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Carbon stocks and flows in native forests and harvested wood products in SE Australia

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Carbon stocks and flows in native forests and harvested wood products in SE Australia

Prepared for

Forest & Wood Products Australia

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Summary for Industry and Policy Makers

Overview

This is the first assessment of the greenhouse (GHG) balance of native forest production in Australia that considers all the key elements of the carbon (C) cycle in forests and harvested wood products (HWPs), plus a cost-benefit analysis that includes socio-economic considerations and C pricing implications. The aim of the project was to track the fate of C from representative native forests in New South Wales (NSW) and Victoria (VIC), in forests managed for multiple use (“production”) and conservation only. The analysis considered C storage in the forest and the effects of disturbance (harvest and fire), as well as C storage in harvested wood products (HWPs) and the effect of substitution of fossil fuels with wood for bioenergy. A complete accounting is important, as much of the debate regarding GHG balances in forests subject to harvesting system stems from different studies including or excluding a number of these key elements from their accounting system, and from the indiscriminate use of default values in the analyses.

Study rationale

There have been many studies recently comparing the GHG outcomes of managing forests for multiple-use (including production) or for conservation only outcomes. There are considerable differences in the conclusions of such studies, which to some extent reflect site-specific conditions, but also the suitability of the values (or methods used to derive the values) underpinning the parameters and the level of inclusiveness of the different components of the C cycle in the forests and HWPs (the “C accounting framework”). There are different C accounting frameworks which, although effective in achieving their specific goals, are limited in scope and generally not intended to cover all life cycle removals and emissions associated with a specific activity – this is typically the realm of life cycle assessments (Table 1).

Table 1. Comparison of scope of accounting frameworks

Parameters	This study (ForestHWP)	International framework (Kyoto)	National Framework (NCAS)
<i>Forest (Above-ground biomass)¹</i>			
Carbon in trees	√	√	√
Carbon in CWD	√	√	√
Emissions due to harvest machinery	√	√ ²	√ ²
Emissions due to fire	√	√	√
Emissions due to decay	√	√	√
<i>HWP</i>			
Carbon storage in HWP in service	√	√	√
Carbon storage in HWP in landfill	√	No	√
Emissions due to log transport and product manufacture	√	√ ²	√ ²
HWP substitution impact (including international leakage)	√	No	No
Fossil fuel displacement benefits – biomass for bioenergy	√	No	No

¹Although there is currently insufficient data to support inclusion of below-ground C dynamics in ForestHWP, it can be easily implemented. ² Emissions are included as part of sectoral GHG emissions reporting, not integrated with forests systems.

The international C accounting framework (the “Kyoto” framework) is the result of negotiations designed to create incentives and share the burden for emission reductions, rather than all-inclusive GHG assessments. For example, despite a range of studies pointing to only small losses of C from timber that is deposited in landfills, under Kyoto accounting rules the C in timber is assumed to be completely oxidised at the time of disposal to landfills, to avoid creating incentives for landfilling. The Kyoto framework also does not directly deal with the substitution impacts associated with the use of HWPs in lieu of greenhouse-intensive materials, and the use of wood biomass in the generation of bioenergy, displacing the use of fossil fuels. Thus the Kyoto framework does not provide an accurate depiction of the true C pathways associated with forest management, and therefore we argue that it should not be used for determining the GHG balance of native forest management. Similarly, national level C accounting in Australia (National Carbon Accounting System – NCAS) does not deal directly with the substitution impacts associated with the use of harvested wood products in lieu of other materials. It also does not deal with international leakage potential (e.g. emission implications associated with increased wood production overseas in the event native wood was no longer harvested in Australia).

In this study we applied a C accounting framework that is based on a life cycle approach, including all key C stocks and flows from the forest / HWP interface. The approach was to select 0.5 ha sites as case studies in native forests areas in NSW and VIC managed for production and conservation. We derived site-specific parameters to support the derivation of above-ground forest C pools and emissions due to harvest and fire; and tracked the fate of C in HWPs derived from the logs produced in the forests. The site-specific forest and commercial log biomass values, while accurate, were not replicated and so a comparison of typical values for similar forest types was conducted to ensure that the values were within the typical range for those forests.

One of the limitations in the assessment of the GHG implications of native forest management in Australia is the lack of a user-friendly, comprehensive tool that captures C dynamics in the forest and HWP pools adequately. In our study we introduce a new tool (“ForestHWP”), which was designed to capture all key elements of the C pools in forests and HWP, allowing immediate integration of parameters and running of simulations that allow for the inclusion of repeated disturbance events (e.g. harvest and fire). The combination of detailed and representative site-specific information, including the dynamics of C in HWP, and a new tool to integrate the forest/HWP parameters allowed a comprehensive assessment of the GHG implications of managing from production versus conservation. This approach was much closer to a “life cycle assessment” framework than the international or national C accounting frameworks (Table 1).

In addition to GHG implications, we considered the socio-economic implications of native forest management, which are often ignored in greenhouse balance studies of native forestry.

Background context and study coverage

The potential role of forestry in mitigating climate change, though substantial, has been largely overlooked in Australian climate change policy. As a result, the cost to society of achieving emissions abatement objectives may increase. When determining the climate impacts of any industry sector, it is important to adopt a true life cycle assessment (LCA) approach, that takes into account all relevant emissions and C removals; i.e. what the atmosphere actually sees.

This study is a follow-up to a previous study by Ximenes et al (2012), which was primarily a desktop study, that used modelled estimates to assess the GHG balance of two key native

forest areas managed for timber production in NSW. The current study addresses a number of gaps identified in Ximenes et al (2012), and is fundamentally different in its approach: data from intensively measured plots were used to extrapolate to wider sub-regional areas. The selected sites were then used as the benchmark against which the long-term C dynamics in the forest and HWP were modelled using a new model (ForestHWP) developed as part of this project. ForestHWP provides a complete system description of forest C dynamics, and the dynamics associated with HWPs, their processing, and their ultimate fate.

This study included the key above-ground forest C pools (live tree biomass, coarse woody debris, litter and harvest residues), the impact of disturbances on those pools (harvest and fire), and the dynamics of C in HWPs in service and in landfills. In addition to the physical tracking of C in forests and HWP, the study also considered the fossil fuel displacement benefits of using biomass for bioenergy, the product substitution impacts and the socio-economic implications of native forest management for the case study regions. The study sites included three paired sites (“production” and “conservation”), with a known history of disturbances (harvest, thinning and wildfire events). The case studies covered one site on the South Coast of NSW (near Eden), one site on the mid-North Coast of NSW (near Wauchope) and one site in the Central Highlands in Victoria (near Toolangi).

Key findings

- *Forest biomass and HWP data*

The comparison of the directly weighed biomass of key native forest species with estimates derived from existing biomass equations revealed that existing equations are generally not reliable and tend to overestimate biomass, especially for trees with large DBH.

Although there is already use of biomass by the mills for bioenergy (kill-drying) and some use offsite, the study highlighted the additional large volumes of harvest and mill-based residues that could be utilised for application such as bioenergy generation.

At the end of their service life, the assumption was that most HWPs are disposed of in landfills. According to the latest research, the C in HWPs in landfills can be considered to be stored for the long-term. However, regardless of the fate of timber at the disposal stage - whether it is recycled, used for energy or landfilled, the GHG impacts will be beneficial.

The HWPs from this study typically required lower fossil-fuel based energy in their extraction and manufacture than alternative materials such as aluminium and concrete. The biggest substitution impacts related to the replacement of hardwood products with imported hardwood (decking and flooring), fibre-cement cladding, concrete slabs and steel and concrete transmission poles.

The GHG mitigation potential of paper products is often dismissed as limited. However, in this study we have shown that when the wood fibre used in paper production is sourced from native forests in SE Asia, the mitigation potential by using Australian native pulpwood is large. This is due to the high emission footprint caused by forest degradation and forest loss in SE Asia, especially when it occurs on peatlands.

- *Socio-economic impacts*

For NSW native forests the socio-economic benefits of management for timber production were higher than those of management for conservation only, with the State Forests on the North Coast of NSW generating more socio-economic value than those at Eden. The cost (economic impact) of transitioning production forests to conservation-only forests for Eden

was \$64M and the loss to the regional economy was \$308M before taking into account carbon abatement. For the north coast, the cost of transition was \$540M and the loss to the regional economy was \$3.36B before taking into account carbon abatement. Transitioning to management for conservation also incurred a sharp decline in regional employment.

Valuation of the carbon abatement benefits relative to business as usual (BAU) were derived using a 65 year modelling period and a low (\$10/tCO₂-e), medium (\$20/tCO₂-e), and high (\$30/tCO₂-e) carbon price.

Assuming a carbon credit was available for avoided emissions by stopping harvest we calculated that a carbon price of \$233 per tCO₂-e (excluding transition costs) was required to support the conservation scenario on the north coast site. At the Eden site the conservation scenario generated less carbon abatement than BAU and so value was lost (rather than gained) when carbon was priced.

When industry value-added benefits (based on BAU) and carbon abatement benefits were combined, the production management scenarios far outperformed the conservation management scenarios. This result was independent of the carbon price (low, medium or high).

- *Scenario modelling (ForestHWP)*

The accrued GHG benefits for the production scenarios compared to the conservation scenarios were greatest for mountain ash, and slightly higher for silvertop ash. Similar to the mountain ash case study, this benefit was driven primarily by the substitution factor for pulp. For blackbutt, the business as usual (BAU) production scenario was approximately 12% lower. Assuming a market for pulpwood and/or use of residual wood for bioenergy, the average GHG emissions over the simulation period were higher for the conservation scenario than for production scenarios. All scenarios, apart from those that involve changing from pulp to biofuel (for silvertop ash and mountain ash), increased net C benefits relative to BAU.

In some cases, changes to the way individual HWPs are managed can have a major impact on the overall GHG balance. For example, favouring the production of electricity poles over sawlogs in the North Coast of NSW resulted in a 30% reduction in GHG emissions. Modelling indicated potential reductions in GHG emissions with changes in assumptions for pallet wood in Victoria, which is currently mulched at the end of its service life, to disposal in landfill or use for bioenergy generation.

The ForestHWP simulation results for each case-study suggest the overall C response to harvesting is context-dependent, and that widely different outcomes are possible depending upon the characteristics of the forest, the harvesting regime, and most importantly the mix of harvested wood products that are produced, their substitution benefits, and their ultimate fate.

Key policy implications

The paper substitution impact is a key component of the GHG balance of sustainably managed forests. Access to existing hardwood plantations in Victoria was not considered economically viable as replacement for the mountain ash and silvertop ash native resource, and there is no realistic prospect for large-scale new plantation establishment. The pulp displacement factor was calculated considering the printing and writing sector of the Australasian market and the implications for sourcing biomass from Asia. The results clearly indicate that ignoring the paper substitution impact would majorly underestimate the current GHG balance of native forestry for the pulp-producing regions.

It is important to adopt a holistic perspective in the analysis of the socio-economic impacts of native forest management, as any potential changes to primary industry production models may have significant implications for regional communities. However in many of the existing analysis of the GHG implications associated with land use options the socio-economic considerations are overlooked. The analysis highlights the importance of considering socio-economic impacts in C policy considerations for regional centres.

The current debate over the C costs and benefits of sustainably managed forests versus management for conservation is overly simplistic; the question is not whether conservation or harvesting produces a more beneficial GHG gas outcome *per se*, because, although harvesting typically produced a more beneficial GHG outcome in the long-term, the outcome could be in either direction depending upon the circumstances. The question should rather be, under what conditions and constraints can timber harvesting, when integrated across the landscape, be optimised to produce the most favourable outcomes.

We demonstrated in this study that one of the ways to enhance the GHG outcome of production forestry is via the increased use of biomass for bioenergy. There are large volumes of harvest slash and mill-based residues available for use. The current business as usual (BAU) scenario for forest harvest residues results in immediate C release (via post-harvest burns), or progressive C loss over time due to natural decay. Similarly the current BAU for much of the available wood-processing residues currently used for low-value applications such as mulch and animal bedding results in release of all the C within 1-3 years, with no net GHG benefit. Thus there are significant opportunities for native forest biomass to play a much larger role in the generation of renewable energy, especially with the recent reinstatement of native forest biomass as an eligible renewable energy source under the Renewable Energy Target (RET). There may also be opportunities in the future for new projects to be supported by a method under the Emissions Reduction Fund (ERF) that credits the fossil-fuel displacement benefits of using biomass for energy displacing the use of fossil fuels, against the baseline of loss of C in the forest via burning or natural decay. This would allow project proponents to choose which scheme (RET or ERF) would be most suitable for a given project.

Conclusions

Under the framework adopted in this study and after considering both BAU and a range of alternative management scenarios, we concluded that the relative differences in the GHG balance of production and conservation scenarios do not warrant policies that aim to halt native forest management for wood production. There is considerable room however for improvement in the GHG outcomes of managing for production, and the work highlights the potential for further industry development that can be coupled with an improved GHG outcome, with multiple benefits. These opportunities could be realised in the forest, in the processing of wood products and in diverting materials to different uses at the end of the life of wood products. The latter involve increased use of biomass for bioenergy, value-adding of processing co-products and changes in waste management. The benefits associated with these activities include:

- Reduction of wastage and increased returns via value-adding of co-products
- Contribution to emission reductions and increased returns by potential participation in national carbon abatement schemes (e.g. RET, ERF)
- Development of new industries, with flow-on benefits to regional centres
- Contribution to energy security

Detailed summary of findings

Background context and study coverage

The potential role of forestry in mitigating climate change, though substantial, has been largely overlooked in Australian climate change policy. As a result, the cost to society of achieving emissions abatement objectives may increase. Forestry can play many roles in mitigating climate change, through maintaining or increasing existing C reservoirs (C in biomass, soil and harvested wood products (HWP)). To quantify the climate change impacts of forestry, we must consider the entire forestry system: the C dynamics of the forest, the life cycle of HWPs, and the substitution benefits of biomass and HWPs. These factors however are sometimes ignored or treated in a simplistic way in much of the available literature. This omission results in an incomplete assessment of the implications of native forest management for wood production, and conclusions that may lead to the promotion of native forest management policy that favours management for conservation outcomes only.

The study assessed changes in C in forests managed for timber production and conservation, taking into account the effects of disturbance (harvest and fire), as well as C storage in HWPs, the substitution effect (for both bioenergy and HWPs) and a cost-benefit analysis. The study sites included three paired sites (“production” and “conservation”), with a known history of disturbances (harvest, thinning and wildfire events). The case studies covered one site on the South Coast of NSW (near Eden), one site on the mid-North Coast of NSW (near Wauchope) and one site in the Central Highlands in Victoria (near Toolangi).

Summary of key findings

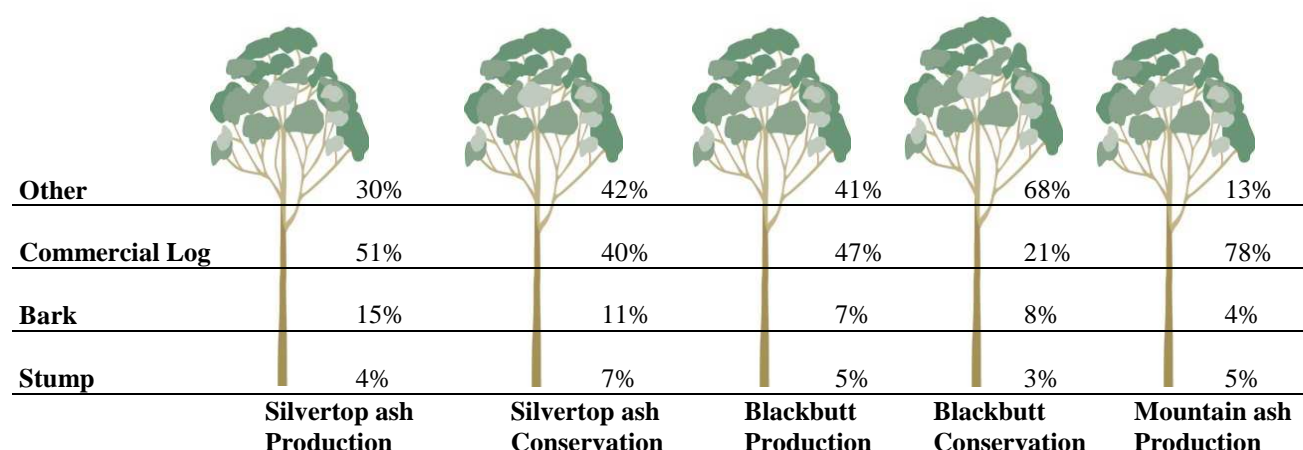
In the sections below we provide brief context and key findings derived from the individual components of the study.

- *Forest biomass*

The lack of directly weighed mature tree biomass is a major impediment for the development of reliable estimates of C stocks in mature native forests. Available allometric equations for key native forest species in Australia typically include trees with a limited DBH range, skewed towards lower DBH classes. Without directly weighed data of mature, large trees, the uncertainty associated with the use of existing allometric equations in the derivation of above-ground biomass for mature forests is high. This is largely because there is insufficient data available to confidently account for the additional variability in the relationship between DBH and biomass due to natural decay (creation of hollows) and loss of limbs in mature trees.

In our study we individually weighed 583 trees, with a range of DBH classes including large mature trees. The mountain ash conservation site was not available for destructive sampling. The breakdown of the biomass components is presented in Figure 1 (proportions based on the total biomass for the site rather than on an individual tree basis). The proportion of biomass in commercial logs from the “production” sites ranged from approximately 50% for the NSW sites to 78% for mountain ash (Figure 1). There was significant variation in the proportion of the above-ground biomass in the bark for the different species – on a dry biomass basis, the values ranged from 4% for *Eucalyptus regnans* (mountain ash) to 7-8% for *Eucalyptus pilularis* (blackbutt) to a high of 11-15% for *Eucalyptus sieberi* (silvertop ash). Considering that in native forest harvest operations the logs are debarked in situ, these figures have a direct impact on the overall production log recoveries for the different species.

Figure 1. Proportion of the biomass allocated to harvest components for the key species from the different case study sites



Comparing the site-specific data with data for standing tree volumes provided by the Forestry Corporation of NSW based on inventory plots in “ready for harvest” mature forest areas showed that the NSW production study sites were representative of silvertop ash and blackbutt forests currently managed for production (Table 1). The percentage breakup of the production and residue components was also comparable with the forecast figures. The Victorian averages provided by VicForests are for pure mountain ash forests for average sized coupes in Toolangi over the past 10 years, and are broadly in agreement with the production site figures in terms of proportion of the production volumes on site (Table 1).

Table 1. Production and residue volumes (excluding dead trees) for all production sites compared to state agency forecasts for similar forest types. Standard errors are shown in parenthesis.

Sites	Volume of commercial logs (m ³ / ha)	Volume of commercial logs (%)	Residue volume (m ³ / ha)	Residue volume (%)	Total Volume (m ³ / ha)
Silvertop ash production	157	55	128	45	285
FCNSW average forecast	165 (17)	56	131 (10)	44	296 (23)
Blackbutt production	177	48	195	52	372
FCNSW average forecast	199 (15)	54	169 (13)	46	368 (21)
Mountain ash production	1107	79	301	21	1407
VicForests Toolangi average forecast	814	75	271	25	1085

The directly weighed biomass figures derived from this study were compared to biomass estimates obtained by applying published biomass equations. Inclusion of large trees in the datasets provided an opportunity to test the robustness of existing equations in predicting biomass for larger trees. For silvertop ash, the published biomass equations, while reasonable at predicting biomass in the lower DBH range, overestimated the biomass as the DBH increased. For blackbutt, all of the equations underestimated both the biomass for trees through the 40-70 cm DBH range and the total biomass for the production site. The biomass estimations for the blackbutt conservation site were more in line with the actual weighed data. For mountain ash trees, an assessment was made for a sub-set of trees with DBH greater than

100 cm – out of the five equations tested, three overestimated biomass to varying extent, one underestimated biomass and one provided a good fit for the data. Thus, biomass estimates of mature native forest stands that are not based on directly-weighed biomass including trees with the largest DBH range are generally not reliable and tend to overestimate biomass.

- *Development of allometric relationships*

For the allometric equations developed from the study data, the impact of using DBH only and DBH and tree height combined as the predictive biomass parameters was tested. Inclusion of height as a combined variable with DBH in the development of the additive biomass equations did not result in higher R^2 for the estimation of whole tree biomass – these were already high with the use of DBH only. Further work is required though to refine the model specifications and parameter estimations from our study, and to refine the handling of data from plots that are spatially clustered – these may improve the correlations further.

- *C storage in harvested wood products (HWP)*

The case study regions varied considerably in the C pathways for the key HWPs produced. To ensure that our analysis of the flow of C from the forest into HWPs reflected current operations, we applied the regions harvest prescriptions and their typical mix of extracted log grades.

The availability of a pulp market for the region is a key factor impacting on production log recoveries. Pulp logs were a major component of the commercial logs extracted for both silvertop ash and mountain ash forests. The ratio of pulp logs to sawlogs (on a C basis) was 70/30 for silvertop ash, and 64/36 for mountain ash. There was no difference between the commercial log recoveries for blackbutt and for silvertop ash (59%) – however if there was a pulp market in the mid-North coast of NSW, the production log recoveries for blackbutt would have been considerably higher. For blackbutt, the volume of sawlogs and poles combined was 53 t C/ ha. Mountain ash had the highest commercial recovery of all the species included in this study.

Key uses of silvertop ash include decking, flooring and structural/cladding products. For blackbutt, there was a greater variety of HWPs produced, with key products including electricity poles, flooring, decking, mining timbers, structural timbers and fencing. For mountain ash, all low-quality sawlogs were processed into pallets, with the high-quality sawlogs being processed into a similar mix of HWPs to blackbutt.

The data used in this study for the determination of the physical C storage in HWPs (other than paper products) was based on information directly supplied by wood-processing facilities and on the latest research findings on the dynamics of the decomposition of HWPs in landfills. Long-term C storage for HWPs in Australia is primarily imparted by the post-service stage of the HWP life (i.e. typically storage in landfill). It is commonly assumed that wood disposed of in landfills decays, generating large quantities of methane; however, research that has specifically targeted the behaviour of HWPs in landfills has demonstrated that C in HWPs in landfills (other than paper products) can be considered to be stored for the long-term, greatly extending the permanence period of C in HWPs.

- *Substitution impacts (bioenergy and HWP)*

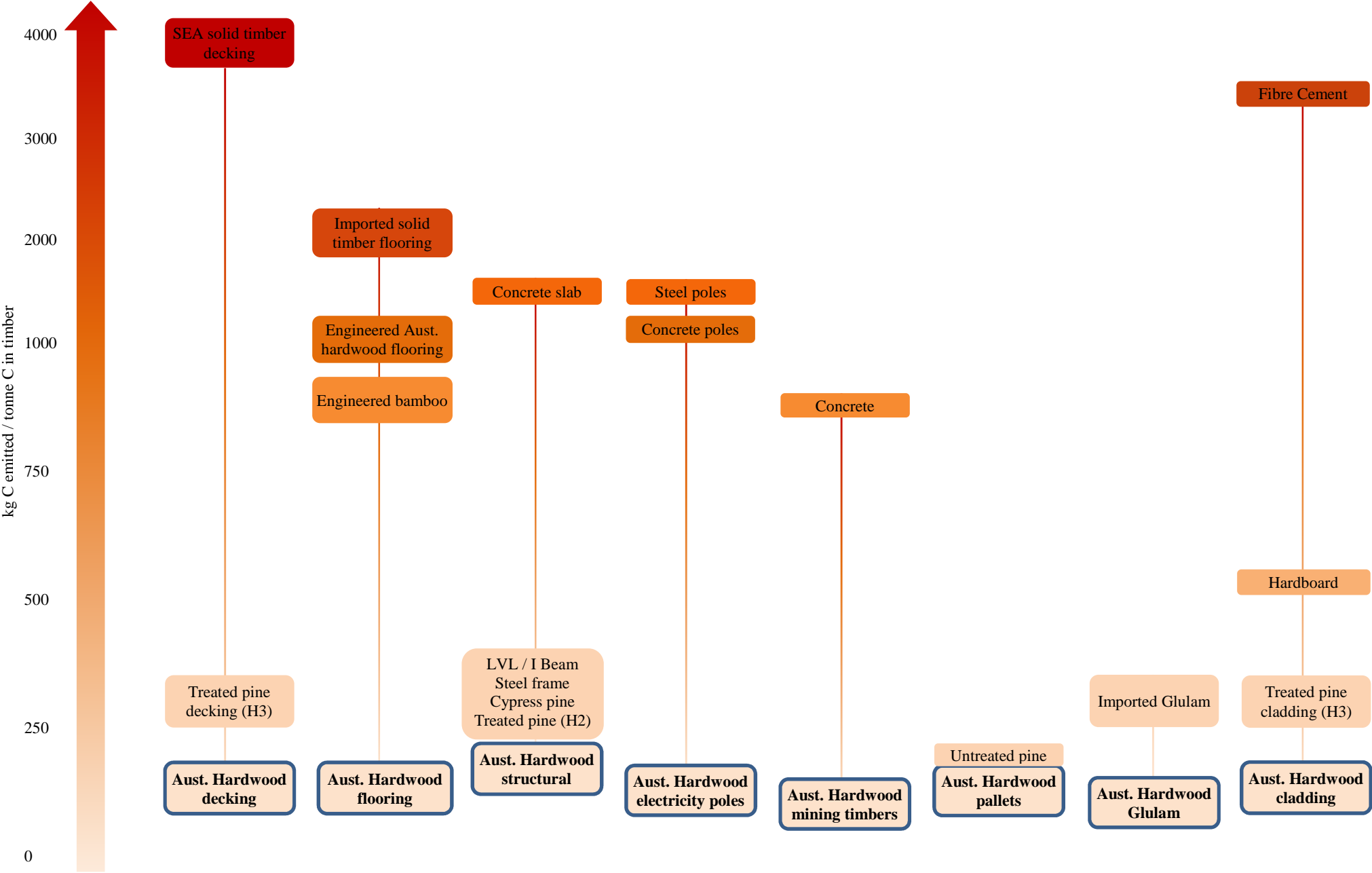
The fossil-fuel displacement factors associated with the use of biomass for bioenergy varied according to assumptions on the use of the biomass; based on current practice, forest biomass was assumed to be used for co-firing with coal for electricity generation; and wood-processing residues and end-of-life wood (as relevant), were assumed to be used either for commercial applications (feedstock for boilers) or in domestic applications (firewood). The

relevant emission factors for each of these utilisation options were included in the calculation of the fossil-fuel displacement factors. Substitution of residential electricity is by far the option which generates the greatest substitution benefit, given the high emission-intensity of coal production required to generate electricity. The results for the regions highlight the significant opportunities that currently exist for native forest biomass to play a much larger role in the generation of renewable energy.

HWPs require comparatively lower fossil-fuel based energy in their extraction and manufacture compared to typically greenhouse-intensive materials such as bricks, aluminium and concrete. The application of the net difference between the emission footprint for HWPs and alternative products is expressed as a product substitution impact. The higher the emission footprint of the alternative products relative to HWPs, the higher the GHG “savings” associated with the use of HWPs. In the calculation of the product substitution factors, region and product-specific emission factors for the key hardwood HWPs were derived. Key alternative products were determined based on extensive market analysis. Replacement markets were often comprised of wood or wood-derived materials, as the native hardwood HWPs often occupy a niche; i.e. consumers who would most likely want a “wood” replacement if they no longer had access to Australian native forest HWPs. Even though the native forest HWPs in most cases had significantly lower emission factors compared to the alternative products identified, the fact that the alternative products were often also HWPs reduced the product substitution impact (with the exception of imported hardwoods from SE Asia and some engineered wood products (EWP)) (Figure 2). This approach contrasts to how product substitution is typically quantified, where product substitution is calculated as the difference in the GHG emission footprint of HWPs versus the use of non-wood materials only.

The substitution impacts on a hectare basis were driven largely by the productivity of the site, the relative efficiencies of biomass recovery in the forest, sawmill recoveries, the types of HWPs produced and the displacement options for each product. The weighted substitution factors ranged from 0.2 t C emitted / t C in HWP for mountain ash to 2.1 t C emitted / t C in HWP for silvertop ash. The high DF for silvertop ash decking (main sawn product for silvertop ash) can be explained largely by the high forest loss emissions associated with the significant proportion of native hardwood from SE Asia that was a likely displacement product. However, given the comparatively low volume of sawlogs per hectare and low sawmill recoveries, the overall product substitution impact for silvertop ash on a hectare basis (6.2 t C / ha) was low compared to other species (9.8 t C/ha for mountain ash and 18.4 t C/ha for blackbutt). The substitution impact, when based on market analyses of product usage in different applications, represents a real mitigation benefit, in the same way the use of sustainably sourced biomass for bioenergy generation represents real mitigation when it displaces the use of fossil fuels.

Figure 2. The emission footprint for Australian hardwood HWPs and their likely replacement products.



- *Substitution impacts (paper products used for printing and writing)*

There is a widespread perception that the GHG mitigation benefits of the production and use of paper products is limited given their typically short service lives. However, the key factor to consider for the determination of the GHG balance of paper products is product substitution. In both the Central Highlands of Victoria and in Eden, the production of pulp logs used in the manufacture of printing and writing paper is part of integrated harvest operations, in forests that are considered to be managed according to international standards for sustainability. Our analysis suggested that the key alternative source of biomass for pulp and paper production was deemed to be primarily in Indonesia. This was based on analyses of the current Australasian supply chain of paper products, use of the existing plantation resource in Australia, the likelihood of the establishment of large scale new plantations, and the current supply of pulpwood from Asia. Pulp and paper production is one of the key industries identified as a driver for deforestation of primary forest, forest degradation and loss of peatlands in SE Asia. It is a reality that paper will continue to need to be produced somewhere into the foreseeable future, especially given the increasing paper consumption forecasts for developing countries such as India and China. If native forest biomass currently sourced from the Central Highlands of Victoria and from Eden was no longer available, logically this would add further pressure to the degraded and depleted forest areas in areas of Asia with comparatively lower standards of forest management.

There is considerable uncertainty about the magnitude of the product substitution factor for paper; thus, every effort was made to ensure that the underlying parameters required for the calculation were conservative. The calculated weighted emission factor for pulp and paper produced in Indonesia ranged from 5.5 to 7.7 t C / t C in pulp logs. The high volume of pulp logs extracted from the Eden and Central Highlands sites, and the high calculated substitution factor for pulp logs resulted in a large substitution impact on a hectare basis. The calculated product substitution impact for pulp production from silvertop ash ranged from 184 t C/ha (peatlands drained) to 252 t C/ha (peatlands burnt). For mountain ash, the figures were much higher, ranging from 1010 t C/ha (peatlands drained) to 1384 t C/ha (peatlands burnt). Ignoring this impact would majorly underestimate the current GHG balance of native forestry in those regions.

- *Key C pathways (after one harvest event)*

In Figures 3-5 we summarise the key pathways for the C from the forests managed for production. The results highlight the diverse nature of the industry in the different regions. Current use of biomass for energy is low in all regions, and the potential for this market is obvious given the existing levels of biomass that are underutilised in all regions. The proportion of C in long-lived HWPs is greatest for blackbutt, and landfill currently represents the main disposal option for those products (with the exception of pallets in Victoria). Paper accounts for approximately 25% and 34% of the total C in the forest–HWP system for silvertop ash and mountain ash forests (Figures 4 and 5).

Figure 3. C flows for blackbutt forests managed for production

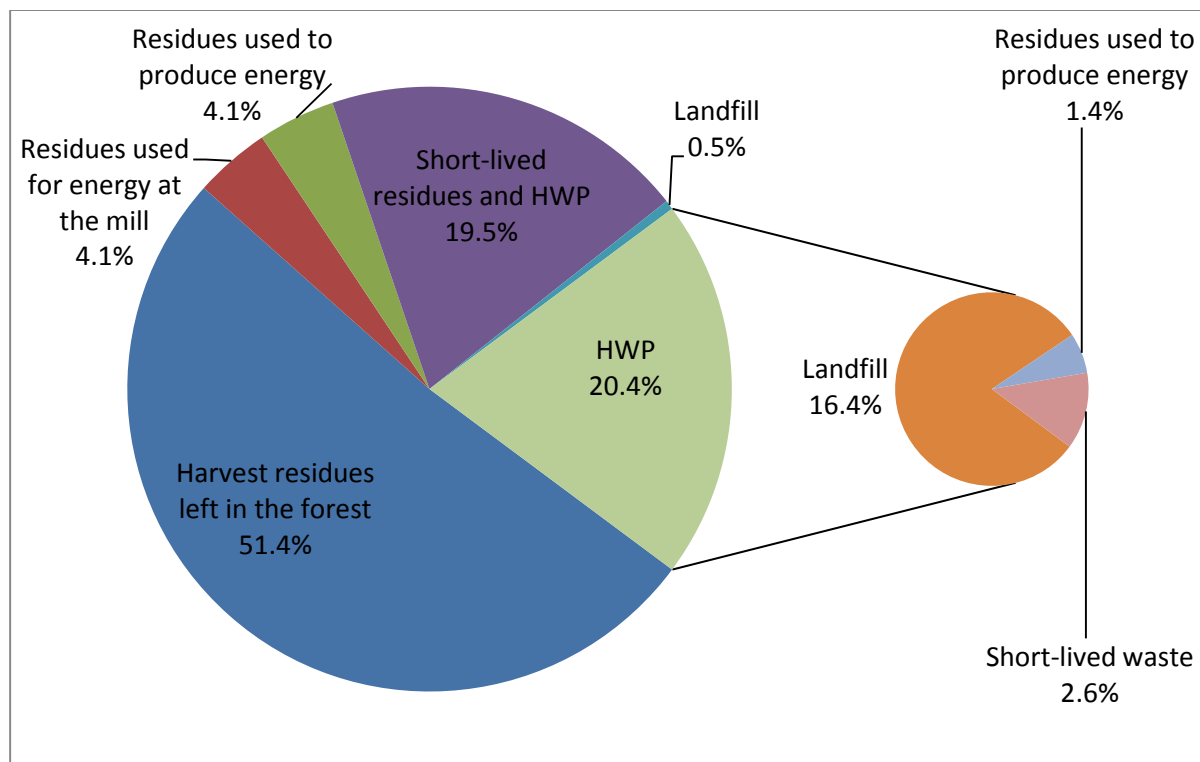


Figure 4. C flows for silvertop ash forests managed for production

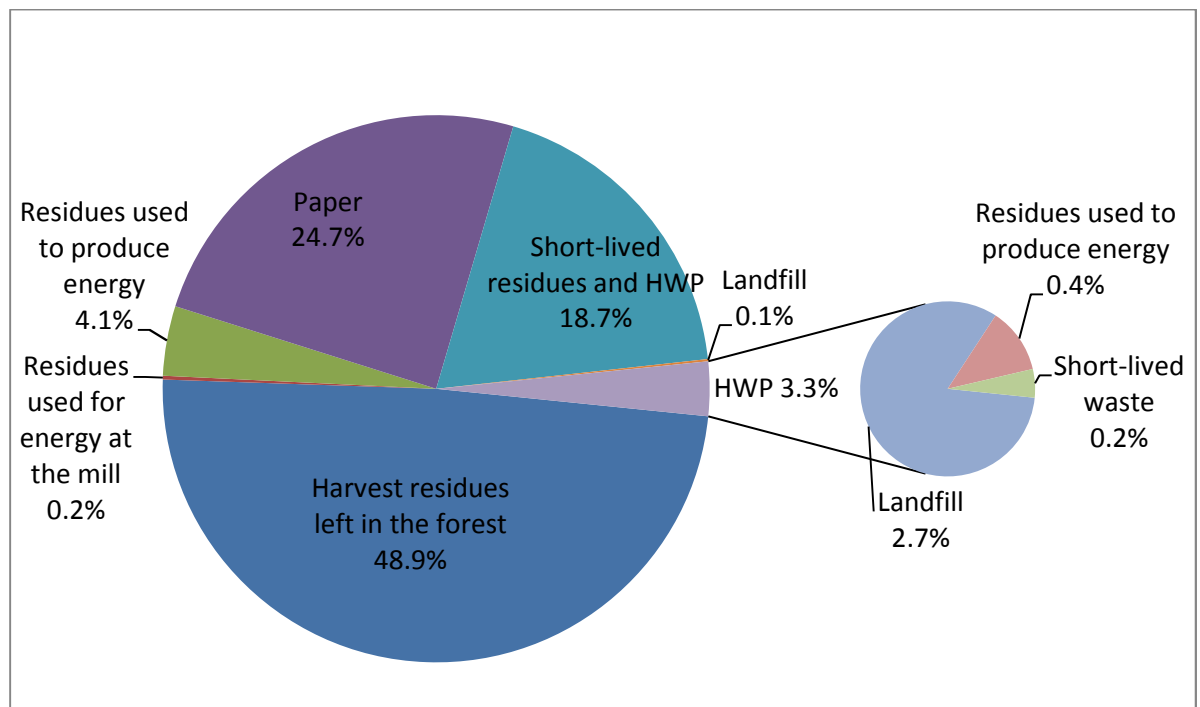
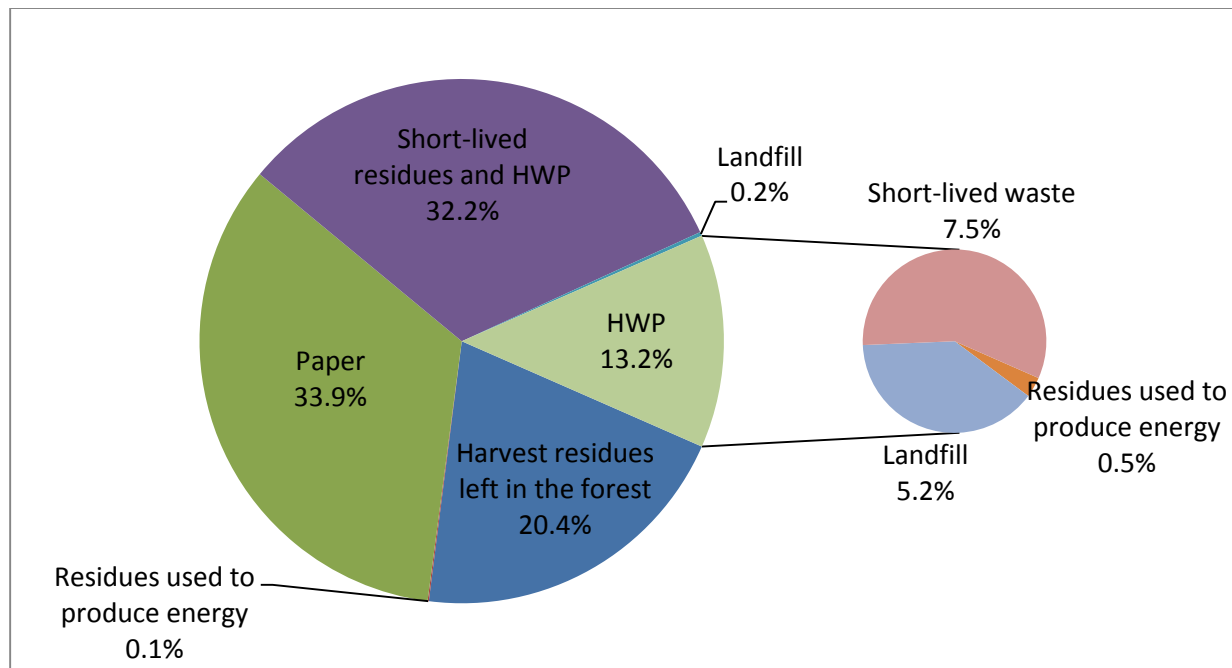


Figure 5. C flows for mountain ash forests managed for production

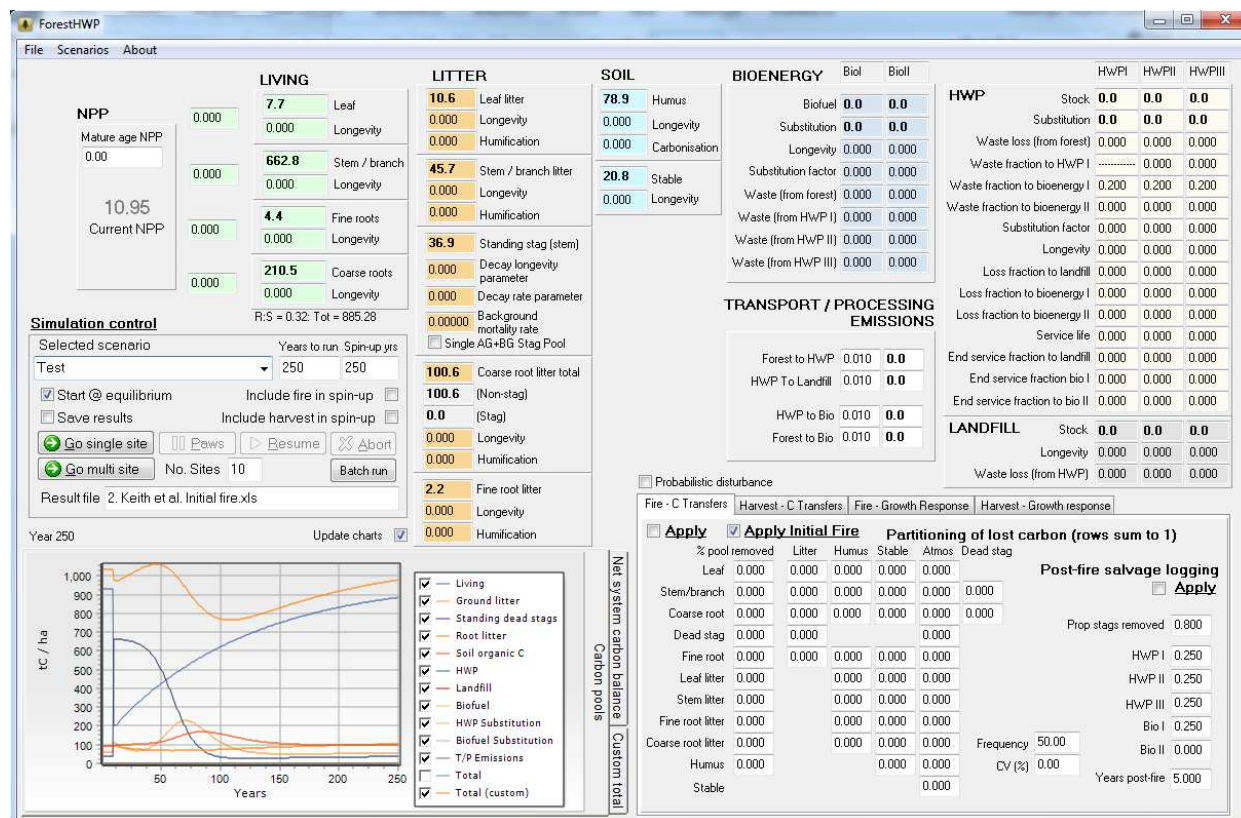


- *ForestHWP – development and verification*

The parameters described above were used in a new model (ForestHWP”) developed to undertake the required integrative analyses (Figure 6). Comprehensive accounting of the whole-of-life GHG balance of production forests requires explicit inclusion of processes and parameters that span the entire forest/HWP system. The development of a new modelling framework can be designed ‘bottom-up’ to ensure that all components of the forest-HWP system are included. This is particularly important, as much of the controversy in recently published research seeking to quantify the full GHG balance of the harvested forest system stems from different studies differentially including/excluding a number of these key processes. Other advantages include the ability to check calculations to ensure they conform to mass-balance principles, and also the flexibility to develop a range of scenarios, and to include a range of potential non-traditional post-harvest pathways such as the utilisation of processing residues for bioenergy, and the calculation of wood product substitution impacts.

Prior to this study, the most comprehensive assessment of the GHG implications of native forest management in Australia was described in Ximenes et al (2012). In that study, the assessment of C flows as a result of native forest management was derived from modelled estimates of the GHG balance of two key native forest areas managed for production in New South Wales for a period of 200 years. The primarily desktop approach in that study differed fundamentally from the one adopted here, where extensive data from intensively measured plots were used to extrapolate to wider sub-regional areas. The selected sites were then used as the benchmark against which the long-term C dynamics in the forest and HWPs were modelled using Forest HWP.

Figure 6. ForestHWP software interface.



- *ForestHWP – modelling of long-term C dynamics from the study sites*

Whilst site-based runs are useful for investigating the temporal dynamics of C in response to e.g. fire and harvesting, it makes comparison across studies difficult, as the C stocks at any given time are a function of the time since last disturbance, and of the overall disturbance regime. To account for this, a number of replicate sites that differed only in the timing of disturbance were run, and the average outcome across those runs is used to allow comparison. This has the effect of averaging over the disturbance-induced fluctuations, thus standardising for the effects of differences in disturbance regime and allowing comparisons across scenarios to be made. For all simulations, 2000 replicate runs were used to quantify the overall scenario outcomes. Scenario results were reported for the predicted C at years 50 and 100, and at year 1000 to provide an estimate of the long term average (LTA) behaviour. For the LTA behaviour, the biofuel offsets, product substitution benefits and the landfill C stocks were set to zero at year 800, and thus the values for these quantities reported for LTA represents accumulation of benefits over a 200 year timeframe. Also, for the LTA behaviour, the C stocks were calculated as the average over the final 200 years of the simulation.

A number of scenarios were considered for modelling with ForestHWP (Table 2). The scenarios were selected based on their likelihood of being implemented in the future. Two of the scenarios can be considered 'reference' or 'baseline'; they are the conservation scenario, where the calibrated models are run with wildfire but without harvesting, and business as usual ('BAU') where both harvesting and fire are included, and where the harvesting parameters are those specific to each case study. There are eight other scenarios that were also applied to each of the case studies, plus a number of scenarios that were specific to each case study. Of the eight shared scenarios, one explored the potential impacts of moving some processing waste and products that had reached the end of their service life into residential bioenergy (Table 2, Scenario 3). There were three scenarios that sought to maximise product

recovery (i.e. increased use of processing waste to generate long-lived products), landfill and biofuel benefits (Table 2, Scenarios 4-6, respectively). Finally, four scenarios were designed to investigate the implication of increasing the incidence of fire (two levels), applied to Scenarios 1 and 2 (Table 2, Scenarios 7-10). The “maximise landfill” scenario only considers long-term C storage in HWPs other than paper products. The rationale for the scenarios that aimed to “maximise landfill” and “maximise bioenergy” was based on the fact that the “maximise landfill” scenario in effect maximises physical C storage as an outcome, whereas the “maximise bioenergy” scenario maximises the C benefit by virtue of fossil-fuel displacement. Thus, the “maximise landfill” scenario is independent of fossil-fuel displacement assumptions, whereas the impacts associated with the “maximise bioenergy” scenario are highly dependent on the types of fossil fuel displaced.

Across all scenarios there is variability over the first 100 years of the simulation; for example in the Victoria case study there is a tendency for C stocks to initially increase, reaching a peak at approximately year 100, before declining again. This pattern reflects the non-equilibrium starting point of the simulations, and in this case is a function of the fire and management histories of the sites post-1939. To exclude such transient dynamics from the comparisons, and thus to ensure that the differences across case studies can be attributed solely to changes in simulated management regime, comparisons amongst the case studies were therefore made based on the LTA summaries.

The accrued GHG benefits were greatest for mountain ash, and slightly higher for silvertop ash compared to the conservation scenario. In both cases this benefit was driven primarily by the substitution factor for pulp. For blackbutt, the BAU production scenario resulted in approximately 12% lower GHG benefit compared to the “conservation scenario; however the BAU harvesting scenario is affected by the current absence of a market for pulpwood and the delay in the introduction of a bioenergy market for that resource. The expectation is that this change will take place in the short to medium term, resulting in BAU scenarios closer to the “50% of forest residue to bioenergy” or “50% of forest residue to pulp” scenarios.

For the pulp-producing areas, a change from pulp production to energy generation resulted in a significant reduction in the GHG benefit, due to the very high product substitution impact associated with paper products. In general, strategies that a) involve increased utilisation of biomass for bioenergy production (without displacing pulp), b) extend the longevity of C in HWPs (by changing production from short-lived products to medium or long-lived products and by storage in landfills) and c) generally minimise waste, will all contribute to a greater net C benefit for all systems studied. This suggests there is considerable room to improve the GHG benefits associated with managing forests for production.

In some cases, changes to the way individual products are treated can have a major impact on the overall GHG balance. For example, favouring the production of electricity poles over the extraction of sawlogs in the North Coast of NSW results in a net C benefit over the long term equivalent to approximately 70% of the long-term average above-ground C stocks for the blackbutt production forests.

Table 2. Summary of the scenarios used in ForestHWP

	<i>Scenario</i>	<i>Description</i>
	<i>Scenarios common to all three case studies</i>	
1	Conservation	Wildfire included but no harvesting.
2	BAU (Business as Usual)	Wildfire included and harvesting as per the observed product recoveries, waste losses, bioenergy usage etc.
3	EoL products and waste to bioenergy	At the end of service life (EoL) all available product is utilised for residential bioenergy, and 70% of dry and green processing waste is utilised for residential bioenergy.
4	Maximise product recovery	70% of dry and green processing waste re-utilised for additional dry product. An example would be use of residues to produce engineered wood products
5	Maximise landfill	At the end of service life all available product is sent to landfill, 70% of dry and green processing waste re-utilised for additional dry product (which eventually ends up in landfill)
6	Maximise bioenergy	At the end of service life all available product and 100% of dry and green processing waste is utilised for residential bioenergy
7	Fire x 1.25 (Consv)	Average fire return time decreased by 0.8 (=1/1.25)
8	Fire x 1.5 (Consv)	Average fire return time decreased by 0.667 (=1/1.5)
9	Fire x 1.25 (BAU)	Average fire return time in the BaU scenario decreased by 0.8 (=1/1.25)
10	Fire x 1.5 (BAU)	Average fire return time in the BaU scenario decreased by 0.667 (=1/1.5)
	<i>Victorian case study scenarios</i>	
11	30% forest residue to bioenergy	30% of forest residues left on site utilised for co-firing with coal for electricity generation
12	50% pulp to bioenergy	50% of the material removed from the forest for pulp is instead utilised for residential bioenergy.
13	100% pulp to bioenergy	100% of the material removed from the forest for pulp is instead utilised for residential bioenergy.
14	EoL pallets to landfill	At the end of service life all pallets (=green) are sent to landfill.
15	EoL pallets to bioenergy	At the end of service life all pallets (=green) are utilised for bioenergy.
	<i>NSW North Coast case study scenarios</i>	
16	50% forest residue to bioenergy	50% of forest residues left on site utilised for co-firing with coal for electricity generation
17	50% forest residue to pulp	50% of forest residues left on site utilised for pulp
18	Increase product to poles	The regional product mix used in the simulations is replaced by the product mix as observed at the coupe-level, which increases the proportion of pole manufacture over 6x.
19	<i>NSW South Coast case study scenarios</i>	
20	30% forest residue to bioenergy	30% of forest residues left on site utilised for co-firing with coal for electricity generation
21	50% pulp to bioenergy	50% of the material removed from the forest for pulp is instead utilised for residential bioenergy.
22	100% pulp to bioenergy	100% of the material removed from the forest for pulp is instead utilised for residential bioenergy.

In Table 3 the LTA C values for the BAU and changed management scenarios are listed.

Table 3. Comparison of the implications of changed management with the BAU scenarios. The BAU value is the LTA total C values. The values for the *Changed management scenarios* are the differences between each scenario and BAU Harvesting (in t C/ha), and the % change (in parentheses). Red values indicate declines relative to BAU.

	Victoria Central Highlands	North Coast Blackbutt	South Coast Silvertop Ash
BAU Conservation (tC/ha)	522.8	247.5	276.7
BAU Harvesting (tC/ha)	835.2	218.5	288.4
<i>Management Scenarios (difference from BAU Harvesting)</i>			
30% of forest residue to bioenergy	10.3 (1.2)	NA	17.0 (5.9)
50% of forest residue to bioenergy	NA	38.4 (17.6)	NA
50% of forest residue to pulp	NA	104.7 (47.9)	NA
50% pulp to bioenergy	-144.1 (-17.3)	NA	-32.8 (-11.4)
100% pulp to bioenergy	-274.6 (-32.9)	NA	-65.7 (-22.8)
All end-of-life products, and all processing waste to bioenergy	23.7 (2.8)	25.9 (11.9)	9.1 (3.2)
Max product to bioenergy	32.9 (3.9)	61.1 (28.0)	12.8 (4.4)
End of service life pallet mulch to bioenergy	10.9 (1.3)	NA	NA
Waste to product (Max product)	13.0 (1.6)	63.0 (28.8)	29.6 (10.3)
Max product to landfill	40.0 (4.8)	64.1 (29.3)	30.6 (10.6)
End of service life pallet mulch to landfill	18.3 (2.2)	NA	NA
Adopt site product spread to yield a greater % of poles	NA	67.9 (31.1)	NA

Considerations on disposal options for HWPs are critical for the analyses of the GHG balance of native forest management. Long-term C storage in HWPs in Australia is largely driven by the C storage that happens in landfills. However, the overall GHG impact will be beneficial whether the product is recycled into another long-lived application (e.g. old floorboards used in recycled furniture), landfilled or burnt to produce energy. In fact, the emission abatement created by diverting wood waste from landfill to energy generation facilities (resulting in fossil fuel displacement) can be higher than if the product is placed in landfill, depending on whether/which fossil fuels are displaced. The relative GHG benefits of energy generation and landfilling will depend primarily on the energy profile of the region. For example, if the HWP was used in Tasmania, where the energy used is predominantly hydro-based (and thus with a low emission profile), landfilling may be preferable from a GHG perspective. In Victoria however, the predominance of brown coal for energy generation would suggest potentially a more beneficial outcome if the HWP was used for energy generation. It is important to note that landfilling results in actual physical storage of C for the long-term, guaranteeing that the C will not be emitted. The net benefits of bioenergy generation as noted above will depend primarily on the alternative energy sources for the particular region.

There are, unsurprisingly, a number of caveats and uncertainties associated with the modelling approach taken here. These arise from the simplifications that are required when translating the complexity of the harvested forest system into a relatively simple set of mathematical equations. Potential sources of uncertainty include the values of parameters that define the stocks and fluxes of carbon, as well as the exclusion of processes that impact on forest growth and structure, such as the impacts of climate change. The overall implications of these uncertainties on the conclusions remains unknown; however the analyses as presented reflect the impacts of different management options as based on the best available data, and although introduction of more complex uncertainty analyses would likely show some of the options to be statistically equivalent in their outcomes, their relative rankings and magnitudes when compared against the BAU scenario would not be expected to change.

- *Socio-economic impacts*

In addition to determining the physical dynamics of C flows in forests and HWPs, it is important to consider the socio-economic implications of native forest management, as any changes to existing primary industry production models may have considerable implications for regional economies. Socio-economic considerations are often overlooked in discussions around the GHG implications of land management. For NSW native forests the socio-economic benefits of management for timber production were higher than those of management for conservation only, with the State Forests on the mid North Coast of NSW generating more socio-economic value than those at Eden. The calculated cost (economic impact) of converting production forests to conservation-only forests for Eden was \$64M and the loss to the regional economy was \$308M. For the mid-north coast, the calculated economic impact of transition was \$540M and the loss to the regional economy was \$3.36B. Transitioning to management for conservation only incurred a sharp decline in regional employment in both regions.

It has been argued that managing forests for conservation only will provide C benefits, which may be associated with a monetary value. It is important to understand how this hypothetical value may be considered against the impacts of reducing or stopping production on the primary and secondary market chains associated with the production of HWPs from native forests. Valuation of the carbon abatement benefits relative to business as usual (BAU) were derived using a 65-year modelling period and a low (\$10/tCO₂-e), medium (\$20/tCO₂-e), and high (\$30/tCO₂-e) carbon price.

Assuming a carbon credit was available for avoided emissions by stopping harvest we calculated that a carbon price of \$233 per tCO₂-e (excluding transition costs) was required to support the conservation scenario on the north coast site. At the Eden site the conservation scenario generated less carbon abatement than BAU and so value was lost (rather than gained) when carbon was priced.

When industry value-added benefits (based on BAU) and carbon abatement benefits were added together, the production management scenarios generated much higher values than the conservation management scenarios. This result was independent of the carbon price (low, medium or high).

Conclusions

The overall conclusion of this study is that the relative differences in the GHG balance of native forests managed for “production” or “conservation” only do not warrant policies that aim to halt native forest management for wood production. In addition to timber production objectives, management of production forests in the study regions also incorporates non-production considerations. These include retention of ecological values, wildfire management, positive socio-economic implications for the regions, generation of revenue to support trained fire-fighting crews and maintenance of roading networks required for quick access to fire fronts.

There is considerable room however for improvement in the GHG outcomes, and the work highlights the potential for further industry development that can be coupled with an improved GHG outcome, with multiple benefits. These opportunities, are present at the forest, processing and disposal levels, and involve primarily increased use of biomass for bioenergy, value-adding of processing co-products and changes in waste management. The benefits associated with these activities include:

- Reduction of wastage and increased returns via value-adding of co-products
- Contribution to emission reductions and increased returns by potential participation in the Emissions Reduction Fund
- Development of new industries, with flow-on benefits to regional centres
- Contribution to energy security

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Introduction

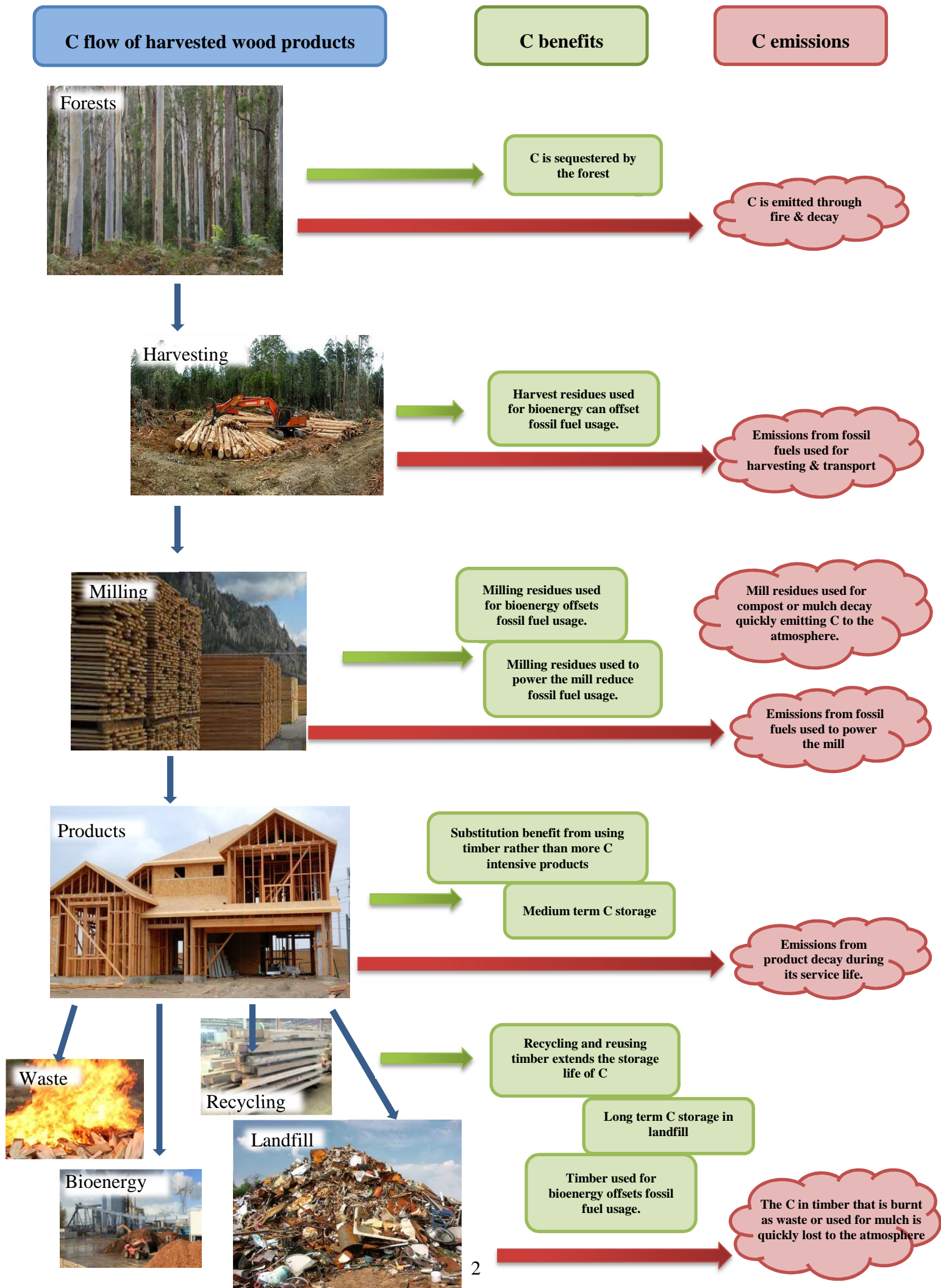
The potential role of forestry in mitigating climate change, though substantial, has been largely overlooked in Australian climate change policy. As a result, the cost to society of achieving greenhouse gas (GHG) emissions abatement objectives may increase. Forestry can play many roles in mitigating climate change, through maintaining or increasing existing C (C) reservoirs, (C in biomass, soil and harvested wood products (HWP)). To quantify the climate change impacts of forestry, we must consider the entire forestry system: the C dynamics of the forest, the life cycle of HWPs, and the substitution benefits of biomass and HWPs (Figure 1). When we quantify the potential implications from changing forest management, we must, similarly, consider the system as a whole, including impacts resulting from indirect land use change and product substitution, and disturbance events. This is important, as much of the controversy in recently published research seeking to quantify the full GHG balance of the harvested forest system stems from different studies differentially including/excluding a number of these key processes. Consideration of socio-economic impacts is also important.

According to the Australia's State of the Forests Report (ABARES 2013), the total area of Australia's native forest both available and suitable for commercial wood production was 36.6 million hectares in 2010–11. This includes 7.5 million hectares of public native forests and 29.1 million hectares of leasehold and private tenure forests both potentially available and suitable for commercial wood production. New South Wales and Victoria combined account for approximately half the total area of multiple-use forests available for harvest in Australia. When harvest restrictions required to maintain non-wood values are taken into account, the net area available and suitable for harvest in multiple-use public native forest (the net harvestable area) is 5.5 million hectares. The area of multiple-use public native forest harvested in Australia in 2010–11 was 79 thousand hectares (approximately 1.4% of the net harvestable area). The main types of logs harvested from Australia's native forests are sawlogs and pulplogs. Other products such as poles used for electricity transmission are dependent on suitability of the species, and market demand.

There are limited reliable datasets describing the above ground biomass of mature Australian native hardwood forests. Roxburgh *et al* (2006) determined the C stocks of 17 sites previously subjected to varying degrees of harvest on the south coast of NSW (Kioloa area) by field survey measurements, with a mean of 214 tonnes above-ground C/ha (excluding coarse woody debris). A study conducted in a similar area (Ximenes *et al* 2005) and based on direct biomass weighing demonstrated that a mature high quality eucalypt forest on the south coast of NSW contained around 200 tonnes above-ground C/ha. The lack of directly weighed mature tree biomass is a major impediment for the development of reliable estimates of C stocks in mature native forests. Typically available allometric equations for key native forest species are limited by the DBH of the trees that may have been included in the development of the equation. Without directly weighed data of large, mature trees, the level of confidence associated with applying existing allometric equations in the derivation of above-ground mature forest biomass is limited.

There are significant differences in the scope of published studies that have attempted to quantify the GHG implications of native forest management (e.g. Dean *et al* 2012; Ximenes *et al* 2012; Keith *et al* 2014). Omission of key components of the C cycle in managed native forests has potentially significant implications in the conclusions drawn. A definitive quantification of the impacts of harvesting, or of the potential response to the removal of a management activity, cannot be made without full accounting of the C gains and losses associated with disturbance events such as harvesting activity, measured against a background of natural variability.

Figure 1. Typical C flows of native hardwood HWPs.



There is also limited information on the long-term physical C storage associated with HWP, and their substitution impacts. It is commonly assumed that physical C storage in HWP is limited to the period in which they are in service. However, the fate of the products after they reach the end of their useful service lives is critical in the determination of long-term storage levels, depending on whether the product is recycled, re-used or disposed of in landfills (Ximenes *et al* 2012).

HWP typically require comparatively low fossil-fuel based energy in their extraction and manufacture compared to greenhouse-intensive materials such as aluminium and concrete. In their meta-analysis of twenty-one European and North-American studies, Sathre and O'Connor (2008) suggest that on average, for each tonne of C in wood products substituted for non-wood products, a GHG reduction of 2.1 tonnes of C is achieved. Similarly, it is important to account for the GHG benefits of the use of biomass for energy that displaces the use of fossil fuels, as the values can be significant (e.g. Ximenes *et al* 2012).

There is a widespread perception that the GHG mitigation benefits of the production and use of paper products is limited given their typically short service lives. Whilst this may be true if only the pool of paper products in service is considered, the product substitution impacts associated with the use of paper products manufactured from native forest biomass in Australia may be significant when the main alternative product is pulp from deforested or degraded areas in SE Asia. Tropical deforestation rates are still high, and the pulp and paper industry is identified as one of the drivers for it (e.g. Lawson *et al* 2014; Persson *et al* 2014).

In addition to determining the physical dynamics of C flows in forests and HWP, it is important to consider the socio-economic implications of native forest management. Any changes to existing primary industry production models may have profound implications for regional economies. It is important to adopt a holistic perspective in the analysis of the socio-economic impacts of native forest management. It has been argued that managing forests for conservation only will provide C benefits, which may be associated with a monetary value. It is important to understand how this hypothetical value may be considered against the impacts of reducing or stopping production of HWP from native forests on the primary and secondary market chains associated with that production.

This is the first assessment of the GHG balance of native forest production in Australia that considers all the key relevant elements of the C cycle in native forests and HWP, plus a cost-benefit analysis that includes socio-economic considerations and C pricing implications. This project is a collaboration between the NSW Department of Primary Industries (NSW DPI), Forestry Corporation of NSW (FCNSW), VicForests, CSIRO, Victorian Department of Sustainability and Environment (DSE) and the Australian Forest Products Association (AFPA). The key aim was to determine C stocks and fluxes in dry sclerophyll forests in SE Australia (NSW and VIC), assessing changes in C in forests managed for production and conservation only, taking into account the effects of harvest, as well as C storage in HWP and the substitution effect.

The native forest areas in NSW and VIC included three paired sites (with each pair including a site that managed for production and the other with minimal or no management history), and with a known history of disturbances (harvest, thinning, prescribed burning and wildfire events). The case studies covered one area on the South Coast of NSW (near Eden), one area on the mid-North Coast of NSW (near Wauchope) and one area in the Central Highlands in Victoria (near Toolangi). A complete accounting is important, as much of the debate

regarding GHG balances in forests subject to harvesting system stems from different studies including or excluding a number of these key elements from their accounting system, and from the indiscriminate application of default values in the analyses.

For each site, the following parameters were developed:

1) *Total above-ground biomass using a biomass-weighing trailer*

This component of the project applied the methodology used in previous research (Ximenes *et al* 2005). Key components of that methodology included the selection of a representative area (approximately 0.5 ha), with all trees individually identified, numbered and measured (DBH and height). Trees and tree components were individually weighed on a biomass-weighing trailer (with the exception of the conservation site at the Central Highlands), and samples of various biomass fractions obtained to determine the moisture and density of the various tree species. These analyses were coupled with the results of analysis conducted for the Federal Department of Environment on the coarse woody debris and litter levels of the same forest types.

The site-specific forest and commercial log biomass values, while very accurate, were not replicated and so a comparison of typical values for similar forest types was conducted to ensure that the values were within the typical range for those forests. In order to mirror harvest prescriptions applied in harvest operations in each relevant region, we adjusted the figures to exclude retained trees. The product mix was also adjusted to reflect the typical mix of commercial logs for each region, in consultation with the FCNSW and VicForests. The adjusted biomass values and product mix were used in the modelling simulations described below to ensure they were regionally relevant.

2) *Site-specific allometric relationships for key species*

This component of the project involved the development and refinement of height-diameter and diameter equations for key native forests species in NSW and VIC, which are not well represented in the existing allometric equations for native species in the study areas.

3) *Long-term C storage and product substitution effect associated with the use of HWPs*

C storage in HWPs was determined for the specific product types from each region as required. This took into consideration biomass flows into the various log products, the mix of the various green and dry and dressed HWPs produced in the various wood-processing facilities, generation of residue and their respective uses, use and disposal of HWPs. Final determination of long-term C storage in the various HWPs was primarily a function of the disposal pathway for each individual product.

The GHG substitution impacts of the use of HWPs from the study regions were determined using market-based information and relevant life-cycle emission data. This took into account the likely substitution scenario for HWPs for each product region (e.g. if native forest harvest was no longer carried out in the mid-north coast of NSW, what would be the most likely product substitution scenarios for blackbutt floor boards, and what would be the GHG implications of such scenarios?).

4) *Product substitution impacts for paper products*

The product substitution impacts for paper products from the case study regions was determined based on analyses of the use of the pulp produced, and a market analysis of the likely replacement scenarios. The GHG balance of extraction of pulp logs to produce the

alternative paper products was based on published data. In the calculation of these factors, we only accounted for emissions associated with forest management. The emissions associated with the manufacture of the paper at the mill were assumed equal in all scenarios. The substitution factor for pulp logs was calculated as the difference in forest management GHG emissions between our case study sites and in SE Asia.

5) *Development and validation of the ForestHWP model*

Comprehensive accounting of the whole-of-life GHG balance of production forests requires explicit inclusion of processes and parameters that span the entire forest – HWP system. This includes:

- * forest growth/decay processes and natural disturbance,
- * the impacts of harvesting on the forest system,
- * accounting for HWPs and their fate, which requires the inclusion of:
 - any emissions related to harvesting, transport and processing,
 - any fossil-fuel displacement benefits from the use of biomass to generate energy,
 - the substitution of wood products for non-wood alternatives.

FullCAM is the model used by the Australian Government to construct Australia's national GHG emissions accounts for the land sector. The original intention was to use FullCAM to model the data from the case studies; however a closer examination of the current version of FullCAM suggested that its capabilities were not ideal for the comprehensive accounting supported by the data developed in this study. The development of a new model (ForestHWP) is described in detail in Section 5.

6) *Modelling the C balance and scenario modelling using ForestHWP*

The main aim of this component of the work was to integrate field data and other components from the project's case studies into a modelling framework capable of including all of the key processes and parameters associated with the forest-HWP system (ForestHWP); in order to provide the capacity to develop comprehensive, long-term GHG accounts for these native production forests, and to use the modelling framework to explore current and potential future harvesting and HWP scenarios.

7) *Socio-economic impacts*

The socio economic implications of managing the forests represented in the case studies for production or conservation purposes alone were investigated. Key considerations included costs of managing the land, impacts on both direct and indirect employment and net present value. These values were aggregated, and applied to determine what the C value would need to be to make managing for conservation a more attractive proposition from an economic perspective than managing for production.

The findings from this project will assist the forest industry to understand climate change policy development that affects its future, and provide guidance on the implications of changes in how the forests and derived HWPs are currently managed on the GHG mitigation credentials of the forest industry, thereby reducing risks and maximising the opportunities for the industry.

Part 1. Total above-ground biomass

Fabiano Ximenes, Michael Maclean, Dave Sargeant, Rebecca Coburn, Matthew Mo (NSW DPI); Michael Ryan (VicForests); Justin Williams (Forestry Corporation of NSW); Ken Boer (previously FCNSW)

In this section we present the results of the total above-ground biomass determinations for the three regions include in the study. This work forms the basis for the analysis presented in the subsequent sections of the report.

1.1 Section summary

- Three paired sites of approximately 0.5 ha each were selected for the study, with each pair including a site that had been managed for production and the other with minimal or no management history. The study sites were located in Eden (NSW), mid-North Coast of NSW (near Wauchope) and the Central Highlands in Victoria (near Toolangi).
- All trees with DBH greater than 10 cm were individually weighed on a biomass-weighing trailer (with the exception of the conservation site at the Central Highlands). The weight of commercial logs and different residue fractions was determined. Above-ground biomass (AGB) was derived after moisture content and density determinations.
- Key findings:
 - Stand density was typically lower and the basal area typically greater for the conservation sites. The silvertop ash production site had the lowest stand density, basal area and average tree height of all the production sites.
 - The production sites were representative of managed silvertop ash, blackbutt and mountain ash forests, based on a comparison with forecast standing tree volumes provided by the State Forest management agencies for similar sites.
 - There was significant variation in the proportion of the AGB in the bark for the different species; the values ranged from 4% for mountain ash, 7-8% for blackbutt, to 11-15% for silvertop ash.
 - The moisture content and basic density of the wood typically decreased from the stump to the crown for all species.
 - The biomass in the NSW conservation sites was significantly higher than that of the NSW production sites. The mountain ash production site had nearly double the biomass of all the other sites reflecting the high productivity of mountain ash.
 - The proportion of biomass in commercial logs from the “production” sites ranged from approximately 50% for the NSW sites to 78% for mountain ash.
 - The biomass estimates from the study sites, although in a few instances consistent with published values for similar forest types, varied considerably with most published values.

- Comparison of the directly weighed data with existing biomass equations revealed that biomass estimates of mature native forest stands that are not based on directly-weighed biomass and that include trees of large DBH are generally not reliable.

1.2 Methods

The following section includes a description of the study sites harvested (Figure 1.1), preliminary total fresh biomass results and an analysis of the production logs obtained. The sites were selected in consultation with local foresters to ensure they were representative of the overall forest stands in the region. The representativeness of the sites was tested again after the biomass determinations by comparisons with sub-regional values provided by the relevant State agencies responsible for managing native forests for production. In Table 1.1 key descriptive information is summarised.

