestration during forest growth was based on the spatial distribution of the age structure of the forest, occurrence and severity of the 2009 wildfire, and a function of carbon accumulation over time. The projected carbon accumulation in the existing forests continuing to grow was between 3.2 to 12.5 MtC over 20 years (range depending on the growth function applied). The estimated carbon loss if the proposed harvesting regime was continued for 20 years would be 1.9 MtC. The net effect of harvesting is to deplete carbon stocks relative to the stocks accumulated from natural forest ecosystem processes, resulting in a net gain in carbon stock of 1.3 MtC to 10.6 MtC (depending on the growth function applied). Hence, the loss of carbon due to harvesting of a small proportion of the area in logging coupes is a significant amount compared with the gain in carbon by forest growth over the whole area.

The emissions from logging the approximately 1,500 ha under the agreed area of harvest were projected to be about 0.42 Mt C or 1.55 Mt CO_2–e over 5 years, and hence about 0.31 Mt CO_2–e yr^{-1}. Australia's mitigation commitment of a 5% emission reduction target for 2020 is 92 Mt CO_2-e yr^{-1} (Australian Treasury 2011).

The area of old growth forest remaining in the catchments is small, approximately 5700 ha or 4.2% of the area, after the 2009 fire. These forests have many conservation values, including providing a unique structural type of vegetation with the world's tallest flowering plant, supporting habitat for rare fauna, and aesthetic qualities, which make the small remaining areas of critical importance.

It likely that Australia will count forest management towards its national mitigation commitments post-2012. Protecting forests from logging, which had been planned under the Timber Release Plans and the Regional Forest Agreement, is a legitimate management activity for both the international Kyoto Protocol Article 3.4 rules and the Australian Carbon Farming Initiative rules. It is an activity that is additional to or has been changed specifically to provide mitigation benefit and the net emissions are lower than the original activity. However, there is uncertainty about the rules for the accounting frameworks for both the international and Australian systems, and hence their capacity to generate carbon credits. Inclusion of forest management in the national accounts will mean that ceasing logging could generate forest management carbon credits that could be used to offset emissions in other sectors. Irrespective of the rules for accounting and trading carbon, protecting native forests from logging will contribute to the global objective of reducing atmospheric GHG emissions.

The data presented for the carbon budget for the montane ash forests of the Central Highlands is an example of the type of information required for the comprehensive accounting that is needed to inform policy decisions. Measuring and reporting this form of comprehensive accounting is gaining support in the international arena and is recognized as a requirement to improve understanding and quantification of the role of the land sector in the global carbon cycle and its potential for mitigation of human-forced climate change.

1. Introduction

Management of native forest ecosystems depends on the objectives of the land use type and trade-offs between competing objectives Lindenmayer and Franklin 2002). In the Central Highlands of Victoria, management of the land within the catchments has the primary objective of supplying a high quantity and quality of water. Within this area, other land uses and ecosystem services may be complementary or detrimental to the primary objective of water supply. Ecosystem services are benefits that human society receives from natural systems. The role of native forest ecosystems in carbon sequestration and storage is currently an important issue. Conservation management of terrestrial ecosystems is an important mitigation response as it both avoids emissions by preventing activities that deplete ecosystem carbon stocks, and maintains natural carbon sequestration processes. Avoiding emissions from all sources in the coming decades is critical if we are to stabilize atmospheric concentrations of carbon dioxide at a level which prevents dangerous climate change (DCCEE 2011). Protecting ecosystem carbon stocks is an ecosystem service that is complementary to the provision of water supply. Costs and benefits derived from all ecosystem services should be quantified to better inform decisions about land management options for native forest ecosystems.

The ecosystem service provided by native forests is a stock of carbon stored in natural, self-regenerating ecosystems. This stock can be considered permanent in the sense that a carbon stock will always exist in a natural ecosystem due to its biodiversity that confers resilience (although the individual carbon atoms are exchanged with the atmosphere over time) (Thompson et al. 2009). The magnitude of the stock may vary spatially and temporally, in response to disturbance regimes, but at the landscape level over long time periods variation is within the bounds of a dynamic equilibrium. In contrast, human-engineered agricultural systems, such as crops and plantations, sequester and store carbon but lack the qualities of self-regeneration and resilience. Carbon that is stored permanently in natural ecosystems cannot cause an increase in the atmospheric concentration of carbon dioxide, when accounted for over long time periods. However, clearing of vegetation and conversion to other land uses results in depletion of natural carbon stocks and emission of carbon dioxide into the atmosphere. These emissions from the land sector are added to the emissions from combustion of fossil fuels. Land use change is the third largest source of greenhouse gas emissions, resulting in approximately 18% of current emissions globally (Pachauri and Reisinger 2007) but 35% of total emissions since 1850 (Houghton 2007, 2008).

Carbon stocks in native forest ecosystems must be quantified to evaluate the impact of land management activities and natural disturbance regimes on the magnitude and stability of the stocks. This will add to the array of land use evaluation methods used to assess the impacts of human activities on other ecosystem services including water supply, biodiversity conservation, and recreational and aesthetic values. Quantifying the carbon stock of a forest ecosystem involves estimation of the carbon stored in all components: living and dead biomass, coarse woody debris, litter and soil. Accounting for these carbon stocks involves an array of complex calculations including estimating the spatial variability in each component related to environmental conditions across the landscape plus the impact of natural and human-induced disturbance regimes on the age structure of the forest. The estimate of the currently existing ecosystem carbon stored at the landscape scale is called the **Current Carbon Stock** (CCS).

Prediction of change over time in ecosystem carbon stocks requires accounting for gains and losses from the stock that result from the natural exchange of carbon through plant growth and decomposition, as well as the temporal variability due to natural and human-induced disturbances. Net losses of carbon from a forest ecosystem due to disturbance, that is, emissions, occur as a result of logging (i.e. harvesting of woody biomass), degradation of soil, increased decomposition rates, and combustion by fire. Net gains in carbon, that is, sequestration or removals from the atmosphere, occur due to a positive net exchange of carbon dioxide by plants through photosynthesis and respiration, resulting in accumulation of carbon in biomass and soil. This positive net exchange of carbon dioxide occurs throughout the life of plants, at a greater rate during early growth and a slower rate as plants age, but it can remain positive even in old growth forests (Carey *et al.* 2001, Zhou *et al.* 2006, Luyssaert *et al.* 2008).

The main disturbance regimes influencing montane ash forests in the Central Highlands of Victoria are logging trees for pulp and sawlogs and fire including prescribed burning and wildfire. Impacts of harvesting trees on ecosystem carbon stocks depend on the logging cycle, silvicultural system, forest type and stand regeneration. In montane ash forests that are 'clearfelled and slash burnt' (80-90% of harvested area), impacts include: (i) removal of woody biomass carbon stocks off-site as input to the production of wood products (pulp and sawlog) and waste residues from processing (sawdust, wood with defect, off-cuts); (ii) conversion of living biomass to dead biomass in the form of 'residue' or 'slash' left on-site (i.e. tree canopy components, bark, stumps, roots and other vegetation), which then have accelerated decay rates; and (iii) emissions of carbon by combustion of 'slash' from post-logging site treatment. Logging depletes the CCS because most of the carbon in a temperate native forest ecosystem is found in the living biomass of large old trees. Logging cycles remove mature eucalypt trees and shift stand demographics so that the forest becomes dominated by regenerating young trees that store less carbon. Following logging, and assuming successful germination of seedlings, regeneration of the forest occurs and carbon dioxide is taken up by regrowth. The management of 'slash' will determine whether carbon emissions are rapid due to combustion or slower due to decomposition.

Impacts of fire on depleting CCS include: (i) combustion of a proportion of the living and dead biomass, depending on fire intensity and fuel type, resulting in emissions of carbon dioxide; and (ii) some loss of organic material off-site due to wind and water erosion. Following a fire in a native forest, natural regeneration and uptake of carbon dioxide occurs in the regrowth.

Both logging and fire disturbances change rates of emissions and sequestration and so over time change the magnitude of the carbon stock in the forest ecosystem. Evaluation of the impacts of disturbance regimes on native forest ecosystem carbon stocks requires understanding and quantifying the processes controlling the magnitude of losses and gains and longevity of each component (living and dead biomass and soil). Disturbances by logging and fire differ in their proportions of carbon stock loss and the resilience of the ecosystem to recover.

Net uptake of carbon by the ecosystem at the landscape scale is called the **Carbon Sequestration Potential** (CSP). Predicting these future carbon stocks requires, in practice, positioning the current carbon stock in relation to an ecosystem carbon accumulation curve as a function of the age structure of the forest (Rhemtulla *et al.* 2009, Conant 2011). This curve is defined by the processes of carbon gains and losses, including rates of forest growth, decomposition, respiration and combustion. Prediction of carbon accumulation at the landscape scale is inclusive of the effects of environmental variability due to climatic gradients, substrate differences, and topography. Estimates of future changes in carbon stocks must also include the impact of proposed management activities.

The carbon stock of a natural ecosystem at the landscape scale is called the **Carbon Carrying Capacity** (CCC); defined as the mass of carbon that can be stored under prevailing environmental conditions and natural disturbance regimes, but excluding disturbance by intensive human land-use activities (Gupta and Rao 1994, Keith *et al.* 2009b). The magnitude of this stock exists in a dynamic equilibrium that varies temporally and spatially in response to disturbance events within a range around a mean value of the stock. The carrying capacity is estimated at the landscape scale and refers to the average conditions within a dynamic equilibrium across a region.

Objectives of our study of carbon stocks in the forest ecosystems within the catchments (Maroondah, O'Shannassy, Upper Yarra, Cement Creek, Armstrong, Starvation Creek, McMahon's Creek, Thompson stages 1, 2, and 3, Tarago) and surrounding forested area in State forests within the Central Highlands region were to:

(i) Calculate the carbon stock in the catchment areas in the context of the forested landscape of the Central Highlands of Victoria. Calculation is based on data from ANU experimental sites and Victorian Department of Sustainability and Environment inventory sites, and then 'up-scaled' to the landscape level. The carbon stock includes all components of living and dead biomass above- and below-ground, litter and coarse woody debris measured at the sites and an estimate of soil carbon.

(ii) Model carbon sequestration during forest growth (i.e. post-disturbance to ecological maturity) based on the age structure of the forest and a function of carbon accumulation over time;

(iii) Quantify carbon losses after the 2009 wildfire based on site measurements of biomass components combusted and up-scaled to the catchment landscape;

(iv) Assess changes in carbon stocks due to logging, particularly the silvicultural systems of clearfelling, thinning and post-fire salvage logging; and

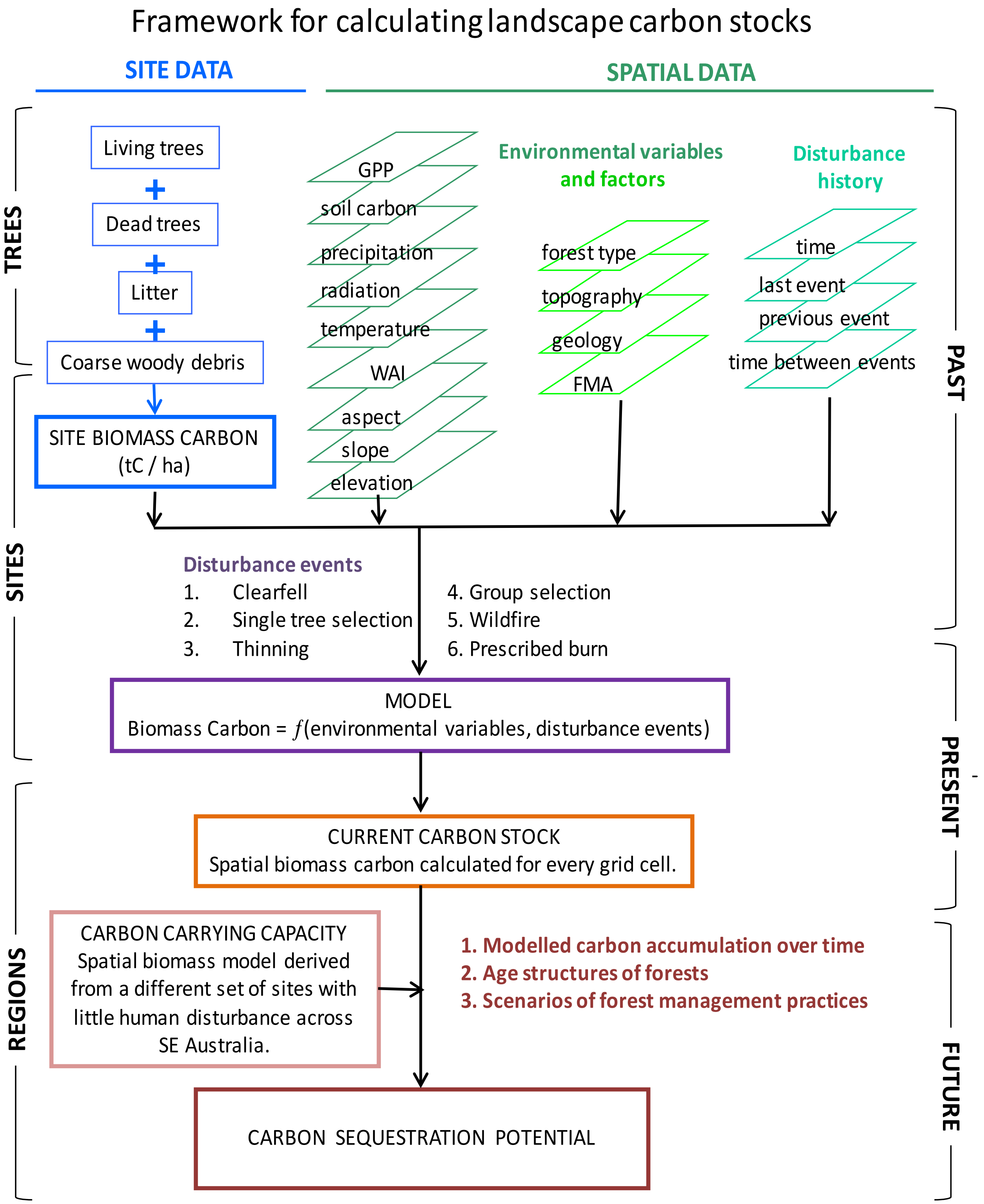
(v) Derive scenarios for the different fire and logging regimes and their impact on carbon stocks over 20, 50 and 150 years.

The study encompasses work at a range of spatial and temporal scales, types of data and analyses to achieve the objectives of a landscape map of the carbon stock and projected changes under different management scenarios. The study was based on the framework illustrated in Figure 1.1.

The sections of this report are divided according to spatial and temporal scales in the analysis of carbon stocks. Each section includes derivation of the data and application of the results to determine the carbon stock at a particular scale. Section 2 provides background information about the environment and history of the montane ash forests of the catchments within the Central Highlands of Victoria. Section 3 describes the methodology for measuring and calculating carbon stocks in trees and other biomass components, and then aggregating to the site scale in native forests. Section 4 uses the carbon component data from our field sites, together with environmental conditions and disturbance history, to construct a spatially-explicit map of carbon stocks at the landscape scale - the Current Carbon Stock. Section 5 quantifies the carbon loss due to combustion of biomass components in the 2009 wildfire. Section 6 includes the temporal scale with derivation of the function of carbon accumulation over time to a maximum potential Carbon Carrying Capacity. Carbon losses due to harvesting are quantified. Carbon Sequestration Potential is predicted under scenarios of change in carbon stock under different forest management regimes. Section 7 discusses how the spatially-explicit carbon stock derived for each catchment and predictions of changes over time resulting from different management practices will help inform policy decisions about management of forests in the water catchments.

There have been differing estimates of the carbon stocks in natural forests in Australia and the impacts of disturbance events. This has led to controversy about the value of protecting carbon stocks in forests, the relative impact of different forms of disturbance in causing emissions, and the mitigation value of carbon sequestration in natural forests. It is therefore important to calculate the current carbon stock as accurately as possible using the best available data and methods, and to document the methodology so that it is repeatable and can be compared with other studies. Accounting for spatial variability in forest age structure resulting from a history of disturbance events is a critical element in calculating current carbon stock and the trajectory of the sequestration potential.

Figure 1.1 A framework describing stages of the study at different scales and integrating different types of data to derive models that relate biomass carbon stocks to environmental conditions and disturbance history at a site, to up-scale across the landscape, and to predict carbon sequestration potential under scenarios of forest management practices. The 'tree' scale includes all plants.



Notes: GPP = Gross Primary Productivity, WAI = water availability index, FMA = forest management area, Disturbance history refers to the time and event type of fire or logging activities.

2. Description of Montane Ash Forests

2.1 Introduction

The location of the study is in the water catchments and surrounding region in the Central Highlands of Victoria. The main forest type consists of montane ash species and their composition and structure are described. The dominant species is *Eucalyptus regnans* (Mountain Ash) and most of the calculations of carbon stock refer to this species. However, it is important to recognise that minor eucalypt species and many mid- and under-storey species contribute to the composition and structure of the forest, and the response of its carbon stock to disturbance and regeneration.

The current structure of forest stands and their carbon stock depend on (i) the environmental conditions for growth at the site, (ii) the timing and type of previous disturbances, and (iii) the success of regeneration after disturbance. The history of disturbance events determines the age, size and density of trees, and hence the carbon stock of forest stands. Wildfire and logging are the main disturbance events that influence mortality and regeneration of these forests. Therefore, the history of these events and their impact on forest biomass are critical components in the assessment of current carbon stocks and prediction of future stocks.

A range in forest ages and stand structures cover the landscape and measurement strategies need to cater for all these types.

2.2 Forest species distribution

The study area includes the catchments of Maroondah, O'Shannassy, Upper Yarra, Armstrong, Cement Creek, McMahon's Creek, Starvation Creek, Tarago, Thompson stages 1, 2 and 3 (Map 1). The forest in this region is predominantly wet sclerohpyll of montane ash species, predominantly *Eucalyptus regnans* (Mountain Ash), but also *E. delegatensis* (Alpine Ash) and *E. nitens* (Shining Gum). The field sites are located mainly in *E. regnans* forest. The dominant height of ash stands is 60-80 m and diameter (at 1.3 m height) is up to 4m, measured at the field sites. Within the catchments there are small areas of other forest types, including wet and dry mixed species forest of *E. obliqua* (Messmate), *E. cypellocarpa* (Mountain Grey Gum), *E. macrorhyncha* (Red Stringybark), *E. radiata* (Narrow-leaf Peppermint), *E. sieberi* (Silvertop Ash), and subalpine forests of *E. pauciflora* (Snowgum). The distribution of forest types is shown in Map 1. The distribution of montane ash forest across the landscape is closely related to climatic conditions (Lindenmayer *et al.* 1996). Mixed-species forests typically occur in warmer areas on mid and upper slopes with high levels of incoming solar radiation, rainforests occur along streamlines in colder higher elevation areas, and montane ash forests occupy an intermediate environmental domain (Mackey *et al.* 2002).

Stands of montane ash forest consist of an overstorey of eucalypts; a mid-storey tree stratum of *Nothofagus cunninghamii* (Myrtle Beech), *Atherosperma moschatum* (Southern Sassafras), *Acacia dealbata* (Black Wattle), *Acacia melanoxylon* (Blackwood), *Acacia frigescens* (Forest Wattle), *Acacia obliquinervia* (Mountain Hickory Wattle); and a tall shrub layer (2m to 15 m) of *Olearia argophhylla* (Musk Daisy Bush), *O. phlogopappa*, *Bedfordia arborescens* (Blanket Leaf), *Pomaderis aspera* (Hazel Pomaderis), *Correa lawrenciana* (Mountain Correa), *Cassinia aculeata* (Dogwood), *Ziera arborescens* (Stinkwood), *Prostanthera* lasianthos (Victorian Christmas Bush), *P. melissifolia, Polyscias sambucifolia* (Elderberry Panax), *Hedycarya angustifolia* (Austral Mulberry), *Dicksonia antarctica* (Soft Tree Fern), *Cyathea australia* (Rough Tree Fern) (Costermans 1994, Lindenmayer and Ough 2006, Mueck 1990, Ough 2002).

2.3 Forest structure

Montane ash forest is multi-layered with an overstorey of eucalypts and mid- and under-storeys of trees and shrubs, which include rainforest species in moist, protected locations (Lindenmayer *et al.* 2000a). Unlike most eucalypt species, ash species (*E. regnans* and *E. delegatensis*) are usually killed if their canopies are largely scorched by fire (Flinn et al 2007). Therefore, the age of stands reflects the fire history as well as the land use disturbance history. However disturbances, particularly fire, can be highly variable spatially, both in extent and intensity. Hence, not all trees are necessarily killed in all fire events and regeneration is not necessarily even-aged. Montane ash forest can be even-aged resulting from widespread mortality and regeneration following an intense disturbance event, but may also be multi-aged following partial tree death and subsequent regeneration alongside surviving trees. Evidence of past disturbance by fire, such as charcoal scars on the butts of large trees indicates a history of disturbance events of varying intensity (McCarthy and Lindenmayer 1998).

Old growth forests have been defined in many ways and include different social and ecological values related to disturbance history, aesthetic qualities, conservation value, successional stage, and structural form. Some definitions used to characterise forest types for land management policy include:

‘forests both little disturbed and ecologically mature’ (RAC 1992)

‘forest that is ecologically mature and has been subjected to negligible unnatural disturbance…in which the upper stratum of overstorey is in the late mature to overmature growth phases’(DAFF 1992 - National Forest Policy Statement).

‘ecologically mature forest where the effects of disturbances are now negligible’ (DAFF 1992, 1998 - Regional Forest Agreements).

The latter definition for the Regional Forest Agreement was the criteria used for mapping old growth areas in our study (see Maps 12 - 15). This definition also included evaluation for National Estate values by the Victorian Department of Conservation and Natural Resources and the Australian Heritage Commission.

Old growth for the montane ash forests was defined by Lindenmayer *et al.* (2009) in terms of the size and prevalence of living overstorey trees; stands where >70% of living overstorey trees exceed 180 years old, and the majority of trees in the stand are greater than 100 cm DBH. These stands almost always have different age cohorts.

The characteristics of old growth montane ash forests also include the multiple layers of the forest of different species groups (Burgman 1996, van Pelt 2007), such as *Nothofagus cunninghamii* (Myrtle Beech), *Atherosperma moschatum* (Southern Sassafras) and tree ferns (*Dicksonia antarctica* and *Cyathea australis*) which can be very old, for example dated at 350 years (Mueck *et al.* 1996). Individual eucalypts have been dated at 309 ±6 years for a late mature growth stage of the tree, and 459 ±8 years for an older growth stage using dendrochronological studies (Banks 1993).

The primary criteria used in our study of forests in Victoria’s Central Highlands are growth stage and level of disturbance. Characteristics of growth stage that contribute to the definition of old growth include older growth stages with stands containing relatively large, old trees and other plants comprising the oldest naturally-achievable growth stage for the ecosystem under a natural regime of fire disclimax; and functional qualities of an ecologically mature ecosystem, such as presence of large crown gaps, presence of tree hollows and fallen trees, and high levels of biodiversity (DSE 1996). The level of disturbance is defined as negligible. However, montane ash forests experience periodic natural disturbance events and these events contribute to the ecological processes that produce old growth characteristics. Old growth ash forests are characterised by overstorey trees usually of multiple ages, which indicates a highly spatially variable pattern of mortality then regeneration as well as survival and recovery of trees (Banks 1993, Lindenmayer *et al.* 2000, Beale 2007).

2.4 Fire History

Wildfires and regeneration events are known to have occurred in the following approximate years: 1750, 1824, 1851, 1895, 1905, 1908, 1918, 1926, 1932, 1939, 1948, 1962, 1983, 2003, 2006 and 2009, within the Central Highlands region (DSE 1996, Lindenmayer 2009). The history of fire occurrence in the ash forest region over the last century pre-2009 is shown by decade in Map 3.

The mapped extent of fire occurrence represents the outer boundary of the burnt area. There is limited information about spatial variation in fire severity and small-scale patchiness of fire occurrence, for example due to topographic and diurnal effects and changes in weather conditions (except for the 2009 wildfire). Information about fire severity is important as it determines the degree of impact at a location. Not all areas designated as 'burnt' result in death of the trees, even for ash tree species (Smith *et al.* 1985, Lindenmayer et al. 2010). For example, our entire study area of the catchments and surrounding region occurred within the boundary of the 1939 fire; however, not all areas were burnt at a sufficiently high intensity to kill the trees. Evidence that individual trees and forest stands survived the 1939 fire is seen by the current age structure of patches of old-growth forest, particularly in the O'Shannassy and Maroondah catchments where many live trees have fire scars from the 1939 fires event. Much of our study area also occurred within the 1983 fire boundary in the south or 2009 fire boundary in the north. A few small areas have experienced two or three fires since 1939.

Ash eucalypt species are usually considered to be killed by fires that scorch more than 50-75% of leaves in the crown (Flinn *et al.* 2007). In areas subject to high severity wildfire, ash trees are completely killed and the species regenerate from seed as even-aged stands (Ashton 1975b). In areas subject to low or moderate fire severity, some or all of the overstorey trees survive and the stand regenerates with more than one age class (McCarthy and Lindenmayer 1998). The estimated mean fire interval is 75 to 150 years for a fire severity that kills ash trees (McCarthy and Lindenmayer 1998, McCarthy *et al.* 1999). Within the boundary of a wildfire McCarthy *et al.* (1999) estimated that approximately half of the trees survive. Hence, the mean interval of all fires is 37 to 75 years.

Fuel reduction burning of the drier mixed forests of the Central Highlands has been practised since the 1960s. Boundary information for these fires is recorded on maps and supported by field reports and aerial observation to determine the degree of disturbance from the mosaic of burning (DSE 1996).

2.5 Logging History

Timber harvesting in the 19th century involved selective logging of *E. regnans* in the more accessible parts of the Central Highlands nearer Melbourne to provide building materials and firewood. Utilisation of ash forests for sawn timber was initially limited by accessibility of this forest type in higher elevation and steeper terrain, and the unstable nature of the timber when it was dried. Demand for timber increased after the discovery of gold in the 1850s and the area of selective harvesting increased. More intensive harvesting of the ash forests did not occur until the 1920s with the introduction of steam powered winches and reconditioning to stabilise sawn timber (Griffiths 1992). Extensive logging of the mature ash forests continued during the 1920s and 1930s driven by the high demand from housing booms.

The wildfire in 1939 killed the majority of the remaining mature stands, but much of this timber was harvested in large-scale salvage operations during the 1940s and 1950s supporting the post-war housing boom (Noble 1977). Salvage logging continued until the dead trees became unsuitable for timber products (Noble 1977).

Extensive areas of regrowth forest originated from the wildfires in 1926, 1932 and 1939. During the 1950s, 1960s and 1970s harvesting occurred in 'understocked' regrowth stands of *E. regnans* and thinning was practised in fully-stocked regrowth stands. Extensive harvesting of *E. delegatensis* began in the early 1950s when supplies of salvage logs from the 1939 burnt *E. regnans* began to diminish. Harvesting of the remaining fully-stocked mature *E. regnans* stands not burnt in 1939 was completed by about 1990. Harvesting of the 1939 regrowth, as 'mature' stands, commenced in the mid-1980s and is currently continuing.

The history of logging events is shown in Map 4 by decade of occurrence over the last century. In the Central Highlands, it is the water catchments designated as conservation reserves that are most likely to have had minimal logging and other human disturbance impacts. Small areas of logging have occurred in the Cement Creek and Armstrong catchments. Extensive logging has occurred over the last 50 years in the Starvation Creek, McMahon's Creek, Thomson and Tarago catchments. In the Maroondah, O'Shannassy and Upper Yarra catchments there has been limited known disturbance from logging. Limited harvesting occurred in the Maroondah in the 1970s/early 1980s for testing water run off under different harvesting regimes. Some of these catchments have been closed to provide the water supply for Melbourne for over 100 years and prior to European settlement, there is little evidence of Aboriginal use of these forests areas (Lindenmayer 2009). There is some uncertainty about the extent to which early logging has been documented. Therefore, some areas that are now in conservation reserves may have been affected by harvesting under a previous land tenure that was not recorded.

The silvicultural system commonly applied in ash forests of Victoria is that of clearfelling and slash burning (Flinn *et al.* 2007). This system was introduced in the 1960s following research into regeneration methods. It was considered to be a simple method that produced regeneration of the forest following harvesting. The felling method involves removal of all commercial trees from a coupe area, usually in one integrated operation. Selected trees are retained for fauna habitat under the Department of Sustainability and Environment Code of Practice (2007a). Seed supply for eucalypt regeneration is either from direct seeding by hand or aerially, from the retained habitat trees or crowns of felled trees, or from planting where seed is scarce or the seedbed is poor. Seedbed preparation is preferentially by burning slash, otherwise by broadscale or selective mechanical disturbance. Slash burning is by high intensity fires in autumn. The rotation length for harvesting is prescribed as 50 years (DPI 2009) where even-aged 1939 regrowth is being harvested, but is often harvested at shorter times.

A range of other silvicultural systems have been used in the Central Highlands as minor practices. 'Single tree selection' is an uneven-aged silvicultural system involving the felling of scattered individual trees across the site, occurring at 10 to 15 year intervals, with regeneration mainly from lignotubers and coppice. 'Seed tree selection' is an even-aged silvicultural system in which all living trees are felled except for a number of uniformly-distributed trees retained to provide seed for regeneration and habitat. 'Group selection' is an even-aged silvicultural system that removes small groups of trees dependant on age and forest structure. 'Thinning from below' removes the smaller and poorly-formed trees from immature stand (20 – 30 years old) and allows the larger better-formed trees to increase their growth, but the harvesting also removes the mid-storey trees and shrub layer. Some thinning produces a commercial product but in other cases the trees remain as waste on-site (Flinn *et al.* 2007, DPI 2012–14).

Regeneration after a disturbance event is critical to allow regrowth of the forest and accumulation of carbon stocks. Early management of harvested mature ash forests often had problems with regeneration. Burning of slash to promote regeneration was practised in the 1950s and 1960s and involved burning a variable proportion of the crowns of felled trees. Since the 1980s, management of harvested forests has included regeneration by burning heaped slash at high-intensity and application of seed (Flinn *et al*. 2007). Regeneration with adequate stocking has been greatly improved through research on silvicultural methods and monitoring. Current practises are considered acceptable under the Victorian Code of Practice for Timber Production (DSE 2007a) that requires monitoring surveys of regeneration and remedial action if stocking is not adequate (Campbell 1997, Flinn *et al.* 2007).

Wood products derived from timber harvesting of ash forests include sawn timber and pulp in varying proportions (DSE 1996). Harvesting is highly variable in terms of wood volume removed due to harvesting methods of clearfelling, single tree or group selection; suitability of the timber for sawlog products; amount of wood degrade; product mix; distance to mills; and accessibility in the landscape. Pulpwood was approximately 13% of the total post-1939 salvage wood product, averaged over the entire ash forest area, as markets for paper were only just being established. This meant that residue levels associated with salvage logging were high and large amounts of dead biomass remained on-site (Flinn *et al.* 2007). Sawlogs comprised approximately 33% of the total wood product in the late 1990s (Flinn *et al.* 2007), since then has reduced to about 28% (DSE 2009), and is continuing to decline. The total amount of sawlog product has remained reasonably steady for the half century of records, but the amount of pulpwood has increased almost 10-fold (Flinn *et al*. 2007).

2.6 Synopsis

Montane ash forests form a mosaic of even-aged and multi-aged stands across the landscape. Even-aged stands result from widespread mortality and regeneration following an intense disturbance event. A multi-aged structure is characteristic of old-growth stands that have derived from spatially variable disturbance events that result in partial mortality and regeneration, and partial survival (Lindenmayer et al. 2000a).

A major source of uncertainty in assessment of carbon stocks is the age of stands and the impact of disturbance events on their biomass and soil carbon content. Fire events are highly variable in their intensity and hence the severity of impacts and extent of tree mortality. Historically, the outer boundary of a fire was recorded but information about the spatial variability of the fire within this bounded area was limited. Timber harvesting has been highly variable in the extent and degree of biomass removal and success of subsequent regeneration. Records were limited in the early period and particularly from salvage logging after the 1939 wildfire. Hence, the current carbon stock of living biomass and especially total biomass (living plus dead) is highly variable in relation to forest age. The various forms of selective harvesting result in uneven-aged stands with limited information about ages of tree cohorts. Assessment of carbon stocks in forest stands in relation to their age makes the assumption that full regeneration occurs, which is not always the case. Therefore, the current carbon stock of a forest depends on environmental conditions and age of the stand, but also depends on the proportion of biomass remaining on-site after disturbance, regeneration method, stocking rate, and success of the regrowth.

3. Measurement of Ecosystem Carbon Stocks at the Site Scale

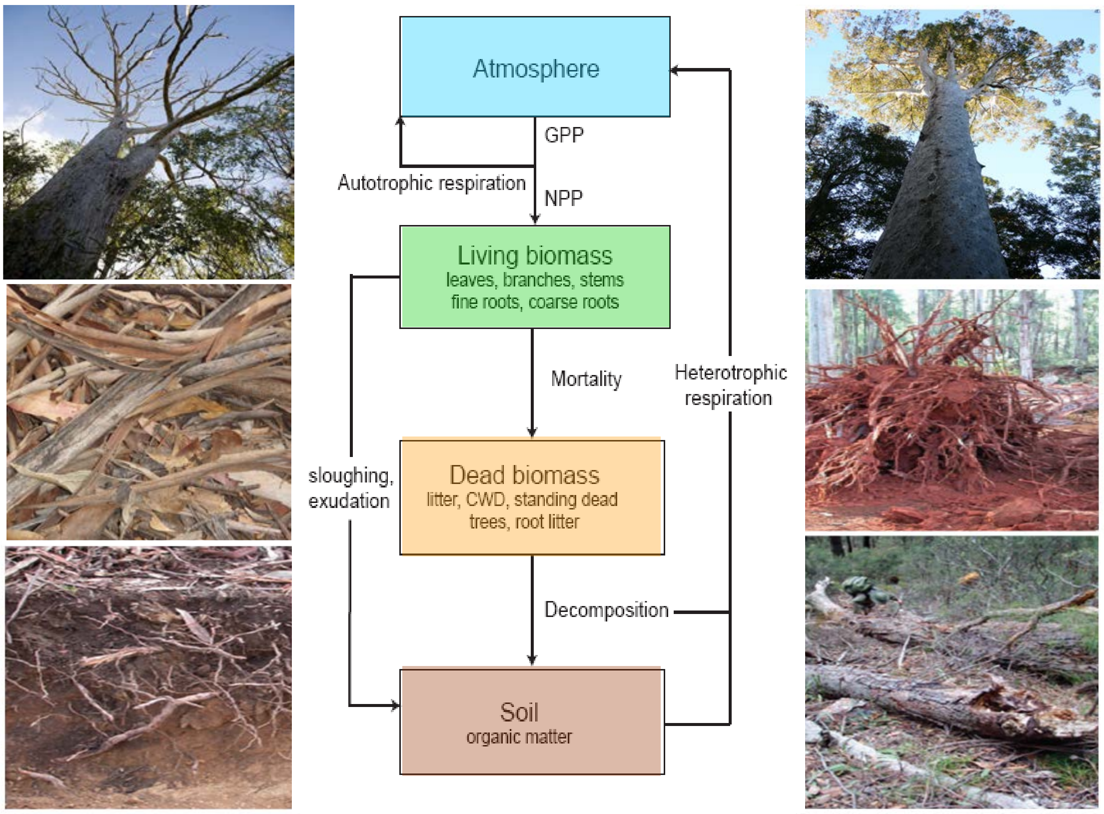
3.1 Introduction

Quantifying the carbon stocks of an ecosystem and upscaling across the landscape requires accounting for all the components as shown in Figure 3.1. Carbon stocks were estimated for each component at each site. Site level carbon stocks were used to relate to environmental conditions and disturbance history of the sites, and to assess the amount of carbon lost by combustion in the 2009 fire. Prediction of carbon stocks across the landscape depends on good quality site data of these measured components; the accuracy of the landscape model will only be as good as the calibration data. Obtaining good quality site data, both from field measurements and application of existing data, was a major objective of this project.

The calculated current carbon stock provides the basis for valuing the ecosystem service of carbon stocks protected in the biosphere, predicting sequestration over time, and assessing carbon loss due to fire. Improving the methodology and addressing issues of concern were priorities in our study, including using the best available data and methods, comparing different sources of data, testing the sensitivity of biomass resulting from different methods, and describing the process in a transparent manner.

Specific methods that were assessed included allometric equations to calculate tree biomass from dimension data, diameter measurements over large buttressed trunks, proportion of internal decay in stem volume, estimation of bark mass combusted, measurement of dead trees, litter and coarse woody debris, and assessment of size structure of forest stands in relation to disturbance history.

Figure 3.1. Stocks and fluxes of the carbon cycle in the forest ecosystem.



3.2 Experimental Design

3.2.1 Sample sites

Data from two sets of sites were used in the analysis to provide both intensive and extensive information on carbon stocks across the region. Spatial modelling of biomass carbon stocks at landscape scales requires a large number of calibration sites across a wide range of environmental conditions and temporal range in time since disturbance in order to capture both the natural and human-induced variability.

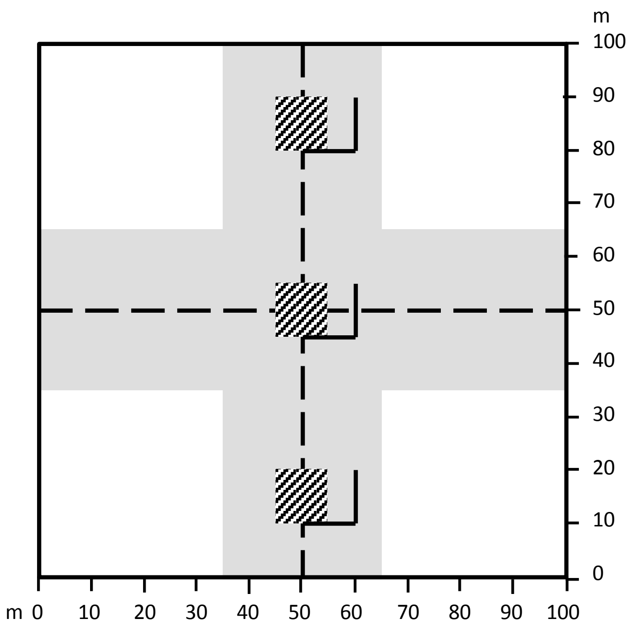
3.2.1.1 ANU Carbon sites

A subset of sites (n = 54) from the ANU's long-term ecological monitoring network of 171 sites, each of 1 ha, in the Central Highlands of Victoria (Lindenmayer 2009, Lindenmayer *et al.* 2003, 2011) was selected for detailed measurements of carbon stocks of all ecosystem components (Map 1). These forest stands are dominated by *E. regnans*. The sites are located across State Forest and National Park both within and outside the catchments to provide suitable forest conditions to fit the criteria of age and fire severity (see Section 3.2.2 and Table 3.1).

Within the 54, 1 ha sites (Figure 3.2), biomass components were sampled using different strategies depending on the size and distribution of different components. Sites were located at a random point that marks the beginning of a central transect of 100 m through the site. A second transect of 100 m was located perpendicular through the 50 m point of the central transect. Three 10 m x 10 m (3 x 0.01 ha) plots were located at the beginning, middle and end of the central transect.

The site treatments and measurements of carbon stocks at the site level, described in section 3, refer to the research at the 54 ANU carbon sites.

Figure 3.2 Design of 1 ha (100m x 100m) experimental sites showing the central and perpendicular transects (dashed lines), 10m x 10m plots (hatched), 10m transects (black lines), and area sampled for large trees 2 x 30m x 100m = 0.51 ha) (grey shaded area). Different biomass components were sampled in different parts of the site.



3.2.1.2 SRFI sites

Existing data from the Statewide Forest Resource Inventory (SFRI) for the Central Highlands Forest Management Area of State Forest (DSE 2007b) were used for additional analysis of carbon stocks in the montane ash forest type (under a data licence agreement with the Forest Resource Analysis and Modelling section of the Department of Sustainability and Environment, Victoria). Sites in the inventory were classified by DSE according to forest type based on aerial photo interpretation. Individual measured trees within sites were assigned to species. We selected sites to represent the montane ash forest type based on species of measured trees (sites containing a majority of *E. regnans* or *E. delegatensis*) as this was considered more accurate than the forest type classification. Other eucalypt species co-occurred at many sites.

The inventory data consisted of (i) inventory data of tree dimensions recorded at a network of plots (SFRI plots, n = 876) (Map 2), (ii) spatial data of forest composition, and (iii) spatial data about forest activity including fire and logging history (i.e. the silvicultural system applied). These sites provided spatial coverage across the region. These existing inventory data were used to calculate carbon stocks, however, the data were collected for a different purpose, namely estimating wood volumes, and so required considerable analysis and application of ancillary data to obtain the required results. No additional data collection was conducted at these sites during the current study.

SRFI sites were located in a stratified random design by forest type. They are variable radius plots with measurements of tree diameter and probability of occurrence of a given sized tree (DSE 2007b). Variable probability sampling is based on the theory that the probability of selecting an individual is dependent on its size (Husch *et al.* 2003). From a central point, trees are assessed to determine whether the ratio of tree radius (DBH/2) : plot radius (distance) exceeds the critical ratio or Basal Area Factor(BAF). An optical wedge instrument helps determine this ratio by projecting a horizontal angle of a fixed size (θ) to determine whether the tree diameter is greater than the projected size. If the ratio exceeds the BAF, then the tree is counted 'in' within its respective plot area.

3.2.2 Treatments

Both forest age and impact of the 2009 wildfire were important for determining the carbon stock of the ecosystem and projected changes over time. Hence, the ANU carbon sites were selected to represent three age categories (1983 regrowth, 1939 regrowth, old growth) and three fire categories (unburnt, low severity fire, high severity fire) in a factorial design. Six sites were sampled from each of the nine categories giving a total of 54 sites.

The 1983 regrowth resulted from clearfell logging or salvage logging after the 1983 wildfire. The 1939 regrowth resulted from wildfire and variable amounts of salvage logging. These are common age classes of montane ash forest and are usually considered even-aged regrowth. However, there were some larger residual trees in the stands that contributed high biomass. Therefore, carbon stock of individual regrowth sites was highly variable depending on the distribution of these residual trees. Old growth sites are multi-aged and have trees originating from pre-1900. Many of the large trees are considered to have regenerated after a fire in approximately 1750 (Lindenmayer *et al.* 1990, 2000). The old growth sites were highly variable in their stand structure and age due to their differing histories of disturbance and regeneration. When selecting sites in the age / fire categories from ANU's network of sites, it was not possible to find old growth sites of similar structure that had been burnt and unburnt. This is partly due to the confounding fact that the 2009 fire occurred in the O'Shannassy catchment where the oldest stands of trees existed. This forest was considered old growth (Lindenmayer *et al.* 2000a) with very large trees dating back to the 1750s plus younger cohorts of trees, and also gaps in the overstorey canopy. There are very few areas of such old-growth forest that remain unburnt and they are difficult to access. Unburnt old growth or mature forest occurs in the Maroondah catchment where the oldest trees likely regenerated in the 1820s and 1850s, and hence are somewhat younger than those in the O'Shannassy catchment.

Fire severity at each site was classified into five classes: 1 – unburnt, 2 and 3 – burnt by low severity fire where the overstorey canopy was mostly unscorched, 4 and 5 – burnt by high severity fire where the overstorey canopy was scorched or combusted. Criteria used for assessing fire severity included degree of crown scorch of each vegetation layer and minimum tip of branches combusted (Table 3.1). The criterion of 'combusted' means the leaves and possibly twigs and small branches on a plant were consumed. The fire severity classes were assessed by on-ground survey within 4 months of the fire.

Table 3.1. Criteria for fire severity categories based on the degree of scorch or combustion of leaves in each vegetation layer.

Category		Over-storey	Mid-storey	Under-storey	Ground cover
Unburnt	1	unburnt	unburnt	unburnt	unburnt
Low	2	unburnt	unburnt	scorch	combusted
Low	3	unburnt/scorch	scorch	Scorch/combusted	combusted
High	4	scorch	combusted	combusted	combusted
High	5	combusted	combusted	combusted	combusted

The assumption that replicate sites (n = 6) within an age / fire category had similar conditions before the 2009 fire was tested by analysing the environmental conditions at the sites. The significant effects of the following variables were tested using analysis of variance (ANOVA, $F_{pr(4,45)} < 0.05$, Least Significant Difference of the Means at 5% level using the Protected Fisher Test, Genstat v.14): GPP, mean temperature, minimum temperature, precipitation, water availability index, radiation, soil organic carbon (SOC), slope, aspect, elevation (variables are defined in Table 4.1).

The only variables that exhibited significant differences for age / fire categories were GPP (F_{pr} = 0.031) and SOC (F_{pr} = 0.004). Among the young regrowth sites before the fire, the sites that remained unburnt had significantly higher GPP than the sites that burnt in low and high severity fires. This difference did not occur for the other age categories of forest sites. SOC differed between specific age / fire categories; with young regrowth unburnt and subject to high severity fire, mature regrowth unburnt and subject to low severity fire, and old-growth subject to low severity fire having significantly lower SOC than young regrowth subject to low severity fire, mature regrowth subject to high severity fire, and old-growth unburnt and subject to high severity fire. However, there was no trend related to the fire treatments.

Factors with categorical data were analysed by a log-linear regression model based on a χ^2 distribution that assessed the frequency of the different levels of each factor in each age / fire category and calculated a deviance ratio where the significance was tested using a χ^2 probability (Genstat v. 14). The following factors were analysed: forest type, topographic position and geology. The factor that exhibited significant differences among age categories was topographic position ($\chi^2_{pr} = 0.048$). Old-growth sites were more likely to occur on mid- and lower-slopes and no sites occurred on ridgetops, whereas young and mature regrowth sites occurred more often on ridgetops than mid- and lower-slopes.

Many of the sites in age / fire categories were clumped geographically, by necessity due to the location of age cohorts of forest and occurrence of the fire. Many of the high severity burnt sites were in the O'Shannassy catchment and many of the unburnt and low severity burn sites were in the Maroondah catchment. In the areas with no recorded disturbance history, there would still have been some disturbances (even if before historical records), and they would have differed in different geographical locations, thus resulting in different age structures of forest stands.

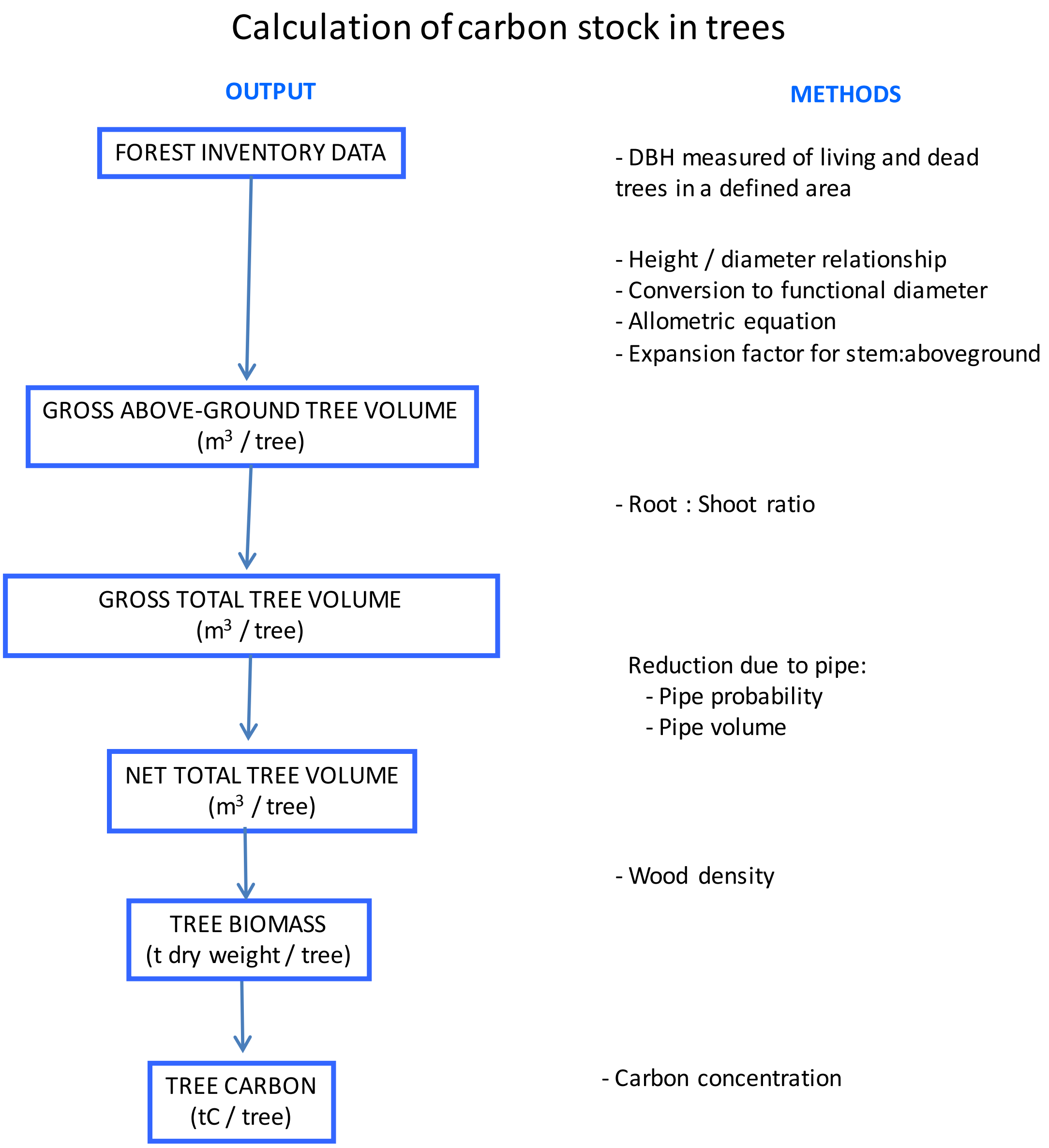
3.3 Measurement of ecosystem carbon components

Biomass components of forest stands were measured to calculate their current carbon stock and to assess the proportion of biomass combusted in the 2009 wildfire. The following components were measured: living and standing dead biomass; coarse woody debris; and the litter layer.

3.3.1 Living and standing dead biomass

The methodology for calculating carbon stocks in living and dead biomass is described as a framework that links estimates of biomass across scales. Figure 3.2 illustrates the stages required for the calculation of tree biomass and the different types of source data including forest inventory and ancillary data. It is important to test and compare different assumptions and sources of data used in the calculation of biomass to determine the sensitivity of the result to different methods of calculation.

Figure 3.3. Framework showing the stages in measurement and calculation of carbon stock in trees.



