Review

Not peer-reviewed version

Carbon, Corporate Forestry and Conservation: Struggles for Science in Sustainability and Benchmarks

Christopher Dean *

Posted Date: 8 October 2024

doi: 10.20944/preprints202410.0555.v1

Keywords: carbon cycle; industry; peer review; quality control; climate change; soil carbon; half-lives; allometric equations; large trees; ecological succession

Preprints.org (www.preprints.org) | NOT PEER-REVIEWED | Posted: 8 October 2024 doi:10.20944/preprints202410.0555.v1

Review

Carbon, Corporate Forestry and Conservation: Struggles for Science in Sustainability and Benchmarks

Christopher Dean

School of Geography, Planning and Spatial Sciences, University of Tasmania, Sandy Bay,
Tasmania 7005, Australia; christopher.dean@utas.edu.au, cdeanspace@gmail.com

Abstract: To help limit further climate change, science must be robust and the best available, because it provides insight and understanding, of our impacts and reparations. Although strong standards are part of science, human frailties and facets of society also contribute. Initially, science aided industrialisation of forests, and conservation-oriented science followed. Some early data and methods suit both purposes, but there are different needs, dimensions and scales to consider. Science struggles to blend these frontiers and societal features, such that peer review cannot be the only quality control. Examples are reviewed in depth under two main headings: sustainability and benchmarks. The main arena is a target for industry and conservation: mixed-forests in Tasmania, Australia. Common flaws leading to faulty conclusions were: mis-representing and not noticing earlier studies, inadequately accounting for spatial dimensions and time, and inappropriate benchmarks or error margins when making comparisons. Example results after refinements are: carbon stocks are not sustained within several cycles of first logging primary forest, succession to rainforest may not reduce carbon more than logging cycles, and the proclaimed highest-carbon site is a common peak. Recognizing the carbon legacy and industrial use together through time is necessary for consideration of alternatives and consequent climate-change mitigation.

Keywords: carbon cycle; industry; peer review; quality control; climate change; soil carbon; large trees; ecological succession; half-lives; allometric equations

---

1. Introduction

To prevent further climate change becoming entrenched, humanity must turn around its trend in carbon emissions by applying only 'robust science' or 'the best available science', i.e., by applying the scientific process correctly [1]. Anthropogenic climate change, which is the warming and extreme weather resulting from our carbon emissions, is forecast to become more frequent [2,3]. Science is important for preventing further climate change because greenhouse gas emissions can be more effectively reduced when they can be better quantified, when their origin can be more precisely attributed, and when the effectiveness of interventions can be forecast [4]. Also, information on what will be the most profitable interventions should be an output of science. Science is a human product and therefore cannot be viewed as somehow infallible, perfectly managed, and devoid of typical human flaws: and similarly for its deployment. The quality of scientific research, is established through a set of standards for experimentation and a sequence of protocols for publishing completed work. This makes science different to many other human activities but it merges with them, and they with it. This paper reviews some frontiers of science and shows some flaws where that science is blended with other characteristics of society: it shows, how to amend the science and the relevance of the new results. This work examines science that is related to carbon dynamics of forests. The first frontier considered is sustainability of forest use and its benchmarks (reference points), which is examined via soil carbon and biomass, and the second is extreme dimensions in forests and its benchmark, used for determining conservation. Some commonalities between these two are the forest

doi:10.20944/preprints202410.0555.v1

type, high public interest, and appropriate benchmarks for use in comparisons of carbon stocks when making decisions.

Carbon and its measurement benchmarks are important in understanding climate change because carbon circulates through forests and the atmosphere, but goes from the forests to the atmosphere sooner if there are extreme disturbances. The focus here is on science that is relevant to the logging of forests that do not have evidence of previous industrial-scale disturbance, generally called logging of 'primary' forests (hereafter, logging). It is a type of land conversion, as it converts primary forest to land with a repeated tree cropping process (logging cycles leading to perpetual, young secondary forest), though it is not conversion in the vernacular usage of the term, which is conversion to non-forest. There is a related term with which the reader may be familiar: 'oldgrowth' forest, which is forest dominated by older trees and naturally fallen coarse woody debris on the ground, typical of old primary forests of that type. Some primary forests are oldgrowth forests, and vice-versa. Other oldgrowth forests are those that have undergone considerable anthropogenic disturbance but have essentially recovered and now have the structural components typical of a later growth-stage. This follows the definitions of Barredo, et al. [5].

Scientists measure the stores and flow of carbon that were in primary forests and any carbon that is later sequestered when the main timber species is regenerated on-site. The carbon may be, for example, in the soil, in wood products, waterways, younger trees, and in waste material, and some will accumulate in the atmosphere as greenhouse gasses. Owing to the many processes and the range of locations involved, there are ample opportunities for both flaws and erudition in the applicable science, and therefore, both flaws and mastery in the style of carbon management adopted by society.

Any blemishes in science, may go unnoticed and are part of the information feedstock for the general public, media, politicians, advocacy organisations (such as environmental NGOs and timber industry representative bodies). The blemishes in climate-change-mitigation science can insinuate themselves into policy and on-ground actions, so it is best to solve them. In a review of the knowledge, perceptions and claims of people who many might expect to be at the forefront of environmental knowledge, namely forestry students and forestry workers, Pernica [6] found that the majority of their information on ecology and the consequences of environmental deterioration was obtained from mass media, especially television (84%). The groups mentioned above are mostly non-scientists, who may not be able to determine which are the most valid scientific facts, and they may not be interested in looking deeper. When comparing the capabilities and responsibilities of ecologists and journalists, Kirkpatrick [7] concludes that it is the responsibility of the media and conservation organisations to translate the results of ecological research into wider communication and action. An alternative view is that while scientists have to maintain their own integrity, they should contribute directly to public debates, and also educate the public about objectivity and scientific abuse [8]. Donald and Graves [9] describe how a fact checking organisation worked with the social media company Meta, to help ensure correct information regarding climate change is delivered to the public. (Some people currently rely on their 'feed'— a software-generated daily compilation of information in social media, in place of reading a newspaper). Although there are fake-news detection algorithms [10], the bottom line may be the veritableness of what scientists provide as feedstock. A major aim of the present work is to assess the use of science and data, to reveal some of the flaws, and to adopt a perspective that includes a larger scale, and thereby indicate amelioration and its impact.

What is typically considered by the general public as climate change science, is the impact of people's greenhouse gas emissions on the climate. Despite the above mention of flaws, compared with some other facets of climate change science, humans' net impact on the climate has been determined to a high degree of quality and accuracy, even though sometimes publicly disputed, i.e., 'climate change denial' [11–13]. Scientists have worked on several other aspects of anthropogenic climate change, e.g.,: (1) how our activities increase greenhouse gas emissions; (2) our future emissions; (3) reducing our emissions and their impact (called 'climate change mitigation'); (4) climate change's past and future impacts on the environment; and (5) ways in which the effects of anthropogenic climate change on the environment, induce even more emissions and further increase climate change (i.e., positive feedback) [14,15].

Any science outputs of numeric values have error margins, which is what some scientists call 'precision' or 'uncertainty'. Some people misinterpret 'uncertainty' as meaning that scientists have misgivings about the fundamentals [3]. As a preliminary consideration, it is important to note that error margins are from two main sources. The first source is the accuracy in reading a single value using an instrument. These are often listed as ± after a value and there are established formula for determining their influence after numerous mathematical operations on data [e.g., 16]. The second source is the imprecision in ascertaining a value from nature by multiple readings, such as from different locations. Each reading may be influenced differently by various characteristics (some unknown) that were not accounted for by using a control experiment. Many scientific reports on forests only account for the second type of error margin such as by citing standard deviations on values. But both types can be important when making comparisons against benchmarks, as considered in the last example in this review. Eighty years ago, new statistical tools, such as standard deviation and the regression equation, being incorporated into forest mensuration for forestry professionals, to the benefit of their profession [17]. Although computer software for the application of those tools continues to be made easier to use, aversion to overstating the implications of statistical measures has been voiced [18,19]. Validation of comparisons in this paper do not rely on those statistical methods for comparing against benchmarks. More-basic scientific protocols remain constant however, such as error analysis for individual measurements.

Omissions and inaccuracies in the development and use of science arise for several reasons (some of which are examined in this review), such as:

(1) experiments may not be designed to consider enough relevant factors owing to a difference in scale (physical and in other dimensions) at which different scientific disciplines normally work [20,21];
(2) there are knowledge hiatuses between and within scientific disciplines, such as between forest carbon modellers and product life-cycle-inventory modellers [22].
(3) the multidisciplinary wavefront of scientific knowledge, may not advance in a coordinated manner, instead its components progress disjointedly [23], possibly leaving a gap which misses some reality.
(4) some relevant science may be either: considered irrelevant or not currently a priority [e.g., off-site effects, 24]; forgotten over the course of time by scientists; or not picked up in literature searches.
(5) some relevant factors may not be programmed, or are incorrectly programmed, into modelling software, which consequently produces incorrect forecasts [25].
(6) although many fields of science involve no public quarrels, some scientists work in areas where controversy and advocacy occur [23], such as at the junction of nature conservation and forestry, and the controversy may lead to unintentional bias in experimental design or interpretation. Advocacy may arise from a personal preference or may be a workplace requirement. For example, the State forestry agency in Tasmania, Australia (Sustainable Timber Tasmania), when advertising for a carbon and ecosystem services analyst said that advocacy and public relations were a duty in the role:

> 'Support advocacy and internal and external knowledge translation on forest management, carbon and climate change.' [26]

Fortunately, there is a consensus for scientific publications that there should be no bias in the application of scientific standard protocols [8,27–31]. Sometimes however, it is left to the filter of peer review, to detect and correct for bias.

This paper looks at struggles with the correctness, relevance and timing of science, which needs to be robust and 'the best available science' [1], in other words: good science. Defining such science, is necessary. The first consideration must be the purpose of science, because that reveals why it needs to be correct. Fundamentally, a reliable understanding of what is around any being augments their usual subconscious decision making, because when needing to stay safe and healthy, it adds to their decision making and allows more judicious choices [32]. Correct information and explanations allow one to be more productive in a range of circumstances, including adapting to new locations and with new materials. Science allows explanations of phenomena that aren't immediately obvious, which in turn allows development of technology and materials for a vast array of purposes. Correct explanations are just as important today, away from primeval evolutionary demands and in a post-development world, for example in how to manage forests and various resources, in relation to anthropogenic climate change and materials for use [9,33].

