Independent Forestry Panel

Public submission

JOHN RAISON		Submission ID: 204714
Organisation:	*N/A*	
Location:	*New South Wales*	
Supporting materials uploaded:	*Attached overleaf*	

Submission date: 10/13/2024 9:21:06 AM

Topic 6. Opportunities to realise carbon and biodiversity benefits and support carbon and biodiversity markets, and mitigate and adapt to climate change risks, including the greenhouse gas emission impacts of different uses of forests and assessment of climate change risks to forests

See attached. Ignore the first attachment which is irrelevant but which I can not remove. The other attachments include my submission , and a key publication referred to in the submission.

Submission to NSW Independent Forestry Panel

Dr John RAISON , Former CSIRO Chief Research Scientist,
[redacted]

13 October 2024

Dear Panel members,

Thank you for the opportunity to provide input to your important work. I am currently travelling overseas on holiday and have limited computing capability; thus, I am using a letter to make my submission to you and not your pro-forma. An attachment (Raison et al., 2024) is also a key part of my submission.

My background is in forest science, forest policy and forest management. I conducted research on the ecology of Australian native forests, including those in SE NSW, over a 40 year period whilst working with CSIRO. I chaired the preparation of the NSW ESFM Report which guided formulation of the RFAs in NSW in the early 2000s and have also reviewed Codes of Forest Practice in the state. I have expertise in forest C accounting and was a co-developer of Australia's National Carbon Accounting System. I am a major contributor to the IPCC GHG accounting guidelines that countries use to report internationally on emissions and C sinks in the land sector.

The main focus of my submission is on effects of harvesting and wildfire on net C emissions from forests, and on impediments to expanding wood production from plantations. I make some brief comments on other issues relevant to assessing the sustainable management of native forests and their mis-use by anti-forestry campaigners.

I have recently conducted a major review of the C balance of managed native forests (Raison, Australian Forestry in press). This is attached, and the summary from the paper included below. It makes important points relevant to your deliberations and dispels several myths being promoted by those opposing the continued harvesting of native forests in NSW.

A critical review of the impacts of sustainable harvesting, non-harvest management and wildfire on net carbon emissions from Australian native forests.

RJ Raison

Abstract

This study provides a critical review of the science underpinning the debate over whether continued harvesting or cessation of harvesting in native forests lead to better overall carbon (C) benefits which are important for climate outcomes.

Assessing the complete C balance associated with harvested native forests requires the application of a full Life Cycle Analysis (LCA) framework that accounts for temporal changes in C stock at the harvested site; C emissions associated with managing and harvesting the forest, transporting and processing harvested wood products (HWP); storage of C in wood products in service and in landfill; any C emissions saved by using residues to generate energy otherwise produced by combustion of fossil fuels; benefits of substituting wood for more C-intensive materials such as steel, aluminium or concrete in construction; and the C footprint of wood products sourced from overseas to replace Australian production. Key considerations in assessing net changes to C stocks within native forests include how to fully account for C flows in harvested forest systems; the concept of C carrying capacity; challenges in estimating changes in soil C and C sequestered in old, unharvested forests; and assessing the effects of wildfire on C stocks and dynamics.

Only one Australian study has adopted a complete LCA approach. That study concluded that harvesting of sustainably managed native forests and the subsequent use of forest biomass to produce HWP or energy can make a positive contribution to mitigating national net C emissions. Other studies claiming that native forest harvesting increases net C emissions have either been incomplete, used inappropriate parameters to estimate components of the total C balance, or overestimated the rate of C gain in older forests and the ability of unharvested forests to store C for the long-term, and consequently have underestimated the C benefits due to wood harvest and use. This has led to the erroneous conclusion that cessation of harvesting would provide better C outcomes than sustainable management for wood production.

A new case estimated the C balance of Victorian 1939 regrowth mountain ash forest managed for sawlog and pulpwood production on a rotation of 75 years. Annual harvesting of 1,000 ha of forest compared with no-harvest (conservation) management results in sustained net C emissions savings equivalent to removing 303,000 cars from the road. These findings contrast with Victorian Government statements that harvesting results in significantly increased C emissions. Closing Victoria's native forest timber industry will therefore have negative outcomes in terms of C emissions and climate.

The management of C in native forests needs to be integrated at the landscape scale with management for other forest values and attributes. Changes to C storage in Australian native forests are driven much more by extensive wildfire than by harvesting. Harvesting affects only a small proportion of the forested landscape, and logs harvested annually from Australian native forests store only about 2.5 Mt CO_2-equivalent, or about 0.6 % of Australia's total net anthropogenic greenhouse gas (GHG) emissions. Additional emissions of C from the decomposition or combustion of slash produced during harvest are about a third of this figure. These C removals from the forest are offset by sequestration of C in new regrowth and are supplemented by benefits derived from the use of harvested wood. In contrast, in very bad fire seasons such as the summer of 2019-20, C emissions were about twice Australia's total annual anthropogenic (i.e. excluding wildfire emissions) GHG emissions and about 200 times greater than C removals in wood plus emissions from logging slash.

When examined at the landscape scale there is no evidence that harvesting leads to increased area burnt, fire severity or C emissions caused by wildfires. However, wildfires in the large and contiguous areas of thick regrowth created after the 'black summer' fire season will pose a major threat to C stocks in all forests during the coming decades.

Timber harvesting, providing it is well conducted in carefully selected parts of the landscape, can provide on-going C benefits. This study supports the global consensus that harvesting of sustainably managed native forests and the subsequent use of biomass to produce HWP or energy can make a positive contribution to mitigating national net C emissions.

1. Carbon emissions from sustainably harvested native forests in NSW.

Some are proposing that cessation of harvesting in NSW native state forests will lead to major reductions in C emissions and that C credits will be created that can be monetised. This is clearly untrue. Only 9.1% of the forest estate in NSW is State Forest and only a part of this is harvested using mainly selective harvesting practices. Privately owned native forests occupy a much larger area (33.8%) some of which is actively managed for wood production. National parks represent 25.1% of the forest estate.

Publicly available data (ABARES, 2024) shows that in 2022-23, about 650,000 cubic metres of log was removed from the State's native forests (public + private). Annual log production from native forests has declined markedly during the last 20 years. We know from detailed research that each cubic metre of harvested wood contains approximately 1 tonne of carbon dioxide equivalent (CO2-e). We also know from extensive field measurements across many forest harvesting operations, that approximately 70% of felled biomass is removed off-site in harvested logs (which is considerably more than the 40% claimed by anti-logging advocates); while the remaining 30% is waste that is left in the forest to be either burnt or to decay over time.

Using the information in the previous paragraph, the total carbon in the felled trees in public forests that produced 650,000 m^3 of logs annually is approximately 0.93 Mt CO2e or about 0.8 % of the annual anthropogenic GHG emissions in NSW. This figure would be an upper limit for C emissions caused by harvesting of native forests in NSW if we assume that all that C was immediately emitted to the atmosphere (which it is not, as discussed below).

Part of the carbon in harvested wood enters long-term storage either whilst in service or in landfill. Wood is also used in construction to substitute for high emission alternative materials (steel, concrete and aluminium) and for creating wood products that might otherwise be sourced from overseas where production emissions are much higher than in Australia. When these factors are properly assessed in a lifecycle analysis, sustainable harvesting does not cause net C emissions but instead leads to greater mitigation of C emissions than an alternative no-harvesting forest management option (Raison, in press).

2. **Role of plantations in wood production**

Plantations (predominantly softwood) currently produce about 85% of the wood harvested annually in NSW, but production has been static for many years resulting in a widening trade deficit in wood products similar to Australia overall. There are many impediments to increasing plantation area and reliable wood production, making it imperative that production from native state forests continues and that support and incentives are provided to unlock the considerable potential to markedly increase wood production from privately owned native forests.

These issues are discussed in a guest editorial to Australian Forestry (Raison and Nambiar, in press) which is copied below.

Australia should source wood from both plantations and native forests.

John Raison and Sadanandan Nambiar

Australia is facing a serious wood shortage that is getting worse. This is because of rising demand from a growing population, stagnating supply from plantations and rapidly declining harvests from native forests because of marked reductions in the area where harvest is permitted.

In 2021-22, imports of wood products cost $6.7 billion, and it is growing 19% year-on-year in real terms, and across all product categories including sawn timber and panels (ABARES, 2024). In contrast, exports of mostly plantation hardwood chips generated only $3 billion. Given national plans to build 1.2 million new houses during the next 5 years and domestic supply constraints, import volumes and costs will continue to increase, so also the trade gap. This doesn't make sense when we could use our own native forest wood and have the resources and capability to do so sustainably.

We currently have 1.72 million hectares of plantations. Of this, about 1 million ha are softwood (mostly pines) managed on cycles of around 25-30 years, primarily for sawn timber with pulpwood as its major by-product. About 0.7 million hectares are hardwoods (mostly eucalypts), managed on cycles of 10-15 years for pulpwood. Almost all softwood is used domestically, and almost all plantation hardwood is exported.

Plantations of softwood and hardwood together account for about 87% of Australia's total wood harvest, including 80% of hardwood pulpwood. The remaining 13% is hardwood sourced from native forests, that contributes 72% of hardwood sawlogs and 20% of hardwood pulpwood (2021-22 figures).

The advocates of a total ban on harvesting of native forests, such as academic David Lindenmayer and anti-forestry activists, suggest that it would be easy to shift all wood production to plantations. This view is simplistic, when we learn from the history of Australian plantation development, the types of wood products produced from plantations, the current stand growth rates, diversity of products we need, and the operation of wood product markets and businesses. For now, we simply don't have sufficient plantations to replace the volumes (or quality) of wood currently obtained from native forests; for the future, expansion of the plantation estate requires overcoming major impediments and has a lag time of many decades.

For a range of bio-physical and economic reasons, only a few eucalypt species are currently grown in plantations and with a few exceptions their wood is generally best suited to pulp rather than sawn wood products, unlike logs from regrowth native forests. Most of the existing estate is not suitable for conversion to a sawn timber production regime. Thus, there is little scope to substitute wood from native forest with that from existing plantations. Hence products are being imported from Asia-Pacific neighbours, North America, South Africa and Chile.

Why is expanding Australia's plantation estate so hard? Major reasons are economic and availability of suitable land. Historically, softwood plantations were established by state governments supported by grants from the Commonwealth on land cleared of native forest specifically for that purpose. Most of these have been sold to the private sector, except in NSW and a very small area in the ACT. State governments no longer invest directly in plantations, and that is unlikely to change. Conversion of native forest to plantations is also no longer possible. Australia's softwood plantation area has been nearly static for the last three decades.

Hardwood plantations expanded during the period 1990 - 2010, mostly on farmland via managed investment schemes (MIS), backed by significant tax incentives (now gone and unlikely to return). Many of these turned out to be unsustainable, financially and otherwise, and often met opposition from sections of rural communities concerned about the loss of land from agriculture. Hardwood plantation area has slowly declined, as some estates reverted to agriculture. Whilst small-scale farm forestry plantings can contribute to wood supply, there has been little adoption of it at scale despite 40 years of promotion.

Future plantation expansion needs to be financially viable, and this is very challenging given the high costs of suitable land and tree establishment, availability of land and the decades-long lead time before an economic return. Land with annual rainfall exceeding 650-700 mm/yr is expensive but required to support economic growth rates for sawlog production. Planting more marginal drier sites results in slower growth rates incurring far greater risk for longer due to the potential for droughts, bushfires, pests and diseases. A 70% reduction in the amount of wood harvested from our native forests over the last 20 years has not stimulated the growth of the plantation resource, as some economists and others argued.

Reliance solely on plantations is risky as they are highly vulnerable to wildfire; for example, 92,000 ha were destroyed in the 2019/20 in NSW (Davey and Sarre, 2020), causing major impacts on industries and markets.