Figure 1.1 Location of study sites

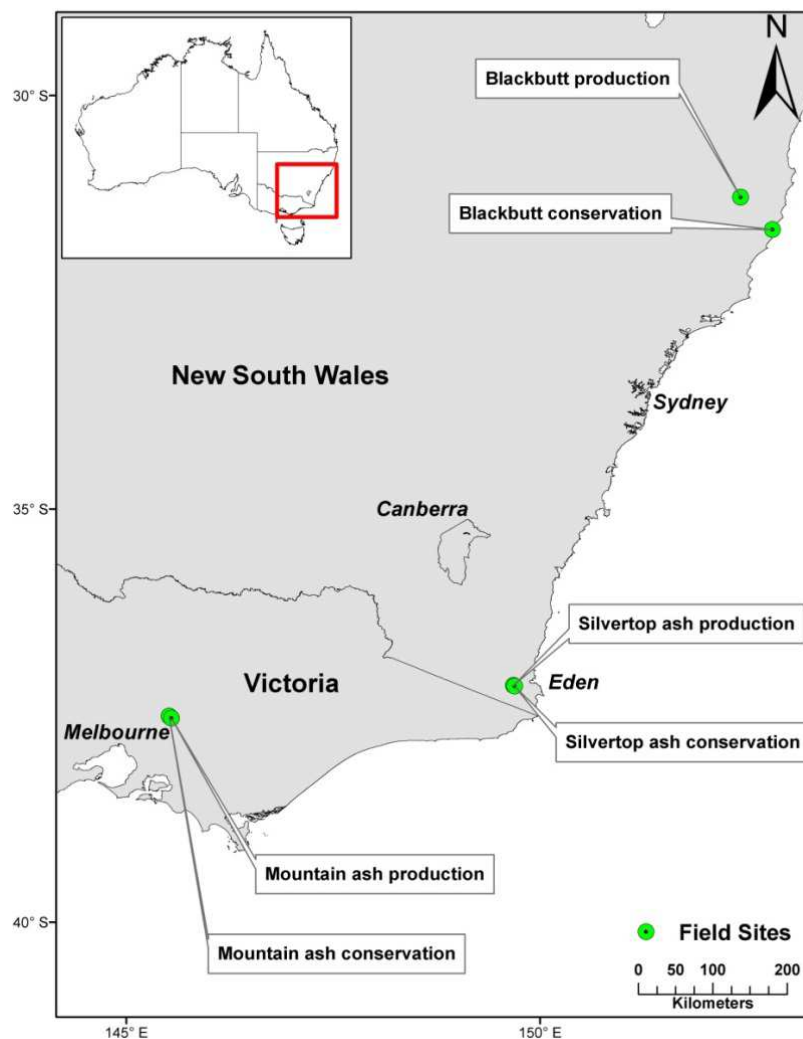


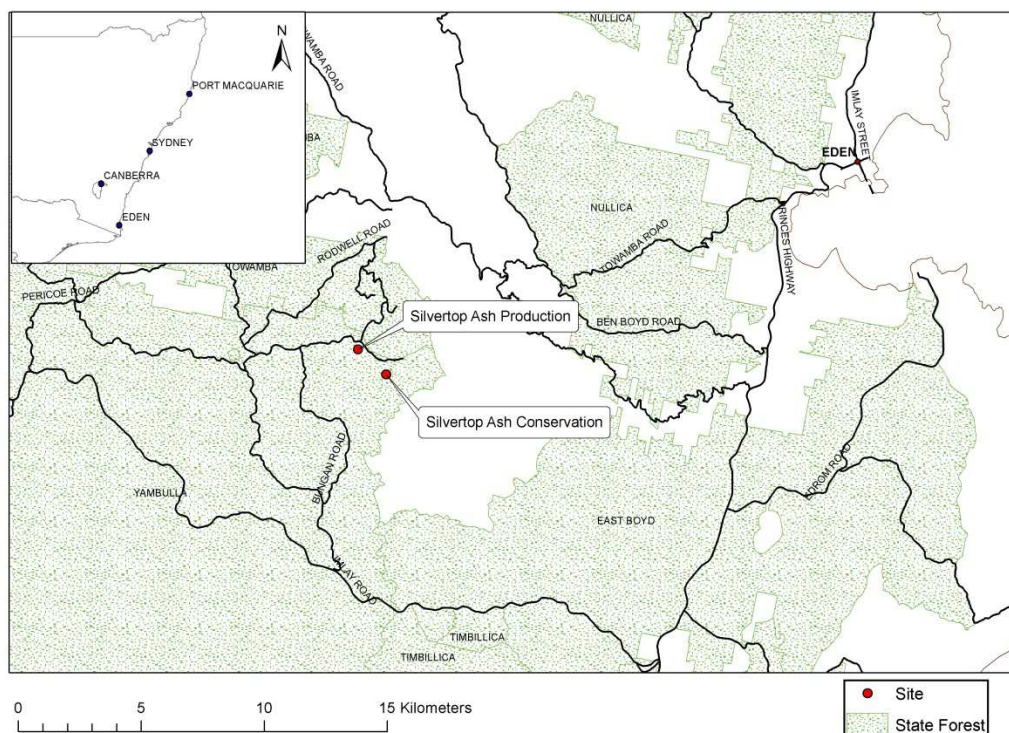
Table 1.1 Site information

Site	Location	Stand age (circa)	Plot dimensions and area	Mean annual rainfall (mm)	No. trees in plot
Silvertop ash production	55S 737919E 5886641N	1950	100.3m x 46.8m (0.47 ha)	1000	103
Silvertop ash conservation	55S 739065E 5885619N	1950-60; 200-250 years*	101m x 47.5m (0.48 ha)	1000	80
Blackbutt production	56S 445400E 6544700N	1950	108m x 46.5m (0.5 ha)	1283	153
Blackbutt conservation	56S 482200E 6501500N	1920-30; >200 years*	80m x 60m (0.48 ha)	1548	128
Mountain ash production	55S 371474E 5845398N	1939	100m x 53m (0.53 ha)	1372	122
Mountain ash conservation	55S 368757E 5847893N	1905-06	100m x 50.5m 0.5 ha	1372	91

* Multi aged stand

1.2.1 NSW South Coast – Eden – Silvertop ash

Paired production and conservation sites were selected to represent “conservation” and “production” scenarios (more details below). The paired NSW south coast biomass sites were located at Yambulla State Forest, approximately 30 km south west of Eden (Figure 1.2).

Figure 1.2. Location of the silvertop ash sites

Silvertop ash production site

The silvertop ash production site was located on a relatively level area, on sandy soil, situated on the south side of Black Range Rd, with most trees being regrowth from approximately 1950. The dominant species was silvertop ash (*Eucalyptus sieberi*) with occasional yellow stringybark (*Eucalyptus muellerana*), river peppermint (*Eucalyptus elata*), messmate (*Eucalyptus obliqua*) and narrow leaf peppermint (*Eucalyptus radiata*). Shrub cover was dominated by saw banksia (*Banksia serrata*) and wattle (*Acacia spp.*), while groundcover was sparse to moderately covered with ferns and grasses. There was evidence of past bushfires and logging. The site was in a compartment scheduled for harvest as part of the Forestry Corporation of NSW (FCNSW) Eden logging operations.

Silvertop ash conservation site

The silvertop ash conservation site (Figure 1.2) was located on the upper slope of a north-western facing rocky ridgetop off Skink Rd. The site contained a number of mature trees with DBH >100cm. The forest was dominated by silvertop ash, with occasional river peppermint and messmate. Shrub cover was dominated by *Acacia sp.* and groundcover by ferns. The forest was 200-250+yrs for the older/dominant cohort and 60-70yrs for the second cohort.

1.2.2 NSW North Coast – Wauchope - Blackbutt

Blackbutt production site

The blackbutt production site was located on a south-west facing slope, in a harvest area at the end of Mcmillians Rd in Mt Boss State Forest (Figure 1.3). Most trees were regrowth from approximately 1950's. The dominant species was blackbutt (*Eucalyptus pilularis*) with occasional tallowwood (*Eucalyptus microcorys*) and Sydney blue gum (*Eucalyptus saligna*). The mid-storey was dominated by *Casuarina spp.*, wattle and rainforest species. Groundcover was a medium to heavy cover with vines and grasses. There was some evidence of past bushfires and logging with many large stumps and charred logging slash present. The site was in a compartment which was scheduled for harvest as part of FCNSW Central region logging operations.

Blackbutt conservation site

The blackbutt conservation site was located on a level sandy area 5km north of the village of North Haven (Figure 1.4) in a compartment which was approved for harvest as part of private native forest logging operations with the Bunyah Local Aboriginal Lands Council. The stand showed little sign of past forest management. The dominant species was blackbutt (, with bloodwood (*Corymbia gummifera*) and tallowwood also commonly present. Shrub cover was dominated by saw banksia and corkwood (*Duboisia myoporoides*). Groundcover was a medium to heavy cover with ferns and grasses. Although there was evidence of past bushfires, the site had not been burnt recently. It was a multi-aged stand, the large bloodwood and blackbutt trees were >200 years old, and the 60-100 cm blackbutt probably circa 1920-30s.

Figure 1.3. Location of the blackbutt production site

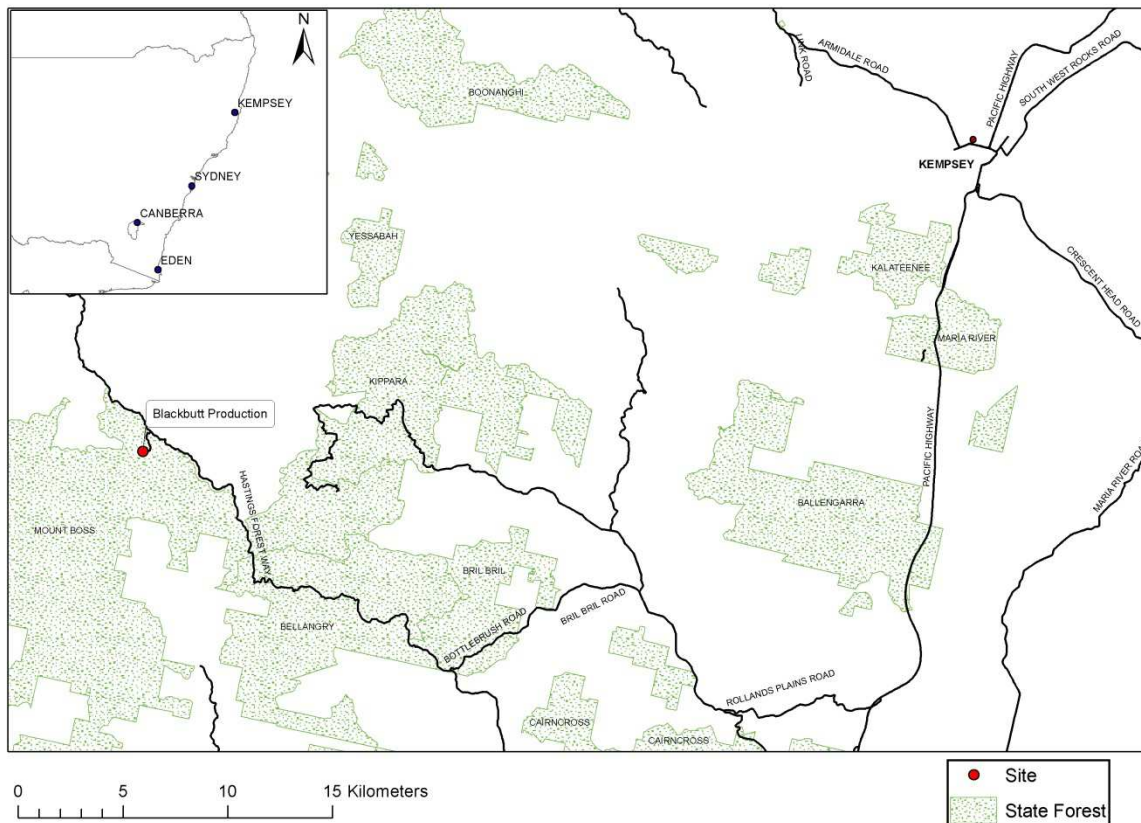
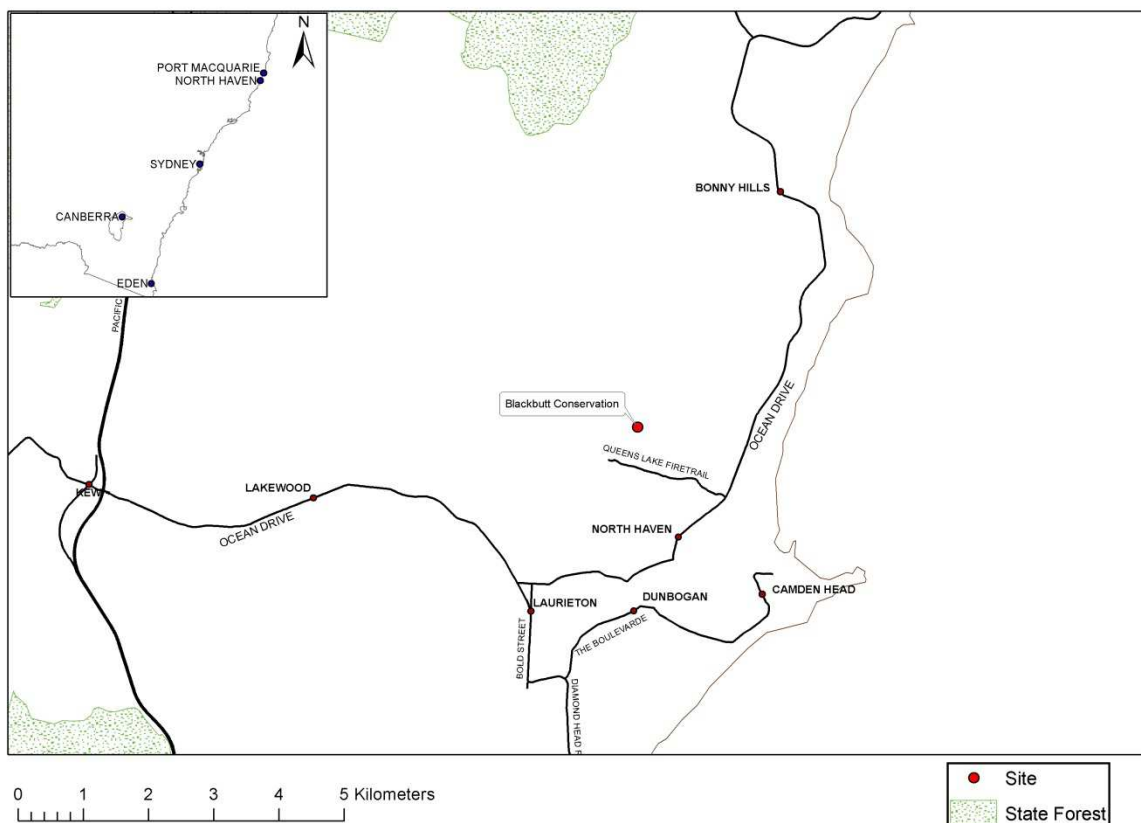


Figure 1.4. Location of the blackbutt conservation site

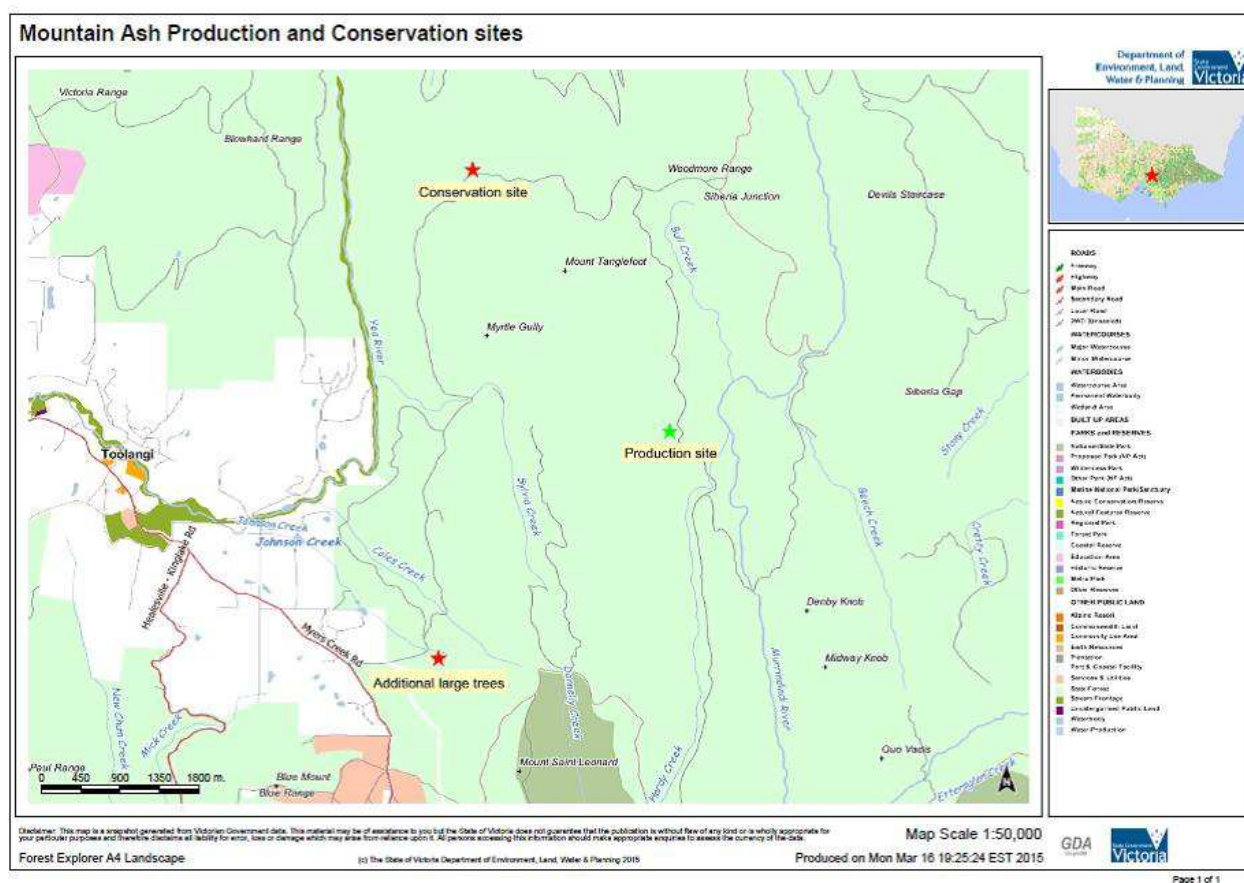


1.2.3. Victoria Central Highlands – Toolangi – Mountain ash

Mountain ash production site

The mountain ash production site was located at Toolangi State Forest, approximately 15 km north of Healesville, Victoria (Figure 1.5). The mountain ash “production” site was located on an east facing slope, in a harvest area off Hardy Creek Rd, with most trees being regrowth from the large 1939 fire. The dominant species was mountain ash (*Eucalyptus regnans*) with a sub canopy of silver wattle (*Acacia dealbata*), hickory wattle (*Acacia obliquinervia*) and mountain pepper (*Tasmannia lanceolata*). Shrub cover was dominated by *Correa spp.* and tree ferns. Groundcover was light with a heavy mulch layer around live trees. The site presented little evidence of past bushfires post 1939. There was evidence of past logging with a few small rotten stumps from a thinning event in the 1960s. The site was in a compartment which was scheduled for harvest as part of VicForests logging operations.

Figure 1.5. Location of the mountain ash production and conservation sites, as well as the site where additional large trees were weighed



Mountain ash conservation site

It was not possible to directly weigh biomass from a significantly older mountain ash site for use as the “conservation” scenario, as old growth mountain ash is rare in the Central Highlands of Victoria. An alternative approach was followed, where a 0.5 ha plot was established adjacent to the “Gun Barrel” coupe, which contained a number of trees with DBH significantly greater than those at the “production site”, due to the stand being primarily

regrowth from the 1905/06 fire. Every tree with a DBH greater than 10 cm was measured. The biomass information derived from the “production” site including three large trees (DBH 101-122cm) located adjacent to the production site as well as ten “additional large trees” from another nearby site (DBH 114-131cm), (Figure 1.5) was used to derive allometrics employed to estimate biomass in the measured trees from the conservation site (“Gun Barrel” coupe).

1.2.4. Plot establishment and tree measurement

At each site a plot of approximately 0.5 ha was established. The sites were selected in consultation with local foresters to ensure they were representative of the overall forest stands in the region, and also to ensure they were in the proximity of sites largely undisturbed by management (conservation scenario). The approach adopted was to attempt to identify the conservation sites first, because of the inherent difficulties associated with finding sites that have been largely undisturbed and that are available for harvest; and then to select a production site, which was guided by the harvest schedule created by the State Forest agencies.

All the standing trees within the plot with a DBH greater than 10cm were identified to species (excluding dead standing trees), numbered and their DBH and height measured. The diameter of the stem was measured using a diameter tape at 1.3m from the base of the tree. The height was measured using a vertex for the NSW sites and a laser instrument for the Victorian sites. The height of the trees was also measured by running a tape measure from the base of the tree to the top of the crown.

1.2.5. Weight determinations

The harvested components of the tree were weighed using a purpose built biomass weighing trailer (Figure 1.6). The trailer has been constructed with a heavy upper frame with each corner being supported by a two-speed drop leg. The nominal dimensions of the trailer are:

Length load space: 2.5 m; length overall: 4.1 m
Width Load space: 1.8 m; width overall: 1.8 m

Figure 1.6. A mountain ash “production” site log being weighed on the biomass weighing trailer.



Both weigh bars are equipped with two load cells with a combined capacity of 5 tonnes. Weight increments of 1.0 kg are displayed on a digital display. Details on the methodology used for weighing tree biomass using the biomass weighing trailer have been described previously in Ximenes *et al* (2008).

Where possible the weight of each log was recorded with the bark on and with the bark removed. The length and diameter of the log was also measured. A visual estimation was made of any decay and or bark loss. For practical reasons the crown component was only weighed with the bark intact. The weight of leaves was included in the weight of the crown. The majority of stumps was not easily removed and therefore could not be weighed, so their diameter and height were measured and an estimate of their biomass was made based on the volume of the stump which was calculated, assuming for practical reasons, they approached the form of a cylinder (Husch *et al* 1972). Density of the base disc extracted was used to determine the mass of the biomass stump.

1.2.6. Sampling and analysis of logs, bark and crowns

To determine site specific moisture and density, samples were taken from a selection of trees from each site ensuring that a range of species and DBH classes were represented. Generally, three disc samples (approximately 50 mm thick) were taken from each tree, one from the base of the tree, one from the middle of the tree and one from the upper stem which we have referred to as the crown. It is important to note that this was not the biological crown but, for trees with production logs, the point where the stem is too small to be of commercial value. Random samples of bark and branches were taken from the trees. All samples were placed in plastic bags and sealed to avoid moisture loss.

1.2.6.1. Density

Samples (“V”-shaped wedges from the pith to the outer edge) cut from discs were used to determine the density of the logs. The green samples were weighed and placed in an oven at 103 ± 2 °C until constant weight (oven-dry weight). The volume of the samples was determined by the water displacement method (ASTM, 1993). The basic density of the samples was calculated as the average of four measurements and expressed as:

$$\text{Basic Density} = \frac{\text{Oven dry weight (g)}}{\text{Green Volume (m}^3\text{)}} \times 1000$$

The same process was used to determine the basic density for bark and branches.

1.2.6.2. Moisture content

Samples used for the determination of density were also used to determine moisture content. The moisture content of bark from the stem was also determined. The moisture content of the samples was determined by oven-drying at 103 ± 2 °C until constant weight in accordance with AS/NZS 1080.1 (1997) and calculated as the average of four measurements expressed as:

$$\text{Moisture Content \%} = \frac{\text{Green weight (g)} - \text{Oven dry weight (g)}}{\text{Green weight (g)}} \times 100$$

1.2.6.3. Sapwood area

The sapwood content of each disc was determined according to AS/NZS 1605:2000 (2000). The assessment was carried out on the discs at four points across their diameter. The sapwood content was expressed as:

$$\text{Sapwood area \%} = \frac{\text{disc area (cm}^2\text{)} - \text{heartwood area (cm}^2\text{)}}{\text{disc area (cm}^2\text{)}} \times 100$$

The sapwood area of each sampled log was calculated as the average sapwood content of the butt and top discs (area basis used, not mass basis).

1.2.6.4. Dry weight biomass calculation

For each tree that was sampled the following was calculated:

Crown moisture content: This was the mathematical average of the moisture content of the crown disc samples and the branch samples.

Stem moisture content: This was the mathematical average of the moisture content of the crown, middle and base disc samples.

Bark moisture content: The mathematical average of the moisture content of the bark samples.

Stump: The volume and basic density was calculated as described previously.

For each study site, the average moisture content and basic density was grouped for each species by DBH class and tree component based on the sampled components. For example for the silver top ash conservation site, each tree in the “DBH class 8 (70-80 cm)” was assigned an average crown moisture content, stem moisture content, stump basic density and bark moisture. Where there was only one tree sampled within a site for a particular DBH class, a region (conservation and production) derived DBH class average was used. Where there was no tree sampled for a specific DBH class, a region derived species average was used.

To calculate the dry biomass for each tree component, the allocated moisture content was applied to the green weight for the crown (which included leaves), and the green weight without bark of the stem component (where possible the base disc moisture was applied to the first log cut from the base of the tree). The following equation was applied to derive dry biomass for the calculated stump volume:

$$\text{Oven Dry Weight (kg)} = \text{Basic Density} \times \text{Green Volume (m}^3\text{)}$$

Where the weight of the bark was known, the relevant bark moisture was applied. Where the weight of the bark was unknown, a bark ratio (of the total weight of the stem) based on species, DBH class and tree component was calculated, from which a bark weight was determined and the relevant moisture content applied.

The total above ground dry biomass for each tree was the sum of each dry weight component. Tree biomass estimates were converted into carbon assuming a default carbon concentration of 50% (Ximenes et al 2008).

1.2.7. Coarse woody debris (CWD) and litter measurement

The CWD of all the sites was determined as part of a parallel project carried out by NSW DPI for the Federal Department of Environment. Four to five 10 x 10 m sub-plots were set up within the 0.5 ha plots for the determination of CWD and litter (Figures 1.7 and 1.8a). CWD was defined as all debris on the forest floor >2.5 cm diameter, stumps <1.3 m in height and bark. All the CWD < 30 cm apart from the bark was classified into two states of decay; (i) 'Sound' if it did not deform under heel pressure or (ii) 'Rotten' if it deformed under heel pressure.

All CWD within the sub-plot with a diameter between 2.5 cm and 30 cm was collected, assessed as to its level of decay and weighed on either a set of field scales (PCS, KLS 150, maximum weight 150 kg and increments of 1 kg – Figure 1.8b) or on the biomass-weighing trailer (Figure 1.8c). Due to the large amount of biomass in CWD with a diameter >30 cm, all large pieces were collected from the broader 0.5 ha plot, thus ensuring that the majority of the biomass was directly weighed. Where possible, the stumps were removed and directly weighed. Samples from a range of diameter classes and decay levels were collected from each of the sub-plots as well as the from the broader plot for determination of moisture content and density.

The CWD levels in the mountain ash conservation site were derived using traditional transect techniques, as direct weighing was not possible.

Within a randomly selected CWD plot, a 5m² litter plot was established. All the litter down to the topsoil which included branches <2.5cm, leaves and organic material was collected and directly weighed. Three random samples were taken from each plot to determine moisture content so that dry biomass could be derived.

Figure 1.7. Plot design showing the general layout, with a rectangular plot of 0.5 ha, five 10 x 10 m subplots

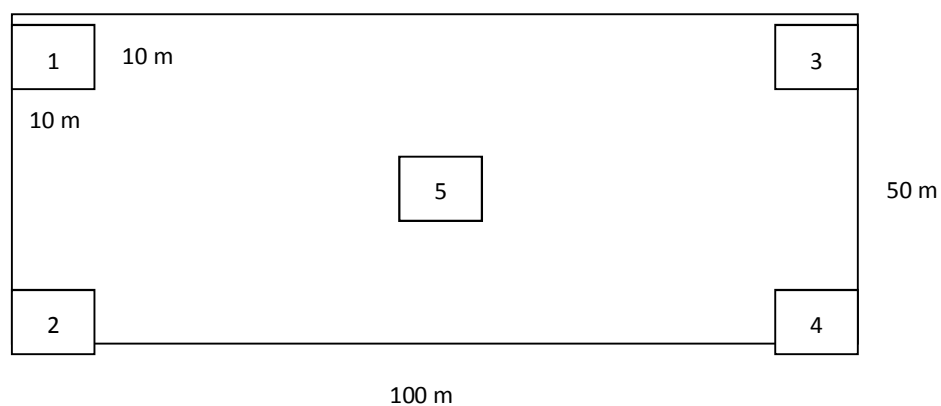


Figure 1.8a. 10 x 10 m sub-plot



1.8b. Weighing CWD <30 cm diam.



1.8c. Weighing CWD >30 cm diam.



The moisture content and density of the samples was calculated as described above. Due to the high porosity of the rotten CWD samples, the pieces were either dipped in wax or wrapped in cling wrap prior to being immersed in water for volume determinations based on water displacement.

1.3 Results

1.3.1 Forest Structure

The stand characteristics for each site are included in Table 1.2. In general the data reflects the differences in maturity between the production and conservation sites, with the total stems per hectare and stand density typically lower for the conservation sites, and the basal area typically greater for the conservation sites (Table 1.2). The standard deviation associated with the tree DBH for the production site was considerably lower than for trees in the conservation site, reflecting the less uniform stand profile for the conservation sites (Table 1.2).

Table 1.2. Stand characteristics for each site.

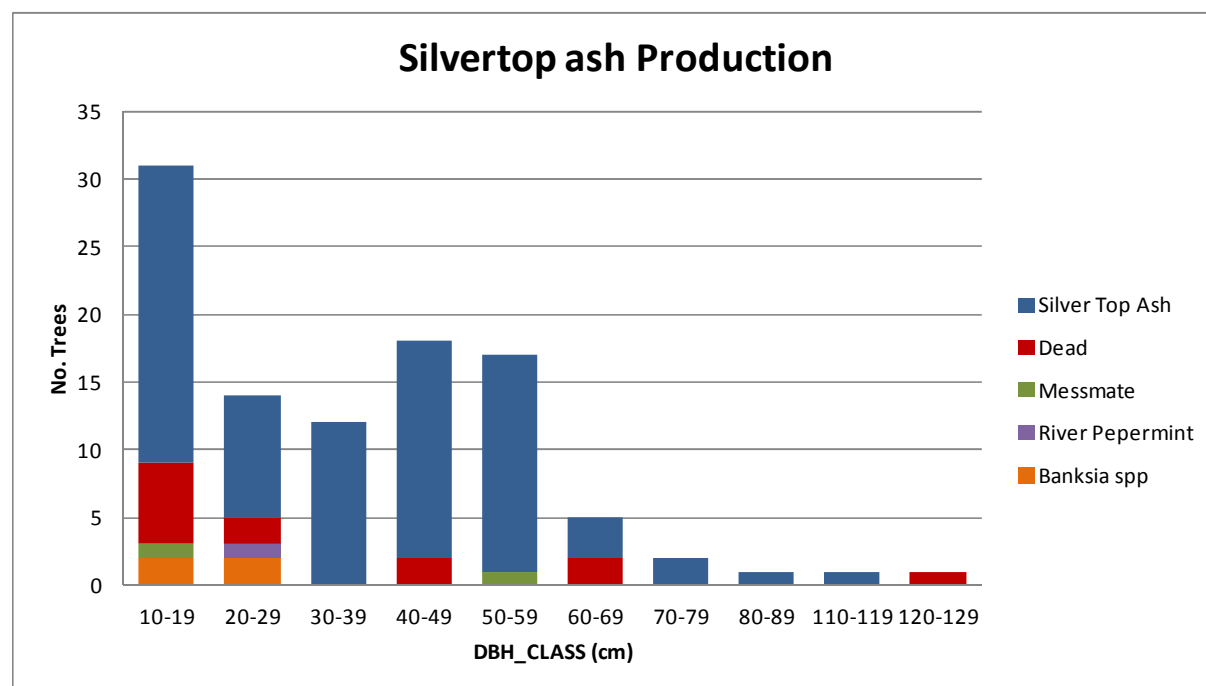
Site	Live Stand Density stems / ha ¹	Dead stems / ha	Basal area m ² /ha	Stand height (m) ²
Silvertop ash conservation	154	13	49	27
Silvertop ash production	191	28	25	23
Blackbutt conservation	242	25	39	29
Blackbutt production	270	38	25	31
Mountain ash conservation	160	22	74	63
Mountain ash production	198	40	62	59

Note. ¹ Stand density is for live trees only. ² Stand height is the average for the dominant species.

NSW South Coast – Eden – Silvertop ash production site

The dominance of silvertop ash across all diameter classes is highlighted in Figure 1.9.

Figure 1.9. DBH distribution of all species for the silvertop ash production site



Approximately 90% of the 103 trees measured (including a number of dead trees and charred stumps) had a DBH <60 cm. The silvertop ash production site had the lowest stand density, basal area and average tree height of all the production sites (Table 1.2). When compared to the conservation site, the higher stand density and lower basal area of the production site is reflective of the DBH distribution indicative of a production forest (Table 1.2, Figure 1.9).

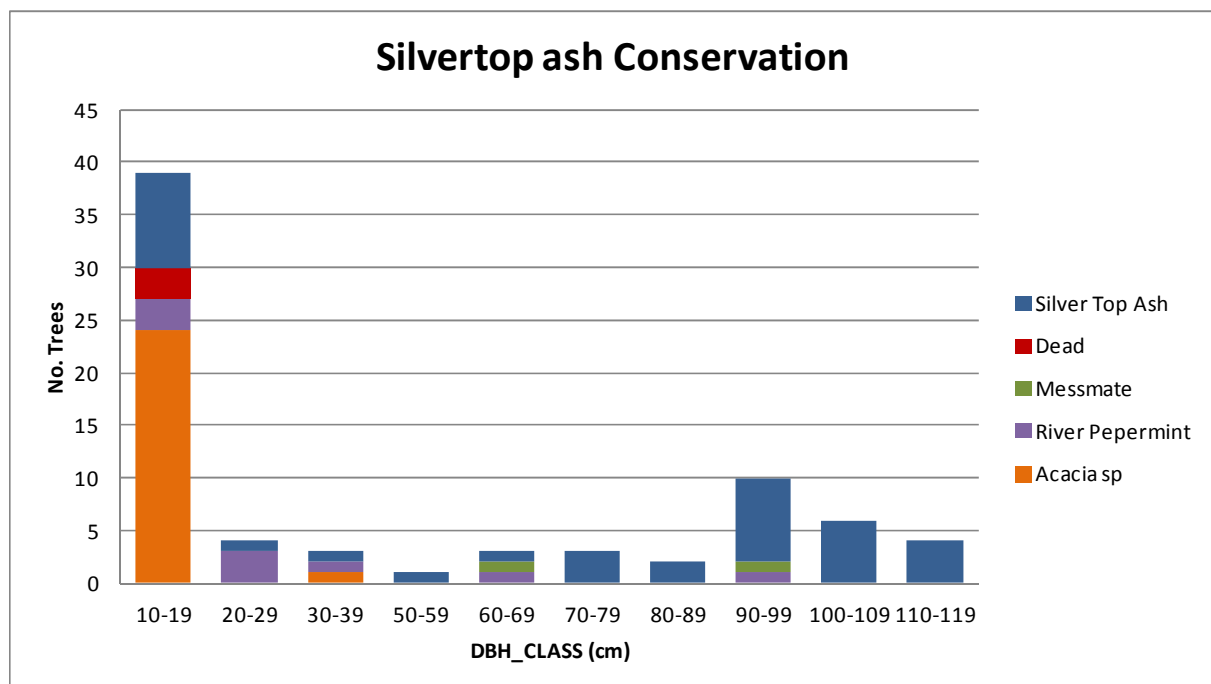
Figure 1.10. Before and after images for the Eden - Silvertop ash production site



NSW South Coast – Eden – Silvertop ash conservation site

A total of 80 trees were measured, with larger diameter trees (DBH >90 cm) representing almost 30% of the stand, and trees with 10-20 cm DBH representing 50% of the stand (Figure 1.11).

Figure 1.11. DBH distribution of all species for the silvertop ash conservation site.



The lower diameter classes were dominated by acacia trees, whereas silvertop ash dominated the larger diameter classes (Figure 1.11). The basal area of 49 m²/ha for a stand density of 154 trees/ha is reflective of a mature forest with the DBH distribution skewed towards the larger diameter classes (Table 1.2, Figure 1.11). The “conservation” site was largely unmanaged for production (as also evidenced by the lack of stumps), with a large proportion of mature trees and some evidence of past fire events (Figure 1.12).

Figure 1.12. Before and after images for the silvertop ash conservation site.



NSW North Coast – Wauchope - Blackbutt production site

A total of 153 trees (including a number of dead trees and charred stumps) were measured, with 95% of these having DBH < 60 cm (Figures 1.13 and 1.14). Blackbutt comprised the majority of the trees measured (63%). The blackbutt production site had a comparative basal area to the silvertop ash production site (25 m²/ha), but with a higher stand density (270 compared to 191 stems/ha). This reflected the larger proportion of trees in the lower DBH classes in the blackbutt production site compared to the silvertop ash site (Table 1.2).

Figure 1.13. DBH distribution by species for the blackbutt production site

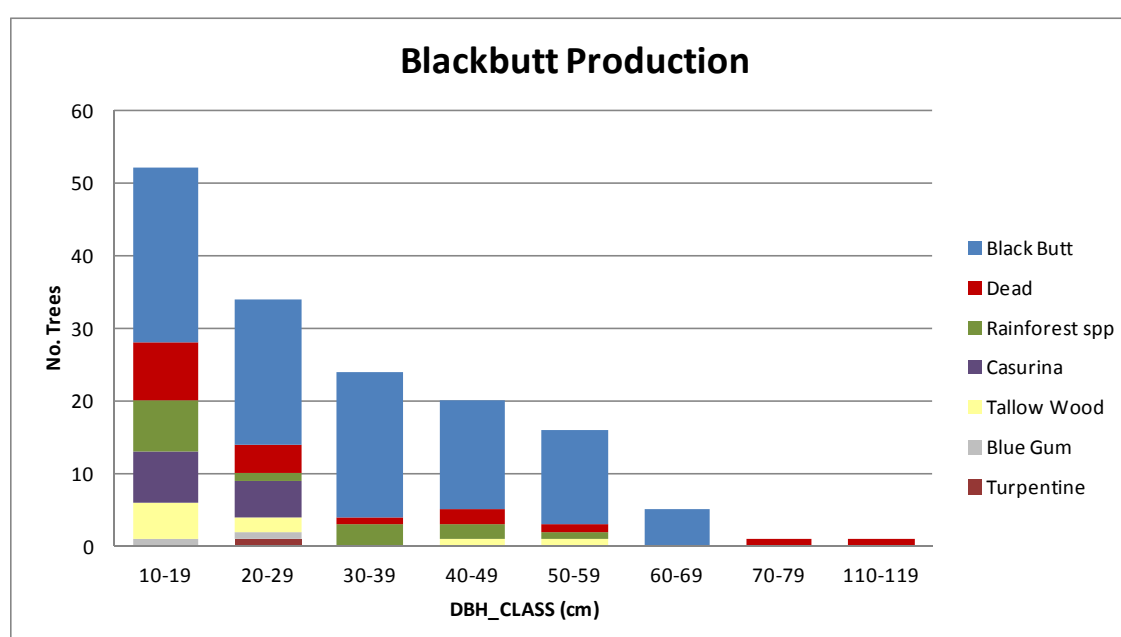


Figure 1.14. Before and after images for the Wauchope blackbutt production site.



NSW North Coast – Wauchope - Blackbutt conservation

The conservation site was dominated by three species: blackbutt (21.8%), bloodwood (18.8%) and tallowwood (19.5%). It had the smallest basal area ($39 \text{ m}^2/\text{ha}$) of all the conservation sites, with the highest stand density (242 trees/ha) (Table 1.2). A total of 128 trees (including a number of dead trees and charred stumps) were measured, with 80% of these having DBH <60cm (Figures 1.15 and 1.16). As expected the conservation site had a higher basal area than the production site ($25 \text{ m}^2/\text{ha}$), reflecting its more mature state (Table 1.2).

Figure 1.15. DBH distribution by species for the Wauchope blackbutt conservation site.

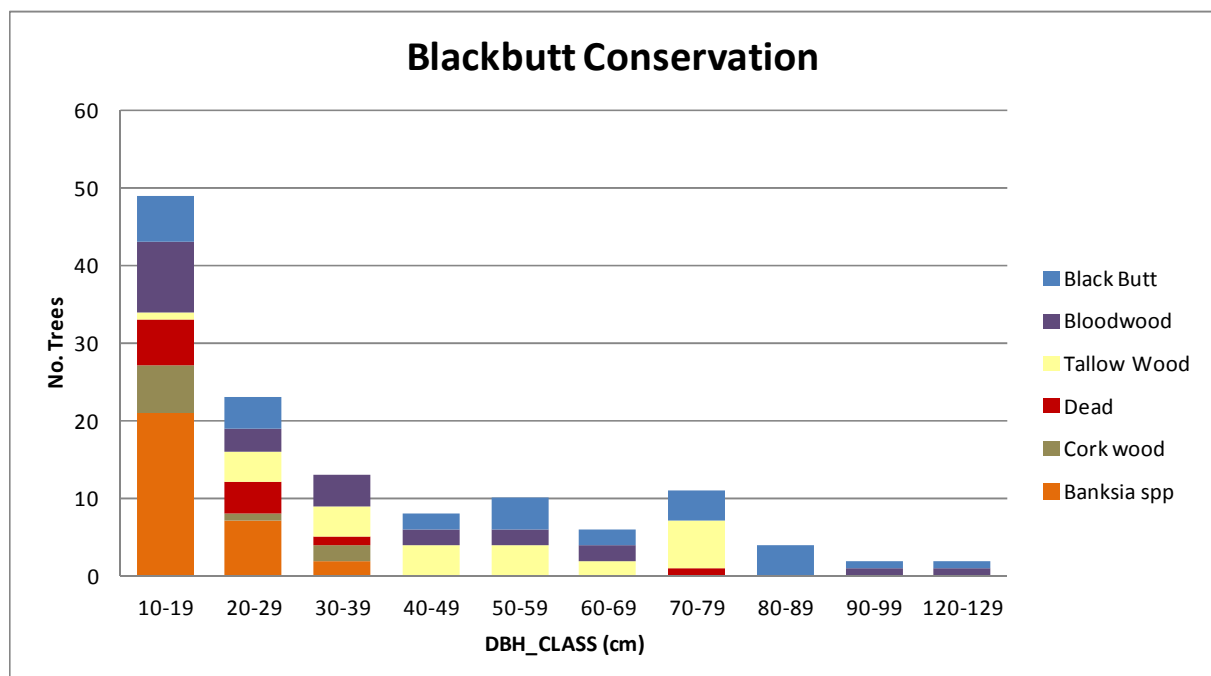


Figure 1.16. Before and after images for the blackbutt conservation site.



Victoria Central Highlands – Toolangi - Mountain ash production site

A total of 120 trees were measured, with 52% of these having DBH <60cm (Figures 1.17 and 1.18). The site was dominated by Mountain ash (66%), with dead standing trees comprising 15% of the total, and mountain pepper and wattle species 23% of the total (Figure 1.17). The stand density (198 stems/ha) was comparable to the other production sites; however the basal area (62 m²/ha) and average tree height (55 m) were much higher (Table 1.2).

Figure 1.17. DBH distribution by species for the mountain ash production site.

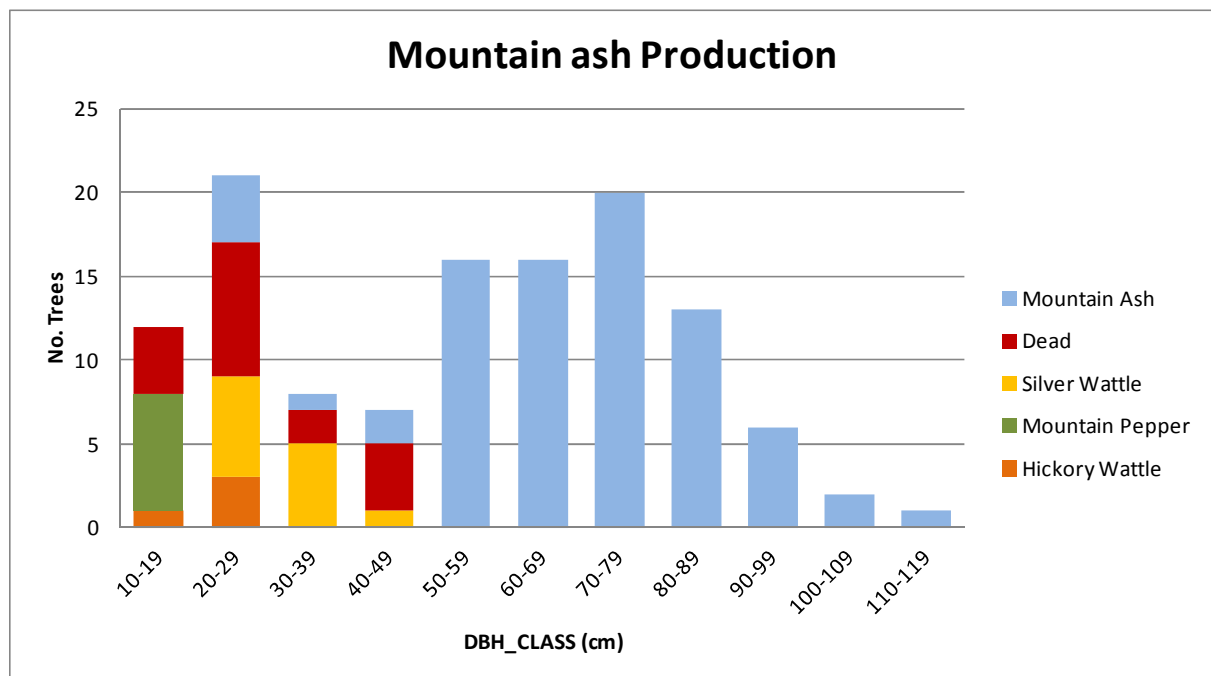


Figure 1.18. Before and after images for the mountain ash production site.



Victoria Central Highlands - Toolangi - Mountain ash conservation

A total of 91 trees were measured, with mountain ash again being the dominant species (43%), with the under and mid-storey species also accounting for 43% of the trees, and 14% of the trees being dead standing (Figures 1.19 and 1.20). The absence of mountain ash in the lower DBH classes was clear evidence of the mature status of the stand (Figure 1.19). The mountain ash conservation site had the highest basal area ($74 \text{ m}^2/\text{ha}$) and average tree height (61 m) for all the sites (Table 1.2).

Figure 1.19. DBH distribution by species for the mountain ash conservation site.

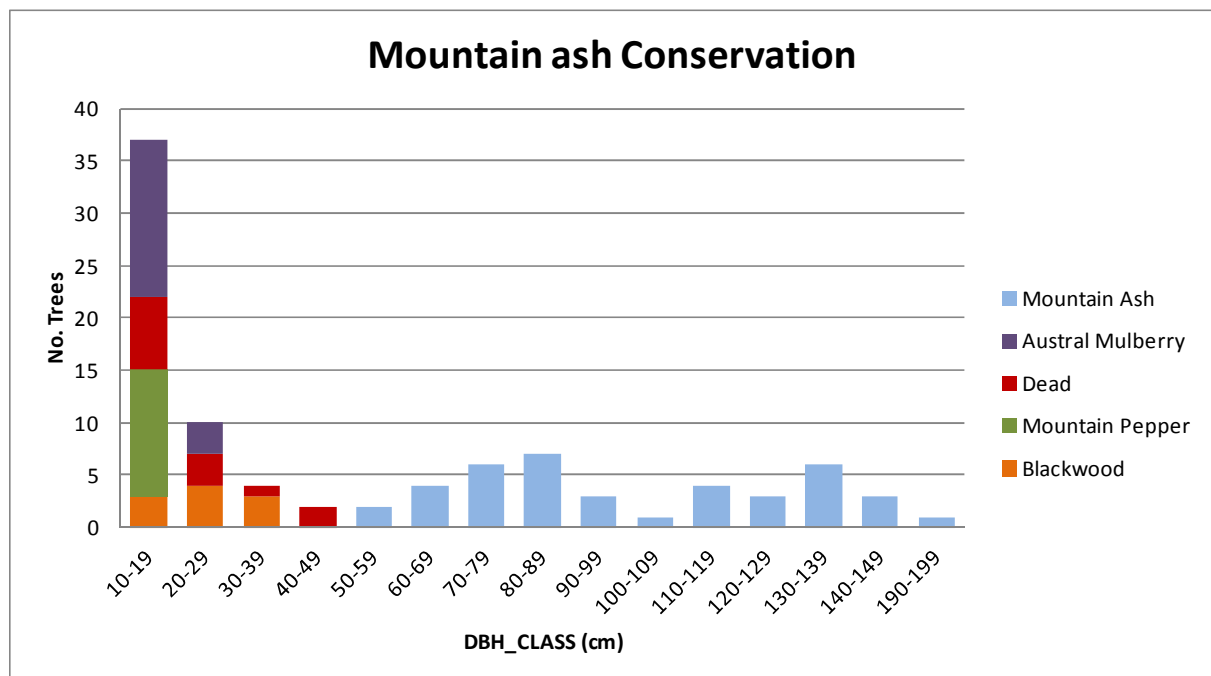


Figure 1.20. Image for the mountain ash conservation site. Note the abundance of tree ferns in the understorey



1.3.2. Total above-ground biomass (AGB)

The above-ground biomass (AGB) “as measured” (i.e. green weights) for each site is included in Table 1.3. The mountain ash conservation site is not included as the biomass was not directly weighed in the field. Biomass estimates were derived instead and are discussed in the ‘Standing above-ground biomass’ section. The live, green AGB in the conservation sites was significantly higher than that of the production sites (Table 1.3). This was unsurprising given the large DBHs of several trees contained in the conservation sites. The CWD and dead tree components were higher for production sites than the conservation sites for silvertop ash and blackbutt, whereas the litter component did not vary much (Table 1.3). CWD accounted for 26% and 32% of the total AGB for the silvertop ash and blackbutt production sites respectively, compared to 8% (silvertop ash) and 7% (blackbutt) for the conservation sites. The mountain ash production site had nearly double the biomass of all the other sites reflecting the high productivity of mountain ash (Table 1.3), with only 6% of the total AGB in the CWD and dead tree component.

Table 1.3. Above ground biomass as measured for each site as fresh weight

Sites	Total live green AGB (t / ha)	Dead trees (t / ha)	CWD (t / ha)	Litter (t/ha)	Total AGB (t / ha)
Silvertop ash conservation	786.2	6.9	63.0	14.5	870.6
Silvertop ash production	320.8	28.0	85.2	14.6	448.6
Blackbutt conservation	674.8	5.4	48.1	21.9	750.2
Blackbutt production	399.0	19.8	170.4	23.4	612.6
Mountain ash production	1483.3	27.5	65.6	12.8	1589.2

In Table 1.4 we compare log and residue volumes derived directly from the production study sites with those obtained from the State Forest agencies for similar forest types, based on inventory plots in “ready for harvest” mature forest areas (all trees with DBH > 10 cm). The FCNSW forecast was averaged over 22 sites for silvertop ash and 53 sites for blackbutt. The Victorian averages, over the past ten years, are for pure average sized mountain ash coupes in the vicinity of the study site.

Both the NSW production sites used for the study were representative of managed silvertop ash and blackbutt forests in the relevant regions, and the percentage breakup of the production and residue components was also comparable with the forecast figures (Table 1.4). The mountain ash volume figures were for average sized mountain ash coupes in Toolangi over the past ten years, and are broadly in agreement with the production site figures in terms of proportion of the production volumes on site (Table 1.4). The total standing tree volume for the study production site was substantially higher than the average, due to the very productive nature of the study site (Table 1.4). The mountain ash production site was thinned in the past, with poorer quality stems and trees that would otherwise have been dead standing removed, resulting in a greater proportion of utilisable timber.

Table 1.4. Commercial logs and residue volumes (excluding dead trees and CWD) for all production sites compared to state agency forecasts for similar forest types. Standard errors are shown in parenthesis.

Sites	Commercial logs (m³ / ha)	Commercial logs (%)	Residue volume (m³ / ha)	Residue volume (%)	Total volume (m³ / ha)
Silvertop ash production	157	55	128	45	285
FCNSW average forecast	165 (17)	56	131 (10)	44	296 (23)
Blackbutt production	177	48	195	52	372
FCNSW average forecast	199 (15)	54	169 (13)	46	368 (21)
Mountain ash production	1107	79	301	21	1407
VicForests Toolangi average forecast	814	75	271	25	1085

1.3.3. Moisture content and Density

The moisture content and basic density of the dominant species in each site are included in Tables 1.5 and 1.6. The moisture content of the wood typically decreased from the stump to the crown - this was particularly pronounced for mountain ash, which is not surprising given the height of the trees (Tables 1.5 and 1.6). The moisture content of the bark was typically considerably higher than for the woody components (Table 1.5). The branch moisture content was also relatively high; however these figures maybe slightly inflated as the branch moistures were derived with the bark intact. The moisture content of the wood from the NSW production sites was higher than that of the equivalent conservation sites (Table 1.5).

Table 1.5. Average moisture content for the dominant species for each site (standard deviation in parenthesis).

Site	Species	Moisture content (%)				
		Stump	Stem	Crown ¹	Branch ²	Bark
Eden production	Silvertop ash	46 (4)	41 (3)	40 (4)	45 (1)	46 (7)
Eden conservation	Silvertop ash	46 (5)	39 (2)	38 (6)	43 (2)	46 (8)
Wauchope production	Blackbutt	43 (5)	41 (3)	39 (4)	45 (2)	56 (6)
Wauchope conservation	Blackbutt	38 (3)	37 (3)	35 (4)	42 (1)	57 (4)
Wauchope conservation	Bloodwood	39 (2)	37 (2)	35 (3)	41 (4)	49 (3)
Wauchope conservation	Tallowwood	42 (2)	40 (1)	38 (2)	NA	50 (7)
Toolangi production	Mountain ash	57 (3)	49 (4)	44 (7)	45 (3)	60 (5)

¹ The moisture content for the crown is the mathematical average of the moisture content of crown discs and branches. ² The bark was not removed from the branch component, thus the moisture content for the branch includes the branch bark.

The basic density of the stump wood was lower than for the stem and crown portions of silvertop ash and mountain ash trees; these differences were not evident for blackbutt (Table 1.6). The basic density of branch wood was typically lower than the stump, stem or crown wood for silvertop ash and blackbutt; however this was reversed for mountain ash (Table 1.6). The basic density of blackbutt from the “conservation” site was the highest of all species tested (Table 1.6). The stem basic density for silvertop ash and mountain ash (Table 1.6) was consistent with values reported in the literature (Stewart et al (1979) and Bootle (1983) for silvertop ash stem wood and Sillett et al (2010) and Keith et al (2014) for mountain ash stem wood). Although the basic density for regrowth blackbutt stem wood from the production site was consistent with the range reported by Davis (1994), the value for the blackbutt stem wood from the “conservation” site was higher than the upper value reported by Davis (1994) of 729 kg/m³, reflecting the more mature status of the trees on that site. The basic density of the silvertop ash thick bark was highest amongst the three species listed in Table 4 (523-541 kg/m³). Stewart et al (1979) reported even higher values (650 kg /m³) for silvertop ash bark sampled in their study.

The sapwood content of blackbutt samples from the blackbutt sites is detailed in Table 1.7 (there were insufficient complete disc samples from the silvertop ash sites). The proportion of sapwood was considerably lower for the sampled trees in the conservation site (Table 1.7), reflecting the more mature status of the trees in the conservation site.

Table 1.6. Average basic density for dominant species for each site (standard deviations in parenthesis).

Site	Species	Basic density (kg / m ³)				
		Stump	Stem	Crown	Branch	Bark
Eden production	Silvertop ash	633 (72)	671 (34)	678 (57)	615 (18)	541 (68)
Eden conservation	Silvertop ash	635 (85)	688 (43)	679 (71)	637 (33)	523 (121)
Wauchope production	Blackbutt	660 (61)	663 (52)	661 (67)	606 (30)	377 (56)
Wauchope conservation	Blackbutt	766 (44)	767 (58)	783 (61)	656 (32)	416 (45)
Wauchope conservation	Bloodwood	750 (33)	770 (35)	745 (81)	599 (25)	482 (46)
Wauchope conservation	Tallowwood	692 (21)	718 (58)	756 (17)	693 (-)	474 (61)
Toolangi production	Mountain ash	474 (43)	533 (45)	567 (55)	590 (24)	380 (39)

¹ The basic density for the crown is the mathematical average of the basic density of crown discs and branches. ² The bark was not removed from the branch component, thus the basic density for the branch includes the branch bark.

Table 1.7. Sapwood (% area) of blackbutt samples taken from the standing live trees (standard deviation in parenthesis).

Site	Species	Sapwood (%)		
		Stump	Stem	Crown
Blackbutt production	Blackbutt	26 (12)	30 (10)	41 (10)
Blackbutt conservation	Blackbutt	13 (7)	14 (6)	16 (10)
Blackbutt conservation	Bloodwood	13 (4)	20 (10)	27 (6)
Blackbutt conservation	Tallowwood	8 (1)	7 (2)	18 (16)

1.3.4 Standing above-ground biomass (0% moisture)

The dry biomass estimates in Table 1.8 were calculated as per the method described earlier and using the moisture and density values reported in Tables 1.5-1.7. The differences between the sites that were evident in the green standing volumes (Table 1.3) are largely reflected in the dry biomass estimates (Table 1.8). The dry biomass in the NSW conservation sites was significantly higher than that of the NSW production sites. When comparing the green volumes to the dry biomass, overall there was a reduction in the relative proportion of bark – this is due to the significantly higher moisture content of the bark compared to the woody component (Table 1.5). As logs in native hardwood harvest operations are debarked in the forest, the differences in bark proportion between species can have a significant impact on

the total amount of harvest residue left in the forest – this is exemplified by the differences between the bark proportions for mountain ash and silvertop ash (Table 1.8).

The crown component represented a larger proportion of the dry biomass (35%) for the NSW conservation sites when compared to their respective production sites (14-22%, Table 1.8). The 'other' residue component for the blackbutt sites was much larger than for the other sites. The Victoria production site not only had the highest total standing biomass per hectare but also had the highest production recovery (Table 1.8).

The dry biomass of 795 t / ha for the Victoria conservation site (Table 1.8) was estimated using allometrics derived from the Victoria production site measurements, plus an additional ten mountain ash trees with a DBH range of 114-131 cm that were weighed on a nearby site (Figure 1.5). The additional trees were measured to increase the number of trees in the larger DBH classes and therefore improve the capacity to predict the biomass of the conservation site, where 45% of the mountain ash had a DBH >100cm. There were a considerable number of tree ferns in the Victoria conservation site. In a recent study, Fedrigo *et al* (2014) estimated the C stocks in above and belowground components of eucalypt forests in the Central Highlands region of Victoria. The estimated C stocks for tree ferns was on average 2.8 t C / ha, with an upper bound (95% confidence interval) of 4.8 t C / ha. The upper bound figure was adopted here as it was likely to be the closest match to our "conservation site" (it was a very "wet" site).

Table 1.8. Dry biomass (t/ha) for each site by tree component.

	Dry biomass (t / ha)										
	Residues								Production		Total
Site	Bark	%	Crown	%	Stump	%	Other ³	%	logs	%	
Silvertop ash production	30	15	46	22	9	4	15	7	105	51	205
Silvertop ash conservation	55	12	165	35	32	7	35	7	189	40	476
Blackbutt production	17	7	35	14	12	5	71	27	123	48	258
Blackbutt conservation	34	8	148	35	11	3	134	32	91	22	418
Mountain ash production	32	4	61	8	34	4	38	5	588	78	753
Mountain ash conservation ¹	31	4	54	7	56	7	17	2	662 ²	81	819

¹ Estimated using allometric relationships derived from the mountain ash production site and from a cluster of large trees in adjacent sites. It does not include the estimated C stocks for tree ferns. ² The "production log" proportion is most likely overstated, as a proportion of the estimated biomass in production logs is likely from trees affected by decay. ³ The 'Other' residues includes non- commercial species, dead and small trees as well as parts of the stem that had no commercial value due to damage during felling, decay or a reflection of the current market for that region.

1.3.5. Biomass breakdown by site

NSW South Coast – Silvertop Ash

Silvertop ash accounted for the majority of the biomass in both the production site (87.6%) and the conservation site (90.1%), (Figures 1.21 and 1.22). A significant proportion of the biomass in the production site was made up of standing stags or burnt stumps (10.3%) and only 2% from other trees (Figure 21).

Figure 1.21. The proportion of dry biomass by species for the silvertop ash production site.

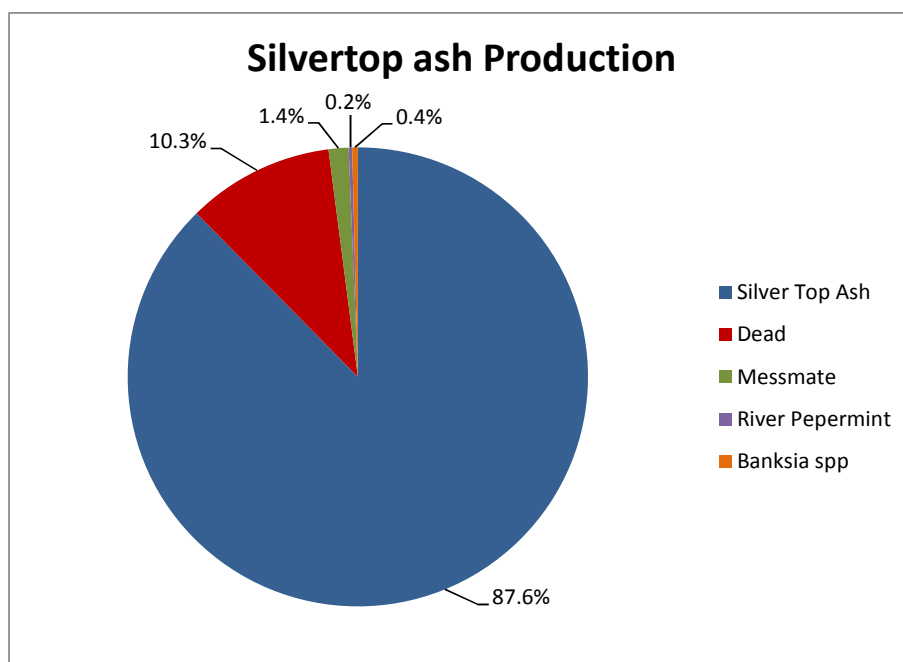
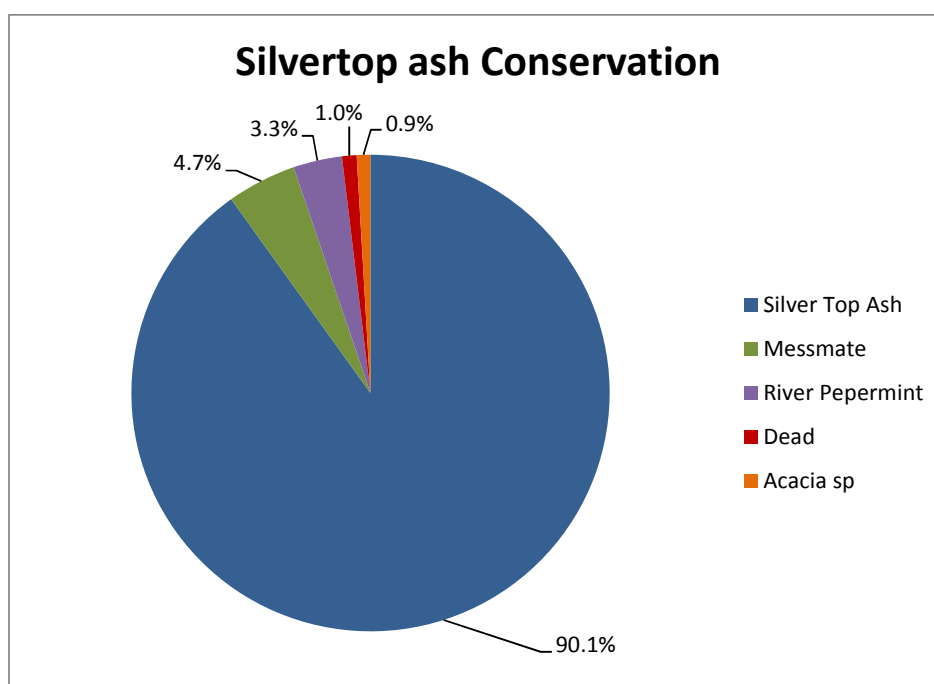


Figure 1.22. The proportion of dry biomass by species for the silvertop ash conservation site.



The majority of the biomass (48%) in the production site was within the DBH range of 40-59 cm. The largest ten trees on the site (10% of total trees) accounted for 31% of the biomass (Figure 1.23). For the conservation site, 77% of the biomass was in the DBH classes > 90cm (Figure 1.24). For both sites the large number of understorey and or regrowth trees in the 10-19 cm range contributes very little to the biomass pool (Figures 1.23 and 1.24).

Figure 1.23. The distribution of biomass and number of trees across the DBH classes for the silvertop ash production site.

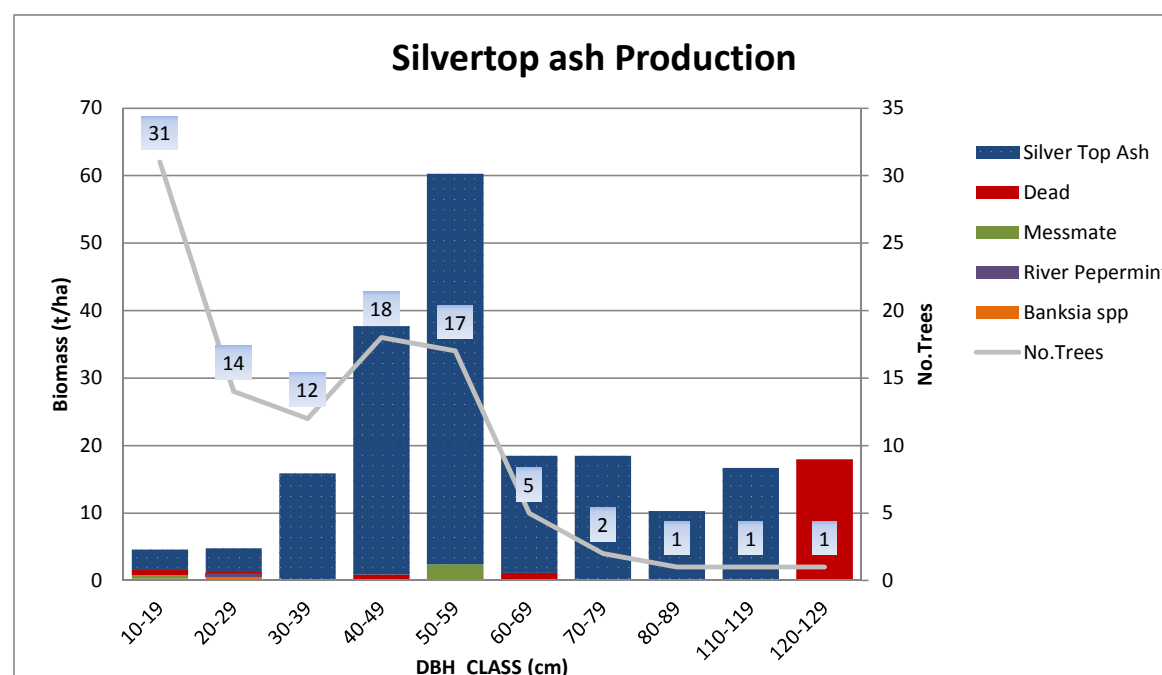
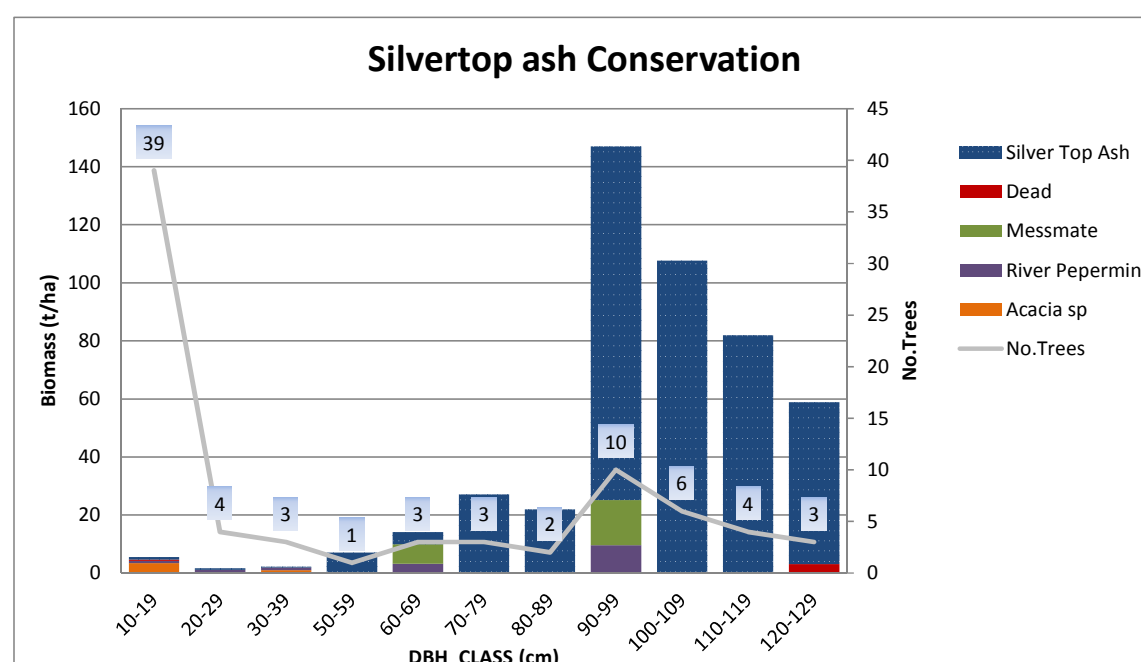
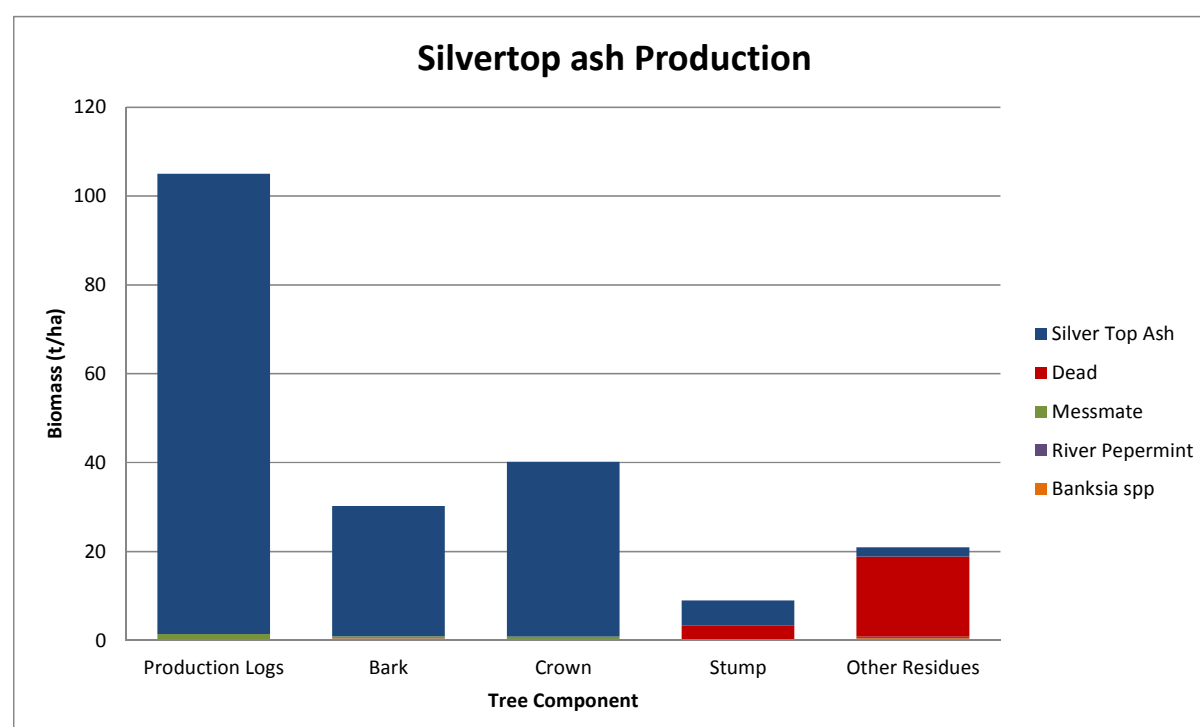


Figure 1.24. The distribution of biomass and number of trees across the DBH classes for the silvertop ash conservation site.



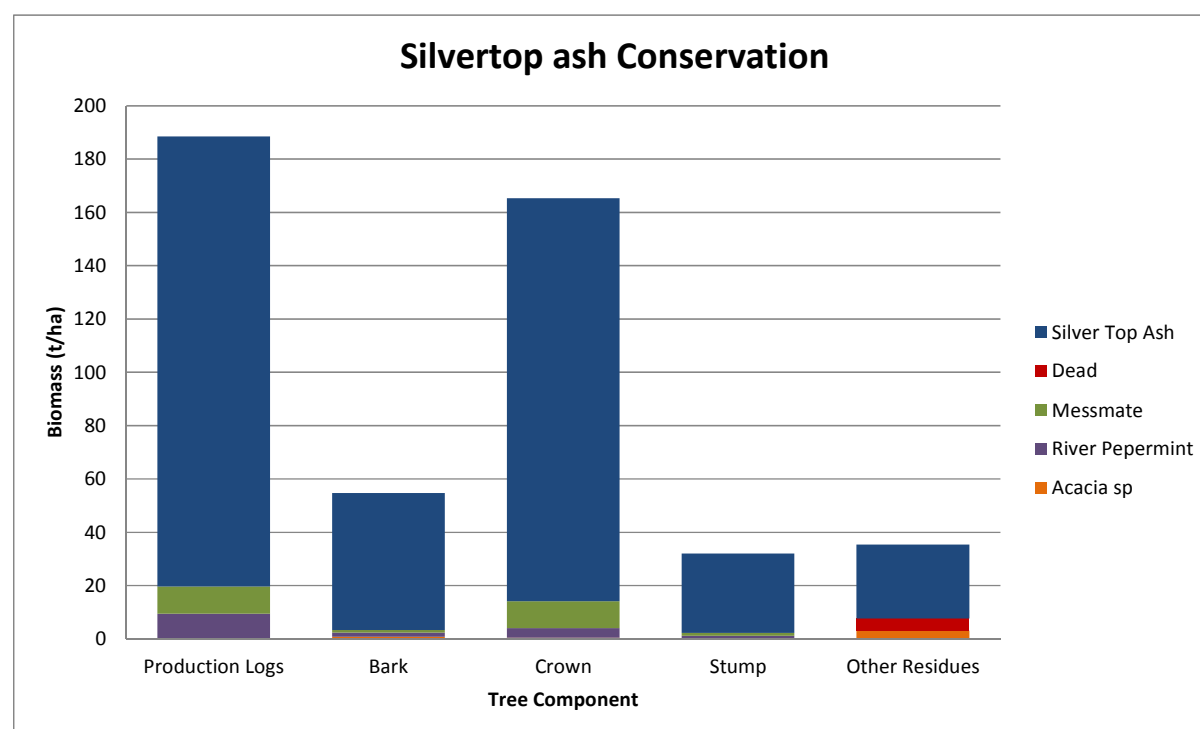
Debarked production logs accounted for over 51% of the biomass in the production site, whereas for the conservation site, it was only slightly higher (40%) than biomass in the crown component (35%) (Figures 1.25 and 1.26). The ‘other’ residue component represented only a small proportion of the biomass in both sites. For the production site this was mainly dead trees, whereas for the conservation site this figure includes a large tree that was not directly weighed on site and its biomass estimated. As the breakup of this tree into separate components was unknown, the whole tree was included in ‘other’ residues, accounting for 24 t/ha of this component. The proportion of the bark in the total biomass was substantial for both sites (12% for the conservation and 15% for production site). As a large proportion of the logs were weighed both before and after debarking, the proportion of biomass in the bark of production logs was determined – on average for silvertop ash, bark represented approximately 21% of the weight of the production log (green basis).