One major purpose of science, is revealing logical, robust reasons for observations, based on more-fundamental robust insights. In this context, robustness of science means that it is resilient to different tests (rather than just being serendipitously correct once), and that it does not provide unobserved information when within the bounds of what it was devised to represent. In philosophical terms, science should provide 'grounded beliefs' [34]. The starting point is therefore with the most basic and simple of observable facts, such as mathematics, then applied mathematics in the form of physics, up through chemistry and then application of that in biology and medicine, all the way building robustly upon what is established below. For example, the cross-sectional area of a tree trunk is approximated by a mathematical formula using the constant π (from mathematics), and applied mathematics and physics are used in instruments to measure the other dimensions of a tree (or to fell it), allowing a calculation of its timber volume, habitat value, wind resistance, and carbon content. Applied sciences are also used to turn some of the tree into paper or to measure how much carbon is kept out of the atmosphere and for how long, if the tree remains intact. The stepwise understanding of science, where each small step is soundly based and verified through scientific protocols, allows people to understand more complex observations including those of our global impact. Understanding causes gives us the opportunity to try and adjust our future. This basic purpose for science, when applied to mitigation (i.e., limiting) of anthropogenic climate change, is summarised in an edict from Francis Bacon:

"Human knowledge and human power meet in one; for where the cause is not known the effect cannot be produced." [35]

The steps of science, which form progress in understanding, are written in scientific publications and computer programs that allow people elsewhere to build on what has been substantiated by others:

'His notebooks indicate that scientific imagining is a human characteristic that might appear in any age but that without the organized companionship of like-minded people, science does not advance', [[36] p105].

Scientific knowledge advances for some individuals in isolation (e.g., Leonardo Da Vinci about 500 years ago), but more generally in contemporary times the broader topic and its application advance faster when networked [37]. Publication allows: 'speculations, theories, hypotheses, observations, and indications of knowledge which still need to be examined and discussed scientifically' [38]. Scientific writing originated at least 3,500 years ago, with Babylonian astronomy written in cuneiform on stone tablets. In contrast, dissemination and quality control of contemporary science is very closely linked with the publication process in scientific journals. That process began about 360 years ago, with Sir Christopher Wren founding The Royal Society in London, which still publishes major scientific journals today, such as the Transactions of the Royal Society series. Quality control of science is also established at a governmental level and by research establishments in many countries. For example, the Australian government has established a code for responsible conduct of research and published several guides to it, focussing on, for example: peer review of research, managing and investigating breaches of the Australian code, and managing data and information [39]. The code describes institutional responsibilities and individuals' appropriate behaviour that can foster 'an honest, ethical and conscientious research culture' to provide a framework for developing high-quality research.

The most common check of research quality is by peer review of manuscripts submitted to journals or conferences for publication. Editors in publishing companies (e.g., Elsevier, MDPI, Springer, Sage, and Taylor and Francis) select usually two or three people to perform a peer review and they are provided with guidelines on how to conduct it. The reviewers should be impartial experts in the topic of the manuscript and they should check its content for aspects such as: appropriate referencing of the scientific literature, originality, relevance, soundness of methodology, informative diagrams, sufficient background for reproducibility by other scientists, and conclusions supported by the data [40,41]. The major outcome is advancement of science and global distribution of reliable information. The peer review process is subject to typical human frailties however, and therefore yields a minor, secondary outcome: less-reliable or less-applicable (limited scope) information, which may have wide negative impacts when combined with some other public influences [41–43]. This review seeks to improve some of that secondary outcome. After publication, rebuttals by other scientists may add some quality control in some circumstances and in extreme cases withdrawal of a published paper may occur. Part of this review has the framework of a rebuttal and part is a synthesis of detail.

Thus far, the explanation of science may appear more like the processes within a computer, and with alternatives being only of lower quality. But both are important, and synergy between the two is productive. There is something else that makes good science but is hard to measure: imagination. It is required for theorising, discovery, experimental design, and conceiving alternative explanations or common links, etc. Using, as an example, the discovery of the concept that planets and stars etc. are formed by the condensation of matter, the historian Webster says:

‘It will be seen that patient, plodding investigation does not form the whole of science, a place exists in it for the widest flights of the scientific imagination.’, [[44] p675].

This can be exemplified by:

‘Progress in science comes from not looking at Nature at face value, but understanding some voyage of discovery to reveal a different viewpoint. From this new vantage, the landscape takes on new aspects and dimensions, leading to fresh insights and new satisfactions. With this vision, the next step is perhaps even more fulfilling—predicting what can be seen from a higher standpoint.’, [[45], p1].

his review does not gauge imagination, but it does bring various work together using a perspective from a bigger scale, hopefully to ameliorate earlier flaws and advance science. In places the information presented in this paper is complex, but context is provided and linked to overarching themes. For the reader to get the most benefit from the present work, a quote from an early proponent of the scientific approach, that helped guide collating information for this paper, is relevant:

“Read not to contradict and confute, nor to believe and take for granted, nor to find talk and discussion, but to weigh and consider.” [[46], p158]

The main part of this review begins by describing the major forest type (Section 2) examined later in the examples of science improvement, and then its industrial usage (Section 3). This includes considering corporate strategy and some missed science. Then the dynamics of soil carbon are discussed, as they relate to forestry activities (Section 4). This establishes the background science necessary to investigate some aspects of science that have been missed in reports relevant to carbon dynamics, forestry and conservation. Examples are given where published interpretations have not sufficiently considered spatial and time dimensions. The sustainability of forestry operations is considered and previously missed aspects of science related to benchmarks are detailed (Section 5), to provide new perspectives. The second part of the review (Section 6) examines claims for extraordinary dimensions (including carbon stocks) of a particular forest stand, that was broached for purposes of conservation. Finally, positive ways forward are summarised.

2. Mixed-forests in Tasmania, Australia: A Target of Industry and then Conservation

The most-often used forest-type example in this work is the main target of the forest industry in Australia in recent decades: mixed-forest in Tasmania. Although a relatively remote country, Australia was the leading supplier of export woodchips for paper production in the Asia-Pacific until 2010, with the majority of hardwood woodchips sourced from its southernmost State of Tasmania, which was Australia’s major wood product by volume [47]. Tasmania has the highest proportion of forest cover amongst Australian States; its forests have on average a higher biomass than other States;

and its southern central, high-biomass forests have been an industry focus. For over a century Tasmania played a major role in the international wood products trade, initially for lumber and then for newsprint and then the hardwood pulp market for paper products [47–49]:

‘TASMANIA possesses, amongst other wonderful natural advantages, extensive Forests of valuable Hardwood, and for its area has the largest supply of timber available for export of all the Australian States. … of which the extent of country estimated to be under Forest is one-fourth, or about four million acres, consisting of some of the finest timber in the world… The principal Hardwood Forests lie within a radius of 60 miles of Hobart, the Capital, in Southern Tasmania, where the greater quantity of better timber is to be found, although there are many extensive beds of fine timber in the North-Western and Western Districts of the Island.’ [48]

Mixed-forest is a major part of an early pulpwood logging concession in the central southern Tasmania, in the Styx, Florentine and Tyenna Valleys [49–51]. Prior to logging, the majority of that mixed-forest was oldgrowth primary forest. The areas with *Eucalyptus regnans* as the eucalypt component were initially the main target [52].

In Australia, ‘tall open-forest’ has its tallest stratum >30 m high, creating a canopy cover of 30–70% [53]. In Tasmania, such forest, the ‘wet-eucalypt’ forest, is subdivided into ‘wet-sclerophyll’ and ‘mixed-forest’ [54,55]. The mixed-forest is a form of rainforest [56], with a closed rainforest understory below an open eucalypt canopy (Figure 1). One could avoid confusion with the more common type of mixed-forest globally—an angiosperm-gymnosperm mix, by using the term ‘clopen forest’ for the Tasmanian mixed-forest. ‘Clopen’ is a term used to describe something that is both open and closed, such as clopen sets, in set theory.

Common rainforest tree species in the mixed-forest are: *Nothofagus cunninghamii* f. Hook. (myrtle beech/ myrtle), *Atherosperma moschatum* Labill. (sassafras) and some *Phyllocladus aspleniifolius* Labill. (celery-top pine). This type of mixed-forest is common in Tasmania and predominates whenever the eucalypt species is *Eucalyptus regnans* (swamp gum/ mountaint ash). Other common eucalypt species in the mixed forest are *E. obliqua* (stringy-bark, messmate), *E. delegatentsis* (gum-top stringy bark/ alpine ash, previously *E. gigantea*), and further from the central ecotone: *E. globulus* (Tasmanian blue gum). Mixed-forest was previously common in Victoria, where it is now rare and termed ‘ecotone’ forest [57,58]. In Australia, mixed-forest also occurs in the States of New South Wales and Queensland, but in narrower patches and with the eucalypts emergent mostly over sub-tropical or tropical rainforest rather than temperate rainforest [55,59].

In Tasmania the largest mixed-forest ecotone is located between lower-water-balance eucalypt forest of the dry east coast woodlands and the higher-rainfall, lower-fire-frequency, west-coast temperate rainforest [55]. Here, in between, fire is naturally infrequent but just frequent enough to regenerate large areas of eucalypts once every few centuries. The edge of the ecotone is not sudden, but incorporates more-frequent fire on its border with the more fire-prone forests, and less fire on its border with the wetter forests and swamplands, with accompanying fragmentation and subtle differences in species distribution. This ecotone in Tasmania is much wider than alpine treeline ecotones, and is more similar in size and genesis to some of the ecotonal forests of the cerrado/Amazon rainforest transition, the northern Brazilian Amazon or the Rainy Zone of western China; but by fitting within Tasmania it is much smaller than the forest tundra ecotone of northern Quebec [60–63].

Another similarity of the mixed-forest ecotone with some other ecotonal forests globally, is that the species from the adjoining forest types have adapted synergistically to each other. For example, the rainforest trees can grow on the buttresses of the tall mature eucalypts, as hemi-epiphytes, using the stemflow [of rainfall] of the *Eucalyptus regnans*, and the eucalypt using the low flammability of the rainforest trees to allocate energy into vertical growth rather than into growing thick bark up the trunk or lignotubers for survival through fire (Figure 1.a) [[59,64–66,67] Figure S2,68].

In relation to carbon accounting, it is important to mention an aspect of the mixed-forests which makes them unique amongst eucalyptus forests in Australia, as it relates to fine-scale carbon concentration. Being of a maritime climate [69], sudden downpours in central Tasmania are often accompanied by strong winds of micro-fronts which means the rainfall is inclined rather than vertical. If one can imagine rainfall at a steep angle in Figure 1.a, it can be visualised how the tall vertical expanse of leaves of the dispersed, mature eucalypts intercepts a larger portion of rain before it reaches the closed rainforest canopy in between them, than if the rain fell vertically. The steep angle of the eucalypt branches, and relatively smooth bark (amongst eucalypt species) of the *E. regnans*, channels it towards the trunk, and subsequently a wavy wall of water flows over the buttress zone, which both the eucalypts and their hemi-epiphytes use. This stemflow is not observed on the rainforest trees between the eucalypts. Thus, carbon from biomass (through root decomposition and exudates) and dissolved organic carbon in the stemflow, concentrate carbon in the soil close to the eucalypts. When these mature eucalypts eventually die, the hemi-epiphytes and moss are notably absent from their buttresses. The different characteristics described above which combine to produce this stemflow process and its effects in Tasmanian mixed-forests, have been observed separately in other forests globally, though not reported in unison [70–76].