Whilst some of the above impediments may be overcome to varying degrees in the future, and there may be opportunities for lifting the productivity of parts of the existing plantation estate, there is little prospect of Australian plantations replacing much of the wood from native forests during the next 20-30 years. Therefore, if use of similar hardwood products is to continue, they will need to be imported at considerable cost. There are also significant risks of high carbon emissions and other adverse environmental impact associated with forestry *practices associated with some of our sources of hardwood from the Asia-Pacific region.*

Every recent global strategy for climate change mitigation proposes increased use of wood, enabling substitution of materials with high embedded carbon emissions such as concrete, steel and aluminium in construction. Long lived wood products are also carbon stores, renewable and recyclable (e.g. Ximenes, 2023). The adoption of sustainably managed forests as a part of the climate change mitigation strategy is built upon solid science, national and international, and that is encapsulated in the policies and recommendations of the IPCC, FAO and others. Yet, we are already seeing a drift from wood to steel frames in construction - a lost opportunity for mitigation.

The best way forward for securing Australia's future wood supply and associated environmental and social benefits is a complementary mix of well managed plantations and sustainable harvesting of a small proportion of native forests in carefully chosen parts of forested landscapes. The ceasing of all harvests in public native forests in Victoria and WA is a lost opportunity for now. However, it remains the best option for NSW, Tasmania and Queensland, and eventually for the country. We have the scientific basis, management knowhow, operational skills, local processing mills and regulatory- monitoring mechanisms for achieving this goal.

ABARES 2024. Australian forest and wood products statistics - DAFF (agriculture.gov.au) [accessed 2024 Aug 16]

Davey S M, Sarre A. 2020. Editorial: the 2019/20 Black Summer bushfires, Australian Forestry, 83:2, 47-51, DOI: 10.1080/00049158.2020.1769899

Ximenes, F. (2023). Forests, plantations, wood products and Australia's carbon balance. Forests and Wood Products Australia, Melbourne. pp. 32. [accessed 2023 Jun 30]. https://fwpa.com.au/wp-content/uploads/2023/08/Forests-Plantations-Wood-Products-and-Australias-Carbon-Balance-08-2023.pdf

3. Other issues

Much research has concluded that regulated and sustainable harvesting is not a major threat to forest biodiversity and that there are several much more important threatening processes such as invasive species, repeated wildfire and land clearing. Other submissions will have provided a detailed treatment of this issue.

Those opposing harvesting of native forests often repeatedly refer to the harvesting of "old" forests with high conservation value – but for decades almost all harvested forests have been regrowth stands regenerated after prior harvesting or wildfire. Claims are also made that harvesting destroys threatened species, but the major efforts to effectively protect them (eg. for koalas in NE NSW – refer to detailed studies by Brad Law and colleagues from NSW DPI) are ignored. Recently a paper [Ward et al. (2024). Shifting baselines clarify the impact of contemporary logging on forest-dependent threatened species. Conservation Science and Practice Volume 6, Issue 9] used an unsubstantiated assumption that any overlap between distribution of threaten species and areas harvested would lead to forest degradation. The study is badly flawed for the following reasons:

(A) The forest degradation map used was produced by combining spatial environmental data with remotely sensed measures and by referencing to sites having intact vegetation. The authors did not derive this coverage – it was produced collaboratively by CSIRO and the Federal Environment Dept. The map claims to represent the capacity of a site to support local wildlife, but it is not clear if or how this has been demonstrated for particular species. It is assumed that a reduction in habitat score negatively impacts species – there are clearly many uncertainties associated with inference drawn when using the degradation map. The degradation map does not capture other threats such as those from feral animals or effects of extensive intense wildfires. These are clearly very important in affecting forest condition across all tenures. There is not a word on the degradation status of conservation forests – instead a complete focus on trying to demonstrate the negative impacts of logging.

(B) There is no map showing degradation status of logged forests. If there is an overlap between the modelled distribution of threatened species and areas logged it is assumed that there are negative impacts on habitat and species. No account is taken of the fact that forests regrow after harvesting and thus habit is dynamic over time. No account is taken of modification of harvesting practices to mitigate impacts on threatened species.

(C) The paper contains several false claims such as that RFA's ignore requirements of the EPBC Act, logging increases the extent and severity of wildfires and increases GHG emissions, and that wood production from native forests is unprofitable!

Overall, the paper applies poor logic and is misleading in the way it casts negative dispersions on the management of native forest for wood production in NSW.

Economic analysis for harvested native forests is often flawed because it excludes value-adding components as well as other important community benefits such as fire protection, roading and water production. Those claiming that wood production in native forests is uneconomic often ignore these other community benefits. Unharvested forests also need to be managed and will incur a cost to the public - the costs/ha are higher than for state forests used for wood production in NSW.

I hope that the Panel find my submission useful. I am happy to provide additional information if requested, and would welcome an opportunity to discuss the points raised with the Panel.

Yours sincerely.

Dr. John Raison, Googong NSW

Impacts of sustainable harvesting, non-harvest management and wildfire on net carbon emissions from Australian native forests: A critical review of the underpinning science

RJ Raison[a]

Contact: R. John Raison; [redacted]

[a]Former Chief Research Scientist, CSIRO

Abstract

This study provides a critical review of the science underpinning the debate over whether continued harvesting or cessation of harvesting in native forests lead to better overall carbon (C) benefits and thus climate outcomes.

Assessing the complete C balance associated with harvested native forests requires the application of a full Life Cycle Analysis (LCA) framework that accounts for temporal changes in C stock at the harvested site; C emissions associated with managing and harvesting the forest, transporting and processing harvested wood products (HWP); storage of C in wood products in service and in landfill; any C emissions saved by using residues to generate energy otherwise produced by combustion of fossil fuels; benefits of substituting wood for more C-intensive materials such as steel, aluminium or concrete in construction; and the C footprint of wood products sourced from overseas to replace Australian production. Key considerations in assessing net changes to C stocks within native forests include how to fully account for C flows in harvested forest systems; the concept of C carrying capacity; challenges in estimating changes in soil C and C sequestered in old, unharvested forests; and assessing the effects of wildfire on C dynamics and C stocks.

Only one Australian study (Ximenes et al., 2016) has adopted a complete LCA approach. That study concluded that harvesting of sustainably managed native forests and the subsequent use of forest biomass to produce HWP or energy can make a positive contribution to mitigating national net C emissions. Other studies claiming that native forest harvesting increases net C emissions have either been incomplete, used inappropriate parameters to estimate components of the total C balance, or overestimated the rate of C gain in older forests and the ability of unharvested forests to store C for the long-term, and consequently have underestimated the C benefits due to wood harvest and use. This has led to the erroneous conclusion that cessation of harvesting would provide better C outcomes than sustainable management for wood production.

Key parameters from Ximenes et al. (2016) were applied in a new case study in Victorian 1939 regrowth mountain ash forest managed for sawlog and pulpwood production on a rotation of 75 years. Annual harvesting of 1,000 ha of forest compared with managing the forest for conservation results in sustained net C emissions savings equivalent to removing 303,000 cars from the road. This contrasts with Victorian Government statements that harvesting results in significantly increased C emissions. Closing Victoria's native forest timber industry will therefore have negative outcomes in terms of C emissions and climate.

The management of C in native forests needs to be integrated at the landscape scale with management for other forest values and attributes. Changes to C storage in Australian native forests are driven much more by extensive wildfire than by harvesting. Harvesting affects only a small proportion of the forested landscape, and logs harvested annually from Australian native forests store only about 4 Mt CO_2-equivalent, or about 1% of Australia's total net anthropogenic greenhouse gas (GHG) emissions. Additional emissions of C from the decomposition or combustion of slash produced during harvest are about a third of this figure. These C removals from the forest are offset by sequestration of C in new regrowth and are supplemented by benefits derived from the use of harvested wood. In contrast, in the bad fire season of 2019-20 C emissions from bushfire were about 150 times greater than emissions due to native forest harvesting. When examined at the landscape scale there is no evidence that harvesting leads to increased area burnt, fire severity or C emissions caused by wildfires. However, wildfires in the large and contiguous areas of thick regrowth created after the 'black summer' fire season will pose a major threat to C stocks in all forests during the coming decades.

Timber harvesting, providing it is well conducted in carefully selected parts of the landscape, can provide useful on-going C benefits. This study supports the global consensus that harvesting of sustainably managed native forests and the subsequent use of biomass to produce HWP or energy can make a positive contribution to mitigating national net C emissions.

Keywords

Australian native forests, carbon, greenhouse gas emissions, sustainable forest management, climate change, fire, harvesting, conservation, perverse forest policy

Introduction

The role that forests play in mitigating climate change is contentious in Australia, partly because of the complexity and logistical challenges of accounting for carbon (C) in native forest ecosystems that are highly spatially and temporally variable, and the need to account for the fate of C in harvested wood. Internationally, it is broadly recognised that forests and forest products can play a major role in the transition to low-carbon economies (e.g., FAO 2016, 2021). The foreword to FAO (2016) states ‘Forests and forest products have a key role to play in mitigation and adaptation, not only because of their double role as sink and source of emissions, but also through the potential for wider use of wood products to displace more fossil fuel intense products. Indeed, a virtuous cycle can be enacted in which forests increase removals of carbon from the atmosphere while sustainable forest management and forest products contribute to enhanced livelihoods and a lower carbon footprint’.

Likewise, the Intergovernmental Panel on Climate Change (IPCC) assessment reports (e.g., IPCC 2022, Olsson et al. 2019) and other major syntheses (IPCC 2018, 2019) have consistently supported sustainable forest management as an important tool for providing significant climate change mitigation.

Forests are an important component of Australia’s national greenhouse gas (GHG) inventory (DISER, 2021) and there are on-going efforts to improve the coverage and accuracy of the accounting systems. C accounting is complex in diverse Australian native forests, that have usually been subject to differing patterns of natural disturbance (fire, storm damage, disease) as well as historical harvesting regimes for a range of wood products (Raison and Squire, 2010). Different groups have made competing claims about the C balance of managed Australian native forests based on use of different assumptions, accounting methods and data. Perverse outcomes can result from basing government policy decisions on analyses based on use of either an inappropriate accounting framework or inappropriate data.

This review analyses the science relevant to assessing the implications for net C emissions of sustainable harvesting, no-harvest management and wildfire in Australian native forests. An appropriate framework for comparing harvest and no-harvest management options is identified and discussed. The focus is on estimating the complete C balance as this is what is relevant to changes in atmospheric carbon dioxide and hence future climate. Key papers are assessed as to how accurately they have conducted their C accounting and how well they have applied this accounting framework, and whether conclusions are soundly based.

This review is structured as follows:

- Description of Australia’s native forests from a C perspective.
- Discussion of the need to apply Life Cycle Analysis (LCA) to harvested forests and HWPs.
- Review of key considerations in estimating temporal changes in C stocks within native forests, including soil and biomass C, and the effects of fire.
- Commentary on four LCA studies comparing the effects of non-harvest and harvest management on the C balance of Australian native forests.

- Presentation of a case study examining the effect on net C emissions of ceasing harvest in Victorian regrowth mountain ash forest.
- Summary of the critical factors for achieving C mitigation benefits.
- Discussion of the international position regarding the role of harvested forests in helping to mitigate climate change.
- Discussion of managing native forests for maintenance of C stocks in the context of broader forest management goals and wildfire.

Australia's native forests from a C perspective

Australia's native forests in 2016 were estimated to cover 132 million hectares or 17% of the land area, with 77% of native forests dominated by eucalypts (ABARES 2023a). In 2016, 28.1 million hectares of native forest were estimated to be available and suitable for wood production, comprising 13.6 million hectares of private forest, 8.2 million hectares of leasehold forest and 6.3 million hectares of multiple-use public forest (MIG and NFISC, 2018). Of this multiple-use public native forest, the net harvestable area after allowing for additional local restrictions was 5.0 million hectares, and the total area of public native forest from which wood was harvested was 73,000 hectares in 2015-16, or 1.5% of the net harvestable area. Average annual total wood removals were 4.1 million cubic metres during the period 2011-12 to 2015-16, which was below sustainable levels (MIG and NFISC, 2018).

Biomass contains about 50% C by mass, so a cubic metre of wood with a density of 500 kg/cubic metre contains about 0.25 tonnes (t) C and the equivalent of about 1 t of sequestered CO_2 (converting C to CO_2 by multiplying by 44/12). Hence the annual wood removal from native forests in 2016 contained about 4.1 Mt CO_2-equivalent, or about 1% of Australia's total anthropogenic net GHG emissions that year (DISER, 2021), plus additional (about 1/3 of that removed in harvested wood) emissions from decomposition or burning of harvesting residues. This quantity of CO_2-equivalent contrasts with emissions of CO_2 in wildfires of about 150 t/ha (Volkova et al., 2022), or approximately 900 Mt CO_2-equivalent from the estimated 6 million ha of native eucalypt forest burned during the 2019/20 wildfires (Davey and Sarre, 2020). These fire emissions were thus about double Australia's total annual anthropogenic net GHG emissions.