3.3.1.1 Inventory data

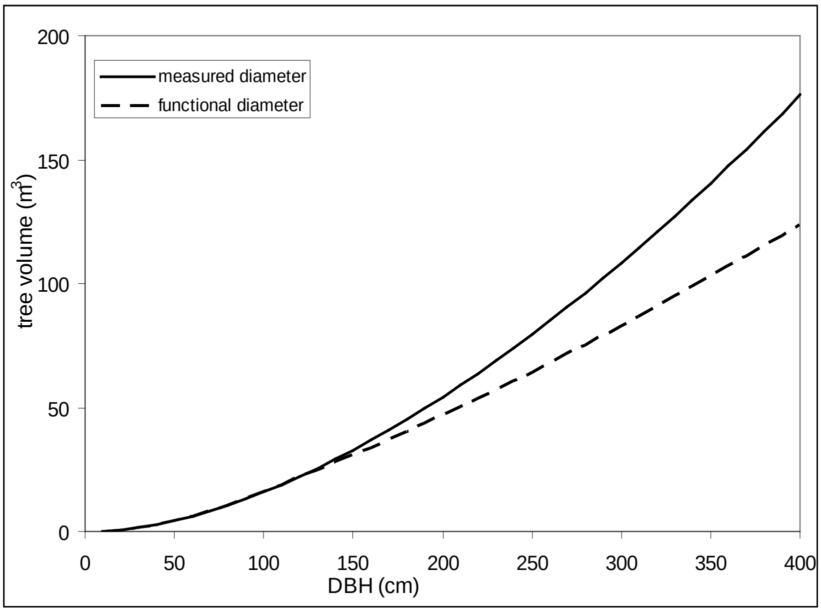
At the 54 ANU carbon sites where biomass components were measured, tree sizes were separated into two categories; less than and greater than 100 cm diameter. They were sampled using different strategies to maximise the area assessed for large trees because they have a highly variable distribution, and also to measure high density small stems in a smaller area. All woody stems 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 x 10 m plots (0.03 ha). Trees greater than 100 cm were measured in the two 100 m transects by 30 m width (0.51 ha) (Trees within the cross-over section of the two transects were only counted once.) (Figure 3.2). Tree diameter was measured at 1.3 m height (using standard DBH measuring methodology) on the uphill side but not on an elevated litter layer, and height was measured with a rangefinder (Nikon Forestry 550 Rangefinder). Measurement of both living and dead trees was important as both contribute to the biomass carbon stock. Biomass of dead under- and mid-storey plants were calculated as stem-only. Biomass of dead over-storey trees was calculated as stem plus branches for trees at the normal canopy height (for a given tree diameter). For trees with a measured height less than the normal canopy because the top section of the stem had broken off, biomass was calculated for stem-only because it was assumed that the branches would also have broken off. Where the stem of the dead trees were no longer intact, due to decay or pieces broken off, the proportion of the dead stem remaining was estimated.

At the 876 SRFI inventory sites, the existing data included living and dead trees greater than 20 cm diameter with measurements of individual tree DBH, the corresponding stem density, and dominant height of the stand. These data were used to calculate tree biomass carbon stock.

Measurements of tree diameter can be problematic in large *E. regnans* that form buttresses with fluted stems (Ashton 1975b). Tape measures of DBH as the circular cross-section do not account for the deficit between the indentations of the buttresses and therefore can be inaccurate. The fractional cross-sectional area deficit can be high (up to 0.6), but is less for the fractional deficit of total stem volume, which increases with tree size and is approximately 0.09 for trees of 3 m diameter and 0.12 for trees greater than 5 m diameter (Dean *et al.* 2003). A reduction in the cross-sectional area based on a 'functional' diameter was measured by Sillett *et al.* (2010) using detailed mapping of the cross-sectional area of tree bases of *E. regnans*. Trees with DBH greater than 122 cm were found to begin forming buttresses. The derived relationship is given in Equation (1) and the difference between measured and functional diameter across a range of tree sizes is illustrated in Figure 3.4. This conversion of functional diameter was calculated for all trees greater than 122 cm DBH in the inventory to use as the independent variable in the volume function.

Functional diameter (cm) = $2.4242 \cdot DBH^{0.8157}$ for tree DBH > 122 cm (1)

Figure 3.4. The effect of using an estimated functional diameter compared with a measured diameter over buttresses, on reducing volume of individual trees of a given size.



3.3.1.2 Allometric equations to predict volume and biomass

Allometric equations relate dimensions of a tree (diameter and height) to its volume or biomass (Keith *et al*. 2000, Snowdon *et al*. 2002). Equations reported in the literature were collated for this investigation. However, there has been little work done on measuring the biomass of ash species because of the difficulties created by their large size. Criteria for selecting appropriate allometric equations included their derivation from trees of similar species, environmental conditions, size range to those trees in the inventory for which biomass is being predicted, and appropriate measures of tree dimensions. It is preferable for an equation to use both diameter and height dimensions when used for predictions over a wide range of environmental conditions. This is because the diameter / height relationship of trees varies in response to variations in environmental conditions.

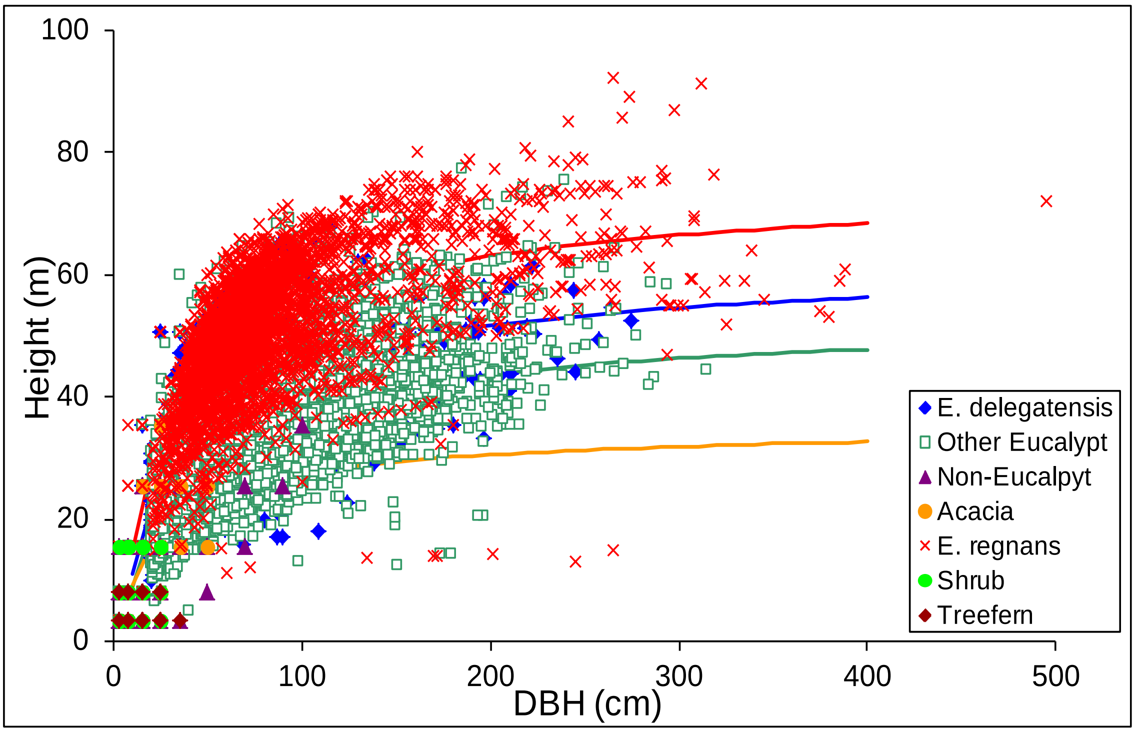
A relationship between DBH and height for living and dead trees was derived by collating various sources of existing tree measurements (n=21669 trees), including the ANU carbon sites from this study, the ANU long-term monitoring sites in the Central Highlands (Lindenamyer 2009, Lindenmayer et al 2011), Keith *et al.* (2009), Dean *et al*. (2003), Sillett *et al*. (2010), and the SFRI dataset. This relationship was used to predict height for tree inventory data where only DBH data were available. The *E. regnans* trees measured by Sillett *et al*. (2010) were significantly taller for a given diameter than trees measured elsewhere. This is likely because Sillett *et al*. (2010) chose an area at Wallaby Creek in Victoria with the tallest *E. regnans* and selected trees of good form for their detailed measurements. The other data sources were from inventory plots where the population of trees was random and therefore exhibited greater variance in the height / DBH relationship. The relationship for dead trees was highly variable due to broken tops of trees; hence a mean height for a species was more appropriate. The relationship is also variable for old, living trees that are prone to stem and crown breakage due to mechanical stresses so that height declines in very large trees (Jacobs 1955, Hickey *et al*. 2000, Lindenmayer and Wood 2010). The form of the relationship is given in Equation (2), the coefficients are in Table 3.2 and the data are illustrated in Figure 3.5.

Model: $\ln Ht = A + B * R^{(\ln DBH)}$ (2)

Table 3.2. Coefficients for Equation (1) relating tree height to DBH with coefficients A and B for living trees, r^2 = 0.768 and R = 0.5056; and a mean height for dead trees.

	Living trees		Dead trees
	A	**B**	**Mean height (m)**
E. regnans	4.361	-8.021	28.2
E. delegatensis	4.174	-8.489	23.1
Other Eucalypts	4.015	-8.806	23.5
Non Eucalypt	3.391	-7.768	6.0
Acacia spp.	3.598	-6.656	11.0

Figure 3.5. .Relationship between height and DBH of individual living trees (n = 21669) as logarithmic regression functions, and showing significant differences in the coefficients between species (*P*<0.001).



Existing information about allometric equations for tall wet forest species was assessed for volume and biomass (Figure 3.6a and b, Table 3.3). These curves show the wide range in predicted volume or biomass for given tree dimensions, and the wide variation that can occur when equations are extrapolated. An equation that provides biomass predictions similar to other equations for small trees may give very different predictions when extrapolated beyond size range for which it was calibrated.

Figure 3.6a. Allometric equations that relate tree diameter (DBH) to tree wood volume. Dean and Roxburgh (2006) (*E. regnans* in Tasmania, maximum size 400 cm), Keith *et al.* (2000) (*E. obliqua* in Tasmania, maximum size 280 cm), Sillett *et al.* (2010) (*E. regnans* in Central Highlands, maximum size 320 cm)), and using independent variables of DBH and Height for *E. regnans* in the DSE (2007b) equation.

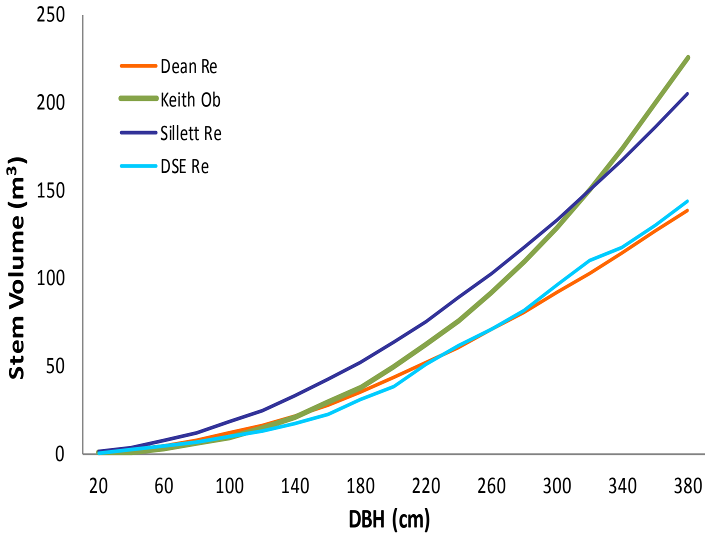


Figure 3.6b. Allometric equations that relate tree (a) diameter (DBH) and (b) diameter and height to biomass from Ashton (1976), Keith *et al*. (2000), Keith *et al*. (2009),Bi *et al.* (2004), Sillett *et al*.(2010), Dean *et al.* (2003), Feller (1980), West *et al*. (1991), where the solid line represents the size distribution of trees sampled and the dashed line is an extrapolation which shows the differences in predictions.

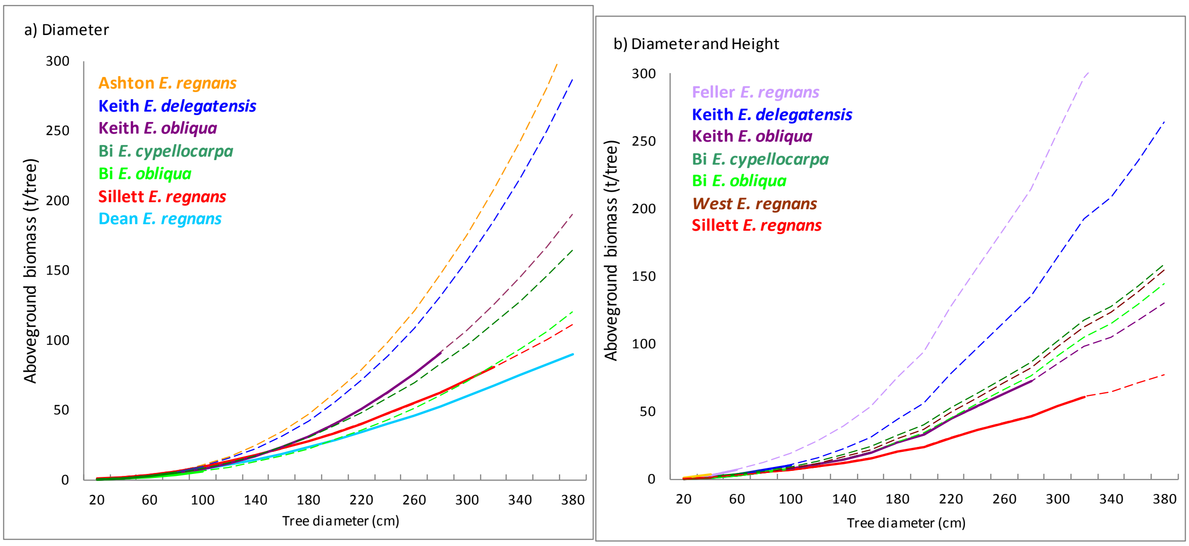


Table 3.3 Allometric equations for tall wet eucalypt species in south-east Australia. DBH is diameter in cm and Girth in cm, Height in m, M is biomass dry weight, V is volume, AGB is aboveground biomass, EMS error mean square.

Reference / Species	Component	Equation	EMS	r^2	n	DBH
Dean and Roxburgh (2006) *E. regnans*	Stem volume (m^3)	$V = 1100x[1+(D/9.2)^2]^{-1}]$	34.2	0.95	24	29-640
Sillett *et al.* (2010) *E. regnans*	Stem & Branch volume (m^3) AGB (kg)	$\text{Ln Vol} = 0.8029 \times \text{Ln}(D^2H) - 7.871$ $\text{Ln M} = 1.859 \times \text{Ln(d)} - 6.3518$ $\text{Ln M} = 0.8102 \times \text{Ln}(D^2.H) - 8.621$	0.02399 0.02977 0.02616	0.96 0.96 0.96	22 22 22	80 – 312 80 – 312 80 - 312
Ashton(1976)*E.regnans*	AGB (kg)	$\text{Log M} = -2.43 + 2.58 \times \text{Log (Girth)}$			11	8-54
Feller (1980) *E. regnans*	Stem mass (kg) Leaf mass (kg) Branch mass (kg)	$M = -45.6 + 248.9 \times D^2.H$ $M = -16.9 + 6.4 \times \text{LnD}$ $M = -42.2 + 25.7 \times D^2.H$			6	17-67
Bi *et al.* (2004) *E. obliqua*	AGB (t)	$M = (\exp(-1.87 + 2.218 \times \text{Ln(D)}) + (\exp(-1.36 + 1.598 \times \text{Ln(D)}) + (\exp(-3.868 + 2.398 \times \text{Ln(D)}) + (\exp(-5.76 + 2.326 \times \text{Ln(D)})/1000$ $M = ((\exp(-4.176 + 0.984 \times \text{Ln}(D^2.H))) + (\exp(-3.102 + 0.716 \times \text{Ln}(D^2.H)))+(\exp(-3.868 + 2.398 \times \text{Ln(D)})) + (\exp(-6.943 + 2.542 \times \text{Ln(D)})))/1000$	$0.762\hat{Y}^{1.408}$ $0.293\hat{Y}^{1.566}$	0.96 0.94	27 26	10-90 10-90
Bi *et al*. (2004) *E. cypellocarpa*	AGB (t)	$M = (\exp(-1.725 + 2.258 \times \text{Ln(D)}) + (\exp(-2.067 + 1.919 \times \text{Ln(D)}) + (\exp(-3.868 + 2.398 \times \text{Ln(D)}) + (\exp(-4.733 + 2.060 \times \text{Ln(D)}))/1000$ $M = ((\exp(-3.372 + 0.939 \times \text{Ln}(D^2.H))) + (\exp(-3.411 + 0.793 \times \text{Ln}(D^2.H))) + (\exp(-3.868 + 2.398 \times \text{Ln(D)})) + (\exp(-4.733 + 2.060 \times \text{Ln(D)})))/1000$	$0.474\hat{Y}^{1.408}$ $0.112\hat{Y}^{1.566}$	0.98 0.98	6 8	10-70 10-70
Keith (2009) *E. delegatensis*	AGB (kg)	$\text{Ln M} = 2.56 \times \text{Ln(D)} - 2.67$ $\text{Ln M} = 1.0734 \times \text{Ln}(D^2.H) - 4.693$	0.05857 0.02575	0.98 0.98	6 6	18-89 18-89
Keith *et al.* (2000) *E. obliqua*	AGB (kg)	$\text{Ln M} = 2.429 \times \text{Ln(D)} - 2.293$ $\text{Ln M} = 0.09418 \times \text{Ln}(D^2.H) - 3.298$	0.0407 0.02186	0.99 0.99	30 30	20-284 20-284
West (1991) *E.regnans*	AGB (kg fresh)	$\text{Ln M} = 6.8787 + 0.7415 \times \text{Ln(h)} + 2.0318 \times \text{Ln(D/100)}$			11	

Allometric equations that were selected for use in the Central Highlands forests are given in Table 3.4 for a range of species or tree forms. There is limited information for individual species except for the dominant eucalypts. Biomass of eucalypt trees was calculated using two methods to assess the difference in the predicted total biomass density.

Method 1: The equation derived by Sillett *et al.* (2010) was selected because the 22 *E. regnans* trees occurred in the ash forests of Victoria and were measured in detail for dimensions and volume. The method of measuring tree diameter and height was thorough and consistent for all trees; the data included large trees; and provided total wood volume in stems and branches. Volume of the measured trees was reduced for decaying wood visible in dead branches and damaged stems, but this was a small proportion overall with a mean of 1%. Decay and hollows that were internal within the stem, and so not visible, were not included in the volume equation. To apply this equation to all the inventory data, height needed to be estimated using Equation (2). Volume of dead trees was calculated using a stem only volume equation.

Method 2: The tree volume function derived by DSE required measured tree diameter and height derived from mean dominant height (MDH) and mean dominant diameter (MDD) of the stand (estimated from air photo interpretation) (Table 3.5). There is no relationship to estimate height of dead trees and so the relationship in Equation (2) and coefficients in Table 3.2 were used. Tree volume represented volume of merchantable wood. The fractional area deficit due to buttressing would be included empirically in this volume equation because it is based on estimates of timber volumes. Estimated stem volume must be converted to total wood volume of stems plus branches, stem base and non-merchantable wood. Biomass of each tree is calculated and multiplied by stems per ha to give the site biomass density (tC ha^{-1}).

Table 3.4. Allometric equations used in this study for prediction of biomass of the range of species in the Central Highlands monitoring sites.

Reference	Component	Equation	EMS	r^2	n	DBH (cm)
Sillett *et al.* (2010) *E. regnans*	Stem & Branch Stem volume (m^3)	Ln Vol = 0.8029 x ln (D^2H) -7.871 Ln Vol = 0.8014 x ln (D^2H) -7.920	0.02399 0.02054	0.96 0.95	22 22	80 – 312 80 – 312
Feller (1980) *Acacia spp.*	Stem (kg)	M_{stem} = 1.9 + 424.9 x D^2H	61.3 (se)	0.98	5	5 - 25
	Branches (kg)	$M_{branches}$ = -24.8 + 209.9 x D^2H	31.0 (se)	0.91	5	5 - 25
	Leaves (kg)	M_{leaves} = 0.6 + 8.0 x D^2H	1.0 (se)	0.94	5	5 - 25
Keith *et al*. (2000) Rainforest and understorey	AGB (kg)	Ln AGB = -1.8957 + 2.3698 x lnD	0.08658	0.97	50	
Beets *et al*. (2011) treeferns	AGB (kgC)	ABG = 0.00270 x $(D^2H)^{1.19}$ root:shoot = 20%				

Height (H) in m for all equations. Diameter (D) in cm for Sillett and Keith equations and in m for Feller equation. Aboveground biomass (AGB).

Table 3.5. Derivation of tree height and tree volume used in the DSE inventory based on measured individual tree diameter (DBH), and estimated mean dominant height (MDH) and mean dominant diameter (MDD) of the stand.

Species	Height Functions
E. regnans	Alpha = Log(MDH x 1.33) / Log(10) Beta = (MDD x ((Log(MDH) / Log(10)) - Alpha)) TreeHt = 10 ^ [(Log(MDH x 1.33) / Log(10)) + (Beta / D)]
E. sieberi	TreeHt = 0.990655 x MDH x [(1 - Exp(-0.018749 x D)) / (1 - Exp(-0.018749 x MDD))] $^{0.54917}$ + 0.600133
Other eucalypts	TreeHt = 10 ^ [(Log(1.14394 x MDH - -5.36888) / Log(10)) + (-1.5607 - 0.08073 x MDD) / D]
	Volume functions
E. regnans, E. obliqua, E. cypellocarpa, E. nitens, E. viminalis, E. dalrympleana	TreeVol = D^2 x H / (10^[2.92 - 870 / (D + 50)^2])
E. delegatensis	TreeVol = 1.0728 - 0.4672 x D + 0.04676 x D^2 - 0.02328 x H + 0.0010515 x H x D^2 + 0.010128 x H x D
E. sieberi, other eucalypts	TreeVol = 10^(2.0656 * Log(D) / Log(10) + 0.8329 * Log(H) / Log(10) - 2.5762)

An expansion factor to convert merchantable wood volume to total wood volume in a tree was derived by combining data for stem and total aboveground biomass (n = 39 trees) from Sillett *et al*. (2010) for *E. regnans*, and Keith (unpubl. data) for *E. delegatensis*. Aboveground mass includes total stemwood and bark from tree base to top. Average ratio of stem mass : aboveground mass was 0.88 g g^{-1} ±0.01 with a range from 0.63 to 0.95 g g^{-1}. Ash species have a strong apical dominance and hence a lower proportion of branches compared with other eucalypt species.

Expansion factor: AGB = stem biomass x 1.136 (3)

The dominant species on all our sites was *E. regnans*. However, minor species occurred such as *E. delegatensis, E. cypellocarpa, E. nitens* and *E. obliqua*. Although equations have been derived for most of these species, it was decided that the use of a single *E. regnans* equation was preferable. This was because the predicted biomass of large trees was more conservative and use of different equations could produce variability in predicted biomass that was an artefact of the equations and not related to environmental conditions for tree growth at a site. Only one equation is known for *Acacia* spp. (*A. obliquinervia* and *A. dealbata* combined) in the Central Highlands (Feller 1980). Equations for other mid- and understorey species are not available, hence a general equation that was derived for rainforest species (Keith *et al*. 2000) has been used for all other species. The mid- and under-storey trees and shrubs represent diverse species and structural forms; however there is no information about the biomass of this type of vegetation.

There are no known relationships between diameter and height for tree ferns (*Cyathea australis* and *Dicksonia antarctica*) (Beets *et al*. 2011). Their stems are reasonably cylindrical, except for a swelling at the base of larger individuals or stems leaning at an angle. Hence, biomass was calculated as the volume of the cylinder multiplied by the wood density. Wood density was measured from disc samples taken from fallen tree ferns (Density = 0.36 g cm^{-3}, se = 0.046 g cm^{-3}, n = 6). The values of biomass calculated were very similar to that calculated using an allometric equation derived for tree ferns in New Zealand of the same genera of *Dicksonia* and *Cyathea* (Beets *et al*. 2011) (Table 3.4).

3.3.1.3 Stem volume reduced by pipes

Calculation of tree biomass from stem volume and wood density, rather than weighing the actual tree, does not account for internal wood decay and piping that is common in large old trees (Mackowski 1987). The term pipe refers to any defects in the heartwood that result in loss of merchantable wood by affecting the grade of sawlog. Hence, it can range from decayed wood to a hollow (DSE 2007a). Most pipes result from wounds to the tree, such as by fire, wind or harvesting damage, animal damage, or abscission of branches, which lead to fungal infection and decay of wood by a range of organisms like termites, fungi and bacteria (Gibbons and Lindenmayer 2002). Ash eucalypts typically shed lower branches as the trees grow in height, and loss of a hollow branch may leave a cavity in the trunk along the zone of abscission (Lindenmayer *et al*. 1993). Heartwood decay is influenced by the age of the tree, degree of damage, species, environmental stresses to growing conditions, and presence of decay organisms (Ambrose 1982). Uncertainty about the magnitude of internal decay and the effect on total biomass in large trees has been a major issue for calculating carbon stocks in native forests. (Dean *et al*. 2003, DSE 2007b)

Eucalypt trees typically begin to form pipes from 120 – 220 years old (Wilkes 1982, Mackowski 1987, Gibbons and Lindenmayer 2002, Looby 2007). In *E. regnans,* pipe formation begins at about 120 – 150 years old shortly after the trees attain maximum height (Ambrose 1982). Hence, the beginnings of pipe formation would be expected in trees greater than 100 cm DBH in *E. regnans*. Large pipes are often not found in trees until they are at least 190 years old (Smith and Lindenmayer 1988). Pipes are most numerous in large old trees, with the maximum age for *E. regnans* about 450 – 500 years (Banks 1983), but in other eucalypt species is up to 700 years (Ambrose 1982).

The SFRI contains a sub-set of trees where a detailed suite of tree characteristics related to defect were recorded and models of pipe occurrence and volume were derived. However, the models can be applied only to trees that have data for all the characteristics related to defect. Hence, new models had to be derived from the DSE data for individual trees with pipe occurrence and dimensions of defect in sawlogs, which used only independent variables for which there were data for all trees in the inventory.

Predicting the reduction in wood volume due to pipe was calculated in two stages.

(1) Probability of pipe occurrence in the tree

The SFRI data contains 2627 trees that were assessed for pipe occurrence. A logistic equation was used as a model to predict probability of pipe occurrence, incorporating only the variables available for all trees in the SFRI database. Two groups of species had significantly different coefficients: Ash – *E. regnans, E. delegatensis, E. obliqua, E. nitens, E. cypellocarpa, E. baxteri*; and Other – *E. sieberi, E. viminalis, E. rubida, E. radiata, E.*

paniculata, E. dives, E. dalrympleana. The probability of pipe presence or absence was related to tree DBH, and a factor of location in Forest Management Area contributed to accounting for the variance.

Model: Pipe probability = $(\exp(\alpha + \beta \ln DBH) / (1+ \exp(\alpha + \beta \ln DBH))$ (4)

where, β=1.951 se = 0.129 t probability <0.001, variance accounted = 80.2%

Table 3.6. Estimates for coefficient α in Equation (4) to calculate probability of pipe occurrence,derived from DSE data for the presence or absence of hollows in 2627 trees.

FMA	Hollows = 0		Hollows = 1	
	Ash	Other	Ash	Other
BM	-9.248	-8.136	-8.691	-7.579
C	-8.556	-7.444	-7.999	-6.887
CG	-8.858	-7.746	-8.301	-7.189
D	-8.603	-7.491	-8.046	-6.934
EG	-7.217	-6.105	-6.660	-5.548
NE	-9.252	-8.140	-8.695	-7.583
T	-8.806	-7.694	-8.249	-7.137

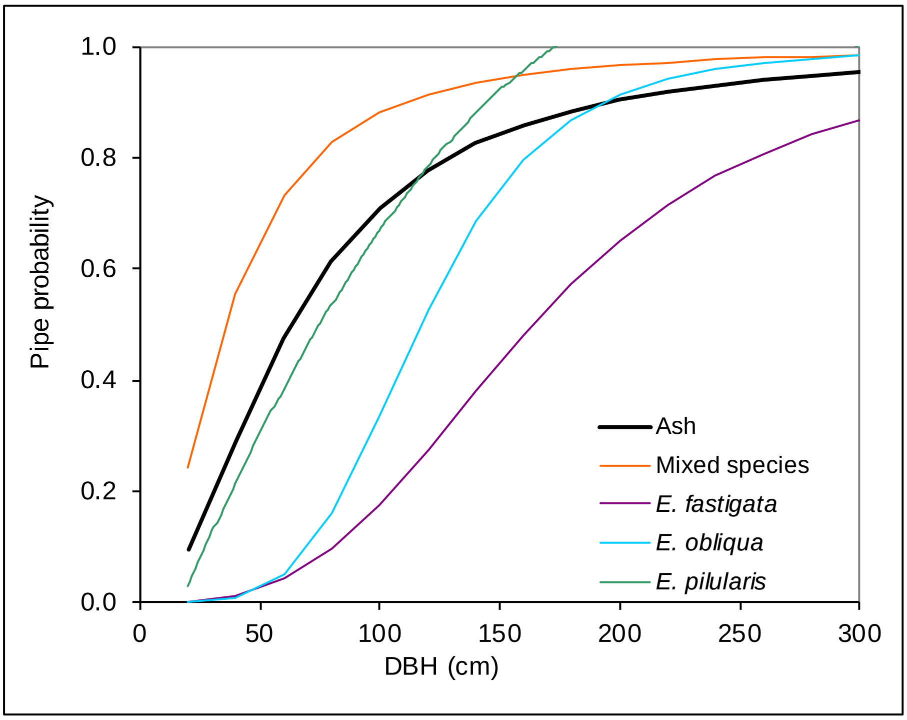
Notes: Forest Management Areas: BM Benalla-Mansfield, C Central, CG Central Gippsland, D Dandenong, EG East Gippsland, NE North East, T Tambo.

The function to predict probability of pipe occurrence is shown in Figure 3.7, and this was compared with other estimates of pipe occurrence from the literature. For three of the species groups (but not *E. fastigata*), very large trees of > 200 cm DBH have a ≥ 90% probability of having pipe (Mackowski 1987, DSE 2007b, Gibbons 2000, 2010). Trees ≥ 100 cm DBH have a much more variable probability depending on the function, and probability changes rapidly with tree size. For ash species there is a 70% probability of pipe occurrence at 100 cm (DSE 2007b). There will be differences in pipe occurrence between species and environmental conditions, but comparison between three independent studies of pipe measurements indicates the likely range in pipe probability related to tree size.

For every tree in the inventory sites, living and dead, this probability function was calculated. If a generated random number was less than the pipe probability, then the tree was assigned a pipe.

Figure 3.7. Probability of pipe occurrence in trees related to their size derived for wet eucalypt forest species.

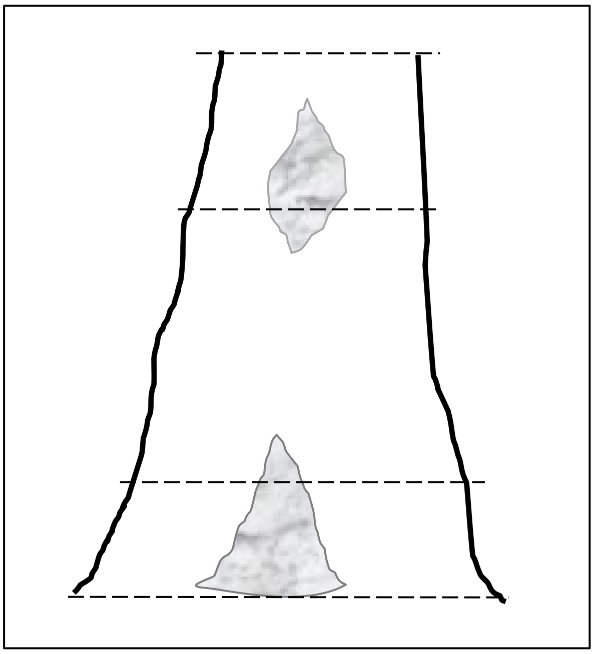
Data sources: Ash and Mixed species (DSE 2007b), *E.fastigata* and *E. obliqua* (Gibbons *et al*. 2000, 2010), *E. pilularis* (Mackowski 1987).



(2) Pipe volume

The SFRI data contains 734 trees with measurements of diameters of internal pipe and the external tree at 1, 2 or 3 heights up the stem, with height 1 usually at 1.3 m. The measured heights were related to sawlog dimensions. An example is shown in Figure 3.8. Data were not available for total tree height. Pipe volume was calculated as truncated cones between pipe diameter measurements, which gave a minimum pipe volume. It is likely that pipe decay extends below 1.3 m to the ground and above height 3. Therefore, pipe volume was extended as a truncated cone between heights 0 and 1.3 m, and as a cone above height 3 by continuing the slope of the lower pipe section but only up to the maximum stem height in the dataset. This extended pipe volume was considered a maximum. For each tree with pipe, minimum and maximum pipe volumes were calculated and related to tree diameter. These data are shown in Figure 3.9.

Figure 3.8. Diagram showing internal defect or pipe in a tree stem (shaded areas) and heights up the stem where diameter measurements were made at the locations where standard sawlogs would be cut (dashed lines).



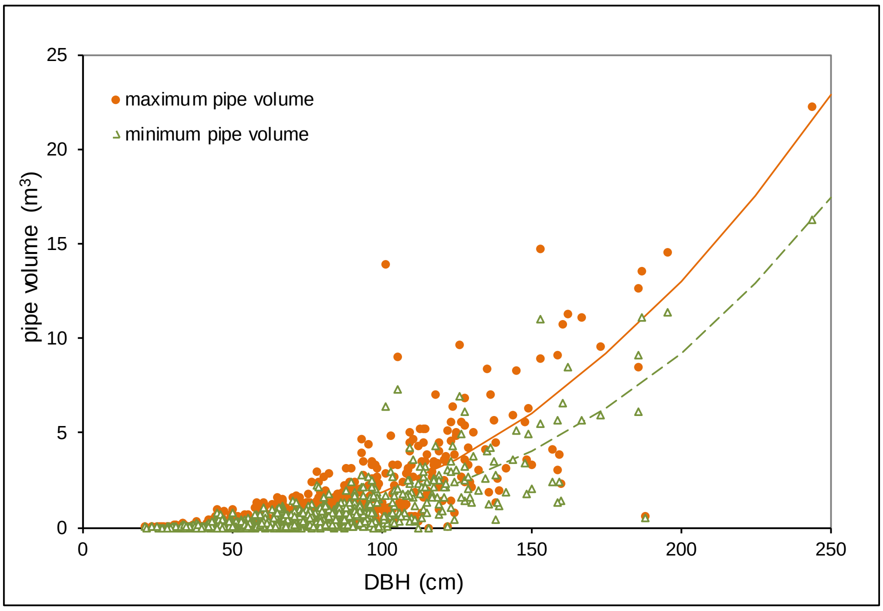
Pipe volume was modelled as a polynomial function of DHB with all species combined. The few large trees in the dataset had a large influence on the regression equation and separating these trees by species groups may cause spurious differences. A range of function forms were tested and the following model provided the best fit to the data and distribution of residuals.

Model: $V_{pipe} = \alpha + \beta_1\ DBH + \beta_2\ DBH^2 + \beta_3\ DBH^3$ (5)

Table 3.7. Coefficients in the pipe volume model (m^3 x 10^{-4}) (F_{pr} < 0.001).

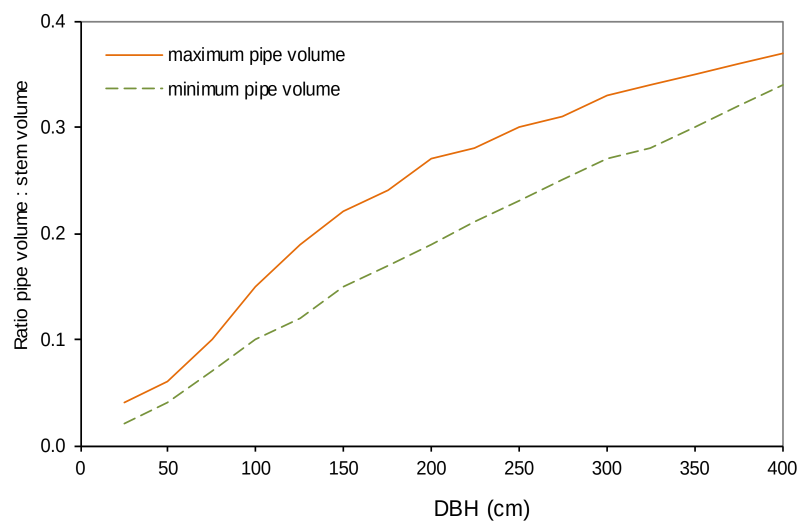
	α	β_1	β_2	β_3	**Variance accounted**
Max. pipe volume					68.0%
coefficient	7076	-319	4.01	0.00326	
SE	4494	149	1.49	0.00447	
Min. pipe volume					67.5%
coefficient	2369	-102	1.13	0.00812	
SE	3107	103	1.03	0.00309	

Figure 3.9. Pipe volume measured for individual trees (n = 734), calculated using minimum and maximum volume estimates, and the relationship with DHB derived from Equation (5).



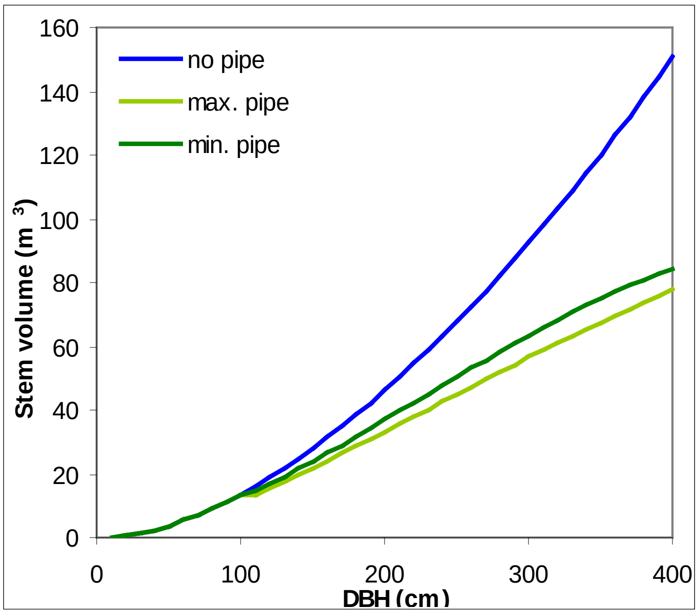
Pipe volume increases as a proportion of stem volume with tree DBH (Figure 10). The predicted proportions of pipe volume using the DSE tree data are in a similar range to that derived by Dean *et al*. (2003) who related the fraction of stem decay to tree age. They estimated the proportion of decay as 0.0002 of the total stem volume at 100 years, 0.015 at 200 years, 0.15 at 300 years and 0.5 at 400 years, which represent similar magnitudes but a more rapid increase in the proportion than in our study.

Figure 3.10. Change in the ratio of pipe volume : stem volume with increasing tree diameter, with the solid line for maximum volume and the dashed line for minimum volume. Values are based on the pipe volume model, Equation (5), and the DSE stem volume model (Table 3.5) across the range of tree sizes in the inventory dataset (20 to 244 cm DBH in pipe tree dataset, and 20 to 400 cm in SFRI inventory dataset).



The reduction in stem volume of individual trees due to piping is shown across a range of sizes based on the minimum and maximum calculated pipe volume (Figure 3.11). Stem volume of individual large trees is greatly reduced due to the volume of piping.

Figure 3.11. Potential reduction in tree volume due to the amount of piping shown as a function of individual tree size. The range of reductions in volumes was based on different assumptions about extrapolating the pipe beyond measurement height.



Stem biomass is the product of volume and wood density. The pipe volume that has been calculated refers to any defect in the wood, which ranges from hollow to various stages of wood decay. The DSE data did not provide information about the proportion of hollow to decayed wood. We have calculated stem biomass using a range of proportions of hollow and decayed wood from 0.5 to 0.8. This is a likely range based on observations of tree hollows and decay at the field sites. Density of decayed wood was 0.2 g cm^{-3}, being the same as decay class 3 for CWD (Section 3.3.2).

Pipe volume was calculated using the minimum and maximum models for every tree in the inventory sites. Biomass of the pipe was calculated using wood densities of 0 g cm^{-3} for hollows and 0.2 g cm^{-3} for decayed wood with proportions of 0.5 to 0.8. The final results of tree biomass were compared and an average used for calculating plot biomass (Table 3.8).

Pipe volume was calculated for living and dead trees, however dead trees often have a lower height for a given diameter because part of the trunk has broken off, and this height was not recorded. Hence, the modelled pipe volume would overestimate the volume of pipe for short dead trees. An average ratio of dead : live height was used to reduce the pipe volume.

Table 3.8. Effect of different assumptions in calculating pipe volume and wood density on tree biomass averaged across all trees and plots in the SRFI dataset.

Pipe model	Average tree biomass (tC ha^{-1})
1. no pipe	313
2. max pipe vol., 50% decayed wood	307
3. max. pipe vol., 80% decayed wood	309
4. min. pipe vol., 50% decayed wood	309
5. min. pipe vol., 80% decayed wood	310

Individual tree biomass can be greatly reduced by internal decay, but as an average across all trees and plots, the reduction is small. The average reduction across all the SFRI plots was 2% and the average across the old growth ANU carbon sites was 9%. The SFRI sites do include trees up to 400 cm DBH but only 5% of trees were greater than 150 cm DBH and hence only a small proportion are likely to be affected by pipe.

Because of the large amount of uncertainty associated with calculation of pipe volume from the data available, likely minimum and maximum volumes were estimated. We compared these estimated pipe volumes with other information in the literature, based on field measurements and structural characteristics of tree trunks, to assess the likely magnitude of the reduction in stem volume due to pipes in *E. regnans*.

Estimates of pipe from studies in different forest types are most readily compared by the ratio of pipe diameter : stem diameter. Pipe diameters in the ash forests calculated from the DSE data in this study have a higher ratio than in *E. pilularis* forest on the NSW north coast (Mackowski 1987) or mixed eucalypt forest in East Gippsland (Gibbons and Lindenmayer 2002). A study in East Gippsland (n = 472 trees) with tree diameter range of 50 – 300 cm revealed a maximum ratio for pipe diameter : stump diameter of 0.6 (Gibbons *et al*. 2000).

Pipe volume calculated from the SFRI field data in the current study was compared with the theoretical structural mechanics of stems to assess whether the field data represent a reasonable magnitude. A study of stem mechanics and bending stresses that explain stem failure in wind concluded that the maximum proportion of stem diameter that could be hollow was 0.7, based on the thickness of remaining stem required for stability (Mattheck *et al.* 1994, 2006). Trees with larger pipes are predicted to collapse to the ground. This proportion was derived both theoretically and from field samples. Failure occurs as shear crack formation followed by cross-sectional flattening and breakage by bending. In the DSE inventory dataset, the ratio of pipe diameter : stem diameter was up to 0.97, and the ratio of pipe : stem cross-sectional areas was up to 0.93. These values suggest that there may be overestimates of pipe diameter as these are living standing trees. However, in the DSE trees measured for pipe, the ratio of pipe volume : total stem volume was $\leq$ 0.7. The range in proportion of pipe volume occurred across the full size range of trees.

3.3.1.4 Biomass carbon

Biomass carbon was calculated for each tree as the product of the wood volume derived from the allometric equation, wood density and carbon concentration of the wood.

Predictions from allometric equations give aboveground biomass. There have been no measurements of root mass in *E. regnans* forests, hence an average root : shoot ratio for eucalypt forests of 0.25 (Raison *et al.* 2001) was used to convert to total tree biomass for living and dead trees.

We used a stem wood density value of 0.520 g cm^{-3}, which was derived as an average of the data for *E. regnans* reported in the literature (Ilic *et al.* 2000, Bootle 2005, Chafe 1985, Mackensen and Bauhus 1999). Branchwood density was measured by Sillett *et al*. (2010), with a mean of 0.677 g cm^{-3}.

Carbon concentration of 0.5 gC g^{-1} was used for all biomass components. Biomass components vary from 0.45 to 0.55 gC g^{-1} and 0.507 gC g^{-1} was the average for heartwood in a range of native species (Gifford 2000) and 0.492 gC g^{-1} for stemwood in *E. delegatensis* (Keith *et al*. 2009a).

The SFRI inventory data included only trees > 20 cm diameter. Biomass of the component < 20 cm of small trees and shrubs was added to the total biomass carbon budget using an average value calculated from the ANU carbon sites where all diameter sizes of plants were measured.

3.3.1.5 Decorticating bark

Decorticating bark is the exfoliated gum bark from the upper stem and large branches of *E. regnans* (Ashton 1975b). This bark peels off, some falls to the ground but some is caught in branch forks and hangs from the tree. The amount of decorticating bark is highly variable but is generally greater in larger trees (Lindenmayer *et al*. 2000c). This is because there is a larger surface area of stem and branch bark and also more perpendicular limbs producing forks. The biomass of decorticating bark is used in Section 5 on carbon content of biomass components combusted by fire.

Biomass of the decorticating bark was estimated in relation to tree DBH based on a non-destructive sampling method (Andrew *et al*. 1979). A ‘standard clump’ of bark was collected and weighed (3.0 kg fresh mass, mean of 8 samples). Trees across a range of DBH size classes (n = 36) were assessed for their number of ‘standard clumps’ of decorticating bark, based on length and number of the strips of bark. Subsamples of bark were weighed fresh and oven-dried to determine a fresh : dry weight ratio of 0.63, which was applied to the fresh weight estimates for each tree. An average carbon concentration for bark of 0.5 g g^{-1} (Gifford 2000) was used to calculate carbon content of the mass of bark. A relationship was derived between tree DBH (D in cm) and dry mass of decorticating bark (M_{rb} in kg).

$$M_{rb} = 0.00013\ D^{2.37}, \quad r^2 = 0.85, \quad \text{for } D = 0 \text{ to } 150 \text{ cm} \qquad (6)$$

Bark mass increases with tree size up to a DBH of about 150 cm, after which the rate of increase in bark mass is highly variable (Figure 3.12). Equation (6) can be used to estimate decorticating bark mass of trees up to 150 cm and this is set as a maximum bark mass for larger trees. Hence, bark mass calculated for large trees may be an underestimate. Bark mass was calculated for all the small trees in the 10m x 10m plots and the large trees in the 0.51 ha transect plots within each site, and then averaged for living trees in each stand age class (Table 3.9).

Figure 3.12. Dry mass of decorticating bark on *E. regnans* in relation to tree size. The relationship in Equation (6) is derived for trees up to 150 cm DBH (solid diamonds). Larger trees do not appear to continue increasing bark biomass (open diamonds).

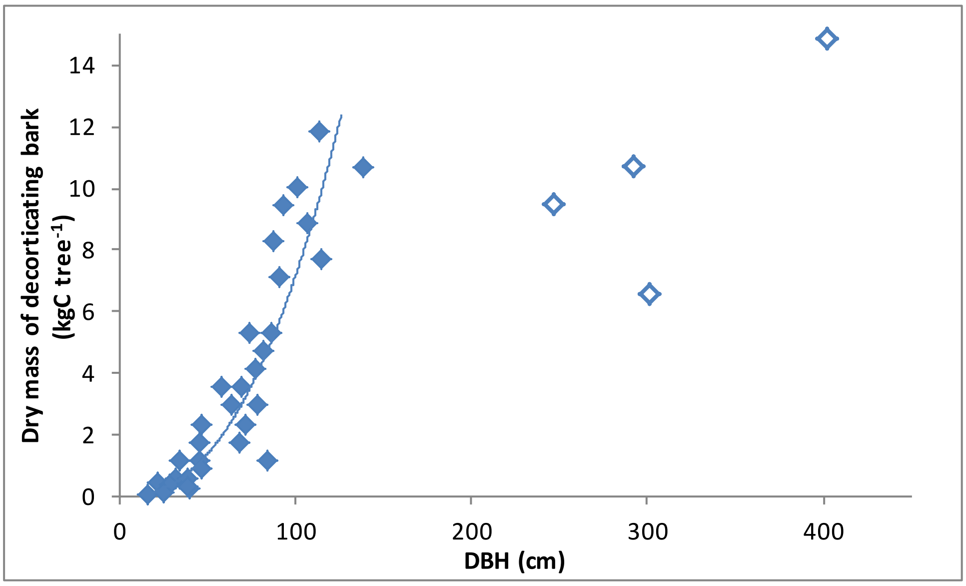


Table 3.9. Average carbon content (tC ha^{-1}) in the mass of decorticating bark on living eucalypt trees in three age classes.

Forest age class	Mean carbon content (tC ha^{-1})	Standard error (tC ha^{-1})
Young regrowth	0.201	0.025
Mature regrowth	0.278	0.047
Old-growth	0.702	0.117

3.3.1.6 Rough bark

Rough bark is the fibrous bark in the lower section of the trunk, often referred to as the sock. The amount of rough bark combusted by fires of different severities was estimated from measurements of bark thickness and density. Bark thickness was measured on trees (n = 3 samples per tree) in each of the three fire categories (n = 30 trees per category) across the range of tree sizes and age categories. The relationship between bark thickness and DBH (Figure 3.13 and Table 3.10) differs due to fire severity, with the burnt trees having reduced bark thickness compared with unburnt trees, particularly large trees greater than 200 cm.

Figure 3.13. Relationship between measured bark thickness and tree diameter from trees in each fire severity category (n = 30 trees per category). The regression equations are given in Table 3.10 as Equation (7).

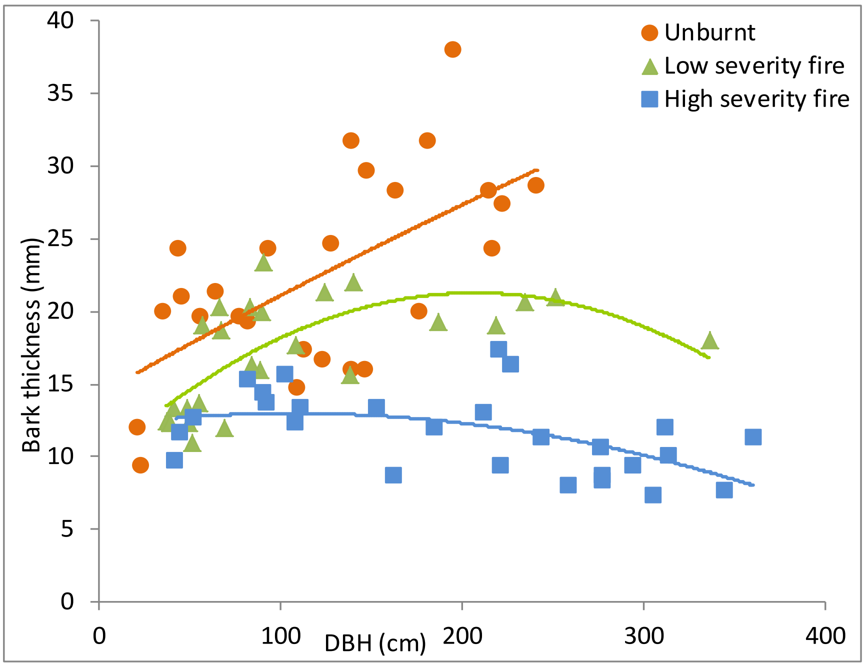


Table 3.10. Equations relating bark thickness (B_t in mm) with tree diameter (D in cm) for different fire categories.