Figure 1.25. Biomass distribution for different species in the silvertop ash production site (0% moisture)¹



¹ The bark component comprises the bark from the stem and stump - the crown bark is included in the crown figure. The ‘Other’ residues includes non- commercial species, dead and small trees as well as parts of the stem that had no commercial value, due to damage during felling, decay or a reflection of the current market in that region.

Figure 1.26. Biomass distribution of the standing component for the silvertop ash conservation site (0% moisture)¹



¹ The bark component comprises the bark from the stem and stump - the crown bark is included in the crown figure. The 'Other' residues includes non- commercial species, dead and small trees as well as parts of the stem that had no commercial value, due to damage during felling, decay or a reflection of the current market in that region.

NSW North Coast – Blackbutt

Blackbutt accounted for the bulk of the biomass in the production site (82.8%), (Figure 1.27). Blackbutt was less dominant in the conservation site, accounting for 48.1% of the total biomass, with bloodwood (23.9%) and tallowwood (24.3%) also accounting for a significant proportion of the biomass (Figure 28). The dead component was again higher for the production site (6.6%) compared to the conservation site (1.1%).

Similarly to the silvertop ash production site, the majority of biomass for the blackbutt conservation site (54%) was within the 40-59 cm DBH range, with 88% in the 30-69 cm classes (Figure 1.29). The eight largest trees in the blackbutt conservation site contributed 40% of the biomass in the site, with the 70-79 cm DBH classes accounting for 20% of the biomass (Figure 1.30). The evidence of past bushfires found on the site is reflected in the distribution of biomass in the various DBH classes, which suggests past disturbance and a multi-aged stand (Figure 1.30).

Figure 1.27. The proportion of dry biomass by species for the blackbutt production site.

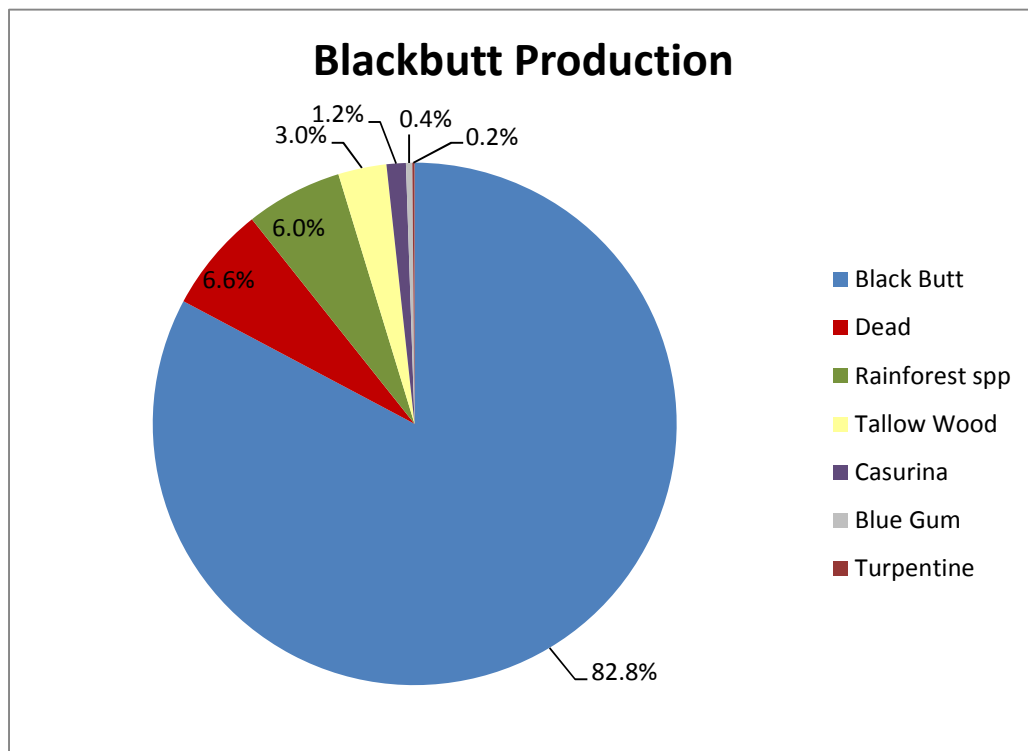


Figure 1.28. The proportion of dry biomass by species for the blackbutt conservation site.

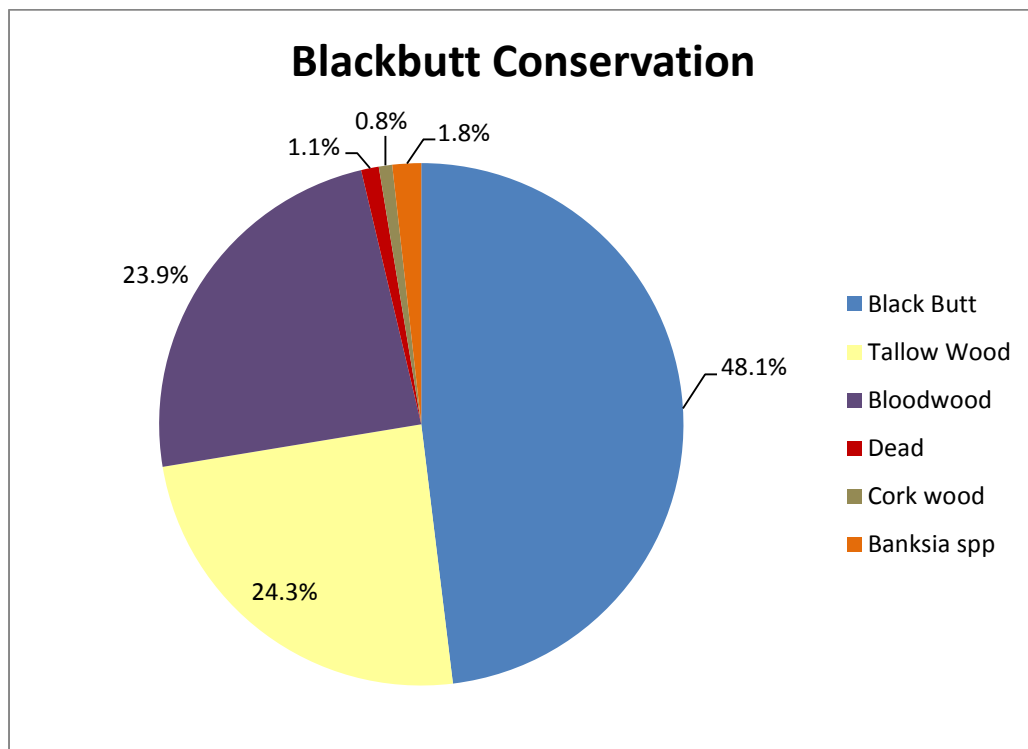


Figure 1.29. The distribution of biomass and number of trees across the DBH classes for the blackbutt production site.

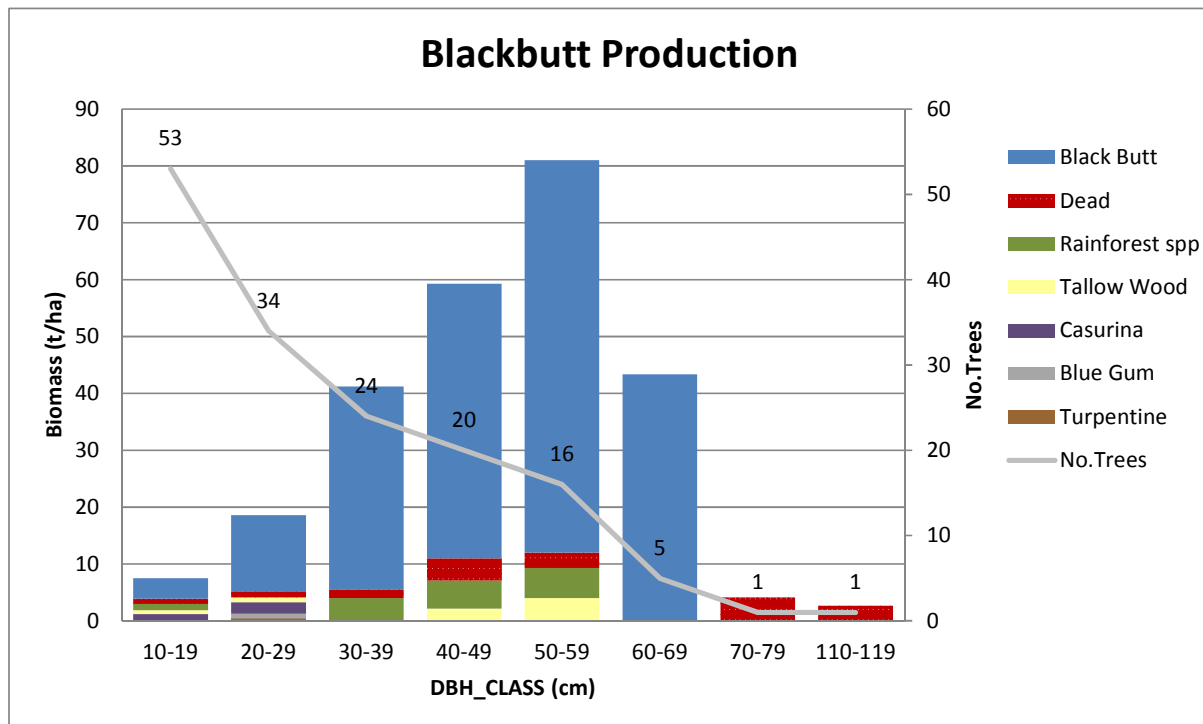
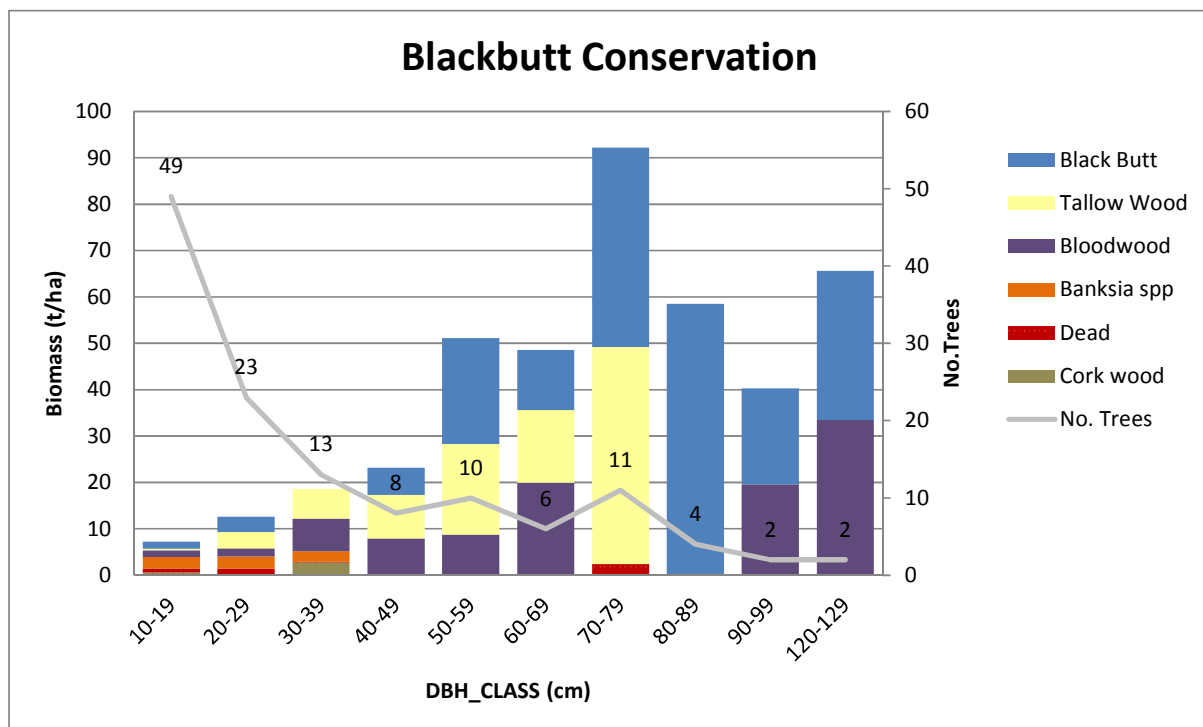


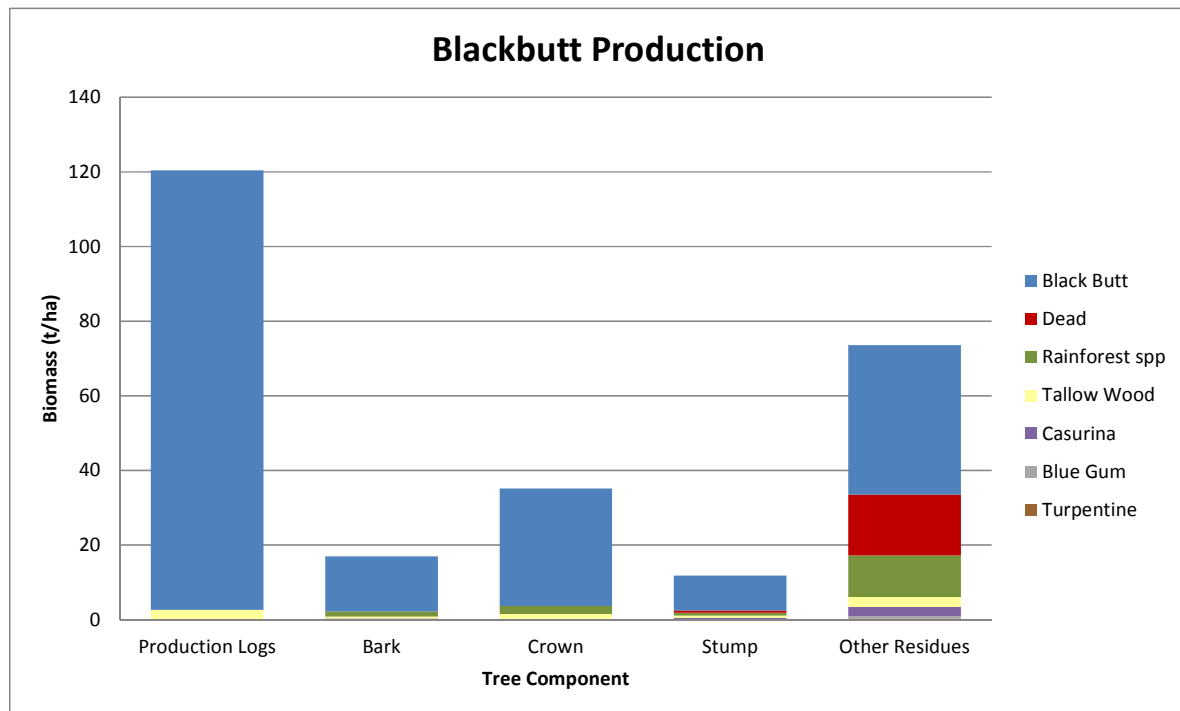
Figure 1.30. The distribution of biomass and number of trees across the DBH classes for the blackbutt conservation site.



Production logs accounted for a much higher proportion of the biomass in the production site, (47%) than in the conservation site (Figures 1.31 and 1.32). Conversely the crown component

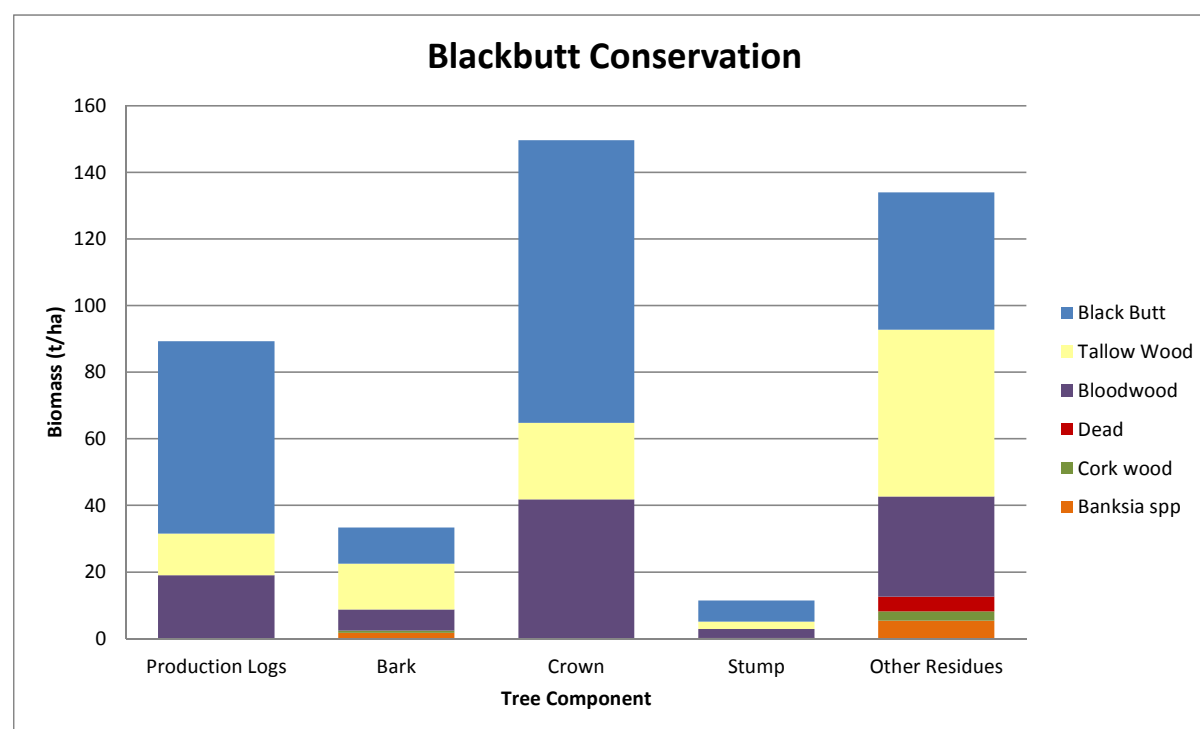
accounted for 14% of the biomass in the production site and 35% for the conservation site. The “Other residue” component was high for both sites, at approximately 30% (Figures 1.31 and 1.32).

Figure 1.31. Biomass distribution of the standing component for the blackbutt production site (0% moisture)¹



¹ The bark component comprises the bark from the stem and stump - the crown bark is included in the crown figure. The ‘Other’ residues includes non- commercial species, dead and small trees as well as parts of the stem that had no commercial value, due to damage during felling, decay or a reflection of the current market in that region.

Figure 1.32. Biomass distribution of the standing component for the blackbutt conservation site (0% moisture)¹



¹ The bark component comprises the bark from the stem and stump - the crown bark is included in the crown figure. The 'Other' residues includes non- commercial species, dead and small trees as well as parts of the stem that had no commercial value, due to damage during felling, decay or a reflection of the current market in that region.

Victoria Central Highlands – Mountain ash

Mountain ash accounted for the vast majority of the biomass in the Victoria production site (over 95%), (Figure 1.33). The number of mountain ash trees in the lower DBH classes for the mountain ash production site was considerably lower than for the dominant species in the NSW production sites (Figure 1.34). The majority of the biomass (48%) was in the 70-89 cm range, with the biomass distribution across the DBH classes approximating a bell curve (Figure 1.34).

The production recovery for mountain ash (78%) was the highest of all the sites (Figure 1.35 and Table 1.7), with the crown and other residues accounting for 8% and 5% of the biomass, respectively.

Figure 1.33. The proportion of dry biomass by species for the mountain ash production site (0% moisture).

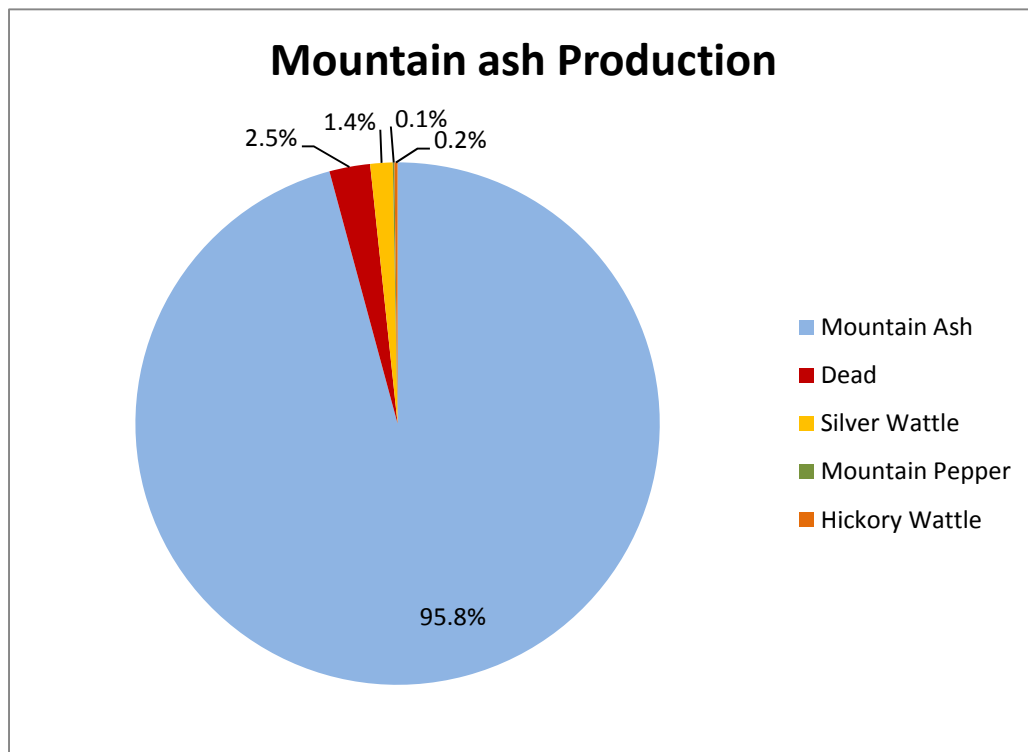


Figure 1.34. The distribution of biomass and number of trees across the DBH classes for the mountain ash production site.

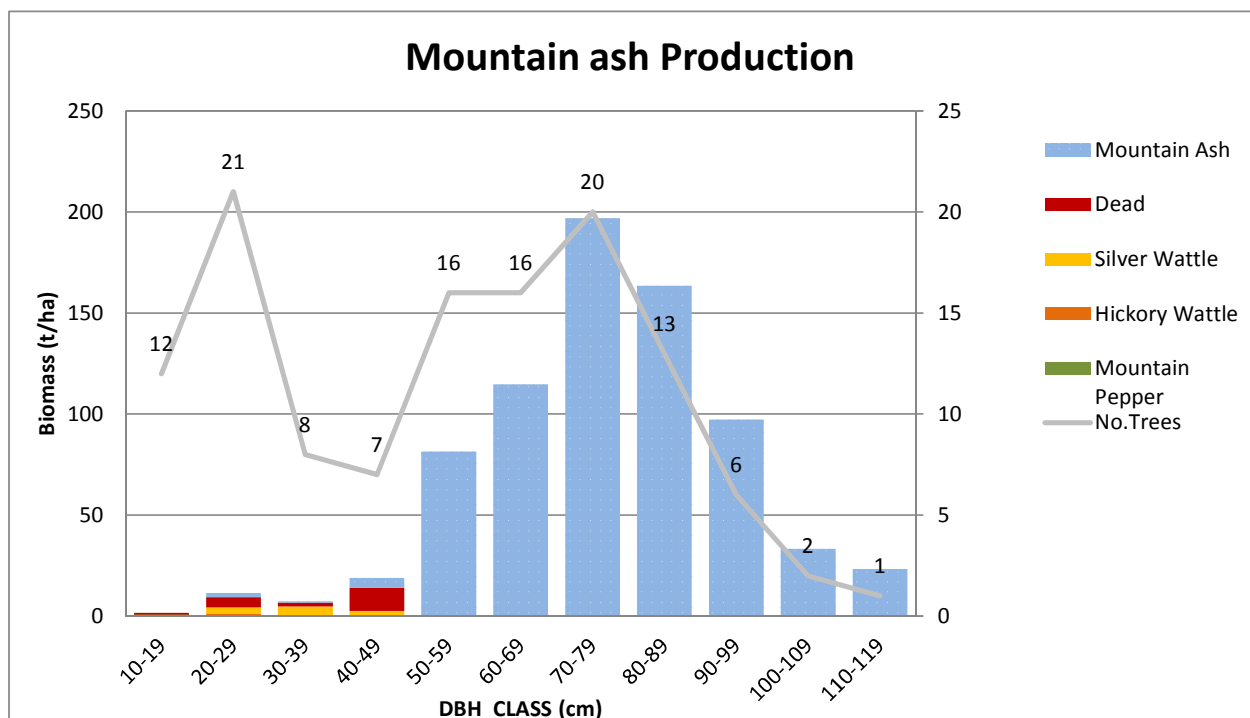
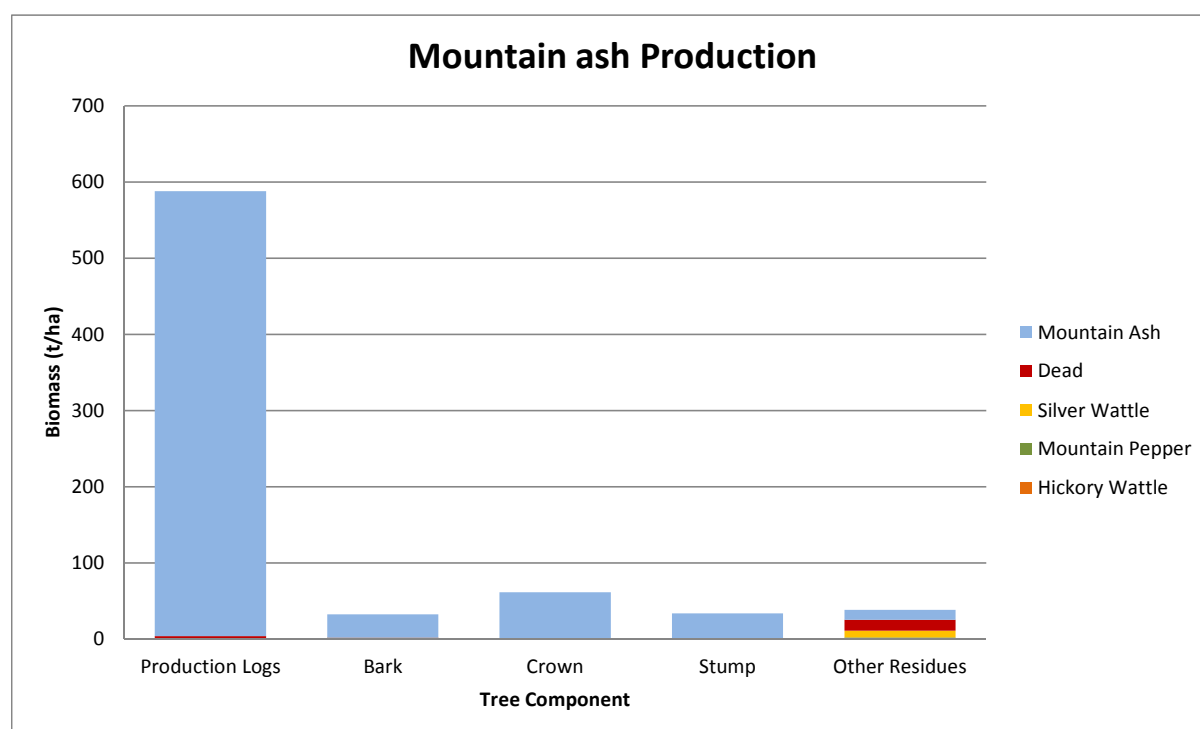


Figure 1.35. Biomass distribution of the standing component for the mountain ash production site (0% moisture)¹



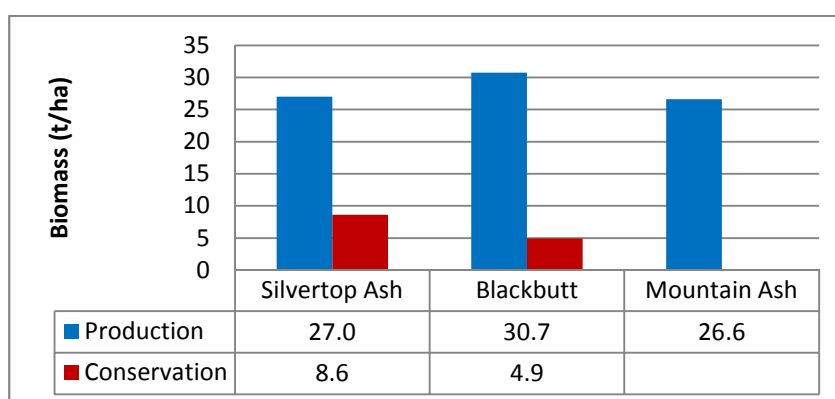
¹ The bark component comprises the bark from the stem and stump - the crown bark is included in the crown figure. The 'Other' residues includes non- commercial species, dead and small trees as well as parts of the stem that had no commercial value, due to damage during felling, decay or a reflection of the current market in that region.

1.3.6. Coarse woody debris (CWD) biomass

A comprehensive separate report has been prepared on the CWD determinations for the study sites, in a parallel project. We include here some key information from the CWD report to assist in the understanding of the relative differences in CWD biomass between sites and to provide a more complete assessment of C stocks in the forests. The amount of CWD biomass < 30cm diameter was larger for NSW "production" forests than for the equivalent "conservation" forests, and was similar across all three production sites (Figure 1.36). The higher levels of CWD < 30 cm diameter in the production sites are likely the result of past management operations such as thinning.

There was a large amount of biomass in the CWD > 30 cm diameter for the blackbutt production forest (Figure 1.37). The DBH distribution of the trees on this site was heavily skewed towards the lower classes, suggesting that harvest of larger, older trees would have taken place in the past, generating large volumes of residues. This is supported by the high harvest residue component (78%) for the blackbutt conservation site (Figure 1.31), which is indicative of volumes of biomass that may have been left on the ground at earlier harvest operations. In the 1950's and 1960's many of the blackbutt-dominated forests in the region were subjected to a TSI (timber stand improvement) treatment (Raison *et al* 2007), resulting in large dead stags that would have naturally fallen over time. Historical harvest records for the area from the FCNSW indeed confirm this, with a TSI treatment occurring in the early 1960's. Also according to historical records, the blackbutt study site was subjected to a thinning treatment approximately 20 years prior to our study.

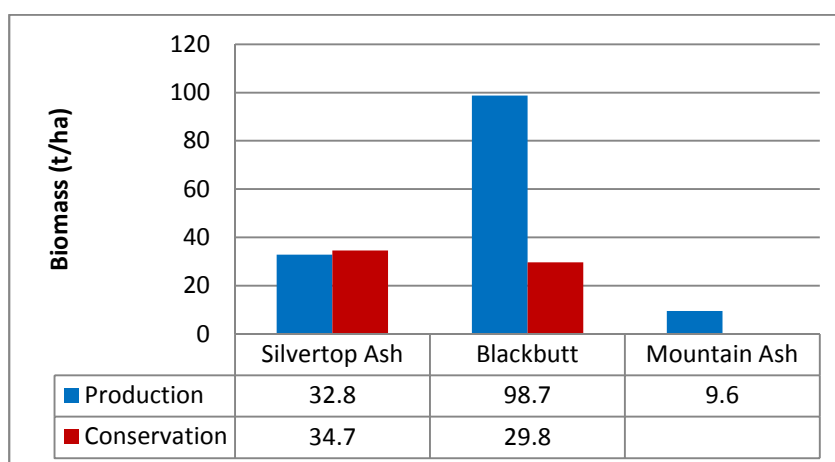
Figure 1.36. CWD biomass < 30 cm diameter (0% moisture content).



The biomass in ground CWD > 30 cm diameter for silvertop ash trees did not differ significantly between the production and conservation sites (Figure 1.36). Overall, the biomass in ground CWD > 30 cm diameter exceeded that of ground CWD < 30 cm diameter for NSW forests (Figures 1.36 and 1.37). This was not the case for mountain ash – following the 1939 fires, timber was salvaged from some of the stands killed (Raison *et al* 2007). The absence of large standing dead trees at the production site suggests that this site was subjected to salvage harvest – the proximity of the site to existing sawmills in Toolangi at the time adds weight to this suggestion. Another possible explanation for this is related to the fact the production site had very healthy trees, with little evidence of significant decay. Thinning in the 1960s would have removed many of the smaller trees, resulting in less competition for the remaining trees and hence less self-thinning. This would have resulted in lower tree mortality, and thus lower levels of large CWD on the ground.

The estimated CWD for the mountain ash conservation site derived using the traditional transect method was very high (316 t / ha), which would bring the total dry biomass of the site to 1111 t / ha. However, there is considerable uncertainty around this figure, as the biomass could not be directly weighed.

Figure 1.37. CWD biomass > 30cm diameter (0% moisture content)



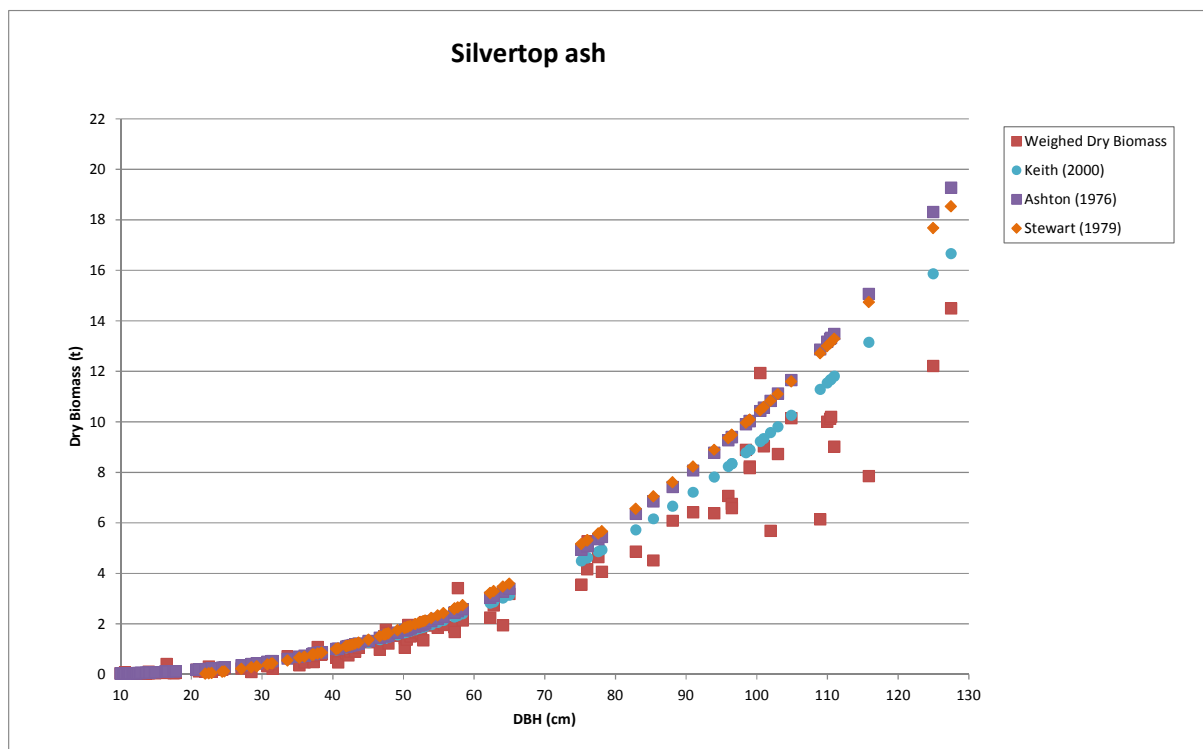
1.3.7. Above ground biomass estimated by biomass equations

For each region two “project specific” additive biomass equations were developed based on the study data; one using DBH as a single predictor, and the other using a combined predictor variable of DBH and height (D2H) (please see Chapter 2 for more details). In addition to the project specific equations, a number of relevant published allometric equations were used, and the estimates of above-ground biomass were compared against the biomass derived from actual weighed data from each site.

NSW South Coast – Silvertop Ash

The silvertop ash weighed data was compared to estimates derived from three available biomass equations (Figure 1.38), as well as the project specific equation (Chapter 2). The weighed data for silvertop ash was compared to estimates derived from three available biomass equations (Figure 1.39). The published biomass equations, while reasonable at predicting biomass in the lower DBH range, overestimated the biomass as the DBH increased (Figure 1.38).

Figure 1.38. A comparison between the dry biomass for silvertop ash for the combined dataset (production and conservation sites) as determined by direct weighing and biomass equations

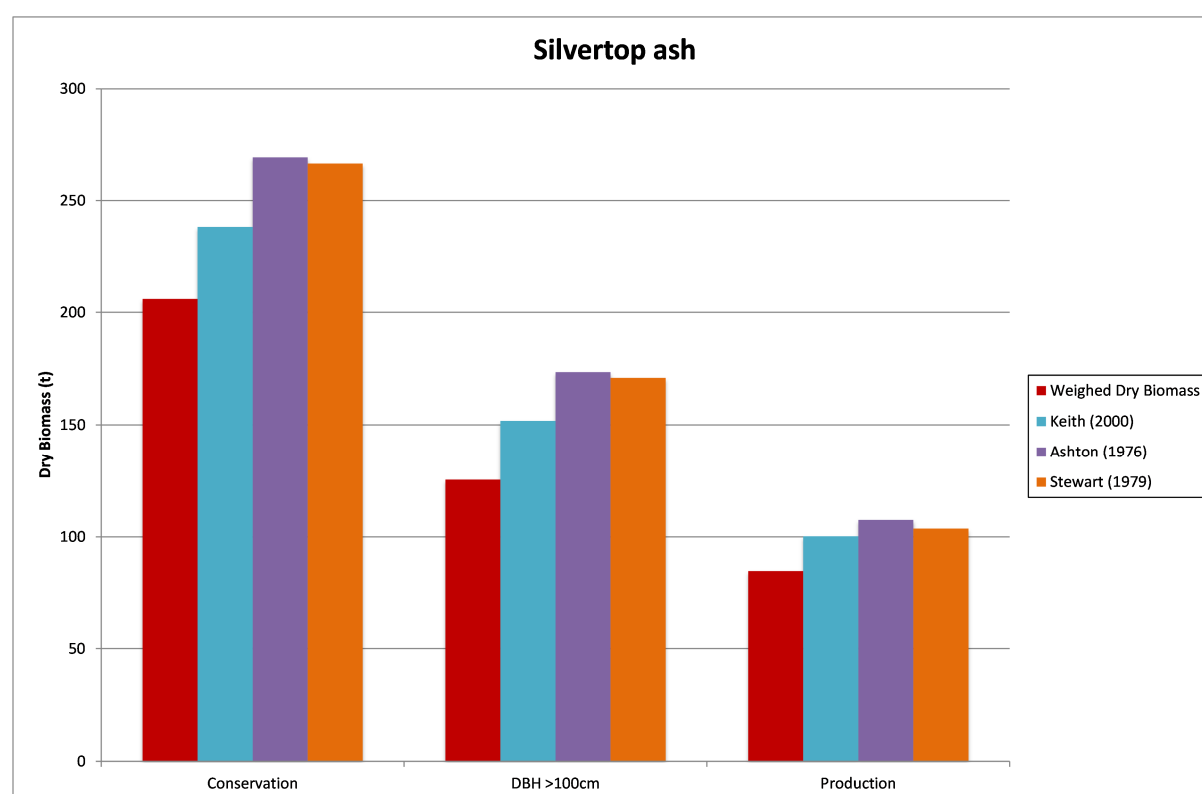


Ashton’s equation (Ashton 1976) was developed for silvertop ash and mountain ash (both sites in VIC), whereas the additive equation proposed by Stewart et al (1979) was developed specifically for silvertop ash (grown in Beenak, VIC), with both equations using DBH as the predictive variable. Ashton’s equation (1976) was based on five 27-year old silvertop ash trees which were felled, with branches and leaves weighed in the field and stem biomass estimated from volume estimations and physical parameters derived from stem discs. Stewart

et al (1979) sampled ten silvertop ash trees located at a study site ten km north of Genoa, eastern VIC. The DBH of the trees ranged between 28 and 89 cm, with the larger trees being more than 100-year old. The diameter of the stem and branches from every tree was measured, and biomass estimated based on destructive sampling of a sub-set of stem discs and branch samples of varying sizes. Keith's equation (Keith et al 2000) is a general equation for native sclerophyll forests (including for the key species in this study), based on 25 records and 135 data points, with biomass of individual trees calculated for trees with DBHs ranging from 10 to 100 cm DBH (10 cm increments).

There was a higher degree of variability in the biomass of the weighed trees for the larger DBH classes (Figure 1.38); this may be partly due to the varying amount of decay in the crown and within the stem as the trees age. However the project specific equation does seem to account for some of this variation, suggesting that the equation may also be driven by tree height (Figure 1.38). All three published equations overestimated the total site biomass (Figure 1.39), more so for the conservation site, which had a greater number of large (Figure 1.39).

Figure 1.39. A comparison between the dry biomass for silvertop ash for the production and conservation sites as well as trees with a DBH >100cm as determined by direct weighing and biomass equations

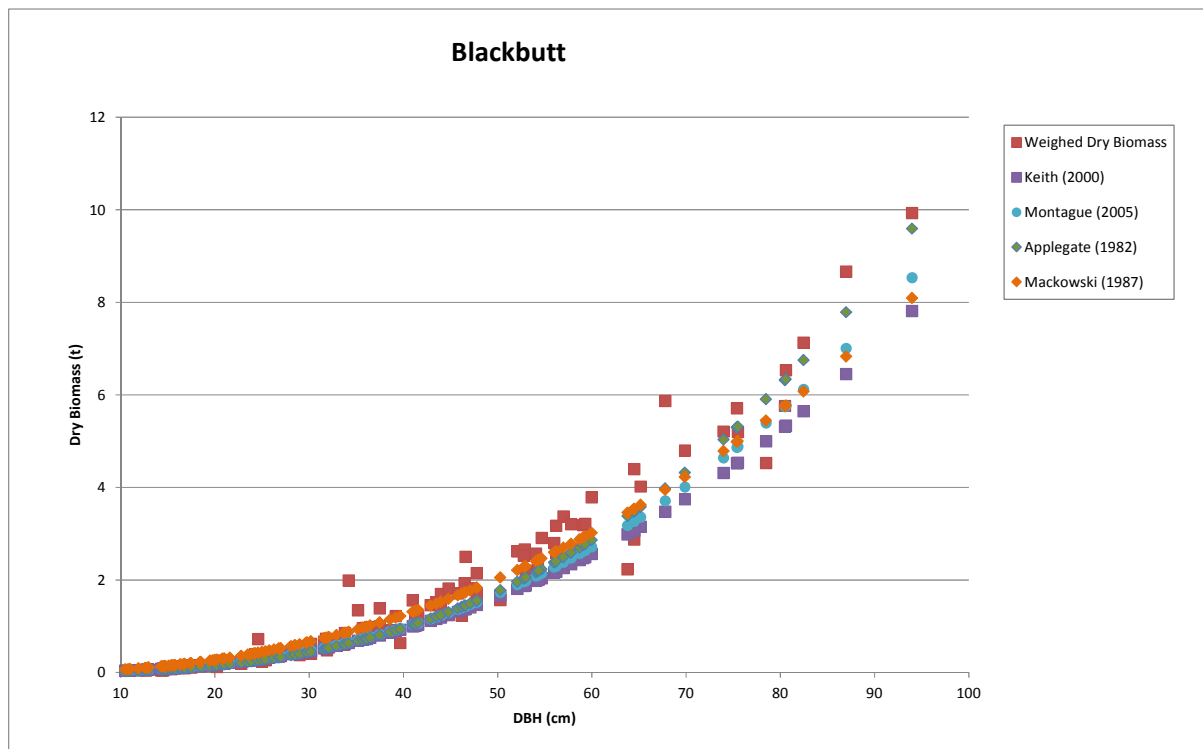


NSW North Coast – Blackbutt

The weighed data for blackbutt was compared to estimates derived from four available equations (Figure 1.40). Applegate (1982) derived a number of equations based on blackbutt biomass data collected from Fraser Island, Queensland - we used the regeneration old growth equation which covered a DBH range of 13-129 cm (Keith et al 2000). Twenty-nine large trees ranging from 12.2-128.9cm were felled; however the DBH class distribution was

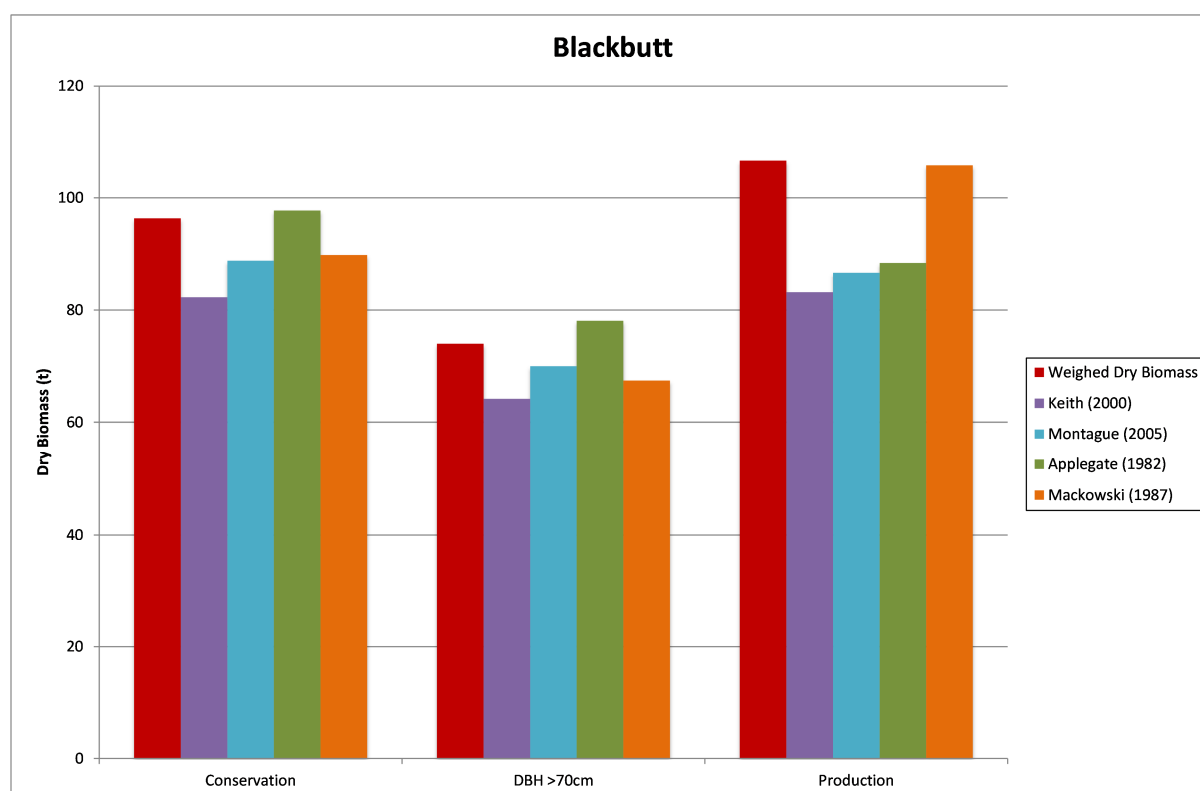
limited with only one tree with DBH greater than 60 cm (128.9 cm). The branches and foliage of each tree were weighed in the field for all but the large tree and samples taken for dry weight determinations. Biomass for the stem was calculated for logs based on the volume of the sapwood, heartwood and bark and density determinations. Montague et al (2005) derived a number of equations specifically for blackbutt across seven study sites (five in the central and north coast of NSW and two at Fraser Island, Queensland), including plantations and native forests - we used the general DBH equation which was fitted across all study sites (DBH range 5-129 cm) without a correction factor. The biomass estimates were based on a mix of direct measurements of the fresh mass of the entire tree and subsampling techniques to estimate the biomass of tree components. Mackowski (1987) proposed a number of equations for blackbutt with specific DBH ranges, derived for blackbutt-dominated forests 30-40 km south east of Grafton, NSW, based on measurements of ninety trees with DBH up to 189.6 cm - we used the equation for a DBH range of 45-135 cm (Keith et al 2000). Stem and branch volume were estimated using a log volume formula, with biomass estimated using published density values.

Figure 1.40. A comparison between the dry biomass for blackbutt for the production and conservation sites combined as determined by direct weighing and biomass equations



All of the equations (with the exception of Mackowski's) underestimate the biomass through the 40-70 cm DBH range and the total biomass for the production site, where the DBH of the largest blackbutt was 70 cm (Figures 1.40 and 1.41). Although Mackowski's equation faithfully predicts biomass in the production site, this was partly due to the underestimation of the biomass for the larger DBH classes, which was offset by overestimation of the biomass for the lower DBH classes (Figure 1.40). The biomass estimations for the conservation site (Figure 1.41) were more in line with the actual weighed data - this is highlighted in Figure 1.40, where the equations generally have a better fit through the larger DBH classes (especially Applegate's).

Figure 1.41. A comparison between the dry biomass blackbutt for the production and conservation sites as determined by direct weighing and biomass equations



Victoria – Mountain ash

The biomass was directly weighed for the mountain ash “production” site and for the “additional large trees”, but as mentioned earlier not for the “conservation” site. The biomass for the trees in the conservation site was derived from the allometric relationship derived using the combined data from the “production” and “large tree” sites (not described here). Feller’s equation (Feller 1980) is a DBH and height equation developed specifically for mountain ash located at the Maroondah catchment area (VIC), based on destructive sampling of six trees of varying size (four trees ranging from 15-30 cm DBH, one around 50 cm and the largest one around 70 cm DBH). The biomass of the crown was derived by a combination of determining the moisture content of a sub-sample of smaller diameter branches, and the use of published density values for the measured larger diameter branches. The biomass of the stem was also calculated based on published density values and stem measurements. The “Sillett” equation (Sillett *et al* 2010) is not actually the equation derived by Sillett *et al*, as the relationship used for whole tree biomass estimations was not made clear in the text. The equation used here was derived using their published biomass data for 22 trees with a DBH range of 80-312 cm, located at Kinglake National Park, VIC (Sillett *et al* 2010). The biomass estimates were based on detailed field measurements (especially of the crown component), with subsampling techniques used to estimate the biomass of tree components.

Biomass estimates varied considerably according to the equation used. Apart from Feller’s equation, the other tested equations performed well when estimating the biomass for the production site, which had a DBH range of 20-117cm (Figures 1.42 and 1.43). Feller’s equation considerably overestimated the biomass in the mountain ash trees (Figures 1.42 and

1.43) - a similar overestimation using Feller's equation was observed by Keith *et al* (2000), who noted that Feller's equation was of an unusual form, which appeared to give erroneous values at high DBH (above 40 cm). The variability in biomass estimates was greater for trees with DBH > 100cm (Figure 1.43) - the Keith and Ashton equations both overestimate biomass while the derived Sillett equation provides a very good estimate.

Figure 1.42. A comparison between the dry biomass for mountain ash for the production and conservation sites combined as determined by direct weighing and biomass equations

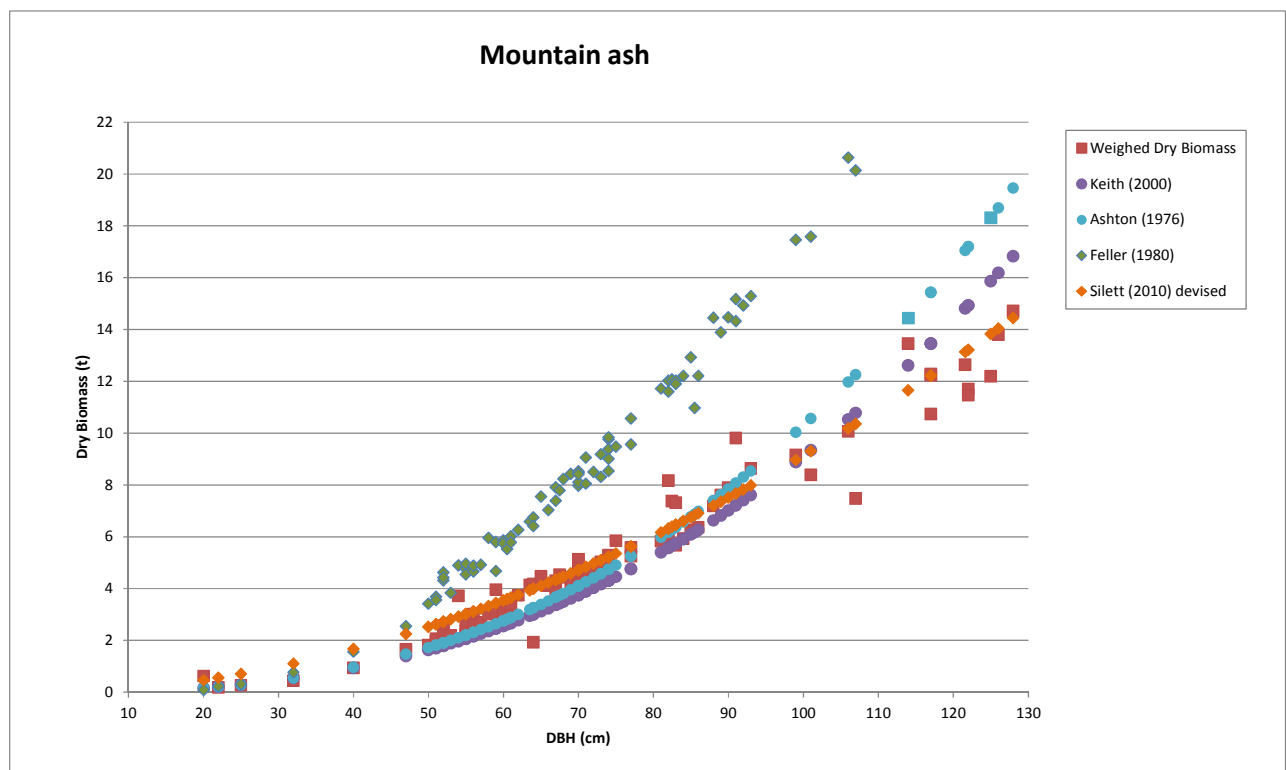
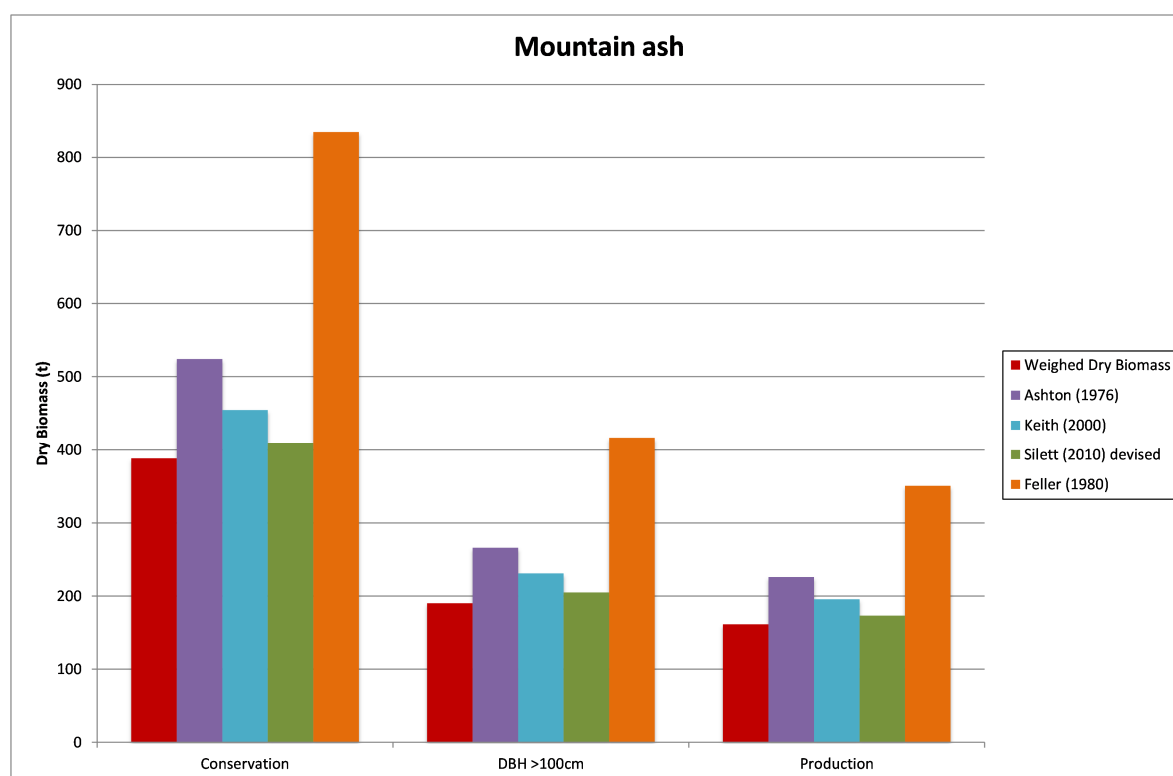


Figure 1.43. A comparison between the dry biomass mountain ash for the “production”, “conservation” and “large tree” sites as determined by direct weighing and biomass equations



1.4 Discussion

A summary of key published above-ground forest carbon estimates for forests comparable to those included in this study is included in Table 6. The estimates varies considerably, ranging from 150 t C / ha for regrowth spotted gum forests in the NSW South Coast to 744 t C / ha for old-growth mountain ash in VIC (Table 1.9).

The range of above-ground C estimates varies considerably between mature native forest types (Table 1.9). For mature silvertop ash dominated forests in the Eden region of NSW which are largely undisturbed by human activity, there is good agreement between the values published by Turner and Lambert (1986), which were based on site-specific allometrics (not presented in their article), and the value for the conservation site in this study (Table 1.9). These values could be assumed to approximate the carbon carrying capacity of those forests. Values reported by Roxburgh *et al* (2006) for the predicted potential carbon carrying capacity for forests around Kioloa (approximately 240 km north of Eden, dominated by species other than silvertop ash) are considerably higher (363.7 t C/ha), although their estimated current carbon stocks were within the range reported above. Roxburgh *et al* (2006) based their estimates on the allometric relationships derived by Ash and Helman (1990), which in turn derive biomass from measured tree volumes and a combination of site-specific and published density values. Ximenes *et al* (2005) determined biomass for a range of mature spotted gum dominated forests around Batemans Bay (approximately 200 km north of Eden) based on direct weighing of the tree components. The values ranged from approximately 119 t C/ha for a low-quality site to 198 t C/ha for a high-quality site. No estimates were found in the literature for biomass in older regrowth blackbutt forests in the North Coast of NSW.

Table 1.9. Summary of published above ground C stock of selected native forests (excluding coarse woody debris and litter)

Forest type	Location	Above ground C (t C ha ⁻¹)		Reference
		Mean	Range	
Mixed	Kioloa, NSW (predicted C carrying capacity)	363.7	±11.2	Roxburgh <i>et al</i> 2006 ¹
<i>E. sieberi</i>	Eden (200-250+yrs for the older/dominant cohort and 60-70yrs for the second cohort.)	238	-	This study (conservation site)
<i>E. sieberi</i> / <i>E. consideniana</i>	Eden, NSW	229	-	Turner & Lambert 1986 ²
<i>E. sieberi</i> / <i>E. obliqua</i>	Eden, NSW	221	-	Turner & Lambert 1986 ²
Mixed	Kioloa, NSW (current C stocks)	210.6	±19.5	Roxburgh <i>et al</i> 2006 ¹
<i>E. agglomerata</i> / <i>E. sieberi</i> / <i>E. muellerana</i>	Eden, NSW	203	-	Turner & Lambert 1986 ²
Dominated by <i>C. maculata</i>	Batemans Bay region, South Coast, NSW (regrowth)	150	119 - 198	Ximenes <i>et al</i> 2005 ³
<i>E. sieberi</i>	Eden (mostly regrowth from the 1950's)	103	-	This study (production site)
<i>E. regnans</i>	Victoria (old-growth)	744	±41	Keith <i>et al</i> 2014 ⁴
<i>E. regnans</i>	Victoria (1905-1906 regrowth)	415	-	This study (conservation site) ⁵
<i>E. regnans</i>	Victoria (1939 regrowth)	373	-	This study (production site)
<i>E. regnans</i>	Victoria (average for 1939 regrowth)	341	±64	Keith <i>et al</i> 2014 ⁴
<i>E. regnans</i>	Victoria (average for 1939 regrowth)	292	234-351	Fedrigio <i>et al</i> 2014
<i>E. pilularis</i>	North Haven, NSW (multi-aged stand – mix of trees >200 years and from circa 1920-30s.	209	-	This study (conservation site)
<i>E. pilularis</i>	Mid-north coast of NSW (mostly 1950's regrowth)	129	-	This study (production site)

¹ Dominant species included spotted gum (*Corymbia maculata*), *Eucalyptus pilularis*, *Eucalyptus sieberi*, *Eucalyptus botryoides* and *Corymbia gummifera*. The area had also been periodically subject to unplanned fires, controlled hazard reduction burning and post-harvest burns. Tree biomass was calculated using the regression equations developed by Ash & Helman (1990) modified to include an adjustment for internal tree decay. ²The stands were fully stocked and carrying near maximum biomass when compared with typical stands in the area. Tree biomass was estimated using tree biomass equations developed for the area. ³Forests dominated by spotted gum, biomass directly weighed for all trees. ⁴Stem and branch volumes were calculated using an allometric equation for mountain ash (Sillett *et al.* 2010) based on tree diameter (accounting for stem buttressing and internal wood decay), and multiplied by wood density and carbon concentration. ⁵ Biomass estimated from site-specific allometrics including trees of large DBH.

The above-ground carbon values reported by Fedrigo *et al* (2014) for mountain ash trees of similar age to our production site were considerably lower (Table 1.9). Their value however is in good agreement with the sub-regional estimates provided by VicForests for pure mountain ash stands in the Central Highlands of Vic (approximately 287 t C/ha, as converted from values reported earlier). There is good agreement between the biomass values reported in this study and those in Keith *et al* (2014) for 1939 regrowth mountain ash forests. However, our study site was of very high productivity as discussed earlier, with substantially more biomass than estimates for mountain ash sites of average productivity. The average carbon stocks suggested by Keith *et al* (2014) for old-growth mountain ash stands (744 t C / ha) are higher than the value calculated for our mountain ash 1905-1906 age class “conservation” site (415 t C / ha), noting that the old growth stands in Keith *et al* (2014) are approximately 250-years old, from fires in the mid 1700’s. For mountain ash, the forest at 250 years is dominated by typically 20 trees/ha, and a relative decrease in biomass is expected, as the tall open eucalypt forest progresses from aggrading to a steady-state (Attiwill 1994). Mountain ash is a relatively fast growing species, with some trees less than 70 years old being more than 80 m tall (Sillett *et al* 2010). According to Sillett *et al* (2015), such rapid growth likely occurs at the expense of fire and decay-resistance (Loehle, 1988), thus reducing their maximum longevity. This, coupled with visual observations of mountain ash old-growth stands, suggests that the onset of decay in old-growth sites is likely to occur relatively early in the old-growth stage of those stands, with significant biomass implications. Ambrose (1982) reports that the development of hollows may not be apparent in mountain ash trees until the trees are at least 120 years old. The findings from Sillett *et al* (2015) suggest that decay is indeed a major factor to consider in the estimation of biomass for old-growth mountain ash stands. They estimated biomass in a 0.73 ha plot in Kinglake National Park, VIC, considered by the authors to represent the upper level of density of large trees for mature mountain ash forests. The maximum estimated aboveground carbon mass was 706 t /ha – this was considered maxima because it did not account for mass loss due to decay in living trees - the authors noticed extensive decay and numerous hollow trunks and limbs. If half of the mountain ash heartwood volume were lost to decay, as suggested by Sillett *et al* (2015), stand-level carbon mass in their plot was reduced to 438 t /ha. This value is similar to the estimated value for our 1905/1906 mountain ash conservation site (410 t C/ha).

When comparing total above ground biomass including CWD, the differences between the NSW production and conservation sites were reduced. The higher proportion of CWD in the production sites is a consequence of management practices, as also observed by others. Bridges (2005) recorded residue volumes of 116 m³ / ha before harvesting increasing to 189 m³ / ha after harvesting and decreasing to 135 m³ / ha following post-harvest burning in a dry sclerophyll forest in Eden. A study in Kioloa, NSW by Woldendorp *et al* (2002), where a plot had been harvested ten years prior to the study had the highest levels of forest floor CWD amongst the five mature native forests included in their study. As previously mentioned there was a large amount of CWD >30 cm observed at the mountain ash conservation site, and an absence of dead standing trees in the larger DBH classes - the largest dead standing tree was 43.5 cm - this possibly a result of a past event altering the structure of the stand. The CWD figure for the mountain ash conservation site of 316.7 t/ha (dry biomass) was estimated using the transect method, and while there was a large amount of CWD observed at the site there is considerable uncertainty as to the accuracy of this figure. Woldendorp *et al* (2002) reported an average value of 109 t/ha for a wet sclerophyll forest; however their report covered a large geographical area, included different stand ages and different management regimes.

The crown encompassed a larger proportion of the total dry biomass (35%) for the NSW conservation sites when compared to their respective production sites (14%, 22%), (Table 1.8). These differences are due to the larger proportion of mature trees with well-developed crowns and a higher portion of decay in the conservation sites, coupled with the absence of management whereby trees with less than optimal form are normally removed during thinning operations. The non-commercial component in the blackbutt sites was much larger than the other sites (Table 1.8, Figure 1.31).

The proportion of biomass in the stump of the different key species ranged from 3-7%. Stumps may have greater longevity than other residue components such as bark and small branches, and generally withstand post-harvest burns with only light surface charring.

The production volumes and residues for the NSW sites were comparable with forecasts provided by the state agencies, although the percentage split of the commercial products varied somewhat. The mountain ash site yielded higher volumes than the average for the area, and the residue component of 301 m³/ha (Table 1.4) or 166 t / ha was within the range 85-205 t / ha of post-harvest residue (slash) levels as described by Raison *et al* (2007).

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Part 2. Biomass Equations

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2.1 Section summary

- The fresh weight for the stump, logs, bark of logs and crown derived from Section 1 was converted to dry weight using the average moisture content for trees of the same species within the same DBH class.
- A system of five nonlinear additive equations was specified for the four biomass components and total tree biomass with DBH as the only predictor.
- To further improve the accuracy in biomass estimation, total tree height was incorporated in the combined predictor variable (D^2H) in another system of additive equations.

Key findings:

- The two systems of additive biomass equations enabled the estimation of log and residue biomass of important commercial tree species in tall open eucalypt forests of south-eastern Australia.
- The accuracy of estimation was generally greater for the log and bark components than for the stump and crown components. For total tree biomass, the accuracy of estimation was generally greater than individual biomass components.
- Adding tree height to the predictor resulted in more accurate estimates for log and bark as the reduction in root mean squared error (RMSE) was 9%-21% for stem biomass and 9%-16% for bark biomass. However, the results were mixed for the stump, crown and total tree biomass.

2.2 Methods

2.2.1 Fresh and dry weight determination

In each plot, all trees with DBH equal to or greater than 10 cm were felled by logging contractors and measured for total tree and component fresh weight using methods described in Ximenes *et al* (2008). For trees of commercial size, one or more commercial logs were cut from each stem by following current commercial log specifications. After the length and diameter of a log were measured, it was weighed first with the bark on and then with the bark removed to obtain the fresh weight of log and bark. Damaged or defect stem sections, if any, were cut and weighed together with the crown. For practical reasons the crown component was only weighed with the bark intact. As stumps could not be weighed, the fresh weight of each stump was estimated from its volume calculated from stump height and diameter. Such complete weighing generated four fresh weight components for each tree of commercial size: (1) stump, (2) stem (i.e. logs without bark), (3) bark of logs and (4) crown which included an unmerchantable upper stem, damaged and off-cut stem sections. The fresh weight for each

component was later converted to dry weight using average moisture content for trees of the same species within the same DBH class. Finally, the total above ground dry biomass for each tree was calculated as the sum of the four dry weight components (Figure. 2.3).

2.2.2 Additive biomass equations

Following the approach of Bi *et al.* (2004), a system of five nonlinear additive equations was specified for the four biomass components and total tree biomass with DBH (D) as the only predictor, as follows:

$$\begin{aligned} Y_1 &= e^{\beta_{10}} D^{\beta_{11}} + \varepsilon_1 \\ Y_2 &= e^{\beta_{20}} D^{\beta_{21}} + \varepsilon_2 \\ Y_3 &= e^{\beta_{30}} D^{\beta_{31}} + \varepsilon_3 \\ Y_4 &= e^{\beta_{40}} D^{\beta_{41}} + \varepsilon_4 \\ Y_5 &= e^{\beta_{10}} D^{\beta_{11}} + e^{\beta_{20}} D^{\beta_{21}} + e^{\beta_{30}} D^{\beta_{31}} + e^{\beta_{40}} D^{\beta_{41}} + \varepsilon_5 \end{aligned} \quad (1)$$

where Y_1 to Y_4 represent the biomass of stump, logs without bark, bark off logs, crown (including damaged stem and off-cuts) in kg respectively, Y_5 is the sum of the four biomass components i.e. the total tree biomass, and β_{ij} are coefficients. As Y_1 to Y_5 represent different biomass components of the same tree, the error terms are inherently correlated and in the matrix algebra notation can be expressed as

$$\boldsymbol{\varepsilon} = [\varepsilon_1, \varepsilon_2, \varepsilon_3, \varepsilon_4, \varepsilon_5]' \quad (2)$$

with the expectation $E(\varepsilon)=0$ and a variance and covariance matrix $E(\varepsilon\varepsilon') = \Sigma$. Each coefficient in the model is shared between two equations as cross-equation constraints are placed on the structural parameters to ensure additivity of biomass estimates i.e. to eliminate the inconsistency between the sum of predicted values for biomass components and the prediction for the total tree biomass. This type of model specification was first reported by Reed and Green (1985) when demonstrating the use of a loss function in forcing additivity among biomass equations. The system of equations specified in model (1) was fitted to the data in the way described in Bi *et al.* (2004) for each species using the generalised method of moments (GMM) that produces efficient parameter estimates under heteroscedastic conditions without any specification of the nature of the heteroscedasticity (Greene 1999).

As demonstrated by Bi *et al.* (2004), Williams *et al.* (2005) and Bi *et al.* (2015), tree height, H, can be used together with diameter in the combined predictor variable, D^2H , to improve the accuracy in biomass estimation. Therefore, another system of additive equations was specified with the combined variable of DBH and tree height, D^2H , replacing D in model (1) such that:

$$\begin{aligned} Y_1 &= e^{\beta_{10}} (D^2H)^{\beta_{11}} + \varepsilon_1 \\ Y_2 &= e^{\beta_{20}} (D^2H)^{\beta_{21}} + \varepsilon_2 \\ Y_3 &= e^{\beta_{30}} (D^2H)^{\beta_{31}} + \varepsilon_3 \\ Y_4 &= e^{\beta_{40}} (D^2H)^{\beta_{41}} + \varepsilon_4 \end{aligned}$$

$$Y_5 = e^{\beta_{10}} (D^2 H)^{\beta_{11}} + e^{\beta_{20}} (D^2 H)^{\beta_{21}} + e^{\beta_{30}} (D^2 H)^{\beta_{31}} + e^{\beta_{40}} (D^2 H)^{\beta_{41}} + \varepsilon_5 \quad (3)$$

Parameters were estimated in the same way as described for model (1). For each biomass component, a generalized R^2 was calculated to indicate the goodness of fit:

$$R^2 = 1 - \frac{\sum_{i=1}^T (Y_i - \hat{Y}_i)^2}{\sum_{i=1}^T (Y_i - \bar{Y})^2} \quad (4)$$

where Y_i and \hat{Y}_i ($i=1, 2, 3, 4, 5$) are the observed and predicted values and \bar{Y} is the mean observed value of dry to fresh weight ratio.

2.3 Results

In Figures 2.1 and 2.2 we present the diameter distribution of the trees and the tree height in relation to DBH of all trees, respectively, from the six plots. In Figure 2.3 we present the total fresh and dry weights for all trees from the six plots. With diameter as the only independent variable in the system of additive biomass equations, the estimated exponent of D for log biomass, i.e. β_{21} in model (1), was 1.82 for *E. regnans*, 1.83 for *E. pilularis* and 2.07 for *E. sieberi* (Table 2.1, Figure. 2.4). The estimated exponent for bark biomass β_{31} was 1.82 for *E. regnans*, 1.83 for *E. pilularis* and 2.07 for *E. sieberi*. In comparison with these two parameters, the exponents for the stump and crown components β_{11} and β_{41} had a greater range and so were more variable. The accuracy of estimation was generally greater for the log and bark components than for the stump and crown components as shown by the fit statistics in Table 2.1. For total tree biomass, the accuracy of estimation was much greater than all individual biomass components.

With the combined variable as the predictor in the system of additive biomass equations, the estimated exponent of $D^2 H$ for stem biomass, i.e. β_{21} in model (3), was 0.86 for *E. regnans*, 0.91 for *E. pilularis* and 0.96 for *E. sieberi* (Table 2.1, Figure. 2.5). The estimated exponent for bark biomass β_{31} was 0.74 for *E. regnans*, 0.88 for *E. pilularis* and 0.93 for *E. sieberi*. The exponents for the stump and crown components β_{11} and β_{41} also had a greater range and were more variable.

Adding tree height to the predictor resulted in more accurate estimates for stem (i.e. log) and bark biomass, but the results were mixed for the stump and crown components. The reduction in root mean squared error (RMSE) was 9%-21% for stem biomass and 9%-16% for bark biomass (Table 2.1). For stump biomass, there was little change in RMSE for *E. pilularis*, an 18% reduction for *E. regnans*, but a 22% increase for *E. sieberi*. For crown biomass, there was a reduction of 31% for *E. regnans*, but a 14% and 55% increase for *E. pilularis* and *E. sieberi* respectively. The change in RMSE for total tree biomass was negligible for *E. pilularis* and *E. regnans*, but a 51% increase for *E. sieberi*.

Figure 2.1. Diameter distribution of all live trees with DBH equal to or greater than 10 cm in each of the six paired conservation and production plots. The number of dead standing trees in each plot is given on the top right hand corner of each panel.