The C stock in Australia's native forests in 2016 was estimated at 22,000 Mt, 85% of which was in non-production native forest and 14% in production native forests (MIG and NFISC, 2018). In contrast, the estimated C stock in softwood and hardwood plantations in 2016 was only 258 Mt. The C stocks in Australia's native forests are therefore approximately 160 times Australia's current net annual anthropogenic GHG emissions. Should future climate change and associated increase in wildfires lead to loss of as little as 5 percent of the C stored in Australian native forests, that would be equivalent to about 8 times current total annual anthropogenic GHG emissions.

A Life Cycle Analysis approach to the effects of harvesting and the use of wood products on net C balance

When assessing the impacts of forest harvesting on net C balance, or net emissions to the atmosphere, it is necessary to apply a full Life Cycle Analysis (LCA) framework that takes account of:

- Changes in C stocks over time at the harvested site due to wood removal, and decay or combustion of residues.
- The GHG emissions associated with managing and harvesting the forest, and transportation and processing, to generate various wood products.
- The storage of C in wood products in service and in landfill, and emissions during decay.
- Any GHG emissions saved by using wood to generate energy otherwise produced by combustion of fossil fuels.
- The benefits of substituting wood for more C-intensive materials such as steel, aluminium, or concrete in construction.
- The C footprint of wood products sourced from overseas if a decision is made not to produce them in Australia.

Partial C accounting, for example that does not properly quantify substitution benefits, can be misleading when the objective is to assess net exchange of C with the atmosphere resulting from a harvesting or no-harvesting decision (Muller et al. 2020).

At the site or coupe level, harvesting reduces C stocks in above-ground biomass (AGB) which then rebuild over time as forests regrow. However, even with regrowth the time-averaged C stock in forests managed for harvest is reduced (Fig. 1). The time-averaged carbon stock is defined as the integral over time of the carbon stock in each phase of a forest management cycle, divided by the duration of the management cycle. The past harvesting of mature eucalypt forests resulting in their conversion to regrowth stands managed on harvest rotations of 80 to 100 years would have lowered average C stocks in AGB. However, today most harvested forests are regrowth stands, either even-aged or more often multi-aged where selection logging methods were used (MIG and NFISC, 2018). Suitable time frames must be used to assess temporal change in C stocks - for forests managed for wood production this is a full forest rotation or longer. Any benefits or disbenefits due to forest harvesting will thus be accumulated over successive rotations. In a sustainably managed native forestry system, although C stocks are highly dynamic on individual harvested areas, C stocks remain stable across the forest estate over time following cycles of harvest and regrowth (Schulze et al., 2022).

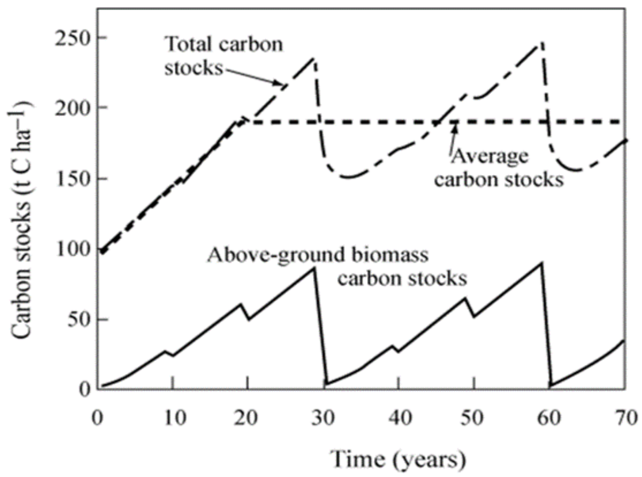

Fig. 1. Schematic representation of a simple (plantation) forest system demonstrating the concept of time-averaged C stocks (GOFI, 2014). In this example a new forest is established on cleared land where initially there is no C in AGB, and 100 tC/ha in the soil. The forest grows and accumulates 100 tC/ha in AGB over 30 years. The forest is then harvested, and all biomass is removed from the site. A new forest is then established, and the cycle of growth and harvest continues. The time-averaged forest C stock (soil + AGB) is shown by the dashed horizontal line in the upper panel.

The growing, harvest and transport of wood and manufacture of HWPs requires much less fossil energy compared to the production of steel, bricks, aluminium, and concrete (e.g. Schultze et al., 2022; Ximenes, 2023), and C is also sequestered in the regrowing forest. Thus, there are opportunities to substitute HWPs for these other materials and obtain GHG benefit. Quantifying the wood product substitution impact in the overall GHG balance of native forestry is similar in principle to estimation of the fossil fuel displacement benefit of using sustainably harvested biomass for bioenergy applications. The C initially sequestered in the bioenergy option is re-emitted on combustion, however its impact is permanent when its use prevents use of alternative options with a higher GHG emission footprint (i.e., fossil fuels). Wood product substitution similarly produces permanent offsets, reducing fossil fuel emissions whilst also retaining C in the HWP in use (Werner *et al.* 2010, Lippke *et al.* 2011). Use of biomass for new bio-products such as green chemicals, solvents and bioplastics can also result in significant C benefits – these are yet to be fully quantified.

The total storage of C in harvested wood products (HWPs) gradually increases over time, especially for long-lived products such as electricity poles and floorboards. If disposed to landfill, those products will continue to store much of their embodied C for a significant period. Fig. 2 provides a summary of the emissions footprint for Australian hardwood HWP.

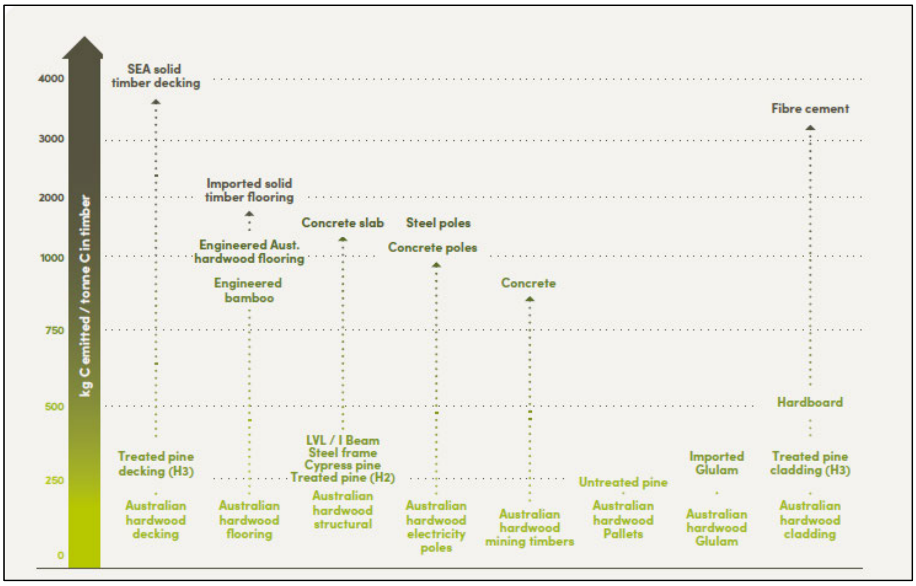

Fig. 2. The emission footprint for Australian hardwood HWPs and the products they could replace (from Ximenes et al., 2016). Southeast Asia (SEA).

A further consideration relates to the need to account for the GHG emissions resulting from production of wood products in another location (known as leakage) if local production is reduced or ceases. This is another form of product substitution: if writing and printing papers are not produced from hardwood grown in Australia, they will likely be produced in SE Asia with potentially greater GHG emissions (see Ximenes et al., 2016). Venn (2023) provides a detailed review of the likely adverse effects on C emissions (and biodiversity) caused by the reduction in harvesting of Australia's native forests and the corresponding increase of hardwood imports from developing countries.

A full LCA approach must be adopted to capture changes in C in harvested forest and all the changes resulting from the production and use of HWPs (Fig. 3).

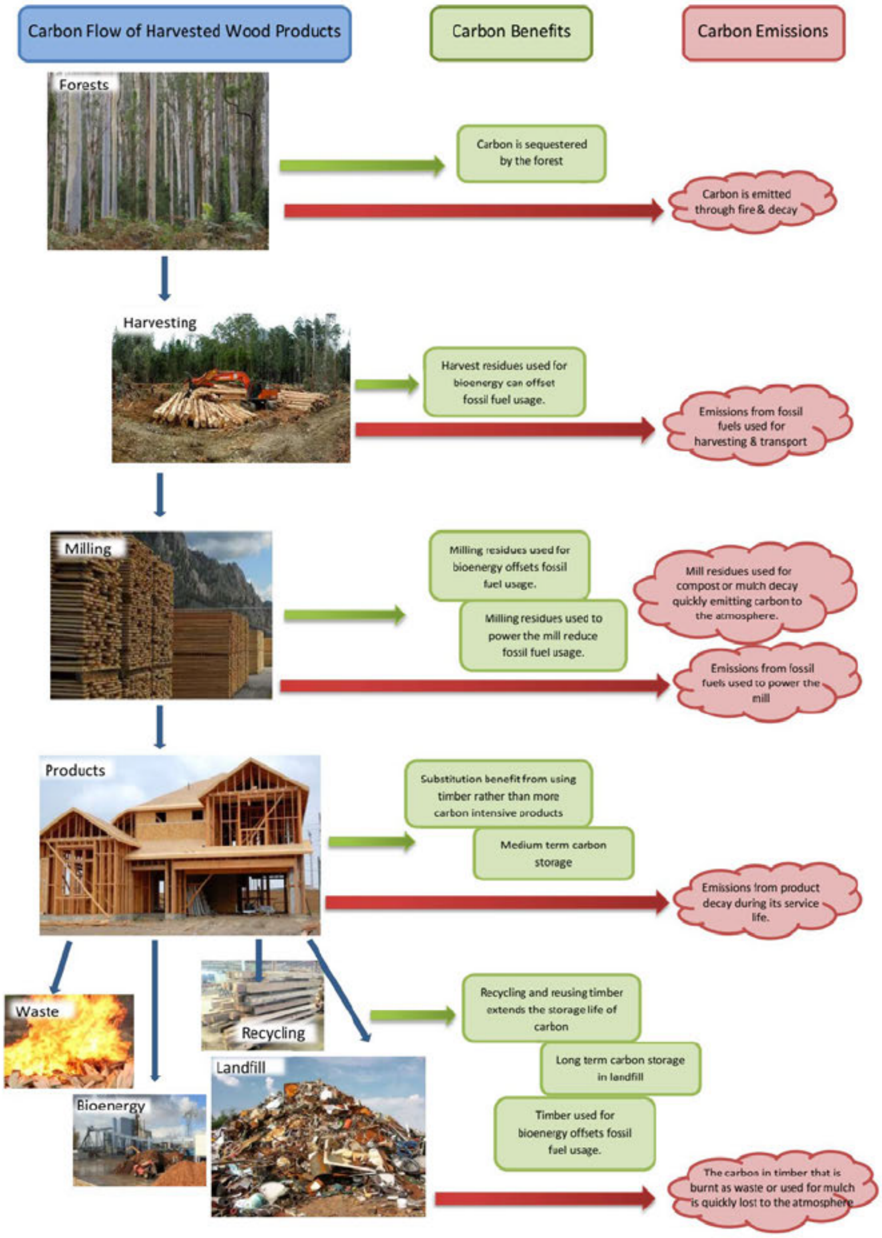

Fig. 3. Typical C flows in harvested Australian native forest systems and resulting from HWP production and use (from Ximenes et al., 2016)

A critical starting point is the amount of biomass felled during the harvesting operation, and how much is removed from the forest as sawlogs or pulpwood. The difference is harvesting residues (slash) left in the forest, which decompose or are burnt, releasing much of the contained C to the atmosphere. Raison and Squire (2010) synthesised available historical data and experience and provided an initial set of approximate default values for the proportions of felled biomass remaining as logging slash or removed from the site for forest types and harvesting systems used in Australian native forests – these have been converted to the proportion of felled biomass removed (Table 1). These values were incorrectly used in the LCA studies of Keith et al. (2014a, 2015), reviewed in a later section.