Fire category	Equation (7)	r^2
Unburnt	B_t = -0.00003 x D^2 + 0.0708 x D + 14.314	0.39
Low severity fire	B_t = -0.0003 x D^2 + 0,1118 x D + 9.7171	0.47
High severity fire	B_t = 0.00008 x D^2 + 0.0166 x D + 12.107	0.28

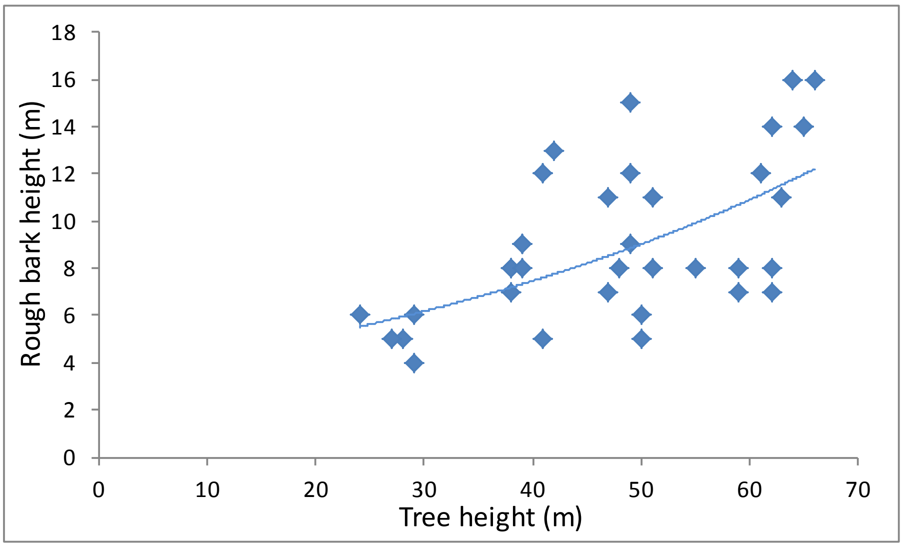
Samples of rough bark were taken from unburnt trees to measure bulk density of the inner green bark and outer fibrous bark. Average thicknesses were 21 mm for inner green bark and 12 mm for outer fibrous bark, with bulk densities given in Table 3.11. Loss of bark biomass from burnt trees in the low and high severity fire categories was calculated as the reduction in bark thickness related to tree DBH, multiplied by the bulk density of fibrous bark for the first 12 mm and then green bark for the remaining loss in thickness. Some severely burnt large trees had the entire bark thickness dried, cracked and peeled off after the fire, indicating that the green inner bark had been partly combusted.

Height of rough bark was calculated for all trees using a relationship between measured total tree height and height of the rough bark (n = 38) (Figure 3.14).

$H_r = 3.5074 \times e^{0.0189} \times H_t,\ \ r^2 = 0.39$ (8)

The proportion of the stem of *E. regnans* with rough bark is highly variable and the relationship in Figure 3.14 provides an approximation for calculating average bark biomass at the site scale but is not accurate for making predictions of individual trees. Surface area of rough bark was calculated for the stem as a truncated cone using tree diameter, height of rough bark and slant height. Volume of bark was calculated using bark thickness. Reduction in bark thickness due to fire of low and high severity was calculated as the difference in thickness between unburnt and burnt for each tree. Loss in bark mass was calculated for all eucalypt trees at the sites, depending on their fire category, using the surface area of rough bark, bark thickness loss and bulk density of green and fibrous bark. Results of bark biomass combusted are given in Section 5.

Figure 3.14. Relationship between height of rough bark up the stem (H_r) and total tree height (H_t). Regression line described by Equation (8).



3.3.2 Coarse Woody Debris

Coarse woody debris (CWD) consists of all woody material ≥ 25 mm diameter on the ground, including fallen logs, branches, bark, stumps < 2 m height, woody material under the litter layer, pieces of charcoal, standing dead trees at an angle < 45̊ to the vertical, and elevated material at any angle if one end touches the ground.

CWD was sampled at the ANU carbon sites in the current study, and existing data was used from the SRFI sites.

(1) ANU carbon sites.

CWD was sampled at the 54 sites in the current study using the line intersect method (van Wagner 1968, McKenzie *et al*. 2001) where every piece of CWD that crossed the transect line was measured. Two size categories were used:

(i) logs < 60 cm diameter were measured along 10m transects extending from two sides of each of the three plots (total length of 60m) (Figure 3.2).

(ii) logs > 60 cm diameter were measured along the 100m central and perpendicular transects (total length of 200m).

The size of each piece of CWD was measured as either a diameter or a length and width. If the log was hollow or had gaps along its length, the proportion missing was estimated. Each piece of CWD was assessed for degree of decay and assigned to the following categories: (1) sound, (2) outer layer showing signs of decay, (3) signs of decay extend to heartwood. The proportion of charcoal on burnt logs was estimated. A total of 4121 pieces of CWD was measured.

The volume of logs was calculated as:

$$V = (\pi^2 \ \Sigma d^2) / 8 L \tag{9}$$

where, V is volume (m^3 m^{-2}), d is log diameter (m), L is length of transect (m) (van Wagner 1968). Most logs were cylindrical and a diameter was measured. For logs with a rectangular cross-section, length and width were measured, and volume was calculated as an ellipse then multiplied by a correction factor for a rectangle. For all logs, total volume was reduced if there was a proportion of the log missing.

Samples of CWD (n = 20) were taken from each of the three decay classes and across the range of diameters to measure bulk density. Discs or sections of logs were cut that sampled the complete cross-section of the log representatively. The volume of sections was measured by water displacement and then dry weight measured to determine bulk density of wood (Table 3.11). Mass of CWD was calculated as the product of volume and bulk density for each wood decay class. Mass of charcoal was calculated for the individual pieces of charcoal and the proportion on burnt logs that consisted of charcoal.

Table 3.11: Bulk density (g cm^{-3}) of biomass components (mean ±standard error).

Coarse Woody Debris			Charcoal	Rough Bark	
Decay 1	Decay 2	Decay 3		Green	Fibrous
0.48 ±0.02	0.28 ±0.02	0.20 ±0.01	0.35 ±0.06	0.34 ±0.02	0.14 ±0.02

An estimation of the CWD biomass combusted at the ANU carbon sites subjected to low and high intensity fire was derived by identifying where logs had been before the fire by the line of charcoal remaining on the ground. CWD pieces that consisted of more than half charcoal and intersected the transect, provided an indication of a charred log line. It was assumed that each line of charcoal represented a log that was combusted. The initial carbon stock of these logs was calculated as representing the mean size of logs on each site. These results are used in Section 5 to determine the amount of biomass carbon combusted.

(2) SFRI sites.

CWD was sampled at 714 sites in montane ash forest with data collected along 3 x 30 m transects per site. Logs greater than 15 cm diameter were recorded and the total volume for the site was calculated. Volume was multiplied by an average tree wood density (*E. regnans* 0.51 g cm^{-3}, *E. delegatensis* 0.54 g cm^{-3}) to give CWD biomass. If a CWD sample was classified as rotten, then the density was reduced by 0.62 (average density of rotten wood derived by Mackensen and Bauhus 1999). The mean value of CWD across the SFRI sites in ash forest was 48 tC ha^{-1}, with a maximum of 292 tC ha^{-1}. SFRI sites in ash forests that did not have data for CWD were assigned an average value for a given dominant tree species and age category.

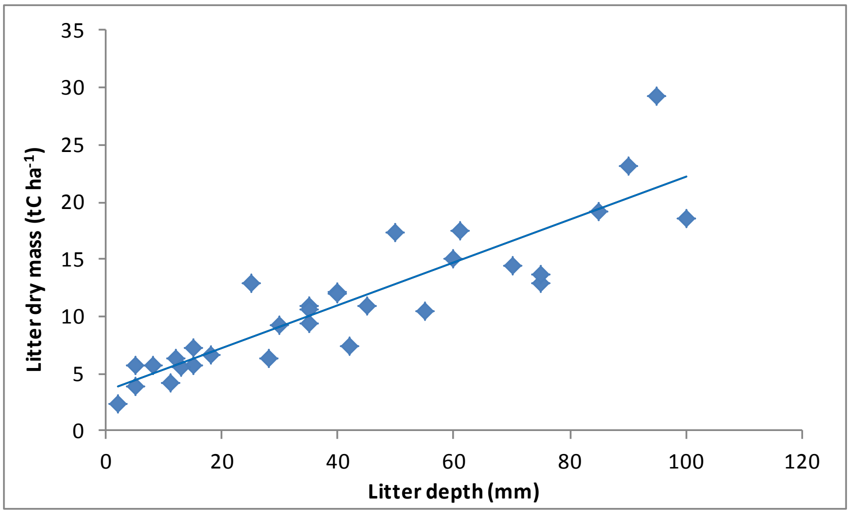
3.3.3 Litter layer

The litter layer consists of organic material less than 25 mm diameter including leaves, twigs, insect detritus, animal scats, and comminuted material that is recognisable as organic material (as distinct from the humus layer of the soil that consists of undifferentiated granular organic material). At the ANU carbon sites, litter depth was sampled at five points along the transects of each 10 m x 10 m plot (25 points per site). The litter layer was compressed manually to compact loose material before measuring depth.

Samples of litter (n = 30) were taken in 0.5 m x 0.5 m quadrats across a selection of the ANU carbon sites and a range of litter depths (2 – 100 mm). Depth of the compressed litter was averaged over the quadrat and related to the dry weight of litter. The boundary between the litter layer and soil is difficult to define in the organic soils of montane ash forests. Hence, after the litter had been removed from the ground, the fragmented material on the soil surface was collected and sieved to provide a consistent differentiation between soil and litter. The following relationship between litter mass (M_L) (tC ha^{-1}) and litter depth (L_d) (cm) was derived and applied to the average litter depths at all the sites.

$M_L = 0.188\ L_d + 3.44$, $r^2 = 0.80$, (n = 30 samples) (10)

Figure 3.13. Relationship between depth of the litter layer (compressed) and biomass. Regression line derived in Equation (10).



At the SFRI inventory sites, there were no data for litter and so an average value based on stand age classes derived from the ANU carbon sites was added to the total biomass carbon stock at each site.

3.3.4 Soil carbon

Data for soil carbon concentration, depth and bulk density in A and B horizons of the mineral soil were obtained from the Australian soil resources map (Australian Soil Resource Information System, McKenzie *et al.* 2005) as a spatial GIS layer for the Central Highlands region.

Table 3.12. Biomass Carbon Stock (tC ha^{-1})

Site number	Live Biomass	Dead Biomass	Litter	CWD	Total Biomass Carbon
Young regrowth - Unburnt					
718	340	26	5	46	417
719	319	17	10	81	426
733	163	15	8	113	299
734	274	40	6	49	368
801	456	45	6	35	542
804	206	62	5	105	378
mean	**293**	**34**	**7**	**71**	**405**
SE	**43**	**8**	**1**	**14**	**33**
Young regrowth - low severity fire					
22	542 *	33	4	42	622
25	252	71	4	72	398
406	266	74	4	76	420
800	296	104	4	44	448
816	378	32	5	87	502
HS08	324	35	4	33	396
mean	**303**	**63**	**4**	**62**	**433**
SE	**22**	**13**	**0**	**9**	**35**
Young regrowth - High severity fire					
802	1	388	4	89	482
803	81	184	4	22	290
806	0	269	4	27	300
HS07	0	138	4	31	172
HS09	8	145	4	15	172
HS10	125	146	4	23	297
mean	**36**	**212**	**4**	**34**	**286**
SE	**22**	**40**	**0**	**11**	**46**
1939 regrowth - Unburnt					
101	542	10	5	29	587
110	217	184	11	95	507
126	335	83	8	26	452
707	325	26	9	79	438
708	517	181	12	186	897
726	618	49	9	60	736
mean	**426**	**89**	**9**	**79**	**603**
SE	**64**	**31**	**1**	**24**	**74**
1939 regrowth - Low severity fire					
1	412	48	8	134	603
712	141	125	6	55	327
728	257	62	4	87	411
744	719	70	6	52	847
745	211	143	5	88	447
749	160	19	4	35	218
mean	**317**	**78**	**6**	**75**	**475**
SE	**90**	**19**	**1**	**15**	**91**
1939 regrowth - High severity fire					
5	16	600	4	87	707
340	9	519	4	68	600
535	3	182	4	64	252
HS03	0	229	4	44	277
HS05	15	630	4	87	735
HS06	2	133	4	43	182
mean	**7**	**382**	**4**	**65**	**459**
SE	**3**	**92**	**0**	**8**	**102**

Old growth - Unburnt					
402	837	28	6	46	917
808	284	29	7	21	341
812	894	19	7	81	1001
813	930	3	7	55	995
814	1081	3	8	36	1129
815	907	165	11	67	1150
mean	**822**	**41**	**8**	**51**	**922**
SE	**113**	**25**	**1**	**9**	**122**
Oldgrowth - Low severity fire					
457	198	136	4	46	384
472	389	120	5	102	615
489	257	70	4	31	361
491	834	20	6	40	899
751	465	107	5	133	710
811	285	93	6	30	414
mean	**405**	**91**	**5**	**63**	**564**
SE	**94**	**17**	**0**	**18**	**88**
Oldgrowth - High severity fire					
2	77	454	4	90	624
470	43	476	3	80	602
471	319	282	4	111	715
810	51	269	4	56	381
HS01	16	321	4	98	438
HS02	0	742	4	30	775
mean	**84**	**424**	**4**	**77**	**589**
SE	**48**	**73**	**0**	**12**	**63**

Notes: * site value was not used in the mean because the site was not characteristic of the tree size structure for the age category.

Litter was measured some time after the fire when scorched leaves had fallen. Hence the litter layer biomass in Table 3.12 does not reflect the amount combusted (see Section 5.2.1).

3.4 Calculation of carbon stocks

The carbon stock in each ecosystem component is summarised in Table 3.12 for each of the 54 ANU carbon sites within the age / fire categories, and a mean and standard error is provided for each category.

Trees contain the largest stock of biomass carbon in the ecosystem with an average proportion of 0.85. Variability in the proportion of biomass in trees is greatest in the young regrowth sites (0.59 – 0.93) and less in the old-growth sites (0.77 – 0.96), which reflects the high variability in CWD remaining after clearfelling at the young regrowth sites. Accounting for both living and dead trees is important in the calculation of ecosystem carbon stocks. Dead trees are not always measured in forest inventories. The proportion of living and dead trees varies with age distribution and disturbance history. The stands unburnt in 2009 had an average of 0.79 to 0.96 of their standing tree biomass in living trees. Living trees comprised 0.80 to 0.82 of tree biomass in stands that had a low severity fire in 2009, which indicated that only a small proportion of trees were killed by the fire. Living trees comprised only 0.02 to 0.14 of tree biomass in stands that had a high severity fire in 2009. All trees were killed in some sites but in other sites that were classified as a high severity fire, there were a few trees that survived, mostly eucalypt species other than *E. regnans*.

The litter values included in the carbon budget are those measured approximately 18 – 24 months after the 2009 wildfire and hence represented the fall of scorched leaves in burnt sites. Litter biomass carbon stock measured in the unburnt sites of 7 tC ha^{-1} in young *E. regnans* regrowth is similar to values reported previously for young regrowth stands (6 tC ha^{-1} Ashton 1976, 7 tC ha^{-1} in Polglase and Attiwill 1992), although Feller (1980) reported 24 tC ha^{-1}. The average carbon stock measured in the current study of 9 tC ha^{-1} in the mature regrowth unburnt sites and 8 tC ha^{-1} in the old-growth unburnt sites is less than that reported previously (10.5 tC ha^{-1} in Ashton 2000, 12 tC ha^{-1} in Polglase and Attiwill 1992). It is estimated that the litter layer was essentially completely combusted in the sites with low and high severity fire (see Section 5). The litter layer rapidly accumulated after the fire with fallen scorched leaves, twigs, fruit and bark (Lindenmayer *et al.* 2010).

The litter layer is a small proportion of the total ecosystem carbon stock as decomposition rates are relatively fast under conditions of high rainfall and reasonably warm temperatures, as well as the decomposition activity of soil fauna such as earthworms, amphipods and crustaceans, and the pedoturbation activity of lyrebirds (Ashton and Bassett 1997). Litter is decomposed within about 4 years in old-growth stands (Polglase and Attiwill 1992) and the leaf component decays within 12 -18 months (Ashton 1975a).

The mean carbon stock in CWD from unburnt forest of 67 tC ha^{-1} is a value between an average reported for wet sclerophyll forests of 55 tC ha^{-1} (n=28, SD=242) (Woldendorp and Keenan 2005), and that from a previous study in the ash forests of the Central Highlands (Lindenmayer *et al*. 1999) of 87 tC ha^{-1}.

The size structure of forest stands is illustrated by the frequency distribution of tree diameter classes (Figure 3.15) and the contribution of biomass carbon of each size class (Figure 3.16). These graphs show that the nominated age of stands based on the disturbance history does not correspond to the size of all trees at the site. Both the 1983 and 1939 regrowth stands have trees greater than 100 cm diameter which must represent residual trees that remained after the logging or fire disturbance event. Hence, the carbon stocks in these forest stands do not represent the accumulated carbon over a given time since a disturbance event, but rather an average carbon stock for a stand with a given history of disturbance that involves regeneration and survival of trees.

Figure 3.15. Frequency size distribution of all woody stems > 2 m height (living and dead) in each age / fire category (n = 6 sites per category). Data for frequency of stems were derived from the three 1m x 1m plots for stems <100 cm DBH (0.03 ha) and the two 100m x 30m transects for stems > 100cm DBH (0.051 ha) in the 1 ha sites.

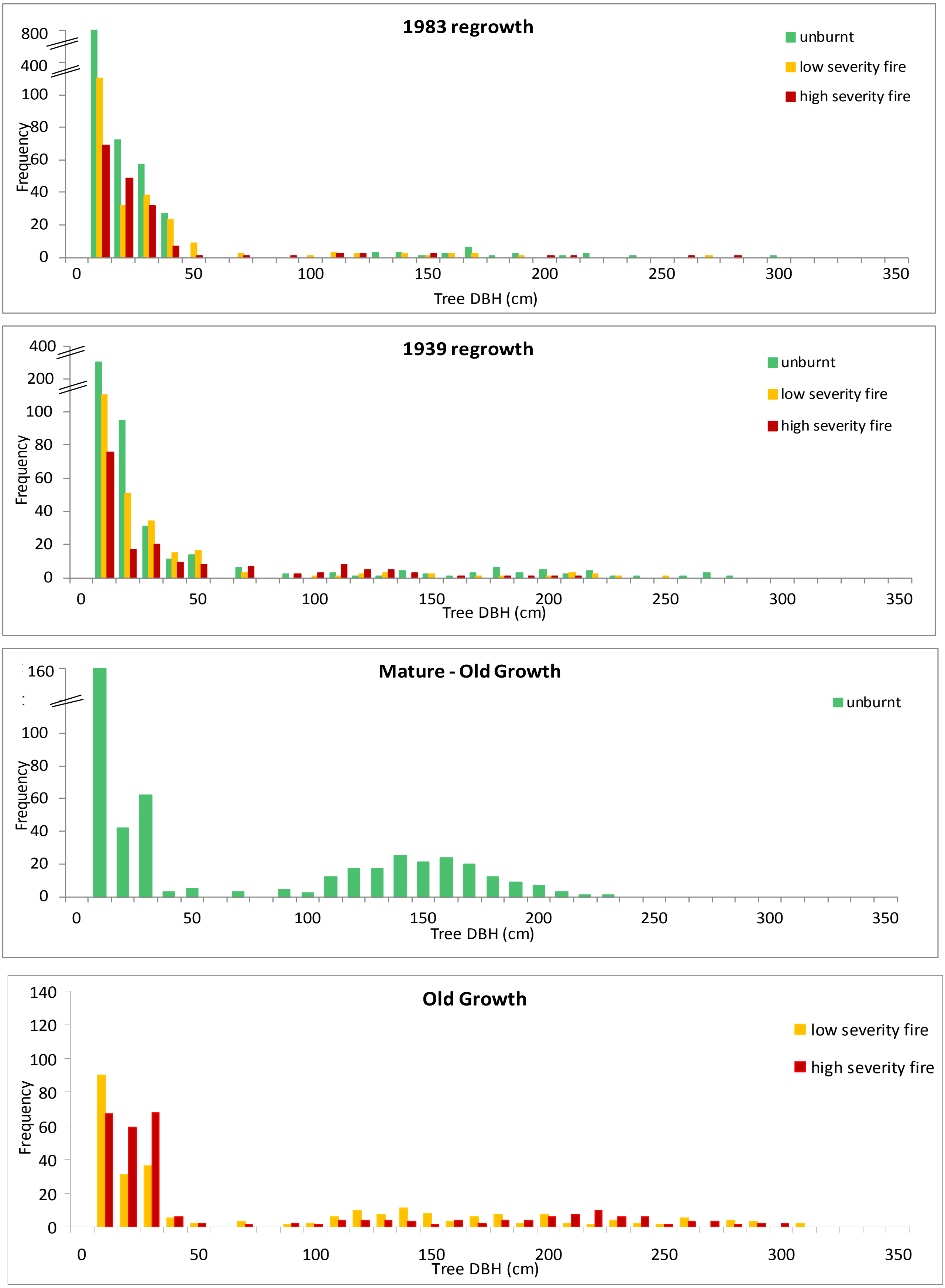
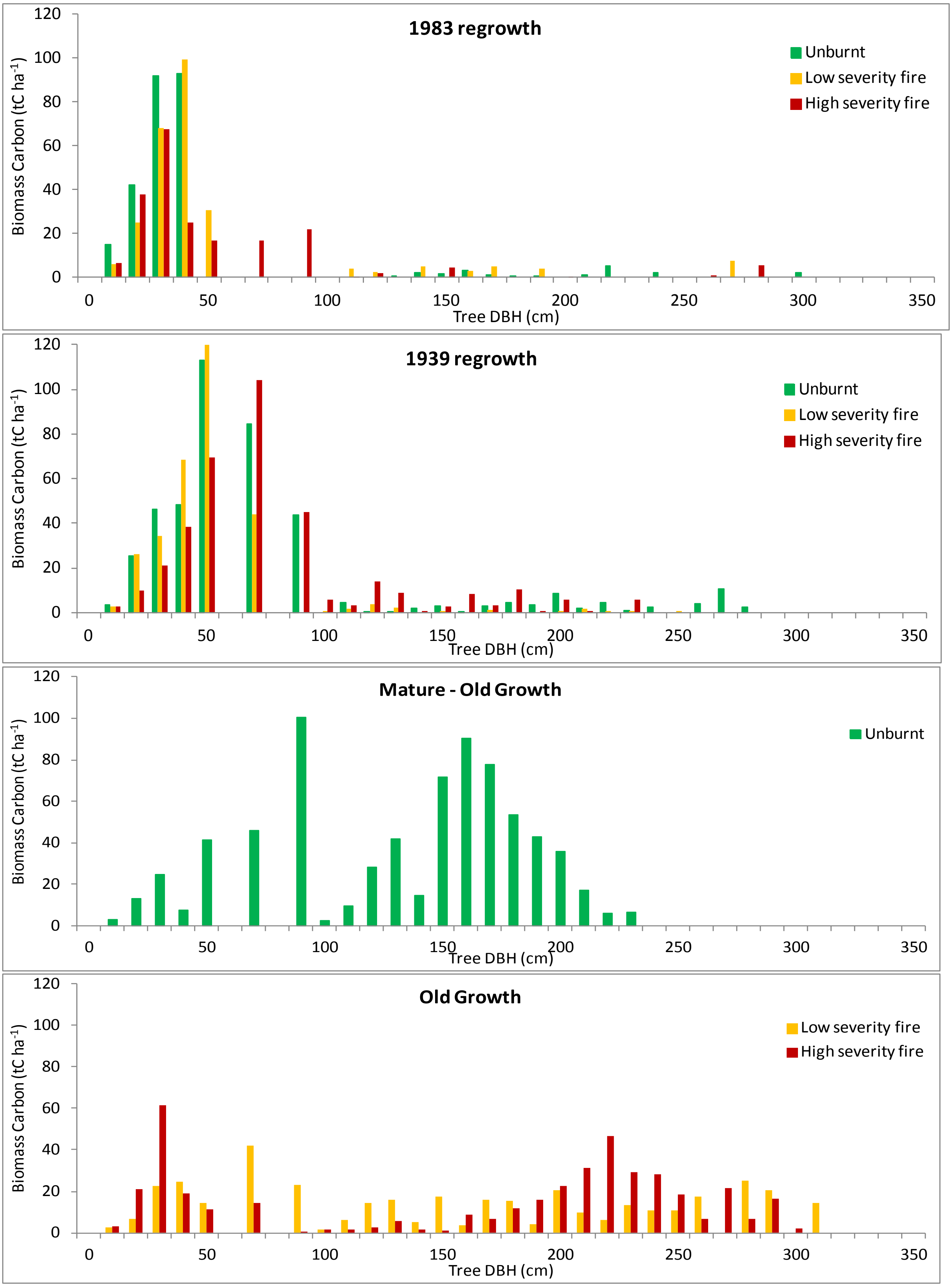
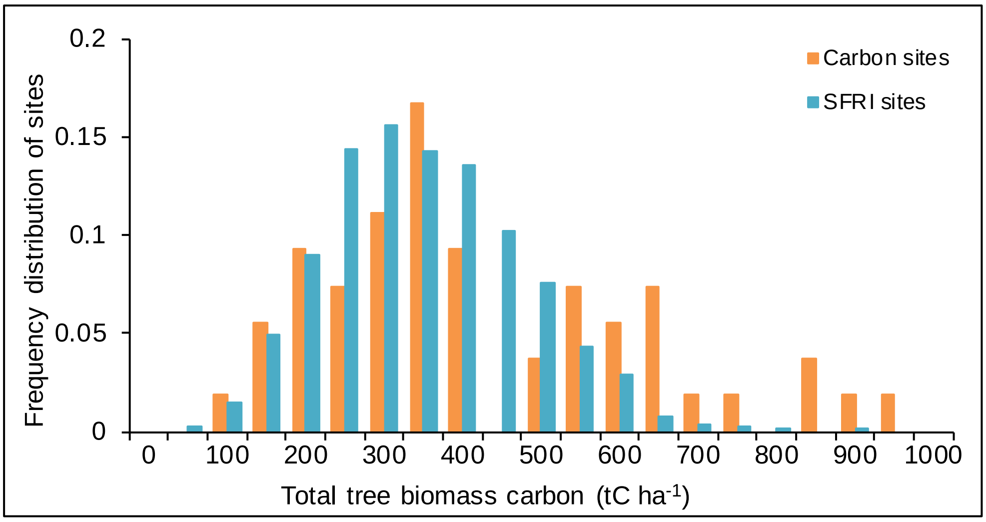


Figure 3.16. Biomass contribution per size class of woody stems > 2 m height (living and dead) in each age / fire category (n = 6 sites per category). Data for carbon stock in trees were derived from the three 1m x 1m plots for stems <100 cm DBH (0.03 ha) and the two 100m x 30m transects for stems > 100cm DBH (0.051 ha) in the 1 ha sites.



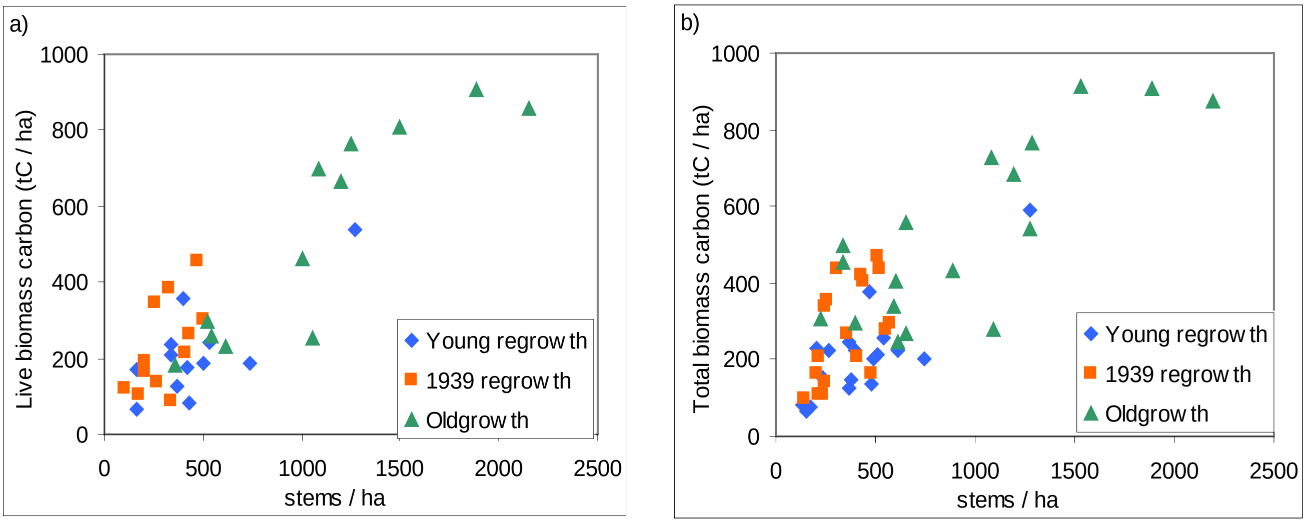
Biomass density at a site scale (1 ha) varied greatly across our study region even within the montane ash forest type. Two sets of sites (ANU carbon sites and SFRI sites) were used as calibration data for the carbon models making it important to assess their relative distributions and comparability (Figure 3.17). The distribution of biomass carbon density is similar distribution between the two sets of sites but there are differences that reflect the location of the sites. The ANU carbon sites were selected to cover a range of ages from young regrowth to old growth but there were probably insufficient sites to provide a normal distribution. It is noteworthy that there is a higher proportion of sites with high biomass density in the old growth carbon sites than in the SFRI sites that are restricted to State Forest. The maximum total biomass in the SRFI sites is 855 tC ha^{-1} compared with 933 tC ha^{-1} in the ANU carbon sites, and less than 0.5% of SFRI sites have biomass greater than 700 tC ha^{-1}.

Figure 3.17. Frequency distribution of total biomass carbon across the ANU carbon sites (n=54) and SFRI sites (n=876).



Forest structure is the main determinant of biomass carbon density because it reflects the size of trees. This is illustrated by the positive relationship between carbon density and tree density (for trees greater than 20 cm DBH) across the age classes of the forest (Figure 3.18). High biomass density occurred in multi-aged old growth forests where a large number of trees from a range of size classes occur, including large old trees, and different species occupy different size / height categories. If all small woody plants less than 20 cm in diameter were included in the stem density count, then the young stands would have much higher stem densities with little addition in biomass.

Figure 3.18. Relationship between site biomass carbon stock and density of woody stems (for stems greater than 20 cm DBH and 2 m height) for the three age classes. (a) Living tree biomass from unburnt and low severity fire treatments, (b) Living + dead tree biomass from unburnt, low and high severity fire treatments. Data for biomass carbon stock in trees were derived from the three 1m x 1m plots for stems <100 cm DBH (0.03 ha) and the two 100m x 30m transects for stems > 100cm DBH (0.051 ha) in the 1 ha sites.



3.5 Synopsis

Calculation of carbon stocks in the current study has attempted to be conservative by applying averages of different functions and likely minimum and maximum values. Significant differences in calculated carbon stocks resulted from use of different functions. Thus, synthesising and testing these functions was informative for identifying reasons for differences in estimates of carbon stocks by different studies.

The allometric equation for *E. regnans* derived by Sillett *et al*. (2010) represented a significant advance in the study of this forest type. However, further sampling of *E. regnans* across a range of environmental conditions and tree forms is needed to test the generality of this equation that used trees from one site. There is currently no information about root biomass in *E. regnans* forests. Allometric equations and information about the biomass of non-eucalypt species, particularly rainforest species, is lacking. The models to predict pipe probability and pipe volume derived in the current study that can be applied to all inventory data represented a significant improvement in our capacity to account for stem decay in the calculation of tree biomass. However, there is still a need for more field data about the amount of pipe and the proportion of hollow and decayed wood.

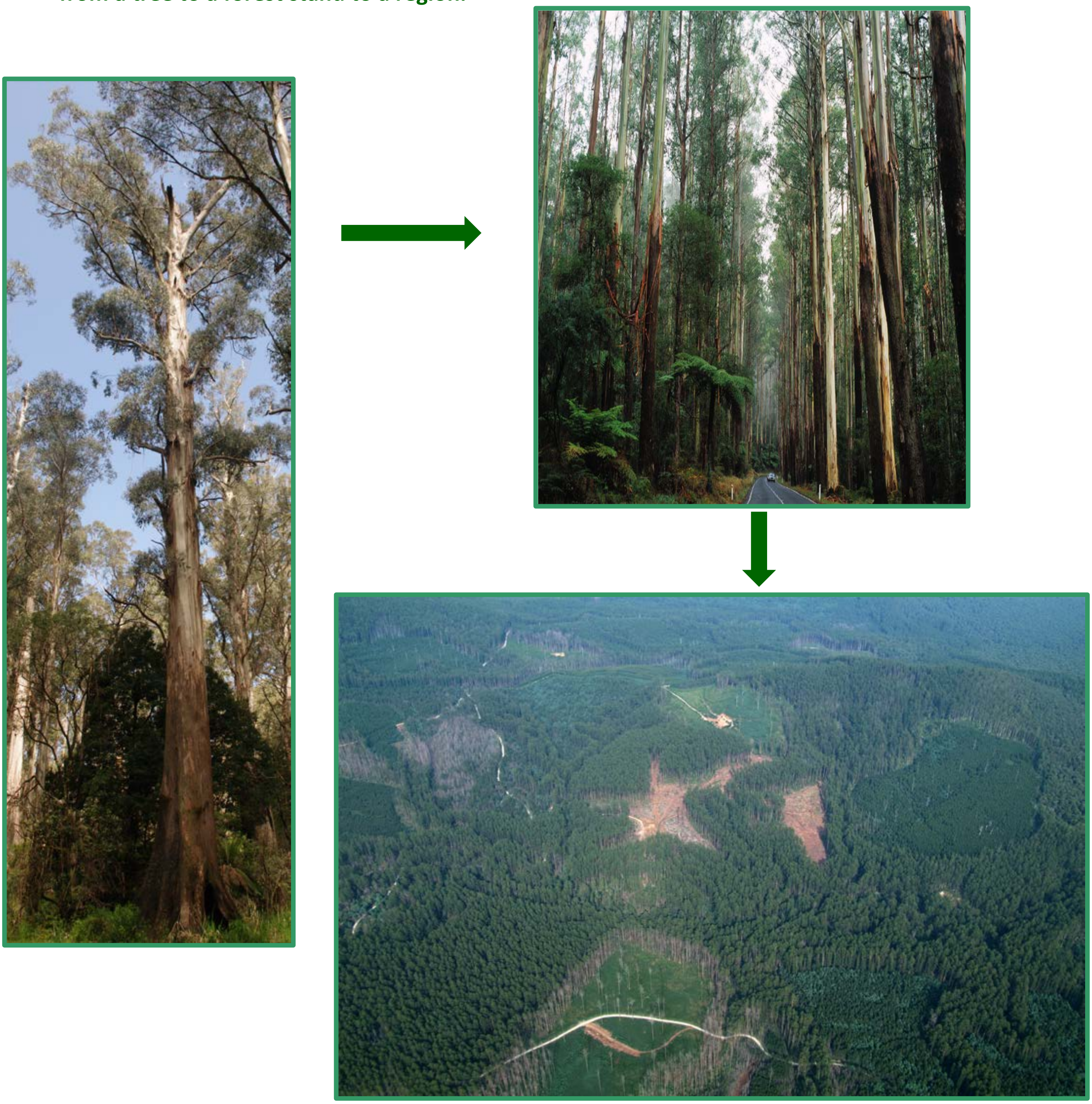
The variability in carbon stocks among sites, which is evident within age / fire categories, is indicative of the heterogeneous nature of the landscape and forest cover. Forest structure at a site can be influenced by factors that are not explained by the spatial environmental data for the site, particularly disturbance factors that are not recorded in historical data. Characteristics of natural and human disturbances are difficult to quantify spatially, for example their magnitude, exact boundaries, and impact on the vegetation. Hence, differences in carbon stocks may not be explained adequately by the available disturbance information.

4. Current Carbon Stock at the Landscape Scale

4.1 Introduction

The next step in the analytical process, following on from the results from figure 3.3, was to use the site-based estimates to generate a map of forest carbon stocks across the landscape within the study region. Spatial modelling of carbon stocks requires spatially-explicit information about environmental variables across the landscape that are related to forest biomass, forest type, and the disturbance history that influences forest age. Disturbance history includes both fires and logging which have complex impacts on forest age structure.

Measurements of biomass are upscaled from a tree to a forest stand to a region.



4.2 Upscaling process

The term 'upscaling' is used to describe the technical procedure of extrapolating values estimated at a network of sites to the landscape level. The upscaling method employed here involved developing an empirical model based on correlating the estimates of carbon stocks at the network of sites (i.e. the ANU carbon sites and SFRI sites) with explanatory variables, that is, variables that statistically explain or account for the spatial variation in carbon stocks. Explanatory variables were used that had GIS layers covering the study region, so that a statistical correlation function could be used to generate estimates of the modelled values across the landscape. The resolution of these modelled landscape-wide values is a function of the scale at which the explanatory variables are mapped and stored in the GIS. Potential explanatory variables include physical environmental conditions, forest types and disturbance history.

In the framework defined in Figure 1.1, upscaling represents the process of using the site biomass data (site data – tree scale), together with the environmental variables and disturbance history (past – spatial data) in a model to produce the Current Carbon Stock (present – regional scale). Landscape scale is defined here by the areal extent, the boundary of the area, and spatial resolution. In our study, the areal extent was approximately 140,000 ha in the Central Highlands of Victoria, with a boundary defined by the water catchments and montane ash forest ecosystem type. The spatial resolution for estimation was 250 m based on the remote sensing imagery used.

4.3 Environmental variables used in upscaling

Environmental variables are used in two stages of upscaling: (i) model development to relate to carbon stocks at each site, and (ii) estimating carbon stocks for every grid cell across the landscape. Spatial layers for the following environmental variables were collated, including variables estimated directly and those derived from other sources: precipitation; water availability index; temperature (minimum, mean, maximum); radiation; elevation; aspect; slope; soil organic carbon content; gross primary productivity; topographic position; geology; forest type; and forest management area. A water availability index (W) was calculated from precipitation and radiation to provide an ecologically meaningful expression of the interaction between precipitation and evaporation in relation to vegetation productivity. The W value was adjusted by a constant, equivalent to the maximum water availability, to provide a positive number (W_{pos}) for calculation of the logarithm used in multiple regression modelling. A summary of these spatial layers and their sources of data is given in Table 4.1 and maps of each variable are provided in the Appendix. Spatial data were standardised to a resolution of 250m. Data for these environmental variables were extracted for all sites (54 carbon sites and 876 SFRI sites).

Table 4.1. Description of spatial data layers used for model development and upscaling of carbon stock estimates across the landscape.

Symbol	Layer description	Spatial resolution	Source
Climate			
P	Annual precipitation, mm yr^{-1}	250m	ANUCLIM; Hutchinson (2005)
Tmin	Minimum monthly near surface air temperature, ^{o}C of the coldest month	250m	ANUCLIM; Hutchinson (2005)
Tmean	Mean monthly near surface air temperature ^{o}C	250m	ANUCLIM; Hutchinson (2005)
W	$W = P - \frac{Q_s}{\rho L}$ mm yr^{-1} where ρ is the density of liquid water (~1000kg m^{-3}) and L is the latent heat of vaporization of water (~2.45 × 10^6 J kg^{-1} H_2O) W is a negative number	1km	Berry and Roderick (2002)
W_{pos}	$W_{pos} = 1380 - W$ mm yr^{-1} W_{pos} allows calculation of the logarithm		
Q_a	Global top of atmosphere solar irradiance (MJ m^{-2} yr^{-1})	10 minute	Roderick (1999)
Q_{s_global}	Solar radiation received at the surface assuming 70% transmittance $Q_{s\_global} = 0.7Q_a$		Anderson (2005)
Radn	Incoming short-wave solar radiation received at the surface, MJ m^{-2} yr^{-1}	250m	SRAD; Wilson & Gallant (2000)
Vegetation			
SFRI	Victorian Department of Sustainability and Environment (DSE) Statewide Forest Resource Inventory (SFRI) Database		Department of Sustainability & Environment (2004, 2007b)

GPP	Gross Primary Productivity (mol CO_2 m^{-2} yr^{-1})	250m	Berry *et al*. (2007)
Topographic			
Elevation	Ground level elevation – 3 Second SRTM Smoothed Digital Elevation Model (DEM-S) Version 1.0.3	3 second	Commonwealth of Australia (Geoscience Australia) 2010
Slope	Slope, degrees, derived from the 3 second digital elevation model	250m	Commonwealth of Australia (Geoscience Australia) 2010
Topo	Topographic position index - calculated from the 3 second digital elevation model Classes: i- ridge top, ii – upper slope, iii – mid slope, iv – lower slope, v –valley bottom	250m	Gallant and Dowling (2003) Hutchinson *et al*. (2008)
Edaphic			
Geology	Classes: i- intrusive igneous, ii - extrusive igneous, iii - metamorphic, iv – sedimentary, v – regolith, vi – fault rocks. Classes derived from the surface lithology descriptions in the tables associated with the digitised map layers	1:1,000,000	Surface geology of Australia 1:1,000,000 scale, (Liu *et al.* 2005)
%SOC	Soil carbon concentration of A and B horizons (%C)	1km	Australian Soil Resource Information System (ASRIS) McKenzie *et al.* (2005)
Sd_A, Sd_B	Soil depth of A and B horizons (m)	1km	ASRIS (McKenzie *et al.* 2005).
BD_A, BD_B	Soil bulk density of A and B horizons (Mg m^{-3})	1km	ASRIS (McKenzie *et al.* 2005).
Sd_{AB}	$Sd_{AB} = Sd_A + Sd_B$ (m)		
SOC_A	$SOC_A = Sd_A BD_A \frac{\%SOC}{100}$ (kg C m^{-2} of ground surface)	250m	Original data resampled to 250m
SOC_B	$SOC_B = Sd_B BD_B \frac{\%SOC}{100}$ (t C ha^{-1} of ground surface)	250m	Original data resampled to 250m

4.4 Models of current carbon stock

A multiple regression model was derived (Genstat v.14) to relate site carbon stock (living and dead tree biomass, coarse woody debris and litter) to site-specific values of the environmental variables (Table 4.1) and disturbance history (including type of disturbance event, time since disturbance, previous event types and times). Disturbance types included wildfire and prescribed burning, and logging using different silvicultural systems (clearfell, single tree selection, group selection, thinning). Fire history of the ash forest in the study region is shown by decade in Map 3. Logging history is shown in Map 4 by decade of occurrence (spatial data from DSE).

Relationships between site carbon stock and environmental variables and disturbance history are complex. To assist separating the effects of disturbance history, the region was divided into areas of different last disturbance event types (that is, most recent event in each grid cell): (1) clearfelling (CF); (2) single tree selection (STS); (3) group selection (GS); (4) thinning (TH); (5) wildfire (WF); and (6) prescribed burn (PB). The management practice of prescribed burning is applied only to areas of mixed species forest and not to ash forests. The areas that have been subject to these disturbance event types of silvicultural systems or fire types have been summarised for each catchment (Table 4.2).

Time since last disturbance is shown in Map 5. Time since disturbance does not necessarily equate with age of a forest stand as not all disturbances kill all trees. Wildfire does not necessarily kill all ash trees (Smith *et al.* 1985, Lindenmayer *et al*. 1990, Mackey *et al*. 2002), and in areas subject to low severity fire, canopy trees may survive (Smith *et al.* 1985). In mixed species forest, which are subject to prescribed or hazard reduction burning, the fire should not kill canopy trees. Single tree selection, group selection and thinning harvesting systems do not kill all trees at a site. Clearfelling is the main harvesting system that removes all trees from the site and should result in even-aged regeneration (Lutze *et al.* 1999, Campbell 1997). However, the site data show that there are sites designated as clearfell in the SFRI inventory up to 50 years ago that have a few trees that are greater than 100 cm or even 300 cm DBH, which must have been left during harvesting. Hence, the relationship between biomass and time since disturbance represents the biomass at the site with a given disturbance history, rather than the biomass of an even-aged stand derived from a regeneration event.

Sites where the age of the trees was reasonably well known were the unburnt ANU carbon sites (n = 18, unburnt treatment in each of the three age categories), and the SFRI clearfell sites (n = 81). These sites were combined to derive a model of carbon accumulation over time. The ANU carbon sites have a higher total carbon for a given age than the SFRI sites by an average of 11% ($F_{1,98}$ = 7.37, P=0.008). Possible reasons include the fact that larger plots were measured on the ANU carbon sites and may provide a better account for the variable distribution of large trees, and some of the ANU carbon sites were located in long-term protected areas that have experienced limited disturbance and hence are likely to have accumulated greater carbon stocks.

Table 4.2. Area (ha) of each last disturbance event, as silvicultural systems and fire types, in each of the catchments pre-2009 wildfire. Data derived from the DSE spatial data of disturbance history (Maps 3 and 4) and using the most recent event in each grid cell.

Catchment	Silvicultural System						Fire		Total area
	Clearfell	Single Tree Selection	Group Selection	Thinning	Total logged	Unlogged	Wildfire	Prescribe burn	
Armstrong	144	99	0	0	243	3921	4151	13	4187
Cement Creek	27	0	0	0	27	788	815	0	809
Maroondah	7	7	0	0	14	17892	17685	221	17942
McMahon's Creek	961	955	46	7	1969	2442	3736	675	4426
Starvation Creek	1050	0	41	27	1118	2518	3636	0	3646
O'Shannassy	0	0	0	0	0	13244	13244	0	13244
Upper Yarra	27	0	0	7	34	34920	30324	4612	34954
Thomson stage 3	3460	28	134	0	3634	11525	14825	328	15159
Thomson stage 1, 1A,2	2648	1268	81	33	4042	29043	30037	3041	33085
Tarago	2258	0	117	0	2375	8185	10560	0	11067

The statistical models derived to predict the current stock of total biomass carbon and graphs of measured versus modelled data for each disturbance event type are given in Appendix 2. The models that have a last disturbance event of wildfire or prescribed burning also include a factor of the previous event when that was clearfelling because previous logging operations reduce the carbon stock. The models for last events of single tree selection and thinning also include previous events of the same type, as this previous logging reduces the carbon stock.

The statistical models to predict biomass carbon stock account for 25 – 65% of the variance in the data, with the range occurring across the six models for types of disturbance events (Appendix 2). The site data of carbon stocks are highly variable because biomass is influenced by a complex array of environmental variables and disturbance history, not all of which are captured in the spatial variables used in the models. The greatest uncertainty is related to the disturbance history and its effect on the size distribution of trees.

Biomass carbon stock predicted across the landscape was restricted to values < 1200tC ha^{-1} to ensure it was within the range of the calibration data from the sites, and thus provided a conservative estimate. An upper limit of 1200 tC ha^{-1} for biomass carbon stocks is comparable with site data from the literature, although there are reports of higher values (Keith *et al.* 2009b).

Most of the catchment areas consist of montane ash forests for which our site data of carbon stocks is appropriate for calibration of the model. However, small areas of other species occur within the catchments, including *E. obliqua* (messmate), *E. radiata* (narrow leaved peppermint) and *E. pauciflora* (snowgum). Carbon stock was calculated over the whole region using the total biomass carbon model calibrated for ash forest and then adjusted in the areas with other species. An adjustment for the carbon stock of other species was estimated using existing allometric volume equations and wood density for the other species (Keith *et al*. 1997, 2000, Bi *et al*. 2004, Illic 2000). The wood density of *E. obliqua* (0.63 g cm^{-3}) is higher than that for *E. regnans* (0.52 g cm^{-3}), the trees are generally smaller so the wood volume for given tree diameter is less, and there is greater branching. This results in similar or higher biomass for *E. obliqua* than *E. regnans*. Hence, no adjustment of the biomass carbon modelled for *E. regnans* was made for the areas of *E. obliqua*. Predicted biomass for *E. pauciflora* from a species-specific allometric equation is approximately one third less than biomass predicted for *E. regnans* by its species-specific allometric equation, for trees up to approximately 100 cm diameter. This concords with the average maximum height of *E. pauciflora* (approximately 20 m) being about one third of the height of *E. regnans* (approximately 60 m). Wood density of *E. pauciflora* (0.527 g cm^{-3}) is similar to *E. regnans*. The biomass of *E. radiata* was assumed to be similar to that of *E. pauciflora*.

The models to predict current biomass carbon stock (living and dead trees, coarse woody debris and litter) were applied to the area of each last disturbance event and then combined for the whole study region (Map 6). The map shows the mean value of total carbon predicted for each grid cell. The uncertainty about this estimated mean value is calculated as the upper and lower 95% confidence limits, which show shifts in the distribution to higher and lower values of total carbon. The frequency distribution of biomass carbon density is shown in Figure 4.1 where the total area integrated under the curves remains the same at 448,500 ha of forest. The confidence limits show the range in carbon density with the lower limit having a higher proportion of the area with low carbon density, and the upper limit having a higher proportion of the area with high carbon density.

Statistics for the spatial biomass carbon density shown in Map 6 and the soil carbon map from ASRIS (McKenzie *et al*. 2005) are summarised for each catchment in Table 4.4. This represents the current carbon stock that takes into account the age of forest stands, forest type and environmental conditions. The mean biomass carbon density varies greatly between catchments, with O'Shannassy and Maroondah having very high mean carbon stocks. The total current carbon stock in the catchment area is 64 MtC in biomass and 18 MtC in the soil.

Figure 4.1. Distribution of total biomass carbon density by area with a mean distribution (green), which is shown in Map 6, and the lower 95% confidence limit (yellow) and the upper 95% confidence limit (blue).

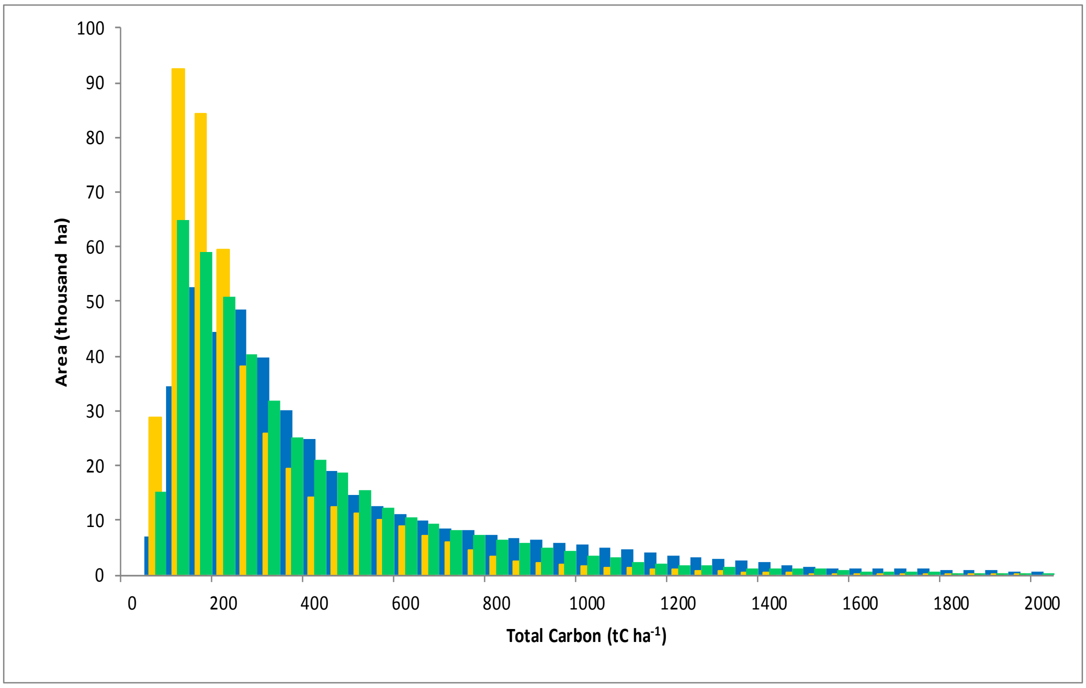


Table 4.4. Carbon stocks in the catchment areas with mean biomass carbon density, standard deviation and the total biomass and soil carbon stock for each catchment area.