(**a**)

(**b**)


(**c**)

Figure 1. Canopy stratification of mixed-forest: open *Eucalyptus regnans* (~72 m high) above closed myrtle and sassafras canopy. Seen from edge of logging coupe SX009C, Styx Valley, Tasmania. (**a**) tallest eucalypt in foreground: DBH= 4.56 m with mature myrtle hemi-epiphyte joint up to 5.4 m height on the right-hand side and sassafras joint up to 4.7 m on left hand side; (**b**) drone view from above closed, lower canopy (courtesy of Darren Turner), tall tree on RHS is same *E. regnans* tree as in (a) on LHS. (**c**) at ground-level: myrtle, sassafras and ferns in-between mature *E. regnans*.

Another aspect of the buttress region of mature eucalypt trees in the mixed-forest warrants mentioning because it relates to allometric equations later in this paper. Tree diameter at 1.3 m from the ground on the high side of the tree (diameter at breast height, DBH) is often used as a variable in allometric equations for tree biomass, carbon or timber volume; and as a measure of growth in long-term inventory plots. (Heights other than 1.3 m may be used in other countries [77]). Spurs develop in the buttress region which link the main trunk to large lateral roots. Cross sections of the buttress region show much folding of the sapwood zone (Figure 2.b). In smaller trees, such as the Hawthorn example in Figure 2.a, this only occurs near ground level. There is a difference of scale which needs to be appreciated in developing science applying to this part of the eucalypt trees. Maximum DBHs observed for *E. delegatensis*, *E. globulus*, *E. obliqua* and *E. regnans* (in alphabetical order) are 5.73, 5.6, 6.53, and 10.78 m in DBH; with maximum heights of 90, 94, 91 and 132 m tall respectively, and wood volumes (not accounting for the possibility of internal hollows) of larger specimens in the high 300s to mid 400s m^3 [78–87]. The *E. obliqua* or *E. regnans* of DBH 2.92 m in Figure 1.b was therefore of modest size but it showed the deep folding of growth layers at least to 2 m up the trunk. The folding has been observed up to 15.4 m above ground (Figure 3.c) [67] and it could possibly occur anywhere in the buttress region. That region on *E. regnans* is evident as the only area aboveground of rough bark, and which can extend up to ~18 m [80], e.g., the trunk of a *E. regnans* tree of DBH 6 m may still have a buttress-spread component of 5% at 14 m height [[88] Equation 5]. Thus, this part of the mature eucalypts in mixed-forest is in between what is typically considered as trunk and roots in smaller trees.

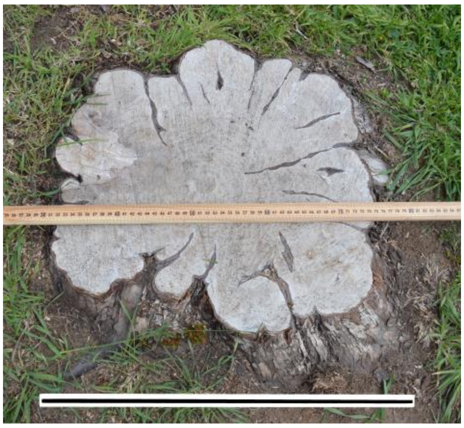
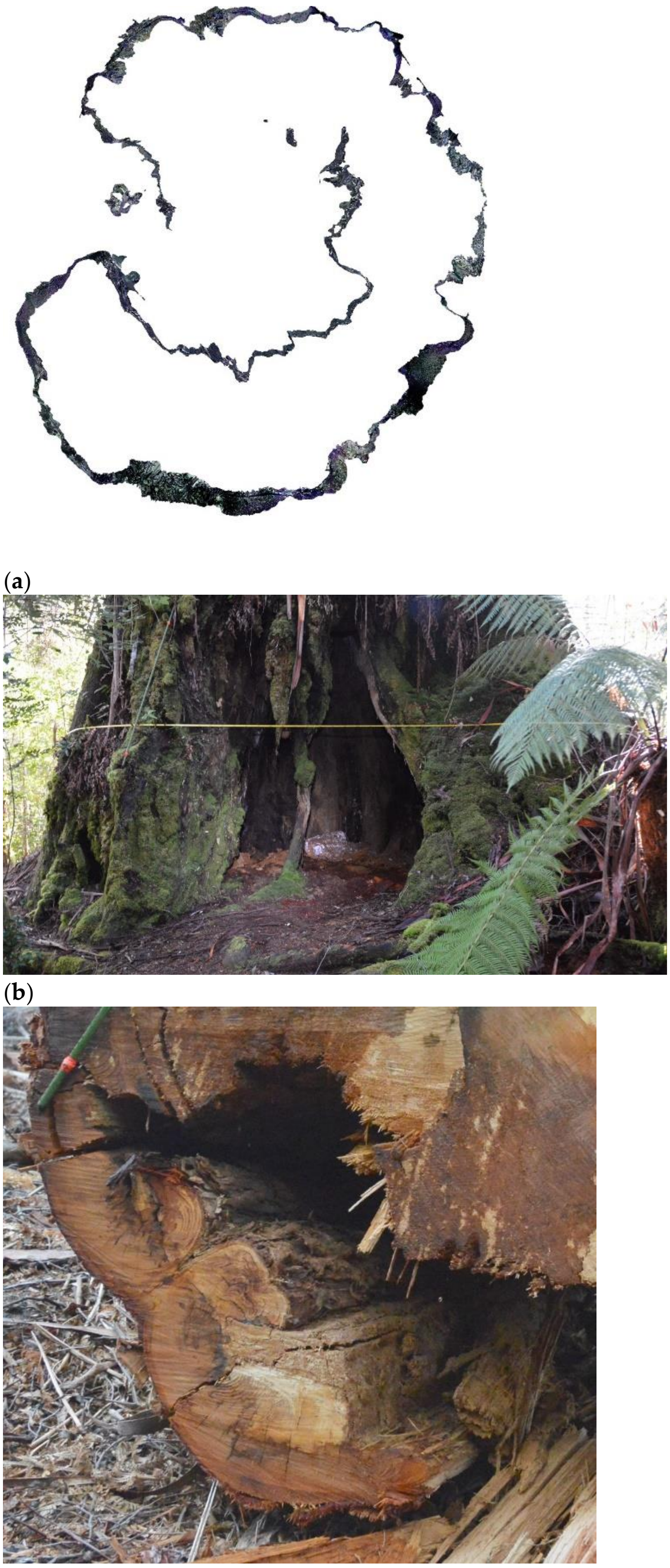


Figure 2. Structure of lower tree trunk extending to lateral roots. (**a**) Urban hawthorn (Crataegus hybrid) Hobart, cut at ground level, diameter= ~0.42 m, scale bar= 0.5 m (**b**) Sector of cross-section of partially hollowed *E. regnans* or *E. obliqua* in logging coupe (operational logging unit) after eucalypt regeneration burn, Styx Valley, DBH 2.92 m, cut near 1.3 m (courtesy of David Green), then pinned in place, dried and sanded. Gaps between lobes due to wood shrinkage and enclosed bark that fell away. Scale bar on left= 1 m. Note wood grain pattern in the lobes such that cambium and phloem are both near the centre of the trunk and on the outside. Insets show chainsawing (top) and fresh solid surface (bottom).

Due to wind stresses the spurs in the buttress region may become pronounced, with flutes in between them [89]. The amount of non-circularity at 1.3 m height is usually about 20%, but is more variable in middle-sized trees, for example *E. regnans* of DBH 4±1 m [[67] Figure 9]. Consequently the amount of carbon in trees of that size will have more uncertainty when derived from allometric equations based on DBH. An example of a more mature tree with a more circular buttress, is the Chapel tree in the Styx Valley, Tasmania (DBH= 6.03, height= 80.1 m, in 2012) (Figure 3.a-b). Such numerous complexities increase error margins in carbon accounting when mature trees are included. If there is a trunk hollow, then due to the deep folding of the cambium zone, growth may occur inwards, into the hollow space, as well as on the outside of the tree (Figure 3.). This has been observed up to at least 15.4m aboveground (the example mentioned above). The occurrence of internal growth adds to the error margins based on allometric equations. If there is no hollow in the buttress region, which appears to be rare in recent times [67], and the cross-section minimally non-circular, then growth may be even and fully represented by the DBH.



(**a**)

(**b**)

(**c**)

Figure 3. Tree growth with only partial contribution to increase in the external diameter, after hollow formation. (**a**) Virtual slice at 1.3 m above ground of 3D model of a *E. regnans* (the Chapel Tree, Styx Valley) DBH 6.03 m, showing minimal buttress spurs and internal growth—the small circle near the entrance to the hollow is a young stem stalk, formed while surrounding wood was decomposing. (**b**) photo of the same tree showing new growth and hollow. (**c**) A cross-section of a *E. regnans* tree at 15.4 m above ground, felled during logging, coupe SX019i, showing new internal growth, including bark, but inside trunk hollow. DBH 3.84 m. Exterior of buttress did not suggest internal hollow, but at 1.3 m aboveground the tree was ~50% hollow.

doi:10.20944/preprints202410.0555.v1

Evidence of large eucalyptus trees in mixed-forest without trunk hollows was secured after they were felled, e.g.,: the 'Helms tree' with a DBH of 6.37 m from the Tyenna Valley [51], age ~390 years; and the 'Maydena butt log', with a DBH of 4.08 m (~40 years after it was felled and assuming it was felled at 1.3 m above ground), on permanent exhibit, Tyenna Valley, Tasmania (Figure 5). Hollows gradually increase in size, such that the large trees don't usually die suddenly (except by severe fire or logging) but if they do, their carbon stock does not suddenly change, except in severe fire or if after felling and milling, a substantial portion of the waste material ('mill residue') is burnt. Although the incorporation of gradual hollow development (with tree decomposition) into carbon accounting is only about two decades old, the phenomenon is global, and was mentioned long ago, e.g.,:

'The monarch oak, the patriarch of trees,
Shoots rising up and spreads by slow degrees;
Three centuries he grows, and three he stays.
Supreme in state, and in three more decays.', Dryden [90] adapted from Chaucer's 14th century 'The Knight's Tale' in 'The Canterbury Tales'.

and: '…its dimensions at the time of writing as thirty feet in circumference on the outside and twenty feet round the hollow within…', [[91] p60].