	Silvicultural system	**Harvest of sawlogs only**	**Harvest of sawlogs and pulpwood**
		Proportion of felled biomass removed (%	
Moist forest types	Clearfall harvest	35	60
	Selection harvest	55	80
Drier forest types	Clearfall harvest	50	70
	Selection harvest	45	75

Table 1. Proportion of felled biomass removed from harvest sites for different eucalypt forest types, silvicultural systems, and utilization regimes (from Raison and Squire 2010).

Removal of felled biomass can be as low as 35% with clear felling in wet forests with no pulp wood market but can be 60% if pulpwood is also harvested. With clear felling of drier forests, about 70% of felled biomass is removed with a pulpwood market. Ximenes et al. (2016) updated estimates for a range of forest types in NSW and Victoria using direct weighing methods. Harvesting a 1939 regrowth mountain ash forest (*Eucalyptus regnans*) stand in Victoria produced a 79:21 ratio of harvested logs to slash. Measurements were checked against regional harvest removal data: for mountain ash, the average for the Toolangi region was a 75:25 ratio of harvested logs to slash.

In addition, May et al. (2012) and England et al. (2013) established that the total GHG emissions across all forest operations required to produce harvested logs or wood chips in native forests is about 3-7% of the amount of CO_2 sequestered in the logs.

The science underpinning the tracking of C in HWP has been evolving rapidly and was recently reviewed and updated by Ximenes (2021b, 2023). When conducting LCA for wood products produced from native forests it is critical to use data relevant to Australian conditions relating to the types of products produced, their service life, disposal options, decay rates in landfill, and benefits of substitution for C-intensive materials. Conduct of a reliable LCA is also critically dependent on sound approaches to estimating spatial and temporal change in C in soil and AGB. These issues are addressed in the following sections.

Key considerations in analysing changes in C stocks within native forests.

(1) General considerations

Estimating changes in C in managed forests is complex (e.g. Puls et al., 2024), and a wide variety of methods and reporting approaches have been adopted globally depending on management and policy objectives (e.g. Peng et al., 2023).

Total forest C stocks are the sum of C in AGB, below ground biomass (roots), deadwood, litter, and soil. Soil C stocks are very significant but are highly variable spatially and thus hard to quantify in terms of temporal change. Disturbances such as harvesting or wildfire might be expected to result in a small short-term reduction in soil C followed by a slow recovery, but understanding of soil carbon dynamics in Australia's native forests is poor (Grant et al., 1995; Raison et al., 2009; Page et al, 2012). Changes in C in AGB are easier to estimate and can be large and highly dynamic over time following forest disturbances.

Given the very high variability in C stocks in native forests, great care is required when extrapolating findings from a limited number of study sites to a forested landscape. Comprehensive studies based on a very large number of inventory sites across the landscape (Moroni et al., 2010; Volkova et al, 2018) show that C stocks in AGB can vary by several-fold for the same forest type and age. Thus, forest C balances are best analysed at the forest estate or landscape scale, which encompass a mosaic of various forest types of differing ages or time since last harvest. Forest growth rates differ depending on site biophysical factors (such as soil fertility and climate) and prior disturbance history. The changes in C stocks in AGB for a forest estate are the summation of changes in each part of the mosaic. In a forest estate managed for sustainable yield of wood the total C stocks in AGB are relatively stable over time (Schulze et al., 2022).

Appropriate inventory methods must be used when estimating C in AGB. In old forests with a low density of very large trees, large sample plots (at least 0.25 ha) and sampling of a significant area of the forest is needed. The study by Keith et al. (2009) was inadequate in this regard. The authors sampled old-growth mountain ash forest in Victoria that had been largely undisturbed for at least 250 years and contained large trees up to 4.86 m in diameter and up to 84 m in height. Tree inventory was based on 318 small (10 x 10 m) plots nested within 53 sites (each of 3 ha) within the O'Shannassy catchment of 13,000 ha. Sample plots were 0.01 ha in size, and in total only 3.2 ha of forest was sampled (0.025% of the total area). In addition, the use of small plots is likely to have resulted in serious errors in the C estimates in forest with a low density of large trees that contain most of the biomass C, due to large edge effects (whether to include or reject a tree on the plot boundary), and the distribution of AGB is unlikely to be normal. A minimum sample plot size of 0.25 ha (25 times larger than used by Keith et al. 2009) was recommended for forests containing very large trees to meet the criterion of normality (Chave et al., 2004). Other estimates of AGB in similar Australian forests have used plots of at least 0.25 ha (Moroni et al., 2010; Sillett et al., 2015; Ximenes et al., 2016; Volkova et al., 2018).

Several authors have proposed the concept of Carbon Carrying Capacity (CCC), defined as the maximum C stock possible under prevailing regimes of natural disturbance (primarily wildfire under Australian conditions). This was first proposed at the site level (Mackey et al. 2008) but applies more sensibly at the landscape level (Roxburgh, 2009). The difference between the CCC and current C stocks was then defined as the C sequestration potential (CSP). The concepts are illustrated in Fig. 4.

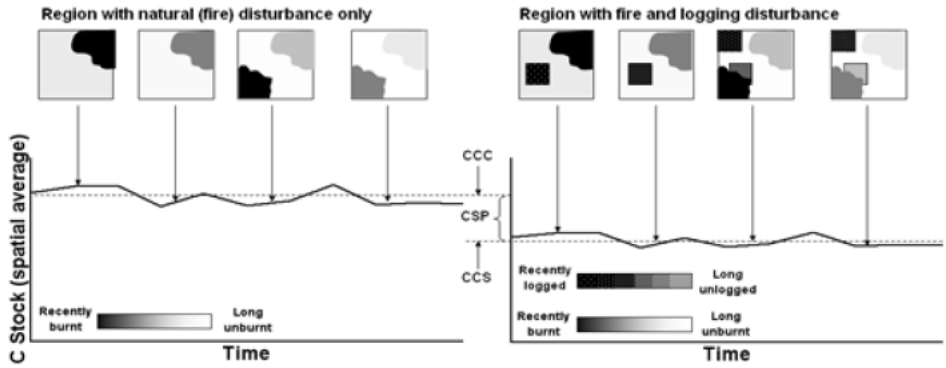

Fig. 4. An illustration of Carbon Carrying Capacity concepts (from Roxburgh, 2009). A hypothetical example is shown with two forested regions both subject to natural disturbance, but with the region on the right also undergoing timber harvesting. The boxes represent the regions and show the spatial and temporal variation in natural disturbance and harvesting. The dotted lines represent the average over space and time of the forest C stocks in each region (Current Carbon Stock in the region with logging). The difference between these lines is the Carbon Sequestration Potential (CSP).

Proponents of a non-harvest approach argue that cessation of harvesting would enable some of the C sequestration potential to be realised. However, these are theoretical concepts with limited practical application:

(a) The CCC is determined by a complex interaction between site growing conditions (including during the critical initial phase of regeneration after wildfire) and the spatially varying impact of previous wildfires. Attempts to use estimates of AGB in long-undisturbed forests (which as discussed later are also hard to estimate reliably) as a measure of CCC, and then to predict CCC spatially based on environmental variables and disturbance history, have not proven successful, explaining a limited (25-65%) proportion of site variance (Keith et al. 2010, 2014a). CCC is essentially unknown at fine scales and only poorly known at landscape scales.

(b) Given that the CCC cannot be reliably estimated, neither can CSP at site or landscape scales.

(c) The maximum CCC is not an endpoint in some forests. Transition from wet forest to rainforest in the absence of fire can be associated with a significant decline in forest C stocks (Moroni et al., 2011; McIntosh and Moroni, 2016).

(d) The CCC of forests is declining because of the increasing frequency of wildfires (Canadell et al., 2021).

(2) Estimating temporal change in soil C stocks

The pool of soil C is large and change to it caused by harvesting or fire is important to forest C balance. However, the impacts of disturbance are complex, variable, ecosystem-specific and difficult to generalize. Moreover, many studies are flawed because of methodological problems, such as those contributing to the reviews of Johnson and Curtis (2001) and Gonzalez-Perez et al. (2004).

Fire and harvesting significantly affect the amount and pattern of inputs of C to the soil from above-ground litter and from root systems (both live and fire-killed roots), as well as rates of loss of soil C caused by changed soil respiration. It is the difference between C inputs and outputs that determines overall change in soil C stocks.

Most studies do not account for both soil and root C when comparing 'soil' C stocks before and after disturbance. In soils sampled in undisturbed forest, most live roots are removed during sieving prior to analysis and thus most root C is not measured as part of the soil C pool. Where fire or harvesting kills vegetation, some dead and decaying roots can pass through the sieve and are thus measured as part of the soil C pool. This renders pre- and post-fire comparisons of soil C invalid because different C pools are being measured and compared before and at different times after fire. The input of C from dead roots is likely to be significant if disturbance kills most of the vegetation, because temperate eucalypt forests have a high root biomass (Mokany et al., 2006). There is input of at least 40 t C/ha from killed roots to the soil on strongly heated 'ashbed' soils during the initial years after clear-fell harvesting and slash burning (Fig. 5).

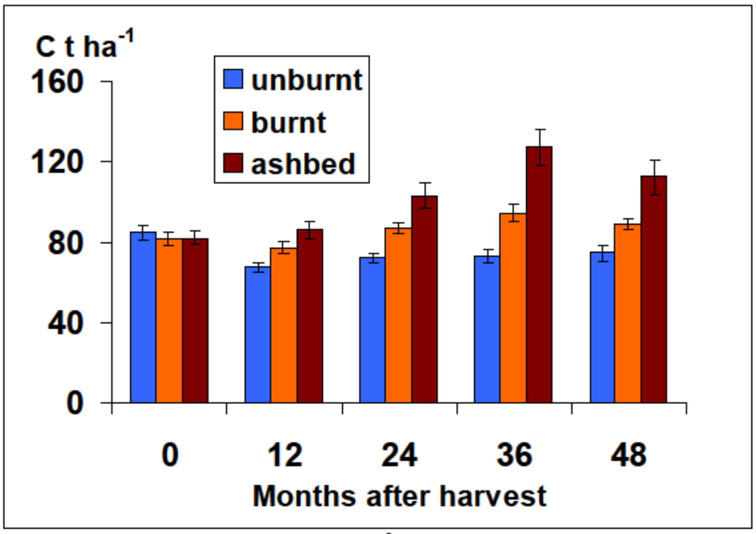

Fig. 5. Changes in soil C stock (tC ha^{-1}, 0-40 cm depth) during the first 4 years after harvesting coastal eucalypt forest followed by fire of differing intensities (from Raison et al., 2009). Ashbeds are those parts of the harvested area where accumulations of woody fuels have burned resulting in intense heating of the soil.

There is insufficient empirical data to estimate the effects of forest harvesting or fire on soil C stocks across the diverse Australian native forest estate. Models of varying complexity have been used to estimate temporal change (Dean et al., 2012; Ximenes et al. 2016; England et al., 2014) but these involve use of critical untested assumptions, leading to little confidence in predictions.

Often ignored are the potentially very important impacts of fire on soil erosion in forests. Soil erosion rates often markedly increase after wildfire, with direct and indirect (mostly longer-term) effects on forest C dynamics (Raison et al., 2009).

An assumption can be made that over the course of a forest rotation soil C stocks do not change [supported by Grant et al. (2005), Norris et al. (2010), Page et al. (2011), England et al. (2014) and McIntosh and Moroni (2016)]. This review therefore focuses attention on above-ground C stocks for which the dynamics over time following disturbance by either harvesting or fire are better known.

(3) Estimating temporal change in biomass C stocks in older forests.