Catchment	Total area (ha)	Mean (tC ha^{-1})	SD (tC ha^{-1})	Biomass carbon (MtC)	Soil carbon (MtC)
Armstrong	4187	323.4	267.2	1.34	0.45
Cement Creek	809	704.0	264.1	0.58	0.14
Maroondah	17942	844.7	578.7	15.06	3.29
McMahon's Creek	4426	345.8	235.5	1.54	0.49
Starvation Creek	3646	431.3	276.5	1.57	0.45
O'Shannassy	13244	935.0	465.1	12.36	2.74
Upper Yarra	34954	421.5	268.3	14.43	4.19
Thomson stage 3	15159	353.8	294.2	4.76	1.59
Thomson 1,1A,2	33085	304.4	210.7	10.07	3.65
Tarago	11067	195.3	141.4	2.05	0.82
Total catchments	138644			63.76	17.79

4.5 Synopsis

Current carbon stock was calculated across the landscape of the catchment area of approximately 450,000 ha to be 64 MtC in biomass and 18 MtC in the soil. The mean biomass carbon density varied spatially with the highest densities of 935 tC ha^{-1} and 845 tC ha^{-1} in the O'Shannassy and Maroondah catchments, respectively. Calculation of the current stock accounted for the forest type, environmental conditions, disturbance history and resulting age of the forest.

However, time since disturbance does not necessarily equate with age of a forest stand as not all disturbances kill all trees. The relationship between biomass and time since disturbance represents the biomass at the site with a given disturbance history, rather than the biomass of an even-aged stand derived from a regeneration event.

The site data of carbon stocks were highly variable because biomass is influenced by a complex array of environmental variables and disturbance history, not all of which are captured in the variables for which there is spatial data that can be modelled. Uncertainty of the impacts of disturbance events on tree size distribution is critical, including the amount of carbon stock loss due to the number and size of tree mortality, and the effect on rates of regeneration, growth and mortality.

5. Impact of Wildfire on Carbon Stocks

5.1 Introduction

Wildfires are a common natural disturbance event across large areas of eucalypt forests in Australia (Bradstock *et al.* 2012) and have been a major factor impacting ecosystems in the Central Highlands (Lindenmayer 2009). The amount of carbon lost from the forest ecosystem is an important issue for quantifying emissions in national and regional carbon accounts and for assessing the differences in emissions from natural and human-induced disturbance events.

The impact of the 2009 wildfire on biomass carbon stocks was assessed by estimating (i) the proportion of each biomass component that was combusted in areas subject to either low or high severity fire, and (ii) the subsequent redistribution of carbon between pools, in particular from living to dead biomass pools. Developing measurement techniques to quantify the carbon combusted was a major research task. Biomass components were measured at the ANU carbon sites and the amount combusted was used as a proportion of the current carbon stock. Spatial estimation of the carbon stock post-2009 fire was based on reductions in the stock according to the fire severity in the areas burnt.

The same forest at Site 470 following high severity fire.

5.2 Biomass components combusted

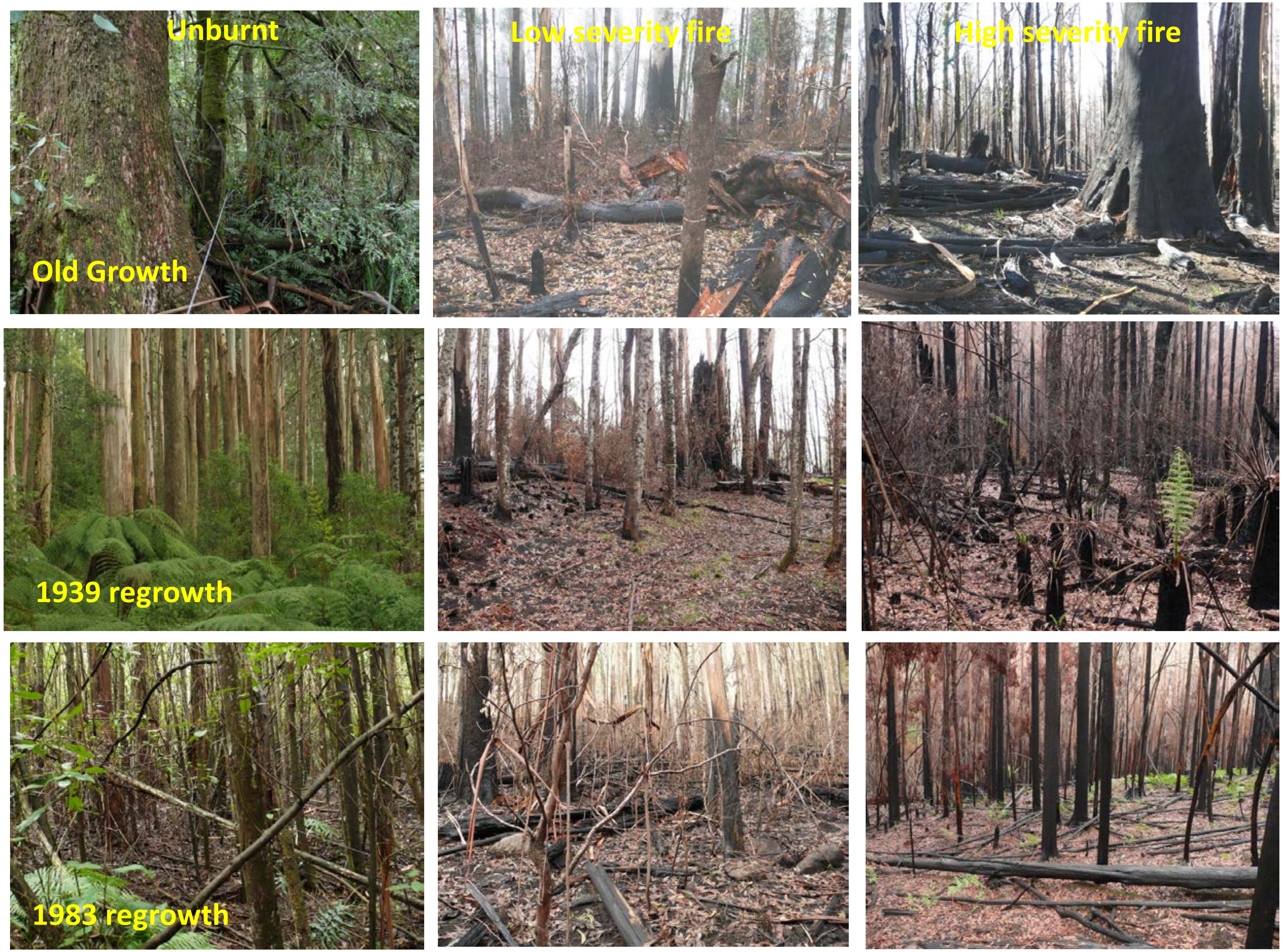
The carbon stock in biomass components combusted was estimated from several sources of evidence: (1) observations and photographs taken after the fire, (2) measurements of components post-fire at burnt and unburnt sites, and (3) comparison of measurements pre- and post-fire. The methods used for each component are described in the following sub-sections and summarised for the ecosystem in Table 5.7. Redistribution of biomass was assessed from the site carbon stock budgets.

Photographs of representative sites in each of the age / fire categories within four months of the fire illustrate many of the biomass components that were changed by the fire (Figure 5.1). In areas burnt by low severity fire, there were mixed proportions of the carbon stock in living and dead vegetation. The proportion was highly variable among sites depending on the severity of the fire, tree size, species present, and their capacity to regenerate. Some stands of *E. regnans,* in fact, had variable proportions of *E. nitens* that regenerated by coppicing. Both green and scorched leaves, twigs and fruit remained in the canopy. Woody shrubs and mid-storey trees remained with leaves and twigs either combusted or scorched. The scorched canopy components and decorticating bark fell to the ground over the weeks and months following the fire. On the ground, most of the litter layer was combusted but fallen charred sticks, CWD, pieces of charcoal and some fragmented woody litter remained. Most of the ground surface was covered by fallen scorched material several months after the fire. This indicated that a large amount of the canopy biomass was transferred to the dead litter pool and was not combusted.

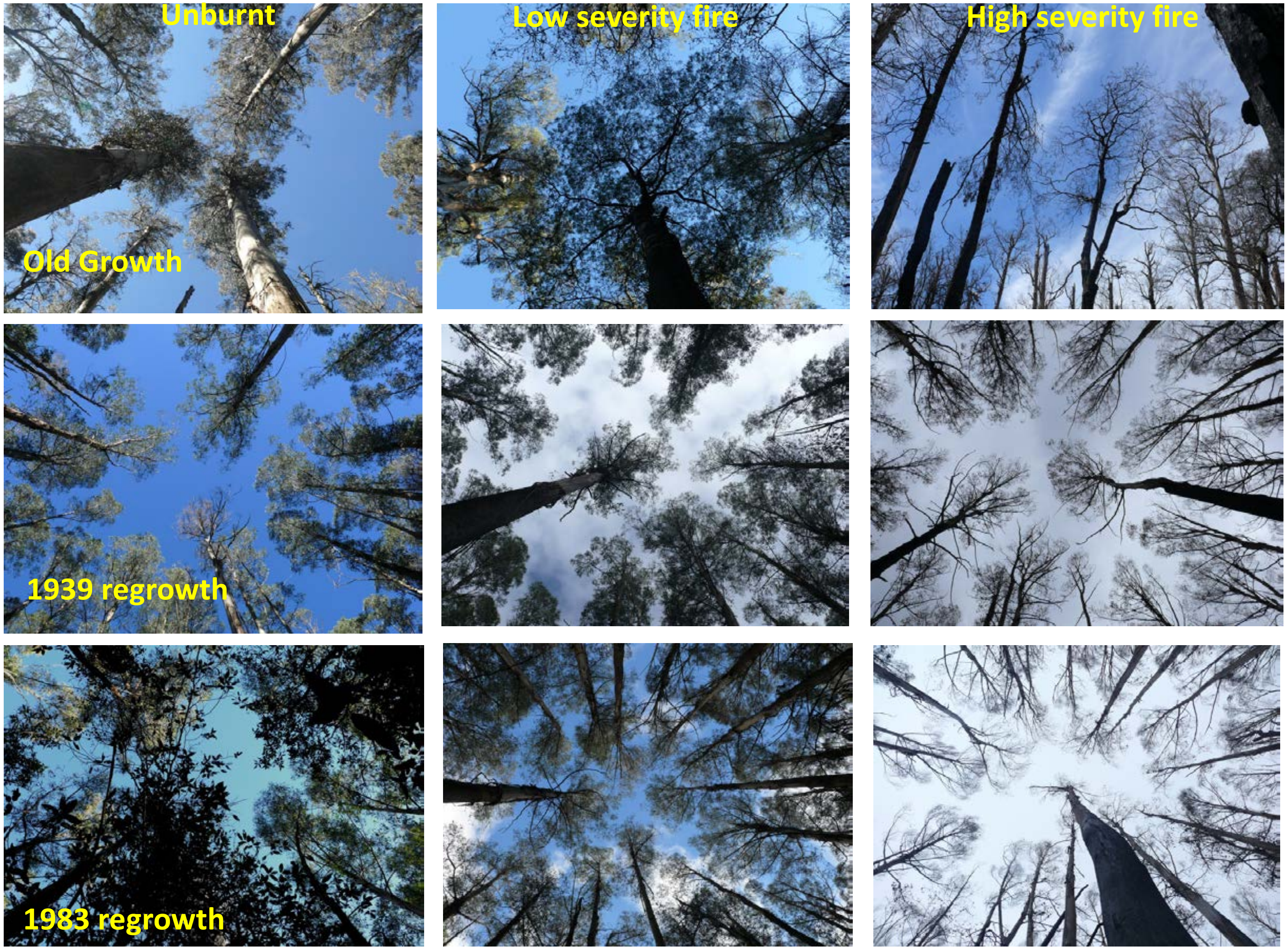
After a high severity fire, most of the leaves and some twigs were combusted, although scorched leaves remained in patches. Many woody shrubs and mid-storey trees remain but with their leaves, twigs and sometimes smaller branches combusted. Also there were examples of entire small trees being combusted and only a stump remaining. Any scorched canopy components fell to the ground after the fire. Decorticating bark, both gum bark from the upper stems and rough bark from the lower stems, peeled away from the stem and fell to the ground over time after the fire and continued to do so for up to three years. The ground surface was similar to the low intensity fire with charred woody material and fallen scorched leaves, but with more bare earth. There were many fallen stems and branches both on the ground and suspended, particularly in the young stands.

Figure 5.1. Photographs of representative sites of each of the age / fire categories approximately 4 months after the fire.

a)Quadrats (1 m x 1 m) showing the litter layer. Leaves on the ground after the fire mainly represent fall of scorched leaves from the canopy.



(b) View through the stand of trees at each site showing the remaining living and dead stems after the fire.



(c) View of the canopy at each site showing the green and scorched leaves and many small twigs and branches remaining after the fire.

5.2.1 Litter layer

Most of the litter layer was combusted in both the low and high severity fire categories at our study sites (Figure 5.1a). This represents a reduction in the carbon stock of about 7 – 9 tC ha^{-1} (Table 3.12). Leaves observed on the ground after the fire mostly represented fallen scorched leaves.

5.2.2 Canopy biomass

An estimation of canopy leaf biomass of approximately 7 tC ha^{-1} was derived from annual litterfall of 4.7 tC ha^{-1} yr^{-1} in ash forests (Ashton 1975a, Polglase and Attiwill 1992) and longevity in the canopy of about 18 months (Jacobs 1955, Ashton 1975b). Most of the canopy leaf biomass was combusted at high severity sites, although there were some trees that retained scorched leaves that later fell to the ground (Figure 5.1a and c). Approximately half or less of the canopy biomass was combusted at low severity sites based on the data from litterfall of scorched material up to two years after the fire (Figure 5.1 a and c). Litter biomass, consisting of scorched material, collected on the ground after the fire was on average 4 – 6 tC ha^{-1} (Table 3.12), which represents more than half of the average canopy biomass. Twig biomass in eucalypt canopies is approximately equivalent to leaf biomass (Keith 1991). If twigs in the canopy of diameter < 4 mm were combusted (see section 5.3), then the carbon stocks combusted for leaves and twigs would be similar, as shown in Table 5.7.

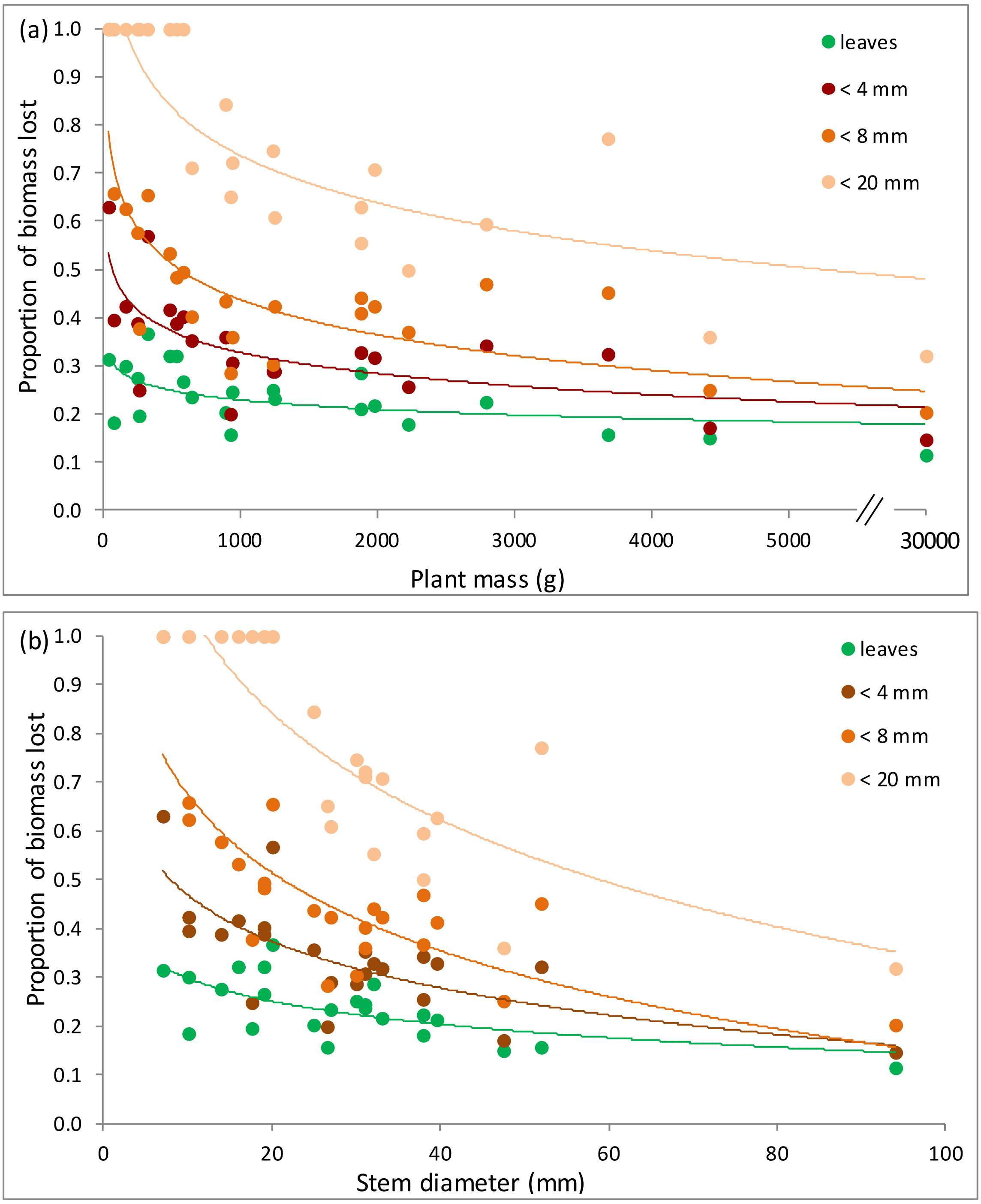
5.2.3 Small stems and branches

The amount of woody stems, branches and twigs combusted from shrubs was assessed at the sites using two sources of data. First, minimum sizes of tips remaining on stems after the fire were measured for shrubs of a range of species (Table 5.1). The maximum branch size combusted was 42 mm and the mean was 4 mm in low severity fire and 8 mm in high severity fire. Second, the proportion of biomass lost from a shrub by combustion of branches was tested by sampling individual shrubs and weighing the whole shrub and then removing leaves and branches of specified diameters (Figure 5.2 a and b). A range of species and sizes of understorey shrubs was sampled.

Table 5.1 Minimum diameter (mm) of tips that remained on stems of shrubs after combustion by low and high severity fire.

	mean	se	minimum	maximum
Low severity fire	4	0.2	1	19
High severity fire	8	0.7	1	42

Figure 5.2. Proportion of biomass lost from a shrub due to removal of leaves and branches in categories of specified diameters plus all smaller components. Biomass lost was related to (a) total biomass of the whole plant, and (b) stem diameter of the plant.



The smallest category of stem diameters measured in the vegetation plot inventory at the ANU carbon sites was 0 - 50 mm. The amount of biomass carbon in these small shrubs is shown in Figure 3.15 with the frequency size distribution of trees in each age / fire category. Combustion of the mean sizes of minimum tips in low (4mm) and high (8 mm) severity fires (from Table 5.1) from these small shrubs resulted in a loss of 0.25 to 0.50 of the shrub biomass in low severity fire, and 0.3 to 0.75 of shrub biomass in high severity fire (range across age categories). This amount of biomass combusted from small shrubs represented a proportion of less than 0.005 of the total tree biomass carbon. The carbon stock in small shrubs and the amount of carbon combusted is given in Table 5.2 and the results of carbon combusted from small stems included in the ecosystem summary (Table 5.7).

The maximum amount of shrub biomass in this size category that could be combusted potentially would be all stems less than 50 mm diameter. This could be estimated from the current amount of biomass in the unburnt sites (Table 5.2). Combustion of this shrub biomass represented about 0.01 of the total tree biomass in 1983 regrowth and less than 0.01 in older regrowth and old growth forest. However, some shrub biomass remained at the burnt sites, which indicated that only a proportion was combusted.

Table 5.2 Carbon stock in stems < 50 mm diameter and their proportion of total tree biomass, averaged for the sites in each age / fire category (n = 6).

	Carbon stock in stems <50 mm (tC ha^{-1})			Proportion of total biomass	Biomass combusted (tC ha^{-1})	
Stand age	Unburnt	Low severity	High severity	Unburnt	Low severity	High severity
1983 regrowth	2.98 ±0.50	1.07 ±0.37	0.01 ±0.01	0.0103 ± 0.0025	1.49 ±0.62	2.24 ±0.50
1939 regrowth	1.12 ±0.34	0.87 ±0.30	0.12 ±0.09	0.0021 ±0.0005	0.56 ±0.45	0.84 ±0.35
Old-growth	0.45 ±0.17	0.30 ±0.11	0.08 ±0.03	0.0007 ±0.0003	0.23 ±0.20	0.34 ±0.17

Small branches combusted on larger diameter shrubs (> 50 mm) were estimated from the proportion of biomass lost when branches of diameter 4 mm (low severity fire) and 8 mm (high severity fire) were removed from the whole shrub (Figure 5.2). Combustion of branches in low severity fire corresponded to approximately 0.15 of the plant biomass, and combustion of branches in high severity fire corresponded to 0.2 of the plant biomass. This carbon stock combusted in small branches and as a proportion of the total tree biomass is shown in Table 5.3 for each of the age categories, and the results for small branches included in the ecosystem summary (Table 5.7).

Table 5.3 Carbon stock in branches of shrubs 50 – 100 mm diameter, with branch diameter < 4 mm combusted in low severity fire and < 8 mm diameter in high severity fire, and the proportion that this represents of the total tree biomass carbon stock, averaged for the sites in each age / fire category (n = 6).

	Carbon stock combusted from stems 50-100mm (tC ha^{-1})		Proportion of total tree biomass carbon	
Stand age	Low severity	High severity	Low severity	High severity
1983 regrowth	11.9 ±1.2	14.0 ±3.0	0.030	0.057
1939 regrowth	11.8 ±2.7	15.0 ±4.0	0.030	0.037
Old-growth	7.6 ±3.4	9.0 ±2.0	0.015	0.018

5.2.4 Bark

Rough fibrous bark on the lower part of *E. regnans* stems can be combusted. The amount of bark lost was calculated by comparing bark thickness related to stem diameter at burnt and unburnt sites, and calculating this in terms of bark mass (Table 5.4). A greater thickness of bark was combusted in high severity fire in all forest age categories. The amount of bark combusted was significantly greater in old growth than in younger stands because there is greater tree biomass and the bark on large trees is thicker and more fibrous, and hence more flammable.

Decorticating bark hanging in the canopy is a small proportion of total tree biomass but it is a component that is combusted in crown fires and so represents a loss of carbon stock. It was assumed that all decorticating bark in the canopy was combusted in both low and high severity fires, based on observations after the fire (Table 5.4). Hence, the bark biomass reflects the tree size distribution at the sites. The largest amounts of decorticating bark occurred in the old growth stands that have large trees. This may be an underestimate because a maximum bark mass was assigned due to the high variability in large trees. This estimation of decorticating bark mass is an approximation but provides information for a biomass component where there was no previous data.

Table 5.4. Carbon stock (tC ha^{-1}) loss due to combustion of rough and decorticating bark on standing trees in fires of low and high severity, averaged for the sites in each age / fire category (n = 6).

	Rough Bark		Decorticating Bark	
Stand age	Low severity	High severity	Low severity	High severity
1983 regrowth	0.56 ±0.19	0.81 ±0.10	0.31 ±0.02	0.17 ±0.05
1939 regrowth	0.57 ±0.13	0.74 ±0.24	0.32 ±0.06	0.40 ±0.10
Old-growth	1.32 ± 0.32	2.90 ±0.43	0.50 ±0.07	0.52 ±0.09

5.2.5 Coarse woody debris

Some CWD on the ground is combusted during wildfire, but both the amount per site and proportion combusted is highly variable spatially. The young regrowth sites subject to high severity fire supported significantly ($P < 0.01$) less CWD than the unburnt sites, with an average of half the amount of CWD (Table 3.12). However, there was no significant difference in amounts of CWD among the other forest age categories, which likely reflects the high variability of this biomass component within age / fire categories rather than the fact that no biomass was combusted.

An alternative approach to estimating the amount of CWD combusted was to identify where logs had been before the fire by the line of charcoal remaining on the ground and assuming that the initial carbon stock of these logs could be represented by the mean size of logs at the site. Carbon stocks in CWD logs that were assumed to be combusted are given in Table 5.5. Much of the CWD in the regrowth stands was derived from waste remaining after logging operations and residual trees. The high carbon stock estimated to be from combusted CWD in the 1983 regrowth sites that were subject to high severity fire concords with the low stock of current CWD post-fire at these sites (Table 3.12).

Table 5.5. Carbon stock (tC ha^{-1}) estimated to have been combusted in CWD.

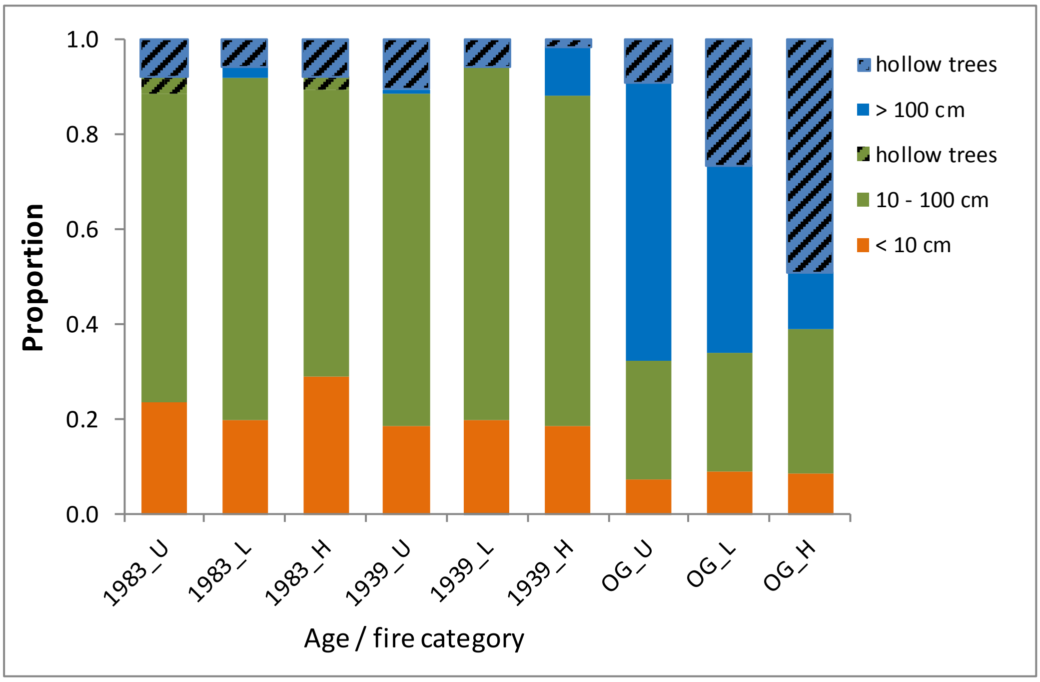
Stand age	Low severity	High severity
1983 regrowth	1.37 ±0.65	2.41 ±1.67
1939 regrowth	1.83 ±0.73	1.70 ±0.48
Old-growth	0.16 ± 0.16	0.43 ±0.33

5.2.6 Trees with hollows

Hollow-bearing trees are an important structural component of montane ash forests (Lindenmayer *et al.* 2011) and are highly susceptible to combustion because the fire is funnelled through the pipe. Both living and dead trees can contain hollows, and they provide habitat for arboreal animals. These individual trees have been identified at each site and their height, diameter and form monitored over a number of years. Data from 2005 and 2011 were compared to assess the change pre- and post-2009 fire. Biomass carbon stock was calculated for individual trees at each site. The difference in carbon stock between years represented a combination of natural mortality, decay and collapse of trees plus combustion in the burnt sites.

The proportion of hollow trees in a stand is highly variable in terms of both number and carbon stock (Lindenmayer *et al*. 1991). These trees represent a specific structural characteristic (Lindenmayer *et al*. 2000a) that is influenced by age, although not exclusively. In the 1983 and 1939 regrowth stands, hollow trees represent remnant trees that remained after harvesting or fire and do not reflect the age of the regrowth stand. Hollow trees represent 0.02 to 0.12 of the total biomass in these regrowth stands (Table 5.6a). In old-growth stands, hollow trees generally consist of the older cohort of trees but may also include damaged younger trees. The proportion is highly variable, from 0.09 to 0.49 of the total tree biomass. Large trees (> 100 cm diameter) comprise 0.61 to 0.68 of the total biomass, hence there are large trees that do not have the structural characteristics of hollow trees (Figure 5.3).

Figure 5.3. Proportion of the carbon stock in forest stands in stem size classes and trees with hollows, averaged for the sites in each age / fire category (n=6 sites per category).



Losses in carbon stock from trees with hollows from 2005 to 2011 were highest in old-growth stands subject to high severity fire (43 tC ha^{-1}) but also high in 1983 regrowth stands both unburnt and subject to high severity fire (Table 5.6b). These losses in carbon stock were highly variable across sites because they depended on initial biomass of the hollow trees in the stand and natural decay and loss of the hollow trees. Distinguishing the effect of natural decay and combustion was more easily interpreted from the proportional loss in carbon stock rather than the absolute amount. The average amount of carbon lost due to combustion in the low and high severity fire was calculated as the difference in loss between burnt and unburnt treatments. This accounted for natural decay and collapse of hollow trees that occurred without fire (Table 5.6b). Carbon loss was also estimated as a proportion of the initial stock in trees with hollows at the site. The proportional loss is significantly ($P < 0.05$) greater in burnt sites in the 1939 regrowth and old-growth sites than in unburnt sites.

The rate of natural loss of hollow trees through mortality, decay and collapse is highly dependent on climate conditions. Mortality of living trees would have been high during the measurement interval from 2005 – 2011 because of drought conditions (Lindenmayer and Wood 2010, Lindenmayer *et al*. 2012).

The amount of combustion and loss of carbon from hollow trees is likely to be greater than from intact trees because the hollow trees have cavities and pipes that allow the stem to ignite easily and continue burning (Mackey *et al.* 2002). Hence, the proportion of combustion of hollow trees is not necessarily applicable to other large trees.

Table 5.6a. Carbon stock (tC ha^{-1}) in hollow trees pre-2009 and calculated as a proportion of the total tree biomass carbon stock at the site. Data are averaged for the sites in each age / fire category (n = 6 sites).

	Carbon stock pre-2009 (tC ha^{-1})			Proportion of total tree biomass carbon		
Stand age	Unburnt	Low severity	High severity	Unburnt	Low severity	High severity
1983 regrowth	36.5 ±10.6	22.6 ±7.3	36.8 ±9.0	0.12 ±0.038	0.06 ±0.020	0.11 ±0.020
1939 regrowth	49.7 ±22.0	22.9 ±7.2	6.9 ±2.2	0.11 ±0.054	0.06 ±0.019	0.02 ±0.003
Old-growth	44.9 ±18.0	122.1 ±40.3	240.9 ±39.0	0.09 ±0.059	0.27 ±0.098	0.49 ±0.064

Table 5.6b. Carbon stock loss (tC ha^{-1}) from trees with hollows, measured before and after the fire, due to natural decay and collapse of dead trees plus combustion in the burnt sites. Data are averaged for the sites in each age / fire category (n=6 sites). Loss of carbon is also shown as a proportion of the initial carbon stock in trees with hollows at the site.

	Carbon stock loss (tC ha^{-1})			Proportional loss		
Stand age	Unburnt	Low severity	High severity	Unburnt	Low severity	High severity
1983 regrowth	19.6 ± 6.5	8.8 ±4.2	20.4 ±9.6	0.32 ±0.16	0.44 ± 0.12	0.54 ±0.09
1939 regrowth	11.3 ±4.9	21.0 ±7.0	2.2 ±1.1	0.25 ±0.04	0.59 ±0.30	0.81 ±0.13
Old-growth	2.3 ± 1.5	14.6 ±6.4	42.7 ±7.7	0.05 ± 0.03	0.18 ±0.04	0.21 ±0.10

5.2.7 Summary of biomass combusted

The estimated maximum carbon stock combusted in low and high severity fires was summarised from each of the biomass components for 1983 and 1939 regrowth and old-growth forest (Table 5.7). The carbon stock combusted as a proportion of the pre-fire total biomass carbon stock provides an indication of the significance of fire as a process of carbon emissions. In low severity fires 6 – 7% of the carbon stock was lost, and in high severity fires 9 – 14% of the carbon stock was lost.

Table 5.7. Summary of biomass components combusted by low and high severity fires (mean ±standard error, n = 6 sites). Total biomass combusted is the sum of all the components combusted, total carbon stock pre-fire is the current carbon stock of the ecosystem before the fire, and proportion combusted is the total biomass combusted divided by the pre-fire carbon stock.

	Mass of carbon combusted (tC ha^{-1})					
Low severity fire	**1983 regrowth**		**1939 regrowth**		**Old growth**	
Litter	6	±1.0	9	±1.0	9	±1.0
canopy leaves	3.5		3.5		3.5	
small stems	1.5	±0.6	0.6	±0.4	0.2	±0.2
small branches	11.9	±1.2	11.8	±2.7	7.6	±3.4
decorticating bark	0.3	±0.02	0.3	±0.06	0.5	±0.07
rough bark	0.6	±0.2	0.6	±0.1	1.3	±0.3
CWD	1.4	±0.6	1.8	±0.7	0.2	± 0.2
Stags	3.8	±0.9	9	±1.7	17.6	±5.2
Total biomass combusted	29	±4.5	36.6	±6.7	39.9	±10.4
Total carbon stock pre-fire	433	±35	475	±91	564	±88
Proportion combusted	0.063		0.072		0.066	
High severity fire	**1983 regrowth**		**1939 regrowth**		**Old growth**	
Litter	6	±1.0	9	±1.0	9	±1.0
canopy leaves	7		7		7	
canopy twigs	7		7		7	
small stems	2.2	±0.5	0.8	±0.3	0.3	±0.2
small branches	14	±3.0	15	±4.0	9	±2.0
decorticating bark	0.3	±0.05	0.4	±0.1	0.5	±0.1
rough bark	0.8	±0.1	0.7	±0.2	2.9	±0.4
CWD	2.4	±1.7	1.7	±0.5	0.4	±0.3
Stags	7	±1.9	14.8	±1.2	21.7	±6.2
Total biomass combusted	46.7	±8.3	56.4	±7.9	57.8	±10.2
Total carbon stock pre-fire	286	±46	459	±102	589	±63
Proportion combusted	0.140		0.109		0.089	

There are few reliable data on fuel consumption during wildfires (Gould and Cheney 2007). Most fine fuel < 25 mm is consumed, but the fraction of larger components consumed depends on fire intensity and spatial variation. Our results of biomass combusted are based on mean values from surveys of sites subjected to low and high severity fire, not a small area uniformly subject to high intensity fire. Comparison with data from an experimental high intensity fire in subalpine *E. delegatensis* forest (O'Loughlin *et al*. 1982) show greater proportions of branches being combusted. In the *E. delegatensis* study, all fine material (< 50 mm) was consumed (22.5 t dry weight ha^{-1}), whereas we estimated that 0.5 and 0.75 of the biomass was combusted in low and high severity fires, respectively. The total carbon stock in this small shrub size category was small, 0.5 to 3 tC ha^{-1}, and the different proportions combusted represented 0.005 to 0.01 of the total biomass carbon stock. Higher proportions of larger diameter material were combusted in the *E. delegatensis* study, including 0.7 of the material 50 – 100 mm (4.3 t dry weight ha^{-1}), 0.57 of the material 100 – 200 mm (7.8 t dry weight ha^{-1}), and 0.26 of the material > 200 mm (14.9 t dry weight ha^{-1}). A proportion of 0.5 of the total ground fuel load of 100 t dry weight ha^{-1} was consumed. In a comparison of studies of fuel consumed during experimental fires in wet sclerophyll forests in Tasmania, Western Australia and NSW, the range was from 0.09 to 0.90, with a mean of 0.45 (Hollis *et al.* 2010). Hence, 0.5 was considered a reasonable average at the landscape scale including spatial variability in fire intensity. In our study in *E. regnans* forest the proportions of material combusted, determined from minimum tips remaining, was lower but the amount of material in these diameter size class was greater. The total amount of biomass carbon combusted in high severity fire, approximately 50 tC ha^{-1} in our study, was about twice that estimated for the *E. delegatensis* forest, although the latter did not include combustion of live standing fuels and bark.

5.3 Redistribution of carbon between pools

The proportions of each of the biomass carbon components within age / fire categories are shown in Figure 5.4. This illustrates the shift among components as forest structure changes with stand age and after fire. Low severity fire resulted in a small shift from living to dead biomass, whereas there was a very large shift after high severity fire. Even in some areas subject to high severity fire small amounts of living biomass survived. There were differences in total carbon stocks between the fire categories but the major change was the redistribution of carbon among pools.

Figure 5.4 Redistribution of carbon stocks between pools in relation to stand age and fire severity, shown as the proportion of carbon in each biomass component. Data are derived from the carbon stock data in Table 3.12in the age / fire categories (n = 6 sites). living biomass (green), dead biomass (blue), litter (orange), CWD (purple).

5.4 Spatial estimation of carbon stocks post-2009 wildfire

The area burnt in the 2009 wildfire is shown in Map 7 derived from DSE spatial data. The distribution of areas subject to low and high severity fire are based on the criteria for fire severity categories in Table 3.1. The areas burnt in low and high severity fire is given for each catchment in Table 5.8.

Table 5.8. Area (ha) of each category of fire severity in the 2009 wildfire in each of the catchments.

Catchment	Low severity	High severity	Unburnt	Total
Armstrong	2392	1608	187	4187
Cement Creek	0	0	809	809
Maroondah	11609	1096	5236	17942
McMahon's Creek	691	0	3735	4426
Starvation Creek	7	0	3639	3646
O'Shannassy	6353	4831	2060	13244
Upper Yarra	1927	199	32827	34954
Thomson stage 3	0	0	15159	15159
Thomson stage 1, 1A,2	0	0	33085	33085
Tarago	2638	3117	5312	11067
Total	25617	10851	102047	138519

The spatial estimate of current carbon stock (Map 6) was reduced in the areas burnt by the proportion of biomass combusted in low and high severity fire for each forest age category (Table 5.7). The post-2009 current carbon stock is shown in Map 8 and the mean carbon density and total carbon stock is given for each catchment in Table 5.9. The total carbon stock in the catchment areas was 63.76 MtC pre-2009 and an estimated 2.42 MtC (approximately 4% across the whole catchment area) was lost by emissions from biomass combustion during the 2009 wildfire.

Table 5.9 Mean carbon density and total biomass carbon stock in each catchment area pre- and post-2009 wildfire.

		Pre-2009		Post-2009	
Catchment	Total area (ha)	Mean (tC ha^{-1})	Biomass carbon (MtC)	Mean (tC ha^{-1})	Biomass carbon (MtC)
Armstrong	4187	323.4	1.34	289.5	1.20
Cement Creek	809	704.0	0.58	704.0	0.58
Maroondah	17942	844.7	15.06	799.9	14.26
McMahon's Creek	4426	345.8	1.54	343.4	1.53
Starvation Creek	3646	431.3	1.57	431.3	1.57
O'Shannassy	13244	935.0	12.36	847.5	11.20
Upper Yarra	34954	421.47	14.43	417.3	14.29
Thomson stage 3	15159	353.8	4.76	353.8	4.76
Thomson stage 1, 1A,2	33085	304.4	10.07	304.4	10.07
Tarago	11067	195.27	2.05	182.2	1.91
Total catchments	138644		63.76		61.34

5.5 Synopsis

Wildfire is a major form of disturbance in the montane forest ecosystem that affects the carbon stock by emissions from combustion of biomass and redistribution of biomass from living to dead carbon pools. The main change is the mortality of nearly all plants subject to high severity fire. This produces a large pool of dead biomass that slowly collapses and decays over many decades, but at the same time regeneration produces new living biomass. The proportion of biomass combusted is small; approximately 7% in forest subject to low severity fire and 9 to 14% in forest subject to high severity fire. In the areas of the water catchments that were burnt, more than twice the area was burnt at low rather than high severity fire. The amount of biomass combusted is highly variable spatially and depends on the fuel type and fuel load, burning conditions such as weather and topography, diurnal effects and the area burnt. An estimated 2.4 MtC (approximately 4% of the pre-2009 current carbon stock across the whole catchment area) was lost by emissions from biomass combustion during the 2009 wildfire.

6. Impacts of Logging on Carbon Stocks

6.1 Introduction

To estimate reductions in carbon stocks due to logging and the accumulation of carbon during regrowth, and hence make predictions of future changes in carbon stocks in forests, data are required to model these dynamics in carbon stocks over time. Carbon accumulation over time is described as a mathematical equation that relates stand age to carbon stock. Age of trees is required to position the forest stand on the carbon accumulation curve. To define the equation for carbon accumulation it is necessary to know the maximum level of the carbon stock at a site in a mature forest given the environmental conditions that influence growth. The reduction in carbon stock due to logging is derived from the amount of biomass removed off-site as harvested product plus the emissions from slash burning. The carbon stock at a site is changed due to logging by the redistribution of biomass components from living to dead carbon pools. In this study, the age of forest stands was determined from the available GIS data layers of harvesting and wildfire disturbance event history (Section 2.4, Maps 4 and 5). However, note that as shown in Section 4.4, stands are not necessarily even-aged in relation to time since last disturbance event due to the legacy of residual trees. Given an estimated age of the stand, the carbon accumulation equation allows prediction of the carbon stock at the site. Scenarios of changes in carbon stock into the future were predicted based on the spatial distribution of the age structure of the forest and the carbon accumulation equation.

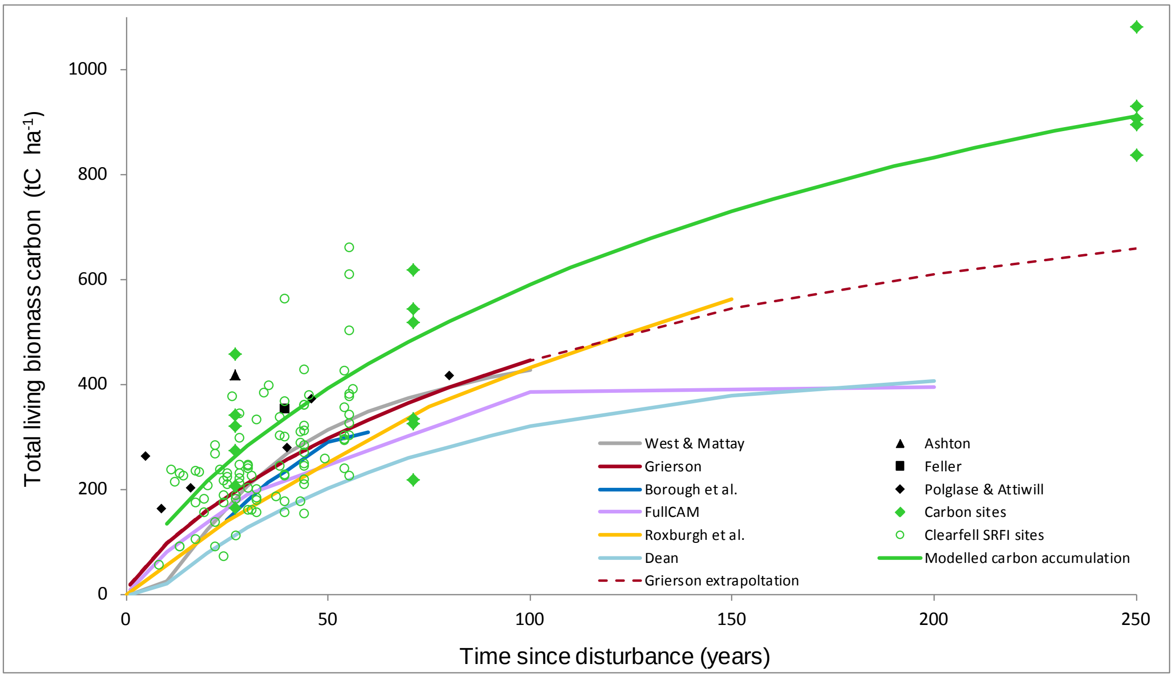
Reduction in the carbon stock of a natural forest due to management practices of clearfelling and slash burning.

6.2 Carbon accumulation function

Prediction of change in carbon stocks over time, or the rate of carbon accumulation in forest stands, is determined by tree growth rates, tree density, and stand size structure. Carbon accumulation over time is defined as the relationship between time since the last disturbance event and total living biomass carbon stock. Site data and growth functions describing *E. regnans* were collated from the literature and combined with data from the unburnt ANU carbon sites and SFRI clearfell sites (where time since disturbance reflects age of the trees most accurately) (Figure 6.1 and Table 6.1).

Figure 6.1. Carbon accumulation over time in *E. regnans* forest stands based on equations and site data from the literature, and data from SFRI clearfell sites and ANU carbon sites unburnt. Details of the data used to derive the equations from the literature are given in Table 6.1.

Legend: West and Mattay (1993); Grierson *et al*. (1992) and extrapolation using an exponential model as the red dashed curve; Borough *et al*. (1984); FullCAM in DCCEE (2010); Roxburgh *et al.* (2010); Dean *et al.* (2003) and Dean and Roxburgh (2006); Ashton (1976); Feller (1980); Polglase and Attiwill (1992); SFRI sites clearfell silvicultural treatment and the ANU carbon sites from the current study in forest unburnt in 2009 were combined to derive the carbon accumulation curve shown in green.



The site data from the current study (green points in Figure 6.1) were combined to derive a carbon accumulation function (green curve in Figure 6.1):

$$\text{Living Biomass Carbon (tC ha}^{-1}\text{)} = 1200 * (1\text{-exp}(-0.0045 * \textit{Time}))^{\wedge 0.7} \qquad (9)$$

Where, *Time* is time since last disturbance event in years.

These sites have similar or higher carbon stocks than estimates from the growth equations in the literature.

The equation by Grierson *et al*. (1992) (Table 6.1) in the form of a polynomial should not be used to estimate carbon stocks in older stands because this requires extrapolation beyond the size range of the field data used for its derivation. The form of the polynomial equation decreases after the maximum tree DBH in the calibration data. Hence, the form of this equation was converted to the Chapman-Richards function (Richards 1959, Janisch and Harmon 2002):

$Y = y_{max}(1-e^{-kx})^{r}$ where, y_{max} is the carbon stock at CCC, e is the base of natural logarithms, 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. The equation derived for *E. regnans* is:

$$\text{Aboveground living biomass carbon (tC ha}^{-1}\text{)} = 620(1-e^{-0.0065*A})^{0.75} \quad (10)$$

This curve is shown in Figure 6.1 (dashed red line) as an extrapolation of the Grierson curve after 100 years. Both Equations (9) and (10) are used to estimate changes in carbon stocks over time to provide a range of predictions.

The equations derived by Dean *et al*. (2003) were based on individual tree data for DBH and age plus stocking rates per hectare. Watson and Vertessy (1996) derived an equation between mean DBH for an even-aged stand and stand age that produced a very similar relationship to that by Dean *et al.* (2003) although using a different function.

Table 6.1. Growth functions reported in the literature for *E. regnans*.

Reference	Equation	Data description	Max. age (years)	Sample no. (n)	r^2
West & Mattay (1996)	V_p = 3.93 -31.2 / A + 0.1146 * SI SI = H{[1-exp(-0.03 * 20)] / [1-exp(-0.03 * A)]}$^{0.902}$	Plot data, fully stocked, monospecific stands, regrowth and plantation	85	215	0.78
Grierson *et al*. (1992)	AGB = 4.99 + 12.83 * A + (-0.0654 * A^2)	Plot data for standing timber volumes, wood density, expansion factor.	76	45	0.95
Borough *et al.*(1984)		Data in the form of yield tables of stem volume for fully stocked, unthinned stands with SI=30m			
FullCAM DCCEE (2010)	6.44 (1-10 yrs), 4.41 (11-30 yrs), 2.23 (31-100 yrs), 0.74 (100-200 yrs), 0 (>200 yrs) tC ha^{-1} yr^{-1}	Default values of growth increments (tC ha^{-1} yr^{-1}) as constants for age classes of Tall Dense Eucalypt Forest			
Dean *et al*. (2003) and Dean and Roxburgh (2006);	DBH(m) = 3.53 –(3.59 * $e^{-0.00323 * Age}$) V_t = 1100 *{1-[1+(DBH(m)/9.2)^2]$^{-1}$} S = exp(10.6 – (1.27 * LN(A)))	Data for DBH and age from Ashton (1976b) in Victoria, coupe data and measured plots in Tasmania. Even-aged stands with no thinning. Data for individual trees and stand stocking related to age.	450	2184 82	0.82 0.92
Watson and Vertessy (1996)	LN DBH(cm) = 1.206 + 0.719 LN(A – 5.04)	Relationship between DBH and age for even-aged stands	230	17	0.99
Roxburgh *et al*. (2010)		Plot data for biomass in Victoria with a fitted growth curve	150		

V_p = stem volume under bark (m^3 ha^{-1}) at the plot scale V_t = stem volume under bark (m^3) for an individual tree A = age (years)
AGB = aboveground biomass carbon (tC ha^{-1}) S = stocking (stem ha^{-1}) SI = site index (mean dominant height at 20 years) (m)
Default factors used in calculations: Mean Site Index of 30m (average value given in West and Mattay (1993), value used in Borough et al (1984), average value calculated for the young regrowth carbon sites in the current study).
Expansion factor for Aboveground biomass : stemwood = 1.18 (mean of 1.17 (Ashton 1976), 1.22 (Feller 1980), 1.14 (Keith and Sillett unpubl. data)
Expansion factor for Total biomass : Aboveground biomass = 1.25 (Snowdon *et al*. 2000); Wood density = 0.5 t m^{-3}; Carbon concentration = 0.5 g g^{-1}

The carbon accumulation function was compared with estimates of growth rates over specific age intervals from reports in the literature to assess the generality of the function (Table 6.2). Carbon accumulation was estimated from gross volume increment data, however it was uncertain whether these estimates of gross volumes included all trees or only merchantable trees in the stand (Raison and Squire 2007). Gross volume was converted to total living biomass carbon stock using the default conversion factors at the end of Table 6.2. Growth rates used by the NGGI are averages that take into account variations in site quality, stocking, fire and management history. However, Flinn *et al*. (2007) consider these growth rates may be underestimates.

Table 6.2. Increments in carbon stocks (tC ha^{-1} yr^{-1}) over specified time intervals, comparing values from various reports in the literature with the carbon accumulation curves (Equation (9)) derived in this study.

Site characteristics	Volume increment (m^3 ha^{-1} yr^{-1})	Biomass increment (tC ha^{-1} yr^{-1})	Reference
High quality regeneration, well-stocked	10 – 20 mean=15	6.8	Borough et al. (1984), Attiwill (1979), West and Mattay (1993)
1-50 yrs 50-80 yrs	11 5.5	2.75 1.37	Feller (1980), Flinn et al. (2007)
0-7 yrs		11	Attiwill (1992)
average	8	2	NGGI (1997)
1-10 yrs 11-30 yrs 31-100 yrs 100-200 yrs >200 yrs		6.44 4.41 2.23 0.74 0	DCCEE (2010)
1-50 yrs 51-100 yrs 101-200 yrs >200 yrs		7.8 4.0 2.4 1.6	Equation (9)

Default conversion factors:
- stem biomass : aboveground biomass for native eucalypt forest = 1.46 (Snowdon et al. 2000, Flinn *et al*. 2007),
- total biomass : aboveground biomass = 1.25 (Snowdon *et al.* 2000),
- wood density = 0.5 t m^{-3},(Illic et al. 2000)
- carbon concentration of 0.5 g g^{-1} (Gifford et al. 1999)

Calculation of carbon accumulation in regrowth stands relies on the assumption of fully-stocked regeneration and hence optimal biomass increment given the site conditions. Hence, estimates of carbon stocks in regrowth forests based on environmental conditions, age and rate of growth will produce overestimates at the landscape level because some areas of regrowth will not be fully stocked.