(**a**) (**b**)

Figure 5. Similar-sized trees with varying hollow size, relating to different carbon stocks. **(a)** A buttress region ('butt') log of DBH ~4.08 m, on permanent display in the Tyenna Valley, showing a nearly 100% solid cross section. 1 m quadrat for qualitative scale only. Photographed in 2002, logged c1965, bark has decomposed and some timber has shrunk, revealing folding of cambium layers in the buttress region. **(b)** 3D model of a logged and burnt *E. regnans* of DBH 4.18 m with ~25% of the carbon lost from the buttress region, in the Styx Valley, developed in earlier work [67,92]. The hollow had no entrance point in the buttress region (so the hollow was not indicated externally). Such 3D models allow: more accurate measurements (e.g., by burl removal); and with sufficient samples, allowed new data types and formation of new allometric equations for C accounting.

A unique attribute of the mature stage of the forests is a concentration of soil organic carbon (SOC) that is not usually measured. It is under large tree trunks or tree buttresses, where it is principally derived from decomposing roots and is usually much more concentrated and deeper in the profile than in between trees [92]. Its expansion with tree growth is shown in Figure 6. The thick layer of reddish-brown lignomor (decomposed, structure-less wood Green, et al. [93]), which is nearly 50% carbon, is derived from decomposing coarse roots (Figure 7),

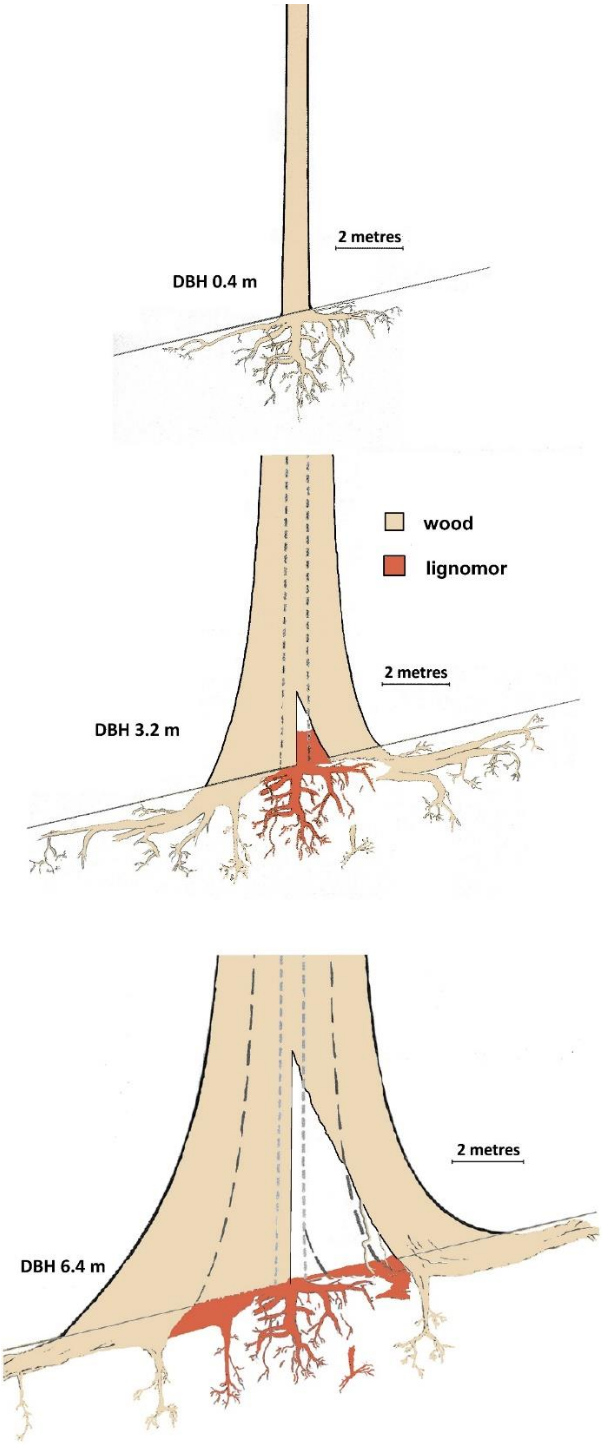


Figure 6. Tree growth of a *E. regnans* and associated concentrated soil organic carbon. Drawing based on Figure 2.b in Dean, et al. [92], but with separated growth stages and extra detail. Aboveground shape drawn from taper formulae adjusted for ground slope [88], roots drawn from observations and literature [67,80]. For the older tree, two scenarios are portrayed: on the left the trunk remains undecomposed and on the right-hand-side the tree has hollowed out.

(a) (b)

(c) (d)

(e) (f)

Figure 7. Examples of stages shown in Figure 6. (**a**) 3D model of an ~0.6 m DBH tree trunk and parts of central tap root and lateral roots (no decomposition), (**b**) lower part of a 3.11m DBH tree filled with lignomor from trunk decomposition (handle of the 2 m long soil auger is visible), (**c**) the edge of the bowl of ligniform from a burnt and fallen tree, showing contrast with surrounding soil, (**d**) pushed over, burnt and wind-rowed medium-sized trunk in a plantation showing large hollow, (**e**) pushed over and burnt medium-sized trunk with small hollow, coupe SX004c (photo: Melinda Lambourne), and (**f**) measuring 0.9 m deep pit in bowl of lignomor under live, hollow *E. regnans*, l(DBH 6.08 m).

In order to refer to the different tree sizes and stand structures in Tasmanian forests, when considering tallies of timber, carbon or stand-type distribution, it is sometimes useful to use a categorisation system developed by the timber industry. It was used heavily in the Tasmanian State-wide mapping of forests by aerial photography interpretation, and the categories are consequently called API types [52,94,95]. In mature stands, eucalypt tree heights are categorised by the acronyms E*n*, corresponding to dominant height classes: E1: > 55 m, E2: 41 to 55 m, E3: 27 to 41 m, E4: 15 to 27 m, E5: < 15 m [52,94]. Eucalypt canopy closures are categorised by crown cover as follows: a=90–100%, b=70–90%, c=50–70%, and d=10–50% [94]. Some of the common rainforest or rainforest understorey types are M+, M- and S, referring to rainforest with myrtles usually > 25 m height, 8–25 m height and scrub, respectively. Thus, for example, a moderately tall mixed-forest stand with a strong rainforest understorey could be labelled as E2b.M+. API types also refer to many other characteristics, such as the year of 'regeneration' for logging operational units with a known date for reseeding of the eucalypt species, or if an area has been strongly affected by wildfire. This allows a range of investigations using GIS (geographic information systems).

3. Industry Activity, Science and Conservation in Mixed-Forest

The Materials and Methods should be described with sufficient details to allow others to replicate and build on the published results. Please note that the publication of your manuscript implicates that you must make all materials, data, computer code, and protocols associated with the publication available to readers. Please disclose at the submission stage any restrictions on the availability of materials or information. New methods and protocols should be described in detail while well-established methods can be briefly described and appropriately cited.

3.1. Allocation of the Southern Central Mixed-Forest Area to Industry

To gain public order and curtail slash-and-burn activity in Tasmania, the government initially set aside 22,000 ha of forest for industrial forestry, then a further 38,000 ha in the early 20th century, then up to 594,000 ha by 1938, of which 46% was pulpwood concessions for private companies to produce newsprint, to save buying it from Canada [48,50]. Only 12% of the large area of forest was allocated to pulpwood concessions (~81,000 ha) was allocated to national parks and scenic reserves. Half of that reserved area was 'in isolated positions where the land is of poor quality' (Counsel, E. A. : Annual Report of Surveyor General and Secretary for Lands, 1906. [50]). Part of the large concession allocation was the Florentine and Styx Valleys' 162,000 ha pulpwood concession. Prior to logging it was mostly even-aged *E. regnan*s—dominated mixed-forest [55,96,97]. Some of that concession included land that was previously in the nearby Mt Field national park [98]. These generally 'mature to over-mature' stands (in forestry terminology) were to provide a century-long new industry sourcing pulpwood, converting them from primary forest into secondary forest with further logging cycles [52,99,100]. Only 0.22% (365 ha) of that area was initially reserved from logging [101]. That small reservation was to conserve 11 particular botanical and cultural assets: 'To provide for values other than wood production such as recreation, fauna, flora, stream protection and historical site.' [Appendix 10 in [52]]. After nearly a century of industrial usage, new species and substances in the vestiges of that concession area are still being discovered [102–105]. Linking logged and unlogged areas of forest, to enable animal recolonisation ,was not undertaken in that concession area [106].

3.2. Historically Missed Science, Filling the Void, and Arising Controversy

One of the earliest portrayals of Australia featured its fauna by 19th century scientists and artists (e.g., Eugene von Gerard, John Gould, Gustav Muchel and Georges Waterhouse). Large areas of native vegetation were drastically altered by industry before some of that fauna could be adequately studied in regions less often accessed, such as in tree canopies, below-ground, and in relation to a forest's 3D structure [e.g., [107,108-113]].

Public controversy about conversion of Australian mature, primary forests to even-aged, eucalypt logging cycles, mounted in the 1970s, simultaneously with the upsurge in logging for woodchip exports [114]. Dargavel [115] states that one part of that controversy was because foresters understood the forest well scientifically, for timber management purposes, but he separates out ecology as a branch of science for which "the foresters' knowledge was limited and partial". At the time, foresters were aware of a deficiency in the knowledge of ecology of eucalypt forests related to timber production, specifically the relationships between species distribution, soil and microclimatic factors, and how species react to different environmental factors and pathogens [116]. Scientists have been experimenting in forests for over 200 years, studying such characteristics as forest soil moisture and the effects of forests on local climate [117]. But when public pressure forced some of their environmental concerns into logging plans it "caught Australia's forest services by surprise", as they were 'unprepared and unskilled for planning in the much more complex and controversial situations that arose' [118]. More research on some aspects of ecology was precipitated by a combination of stakeholder factors: the forest owners (the public) became more interested in forest management following the peak extraction for woodchips in Australia, the forest industries were approaching the vestiges of primary forests in their original logging concessions, and privatisation of plantations meant a change in research funding.