The C stock in AGB of younger forests, with trees of diameter at breast height (DBH) <~0.8 m, is commonly estimated by applying allometric equations to forest inventory data (usually data on tree diameters measured on replicated sample plots of known area), but sometimes trees are weighed directly on sample plots (Ximenes et al., 2016). Many estimates of AGB of younger stands are available, but the challenge of estimating forest biomass is much greater in old forests for which relatively few and likely unreliable estimates are available

The major uncertainties associated with estimating C in AGB in older forests result from the presence of very large trees (DBH from 1-5 m) and greater amounts of wood decay in boles and large branches. There are no allometric equations for such large trees that are based on weighing either the whole tree or sections of it that could be scaled up. All estimates of biomass for very large trees are based on estimates of stem volume (e.g., Dean et al., 2011; Sillett et al., 2015), then converting that to biomass by multiplying by wood density. Often only a single wood density value is applied, which likely creates significant errors, especially when the wood density value is derived from small trees with solid wood (e.g. those compiled by Ilic et al., 2000).

Decay of trunks and large branches increases as eucalypt trees age (Roxburgh et al. 2006; Sillett et al., 2015). Despite claims to the contrary, no study has dealt adequately with correcting for decay when estimating AGB of large trees. Roxburgh et al. (2006) modified allometric equations applied to spotted gum (*Corymbia maculata)* to allow for internal decay based on visual observations on 527 trees destructively sampled by Ximenes (pers. comm.). Ximenes estimated that defect commenced in trees >50 cm DBH, and that for trees of >120 cm DBH tree mass was likely only about 50% of that estimated using allometric equations.

Keith et al. (2009) attempted to adjust for the loss of C caused by wood decay in old mountain in ash trees located in Victoria. They based their adjustments on broad observations of decay made in spotted gum forest on the south coast of NSW. How reductions were made is not clearly described. Even with adjustment their reported AGB biomass estimates exceeded all other estimates reported for mountain ash forests (Volkova et al., 2018). Sillett et al. (2015) acknowledged decay as a major issue in old eucalypts, leading them to conclude that several of the very high estimates of biomass C stock in old mountain ash stands (e.g., those of Keith et al., 2014a) were unlikely to be correct. Sillett et al. concluded that estimates should be seen as 'maxima' rather the true value.

Ximenes et al. (2016) weighed whole trees at a range of sites in NSW and Victoria and compared direct measures of biomass with those estimated by a range of existing allometric equations. They found that existing equations were often poor predictors for large trees (DBH >~1 m) and concluded that previous studies overestimated biomass, and that the estimates of Keith et al. (2015) were too high.

The above conclusions are important because calculating C benefits from no-harvest management requires reliable estimates of C accumulation in large trees in forests older than 100 years. Since the reliability of estimates declines in older forests, forecasting the rate of C gain as forests age is problematic. Keith et al. (2014a, 2015) fitted curves to field estimates of C in AGB (Figures 6 and 7), but with a paucity of data and highly variable estimates beyond age 80 years. In both cases the few observations at age 200 or 250 years exert enormous leverage on the model fitted. The approach used is not appropriate and their estimates of C gain in older forests are thus highly uncertain.

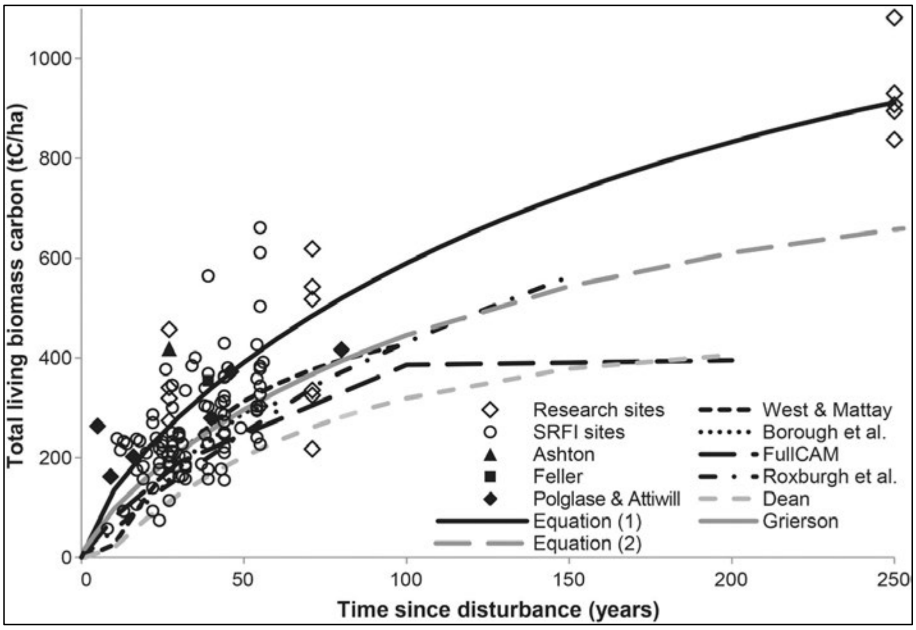

Fig. 6. C accumulation curves for mountain ash forests used by Keith et al. (2014a, 2015). Two curves were used: equation (1) and equation (2) to estimate the long-term patterns of C accumulation in AGB.

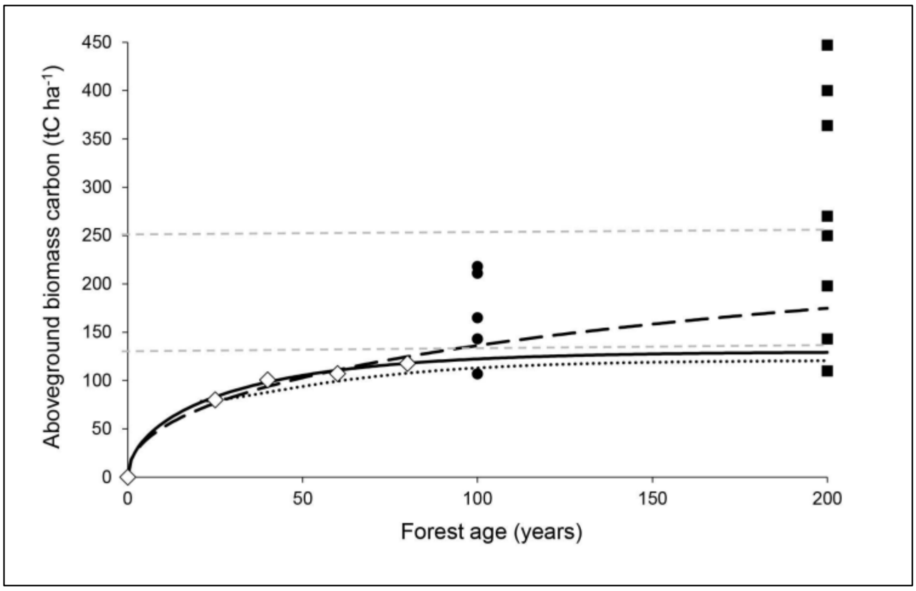

Fig. 7. C accumulation curves used by Keith et al. (2014a, 2015) for regrowing forests on the south coast of NSW. The authors used one Equation (S2-1) derived from the maximum C stock of Ximenes et al. (2012) (solid black line). The other equation (S2-2) was derived from site measurements in undisturbed forest (dashed black line). Site data of biomass carbon stocks included: undisturbed sites (black squares), sites selectively harvested at an unknown time (black circles), and inventory data of biomass up to 80 years (open diamonds). The simulated C stock in a 'conservation forest' (dotted black curve) was used by Ximenes et al. (2012).

A major synthesis of the C dynamics in Victorian mountain ash forests (the most studied forest type in Australia), based on 635 field plots and a review of other published data, was published by Volkova et al. (2018). They demonstrated wide variation in C stocks at different locations across the landscape for forests of the same age (at least 5-fold variation at ages 80, 150 and 250 years) reflecting growing conditions (soil fertility and micro-climate) and other factors (Fig. 8). Age is clearly not a good predictor of site biomass accumulation, and age can only be used to give a broad indication at the landscape scale. Broadly, C stocks in live biomass of regenerating mountain ash increase rapidly to about 300 t/ha by age 100 years, then the rate of accumulation flattens, with stocks reaching about 400 t/ha at age 250 years. The average landscape-scale maximum C storage in AGB would be about 450 t/ha, and the difference between C stocks at age 80 and 250 years only a little more than 100 t/ha. Much higher C stocks can occur at specific locations with favourable hydrologic conditions, deeper and more fertile soils, and topography that shields trees from wind and intense fire, but such sites are not very common across the distribution of mountain ash forest. Moroni et al. (2010) similarly concluded based on biomass estimates for 3,500 field inventory plots in a wide range of forest associations in Tasmania that very high C densities (~470 tC/ha) only occurred on about 0.2 % of the forest area.

The data used by Keith et al. (2014a, 2015) [Fig.6] contrast sharply with those presented by Volkova et al. (2018) [Fig. 8] for mountain ash forests. At age 250 years, equations (1) and (2) of Keith et al. predict C stocks in live biomass to be about 900 and 650 tC/ha, respectively. The estimated landscape average at the same age, based on estimates at many more locations, by Volkova et al. is only about 400 tC/ha. Thus, the model applied by Keith et al. (2014a, 2015) likely overestimates maximum C stocks by ~60-120%, resulting in a large overestimate of the rate of C gain in AGB and consequently inflated estimates of the C benefits accruing from non-harvest management.

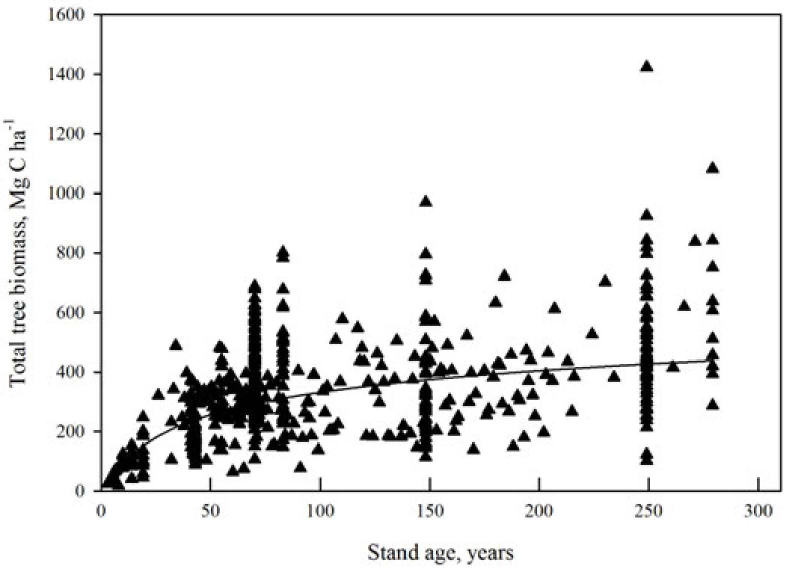

Fig. 8. The C accumulation curve for mountain ash forest in Victoria derived by Volkova et al. (2018). The solid line is the best fit to the data and takes account of the large variation in stand biomass across the landscape for stands of the same age.

Estimates of C stocks in live and dead biomass in old-growth mountain ash forests made in earlier work by Keith et al., (2009, 2010) are as high as 3,000 t/ha. These are much greater than any other reported estimates, probably due to several sources of estimation error including inappropriate inventory methods and application of unreliable allometric equations.

Analysis of C gain in harvested forests must span several hundred years to account for the gradual on-going accumulation of biomass C, or the potential loss of biomass C resulting from succession to communities with much lower C stock in AGB (Moroni et al, 2010; McIntosh and Moroni, 2016). In contrast with a harvesting regime, benefits to C storage with non-harvest management can only occur once, until maximum C storage is reached. Further, such management carries the risk or inevitability that wildfire will intervene, resulting in a significant amount of the sequestered C being returned to the atmosphere due to combustion or to decay of fire-killed vegetation.

(4) The effects of fire on forest C dynamics

In the highly fire-prone Australian environment, the effect of fire, both planned fire and unplanned wildfire, on forest C dynamics must be considered (Raison et al. 2009). Direct effects of fire are usually short-term, such as combustion of organic matter and killing of vegetation. Longer-term effects are due to changes in micro-climate, decomposition of killed vegetation and C sequestration by regenerating vegetation and have consequences for the C cycle for decades or centuries. Further complexity is caused by whether forests are largely killed by wildfire and regenerate from seed, or whether individual trees survive and resprout; and by interactions between prior harvesting and wildfire that can then affect C emissions in subsequent wildfires (Wilson and Bradstock, 2022).