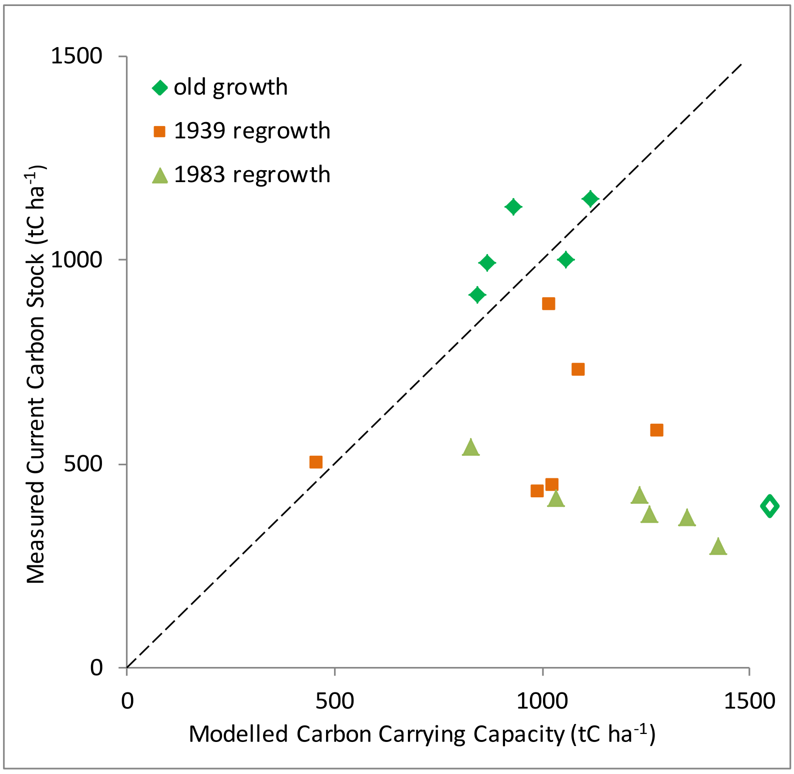
6.3 Carbon Carrying Capacity

Carbon carrying capacity (CCC) is defined as the mass of carbon that can be stored in a natural ecosystem under prevailing environmental conditions and natural disturbance regimes, but excluding disturbance by intensive human land-use activities (Gupta and Rao 1994, Keith *et al*. 2009b). It is a landscape-scale metric that refers to the average conditions within a dynamic equilibrium across a region. Hence, CCC provides a baseline for estimating the impacts of human disturbance regimes on carbon stocks. Once the natural CCC is established, it is possible to calculate the potential increase in carbon storage that would occur if forest management was changed so that emissions are avoided or reduced. This potential increase is called the carbon sequestration potential (CSP).

Age of forest stands is an important factor influencing CCC. The maximum age of *E. regnans* in the ANU carbon sites is approximately 250 years, reflecting regeneration following a wildfire in approximately 1750. Evidence of the maximum age of forest stands of *E. regnans* and other eucalypts indicates up to 400 - 500 years (Gilbert 1958, Ogden 1978, Banks 1983, Wellington and Noble 1985, Looby 2007, Wood *et al*. 2010). Continuing rates of carbon accumulation or positive net ecosystem carbon exchange has been demonstrated in old growth forests (Luyssaert *et al*.2008, Phillips *et al*. 1998). The increment in total ecosystem carbon depends on the rates of photosynthesis, respiration, recruitment and mortality. Ecosystem respiration encompasses respiration from living biomass, soil respiration and the rate of decomposition of dead biomass (Keith *et al*. 2009a).

Spatial estimation of carbon carrying capacity for the Central Highlands region in this study was derived from an analysis of natural eucalypt forests in south-east Australia (Keith *et al*. 2009b) that used calibration data from sites that had minimal disturbance from human landuse activities. The Keith *et al.* (2009) model was used to derive CCC for the ANU carbon sites that were unburnt in the 2009 fire. The comparison between the modelled values of CCC and the measured carbon stocks at the sites is shown in Figure 6.2. The model provides a reasonable prediction of carbon stocks for most old growth sites and some 1939 regrowth. The younger sites have lower measured carbon stocks than the modelled value for carrying capacity which represents maximum average age of forest stands. The difference between the measured current carbon stock in the younger stands and the CCC represents the carbon sequestration potential or gain in carbon that could be achieved if the forest continued to regrow undisturbed. Spatial estimates of CCC in each catchment are given in Table 6.4 and Figure 6.3.

Figure 6.2 Comparison of modelled carbon carrying capacity and measured carbon stocks at the ANU carbon sites in forest unburnt in 2009 in the Central Highlands in three age categories. Dashed line represents the 1:1 line comparing measured and modelled values. Open green diamond represents an outlying site of old growth with a much higher modelled CCC than measured carbon stock.



6.4 Biomass removal by logging

In the context of accounting for carbon stocks within catchment forests, harvesting represents removal of biomass carbon off-site. Timber harvesting utilizes various silvicultural systems in the Central Highlands, as defined in Section 4.3. Change in carbon stock at a site due to logging depends on the silvicultural system, forest type, form and age of trees, market demand and proximity to processing facilities. Here, we concentrate on the most common practice in the Central Highlands of clearfelling and slash burning in montane ash forests (Lutze *et al*. 1999).

The total area within the water catchments that has been logged since the 2009 wildfire is 608 ha, shown in Map 9. The silvicultural systems utilized include salvage logging within the fire footprint area and clearfell and seed tree retention logging in unburnt areas (Table 6.3).

Table 6.3. Area (ha) of forest in each of the catchments that has been subject to logging since 2009, classified by the main silvicultural systems. Spatial data of logging history is from DSE.

Catchment	**Silvicultural System**			
	Clearfell salvage	Clearfell	Seed tree retention	Total logged
Armstrong	41	0	0	41
Cement Creek	0	0	0	0
Maroondah	0	0	0	0
McMahon's Creek	0	0	0	0
Starvation Creek	0	24	12	36
O'Shannassy	0	0	0	0
Upper Yarra	0	0	0	0
Thomson stage 3	0	47	12	59
Thomson stage 1, 1A,2	0	236	12	248
Tarago	165	24	35	224

The area proposed for logging is determined by the Timber Release Plans (TRP) designated by the Department of Sustainability and Environment (DSE 2012). The TRP defines the area available for harvest, and within this area there is an agreed limit to the hectares actually harvested each year. For the period 2011 to 2016, this includes both salvage logging and logging in unburnt areas, with approximately one-third of this area having been logged previously (Map 10).

The area proposed for logging in the TRP is 1025 ha yr^{-1} and the agreed limit is 285 ha yr^{-1}, that is, 29% of the proposed area is actually harvested (150 ha in Thomson; 67 ha in Cement, Armstrong, Starvation and McMahon's; and 78 ha in Tarago) (Melbourne Water pers. comm.). The areas of best quality timber within the TRP are selected, but the actual area harvested is not spatially defined. The total carbon stock in the TRP area within the catchments is estimated to be 2,300,000 tC (Table 6.4a) and so the carbon stock actually harvested would be approximately 670,000 tC (this is likely an underestimate if the best timber is selected for harvest). The silvicultural systems to be used in the proposed logging area are specified in Table 6.4b, with each system as a proportion of the total TRP area. Most of the area proposed for logging is for clearfell and clearfell salvage logging, with smaller areas for seed tree retention, thinning and roads. Thinning is proposed only in the

Thomson and Tarago catchments as a small proportion of the area logged, which would account for approximately 8,000 tC.

Table 6.4a). Areas of the catchment that have been logged (to December 2011) and are proposed for logging in the Timber Release Plans (TRP) by DSE for 2011 to 2016, and the predicted carbon stock in this area of forest.

Catchment	Total area (ha)	Area logged to 2011 (ha)	% catchment area logged to 2011	Area of proposed logging TRP (ha)	% catchment area in TRP	Mean ($tC\ ha^{-1}$)	SD ($tC\ ha^{-1}$)	Total carbon in TRP area (Mt C)
Armstrong	4187	285	6.87	318	7.67	543	306	187
Cement Creek	809	27	3.24	166	20.24	846	265	164
Maroondah	17942	13	0.07	0	0	0	0	0
McMahon’s Creek	4426	1969	44.12	292	6.54	510	159	128
Starvation Creek	3646	1154	31.66	305	8.37	648	319	190
O’Shannassy	13244	20	0.15	0	0	0	0	0
Upper Yarra	34954	0	0	0	0	0	0	0
Thomson stage 3	15159	3693	27.47	577	4.29	591	251	355
Thomson stage 1, 1A,2	33085	4289	12.96	1843	5.57	453	208	841
Tarago	11067	2599	24.74	1565	14.89	256	126	388
Total catchments	138644	14068	10.39	5125	3.78			2312

Table 6.4b) Proportion of the TRP area within each silvicultural system based on spatial data from DSE.

Catchment	Area of TRP (ha)	Proportion of TRP area in each silvicultural system				
		Clearfell	Clearfell Salvage	Seed Tree	Thinning	Roads
Armstrong	318	0.41	0.59			
Cement Creek	166	0.88	0.12			
Maroondah						
McMahon's Creek	292	0.74		0.26		
Starvation Creek	305	0.89		0.11		
O'Shannassy						
Upper Yarra						
Thomson stage 3	577	0.92				0.04
Thomson stage 1, 1A,2	1843	0.89		0.03	0.06	0.02
Tarago	1565	0.43	0.44	0.06	0.04	0.04

The areas proposed for logging do not occur in the areas of highest biomass carbon density, as these are in the protected water catchments. However, some of the areas proposed for future logging in the TRPs occur in relatively high carbon-dense forest. The carbon density of areas proposed for logging is related to (i) the previous logging history which is a major determinant of the current age of the forest, (ii) the species composition which affects tree longevity, density and growth rates, and (iii) land tenure which determines previous disturbance by logging (Map 11).

The proportion of total biomass that is removed off-site as harvested sawlog plus pulpwood product and the proportion remaining on-site as slash is important for determining the shift in pools of carbon and their longevity. The default value used for moist high quality forest under integrated harvesting for sawlog plus pulpwood is 0.4 removed and 0.6 remaining as slash. The range in the proportion of slash estimated in native forests ranges from 0.2 to 0.8 (Raison and Squires 2007). The reported proportion removed off-site as harvested product is 0.44 in ash forests (Flinn *et al*. 2007). The proportion was also estimated from data in the current study based on the average total biomass carbon density in the areas proposed for harvesting (450 tC ha^{-1} from Table 6.4a), and the reported average wood product harvested in clearfelling operations (600 m^3 ha^{-1} from DSE (2009), [equivalent to a carbon stock of 150 tC ha^{-1} using conversion factors of wood density=0.5 t m^{-3}, carbon concentration=0.5 g g^{-1}]). This gives the proportion of the carbon stock removed from site by clearfell harvesting as 0.33.

The few reported studies of the proportion of total biomass removed off-site by harvesting in other forest types show similar proportions, but with a wide range. At an experimental site in tall wet regrowth forest on the south coast of NSW, the proportion of wood product to aboveground biomass and total biomass was 0.58 and 0. 46 in *Corymbia maculata* (Spotted Gum), 0.70 and 0.56 in *E. obliqua* (Messmate), and 0.45 and 0.36 in *E. pilularis* (Blackbutt), respectively (Ximenes *et al* 2004, Ximenes and Gardner 2005, Ximenes *et al.* 2008b)(using an average root : shoot ratio of 0.25 (Snowdon *et al.* 2000)). The proportions of 0.46, 0.56 and 0.36 were for living biomass whereas the proportion calculated in the current study in ash forests was for total biomass. A study in *E. regnans* forest with temperate rainforest understorey in Tasmania found 0.26 of the aboveground biomass was removed as wood product (Green 2002). Expansion factors, which relate the proportion of aboveground biomass that is converted into wood products, reported in the literature range from 0.5 to 0.77 (Snowdon *et al.* 2000). Amounts of biomass remaining on-site as slash can be very high. For example, the reported range in Victorian Ash forests was an average of 42 – 102 tC ha^{-1} but can exceed 275 tC ha^{-1} (Flinn *et al.* 2007).

There is a high degree of uncertainty in the estimate of the proportion of total biomass that is removed by harvesting operations because few field data have been collected on the biomass that remains on site. The proportion is highly variable depending on forest type and age of the stand, which determine the proportion of branchwood, stem defect, commercial species, and amount of understorey.

The current market demand for types of products and the economics of transporting to processing facilities influence the products harvested. The proportion removed is usually less in old multi-aged forests than in even-aged regrowth forests. The remaining biomass on-site or slash consists of stumps, crowns, branches, bark, wood with defect, non-commercial species and understorey. The remaining biomass on-site has not been considered an important forest management issue because it is not relevant to industrial wood supply products, but it is important in terms of the impact of land use activities on forest carbon stocks.

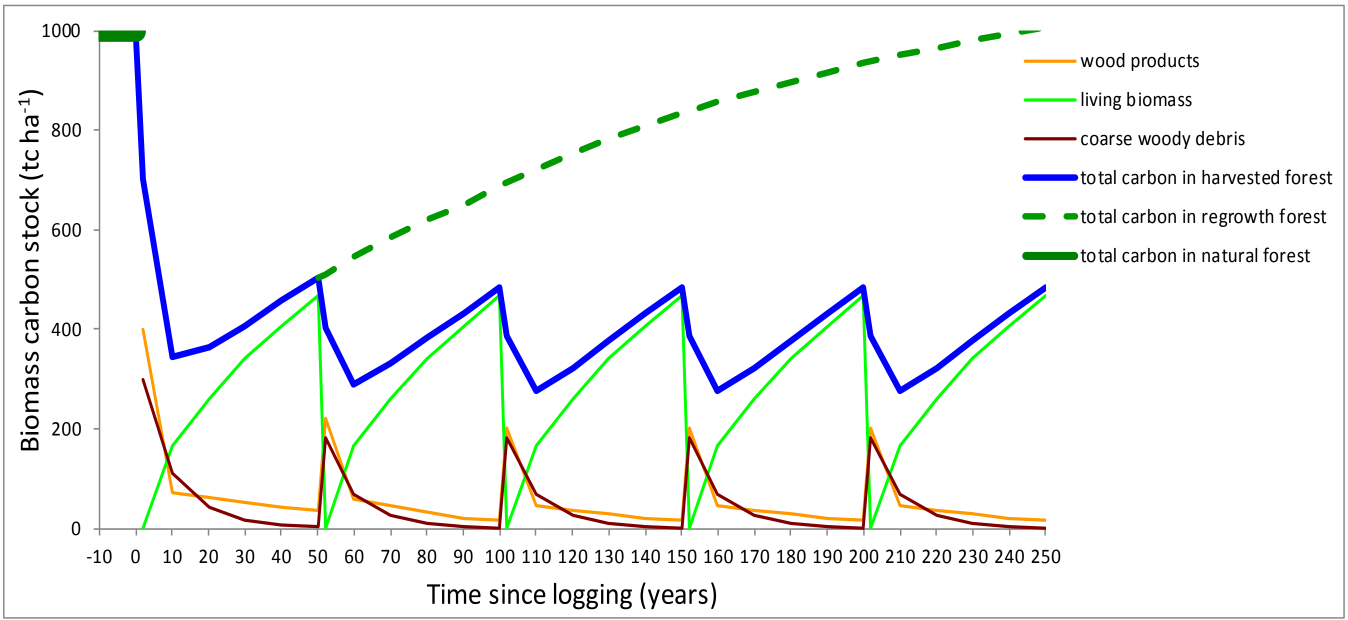
Much of the harvested area in the Central Highlands is burnt by intense slash fires, a practice that became common after 1965 (Flinn *et al.* 2007). Slash is burnt to clear the ground surface, release seed stored in capsules by heating, heat the soil and produce a layer of ash that improves conditions for seed germination (Slijepcevic 2001). The biomass components combusted include the slash produced from harvesting, understorey biomass, fallen dead trees, coarse woody debris and litter layer. An approximate burning efficiency is 0.5 of all available fuel present over the total area that is combusted (Gould and Cheney 2007). Burning efficiencies were estimated to be 0.7 for fine fuels (< 100 mm with 1.0 loss over 0.8 of coupe area for 0.9 of coupes), and 0.5 for CWD (> 100 mm with 0.8 loss over 0.7 of coupe area for 0.9 of coupes) (Flinn *et al.* 2007). In a specific study of burning efficiencies in tall wet eucalypt forest of *E. obliqua* in Tasmania, Slijepcevic (2001) found that 0.58 to 0.63 of the biomass carbon in slash was emitted to the atmosphere. In tall wet forest of *E.*

diversicolor in Western Australia, the burning efficiency of slash fires was estimated to be 0.31 to 0.89 (McCaw *et al.* 1997).

The dead biomass remaining on-site, including aboveground biomass that was not combusted during slash burning plus belowground biomass (either the 0.5 that is not combusted or the total amount if there is no slash burning) decomposes over time and emits carbon dioxide to the atmosphere. Decomposition rates vary depending on the size and location of the dead biomass components. Information about decomposition rates of large woody material is scarce. The half- life for coarse root biomass in moist eucalypt forests is estimated to be approximately 50 years, and time for complete decomposition and emission of carbon dioxide is not known (Ximenes and Gardner 2006). Decomposition of *E. regnans* coarse woody debris for logs 10 – 30 cm diameter on the ground surface has an average lifetime of 43 years (Mackensen *et al*. 2003, Mackensen and Bauhus 2003). However, coarse woody debris decomposition rates are highly dependent on size of the log and the micro-climate conditions at the site.

The changes in carbon stock of total biomass following clearfelling and slash burning on a 50 year rotation were summarised in a model with the output shown in Figure 6.3. Initial carbon stock before logging was the average value of total biomass carbon at 250 years (as shown in Figure 6.1 for living biomass carbon). The proportions of biomass carbon lost from the site included 0.4 of total biomass as wood product removed and 0.5 of the remaining slash due to combustion. Remaining dead biomass was assumed to decompose with a decay rate constant of 0.1 (default value used in NCAS, Richards and Brack 2004). Accumulation of carbon over time by the regenerating forest follows the function in Equation (9) but calibrated for total biomass carbon (living plus dead biomass). The results of this modelled output show that the carbon stock in a repeatedly harvested forest never regains the magnitude of the initial carbon stock (Figure 6.1). As a silvicultural system averaged spatially across the landscape with areas at different times since logging, the average carbon stock is 0.39 of the initial stock. After a single logging event, accumulation of carbon takes 250 years to regain the initial stock. (This time period was defined by the assumption that the initial stock was that at 250 years in Equation (9), and the rate at which carbon continues to accumulate in older forests is not known.) An assumption has been made in the model that carbon accumulation in the regenerating forest continues at the same rate during each rotation, that is, there is no reduction in site productivity. However, there is little evidence to support this assumption in montane ash forests because there have not been sufficient areas harvested and regenerated on more than one rotation. If it is shown that productivity does decline, then there would be a corresponding decrease in carbon stocks.

Figure 6.3. Changes in biomass carbon stock under a scenario of clearfelling and slash burning on a 50 year rotation (Timber Industry Strategy State of Victoria, DPI 2009). Carbon stock in the natural forest is reduced by logging and will either remain at a lower level under continued harvesting rotations, or increase in the regrowing forest to eventually return to the initial stock. The carbon stock under the harvesting rotation is separated into the components of living biomass, coarse woody debris and wood products. The proportions allocated to each of these components and their rates of stock change are derived from the data in Figure 6.4 and the accompanying text. Sustainable production is assumed over the course of the rotations.



Thinning operations remove an average of 0.5 of the basal area of trees, and damage an average maximum of 0.15 of retained trees (DSE 2009). The area of thinning operations is an average of 0.14 of the total area harvested. [Source data from DSE 2009 averaged for the period 2002 – 03 to 2008 – 09, for forest management areas of Central, Dandenong and Central Gippsland, and for forest types of *E. regnans, E. delegatensis* and mixed species].

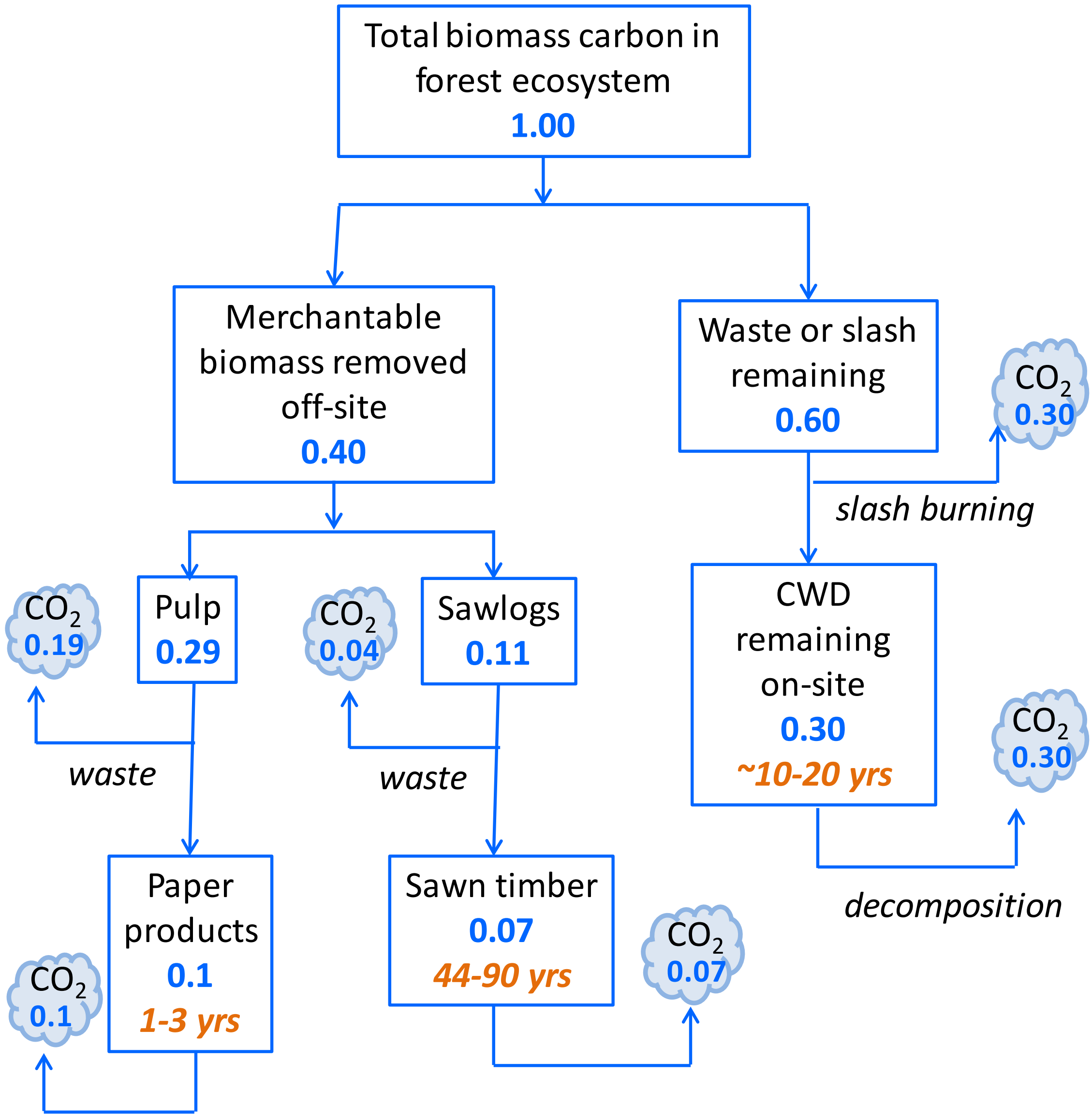
The area proposed for harvesting within the catchments contains approximately 2,312,000 tC (Table 6.3a), and the majority of this area is designated for clearfelling (Table 6.4b). Using the proportion of biomass removed off-site by harvesting as 0.4 of the total biomass, this equates to 924,800 tC, of which 161,840 tC (i.e. 17.5%) would be stored in long-term timber products (lifetimes in Figure 6.4). As slash burning is practised in most clearfelled areas, this represents emissions of 693,600 tC. The carbon stock emitted to the atmosphere within a few years is 1,456,560 tC due to slash burning, combustion or decomposition of waste during processing, and decomposition of paper products. The remaining carbon stock of 855,440 tC in timber products and coarse woody debris remaining on the ground will decompose at a slow rate (see component lifetimes in Figure 6.4).

The soil carbon stock increases initially after logging due to the addition of dead biomass that progressively decomposes. In the longer term, the soil carbon stock is reduced due to mechanical disturbance during harvesting operations and changes in micro-climate that cause increased rates of soil respiration, and reduced inputs of litter from the regenerating aboveground biomass. These impacts have been studied in *E. regnans* forests in the Central Highlands (Rab 1994, 1999, 2004) and in other forest types overseas (Olsson *et al*. 1996, Diochon *et al*. 2009). Rab (2004) found that 10 years after harvesting, soil carbon was still reduced by 0.3 to 0.4 compared with pre-harvest levels. Diochon *et al*. (2009) found that soil carbon was reduced for 15 to 45 years after harvesting in a spruce (*Picea rubens*) forest in Canada, and then gradually increased and reached the pre-harvest stock after 125 years (assumuing a single harvesting event). Hence, in the silvicultural system of an 80 year rotation, the soil carbon stock would be permanently reduced.

Biomass removed off-site represents a loss of carbon stock from the catchment area and land tenure. From the perspective of the national or global carbon balance, this biomass removed as wood products remains a stock of carbon in the land sector until it is converted to carbon dioxide. The time this takes depends on the longevity of the wood products. The proportion of biomass harvested that goes into different products depends on the tree species, age and form, proximity to processing facilities, and supply quota agreements. Of the merchantable roundwood removed off-site in the Central Highlands, approximately 0.28 is used for sawlogs (used for sawn timber, plywood and veneer) and the remainder of 0.72 for paper pulp [derived from data in DSE documents Monitoring Annual Harvest Performance for Central and Dandenong Forest Management Areas from 2003 to 2008 (DSE 2009)]. (This proportion of sawlogs may have decreased in recent years.) On average one-third of sawlogs (i.e. 0.09 of biomass in wood products removed from the forest) produce long-term products with lifetimes of 44 to 90 years, and two-thirds (i.e. 0.18 of biomass in wood products removed from the forest) goes to waste and short-lived products such as paper and packaging (Skog and Nicholson 1998, Jaako Pöyry 1999, 2000, Ximenes *et al.* 2005, 2008a, Richards *et al.* 2007). Similarly, approximately one-third of wood product that goes to pulp processing becomes waste (i.e. 0.24 of biomass in wood products removed from the forest). Other shorter-lived products have estimated lifetimes of 10 years for wood packaging and less than 3 years for packaging, cardboard, paper (Jaako Pöyry 1999, 2000, Richards *et al.* 2007). Only 0.4 of paper products are printing and writing paper that may have lifetimes of several years, the other 0.6 is household and sanitary paper, newsprint and packaging (Jaako Pöyry 1999, 2000, Australian Paper Industry 2004).

At each stage of processing of wood products, waste is produced, such as woodchips, shavings and sawdust that have short product lifetimes. At the end of the life of each type of product, the carbon in the wood or paper may be combusted as waste or decompose and hence release carbon dioxide. Rates of decomposition are highly variable depending on the size of the product and the conditions at the site. Decay rates in landfill have been estimated at 0.002 (Ximenes *et al*. 2008c). This means that the wood product with a long lifetime represents less than 0.1 of the biomass removed off-site or less than 0.04 of the initial total biomass in the forest. Most (63%) of the initial biomass carbon stock in a forest that is harvested is emitted to the atmosphere within 3 years due to combustion or decomposition of waste and pulp products. In a harvesting system, the amount of carbon stored in products for longer than the rotation length is negligible, hence there is no accumulation of carbon under this system.

Figure 6.4. Fate of biomass carbon during harvesting and processing of wood products. Numbers in blue represent the proportion of the total biomass carbon in the forest that remains in each component. Numbers in red are the average lifetime of the carbon pool (see data sources in text).



6.5 Scenarios of forest management

Scenarios of the carbon sequestration potential in the catchments were developed over different time periods and management practices to quantify the impacts of forest management on ecosystem carbon stocks. We compared carbon stocks currently, predictions for 20, 50 and 150 years into the future, and the carbon carrying capacity (CCC). Two carbon accumulation functions (Equations (9) and (10)) were used to estimate future carbon stocks, representing data from the current study and an average function from the literature, respectively. Calculation by these two functions provides an indication of the likely range in predicted carbon stocks given the uncertainty associated with the calibration data.

Projections of carbon stocks for 20, 50, 150 years into the future were calculated using forest age based on the time since the last harvesting or wildfire disturbance event and the current carbon stock as the initial value. The shape of the carbon accumulation functions (Equations (9) and (10)) were assumed to be constant irrespective of the initial carbon stock at a given age; that is the growth curves at each location were parallel. The dominant canopy trees of the forest in the majority of the study area are less than 73 years reflecting regeneration after the 1939 wildfire or more recent harvesting. Hence, it is appropriate for carbon accumulation to follow the functions in Equations (9) and (10).

Two sets of projections were calculated, based on:

1) pre-2009 condition, assuming the fire had not occurred, using the current carbon stock (Map 6) and then allowing the forest to continue growing and accumulating carbon.
2) post-2009 condition after the fire using the current carbon stock after it had been reduced due to combustion of biomass (Map 8) plus the carbon accumulation calculated from the current age in unburnt and low severity fire areas and from 0 years for regeneration in high severity fire areas.

The projected stocks of carbon in the catchments (Table 6.5) were calculated for the current carbon stock pre-2009 and post-2009, for projections over 20, 50 and 150 years using the carbon accumulation functions from Equations (9) and (10), and the potential carbon carrying capacity.

In the burnt areas, the carbon stock was initially reduced by the amount of biomass combusted, after which forest biomass growth continued from a combination of living trees or newly germinated trees. It is important to note that *E. regnans* and *E. delegatensis* trees are not like most eucalypt species which usually survive having their canopies scorched. These species usually die if the canopy is scorched and the stand regenerates naturally from germination of seed following the fire (Gill 1981). The dead biomass remaining after the fire was not changed over time, whereas it should decrease due to decomposition. However, as discussed above, rates of decomposition of different components of dead biomass are highly uncertain. Hence, the post-fire projections are likely to be overestimates for the longer time periods. After more than 50 years, approximately half of the smaller (< 30 cm diameter) dead fallen logs are likely to have decomposed (Mackensen *et al.* 2003, Mackensen and Bauhus 2003). However, the majority of the carbon stock is in larger logs and standing dead trees about which there is little information on rates of falling or decomposition. It is unlikely that much of these larger components would decompose within 50 years.

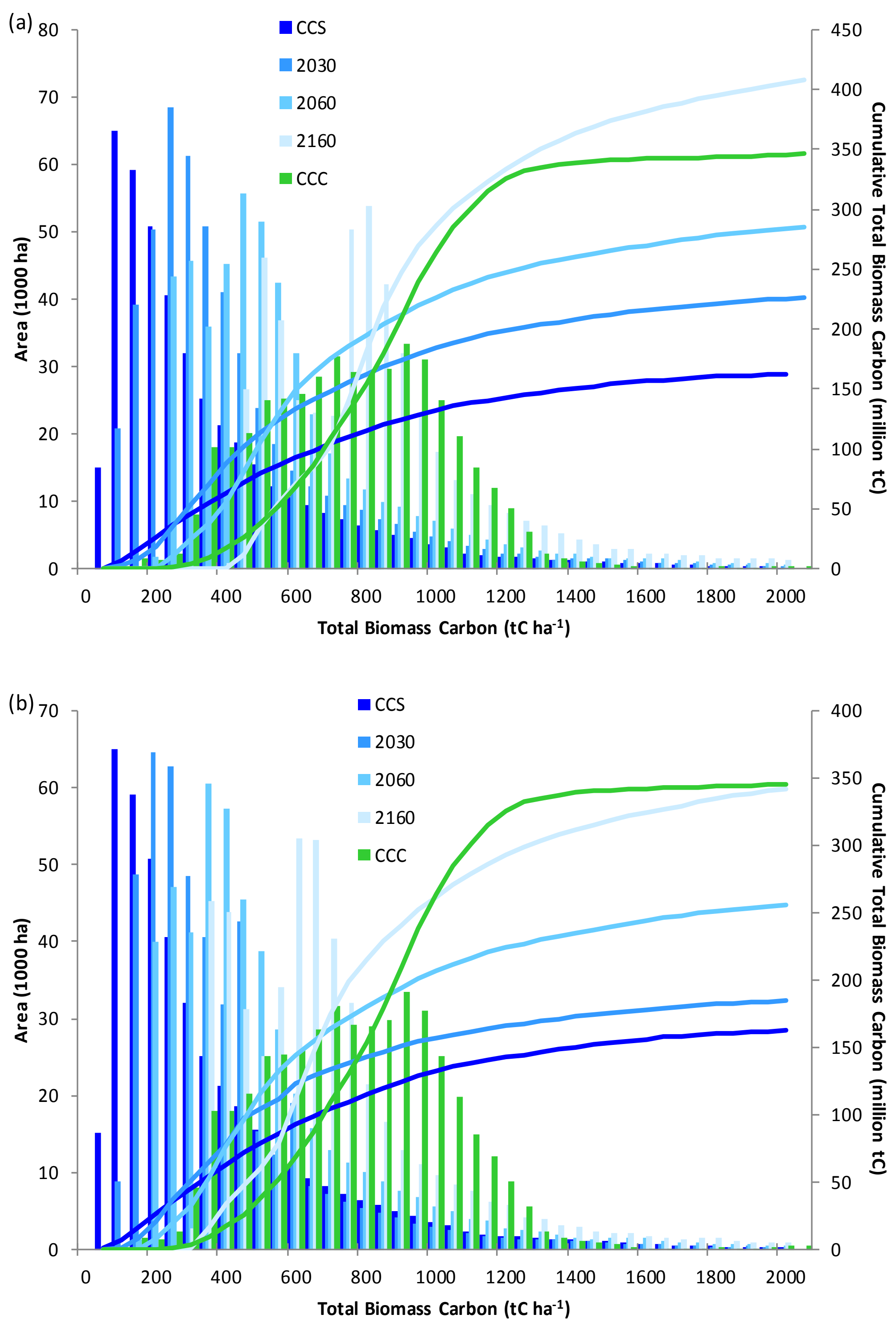
The shift in carbon stocks over time is illustrated in Figure 6.5 by the frequency distribution of carbon density across the whole study area (defined as the rectangle in the maps) for each of the scenario time periods. The age cohorts of trees that constitute the current carbon stock (CCS) continued to grow and accumulate carbon over time for 20, 50 and 150 years, and hence have similar shaped distributions but with increasing carbon density. The CCC was estimated for natural forest with the specified site conditions for each grid cell, and was not based on the existing age structure of the forest at each grid cell. The cumulative total biomass carbon stock for each of the scenario time periods illustrated the differences in final stocks as carbon accumulated over time in the growing forest.

If logging continued in the catchment areas at the current agreed limit of the actual area harvested, this would represent loss of carbon stock off-site of 469,000 tC over 5 years (proportions of 0.4 removed in wood products and 0.3 combusted in slash burning - Section 6.4) or 1,876,000 tC over 20 years. This loss of carbon off-site due to logging of coupes can be compared with the projected accumulation of carbon by the growing forest over the whole catchment area of 12,460,000 tC over 20 years using Equation (9) or 3,170,000 tC using Equation (10). The net change in carbon stock ranges from gains of 10,584,000 tC to 1,294,000 tC, depending on the growth equation used (Section 6.2). Hence, the loss of carbon due to logging represents 0.15 to 0.59 of the gain in carbon by forest growth over 20 years. The net effect of logging is to deplete carbon stocks relative to the stocks accumulated from natural forest ecosystem processes (and taking into account carbon stored in wood products) .

Table 6.5. Carbon stocks within each catchment area for the estimated Current Carbon Stock (CCS), Carbon Carrying Capacity (CCC), and predicted for +20 years (2030), +50 years (2060) and +150 years (2160), based on the current carbon stock pre-2009 fire and post-2009 fire with a reduction due to biomass combusted.

	Total carbon (MtC)											
Catchment	Pre-2009 fire condition								Post-2009 fire condition			
	CCS	Equation (9)			Equation (10)			CCC	CCS	Equation (9)		
		2030	2060	2160	2030	2060	2160			2030	2060	2160
Armstrong	1.34	2.13	2.81	4.14	1.73	2.48	3.45	3.57	1.20	1.97	2.68	4.06
Cement Creek	0.58	0.61	0.68	0.85	0.58	0.64	0.75	1.02	0.58	0.64	0.71	0.89
Maroondah	15.06	16.74	19.27	24.37	15.08	18.05	21.69	18.30	14.26	17.00	19.67	25.01
McMahon's Creek	1.54	1.96	2.53	3.72	1.67	2.28	3.11	4.07	1.53	2.06	2.64	3.88
Starvation Creek	1.57	1.87	2.33	3.31	1.63	2.13	2.82	3.50	1.57	1.99	2.47	3.49
O'Shannassy	12.36	14.15	16.19	20.22	12.57	15.21	18.13	16.10	11.20	13.47	15.63	19.88
Upper Yarra	14.43	16.92	20.65	28.72	14.94	18.89	24.38	27.63	14.29	17.68	21.53	29.88
Thomson stage 3	4.76	5.89	7.26	10.29	5.10	6.62	8.65	9.66	4.76	5.95	7.38	10.52
Thomson stage 1, 1A,2	10.07	12.56	15.84	23.14	10.92	14.29	19.17	25.00	10.07	12.89	16.32	23.93
Tarago	2.05	3.38	4.80	7.68	2.80	4.12	6.16	7.79	1.91	3.38	4.86	7.86
Total catchments	63.76	76.22	92.38	126.43	66.93	84.71	108.31	116.64	61.34	77.03	93.89	129.39

Figure 6.5 Frequency distribution of carbon stocks and the cumulative total stock in the whole study area based on carbon accumulation calculated from (a) Equation (9), and (b) Equation (10). Scenarios for different time periods include: current carbon stock (CCS), + 20 years in 2030, + 50 years in 2060, + 150 years in 2160, carbon carrying capacity (CCC).



6.6 Synopsis

Carbon accumulation over time was defined as the relationship between time since the last disturbance event and total living biomass carbon stock. The function was calibrated with site data that was known reasonably well as a regeneration event such as high severity wildfire or clearfell logging. A range of growth functions have been reported in the literature and they result in large differences in predicted carbon accumulation. Caution is required in applying functions beyond the size range of trees from which they were calibrated and evaluating the shape of the curve at large tree sizes. Adequate calibration data of carbon stocks at sites of known stand age are critical for improving the accuracy of these functions.

Impacts of human disturbance regimes on carbon stocks can be assessed against a baseline of the carbon carrying capacity. This refers to the maximum level of the accumulated carbon stock in a mature ecosystem given the environmental conditions that influence growth and natural disturbance regimes. The potential increase in carbon storage was calculated under scenarios of changed forest management such that emissions were avoided or reduced.

Timber harvesting represents removal of an average of 0.4 of the total biomass carbon stock off-site as wood products. In most ash forests, the slash remaining on-site is burnt and approximately half the carbon stock is combusted resulting in emission of carbon dioxide. The proportion of total biomass that is utilized as long-term timber products is about 0.07. These products have a longevity of 44 to 90 years which is substantially shorter than the lifespan of the trees. The carbon stock in long-term timber products is very small compared with the total biomass in the forest because large proportions of the biomass become waste at each stage of the process. In a harvesting system, the amount of carbon stored in products for longer than the rotation length is negligible; hence there is no accumulation of carbon under this system.

The initial proportion of wood products removed off-site is highly variable and uncertain; collection of field data on the amount of biomass remaining on-site should be a priority to improve carbon accounting of forest harvest management. Another factor in the uncertainty of carbon accounting is the decomposition rate of coarse woody debris, including slash remaining after burning, as there is little data available.

The area that has been harvested post-2009 fire to 2011 is 608 ha. The area agreed for future logging from 2011 to 2016 is 1486 ha. The total carbon stock in the area proposed for logging within the catchments is estimated to be 670,000 tC. From this amount 422,100 tC would be emitted to the atmosphere within a few years due to slash burning, combustion or decomposition of waste during processing, and decomposition of paper products. The remaining carbon stock of 247,900 tC in timber products and coarse woody debris remaining on the ground will decompose at a slow rate.

Forest harvested on a 50 year rotation has a maximum carbon stock that is 0.48 of the initial stock in the natural forest, and the average carbon stock across a region consisting of forest stands harvested at different times would be 0.39 of the initial stock. The projected loss of carbon stock off-site due to clearfelling is 1,876,000 tC over 20 years at the current rate of harvesting. This loss from designated coupes is a significant amount compared with the accumulation of carbon from growth of the forest over the whole area of the catchments that is predicted to range from 3,200,000 to 12,500,000 tC (range depending on the growth function applied).

7. Management Implications

7.1 Introduction

In the current national and international political setting, this is an important time to obtain data quantifying carbon stocks and stock changes due to changes in forest management activities, such as logging and the impact of fire regimes. These data are required to inform policy makers about the consequences of decisions and to maximise the opportunities to realise economic benefits derived from carbon pricing and associated funds or offset schemes. We summarise the current status of national and international policy negotiations and decisions in the context of the mitigation value of management options for native forests in Australia.

7.2 Carbon stocks

The protection of native forest ecosystem carbon stocks yields climate change mitigation value by :

i) Avoiding emissions from the industrial logging of native forests; and
ii) Sequestration of atmospheric carbon by the growing forest uninterrupted by logging and the associated depletion of organic forest ecosystem carbon stocks.

The mitigation value of carbon stored in terrestrial ecosystems (referred to as the land sector by the UNFCCC) can be realized in complementary ways at different scales:

i) Globally, management of the land sector to reduce carbon emissions and increase sequestration is a vital part of a comprehensive approach to addressing the climate change mitigation problem which complements reductions in fossil fuel emissions. When logged forests are allowed to regrow naturally, the previously depleted carbon stocks are replenished;
ii) Nationally, avoiding and reducing land carbon emissions can be registered in Australia’s national carbon accounts and contribute towards meeting mitigation targets such as those negotiated under the Kyoto Protocol; and
iii) Land carbon credits from changing forest management are eligible under voluntary carbon markets where removal of carbon from the atmosphere into the land sector is used as an offset for fossil fuel emissions; and
iv) Land carbon credits from changing forest management in public forest may be eligible under formal schemes depending on various policy decisions taken by the Australian government and the international community in the coming years.

International mitigation policies

Australia currently submits two national carbon accounts to the international community:

i) UNFCCC (United Nations Framework Convention on Climate Change) accounts include all carbon stocks and stock changes in the Land Use, Land Use Change and Forestry (LULUCF) sector; and
ii) Kyoto Protocol accounts that include afforestation, reforestation and deforestation under Article 3.3 during the first commitment period (2008 – 2012). These activities refer to a permanent change in land use. Changes in carbon stocks due to management of native forests, such as logging, are not included in these accounts.

It is the Kyoto Protocol accounts that are relevant for policy because they are used for determining compliance with Australia's international obligations for emission reduction targets. The land sector was divided into two Articles in the Protocol: Article 3.3 accounted for land use change by afforestation, reforestation and deforestation; and Article 3.4 accounted for carbon stock change due to land management, including forests, with no change in land use. Under the international agreement, Article 3.4 was optional for the first commitment period (2008 – 2012) and Australia opted not to include this activity in the national accounts. This meant that the accounts for the UNFCCC and the Kyoto Protocol were different because of the inclusion or exclusion of forest management.

Note that under the Kyoto Protocol definition of 'forest', clearfelling a native forest does not constitute deforestation so long as the land use remains 'forestry'. This definition also allows native forest to be cleared and replaced with a monoculture of exotic trees without 'deforestation' being deemed to have occurred. As noted above, the emissions from other silvicultural regimes that involve selective logging are also not covered by Article 3.3. These are unfortunate anomalies which, among other perverse outcomes, create loopholes that allow significant emissions to occur without being included in the accounts. Implementation of Article 3.4 is therefore increasingly seen as necessary in order to stop these loopholes and ensure national carbon accounts are more comprehensive and accurate.

At the UNFCCC 17th Conference of the Parties in Durban (UNFCCC 2012) a decision was made to introduce a new round of changes to the GHG inventory rules. One of the main changes was to make Article 3.4 accounting for forest management mandatory in the second commitment period (2012 – 2017 or 2020). However, a second commitment period for the Kyoto Protocol has not been agreed upon by the signatories to the Kyoto Protocol, and the Australian Government has not decided whether to participate in the second commitment period.

Accounting for forest management under Article 3.4 will be based on reference levels whereby credits and debits will be determined on the basis of the deviation of net emissions from the reference level. The reference level is pre-set and represents an estimate of the forest management net emissions over the commitment period assuming no change in policies since 31 December 2009 (UNFCCC 2010). Rules for setting the reference level allow for some options, such as the treatment of natural disturbances. Australia has used the option for natural disturbances where net emissions from natural disturbances can be excluded above a pre-set disturbance baseline. The Australian Government has determined a forest management reference level of 4.7 Mt CO_2-e yr^{-1} over the period 2012 – 2020, which was submitted at Durban. This calculation was derived from carbon stock changes in the live biomass and debris pools in native forests using the mean national harvest rate from 2002 – 2009, and carbon stock changes in the harvested wood products pool using the 2008 wood production levels (Australian Government 2011). Under this approach, carbon credits are achieved if the net emissions during the commitment period are below the reference level.

Under the Durban proposal, carbon credits from forest management will be subject to a cap equal to 3.5% of base year (1990) emissions (excluding LULUCF in Article 3.3), which equates to a limit of 14.6 Mt CO_2-e yr^{-1} for Australia over the second commitment period; Australia's 1990 base year estimate under the Kyoto Protocol is 416.2 Mt CO_2 –e (UNFCCC 2009) (Macintosh 2012). Reductions in emissions due to changes in forest management in excess of this limit would not count towards Australia's emissions reduction target under these new LULUCF rules. An estimate of the net carbon stock change over the period 2012 - 2020 that is derived from native forests in Australia with the rate of harvesting remaining at 2010 levels is a net sequestration or removal of 12 Mt CO_2-e yr^{-1} (Macintosh 2011c). While there are uncertainties associated with this estimate this, albeit approximate estimate, suggests that the new rules would provide only modest incentive to change forest management practices in ways that encourage forest protection of and associated avoided emissions.

The impact of the LULUCF rules agreed in Durban on forest policy in Australia will depend on (i) whether Australia continues to participate in the Kyoto Protocol, and (ii) whether in fact a second commitment period does go ahead. Australia had indicated before Durban that it would not enter a second commitment period of the Kyoto Protocol unless all major emitters participated. The European Union is expected to participate, but other nations such as the Unites States, Canada, Japan and Russia are now unlikely to participate in the Kyoto Protocol.

At Durban a new group was established, the Ad hoc Working Group on the Durban Platform for Enhanced Action that will negotiate, by 2015, a new international agreement that will take effect in 2020. Developed nations are likely to adopt a variety of approaches to accounting and mitigation commitments in the intervening period (Macintosh 2012). Australia may choose an accounting framework that suits it interests and circumstances better than the LULUCF rules. However, for various reasons (practical, diplomatic), there will be pressure for Australia not to stray too far from ongoing UNFCCC negotiated LULUCF decisions. Therefore, we agree with Macintosh's (2012) suggestion that an Australian framework would include forest management, be based on a reference level, and credits would be uncapped.

National mitigation policies

The Australian national legislation of the Clean Energy Act (2011) provides incentives for mitigation by forest management activities in the land sector:

i) Land carbon credits that are Kyoto compliant can be traded as offsets for emissions, and non-Kyoto credits can be traded on the voluntary market or be purchased by the government.
ii) Protecting and restoring native ecosystems for the co-benefits of biodiversity and carbon is being funded under the Biodiversity Fund. Permitted activities include restoring carbon stocks, avoiding degradation, and facilitating natural adaptation of biodiversity.

Carbon credits under the Carbon Farming Initiative (CFI Act 2011) are termed Australian Carbon Credit Units (ACCUs) and are divided into two types depending on their Kyoto compliance or not. Kyoto ACCUs are issued for activities that meet Australia's mitigation targets, and can be used to meet carbon pricing scheme liabilities under the Clean Energy Act 2011. Non-Kyoto ACCUs are issued for non-compliant activities and can be sold in voluntary markets or purchased by the Australian Government through the CFI Non-Kyoto Carbon Fund. The carbon emissions permits will have a fixed price of \$23 / tCO_2-e from 1 July 2012 to 30 June 2015, and after 1 July 2015 the price will be determined by the markets of carbon trading.

To gain carbon credits under the CFI, land management activities must demonstrate 'additionality'; that is, the activity is additional to existing activities or management has been changed specifically to provide mitigation benefit. Changing forest management to cease harvesting meets the 'additionality' criteria because it is changing the current management practice that is designated in the Timber Release Plans and the legal obligations for scheduled harvesting under the Regional Forest Agreement. Compliance with this regulation means that the activity is declared an eligible offsets project.

Carbon credits from the project will be determined on the basis of a baseline set in accordance with a methodology approved by the Domestic Offsets Integrity Committee. At present no relevant methodology has been published by the Department of Climate Change and Energy Efficiency for forest management projects involving the cessation of logging in native forest and avoidance of emissions. This baseline should represent an estimate of net emissions in the absence of the project, which in this case means the emissions from continued logging. Hence, the CFI baseline differs from the Kyoto Protocol reference level in terms of the time periods covered and activities accounted. The reference level is a past average (2002 – 2009) harvest rate to which activities are compared. The baseline is a projected future level to which new activities are compared. Kyoto ACCUs issued for forest management projects, therefore, should be a sub-component of the forest management credits recorded against Australia's mitigation targets.

Avoiding emissions by ceasing the forest logging that had been planned would comply with Kyoto ACCUs and would contribute to reducing net emissions from the 'uncovered sector' (that is, not subject to the carbon pricing scheme). This would lower the economic cost associated with Australia meeting international mitigation commitments, if the accounting rules included forest management without a cap.

Comparison of net emissions

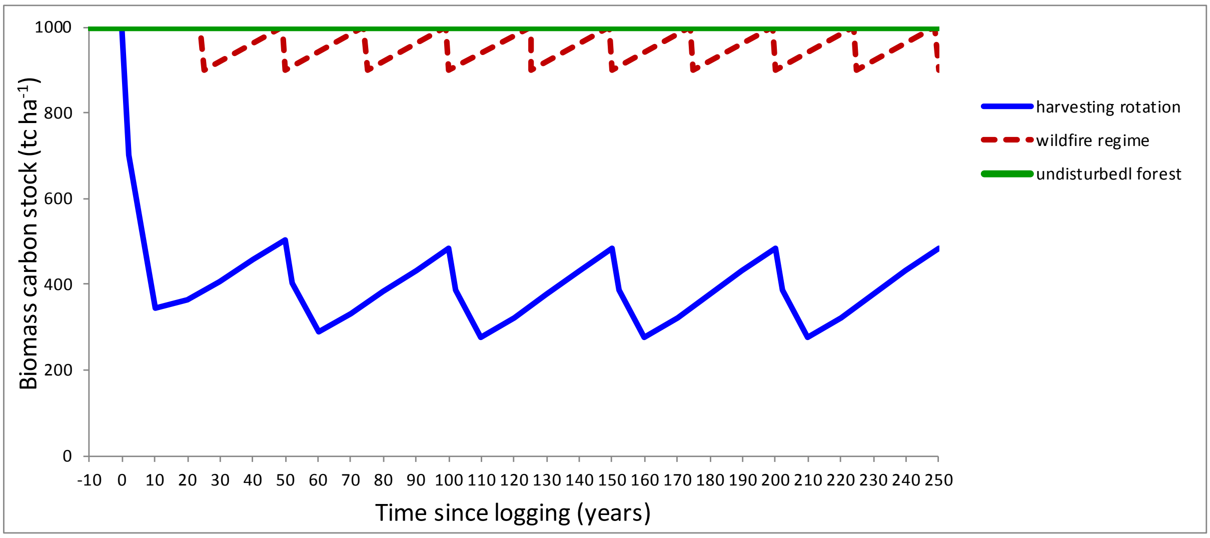
The emissions from logging the approximately 1,500 ha of the agreed future harvesting area were projected to be 0.42 Mt C or 1.55 Mt CO_2 over 5 years, and hence about 0.31 Mt CO_2 yr^{-1}. This amount can be compared with the emissions from a coal-fired power station of 6.2 Mt CO_2 yr^{-1} (calculated for Stanwell Power Station in Queensland, a black coal-fired power station with 1434 MW capacity and generated 8063 GWh of electricity in 2009-10 – calculated by Macintosh 2012). Australia's mitigation commitment of a 5% emission reduction target for 2020 is 92 Mt CO_2-e yr^{-1} or a cumulative target of 737 Mt CO_2 over 2013 – 2020 (Australian Treasury 2011).