In Tasmania, in the mid-twentieth century peer reviewed scientific publications were mainly on silviculture and logging methods but included some aspects of local ecology [55,119–121]. Ecology has only in the last four decades begun to be part of forest management [122]. For all intents and purposes, the focus meanwhile had been on productivity goals [123]. For example, Keith, et al. [124], with reference to the availability of allometric equations, note there is 'a bias towards production forest species and small trees', and the lack of data for large trees means assessment of mature forests is hampered. Globally, for the late-twentieth and early twenty-first centuries, a bibliographic analysis of science publications revealed several trends in forest research. From 1956–2019 there was a trend of less research on sustained-yield to an increase in forest ecology [125]; and in the second half of the period 1970–2005 compared with the first half, there were an increased number of publications on ecology of tropical rainforests, landscape habitat fragmentation, elevated CO_2, and climate change [23]. Currently, a wide range of research topics related to forestry is primarily funded by the Australian federal government's CSIRO, Australian Research Council and Rural Industries Research & Development Corporation; some State government departments; and more independently, at universities. The major focus is still on the traditional topics of silviculture and wood products but also the renewed one of ecology, plus the newer topics of the carbon cycle and the life-cycle of wood-products. The legacy of the historical bias in application of science to forests is that current-day society struggles to integrate ecological concepts into government policies on industrial forestry: even amidst anthropogenic climate change the two appear immiscible.

The partitioning of forest components in science is not only from wood production. Biodiversity of forests has traditionally been studied in terms of the numbers of individual species or types of ecosystems, and their spatial distribution over map projections, i.e., within two dimensions (2D) [e.g., [126,127,128]]. Forest ecosystems are still often catalogued or described according to the most common tall plant species present [e.g., [129,130]]. This could be because people undertaking science are generally very much non-arboreal, diurnal, surface-based and traversing the vertical axis (above or below ground) is not a region to which they are intricately accustomed [e.g., [107,131,132]]. They rely on specially designed tools or local indigenous people, to measure outside of their 2D oecumene [e.g., [133,134]]. For example, regarding public demands on State-organised forestry in Australia, Dargavel [115] states:

> 'What the foresters lacked the most was detailed knowledge about the arboreal wildlife, and even their casual knowledge was limited because much of the wildlife is nocturnal, when most foresters have gone home.'

This was reiterated in 2022–23, when it was noted that the State forest agency for Victoria stated it was too dangerous to survey for nocturnal fauna at nighttime, though it was done by academic scientists [[135], p38-39].

Consequently, the 3D aspect of forest biodiversity has historically come in as a poor second in scientific reports. Biodiversity of forest canopies was recorded in the late 20th century but the technology of that era didn't allow sufficient expanse in 2D, for the data collection to encompass 2D heterogeneity [132]. This century though, scientific endeavour is burgeoning in the vertical dimension, at least aboveground, and with significant findings: for example, canopy insect diversity from DNA in rainfall or from climbing using modern apparatus [136-138.

[136–138], canopy structure of mature forests using long-boom tower cranes or LiDAR [139,140], taking advantage of logging to compress 3D habitat information into 2D [141], and contrasts in animal diversity between measurements from the ground and from the canopy— sometimes with previously unrecognised differences between before and after logging

[142–148]. Most of the science reviewed in the first part of this paper, is on the negative vertical axis, namely soil carbon effects, because recent work there has resulted in controversy.

3.3. Public vs. Industrial Interest, in Determining Scientific Endeavours

In Australia, most State governments manage timber extraction from the public forests as a 'government trading enterprise', or 'government business enterprise', or 'state business corporation', each based on the model of corporations. These enterprises are the cores of what were previously the State forestry departments. The administration of the more-peripheral parts varies between the States (e.g., either privatised, contracted out or handled by a different government department). Each corporation has two main areas of activity that may overlap spatially: use of existing forests (native-forest logging) and planting forests before using them (plantation and native-regeneration forestry). South Australia, the Northern Territory and the Australian Capital Territory don't have government commercial native forest logging enterprises, just plantations.

Globally, government management of forests has always been intertwined with the forest industries, with access to primary forests remaining controversial [149,150]. When public objections rose steeply in Australia there was a desire in the forest industry to avoid conflicts similar to those experienced in the U.S.A., which had reduced the power of the industry. In order to maintain forestry's control over public forests in Australia, Florence [116] suggested that localised environmental considerations, including ecology and maintenance or enhancement of non-timber values, be factored into logging plans, and that the essence of the precautionary principle be followed. But this recommendation was not heeded to any notable degree, and not such that it mollified biologists or conservationists. Instead, another avenue expanded: Australian forest industries have directed a greater amount of funding away from research and into marketing, including public relations (PR) through representative bodies such as Forest and Wood Products Australia (FWPA) [151]. Here, the Bernays [152] definition of PR is followed: as 'the attempt, by information, persuasion, and adjustment, to engineer public support for an activity, cause, movement or institution.'

More generally, the divisive ambience that pervades society, with regards to existing large industries and carbon emissions, climate change, industrial land use and conservation, is part of humanity's larger struggle of industrialisation vs. environment. The struggle between forestry causes and environmental causes includes: industries' and conservationists' public relations, corporate defences against environmentalists, government alignments, litigation, forestry blockades, animal rights activism, industrial crime and corruption, environmental damage and human fatalities [e.g., [7,153,154-168]]. For any large industry that interacts with the environment, a portion of humanity may see that industry as an aggressor against the environment and therefore react, but that reaction may surprise the industry who in turn see the public's reaction as a force against which they must act, and that may entail public relations exercises [169]. Both sides may refute each other's PR [170,171]. Within this paper some scientific reports are described which are aligned with PR campaigns, and detail is examined in order to improve the science and help to resolve outstanding issues.

The process of scientific publication in journals, can be used both by people preferring a particular industrial development and by people preferring conservation. But conservation-oriented scientific studies more often seem to be reactionary, purely in response to industry's impacts, rather than science driven for the need to know about nature before using it. For example, research interest in ecology in Victoria (a State of Australia) is strongly correlated to the amount of wood products a forest has yielded, more-so than to the financial returns from logging. Bennett and Adams [172] suggested this was due to logging intensity— its associated environmental impacts and opportunities for study, or that more people settled near the more-productive areas [and thus were closer to such logging]. One technique is to use science to produce benchmarks for 'ecological importance' and species endangerment. These combine with public preference to determine policy [173]. One such study is reviewed below (Section 6).

Corporate responsibility (CR) is multifaceted, including, inter alia, responsibility to shareholders, financial sustainability, and corporate social responsibility (CSR). CR may require a corporation to adjust the market place (called 'market management' or 'issues management' [174]) through a range of strategies, not only in trade competition but even such processes as eliminating standards bodies and alternative products, and modification of government regulation [175,176]. Within that process, some corporations may subvert some of the parts of science that support conservation but that will cause adverse financial performance (e.g., by sidelining, omitting or discrediting its relevance or veracity): and more generally, a range of methods may be used to align science to satisfy the needs of a large organisation [114,166,177].

Often science only occurs after corporations have begun to implement their proposals. This science sometimes provides clarification, may assess alternatives, or may investigate side effects. Just as a conservation organisation may not feel obliged to consider the financial implications of nature conservation, so too, a corporation may not devise methods of resource use that entail comprehensive environmental knowledge and long-term protection of the environment, unless demanded by their stakeholders [e.g., [176]]. Thus, science may simply not be considered prior to, for example, industrial activities that produce carbon emissions. In this context, this review looks at some scientific publications that may inadvertently function as part of industry's more-general, issues management [174].

Goals of corporate responsibility have moved from solely profit to include social and environmental concerns, towards CSR and the development of a 'social license' for corporate activity. From the late 1990s major international organisations such as the European Commission and the United Nations urged large corporations to include CSR in their core strategies, in consultation with stakeholders [178,179]. Centuries ago the various benefits of forests made the general public a stakeholder in forestry and forests [[180] p 10, [181] p xi]. Sustainability is currently one of the expectations critical to social license, and part of this review examines how that expectation is met.

The World Bank suggests that sustainability should be included in CSR but doesn't specify if that is environmental sustainability, and some companies interpret sustainability as meaning business viability [182,183]. Some industries rely on the older targets and methods of corporate responsibility, against a background of legal requirements which have not shifted markedly to reflect CSR. For example, when solutions for the global syndemic (the combination of the obesity, undernutrition and climate change pandemics) were reviewed, a change in management of CSR was recommended:

'Corporate social responsibility efforts, which are too often marketing exercises, need to evolve into a stronger accountability model, in which targets and performance criteria are independently specified, monitored, and publicly shared.' [184]

In corporate forestry, one PR mechanism is 'generic marketing', where plantation forestry and logging of primary forests are not differentiated in marketing material. It is designed to counterbalance 'anti-logging activism and competition for market share' [e.g., [185–187]]. This avoids recognising the legacy carbon from the primary forest by considering the secondary forest as long-term plantation land, but the legacy carbon is considered in this review.

A more-general industrial public relations tool is to embellish knowledge gaps in controversial areas, such as exaggerating the significance of scientific uncertainty, or even creating it [188]. When applied to climate change it increases the momentum of avoidance of climate change mitigation by humanity [12,189,190]. Knowledge gaps need to be handled carefully in scientific endeavours, else the science created may be lopsided, which may lead to lopsided policy. For example, change in soil organic carbon (ΔSOC) with logging hasn't been studied to the same precision as change in biomass, and thus ignoring it in financial accounting is appropriate, due to insufficient precision for investment. However, not accounting for it in scientific forecasts is an omission, as both the value and the error margin could be mentioned. In this review, the ΔSOC accompanying conversion of primary forest to secondary forest with logging cycles is considered, and contrasted against its omission.

3.4. Conservation in the Mixed-Forests

Although conserving forests from industrial activity is often dressed as a recent idea, it is centuries or millennia old. Ancient civilisations such as those of Greece, Rome, India, China and Maya also exercised forms of forest conservation, reforestation and plantations, though for more local benefit and society-oriented reasons than are the national parks of today [191–194]. These conservation initiatives were concomitant with neighbouring major forest attrition, deforestation and related land degradation issues, and consequent adjustments to societies [191,195–197]. Forest conservation was suggested in the 13th and 14th centuries in England, to keep forests intact for some people's needs, in the face of flora and fauna attrition by local residents [198]. More forest conservation was later proposed in that same region, but for forest amenity's sake, and again to counteract attrition by local residents [198]. Sourcing of forest products by local people was mainly for local usage, but also for selling further afield. In the late eighteenth and early nineteenth century, forest conservation, less damaging use of forests and even reforestation, was proposed in Brazil, due to: land becoming far less productive after slash and burn agriculture, possibly medicinally important plant species being lost, alternative uses for the timber, and important fauna species declining [199].