The effects of managed fire will differ in native forests managed for conservation of C, or for wood production. Low-intensity fire (prescribed burning) is used to reduce fuel loads in most forest types. In many forests managed for wood production planned fire of higher intensity is also used to both reduce fuels and subsequent fire risk and to assist regeneration following harvesting, especially in wetter forest types. Wildfires have the greatest impacts, especially where they lead to stand replacement, and can affect all forest types. The way in which fire is treated in Australia's national C accounting is complex and based on some key assumptions which require re-examination (Bowman et al, 2023).

Key mechanisms by which fire affects forest C budgets are:

- Volatilization of C during combustion of vegetation, litter, coarse woody debris (CWD), and organic matter in surface soils when they are very dry and fires are intense. When forest fuels are combusted, essentially all the C in the fuel is volatilized (Raison et al., 1985).
- Conversion of about 1% of the C in combusted fuels to relatively inert char, which persists for very long periods in soils and sediments (Forbes et al., 2006).
- Promotion of regrowth and increased C fixation from the atmosphere (Fairman et al., 2022; Volkova et al., 2018).
- Loss of soil C stocks in the short-term due to increased soil respiration after disturbance, or erosion.
- Reductions in forest growth due to loss of soil fertility.

Keith et al. (2015) argue that forest C stocks are relatively stable over time even when forests are subject to repeated wildfires, because wildfires only consume a relatively small fraction of the C in AGB, and that this loss is replaced by C accumulating in new regrowth within a decade. While this may be largely true in the short-term, it does not follow that C stocks will be maintained in the longer term because there are significant ongoing losses of C that also occur in forests killed or damaged by intense wildfire. Further, as discussed below, repeated wildfires can also affect forest species composition and growth rates resulting in lowering of C stocks.

Killed biomass (both above- and below-ground) will continue to decay over time. These losses must be deducted from the C accumulated in regrowth to estimate the net effect. Standing killed trees are the main stock of C in AGB after wildfire in 'ash-type' forests – these start to decay whilst trees are still standing but decay accelerates after the trunks fall, and they will then also be more likely to be consumed in subsequent fires burning under drought conditions. As an example, 400 tC/ha in killed standing trees decaying at 1% per year releases 4 tC/ha/yr to the atmosphere for multiple decades. Such a rate of C loss is a significant fraction of net primary productivity in older mountain ash forests (Volkova et al., 2018) and contributes to the rapid tapering off in C gain above-ground (Fig. 8). If the argument put forward by Keith et al. (2015) were correct, mountain ash forests burnt by wildfire but not subsequently logged would have similar C stock in AGB to that in long-unburnt forests. This is clearly not so (Ximenes et al., 2016). C stocks will be relatively more stable over time in re-sprouting forests where there is generally not high mortality due to wildfire (Wilson and Bradstock, 2022).

Frequent wildfire may also lower the capacity of regrowth forests to recover C stocks. In ash-type eucalypt forests which are obligate seeders, a second wildfire before viable seed is produced can lead to succession to less productive vegetation types (Bowman et al., 2014). There is also mounting evidence that fires of high severity coupled with drought can lower the regenerative capacity and C gain of multi-aged mixed species forests resprouting after wildfire (Bennett et al., 2016; Fairman et al., 2022; Bendall et al., 2022).

Wildfires that result in stand replacement in ash-type eucalypt forests create conditions (reduced litter inputs, moister and warmer soil conditions) that increase the loss of soil C by respiration in the short term (Raison et al. 2009), but soil C stocks are likely to recover over subsequent decades.

Several modelling studies of the effects of wildfire in wet eucalypt forests predict temporal swings in total forest C of up to ~500 t/ha over the course of a century (Dean et al., 2012; Ximenes et al., 2016). Forest C stocks are clearly not stable under a regime of repeated wildfires.

There has been much debate in recent years on the very important question as to whether harvesting makes native forests more flammable, and thus increases the risks from wildfires at the landscape scale (Lindenmayer et al., 2006, 2022;

Bradstock and Price, 2014; Taylor et al., 2014; Attiwill et al., 2014; Tolhurst and McCarthy, 2016; Keenan et al., 2021; Bowman et al., 2016, 2021, 2022). If harvesting did increase landscape fire, this would have a negative impact on many forest values and the increased C emissions would need to be included in the LCA for harvested forests. It is noteworthy that 3 major government enquiries did not establish any link between timber harvesting and fire risk (Keenan et al. 2021) and that during the last 2 decades when native forest harvesting has declined markedly the area burnt by wildfires has more than doubled (MIG and NFISC, 2018). At the scale of the harvested area, fuel loads and fire risk are likely to be greater in the short-term after harvesting, but of greater significance is whether harvesting, which creates a landscape mosaic of forests of differing ages (or time since the last harvest event) over rotations of 80-100 years poses a greater fire risk across the landscape during severe fire weather. Key insights were gained from analysis of fire spread and severity during the extensive wildfires of the summer of 2019/20 in SE Australia (often referred to as the Black Summer fires) which burnt about 7 million ha (Davey and Sarre, 2020) of mostly eucalypt forest in NSW and Victoria.

Lindenmayer et al. (2020) re-opened the debate after the Black Summer fires by claiming that prior logging and forest management had made the fires worse, and that this was a further reason for stopping harvesting of native forests. They did not provide any direct evidence based on the actual fires. Keenan et al. (2021) reviewed evidence for any link between timber harvesting and fire extent and severity and concluded that at the landscape scale there was none, with the dominant factors driving fire behavior being 3 years of prior drought, and local topography. A further detailed study by Bowman et al. (2021a) involving 3 regions (total of 2.35 million ha) across NSW and Victoria impacted by the wildfires found that prior timber harvesting had only a very small effect on severe canopy damage, with topographic factors and fire weather the dominant factors affecting fire spread and severity. Abram et al. (2021) concluded that the fires were probably exacerbated by climate change, whilst Adams et al. (2020) reported that in NSW wildfires burned intensely across all tenures and argued that increase in fuel loads in recent decades was a strong contributing factor.

Lindenmayer et al. (2022) made further claims that prior logging increased the severity of wildfires and re-analyzed some of the data of Bowman et al. (2021) to support their argument based on extrapolation of research from tall mountain ash forest. Bowman et al. (2022) provided a strong and convincing rebuttal, showing that Lindenmayer et al. (2022) had mis-analyzed their data and used it out of statistical context. They confirmed that prior harvesting had a negligible effect on the severity of wildfire at landscape scales and concluded by stating: ‘Solely focusing scientific and media attention on the small, and highly variable, relationship between past logging and fire severity distracts from evidence-based policy regarding options for managing future fire risks.’

Bowman et al. (2020) noted that the widespread use of prescribed burning to reduce fire risk also causes C emissions. Prescribed burning can reduce fire severity at the local scale (Hislop et al., 2020), but its effectiveness at landscape scales under extreme fire weather is still debated (e.g., Tolhurst and McCarthy, 2016; Morgan et al., 2020). Volkova et al. (2021) used a detailed model to explore the relationship between prescribed burning and C balance in eucalypt forests. They concluded that prescribed burning has limited potential to reduce C emissions from wildfire, but also that using it to achieve other forest management objectives incurred no C emissions penalty. This is an important finding, but of course there are many assumptions embedded in such models, particularly relating to impacts of fire on below-ground processes including nutrient cycling which affect subsequent rates of C sequestration into vegetation and soils.

There are some critical conclusions pertinent to maintaining C stocks and other forest values in native forests that emerge from the above discussion. Firstly, wildfire can have major impacts on forest C stocks that can persist for centuries. Secondly, under sustained severe fire weather all forests across the landscape will burn at high intensity and severity resulting in large C emissions irrespective of whether they have a history of harvesting or not. Thirdly, management of fire will be more challenging in future under a predicted warmer and drier climate in SE Australia - both the area burnt and the number of megafires (> 1 million ha burnt) have increased in Australia in recent decades (Morgan et al. 2020, Canadell et al., 2021). Unless ways can be found to mitigate escalating bushfires, C storage in all native forests (irrespective of management intent) is clearly threatened and C stocks will almost certainly decline into the future. Fourthly, the threat from wildfire to any C emissions abatement benefit derived from forests will be greater under a non-harvest management scenario which relies on the protection of forest C stocks over a very long time into the future. Harvested forests managed on rotations of 80-100 years provide more opportunities for any benefits related to the use of harvested wood to be 'banked' progressively over time as C stocks in HWP in use and landfill, and emissions savings from reduced use of fossil fuels or other high energy-embedded material.

Commentary on four LCA studies comparing the effects of non-harvest (conservation) management and sustainable harvesting on the C balance of Australian native forests.

Ximenes et al. (2012). This was the first detailed LCA conducted in Australian native forests. Extensive field measurements were used to model the C balance over 200 years under either selective harvesting or no harvest management in production forests on the north and south coast regions of NSW that accounted for 12% of the native forest estate available for harvest in NSW and 25% of the volume of sawlogs produced. The study included the effects of product substitution and bioenergy offsets.

Simulation of forest growth and C dynamics was based on forest inventory data and growth predictions from the empirical model FRAMES (Forest Resource and Management Evaluation System) used by Forestry Corporation of NSW (see Forestry Corporation of NSW 2016). Rates of C gain in the conservation forest beyond 20-40 years after the current state are low, and both case studies showed a significant C benefit of forest harvesting and HWP use compared with conservation. Key components to the net benefit increase progressively beyond 40 years, with the product substitution effect large and on-going (Fig. 9), especially if alternative products are sourced from production systems where C emissions are high.

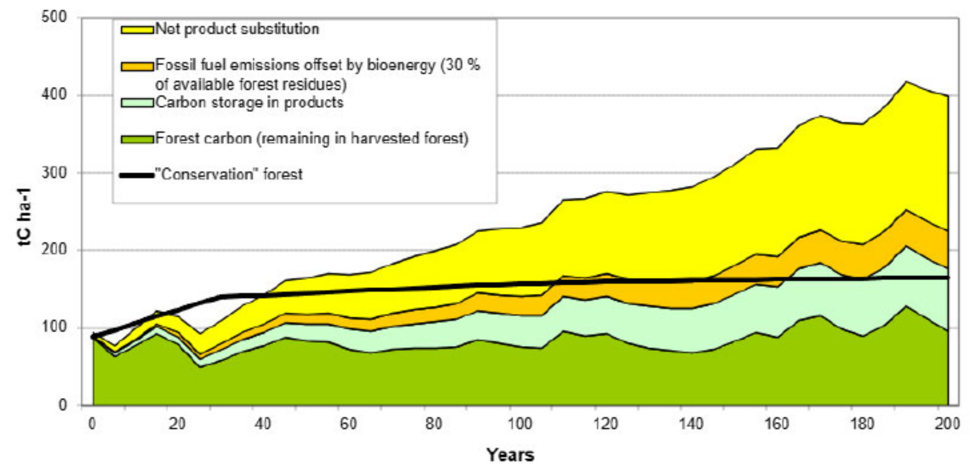

Fig. 9. C implications of the 'conservation' and 'wood production' scenarios (t C/ha sequestered or displaced) for north coast forests modelled over a 200-year period (Ximenes et al., 2012).

The rate of C accumulation in AGB is quite slow in these NSW forest types and so is total C storage. This contrasts with that seen in more productive Victorian mountain ash forests (Fig. 8). Slower growth rates and lower total C gain limit benefits of the no-harvest management option.

Keith et al. (2014a). The authors used mountain ash forests in the Central Highlands of Victoria as a case study for comparing the effects of harvesting or conservation on C stocks. The authors concluded that conservation resulted in the best C outcomes, but there are 6 important limitations to this study:

(1) Use of an inappropriate split between felled biomass removed in harvest and biomass left on site as slash.

The authors claim to have used the generalized default value for the proportion of biomass removed from wet forests subject to sawlog plus pulpwood harvest proposed by Raison and Squire (2010). This value was 60% (Table 1), but Keith et al. appear to have misunderstood the default and instead used 40% of total biomass. The field study by Ximenes et al. (2016) also found that the fraction of biomass removed was much higher (nearly 80%). Use of 40% for biomass utilization after harvest overestimates the amount of slash that is combusted or decays to releases C to the atmosphere. The authors used a pulp:sawlog ratio of 72:28, but Ximenes et al. (2016) found the proportion of biomass in sawlogs ranged from 36 to around 50%. Keith et al. also assumed only 4% of the initial C stock is destined for sawn timber. Ximenes et al. (2016) showed that this figure ranges from 8.5% (sawn timber from high-quality sawlogs only) to 17% (sawn timber from all sawlogs).