7.3 Fire management

Wildfire affects the carbon budget of forest ecosystems due to losses of carbon stocks and transfers between carbon pools. Our analyses revealed that between 6 - 14% (depending on fire severity and forest age) of total forest ecosystem carbon stocks were combusted during wildfire and emitted to the atmosphere. The 2009 wildfire burnt about one-quarter of the catchment area and resulted in an estimated emission of 2.42 MtC or 4% of the total biomass carbon stock. Much of the biomass carbon combusted was in relatively short-lived pools in the canopy, litter layer and shrubs, which have a rapid turnover and regenerate quickly. Carbon stocks in some biomass components that remained after fire were transferred from living to dead biomass pools. This transfer changed the dynamics of the carbon cycle in the ecosystem in terms of rates of growth resulting in gains of carbon and rates of decomposition and respiration resulting in losses of carbon. However, our data suggested that the impact of wildfire on total carbon stocks was relatively small across the landscape and over the timescales of the frequencies of major wildfires.

The difference in the dynamics of the carbon stock at a site between a forest managed on a 50 year harvesting rotation and a forest subject to frequent wildfire events (25 year frequency) are illustrated in Figure 7.1. The biomass carbon stock for the harvesting rotation is the same as that calculated for Figure 6.3, with 0.4 of the biomass removed off-site and 0.5 of the slash combusted. Short-lived (< 3 years) products constitute 0.63 of the total carbon stock and the amounts of wood products remaining at the end of a rotation were negligible. The carbon loss from a site during wildfire was an average of 0.1 of the total and this would be restored during the inter-fire period by regeneration. A fire frequency of 25 years is a very short return interval at a site, but has been used to illustrate a maximum effect of fire on carbon stocks. This scenario is not indicative of the regenerative capacity of the forest at this fire frequency.

Figure 7.1. Comparison of the biomass carbon stock dynamics of a forest site under regimes of frequent wildfire or harvesting rotations (clearfelling and slash burning). Source data for the scenarios from Figure 6.3 and Table 5.7.



7.4 Old growth forest

Old growth forest was identified and mapped under the Regional Forest Agreement (RFA) process (DSE 1996). Within the catchment areas, the mapped old growth in all forest types was 10940 ha or 8.1% of the area in 2003 (Map 12) mainly in the O'Shannassy and Maroondah catchments. The area of wet forest old growth (classes montane damp forest, montane wet forest, wet forest, damp forest, riparian forest) (DSE 1996) was 6860 ha, mainly in the O'Shannassy and Watts River (southern Maroondah) catchments. This represented 1.8% of the total area of these vegetation classes.

The area of old growth in all forest types remaining after the 2009 wildfire was 5700 ha or 4.2% of the area (Map 14) (Table 7.1). The areas of old growth in the larger region of the ash forest type in Victoria in 2003 and 2009 are shown in Maps 13 and 15, respectively. Low severity fire in montane ash forests does not kill the trees and hence does not necessarily negate the old growth status of the forest. It appears that some areas of old growth forest within the low fire severity footprint area remained classified as old growth while other areas were removed. This likely reflects the variability in fire severity and impact on the forest, as well as a level of uncertainty in classification of old growth.

Based on the spatial information about harvesting and wildfire disturbance history, there are no areas within the study region that have experienced no fire or logging events. However, the whole study region was included within the boundaries of the 1939 fire and it is known that not all areas burnt or burnt at a severity sufficient to kill all trees in affected stands. There are no detailed maps of the area burnt by the 1939 wildfire and the outer boundaries were extracted from historical records (DSE 1996). The existing old growth within the study area must have survived this fire.

Old growth status of forest areas used for mapping was based on information from remote sensing, field survey and archival research. Age of forest stands was assessed indirectly through the relative stages of growth of the overstorey with a requirement for the senescing stage of trees in an old growth stand. The senescing stage was defined using a surrogate of crown form (size, shape and composition) that could be mapped by aerial photography (DSE 1996).

The areas proposed for logging under the Timber Release Plans show that a very small area occurs in the mapped old growth zone; 5.03 ha in Armstrong Creek and 3.05 ha in Thomson stages 1 and 2. The overlap of these small areas could be real or could represent inaccuracies in the boundaries of either map.

Table 7.1. Area of old growth forest mapped by DSE in 2003 and after the 2009 wildfire.

Catchment	Area of old growth (ha)		Proportion of total area	
	2003	2009	2003	2009
Armstrong	25	0	0.6	0
Cement Creek	40	40	4.9	4.9
Maroondah	1644	1571	9.2	8.8
McMahon’s Creek	13	13	0.3	0.3
Starvation Creek	0	0	0	0
O’Shannassy	5375	368	40.7	2.8
Upper Yarra	844	815	2.5	2.4
Thomson stage 3	688	635	5.1	4.7
Thomson stage 1, 1A,2	2306	2252	7.0	6.8
Tarago	7	7	0.06	0.06
Total catchments	10941	5701	8.1	4.2
Total study region	24204	8763	4.5	1.6

Areas of old growth remaining in the montane ash forests are very small. These forests have many conservation values in addition to being among the world's most carbon-dense forest (Keith *et al.* 2009b), including as a unique structural type of vegetation with the world's tallest flowering plant (Beale 2007), biodiversity, habitat for rare fauna, and aesthetic qualities, all of which are critically endangered given the small area remaining.

7.5 Synopsis

It is likely that Australia will count forest management towards its national mitigation commitments post-2012, however, there is uncertainty about the rules for the accounting framework. Inclusion of forest management in the national accounts will mean that ceasing logging could generate forest management carbon credits that could be used to offset emissions in other sectors.

Changing forest management policy to avoid emissions from logging will contribute to the global objective of reducing atmospheric GHG emissions. The extent to which changing forest management will generate carbon credits that can be used as offsets and added to meeting Australia's mitigation target will depend on the accounting rules. It is not known what options for accounting rules will be adopted by Australia, but a new accounting framework may be considered.

From the Victorian state perspective, changing land management activities to avoid emissions and increase sequestration of carbon contributes to commitments made regarding targets for reducing emissions. The estimated carbon credits for the catchment areas that are agreed for future logging are 84,420 t C yr^{-1} or 309,540 t CO_2 yr^{-1} of emissions from 295 ha harvested per year, which is equivalent to a carbon emissions density of 286 tC ha^{-1}. By comparison, the estimated total emissions from the 2009 wildfire, were 2,420,000 tC over the 138,500 ha that were burnt, which is equivalent to a carbon emissions density of 17 tC ha^{-1}. Hence, an equivalent emission to that created by the wildfire would result from 29 years of logging over about 6% of the area that was burnt. It is also relevant to note that logging emissions are from long-lived carbon stocks in wood which take decades to centuries to recover. Emissions from wildfires tend to be from short-lived pools which are replenished in less than a decade.

In the international arena, there is growing support for full carbon accounting of all land use types and management activities in national and international carbon accounts. Consequently, land carbon accounting rules are under revision since the Durban conference (UNFCCC 2012). It is likely that the international community will move to a comprehensive accounting system for the whole land sector as it is increasingly being recognized that this is needed to better understand and quantify the role of the land sector in the global carbon cycle and its potential for mitigation of human-forced climate change. The data presented for the carbon budget for the montane ash forests of the Central Highlands is an example of the type of data required if such comprehensive accounting is to inform policy and management decisions.

References

Ambrose G.J. 1982. An ecological and behavioural study of vertebrates using hollows in eucalypt branches. PhD thesis, LaTrobe University, Melbourne.

Anderson MG (2005) *Encyclopedia of hydrological sciences*. J. Wiley, Chichester, England.

Andrew M.H., Noble I.R., Lange R.T. 1979. A non-destructive method for estimating the weight of forage on shrubs. *Aust. Rangelands J*. 1(3): 225-231.

Ashton D. H. 1975a. Studies of litter in *Eucalyptus regnans* forests. *Aust. J. Bot*. 23: 413 – 433.

Ashton D.H. 1975b. The seasonal growth of *Eucalyptus regnans* F. Muell. *Aust. J. Bot*. 23: 239 – 252.

Ashton D.H. 1976a. Phosphorus in forest ecosystems at Beenak, Victoria. *Journal of Ecology* 64: 171 – 186.

Ashton D.H. 1976b. The development of even-aged stands of *Eucalyptus regnans* (F.Muell.) in Central Victoria. *Australian Journal of Botany* 24: 397 – 414.

Ashton D. H. 2000. The Big Ash forest, Wallaby Creek, Victoria – changes during one lifetime. *Aust. J. Bot*. 48: 1 – 26.

Ashton D.H. and Bassett O.D. 1997 The effect of foraging by the superb lyrebird (*Menura novae-hollandiae*) in *Eucalyptus regnans* forests at Beenak, Victoria. *Australian Journal of Ecology* 22: 383 – 394.

Attiwill P.M. 1979. Nutrient cycling in a *Eucalyptus obliqua* (L'Herit) forests. III Growth, biomass and net primary production. *Aust. J. Botany* 27: 439-458.

Attiwill P.M. 1992. Productivity of *Eucalyptus regnans* forest regenerating after bushfire. *South African Forestry Journal* 160: 1-6.

Australian Government 2011a. Submission to the AWG-KP, September 2011, Forest Management Reference Level Submission, UNFCCC Secretariat, Bonn, Germany. http://unfccc.int/bodies/awg-kp/items/5896.php

Australian Government 2011b. Clean Energy Act November 2011. http://www.comlaw.gov.au/Details/C2011A00131

Australian Government 2011c. Carbon Credits (Carbon Farming Initiative) Act May 2012. http://www.comlaw.gov.au/Details/C2012C00417

Australian Paper Industry 2004. Production Statistics 2003-04. http://www.a3p.asn.au/admin/assets/pdf/Stats/Statistics2003-04PaperProduction.pdf

Australian Treasury 2011. *Strong growth, low pollution: modelling a carbon price*. Australian Government, Canberra.

Banks J.C.G. 1993. Tree ring analysis of two Mountain Ash trees *Eucalyptus regnans* (F.Muell.) from the Watts and O'Shannassy catchments, Central Highlands, Victoria. Report to the Central Highlands Old Growth Forest Project, Department of Conservation and Natural Resources, Melbourne.

Beale R. 2007. *If Trees Could Speak. Stories of Australia's Greatest Trees*. Allen & Unwin, Sydney.

Beets P.N., Brandon A.M., Goulding C.J., Kimberley M.O., Paul T.S.H., Searles N. 2011. The inventory of carbon stock in New Zealand's post-1989 planted forest for reporting under the Kyoto Protocol. *Forest Ecology and Management* 262: 1119 – 1130.

Berry SL, Roderick ML 2002. Estimating mixtures of leaf functional types using continental-scale satellite and climatic data. *Global Ecology and Biogeography*, **11**, 23-40.

Berry S, Mackey BG, Brown T 2007. Potential applications of remotely sensed vegetation greenness to habitat analysis and the conservation of dispersive fauna. *Pacific Conservation Biology*, **13**(2), 120-127.

Bi H., Turner J., Lambert M.J. 2004. Additive biomass equations for native eucalypt forest trees of temperate Australia. *Trees* 18: 467 – 479.

Bootle, K.R. (2005) *Wood in Australia: types, properties and uses.* McGraw-Hill, Sydney

Borough C.J., Brown W.E., Brown A.G., Incoll W.D., May J.R., Bird T. 1984. Yield statistics. Ch 10. In: Hillis W.E. and Brown A.G. *Eucalypts for Wood Production*. CSIRO, Academic Press, Sydney. pp201 – 228.

Bradstock R.A., Gill A.M., Williams R.J. 2012. Flammable Australia: fire regimes, biodiversity and ecosystems in a changing world. CSIRO Publishing, Collingwood, 333 pp.

Burgman M.A. 1996. Characterisation and delineation of the eucalypt old-growth forest estate in Australia: a review. *Forest Ecology and Management* 83(3): 149 – 161.

Campbell R.G. 1997. Evaluation and development of sustainable silvicultural systems for mountain ash forests. Discussion Paper. In VSP Technical Report No. 28. Forest Service, Department of Natural Resources and Environment, Melbourne.

Carey E.V., Sala A., Keane R., Callaway R.M. 2001. Are old forests underestimated as global carbon sinks? *Global Change Biology* 7: 339 – 344.

Chafe S.C. 1985. The distribution and interrelationship of collapse, volumetric shrinkage, moisture content and density in trees of *Eucalyptus regnans* F. Muell. *Wood Science and Technology*, 19: 329-345.

Commonwealth of Australia (Geoscience Australia) (2010) Digital Elevation Models User Guide. 1 second DSM, DEM & DEM-S. 3 second DSM, DEM & DEM-S. Version 1.0.3. Geoscience Australia, Canberra.

Conant R.T. 2011. Sequestration through forestry and agriculture. *WIREs Climate Change* 2(2): 238 – 254.

Costermans L. 1994. *Native Trees and Shrubs of South-eastern Australia*. 2nd edn. Rigby, Sydney.

Dean C., Roxburgh S.H., Mackey B. 2003. Growth modelling of Eucalyptus regnans for carbon accounting at the landscape scale. In: Amaroo A., Reed D., Soares P. (eds.) *Modelling Forest Systems*, CAB International, p27 – 39.

Dean C. and Roxburgh S.H. 2006. Improving visualisation of mature, high-carbon sequestering forests. *FBMIS* 1: 48-69.

Department of Agriculture, Fisheries and Forestry 1992. National Forest Policy Statement. <http://www.daff.gov.au/rfa/publications/nfp-statement> (accessed 8/5/2012)

Department of Agriculture, Fisheries and Forestry 1997. Regional Forest Agreements. <http://www.daff.gov.au/rfa> (accessed 8/5/2012)

Department of Agriculture, Fisheries and Forestry 1998. Regional Forest Agreement for Central Highlands region Victoria. <http://www.daff.gov.au/rfa/regions/vic-centralhighlands> (accessed 8/5/2012)

Department of Climate Change and Energy Efficiency 2010. Australian National Greenhouse Accounts. National Inventory Report Vol. 2. , Australian Government, Canberra.

Department of Climate Change and Energy Efficiency 2011. The Critical Decade: Climate science, risks and responses. Climate Commission, Commonwealth of Australia, Canberra.

Department of Primary Industries, Victoria, DPI 2009. Victoria's Timber Industry Strategy, State of Victoria. http://www.dpi.vic.gov.au/__data/assets/pdf_file/0011/14150/TIS-Revised-Version-21-Dec-09.pdf (accessed 25/5/2012)

Department of Primary Industries, Victoria, DPI 2012. Wood Utilisation Plan: Central Forest Management Area 2012-13 to 2013-14.

Department of Sustainability and Environment 1996. Study of old growth forest in Victoria's Central Highlands. http://www.dse.vic.gov.au/forests/publications/research-reports/study-of-old-growth-forests (accessed 8/5/2012)

Department of Sustainability and Environment 2004. Victoria's Statewide Forest Resource Inventory – Field Inventory Procedures, Parks and Forests Report Series 04-1, Victorian Government.

Department of Sustainability and Environment 2007. Code of Practice for Timber Production. State of Victoria.

Department of Sustainability and Environment 2007. Victoria's Statewide Forest Resource Inventory – Central, Central Gippsland and Dandenong Forest Management Areas, Natural Resources Report Series 07-1, Victorian Government.

Department of Sustainability and Environment (2009) Monitoring Annual Harvesting Performance in Victoria's State Forests Technical Reports for Forest Management Areas of Central, Dandenong and Central Gippsland.

Department of Sustainability and Environment (2012) Forest Explorer Map. http://nremap-sc.nre.vic.gov.au/MapShare.v2/imf.jsp?site=forestexplorer (accessed 16/5/2012)

Diochon A., Kellman L., Beltrami H. 2009. Looking deeper: An investigation of soil carbon losses following harvesting from a managed northeastern red spruce (*Picea rubens* Sarg.) forest chronosequence. *Forest Ecology and Management* 257: 413 – 420.

Feller M.C. 1980 Biomass and nutrient distribution in two eucalypt forest ecosystems. *Aust. J. Ecology* 5: 309-333.

Flinn D., Squire R., Wareing K. 2007. Victoria. Ch 3 in Forest Management in Australia: Implications for Carbon Budgets, eds. R.J. Raison and R.O. Squire. National Carbon

Accounting System Technical Report No. 32, Australian Greenhouse Office, Canberra, pp103 – 146.

Friedlingstein P, Houghton RA, Marland G, Hackler J, Boden TA, Conway TJ, Canadell JG, Raupach MR, Ciais P, Le Quéré C. Update on CO2 emissions**.** *Nature Geoscience*, DOI 10.1038/ngeo_1022, Online 21 November 2010. http://www.nature.com/ngeo/journal/vaop/ncurrent/full/ngeo1022.html

Gallant J.C. and Dowling T.I. 2003. A multi resolution index of valley bottom flatness for mapping depositional areas. *Water Resources Research*, **39**(12), 1347-1361.

Gibbons P. and Lindenmayer D.B. 2002 *Tree hollows and wildlife conservation in Australia*. CSIRO Publications, Collingwood.

Gibbons P., Lindenmayer D.L., Barry S.C., Tanton M.T. 2000. Hollow formation in eucalypts from temperate forests in southeastern Australia. *Pacific Conservation Biology* 6: 218-228.

Gibbons P., McElhinny C., Lindenmayer D.B. 2010. What strategies are effective for perpetuating structures provided by old trees in harvested forests? A case study on trees with hollows in south-east Australia. *For. Ecol. Manage*. 260: 975-982.

Gifford R.M. 2000. Carbon contents of above-ground tissues of forest and woodland trees. National Carbon Accounting System Technical Report No. 22, Australian Greenhouse Office, Canberra.

Gilbert J.M. 1958. Eucalypt-rainforest relationships and the regeneration of eucalypts. PhD Thesis, Botany Department, University of Tasmania, Australia.

Gill A.M. 1981. Adaptive responses of Australian vascular plant species to fires. In: *Fire and the Australian Biota*. Eds. A.M. Gill, R.H. Groves, I.R. Noble. Australian Academy of Sciences, Canberra. pp. 243 – 271.

Gould J. and Cheney P. Fire Management in Australian Forests, Ch 8 In: *Forest Management in Australia: Implications for Carbon Budgets*, eds. R.J. Raison and R.O. Squire. National Carbon Accounting System Technical Report No. 32, Australian Greenhouse Office, Canberra, pp 341 - 368.

Green G. 2002. Esperence 74D (EPO74D) Logging coupe inventory. Timber workers for forests. Web publication available at http://www.twff.com.au/epo74d.pdf

Grierson P.F., Adams M.A., Attiwill P.M. 1992. Estimates of carbon storage in the above-ground biomass of Victoria's forests. *Aust. J. Bot*. 40: 631-640.

Griffiths T. 1992. *Secrets of the Forest: discovering history of Melbourne's Ash Range*. Allen and Unwin, Sydney.

Gupta R.K. and Rao D.L.N. 1994. Potential of wastelands for sequestering carbon by reforestation. *Current Science* 66: 378 – 380.

Hickey J.E., Kostoglou P., Sargison G.J. 2000. Tasmania's tallest trees. *Tasforests* 12: 105 – 121.

Hollis J.J., Matthews S., Ottmar R.D., Prichard S.J., Slijepcevic A., Burrows N.D., Ward B., Tolhurst K.G., Anderson W.R., Gould J.S. 2010. Testing woody fuel consumption models for application in Australian southern eucalypt forest fires. Forest Ecology and Management 260: 948 – 964.

Houghton R.A. 2007. Balancing the global carbon budget. *Annual Review of Earth and Planetary Science* 35: 313 – 347.

Houghton R.A. 2008. Carbon flux to the atmosphere from land-use changes: 1850-2005. TRENDS: A Compendium of Data on Global Change. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy, Oak Ridge, Tenn., USA. [http://cdiac.ornl.gov/trends/landuse/houghton/houghton.html] (accessed 19/4/2012)

Husch B., Beers T.W., Kershaw J.A., Hoboken N.J. 2003. *Forest Mensuration*. Wiley, Chichester, 443p.

Hutchinson M.F. 2005. ANUCLIM ver. 5.1. Centre for Resource and Environmental Studies, The Australian National University. <http://cres.anu.edu.au/outputs/anuclim.php.>

Hutchinson M.F., Stein J, Stein J.L., Anderson H., Tickle P. 2008. Geodata 9 second DEM and D8 User Guide, Fenner School of Environment and Society, ANU and Geoscience Australia. <http://fennerschool.anu.edu.au/publications/software/>

Illic J., Boland D., McDonald M., Downes G., Blakemore P. 2000. Woody density phase 1 – state of knowledge. . National Carbon Accounting System Technical Report No. 18, Australian Greenhouse Office, Canberra.

Jaakko Pöyry Consulting 1999. Usage and lifecycle of wood products. National Carbon Accounting System, Technical Report No. 8, Australian Greenhouse Office, Commonwealth of Australia, Canberra

Jaakko Pöyry Consulting 2000. Analysis of wood product accounting options for the nation accounting system. National Carbon Accounting System, Technical Report No. 24, Australian Greenhouse Office, Commonwealth of Australia, Canberra.

Jacobs M. 1955. *Growth Habits of the Eucalypts*. Forestry and Timber Bureau, Canberra.

Janisch J.E. and Harmon M.E. 2002. Successional changes in live and dead wood carbon stores: implications for net ecosystem productivity. *Tree Physiology* 22: 77 – 89.

Keith H. 1991. Effects of fire and fertilization on nitrogen cycling and tree growth in a subalpine eucalypt forest. PhD Thesis, Australian National University, Canberra. 369 pp.

Keith H., Raison R.J., Jacobsen K.L. 1997. Allocation of carbon in a mature eucalypt forest and some effects of soil phosphorus availability. *Plant and Soil* 196: 81–99.

Keith H., Barrett D., Keenan R. 2000. Review of allometric relationships for estimating woody biomass for New South Wales, the Australian Capital Territory, Victoria, Tasmania and South Australia. National Carbon Accounting System Technical Report No. 5b, Australian Greenhouse Office, Canberra.

Keith H., Leuning R. Jacobsen K.L., Cleugh H.A., van Gorsel E., Raison R.J., Medlyn B.E., Winters A., Keitel C. 2009a. Multiple measurements constrain estimates of net carbon exchange by a Eucalyptus forest. *Agricultural and Forest Meteorology* 149 (2007) 535 – 558.

Keith H, Mackey BG, Lindenmayer DB (2009b) Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon dense forests. *Proceedings of the National Academy of Science.* 106(28), 11635-11640.

Le Quéré C, Raupach MR, Canadell JG, Marland G *et al.* (2009) Trends in the sources and sinks of carbon dioxide. *Nature Geosciences* 2: 831 – 836. , doi: 10.1038/ngeo689. http://www.nature.com/ngeo/journal/v2/n12/full/ngeo689.html

Lindenmayer D.B., Cunningham R.B., Tanton M.T., Smith A.P., Nix H.A. 1990a. The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-eastern Australia. I.Factors influencing the occupancy of trees with hollows. *Biological Conservation* 54(2): 111 - 131.

Lindenmayer D.B., Cunningham R.B., Tanton M.T., Smith A.P. 1990b. The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-eastern Australia. II. The loss of trees with hollows and its implications for the conservation of Leadbeater's possum *Gymnobelideus leadbeateri* McCoy (Marsupialia, Petauridae). *Biological Conservation* 54(2): 133 – 145.

Lindenmayer D.B., Cunningham R.B., Nix H.A., Tanton M.T., Smith A.P. 1991. Predicting the abundance of hollow0bearing trees in montane ash forests of south-east Australia. *Australian Journal of Ecology* 16: 91 – 98.

Lindenmayer D.B., Cunningham R.B., Donnelly C.F., Tanton M.T., Nix H.A. 1993. The abundance and development of cavities in Eucalyptus trees: a case study in the montane forests of Victoria, southeastern Australia. *For. Ecol. Manag*. 60: 77-104.

Lindenmayer D.B., Mackey B.M., Nix H.A. 1996. Climatic analyses of the distribution of four commercially important wood production eucalypt trees from south-eastern Australia. *Australian Forestry* 59: 11 – 26.

Lindenmayer D.B., Incoll R.D., Cunningham R.B., Donnelly C.F. 1999. Attributes of logs on the floor of Australian Mountain Ash (*Eucalyptus regnans*) forests of different ages. *Forest Ecology and Management* 123: 195 – 203.

Lindenmayer D.B., Cunningham R.B., Donnelly C.F., Franklin J.F. 2000a. Structural features of old growth Australian montane ash forests. *Forest Ecology and Management* 134 (1-3): 189 – 204.

Lindenmayer D.B., Cunningham R.B., Pope M.L., Gibbons P., Donnelly C.F. 2000b. Cavity sizes and types in Australian eucalypts from wet and dry forest types: a simple of rule of thumb for estimating size and number of cavities. *Forest Ecology and Management* 137: 139 – 150.

Lindenmayer D.B. and Franklin J.F. 2002. *Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach.* Island Press, Washington, DC.

Lindenamyer D.B., Claridge A.W., Gilmore A.M., Michael D.R., Lindenmayer B.D. 2002. The ecological role of logs in Australian forests and the potential impacts of harvesting intensification on log-using biota. *Pacific Conservation Biology* 8(2): 121 – 140.

Lindenmayer D.B., Cunningham R.B., MacGregor C., Incoll R.D., Michael D. 2003. A survey design for monitoring the abundance of arboreal marsupials in the Central highlands of Victoria. *Biological Conservation* 110: 161 – 167.

Lindenmayer D.B. and Ough K. 2006. Salvage harvesting in the montane ash forests of the Central highlands of Victoria, south-eastern Australia. *Conservation Biology* 20: 1005 – 1015.

Lindenmayer D.B. 2009. Old forests, new perspectives: insights from the wet ash forests of the Central Highlands of Victoria. *Forest Ecology and Management* 258: 357 – 365.

Lindenmayer D.B. 2009. *Forest Pattern and Ecological Processes: A Synthesis of 25 Years of Research*. CSIRO Publishing, Collingwood, Australia.

Lindenmayer D.B. and Wood J.T. 2010. Long-term patterns in the decay, collapse, and abundance of trees with hollows in the mountain ash (*Eucalyptus regnans*) forests of Victoria, southeastern Australia. *Canadian Journal of Forest Research* 40: 48 – 54.

Lindenmayer D.B., Blair D., McBurney L., Banks S. 2010. *Forest Phoenix: How a great forest recovers after wildfire*. CSIRO Publishing, Melbourne, 128 pp.

Lindenmayer D.B., Hobbs R.J., Likens G.E., Krebs C.J., Banks S.C. 2011. Newly discovered landscape traps produce regime. shifts in wet forests. *Proceedings of the National Academy of Sciences, USA* 108(38): 15887 – 15891.

Lindenamyer D.B., Wood J., McBurney L., Michael D., Crane M., MacGregor C., Montague-Drake R., Gibbons P. 2011. Cross-sectional vs. longitudinal research: a case study of trees with hollows and marsupials in Australian forests. *Ecological Monographs* 81(4): 557–580.

Lindenmayer D.B. et al. 2012. Interacting factors driving a major loss of large trees with cavities in a forest ecosystem. *Public Library of Science 1* (in review).

Liu SF, Retter AJ, Raymond OL, Stanley SP, Phillips D (2005c). Surface Geology of Australia 1:1,000,000 scale, Victoria - 2nd Edition [Digital Dataset], Canberra: Geoscience Australia. . http://www.ga.gov.au

Looby M.J. 2007. Tree hollows, tree dimensions and tree age in *Eucalyptus microcarpa* Maiden (Grey box) in Victoria. Masters Thesis, School of Resource Management, Faculty of Land and Food Resources, University of Melbourne.

Lutze M.T., Campbell R.G., Fagg P.C. 1999. Development of silvicultural in the native state forests of Victoria. *Australian Forestry* 62: 236 – 244.

Luyssaert S., Schulze E-D., Börner A., Knohl A., Hessenmöller D., Law B.E., Cias P., Grace J. 2008. Old-growth forests as global carbon sinks. *Nature* 455: 213 – 215.

McCarthy M.A. and Lindenmayer D.B. 1998. Multi-aged mountain ash forest, wildlife conservation and timber harvesting. *Forest Ecology and Management* 104: 43 – 56.

McCarthy M.A., Gill A.M., Lindenmayer D.B. 1999. Fires regimes in mountain ash forest: evidence from forest age structure, extinction models and wildlife habitat. *Forest Ecology and Management* 124: 193 – 203.

McCaw W.L., Smith R.H., Neal J.E., 1997. Prescribed burning of thinning slash in regrowth stands of karri (*Eucalyptus diversicolor*). 1. Fire characteristics, fuel consumption and tree damage. International Journal of Wildland Fire 7(1): 29 – 40.

Mackensen J. and Bauhus J. 1999. The decay of coarse woody debris. National Carbon Accounting System Technical Report No. 6. Australian Greenhouse Office, Canberra.

Mackensen J. and Bauhus J. 2003. Density loss and respiration rates in coarse woody debris of *Pinus radiata, Eucalyptus regnans* and *Eucalyptus maculata*. *Soil Biology and Biochemistry* 35: 177 – 186.

Mackensen J., Bauhus J., Webber E. 2003. Decomposition rates of coarse woody debris – A review with particular emphasis on Australian tree species. *Aust. J. Bot*. 51: 27 – 37.

McKenzie N., Ryan P., Fogarty P., Wood J. 2001. Sampling, measurement and analytical protocols for carbon estimation in soil, litter ad coarse woody debris. National Carbon Accounting System Technical Report No. 14., Australian Greenhouse Office, Canberra.

McKenzie NJ, Jacquier DW, Maschmedt DJ, Griffin EA, Brough DM (2005) ASRIS Technical specifications Version 1.5, October 2005. <http://www.asris.csiro.au/>

Mackey BG, Lindenmayer DB, Gill AM, McCarthy AM, Lindesay JA (2002) *Wildfire, Fire and Future Climate: A forest ecosystem analysis*. CSIRO Publ., Melbourne.

Mackowski C.W. 1987. Wildlife hollows and timber management in Blackbutt forest. Masters' Thesis, University of New England, Armidale, NSW.

Macintosh A. 2011. Potential carbon credits from reducing native forest harvesting in Australia. ANU Centre for Climate Law and Policy, Working Paper No. 2011/1. ANU, Canberra.

Macintosh A. 2012. Tasmanian Forests Intergovernmental Agreement: An assessment of its carbon value. ANU Centre for Climate Law and Policy, Canberra.

Mattheck C., Bethge K., West P.W. 1994. Breakage of hollow tree stems. *Trees* 9: 47-50.

Mattheck C., Bethge K., Tesari I. (2006). Shear effects on failure of hollow trees. *Trees* 20: 329-333.

Mueck S.G. 1990. The floristic composition of mountain ash and alpine ash forests in Victoria. SSP Technical Report No. 4 Department of Conservation and Environment, Melbourne.

Mueck S.G., Ough K., Banks J.C.G. 1996. How old are wet forest understoreys? *Australian Journal of Ecology* 21: 354 – 348.

NGGI 1997. National Greenhouse Gas Inventory committee. Land Use Change and Forestry, workbook 4.2 (revision 2) for carbon dioxide from the biosphere. Environment Australia.

Noble W.S. 1977 *Ordeal by Fire. The Week a State Burned Up*. Hawthorn Press, Melbourne.

Ogden J. 1978. On the dendrochronological potential of Australian trees. *Aust. J. Ecology*. 3: 339 – 356.

O'Loughlin E.M., Cheney N.P., Burns J. 1982. The bushranger experiment: Hydrological response of a eucalypt catchment. Proceedings of the First National Symposium on Forest Hydrology, Melbourne.

Olsson B.A., Staaf H., Lundkvist H., Bengtsson J., Rosen K. 1996. Carbon and nitrogen in coniferous forest soils after clearfelling and harvests of different intensity. *Forest Ecology and Management* 82: 19 – 32.

Ough K. 2002. Regeneration of wet forest flora a decade after clearfelling or wildfire: is there a difference? *Australian Journal of Botany* 49: 645 – 664.

Pachauri R.K. and Reisinger A. 2007. The fourth Assessment Report: Synthesis Report. Contribution from working Groups I, II and III. Intergovernmental Panel on Climate Change, Geneva.

Phillips O.L., Malhi Y., Higuchi N., Laurance W.F., Nunez P.V., Vasquez R.M., Laurance S.G., Ferreira L.V., Stern M, Brown S., Grace J. 1998. Changes in the carbon balance of tropical forests: evidence from long-term plots. *Science* 282:439 – 442.

Polglase P.J. and Attiwill P.M. 1992. Nitrogen and phosphorus cycling in relation to stand age of *Eucalyptus regnans* F.Muell. *Plant and Soil* 142: 157 – 166.

Rab M.A. 1994. Changes in physical properties of a soil associated with logging of *Eucalyptus regnans* forest in southeastern Australia. *Forest Ecology and Management* 70: 215 - 229.

Rab M.A. 1999 Measures and operating standards for assessing Montreal process soil sustainability indicators with reference to Victorian Central Highlands forest, southeastern Australia. *Forest Ecology and Management* 117: 53 – 73

Rab M.A. 2004 Recovery of soil physical properties from compaction and soil profile disturbance caused by logging of native forest in Victorian Central Highlands, Australia. *Forest Ecology and Management* 191: 329 – 340.

Raison R.J. and Squire R.O. eds. 2007. Forest Management in Australia: Implications for Carbon Budgets. National Carbon Accounting System Technical Report No. 32, Australian Greenhouse Office, Canberra.

Resource Assessment Commission 1992. Forest and Timber Inquiry. Draft Report Vol. 2, Australian Government Publishing Service, Canberra.

Rhemtulla J.M., Miadenoff D.J., Clayton M.K. 2009. Historical forest baselines reveal potential for continued carbon sequestration. *Proceedings of the National Academy of Sciences* 106: 6082 – 6087.

Richards F. 1959. A flexible growth function for empirical use. *J. Experimental Botany* 10: 290 – 300.

Richards G.P. and Brack C.L. 2004. A modelled carbon account for Australia’s post-1990 plantation estate. *Australian Forestry* 67: 289 – 300.

Richards G.P., Borough C., Evans D., Reddin A., Ximenes F., Gardner D. 2007. Developing a carbon stocks and flows model for Australian wood products. *Australian Forestry* 70: 108 – 119.

Roderick M.L. 1999. Estimating the diffuse component from daily and monthly measurements of global radiation. *Agricultural and Forest Meteorology* 95: 169 – 185.

Roxburgh S.H., England J.R., Paul K. 2010. Developing capability to predict biomass carbon in biodiverse plantings and native forest ecosystems. CSIRO National Research Flagships

Sustainable Agriculture. Report prepared for the Victorian Department of Sustainability and Environment, Environmental Policy and Climate Change Branch.

Skog K.E. and Nicholson G.A. 1998 Carbon cycling through wood products: The role of wood and paper products in carbon sequestration. *Forest Products Journal* 48: 75 – 83.

Sillett S.C., van Pelt R., Koch G.W., Ambrose A.R., Carroll A.L., Antoine M.E., Mifsud B.M. 2010. Increasing wood production through old age in tall trees. *Forest Ecology and Management* 259: 976 – 994.

Slijepcevic A. 2001. Loss of carbon during controlled regeneration burns in *Eucalyptus obliqua* forest. *Tasforests* 13: 281 – 290.

Smith A.P. and Lindenmayer D.B. 1988. Tree hollow requirements of Leadbeater's possum and other possums and gliders in timber production ash forests of the Victorian Central Highlands. *Aust. J. Wildl. Res*. 15(4): 347-362.

Smith R.B. and Woodgate P. 1985. Appraisal of fire damage for timber salvage by remote sensing in mountain ash forests. *Australian Forestry* 48: 252 – 263.

Snowdon P., Eamus D., Gibbons P., Khanna P., Keith H., Raison J., Kirschbaum M. 2000. Synthesis of allometrics, review of root biomass and design of future woody biomass sampling strategies. National Carbon Accounting System Technical report No. 17, Australian Greenhouse Office, Canberra.

Snowdon P., Raison J., Keith H., Ritson P., Grierson P., Adams M., Montagu K., Bi H., Burrows W., Eamus D. 2002. Protocol for sampling tree and stand biomass. National Carbon Accounting System Technical report No. 31, Australian Greenhouse Office, Canberra.

Thompson I., Mackey B., McNulty S., Mosseler A. 2009. Forest resilience, biodiversity and climate change. A synthesis of the biodiversity / resilience / stability relationship in forest ecosystems. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series No. 43, 67 pp.

United Nations, 2009. Framework Convention on Climate Change. Report of the review of the initial report of Australia. FCCC/IRR/2077/AUS. UNFCCC Secretariat, Bonn, Germany.

United Nations, 2010. Framework Convention on Climate Change. The Cancun Agreements: Land use, land-use change and forestry, Decision 2/CMP.6. [http://UNFCCC/KP/CMP/2010/12/Add.1]

United Nations, 2012. Framework Convention on Climate Change, Report of the Conference of the Parties serving as the meeting of the Parties to the Kyoto Protocol on its seventh session, held in Durban from 28 November to 11 December 2011. [http://unfccc.int/resource/docs/2011/cmp7/eng/10a01.pdf]

Van Pelt R. 2007. Identifying mature and old forests in Western Washington. Department of Natural Resources, Olympia, Washington State.

Van Wagner C.E. 1968. The line intersect method in forest fuel sampling. *Forest Science* 14: 20-26

Watson F.G.R. and Vertessy R.A. 1996 Estimating leaf area index from stem diameter measurements in Mountain Ash forest. Report 96/7, Cooperative Research Centre for Catchment Hydrology.

Wellington A.B. and Noble I.R. 1985. Post-fire recruitment and mortality in a population of the mallee *Eucalyptus incrassata* in semi-arid, south-eastern *Australia. J Ecol*. 73: 645 – 656.

West P.W., Wells K.F., Cameron D.M., Rance S.J., Turnbull C.R.A., Beadle C.L. 1991. Predicting tree diameter and height from aboveground biomass for four eucalypt species*. Trees Structure and Function* 5: 30 – 35.

West P.W. and Mattay J.P. 1996. Yield prediction models and comparative growth rates for six eucalypt species. *Aust. Forestry*. 56: 210-225.

Wilkes J. 1982. Pattern and process of heart rot in *Eucalyptus microcorys*. *Aust. For*. 45(1) 51: 56.

Wilson J.P. and Gallant J.C. 2000. *Terrain Analysis: principles and applications*. Edited by John P. Wilson and John C. Gallant, Wiley, New York.

Woldendorp G. and Keenan R.J. 2005. Coarse woody debris in Australian forest ecosystems: a review. *Austral Ecology* 30: 834 – 843.

Wood S.W., Hua Q., Allen K.J, Bowman D.M.J.S. 2010. Age and growth of a fire prone Tasmanian temperate old-growth forest stand dominated by *Eucalyptus regnans*, the world's tallest angiosperm. *Forest Ecology and Management* 260: 438 – 447.

Ximenes F. Gardner W.D., Marchant J.F. 2004 Total biomass measurement and recovery of biomass in log products in Spotted Gum (*Corymbia maculata*) forests of SE NSW. National Carbon Accounting System, Technical Report No. 47, Australian Greenhouse Office, Canberra.

Ximenes F.A. and Gardner W.D. 2005.Recovery of biomass as green sawn boards after milling of Spotted Gum (*Corymbia maculata*) sawlogs from NSW south coast forests. National Carbon Accounting System Technical report No. 48, Australian Greenhouse Office, Canberra.

Ximenes F. Gardner W.D., Marchant J.F. 2005. Carbon flow following the harvest of Blackbutt trees and their conversion into sawn products. Forest Resources Research – Research Paper No. 41, pp38. Forests NSW, Sydney.

Ximenes F.A. and Gardner W.D. 2006. The decay of coarse woody roots following harvest in a range of forest types. National Carbon Accounting System Technical report No. 49, Australian Greenhouse Office, Canberra.

Ximenes F., Kapambwe M., Keenan R. 2008a. Timber use in residential construction and demolition. BDP Environmental Design Guide, PRO 36: 1 – 8.

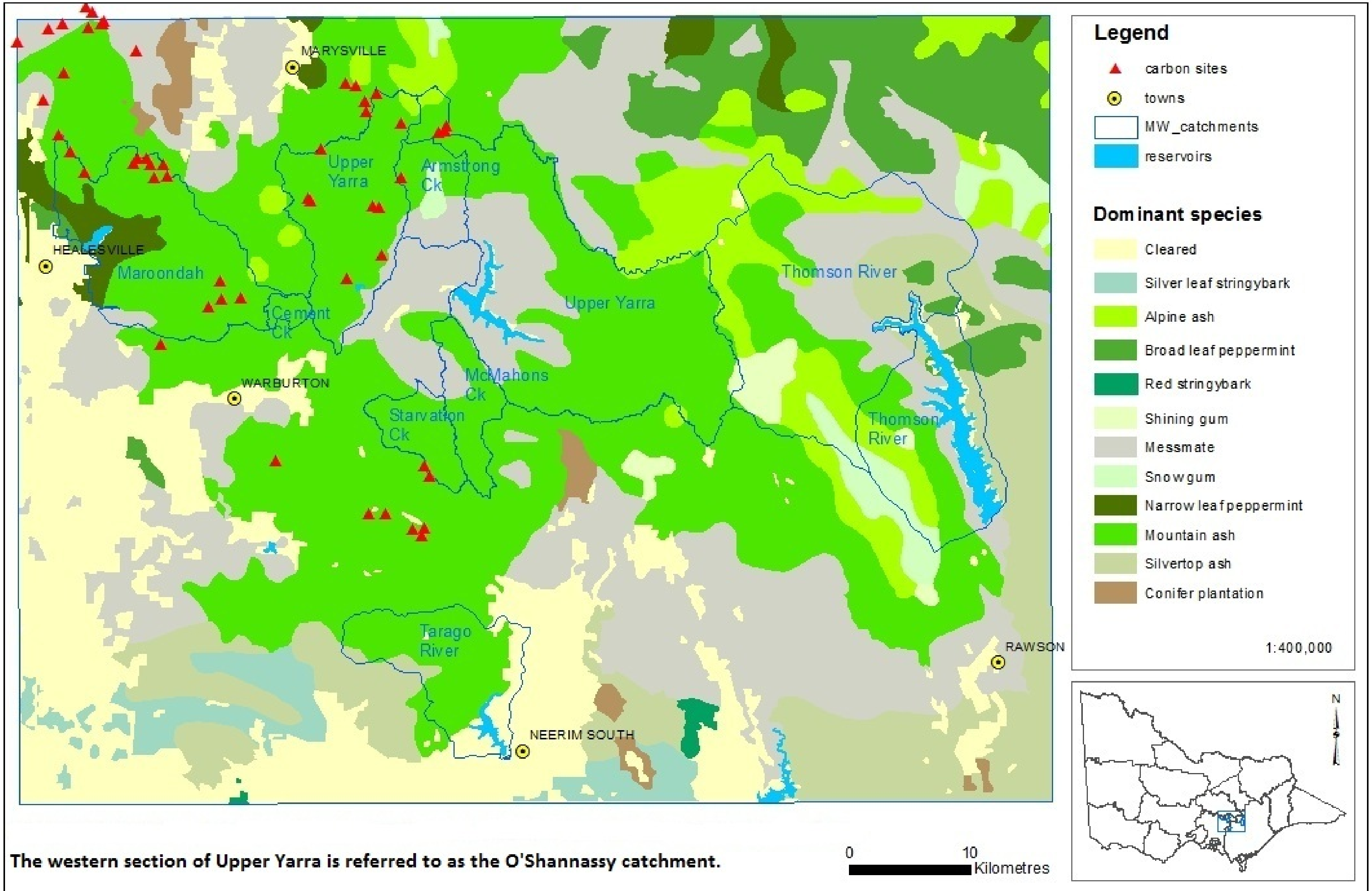
Ximenes X., Gardner W.D., Kathuria A. 2008b. Proportion of aboveground biomass in commercial logs and residues following the harvest of five commercial forest species in Australia. *Forest Ecology and Management* 256: 335 – 346.

Ximenes F., Gardner W.D., Cowie A.L. 2008c. The decomposition of wood products in landfills in Sydney, Australia. *Waste Management* 28: 2344 – 2354.

Zhou G., Liu S., Li Z., Zhang D., Tang X., Zhou C., Yan J., Mo J. 2006. Old-growth forests can accumulate carbon in soils. *Science* 314: 1417.

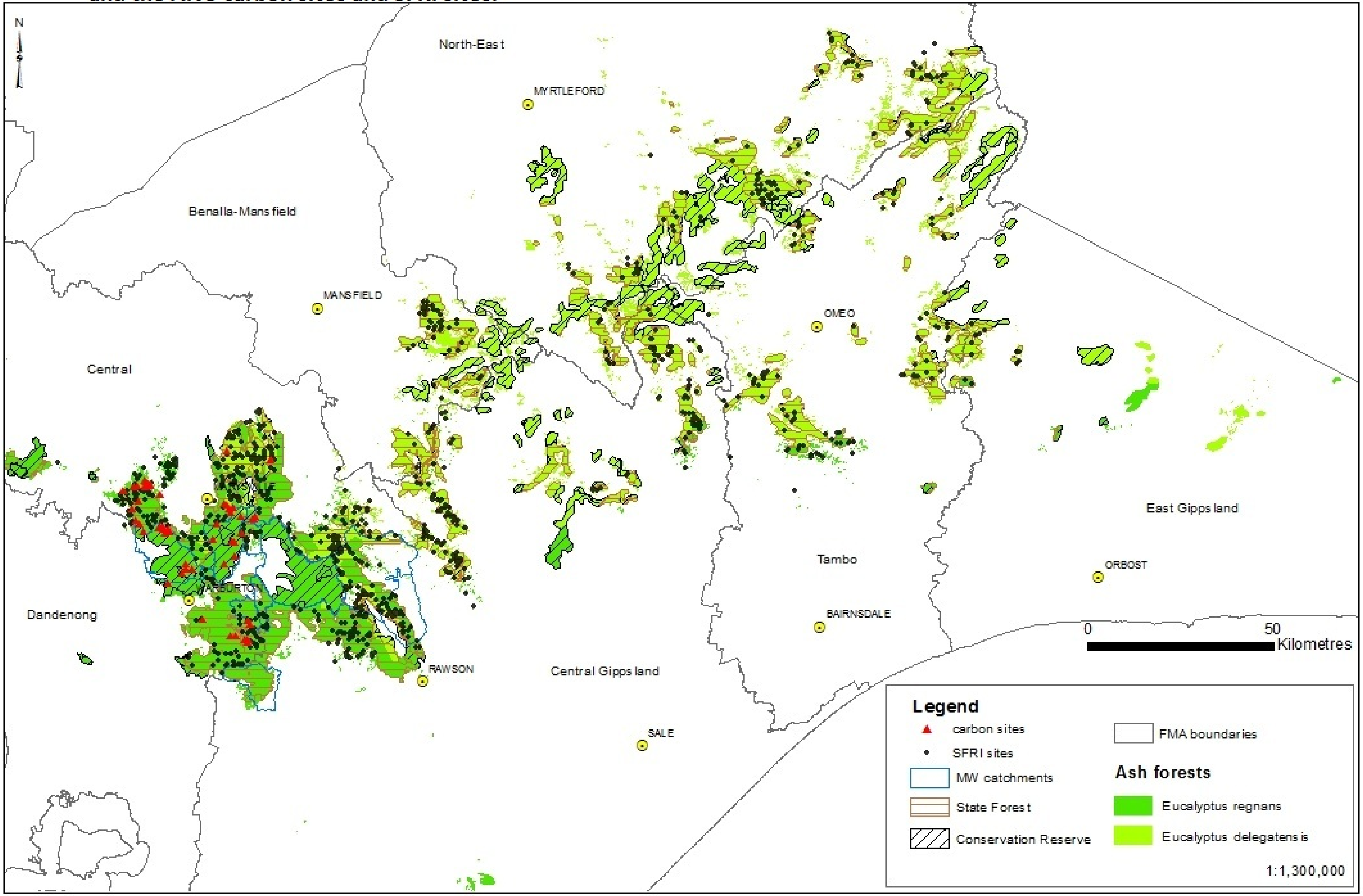
Maps

Map 1: Location of the catchments supplying water to Melbourne, the ANU carbon sites, and forest species distribution.

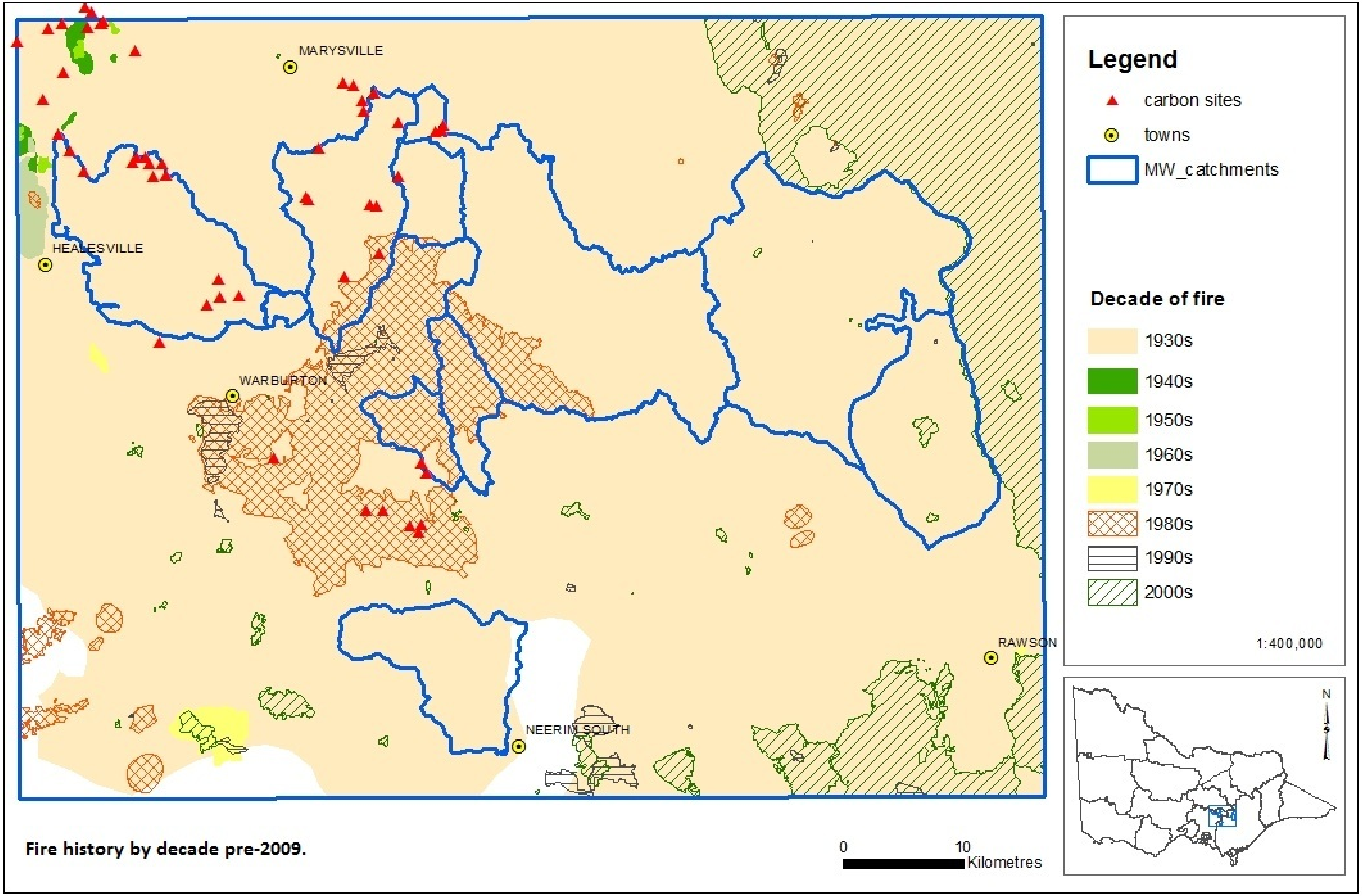


The western section of Upper Yarra is referred to as the O'Shannassy catchment.