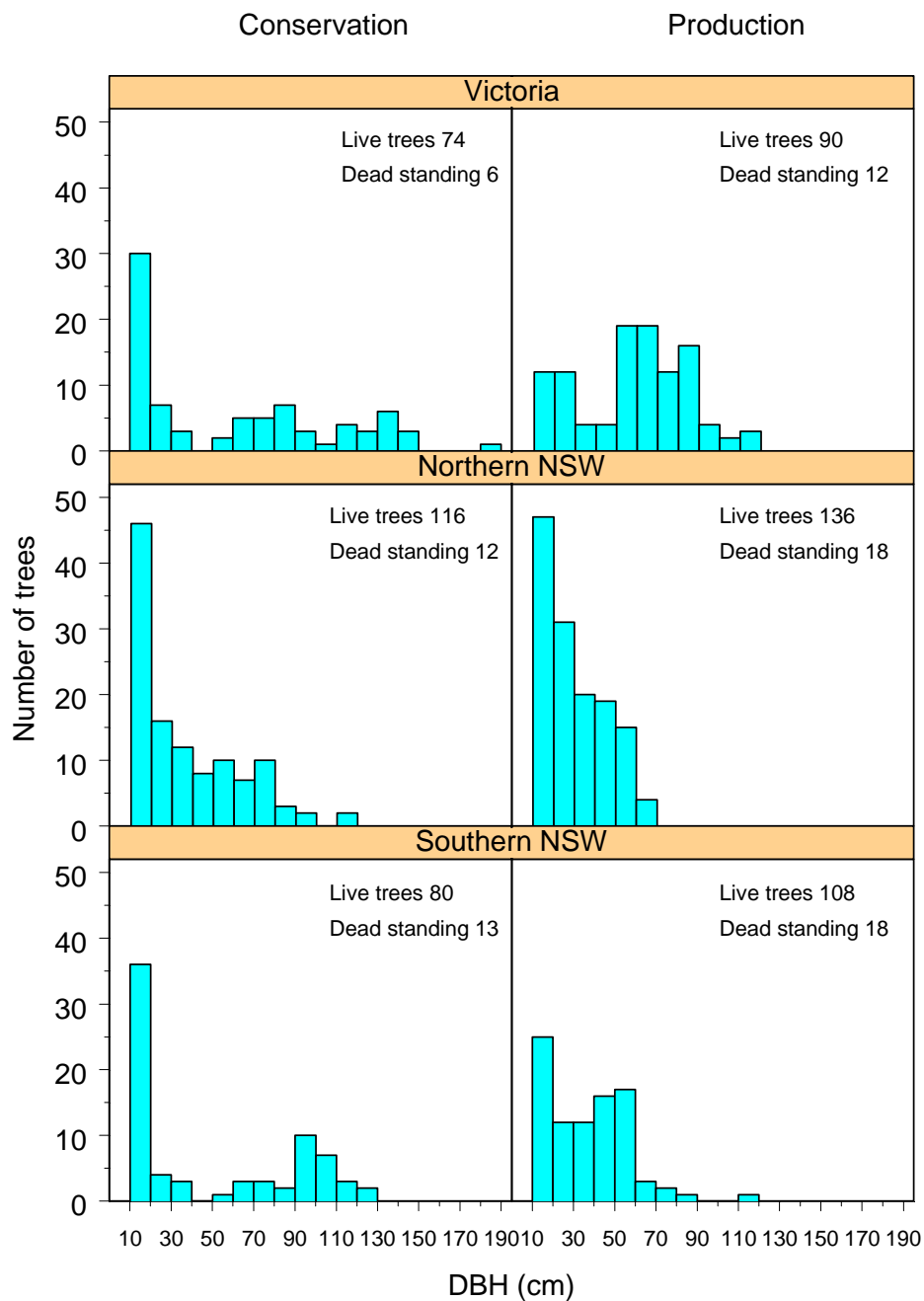


Figure 2.2. Tree height in relation to DBH in each of the six paired conservation and production plots. Points showing zero height indicate that height measurements were not taken for these trees.

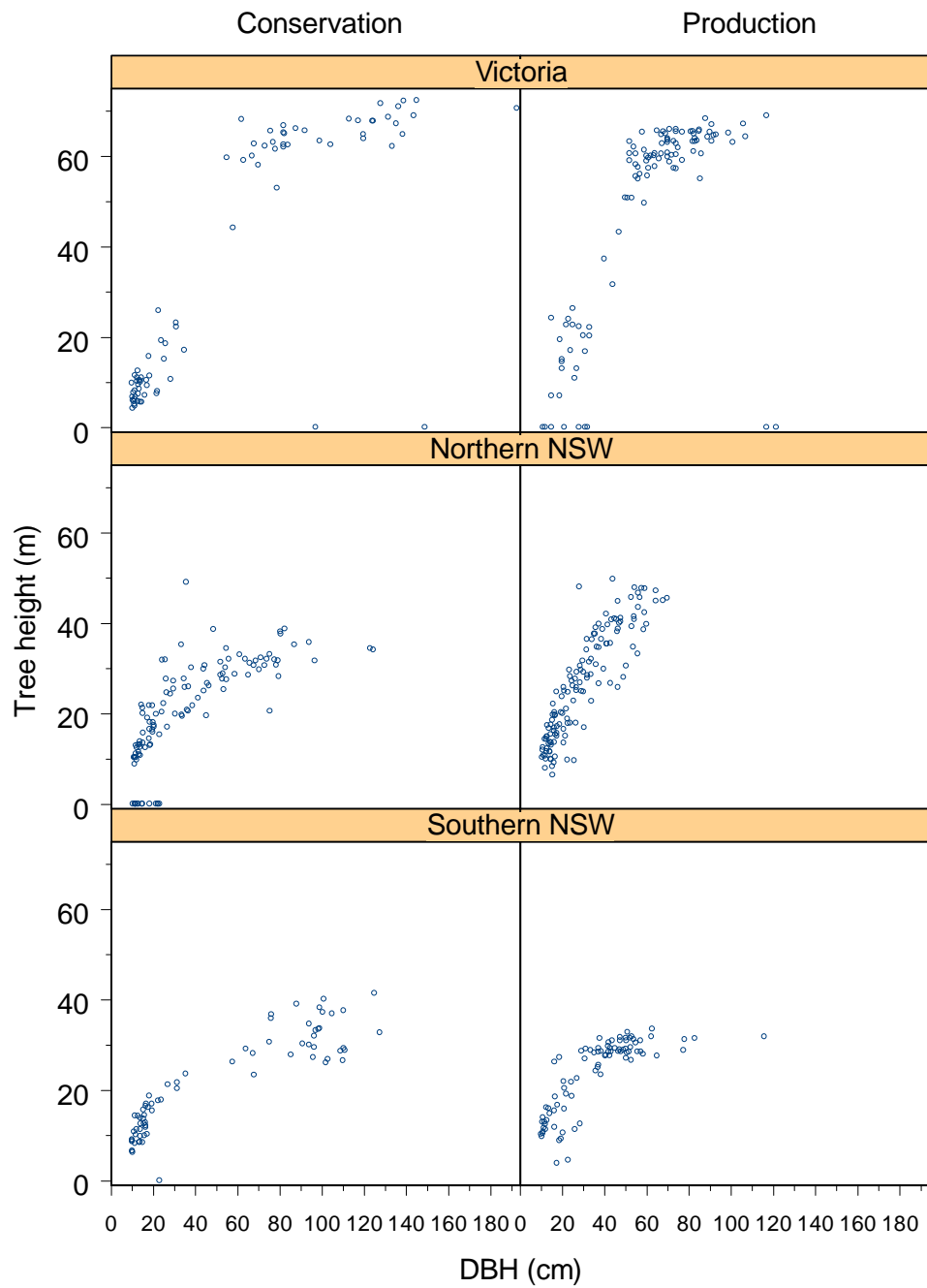


Figure 2.3. Total aboveground fresh weight (blue) and dry weight (red brown) of all individual trees that were weighed in each plot.

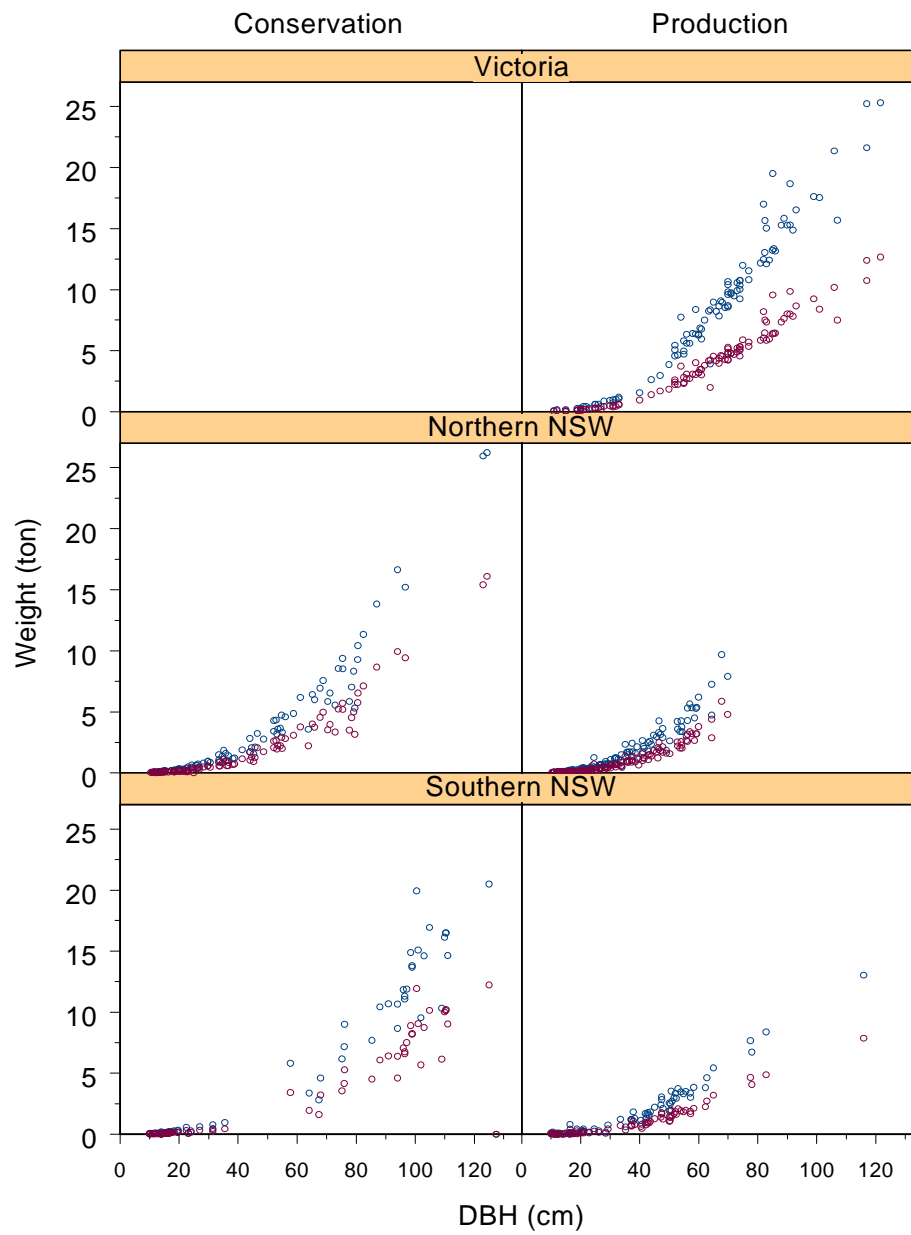


Figure 2.4. Observed values of biomass components and total tree biomass plotted against DBH for the three species together with the fitted curves.

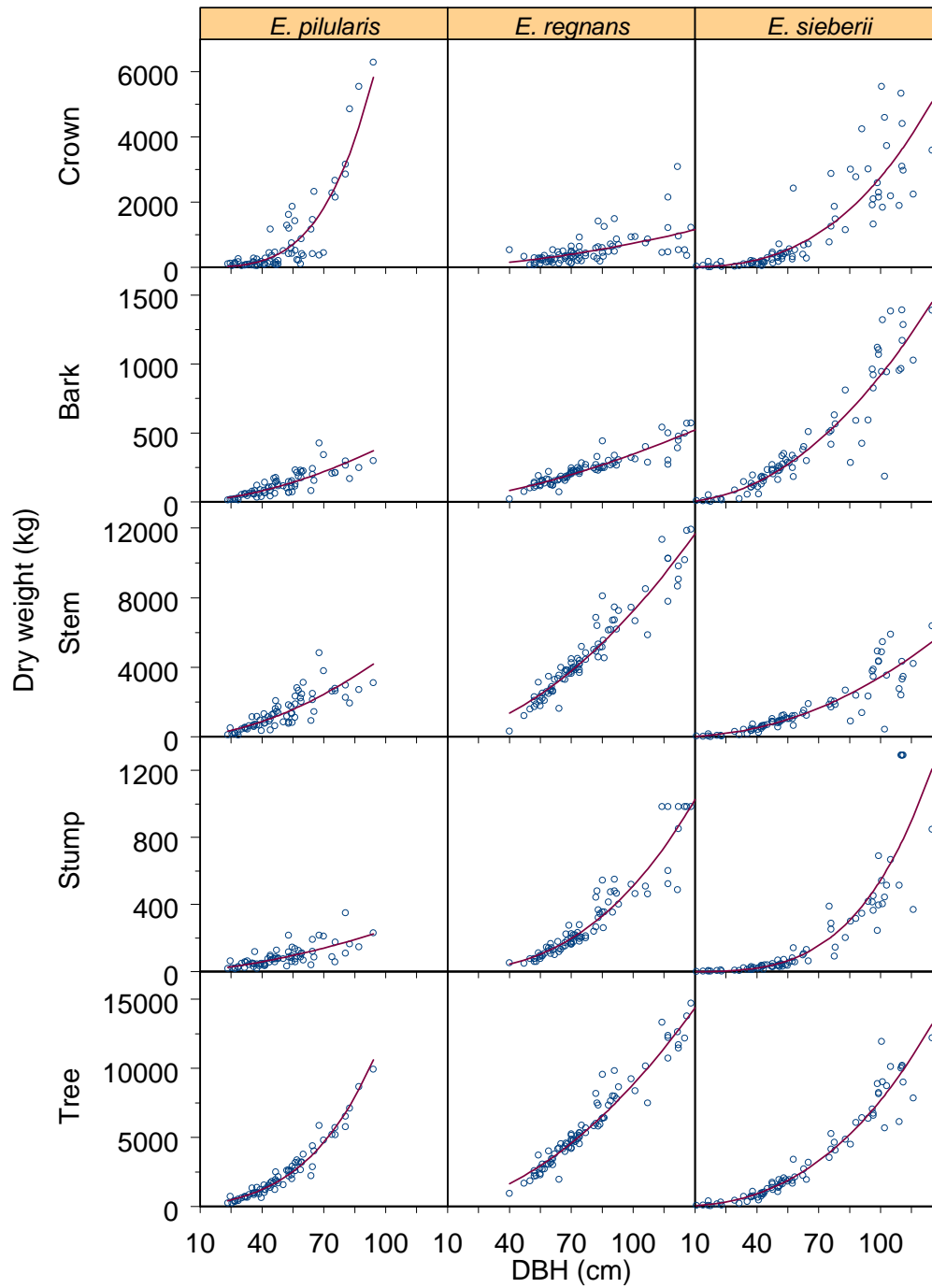


Figure 2.5. Observed values of component and total tree biomass plotted against the combined variable, D^2H , together with the fitted curves on natural logarithmic scales for the three species.

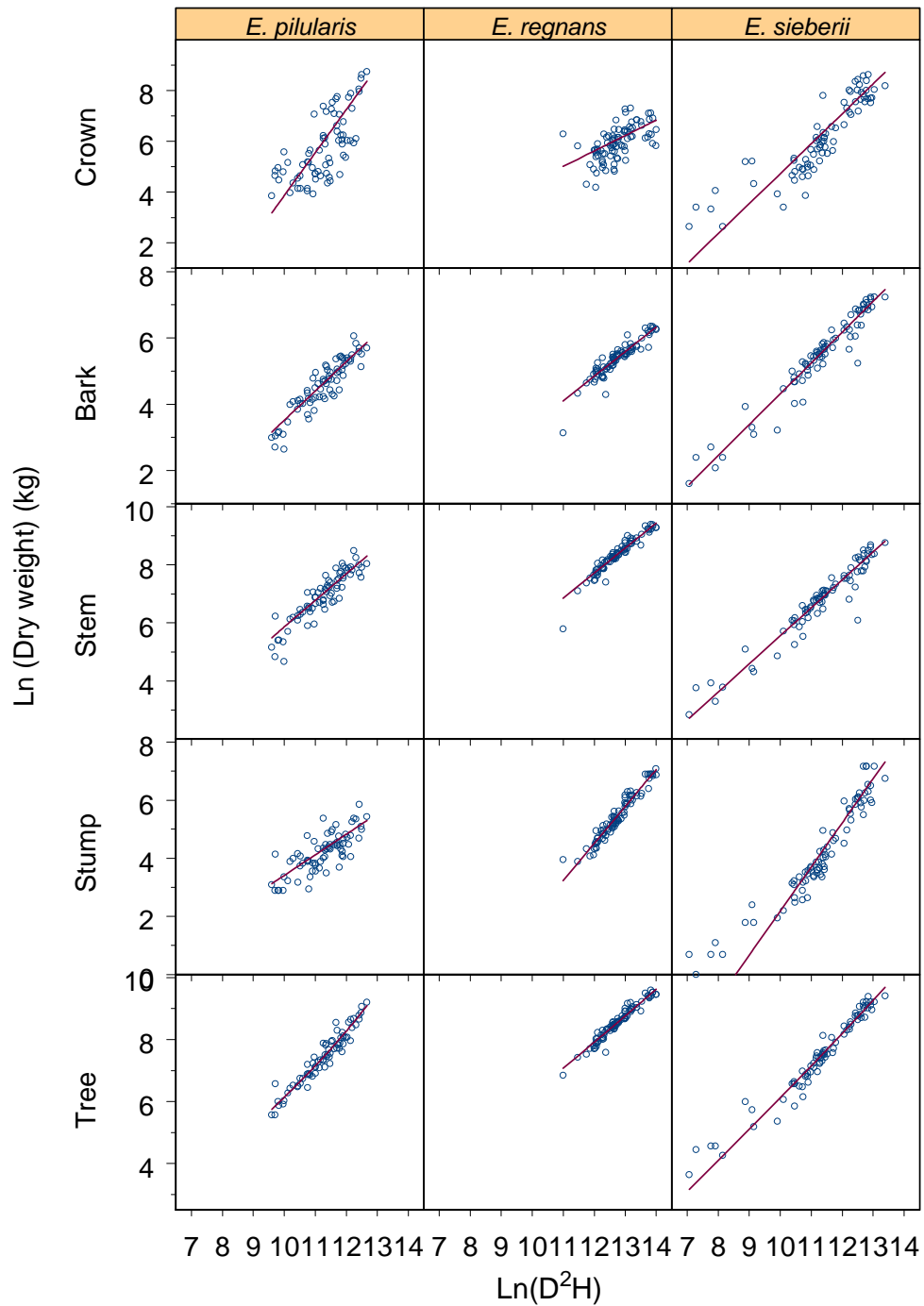


Table 2.1. Parameter estimates for the two systems of additive biomass equations of each species. DBH is the independent variable in one system, and the combined variable D^2H is the predictor in another. RMSE stands for root mean squared error.

Species	Predictor	N	i	Component	β_{i0}	β_{i1}	RMSE	R^2
<i>E. pilularis</i>	DBH	72	1	stump	-1.6938	1.5633	41.7	0.52
			2	stem	0.0460	1.8257	604.1	0.62
			3	bark	-2.1772	1.7821	50.4	0.66
			4	crown	-9.2652	3.9480	482.3	0.85
			5	tree			427.8	0.96
<i>E. regnans</i>	DBH	87	1	stump	-6.0148	2.6601	86.2	0.90
			2	stem	0.5057	1.8200	777.9	0.92
			3	bark	-1.3226	1.5572	46.1	0.86
			4	crown	-1.2664	1.7083	373.6	0.32
			5	tree			855.1	0.94
<i>E. sieberi</i>	DBH	78	1	stump	-10.0985	3.5611	153.9	0.77
			2	stem	-1.3748	2.0671	718.2	0.79
			3	bark	-2.5401	2.0331	147.7	0.86
			4	crown	-4.6137	2.7228	745.0	0.72
			5	tree			905.2	0.93
<i>E. pilularis</i>	D^2H	72	1	stump	-3.6695	0.7081	41.9	0.52
			2	stem	-3.2791	0.9142	514.3	0.73
			3	bark	-5.3047	0.8823	42.4	0.76
			4	crown	-13.0345	1.6900	747.8	0.64
			5	tree			646.9	0.90
<i>E. regnans</i>	D^2H	85	1	stump	-10.7815	1.2742	71.0	0.94
			2	stem	-2.6107	0.8605	709.6	0.93
			3	bark	-4.0560	0.7418	41.8	0.89
			4	crown	-1.6637	0.6069	257.9	0.31
			5	tree			837.6	0.94
<i>E. sieberi</i>	D^2H	78	1	stump	-13.0203	1.5202	187.1	0.65
			2	stem	-4.0905	0.9643	566.3	0.87
			3	bark	-4.9874	0.9303	133.0	0.89
			4	crown	-7.0648	1.1791	849.2	0.64
			5	tree			912.2	0.93

2.4 References

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Part 3. Long term C storage in HWPs and product substitution

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In this section, we move away from the forest and track the fate of the C in the commercial logs obtained from each of the study sites. The analysis includes all key emissions associated with the transport and processing of the biomass, the physical C storage in harvested wood products (HWP), and the substitution impacts associated with the use of biomass for bioenergy and with the use of HWPs in lieu of alternative products.

3.1 Section summary

- Determination of the C storage in HWPs was based on production information supplied by wood-processing facilities most likely to process the logs from the study sites, and also on default service lives for the different HWPs and on the fate of HWPs post-service (primarily based on the dynamics of the decomposition of HWPs in landfills).
- The product substitution factors applied to the HWPs were determined taking into account key geographical markets, key competitors for HWP and likely replacement scenarios. The emission footprint of the HWPs and replacement products were taken from the literature.
- Fossil fuel displacement factors for bioenergy generation from wood biomass (wood-processing residues) were derived for commercial applications (feedstock for boilers) or domestic applications (firewood).
- Key findings:
 - Pulp logs were a major component of the commercial logs extracted for both silvertop ash and mountain ash forests. The ratio of pulp logs to sawlogs (on a C basis) was 70/30 for silvertop ash, and 64/36 for mountain ash. For blackbutt, the typical volume of sawlogs and poles combined was 53 t C/ ha. The highest commercial recovery was for mountain ash.
 - Key HWPs (other than paper) from silvertop ash included decking, flooring and structural/cladding applications. For blackbutt, there was a greater variety of HWPs produced, with electricity poles, flooring, decking, mining timbers, structural timbers and fencing accounting for the majority of the C. For mountain ash, all low-quality sawlogs were processed into pallets, with the high-quality sawlogs being processed into a similar mix of HWPs to blackbutt.
 - The production of electricity poles in NSW has a dual GHG benefit of very high long-term C storage factors and a high product substitution benefit. Pallets produced in Victoria, on the other hand, have insignificant long-term storage benefits. However, if pallets were disposed of in landfill as opposed to being used as mulch at the end of their service life, the long-term C storage in HWPs for the mountain ash site would increase.
 - Use of wood-processing residues for residential energy generation as firewood has a much higher substitution benefit than its use for commercial applications (boilers), given the high emission-intensity of coal production in electricity

generation, as opposed to the typical displacement of natural gas in commercial applications (natural gas has a lower GHG emission footprint than coal).

- The substitution impact associated with the use of wood-processing residues for domestic heating or energy generation offsite ranged from 1.7 t C/ha for mountain ash to 3.3 t C/ha for blackbutt. The Substitution impact associated with the potential use of harvest slash (30% of total) for electricity co-generation was much higher, ranging from 11 t C/ha for silvertop ash to 18 t C/ha for mountain ash.
- There was comparatively little variation in the emission factors for most hardwood HWPs from the study sites, ranging from 0.13 to 0.23 kg C emitted for every tonne of C in sawn hardwood HWPs.
- The net difference between the emission footprint for HWPs and alternative products is expressed as the product substitution impact. The higher the emission footprint of the alternative products relative to HWPs, the higher the GHG “savings” associated with the use of HWPs, and the higher the substitution factors.
- The high emission factor for imported tropical hardwood HWPs reflects the assumption that a significant proportion of those products are likely to be manufactured from wood originated from deforestation or highly degraded forests in SE Asia.
- The weighted substitution factors ranged from 0.2 t C / t C in HWP for mountain ash to 2.1 t C / t C in HWP for silvertop ash – mostly due to the likely displacement by SE Asia hardwood. On a hectare basis though, product substitution impact for silvertop ash was low compared to other species (due to low site productivity and low sawmill recoveries).
- The substitution impacts ranged from 6.2 t C / ha for silvertop ash to 18.4 t C/ha for blackbutt HWPs.
- Long-term C storage for HWPs in Australia is primarily imparted by the post-service stage of the HWP life (i.e. storage in landfill). However, the overall GHG impact will be positive whether the product is recycled into another long-lived application (e.g. old floorboards used in recycled furniture), landfilled or burnt to produce energy.
- The substitution impact, when based on market analyses of product usage in different applications, represents a real mitigation benefit, in the same way the use of sustainably sourced biomass for bioenergy generation represents real mitigation when it displaces the use of fossil fuels.

3.2. Methods

3.2.1. Long-term C storage in harvested wood products (HWP)

The long-term C storage in HWPs was determined by following these steps:

- Determination of the weight of commercial logs (sawlogs, pulp logs and poles) was done by direct weighing in the field, followed by the determination of dry weight based on the moisture content and basic density of the logs.
- Determination of the recovery of sawn boards at the green and dry mills (where relevant) of the sawmills as required (in direct consultation with the relevant facilities and using information from previous studies).
- Determination of the volume and mass of the main types of green and dry and dressed sawn boards (where relevant) produced in sawmills, and their relative proportion as related to the original weight of sawlogs (in direct consultation with the relevant facilities).
- Key disposal pathways (previous waste studies).
- Application of C storage factors to HWPs disposed of in landfills (Wang *et al* 2011, Ximenes *et al* 2013).

3.2.2. Product substitution factors

The product substitution factors applied to the HWPs produced (except for pulp logs, which will be included in the next Chapter of this report) were determined by following these steps:

- Determination of key geographical markets for products (in consultation with wood processors).
- Determination of key competitors for each product type and likely replacement scenarios in consultation with the relevant wood-processors and expert market analysts. Where there were multiple replacement products the relative market proportions were given.
- Determination of the GHG balance of the HWPs produced from the study sites, expressed as tonne of C / m³ in HWP (based on previous studies, especially Ximenes *et al* 2010 and Tucker *et al* 2009). This was a cradle to gate assessment, including emissions associated with the establishment, management, harvest, log transport and mill-based emissions.
- The emission footprint of the HWPs was expressed as tonne C / tonne C in HWP, in order to make the results comparable with the other components of the assessment. For example, one tonne of C in the hardwood HWPs from the north Coast of NSW equals approximately 2.8 m³ of hardwood HWPs.
- Energy use of processing residues (residential and commercial), other than energy generated for internal consumption at the mill. More details on this are given below.

- Determination of the GHG balance of the likely replacement products as alternatives to the HWPs from the study sites (expressed as tonne of C / m³ of product). This was a cradle to gate assessment based on an extensive search of the literature (peer-reviewed journals, direct LCA data, environmental product declaration data and industry reports). When determining the GHG balance of the replacement products, a functional equivalence principle was applied. As an example for the sub-floor option:
 1. We determined the volume (and C) in hardwood bearers and joists used in a 100 m² house.
 2. If the replacement product (e.g. LVL) was wood-based, then we also determined the volume (and C) in the LVL used in a 100 m² house as sub-flooring.
 3. If the replacement product was not wood-based (e.g. concrete), then we determined the volume of concrete used as a slab for a 100 m² house.
- Determination of the weighted GHG balance of the likely replacement products based on their likely respective gains in the share of the market if harvesting in native forests was stopped in Australia (expressed as tonne of C / m³ of product).
- Determination of the substitution factor, as the difference between the weighted GHG balance of the alternative materials to the HWPs from the sites and the emission footprint of the HWPs from each site.

In our assessment, the derivation of emission footprints for some HWPs and for energy was more complex, with a number of steps required before reaching a final emission footprint. Below we include a more detailed description of how those factors were derived.

3.2.3. Energy

Fossil fuel displacement factors for bioenergy generation from wood biomass (wood-processing residues) were derived for residential and commercial energy generation (other than biomass used to generate energy for internal mill usage). Residential energy generation here refers to the use of residues for home heating (firewood), whereas commercial energy generation involves use of the sale of residues for use offsite in boilers. The assessment took into account the likely type of fossil fuel that would be displaced in each case. Information on the proportion of residues used for energy generation was obtained directly from sawmills in NSW and Victoria. This was combined with estimates of residues beyond the mill, such as those generated during construction and demolition activities.

The method described in the National Greenhouse Accounts (NGA) Factors 2014 was followed to calculate the emissions from using timber residues to produce energy (Comm. Aus. 2014). This included the emissions from the combustion of wood plus the emissions from the vehicles used to transport the wood (applied only to firewood as the use of residues for commercial applications typically involves short transport distances). Under the 2006 IPCC Guidelines, the emission factor for CO₂ released from the combustion of biogenic C fuels is zero (Comm. Aus. 2014). Therefore only nitrous oxide (N₂O) and methane (CH₄) emissions that arise from the combustion of wood were accounted for. All emissions factors were converted a per tonne of C in wood basis, assuming a moisture content of 15 % for firewood and 30 % for boiler timber, and C content for dry wood of 50 %. The parameters required for the derivation of emission factors for determining the energy output from residues (on a GJ basis) are presented in Table 3.1. In Table 3.2 we present the GHG

emission factors that were derived for the use of residues and for the alternative use of fossil fuels. The factors were expressed on a tonne of C basis; this means that for residues, the emission factors in Table 3.2 reflect the GHG emissions associated with the use of the equivalent of one tonne of C in residues. For fossil fuels, the emission factors presented are for the use of the equivalent amount of fossil fuel required to generate the same output of energy as that achieved by the use of one tonne of C in residues. This was required to ensure consistency with the way the results for other parts of the analysis were expressed.

Table 3.1 Parameters used for the calculation of emission factors for residues used for residential heating (firewood) or commercial heating (boilers).

Residue use	Emission factor kg CO ₂ -e / GJ ¹		Energy content GJ / t ²	Efficiency factor % ³
	CH ₄	N ₂ O		
Firewood (Residential)	17.85	0.57	16.2	62
Boilers (Commercial)	0.08	1.22	10.4	70

¹ The emission factors for firewood were derived from the Australian Methodology for the Estimation of Greenhouse Gas Emissions and Sinks 2006 – Energy (Stationary sources), (Commonwealth of Australia 2007). The NGA factors (Comm. Aus. 2014) were used for commercial residues. ² The energy content figures are as per the NGA factors (Comm. Aus. 2014). ³ The residential efficiency factor was derived from Paul *et al* 2003; the commercial boiler efficiency factor was derived from the FAO (FAO 1990).

Table 3.2 Emission factors (expressed as CO₂ emitted per tonne of C in the biomass) for residues and the fossil fuel types displaced by their use

Energy market	Emissions (kg CO ₂ -e/t C in biomass used)		Emissions (kg CO ₂ -e/ t C in biomass ³)			
	Residues		Fossil Fuels			
	Firewood ¹	Commercial residues	Electricity	Natural gas	Heating oil	Weighted emissions ²
Residential	715	-	6435	1200	-	3818
Commercial	-	38.83	-	1067	1435	1141

¹ The emissions for firewood include 13.1 kg CO₂-e/t C for transport based on 0.07 L/km/tonne firewood and a 30 km round trip. ² The weighted emissions for the residential energy market were calculated based on a 50/ 50 split between electricity and natural gas. For commercial boilers this was 80 % for natural gas and 20 % for heating oil. ³ This factor reflects the emissions associated with the use of the amount of fossil fuel required to produce the same amount of energy that one tonne of C in residues would produce

3.2.4. HWP from South East Asian (SEA) tropical forests

Analysis of the likely displacement scenarios for native forest hardwood from the study regions revealed a high likelihood of replacement with SE Asia timber for some applications. There are serious concerns over the rate of deforestation, C emissions and loss of biodiversity in Southeast Asia (SEA), where tropical forests comprise 60% of the forested area (FAO 2001). While in 1990 SEA had an estimated 268 Mha of forest cover, by 2010 this had dropped to 236 Mha (Stibig *et al* 2014), with an estimated net annual deforestation rate (2000-2010) ranging from 1.0- 1.45 Mha (Stibig *et al* 2014, FAO 2010).

In deriving an emissions factor for the timber extracted from native forests in SEA for sawnwood production, we considered that a high proportion of the timber extracted originated from deforestation activities, and also from degraded areas, with a smaller proportion extracted sustainably. Based on that we applied three harvest intensities; (i) low intensity harvest, (ii) high intensity harvest and (iii) deforestation (Appendix 1). We calculated a net weighted emissions factor for each harvest intensity based on the industrial sawlog production volumes for each country for 2013 (Appendix 1 -Table 3.3). We then weighted the three harvest intensity emissions factors for SEA to derive a single factor (Appendix 1 -Table 3.3).

For the purposes of this report we defined low intensity harvest as that which is done with minimal disturbance and with extraction rates and harvest rotations at a sustainable level. On that basis the C extracted is equal to the C sequestered, and the emission sources are only those derived from the fossil fuels used in the machinery for harvest, transport and processing. For high intensity harvest, in addition to the fossil fuel emissions C is lost through poor harvest techniques and extraction levels greater than those deemed as best practice for the region. In order to derive the deforestation emission factor we assumed that the area has been clear felled, with no regeneration and all the above ground C has been lost.

A net emissions factor for each harvest intensity was derived making provisions for the C that is sequestered following high intensity harvest and due to land use change following deforestation. The three harvest intensity emissions factors were calculated for each SE Asian country and a weighted average based on the industrial sawlog production for each country was applied to obtain a factor for SE Asia (Appendix 1).

The proportion of industrial sawlog that is produced as a result of each harvest intensity is difficult to determine, as data pertaining to the origin of timber (i.e. plantation timber, selectively harvested timber or through legal land conversion) is not officially recorded. We therefore relied on published data from a number of sources, and based on this the harvest intensities were weighted as follows; 10% of production from low intensity harvest, 45% from high intensity and 45% from deforestation (Appendix 1).

A summary of key emission factors used is given in Table 3.3 below; a more detailed description of the method and a discussion on the complexity and uncertainty surrounding the factor can be found in Appendix 1.

Table 3.3 Weighted net emission factors for each harvest intensity and the total for SEA including an emission factor for shipping.

	Production m ³ ¹	Net emissions ² (kg CO ₂ -e/t C in wood)				
		Low intensity harvest	High intensity harvest	Deforestation	Shipping emissions ³	Total emissions
Sawlogs & veneer logs	38,642,629	62	4,338	10,657	243	15,301

¹ The production volumes are estimated based on FAO sawlog & veneer log figures from FAOSTAT for 2013 and FAO plantation data for 2012 (FAO 2015, Jurgensen 2014). ² The net emissions allow for sequestration through regeneration in high intensity harvest and land use change following deforestation. ³ Based on factors contained in Ximenes and Brooks (2010)

3.2.5 Preservative-treated wood

The HWPs that were preservative-treated were grouped under 4 categories and classified under various hazard classes relative to the exposure of the product use:

- Framing - H2 and H2F – Softwood; Decking - H3 – Softwood; Fencing - H3 – Softwood; Poles - H5 – Hardwood

A summary of the treatment levels and hazard classes is shown in Appendix 2. A description of the steps used in the calculation of the emission factors for each preservative type is also included in Appendix 2. The emission factors for the key preservatives used are presented in Table 3.4, whereas the emission factors used for each treated HWP is included in Table 3.5.

Table 3.4 Emission factors (expressed on a kg of total active ingredient basis) for key wood-preservative formulations

Preservative	Emission factor (kg CO ₂ -e / kg of total active ingredient)	Reference
CCA	4.7	McCallum (2010)
ACQ	19.1	Bolin and Smith (2010)
Boron	4.5	McCallum (2010)
LOSP	90.4	McCallum (2010)
Creosote	2.2 ¹	Bolin and Smith (2013)

¹ This refers to the use of 1 kg of the product “creosote”.

Table 3.5 Emission factors for the HWPs included in the analysis that were preservative-treated

HWP	Kg C/ tonne of C (due to treatment only) ¹
Treated pine, framing (H2, H2F) ²	7.4
Treated pine, decking (H3)	26.3
Treated pine, fencing (H3)	15.1
Treated hardwood poles (H5, CCA)	17.5
Treated hardwood poles (H5, creosote)	117.9

¹ does not include emissions associated with harvest, transport and manufacture – these are included in the calculations of the product displacement factors. ² mean of CCA, ACQ, and boron

3.2.6. Electricity poles

Emission factors were derived for electricity transmission poles made using hardwood, concrete and steel. The poles were all assumed to be 14 m in length. The key factors associated with the different poles are listed in Table 3.6 below:

Table 3.6 Weight and emission factors for hardwood, steel and concrete power poles

Pole type	Assumed weight (kg)	Emission factor (kg CO₂-e / kg of material)
Hardwood	1091	0.52 (this study)
Steel	651 (Sureline 2011)	2.62 (Bolin and Smith 2011)
Concrete	2645 (Rocla 2007)	0.15 (Bolin and Smith 2011)

3.3. Results

3.3.1 Commercial Logs

The results below (expressed in tonnes of C in logs per hectare) are presented both as actual site production (SI) and regionally typical production (RG), as defined below:

a) Actual site production (SI)

Sawlogs, pulp log and/or pole logs from all commercial trees in the study sites were included. This scenario reflected the actual flow of C from the study sites and the fact that every tree with DBH greater than 10 cm was harvested and weighed.

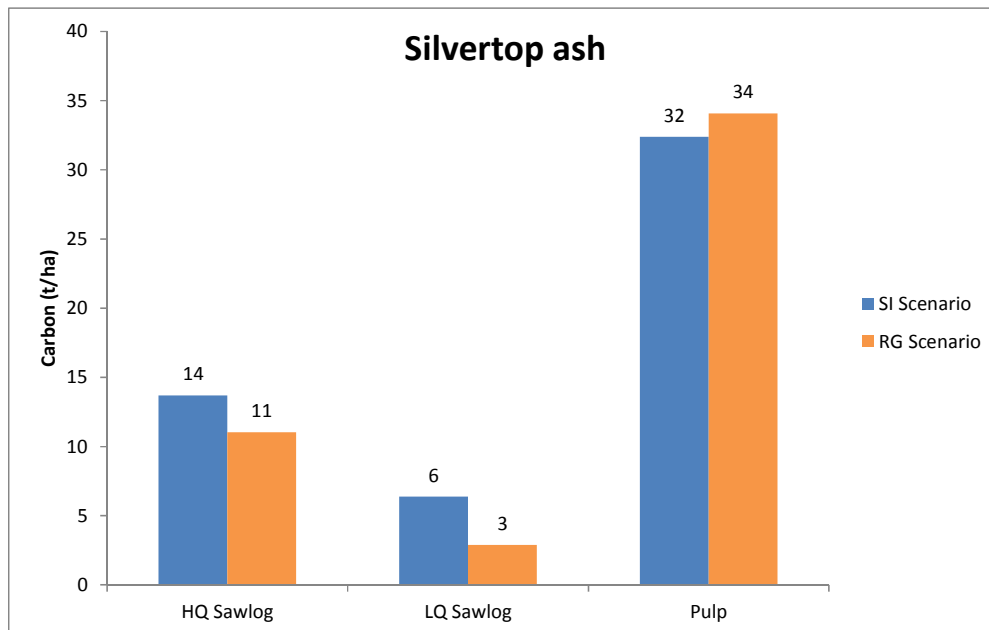
b) Regionally typical production (RG)

In order to mirror harvest prescriptions applied in harvest operations in each relevant region, we adjusted the figures to exclude retained trees. The product mix was also adjusted to reflect the typical mix of commercial logs for each region, in consultation with the FCNSW and VicForests.

Silvertop Ash- NSW South Coast

Silvertop ash accounted for the vast majority of the C in production logs harvested in Yambulla State Forest (99%). As expected pulp logs accounted for the majority of the C in the silvertop ash production logs, for both the SI and RG scenarios (Figure 3.1).

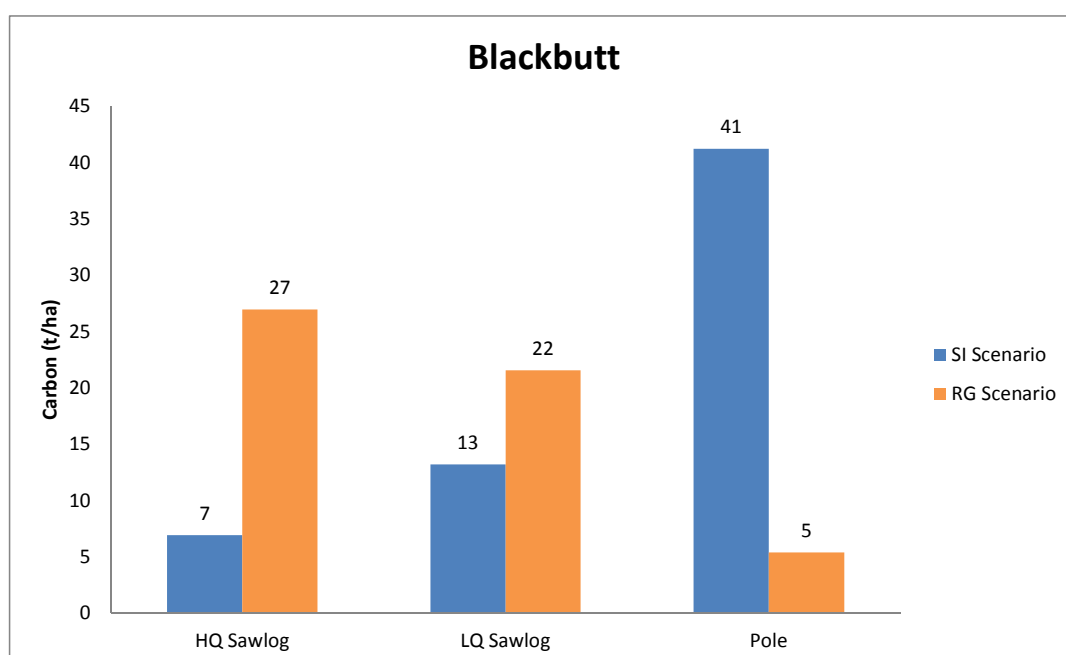
Figure 3.1. Silvertop ash production logs from Yambulla State Forest derived from actual site data and adjusted regional average production



Blackbutt- NSW North Coast

Blackbutt accounted for the vast majority of the C in production logs harvested in Mt Boss State Forest (96%). The majority of the production logs in the SI scenario were destined for use as electricity poles (Figure 3.2). This changed dramatically for the RG scenario, with a substantial shift towards high-quality (HQ) and low-quality (LQ) sawlogs (Figure 3.2).

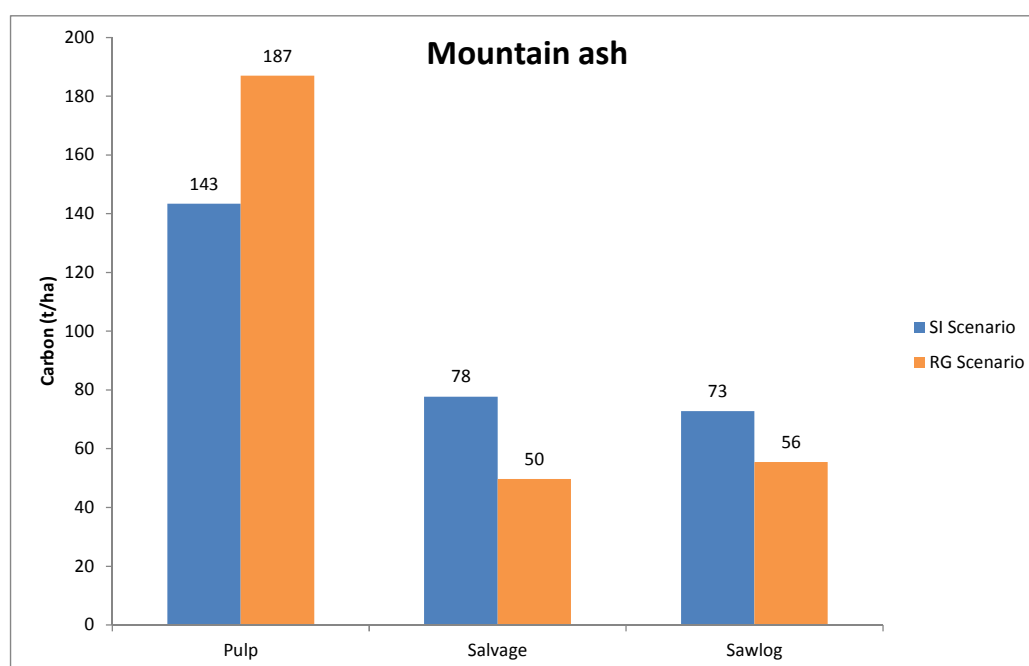
Figure 3.2. Blackbutt production logs from Mt Boss State Forest derived from actual site data and adjusted regional average production



Mountain Ash- Victoria Central Highlands

Approximately half of the C in the mountain ash production logs was destined for use as pulp, with the remainder equally divided between low (E Grade) and high quality (A-D grade) sawlogs (Figure 3.3). In the RG scenario the volume of pulp logs increased reflecting average volumes for the Central Highlands (Figure 3.3).

Figure 3.3. Mountain ash production logs from Toolangi State Forest – derived from actual site data and adjusted regional average production



The recovery of production logs was expressed as a percentage of the harvested production trees (RG scenario) and as a percentage of the total plot standing (SI scenario), (Table 3.7).

Table 3.7. Recovery of biomass in production logs for the RG and SI scenarios in each study region on a dry weight basis.

Species	Recovery (%); assuming harvest prescriptions (RG)	Recovery (%); actual site production (SI)
Silvertop ash (Yambulla State Forest)	58.8	51.2
Blackbutt (Mt Boss State Forest)	59.3	47.5
Mountain ash (Toolangi State Forest)	82.5	78.1

The recovery varied significantly between the NSW species and mountain ash in Victoria (Table 3.7). Mountain ash had the highest recovery levels, largely as a reflection of the characteristics of the species (especially low levels of branching in the crown), but also due to the presence of an active pulp market. Availability of a pulp market would have significantly increased the recovery for blackbutt (Table 3.7).

A breakup of the commercial logs as derived for the study sites is given in Table 3.8, along with actual harvest data provided by the FCNSW and VicForests. The data from the FCNSW for the mid-north coast covered a broad geographical area and suggests that the study “production” site yielded a slightly higher proportion of high quality logs than the average blackbutt forest in that region. Logs are graded as poles or HQ sawlogs primarily depending on market demand at the time of harvest. Comparable data for Eden was limited due to a large proportion of the harvest in the region being derived from thinning operations, which yields a larger proportion of pulp (~90%). While the FCNSW data and the Eden production site are comparable (Table 3.8), the analysis is limited due to the restricted number of comparative sites.

The mountain ash Toolangi SF site had a higher proportion of sawlogs than the average for similar Central Highlands forests provided by VicForests (Table 3.8). The Toolangi SF study site was thinned in the past, with the removal of poorer quality stems and trees that would have been dead standing, resulting in increased sawlog recoveries (Table 3.8). Distances to pulpwood markets also play an important role in the pulp yields for different coupes. The increased haulage costs for coupes further from markets often means reduced commercial utilisation of non-sawlog components.

Table 3.8. Percentage breakup of the commercial component of the harvest from the study sites compared with actual harvest data provided by the state agencies for similar sites (based on green volumes, excludes dead trees).

Commercial Product	Silvertop ash		Blackbutt		Mountain ash	
	FCNSW actual (%)	Yambulla SF site production (%)	FCNSW actual (%)	Mt Boss SF site production (%)	Vic Forests (%)	Toolangi SF site (%)
LQ Sawlogs	5	12	36	22	14	26
HQ Sawlogs	23	27	50	11	23	25
Poles	-	-	10	67	-	-
Pulp	71	61	-	-	63	49
Residue	1	-	4	-	-	-

3.3.2. Production of HWPs

In order to ensure the results were compared on the same basis, the HWPs were expressed as tonnes of C of HWPs produced per hectare (finished HWPs leaving the “sawmill gate”). As for section 3.2.1, these are presented as actual site production (SI) and regionally typical production (RG).

For silvertop ash, a large proportion of the C in production logs was destined for pulp and paper use (Figure 3.4), with a comparatively small proportion of the C in sawn hardwood products. Under the RG scenario, there was a small increase in the pulp produced and a small decrease in the sawn hardwood produced, with only 4.5 t C/ha excluded from the harvest (Figure 3.4).

Poles accounted for the bulk of the blackbutt HWP's under the SI scenario (Figure 3.5). As described in section 3.2.1, the decision on whether the logs are graded as “poles” or “high-quality sawlogs” depends primarily on the market demands. Typically though for this study region there is a more even distribution between poles and high-quality sawlogs than in the SI scenario (RG scenario, Figure 3.5). The impact of applying typical harvest prescriptions (RG Scenario) is significant, with 7.4 tC / ha excluded from the harvest. More importantly under the RG scenario there is a large drop in the C in poles, and a comparatively smaller increase in the C in the other products (Figure 3.5). The main reason why the magnitude of the loss of C storage in the poles was not mirrored by increases in the other products is due to high recovery rates in the production of electricity poles, with very little residue produced.

For mountain ash, all low quality sawlogs are processed into pallets, resulting in pallets being overall the main sawn product obtained from the site (Figure 3.6). Processing of the higher quality sawlogs generates a range of both sawn hardwood products and engineered wood products (EWP) produced using offcuts (Figure 3.6). Pulp accounted for a large proportion of the overall product mix (Figure 3.6). The key impact of the RG scenario was an increase in the volume of pulp and as a result a reduction in the volumes of sawn products (Figure 3.6).

Figure 3.4 Silvertop ash HWP's from Yambulla State Forest derived from actual site data and adjusted regional average production

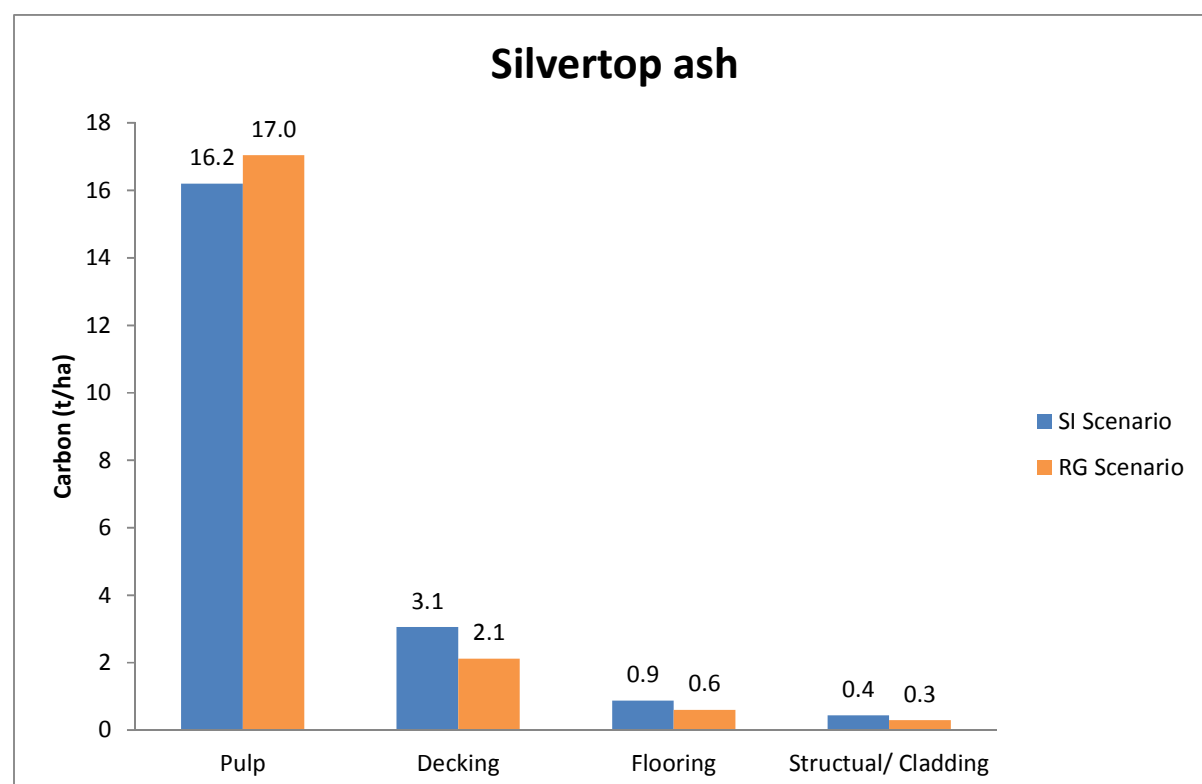


Figure 3.5 Blackbutt HWP from Mt Boss State Forest – derived from actual site data and adjusted regional average production

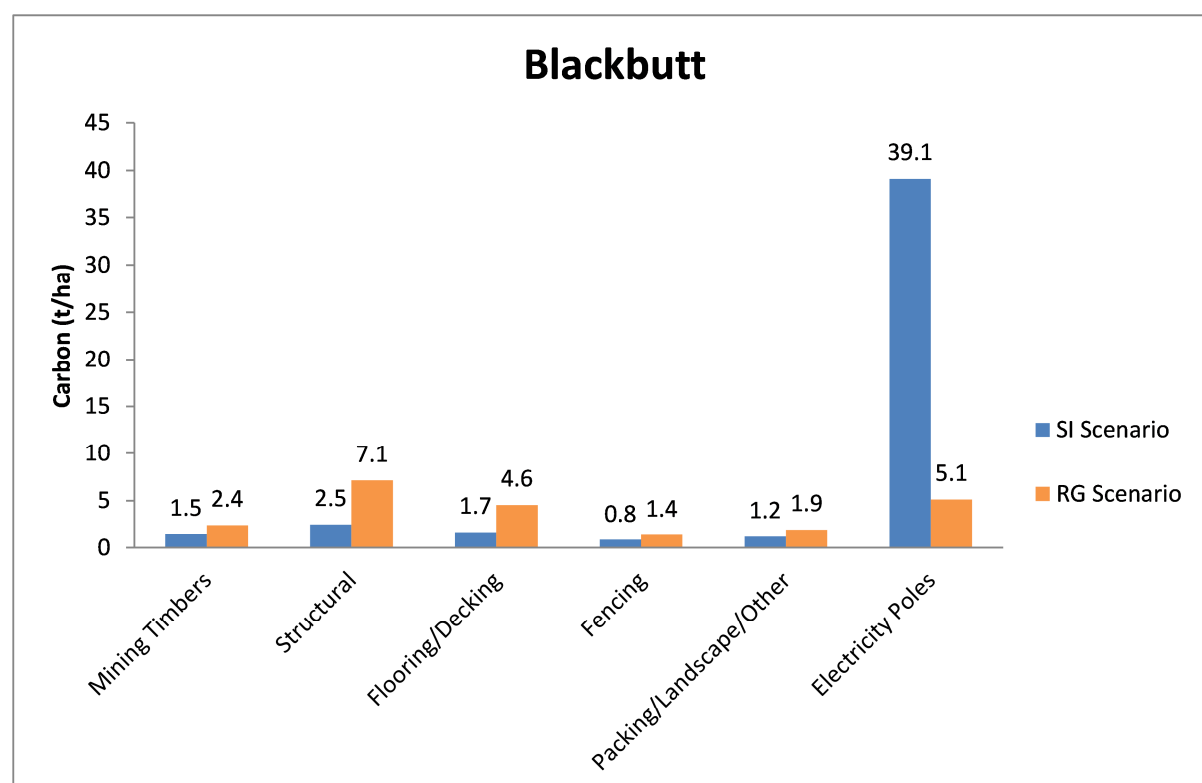
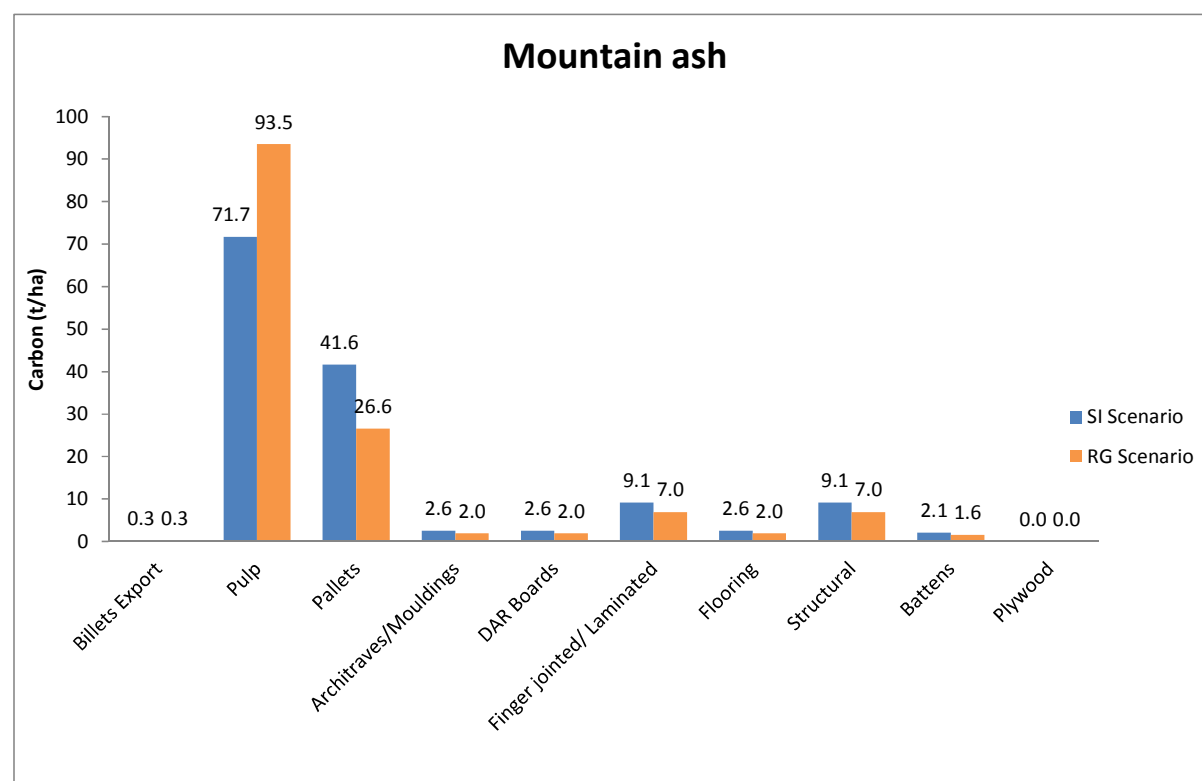


Figure 3.6 Mountain ash HWP from Toolangi State Forest – derived from actual site data and adjusted regional average production



3.3.3. Long-term C storage (LTCS) in HWPs.

Long-term C storage (LTCS) in HWPs (other than paper products) under both the SI and RG scenarios is shown in Table 3.7. Calculation of the LTCS for the various HWPs is based on the proportion of HWPs disposed of in landfills at the end of their service life. The C in HWPs (other than paper products) is stored indefinitely in landfills (e.g. Wang *et al* 2011, Ximenes *et al* 2013, Ximenes *et al* 2015).

There was a large variation in the total LTCS for the different species under the SI scenario, with blackbutt HWPs having the highest LTCS (43 t C /ha). For silvertop ash, decking accounted for the majority of the LTCS, whereas for blackbutt, electricity poles were by far the biggest contributor (Table 3.7). For mountain ash, both structural / cladding materials, and finger-jointed / laminated products were the greatest contributors to LTCS (Table 3.7).

The RG scenario had a significant impact especially on blackbutt HWPs (Table 3.7). That is primarily due to the shift from poles to sawlogs; the overall sawn product yields are much lower than using very long parts of the stem with minimal processing as electricity poles, which mainly are disposed of in landfills at the end of their functional lives. As a result, after adjusting the product mix to reflect the regional averages, blackbutt structural / cladding HWPs have higher LTCS than electricity poles (Table 3.7). Under the RG scenario the total LTCS for silvertop ash and mountain ash decreased by 29 and 24%, respectively (Tables 3.7).

Table 3.7 Long-term C storage in HWPs produced from sawlogs and poles for actual site production (SI) and regionally typical production (RG) scenarios.

HWP	LTCS (Landfill, t C / ha)					
	Silvertop ash		Blackbutt		Mountain ash	
	SI	RG	SI	RG	SI	RG
Decking	2.5	1.7	0.7	1.9	-	-
Flooring	0.7	0.5	0.7	1.9	2.2	1.6
Structural/ Cladding	0.4	0.3	2.1	6.0	7.7	5.9
Mining timbers	-	-	1.4	2.3	-	-
Fencing	-	-	0.6	0.9	-	-
Packing/Landscape/Other	-	-	0.2	0.3	-	-
Electricity poles	-	-	37.2	4.9	-	-
Architraves/Mouldings	-	-	-	-	2.3	1.8
Dressed all round (DAR) boards	-	-	-	-	2.2	1.7
Finger jointed/ Laminated	-	-	-	-	8.2	6.3
Battens	-	-	-	-	1.8	1.4
Plywood	-	-	-	-	0.0	0.0
Residues	0.2	0.2	0.2	0.6	1.2	0.9
Total	3.8	2.7	43.0	18.7	25.7	19.6

3.3.4 Market Information

In Tables 3.8 - 3.11 we present the market share for the key product categories for the different regions in this study, following consultation with wood market experts and industry. Key sources consulted for the development of this data included the URS Timber Market Survey (2014) and the Bis-Shrapnel report (2014) “Sawn Timber in Australia (2014-2028). It is clear from the analysis that there is plenty of scope for increasing the market share for the majority of hardwood HWP (Tables 3.8- 3.11). In addition, the proportion of the market that is occupied by products other than hardwood products provides a good indication of the likely replacement alternatives in the event native hardwood HWPs were no longer available.

Table 3.8 Current market share in Victoria, NSW south coast and NSW north coast for decking and floor coverings

Products	Potential substitution products	% Share of domestic market		
		Victoria	NSW South Coast	NSW North Coast
Decking	Treated pine	35%	35%	35%
	Merbau	25%	30%	25%
	Balau	5%	5%	10%
	Cypress pine	5%	0%	0%
	Concrete	5%	5%	5%
	Composite (PVC)	5%	0%	5%
	<i>Domestic hardwood</i>	20%	25%	20%
Floor Coverings	Carpet	39%	35%	37%
	Tiles	27%	25%	24%
	Engineered - bamboo	7%	6%	7%
	Engineered flooring (imported)	5%	5%	7%
	Imported solid timber floor	3%	4%	4%
	Cypress	3%	5%	3%
	Laminate (melamine infused paper on top of wood chip composite)	1%	0%	2%
	Engineered flooring - ply core with hardwood top (0.6-6mm) (Australian Hardwood)	1%	2%	1%
	Vinyl	1%	1%	1%
	<i>Domestic hardwood</i>	13%	17%	14%

Table 3.9 Current market share in Victoria, NSW south coast and NSW north coast for cladding, architraves and mouldings, finger jointed/laminated products and lining

Products	Potential substitution products	% Share of domestic market		
		Victoria	NSW South Coast	NSW North Coast
Cladding	Fibre cement	20%	20%	20%
	Hardboard	15%	15%	15%
	Plywood	15%	15%	15%
	Radiata pine	15%	15%	15%
	Steel	10%	10%	10%
	Aluminium	10%	10%	10%
	Imported timber	5%	5%	5%
	Cypress	5%	5%	5%
	<i>Domestic hardwood</i>	5%	5%	5%
Architraves/Mouldings	MDF	87%	55%	30%
	Clear pine	4%	10%	18%
	FJ Radiata pine	2%	10%	30%
	FJ Hoop pine	1%	0%	3%
	Meranti	0%	10%	12%
	Western red cedar	0%	10%	6%
	<i>Domestic hardwood</i>	6%	5%	1%
Finger jointed/ Laminated	Imported	50%	90%	90%
	<i>Domestic hardwood</i>	50%	10%	10%
Lining	Western red cedar	60%	60%	60%
	<i>Domestic hardwood</i>	40%	40%	40%

Table 3.10 Current market share in Victoria, NSW south coast and NSW north coast for structural products

Products	Potential substitution products	% Share of domestic market		
		Victoria	NSW South Coast	NSW North Coast
Structural-Framing (wall)	Pine	87%	88%	90%
	Steel	4%	3%	5%
	Concrete/Cement	1%	0%	1%
	Cypress	1%	1%	1%
	<i>Domestic hardwood</i>	7%	8%	3%
Structural-Framing (roof)	Pine	80%	75%	78%
	LVL/ I-beams	8%	9%	9%
	Steel	4%	5%	5%
	<i>Domestic hardwood</i>	8%	11%	8%
Battens	Pine	40%	80%	90%
	<i>Domestic hardwood</i>	60%	20%	10%
Sub-Floor Products	Pine	16%	43%	44%
	LVL/ I-beams	13%	28%	25%
	Parallel chord truss	16%	0%	2%
	Concrete (suspended sections)	13%	15%	12%
	Steel	2%	2%	2%
	<i>Domestic hardwood</i>	40%	12%	15%

The likely resulting market shares for products other than native hardwood products, if they were no longer available, are provided in Table 3.12. The data in Table 3.12 is based on the “domestic hardwood” component of Tables 3.8 - 3.11, taking into account factors such as “consumer loyalty”, i.e. the consumer is a “committed” hardwood user and is unlikely to purchase a non-hardwood product unless none is available in the market. Using the “floor coverings” market as an example (Table 3.8), domestic hardwood accounts for a relative small proportion of the overall market (13-17%). Tiles and carpet have the biggest share of the floor covering market for the regions studied. However, it would be reasonable to assume that a majority of those consumers who have a hardwood floor would be unlikely to switch to carpet or tiles should the native hardwood floor no longer be available. This is reflected in the figures in Table 3.12, where either imported hardwood floors or engineered wood products are listed as the most likely products to have an increased market share. Another important consideration is that the substitution options do not necessarily mean that the alternative products will perform to the same extent as the native hardwood product - compromises in performance may be associated with some of the likely replacement options.

Table 3.11 Current market share in Victoria, NSW south coast and NSW north coast for fencing, plywood, electricity poles, mining timbers, packing and pallets

Products	Potential substitution products	% Share of domestic market		
		Victoria	NSW South Coast	NSW North Coast
Fencing	Treated pine	75%	65%	75%
	Steel	20%	10%	10%
	<i>Domestic hardwood</i>	5%	25%	15%
Plywood for formwork and bracing	Tropical hardwoods	40%	20%	20%
	Pine	40%	40%	30%
	<i>Domestic hardwood</i>	20%	40%	50%
Electricity poles	concrete	28%	7%	7%
	Steel	2%	3%	3%
	<i>Domestic hardwood</i>	70%	90%	90%
Mining timbers	Other domestic timber	60%	30%	30%
	Concrete	40%	20%	20%
	<i>Domestic hardwood</i>	0%	50%	50%
Packing/Landscape/Other	Treated pine	50%	50%	50%
	Pine	30%	30%	30%
	<i>Domestic hardwood</i>	20%	20%	20%
Pallets	Pine	15%	15%	20%
	Plastic	3%	2%	4%
	<i>Domestic hardwood</i>	82%	83%	76%

Table 3.12. Likely substitution products for HWPs from the study sites

Products	Most likely substitution products	Likely market share
Decking	Merbau	50%
	Treated pine	40%
	Balau	10%
Floor coverings	Engineered flooring (imported)	50%
	Imported solid timber floor	30%
	Engineered - bamboo	20%
Cladding	Hardboard	40%
	Fibre cement	40%
	Radiata pine	20%
Architraves/Mouldings	FJ Radiata pine	40%
	MDF	40%
	Clear pine	20%
Finger jointed/ Laminated	Imported glulam	100%
Lining	Western red cedar	100%
Structural- Framing (wall)	Pine	80%
	Steel	15%
	Cypress	5%
Structural- Framing (roof)	LVL/ I-beams	60%
	Pine	20%
	Steel	20%
Battens	Pine	100%
Sub-floor products	LVL/ I-beams	40%
	Pine	40%
	Concrete (suspended sections)	20%
Fencing	Treated pine	60%
	Steel	40%
Plywood for formwork and bracing	Tropical hardwoods	50%
	Pine	50%
Electricity poles	Concrete	50%
	Steel	50%
Mining timbers	Concrete	100%
Packing/Landscape/Other	Treated pine	60%
	Pine	40%
Pallets	Pine	100%
	Plastic	0%

3.3.5 Substitution impacts

In Table 3.13 we report on the emission factors associated with the extraction, transport and manufacture of the native hardwood HWPs from each region (“cradle to gate” approach). The emission factors in Table 3.13 do not include fossil-fuel displacement factors for renewable energy produced outside the mills from wood-processing residues for commercial or residential applications – these factors are detailed separately in Table 3.16. We do not provide substitution factors for paper products in this section – these will be discussed in detail in Chapter 4 of this report.

There is comparatively little variation in the emission factors for most hardwood HWPs included in Table 3.13, ranging from 133 to 226 kg C / tonne of C for sawn hardwood HWPs. The emission factor for plywood is higher because the emission intensity of production for EWP is typically higher than for sawn products (Table 3.13).

Table 3.13 Emission footprints (kg C emitted per one tonne of C in the products) for the hardwood HWPs produced from the study sites

Hardwood HWPs	kg C / tonne C	References
Finger jointed / Laminated (GLULAM)	133.1	Tucker et al 2009, Ximenes <i>et al</i> 2010; this study
Mining timbers	138.5	Ximenes <i>et al</i> 2010; this study
Fencing	138.5	Ximenes <i>et al</i> 2010; this study
Packing/Landscape/Other	138.5	Ximenes <i>et al</i> 2010; this study
Electricity poles	149.9	Ximenes <i>et al</i> 2010; this study; Bolin and Smith 2011
Decking	156.0	Ximenes <i>et al</i> 2010; this study
Flooring	156.0	Ximenes <i>et al</i> 2010; this study
Cladding	156.0	Ximenes <i>et al</i> 2010; this study
Pallets	162.7	Ximenes <i>et al</i> 2010; this study
Structural - Sub floor	182.1	Ximenes <i>et al</i> 2010; this study
Structural - Wall	182.1	Ximenes <i>et al</i> 2010; this study
Structural - Roof	182.1	Ximenes <i>et al</i> 2010; this study
Architraves / Mouldings	207.4	Ximenes <i>et al</i> 2010; this study
DAR boards	207.4	Ximenes <i>et al</i> 2010; this study
Battens	225.8	Ximenes <i>et al</i> 2010; this study
Plywood	316.3	Tucker et al 2009; this study

In Table 3.14 we report on the emission factors for the various alternative wood and non-wood products identified as the most likely replacement products for the HWPs from the study sites. The factors vary significantly, both for HWPs and non-timber products (Table 3.14). The high emission factor for imported tropical hardwood HWPs reflects the assumption that a significant proportion of those products are likely to originate from deforested areas or areas where forestry practice is unsustainable in SE Asia (for more details please see section 3.1.4 in the Methods section and Appendix 1). The emission factors for EWPs are weighted averages accounting for the current proportion of the market occupied by domestic and imported EWPs (Table 3.14). A number of non-timber products have emission factors in excess of one tonne of C per equivalent tonne of C (Table 3.14). Emission factors

for products that use the same feedstock (e.g. concrete) vary depending on the function provided (e.g. for concrete: electricity pole, concrete slab, mining) and the quantity of C in the equivalent hardwood HWP.

Table 3.14 Emission footprints (kg C emitted per one tonne of C in the products) for the alternative wood and non-wood products identified as the most likely replacement products for the HWPs from the study sites

HWPs		References	Non-timber products		References
Product	kg C / tonne C		Product	kg C / tonne C	
Untreated pine	183.8	Ximenes <i>et al</i> 2010	LVL/ I Beam	220.9	Wilson and Dancer (2005), Tucker <i>et al</i> 2009; Puettman <i>et al</i> 2013a
FJ pine	209.5	Tucker <i>et al</i> 2009; Puettman <i>et al</i> 2013	Steel (wall frames)	277.1	Ximenes and Grant (2013)
Cypress pine	286.7	Ximenes <i>et al</i> 2010	Engineered - Bamboo	883.8	Vogtländer and van der Lugt (2014)
Pine (H2F treatment); Radiata pine (H3)	288.1	Ximenes <i>et al</i> 2010; McCallum 2010; Bolin and Smith 2010	Concrete (mining shafts)	922.9	Bolin and Smith (2011)
Plywood-pine	316.3	Tucker <i>et al</i> 2009	Steel - colorbond	1045.9	OECD (2001)
GLULAM imported	322.5	Puettman <i>et al</i> 2013	Concrete poles	1102.5	Bolin and Smith (2011)
Plywood-Hardwood from SE Asia	480.4	ITTO (2014)	Steel poles	1358.3	Bolin and Smith (2011)
Hardboard	527.1	Bergman (2005)	Concrete (floor slab)	1548.9	Bolin and Smith (2011)
Engineered flooring	1055.4	Bergman and Bowe (2011)	Fibre Cement	3568.1	CertainTeed (2011)
Imported solid timber (50% SE Asia, 50% Northern Hemisphere)	2227.6	References listed in Appendix 1.			
Imported solid timber from SE Asia	4173.0	References listed in Appendix 1.			

In Table 3.15 we report on the emission footprint associated with the use of alternative energy sources to hardwood wood-processing residues. Those factors reflect the emission intensities associated with the different fossil fuels considered. Substitution of residential electricity is by far the option which generates the greatest substitution benefit, given the high emission-intensity of coal production (Table 3.15).

Table 3.15 Emission footprints (kg C emitted per one tonne of C in the residues) for the energy generation alternatives identified as the most likely replacement options for bioenergy from the hardwood wood-processing residues.

Energy		References
Product	kg C / tonne C	
Natural gas (commercial)	291.0	FAO (1990), NGA (2014)
Natural gas (residential)	327.4	NGA (2014), Paul <i>et al</i> (2003)
Heating oil (commercial)	391.4	FAO (1990), NGA (2014)
Electricity (residential)	1755.0	NGA (2014), Paul <i>et al</i> (2003)

In Table 3.16 we combine the factors presented in Tables 3.13 - 3.15 to derive weighted displacement factors (DF) for each HWP, under the SI and RG scenarios (Table 3.16). The higher the DF, the higher the emissions associated with the alternative product. The weighted DF for HWPs for each region (“Total HWP” cell) ranged from 0.18 t C / t C in HWP for mountain ash to 2.07 for silvertop ash. The RG scenario did not have a significant impact on either the HWP or the fossil fuel DF for silvertop ash or mountain ash, and resulted in slightly lower factors for blackbutt (Tables 3.16).

Table 3.16 HWP and fossil fuel (energy) displacement factors (DF) (tonnes of C “saved” per tonne of C) for each study site for the actual site production (SI) and regionally typical production (RG) scenarios.