The use of inappropriate wood recovery rates and/or pulp:sawlog ratios negatively impact the estimated C balance of harvested forests and inflate the proposed benefits of forest conservation. Consequently, the analysis and conclusions provided by Keith et al. are unrealistic.

(2) The rates of C accumulation assumed for older forests are much too high.

As discussed earlier, Keith et al. adopted unrealistically high rates of C accumulation in AGB beyond 80 years of age. They use their 5 observations for 250-year-old stands as the sole basis for extrapolating AGB beyond 80 years of age. Their estimates are much greater than the projections of all other authors (Fig. 6). Subsequent analysis by Volkova et al. (2018) based on many more field estimates for older stands show much lower rates of C gain as forests age (Fig. 8). Use by Keith et al. of these high rates of C accumulation greatly inflates the estimated benefit of forest conservation.

(3) The C accounting approach used for harvested forests was incomplete.

Keith et al. did not consider the use of forest residues for bioenergy, as a substitute for fossil fuels. They also did not include substitution of wood for more energy-intensive products such as steel, concrete and aluminium, or the C costs of obtaining wood products (including high-quality paper) from alternative sources if the forests were no longer harvested. Ximenes et al. (2016) showed that these factors are a major contributor to the net positive C effects of the production and use of HWP in mountain ash forests.

(4) Decay rates applied to wood and paper in landfills were unrealistically high.

See discussion by Ximenes et al. (2016) and Ximenes (2021) for further details.

(5) Use of incorrect contextual information leads to inappropriate conclusions.

The impression in Keith et al. is that logging at the time was mainly of old-growth forests leading to a large C debt that can never be recovered. The reality is that almost all harvesting at that time was in regrowth mountain ash forests. Further, the authors analysed harvest rotations of 50 years, whereas planned rotations at that time were 60-100 years. Use of shorter rotations reduced the time-averaged C stock in forests under a harvesting scenario and hence favours a non-harvesting scenario.

Keith et al. (2015). In this study, the C consequences of a range of native forest management scenarios were simulated over a 100-year period using current commercial forestry practice as the baseline, for mixed-eucalypt forests in New South Wales and mountain ash forests in Victoria. The production scenario included C stocks in wood and paper products, waste in landfill, and bioenergy that substituted for fossil fuel energy (the C impacts of importing forest products if they are not produced in Australia was not included).

The authors concluded that none of the parameter values tested could increase the C stock in HWP in use and landfill and the C savings from fossil-fuel substitution sufficiently to compare with the increase in C stock due to changing management to a no-harvest regime, and that the "no-harvest" approach therefore led to a superior climate mitigation outcome.

However, the major problems identified above for Keith et al. (2014a) also apply to this paper. Unfortunately, this invalid C accounting model has also been used in subsequent work to promote the non-harvest approach in mountain ash forests (Keith et al., 2022). The analysis by Keith et al. (2015) is also limited as it covers the impact of only one harvest cycle. Malmsheimer et al. (2014) found – as have others – that 'extending the period of analysis through multiple rotations can be critical to understanding the short- and long-term GHG implications of using forest-based products.' Product substitution impacts due to the use of paper products were also not included – these are shown by Ximenes et al. (2016) to contribute significantly to the C balance of mountain ash production forests.

Although Keith et al. (2015) provide estimates for fossil fuel displacement benefits due to the use of residues for bioenergy, several arguments against the use of biomass for bioenergy are also put forward. They argue the need to account for the time taken to replenish the C released when the biomass is used for bioenergy. This is not a significant consideration for a sustainably managed system if a landscape assessment is conducted (not just the harvested site in isolation), where emissions and removals are balanced over time. Keith et al. (2015) also argue that removal of biomass for bioenergy may lead to nutrient deficiencies in the forest. That is likely to be the case if high-nutrient biomass fractions such as bark, foliage and small twigs are removed, but those biomass fractions are unsuitable for energy generation purposes and would be left in the forest. Ximenes et al. (2017) stated that 'retention and management of bark on site, retention of leaves and minimising post-harvest regeneration burns are identified as key actions to minimise any impacts on nutrient availability due to extraction of biomass'.

In summary, the analysis conducted by Keith et al. (2015) does not provide a reliable assessment of the comparative impacts of no-harvest and harvest management on net C balance.

Ximenes et al. (2016). This detailed study examined the C balance in native forest under both no-harvest and wood production management. It considered all key elements of the C cycle both within forests and in HWP, including substitution within the Australian context. A model (ForestHWP) was developed to simulate changes in forest C pools and changes in C outside the forest, and the study included a cost-benefit analysis of socio-economic and carbon pricing implications.

The authors used three case studies: mountain ash forest regenerated after the 1939 wildfires in Victoria (Fig. 10); forests dominated by silvertop ash on the south coast of NSW; and moist/wet forests dominated by blackbutt on the mid-north coast of NSW. The three case study areas approach had paired 0.5 ha 'production' and 'conservation' sites. Forest biomass data was derived from extensive field work involving direct weighing of hundreds of trees. Site-specific parameters estimated above-ground forest C pools and emissions due to harvest and fire, and the fate of C in HWPs derived from the forests was tracked. The site-specific data were not replicated but were compared with data for similar forest types and from regional commercial forest operations to check that the values were within the wider ranges observed.

This study used the same models for C accumulation in old forests as Keith et al. (2014a), resulting in an overestimation of the C benefits of unharvested forests.

Fig. 10. Conservation site in 1939 regrowth mountain ash forest studied by Ximenes et al. (2016).

Table 2 summarises outcomes for business-as-usual (BAU) and alternative management scenarios. The data are long-term averages across the forest estate, all expressed as effective C stocks from managing one hectare (tC/ha) and based on harvest rotations of 65 years in north and south coast New South Wales and 75 years in Victorian mountain ash.

The C benefits were greatest for mountain ash harvest scenarios, with a slight benefit for the silvertop ash harvest option. For mountain ash in Victoria, the conservation forest is estimated to contain a long-term average of 522 tC/ha in AGB, whilst in harvested forests the equivalent value is 835 tC/ha, representing the net effects of changes in C stock in the forest (negative), producing and using HWPs (positive) and benefits accruing from product substitution (positive).

For blackbutt, the harvest option resulted in slightly lower benefits than the conservation option. However, use of a portion of blackbutt harvest residues for bioenergy led to the "harvest" option having superior C mitigation outcomes. Further, most of the "harvest" alternative utilization scenarios modelled resulted in increased net C benefits.

	Victoria Central Highlands Mtn Ash	North Coast Blackbutt	South Coast Silvertop Ash
BAU No harvest (tonnes C/ha)	522.8	247.5	276.7
BAU Harvesting (tonnes C/ha)	835.2	218.5	288.4
Differences due to non-BAU HWP management scenarios (% difference from BAU Harvesting)			
30% of forest residue to bioenergy	(1.2)	NA	(5.9)
50% of forest residue to bioenergy	NA	(17.6)	NA
50% of forest residue to pulp	NA	(47.9)	NA
50% pulp to bioenergy	(-17.3)	NA	(-11.4)
100% pulp to bioenergy	(-32.9)	NA	(-22.8)
End-of-life products and all waste to bioenergy	(2.8)	(11.9)	9.1 (3.2)
Max products to bioenergy	(3.9)	(28.0)	(4.4)
End-of-life pallet mulch to bioenergy	(1.3)	NA	NA
Waste to product (max product)	(1.6)	(28.8)	(10.3)
Max product to landfill	(4.8)	(29.3)	(10.6)
Site product spread for a greater % of poles	NA	(3.1)	NA

Table. 2. Comparison of the implications of changed forest management and wood utilization compared with the Business as Usual (BAU) management scenarios. For the different wood utilization scenarios, the percentage change is shown, and negative values indicate poorer C outcomes relative to BAU. Table from Ximenes, 2021b – adapted from the detailed studies of Ximenes et al. (2016). Non – applicable (NA).

Other key conclusions from the study were:

- Existing allometric equations tend to overestimate biomass in the forest types studied, especially for large diameter trees. For trees >1 m in diameter, the weighed mass was always less than that estimated by all existing allometric equations. Large trees were also more variable in their mass, suggesting significant and variable wood decay.
- Operations produce large volumes of harvest and mill residues that could be utilised for applications such as bioenergy generation.
- The majority of HWPs are eventually disposed of in landfills, creating a long-term reservoir of C.
- The HWPs produced 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 by imported hardwood products (decking and flooring), fibre-cement cladding, concrete slabs and steel and concrete transmission poles.

- Scenarios for conservation management included increased emissions resulting from import of paper from unsustainable wood sources from south-east Asia.

Ximenes et al. (2016) concluded that the differences in the C balance between wood production and no harvest scenarios did not warrant policies to halt native forest management for wood production. They identified considerable room for improving forest management and wood utilization for better C outcomes in production forests, including better recovery of felled biomass, as well as for industry development to provide superior C outcomes and multiple other benefits.

The effect of closing Victoria's native forest timber industry on net C emissions

This section uses the findings of Ximenes et al. (2016) on mountain ash forest to analyse the impacts on C emissions of ceasing harvesting in these forests in Victoria.

The Victorian State Government decision to close the native forest timber industry was partially based on the belief that this would deliver a considerable reduction in C emissions (D'Ambrosio 2019; Hansard, Victoria 2019). Similar arguments have been made by environmental groups in Tasmania, NSW and WA (e.g. Cross et al., 2023). A subsequent report by Sanger (2022) in Victoria made similar claims and was proposed as a contribution to Victoria's 2035 Climate Action Target (Independent Expert Panel for Victoria's 2035 Emissions reduction Target 2023).

The Victorian Government has estimated that closing the state's native forest timber industry will reduce C emissions by an amount equivalent to taking 730,000 cars off the state's roads for the next 25 years (D'Ambrosio 2019). The Government has not disclosed the science that informs this expectation, but at 3.2 tonnes of CO_2-equivalent emitted per car per annum (Sanger et al. 2022) the Government must expect the closure of the industry to reduce C emissions by 2.336 million tonnes of CO_2-equivalent per annum. This is not supported by detailed Life Cycle Analysis (LCA) of the carbon stocks and flows associated with native forest wood production in SE Australia (Ximenes et al. 2016).

Box 1 provides a summary of the likely C outcome from cessation of harvesting by applying the findings of Ximenes et al. (2016) to a realistic harvesting scenario for mountain ash forests.

Box 1. Impact of the Victorian Government's native forest harvesting decision on C emissions from mountain ash forests

The following analysis is based on data from detailed field studies and modelling in 1939-regrowth mountain ash forests in Central Victoria (Ximenes et al., 2016), plus unpublished data from F. Ximenes on in-service storage of C in HWP. Car equivalents are calculated based on a conversion factor of 3.2 tonnes of CO_2-equivalent emitted per car per annum (Sanger, 2022).

A **A harvesting rate of 1,000 hectares per year was applied to regrowth mountain ash forest managed on a 75-year rotation (total harvested area of 75,000 ha), producing a mixed yield of sawlogs and pulpwood used for domestic paper making at Gippsland's Maryvale mill.** Data were taken from the LCA of Ximenes et al. (2016) that compares net C emissions from harvesting or not harvesting regrowth mountain ash forest under a BAU scenario that existed at that time (Table 2). The C changes shown below reflect the effects of an annual harvest of 1000 ha and are sustained across the entire forest rotation during which 75,000 ha are harvested. A larger annual harvest area (and thus total harvested forest estate) would confer a larger C change. All forests (harvested and non-harvested) are assumed subject to low-frequency wildfire.

B **Net annual loss in forest C stock due to log removal and burning and decay of harvest residues is 0.381 Mt CO_2-equivalent (equates to an extra 119,000 cars on the road).** This quantifies the difference in forest C storage between managing the forests on a 75-year cycle of harvest and regeneration versus leaving them unharvested. This value is likely an over-estimate because, as discussed earlier, the C increment for unharvested forest used by Ximenes et al. (2016) was too high.