Map 2: Location of the ash forest type in the water catchments, State forests and Conservation Reserves in Victoria, and the ANU carbon sites and SFRI sites.

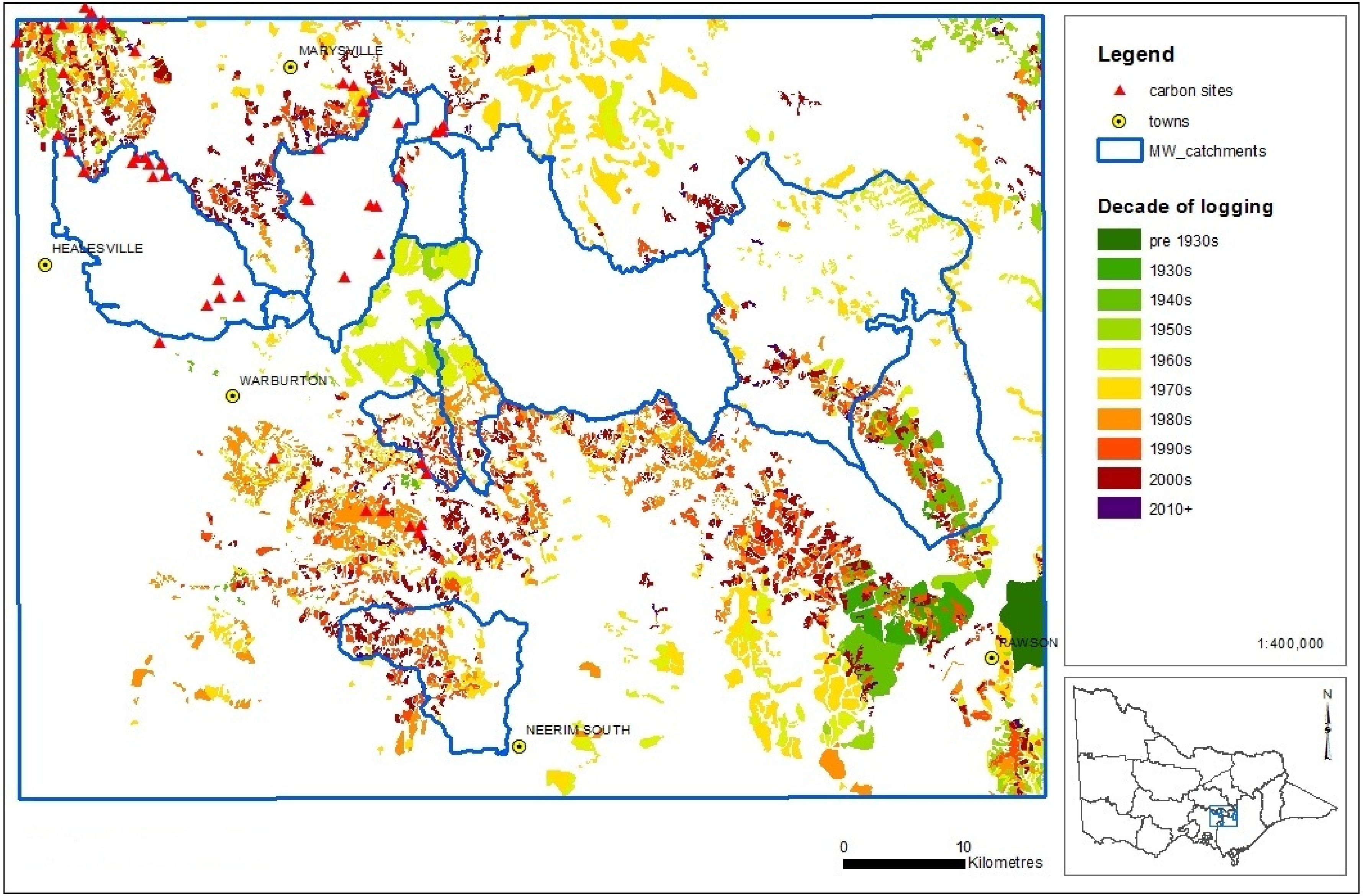


Map 3: Fire history by decade pre-2009 around the water catchments.

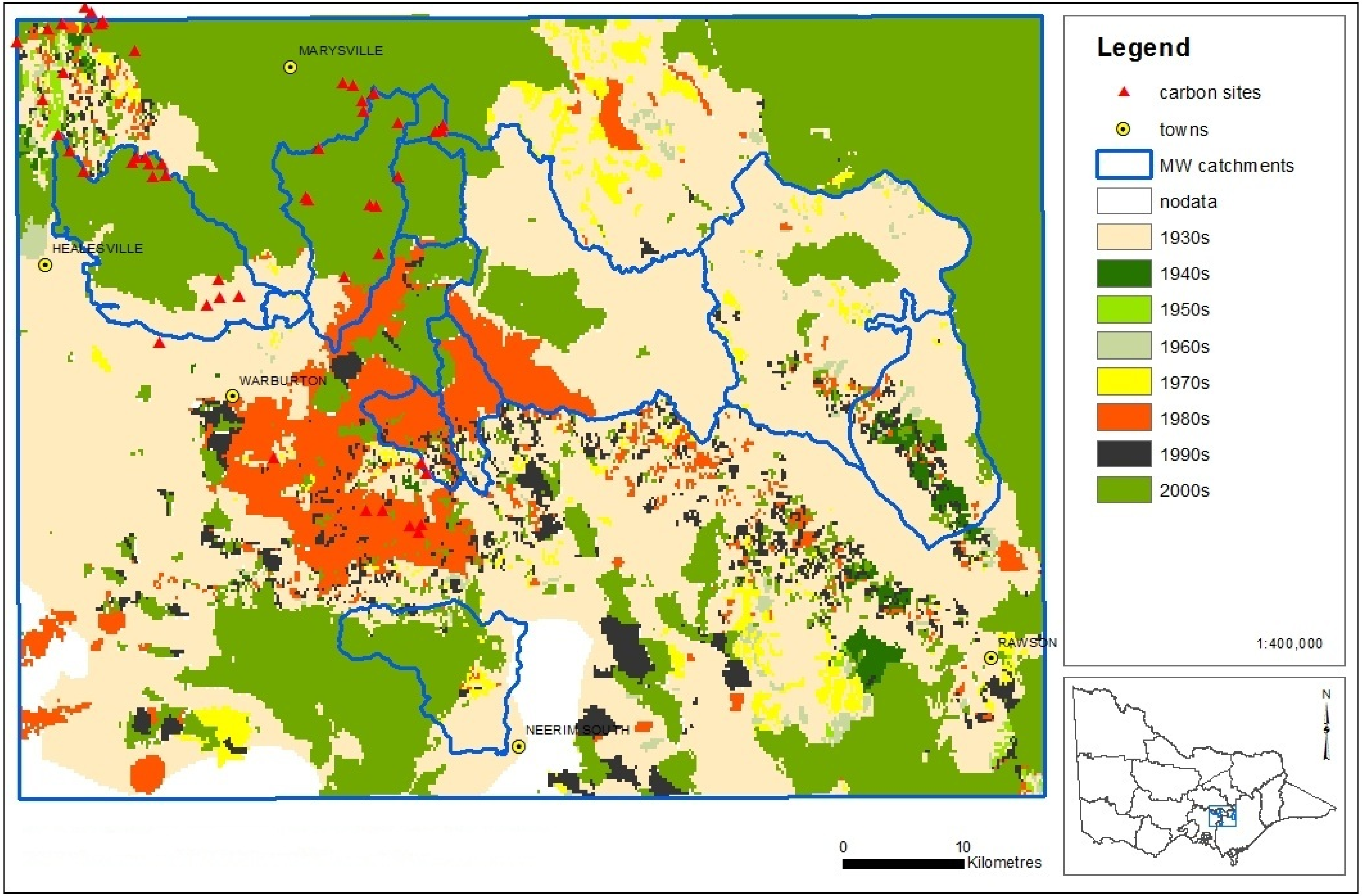


Fire history by decade pre-2009.

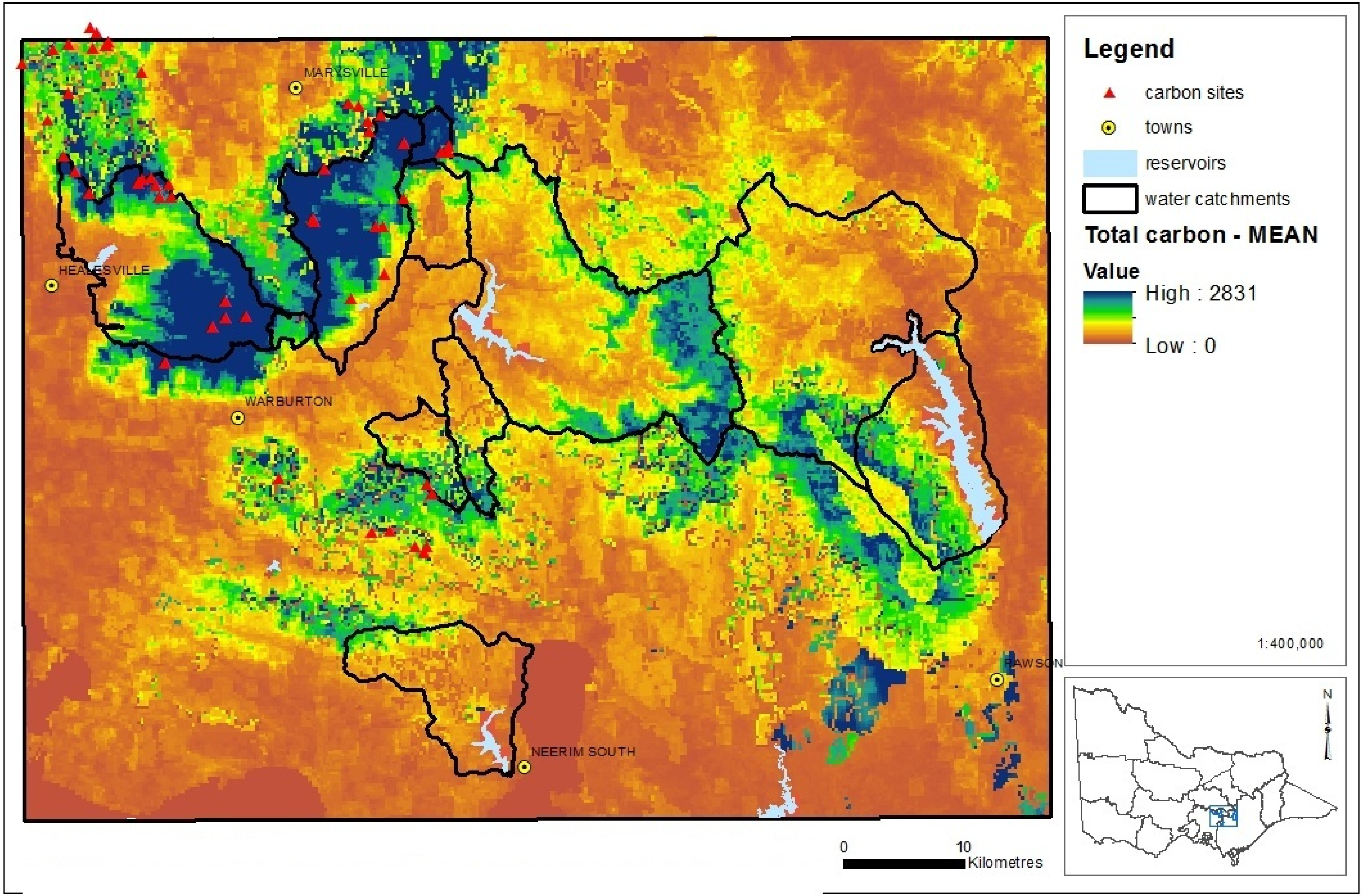
Map 4: Logging history by decade around the water catchments.



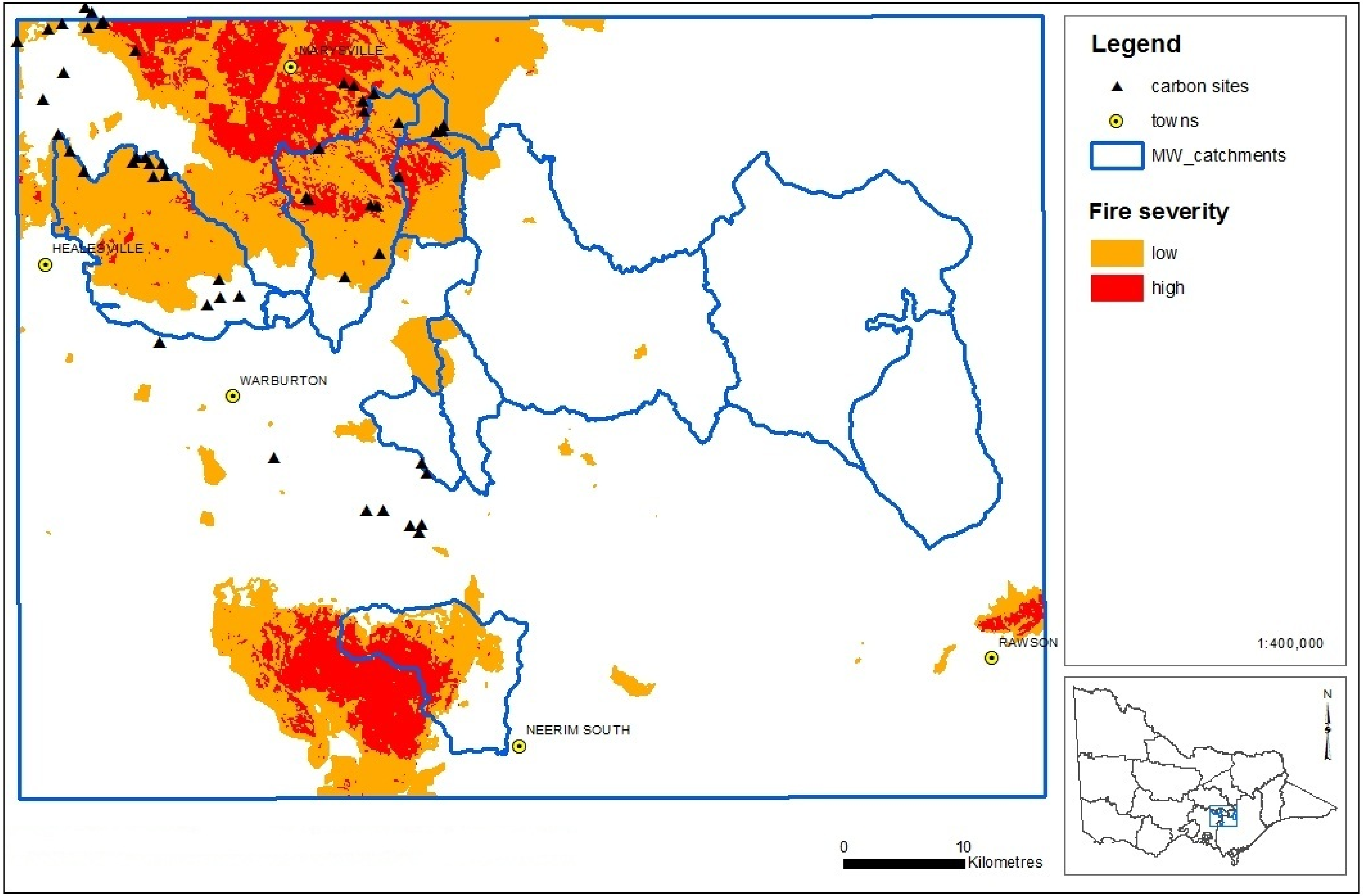
Map 5: Decade of last disturbance event, either logging or fire including 2009.



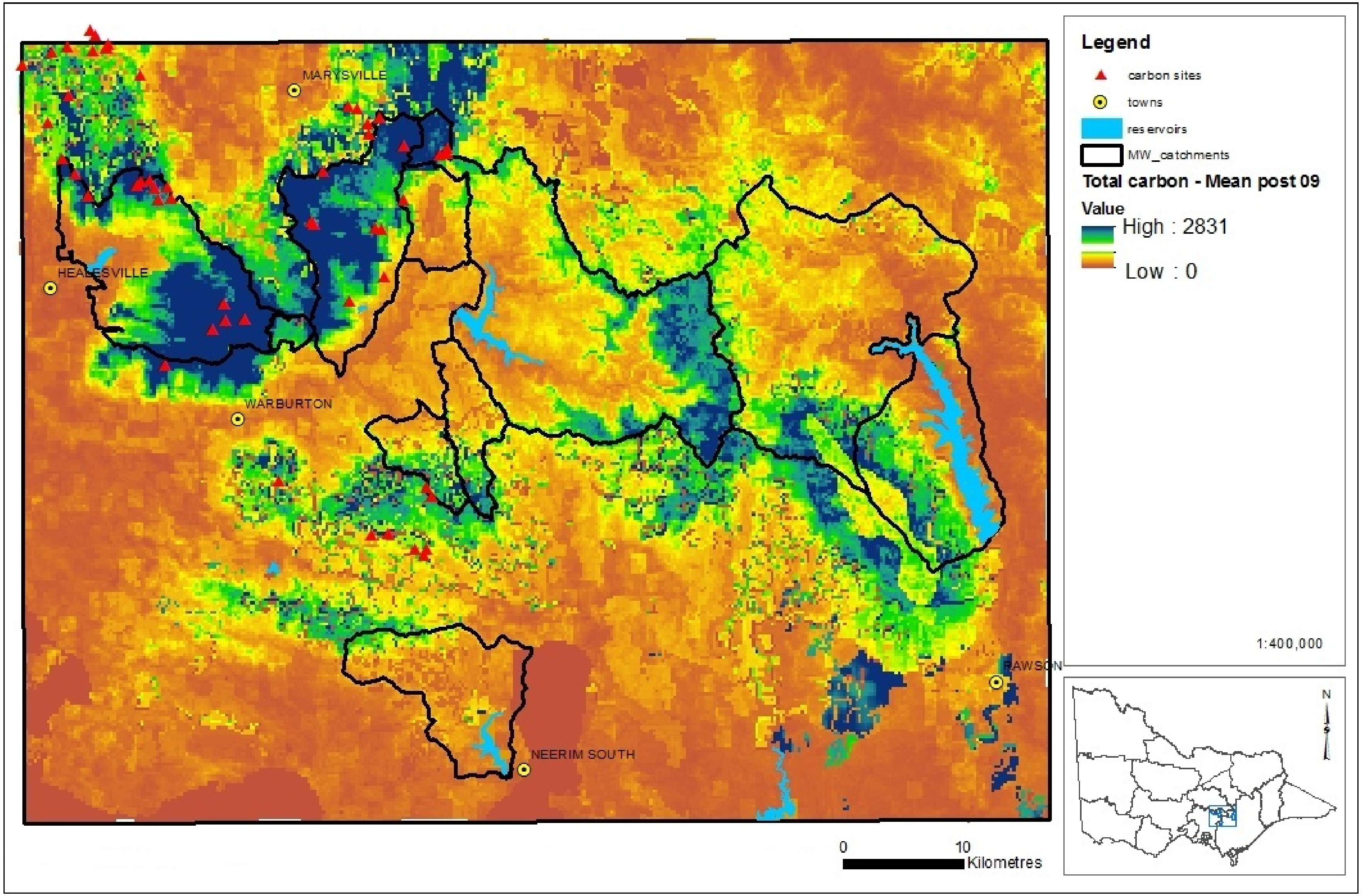
Map 6: Mean biomass carbon density (t C / ha) in the region of the water catchments.



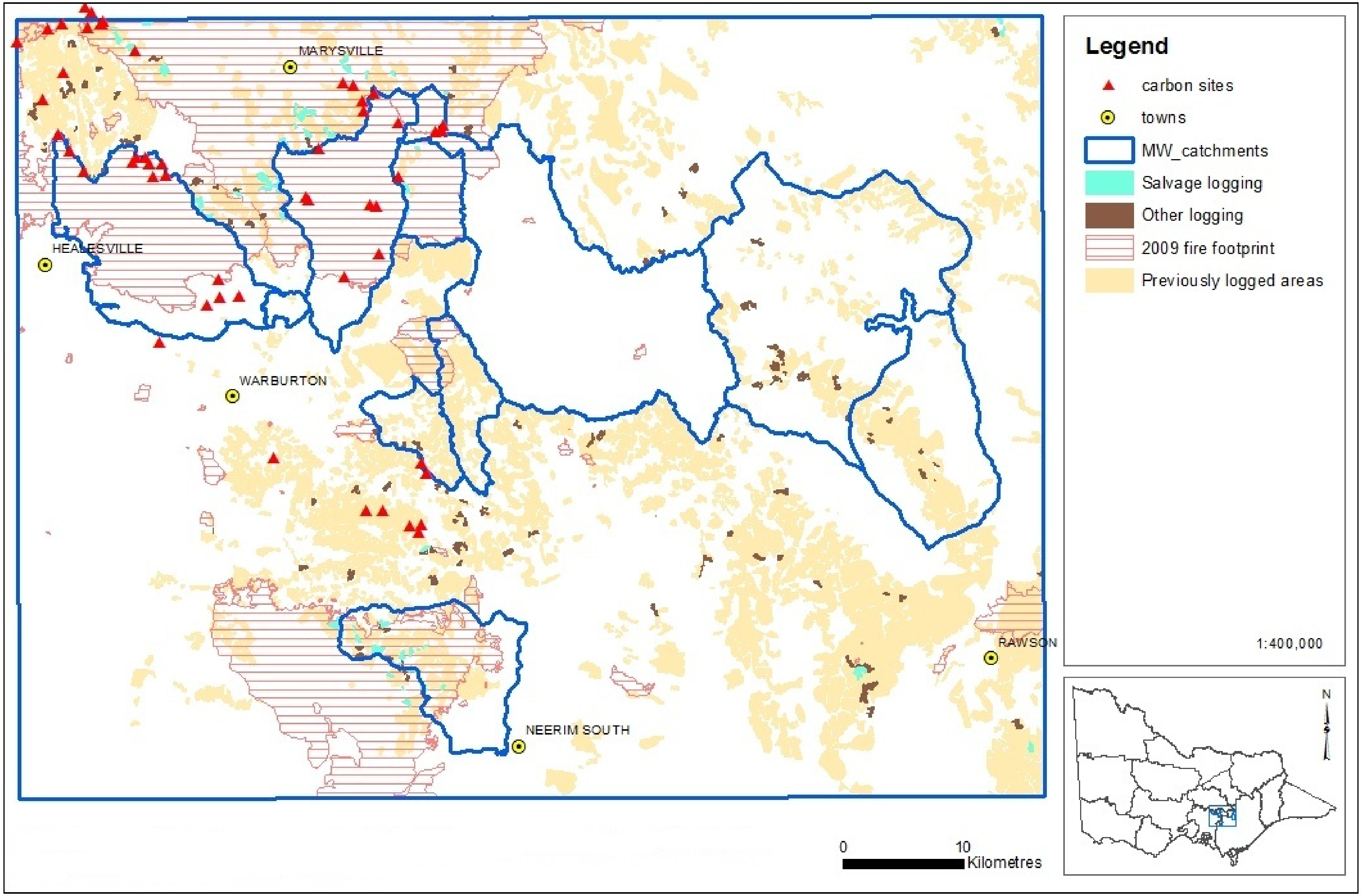
Map 7: Area of the 2009 wildfire classified as low and high severity fire.



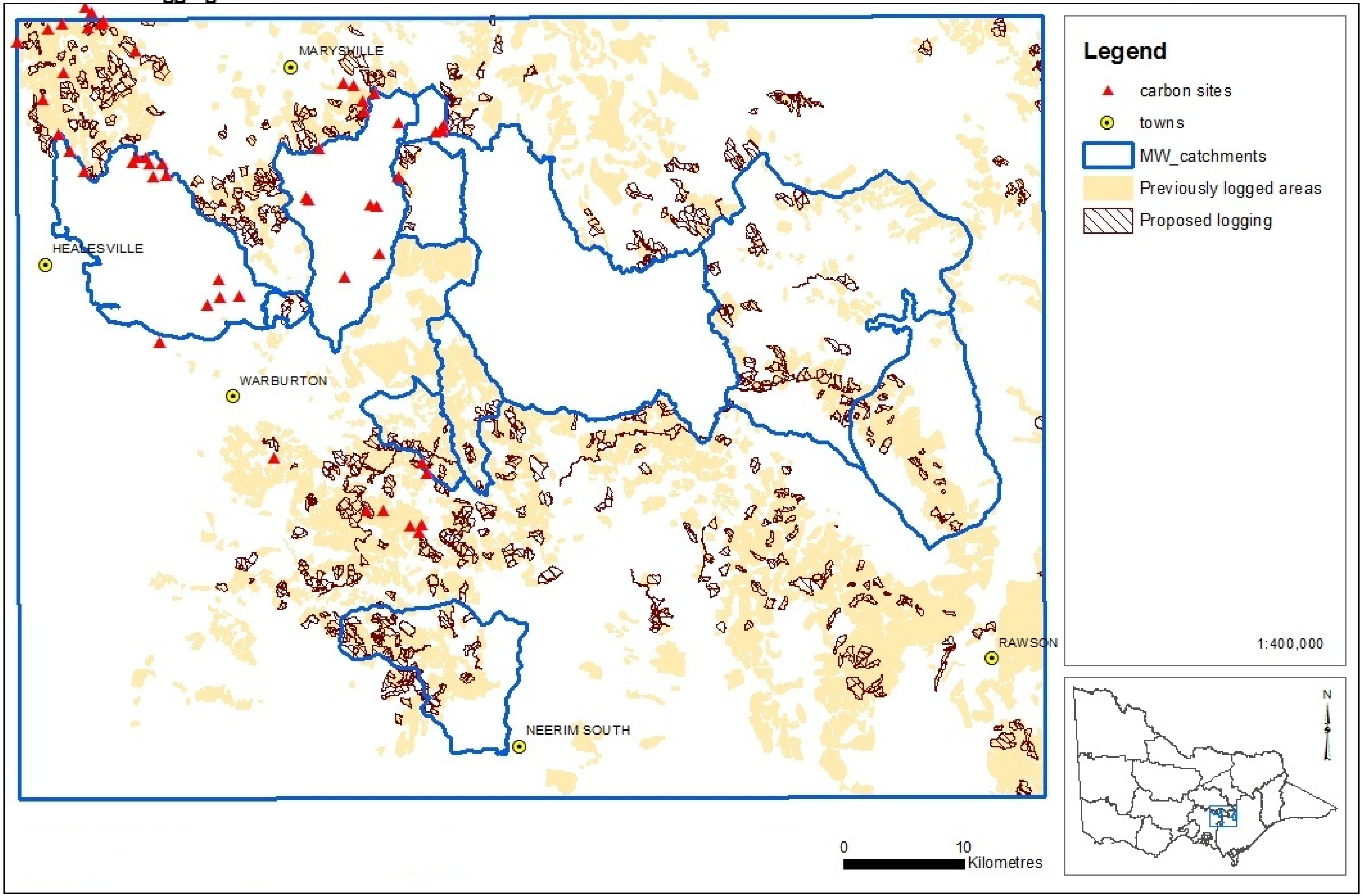
Map 8: Mean biomass carbon density (t C / ha) after the 2009 wildfire in the region of the water catchments.



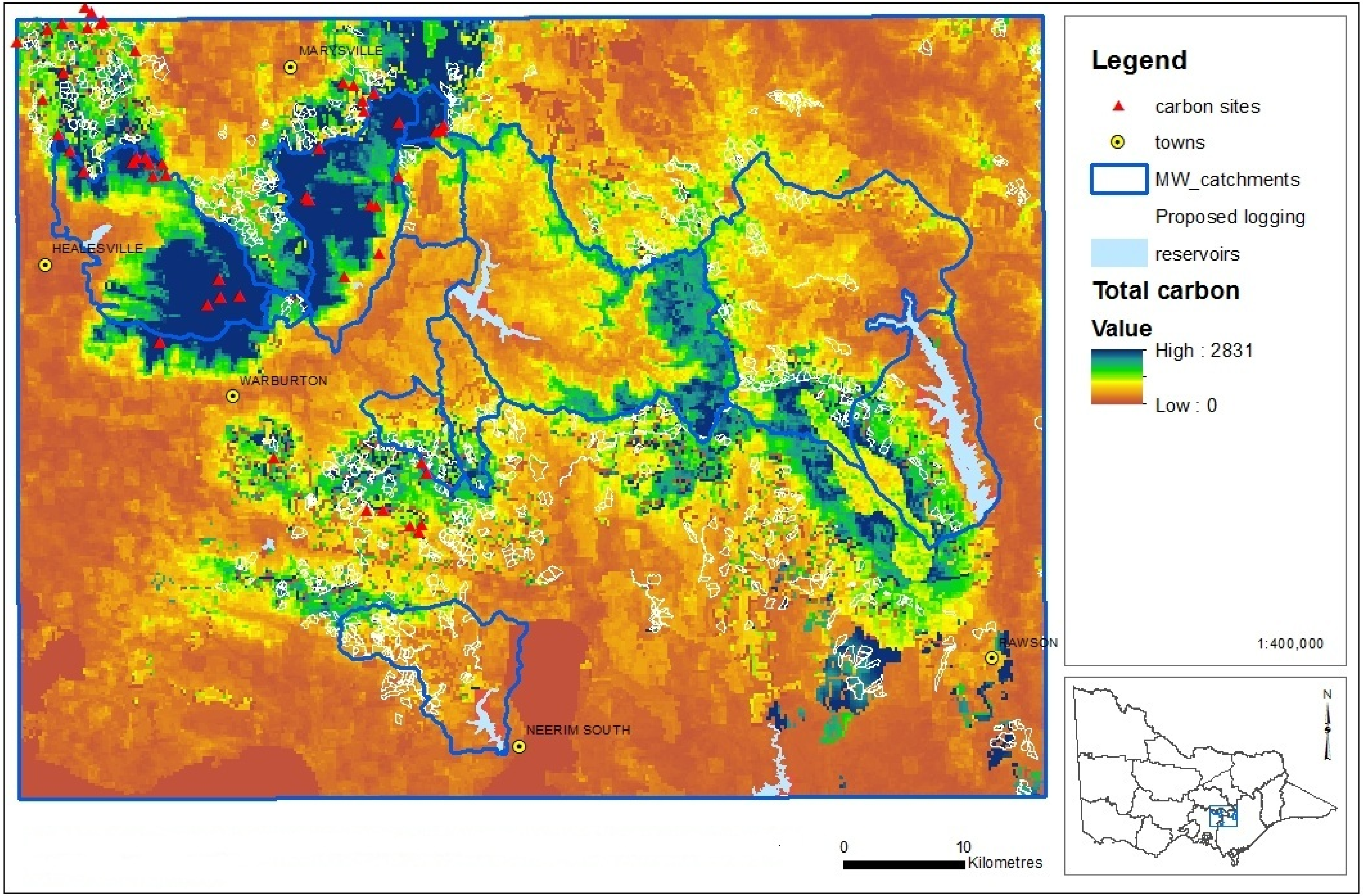
Map 9: Areas that have been logged since the 2009 wildfire as salvage logging within the fire footprint and logging in other areas.



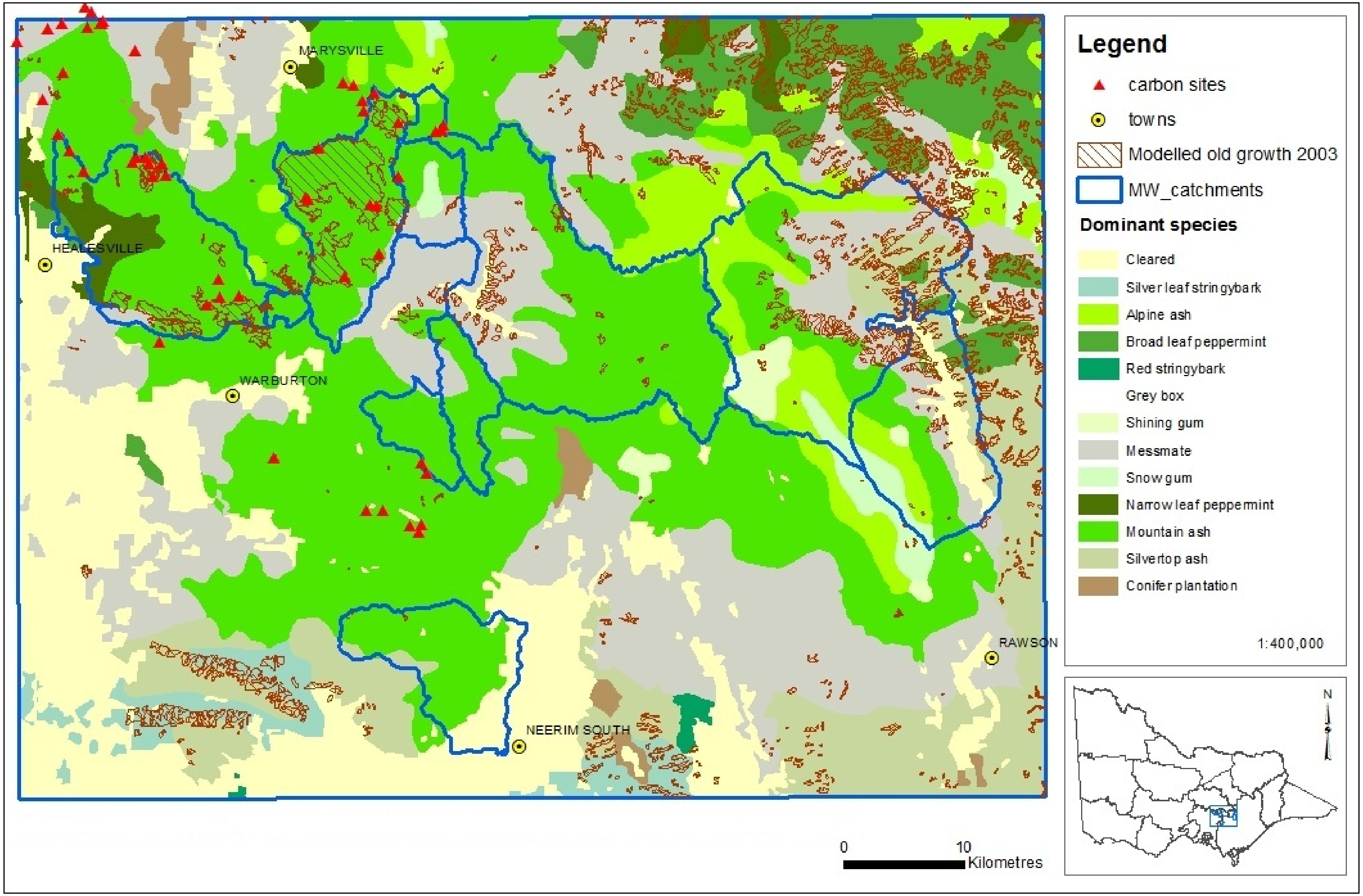
Map 10: Areas proposed for logging in the Timber Release Plans for 2011 - 2016, including salvage logging within the fire footprint and logging in the other areas.



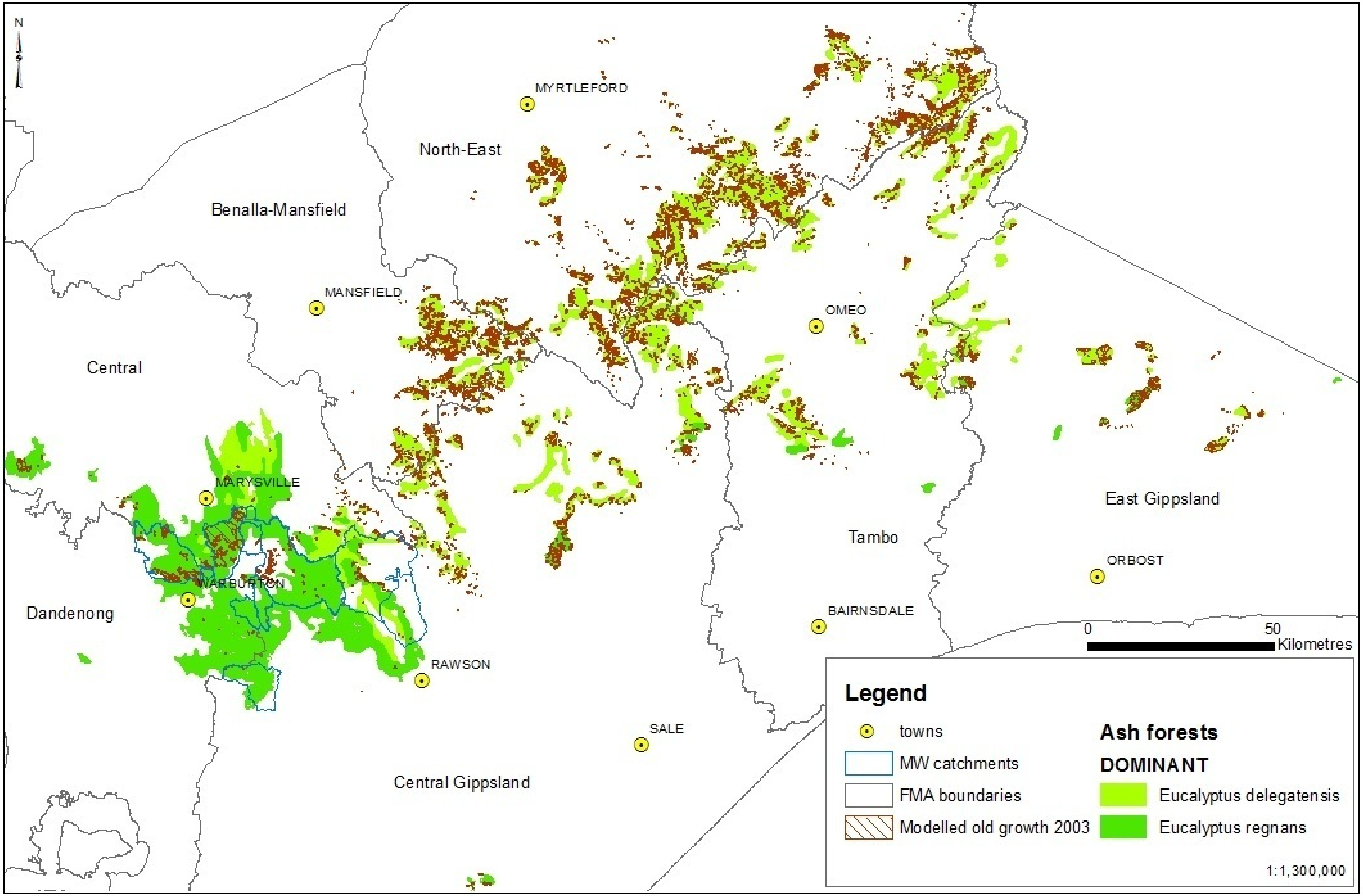
Map 11: Area proposed for logging (white boundaries) in relation to the spatial distribution of biomass carbon density (t C / ha).



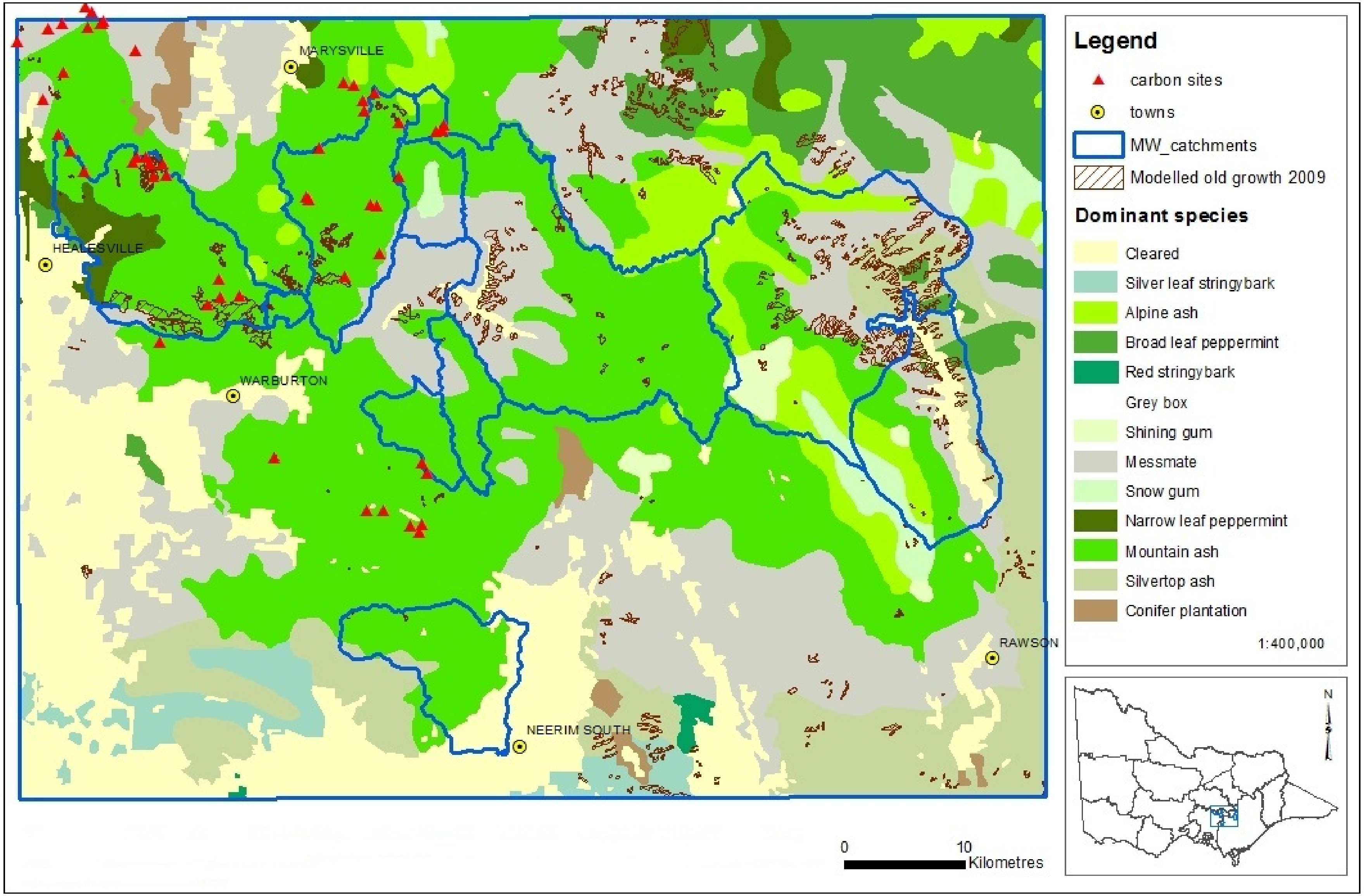
Map 12: Areas mapped as old growth forest by DSE in 2003 within the water catchment region.



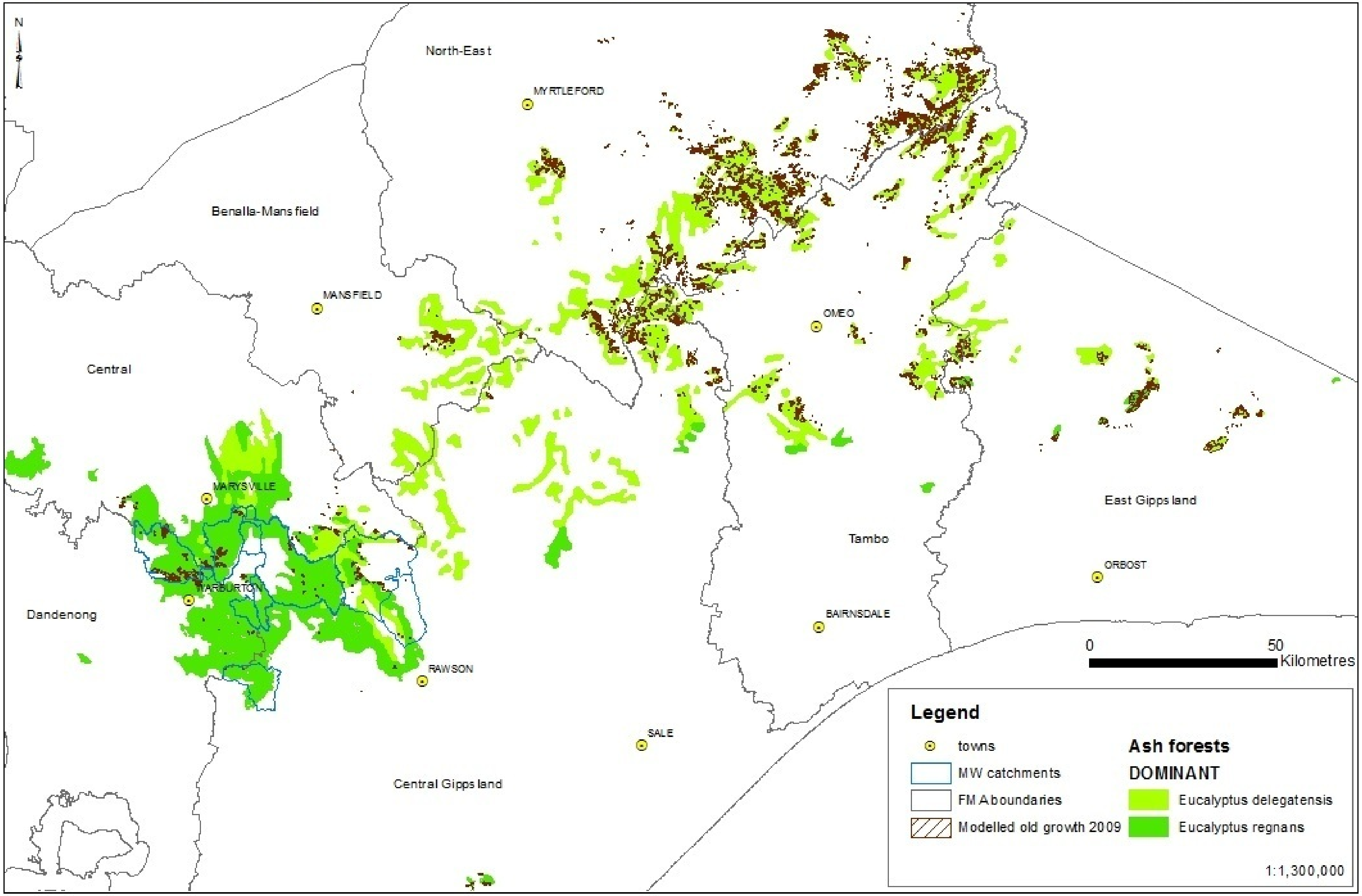
Map 13: Areas mapped as old growth ash forest in Victoria by DSE in 2003.



Map 14: Areas mapped as old growth by DSE in 2009 after the wildfire in the water catchment region.



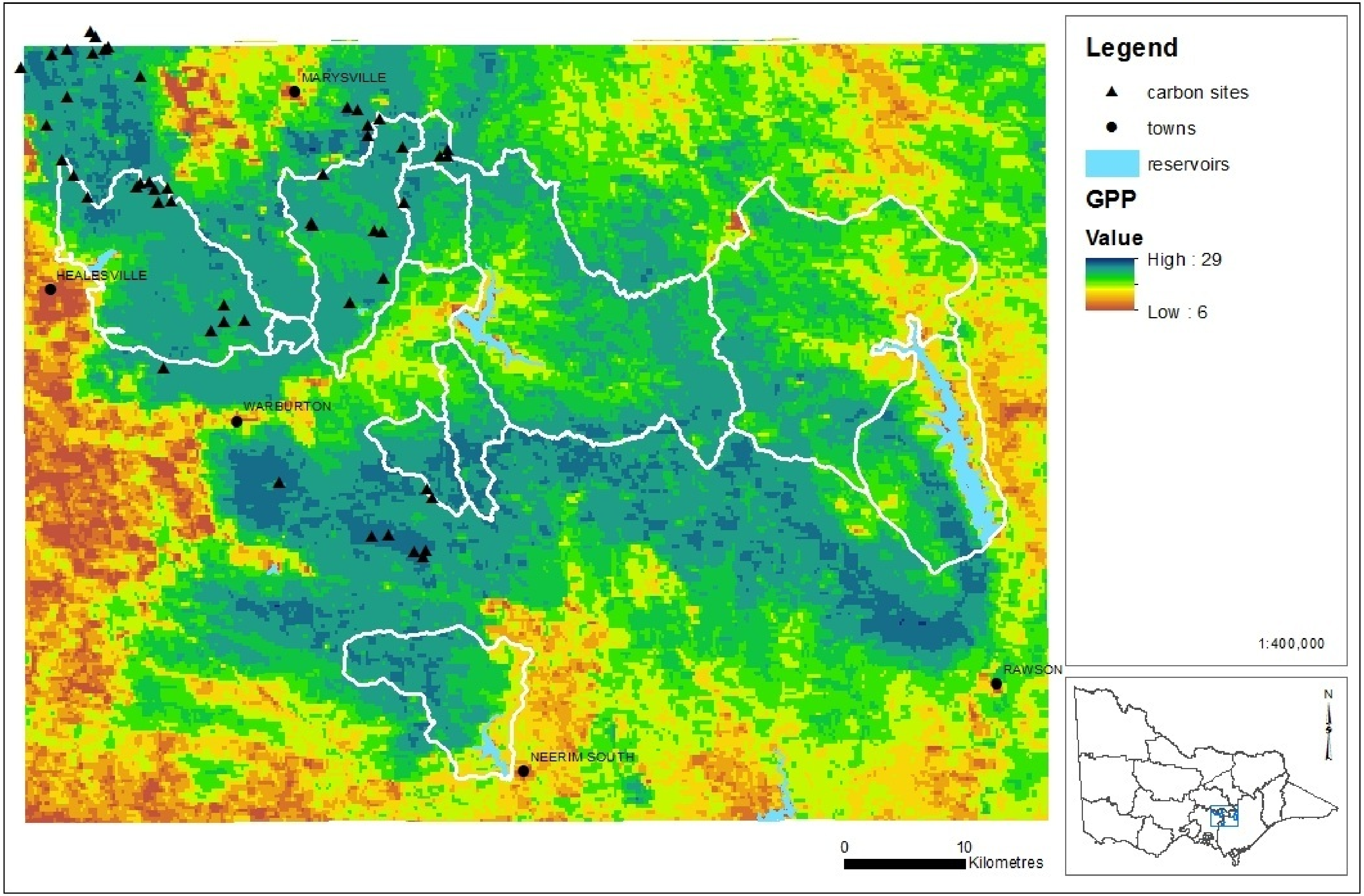
Map 15: Areas mapped as old growth ash forest in Victoria by DSE in 2009 after the wildfire.