HWP	DF (tonne C / tonne C in HWP)					
	Silvertop ash		Blackbutt		Mountain ash	
	SI	RG	SI	RG	SI	RG
Decking	1.74	1.74	0.05	0.27	0.04	0.05
Flooring	0.24	0.24	0.02	0.13	-	
Cladding	0.08	0.08	-		-	
Structural-Sub floor	0.01	0.01	0.01	0.05	0.02	0.02
Structural- Wall	0.00	0.0	0.00	0.02	0.00	0.00
Structural-Roof	0.00	0.0	0.00	0.01	0.00	0.00
Mining timbers	-	-	0.03	0.09	-	-
Fencing	-	-	0.01	0.02	-	-
Packing/Landscape /Other	-	-	0.00	0.00	-	-
Electricity poles	-	-	0.93	0.27	-	-
Architraves/Mouldings	-	-	-	-	0.00	0.00
DAR boards	-	-	-	-	0.07	0.08
Finger jointed/ Laminated (GLULAM)	-	-	-	-	0.02	0.03
Battens	-	-	-	-	0.00	0.00
Plywood	-	-	-	-	0.00	0.00
Pallets	-	-	-	-	0.01	0.01
Total HWP	2.07	2.07	1.05	0.87	0.18	0.19
Residential energy	0.34	0.29	0.59	0.37	0.85	0.85
Commercial energy	0.18	0.20	0.09	0.17	0.00	0.00
Total energy	0.52	0.49	0.68	0.54	0.85	0.85

The high DF for silvertop ash decking can be explained largely by the significant proportion of native hardwood from SE Asia that was a likely displacement product. Although the un-weighted DF for blackbutt decking was identical to that for silvertop ash decking, its impact on the “total HWP” factor was much less than for silvertop ash. This is due to blackbutt decking accounting for a much lower proportion of the sawn HWP output compared to silvertop ash decking. The comparatively lower DF for mountain ash is largely due to the low emission intensity associated with the main replacement product for hardwood pallets (untreated pine).

When the factors listed in Table 3.16 are combined with the total C in HWPs from each site on a hectare basis, the total contribution of “product substitution” can be quantified (Table 3.17). The higher the value, the greater the benefit associated with the use of the HWP. Under the SI scenario, the total displacement factor for blackbutt was much greater than for the other species, due to the very large impact of electricity poles. For silvertop ash and mountain ash, decking and DAR boards respectively accounted for the greatest substitution impact (Tables 3.17).

Adjusting for harvest prescriptions and for the average product mixes for the wider regions reduced the overall substitution impacts for all regions, most markedly for blackbutt. This was primarily due to the switch of a large proportion of commercial logs from poles to sawlogs, resulting in decking having slightly higher substitution impacts compared to electricity poles (Table 3.17). For silvertop ash and mountain ash, the reductions were primarily due to an increase in the amount of pulpwood under the RG scenario.

Table 3.17 Substitution impacts (t C/ha) associated with the use of HWPs from each study site for the actual site production (SI) and regionally typical (RG) scenarios.

HWP	Substitution impact (t C / ha)					
	Silvertop ash		Blackbutt		Mountain ash	
	SI	RG	SI	RG	SI	RG
Decking	7.6	5.2	2.1	5.7	-	-
Flooring	1.1	0.7	1.0	2.8	3.0	2.3
Structural/ Cladding	0.4	0.3	0.6	1.7	1.5	1.1
Mining timbers	-	-	1.2	1.9	-	-
Fencing	-	-	0.4	0.7	-	-
Packing/Landscape/Other	-	-	0.2	0.3	-	-
Electricity Poles	-	-	42.3	5.5	-	-
Architraves/Mouldings	-	-	-	-	0.3	0.2
DAR Boards	-	-	-	-	5.3	4.0
Finger jointed/ Laminated	-	-	-	-	1.7	1.3
Battens	-	-	-	-	0.2	0.2
Plywood	-	-	-	-	0.0	0.0
Pallets	-	-	-	-	0.9	0.6
Total	9.0	6.2	47.7	18.4	13.0	9.8

3.3.6 Use of residues for bioenergy generation (ex-mill)

In Table 3.18 we report on the use of residues from wood-processing operations for bioenergy generation other than residues burnt for drying the timber in the kilns. The residues may be used for home heating as firewood (e.g. timber offcuts) or commercially in boilers (e.g. in brick manufacture). The amount of C in residues used for energy increases for blackbutt under the RG scenario (Table 3.18), reflecting the higher proportion of sawlogs to electricity poles under that scenario (Tables 3.18).

Table 3.18 Use of wood-processing residues for domestic heating or energy generation offsite for actual site production (SI) and regionally typical production (RG).

Energy use	Energy (t C / ha used for energy)					
	Silvertop ash		Blackbutt		Mountain ash	
	SI	RG	SI	RG	SI	RG
Residential	2.1	1.5	3.1	2.7	2.6	2.0
Commercial	3.2	2.8	1.4	3.4	0.0	0.0
Total	5.3	4.3	4.5	6.1	2.6	2.0

3.3.7 Substitution impact (SUI) from the use of wood-processing residues to generate energy

The SUI associated with the use of wood-processing residues to generate energy is dependent on the fossil fuel that is displaced. In the case of the residues used here, generation of bioenergy for commercial purposes was assumed to result in the displacement of natural gas (80%) and heating oil (20%). In the case of the use of residues for home heating, they were assumed to displace the use of electricity (50%) or natural gas (50%). The weighted DF for residential purposes was significantly higher than that for commercial purposes. Thus, the higher the proportion of residues used for home heating, the higher the SUI (Table 3.19).

Under the RG scenario the SUI for silvertop ash and mountain ash was slightly lower (due to the higher proportion of pulp) and slightly higher for blackbutt (due to the higher proportion of sawlogs compared to poles), (Table 3.19).

Table 3.19 Substitution impact (SUI) associated with the use of wood-processing residues for domestic heating or energy generation offsite for actual site production (SI) and regionally typical production (RG) scenarios.

Energy use	SUI of using wood processing residues for energy (t C / ha)					
	Silvertop ash		Blackbutt		Mountain ash	
	SI	RG	SI	RG	SI	RG
Residential	1.8	1.2	2.7	2.3	2.2	1.7
Commercial	1.0	0.9	0.4	1.0	0.0	0.0
Total	2.8	2.1	3.1	3.3	2.2	1.7

3.3.8 Substitution impact (SUI) from the potential use of harvest slash to generate energy

In Table 3.20 we report on the SUI associated with the potential use of harvest slash to generate energy (electricity co-generation with coal). The potential benefits are greater than those from using wood-processing residues (Table 3.20), because of the significantly higher volumes of forest-based biomass potentially available. The SUI ranged from 12.0 t C/ha for silvertop ash to 19.8 t C / ha for mountain ash (Table 3.20). Under the RG scenario the SUI for each region was reduced (Table 3.20). The figures below do not take into account the potential SUI associated with diverting pulp logs to bioenergy generation - this will be assessed in Chapter 6.

Table 3.20 Substitution impact (SUI) associated with the potential use of harvest slash (30% of total) for electricity co-generation for actual site production (SI) and regionally typical production (RG) scenarios.

	SUI of using harvest slash for energy (t C / ha)					
	Silvertop ash		Blackbutt		Mountain ash	
	SI	RG	SI	RG	SI	RG
C in harvest slash	50.1	46.0	67.6	57.6	82.6	75.0
Harvest slash used for energy	15.0	13.8	20.3	17.3	24.8	22.5
SUI	12.0	11.0	16.2	13.8	19.8	18.0

3.2.8 Summary of key C flows

Table 3.21 combines the information presented above and in previous components of this report to present a summary of key components of the C flows for the species involved in this study. It includes both C stocks and avoided emissions (substitution impacts for products and energy). These factors will be used for modelling the long-term impacts of native forest management, taking into account both harvest and wildfire events (Chapter 6). Product substitution factors for pulp and paper products are included in Chapter 4.

Table 3.21 Key C stocks and flows (t C / ha) for each study region for actual site production (SI) and regionally typical production (RG) scenarios.

C Flow	Key C flows (t C / ha)					
	Silvertop ash		Blackbutt		Mountain ash	
	SI	RG	SI	RG	SI	RG
Conservation forest	238	238	209	209	398	398
Coarse woody debris (conserv. forest)	22	22	17	17	NA ¹	NA ¹
Production forest	103	103	129	129	377	377
Coarse woody debris (prod. forest)	30	30	65	65	18	18
Trees retained due to harvest prescriptions	-	9	-	18	-	10
Long-term storage in HWP	4	3	43	19	26	20
SUI-Production substitution	9	6	48	18	13	10
SUI-Energy (mill residues)	3	2	3	3	2	2
SUI-Energy (30% slash)	12	11	16	14	20	18

Note ¹CWD was not determined for the Mountain ash conservation forest using the same methods as for blackbutt and silvertop ash.

3.4. Discussion

In this section we did not consider temporary C storage in HWPs in service. At any given time there is a quantum of C in HWPs in service, with constant additions and removals. In the long-term assessment of the dynamics of C flows from the study sites included here, both temporary C storage and long-term C storage (i.e. in landfills) need to be considered – this assessment is included in the analysis presented in Chapter 6.

Key replacement markets were often comprised of wood or wood-derived materials, as the native hardwood products often occupy a niche; i.e. consumers who would most likely want a “wood” replacement if they no longer had access to Australian native forest HWPs. Even though the native forest hardwood products in most cases had significantly lower displacement factors compared to the alternative products identified, the fact that the alternative products were often also wood and wood products reduced the impact of the product substitution (with the exception of imported hardwoods from SE Asia and some EWPs). This approach is different to that described in Sathre and O’Connor’s meta-analysis of twenty-one international studies (Sathre and O’Connor 2010), which only considered product substitution that assumed the use of wood versus the use of non-wood materials. Sathre and O’Connor (2010) determined that displacement factors range from a low of 2.3 to a high of 15 tonne C / tonne of C in HWP, with most lying in the range of 1.0 to 3.0 t C / t C in HWP, with an average of 2.1 t C / t C in HWP. In addition to the differences in the scope of the studies (i.e. whether replacement products include other HWPs), differences in the

energy sources used for the manufacture of different products can have a major impact in the comparisons between studies. This is because the energy profile can differ substantially between countries and also between regions in the same country. That is why it is important, where possible, to use regionally relevant factors when considering substitution factors.

Jönsson *et al* (1997) determined the displacement factors for Swedish solid wood flooring, using a functional unit of 1 m² flooring during one year of operation. The values ranged from 0.2 (linoleum) to 0.7 (vinyl flooring) t C / t C in HWP. In our study, the displacement factors ranged from 0.7 t C / t C in HWP (bamboo flooring as the alternative) to 2 t C / t C in HWP for imported hardwood flooring. The high figures for imported hardwood flooring are due to the high component of sawlogs from SE Asia used to manufacture the floorboards.

Scharai-Rad and Welling (2002) calculated the substitution factors for wood utility poles compared to steel utility poles as 1.6 t C / t C in HWP – in our study the equivalent figure was 1.2 t C / t C in HWP. Künniger and Richter (1995) conducted a comprehensive study of a range of different types of roundwood poles in Switzerland, and the middle range of their estimates for roundwood utility poles ranged from 0.6 to 4.4 t C / t C in HWP.

The production of electricity poles in NSW has a dual GHG benefit of very high long-term C storage and a high product substitution benefit. Pallets produced in Victoria, on the other hand, have insignificant long-term storage benefits. Although the true economic life of pallets is difficult to determine because of the frequent need to repair them and variability in how they are treated in service, it is commonly assumed that on average they have a relatively short service life. Furthermore, pallets from Victoria are mulched at the end of their service life and used in gardens. They also have low product substitution impacts, as the main alternative scenario is the use of untreated pine, which has a low emission factor associated with its production. It should be noted though that there may be some compromises in durability and longevity associated with a switch from hardwood to softwood pallets. If pallets were disposed of in landfill as opposed to being used as mulch at the end of their service life, the impact on the overall long-term C storage in HWPs for the mountain ash site would increase. However, companies with responsibility for the disposal of pallets may have sustainability policies in place that favour minimising sending waste to landfills and seeking an alternative use for the pallets at the end of their service life.

Long-term C storage for HWPs in Australia is primarily imparted by the post-service stage of the HWP life (i.e. storage in landfill). However, the overall GHG impact will be beneficial whether the product is recycled into another long-lived application (e.g. old floorboards used in recycled furniture), landfilled or burnt to produce energy. In fact, the emission abatement created by diverting wood waste from landfill to energy generation facilities (resulting in fossil fuel displacement) is likely to be higher than if the product is placed in landfill, depending on whether/which fossil fuels are displaced.

In this section we report on the potential GHG benefits associated with the extraction of forest harvest residues for bioenergy generation - removal of biomass may also play an important role in reducing the impact of wildfires. This removal would need to take into account the impact of extraction on biodiversity and on nutrition levels for future rotations.

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Part 4. Pulp and paper products - product substitution

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4.1. Section summary

In chapter 3 we devised product substitution factors for HWP and biomass used for bioenergy. In this section we consider the product substitution impacts associated with the harvest of pulpwood. Pulp logs represented a significant proportion of the biomass in native regrowth silvertop ash and mountain ash forests, with currently approximately 1.1 million m³ of pulp logs harvested from Eden and Victoria (ABARES 2015). The dynamics of C flows in the paper products from the case studies included here were originally outside the scope of this analysis. This was partly due to the perception that the GHG mitigation benefits of the production and use of paper products was limited given their typically short service lives. However, the product substitution impacts associated with the use of pulp logs from native forest biomass may be significant, when the relative emissions associated with the extraction of pulp logs elsewhere are taken into account. In this section of the report we present estimates for the substitution impact of using pulp logs from native forest biomass from Eden, NSW and from the Central Highlands of Victoria.

Currently mountain ash pulp logs are mostly used locally by Australian Paper in the production of printing and writing paper (primarily the Reflex brand of copy paper), which is also consumed locally. Silvertop ash woodchips are exported to Asia (predominantly Japan and China), where they are also used for the manufacture of printing and writing paper.

Based on recent assessments conducted for Victoria, we have deemed that sourcing pulpwood from existing plantations in Victoria or elsewhere in Australia, or importing pulpwood to replace the native regrowth pulplogs was financially unviable. A similar conclusion was reached in relation to the possibility of new hardwood pulp plantation development to supply the local market, given the recent trends in plantation establishment and negative prospects. Further discussion on these topics is included below.

The results below need to be interpreted in the context of the type of paper (printing and writing paper) that is typically manufactured from the pulplogs extracted from both the Central Highlands of Victoria and the Eden region of NSW. The pulp and paper industry has complex market structures, and what may be typical of a particular sector of the industry (e.g. packaging) may be completely different for another sector. All efforts were made to understand the potential implications of stopping this production in the context of current usage and trade patterns for printing and writing paper as it relates to Australia. As complete trade information for printing and writing paper is not readily available due to confidentiality issues, our analysis of market trends was developed based on discussions with industry and market analysts. Due to time constraints we restricted our analysis to implications in the Australasian region, given the importance of the region in terms of volumes of pulp and paper production, its proximity and the historical trade links with Australia. Discussion on potential market implications associated with potential substitution scenarios involving other regions, especially South America, was outside the scope of the analysis for this report. A more comprehensive analysis will be included in a future manuscript.

- *Key findings*

- In this study the key alternative source of pulpwood for paper production (printing and writing paper) was identified as SE Asia, primarily in Indonesia. Indonesia is also the key pulp and paper producing and exporting country in SE Asia.
- The calculated weighted emission factor for pulp and paper produced in Indonesia range from 5.5 to 7.7 t C / t C in pulp logs. These figures are consistent with the few factors available in the literature for pulp production in SE Asia.
- The product substitution impact for pulp production from silvertop ash ranged from 184 t C/ha to 252 t C/ha. For mountain ash, the figures were much higher, ranging from 1010 t C/ha to 1384 t C/ha.
- The inclusion of the product substitution impact for pulp biomass has a very large impact on the GHG balance of production forestry in the regions where pulp logs are extracted.

4.2. Methods and industry background

4.2.1. Pulp log substitution factors

The substitution factors applied to the pulp logs were determined by following these steps:

- Determination of key product types (printing and writing paper) and geographical markets for products (in consultation with Australian Paper and pulp and paper market analysts).
- Determination of the likely source of pulpwood should harvesting cease in the mountain ash and silvertop ash forests of south east Australia (in consultation with Australian Paper and pulp and paper market analysts).
- Determination of the GHG balance associated with the production of the alternative pulp and paper products. In the calculation of these factors, we only took into account the emissions associated with forest management (i.e. any C losses associated with the harvest of the SE Asia native forests, and where relevant subsequent sequestration in growing plantations managed for pulp production). The emissions associated with the manufacture of the paper at the pulp mill were assumed equal in both scenarios (i.e. production in Australia and overseas).
- Key potential sources of biomass for pulp and paper production in SE Asia include native forests on mineral soil and on peatlands (Indonesia) and plantations, the majority of which have been established post 1990 on former forested lands.
- The emission footprint of converting native forest on mineral soil and peatlands was determined based on published above ground C stocks for the relevant native forest types and published emission factors for peatland drainage and or burning where relevant.

- C sequestration in the plantations established was calculated based on published values for key species used for pulp production.
- In order to calculate an emissions factor for each tonne of C in pulp logs, harvest yields for pulp from native forests were estimated based on the knowledge of typical recovery values for sawlogs in Indonesia.
- The substitution factor for pulp logs was calculated as the difference between the impact on forest management between the case study sites in Australia and in SE Asia.

4.2.2 Likely replacement scenarios for printing and writing paper produced from native forest biomass in Victoria and in Eden.

The exact implications of removing Australian native hardwood pulpwood from the market are of course unknown and impacted by local and global factors; however based on an assessment of the local plantation profile, current major suppliers in Asia and the shifting trade patterns, we have concluded that biomass from SE Asia would most likely fill the immediate to medium-term gap in the market. In the sections below we discuss current markets and likelihood of different substitution scenarios.

4.2.2.1 Current production in Australia

Cessation of harvest in silvertop ash and mountain ash forests in Eden and Victoria would remove approximately 1.1 million m³ of pulp logs from the market (ABARES 2015). In both the Central Highlands of Victoria and in Eden, the production of pulp logs is part of sawlog-driven integrated harvest operations, in forests that are considered to be sustainably harvested. Currently most of the mountain ash pulp logs are used locally by Australian Paper in the production of printing and writing paper (“Reflex” brand). In 2013 Australian Paper produced 619, 000 tonnes of paper, with native regrowth forests supplying 34% of the fibre (Australian Paper 2013).

Silvertop ash woodchips are exported to Asia (primarily Japan and China). The export market for woodchips from Australia has recovered recently following a contraction in 2012 – key factors for this resurgence include the drop in the Australian dollar, the quality of resource, consistency of supply, reduction in production costs, low sovereign risk and the recent availability of very large and efficient woodchip carrier vessels (Flynn 2015, Industry Edge 2015b, Grindlay 2015).

4.2.2.2. Substitution with plantation hardwood pulplogs from existing or new plantations in Australia

The first logical assumption to analyse is whether native pulpwood from the case studies is likely to be replaced by pulpwood from existing or new plantations in Australia. Existing plantations in western Victoria were established to supply export pulpwood markets in the Asia-Pacific, mainly Japan, with location, species and management regimes optimised to suit those markets (Poyry 2011). The impact on supply from those plantations should harvesting cease in the mountain ash forests of Victoria has been addressed in two recent reports; one in 2010 by the National Institute of Economic and Industry Research (NIEIR 2010), the other in response to this review by Poyry Management Consulting (2011). Both reports agreed that supply from the current plantation estate in western Victoria is theoretically sufficient to meet the demand of Australian Paper now and into the near future. However the financial viability

and long-term sustainability of supply to Australian Paper is questionable. Poyry (2011) concluded that the transportation costs would double if the pulpwood had to be transported from plantations in Western Victoria to the Australian Paper. Also cost advantages of native forest pulp logs over commercial plantation logs is a major impediment for transitioning to a greater utilisation of the plantation resource, with stumpage prices for native pulpwood much lower than for plantation pulpwood (NIEIR 2010). These differences are primarily related to the high costs of plantation establishment and management, as well as land acquisition costs associated with plantation pulpwood. Furthermore, breaking existing long-term pulpwood supply contracts between VicForests and Australia Paper would incur compensation claims and further costs. These three key factors strongly suggest that transitioning from native pulpwood to plantation pulpwood to supply Australian Paper is not financially viable. Transitioning to a plantation estate to replace the current large volumes of native regrowth pulpwood from the Eden region (approximately 270,000 tonnes) is also not financially viable, due to similar reasons as outlined above.

As the blue gum plantations established under the MIS reach maturity, their harvest is causing a temporary glut in the hardwood chip market in Australia (Grindlay 2015). These plantations are feeding the export market following significant investment in chipping mills and port infrastructure (Grindlay 2015). However, there is reluctance to retain many of the existing hardwood pulp plantations following the collapse of the MIS, and the establishment of large new areas of hardwood pulp plantations is currently considered highly unlikely given the high costs of plantation establishment (Poyry 2011, Comm. 2015a & c). Hardwood plantation pulpwood available from Western Victoria was expected to peak by 2015, and expected to decline considerably (50% reduction) after 2020 due to reduced reinvestment (Poyry 2011).

Thus, while it is generally agreed that the current west Victorian plantation estate has the capacity to supply Australian Paper with pulp logs now and into the near future, it is neither financially viable nor sustainable in the long term. A similar conclusion is reached for the Eden resource. As a result, we need to look into potential replacement scenarios overseas.

4.2.2.3. Substitution with hardwood pulplogs from SE Asia

In this study we have restricted our assessment to Asia, given the importance of the region in terms of volumes of woodchip and pulp and paper production, its proximity to Australia and the historical trade links with Australia. Below we discuss the woodchip and pulp and paper industry for key countries in the region.

4.2.2.3.1. Vietnam

Changes in trade patterns, driven primarily by the emergence of China have seen Vietnam overtake Australia as the largest supplier of hardwood chips to Asia (Massey 2012, Flynn 2014, FAOSTAT 2015). However there has been growing concern over the long term sustainability of supply from Vietnam. This has been fuelled by recent government policy to move away from low value exports to more integrated processing and value adding, the reduction in overseas funding, the high dependence on Acacia (which is currently being replaced in Indonesia and Malaysia due to disease) and the rapid rates of harvesting over recent years (Harwood & Nambiar 2014b, Viet Nam News 2013, Flynn 2015, Hawkins Wright 2013, Tran & To 2013, New Forests 2015). Permits for new chipping facilities are becoming more difficult to acquire (Hawkins Wright 2013) and there has been discussion around woodchip export tariffs being put in place (Hawkins Wright 2013, Tran & To 2013). Recent trade figures suggest that Vietnam's supply may have peaked with exports declining

in 2014 (New Forests 2015). The quality of the woodchip coming from Vietnam's *Acacia* plantations also suggests that it would be an unsuitable replacement for Australia's hardwood chip, which is primarily used in the production of high quality writing paper. This has been suggested to be currently limiting Vietnam's incursion into the Japanese market (Tran & To 2013, Industry Edge 2014).

4.2.2.3.2. *Thailand*

Thailand has a more established processing industry than Vietnam, with three major pulp mills producing over 1 million tonnes per year, and Siam Pulp and Paper the largest integrated pulp manufacturer in Thailand with facilities in other ASEAN countries (Harwood & Nambiar 2014b, Woods *et al* 2011). Thailand has increased its exports of hardwood chips over recent years, from a predominantly *Eucalyptus* plantation estate (Harwood & Nambiar 2014b, Woods *et al* 2011). Between 80-90% of pulpwood comes from smallholder farmers, and with the industry accepting logs down to 2.5cm diameter, rotations can be as short as 3 years (Woods *et al* 2011). However, there is uncertainty in the market surrounding the supply from Thailand (Hawkins Wright 2013); this may be partly due to the flexibility of farmers to move to other crops relatively quickly depending on market conditions. Contract farming tries to address this by securing supply, but penalties for breach of contract are not normally enforced and approximately two thirds of plantations are not under any contract (Boulay & Tacconi 2012, Woods *et al* 2011). Harwood & Nambiar (2014b) identified limitations in the Thailand's plantation industry; in particular an absence of coherent research and development has meant that productivity is well below its potential and the genetic research needed to develop new clonal varieties is not occurring. Similar to Vietnam, the uncertainty surrounding the supply from Thailand is reflected in a reduced estimated forecast range of 3-6 Mt per annum (Hawkins Wright 2013).

4.2.2.3.3. *Indonesia*

Indonesia is one of the top four exporters of hardwood chips to China. In 2013 90% of Indonesia's 1.6 Mm³ of exported woodchip went to China (FAOSTAT 2015). Indonesia differs from Vietnam and Thailand in that it still has significant areas of native forest and continues to source timber from those forests (FAO 2011). The government has encouraged a move away from native forest timber by providing financial incentives for plantation establishment (Obidzinski *et al* 2012). According to Obidzinski *et al* (2012), this has been largely abused, with only 4.9 Mha of the total of 10 M ha of land allocated for plantation establishment actually planted. Where plantations have been established (typically *Acacia mangium*), the quality and stocking rates have been poor and of low productivity, with the onset of widespread stand mortality due to fungal disease (*Ganoderma* root rot and *Ceratocystis* stem wilt/canker), (Harwood and Nambiar 2014b). As a result, many *Acacia* plantations are currently being replaced by *Eucalypt spp.* plantations (Harwood and Nambiar 2014b). There is no data on the origin of the pulpwood (either as chip or pulp) exported from Indonesia. However, based on historical patterns of forest loss within the bounds of fibre concessions, the vast majority of pulpwood has been either directly sourced from native forests or from plantations that have been established on lands cleared post 1990 (Abood *et al* 2014, Obidzinski *et al* 2012, Barr 2007).

4.2.2.3.4. *China*

Another potential source of pulp biomass is China. The total area of *Eucalyptus* plantations in China doubled between 2006-2012, and in 2013 it was estimated to be 4 million hectares (Harwood and Nambiar 2014b, Flynn 2013). The productivity and sustainability of these plantations to pulp production face similar issues as in other SE Asian countries, with small

numbers of clones being planted in large areas and generally poor land management practices. In addition, frost susceptibility and topographically diverse and often nutritionally poor plantation sites pose additional challenges (Hardwood and Nambiar 2014b, Flynn 2013). Between 2007-2014, imports of woodchips and pulp increased proportionally more than consumption (61% and 16% respectively), indicating that China has been unable to increase their production capacity to meet domestic demand over this period (FAOSTAT2015). The demand for domestic hardwood plantation timber is high, with competition from the veneer and plywood industries (Hardwood and Nambiar 2014b, Flynn 2013).

4.2.3. Trends in printing and writing paper consumption and production

Consumption of printing and writing paper in developed countries is declining as paper is replaced by electronic devices. Globally this has been largely offset by India and China's increased consumption, which for paper and paperboard, grew by 8% and 9.5% respectively, while global consumption increased by 1.3% (FAOSTAT2015, Valois 2012). Strong demand in the sector is expected to continue into the future, as the per capita GDP for both India and China is well below that of other countries (GDP being a common indicator to paper consumption, (Flynn 2013, New Forests 2014, 2015)).

In line with consumption trends, production of writing paper in developed countries is declining, with Europe and North America reducing paper production capacity, while at the same time China has been replacing outdated inefficient mills and increasing their capacity (Valois 2012, PTT 2015, New Forests 2015, Hawkins & Wright 2015). China is the only major paper and paperboard producing country to have increased production since the year 2000 (FAOSTAT 2015). This is driving China's demand for hardwood chip, which is currently being met by Australia and SE Asia (FAOSTAT 2015). Historically Japan has been the dominant importer of Australian hardwood chips, receiving 86% of Australia's exports in 2008; however by 2014, 45% of Australia's hardwood chip exports went to China (ABARES 2015, Industry Edge 2015c).

There are concerns about the implications of China's increased demand for imported woodchips, with Sun et al (2010) suggesting that it could drive demand for illegal "virgin wood and pulp". China is considered to be the world's largest importer of illegal timber (EIA 2012). Lawson et al (2010) estimated that approximately 20 Mm³ of illegally sourced wood products (logs, sawn timber, plywood and veneer) were imported into China in 2008. A combined total of nearly 25% of China's wood pulp in 2012 came from Indonesia and Brazil (Margono et al 2014). Although China's State Forestry Administration (SFA) has engaged with the international community and the Chinese Government has announced initiatives to combat illegal logging through a Memorandum of Understanding with both Myanmar and Indonesia, so far it has not put in place legislation prohibiting illegal trade into and within the country (EIA 2012, Lawson et al 2010). China's state owned companies in 2007 imported 46% of the total tropical log imports, including timber from high risk countries (EIA 2012).

Within the pulp and paper industry, China's strongest commercial ties within SE Asia are with Indonesia. In 2012, 75% of China's hardwood chip imports went to just two mills, APRIL at Rizhao and APP on Hainan Island (Flynn 2013, Hawkins Wright 2013). These are two of the world's largest vertically integrated pulp and paper companies, with origins in Indonesia. APP has a total operating capacity of over 19 Mt per annum and market their products to over 120 countries (APP 2013). APRIL operates one of the largest pulp mills in the world, located in Riau Sumatra (the pulp capital of Indonesia) with a capacity of 2.8 Mt

per annum. Both APP and APRIL have been found to be either directly involved or linked to illegal forest conversion (Lawson *et al* 2014). Under the government's reclassification of degraded forests (mostly logged over forests) policy, both companies can also legally remove native vegetation from areas re-classified as plantation concessions. Both companies have taken steps to address global concerns over this - APP released its 'Forest Conservation Policy' in February 2013, halting natural forest clearance across its supply chain while High Conservation Value (HCV) assessments are conducted and non-forested areas are identified for development (APP 2013). APRIL has made similar commitments, halting harvesting of mixed hardwoods in May 2015 and declaring that future development will be on non-forested areas only (APRIL 2015). Both companies have made similar commitments in the past, whereby 100% of their supply would come from Acacia plantations by 2007 (APP) and 2009 (APRIL) (Barr 2007). These current commitments coincide with a slowing of the Chinese economy and subsequent reduction in demand. The business model of the largest pulp and paper companies, which are often multinationals supplying many different markets, relies to some extent on the willingness of major consuming markets to purchase paper that is not certified. Considering the growing consumption by China and India and their increasing reliance on imports to meet demand, any measures put in place in Australia to discourage the use of illegally sourced biomass will have little impact globally unless those measures are implemented in all major importing countries.

Indonesia is well positioned both geographically and commercially, to supply China with hardwood chips and pulp. Indonesia's move to Eucalypt plantations combined with its native forest resources place it in a much better position than other SE Asian countries, such as Vietnam and Thailand, to increase the supply of high quality wood chips and or virgin pulp fibre relatively quickly. Thus, below we detail the steps taken to derive an emission factor for future supply of woodchips and pulp from Indonesia primarily to China. This factor was used to derive the product substitution impact if native pulp logs were no longer extracted from the Central Highlands of Victoria and Eden.

4.2.4. Derivation of an emissions factor for pulp logs extracted in Indonesia

It is difficult to ascertain the specific origin of timber used in the pulp and paper industry within Indonesia, as detailed information is not available (Lawson *et al* 2014, Persson *et al* 2014a), with official data often being unreliable (Wicke *et al* 2011). The increased availability of satellite imagery has allowed researchers to estimate areas of deforestation and degradation. Hansen *et al* (2009) estimated that 21.3 Mha of forest area had been cleared in Indonesia between 1990-2005, and Miettinen *et al* (2011) reported a loss of 8.8 Mha in Indonesia from 2000-2010. Margono *et al* (2012) reported that 7.54 Mha of primary forest was deforested and 2.31 Mha degraded in Sumatra from 1990-2010 with the majority of this (5.43 Mha or 72%) deforested during the 1990's. Thus while the ability to estimate land use change has improved with the use of satellite imagery, the reported values vary due to inconsistencies in the definitions used when stratifying remotely sensed data, the use of different data sets, and the potential for misclassification as assessments are based on a snapshot in time.

Attempts have been made to use remote sensing information to link it to the drivers of land use change and quantitatively apportion this to the major industries such as palm oil, pulp and timber. To date much of the industry-specific research has focused on the palm oil industry. Carlson *et al* (2013) calculated a net emission of 0.4 Gt C from the conversion of 3.15 Mha of forested land to oil palm plantations in Kalimantan (127 t C emitted / ha) from 1990-2010,

with peatland conversion accounting for 26% of this. Interestingly Margono *et al* (2012) reported that 28% of land conversion in Sumatra from 1990-2010 occurred during the 2000's, whereas Carlson *et al* (2013) estimated that 76% of land conversion by oil palm in Kalimantan occurred during the 2000's. This suggests that land conversion driven by plantations has occurred at different temporal rates across different geographical areas. Wicke *et al* (2011) attempted to quantify land use change in Indonesia and Malaysia from 1975-2005, and in particular the role that palm oil has played. They estimated that forest cover decreased by 39 Mha from 1975 - 2005 while agricultural land increased by 10 Mha over this period, with palm oil accounting for approximately half of this expansion. Abood *et al* (2014) apportioned forest loss to multiple industries within Indonesia to the extent that it occurs within industrial concessions for the period 2000-2010. Deforestation within concessions, whereby forests are converted to non-forest land cover, accounted for 44.7% of total forest loss, with fibre plantation and logging concessions and palm oil plantations accounting for approximately 12% of the total forest loss each, with the balance coming from mining (2.1%) and mixed concessions (6.3%).

There are difficulties in directly attributing forest loss and degradation to a specific industry. Estimates from Abood *et al* (2014) are limited to the confines of concessions boundaries. Delays in plantation establishment can result in underestimation of an industry's contribution (Abood *et al* 2014, Lawson *et al* 2014, Wicke *et al* 2011). Furthermore, the use of fire to clear lands often spread beyond the intended area, making industry apportioning difficult (Lawson *et al* 2014). This is further complicated by evidence of the practice of acquiring plantation licences as a means to access timber, with no intention of fulfilling the licence requirements of plantation establishment (Lawson *et al* 2014, Wicke *et al* 2011, Persson *et al* 2014a).

A number of researchers have attempted to attribute land use change and its associated emissions to a commodity. Persson *et al* (2014a) developed a method for calculating a land use change C footprint for agricultural commodities and applied it to beef and soy from Brazil and palm oil from Indonesia. They estimated that 7.5 t CO₂ was emitted per tonne of palm oil extracted from Indonesia using a ten year amortisation period. In a working paper for the Centre for Global Development, they applied this method to Indonesian pulp and paper, and for 2009 estimated that 46 t CO₂ was emitted for every tonne of pulp and paper produced (Persson *et al* 2014b). However it is unclear from the study whether the emissions were expressed on a green tonne of pulp basis or on an air-dried finished product basis. Depending on the values used, the emission factor would be equivalent to a range of 4.3 to 12.5 t C / t C in pulp and paper. The Rainforest Action Network and the Japan Tropical Forest Action Group estimated emission intensities in the range of 16-21 t CO₂ (or 4.4 – 5.7 t C) per tonne of paper (RAN & JATAN 2010). As there is considerable uncertainty surrounding the calculation of these figures and how they are expressed, they should be treated as indicative only.

As there is currently no direct means of knowing precisely the origin of the timber used to produce pulp and paper, we have drawn on a number of publications in an attempt to derive an emission factor. The most challenging aspect is deriving an estimate of the proportionate origin of the timber used. We apportioned pulp into two broad sources: pulp from plantations established prior to 1990 (the default base year used for emission reporting under the Kyoto Protocol) and pulp from lands that are either currently being converted to plantations or have been converted post 1990. Within the post 1990 classification we have made further distinction as to the origin of the pulp based on the condition of the forest being converted, and whether it was on peatlands or mineral soil.

4.2.4.1. Proportion of pulp from plantations established prior to 1990

Obidzinski *et al* (2012) estimated the proportion of timber for the pulp and paper industry coming from plantations and from natural forests annually from 1997-2010. For 1997 the proportion of timber for pulp and paper from plantations approximated 5% - assuming they were established prior to 1990 (which is probable given a seven-year rotation period) and that they are still productive, in 2010 these plantations accounted for 2% of total timber production.

4.2.4.2. Proportion of pulp from lands that are either currently being converted or have been converted from forested lands to plantations post 1990

Of the 98% of pulp sourced from previously forested lands (post 1990), 5% was deemed to originate from primary intact forest and 95% from primary degraded forest. This is based on the assessment by Margono *et al* (2012) of forest cover loss in Sumatra from 1990-2010, where 95% of forest loss during this period was primary degraded forest, meaning that forests were typically logged prior to clearing. Their assessment of the whole of Indonesia for 2000-2010 found that 98% of forest loss was primary degraded forest. This notion that the majority of primary intact forest is firstly degraded through selective logging of high value species for the sawn timber industry with the pulp industry converting already degraded lands is supported by many authors (Persson *et al* 2014, Lawson *et al* 2014, Agus 2013, Carlson *et al* 2013). It is also consistent with 'The State of the Forest: Indonesia' report in 2002, which details Indonesia's forest history and specifically the establishment of large scale commercial logging concessions during the 1970's, which saw the extraction of logs increase from 4 Mm³ in 1967 to 28 Mm³ in 1977, with the majority of this exported. A ban on the export of logs in the early 1980's to encourage processing industries within Indonesia saw a shift into plywood production and the number of plymills increase from a total of 21 in 1979 to 101 in 1985. Therefore it is likely that the majority of primary forest that has been cleared, particularly easily accessible forest, had already been selectively logged by 1990. Obidzinski *et al* (2011) suggests that pulp mill timber is either sourced from plantations or conversion of natural forest, implying that none of the timber is obtained through selective logging, which seems likely considering the additional expense involved in selective logging compared to forest clearing.

To account for the emissions associated with forest loss from peatlands, we used the assessment by Abood *et al* (2014) of forest loss during 2000-2010 within concession boundaries attributable to the fibre industry. They found that ~0.665 Mha (35%) of the 1.9 Mha attributable to the fibre industry occurred on peatlands. It is assumed that the balance of 65% was from forests on mineral soils. Miettinen *et al* (2011) reported a 25% loss of forest on peatlands in their study area which covered Malaysia and Indonesia for 2000-2010 - when looking specifically at two provinces in Sumatra the forest loss on peatlands was 40%. Margono *et al* (2014) estimated that 43% of Indonesia's total loss of primary forest occurred in wetlands (peat swamp, freshwater swamp and mangrove forest) although for Sumatra the proportion from peatlands was higher at 54%. Thus, while estimates from Abood *et al* (2014) of 35% may be conservative, it is the only study which specifically attributes loss to the fibre industry and therefore the figure we have used.

4.2.4.3. Net emissions from forested lands converted to plantations

Gross above ground (ABG) C emissions for primary forest were based on C stocks of 229 t C/ha for primary forests on mineral soils and 180 t C/ha for primary forests on peatlands (Carlson *et al* 2013). Gross C stock emissions for peatlands include emissions due to drainage and burning: 235 t C/ha for peatland drainage (23.5 t C/ha/yr assuming ten years of emissions, although emissions can continue on for much longer), and 203 t C/ha for peatlands burnt (Carlson *et al* 2013). The C stocks for degraded lands were assumed to be on average 40% of primary forests (Carlson *et al* 2013, Lasco *et al* 2002).

In calculating a net emission factor, allowance was made for the C sequestered in the pulpwood plantations that were established after the land was cleared. With an average rotation of 7 and 10 years respectively, the ABG C figures for *A. mangium* and *Eucalyptus spp.* plantations range from 55-75 t C/ha at the time of harvest (Krisnawati *et al*, 2011, Harwood and Nambiar 2014a, 2014b, Lasco *et al* 2002). Assuming they are managed on continuous rotations, the long-term average ABG C stocks would be 50% of this. An average standing C stock of 38 t C/ha for pulpwood plantations was derived based on 80% of pulp plantations being *Acacia mangium*, with the balance comprised of *Eucalyptus sp.* (Harwood and Nambiar 2014b, Krisnawati *et al* 2011). Further, we have assumed that all land cleared due to pulpwood extraction is converted to plantations; this however is not always the case with only half of the allocated timber plantation area reported as planted in 2010 (Obidzinski *et al* 2012). Thus the C sequestered in these areas could be lower and therefore our calculations are conservative.

In deriving an emissions factor for each tonne of C in the pulpwood extracted we were unable to find harvest yields specifically for pulp from natural forests. Harvest yields for sawlogs in Indonesia are typically 70 m³/ha (this takes into account the fact that nearly all easily accessible areas have already been logged once and second rotation harvesting has commenced), (Ruslandi *et al* 2011). Based on the assumption that the recovery for pulp is greater than for sawlogs (given the lower diameter cut-off for extraction of pulp logs), and also that there would be a broader range of species suitable for pulp production, we have adopted a harvest yield of 120 m³/ha (41 tC/ha) for pulp (Tables 4.2 and 4.3). This was proportionally adjusted for harvesting on degraded lands.

The emissions associated with each source of pulp were weighted based on their proportionate contribution to the total pulp extracted to derive a single emission factor.

4.3. Results

4.3.1. Production of pulp and paper from the case study sites

Pulp logs were extracted from both the silvertop ash and mountain ash production sites. Although pulp grade logs were also present in the blackbutt production site, at the time of the study there was no market for the pulp logs. In Table 4.1 we include the pulp log production volumes and the total C in pulp logs, expressed both “as extracted” from each site (SI) and also after an adjustment to account for the average regional production (RG), and also an estimate of the production of finished printing and writing paper.

Table 4.1 Volume and total C in silvertop ash and mountain ash pulp logs from the study sites

Species	Source	Volume of pulp logs (m ³ /ha)	C in pulp logs (t C / ha)	Printing and writing paper (tonnes/ha)
Silvertop ash	SI	95.5	32.4	54.0
Silvertop ash	RG	102.6	34.1	56.8
Mountain ash	SI	543.4	143.4	239
Mountain ash	RG	708.2	187.0	311.6

In Tables 4.2 and 4.3 we include the emission factors associated with the extraction of timber used for the production of pulp and paper in SE Asia. The calculated weighted emission factor for pulp and paper produced in Indonesia range from 5.5 to 7.7 t C / t C in pulp logs, assuming peatlands are drained and burnt, respectively (Tables 4.2 and 4.3). When expressed on a tonne of printing and writing paper basis, the figures range from 4.1 to 5.7 t C / t of finished paper.

The substitution impacts associated with the extraction of silvertop ash and mountain ash pulp logs are presented in Table 4.4. The figures for silvertop ash range from a minimum of 175 t C/ha (peatlands drained, SI) to 252.3 t C/ha (peatlands burnt, RG). For mountain ash, the figures range from a minimum of 774.4 t C/ha (peatlands drained, SI) to 1383.8 t C/ha (peatlands burnt, RG).

Table 4.2 Emissions factor for Indonesian pulpwood assuming peatlands are drained.

Source of pulp	%	Gross Emissions t C/ha	Sequestration from plantations t C/ha	Net emissions t C/ha	Harvest rates native timber t C/ha	Emissions t C Emitted / t C in wood extracted	Weighted net emissions t C/ t C
Pulp from plantations pre 1990 (2%)							
Plantation	2	-	-	-	-	-	0.0
Pulp from forests converted to plantations post 1990 (98%)							
<i>Primary forest (5%)</i>							
Mineral soil (65%)	3	229	38	191	41	5	0.1
Peat lands (35%)	2	415	38	377	32	12	0.2
<i>Degraded lands (95%)</i>							
Mineral soil (65%)	61	92	38	54	16	3	2.0
Peat lands (35%)	33	166	38	128	13	10	3.2
Total	100	Total weighted emissions (displacement factor)					5.5

Note. ¹ Long-term average C stocks assuming the plantations are managed on continuous rotations

Table 4.3 Emissions factor for Indonesian pulpwood assuming peatlands are burnt.

Source of pulp	%	Gross Emissions t C/ha	Sequestration from plantations t C/ha	Net emissions t C/ha	Harvest rates native timber t C/ha	Emissions t C Emitted / t C in wood extracted	Weighted net emissions t C/ t C
Pulp from plantations pre 1990 (2%)							
Plantation	2	-	-	-	-	-	0.0
Pulp from forests converted to plantations post 1990 (98%)							
<i>Primary forest (5%)</i>							
Mineral soil (65%)	3	229	38	191	41	5	0.1
Peat lands (35%)	2	618	38	580	32	18	0.3
<i>Degraded lands (95%)</i>							
Mineral soil (65%)	61	92	38	54	16	3	2.0
Peat lands (35%)	33	247	38	210	13	16	5.3
Total	100	Total weighted emissions (displacement factor)					7.7

Note. ¹ Long-term average C stocks

Table 4.4 Substitution impacts for the silvertop ash and mountain ash pulp logs

Source	Displacement factor (t C emitted / t C in pulp logs)	Substitution impact (t C / ha) Silvertop ash (SI)	Substitution impact (t C / ha) Silvertop ash (RG)	Substitution impact (t C / ha) Mountain ash (SI)	Substitution impact (t C / ha) Mountain ash (RG)
A ¹	5.5	178.2	187.6	788.7	1028.5
B ²	7.7	249.5	262.57	1104.18	1439.9

Note. ¹ Peatlands drained. ² Peatlands burnt

4.4 Discussion

The emission intensity for printing and writing paper was calculated for the extraction of pulp logs rather than on a finished product basis, as the only emissions that were accounted for were those related to loss of biomass in forest conversion. The assumption was that the energy and emission footprint of production at the pulp and mill level does not differ substantially between large, modern pulp mills, irrespective of which country they are based on. Also by expressing the results on a “tonne of C in pulp logs” basis, we minimised the opportunities for calculation errors associated with assumptions regarding recoveries at the mill level and the final organic C concentration of the finished paper products.

The substitution factors for pulp logs calculated for this report, although high compared to the factors calculated for other HWP in section 3, are consistent with the few factors available in the literature. Persson *et al* (2014b) reported an emission factor of 46 t CO₂ emitted per tonne of C in pulp and paper. It is not clear though on what basis Persson *et al* (2014b) expressed the pulp and paper figures. Similarly, the Rainforest Action Network and the Japan Tropical Forest Action Group assessed the C footprint reported by APP, reporting emission intensities on average of approximately 5.0 t C per tonne of paper (RAN & JATAN 2010). If expressed on a tonnes of C in pulp logs, this figure would increase to approximately 6.8 t C emitted per tonne of C in pulp logs. Thus the factors used in this report (5.5-7.7 t C emitted / t C in pulp logs) can be regarded as consistent with previous estimates.

When considering the emission footprint of production in a particular country, it is important to consider implications of national policy on the wider region. The issue of leakage is a common factor reported in SE Asia, where deforestation may be shifted from one country to other countries in the same region or other regions of the world where forest practices are unsustainable. The global nature of the supply chain for commodities such as paper products makes it especially challenging to model future changes in the emission footprint of production.

The high volume of pulp logs extracted from the Eden and Central Highlands sites, and the high calculated substitution factor for pulp logs resulted in a large substitution impact as a result of the extraction of silvertop ash and mountain ash pulp logs. The higher the volume of pulp logs extracted from the sustainably managed forests in Eden and the Central Highlands, the higher the substitution impacts associated with the use of that resource, which in turn can have the benefit of reducing the pressure for high emission intensity pulp and paper production in SE Asia.

In addition to the quantification of the pulp substitution benefits, another important aspect of the C cycle in the pulp logs areas relates to the post-service stage. The fate of the paper product at the point of disposal may have significant implications for the overall emission intensity of pulp production. Although the levels of recycling of paper are high in Australia (e.g. approximately 78% for cardboard – NRRS 2014), there are only a limited number of times that the same fibre may be recycled. Most paper products are eventually disposed of in landfills, as waste to energy facilities are not yet a widespread commercial reality in Australia. Most large-capacity landfills in Australia have state of the art methane capture facilities, with many generating electricity from the methane generated. Thus, it is important to understand the dynamics of anaerobic decay of paper products in landfills. There are varying estimates of the decay of the various types of paper products, but it is generally accepted that the lower the lignin concentration in the paper, the higher the likelihood of

decay (Wang *et al* 2015). Assuming that at least a portion of the paper in landfills will decay and generate methane, and if electricity is generated as a result, this will be renewable energy displacing the use of fossil fuels. The combined impact of the fossil-fuel displacement from the use of methane to generate energy and the C storage in the portion of the paper that does not degrade may be significant, and would potentially increase the overall GHG benefit of paper production and use even further. However quantification of these impacts was outside the scope of this report.

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Part 5. ForestHWP: Model Description and Verification

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5.1. Section Summary

In this section of the report a new C (C) accounting model for integrating forest growth and decay processes with harvested wood products (HWPs), landfall, and bioenergy is described (ForestHWP). The model includes forest growth/decay processes and natural disturbance, the impacts of harvesting on the forest system, and a full life cycle analysis of all HWPs and their ultimate fate. Also included is the capacity to account for transport and processing emissions associated with harvesting and product manufacture, and the inclusion of product substitution factors and fossil fuel offsets associated with the use of forest residues for bioenergy.

Verification of the model, to ensure that all model equations were correctly implemented and that the model structure was flexible enough to include a wide range of forest types and HWP scenarios, was achieved through replicating the results from three published studies, with parameter values and other constraints obtained from the original reports. A full description of the model equations is given in appendix 3.

5.2. Introduction

Comprehensive accounting of the whole-of-life GHG balance of production forests requires explicit inclusion of processes and parameters that span the entire forest/harvested wood product (HWP) system. This includes forest growth/decay processes and natural disturbance, the impacts of harvesting on the forest system, and a full life cycle analysis of all HWPs and their fate. It also requires inclusion of any emissions costs related to harvesting, transport and processing, and any emissions benefits accruing from the use of wood products as biofuels, and the substitution of wood products for non-wood alternatives.

During the planning phase for this project it was proposed to use the Australian Government's C accounting model 'FullCAM' as the tool for undertaking such whole-of-life assessments. However preliminary explorations using FullCAM suggested a number of simplifications built into this model could potentially compromise the detailed, site-based analyses required for this project. Most notable were the inability of FullCAM to simulate the time-lagged decay of standing dead trees and associated coarse woody debris (CWD) dynamics; difficulties in prescribing HWP decay dynamics and in-use storage; and difficulties with integrating the HWP dynamics with landfill. Of these the former is particularly important for describing the C dynamics of Ash-dominated forests, where cohorts of standing dead trees can persist for years-to-decades following fire (with associated relatively slow decay), before collapsing and decaying at much faster rates. Also, in many regenerating forests there is a peak in NPP in the years following disturbance (in the case of Ash forests this results from mass seedling regeneration, with correspondingly elevated leaf area indices and photosynthesis). This peak in NPP leads to an acceleration in C turnover that significantly modifies the pattern and timing of biomass accumulation. Prescribing post-disturbance NPP profiles in FullCAM is technically possible through manual adjustment of the Forest Productivity Index, but it is a relatively complex and error-prone process.

To overcome these limitations, the project team in consultation with the steering committee decided to develop a new C accounting model ('ForestHWP') to undertake the required integrative analyses. The development of a new modelling framework brings with it a number of advantages. First, it can be designed 'bottom-up' to ensure that all components of the forest-HWP system are included (forest dynamics including growth and wildfire, HWP, landfill, bioenergy offsets, product substitution, etc.). This is particularly important, as much of the controversy in recently published research seeking to quantify the full GHG balance of the harvested forest system stems from different studies differentially including/excluding a number of these key processes (see e.g. Dean *et al.* 2012; Ximenes *et al.* 2012; Keith *et al.* 2014a,b); the most notable of these being the handling of the impact of wildfire, the role of C storage in wood products in landfill, and the product substitution effect. Second, by including all sources and sinks, the calculations can be checked to ensure they conform to mass-balance, and hence that they provide an unbiased estimate of the net impact on the atmosphere. Finally, a customised solution provides the flexibility to develop a range of scenarios, and to include a range of potential non-traditional post-harvest pathways such as the utilisation of processing residues for bioenergy, and the calculation of any wood product substitution benefits/costs.

The purpose of this section of the report is to introduce and describe the ForestHWP model, and to present some verification analyses that confirms the model has been correctly implemented and is fit for purpose. Note that verification is distinct from validation, which is a test of how generically valid a set of parameters are when applied to novel situations. Whilst validation is an important component of general model testing, the lack of independent data restricted opportunities for validation in this study. The use of ForestHWP for integrating the data from the three case studies, including scenario exploration of different HWP pathways (e.g. utilisation of processing waste for bioenergy; changes to the product mix), is presented in the next section.

5.3. Forest HWP Model Description

ForestHWP seeks to provide a complete system description of both the forest C dynamics, and the dynamics associated with off-site HWPs, their processing, and their ultimate fate.

A three-stage process to the development and use of the model was adopted, of which stages 1 and 2 are covered in this section:

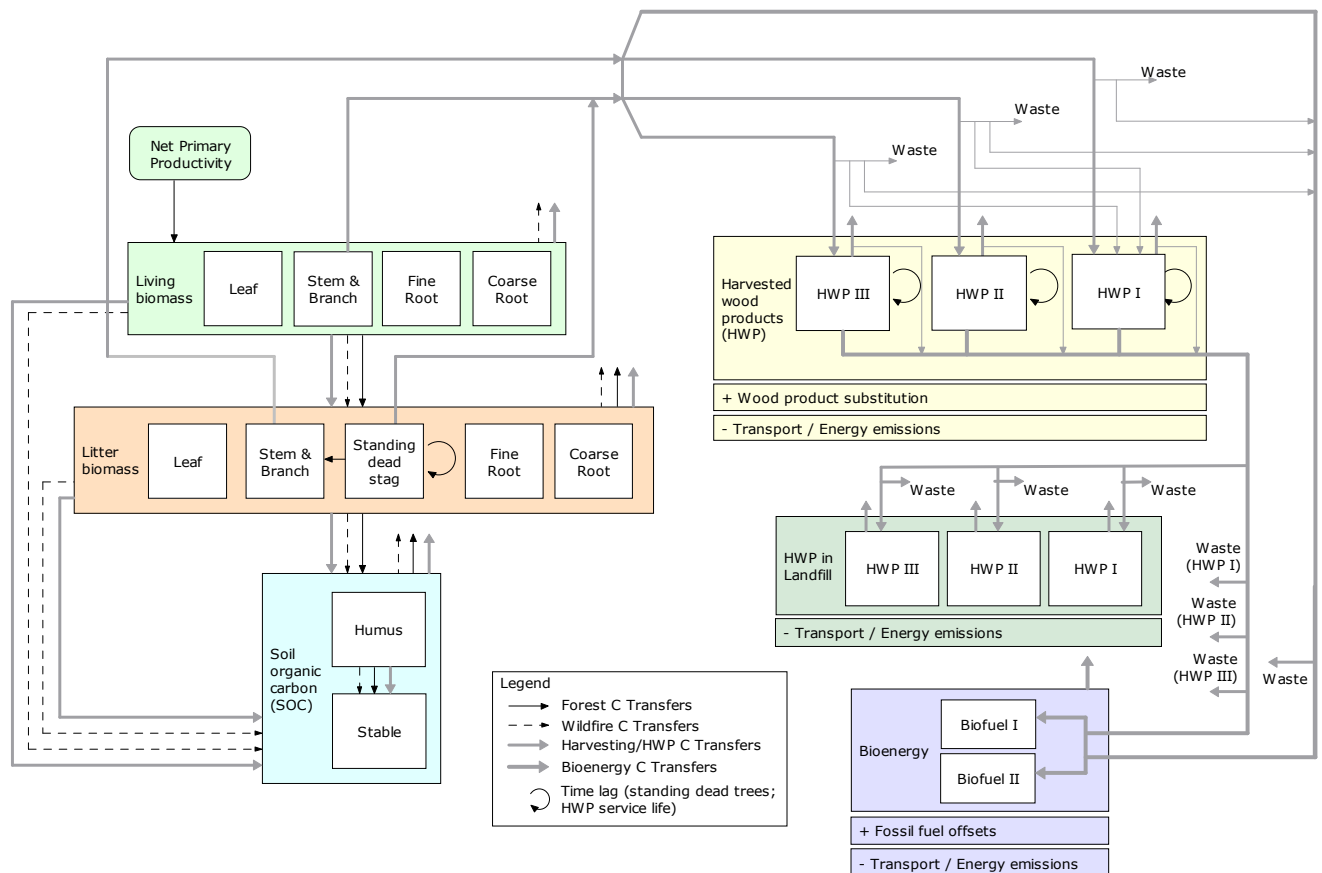
1. Development of the model framework and subsequent coding that includes (a) forest growth and recovery from harvesting and natural disturbance; (b) the dynamics of HWP in-service and at end-of-life, including landfill; (c) potential benefits arising from product substitution of wood products for more energy intensive materials, and fossil-fuel offset benefits from burning wood for bioenergy; and (d) potential emissions and waste costs associated with harvesting and processing of timber for HWP and for bioenergy.
2. Verification of the model through replicating the results from previous analyses, using parameters extracted from those studies. Verification is the process of ensuring that a new model is operating as expected, and that it is therefore "fit for purpose".

- Integration of the case study data within the modelling framework to quantify the long-term GHG account implications of these production forest systems, and use of the modelling framework to explore current and potential future harvesting and HWP scenarios.

A schematic of the model is shown in Figure 5.1, and a full description is given in the Appendix 3. Only a summary of the key features is given below.

There are 19 state variables in the model that represent the pools of C in the forest and HWP sub-systems. For the forest sub-system there are four living biomass pools (stem + branch - referred to hereafter simply as stem; leaf, fine root and coarse root); five litter pools (leaf, stem, standing dead stag, fine root, coarse root) and two soil pools (humus and stable). For the HWP sub-system there are three in-service harvested wood products (labelled I, II and III, but which could be calibrated to represent fast turnover products (e.g. ‘pulp’), medium turnover products (e.g. ‘pallets’) or long-lived products (e.g. construction timber); three landfill pools corresponding to each of the HWP’s (fast, medium, slow) and two biofuel pools. Because biomass for biofuel typically has a short life before it is combusted, the size of this pool is in practice negligible (although the rate of consumption of wood for bioenergy could be high).

Figure 5.1. Overview of the ForestHWP model. The left-hand-side of the figure represents the stocks and flows of C in the forest sub-system, and the right-hand-side are the stocks and flows associated with harvested wood products (HWP), landfill and bioenergy.



The dynamics of each state variable take the general form:

$$\frac{dC_i}{dt} = \sum Gains - \sum Losses$$

Where C_i is the C stock of pool i (living biomass, litter, soil, HWP, landfill or biofuel) and $\frac{dC_i}{dt}$ describes how fast the pool is changing. Integrating this equation over time gives the C storage. A summary of the typical gains and losses for each pool are given in Table 5.1, and are separated into continuous and event-driven or episodic. In ForestHWP episodic gains and losses occur due to harvesting events or to natural disturbance (fire). Continuous gains and losses include growth, litterfall and decomposition (Appendix 3).

Table 5.1. Summary of main gains and losses of C in ForestHWP.

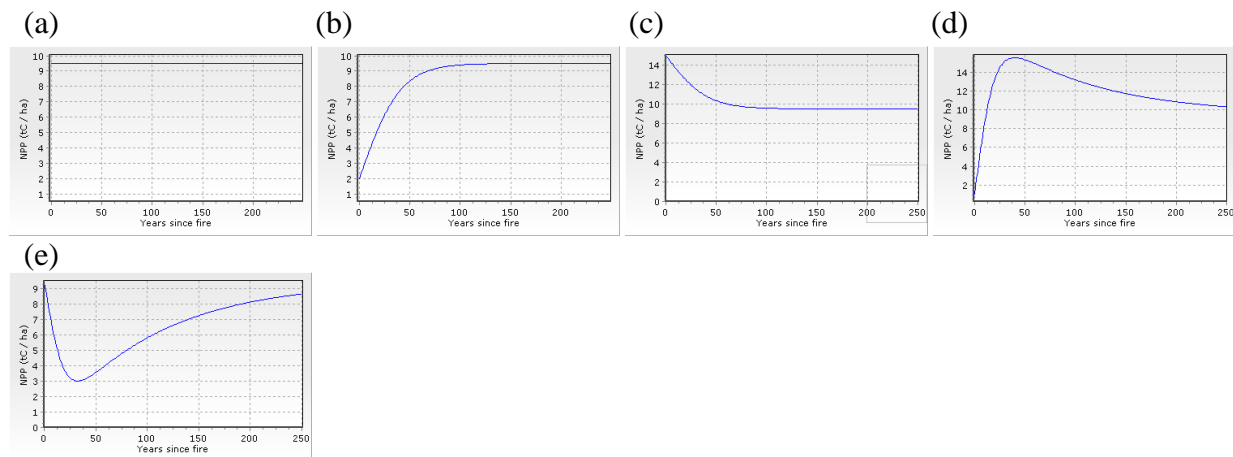
Pool	Continuous		Episodic	
	Gains	Losses	Gains	Losses
Living (includes leaf, stem and root)	New growth (NPP)	Litterfall, Mortality	-	Harvest (to litter, soil, HWP, bioenergy, and atmosphere). Fire (to atmosphere, litter, and soil).
Litter (includes leaf, stem, standing dead trees, and root)	Litterfall	Decomposition, Humification (losses from decomposing litter)	From living (Harvest and Fire)	Harvest (to soil, atmosphere, bioenergy, HWP); Fire (to soil, atmosphere).
Soil	Humification (inputs from decomposing litter)	Decomposition	From living and litter (Harvest and Fire)	Harvest and Fire to atmosphere.
HWP	-	In-service life decomposition	From harvesting	Waste losses during processing. Losses at end of service life to landfill, atmosphere, bioenergy.
Landfill	-	Decomposition	From HWP at end of service life	-
Bioenergy	-	Combustion	From harvesting, waste processing of HWP, HWP at end of service life.	-

5.3.1. The forest sub-system

A key component of the forest sub-system is the rate at which new growth is added to the living biomass, i.e. Net Primary Productivity (NPP). The dynamics of NPP following perturbations such as harvesting and fire are variable, and in ForestHWP an empirical NPP function provides this flexibility (Figure 5.2, Appendix 3). Of the possible responses the most important is depicted in Figure 5.2d, showing a peak in NPP in the years following perturbation, followed by a gradual decline to pre-disturbance values. This is a generic growth response of forests following a stand-replacing disturbance (e.g. Chen *et al.* 2002, He *et al.* 2012), and reflects higher rates of sequestration of C in young regenerating forests, driven by increased stem densities and increased leaf area.

Another key component of the forest sub-system is the inclusion of a standing dead tree (stag) pool. For this pool there is a time lag between the event that created the stags, and the time at which the decomposition of the stags starts to accelerate (Keith *et al.* 2014a,b). Following Keith *et al.* (2014a,b) a sigmoidal function is used to simulate this dynamic. In ForestHWP, dead stags can also be continuously created through a constant mortality fraction.

Figure 5.2. Range of post-disturbance NPP responses able to be simulated by the NPP-response function. (a) no growth response. (b) sharp decline followed by increase to pre-disturbance level. (c) sharp increase followed by decrease to pre-disturbance level. (d) the traditional forest growth response with an initial decline, followed by a peak NPP as the forest regenerates, then with a gradual decline as the forest matures. (e) Gradual decline followed by increase to pre-disturbance level.



During fire and harvesting events living biomass can be either combusted, removed from the site for HWP processing or for bioenergy, or can be transferred into the litter pool. Litter biomass can similarly be combusted or removed off-site. The model also allows for a fraction of living and litter C to be transferred to the soil organic C (SOC) pool, e.g. via the production of char from biomass burning.

5.3.2. The HWP sub-system

HWPs are created only during harvesting events. Losses of C to the atmosphere as waste during processing can occur, with an option for some or all of this waste to be utilised for bioenergy. Additionally, waste products from two of the HWP pools (I & II) can also be

converted into the third (III). If desired, decomposition of each HWP pool can occur in-service. At the end of the prescribed service life (which might implicitly include any recycling) the HWP can be transferred to landfill, to biofuel, or be lost back to the atmosphere; or any combination of the above. There are additional waste losses associated with the processing of end-of-life HWP for landfill and for bioenergy. Because of the discontinuous nature of gains and losses to HWPs, in ForestHWP each harvest cohort is kept separate to facilitate the tracking of service life and the subsequent calculation of post-service fate.

5.3.3 Extra-system calculations

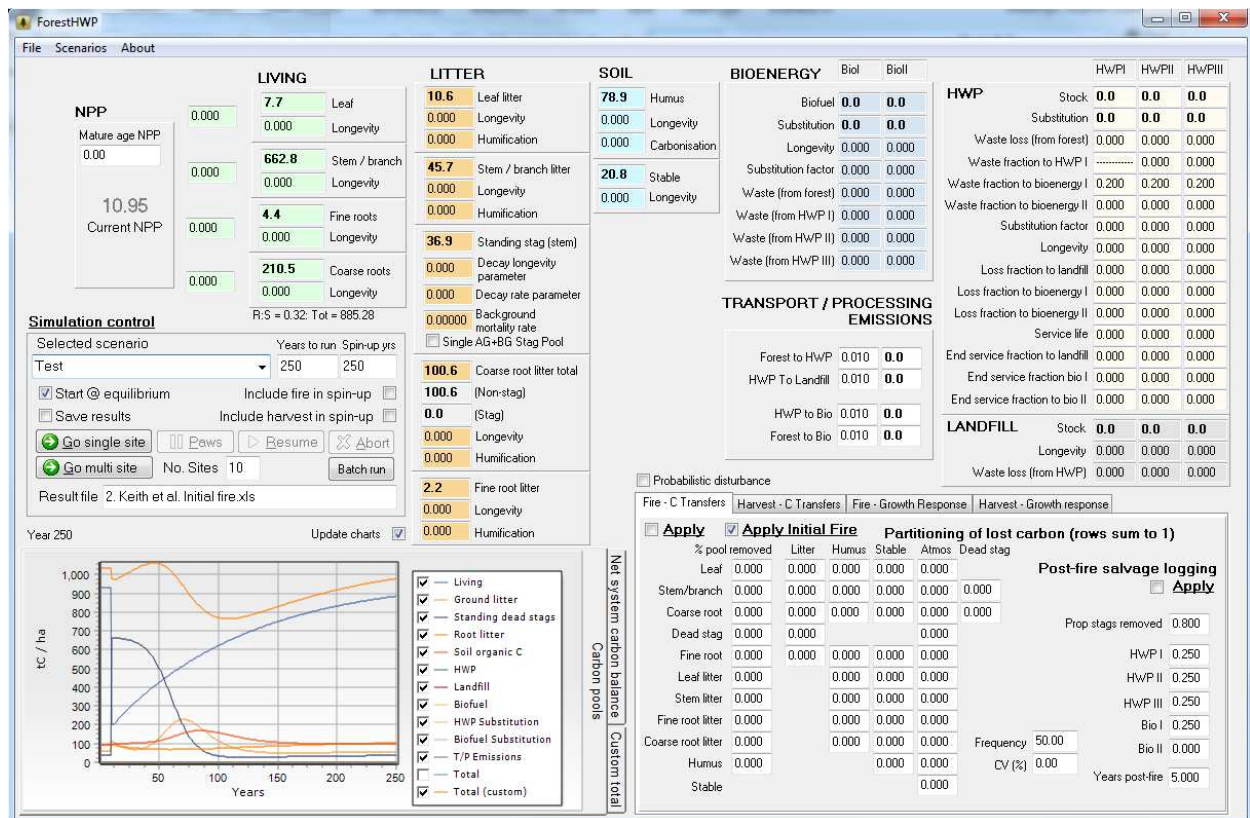
The two sub-systems of ForestHWP define a closed system that conforms to conservation of mass, where under any stationary (i.e. constant over the long term) regime of growth, harvesting and fire the system will eventually settle down to a long-term steady state where the total gains of C (from NPP) equal the total system emissions (which include decomposition, disturbance losses, and all emissions associated with the processing and fate of HWP, landfill and biofuels).

There are additionally two classes of calculations that are ‘extra-system’; i.e. they involve the fluxes of C outside of the forest-HWP system boundary. The first are the costs or benefits associated with the substitution of wood for other materials and fossil-fuel offset benefits associated with burning biomass for bioenergy. The second are fossil fuel emissions associated with the harvesting, processing and transport of HWPs. These are implemented in ForestHWP as simple product displacement factors, fossil fuel displacement factors, and emissions factors associated with the handling of HWP materials respectively (Ximenes *et al.* 2012).

The equations detailed in the Appendix 3 were implemented as computer code using the software development platform Embarcadero Delphi XE7.

An image of the ForestHWP interface is shown in Figure 5.3. The simultaneous visibility of all model input parameters and output results is intentional, and allows changes to scenario settings to be implemented without the need to navigate through parameter tab pages, or external parameter initialisation files.

Figure 5.3. ForestHWP software interface.



5.4. ForestHWP Model Verification

Model verification is the process of confirming that a given model is correctly implemented, and is producing results that are consistent with some pre-existing measure or test; i.e. that the model is fit for purpose. ForestHWP is both flexible and generic, and is designed to be comprehensive in its description of the total forest-HWP system. The verification procedure for ForestHWP involved calibrating the model parameters to replicate results from three previous case studies that collated detailed data and modelled the dynamics of (1) harvested Mountain Ash (*Eucalyptus regnans*) forests in Victoria (Keith *et al.* 2014a,b) and (2) Tasmania (Dean *et al.* 2012). These two case studies presented results in the form of a generalised or 'average' forest plot, and thus showed the temporal variability in C storage in response to specific harvesting and fire events. These two studies therefore explicitly included wildfire impacts, and reported detailed information on the temporal dynamics of C stores and fluxes within the forest and HWP sub-systems, allowing calibration of the major living and non-living C pools. The third case study (3), that of Ximenes *et al.* (2012), modelled the dynamics of harvested native forests at a regional scale in New South Wales. In this case study historical data on the regional harvesting activity was combined across multiple coupes, and the results were aggregated into a single figure that presented the regional average C storage.

Note that the verification process makes no claims as to the validity or otherwise of the data, assumptions, models or parameters used in underlying validation case studies. The verification process simply accepts the results as reported, and uses them as a yardstick

against which the model structure and behaviour can be assessed. Note that the model calibrations reported here for the purposes of model validation were conducted separately to those for the case study analyses based on the project results, which are described in the next chapter.

5.4.1. Verification procedure

The verification procedure involved calibrating the model parameters such that model outputs (the temporal changes in the various C pools) matched as closely as possible those reported in the original study. The calibration process was achieved by fixing ‘known’ parameters as reported in the original studies, and then manually tuning the remaining parameters (under constraints to ensure parameter values remain ecologically sensible) until congruency in the predictions was achieved.

(1) Victorian Mountain Ash case study

For the purposes of verification, ForestHWP was calibrated to the dynamics of the ‘average forest’ (Figure 5.4), as summarised graphically in Figure 10 of Keith *et al.* (2014a), Figure 7 in Keith *et al.* (2014b), and the ‘average mature forest’ data summarised in Table 2 of Keith *et al.* (2014a). The model parameters summarising the transfer and waste pathways of the harvested wood products were taken from Figure 8 in Keith *et al.* (2014a), and other parameters such as the CWD decomposition rates and fire combustion factors were obtained from descriptions in the text, and from Keith *et al.* (2014b).

A key driver of the C dynamics is the pattern of biomass accumulation over time, which in the absence of disturbance is defined as the balance between the rate of new biomass addition from photosynthesis (NPP), and losses due to litterfall/mortality. For model verification the biomass accumulation curve is given by Equation 2 of Keith *et al.* (2014a),

$$\text{Living biomass} = 620 \times (1 - e^{-0.0065 \times \text{age}})^{0.75} \times 1.25$$

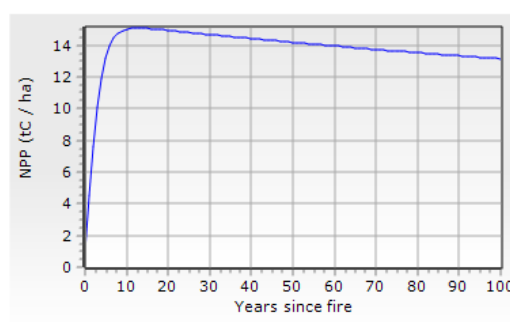
where 620 (t/ha) represents the above-ground C mass at steady state, and 1.25 is the expansion factor for root biomass. Overall, the verification procedure involved first calibrating the model for the forest sub-system in the absence of fire, to generate predictions for the forest at maturity. Fire was then introduced, requiring additional calibration of the forest sub-system parameters associated with fire impacts, post-fire growth response, and the production of dead stags. The forest sub-system parameters were then fixed, and the harvesting event parameters were calibrated, including the HWP stocks and transfers. A subset of the calibrated parameters that determine the dynamics of the living biomass pools is given in Table 5.2.

The standing stocks of living biomass and litter have been relatively well measured for Mountain Ash forests, providing strong constraints for the parameter value estimates. Of particular note is the NPP (Table 5.2; 9.5 tC/ha/yr), and the resulting NPP post-disturbance recovery trend (Figure 5.5a). The shape of the relationship in Figure 5.5 is a direct consequence of the interplay between the biomass accumulation function, and the constraints imposed by the litter and biomass measurements for the different tree components. Although there is limited data on the actual time-course of NPP following fire, Figure 5.5 is consistent in both magnitude and shape with alternative estimates based on combining biomass increments, litterfall and mortality (Polglase *et al.*, unpublished data; Haverd *et al.* 2013), and with estimates from other forest types (Chen *et al.* 2002, He *et al.* 2012).

Table 5.2. Calibrated model parameters controlling living biomass C dynamics in the three case studies. See Appendix 3 for parameter definitions.

Parameter	Keith <i>et al.</i> (2014)	Dean <i>et al.</i> (2015)	Ximenes <i>et al.</i> (2013) – North Coast	Ximenes <i>et al.</i> (2013) – South Coast
NPP at maturity (tC / ha / yr)	9.5	8.0	8.9	6.85
Fractional allocation of NPP to leaf	0.35	0.35	0.50	0.50
Fractional allocation of NPP to stems / branches	0.35	0.45	0.22	0.22
Fractional allocation of NPP to fine roots	0.1	0.1	0.25	0.25
Fractional allocation of NPP to coarse roots	0.2	0.1	0.03	0.03
Leaf lifetime (yrs)	2	2	0.67	0.67
Stem / branch lifetime (yrs)	274	290	120	120
Fine root lifetime (yrs)	4	4	4	4
Coarse root lifetime (yrs)	105	70	120	120

Figure 5.4. Calibrated NPP response curve for the study of Keith *et al.* (2014).



(2) Tasmanian Mountain Ash case study

The presentation of the Dean *et al.* (2012) CAR4D model results for their Tasmanian study was similar to the format presented in Keith *et al.* (2014), allowing a similar calibration process to be used. Also, given the similarity in forest type between the two studies, the Victorian Ash calibration was used as an initial starting point for the Tasmanian study, with the parameters modified only when required in order to match the magnitude of the C stocks and their temporal dynamics. Because ForestHWP does not separate understorey and overstorey biomass (as is done in CAR4D), these two biomass components were summed to yield total biomass. ForestHWP model parameters for fire frequency, harvesting frequency, soil organic C (SOC) loss due to harvesting activity, and HWP decay rates were as described in Dean *et al.* (2012). The parameters associated with the forest biomass dynamics are also given in Table 5.2.