C **Annual C storage and reduced emissions due to local production and use of HWP is 0.218 Mt CO_2-equivalent (equates 68,000 less cars on the road).** This quantifies the positive effects of HWPs in storing carbon, plus the reduction of emissions from bioenergy and use of C-intensive alternative materials, such as steel, concrete and aluminium.

D **Annual C savings due to reduced global emissions from local production of high-quality paper compared to importing it is 1.133 Mt CO_2-equivalent (equates to 354,000 less cars on the road).** This quantifies the positive effects of superior forest management associated with local production of high-quality writing and printing papers relative to the higher C emissions associated with pulp and paper production in SE Asia (Ximenes et al., 2016). Victoria will most likely import paper products from SE Asia now that the local timber industry has closed (Venn, 2023).

E **Sustainable Net Annual C benefit from annual harvest of 1,000 hectares of Victorian mountain ash regrowth forest is 0.97 Mt CO_2[1]-equivalent (equates to 303,000 less cars on Victoria's roads).**

Ceasing harvest of mountain ash forests in Victoria will therefore increase net C emissions, unlike the claim of the Victorian Government. This conclusion also contrasts with that of Sanger (2022). Closing Victoria's native forest timber

[1] Calculation 0.97=1.133+0.218-0.381

industry may also degrade forests in SE Asia because of the import of paper produced there (Venn, 2023).

Critical factors for achieving and estimating net C mitigation benefits

This summary is based on the analysis earlier in this paper of several LCA studies and review of key considerations for assessing changes in C stocks in native forests managed for wood production or subjected to no-harvest management.

(1) Unharvested forests

Two critical factors affect the viability of a non-harvest strategy in providing reliable climate change mitigation in forests.

(i) Reliable estimation of C accumulation in older forests

Estimating C accumulation in forest AGB beyond about age 80 years is problematic. The relatively few estimates made are unreliable because of a lack of validated allometric equations for large trees and a failure to adjust for increasing levels of wood decay as trees age. Most biomass estimates in old forests are over-estimates (Ximenes et al., 2018). Models such as those used by Keith et al. (2014a, 2015) for making forecasts of C in AGB a hundred or more years into the future are subject to large and undefined uncertainties. Several authors have concluded that the C stock used by Keith et al. (2014a, 2015) for old-growth mountain ash is too high (Sillett et al., 2018; Ximenes et al., 2016).

(ii) Stability of forest C stocks after wildfires

The conservation argument requires that the accumulated C stock in older forests is not depleted by wildfire. This is most unlikely.

(2) Harvested forests

A complete LCA must be applied to enable accurate tracking of C changes both in the forest and in harvested wood over a full forest rotation. Most Australian studies have not done this. Use of appropriate figures for biomass removal in harvested sawlogs and pulp logs is critical to conducting an accurate LCA.

(i) A high utilization of felled biomass is needed. Unless this occurs, much felled biomass will either be burnt or will decay in the forest, rapidly returning C to the atmosphere, and benefits from use of HWPs will be reduced. There are strong market drivers for high utilization of harvested biomass.

(ii) Forests need to be protected from wildfire so that a sustainable yield of merchantable biomass can be harvested over the long-term. Unless this occurs, forest C stocks will not be maintained, and benefits derived from use of HWPs will not be realized.

(iii) Wherever feasible, HWPs produced need to be used to substitute to materials with a higher C footprint such as steel, concrete, and aluminium. Current government policies are encouraging this.

International position on the role of managed forests in contributing to the mitigation of C emissions.

There are numerous international studies comparing the GHG implications of managing forests under harvest and non-harvest scenarios (Olsson et al. 2019), Krug et al. (2012), Ameray et al. (2021), Schulze et al. (2022), Peng et al. (2023), providing useful insights into international discussion about the role of forest management in the global C cycle. The review of Ximenes (2021b) stressed that the completeness of C accounting and timescales of analyses strongly affected conclusions drawn.

Oliver et al. (2014) undertook a global assessment of forest management strategies and found that substitution of wood for other construction materials generated higher levels of avoided emissions compared to storage of C in HWPs or use of wood for bioenergy; and that overall harvested forests generated greater emissions savings than unharvested forests. In Canada, comprehensive studies considering all key C removals and emissions pathways concluded that the most effective strategies for climate change mitigation in the medium to long-term were those that involved optimum sustainable utilization of forest biomass (Smyth et al. 2014, 2020).

Ximenes (2021b) identified many international documents outlining how accounting for C changes in managed native forests fits within national and international frameworks and sustainability principles. For example, a recent report by the FAO (2021) provides a useful overview of how countries planned contributions to the Paris Agreement can include forests and HWPs. Voluntary commitments to emissions reductions are expressed as nationally determined contributions (NDCs), and about 70% of the countries that submitted NDCs to the UNFCCC included production forests in their planned contributions. There is strong commitment to using forests to achieve long-term GHG emissions reduction goals in Europe. The EU Forest Strategy for 2030 (EU, 2021) stated that **‘**GHG emissions and removals by forests and forest products will play a crucial role in reaching the ambitious net removal target for the Union of -310 million tonnes of carbon dioxide-equivalents’.

The high-level statements by the IPCC and FAO represent a collective international view on the positive role that sustainably managed production forests play in mitigating climate change, drawing on recent underlying science. This role will likely be strengthened over time as more of currently under-utilised forestry biomass (harvest and processing residues) is used to manufacture novel bio-products that will displace products produced using fossil fuels.

Management of forests for C mitigation benefits in the context of broader management goals

Sustainable forest management seeks to achieve environmental outcomes, promote economic development, and maintain the social values of forests, to meet the needs of society without compromising the ability of future generations to meet their needs. This reflects the principal objectives of the United Nations Convention on Biological Diversity and is also an important approach to manage C in the context of the United Nations Framework Convention on Climate Change (Parrotta et al. 2012; MIG and NFISC, 2018). Maintenance of forest contribution to global C cycles was an important consideration in sustainable forest management during the development of Regional Forest Agreements in Australia and in their application (Davey, 2018).

Applying sustainable forest management to Australian native forests is complex due to competing objectives and societal requirements. Management objectives will vary substantially from place to place, and management for C will not always be the highest priority. For example, it may be beneficial to thin forests (which lowers C stocks, unless the felled biomass can be used productively) to increase water supply and counteract the effects of a drying climate. Moroni et al. (2010) noted that managing forests solely for C storage will lead to distorted policy and environmental outcomes; for example, a range of forest age-classes, some with lower C stocks, is needed for enhanced biodiversity outcomes.

Many have proposed a more active and adaptive approach to management of forests on all tenures to deal with a range of threats to forest C and other values (e.g., Gonsalves et al., 2018; Vance, 2018; Morgan et al., 2020; Jackson et al., 2021). Such an approach will require a major increase in resources for planning and implementation of activities on the ground. The sustainable commercial harvest and use of wood products, including of thinned trees, can play a role, especially with the major shortage of wood in Australia and internationally. Income from wood production can help offset forest management costs including improved fire management that contribute directly to climate change mitigation.

Those opposing the harvesting of Australian native forest suggest that HWPs sourced from them could be replaced with products from existing or new plantations. This reflects a poor understanding of both the Australian and the international forestry sectors – Venn (2023) provided a recent comprehensive review of the key issues. Most wood produced from short-rotation hardwood plantations is currently profitably exported due to strong international demand. The species grown in existing plantations are usually not suitable for replacing products such as flooring and external decking, for which native forest timbers are used. Further, Australia's plantation estate has been slowly declining since about 2014 (MIG and NFISC, 2018; ABARES, 2023b) and there are significant barriers to expanding the plantation estate (Venn, 2023). Brizga et al. (2019) provided a detailed review of the situation in Victoria including differing views on the future role of plantations for wood supply. The high cost of suitable land (locations with annual rainfall > 800 mm/yr needed to support good growth of trees for sawlog production) and resistance of landowners to planting trees are key impediments. Plantations are also highly vulnerable to loss from wildfire; 92,000 ha were destroyed in the 2019/20 wildfires in NSW (Davey and Sarre, 2020), resulting in serious interruption to wood supply (Bowman et al., 2022). Whilst some of these impediments may be overcome in the future, there is little prospect of Australian plantations replacing much of the hardwood supplied from native forests during the next 20-30 years. Therefore, if other HWPs are to replace those currently produced from native forests, these are likely to be imported, with a significant risk of C 'leakage' due to increased C emissions from forests harvested (often unsustainably) outside Australia (Venn, 2023). Such risks were discussed by Kastner *et al.* (2011), who stated that 'policies aiming at increasing national forest stocks, should include careful assessments if and to what extent this forest return will be facilitated by increasing risks and vulnerabilities in distant places'.

The greatest threat to the maintenance of C stocks in Australia's native forests is from extensive wildfire, rather than from harvesting which affects only a small proportion of the forested landscape. Logs harvested annually from Australian native forests contain only about 4 Mt CO_2-equivalent, or 1% of national GHG emissions. Additional emissions of C from the decomposition or combustion of logging slash are about 1/3 of this figure. These C removals from the forest are offset by sequestration of C into new regrowth, or by C benefits derived from the use of harvested biomass. In contrast, in bad fire seasons such as the summer of 2019-20, C emissions were twice Australia's total annual anthropogenic (i.e. excluding wildfire emissions) GHG emissions and about 150 times greater than C removals in wood and emissions from logging slash.

Young forests regenerating after either harvesting or wildfire may be more flammable than older forests for several decades, but young regrowth forest created by harvesting is dispersed and occupies only a small fraction of the landscape, whereas young forests generated by extensive wildfires in 2019-2020 now occupy a large and quite contiguous area (Davey and Sarre, 2020). Wildfires in the large and contiguous areas of thick regrowth created after the 'black summer' fire season will pose a major threat to C stocks in all forests during the coming decades. Control of further wildfire in such forests will be a major challenge in future decades (Bowman et al., 2022), and there is a major risk and threat to C stocks in all forests during that time. Frequent and severe fires not only inhibit the rebuilding of forest C stocks but can also drive forest succession toward communities with fewer trees and lower rates of net C fixation and C storage (Bowman et al., 2014; Fairman et al., 2022).

Conclusions

The scientific evidence does not support the contention that management of Australia's native forests under a no- harvest regime would result in greater C mitigation benefits than management for timber harvesting. This conclusion is consistent with the international position. There are also a range of management options that could improve C outcomes in harvested forests.

Several well- publicised Australian studies did not conduct a full Life-Cycle Analysis, and used inappropriate parameters in key parts of their analysis, resulting in marked overestimation of C gains over time in unharvested forests and underestimation of C benefits associated with the harvest and use of wood. Their conclusion that unharvested forests provide a better C mitigation outcome than management for wood production is not valid.

The Australian Forest sector needs to become better equipped to conduct timely, detailed analysis of complex issues relating to the consequences of native forest management for future climate, management of risks from wildfire, water security, and maintenance of landscape biodiversity; and to effectively communicate findings to the public and to policy makers. These are significant tasks, requiring input from high-calibre scientists and adequate resourcing. Without such critical regular reviews and associated communication, flawed scientific conclusions can persist, and shaky foundations can be used to support subsequent decisions. The false contention that cessation of harvesting in native forests provides better C outcomes than sustainable harvesting is a good example – it has lingered for too long and has had important negative policy outcomes.

The management of C in native forests needs to be sensibly integrated at the landscape scale with management for other forest values, and to meet the socio-economic needs of local and rural communities. Management for C alone will provide distorted outcomes. Management priorities will vary greatly in specific locations and maximising C stocks will generally not be the only high priority. Timber harvesting, providing it is well conducted, is a valid forest use in carefully selected parts of the landscape and can provide useful on-going C mitigation benefits now and into the future.

Acknowledgements

The author thanks Dr. Fabiano Ximenes (NSW DPI, Forest Science), Dr. Stephen Roxburgh (CSIRO), Dr. Gary Richards (FlintPro), Dr. Stuart Davey and Dr Steve Read for useful suggestions during the preparation of this review. Dr Read edited and improved the final version of the manuscript. Dr. Ximenes provided unpublished data on C storage in HWP in service that supported the Victorian case study.

Comments from 8 reviewers helped improve the manuscript.

Funding

This work was partly supported by Victorian Hardwood Sawmillers, who also had initial input into defining the scope of this study. All subsequent research and preparation of this review was conducted independently by the author.

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