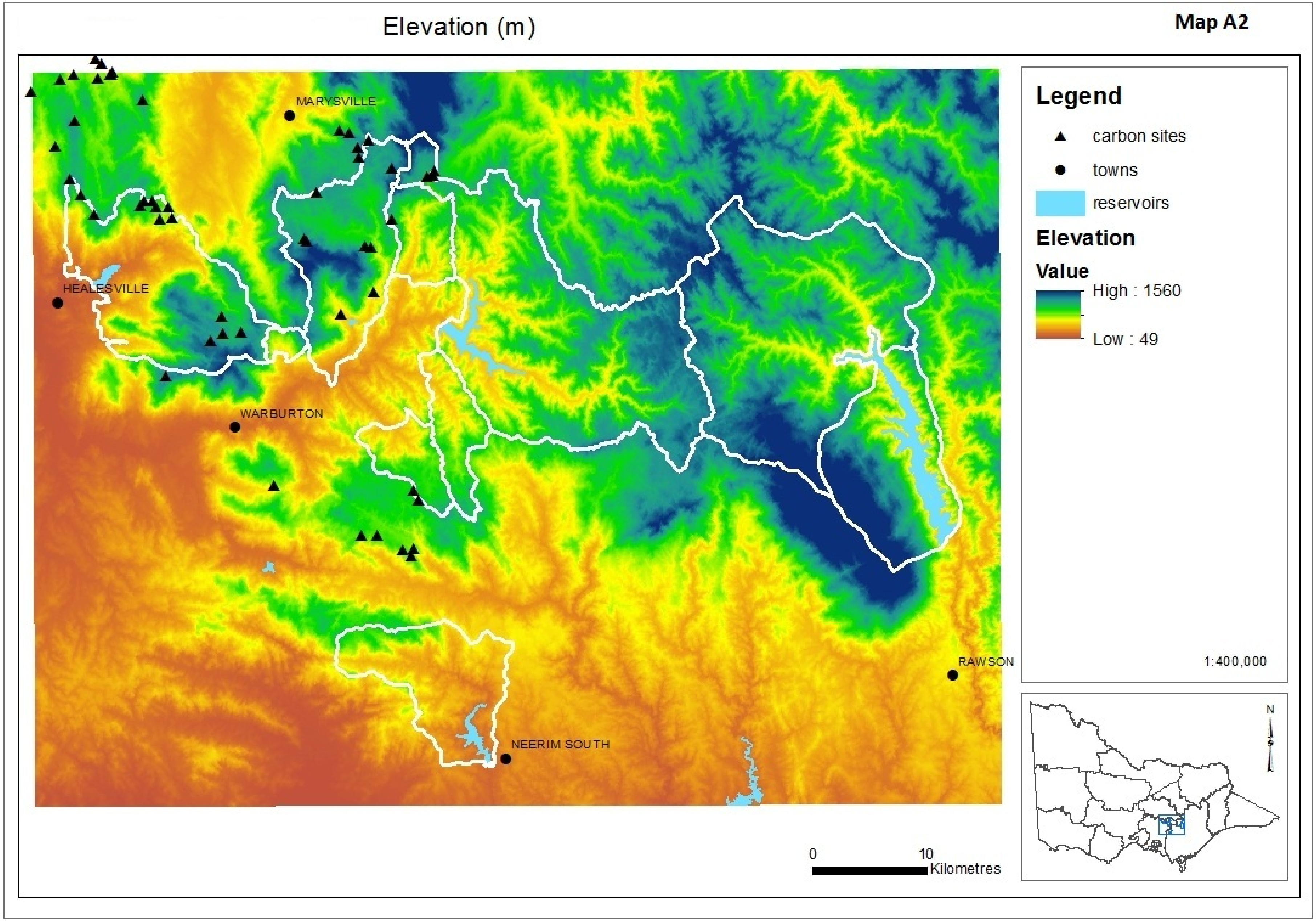
Appendix 1. Maps of Environmental Variables as defined in Table 4.1.

Gross Primary Productivity (GPP) ($t\ C\ ha^{-1}\ year^{-1}$)

Map A1

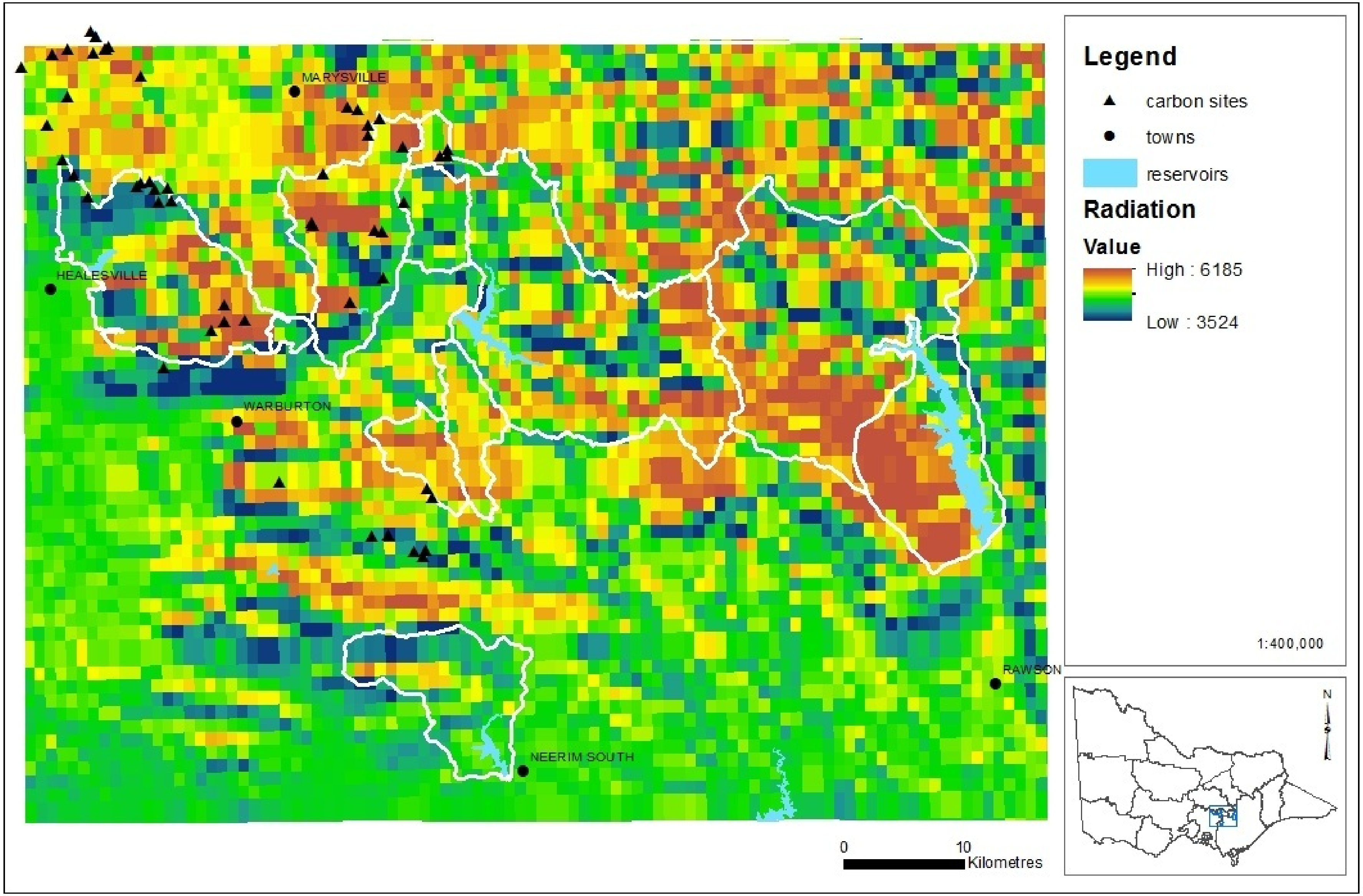


Map A2


Elevation (m)
MARYSVILLE
HEALESVILLE
WARBURTON
RAWSON
NEERIM SOUTH
Legend
carbon sites
towns
reservoirs
Elevation
Value
High : 1560
Low : 49
1:400,000
N
0
10
Kilometres

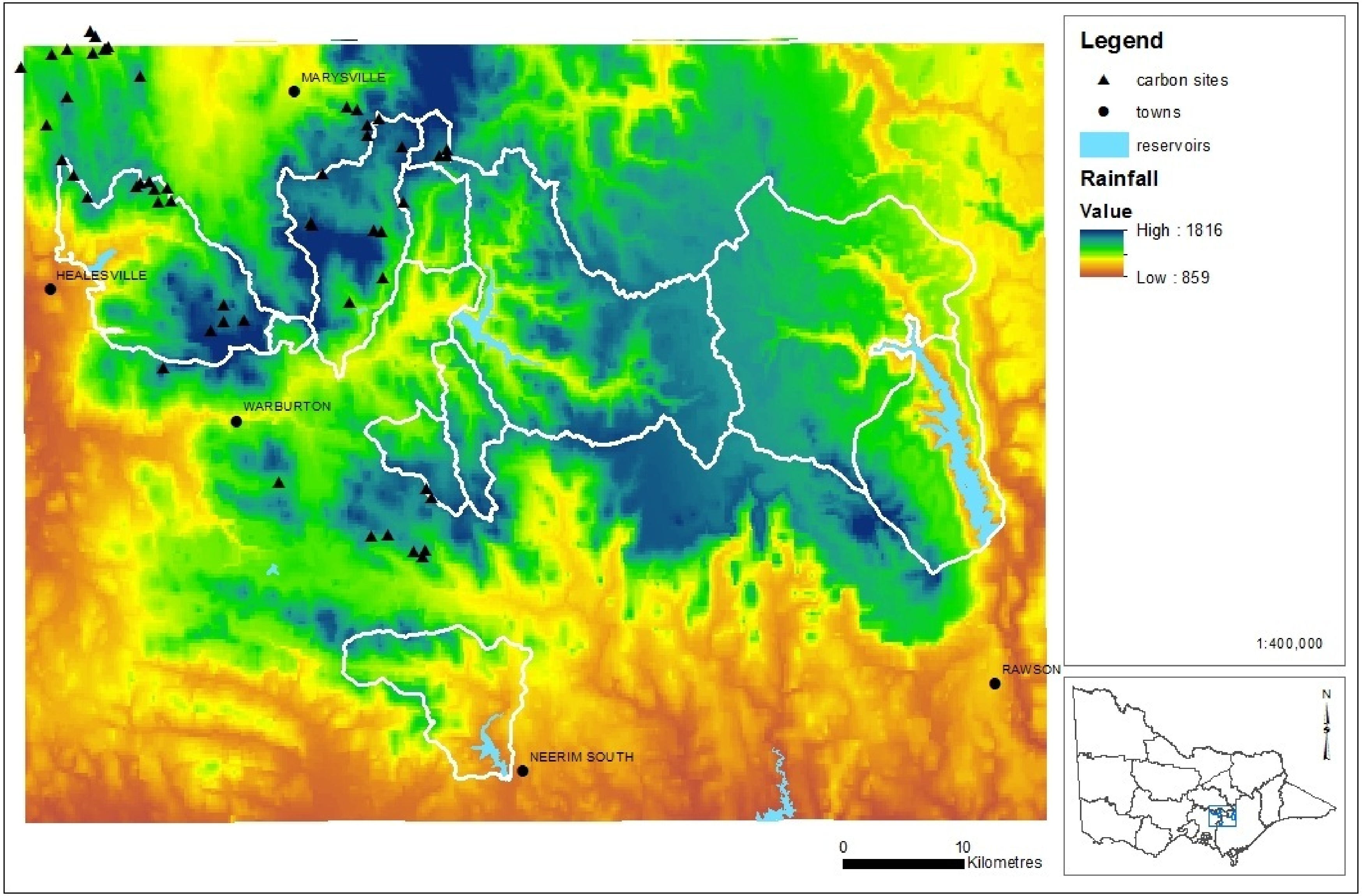
Map A3

Incoming solar radiation (mJ m^{-2} $year^{-1}$)



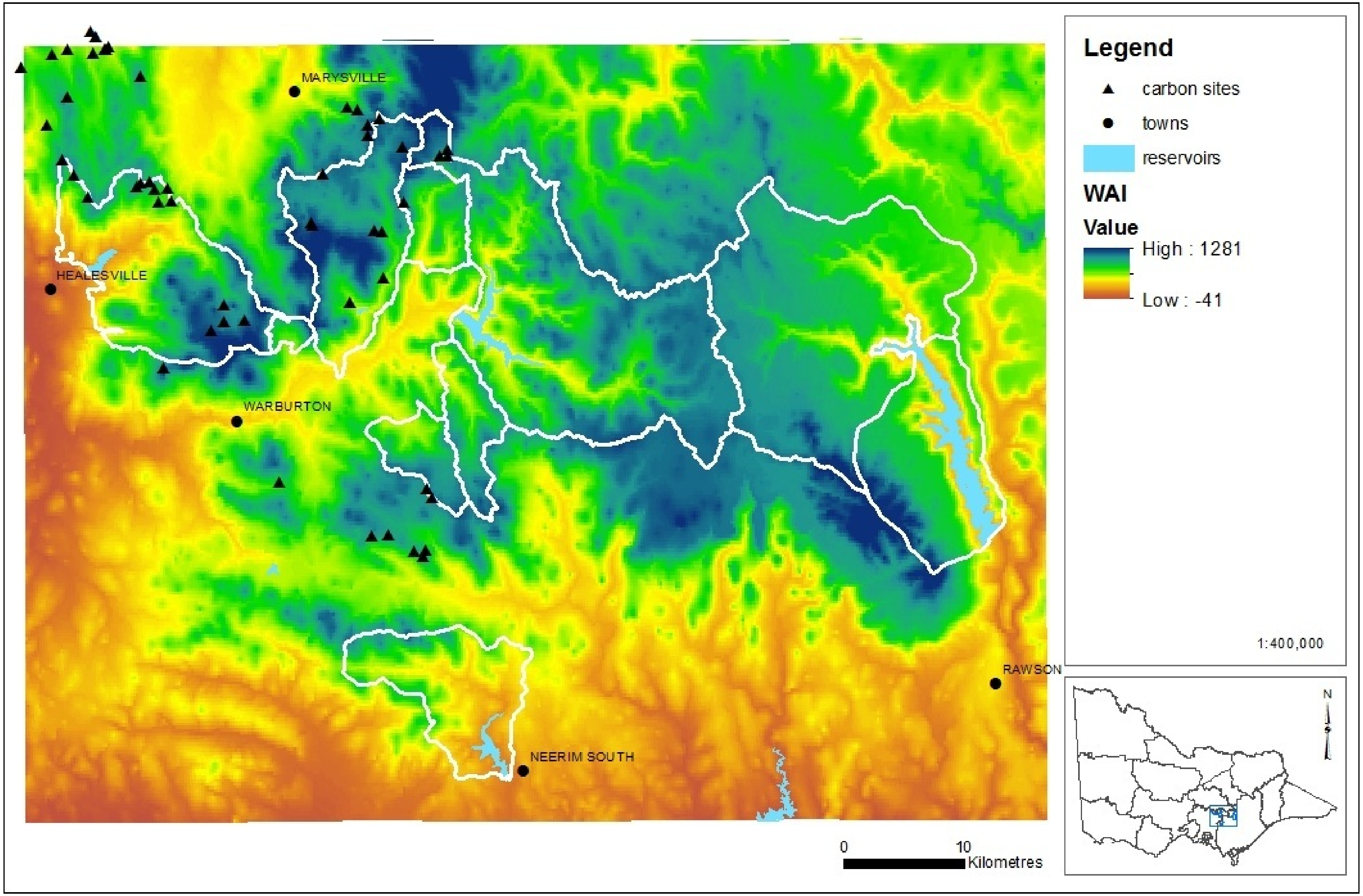
Annual rainfall (mm)

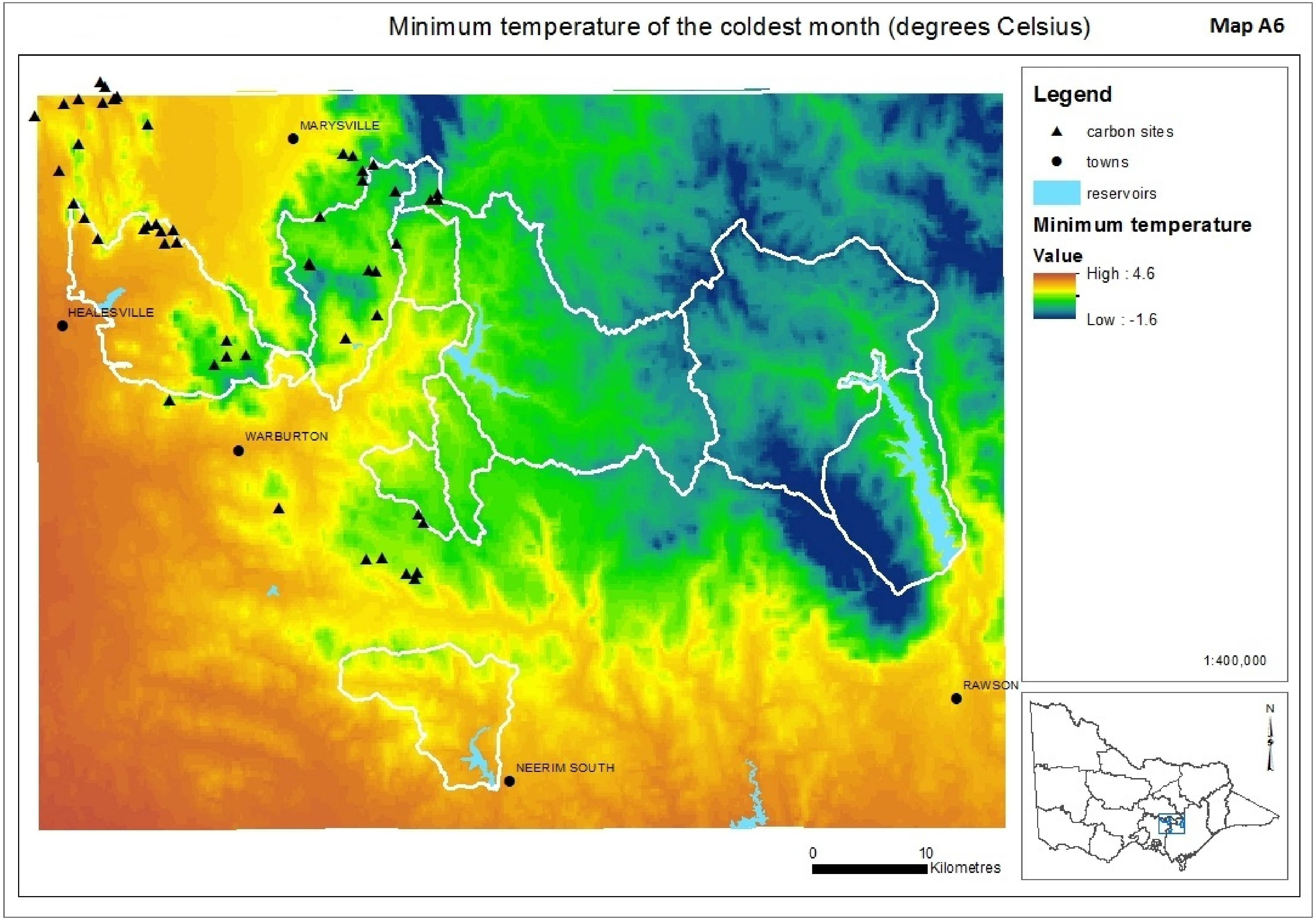
Map A4



Water Availability Index (mm year^{-1})

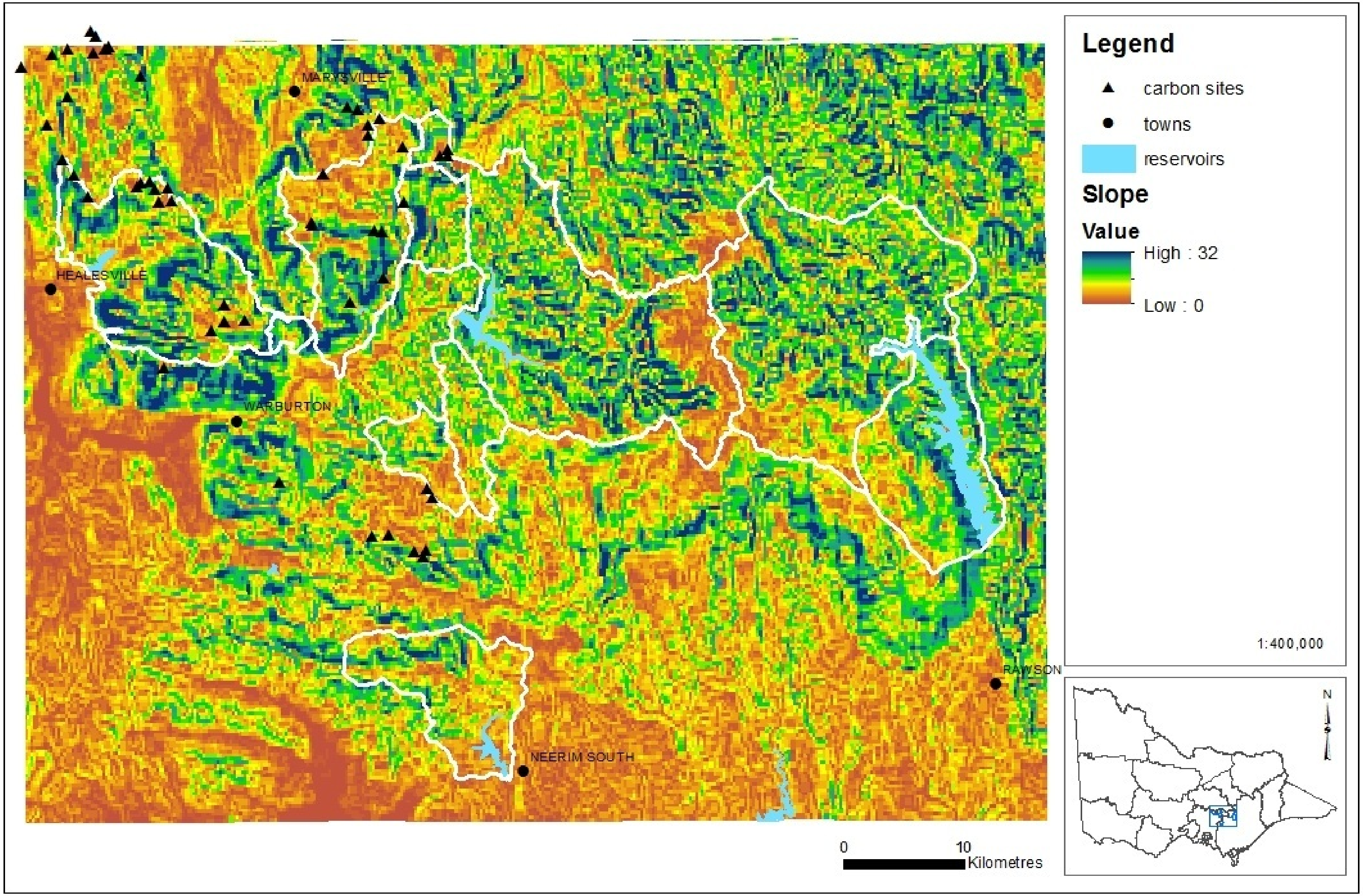
Map A5




Minimum temperature of the coldest month (degrees Celsius)
Map A6
Legend
carbon sites
towns
reservoirs
Minimum temperature
Value
High : 4.6
Low : -1.6
1:400,000
MARYSVILLE
HEALESVILLE
WARBURTON
RAWSON
NEERIM SOUTH
0
10
Kilometres
N

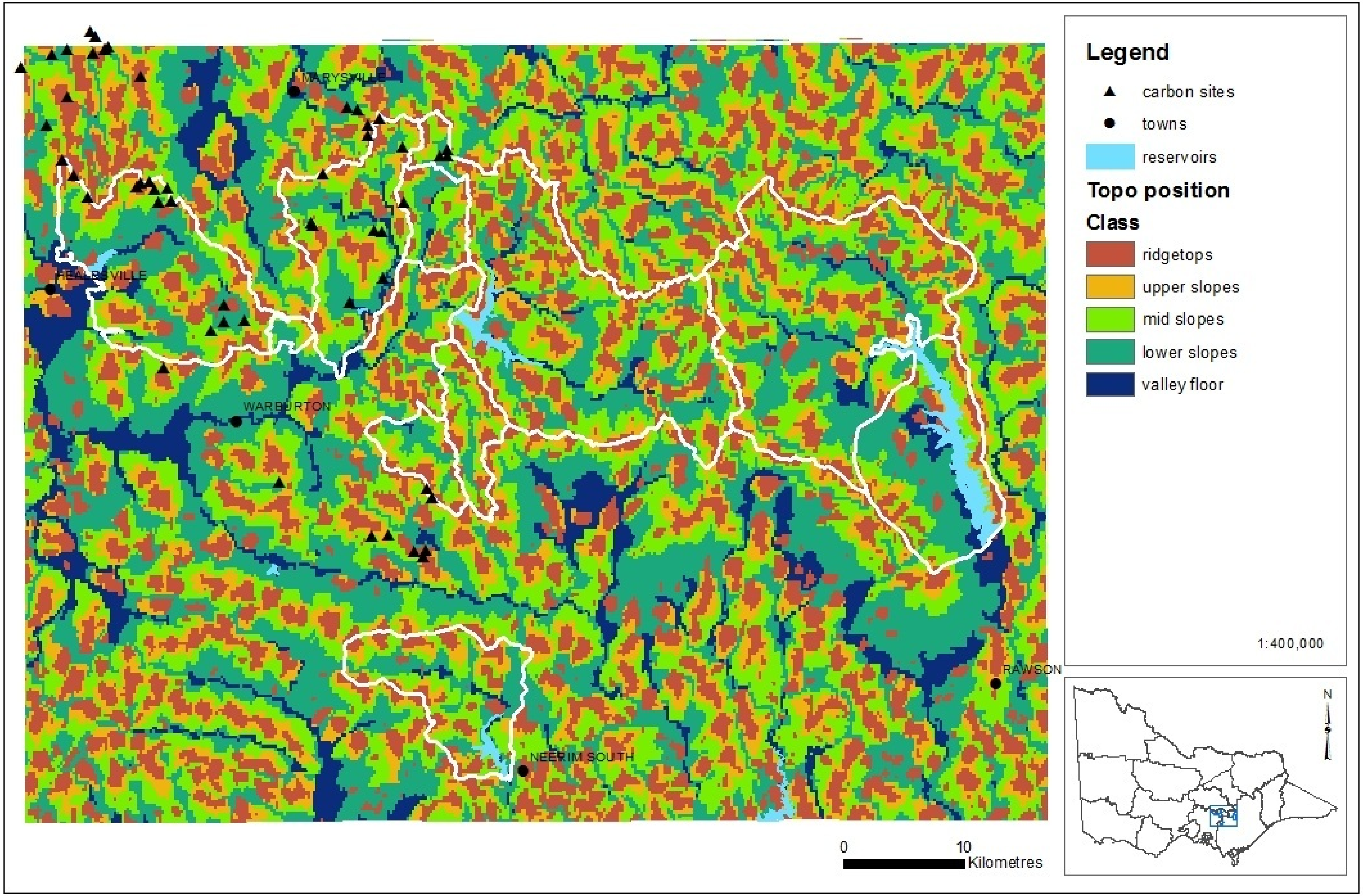
Slope (degrees)

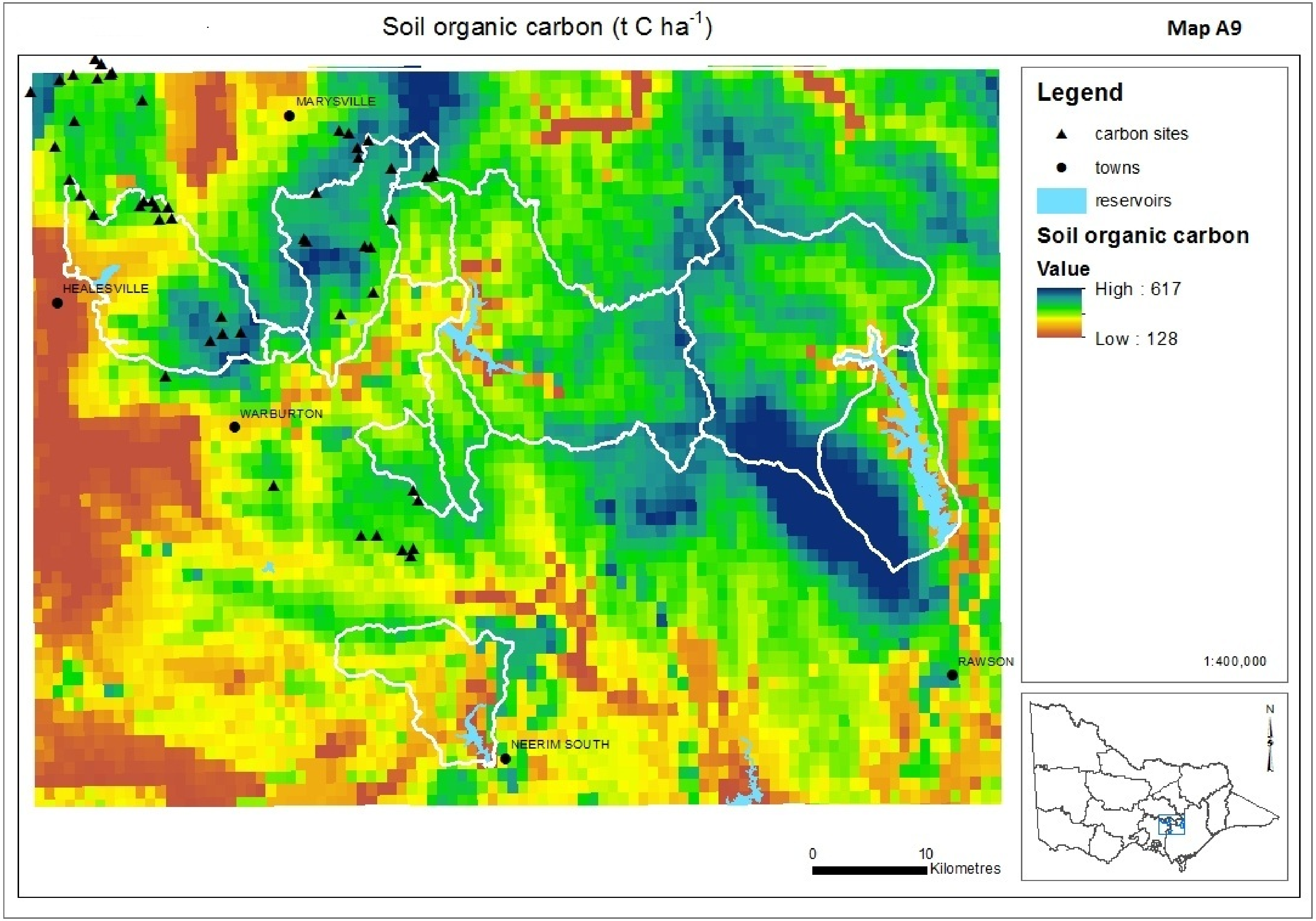
Map A7



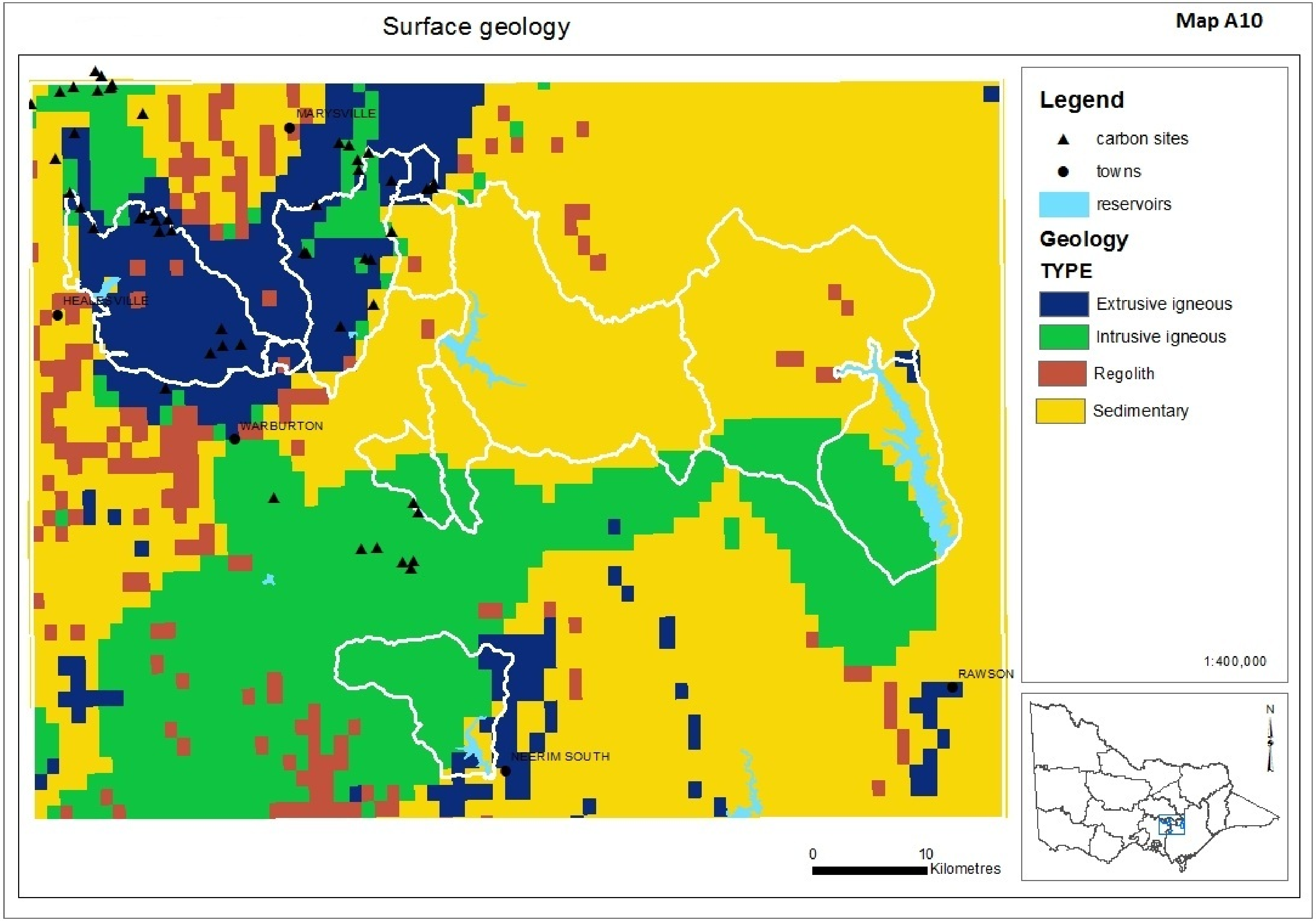
Topographic position

Map A8





Soil organic carbon (t C ha-1)
Map A9
MARYSVILLE
HEALESVILLE
WARBURTON
NEERIM SOUTH
RAWSON
Legend
carbon sites
towns
reservoirs
Soil organic carbon
Value
High : 617
Low : 128
1:400,000
0
10
Kilometres
N



Surface geology
Map A10
MARYSVILLE
HEALESVILLE
WARBURTON
RAWSON
NEERIM SOUTH
Legend
carbon sites
towns
reservoirs
Geology
TYPE
Extrusive igneous
Intrusive igneous
Regolith
Sedimentary
1:400,000
0
10
Kilometres
N

Appendix 2: Statistical models to predict total biomass carbon stock

Carbon stock includes living and dead biomass, litter and coarse woody debris.

Model coefficients for each disturbance event type.

1. Clearfelling plus ANU carbon sites of known age

i) Unburnt ($TotC_U$)

Ln $TotC_U$ = a + b_1 age + b_2 GPP + b_3 soc + b_5 radiation + b_6 mintemp + b_{11} elevation

Values of the regression constant a are specific to each factorial level of geology.

variance accounted for is 64.8%, se = 0.263, df = 9,99 , n= 108

Parameter	coefficient	estimate	SE	v.r.	F_{pr}
Variates					
age	b_1	0.003783	0.000785	145.19	<0.001
GPP	b_2	0.0103	0.0434	12.44	<0.001
soc	b_3	0.001782	0.000569	8.86	0.004
radiation	b_5	0.00000164	0.0000452	5.61	0.020
mintemp	b_6	0.2442	0.0847	4.81	0.031
elevation	b_{11}	0.001205	0.000381	13.96	<0.001
Factors					
constant	a	see table for a			
geology			1.22	5.15	0.007

Standard error of the estimate

Values of the regression constant a:

Geology						
1	2	3	4	5	6	SED
3.554	3.327	3.554	3.39	3.554	3.554	0.097

Standard error of the difference

$TotC_U = \exp(LnTotC_U)$

ii) Low severity fire ($TotC_L$)

Ln $TotC_L$ = 6.1587

Regression equations calculated for the difference in total carbon between unburnt and burnt conditions.

$\Delta_{U\text{-}L} = (LnTotC_U) - (LnTotC_L)$

$\Delta_{U\text{-}L} = a + b_1$ age $+ b_2$ GPP $+ b_3$ soc $+ b_5$ radiation $+ b_6$ mintemp $+ b_{11}$ elevation

Parameter	**coefficient**	**estimate**	**SE**	**v.r.**	F_{pr}
Variates					
age	b_1	0.003783	0.000785	145.19	<0.001
GPP	b_2	0.0103	0.0434	12.44	<0.001
soc	b_3	0.001782	0.000569	8.86	0.004
radiation	b_5	0.00000164	0.0000452	5.61	0.020
mintemp	b_6	0.2442	0.0847	4.81	0.031
elevation	b_{11}	0.001205	0.000381	13.96	<0.001
Factors					
constant	a	see table for a			
geology			1.22	5.15	0.007

Standard error of the estimate

Values of the regression constant a:

Geology						
1	2	3	4	5	6	SED
-2.6047	-2.9317	-2.7687	-2.7687	-2.7687	-2.7687	0.097

Standard error of the difference

$TotC_L = \exp(LnTotC_U - \Delta_{U\text{-}L})$

iii) High severity fire

$LnTotC_H = 5.627 + 0.003 * age$

$\Delta_{U\text{-}H} = (LnTotC_U) - (LnTotC_H)$

$\Delta_{U\text{-}H} = a + b_1$ age $+ b_2$ GPP $+ b_3$ soc $+ b_5$ radiation $+ b_6$ mintemp $+ b_{11}$ elevation

Parameter	**coefficient**	**estimate**	**SE**	**v.r.**	F_{pr}
Variates					
age	b_1	0.000783	0.000785	145.19	<0.001
GPP	b_2	0.0103	0.0434	12.44	<0.001
soc	b_3	0.001782	0.000569	8.86	0.004
radiation	b_5	0.00000164	0.0000452	5.61	0.020
mintemp	b_6	0.2442	0.0847	4.81	0.031
elevation	b_{11}	0.001205	0.000381	13.96	<0.001
Factors					
constant	a	see table for a			
geology			1.22	5.15	0.007

Standard error of the estimate

Values of the regression constant a:

Geology						
1	2	3	4	5	6	SED
-2.073	-2.400	-2.237	-2.237	-2.237	-2.237	0.097

Standard error of the difference

$TotC_H = \exp(LnTotC_U - \Delta_{U\text{-}H})$

2. Single Tree Selection

Ln TotC = a + b_2 GPP + b_{11} elevation + b_6 mintemp + b_8 W

Values of the regression constant a are specific to each factorial combination of forest type and number of previous single tree selection events.

variance accounted for is 46.3%, se = 0.181, df = 8,22 , n= 31

Parameter	**coefficient**	**estimate**	**SE**	**v.r.**	F_{pr}
Variates					
GPP	b_2	0.1472	0.0627	2.73	0.113
elevation	b_{11}	0.001041	0.000449	3.11	0.091
W	b_8	-0.000495	0.000233	7.54	0.012
mintemp	b_6	0.1289	0.0622	2.96	0.099
Factors					
constant	a	see table for a	1.69		
forest type				5.15	0.008
no. STS events				2.08	0.164

Standard error of the estimate

Values of the regression constant a:

no. STS events	Forest Type						
	Er_p	Er_d	Er_m	Ed_p	Ed_d	Ed_m	SED
1	1.270	0.876	1.270	1.335	1.149	1.270	
2	1.562	1.168	1.562	1.627	1.441	1.562	0.203
SED		0.130		0.179	0.163		

Standard error of the difference

3. Thinning

Ln TotC = $a + b_9$ aspect

Values of the regression constant *a* are specific to each factorial level of topography and forest type and number of thinning events previously.

variance accounted for is 40.2%, se = 0.254, df = 8,10, n= 19.

Parameter	coefficient	estimate	SE	v.r.	F_{pr}
Variates					
aspect	b_9	0.000992	0.000901	3.44	0.094
Factors					
constant	*a*	see table for *a*	0.272		
topography				2.87	0.090
forest type				1.74	0.222
no. thinnings				2.83	0.124

standard error of the estimate

Values of the regression constant *a*:

No. thinning events	Topographic Position	Forest Type						
		Er_p	Er_d	Er_m	Ed_p	Ed_d	Ed_m	SED
1	1	5.730	5.929	5.730	6.009	5.494	5.730	
	2	5.324	5.523	5.324	5.603	5.088	5.324	0.192
	3	5.854	6.053	5.854	6.133	5.618	5.854	0.217
	4	6.061	6.260	6.061	6.340	5.825	6.061	0.163
	6	5.730	5.929	5.730	6.009	5.494	5.730	
2	1	6.360	6.559	6.360	6.639	6.124	6.360	
	2	5.954	6.153	5.954	6.233	5.718	5.954	0.192
	3	6.484	6.683	6.484	6.763	6.248	6.484	0.217
	4	6.691	6.890	6.691	6.970	6.455	6.691	0.163
	6	6.360	6.559	6.360	6.639	6.124	6.360	
	SED		0.310		0.166	0.248		

Standard error of the difference

4. Group selection

Ln TotC = $a + b_3$ soc + b_4 precipitation

variance accounted for is 85.5%, se = 0.185, df = 2, 4, n= 7.

Parameter	coefficient	estimate	SE	v.r.	F_{pr}
Variates					
soc	b_3	-0.00615	0.00126	33.59	0.004
precipitation	b_4	-0.00198	0.00103	3.70	0.127
constant	a	10.95	1.30		

standard error of the estimate

5. Wildfire

Ln TotC = $a + b_1$ age + b_2 GPP + b_5 radiation + b_{10} slope + b_{11} elevation + b_6 mintemp

Values of the regression constant a are specific to each factorial combination of geology, FMA and number of clearfell events previously.

variance accounted for is 24.9%, se = 0.293, df = 18, 518, n= 540.

Parameter	coefficient	estimate	SE	v.r.	F_{pr}
Variates					
age	b_1	0.00029	0.00402	6.77	0.010
GPP	b_2	0.228	0.068	19.59	<0.001
radiation	b_5	0.0001714	0.000068	58.38	<0.001
slope	b_{10}	-0.0199	0.0066	13.50	<0.001
elevation	b_{11}	0.00290	0.000576	30.80	<0.001
mintemp	b_6	0.356	0.140	32.86	<0.001
Factors					
constant	a	see table for a	0.538		
geology				4.78	<0.001
FMA				2.26	0.036
no. clearfells				1.40	0.238

standard error of the estimate

Values of the regression constant *a*:

No. clearfell events previously	**Geology**	**Forest Management Area**							
		C	CG	D	EG	NE	T	BM	SED
0	1	-3.697	-3.663	-3.416	-2.465	-3.587	-3.749	-4.167	
	2	-3.314	-3.280	-3.033	-2.081	-3.203	-3.366	-3.783	0.0415
	3	-3.824	-3.790	-3.543	-2.592	-3.714	-3.876	-4.294	0.155
	4	-3.888	-3.855	-3.608	-2.656	-3.778	-3.941	-4.358	0.0352
	5	-3.251	-3.217	-2.971	-2.019	-3.141	-3.303	-3.721	0.175
	6	-3.697	-3.663	-3.416	-2.465	-3.587	-3.749	-4.167	
1	1	-4.625	-4.591	-4.344	-3.393	-4.515	-4.677	-5.095	
	2	-4.241	-4.208	-3.961	-3.009	-4.131	-4.294	-4.711	0.0415
	3	-4.752	-4.718	-4.471	-3.519	-4.642	-4.804	-5.221	0.155
	4	-4.816	-4.783	-4.536	-3.584	-4.706	-4.868	-5.286	0.0352
	5	-4.179	-4.145	-3.899	-2.947	-4.069	-4.231	-4.649	0.175
	6	-4.625	-4.591	-4.344	-3.393	-4.515	-4.677	-5.095	
	SED		0.0465	0.0519	0.181	0.0718	0.0848	0.0687	

Standard error of the difference

6. Prescribed Burn

Ln TotC = $a + b_1$ age + b_3 soc + b_{11} elevation + b_6 mintemp + b_7 meantemp + b_8 W

Values of the regression constant a are specific to each factorial combination of topographic position, geology and number of clearfell events previously.

variance accounted for is 46.7%, se = 0.194, df = 13,36, n= 50

Parameter	coefficient	estimate	SE	v.r.	F_{pr}
Variates					
age	b_1	0.000030	0.000514	13.00	<0.001
soc	b_3	0.001444	0.00050	7.34	0.010
elevation	b_{11}	0.001736	0.000522	2.29	0.139
mintemp	b_6	0.0631	0.0600	4.80	0.035
meantemp	b_7	0.2251	0.0879	1.59	0.216
W	b_8	0.000225	0.000183	2.13	0.153
Factors					
constant	a	see table for a	1.42		
geology				4.20	0.012
topography				2.26	0.098
no. clearfells				5.37	0.026

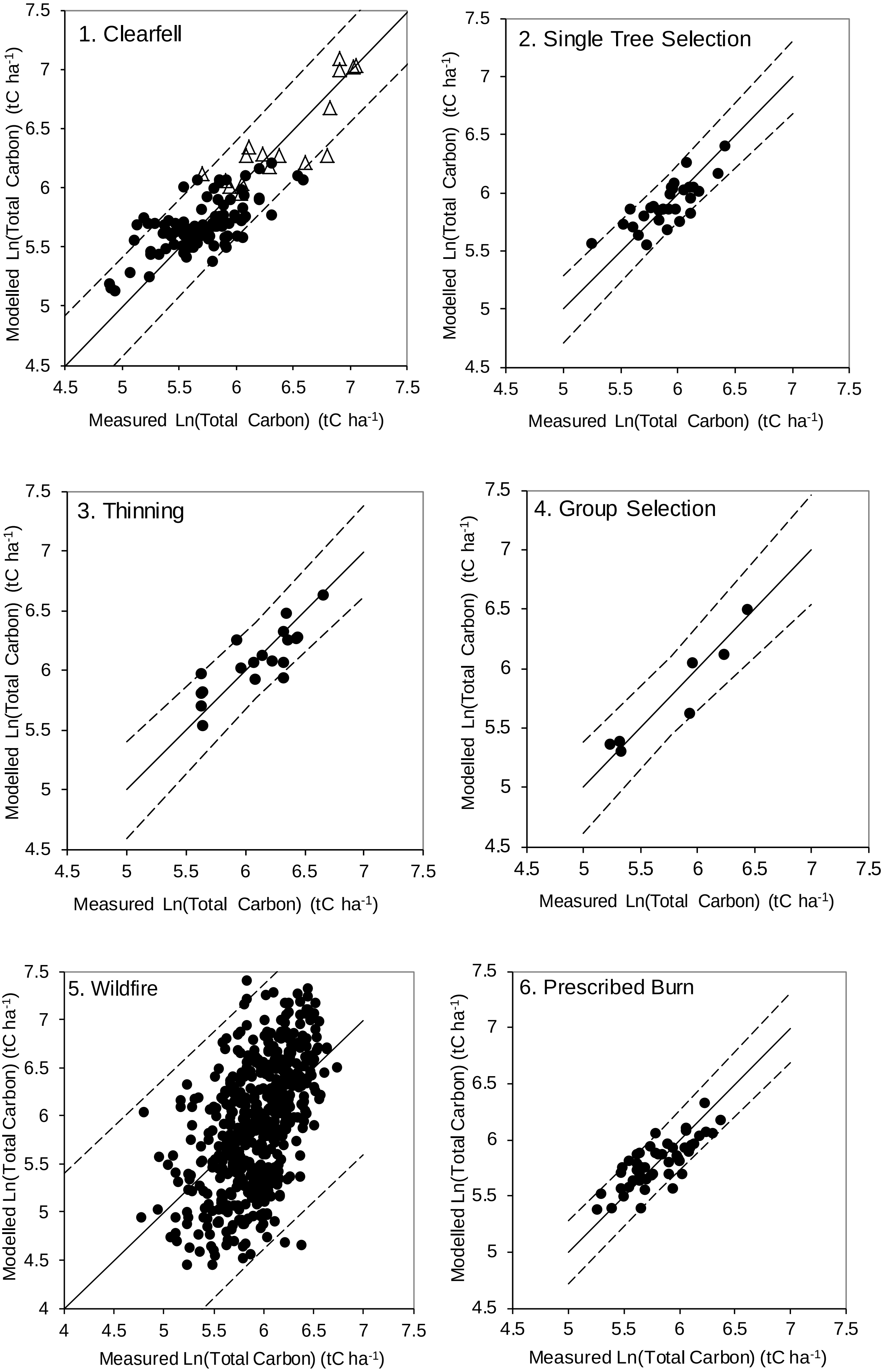
standard error of the estimate

Values of the regression constant a:

No. clearfell events previously	Topographic Position	Geology						
		1	2	3	4	5	6	SED
0	1	1.03	1.281	0.724	1.014	1.03	1.03	1.39
	2	0.845	1.096	0.539	0.828	0.845	0.845	0.0681
	3	0.985	1.236	0.679	0.969	0.985	0.985	0.108
	4	0.916	1.167	0.610	0.900	0.916	0.916	0.118
	6	1.03	1.281	0.724	1.014	1.03	1.03	
1	1	0.534	0.785	0.228	0.518	0.534	0.534	1.39
	2	0.349	0.600	0.043	0.332	0.349	0.349	0.0681
	3	0.489	0.740	0.183	0.473	0.489	0.489	0.108
	4	0.420	0.671	0.114	0.404	0.420	0.420	0.118
	6	0.534	0.785	0.228	0.518	0.534	0.534	
	SED	1.39	0.129	0.157	0.0914			

Standard error of the difference

Figure A2.1: Statistical models to predict total carbon stock categorised by the previous disturbance event. Values are shown for measured versus modelled (using a logarithmic transformation) with the 1:1 solid line and the 95% confidence intervals as dashed lines. Site data shown are SRFI (●) and carbon sites (Δ).


1. Clearfell
Modelled Ln(Total Carbon) (tC ha-1)
Measured Ln(Total Carbon) (tC ha-1)
2. Single Tree Selection
Modelled Ln(Total Carbon) (tC ha-1)
Measured Ln(Total Carbon) (tC ha-1)
3. Thinning
Modelled Ln(Total Carbon) (tC ha-1)
Measured Ln(Total Carbon) (tC ha-1)
4. Group Selection
Modelled Ln(Total Carbon) (tC ha-1)
Measured Ln(Total Carbon) (tC ha-1)
5. Wildfire
Modelled Ln(Total Carbon) (tC ha-1)
Measured Ln(Total Carbon) (tC ha-1)
6. Prescribed Burn
Modelled Ln(Total Carbon) (tC ha-1)
Measured Ln(Total Carbon) (tC ha-1)