(3) New South Wales north and south coast regional case studies

Because the results presented in Ximenes *et al.* (2012) were regional averages calculated over a number of separate harvesting events (without detailed site-specific dynamics of loss and recovery from specific events), the calibration process was slightly different. First, the forest biomass in the ‘Conservation’ scenario was calibrated to yield the observed above-ground C

stock, and then the harvesting impacts were added with parameters obtained from the values reported in the paper, which included the forest extraction percentages and HWP processing losses (section 2.7 of Ximenes *et al.* 2012). The HWP and fossil fuel product substitution factors (not included in the previous two case studies) were also obtained from the text (section 2.9 of Ximenes *et al.* 2012).

Because of the absence of reliable ground-based estimates of forest NPP, the forest biomass dynamics were calibrated by initially setting NPP to values estimated from the continental modelling study of Haverd *et al.* (2013), which yield values of 10.7 and 9.3 tC/ha/yr for the north and south coast regional sites respectively. In the conservation scenario wildfire was added with an average return time of 75 years, and with living above-ground biomass losses of 10%, 95% losses of fine (leaf) litter, and 50% losses of coarse woody debris. During the calibration process it was necessary to reduce the initial NPP estimates by approximately 20-25% (Table 5.2), in order for the model to satisfactorily replicate the values reported in the original study.

To replicate the regional averaging in this case study, ForestHWP was calibrated to represent the average case, and then 300 replicate simulations were run and the results averaged – with the only difference between the replicate runs being the random timing of the initial fire and harvest events. The conservation scenario was simulated by running the model for 1000 years with harvesting enabled, and then switching the harvesting off to allow forest recovery over the 200 year simulation period.

5.4.2 Verification results

The results of the verification process for all three case studies showed close agreement between the original study and the ForestHWP outputs (Figures 5.5-5.9). For the Victorian case study the peak in total biomass at approximately 40 years post-fire results from the lagged-decay dynamics of the standing dead stags, combined with a rapidly regenerating forest. The forest harvesting dynamics, based on the assumptions and parameter values presented in Keith *et al.* (2014a), shows similarly good agreement (Figure 5.6).

Figure 5.5. Comparison of the changes in ecosystem C storage post-fire between the original study of Keith *et al.* (2014a) (fine line) and the calibrated ForestHWP model (thick line). A fire event is simulated at time 10.

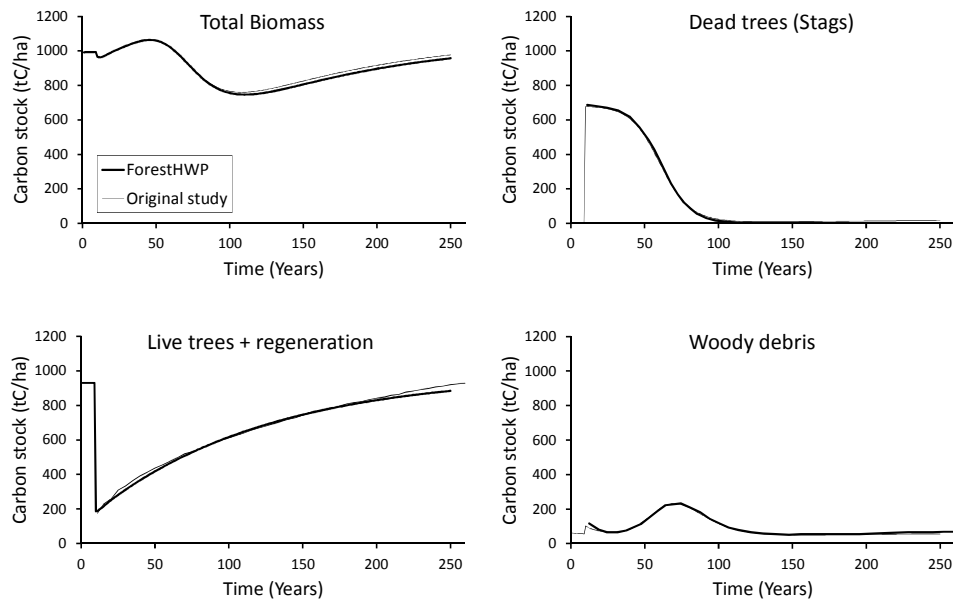
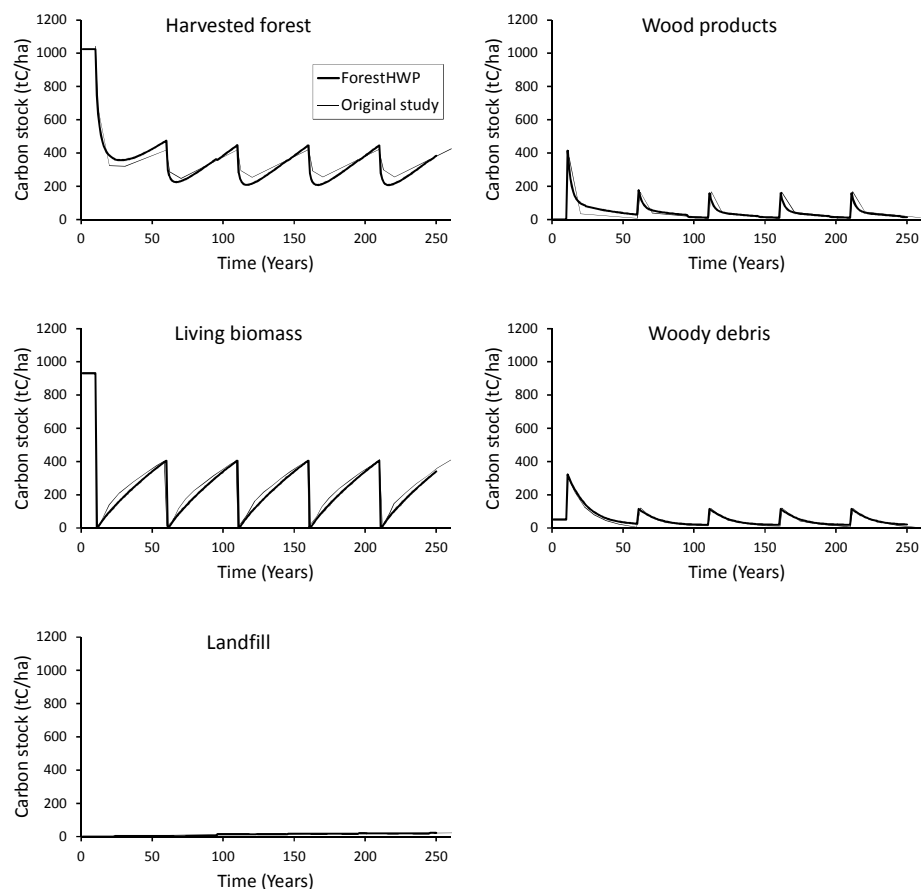


Figure 5.6. Comparison of the changes in ecosystem C storage in response to repeated harvest between the original study of Keith *et al.* (2014a) (fine line) and the calibrated ForestHWP model (thick line). Harvesting events occur at 50-year intervals¹.



¹Note that in this figure the HWP pool follows the fate of the total C removed off-site, and not the actual processed products (H. Keith, *Pers. Comm.*).

The comparison of the Forest HWP results with the results from Dean *et al.* (2012) showed similarly good agreement (Figure 5.8). Of particular significance is the agreement for the simulations that explored the sensitivities of the results to changes in the model parameter defining soil organic C (SOC) losses following harvest, and sensitivity to changes in the HWP longevity parameter (Figure 5.9). This indicates that ForestHWP correctly captures the dynamic nature of the system responses to perturbation (and not just matching the state variable under static parameter values), which is a strong verification test of the underlying model structure.

For the Ximenes *et al.* (2012) study there was less detailed information on which to base the calibration, and because the original study was based on actual historical harvesting events (and not a generalised representation of forest dynamics as in the other two case studies) the degree of fit for the forest biomass was less precise (Figure 5.9). Nevertheless the ForestHWP model was able to capture the major characteristics of both the conservation and harvesting forest biomass dynamics, and captured very closely the C storage in HWP, and the product and fossil fuel substitution benefits.

Figure 5.7. Comparison of the changes in ecosystem C storage during repeated fire (up until year 1200), with subsequent harvesting at 80-year intervals between ForestHWP (thick line) and the CAR4D model results (thin line) as reported in Dean *et al.* (2012).

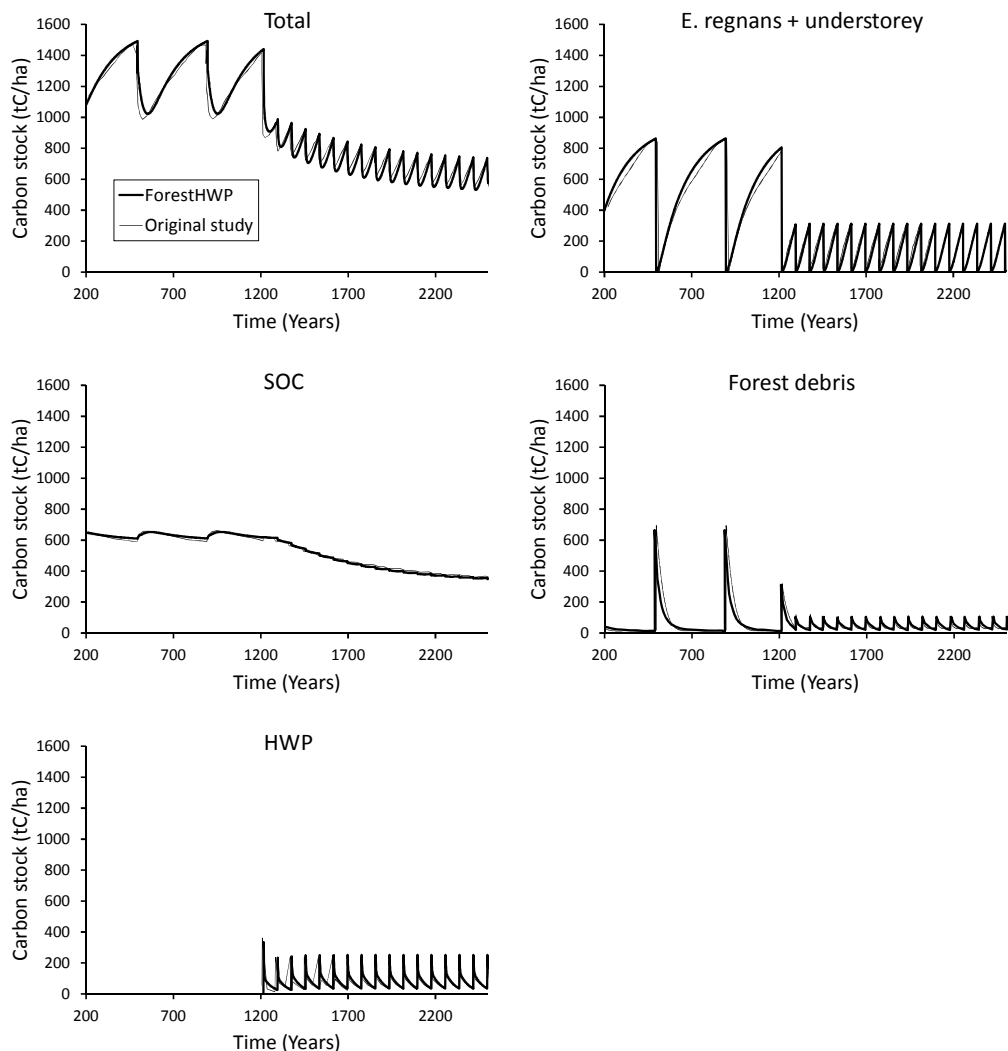


Figure 5.8. Comparison of the changes in ecosystem C storage in response to varying the % loss of soil organic C (SOC) and HWP product longevity between the results of the original study of Dean et al. (2012) (thin line) and ForestHWP (thick line).

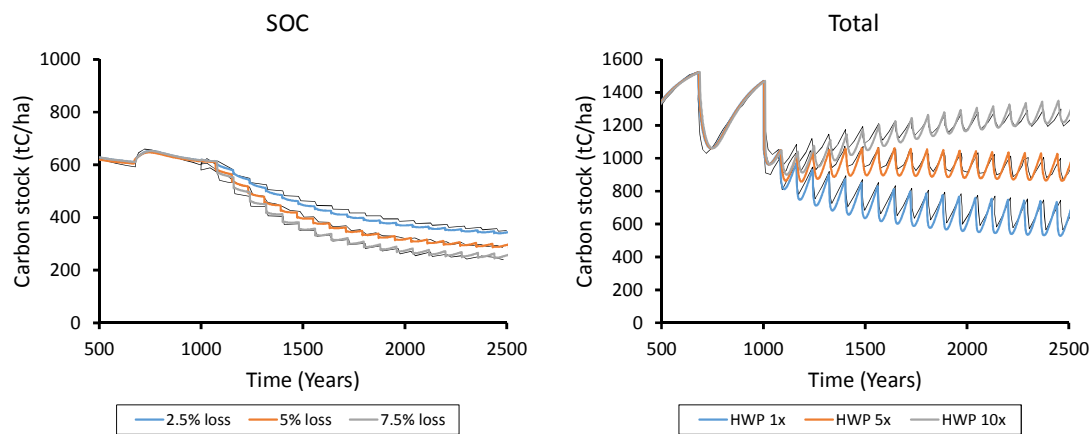


Figure 5.9. Comparison of the changes in ecosystem C storage between ForestHWP (thick line) and the results of Ximenes *et al.* (2012), for the north coast and south coast regional studies.

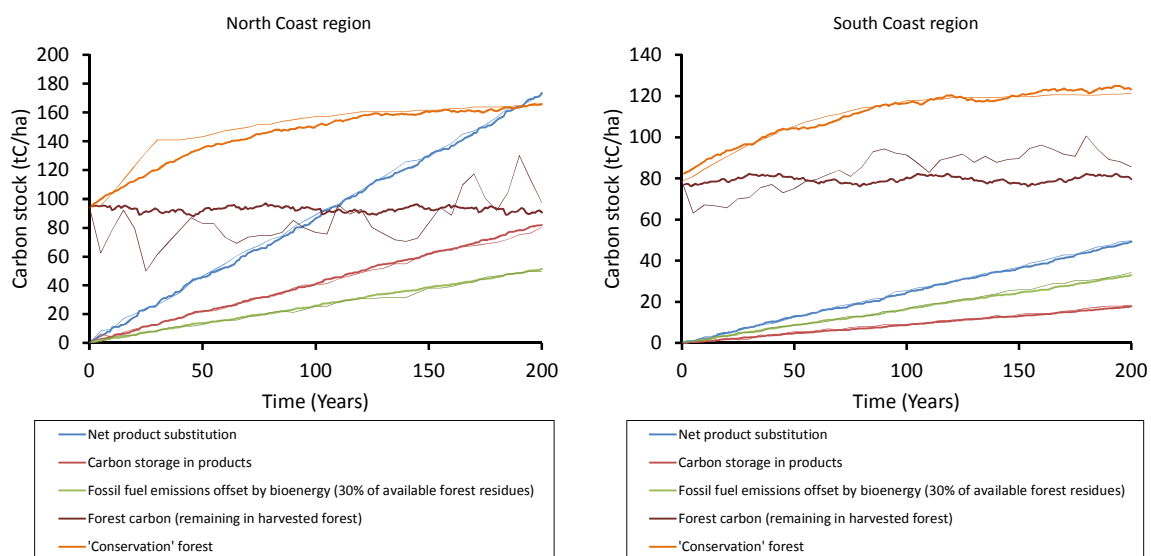
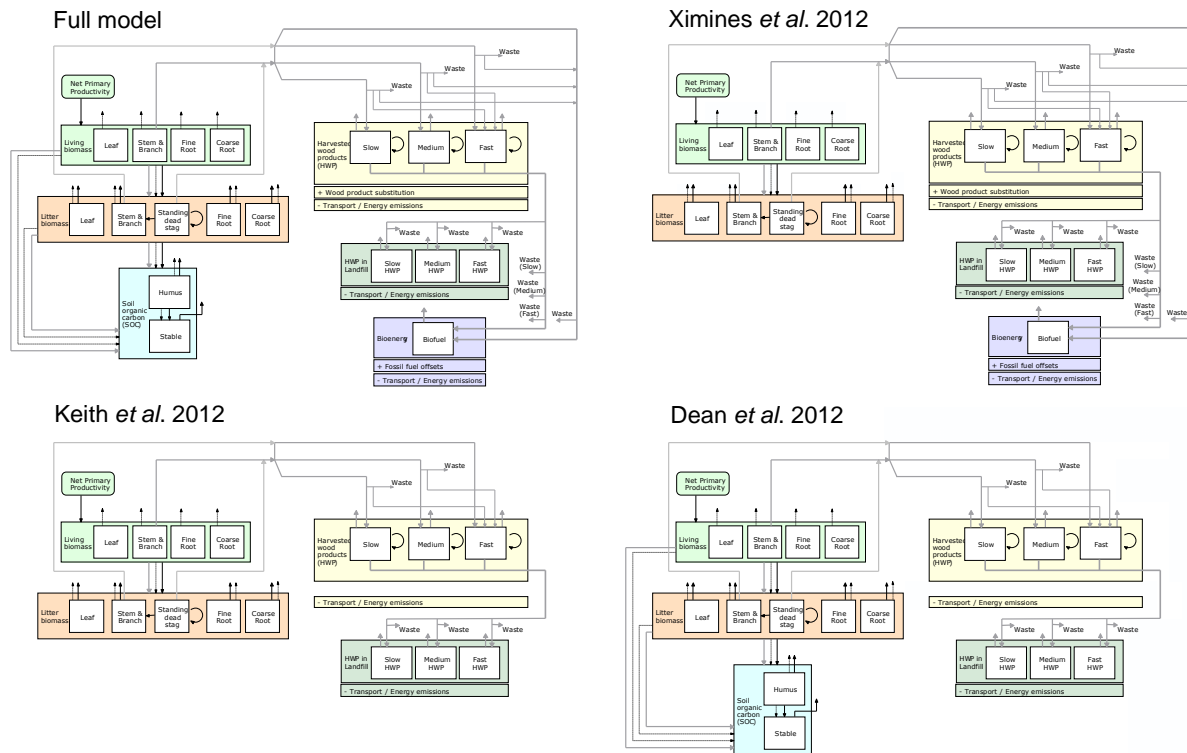


Figure 5.10. Comparison of included/excluded processes across three published studies seeking to quantify the net GHG balance of the full forest-HWP system. Key processes not included in each of the studies (relative to the full model) have been deleted. Note this figure is designed to provide an overview of the various modelling approaches - the details of the full model can be read more clearly in Figure 5.1.



5.5 Discussion

It is significant to note that the three case studies used in the verification of the ForestHWP model differed in their conclusions regarding the role of HWP in contributing to the net GHG balance. Ximenes *et al.* (2012) concluded sustainably managed forests could generate net C benefits over the long term, whereas Dean *et al.* (2012) and Keith *et al.* (2014a) concluded the opposite.

Although a more detailed analysis of these studies (including an assessment of their underlying data and assumptions) is beyond the scope of this current chapter, as a first step towards understanding how the different studies reached opposing conclusions it is instructive to compare the range of processes that were included in each of the studies against the full model (Figure 5.10). Of the three studies Ximenes *et al.* (2012) was the only one to include effects of product substitution and biofuel offsets. The studies of Dean *et al.* (2012) and Keith *et al.* (2014a) both included comprehensive descriptions of forest C dynamics, and also included both HWPs and landfill, but did not include biofuels nor product substitution (note that in Dean *et al.* (2012) landfill is not considered explicitly; rather, the landfill dynamics are embedded within a composite ‘wood products’ pool that combines both in-service HWP and landfill (C. Dean Pers. Comm)). Comparing results across the three studies is therefore hampered by these fundamental differences in the specification of the system boundary. There are also important differences in values of key parameters across these studies, such as the assumed time it takes for products to decay in landfill, and the mix of HWP being generated. To provide an equitable comparison, the range of processes included in the analyses and key parameter values needs to be standardised. This can be achieved in ForestHWP by switching off and on the appropriate model components, and by harmonising parameter values. For example, in the calibration of Keith *et al.* (2014a) in the previous section the soil organic C, biofuels and product substitution sub-models are excluded from the model runs.

The application of ForestHWP to the case studies in this project, and a description of the various scenarios that are explored, is given in the next section of this report.

5.6 Summary

ForestHWP, a new C accounting model for the comprehensive assessment of the full forest – harvested wood products system, has been designed and implemented in software. A summary of the model is provided, and a full description is given in the Appendix 3.

Verifying that the ForestHWP model can accurately replicate previous analyses provides confidence that the equations have been correctly implemented, and that the core functionality has been adequately specified and represented. Demonstrating that the model can adequately capture the dynamics of all the C pools is a particularly strong verification test for ForestHWP, given the inter-connectedness and dependency among the processes (Figure 5.1).

The model was verified by replicating the results from three published studies. Following manual calibration of the model parameters, ForestHWP was able to faithfully replicate the results as presented in those original studies. The verification procedure therefore confirms

the model equations (detailed in the Appendix 3) have been correctly implemented in the computer code, and that the model is both flexible and comprehensive in its representation of the forest and HWP C dynamics.

5.7 References

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Part 6. ForestHWP: Case Studies and Scenarios

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6.1 Section Summary

This section describes how the field data and associated measurements from the three case studies (Victoria Central Highlands Mountain Ash, NSW North Coast Blackbutt, NSW South Coast Silvertop Ash) were incorporated into the ForestHWP model described in part 5, and how ForestHWP was used to explore the implications of various management options on the total system C (C) balance.

For all three case studies model calibration followed the same general procedure, whereby the parameters specifying the forest C stores and fluxes (in the absence of harvesting) were determined by a combination of model inversion (whereby the model parameters were manually adjusted such that model predictions matched the observation) and direct parameter estimation. Once the forest subsystem was calibrated, then the parameters specifying the HWP sub-system, such as harvest forest removals, HWP service lives, and product substitution factors were specified, based on case study observations and literature survey.

Two ‘reference’ or ‘baseline’ scenarios were considered; they were the conservation scenario, where the calibrated models were run with wildfire but without harvesting, and business as usual (‘BAU’) where both harvesting and fire are included, and where the harvesting parameters are those specific to each case study. Within the ForestHWP model wildfire and harvesting events occur stochastically through time, and therefore comparisons between scenarios were made based on the average of 1000 replicate runs. To remove the influence of transient dynamics, scenario comparisons were based on the long-term average behaviour (although comparisons based at years 50 and 100 showed the same trends). For each scenario the model ‘spin up’ involved running the forest-only sub-system for 1000 years with the appropriate wildfire frequency, and then prescribing major known harvesting and/or wildfire events that occurred from 1939 (for Victoria) or 1950 (for the NSW case studies), with the scenarios beginning at a nominal year of 2012.

In addition to the baseline scenarios, eight additional scenarios common to all three case studies were explored. Of these, one explored the potential impacts of moving processing waste and a proportion of products that had reached the end of their service life into residential bioenergy. There were three scenarios that sought to maximise (within constraints) either product recovery (i.e. increased use of processing waste to generate long-lived products), landfill benefits, or biofuel benefits. Finally, four scenarios were designed to investigate the implications of increasing the incidence of fire (two levels), applied to the conservation and BAU scenarios. For the Victorian Central Highlands case study additional scenarios included varying proportions of the current pulp logs to produce bioenergy, and options around the end use of pallets. For the North Coast Blackbutt case study additional scenarios involved the utilisation forest residues for bioenergy and pulp, and increasing the proportion of logs extracted for use as electricity transmission poles. For the South Coast Silver Top Ash case study additional scenarios involved utilising varying proportions of the current pulp logs to produce bioenergy, and implications of using a proportion of forest harvest residues for co-firing with coal for electricity generation.

Comparison of the BAU scenarios with the conservation scenario indicated for the Victorian and New South Wales South Coast case studies that harvested forests yield a greater GHG benefit compared to management for conservation, with this effect driven primarily by the

product substitution factor associated with pulp production, which is relatively high due to the most likely alternative market over the short term (assumed to be one rotation length in the simulations) being the production of pulp from Southeast Asia, which has a significantly higher GHG cost compared to Australian forestry practices. Scenarios that increased utilisation of residues for bioenergy or that increased storage in landfill consistently improved further the benefits of harvested forests relative to the conservation scenario.

For the New South Wales North Coast case study the business as usual scenario had a net GHG balance lower than the conservation scenario. For This case study large amounts of residue were left in the forest; utilisation of this residue for pulp or bioenergy reversed the balance, as did scenarios associated with increasing the longevity of HWPs in the system (either through the production of longer-lived products, or sequestration in landfill). For the North Coast case study there are therefore a number of management options that could feasibly be employed to significantly improve the net GHG balance of the harvested forest, though in some cases to be realised this will require the establishment of a viable market.

Overall, the ForestHWP simulation results suggest the overall C response to harvesting is context-dependent, and that different outcomes are possible depending upon the characteristics of the forest, the harvesting regime, and most importantly the mix of harvested wood products that are produced, their substitution benefits, and their ultimate fate.

6.2 Introduction

A description of the ForestHWP model for integrating the C balance across both the forest and harvested wood product (HWP) systems was given in Part 5. This section describes how the field data and associated measurements from the three case studies (Victoria Central Highlands Mountain Ash, NSW North Coast Blackbutt, NSW South Coast Silvertop Ash) were incorporated into ForestHWP, and how the model was used to explore the implications of various management options on the total system C balance.

The approximate areas over which the case studies can be considered broadly representative with respect to forest type, disturbance regime and management regime are 79,000 ha for Victoria Central Highlands, 36,000 ha for NSW North Coast Blackbutt, and 56,000 ha for South Coast Silvertop Ash.

6.3 Calibration of ForestHWP to the case studies

The case study data was integrated with ForestHWP via the following steps:

- Calibration of the forest C equations of the model to case study field observations and ancillary data (estimates of Net Primary Productivity (NPP), and parameter estimates associated with the wildfire regime).
- Setting of parameters associated with harvesting removals from the forest, processing waste losses, product service lives, bioenergy offset factors and HWP product displacement factors, and post service-life fate etc. All analyses in this chapter were based on the regional production (RG) scenarios described in Part 3
- Exploration of the C implications of current practice (i.e. BAU or ‘Business as Usual’) and a range of possible alternative management scenarios, with a primary focus on the process of harvesting, and the post-harvest treatment of HWPs.

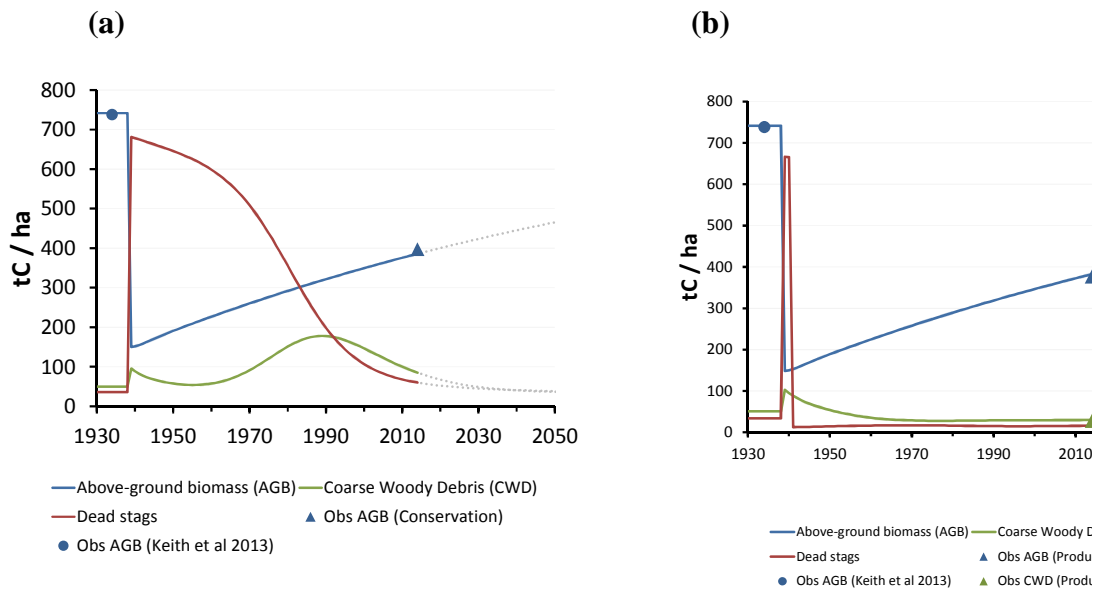
6.3.1 Forest dynamics – Victoria Central Highlands

The Victorian production and conservation sites were both impacted by the 1939 bushfires although with differing intensities, with the conservation site subject to a lower intensity non-stand replacing fire. The production site was also subject to post-fire salvage logging, whereas in the conservation site the natural process of standing dead stag collapse and decay was allowed to proceed. Whilst current living above-ground biomass was similar between the two sites (at approximately 350-400 tC/ha, Part 1), the forest structure was quite different (with a more even-aged and even-sized distribution of individuals in the production site, and with a very high contribution of coarse woody debris (CWD) biomass in the conservation site (Part 1)). Note that, unlike the other two cases studies, destructive sampling of CWD was not possible at the Victorian conservation site, and therefore no direct estimate of CWD biomass are available.

The Keith *et al.* (2014a) model calibration presented in Part 5 was used as the starting point for the calibration of the Victorian case study site. The process involved setting the NPP (9.88 tC/ha/yr), as estimated by the continental model of Haverd *et al.* (2013). The ForestHWP model was then initiated at steady state using data from Keith *et al.* (2014a) (i.e. mature forest in the absence of fire, with an above-ground biomass estimate of approximately 740 tC/ha) and a fire was imposed at 1939, with fire loss parameters given in Keith *et al.* (2014a). For the conservation scenario the site was allowed to recover naturally, and the observed above-ground biomass and CWD estimates were compared at year 2013 – the year at which the field measurements were taken (Figure 6.1b; note that the biomass estimates for the conservation site were based on measurements and allometric models rather than direct weighing). A similar procedure was used for the production site, with the exception that a salvage logging event was imposed five years post-fire (Figure 6.1a).

Using the initial Keith *et al.* (2014a) parameter set the predictions for both the production and conservation sites were close to those observed, and only required minor changes to the NPP allocation fractions (parameters 2-5, Table 6.1) and living biomass longevities (parameters 6-9, Table 6.1). The root parameters were adjusted to ensure a root:shoot ratio of 0.25. Although CWD was not measured for the conservation site, due to the reasons discussed above, the calibrated model suggests a C stock of 100-200 tC/ha.

Figure 6.1. Site calibration for the Victoria case study. (a) Conservation forest site. (b) Production forest site. The circle and triangle symbols show the values for field-based observations, and the lines are the calibrated ForestHWP model predictions.



6.3.2 Forest dynamics – NSW North Coast Blackbutt

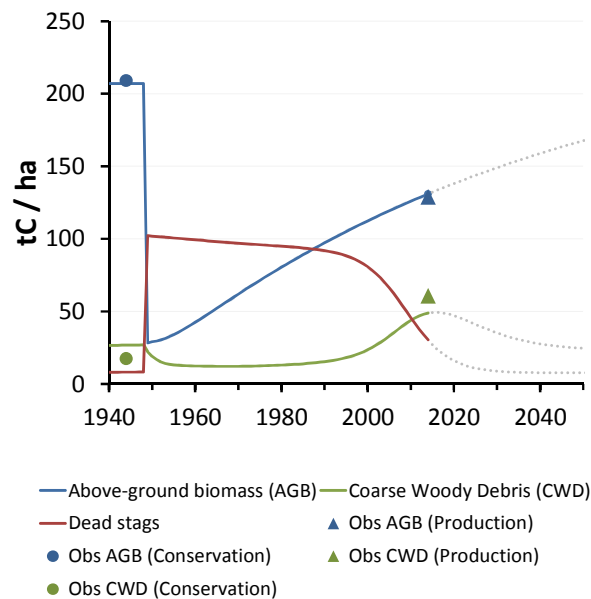
The forest calibration process for the NSW North Coast case study required a slightly different approach due to differences in the history of the production and conservation sites, and differences in the impacts of wildfire. Unlike Victoria (where both sites had undergone a similar disturbance history), the NSW North Coast conservation site was located within an area that had not been significantly disturbed, whereas the production site had undergone harvesting in the 1950s, with subsequent silvicultural management including ringbarking of remaining trees sometime after 1950, and a thinning event in the mid-1990s. Both production and conservation sites were likely to have been subject to wildfire over the last century, however unlike the Victorian Ash forests that suffer significant mortality from fire, the impact of fire on the NSW re-sprouting forests is predominantly limited to the fuel layer, with only moderate mortality assumed (10%).

The model calibration therefore assumed the observed above-ground biomass and CWD for the conservation site was representative of the stocks at the time of initial harvest in the production site (Figure 6.2). NPP was again estimated from Haverd *et al.* (2013), and as per the Victorian site the NPP allocation and longevity parameters were modified in order to attain the observed site biomass values, at the appropriate time since historical harvest (Table 6.1; Figure 6.2).

The timing of the application of the post-1950 treatment at the NSW North Coast production site was uncertain, and therefore the year of application of the ring-barking treatment and the stag decay parameter were adjusted such that the predicted CWD approximated that of the observation at 2013.

Prior to the simulated harvesting event at 1950, the model was run for 1000 years, with wildfires occurring at random with a mean frequency of 75 years.

Figure 6.2. Site calibration for the North Coast case study. The circle and triangle symbols show the values for field-based observations, and the lines are the calibrated ForestHWP model predictions.



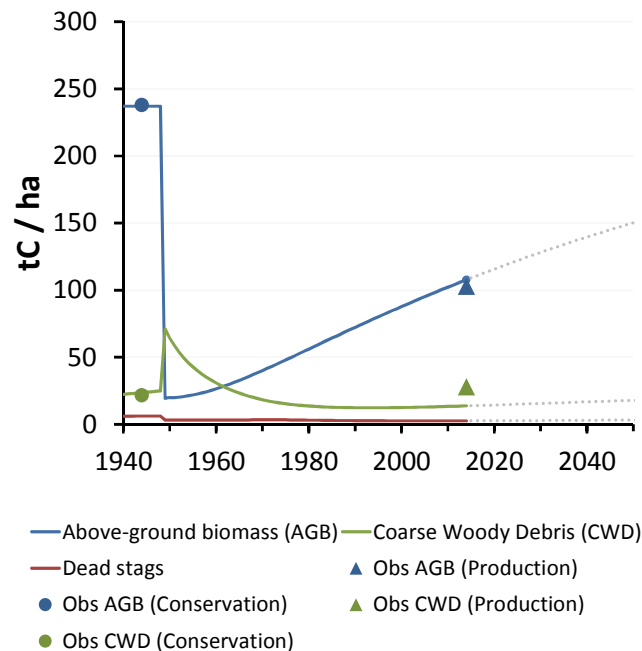
6.3.3 NSW South Coast Silvertop Ash

The history and therefore calibration of the NSW South Coast sites was similar to the NSW North Coast, with a history of harvesting in the 1950s in the production site, although without the subsequent application of ringbarking. As with the previous case studies, only minor adjustment of the default parameters was required in order for the model predicted and observed biomass values to align (Table 6.1, Figure 6.3). As per the NSW North Coast case study, prior to the simulated harvesting event at 1950, the model was run for 1000 years, with wildfires occurring at random with a mean frequency of 75 years.

The model parameters associated with the impacts of harvesting on the forest, and the subsequent processing and fate of the HWP, including the calculation of product substitution factors and bioenergy offsets (Table 6.1, parameters 18-49), were derived from information given in Parts 3 and 4. A brief example is provided below to illustrate how the calculations occur, and how the parameters operate.

Using the NSW South Coast as an example, when the production forest was harvested in 2013 there was 103 tC/ha available for harvest. Of this, 94 tC/ha was actually impacted by the harvesting event ($94.1/102.7 = 0.92$; parameter 18, Table 1). Of this 94.1, 48.0 tC/ha was removed off-site for processing, comprising 34.1 tC/ha to pulp, and 13.9 tC/ha as sawlogs. In terms of ForestHWP parameters this translates to $48.0/94.1 = 0.51$ removed off site as product in the form of pulp ($34.1/94.1 = 0.362$; Parameter 24) and dry board ($13.9/94.1 = 0.148$; parameter 21). Similar calculations are carried through the processing of logs into HWPs, the passage of processing waste and product at the end of its service life into biofuel, and the eventual fate of HWP into either landfill, or release back to the atmosphere.

Figure 6.3. Site calibration for the South Coast case study. The circle and triangle symbols show the values for field-based observations, and the lines are the calibrated ForestHWP model predictions.



6.3.4 Harvest removals, HWP and landfill dynamics, and bioenergy and product substitution offsets.

There are some model parameters that require further explanation. The first is the product substitution factor for pulp, which is 6.4 kg C offset / kg C in pulp (Table 6.1). This factor is applied to the amount of material removed from the forest for pulp processing, i.e. it is applied to the ‘gross’ forest removal fraction. The remaining product substitution factors (i.e. for green and dry board, and poles) are applied to the post-processing product mass, i.e. these factors are applied to the ‘net’ forest removal fraction, after processing waste losses have been deducted. This difference in calculation method between the product streams is due to the difficulties in deriving suitable ‘net’ substitution factors for finished paper products when analysing international datasets (see Part 4). Note that the pulp substitution factor is only applied for a limited time in the simulation scenarios (a single rotation: 75 years for Victoria, 65 years for NSW), rather than over successive rotations. This is in recognition of the transient nature of the markets and the expectation that in the long-term, pulp and paper production in SE Asia will be based on the use of more sustainably sourced biomass.

Table 6.1. Selected Forest HWP parameter values. Parameters 1-13 were calibrated to ensure ForestHWP matched the observed aboveground biomass and coarse woody debris C stocks, and temporal dynamics of those stocks from historical disturbance. The wildfire parameters (14-17) were derived from the literature, and the remaining parameters that define the harvesting impacts and HWP dynamics (18-49) were based on data collected as part of this study, and modified to reflect typical regional production (RG scenario, Parts 3 & 4).

	Parameter	Victoria	NSW South Coast	NSW North Coast
	<i>Forest growth parameters</i>			
1	NPP ² at maturity (tC / ha / yr)	9.88	8.82	9.50
2	Fractional allocation of NPP to leaf	0.35	0.35	0.35
3	Fractional allocation of NPP to stems / branches	0.35	0.25	0.25
4	Fractional allocation of NPP to fine roots	0.10	0.25	0.25
5	Fractional allocation of NPP to coarse roots	0.20	0.15	0.15
6	Leaf lifetime (yrs)	2	0.67	0.67
7	Stem / branch lifetime (yrs)	260	135	105
8	Fine root lifetime (yrs)	4	1	1
9	Coarse root lifetime (yrs) ¹	105	25	25
10	Leaf litter longevity (yrs)	2.75	2	2
11	Stem litter longevity (yrs)	12	13	11
12	Coarse root litter longevity (yrs) ¹	52	25	25
13	Fine root litter longevity (yrs)	2	2	2
	<i>Wildfire parameters</i>			
14	Mean fire return interval (yrs)	112	75	75
15	Living biomass loss fraction	0.8	0.1	0.1
16	Coarse woody debris loss fraction	0.5	0.5	0.5
17	Fine litter loss fraction	0.95	0.95	0.95
	<i>HWP parameters</i>			
18	Rotation length (years)	75	65	65
19	% of coupe affected by harvest	0.98	0.92	0.86
20	% of landscape harvested (adjustment for riparian exclusions zones, etc.)	0.56	0.55	0.60
21	Proportion harvested material left on site	0.204	0.490	0.517
22	Proportion extracted to dry boards	0.160	0.148	0.242
23	Proportion extracted to green boards	0.127	-	0.193

¹These parameters were adjusted to maintain root:shoot ratios of approximately 0.25, and thus to ensure model predictions stayed within realistic bounds given no root harvesting was undertaken. There is limited empirical knowledge of the actual turnover rates of living and dead roots against which to validate these parameters.

²These NPP estimates are broadly consistent with current knowledge of forest growth. For example Havard et al. (2013) report an empirically-derived estimate for Victorian Ash forests of 8-12 tC/ha/yr, and for a range of US forests NPP is approximately 6-8 tC/ha/yr. Although the US estimates are lower relative to those reported in the table, this is to be expected given the better growth conditions at the Australian sites, with warmer temperatures, and non-deciduous species.

Table 6.1 continued.

	Parameter	Victoria	NSW South Coast	NSW North Coast
24	Proportion extracted to poles	-	-	0.048
25	Proportion extracted to pulp	0.509	0.362	-
26	Processing waste fraction loss – dry boards	0.540	0.783	0.663
27	Processing waste fraction to residential bioenergy – dry boards	0.004	0.05	0.037
28	Processing waste fraction to commercial bioenergy – dry boards	0	0.10	0.11
29	Processing waste fraction loss – green boards	0.429	-	0.612
30	Processing waste fraction to residential bioenergy – green boards	0.006	-	0.037
31	Processing waste fraction to commercial bioenergy – green boards	0	-	0.11
32	In-service life loss (%) – dry boards	5	5	5
33	In-service life loss (%) – green boards	5	-	5
34	In-service life loss (%) – poles	-	-	5
35	In-service life loss (%) – pulp	100	100	-
36	Service life – dry boards (yrs)	50 ²	29 ³	50
37	Service life – green boards (yrs)	7.5 ⁴	-	25
38	Service life – poles - (yrs)	-	-	70
39	Service life – pulp (yrs)	1	1	-
40	End service life fraction to landfill – dry boards	0.916	0.872	0.88
41	End service life fraction to residential biofuel – dry boards	0.084	0.128	0.12
42	End service life fraction to landfill – green boards	0	-	0.922
43	End service life fraction to residential bioenergy – green boards	0	-	0.078
44	Dry board product substitution factor (Kg C offset / Kg C manufactured)	0.43	2.07	1.04
45	Green board product substitution factor (Kg C offset / Kg C manufactured)	0.02	-	0.45
46	Poles product substitution factor (Kg C offset / Kg C manufactured)	-	-	1.08
47	Pulp product substitution factor (Kg C offset / Kg C in pulp harvested)	6.4	6.4	-
48	Residential bioenergy fossil fuel offset factor (Kg C offset / Kg C combusted)	0.85	0.85	0.85
49	Commercial bioenergy fossil fuel offset factor (Kg C offset / Kg C combusted)	0.30	0.30	0.30

³Estimated aggregate service life of various wood products (decking, flooring, structural/cladding, etc.). ⁴Estimated service life of pallets.

The second parameter that requires some explanation is the decomposition loss of HWP in landfill, which is assumed to be negligible (and are thus not present in Table 6.1). This contrasts with other factors, such as the default factor used by the Intergovernmental Panel on Climate Change (IPCC) for decomposition of organic materials in landfills of 50% (IPCC 2006). The IPCC factor is designed to be applied to the organic waste stream as a whole, and thus, accounts for both easily digestible materials (e.g. food waste), as well as those considered recalcitrant (e.g. woody material). The IPCC recommends using waste-specific factors if they are available (IPCC 2006). The latest research strongly suggests actual wood decomposition in landfill is minimal, and landfills are in fact a potentially strong C sink for wood and engineered wood products (e.g. Ximenes *et al* 2008, Wang *et al* 2011, Ximenes *et al* 2013, Ximenes *et al* 2015; see also Part 3).

The final set of parameters that require clarification are the GHG emissions associated with harvesting operations, and the transporting and processing of HWP. Whilst these can be explicitly included within ForestHWP, for the three case studies these losses have been subsumed within the calculation of the product substitution factors.

6.4 Description of Scenarios

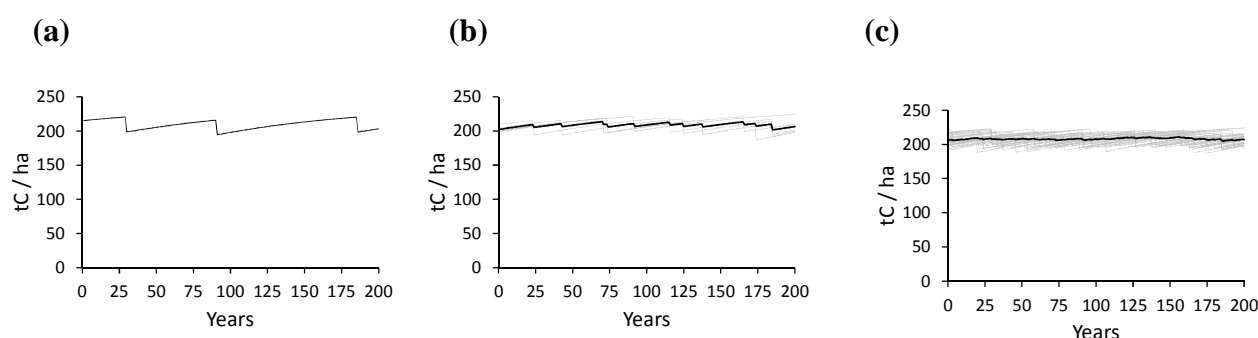
6.4.1 Overall approach.

ForestHWP is essentially a site-based (or non-spatial) model, where the fate of C for a given location is modelled through time. In Figures 6.1-6.3 above the ‘site’ represents the 0.5ha plots measured as part of this study, and in Part 5, Figures 5.5-5.8 the spatial scale is undefined, but represents an ‘average’ plot from a *Eucalyptus regnans* forest in Victoria and Tasmania, respectively.

Whilst such site-based runs are useful for investigating the temporal dynamics of C in response to e.g. fire and harvesting, it makes comparison across studies difficult, as the C stocks at any given time are a function of the time since last disturbance, and of the overall disturbance regime. To account for this in the scenario investigations below, a number of replicate sites that differ only in the timing of disturbance are run, and the average outcome across those runs is used to allow comparison. This has the effect of averaging over the disturbance-induced fluctuations (Figure 6.4), thus standardising for the effects of differences in disturbance regime and allowing comparisons across scenarios to be made. An example of a similar approach is also given in Part 5 Figure 5.9, for the Ximenes *et al.* (2012) study. For all simulations, 2000 replicate runs are used to quantify the overall scenario outcomes, with the only difference between runs being the timing of fire and harvesting events

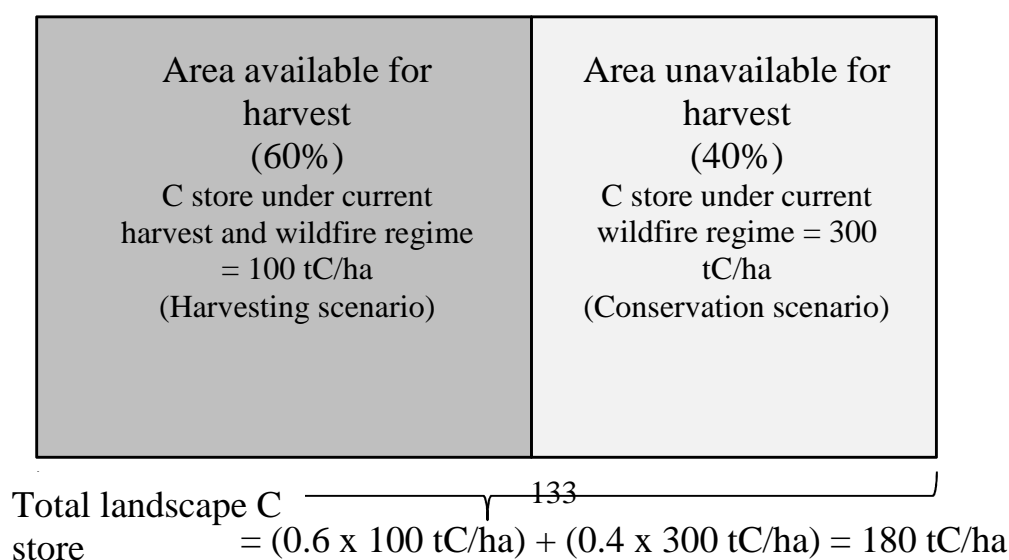
A simple interpretation of this approach is that the scenario results represent the average outcome of a number of patches within a region, and where disturbances occur independently across patches.

Figure 6.4. Summary of how average ForestHWP runs are used to compare scenarios. In (a) a single ‘site’ is simulated, showing three fire events over time that remove biomass, with subsequent recovery during the inter-fire period. Single site runs such as this demonstrate the temporal variability in biomass in response to perturbation, but the biomass at any given time is contingent upon the timing of the last fire, which in the model is determined at random. (b) In order to obtain an estimate of the overall or ‘average’ impacts of the perturbation, and thus to allow different scenarios to be compared, the average over a number of replicate site runs is determined, where replicates differ only in the timing of disturbance. In (b) the average over five such replicate ‘sites’ is shown, with the fluctuations starting to be smoothed out (dark line). In (c) the average over 30 replicate sites is shown, and suggests the expected or ‘average’ biomass under this fire regime is 203 tC/ha.



To further elaborate on this landscape-level interpretation of the scenarios, for each of the case studies an estimate was available of the total regional area that was unavailable for harvest, due to e.g. sensitive environmental areas such as riparian zones. These estimates are given in Table 6.1 as Parameter 20. To take this landscape-scale variability into account the total C store is calculated as the area-weighted average across both the harvested and non-harvested areas (Figure 6.5).

Figure 6.5. Explanation of the landscape-based calculation on which the results are presented. The dark grey area represents the fraction of the total landscape that is available for harvest (60% in this example; Table 1, parameter 19), and the light grey area is the area within the landscape where harvesting is excluded (riparian exclusion zones, etc.). The total landscape C store is the area-weighted average of the C store in the two sub-areas. For simplicity, and in the absence of supporting data, it is assumed the wildfire regime is equivalent across the two sub-regions. For each case study the C store in the unavailable fraction is given by the conservation scenario, and the C store in the available fraction by the various harvesting scenarios (see Table 6.2).



For each scenario ForestHWP was initialised as described in the calibration procedure above, and run until 2013, i.e. as far as the triangle symbols in Figures 6.1-6.3 above. At this point the biofuel offsets, product substitution benefits and the landfill C stocks were set to zero, and then the model advanced for a further 1000 years (in order to provide an estimate of the mean implications of each scenario). Scenario results are reported below for the predicted C at years 50 and 100, and at year 1000 to provide an estimate of the long term average (LTA) behaviour. For the LTA behaviour, the biofuel offsets, product substitution benefits and the landfill C stocks were set to zero at year 800, and thus the values for these quantities reported for LTA represents accumulation of benefits over a 200 year timeframe. Also, for the LTA behaviour, the C stocks were calculated as the average over the final 200 years of the simulation.

6.4.2. Description of Scenarios.

Two of the scenarios can be considered ‘reference’ or ‘baseline’; they are the conservation scenario, where the calibrated models are run with wildfire but without harvesting, and business as usual (‘BAU’) where both harvesting and fire are included, and where the harvesting parameters are those specific to each case study (Table 6.1). These scenarios are described in Table 6.2, Scenarios 1 & 2. Note that In Part 3 the actual mix of products harvested from each 0.5ha coupe was reported (SI parameters), as well as an adjustment to better represent the regional product mix (RG parameters). All of the scenarios involving harvesting are based on the RG parameters, with the exception of one of the NSW North Coast Scenarios (see below).

There are eight other scenarios that were also applied to each of the case studies (Table 6.2, Scenarios 3-10), plus a number of scenarios that were specific to each case study.

Of the eight shared scenarios, one explored the potential impacts of moving some processing waste and products that had reached the end of their service life into residential bioenergy (Table 6.2, Scenario 3). There were three scenarios that sought to maximise product recovery (i.e. increased use of processing waste to generate long-lived products), landfill and biofuel benefits (Table 6.2, Scenarios 4-6, respectively). Finally, four scenarios were designed to investigate the implication of increasing the incidence of fire (two levels), applied to Scenarios 1 and 2 (Table 6.2, Scenarios 7-10).

The “maximise landfill” scenario only considers long-term C storage in HWPs other than paper products. The rationale for the scenarios that aimed to “maximise landfill” and “maximise bioenergy” was based on the fact that the “maximise landfill” scenario in effect maximises physical C storage as an outcome, whereas the “maximise bioenergy” scenario maximises the C benefit by virtue of fossil-fuel displacement. Thus, the “landfill” scenario is independent of fossil-fuel displacement assumptions, whereas the impacts associated with the “maximise bioenergy” scenario are highly dependent on the types of fossil fuel displaced.

No scenarios involving the alteration of harvesting rotation lengths were conducted, in order to retain the focus on HWP management. The sensitivity of model outcomes to rotation length could be considered in the future as part of broader model uncertainty analysis (see section 6.5).

The case study-specific scenarios are described below.

Victoria Central Highlands

The scenarios specific for mountain ash in the Victoria Central Highlands explored utilising varying proportions of the current pulp logs to produce bioenergy (Scenarios 12-13), and options around the end use of pallets produced (Scenarios 14-15). These two products were selected because they both represent currently the largest volumes of products derived from those forests, and the changed management practices modelled represent realistic scenarios. In addition to these scenarios, we also considered the implications of using a proportion of the high volume of forest harvest residues for co-firing with coal for electricity generation (Scenario 11).

North Coast Blackbutt

The scenarios specific for the blackbutt stands on the North Coast of NSW targeted options for utilisation of the large volumes of forest residues currently left in the forest after harvest operations (Scenarios 16-17), and increasing the proportion of logs extracted for use as electricity transmission poles (Scenario 18). Currently there is no market for pulpwood on the North Coast of NSW, resulting in considerably higher volumes of harvest residues for that region compared to the other case studies. Increased use of biomass also has implications for the long-term stocks in the residues in the forest.

Table 6.2. Summary of the scenarios explored.

	<i>Scenario</i>	<i>Description</i>
	<i>Scenarios common to all three case studies</i>	
1	Conservation	Wildfire included but no harvesting.
2	BaU (Business as Usual)	Wildfire included and harvesting as per the observed product recoveries, waste losses, bioenergy usage etc.
3	EoL products and waste to bioenergy	At the end of service life (EoL) all available product is utilised for residential bioenergy, and 70% of dry and green processing waste is utilised for residential bioenergy.
4	Maximise product recovery	70% of dry and green processing waste re-utilised for additional dry product. An example would be use of residues to produce engineered wood products
5	Maximise landfill	At the end of service life all available product is sent to landfill, 70% of dry and green processing waste re-utilised for additional dry product (which eventually ends up in landfill)
6	Maximise bioenergy	At the end of service life all available product is utilised for residential bioenergy, and 100% of dry and green processing waste is utilised for residential bioenergy.
7	Fire x 1.25 (Consv)	Average fire return time in the conservation scenario decreased by 0.8 (=1/1.25)
8	Fire x 1.5 (Consv)	Average fire return time in the conservation scenario decreased by 0.667 (=1/1.5)
9	Fire x 1.25 (BAU)	Average fire return time in the BaU scenario decreased by 0.8 (=1/1.25)
10	Fire x 1.5 (BAU)	Average fire return time in the BaU scenario decreased by 0.667 (=1/1.5)

Table 6.2. Continued

	<i>Scenario</i>	<i>Description</i>
	<i>Victorian case study scenarios</i>	
11	30% forest residue to bioenergy	30% of forest residues left on site utilised for co-firing with coal for electricity generation
12	50% pulp to bioenergy	50% of the material removed from the forest for pulp is instead utilised for residential bioenergy.
13	100% pulp to bioenergy	100% of the material removed from the forest for pulp is instead utilised for residential bioenergy.
14	EoL pallets to landfill	At the end of service life all pallets (=green) are sent to landfill.
15	EoL pallets to bioenergy	At the end of service life all pallets (=green) are utilised for commercial bioenergy.
	<i>NSW North Coast case study scenarios</i>	
16	50% forest residue to bioenergy	50% of forest residues left on site utilised for co-firing with coal for electricity generation
17	50% forest residue to pulp	50% of forest residues left on site utilised for pulp
18	Increase product to poles	The regional product mix (Part 3, section 3.2) used in the simulations is replaced by the product mix as observed at the coupe-level, which increases the proportion of pole manufacture over 6x.
19	<i>NSW South Coast case study scenarios</i>	
20	30% forest residue to bioenergy	30% of forest residues left on site utilised for co-firing with coal for electricity generation
21	50% pulp to bioenergy	50% of the material removed from the forest for pulp is instead utilised for residential bioenergy.
22	100% pulp to bioenergy	100% of the material removed from the forest for pulp is instead utilised for residential bioenergy.

South Coast Silvertop Ash

The scenarios specific for silvertop ash stands in Eden explored utilising varying proportions of the current pulp logs to produce bioenergy (Scenarios 20-21). In addition to these scenarios, we also considered the implications of using a proportion of the high volume of forest harvest residues for co-firing with coal for electricity generation (Scenario 20).

6.5 Scenario results

The Scenario results for each case study are summarised in Tables 6.3-6.5 and Figures 6.6-6.9 below. Across all scenarios there is variability over the first 100 years of the simulation. For example in the Victoria case study there is a tendency for C stocks to initially increase, reaching a peak at approximately year 100, before declining again (as reflected in a lower total C LTA compared with year 100) (Table 6.3). This pattern reflects the non-equilibrium

starting point of the simulations, and in this case is a function of the fire and management histories of the sites post-1939. To exclude such transient dynamics from the comparisons, and thus to ensure that the differences across case studies can be attributed solely to changes in simulated management regime, comparisons amongst the case studies are therefore made based on the LTA summaries. ‘Total C’ refers to the sum of the above ground forest C + the off-site C, the latter of which includes both the C storage of products in-service and in landfill, as well as the accrued bioenergy and HWP substitution benefits (for LTA this represents the accrual over 200 years).

6.5.1 Victoria Central Highlands

The total above ground forest C under the conservation scenario was 523 tC/ha, with an embedded average fire return interval of 112 years (and with the limitation that a fire requires at least 10 years of fuel accumulation for it to carry). Under a regime of ‘business as usual’ harvesting with a 75 year rotation length the expected above ground forest C storage declines to 384 tC/ha; however with the inclusion of the off-site C the total C under business as usual management is 835 tC/ha, representing, over a 200 year period, a 65% increase in C storage, or an equivalent of 312 tC/ha (Table 6.3). The majority of this increase is due to pulp and paper HWP (Figure 6), which has a relatively high product substitution factor (6.4 tC offset / tC pulp). The importance of pulp for the overall C balance of these managed forests is considered in greater detail in the discussion.

Some further observations can be made on these results. First, the reduction of aboveground forest biomass from 523 tC/ha (conservation) to 384 tC/ha (BAU) (Table 6.3) is perhaps not as great as one might expect given fire predominantly redistributes C within the forest, particularly between the living and dead stag pools (Keith *et al.* 2014a), whereas harvesting results in removal of biomass C from the forest system. This can be reconciled by noting within the Victorian case study simulations there was a 50 year buffer period between wildfire and allowable harvest, and therefore when integrated over time (e.g. a 100 year period) the disturbances that occur tend to be either fire or harvesting. In effect, the imposition of wildfire within the harvested forest system leads to an effective increase in rotation length, as some harvesting opportunities are lost due to the occurrence of fire. Second, ForestHWP was specifically designed to include the full system C balance in order to comprehensively account for all stocks and fluxes between the forest and HWP subsystems (Part 5). However a lack of data on roots and soil organic C for the case studies meant that these belowground components were not included in the results summary. Despite this, approximate ‘best guess’ parameters were included for the belowground dynamics, and the implications on the overall results of including belowground C are shown in the last column of Table 6.3 (and Tables 6.4-6.5 for the remaining case studies). These results are displayed in grey text to highlight the large uncertainties surrounding this part of the analysis.

Overall, the results suggest inclusion of the belowground root and soil organic C dynamics results in a reduction in the total C benefit (but only rarely was this sufficient to change the sign of the net C balance from sink to source, or *vice versa*). This implies that under the current parameter settings any increase in the rate of SOC input due to e.g. increased production of litter and CWD from harvesting is insufficient to compensate for the additional losses incurred through the decomposition of the roots of harvested trees, and losses of soil organic C due to reduced aboveground biomass (and thus reduced litter input). There are also, potentially, direct erosional losses associated with the harvesting process itself, which were not included in the modelling. It is important to note that, in order to be conservative, the

default decay rates of coarse root biomass in ForestHWP are relatively fast (with longevity of approximately 25-50 years). However there is significant evidence that the biomass in roots of native hardwood trees that are harvested is much more resistant to decay than typically assumed (Ximenes *et al* 2004), resulting in much slower rates of C loss over time and comparatively smaller losses. Preliminary ForestHWP analyses suggest modification of the root decay rates to reflect these results has the potential to significantly alter the magnitude of the net belowground C balance, further highlighting the sensitivity of the belowground analyses to the assumptions adopted. The overall role of belowground C in the total system C balance remains a large gap in our knowledge.

Of primary interest in this study is the comparison of the business as usual (BAU) results with a range of alternative management options (Table 6.6).

Utilising 50% or 100% the biomass currently used for pulp and paper production for the generation of energy has an overall lower GHG benefit than BAU, with reductions of -144 and -275 tC/ha respectively (representing declines, relative to BAU, of -17% and -33%, respectively) (Table 6.6). Utilising 30% of the forest harvest residues for bioenergy resulted in a modest increase in total C of 10 tC/ha ($\approx 1\%$), as did diverting the use of pallets from mulch to bioenergy at the end of their service life (11 tC/ha, or $\approx 1\%$). Greater benefits were predicted if all end-of-life products and processing waste were utilised for bioenergy (24 tC/ha, or 3%), or if bioenergy opportunities were maximised (33 tC/ha, or 4%). The declines in GHG benefit through the diversion of pulp and paper production to energy production result from the high pulp product substitution factor. Note because the pulp product substitution factor was limited to the first harvest rotation, it is anticipated the application of the product substitution factor associated with the use of biomass for energy and the direct displacement of fossil fuels will have greater application in the long-term. Maximising the product recovery through making long-lived products from residues has a net C benefit of approximately 13 tC/ha (2% over BAU), and maximising storage in landfill yielded a benefit of approximately 40 tC/ha (5% over BAU).

Increasing the frequency of wildfire within the BAU scenario by 1.25x and 1.5x led to declines in total C of -83 tC/ha and -109 tC/ha (or -10 and -13 % of BAU) respectively. This was driven primarily through reduced opportunities for harvesting due to fire disturbance. As expected, increases in the fire frequency lead to overall lower C stocks in the forest sub-system, by approximately 7% (Figure 6.7). These fire scenarios embed a large number of uncertainties and should be regarded as indicative only; these uncertainties and their possible implications for the results are considered further in the Discussion.

Table 6.3. Summary of scenario results for the Victoria case study. ‘Year of assessment’ refers to the predicted C stores (all in t C/ha) at 50 years, 100 years and the long term expectation (LTA). Total C (A+B) refers to all on-site plus off-site C storage, excluding forest below-ground C. The ‘%Change in total C’ is the percentage change of each scenario relative to the conservation scenario.

*Although the belowground C stock estimates are uncertain (see text), they are included at the end of the table in grey font to provide an approximate indication of the full ecosystem/HWP C balance.

Scenario	Year of assessment	Biofuel substitution	Non-pulp HWP substitution	Pulp-HWP substitution	Landfill + in-service HWP storage	(A) Total off-site C	(B) Total above-ground forest C	Total C (A + B)	% change in Total C excluding belowground (roots, root litter, SOC)	% change in Total C including belowground (roots, root litter, SOC)*
<i>Baseline scenarios</i>										
Conservation	50	0.0	0.0	0.0	0.0	0.0	517.9	517.9		
	100	0.0	0.0	0.0	0.0	0.0	526.4	526.4		
	LTA	0.0	0.0	0.0	0.0	0.0	522.8	522.8		
BAU Harvesting (75 year rotation)	50	0.1	4.2	415.9	11.9	432.1	410.6	842.7	62.7	30.8
	100	0.8	7.8	609.4	18.0	636.0	381.2	1017.1	93.2	47.1
	LTA	1.5	11.2	415.1	23.8	451.6	383.6	835.2	59.7	27.1
<i>Biofuel Scenarios</i>										
30% of forest residue to bioenergy	50	6.6	4.2	411.5	11.7	434.0	411.2	845.2	63.2	31.1
	100	12.9	7.7	604.4	17.5	642.5	381.2	1023.7	94.5	47.8
	LTA	18.8	11.1	408.0	23.6	461.6	383.9	845.5	61.7	28.2
50% pulp to residential biofuel	50	27.8	4.2	208.4	11.1	251.5	410.2	661.7	27.8	10.8
	100	51.3	7.7	302.7	17.4	379.1	382.4	761.6	44.7	19.2
	LTA	72.6	10.9	201.1	22.9	307.5	383.4	691.0	32.2	11.3
100% pulp to residential biofuel	50	54.8	4.2	0.0	10.3	69.3	413.0	482.3	-6.9	-9.0
	100	100.2	7.6	0.0	16.4	124.2	382.9	507.2	-3.6	-8.5
	LTA	143.0	10.8	0.0	22.4	176.2	384.4	560.6	7.2	-2.9
All end-of-life products, and all processing waste to residential biofuels	50	17.6	4.3	421.9	11.9	455.7	408.7	864.4	66.9	33.1
	100	40.9	7.8	608.7	9.2	666.6	381.7	1048.3	99.2	50.5
	LTA	61.6	10.8	394.7	7.3	474.4	384.5	858.9	64.3	29.7

Table 6.3. Continued.

Scenario	Year of assessment	Biofuel substitution	Non-pulp HWP substitution	Pulp-HWP substitution	Landfill + in-service HWP storage	(A) Total off-site C	(B) Total above-ground forest C	Total C (A + B)	% change in Total C excluding belowground (roots, root litter, SOC)	% change in Total C including belowground (roots, root litter, SOC)*
<i>Product scenarios</i>										
End of service life pallet mulch	50	0.1	3.8	371.4	17.6	392.9	423.4	816.3	57.6	28.4
to landfill	100	0.7	7.4	597.1	32.4	637.6	385.5	1023.1	94.4	48.1
	LTA	1.4	11.0	409.5	47.8	469.7	383.8	853.5	63.2	29.1
End of service life pallet mulch	50	2.3	4.1	404.2	11.2	421.9	412.2	834.1	61.0	30.0
to commercial bioenergy	100	5.5	7.6	594.7	17.5	625.2	380.7	1005.9	91.1	45.8
	LTA	8.5	11.2	419.1	24.1	462.9	383.2	846.1	61.8	28.3
Green processing waste to dry product	50	0.1	9.0	389.4	22.7	421.2	415.6	836.8	61.6	30.5
(Max product)	100	1.5	16.1	559.2	35.5	612.3	393.5	1005.9	91.1	46.6
	LTA	3.0	24.5	384.6	51.7	463.8	384.5	848.2	62.2	28.6
Max product to landfill	50	0.1	9.5	411.1	31.4	452.1	409.3	861.4	66.3	32.8
	100	0.1	16.9	557.4	53.8	628.3	379.4	1007.7	91.4	46.0
	LTA	0.2	24.8	389.2	78.8	493.0	382.2	875.2	67.4	31.4
Max product to residential bioenergy	50	20.4	3.9	385.7	10.9	420.9	417.2	838.2	61.8	30.6
	100	46.5	7.3	587.3	9.9	651.0	385.2	1036.2	96.9	49.4
	LTA	72.5	10.8	391.6	7.7	482.6	385.5	868.1	66.0	30.8

Table 6.3. Continued.

Scenario	Year of assessment	Biofuel substitution	Non-pulp HWP substitution	Pulp-HWP substitution	Landfill + in-service HWP storage	(A) Total off-site C	(B) Total above-ground forest C	Total C (A + B)	% change in Total C excluding belowground (roots, root litter, SOC)	% change in Total C including belowground (roots, root litter, SOC)*
<i>Fire scenarios</i>										
Increase fire frequency by 1.25x	50	0.1	3.7	364.0	9.6	377.4	419.2	796.6	53.8	26.1
(Applied to BAU Harvesting))	100	0.7	7.0	552.2	16.8	576.7	383.1	959.8	82.3	41.0
	LTA	1.2	9.6	333.4	20.0	364.2	388.1	752.4	43.9	18.5
Increase fire frequency by 1.5x	50	0.1	3.5	340.1	9.5	353.1	421.8	774.9	49.6	23.9
(Applied to BAU Harvesting))	100	0.6	6.7	531.5	14.5	553.4	382.3	935.6	77.8	38.4
	LTA	1.1	8.6	310.6	18.8	339.2	387.2	726.3	38.9	15.7
Increase fire frequency by 1.25x	50	0.0	0.0	0.0	0.0	0.0	510.9	510.9	-1.4	-5.9
(Applied to Conservation)	100	0.0	0.0	0.0	0.0	0.0	506.9	506.9	-3.7	-8.5
	LTA	0.0	0.0	0.0	0.0	0.0	504.0	504.0	-3.6	-26.0
Increase fire frequency by 1.5x	50	0.0	0.0	0.0	0.0	0.0	512.2	512.2	-1.1	-5.5
(Applied to Conservation)	100	0.0	0.0	0.0	0.0	0.0	505.2	505.2	-4.0	-8.7
	LTA	0.0	0.0	0.0	0.0	0.0	489.2	489.2	-6.4	-27.7

Figure 6.6. Summary of the long-term average results for the Victorian case study, for each of the major C balance components. An explanation of the scenarios is given in Table 6.2.

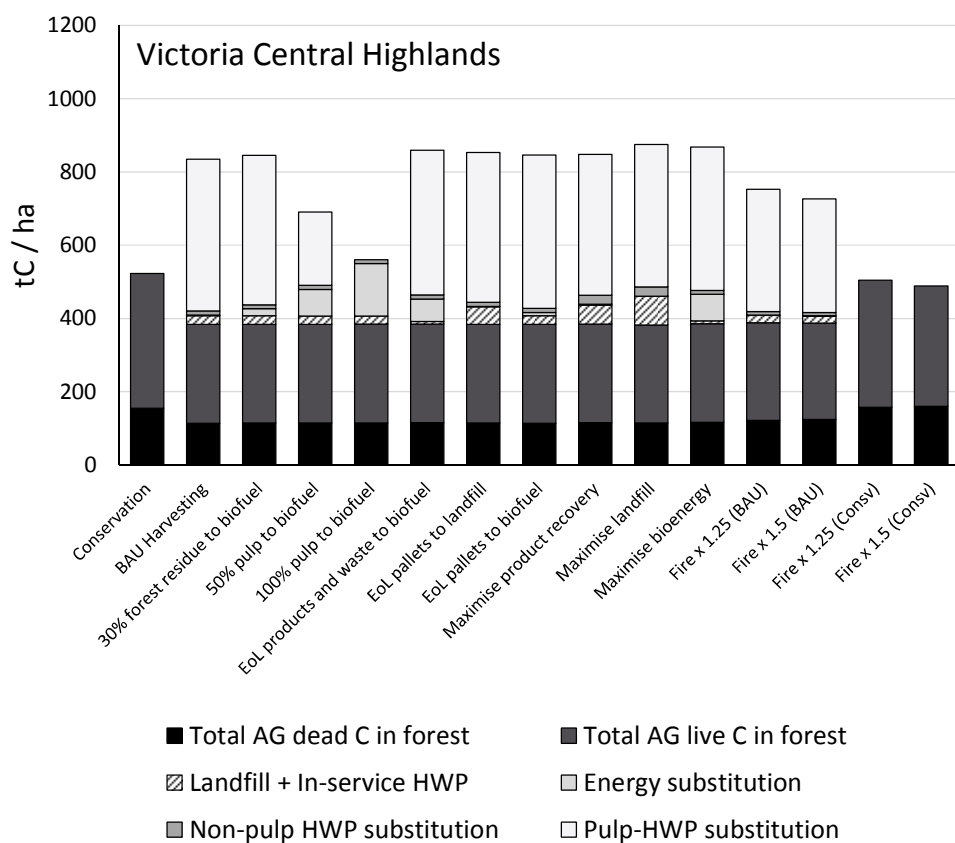
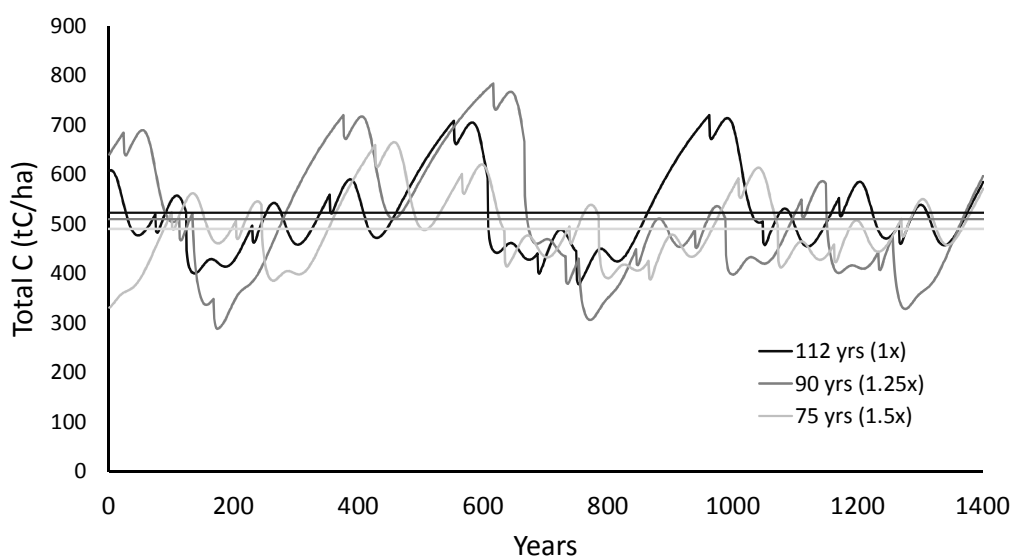


Figure 6.7. Long-term average (LTA) impacts of changing wildfire frequency on total C for the Victoria Central highlands for the conservation case study. Inter-fire intervals (IFI) are 112 years (default), 90 years (1.25x) and 75 years (1.5x). Fire occurs randomly through time with a probability $1/\text{IFI}$, and minimum time since last fire = 10 years.



6.5.2. North Coast Blackbutt

In contrast to the Victorian Central Highlands case study, where the accrued GHG benefits for the BAU harvesting scenario were significantly greater than the corresponding conservation scenario, in the North Coast Blackbutt case study the BAU scenario was approximately 12% lower (248 tC/ha vs. 219 tC/ha, Table 6.4 & Figure 6.8). However the BAU harvesting scenario is affected by the current absence of a market for pulpwood and the delay in the introduction of a bioenergy market for that resource. The expectation is that this change will take place in the short to medium term, resulting in BAU scenarios closer to the “50% of forest residue to bioenergy” or “50% of forest residue to pulp” scenarios. These scenarios generated GHG benefits of 38 tC/ha (18% above BAU) and 105 tC/ha (48% above BAU) respectively, and were also both in excess of the conservation scenario. The contribution of pulp product in these projections is again notable.

The “Maximising product recovery” scenario, by making long-lived products from residues has a net C benefit of approximately 63 t C/ha (29% increase over BAU). This benefit is similar to that of the “maximising landfill” scenario (64 tC/ha; 29% increase over BAU) and “maximising bioenergy” scenario (61 tC/ha; 28% increase over BAU). Utilising processing residues and end-of-life products for bioenergy yielded a lower benefit, of 26 tC/ha over BAU (12%).

Extraction of logs for use as electricity transmission poles rather than sawlogs for sawn products also has a substantial impact, resulting in an additional net C benefit of approximately 68 t C/ha (31 % increase over BAU). This is primarily due to the large proportion of transmission poles that are deposited in landfill at the end of their service life (0.92; Table 6.1, parameter 19), which generates a long-term storage pool for this product.