As early as 1925 there was public disquiet about the misleading information in forest industry PR [200], such as against the notion that forests cannot regrow and maintain themselves without human intervention by logging:

'A forest may and often does maintain itself unimpaired century after century. If this were not so, why was a large part of this country covered with magnificent forest with trees several to many centuries old when the first settlers came? Can we doubt that it would still be so but for human interference?...No more nonsense can be disseminated than the idea that if we do not hurry up and cut the rest of our dwindling pernicious supply of timber the forests are going to fall down and rot like a crop of weeds. … We shall never get any real conservation in this country until people wake up to a realization of how the tentacles of commercial interests have penetrated, not only the branches of our government, but also most of the conservation organizations.' [200]

Similar reasoning is published by advocates of corporate forestry today, when they imply that mixed-forest will turn into lower-carbon-stock rainforest and emit carbon if not logged [e.g., [201]]. We examine the carbon balance for that process in this review.

All remnant primary forest on public land in Australia, is currently sought after by both the forest industries and those interested in conservation. National parks and other reserved forest land such as world heritage areas, have become newly contested land [202]. The goal is represented through industry mantras such as 'a shared vision' [203,204]. A media release by Forestry Australia (previously Institute of Foresters Australia) included:

'The paper Reshaping forest management in Australia to provide nature-based solutions to global challenges, by Dr William Jackson and other members of the IFA/AFG, says it is time to move beyond the era of conflict and develop more holistic approaches that encompass all forest values, such as water, biodiversity, tourism and forest products, across the landscape.' [205]

In contrast, the article they cite states:

'To be clear, active management is not a call for commercial timber harvesting in national parks and conservation reserves.' [206]

Overall, conservation of primary forests did not prevail in Australia because "The forest industry had the financial advantage in being able to have offices and professional lobbyists at a level the environmental movement could never match. It presented itself as the defender of employment, in ways that were as specious as the appealing pictures and that were also persuasive." [115]. Demand grew to conserve some more forest from timber felling following intensification of logging in Australia for export pulpwood in the 1970s and 1980s, which peaked in the 90s and included further forays into national parks [207,208]. Consequently, in the 1980s some forest neighbouring pulpwood concessions in Tasmania and some national park land were assigned World Heritage status [209]. The most notable of these was an area of tall, mature mixed-forest (with 41–80 m tall eucalypts) intermingled with myrtle-dominated rainforest, bordering the northwest side of the Florentine Valley (Figure 8). Without such reservation, designed by Kirkpatrick, it would have met the same fate as nearly all the mixed-forest in that valley— clearfell logging conversion to young, production forest cycles (Figure 8). This century, the public's awareness of dwindling primary forest acreage in the concessions caused further demand for World Heritage assignments, even including land that had earlier been selectively logged [210]. The extension was achieved and some its components are described below.

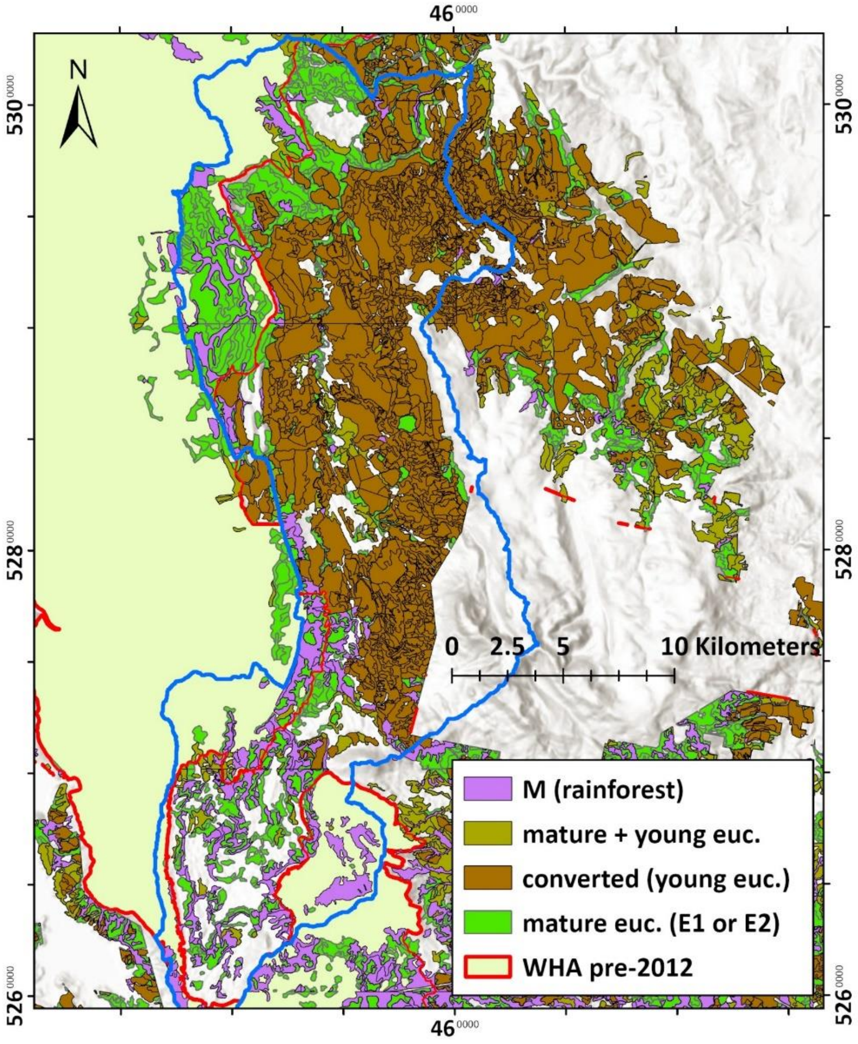


Figure 8. Logging extent (brown) in the Florentine Valley catchment (blue outline) as of 2012. Note the area of mature, tall mixed-forest (bright green) in the top left of the Florentine Valley that was reserved from logging and allocated into the World Heritage Area (yellow with red outline). Forest type mapping was from aerial photography interpretation by the State forestry agency, Forestry Tasmania. Projection: GDA 1994, MGA, Zone 55.

4. Dynamics of Soil Carbon Relevant to Measuring Forestry Effects

4.1. Time Periods for Measurement of Change in SOC

In the last thirty years, there has been an approximately 12-fold increase in interest soil related to climate change, followed by 10-fold for soil-and-biodiversity, and then to a much lesser extent,

~5.5-fold increase for soil organic carbon specifically; soil-and-logging, soil-and-ecosystems and soil-and-silviculture [211].

The change in soil organic carbon (ΔSOC) accompanying industrial forestry is gradually being measured more comprehensively. The time required for change is a complicating factor that must be considered when designing experiments aiming to detect ΔSOC. It is necessary to consider the spatiotemporal dynamics of SOC so that the different results from various experiments on SOC can be interpreted. Most experiments so far have been conducted over time periods such as 1 to 50 years after logging, which is when decomposition of the fresh logging debris creates a peak in SOC. These periods have been considered to be short, medium or even long-term [212–214].

The upper mineral soil is most responsive to surface changes. For example, it shows a short-term peak in SOC after a brief, disturbance-driven decline, after both logging and fire [[215], Figure 4, [216-218]]. Therefore, experiments examining only surface soil are more likely to indicate change, if done at timescales typical of many experiments (the 1-to-50 years mentioned above).

Change in SOC, over most of the soil profile, in response to long-term change in forest biomass (not just a single disturbance such as a wildfire), has a time scale longer than that of a human generation and therefore longer than most experiments— typically several centuries to millennia [215,219]. The fact that SOC changes so much slower than does biomass, has been offered as an explanation, along with natural spatiotemporal variability, as to why many experiments have not found any ΔSOC due to forestry activities [220]. The reasons for the relatively slow change in SOC are three-fold:

(a) coarse tree roots take time to grow and then decompose to contribute to soil carbon stocks, and it is the new tree stock for which one requires a corresponding value of SOC;
(b) the half-life of the slowly decomposing part of SOC, which forms the majority of total SOC, is in the order of several centuries [e.g., [221]], which is much longer than that of biomass and therefore its change is also slower; and
(c) the SOC in the lower soil-profile can be a substantial part of the total and therefore must be included in measurements [222–224], but SOC further down usually has an even longer half-life than the SOC higher up [223,225], except possibly when it is chemically disturbed [226].

Due to differences in water availability and temperature, decomposition of molecules that reside in the soil and were derived from trees, is likely to be faster in tropical climates and slower in more-polar climates. The timescales involved over the whole soil profile are exhibited in carbon isotope measurements of carbon age at different depths [223,227]. For example, in a warm, moist tropical environment for 0.7 to 1 m depths:

‘In deep layers, the age distribution reveals the small but non-negligible direct incorporation of photosynthetically fixed carbon through deep roots or soluble carbon (for the youngest carbon), and the predominance of carbon that is older than 1,000 years.’ [223]

In this environment, mid-level (0.2 to 0.7 m) has SOC of mostly intermediate ages of 100 to 1,000 years, which leaves the surface soil as the most responsive to change [223,227]. In a temperate forest, mean SOC ages were <60, 100 and 550 years for the Ah horizon (uppermost mineral soil), 0–0.1 m and 0.1–0.3 m depths, respectively [227]. In temperate forests, SOC turnover times of ~200 years have been observed to be long enough to prevent detection of management effects in experiments spanning only several decades [225]. This confirms the notion that experiments to detect change even as high up as 0.1–0.3 m, need to sample over several centuries to measure ΔSOC.

For the purpose of modelling the carbon cycle for carbon accounting, the change in SOC due to the short-term activity of logging, including the intense regeneration burn, can be considered separately from the longer-term effect of reduced biomass after logging. There are multiple effects around the time of logging, such as additions to SOC from the extra charcoal (if there is burning) and decomposing timber [228]. The timber, bark and leaves left on site after logging is collectively called ‘forest residue’ or ‘logging residue’ in forest industry terminology. For calculation purposes, the loss from soil turbation during logging, post-logging grading for plantations, and fire due to clearfell, burn and sow (CBS) type logging, must also be separated, from the additions from logging residue.

doi:10.20944/preprints202410.0555.v1

This separation requires care in experimental design because the factors are best measured separately— the contributing factors must be discerned in the data collected.

The SOC in the soil profile can be divided into categories called pools, where a pool is a group of organic compounds whose stability is delineated by having a similar half-life, and some of the pools may contribute their carbon to each other or directly to carbon in the atmosphere [e.g., [229]]. In general, for the soil profile, the half-time for sequestration or loss of SOC is mathematically equal to the half-life of the slowest changing SOC pool that forms a substantial part of the SOC stock [[230,231], p43]. This means that the time it takes for SOC to reach a long-term steady-state equilibrium, that corresponds to the prevalent, long-term-average biomass cover is proportional to the half-life of the slow SOC pool.