The effect of increasing the frequency of wildfire was a small decline in total C (Table 6.5, Figure 6.8). The effect is less than for Victoria due to much lower fire severity impacts on the forest biomass pools (Table 6.1, parameter 15) and the removal of a buffering period between a wildfire event and a harvesting event (on the assumption wildfires in the predominantly resprouting NSW forests have a minor impact on the mature trees, and thus a minor impact on the ability to harvest soon after wildfire).

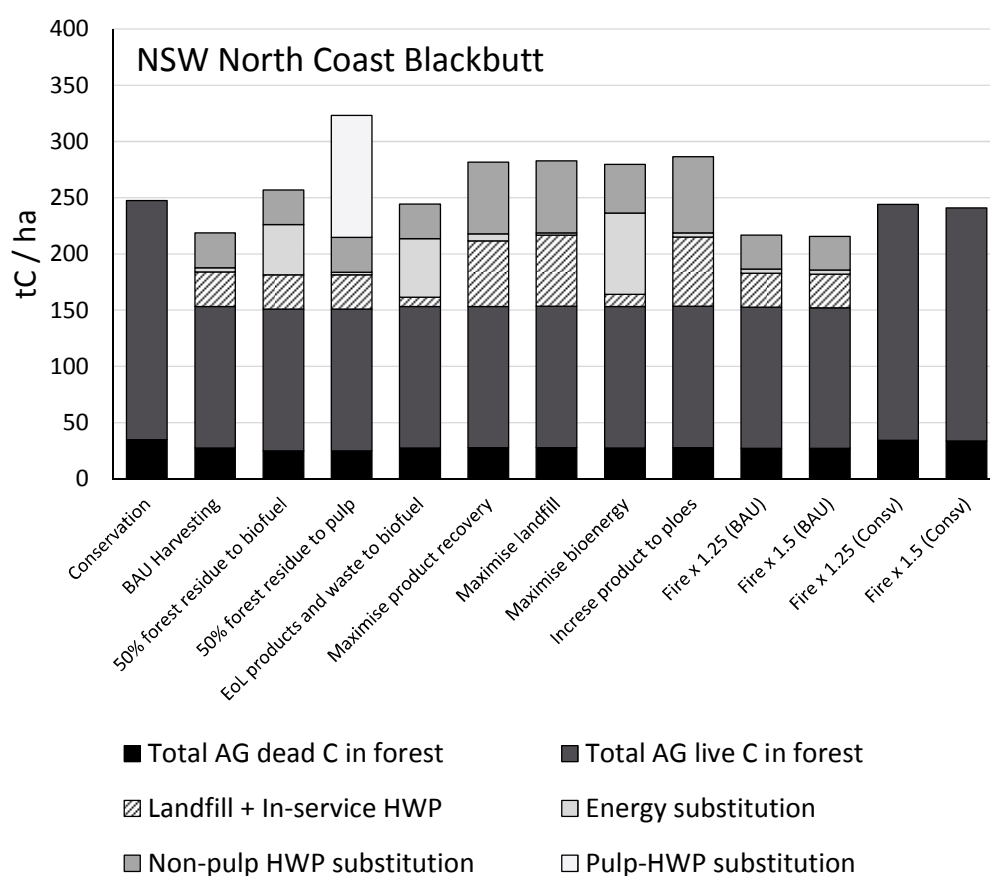
Table 6.4. Summary of scenario results for the NSW North Coast case study. See Table 3 caption for details.

Scenario	Year of assessment	Biofuel substitution	Non-pulp HWP substitution	Pulp-HWP substitution	Landfill + in-service HWP storage	(A) Total off-site C	(B) Total above-ground forest C	Total C (A + B)	% change in Total C excluding belowground (roots, root litter, SOC)	% change in Total C including belowground (roots, root litter, SOC)*
<i>Baseline scenarios</i>										
Conservation	50	0.0	0.0		0.0	0.0	253.7	253.7		
	100	0.0	0.0		0.0	0.0	250.7	250.7		
	LTA	0.0	0.0		0.0	0.0	247.5	247.5		
BAU Harvesting (65 year rotation)	50	1.2	10.1		11.0	22.4	170.7	193.1	-23.9	-8.9
	100	2.5	19.0		19.3	40.9	156.4	197.3	-21.3	-8.8
	LTA	3.8	30.7		30.5	65.1	153.5	218.5	-11.7	-7.0
<i>Biofuel Scenarios</i>										
50% of forest residue to Bioenergy	50	13.2	9.2		10.0	32.4	162.3	194.7	-23.3	-9.3
	100	25.5	17.7		18.0	61.3	153.1	214.3	-14.5	-7.1
	LTA	44.4	30.9		30.8	106.1	150.8	256.9	3.8	-2.4
50% of forest residue to pulp	50	0.4	9.1	100.5	10.6	120.7	163.2	283.9	11.9	3.7
	100	1.3	17.6	131.8	18.4	169.1	153.3	322.4	28.6	8.6
	LTA	2.6	30.7	108.6	30.6	172.4	150.8	323.2	30.6	7.3
All end-of-life products and all processing waste to bioenergy	50	12.5	10.5		10.0	33.0	168.9	201.9	-20.4	-7.7
	100	29.4	19.1		9.0	57.5	156.3	213.8	-14.7	-6.4
	LTA	52.3	30.8		7.8	90.9	153.5	244.4	-1.3	-3.2
<i>Product scenarios</i>										
Waste to product (Max product)	50	1.2	21.4		21.7	44.3	169.9	214.2	-15.6	-5.9
	100	3.5	39.3		36.8	79.6	156.7	236.3	-5.7	-3.1
	LTA	6.1	63.8		58.2	128.1	153.5	281.5	13.8	2.3

Table 6.4. Continued.

Scenario	Year of assessment	Biofuel substitution	Non-pulp HWP substitution	Pulp-HWP substitution	Landfill + in-service HWP storage	(A) Total off-site C	(B) Total above-ground forest C	Total C (A + B)	% change in Total C excluding belowground (roots, root litter, SOC)	% change in Total C including belowground (roots, root litter, SOC)*
Max product to landfill	50	1.1	21.7		22.1	44.9	169.2	214.1	-15.6	-5.9
	100	1.7	39.6		39.2	80.5	156.5	237.1	-5.4	-3.0
	LTA	2.1	63.9		62.9	129.0	153.6	282.6	14.2	2.4
Max product to bioenergy	50	16.3	14.8		13.9	45.1	169.0	214.1	-15.6	-5.9
	100	40.0	26.9		12.3	79.3	156.6	235.8	-5.9	-3.2
	LTA	72.1	43.4		10.7	126.2	153.5	279.6	13.0	2.0
Adopt site product spread, to yield a greater proportion of poles	50	1.2	22.2		21.0	44.5	170.9	215.4	-15.1	-5.7
	100	2.4	41.7		38.2	82.3	156.5	238.8	-4.7	-2.8
	LTA	3.8	67.5		61.4	132.7	153.6	286.4	15.7	3.0
<i>Fire scenarios</i>										
Increase fire frequency by 1.25x (Applied to BAU Harvesting))	50	1.2	10.1		11.0	22.2	169.5	191.7	-24.5	-9.5
	100	2.5	18.7		19.0	40.2	156.0	196.1	-21.8	-9.3
	LTA	3.7	30.2		30.1	64.0	152.8	216.8	-12.4	-7.6
Increase fire frequency by 1.5x (Applied to BAU Harvesting))	50	1.2	10.2		11.2	22.6	168.1	190.6	-24.9	-9.9
	100	2.5	18.5		18.8	39.8	155.7	195.5	-22.0	-9.7
	LTA	3.7	29.9		29.8	63.4	152.1	215.6	-12.9	-8.1
Increase fire frequency by 1.25x (Applied to Conservation)	50	0.0	0.0		0.0	0.0	251.3	251.3	-1.0	-0.6
	100	0.0	0.0		0.0	0.0	247.6	247.6	-1.3	-0.8
	LTA	0.0	0.0		0.0	0.0	243.9	243.9	-1.4	-0.9
Increase fire frequency by 1.5x (Applied to Conservation)	50	0.0	0.0		0.0	0.0	249.2	249.2	-1.8	-1.2
	100	0.0	0.0		0.0	0.0	245.2	245.2	-2.2	-1.4
	LTA	0.0	0.0		0.0	0.0	241.1	241.1	-2.6	-1.6

Figure 6.8. Summary of the long-term average results for the NSW North Coast case study, for each of the major C balance components. An explanation of the scenarios is given in Table 2.



6.5.3. South Coast Silvertop Ash

For the South Coast Silvertop Ash case study the BAU scenario generated a C benefit slightly in excess of the conservation scenario (289 tC/ha vs 277 tC/ha) (Table 6.6 & Figure 6.9). Similar to the Victorian Central Highlands case study, this benefit was driven primarily by the substitution factor for pulp. This is also reflected in the scenarios where 50% or 100% of the pulp product stream is instead utilised for bioenergy; when compared to the BAU scenario these scenarios generated a lower GHG benefit of -33 tC/ha (11% lower) and -66 tC/ha (23% lower), respectively.

Use of 30% of the forest harvest residues for bioenergy results in an additional net C benefit of approximately 17 tC/ha (6% increase over BAU). The “Maximising bioenergy” scenario and utilising processing residues and end-of-life products for bioenergy yielded benefits of a similar magnitude (13 tC/ha over BAU (4%) and 9 tC/ha over BAU (3%), respectively).

The “Maximising landfill” and “maximising product recovery” scenarios have similar impacts, with net C benefits of approximately 31 t C/ha (11 % increase over BAU) and 30 t C/ha (10 % increase over BAU), respectively. The response to changing the wildfire frequency for the South Coast case study was very similar to that the North Coast case study, due to the similarities in forest types and wildfire regimes (Table 6.6).

Table 6.5. Summary of scenario results for the NSW South Coast case study. See Table 3 caption for details.

Scenario	Year of assessment	Biofuel substitution	Non-pulp HWP substitution	Pulp-HWP substitution	Landfill + in-service HWP storage	(A) Total off-site C	(B) Total above-ground forest C	Total C (A + B)	% change in Total C excluding belowground (roots, root litter, SOC)	% change in Total C including belowground (roots, root litter, SOC)*
<i>Baseline scenarios</i>										
Conservation	50	0.0	0.0	0.0	0.0	0.0	285.9	285.9		
	100	0.0	0.0	0.0	0.0	0.0	282.3	282.3		
	LTA	0.0	0.0	0.0	0.0	0.0	276.7	276.7		
BAU Harvesting (65 year rotation) Pulp DF = 6.4	50	0.5	3.5	121.0	1.9	126.8	178.6	305.4	6.8	0.2
	100	1.1	6.4	159.7	2.9	170.0	165.0	335.1	18.7	4.3
	LTA	1.7	9.8	111.2	4.2	126.9	161.5	288.4	4.3	-4.6
<i>Biofuel Scenarios</i>										
30% of forest residue to bioenergy	50	6.8	3.4	116.9	1.8	128.8	178.1	306.9	7.3	0.0
	100	13.0	6.3	160.4	2.8	182.5	164.0	346.6	22.8	5.6
	LTA	20.2	9.9	110.9	4.2	145.1	160.2	305.4	10.4	-2.5
50% pulp to bioenergy	50	8.2	3.3	58.1	1.6	71.2	180.4	251.7	-12.0	-8.0
	100	15.7	6.3	79.5	2.7	104.2	165.8	270.0	-4.4	-5.7
	LTA	24.5	9.9	55.6	4.1	94.1	161.5	255.6	-7.6	-9.7
100% pulp to bioenergy	50	16.4	3.4	0.0	1.5	21.3	179.4	200.7	-29.8	-15.8
	100	30.5	6.4	0.0	2.6	39.5	165.7	205.2	-27.3	-15.6
	LTA	47.3	9.9	0.0	4.0	61.2	161.5	222.7	-19.5	-14.8
All end-of-life products, and all processing waste to bioenergy	50	4.2	3.4	117.4	1.2	126.1	179.9	306.1	7.1	0.3
	100	8.9	6.4	160.8	0.9	176.9	165.4	342.3	21.2	5.4
	LTA	14.0	9.9	111.2	0.9	136.0	161.6	297.5	7.5	-3.2
Waste to product (Max product)	50	1.2	10.2	119.1	4.8	135.3	179.2	314.5	10.0	1.6
	100	2.6	19.2	159.9	8.2	189.8	165.2	355.0	25.8	7.4
	LTA	4.0	29.5	110.7	12.2	156.4	161.6	318.0	14.9	0.0

Table 6.5. Continued.

Scenario	Year of assessment	Biofuel substitution	Non-pulp HWP substitution	Pulp-HWP substitution	Landfill + in-service HWP storage	(A) Total off-site C	(B) Total above-ground forest C	Total C (A + B)	% change in Total C excluding belowground (roots, root litter, SOC)	% change in Total C including belowground (roots, root litter, SOC)*
Max product to landfill	50	1.0	10.2	119.1	5.0	135.3	179.3	314.6	10.1	1.6
	100	1.8	19.2	160.0	9.0	190.0	164.9	355.0	25.7	7.3
	LTA	2.8	29.6	111.4	13.8	157.5	161.5	319.0	15.3	0.1
Max product to bioenergy	50	5.6	3.4	119.7	1.3	130.1	179.4	309.5	8.3	0.8
	100	11.4	6.4	159.2	0.8	177.7	165.6	343.3	21.6	5.6
	LTA	17.8	9.8	111.1	0.8	139.6	161.6	301.2	8.9	-2.7
<i>Fire scenarios</i>										
Increase fire frequency by 1.25x (Applied to BAU Harvesting))	50	0.5	3.3	116.6	1.8	122.2	179.3	301.5	5.5	-0.7
	100	1.0	6.3	157.9	2.8	168.0	165.2	333.2	18.0	3.8
	LTA	1.7	9.8	109.6	4.2	125.2	161.4	286.6	3.6	-5.1
Increase fire frequency by 1.5x (Applied to BAU Harvesting))	50	0.5	3.4	117.9	1.7	123.5	177.8	301.4	5.4	-1.0
	100	1.0	6.3	156.5	2.8	166.5	164.8	331.3	17.4	3.2
	LTA	1.6	9.7	109.4	4.2	125.0	161.1	286.0	3.4	-5.3
Increase fire frequency by 1.25x (Applied to Conservation)	50	0.0	0.0	0.0	0.0	0.0	282.0	282.0	-1.4	-0.9
	100	0.0	0.0	0.0	0.0	0.0	279.0	279.0	-1.2	-0.8
	LTA	0.0	0.0	0.0	0.0	0.0	272.7	272.7	-1.4	-0.9
Increase fire frequency by 1.5x (Applied to Conservation)	50	0.0	0.0	0.0	0.0	0.0	279.1	279.1	-2.4	-1.6
	100	0.0	0.0	0.0	0.0	0.0	275.1	275.1	-2.6	-1.7
	LTA	0.0	0.0	0.0	0.0	0.0	269.4	269.4	-2.6	-1.7

Figure 6.9. Summary of the long-term average results for the NSW South Coast case study, for each of the major C balance components. An explanation of the scenarios is given in Table 6.2.

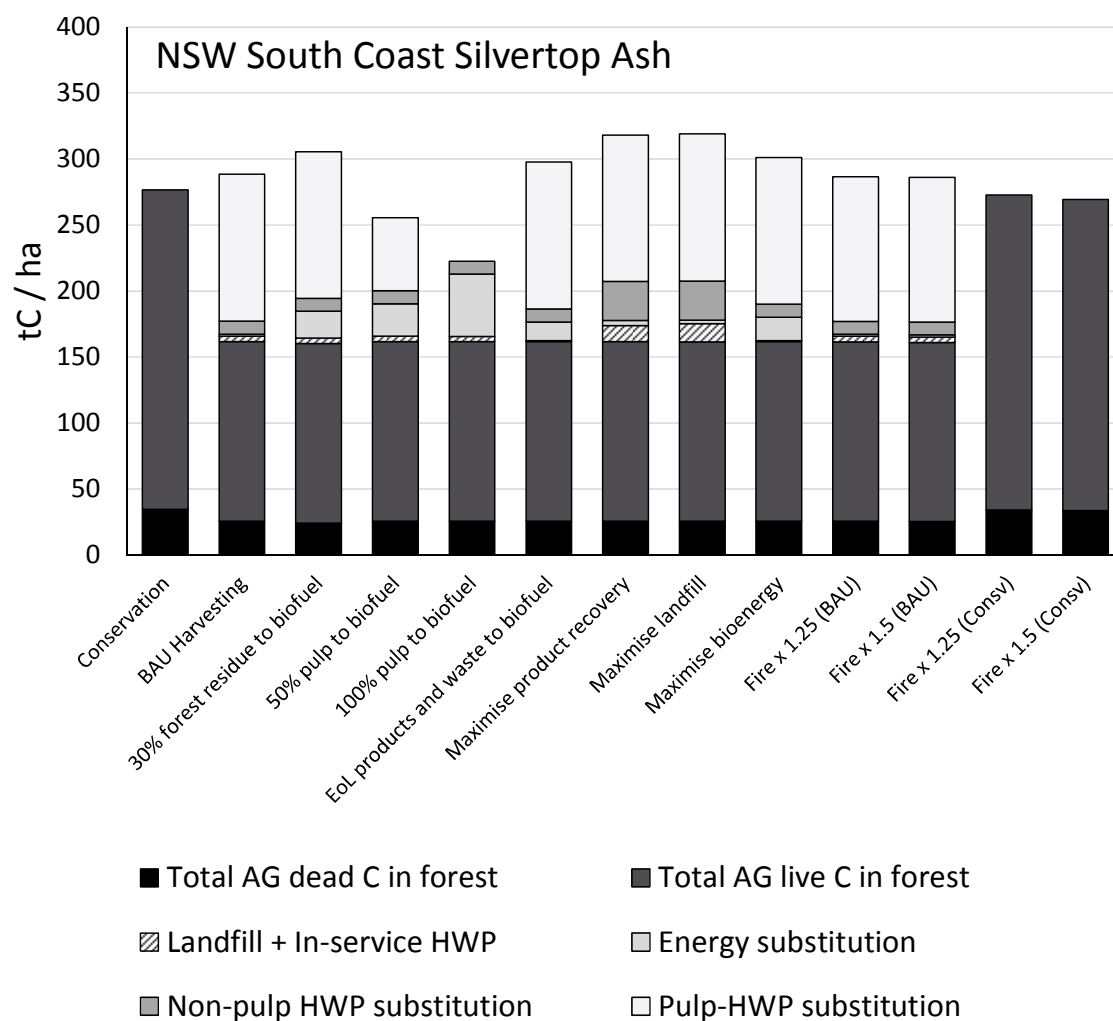


Table 6.6. Comparison of the implications of changed management with the business as usual (BAU) scenario. The BAU value is the LTA total above-ground + HWP C storage, and includes the total bioenergy offset and substitution benefits summed over 200 years. The values for the *Changed management scenarios* are the differences between each scenario and BAU (in t/ha), and the % change (In parentheses). Red values indicate declines relative to BAU.

	Victoria Central Highlands	North Coast Blackbutt	South Coast Silvertop Ash
BAU Harvesting (tC/ha)	835.2	218.5	288.4
<i>Management Scenarios (difference from BAU)</i>			
30% of forest residue to bioenergy	10.3 (1.2)	NA	17.0 (5.9)
50% of forest residue to bioenergy	NA	38.4 (17.6)	NA
50% of forest residue to pulp	NA	104.7 (47.9)	NA
50% pulp to bioenergy	-144.1 (-17.3)	NA	-32.8 (-11.4)
100% pulp to bioenergy	-274.6 (-32.9)	NA	-65.7 (-22.8)
All end-of-life products, and all processing waste to bioenergy	23.7 (2.8)	25.9 (11.9)	9.1 (3.2)
Max product to bioenergy	32.9 (3.9)	61.1 (28.0)	12.8 (4.4)
End of service life pallet mulch to bioenergy	10.9 (1.3)	NA	NA
Waste to product (Max product)	13.0 (1.6)	63.0 (28.8)	29.6 (10.3)
Max product to landfill	40.0 (4.8)	64.1 (29.3)	30.6 (10.6)
End of service life pallet mulch to landfill	18.3 (2.2)	NA	NA
Adopt site product spread, to yield a greater proportion of poles	NA	67.9 (31.1)	NA

6.6 Discussion

The ForestHWP model was applied to the three case studies to explore the implications of a range of alternative management scenarios on the total system C balance, inclusive of both on-site forest C and the impacts of wildfire, as well as off-site C in HWP and biofuels and associated substitution benefits. The model was calibrated at the ‘landscape’ scale, whereby the results reflect the area-weighted average C balance of the sub-areas within each case study that were available for harvest, as well as the sub-areas that were protected from timber harvesting. The model calibration was further based on the regionally-adjusted harvested parameters, as described in Part 3.

For comparative purposes the long-term average response of each scenario was calculated, to provide an estimate of the average or expected outcome for the different harvesting and HWP management regimes. This was necessary to allow unambiguous comparisons to be made across scenarios and between case studies. Note that because of the spatial scale of these calibrations the results should not be applied at the project level, nor to provide estimates of potential GHG emissions costs or benefits at project-level scales; for such applications project-specific calibrations and analyses are required. Although the projected C stocks reflect the expected long-term average outcomes, C storage of HWPs in landfills, the bioenergy offsets and product substitution factors accrue continuously through time; therefore the values that are reported for these quantities represent the summed benefits over a 200 year period.

The ForestHWP simulation results for each case-study suggest the overall C response to harvesting is context-dependent, and that widely different outcomes are possible depending upon the characteristics of the forest, the harvesting regime, and most importantly the mix of harvested wood products that are produced, their substitution benefits, and their ultimate fate. The current debate over the C costs and benefits of sustainably managed forests versus management for conservation is therefore overly simplistic; the question is not whether conservation or harvesting produces a more beneficial GHG outcome *per se*, because, although harvesting typically produced a more beneficial GHG outcome in the long-term, the outcome could be in either direction depending upon the circumstances. The question should rather be, under what conditions and constraints can forest management, when integrated across the landscape, be optimised to produce the most favourable outcomes. The focus of this study has been on outcomes specific to GHG balance, but it should be borne in mind that, in general, a more holistic approach is required that includes a broader array of forest values such as recreational amenity, timber products, maintenance of transport corridors for e.g. fire management, biodiversity conservation, and socio-economic outcomes.

The context-dependence of the results is best seen when comparing the summary figures for each case study (Figures 6.6, 6.8 and 6.9). The Victorian Central Highlands case study (Figure 6.6) showed significant potential for positive GHG outcomes, which were driven primarily by the product substitution costs associated with sourcing alternative pulp supplies from Asia (see below). The profile of the scenario responses for the NSW silvertop ash case study (Figure 6.9) was broadly similar to that for Victoria, with a similarly important role for pulp. This was despite the different forest types and associated wildfire regimes between these two case studies. In contrast the NSW North Coast case study BAU scenario produced lower GHG benefits than management for conservation, however it can be noted that the current harvesting regime does not fully utilise the available timber source due to market constraints, for example the lack of a pulp market in this region. For all three case studies changing either the utilisation of forest residues or the post-harvest processing and management of HWPs to optimise opportunities for bioenergy, or to increase the storage of end-of-service-life products in landfill, increased GHG benefits up to 30% above business as usual. Overall, the results

from the simulations suggested a number of options are potentially available for improving the GHG balance of harvested native forests above current business as usual (BAU) within the case study regions. Some of these, such as diverting processing residues to longer-lived products, or altering the product mix to favour long-lived products (such as the scenario investigating an increase in pole production for the New South Wales North Coast case study) could be implemented within current market constraints; whereas other options, such as the utilisation of residues and/or end-of-life products for bioenergy still lack a suitable market.

All scenarios, apart from those that involve changing from pulp to biofuel (for silvertop ash and mountain ash), increased net C benefits relative to BAU. In some cases, changes to the way individual products are treated can have a major impact on the overall C balance. For example, favouring the production of electricity poles over the extraction of sawlogs in the North Coast of NSW results in a net C benefit over the long term equivalent to approximately 30% (or approximately 70 tC/ha) above expected long-term average total C stocks for the blackbutt forests under current production. A moderate effect is also observed for the change of waste management for pallets in Victoria, where changing from mulch production to landfilling or to bioenergy generation represents a net change of approximately 10 – 20 tC/ha. Unlike the case of electricity poles, where both long-term C storage and product substitution contribute to the net C benefit, for pallets the net C benefit is achieved without the contribution of product substitution, as the likely replacement (untreated pine) has a similar emission footprint to untreated mountain ash. In general, strategies that involve increased utilisation of biomass for bioenergy production, extending the longevity of C in the HWPs (by changing production from short-lived products to medium to long-lived products and by storage in landfills) and general minimisation of waste all contribute to a greater net C benefit for all systems studied.

A major finding from the ForestHWP simulations was the important contribution of paper products to the net GHG balance. This is often ignored based on their typically short-service life, however the key factor to consider for the determination of the GHG balance of paper products is product substitution. In both the Central Highlands of Victoria and in Eden the production of pulp logs is part of integrated harvest operations, with forests that are considered to be sustainably harvested. In contrast, the key alternative market for paper production is based in SE Asia, primarily in Indonesia. As explored in detail in Part 4, the emissions footprint associated with the extraction of biomass for pulp and paper production in SE Asia is very large, due to high levels of deforestation of primary forest, forest degradation and loss of peatlands. It is important to acknowledge that, in a global economy, any changes to an important market in an individual country will have flow-on effects in different countries. As detailed in Part 4, the market forecasts are for increased paper consumption globally, fuelled by the emerging middle class in large countries such as China, India and Brazil. It is a reality that paper will continue to need to be produced somewhere into the foreseeable future. If native forest biomass currently sourced from the Central Highlands of Victoria and from Eden was no longer available, logically this would only add further pressure to the degraded and depleted forest areas in other areas of the world with comparatively lower standards of forest management, and with subsequent GHG implications.

If the choice for the management of the HWP post-service is between a short-lived product with no product substitution benefits (e.g. mulch), bioenergy generation or landfill, it is clear that the production of a short-lived product which decays quickly in service is the least preferable option from a GHG perspective. The choice between energy generation and landfilling will depend primarily on the energy profile of the region. For example, if the HWP was used in Tasmania, where the energy used is predominantly hydro –based (and thus with a low emission profile), landfilling would be preferable from a GHG perspective. In Victoria

however, the predominance of brown coal for energy generation would suggest potentially a more beneficial outcome if the HWP was used for energy generation. It is important to note that landfilling results in actual physical storage of C for the long-term, guaranteeing that the C will not be emitted. The net benefits of bioenergy generation as noted above will depend primarily on the alternative energy sources for the particular region. Ultimately, whether the HWP is recycled, used for energy or landfilled, the overall net GHG outcome will be positive.

A number of studies have attempted to quantify the overall GHG implications of native forest harvesting for Australian forests (Ximenes *et al.* 2012; Dean & Wardell-Johnson 2010; Dean *et al.* 2003, 2012; Keith *et al.* 2014a), with different authors arriving at opposing conclusions regarding the net emissions costs and/or benefits of harvesting compared with non-harvesting. There are at least three reasons for this divergence of opinion. First, it is perhaps unreasonable to expect a simple generalised response; for example in this study we found that emissions and the potential for HWP to generate GHG benefits varied greatly with forest type and product mix, as discussed above. The second reason is that different authors have made fundamentally different assumptions regarding key parameters. For example Keith *et al.* (2014a) have assumed timber in landfill decomposes with a half-life in the order of decades to centuries (consistent with previously accepted assumptions), whereas recent research has demonstrated landfill is in fact a strong sink for timber, with minimal *in situ* decomposition (e.g. Ximenes *et al.* 2008, Wang *et al.* 2011, Ximenes *et al.* 2013, Ximenes *et al.* 2015). Finally, and perhaps most fundamentally, most studies have considered only part of the total Forest-HWP system, and therefore comparisons across studies are confounded because they do not overlap in the various processes that the authors elected to either include or exclude from their particular analysis. For example Ximenes *et al.* (2012) included most of the processes summarised in Figure 5.1 (Section 5), with the exception of belowground biomass and soil organic C. In contrast Dean *et al.* (2012a) in their modelling included a comprehensive description of the forest C system, including below-ground and HWP's in landfill, but did not include product substitution effects nor the potential for bioenergy offsets. Keith *et al.* (2014) in their study included HWP in landfill, but neither product substitution nor bioenergy. Because of these differences in the specification of the system boundary across studies it is difficult to make sensible comparisons. Indeed, it was partly to address the difficulty of including all key processes associated with both the forest and HWP subsystems that the ForestHWP model was developed.

6.7 Caveats

The major strength of the ForestHWP model is that it was designed to comprehensively account for all C stocks and fluxes within both the forest and the harvested wood product subsystems, thus providing the capacity to estimate the total net GHG implications of different management options. However describing in relatively simple mathematical terms the complexity of forest growth, debris decay, wildfire, and the complexity of describing the processes underlying the production of HWPs, their in-service use and ultimate disposal is challenging. It is therefore unavoidable that many simplifications had to be made. The potential implications of some of these simplifications for the simulation results are discussed below. Most of these relate to the forest-sub-system, given the data and processes underlying the use and disposal of HWPs are relatively more tractable.

Process description vs empiricism

To keep the problem manageable, the description of processes within ForestHWP is kept to a minimum. For example there are no climate inputs and forest growth is specified *a priori* through a user-defined function, and the decay of organic matter is assumed to follow an exponential curve defined by a single invariant rate parameter. In this sense ForestHWP is perhaps better viewed as a C calculator rather than a model of forest C growth / HWP. This was both intentional and necessary in order to provide a consistent and transparent framework within which all of the necessary calculations spanning both the forest and the HWP subsystems could be embedded. Because of this our 200 year analysis timeframe and use of long-term average outcomes should not be interpreted as specific predictions of forest growth or HWP storage over the coming decades; rather, it is a mathematical device to allow the fundamental differences among the different management options to be expressed.

The treatment of wildfire

Wildfire is a key environmental factor in all Australian temperate forests, but the processes underlying the ignition, spread, and extinguishment of wildfires are exceedingly complicated and involve the distribution of fuels across landscapes, weather conditions including wind speed and direction, topographic complexity, etc. In contrast, the implementation of fire within ForestHWP was exceedingly simple, involving ‘fires’ that occur at random through time with a given frequency, and with consumption and transfers of forest C stocks that are specified by the user. The aim of this simple fire algorithm was therefore to try and capture the key impacts of fire on the forest system, without any attempt to provide a predictive capacity of when fires might occur as a function of e.g. prevailing weather conditions.

The scenarios involving alterations to fire frequency were also very simplistic (Table 6.2), in that they did not account for the possibility of simultaneous changes to other aspects of the fire regime, such as the severity of the impact on the forest C stores. In reality, fires are often associated with drought (or at the least prolonged periods of dry conditions) and thus there is a functional link between forest growth, forest C dynamics, and fire behaviour. Another key aspect that was not considered in the fire simulations was the potential for changes in fire regime to affect forest structure and composition (e.g. Bowman et al 2014), thus potentially leading to further alterations in fire behaviour. These are all important considerations and they potentially have implications for both forest structure and function, as well as the ongoing supply of timber services.

Belowground C and CWD

Because ForestHWP is a full system C model, predictions of belowground C (roots and soil organic C) were available and were presented in the summary tables (Tables 3-5, grey values). However there is a lack of empirical data for C stocks and fluxes in either roots or soil, and thus very large uncertainties over the default parameter values adopted in Table 1. It is for this reason we chose to exclude belowground C when presenting and discussing the main body of the results. For example, England *et al.* (2013) reviewed the evidence for changes in soil C in response to harvesting in temperate Australian forests and concluded there was at present insufficient data to support definitive conclusions. Although this review, like many others, indicates minimal change in soil C in response to harvesting, changes in recalcitrant C pools in response to perturbation may take decades or centuries to become apparent, as indicated by a number of modelling studies (e.g. Dean *et al.* 2012b; Hibbard et al. 2003). Belowground C in forests is a very important component of the total system C balance, and is likely impacted by harvesting. However further work is required before the direction and magnitude of these impacts can be predicted with any certainty.

Coarse woody debris is also an important component of the total C balance, particularly in the Victorian ash forest case study where the creation and decay of woody material is a dominant process. There are difficulties in measuring the amount of forest CWD, given it is spatially highly variable, thus precise measurements are difficult to obtain. There are additionally difficulties in either directly weighing or estimating the bulk density of fallen timber (if mass is being estimated from volume), particularly when it is large. There are also few data documenting the time course of decay of dead woody material in temperate forests, partly because of methodological issues, and partly because of the decadal timescales involved. Again, more data is required.

Uncertainty

The dynamics of C belowground are not the only source of uncertainty. Other important sources include:

- 1 Uncertainty and representation and or inclusion of key processes

Because of the simplicity of the ForestHWP modelling approach there are many processes that were either excluded from the calculations, or that were expressed using simple functional forms. Examples of processes that were excluded include the potential for changes in forest growth and structure due to climate change and changing atmospheric C dioxide concentrations, and changes in the rates of decomposition due to climatic variability. Examples of processes that were simplistically expressed include the assumed exponential decay of both forest C stocks, as well as in-service losses of HWP.

- 2 Uncertainty in parameter values

A number of the parameters controlling the dynamics of C in the forest subsystem were estimated through ‘model –inversion’ - a process where the model outputs (the C stocks in e.g. the trees or litter) are considered fixed, and the model parameters are adjusted in such a way that the predictions match these observations. Whilst this provides some internal consistency ensuring that the predictions are consistent with reality, there is nevertheless the possibility of parameters ‘trading off’ against one another, and thus for the values and resulting temporal behaviour to be ‘incorrect’. Some of the model parameters were able to be estimated directly using available data. Examples include the estimation of net primary productivity using previous work, and the setting of the majority of the HWP parameters from data collected as part of this project. These data are also subject to uncertainty, which will propagate into the analysis.

The degree to which these uncertainties could influence the outcomes and hence conclusions of the modelling remains unknown, but could be addressed through a formal uncertainty analysis of the modelling, to give some indication of the likely confidence bounds around the model outputs.

- 3 Uncertainty due to the representativeness of the calibration data

The basis for the modelling of each of the case studies involved an inherent assumption that the conservation field data is an adequate representation of the forest structure and dynamics in the absence of harvesting, and in particular the harvested paired-plot. For each of the case studies this involved first calibrating the model to replicate the C stocks as measured in the conservation sites (taking into account previous fire and management history), and then applying a range of harvesting

scenarios to those calibrations. Testing the validity of this assumption would require extensive additional field survey combined with an assessment of historical disturbances. This is problematic because one of the strengths of this study is that biomass estimates are based on direct harvest and measurement, thus avoiding potential additional large uncertainties associated with the use of e.g. allometric equations or expansion factors; however such activities are extremely time-consuming and expensive, and for estimating the ‘conservation’ baseline are for the most part impractical due to their protected status.

4 Uncertainty in markets and other external drivers

Although not related to the structure of the ForestHWP model or its parameters, another important source of uncertainty is the influence of external drivers such as changes in forest policy, and market opportunities. The 22 scenarios described above go some way to exploring the potential sensitivities; however within each scenario the assumption was made that these externalities remained constant, which is unrealistic. A valuable extension study would be to explore the sensitivity of the results to a relaxation of this assumption, although deriving sets of plausible future scenarios could prove challenging.

The overall implications of including some or all of these uncertainties on the conclusions remains unknown; however the analyses as presented reflect the impacts of different management options as based on the best available data, and although introduction of more complex uncertainty analyses, such as Monte-Carlo sensitivity experiments, would likely show some of the options to be statistically equivalent in their outcomes, their relative rankings and magnitudes when compared against the business as usual scenario would not be expected to change.

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Part 7. Socio-economic considerations

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7.1 Overview

There is a range of different ways that native forests may be managed for carbon storage and socio-economic benefits. Using the findings from two State Forest sites (one at Eden dominated by silvertop ash and the other near Wauchope dominated by coastal blackbutt) this study sought to evaluate and rank a suite of different native forest management scenarios using socio-economic indices and carbon pricing as the basis of comparison. A similar analyses is expected to be conducted for the mountain ash forests in the Central Highlands of Victoria in the future.

Analysis for the study was undertaken in three steps:

- i. A general comparison of the regional economic impact of “production forests” (State Forests) and forests managed for conservation only (National Parks).
- ii. Detailing the costs and benefits of transitioning State Forests to National Park.
- iii. Quantifying the effect of a carbon price on eleven alternative forest management scenarios using two different carbon accounting methodologies with business as usual (BAU) as the control.

7.2 Executive Summary

This study found that State Forests generate greater socio-economic benefits than National Parks and that the State Forests on the north coast generate more socio-economic value than those at Eden. The greater benefits arising from north coast State Forests were due to a better quality timber resource and a higher proportion of domestic value adding.

Converting State Forests to National Parks was found to cause a reduction of socio-economic value and a decline in regional employment at Eden and on the north coast. The greatest loss of jobs and socio-economic value occurred on the north coast. Converting State Forests to National Parks triggered publicly funded transition costs. These costs were very large when applied at a regional scale - \$65M at Eden and \$540M on the North Coast (NPVs modelled over 20 years using a 7.5% discount rate).

A true fate carbon accounting methodology (what the atmosphere sees) was used to quantify the carbon abatement arising from eleven alternative management scenarios relative to business as usual (BAU). Under true fate accounting, eight of the production based management scenarios and one of the conservation scenarios generated positive carbon abatement relative to BAU. At Eden, the conservation scenario and two of the production scenarios had less carbon storage than under BAU. On the north coast all of the alternative management scenarios were positive (including the conservation scenario).

Valuation of the carbon abatement benefits (relative to business as usual) were derived using a 65 year modelling period and a low (\$10/t CO₂-e) (Figures 7.1 and 7.2), medium (\$20/t CO₂-e), and high (\$30/t CO₂-e) carbon price. The results were presented as net present values (NPVs) based on a 7.5% discount rate.

Figure 7.1 – Net present value of carbon abatement relative to business as usual with carbon priced at \$10/tO2-e - Eden

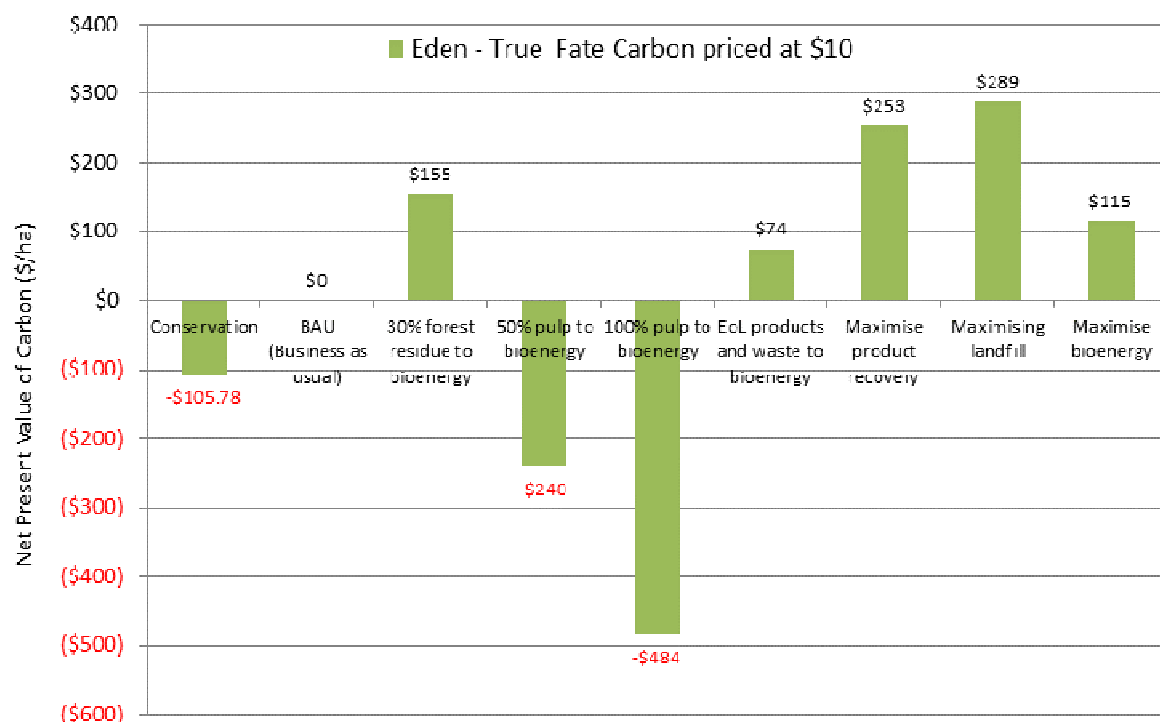
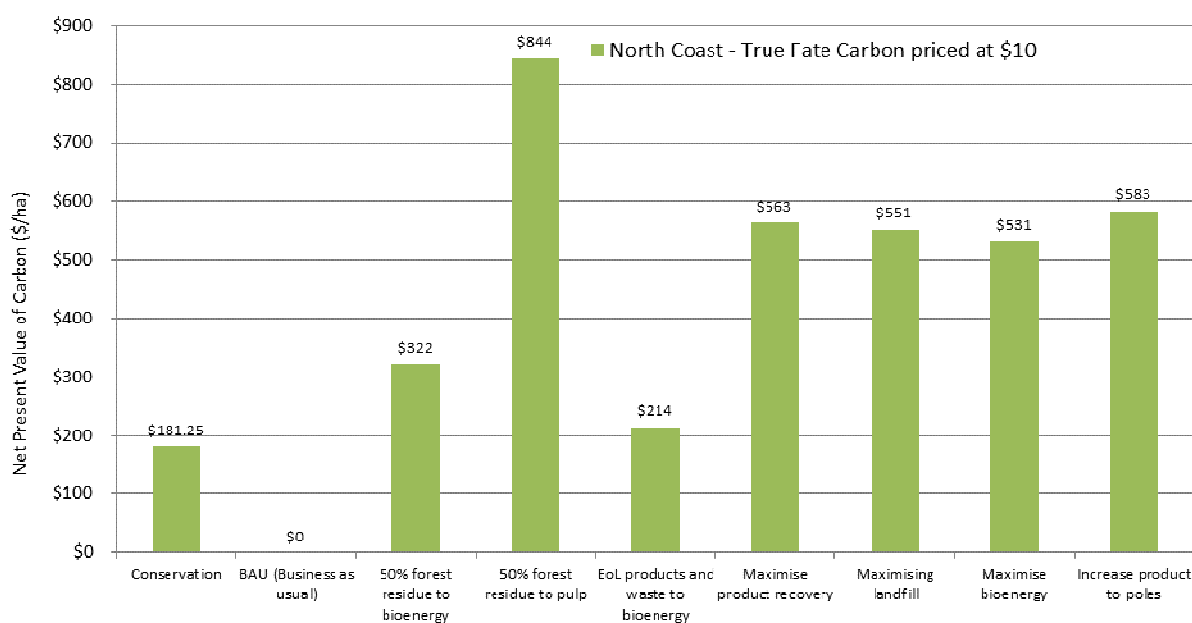


Figure 7.2 – Net present value of carbon abatement relative to business as usual with carbon priced at \$10/tO2-e – North Coast



The most favourable carbon abatement scenario occurred on the north coast and involved *utilising 50% of the forest residues left in the forest for pulpwood*. When applied at a regional scale this scenario added \$669M in value with carbon priced at \$10/tCO₂-e.

When carbon abatement benefits were added to existing industry value-added benefits (based on BAU) the alternative production management scenarios far exceeded the conservation management scenarios. From this finding it was concluded that the transfer of State Forests to National Parks for carbon abatement benefit was unsupported on either financial or socio-economic grounds.

7.3 Method

The method used for this study was based on financial modelling. To facilitate comparison all modelling was undertaken on a 'per hectare' basis. Key modelling assumptions are set out below with further detail provided within Appendix 4.

Net present values (NPVs) were calculated using discounted cash flow analysis. To aid clarity analysis was divided into three discrete steps:

- i. Valuing the regional economic contribution of State Forests and National Parks under business as usual.
- ii. Detailing the costs and benefits of transitioning State Forest to National Park.
- iii. Quantifying the effect of a carbon price on eleven alternative forest management scenarios using two different carbon accounting methodologies and business as usual (BAU) as the control.

A real discount rate of 7.5% was used. 7.5% is commonly applied to the valuation of long term forestry projects within Australia. At present a 7.5% discount rate will cover inflation (10-year Australian Bond yield rate) +5% and is comparable to the weighted average cost of capital of many Australian companies.

Benchmarking the economic impact of NSW National Parks & Reserves and State Forest has not been previously attempted. A lack of common economic indicators is one reason why this has not occurred. This study attempted to circumvent this constraint by sourcing relevant data from a range of existing reports and sources and combining it into a single valuation.

The limitation of this approach was its reliance on data of different ages and in some cases differing qualities. In an attempt to address these issues CPI adjustments were applied (so that all figures are reported in 2014 dollars) and detailed documentation provided on all key modelling assumptions.

A key strength of the method was its reliance on high quality empirical measurement data collected from the forests around Eden and Wauchope and applied using a state of the art CSIRO model called ForestHWP. The results from the model provided a complete system profile of the carbon dynamics of the measured forest sites as well as the harvested wood products (HWPs) they were generated from them.

By not relying on a single data source and by not deriving data from first principles the method has maintained a level of independence and objectivity that would not have been otherwise achievable.

7.3.1. Indicative comparison of the economic contribution of State Forests and National Parks

This method was undertaken by way of a valuation of the forested estates under National Park and State Forest management on the Far South Coast (Eden RFA Region) and on the NSW North Coast (North East RFA Region).

Direct Value-added was used as the common denominator for measuring economic impact. The direct value added of an industry, also referred to as gross domestic product (GDP) by industry, is the contribution of a private industry or government sector to overall GDP. The U.S Department of Commerce clearly defines *industry value added* as the difference between an industry's gross output (consisting of sales or receipts and other operating income, commodity taxes, and inventory change) and the cost of its intermediate inputs (including energy, raw materials, semi-finished goods, and services that are purchased from all sources).

For the purpose of economic evaluation, the management of NSW National Parks & Reserves was treated as an industry. The direct value-added effect of NSW National Parks & Reserves was based on actual government expenditure on management and estimated expenditure related to National Park visitation independently valued using input-output analysis. Values were sourced directly from two reports - produced in 2006 and commissioned by the then NSW Department of Environment and Conservation (DEC) - one for the NSW Far South Coast and the other for North-East NSW.

NSW DEC Reports:

- Powell, R., Chalmers, L. & Bentham, A. (2006) Impact of National Parks on the regional economies of the NSW Far South Coast. Final Report to the NSW Department of Environment and Conservation. Centre for Agricultural and Regional Economics Pty Ltd. September 2006. 56pp.
- Gillespie Economics (2006) Impacts of Protected Area on the Regional Economy of North-East NSW. A study prepared for the NSW Department of Environment and Conservation. August 2006 43pp including appendices.

The direct value-added of State Forests was based on the value-added of the timber industry and the value-added of State Forest visitor expenditure (i.e. estimated expenditure associated with visitation to the estate). Other contributors to the value-added of State Forests including grazing and apiary were not included due to their relatively small economic contributions.

The value-added of the timber industry was based on timber production levels average over three years (2011/12, 2012/13 and 2013/14) for Eden and for the NSW North Coast. The value of timber production was calculated using national data published by the Australian Bureau of Resource Economics and Sciences (ABARES) on the value-added of the timber industry. The method for deriving these values is detailed at Appendix 4.1.

7.3.2. Detailing the costs and benefits of transitioning State Forests to National Parks

The objective of this component of the study was to model the costs and benefits of transitioning existing State Forests to National Park for the purpose of achieving a forest carbon abatement benefit. This method involved three steps:

- i. Valuing Eden and the North Coast State Forests under business as usual (BAU).

- ii. Valuing the transition from State Forests to National Parks in terms of jobs, structural adjustment and unemployment costs.
- iii. Revaluing the forests at Eden and the North Coast as National Parks.

A summary of all key modelling parameters is provided at Appendix 4.1.

7.3.2.1. Business as usual (BAU)

The value-added of State Forest calculated in the first component of this study was used as the value of BAU. Employment levels for BAU were sourced from ABARES Australian Forest & Wood Product Statistics Socioeconomic Tables index. The original source of the ABARES data was the 2011 National Census. Employment level assumptions are detailed at Appendix 4.3.

7.3.2.2. Transition - Structural adjustment & 'Unemployment Benefit' costs

Under the transitioning scenario timber harvesting ceases and employees directly affected are eligible for financial assistance. There have been numerous precedents for financial assistance, the most recent occurring in 2010 when around 100,000 hectares of State Forest in the NSW Riverina Region was transferred to National Park. This study applied compensation in accord with the 2010 River Red Gum decision exit assistance guidelines (Appendix 4.4).

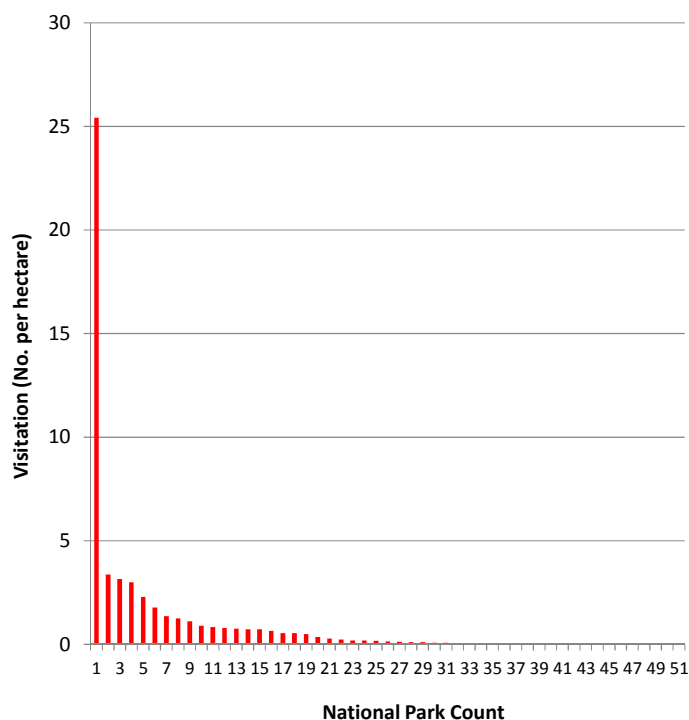
Displaced forestry workers were also assumed to be eligible for unemployment benefits. Details of these entitlements and the proportion of persons affected are provided at Appendix 4.5.

7.3.2.3. Revaluing State Forests as National Parks

Revaluing State Forests as National Parks involved identifying values that stay the same and quantifying those that will change:

- i. The loss of the regional timber industry (value added) was assumed as it is dependent on the forest remaining dedicated as State Forest.
- ii. National Park management expenditure was identified as a new source of value added for the regional economy. The contribution of this source was assumed to be the same as existing National Park (on a \$ per hectare basis) and based on the work of Powell et al (2006) on the Far South Coast and Gillespie Economics (2006) for the north coast.
- iii. Visitor expenditure value added was assumed not to change however there is some evidence to suggest that it may decline. Data on visitation levels (Gillespie 2006) shows that within coastal forest regions 70% of the visitation to National Parks & Reserves occurs on only 10% of the estate and that over half of all National Parks receive very low visitation levels (Figure 7.3). National Parks that receive high visitation tends to be close to major population centres, are easy to access and or have iconic attributes such as waterfalls, majestic lookouts or adjoin the coast. State Forests in contrast contain fewer sites with iconic attributes. Road access to State Forests is better and more uniform than National Parks resulting in more uniform visitation. State Forests are also more amenable for visitors who wish to car camp, hunt, bring their pets and recreational equipment (e.g. 4wd, mountain bikes and motor bikes). When State Forests are converted to National Park roads are typically closed and tighter restrictions on forest use are imposed.

Figure 7.3 – Visitation in National Parks and reserves in NE NSW (Gillespie 2006)



- iv. Generating income from carbon abatement credits was quantified under a low (\$10/tCO₂-e) medium (\$20/t CO₂-e) and high (\$30/t CO₂-e) carbon price using an “true fate” accounting methodology. Details of this accounting method have been provided in the next section.
- v. Changes in employment levels were assumed based on the loss of timber industry jobs and the creation of jobs in National Park land management. Calculations were undertaken in accord with the methodology detailed in Appendix 4.5. For the reasons outlined above, no change was assumed for the number of jobs associated with forest visitation.

7.3.3. The effect of a carbon price on eleven alternative forest management scenarios

7.3.3.1. Carbon accounting

Carbon pricing analysis undertaken for this study was based on the carbon accounting outputs of a new model called ForestHWP. This model used forest measurements taken from two native forests sites, one near Eden in the State’s south-east and the other near Wauchope on the State’s mid north coast. From these measurements the ForestHWP model generated a complete system profile of the carbon dynamics.

The carbon accounts generated from the ForestHWP model were termed the true fate carbon accounts. These accounts incorporated all of the major pools and fluxes of carbon that the atmosphere sees (excluding below ground) including both onsite and offsite carbon pathways. Offsite storage included harvested wood products in service and in landfill and emission savings arising from product substitution and fossil fuel displacement due to the use of biomass to generate energy.

A carbon accounting method that may be recognised under the Commonwealth's Emission Reduction Fund (ERF) was beyond the scope of this project. No methodologies have been as yet approved under the ERF for native forest management activities.

7.3.3.2. Carbon pricing

For analysis purposes NPVs were calculated for each scenario using three different carbon prices, namely \$10, \$20 and \$30 per tCO₂-e.

Over the last ten years Australia carbon prices have ranged between \$5 and \$25 per tonne of carbon dioxide equivalent (tCO₂-e). Under the NSW Greenhouse Gas Reduction Scheme which operated between 2005 and 2013 most carbon was traded at between \$5 and \$14 per tCO₂-e. Fixed pricing under the Carbon Farming Initiative (CFI) went as high as \$25 per tCO₂-e before being abolished.

Over the last 2 years (July 2013 to June 2015) the price of European Union Allowances (EUAs) has averaged around €6.00/tCO₂-e or \$AU10/tCO₂-e. The current price of EUAs is around 8.36/tCO₂-e or \$AU13.95/tCO₂-e

Price signals from the Commonwealth Government are provided through its ERF auction. The Commonwealth Government has allocated \$2.55 billion to underpin the ERF. In the first auction in April 2015, \$13.95 per tCO₂-e was the average price paid. In the second auction in November 2015 the average price paid was \$12.25 per tCO₂-e.

7.3.3.3 Modelling scenarios

The two most common forms of public forest management are – State Forests managed to produce hardwood timber products, and National Parks managed as reserves for biodiversity conservation. Within State Forests there are a suite of possible variations on the BAU harvesting model. The eleven alternatives that were analysed for this study have been detailed in Table 7.1. Eight of these scenarios were applied at Eden (three of these were unique to that site). Eight scenarios were also applied to the North Coast site with three also being unique to that site.

7.3.3.4. Modelling period

Financial modelling was undertaken over a 65 year period. This period was selected as it represented an average native forestry rotation length in NSW.

7.3.3.5. Modelling exclusions

Modelling the specific costs and benefits of each modified harvesting scenario (2 to 11) was beyond the scope of the project. To implement any of the alternative harvest management scenarios would require new capital investment and new investment in human resources that would add value to the BAU. For this reason using the socio-economic value of BAU as a surrogate for an alternative harvest management scenario should be treated as a conservative underestimate.

Table 7.1- State Forest forest management modelling scenarios

Forest Mgt Scenario		Scenario description	Sites where scenario was applied
	BAU	Business as usual timber harvesting on State Forest	Eden and Mid-North Coast
i.	Conservation	Reservation of State Forest in National Park	Eden and Mid-North Coast
ii.	30% forest residue to bioenergy	30% of forest residues left on site utilised for co-firing with coal for electricity generation	Eden
iii.	50% forest residue to bioenergy	50% of forest residues left on site utilised for co-firing with coal for electricity generation	Mid-North Coast
iv.	50% pulp to bioenergy	50% of the material removed from the forest for pulp is instead utilised for residential bioenergy.	Eden
v.	50% forest residue to pulp	50% of forest residues left on site utilised for pulp	Mid-North Coast
vi.	100% pulp to bioenergy	100% of the material removed from the forest for pulp is instead utilised for residential bioenergy.	Eden
vii.	EoL products and waste to bioenergy	At the end of service life (EoL) all available product is utilised for residential bioenergy, and 70% of dry and green processing waste is utilised for residential bioenergy.	Eden and Mid-North
viii.	Maximise product recovery	70% of dry and green processing waste re-utilised for additional dry product. An example would be use of residues to produce engineered wood products	Eden and Mid-North Coast
ix.	Maximising landfill	At the end of service life all available product is sent to landfill, 70% of dry and green processing waste reutilised for additional dry product (which eventually ends up in landfill)	Eden and Mid-North Coast
x.	Maximise bioenergy	At the end of service life all available product is utilised for residential bioenergy, and 100% of dry and green processing waste is utilised for residential bioenergy.	Eden and Mid-North Coast
xi.	Increase product to poles	The regional product mix used in the simulations is replaced by the product mix as observed at the coupe level, which increases the proportion of pole manufacture over 6x.	Mid-North Coast

7.4. Results

7.4.1. Indicative comparison of the economic contribution of State Forest and National Park

The economic contribution of State Forests and National Parks were separately derived for the NSW Far South Coast and for the NSW North Coast regions. Results have been presented in NPV\$ per hectare rather than regional totals to accommodate differences in geographic boundaries and estate sizes (Figures 7.4 and 7.5).

Figure 7.4 – Economic Impact of native forests managed as State Forest and National Park on the NSW Far South Coast

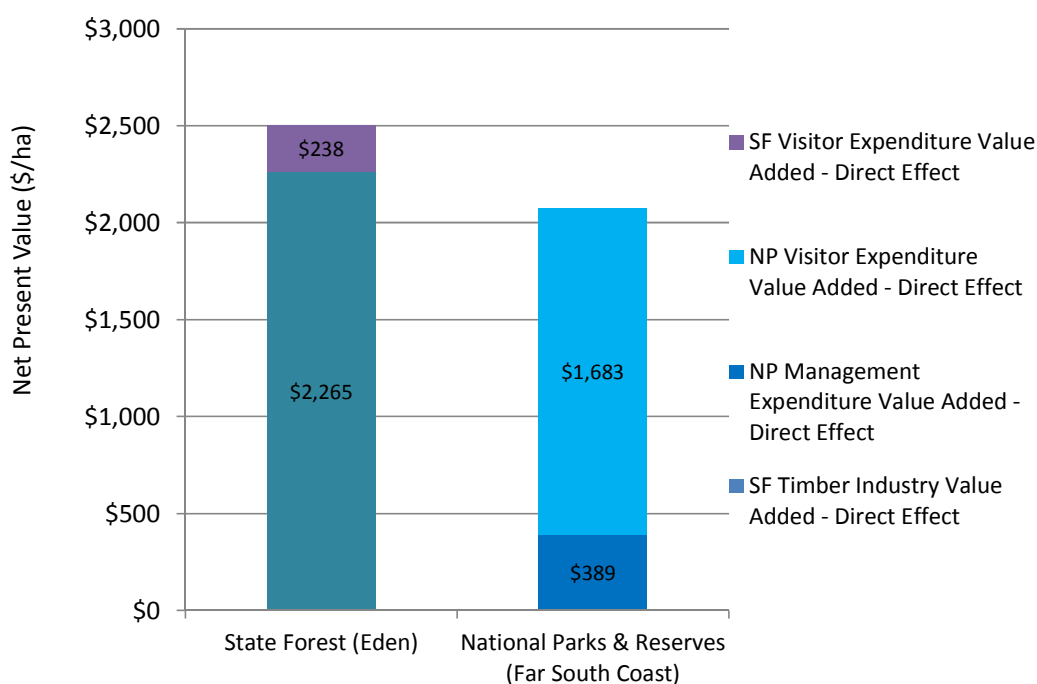
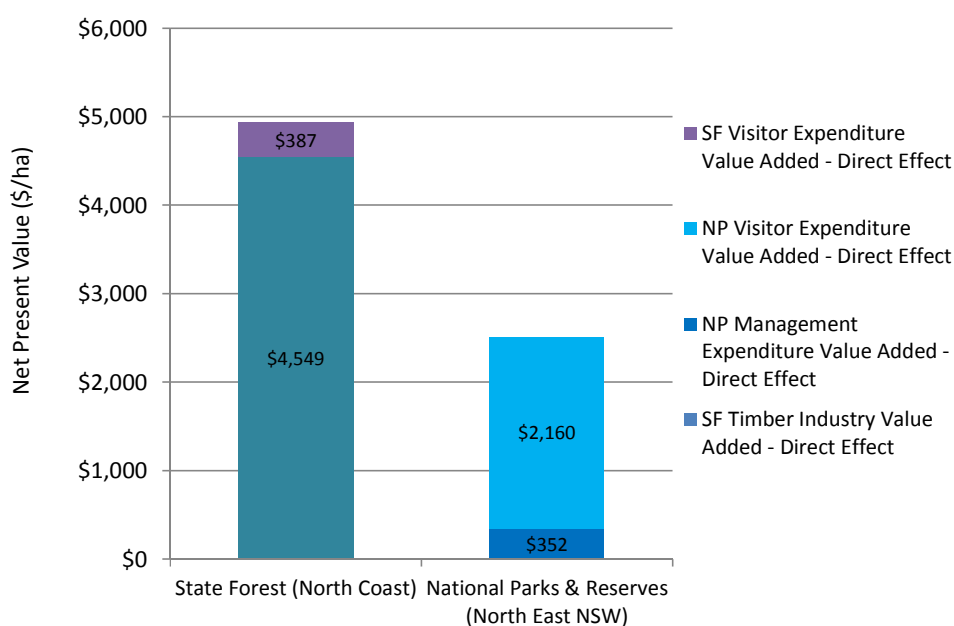


Figure 7.5 – Economic Impact of native forests managed as State Forest and National Park on the NSW North Coast



7.4.2. Quantifying the costs and benefits of transitioning State Forests to National Parks

The cost (economic impact) of converting one hectare of State Forest to National Park is shown at Figure 7.6 for Eden and at Figure 7.7 for the North Coast. These figures exclude consideration of any carbon abatement benefit.

Figure 7.6 – Economic effect of converting Eden State Forests to National Parks – before during and after conversion

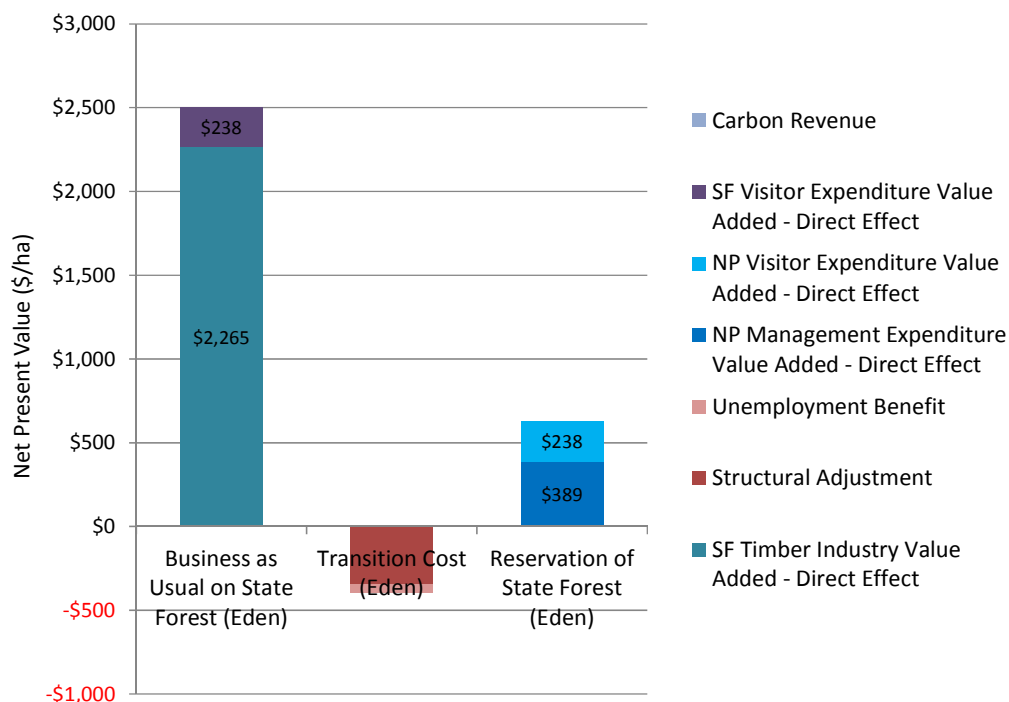
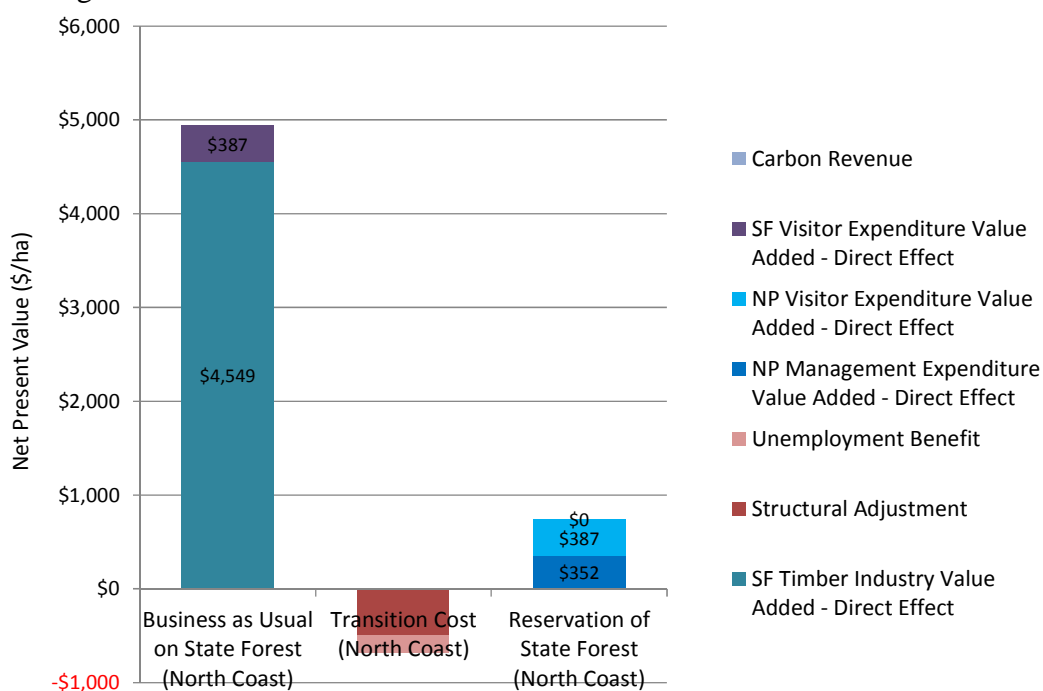


Figure 7.7 – Economic effect of converting North Coast State Forests to National Parks – before during and after conversion

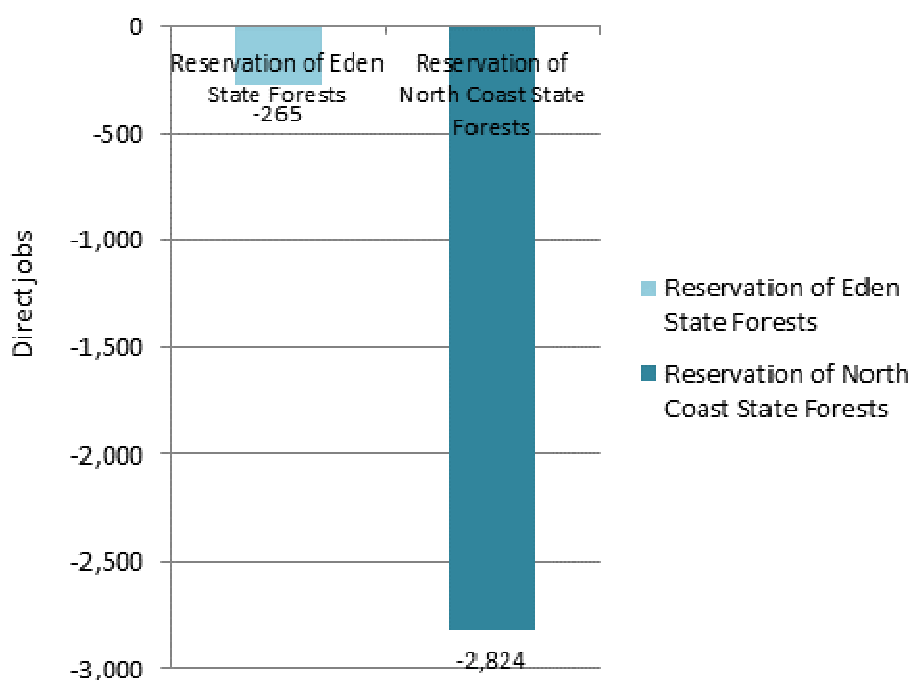


Extrapolating the values in Figure 7.6 to the Eden Region (164,000 hectares of State Forest) the cost of transition is \$64M (NPV over 65 yrs @ 7.5% DR) and the loss to the regional economy is \$308M (NPV over 65 yrs @ 7.5% DR).

Extrapolating the values in Figure 7.7 to the North Coast Region (792,361 hectares of State Forest) the cost of transition is \$540M (NPV over 65 yrs @ 7.5% DR) and the loss to the regional economy is \$3.36B (NPV over 65 yrs @ 7.5% DR).

The effect of transitioning State Forests to National Parks on regional employment is illustrated in Figure 7.8. It shows the loss of 265 jobs in Eden and 2824 jobs on the north coast. Quantifying the impact on city based timber industry jobs was beyond the scope of this study.

Figure 7.8 –Impact on regional employment from converting Eden and North Coast State Forests to National Parks



7.4.3. The effect of a carbon price on eleven alternative forest management scenarios

7.4.3.1. Carbon accounting

As explained in the methodology, carbon accounting was based on the True Fate carbon accounts generated from CSIRO's ForestHWP model using the field measurements taken at Eden and Wauchope.

The amount of carbon abatement generated for each of the eleven alternative forest management scenarios are detailed in Figure 7.9 for Eden and Figure 7.10 for the North Coast.

Figure 7.9 – Long term carbon abatement by management scenario and source - Eden

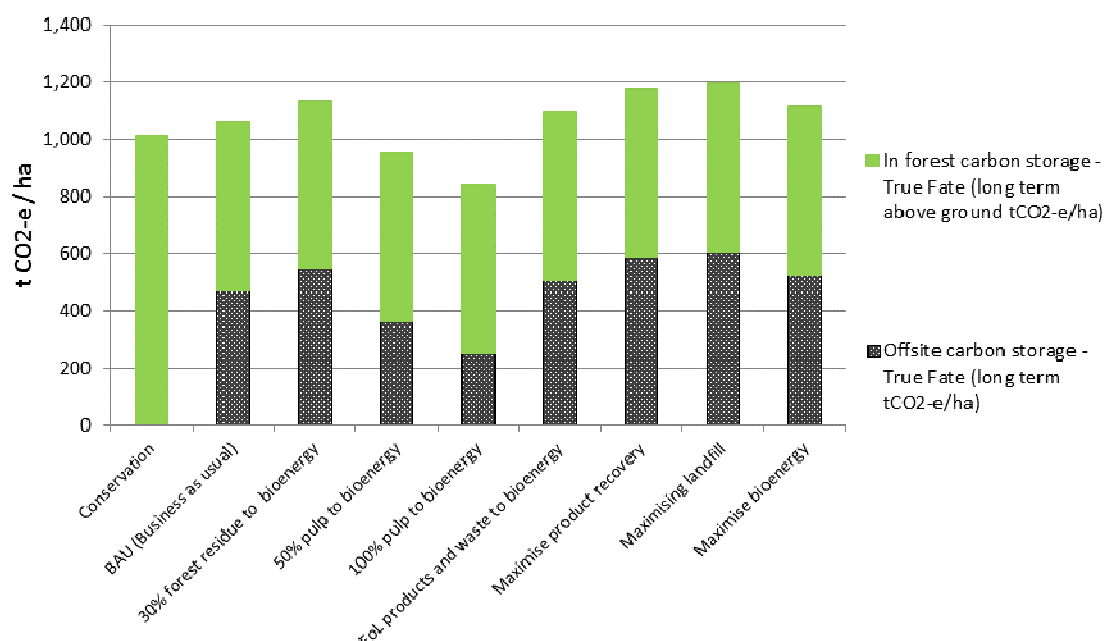
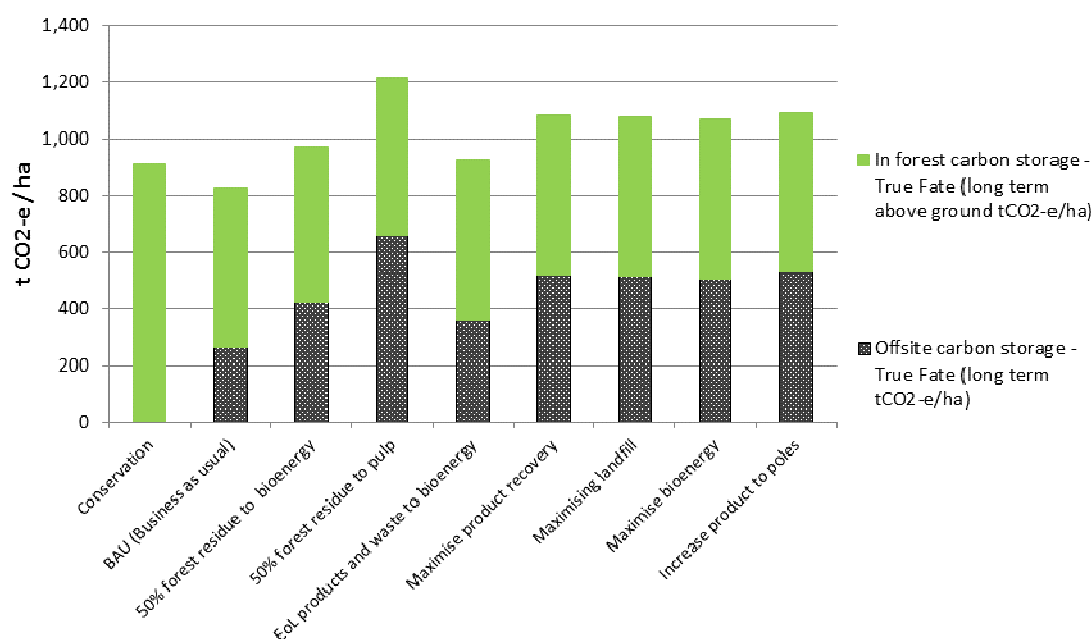


Figure 7.10 - Long term carbon abatement by management scenario and source – Wauchope



7.4.3.2. Carbon pricing

The effect of placing a price on carbon for the eleven alternative forest management scenarios is best demonstrated through graphical illustration. Figures 7.11 to 7.13 respectively show the effect of carbon priced at \$10/tCO₂-e, \$20/tCO₂-e and \$30/tCO₂-e.