A simple calculation can show how long it takes to achieve some sort of equilibrium in SOC, depending on that slow SOC pool: assuming first order kinetics, such as simple exponential decay of soil carbon, the time to 95% of the long-term steady state, is ln(0.05)/(ln(0.5) (= 4.3) times the half-life of the slow SOC pool. This is the mathematical reason why the initialisation period ('spin-up' time) of soil carbon models is usually from around 1,000 to 3,000 years, and longer periods can provide more certainty [232–235].

This time scale, of around 1,000 years or more, is not unique to assessing ΔSOC with land cover change: it appears elsewhere in the science of anthropogenic climate change. It is the period after which, if we promptly and completely stop adding our greenhouse gas emissions to the atmosphere, anthropogenic warming and atmospheric CO_2 will notably begin to dissipate (though in this case it is related mostly to the atmosphere-ocean C exchange) [236,237]. Thus, there are two reasons to be aware of that time scale when considering our impacts, both past and future.

This temporal characteristic of the slow SOC pool also means that multi-century and millennial time spans are necessary for ΔSOC experiments that aim to determine the effect on the whole soil profile due to long-term change in the forest biomass, such as conversion to logging cycles to produce wood products or bioenergy, reforestation, afforestation, or long-term change in fire regime. Also due to such time scales, long-term averages must be used when calculating the effects using modelling/simulation/forecasting studies. It is the new, long-term-average SOC, either modelled, or observed through space-for-time substitution studies, that must be used to represent the real effects of management. To shortcut that long duration, one could claim that no change will occur lower down in the soil profile where half-lives are longer, but that would be denying that the SOC originated from trees.

From these time scales, rather than from the viewpoint of an individual person's typical experiment, the first 50 years after logging, mentioned at the beginning of this section, is short-term.

4.3. Linking ΔSOC to Change in Biomass, and Measuring It

Broadscale datasets where SOC can be compared against forest biomass, and where other factors are normalised, show that SOC is positively correlated with forest biomass per unit area [238,239]. Thus, where time-averaged, long-term biomass is reduced by intensive logging then it would be expected that SOC must also decrease. A long-term decline in SOC with subsequent harvest cycles over several centuries is because there's less wood available (compared with the primary forest), both from fallen branches and trunks and from coarse roots of those trunks, to decompose into soil carbon [240–243]. That is the biological perspective, and it is mirrored in computer models, where mass is stored in matrices for live biomass carbon, dead biomass carbon, SOC pools and the atmosphere, with periodic movement of carbon between them according to half-lives of the pools.

Within the numerous findings from experiments on intensive logging and SOC that were reviewed in Dean, et al. [239], any seeming inconsistency from the positive correlation can be explained. For example, upon examining purportedly paired sites Leuschner, et al. [244] found a long-term drop in SOC over a few centuries of logging but did not find a correlation between SOC and biomass. However, their 'low' biomass values were not time averaged over the logging cycle but were near the end of the cycle and on average, statistically close to the primary forest values. Assuming a Chapman-Richards type growth function, the time average biomass for the logging cycle is more likely to be ~2/3 of the biomass at the time of logging. That lower value would have been more likely to reveal a positive correlation between biomass and SOC. Because of the laggardness of ΔSOC, average values over longer time periods are better indicators than values at a point in time. They are also more relevant to the effects of land use on climate change.

There are of course natural, long-term changes in SOC [245] that create a background against which one would observe the trend due to intensive logging. Where some of the forest biomass due to logging is moved off site, is not burnt as mill waste, and becomes a wood-product, then eventually decomposes in landfill, it can contribute to off-site SOC. For the mathematics of carbon accounting, this off-site SOC should be added to the forest's on-site SOC.

The question arises, when will the ΔSOC due to management, become measurable by experimentation? Several cycles of reduced growth and decomposition under conditions of reduced biomass [due to logging of primary then secondary forest], are needed to change SOC sufficiently for it to show up empirically [239]. One can merge the outputs from different reports shown in Figure 2 of Dean, et al. [239] based on cycle number of the secondary-forest logging [models from: [47,241,242,246-248]]. To normalise the different data sets along the time axis, the common logging cycle duration was chosen as 80 years. The merged data provide a rough estimate of the fate of SOC for a range of species, range of soil depths and range of cycle lengths as a function of logging cycle number. This average trend can be formulated in equation form (Eq1) using Eureqa [249,250] (portrayed in Figure 9):

$$SOC\% = 1{,}965/(14.92 + lc) + 8.944\text{sqrt}(0.0875 + lc) - 34.21 \quad (1)$$

where $SOC\%$= percentage of the original, long-term-average, primary forest SOC, lc= logging cycle [number] from 1 to 35 cycles (i.e., up to 2,800 years since first logging), R^2= 0.95, Correlation coefficient= 0.97, Maximum error= 32.05, Mean squared error= 11.92, Mean absolute error= 1.475, Coefficients= 3.

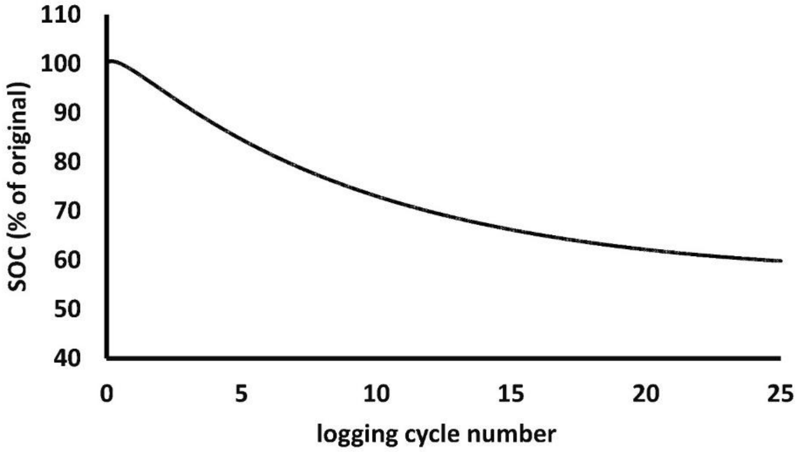


Figure 9. Change in SOC, as a percent of the time-averaged-mean of the original primary forest SOC, versus cycle number of logging the secondary forest, with 0 being the logging of the primary forest. This average line was from data from the models in Figure of Dean, et al. [239] cited in the main text here; data were first converted to the relative time through each logging cycle.

The model outputs merged to create Figure 9 had cycle lengths from 50 to 100 years. For other cycle lengths the curve may extend or contract depending on the time balance between coarse root growth, and root and SOC decomposition, within each cycle. Other causes of variation in the curve will be, for example, the local environment, species, and logging procedures.

Quantitatively, Equation 1 and the curve in Figure 9 indicate that after 3 cycles of intensive logging there will be a 9% drop in SOC, 18% after 6 cycles, and 30% after 12 cycles. The trend implies that even where people are no longer logging primary forest, they will still be losing carbon if persisting with logging of the secondary forest, except after many centuries. In contrast, conversion of forest to cropland can incur a similar ΔSOC after only ~20 years [251], making it much easier to measure, but even there, the decline continues in subsequent years before levelling off [[230], Figure 8, [252–254]]. The gradual decline in Figure 9 reveals that in order to measure ΔSOC with scientific significance, one may need to measure over several logging cycles. SOC shows a peak in the first cycle due to decomposition of the large amount of debris from primary-forest logging (unless it is mostly burnt), so it is pointless trying to detect ΔSOC that soon. This explains why many review papers indicate ambivalence about ΔSOC with logging, which is echoed in the finding of Leuschner, et al. [244] that: '… the long management history of nearly all Central European forests often hinders the detection of legacy effects, unless true primeval forests are used as a reference.'

A few studies have observed the cumulative effects of several logging cycles, which allows comparison with Figure 9. Leuschner, et al. [244], for the upper soil profile down to 0.5 m depth report a 13.5% drop in SOC after 'several' centuries of logging (which may correspond to three or four cycles). However, that change was averaged over three different locations then converted to a percentage, whereas if the percentages had been calculated first for each of the three locations and then those averaged, the result might differ in magnitude. For the upper soil profile down to 0.55 m Ferré, et al. [255] note a drop in SOC of ~40% over 3 or 4 logging cycles over 37 years, but that includes soil manipulation through ploughing and fertiliser addition. For the upper soil profile down to 0.6 m, Vario, et al. [256] reported a drop in SOC of up to -24% for the second logging cycle. These changes are comparable with the trends shown in Figure 9 but are higher in magnitude, considering the number of logging cycles passed.

An example of the ambivalence arising from short-term experiments is in a recent review of ΔSOC, with different anthropogenic activities [245]. Their overall finding concurred with that of Dean, et al. [239], in that intensive logging of a primary forest followed by repeated logging cycles decreases soil carbon compared with the earlier, long-term average for the primary forest. However, they found that the change was mostly in the topsoil, whereas if allowing sufficient time for the whole soil profile to match that of the new forest cover, then that whole profile is more likely to change (as lower down changes more slowly). Conversely, another recent examination of ΔSOC with logging, though only thinning, which considered mostly short-term experiments [257], could not find any statistically significant effects of logging. On considering the dynamics described above, short-term and surface-soil experiments cannot possibly show the long-term, whole-profile effects on SOC from logging. Thus, although Lei, et al. [257], cited Dean, et al. [239], they didn't take onboard the timelines for assessing change mentioned therein.

Most chronosequence studies compare forest SOC at different ages since the first logging, against the SOC of primary forest, but only for one logging cycle [[258-260],e.g., [261,262]]. There are few experiments on ΔSOC that span several logging cycles. They would probably use space-for-time substitution, but the non-equivalence between sites is the bane of such experiments. Often, one of the pairs has already been considered inferior for resource extraction. Undeveloped land has a long history of being less productive than developed land [50,263–266]. It will be difficult to find data in the form of equivalent paired sites, to compare with the modelling in Figure 9. A typical example is presented here. For selective logging over about two centuries (i.e., no definitive cycles) Christophel, et al. [267] found an increase in the upper mineral soil to 0.3 m depth and a decrease in the organic layer above, and combined there is a net decrease with the logging. However, the paired sites may not have been equivalent because the mineral soil in the unmanaged forest sites had 'larger stone contents' [267]. Regarding that comparison of sties, the increase in stoniness % volume for the primary forests compared with the managed sites, averaged across the three locations in Christophel, et al. [267], for the Ah, AB and BC horizons was 14%, 10% and 21%, respectively. These amounts are enough to indicate that the productivity on the sites could be different and if so, then the organic matter returns to the soil will also be different. Also, as sampling an equivalent mass of soil when making SOC comparisons is sometimes necessary [268], the soil sampling depth would need to be between 10% and 21% deeper than -0.3 m in the stonier primary forest.