Of the eight scenarios modelled at Eden five were found to have greater carbon revenue benefits than the BAU, namely, *30% forest residue to bioenergy*, *EoL products and waste to bioenergy*, *maximise product recovery*, *maximising landfill*, and *maximise bioenergy*. The

remaining three scenarios, namely: *conservation*; *50% pulp to bioenergy*; and *100% pulp to bioenergy* all had lower NPVs than the BAU scenario. On the North Coast increasing the price of carbon had a positive effect on the NPVs of all of the management scenarios relative to BAU.

Pricing carbon had a greater effect on the North Coast NPVs than it did on the Eden NPVs. For example, pricing carbon at \$30/tCO₂-e increased the NPV of the *50% residue to pulpwood* scenario by \$2,532 per hectare on the North Coast. In contrast, the highest NPV at Eden was the *maximising landfill* scenario. At a carbon price of \$30/tCO₂-e the NPV of this scenario increased by \$866 per hectare.

Figure 7.11 Value of carbon abatement for eleven alternative forest management scenarios with carbon priced at \$10/tCO₂-e

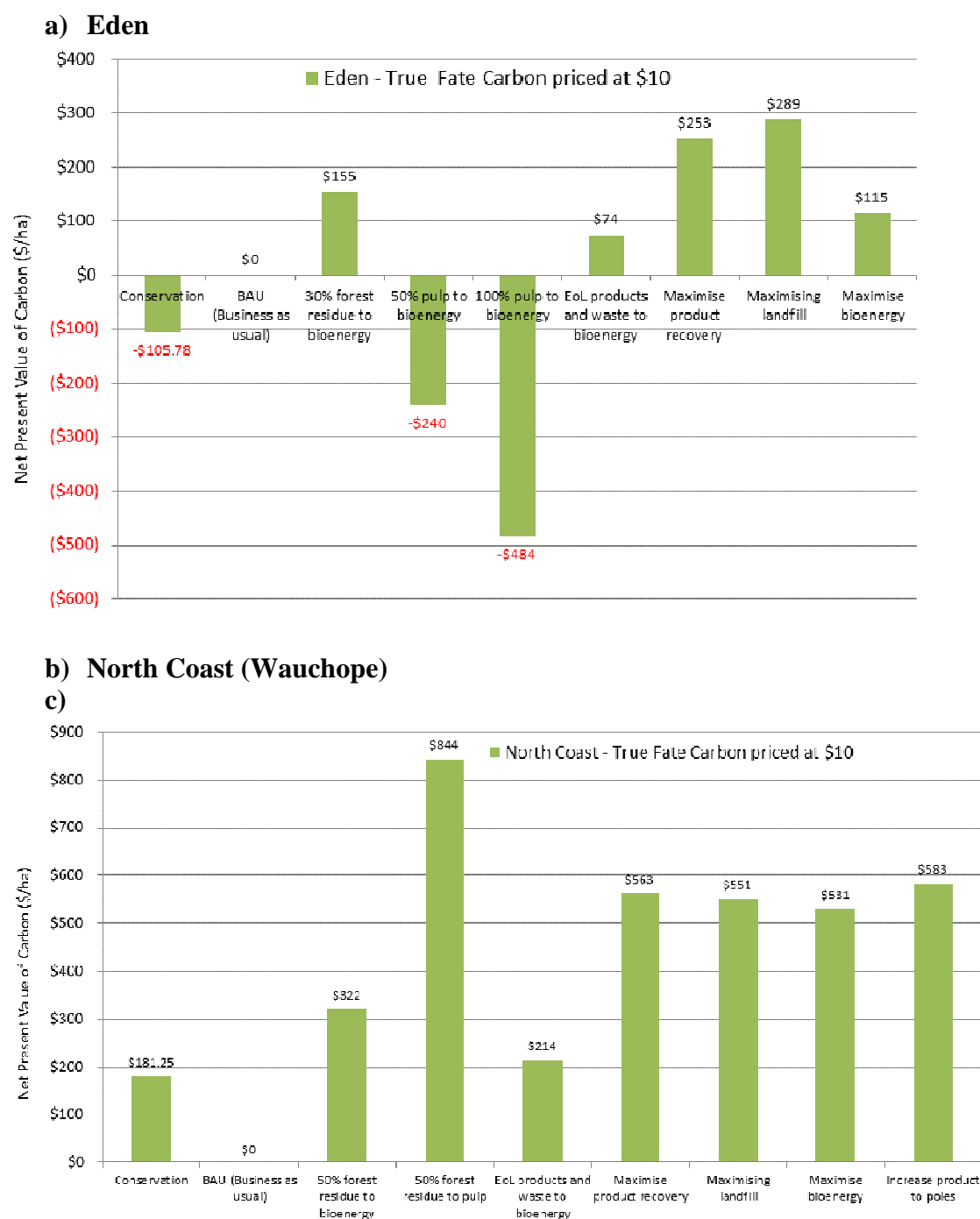
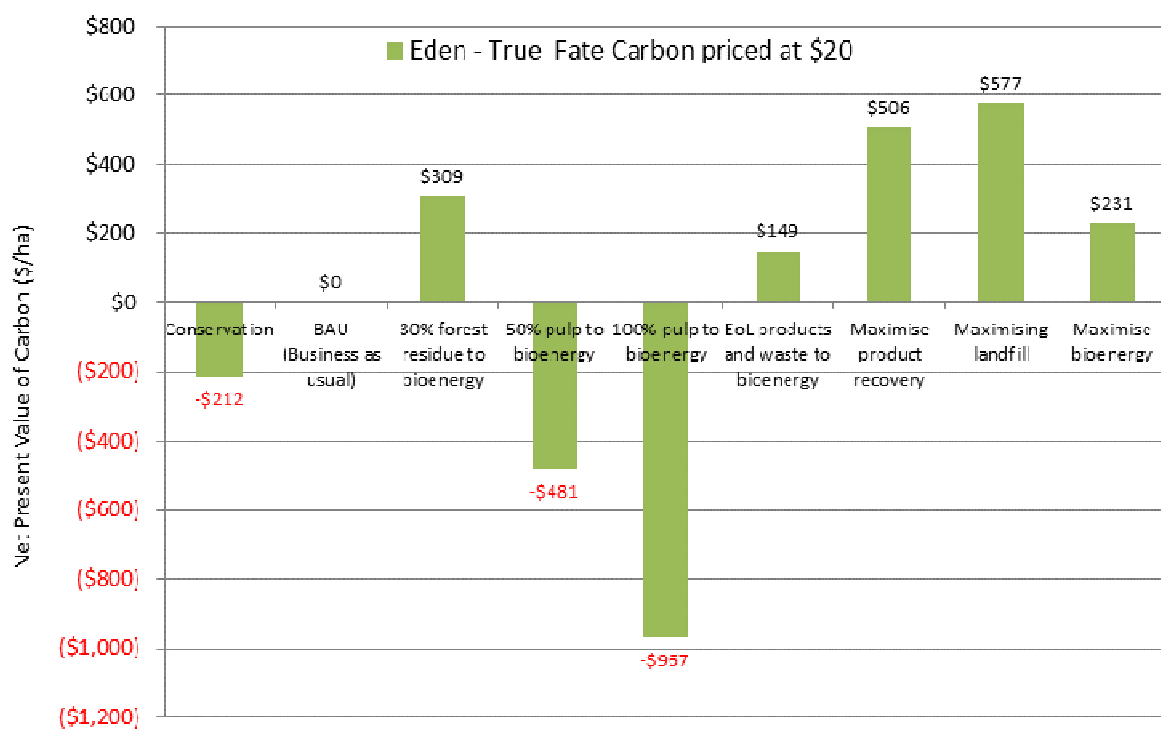


Figure 7.12 Value of carbon abatement for eleven alternative forest management scenarios with carbon priced at \$20/tCO₂-e

a) Eden



b) North Coast (Wauchope)

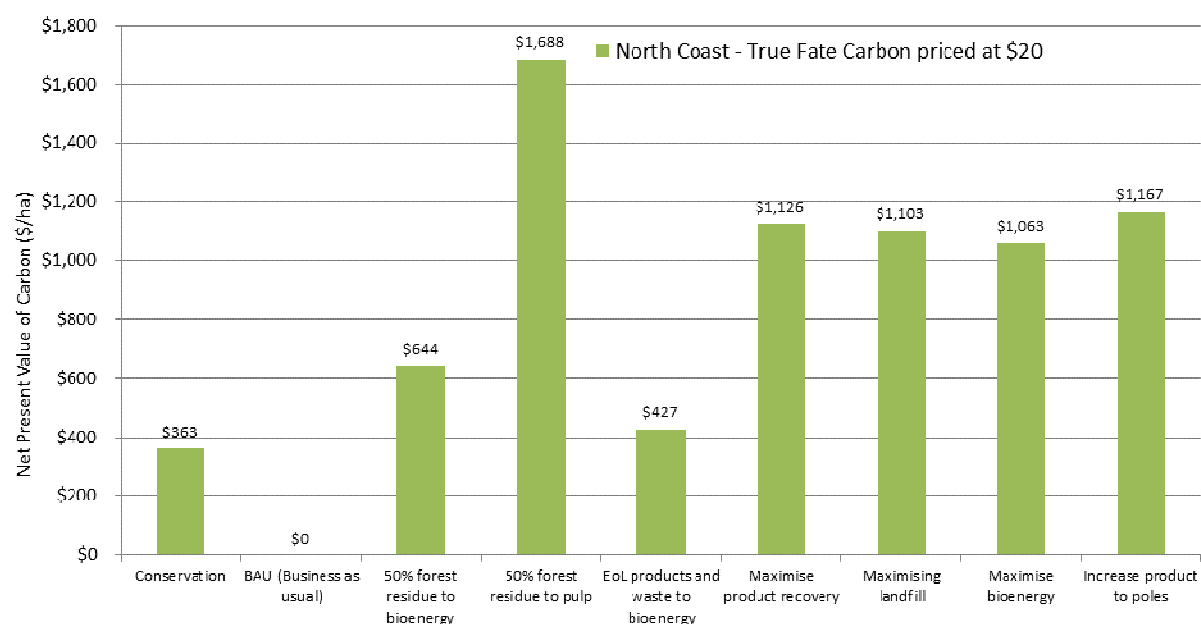
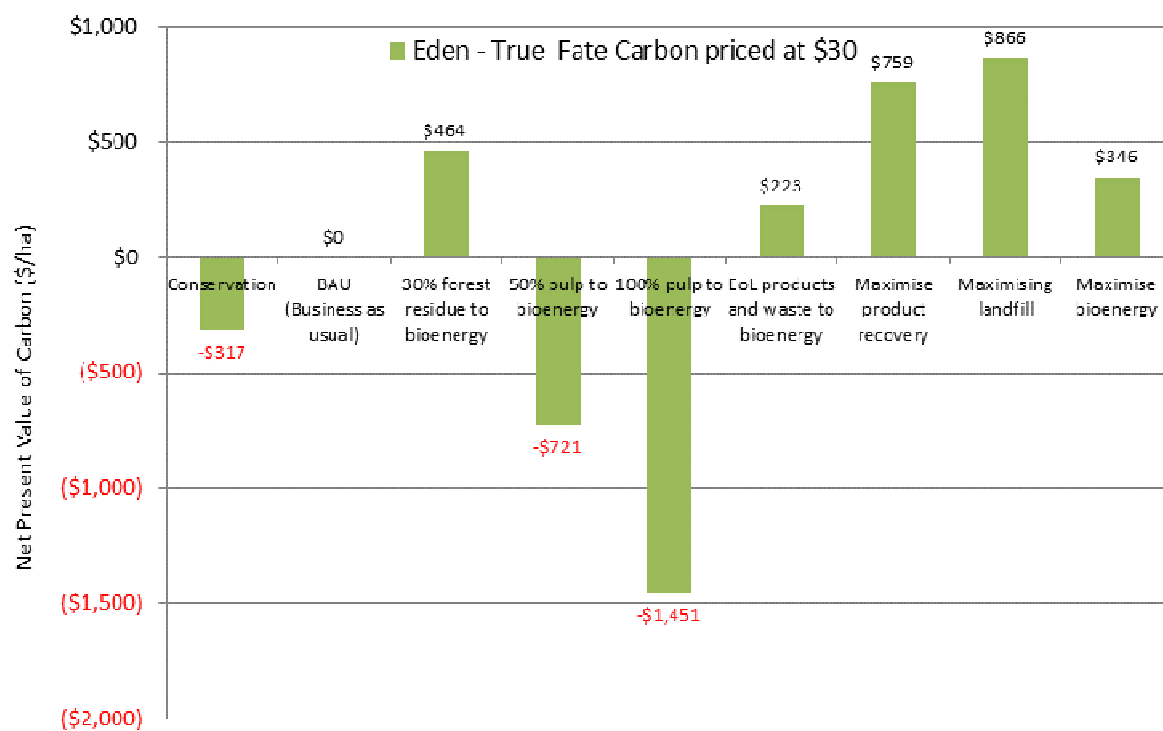
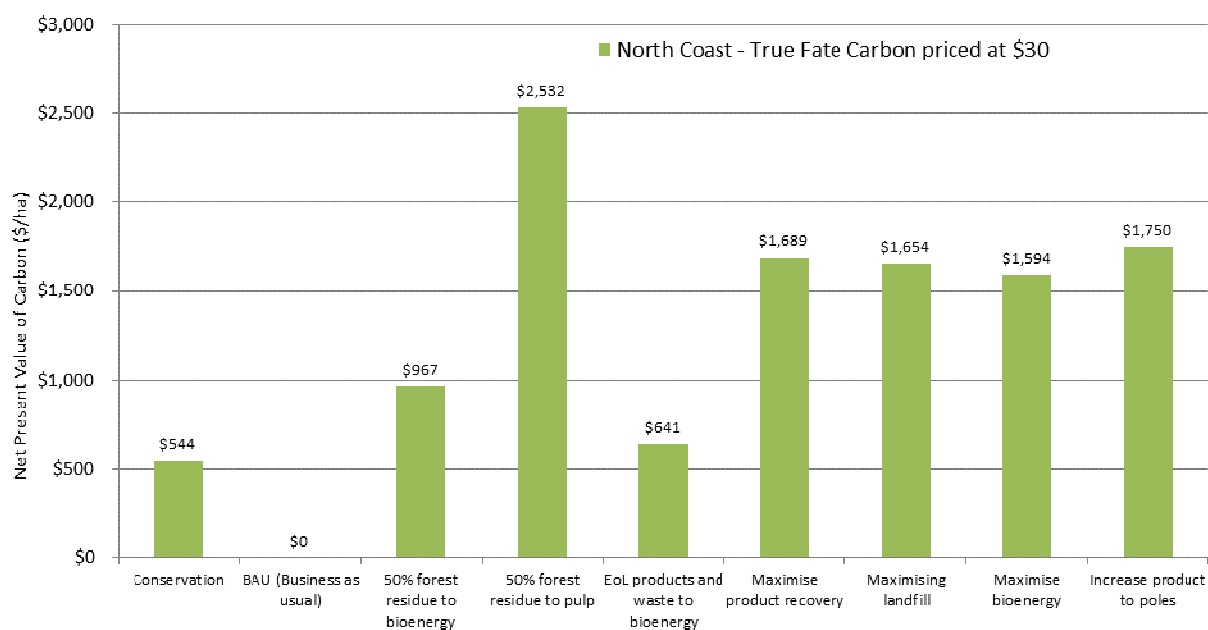


Figure 7.13 Value of carbon abatement for eleven alternative forest management scenarios with carbon priced at \$30/tCO₂-e

a) Eden

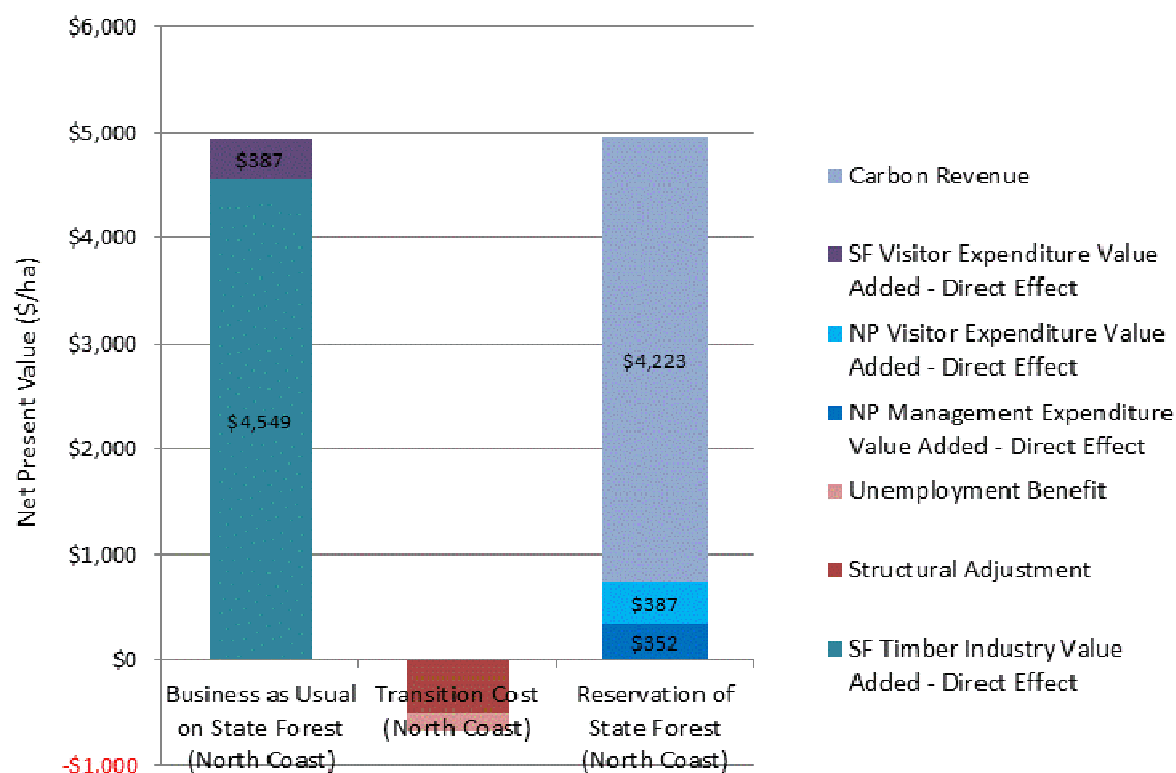


b) North Coast (Wauchope)



For the conservation scenario, no financial benefit was generated from pricing carbon at Eden (as it was less than BAU). On the north coast the conservation scenario required a carbon price of \$233/tCO₂-e for it to break even with BAU (Figure 7.14)

Figure 7.14 – Economic Impact of converting North Coast State Forests to National Parks – with carbon priced at \$233/tCO₂-e



7.4.3.5. Comparison of results – Perkins and Macintosh (2013)

Only one other study has attempted to compare the value of State Forests with and without timber harvesting. This study was published in 2013 by The Australia Institute (Perkins and Macintosh 2013).

The report by Perkins and Macintosh was based on timber harvesting in the Southern Forest Region (SFR) of New South Wales which encompasses the Eden Region (and the Eden study site). It quantified the changes in forest carbon associated with the conservation of SFR State Forests and estimated the amount of carbon abatement that may be recognised under the Commonwealth's CFI (now known as the ERF) and what it might be worth. It then compared this value with the estimated worth of the timber industry that was reliant on the SFR's State Forest wood resources.

Perkins and Macintosh (2013) found that stopping harvesting and using the native forests of the SFR to generate carbon credits offered a viable alternative to commercial forestry. This conclusion is at odds with the findings of this study.

The two studies diverged in their approach to the accounting and valuation of carbon. In the Australia Institute paper Perkins and Macintosh focused on in-forest carbon abatement (e.g. as shown in figures 7.9 and 7.10). In contrast this paper took a holistic approach giving equal weighting to in-forest and off-site carbon storage sources.

The two studies also diverged in their approach to the valuation of the timber industry. The valuation in Perkins and Macintosh's paper was based on forest harvesting and primary wood processing. These components of the industry only account for around one quarter of the industry's total value added (see ABARES AFWPS, 2015).

Perkins and Macintosh's report was also based on a predictive model that assumed that the SFR timber industry was in a precarious financial position and at imminent risk of collapse. At the time a high Australian dollar and weak global demand (linked to the global financial crisis) had caused a reduction in demand for State Forests logs. Since then the Australian dollar has fallen, the domestic market has improved and demand for log products has been restored.

This study did not attempt to predict the future profitability or viability of the timber industry. It estimated the direct contribution that native timber harvesting makes to the economy (industry value added) based on its actual economic performance over the last three years (2011/12, 2012/13 and 2013/14).

Other differences between this study and Perkins and Macintosh's included the examination of socio-economic costs. This study took into account transition costs and the cost of job losses which impose significant direct costs on government - Perkins and Macintosh's paper did not do this.

7.5. Discussion & Conclusions

7.5.1. Indicative comparison of the economic contribution of State Forest and National Parks & Reserves

Public native forests were found to generate positive regional economic impacts when managed as either State Forest or National Parks & Reserves. For the sites studied, the economic benefits of the timber industry were found to be greater than the economic benefits of visitation (recreation and tourism).

For State Forests over 80% of the value in timber industry value added was found to be generated during the manufacture of wood products. This is also where most of the jobs were concentrated.

For National Parks & Reserves high visitation levels were the primary driver of their economic benefit. The study found that most of this economic benefit is generated from a small number of high profile sites. Sites that attract high visitation tend to be close to major population centres, easy to access and or have iconic attributes such as waterfalls, majestic lookouts or adjoin the coast. On State Forests there are lesser number of these types of sites.

On State Forest recreation and tourism is not the primary business, so it was not surprising to find that the data on visitation was less comprehensive. The State Forests data used for this study was limited to recorded visitation at specific recreation sites. This may have resulted in an under-reporting of the economic impact of visitation on State Forests relative to National Parks.

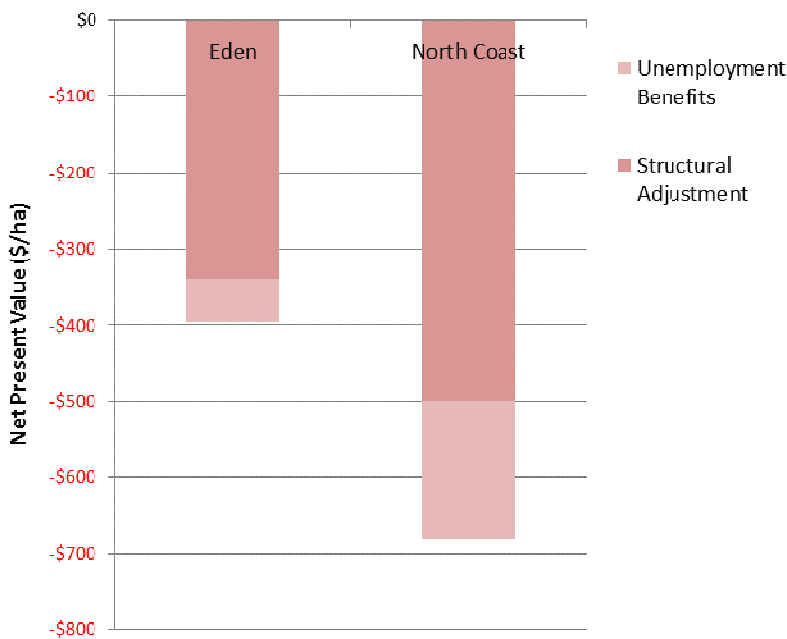
7.5.2. Quantifying the costs and benefits of transitioning State Forests to National Parks

For this study there was insufficient evidence to support either a drop or rise in visitation when State Forests were transferred to National Parks. As a consequence the calculated impact of visitation was left unchanged. State Forests attract visitors who seek different experiences to those that are available in National Parks. State forest visitors include those who enjoy car camping, car rallies, hunting, and those who like to bring their pets and their own recreational equipment (e.g. 4wd, mountain bikes and motor bikes). When State Forests are converted to National Parks, greater restrictions on forest use may result in a decline in certain recreational activities. Depending on the forest these activities may or may not be supplemented with forest users seeking alternative experiences (e.g. bushwalkers). The transfer of the Barmah State forest into the Murray National Park in 2010 is one example where visitation levels were impacted. In this case the Mathoura tourism information centre recorded a 20% drop in visitation averaged over a period of three years.

A more definitive change arising from the transition of State Forests to National Parks was the loss of the regional timber industry and the replacement of some of the native wood with imported timber. It could be argued that some of the imported wood products that replace domestic production would be subject to domestic value-adding. Analysis of the characteristics of existing imports suggests however that the bulk of the value-adding is likely to occur within the country of origin. Where domestic value-adding of imported timber does arise it is likely to be undertaken in the city, and as such does not generate a regional value-added benefit.

The cost of transitioning (Figure 7.15) State Forests to National Parks creates a financial disincentive for those who incur the cost. Apart from the obvious direct effect on individuals, structural adjustment is a one-off cost that is borne by the State. In contrast unemployment benefits are incurred as an ongoing cost by the Commonwealth. This split in responsibility means that when a decision to change forest tenure is made by the State, the legacy of that decision will be borne by both the individual and the Commonwealth.

Figure 7.15 – Cost of transitioning State Forests to National Parks at Eden and on the North Coast



The overall costs of transition were found to be dependent on the site. The following provides some explanation of the differences:

- forests on the North Coast were found to be more productive, generating more logs of a higher quality than at Eden;
- on the North Coast a greater proportion of logs were suitable for and subject to domestic value adding;
- at Eden the majority of the logs were exported as pulpwood. This prevented the capture of domestic value-adding benefits.

7.5.3 The effect of a carbon price on eleven alternative forest management scenarios

By placing a price on carbon it was possible to assess the relative value of carbon abatement under conservation and alternative production management scenarios. The true fate accounting method provided a holistic account of all major carbon pathways, reflecting the emissions that the atmosphere actually sees.

Using the true fate method revealed that industry-based carbon abatement options are superior to conservation options. More specifically, the method revealed that any management scenario which improves utilisation, both in forest harvesting and in wood processing, will generate a positive GHG mitigation outcome. Maximising the storage of carbon in harvested wood products was also shown to be beneficial regardless of whether the products were in service, buried in landfill or used for energy at the end of their service life.

The use of the true fate accounting method generated some interesting and in some cases unexpected results. For example, the best performing scenario was the utilisation of *50% forest residue to pulp* on the north coast, while the worst performing scenario was the utilisation of *100% pulpwood as bioenergy* at Eden. The driver behind these results was the very high substitution factor associated with the use of native forest pulp (Chapter 4), which was considerably higher than the substitution impact associated with the use of biomass for bioenergy.

The pulp substitution impact is calculated based on the assumption of substitution with biomass from SE Asia, from deforestation activities or from plantations recently established on previously virgin forests. If more of the demand can be met by using pulpwood from sustainably managed forests, then the pulp substitution impact will be reduced. Over time it may be assumed that the benefits of this substitution effect may dissipate. When the effect of the substitution effect is reduced the case for alternative production management scenarios such as the use of biomass for bioenergy is strengthened.

7.5.4 Key messages

- i. Analysis of the economic value of native State Forest sites at Eden and on the North Coast revealed that the highest economic impacts are achieved under continuing timber production.
- ii. Transitioning State Forests to National Parks was not justified on socio-economic grounds nor on the basis of carbon abatement.
- iii. Recognising carbon abatement opportunities through production based management scenarios was found to generate the greatest economic outcomes and the largest amount of carbon abatement.

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8. General Discussion

This study focused on several key aspects of the C cycle in native forests managed for production or conservation only. In the previous chapters the discussion sections included key points relevant to the interpretation of the results. In this section we expand the discussion to include broader perspectives and to highlight what we consider to be key messages from the study, including recommendations for future research. Firstly we list the key aspects of the work that made it novel in comparison to previously published studies:

- 1) *Over 500 mature native forests trees directly weighed.*
Most studies rely on published allometric relationships and biomass equations or non-destructive data to determine biomass in mature native forests.
- 2) *Directly weighed commercial logs and a detailed breakdown of the biomass in harvest residue types (stump, bark, crown, defective logs)*
This level of detail is never found in forest C studies.
- 3) *Landscape-level approach in the assessment of C flows, including analysis of forests under management for production that are not part of the net harvest area.*
Many studies adopt a stand-level approach in the assessment of long-term C flows, and ignore areas zoned as production forest which are not harvested.
- 4) *Creation of a new dynamic landscape-level model that captures all key dynamics in both the forest and wood products, using a life cycle assessment approach*
For Australia, typically FullCAM, with its limitations, is used in the estimation of C in native forests.
- 5) *Inclusion of fire effects and potential changes in fire frequency over time*
The impact of natural disturbance events is seldom explicitly included in greenhouse (GHG) balance studies of forests in Australia.
- 6) *Inclusion of directly weighed coarse woody debris (CWD) and litter data.*
Traditional CWD and litter and derived indirectly from transect methods.
- 7) *Detailed analysis of the C dynamics in wood at wood-processing facilities, in terms of recoveries in sawn products and biomass in residue pools*
Most studies rely on other published data or assumptions about usage rather than directly collecting it from sawmills.
- 8) *Analysis of the C implications of the post-service options for HWPs, including the latest findings on the area of the decomposition of HWPs in landfills.*
Most studies only consider C in HWPs in service, or rely on outdated default landfill decay factors.
- 9) *Detailed market-based analysis of alternative supply scenarios for native forest HWPs, in the event they were no longer produced.*
Typically similar analyses rely on generic market information.

10) Calculation of market-based product substitution factors

Most studies only include non-wood products in the calculations of substitution factors.

11) Calculation of substitution factors for paper products produced from Australian native forest biomass

C stored in paper products is often assumed to have no GHG abatement value in assessments of the GHG emissions from production forestry.

12) A more complete analysis of costs and benefits of timber harvesting in native forest.

The few studies of the costs and benefit of native forest management have been limited in scope. The analysis also includes the calculation of the C price required to offset the economic benefits of managing for production.

8.1. Biomass and C in mature native forest stands

The vast majority of the literature concerning the estimation of biomass for mature forests does not include biomass relationships based on directly weighed trees, and as a result they carry a significant degree of uncertainty. When estimating biomass in mature forests, it is critical to utilise allometric equations that include the range of DBHs of trees found in the site, and that are species-specific. Biomass in mature trees is typically more variable and challenging to estimate than for younger stands, as the structure of crowns becomes more complex with age. Another key factor is the impact of decay, which is not uniform, increasing as the trees grow older, resulting in loss of biomass, through the formation of hollows and eventual loss of limbs. Thus, it is important to include data derived from destructively weighed biomass in the development of allometric equations. This was illustrated here by the comparisons between directly weighed biomass and the use of allometric equations selected from the literature as the best match for those sites. In the case of Eden for example, the equations overestimated the biomass for the “conservation” site, with larger DBH trees. The effect was not as pronounced for the production sites, which contained a smaller proportion of large DBH trees.

While the standing biomass for the mountain ash conservation site was slightly higher than for the production site, the difference was much less than for the other sites. This may be a reflection of the higher growth rates of mountain ash during the first 50-60 years, reaching maturity between 80-120 years (Mifsud 2003, Raison *et al* 2007). The Victoria production site was approximately 75 years old at the time of harvest. Secondly the mountain ash conservation site was comprised of two distinct age classes - this is supported by the fact that there was a large amount of CWD with a DBH >30 cm observed at the site and an absence of any large dead standing trees, suggesting the likelihood of a significant past natural disturbance event. Such event would have resulted in partial tree death and triggered regeneration, thus producing a multi aged stand and reducing the total amount of standing biomass. Both the 1939 and 1926 fires burnt through this site but at a low intensity. Adjacent stumps have been dated through ring counts to originate from both these fires.

It is difficult to determine though how much more biomass a site for which the age difference in relation to the production site was significantly greater than the 33-34 year gap in this study would sustain, as decay would potentially become a major factor reducing tree biomass as mountain ash trees mature. However this could not be tested in this study as unfortunately we did not have access to a site with trees as mature as for the NSW “conservation” sites. For mountain ash, the forest at 250 years is dominated by perhaps 20 trees/ha, and a relative

decrease in biomass is expected, as the tall open eucalypt forest progresses from aggrading to a steady-state (Attiwill 1994). Mountain ash is a relatively fast growing species, with some trees less than 70 years old being more than 80 m tall (Sillett *et al* 2010). According to Sillett *et al* (2015), such rapid growth likely occurs at the expense of fire and decay-resistance (Loehle, 1988), thus reducing their maximum longevity. This, coupled with visual observations of mountain ash old-growth stands, suggests that the onset of decay in old-growth sites is likely to occur relatively early in the old-growth stage of those stands, with significant biomass implications. The findings from Sillett *et al* (2015) suggest that decay is indeed a major factor to consider in the estimation of biomass for very old mountain ash stands. The authors identified limitations in the development of biomass equations for mountain ash 60– 100 m tall based on 27 trees distributed over a wide area. Those were associated with difficulties in the replication of certain measurements from year to year, no accounting for decay, and increased risk of biomass overestimation especially for the irregular crowns. Notwithstanding these limitations, the equations were applied to a 0.73 ha plot in Kinglake National Park to estimate the aboveground biomass and C mass of this forest prior to the 2009 fire. This plot was considered by the authors to represent the upper level of density of large trees for mature mountain ash forests in Victoria. The maximum aboveground C mass was 706 t /ha, which was considered maxima because as stated above it does not account for mass loss due to decay in living trees. The authors noticed extensive decay and numerous hollow trunks and limbs. If half of the mountain ash heartwood volume were lost to decay, stand-level C mass is reduced to 438 t /ha. This value is similar to the estimated value for our 1905/1906 mountain ash conservation site (410 t C/ha).

Key Message 1

Biomass estimates of mature native forest stands that are not based on directly-weighed biomass including trees with the largest DBH range are generally not reliable.

8.2. Inclusion of CWD

The inclusion of C in CWD pools has important implications for the estimation of total above-ground C in native forests. Although the inclusion of CWD and litter had immediately opposite effects on the total above-ground forest C for the NSW and VIC sites following a harvest event (increased totals for NSW production forests compared to conservation sites, and reduced totals for VIC production forests compared to conservation sites), in the long-term, when the long periods between harvest events and relatively shorter decay timeframe for residues are considered, the relative differences between production and conservation sites are changed. For Victoria, the long-term C in CWD averages for the conservation sites are almost double those of equivalent production sites, whereas for the NSW sites the differences are much less, but with long-term C in CWD in the conservation sites also higher than for the equivalent production sites. Inclusion of CWD and litter increased the overall differences in the total above-ground forest C for NSW production forests compared to conservation sites slightly (by approximately 2%) and by 8% for mountain ash forests. However there is considerable uncertainty around the decay dynamics of CWD in native forests – this is an area that requires further investigation.

8.3. Development of allometric equations

The development of allometric equations is key to the ability to extrapolate biomass estimates from the plot level to a landscape level. According to Fedrigo *et al* (2014), of the limited species-specific allometrics for Australian trees, uncertainty stems from the applicability of allometric equations across a range of diameters (especially for large trees), species

composition, accounting of additive elements, and the representativeness of locations from which the trees were harvested. As stated earlier, the inability to account for the impacts of decay in mature trees is another major limitation of commonly used allometric equations.

The allometric equations presented here were based on data for hundreds of trees individually weighed. In our study, inclusion of height as a combined variable with DBH in the development of the additive biomass equations did not result in higher R^2 for the estimation of whole tree biomass – these were already high with the inclusion of DBH only. Fedrigo *et al* (2014) found that equations that use only DBH as an input variable resulted in a wide range of C estimates for trees of the same species and measured height – as a result of distortions created by stem breakage at height, as observed for both *Eucalyptus* spp. and rainforest species (Keith *et al*, 2000, Woldendorp and Keenan 2005). Sillett *et al* (2015) report that in at least two cases where sufficient effort has been expended to develop equations suitable for species in old-growth forests, measurements besides DBH (e.g., crown volume) provide stronger prediction of biomass and other aboveground attributes. Further work is required though to refine the model specifications and parameter estimations from our study, and refinements in the handling of data from plots that are spatially clustered – these may improve the correlations further. The analysis also needs to be expanded further to include other species present in the study sites.

8.4. HWP – Long-term C storage

Having discussed key aspects related to biomass estimation in mature native forest stands, we turn our attention to HWPs. The C dynamics in HWPs have been ignored or treated in a simplistic way in a range of earlier studies that attempt to quantify the GHG implications of forest management for wood production in Australia (e.g. Mackey *et al* 2009, Keith *et al* 2014). This omission resulted in an incomplete assessment of the implications of native forest management for wood production and the promotion of a forest management policy that favoured management of native forests for conservation only. Examples of key parameters often ignored or erroneously applied include the role of C storage in HWPs in landfill, the fossil-fuel displacement effect created by the use of biomass for energy and the product substitution effect. Ximenes *et al* (2012) demonstrated the impact of the inclusion of C storage in HWPs and the product substitution effect on the relative GHG balance of native forest management, leading to vastly different conclusions from the studies mentioned above. However, Ximenes *et al* (2012) also highlighted areas where improvements could be made to refine the data underpinning those analyses further.

The data used in this study for the determination of the physical C storage in HWPs was based on information directly supplied by wood-processing facilities and on the latest research findings on the dynamics of the decomposition of HWPs in landfills. Long-term C storage for HWPs in Australia is primarily imparted by the post-service stage of the HWP life (i.e. storage in landfill). It is commonly assumed that wood in landfills decays, generating large quantities of methane; however, research that has specifically targeted the decay dynamics of HWPs in landfills has demonstrated that C in HWPs in landfills (other than paper products) can be considered to be stored for the long-term, with a real mitigation benefit. However, landfilling is not the only way to achieve this mitigation benefit. There will be benefits also when the product is recycled into another long-lived application (e.g. old floorboards used in recycled furniture), or burnt to produce energy. The emission abatement created by diverting wood waste from landfill to energy generation facilities (resulting in fossil fuel displacement) may be higher than if the product is placed in landfill, depending on whether/which fossil fuels are displaced. For paper products, the dynamics of decay in landfill

are significantly more complex than for non-paper HWPs, and to a large extent driven by the extent to which lignin is present in the paper product (e.g. Wang et al 2015). Another factor to consider includes the extent of methane recovery (most modern landfills recover methane and flare and/or generate electricity from the methane recovered, displacing the use of fossil fuels), which combined with a level of long-term C storage, would likely result in a positive outcome from a GHG perspective for paper products as well. However this level of analysis was outside the scope of this study.

Key Message 2

Considerations on disposal options for HWPs are critical for the analyses of the GHG balance of native forest management. Long-term C storage in HWPs in Australia is largely driven by the C storage that happens in landfills. However, whether the HWP is placed in landfills, recycled, or used to generate bioenergy, the GHG outcomes will be positive.

8.5. Substitution impacts - biomass for bioenergy

In addition to understanding the long-term implications of the physical C storage in HWPs, the use of biomass to generate bioenergy is another important aspect of the quantification of the GHG impacts of native forest management. Unlike C storage in HWPs, the mitigation benefit of the use of biomass for bioenergy is dependent primarily on whether/which fossil fuel is displaced. Lippke et al (2011) report that when wood is used to displace coal, which would be the likely offset from using forest residues in utility-power generation, approximately 1.9 tons CO₂ is displaced for every ton of forest residues used (or approximately 1.0 t C / t C in harvest residues), which is similar to the figure used in our study for use of harvest slash in electricity generation with coal (0.8 t C / t C in forest residues, derived in Ximenes et al 2012). The life cycle information collected in wood-processing mills by Lippke *et al* (2011) suggests a reduction of approximately 1.2 tons of CO₂ for every 1.0 ton of wood biofuel used in product-processing mills (approximately 0.7 t C / t C in processing residues), which is in agreement with factors derived by Klein et al (2013) for forests in Bavaria. These values are consistent with values we derived in our study for commercial energy generation that displaces the use of natural gas or heating oil (between 0.5-0.7 t C / t C in processing residues). The average substitution factors used by Oliver et al (2014) for the calculation of the impact of using residues instead of natural gas or heating oil were higher the factors we derived here. Thus the energy emission factors used in our study are either consistent with other published estimates or lower.

8.6. Substitution impacts – HWP other than paper

In addition to the fossil-fuel displacement benefits of using biomass for energy generation, it is also important to consider the GHG mitigation benefits associated with the use of HWPs in lieu of more GHG-intensive alternatives. The substitution impacts determined for HWPs produced in each case study region varied considerably. Without considering biomass used for pulp production, the substitution impacts on a hectare basis were driven largely by the productivity of the site, the relative efficiencies of biomass recovery in the forest, sawmill recoveries, the types of HWPs produced and the displacement options for each product.

Considerable efforts were made to derive region and product-specific product substitution factors for the key hardwood HWPs from each region. Key replacement markets were often comprised of wood or wood-derived materials, as the native hardwood HWPs often occupy a

niche; i.e. consumers who would most likely want a “wood” replacement if they no longer had access to Australian native forest HWP. Even though the native forest HWPs in most cases had significantly lower displacement factors compared to the alternative products identified, the fact that the alternative products were often also HWPs reduced the potential impact of the product substitution (with the exception of imported hardwoods from SE Asia and some EWPs). This approach contrasts to how product substitution is typically quantified, where product substitution is calculated as the difference in the GHG emission footprint of HWPs versus the use of non-wood materials. (e.g. Sathre and O’Connor 2010; Lippke *et al* 2011; Oliver *et al* 2014). Using this approach, Sathre and O’Connor’s meta-analysis of twenty-one international studies (Sathre and O’Connor 2010) concluded that displacement factors range from a low of - 2.3 to a high of 15 tonne C / tonne of C in HWP, with most lying in the range of 1.0 to 3.0 t C / t C in HWP, with an average of 2.1 t C / t C in HWP. The weighted substitution factors derived for each case study region in this report (excluding paper products) ranged from 0.2 for mountain ash HWPs to 2.1 t C / C in HWP for silvertop ash. The factors derived for this study are conservative, since as mentioned earlier they take into account the relative emission footprint of all realistic alternative products. This is exemplified in the calculation of the substitution impacts of producing pallets made from mountain ash; the most likely replacement product was untreated pine, which has a very similar emission footprint to the hardwood pallets, and thus resulting in negligible substitution impacts. Under the typical approach adopted in most other studies, we would have used plastic pallets as the most likely non-wood replacement, generating a large substitution impact, as plastic is made from oil (a fossil fuel) and is much more emission-intensive in its manufacture than wood pallets (e.g. Philip 2010).

There has been some debate about the use of product substitution factors and the validity of some of the assumptions underlying the calculations (most notably Law and Harmon 2011). Some of the key areas for discussion include how the substitution factors are applied in the context of the forest C dynamics, considerations on long-term C storage in landfills, and the application of additionality and leakage principles. These points are discussed in detail below.

Law and Harmon (2011) put forward a number of arguments cautioning against the use of product substitution factors. They argue that since the forest has a maximum C carrying capacity, eventually just the growth in C stores in HWPs and fossil-fuel offsets would exceed old forest C, although it could take centuries to happen, even using the most generous substitution effects. The assertion that it would take centuries for the production scenario to incur a more beneficial GHG outcome than in a conservation-only scenario is certainly not true for the systems included in Ximenes *et al* (2012) and not true for the systems considered in this study. It is misleading to derive comparisons using a “fixed” forest stand approach, where the fate of C throughout the cycles of harvest and growth is tracked. In reality, native forest management typically involves the management of forest stands at various age classes, as harvest takes place each year in different parts of the landscape. Therefore, it is preferable to consider the long-term average C stocks for the forest, which are reflective of the long-term impact of management. It is important to remember that in most developed countries, including Australia, native forests have been managed for wood production for many decades. Another reality that is often ignored when we consider a “fixed stand” approach is that a substantial proportion of the total forest estate managed for production often includes areas that are not accessed for harvest, due to factors such as proximity to riparian zones, steepness of the terrain, high-value conservation areas, aboriginal heritage and due to biodiversity considerations. For our case studies, accounting for these resulted in typically 50% of the area of the forests managed for production all the case study areas deemed not accessible for harvest.

Another argument raised by Law and Harmon (2011) is that, in substitution impact calculations, it is often tacitly assumed that wood that is removed from forests and used in long-term HWPs, specifically buildings, continues to accumulate infinitely over time. This statement refers to the saturation or steady state that eventually takes place when HWPs in service are considered. Although this is the case for HWPs in service, it is certainly not true when the fate of HWPs post-service is considered. Implied by Law and Harmon (2011) is the assumption that the C in the HWP will at some point be returned to the atmosphere. Whilst that is true if the HWP decays in service or is burnt, the reality is that disposal in landfills as discussed earlier will result in long-term C storage, that does accumulate over time. Furthermore, use of the redundant HWP to produce bioenergy has immediate and irreversible GHG benefits by typically displacing the use of fossil fuels.

Law and Harmon contend that while building C stores have increased in many areas (e.g., the USA), this is largely because more forest area is being harvested, and not because the harvest related stores per harvest area are increasing. As substitution impacts have traditionally been evaluated on a fixed area basis and under the assumption of sustainable forest management, the origin of the timber does not influence the calculations. They also argue that in most studies, the substitution offset is calculated based on the assumption that each time a house is to be built, the preference is for nonwood materials. This results in an estimate of the maximum substitution effect possible, but does not account for actual preferences for building materials. The assumption that substituting for non-wood products will always lead to a maximum substitution effect is not true, as evidenced in our study by the very high emission factors associated with the use of unsustainably harvested HWPs from tropical forests. Notwithstanding this fact, once again this argument is not relevant to our study, since we considered market-based alternatives to the HWPs in the case studies, regardless of whether they were wood on non-wood based materials.

Finally, Law and Harmon state that current analyses of substitution effects ignore the principle of permanence and the effects of additionality (whether wooden buildings are initially present). According to them, given that many forests have already been harvested to produce HWPs, replacing wooden buildings with more wooden buildings results in no additional substitution effect. This line of thinking applies only in the situation where a specific HWP has complete monopoly over a market sector – in reality this will never be the case. As seen in the detailed market analyses in Chapter 3, the reality for the case study areas included here is typically the opposite, with native hardwood HWPs usually occupying a comparatively small segment of the market. The principle of permanence, by definition, cannot apply to substitution factors, as it does not involve physical storage of C. Quantification of the product substitution impact that happens at any given year is no different from the displacement of fossil fuels by bioenergy; the principle is the same. The C in the bioenergy option is burnt and emitted immediately back into the atmosphere; however its impact is permanent, as its use prevented the use of alternative options with a much higher GHG balance (i.e. fossil fuels). Substitution produces permanent offsets, by generating reductions in the one-way flow of fossil emissions at the time of harvest and wood use, independent of the products useful life, whilst retaining the C in the HWP (Lippke *et al* 2011, Werner *et al* 2010). Each time there is a choice between a product with a low GHG intensity and a product with a high GHG intensity, by choosing products with lower GHG intensity the consumer has a concrete impact on GHG emissions, by reducing the demand for GHG-intensive products. As pointed out by Lippke *et al* (2011), substitution for energy has been more readily accepted since the biomass being used results in only a narrow range of impacts across a few different fossil energy uses, reducing the range of uncertainty. Ironically, by contrast, the substitution of wood for fossil intensive products not only results in storage of C

in the product for the life of the product, but it also displaces emissions from fossil intensive products, with much higher leverage than using wood for bioenergy.

Key Message 3

The substitution impact, when based on market analyses of product usage in different applications, represents a real mitigation benefit, in the same way the use of sustainably sourced biomass for bioenergy generation represents real mitigation when it displaces the use of fossil fuels.

8.7. Substitution impacts – pulp and paper

Although many studies consider the product substitution impacts associated with the use of sawn timber and engineered wood products, the contribution of paper products to the GHG balance of production forest systems is often ignored due to their typically short service life. However, the key factor to consider for the determination of the GHG balance of paper products is product substitution. In both the Central Highlands of Victoria and in Eden, the production of pulp logs is part of integrated harvest operations, in forests that are considered to be sustainably harvested. In contrast, based on an extensive analysis of current and future markets, we concluded that the key alternative market for paper production is based in SE Asia, primarily in Indonesia. The emission footprint associated with the extraction of biomass for pulp and paper production in SE Asia is very large, due to high levels of deforestation of primary forest, forest degradation and loss of peatlands. It is important to acknowledge that, in a global economy, any changes to an important market in an individual country will have flow-on effects in different countries. The market forecasts are for increased paper consumption globally, fuelled by the emerging middle class in large countries such as China, India and Brazil. It is a reality that paper will continue to need to be produced somewhere into the foreseeable future. If native forest biomass currently sourced from the Central Highlands of Victoria and from Eden was no longer available, logically this would add further pressure to the degraded and depleted forest areas in other areas of the world with comparatively lower standards of forest management.

There is considerable uncertainty though about the magnitude of the product substitution factor for paper, as discussed in detail in Chapter 4. In the calculation of the factor, every effort was made to ensure that the underlying parameters required were conservative. In order to test the impact of the magnitude of this factor on the relative GHG balance of the mountain ash and silvertop ash forests, we tested the use of a substitution factor that was half the average factor used here. For mountain ash, the relative differences between the GHG balance of the production and conservation forests would be reduced from 58.4% to 19.5%, whereas for silvertop ash, halving the paper substitution factor would result in a switch from a benefit of 4.8% for production forests, to a benefit in the order of 15.2% for conservation forests. This demonstrates the significance of the pulp substitution factor in the overall GHG balance assessment of those forests.

Key Message 4

The inclusion of the product substitution impact for pulp biomass has a very large impact on the GHG balance of production forestry in the regions where pulp logs are extracted. Ignoring this impact would majorly underestimate the GHG benefits of native forestry in those regions.

8.8. Forest HWP

The parameters described above were used in a new model – “ForestHWP” - developed to undertake the required integrative analyses for each study region. Prior to this study, the most comprehensive assessment of the GHG implications of native forest management in Australia was described in Ximenes *et al* (2012). In that study, the assessment of C flows as a result of native forest management was derived from modelled estimates of the GHG balance of two key native forest areas in New South Wales for a period of 200 years. The above-ground biomass C predictions of large forest areas were derived from an empirical model used to calculate long-term wood supply volumes from native forests. That approach differed fundamentally from the one adopted in this study, where we selected large enough plots to account for the typical profile of the forest types selected, individually weighed every tree in the plot with DBH greater than 10 cm, and also comprehensively accounted for the coarse woody debris and fine litter in the sites. Paired sites were selected based on their maturity; sites managed for production, deemed to be of “average” quality and “ready for harvest” were selected for the “production” scenario; matching sites (based on species type and location) that were largely unmanaged, undisturbed by recent fire events and which contained a significant cohort of trees that were substantially older than the equivalent “production” site were selected. These sites were then used as the benchmark against which the long-term C dynamics in the forest and HWPs were modelled using ForestHWP, a model specifically created for this project.

Additional key differences between the current study and that of Ximenes *et al* (2012) include the incorporation in this study of:

- Estimates for C in the coarse woody debris and fine litter pools;
- Site-level determination of live forest biomass
- Long-term impact of fire events
- Temporary long-term C stocks in HWPs in service
- Product-specific substitution impacts, including for pulp
- Cost-benefit analyses associated with native forest management

The analyses conducted here were extrapolated from a limited number of sites to a significantly larger area of forests, which were deemed to be a good match for the study sites. The good agreement between standing volumes and production log recoveries for the study sites and for the extrapolated areas (as obtained from the relevant State Forest agencies) provided the confidence required for the modelling to be extended to the larger relevant areas.

The C in the production management areas that are not subjected to harvest for a number of reasons (e.g. harvest prescriptions, aboriginal heritage, biodiversity considerations) was modelled using the same assumptions as for the conservation scenario, reaching similar C stocks. The impact of including these areas in the modelling for the production scenario is an increase in the forest C for the production scenarios, coupled with a reduction in the HWP production on a hectare basis. In our study, the average long-term forest C for the NSW sites was approximately 60% that of the conservation equivalent, whereas for mountain ash in the

Central Highlands of Victoria the figure was approximately 73%. In a study for North American species where only the areas actually available for harvest were modelled, there was a reduction in the average C across the rotation to less than half that of an unharvested forest depending upon region, forest type and management treatment (Lippke *et al* 2011). It is important to account for the C in the areas that are not available for harvest within the production estate, as this area is significant (approximately 50% of the total for all study areas here), with important non-production values that are supported by the commercial returns from selling the production logs.

A number of scenarios were considered in the modelling with ForestHWP in addition to the BAU scenarios – these included changes in biomass usage for bioenergy, the impact of switching product types and disposal options. The results indicated there is considerable room to improve the C benefits associated with managing forests for production. There are a number of management strategies at the forest level, at the wood-processing facilities and at the disposal stage that could make a considerable difference to the net impact of production. All scenarios that do not involve changing from pulp to biofuel (for silvertop ash and mountain ash) result in net C benefits. In some cases, changes to the way individual products are treated can have a major impact on the overall C balance. For example, favouring the production of electricity poles over the extraction of sawlogs in the North Coast of NSW results in a net C benefit over the long term equivalent to approximately 70% of the long-term average above-ground C stocks for the *Eucalyptus pilularis* (blackbutt) production forests.

Key Message 5

In general, strategies that involve increased utilisation of biomass for bioenergy production, extending the longevity of C in the HWPs (by changing production from short-lived products to medium to long-lived products and by storage in landfills) and general minimisation of waste all contribute to a greater net C benefit for all systems studied.

8.9. Findings from similar international studies

In addition to the key references for Australia listed in Chapter 4, there have been a number of studies, primarily in North America and in Europe, which have attempted to quantify the GHG implications of management of forests for production or conservation purposes only. In two separate studies in Canada and in the US (Colombo *et al* (2012) and Krankina *et al* (2012)), both the product substitution impact and fossil fuel displacement impacts due to the use of residues or redundant HWP for energy production were ignored. Krankina *et al* (2012) assessed changes in forest C stores in old-growth forests, with the aim to provide an upper limit for C storage. The main aim of the study was to analyse the effect on forest sector C stores of varying levels of timber harvest in federally managed forest lands within western Oregon and western Washington. The conservation scenario resulted in an annual increase in C of 2.49 Tg C/year, whereas the production scenario resulted in an annual loss in the order of 2.17 Tg C/year.

Colombo *et al* (2012) investigated the effects of management strategies on trade-offs between forest C stocks and ecological sustainability under a number of management scenarios, for 3.4 Mha of forests in north-eastern Ontario, Canada, at the interface between the temperate hardwood and boreal forest zones. The analysis of C dynamics in HWPs was limited, as it considered only long-term C storage in HWPs in landfills and ignored the product substitution and fossil fuel displacement impacts associated with the HWPs. Thus the impact of management for product was underestimated. After 100 years of adding C in HWPs to that in regenerating forests, total C storage was equivalent or greater than the forest C in protected

areas. The study also seemed to ignore C stocks in CWD. Klein *et al* (2014), on the other hand, did not consider long-term C storage in landfills, as landfilling is not a common practice in Germany. However they did consider both product substitution and fossil fuel displacement impacts in the assessment of the impacts of management of key species in the Bavarian region. Mitigation effects in unmanaged spruce and beech stands did not differ in the first decades from their managed counterparts, but were below that from stands managed for production in the long term (Klein *et al* 2014).

8.10. Cascading approach

There are many studies, especially in Europe, that advocate a ‘cascading’ approach as a way of deciding the most GHG-friendly uses for HWPs. This approach favours long-term C storage in HWPs in service (preferably in structural components), simultaneous product substitution effects, and importantly, at the point of disposal energy use with energy substitution effects (e.g. Sathre and Gustavsson 2006, Werner *et al* 2010, Klein *et al* 2013). This approach may not always result in the best outcome from a GHG perspective. If the choice for the management of the HWP post-service is between a short-lived product (e.g. mulch), bioenergy generation or landfill, it is clear that the production of a short-lived product which decays quickly in service is the least preferable option from a GHG perspective. The choice between energy generation and landfilling will depend primarily on the energy profile of the region. For example, if the HWP was used in a location in Tasmania, where the energy used was predominantly hydro-based (and thus with a low emission profile), landfilling is likely preferable from a GHG perspective. In Victoria however, the predominance of brown coal for energy generation would suggest potentially a more beneficial outcome if the HWP was used for energy generation instead of landfilling. It is important to note that landfilling results in actual physical storage of C for the long-term, guaranteeing that the C will not be emitted. The net benefits of bioenergy generation as noted earlier will depend primarily on the alternative energy sources for the particular region. Furthermore, the energy substitution impact may change in a relatively quickly period if the energy matrix for a specific region changes, whereas physical long-term C storage is more secure. Thus caution should be exercised when adopting approaches like cascading in a dogmatic manner when deciding on the best GHG outcomes for HWP management without proper consideration of the energy mix and market situation of the region in question. This may lead to sub-optimal GHG mitigation outcomes.

8.11. Accounting issues – timing

The timing of the relative impacts of management can be greatly influenced by the selected starting point for the simulations; for instance the creation of a C debt that occurs when stand-level simulations start immediately after harvest will take some time to be overcome and potentially bias outcomes. That is why it is more appropriate to use long-term average stocks for the various pools in the assessment of the relative impacts of management, as that reflects reality more closely (i.e. most commercial forest stands have been managed for production forest for decades). Using a single stand or project to model a wood supply system can severely distort system-wide outcomes (Lucier 2010). At the scale of a wood supply area, sustained-yield forestry and sustainable management systems keep growth and removals in balance, and the loss of C from harvests in any given year is equal to gains in C elsewhere in the area (Malmsheimer *et al* 2014). The benefits associated with using products from forest-based systems, however, often continue to accumulate, as demonstrated in this study and in previous studies (e.g. Schlmadinger and Marland 1996, Lippke *et al.* 2010, 2011). Thus, the simulation periods need to be long enough to properly capture this effect. Extending the

period of analysis through multiple rotations can be critical to understanding the short-term and long-term GHG implications of using forest-based products (Malmsheimer *et al* 2014).

8.12. Socio-economic implications

In addition to understanding the GHG implications of native forests management, it is important to consider the socio-economic implications associated with managing for production or for conservation only. Too often in the discussion around the implications of land management from a GHG perspective, the socio-economic impacts are ignored. In the case of the NSW native forests included in the cost-benefit assessment, the beneficial socio-economic impacts of management for production, under the assumptions used, far outweigh the impacts associated with management for conservation only.

Whenever a new business proposition is made in any sector, a comprehensive cost-benefit analysis is always required. Yet, in considerations regarding the future of a whole industry sector such as native forestry based on GHG issues, this is often overlooked. It has been argued that managing forests for conservation only will provide C benefits, which may be associated with a monetary value. It is important to understand how this hypothetical value may be considered against the impacts of reducing or stopping production on the primary and secondary market chains associated with the production of HWPs from native forests.

For a scenario where there is a C credit associated with the additional growth assumed in the case native forest harvest was stopped, it must be an improvement on BAU to be justified on economic grounds. At Eden the “conservation” management scenario was unable to outperform the BAU regardless of the C price while for the north coast site a C price of \$233 per tCO₂-e was required to outperform the BAU.

Key Message 6

When industry value-added benefits and C abatement benefits are added together, the production management scenarios generate much higher value than the conservation management scenarios, independent of the carbon price (low, medium or high).

8.13. Potential climate change impacts

The impact of climate change on the future management of the forests included here was outside the scope of the study. Climate change may have especially significant impacts on the frequency and severity of wildfires, with important implications for the long-term average C stocks in those forests. Anticipated increases in natural disturbance resulting from global warming may further reduce the climate change mitigation potential of forest conservation in disturbance-prone ecosystems (Sharma *et al* 2013). Sharma *et al* (2013) also observed that, on the other hand, global warming may cause an increase in forest productivity in some areas, as observed by Hember *et al.* (2012) for Coastal Douglas fir and Western Hemlock on coastal BC. This which would result in an increased uptake of CO₂ sequestration rates by these forests. According to Fedrigo *et al* (2014), predicted changes towards more frequent and intense wildfires under climate change could facilitate the death and collapse of large trees, and limit recruitment of both eucalypt and rainforest species, leading to new stand structures and potentially a new C dynamic in these forest landscapes. The outcome of such changes would likely result in reductions in the C-carrying capacity of the new forest stands, especially due to the impact of more frequent and intense wildfires. The magnitude of such changes in the temperate forests of south-eastern Australia needs to be better understood.

9. Conclusions

This study has contributed a number of novel aspects to the understanding of the C dynamics of native forest management in Australia. Such comprehensive assessment is required in order to more fully understand the GHG implications of native forest management.

The forest biomass estimates were as much as possible based on direct weighing of the relevant C pools. This approach ensures a degree of certainty in the results that cannot be associated with previous studies that relied on indirect measurements only, or with studies that relied on existing allometric equations, which typically do not include the range of sizes that exist in mature native forests. This ensured that the results from the different case study areas could be compared on the same basis. We demonstrated that existing allometric equations for the relevant species in this study are generally poor at estimating biomass for mature trees. We demonstrated that inclusion of height in the biomass equations generally did not result in substantially improved correlations in our site-specific allometric equations, as the R^2 obtained using DBH alone was already high for all species.

We demonstrated that conservation forests hold significantly more C in the long-term than the equivalent forests managed for production in NSW, and to a lesser extent in Victoria, when forest C only was considered. There were considerable differences in the C stocks both in the forests and offsite for all case studies. This demonstrated the importance of selecting case study areas that covered species that were intrinsically different in characteristics, in their response to management and natural disturbance, HWP mix and fate of HWP post-service. The mountain ash production forest was highly productive compared to the NSW production forests. The natural form of mountain ash trees (low branching, low bark formation, straightness) contributes substantially to a very high recovery after harvest. The silvertop ash production forests to some extent had opposite characteristics to the production mountain ash forest, with significant branching, very high bark formation (approximately 3x that for mountain ash) and with considerable decay. All these factors resulted in lower biomass estimates for the production silvertop ash site, and much lower recoveries than for mountain ash. The log recovery for North coast blackbutt was lower than for the other regions due to a lack of a pulp or biomass market for that region. Inclusion of those markets in the future may substantially change the overall net C benefit for those forests.

The recoveries for silvertop ash at the mill were low primarily due to the fact that all green rough sawn boards were dried and dressed (i.e. no products were sold “green”). Silvertop ash decking is high-valued for its bushfire resistance. For blackbutt forests, a change in management to target extraction of poles for electricity transmission rather than sawlogs had a major impact for the C footprint of those forests.

We demonstrated that the use of product substitution in the context of the analysis of the C footprint of forestry systems needs to be regionalised, ensuring scenarios are realistic, based on market research and inclusive of both wood and non-wood alternatives. We have also demonstrated that accounting for the long-term C storage in HWPs in landfills can have a significant impact on the C footprint of production systems.

Depending on the region, we demonstrated that optimising physical C storage (long-lived HWPs plus landfill) leads to a more positive outcome than optimising diversion of biomass to bioenergy generation. This suggests that a “cascading” approach as suggested by some authors may not always lead to the most optimal GHG outcomes. Ultimately though, any of the key post-service options will lead to a positive outcome from a GHG perspective.

The inclusion of paper substitution impacts for the case studies where pulplogs were extracted was the most important factor driving the C footprint for those regions. Although there is considerable uncertainty around the calculation of the product substitution factors for paper, they were typically consistent with those reported in the literature for SE Asia. Thus it is important to highlight the important GHG contribution that extraction of pulp logs from those regions make, especially given that paper products are typically dismissed as short-lived products that do not have a GHG mitigation role.

The impact of wildfires on long-term forest C estimates was not as large as potentially expected. However it is important to understand that the analysis was somewhat limited, as factors such as potential changes in severity, potential for co-occurring effects of droughts and pests and potential failure for systems to recover after repeated fires were not tested.

The overall conclusion of this study is that the relative differences in the GHG balance of the case study regions do not warrant policies that aim to halt native forest management for wood production. In addition to timber production objectives, management of production forests in the study regions also takes into consideration non-production factors. These include retention of ecological values, wildfire management, positive socio-economic implications for the regions, generation of revenue to support trained fire-fighting crews and maintenance of roading networks required for quicker access to fire fronts. There is considerable room however for improvement in the GHG outcomes for all the three case study regions included in this study. These could be concretely achieved by a combination of reduced wastage, increased recoveries, increased physical C storage in HWP and increased fossil-fuel displacement via the use of biomass to produce energy.

9.1. Sources of uncertainty and recommendations for further research

There are a number of areas highlighted in this study that would benefit from further research. The differences in many aspects of the C dynamics both in the forests and in HWP for the key native forest production areas suggests that extrapolation of the results into other regions may not be warranted without similar analyses being conducted. There would be value in conducting similar analyses for other native forest types in NSW (e.g. river red gum, cypress pine, spotted gum), spotted gum forests in Queensland, “Tasmanian oak” (alpine ash, mountain ash and messmate) in Tasmania and jarrah and karri in Western Australia. Such studies would be useful in determining the current GHG balance of production for those areas, and highlighting opportunities to improve on current management practices.

As pointed out in Chapter 5, there is significant uncertainty around the dynamics of below-ground C (soil and roots) and the impact of management over time. Inclusion of below-ground C dynamics in the overall assessment of the C implications of native forest management would make the analysis more complete, and would also enable the development of strategies that impact on below-ground C and that optimise the overall GHG outcomes.

There is considerable uncertainty about the patterns of decay onset for different native forest species, and importantly, its impact on biomass loss as the trees grow older. This is one of the key drivers for C estimations in older forests. In recognition of the difficulties in accessing a large enough number of mature trees in varying stages of decay for direct weighing, a new approach is required that would result in greater confidence in the application of allometric relationships for those larger trees. Such an approach may involve a combination of innovative techniques (e.g. acoustic devices, modified micro-drilling). This is potentially the area of mature forest C research that would probably result in the biggest contribution towards

reducing current uncertainties in forest C estimates. Decay in mature forests may be correlated to age, species, basal area and management interventions (e.g. thinning) – these relationships need to be examined further.

The dynamics of C in dead stags post fire is poorly understood and has significant implications for the C balance, especially in mountain ash forests, which are subjected to stand-replacing wildfires. Related to this, one of the limitations of our study was in the underlying assumption that the forest systems always recovered from natural disturbances. It is clear from evidence in alpine ash forests for example (Bowman *et al* 2014) that repeated wildfires in quick succession may result in landscape-wide loss of mature forests. This will have potentially significant impacts on long-term C stocks, especially in forests that are subjected to stand-replacing fires and which are dependent on regeneration from seed.

The study highlighted opportunities associated with the use of the large volumes of residues produced both as a result of forest harvest operations and sawmilling. Further research on potential opportunities for conversion of residues into higher value, long-lived products may be warranted. Such projects may be eligible to claim C credits under the Emissions Reduction Fund, as currently the C storage value of the biomass that is left to rot or burn in the forest, or processed into a low-value, short-lived application at the sawmill (e.g. mulch), is limited.

It is important to acknowledge that the product substitution factors for individual HWPs may change significantly as a result of changes in the market. Thus, it is important to revisit the market conditions regularly by conducting market-based research. A significant change in the market share for different products would require revisiting the product substitution factors.

The product substitution calculations relied to some extent on published values that were the best match for the respective products. However, there is considerable room for refining these factors by identifying key products or production systems to focus on for emission footprint studies. In particular there are considerable uncertainties associated with the calculation of emission footprint factors for pulp and paper production in SE Asia. A more comprehensive assessment would require a more thorough analysis of the linkages between deforestation and different industry sectors. This is complicated here by the fact that there are multiple drivers for deforestation – sawlog extraction, pulp, agriculture, establishment of timber plantations and establishment of palm oil plantations to name a few. Considering that almost all deforestation data is at a large scale (whole countries), and is largely now derived from remote sensing (which should only improve over time), the estimates on total emissions due to deforestation have a higher degree of confidence associated with them. To this end in the future analyses should focus on a comprehensive land-use change assessment, with refinements of the amount of timber extracted before being cleared or burnt. Ideally such assessment would combine remote sensing data, land tenure information, timber volumes/extraction rates, fire data and locations of mills. Although one would still need to take into account the varying quality of country-specific data and remaining uncertainties regarding drivers within the timber industry, such assessment would represent a significant improvement on current knowledge.

In this study we have not considered the life cycle of C in paper products beyond product substitution factors. Other factors to be considered and which were outside the scope of this work include emissions due to the manufacturing process of paper, service life of paper, recycling levels for different types of paper, and perhaps most importantly, the decomposition dynamics in landfills (with consideration of the recovery of methane to produce electricity). This would ensure a more complete comparison of the GHG balance of paper production in Australia compared to paper production in other key paper producing regions.

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Glossary

Above Ground Biomass (AGB): Biomass is the biological material from living or recently living organisms. Above ground biomass is that portion which is above the soil layer including the stem, stump, branches, bark, seeds and foliage. It can be expressed as green/ fresh or dried biomass and it may include dead material such as dead standing trees and coarse woody debris.

Actual site production (SI): Used in Section 3, all sawlogs, pulp log and pole logs from the study sites were included, and the actual distribution of logs from the sites was used.

Allometric/ Biomass Equation: Allometric relationships/ equations are used to estimate biomass from an easily measured attribute such as DBH.

Basic Density: The density of wood expressed as:
$$\frac{\text{Oven dry weight (kg)}}{\text{Green Volume (m}^3\text{)}}$$

Basal Area: The total cross-sectional area of all stems in a stand measured at breast height, and expressed per unit of land area (typically square metres per hectare).

Below Ground Biomass: Live biological material in the soil layer normally restricted to root material and is generally limited by size (>2mm).

Bioenergy: Bioenergy is renewable energy made available from materials derived from biological sources.

Biomass: Biomass is the biological material from living or recently living organisms.

C Stock: The quantity of C in a “pool” which has the capacity to accumulate or release C.

Commercial Logs: Logs that have a commercial value.

Concessions (Logging, Palm Oil etc): An area that is zoned for a particular activity, i.e. for the establishment of palm oil plantations or for the extraction of timber.

Conservation Forest: An area of forest that is not subjected to anthropogenic disturbance activities (with the exception of fire hazard reduction activities).

Coarse Woody Debris: All debris on the forest floor >2.5 cm diameter (including dead stags), stumps <1.3 m in height and bark.

Crown: For Section 1 of the report it is defined for trees with production logs, as the point where the stem is too small to be of commercial value. Note that this differs from the definition for the biological crown.

Deforestation: The conversion of forest to another land use or the permanent reduction of the tree canopy cover below a certain threshold.

Diameter at Breast Height (DBH): Diameter of the stem (over bark) measured at 1.3m height above the ground.

Forest Conversion: The conversion of virgin forest areas to non-forest or plantations.

Forest Cover: As per the FAO definition “Land with tree crown cover (or equivalent stocking level) of more than 10 percent and area of more than 0.5 hectares (ha). The trees should be able to reach a minimum height of 5 meters (m) at maturity *in situ*.” It includes both open and closed forest types and young natural stands as well as plantations. Please note that variations of this definition are often used.

Forest Degradation: A reduction in the ecological functioning of the forest. Or in our case perhaps “harvest activity that results in the reduction of forest biomass in the long-term”.

Fossil fuel displacement Factor: The emission savings incurred by using renewable energy instead of fossil fuels. The factor value will depend on the relative efficiencies associated with the use of the renewable energy source and the emission footprint of the fossil fuel displaced.

Greenhouse Gases (GHG): Any of the atmospheric gases that contribute to the greenhouse effect by absorbing infrared radiation, including C dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O).

Harvest Prescription: A set of guidelines to be followed when undertaking harvesting activities, including retention of a number of trees for habitat or as a seed source.

Harvested Wood Product (HWP): Includes all wood material that is removed from a harvested area.

High Intensity Harvest: Harvesting using poor harvest techniques and extraction levels greater than those deemed as best practice for the region

HQ (High Quality) Sawlog: The grading specifications vary across regions, but generally it is a log that is large enough to be sawn into boards with minimal defects, yielding a higher proportion of higher quality HWPs than low quality sawlogs”.

Industrial Sawlog (inc. Veneer Logs): As per the FAO definition “roundwood that will be sawn (or chipped) lengthways for the manufacture of sawnwood or railway sleepers (ties) or used for the production of veneer”.

Land Use Change: Generally refers to anthropogenic changes to the natural environment (i.e. the conversion of natural forest to agricultural land).

Long Term C Storage: In this study this is defined as the C that is stored in HWPs (other than paper products) in landfills

Low Intensity Harvest: Harvesting is done with minimal disturbance, with extraction rates and rotation length at a sustainable level.

LQ (Low Quality) Sawlog: The grading specifications vary across regions but generally it is a log that is large enough to be sawn into boards with some defects or imperfections, yielding a higher proportion of lower quality HWPs.

Multi-aged stand: A stand where two or more age or species groups can be distinguished within the stand, although the boundaries may not be clearly defined.

Natural/ Native Forest: A forest composed of indigenous trees and not classified as a forest plantation.

Net Primary Productivity: The rate at which an ecosystem accumulates energy or biomass, less the energy it uses for the process of respiration.

Peatland: Peatland is an area which contains a naturally occurring peat layer at its surface. Peat is a soil with a high organic C content and water holding capacity and a low mineral content..

Peatland Burning: The surface of an intact peatland is usually too wet to burn, however by removing vegetation or through drainage the loss of soil moisture increases its potential to burn. The use of fire to clear peatland vegetation can also cause the organic peat soil to combust. Peatlands are potentially a large source of C emissions due to the thickness of the peat.

Peatland Drainage: Peatland drainage, usually through the construction of channels, lowers the water table exposing the organic soil to the atmosphere resulting in C being emitted to the atmosphere through oxidisation as CO₂.

Pole (Utility): Transmission poles used in the electricity network to support overhead power lines.

Primary Intact Forest: A mature native forest >5ha containing indigenous tree species where there are no clearly visible signs of human activities and the ecological processes are primarily intact.

Primary Degraded Forest: A primary forest that have been fragmented and or subjected to human disturbances that have altered the forest composition and structure.

Product Substitution (Factor): The emission savings incurred by using a HWP instead of alternative products. The factor value will depend on the emission footprint associated with the extraction, transport and manufacture of the HWP and the emission footprint of the alternative product displaced.

Production Forest: A forest that is managed to produce timber for commercial purposes.

Production Logs: Logs that have a commercial value.

Regionally typical production (RG): Used in Section 3, the quantity of timber extracted and the product mix from each production site are adjusted to mirror typical harvest operations in each region.

Residues (forest and mill): Forest residues include the bark , crown and stump and may also include non- commercial species, dead and small trees as well as parts of the stem that had no commercial value due to damage during felling, decay or a reflection of the current market for that region. Mill residues are generated when the logs are cut into rough green sawn boards

(offcuts, sawdust, log hearts), and where applicable when the boards are dried and dressed (offcuts, shavings and sawdust).

Root Shoot Ratio: The ratio of below and above ground biomass in a plant. This value varies according to species and age of the trees.

Salvage Logging: The removal of logs from trees that have been damaged by a natural disturbance such as wildfire, severe wind etc.

Selective Harvesting: Harvesting that removes only a selected portion of the trees in a stand.

Southeast Asia (SEA): As per the FAO definition includes:

Brunei Darussalam
Myanmar
Indonesia
Cambodia
Lao People's
Democratic
Republic
Malaysia
Philippines
Thailand
Viet Nam

Stand Density: A measure of the stocking of a stand of trees based on the number of trees per unit area (ha).

Stand Height/ Stand Height of Dominate Species: The average height of all trees in the stand (site)/ The average height of the dominate species in the stand (site).