It is often inappropriate to ignore earlier land cover when portraying the carbon forecast for forestry, because of the centuries required for SOC stock to reach a new dynamic equilibrium [e.g., [230,269,270,271]]. In ecology, where there is a major disturbance followed by a long-term change, and the flora and fauna species take time to reach a new equilibrium, that period is called 'relaxation time' [272,273]. That same term will be adopted here, for the forest soil organic carbon to adjust after conversion from the long-term average of a primary forest to that of the long-term secondary forest under logging cycles.

There are ~2 Mha of plantations in Australia, and ~5 Mha of mostly hardwood production forest on public land, plus between 22 and 108 Mha of privately owned or leased forest but most of which is not commercially viable [274–276]. Most of that 7 Mha of forest production land was primary forest prior to logging. And as Australia only began converting primary forests about 200 years ago, much of that area is still in the first few cycles, and consequently Australia has much more carbon to emit within the relaxation time [if it continues logging on that land].

An appropriate baseline corresponding to somewhere within the relaxation time must be included when showing a carbon forecast for a particular activity. It is often not included in bioenergy and forest carbon accounts [277]. Indeed, forest debris, waste from mills and almost any timber from forests is called 'renewable energy' and burning it is considered 'carbon neutral' by many corporate forestry proponents, because the trees can be regrown [e.g., [278]]. But this neglects the decreasing soil carbon during the relaxation time. Time segments and baselines from within the relaxation time could be included in corporate carbon accounting. If the date of the original forest conversion to logging cycles is known, then one can estimate where in Figure 9 the current logging is acting. This would allow up-to-date life-cycle-analysis calculations for the carbon footprint of recent wood products, which could be conveyed to customers or used in national accounts.

As a corollary to the long-term emission upon conversion of primary forest to logging cycles, another perspective is offered by considering the time taken to replenish emitted carbon. Long relaxation times are also required upon soil recovery, for definitive results— to separate the signal from the background noise. Recovery requires several generations of trees to grow and decompose to sequester the carbon into the soil, as shown in the model output in Figure 10 (adapted from Dean, et al. [230]). For diagrammatic simplicity, the SOC depletion in this example is from deforestation, rather than from logging cycles. The modelled SOC in Figure 10 starts at 0 Mg ha-1 before any vegetation contributes carbon to it, to show the spin-up time for models (mentioned above), but less time would be needed where there is nearly as much legacy carbon as in the future forest. The ripples are due to sudden death of the stand (such as from a stand-replacing fire) but the same overall shape of the sequestration and emission occurs when modelled as annual contributions of SOC to the soil [230].

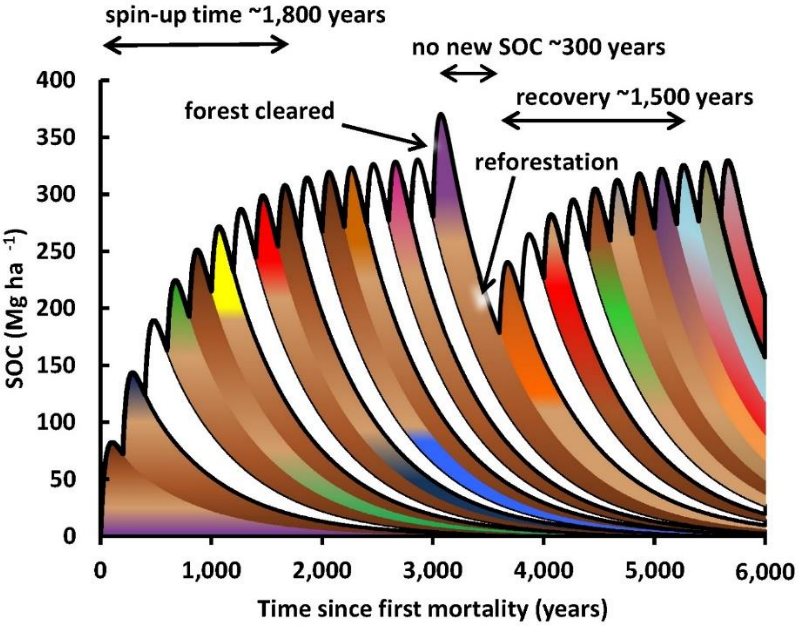


Figure 10. Output from a simple model in Microsoft-Excel: a three-pool SOC system with half-lives of 2, 50 and 500 years where SOC was created from decomposing roots of trees that died and regrew every 200 years. Colours differentiate successive generations of trees. After ~3,000 years there was deforestation and the site left barren for 200 years (e.g., a wide forest road). Recovery of SOC took nearly 1,500 years (the 'relaxation time') because several generations of forest had to regrow, mature, and decompose, to supply new SOC.

4.4. Points of Reference for SOC Measurement

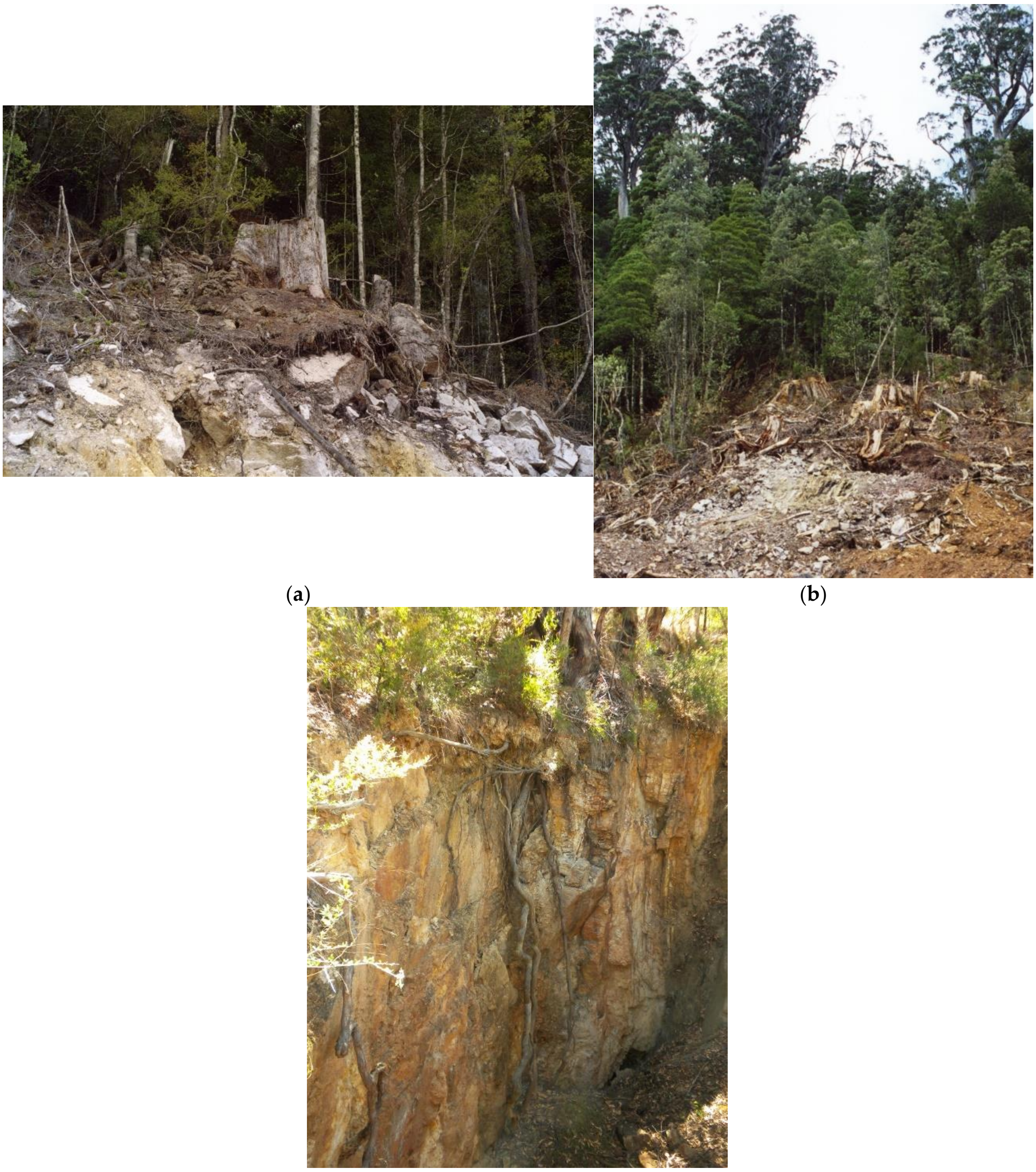
4.4.1. Difficult Locations: Under Large Trees and Deeper Down

Another reason for ambiguity over ΔSOC with logging of primary forests is that, before logging, SOC is not usually measured under large tree trunks where it is usually more concentrated and deeper in the profile than in between trees, nor is it measured under large coarse woody debris (CWD) [92,279] (Figure 7). Therefore, pedogenic patches [279] of concentrated SOC could be missed before logging. During logging the large stumps and CWD are sometimes upturned, broken or bulldozed aside, such that the lignomor underneath is accessible to random soil sampling (Figure 7.d-e). Approximately 4% of the SOC could be missed prior to logging because of this difference in measurement before and after logging [Figure 11 in [92]]. This could cause inaccuracy and imprecision, or introduce bias when assessing the impact of land use on SOC.

Depth is another inadequately represented dimension when measuring SOC [224,261]. The international, minimum recommended depth for measuring SOC change is 0.3 m, principally for international comparisons [280], with a more general recommendation of 1 m [281]. Depending on the tree species, soil profile and fragmentation of bedrock, the roots and therefore a substantial portion of the SOC from decomposition of old roots or infiltration of surface water containing tree carbon, can be below a metre depth [92,224,282–291]. The depth for SOC measurement should be guided by such factors [224,290–292].

To measure 90% of SOC in a mixed-forest in Tasmania it was necessary to include soil down to ~1.6 m depth [92]. The depth to include at least 95% of the root biomass for *Eucalyptus* species is often over 2 m or deeper where the soil depth or fracturing of bedrock permits [92,282] (Figure 11.c). For a site in the Brazilian Amazon where SOC was measured to -10 m, only 21% of the SOC was in the top 0.3 m, ~50% to -1 m, and 16% was below -3 m [293]. Where soil or bedrock and species permit, roots of woody species extend to 10–20 m depth [294,295], which will create pedogenic high-SOC patches after root decomposition.

The stoniness of many forest soils is often an impediment to SOC sampling. The location in Figure 11 (a) and (b) with a very thin layer (~0.2 m) of mineral soil supported forests of *Eucalyptus regnans* and *Eucalyptus obliqua*, though only to ~45 m height, rather than the 75 m typical of mixed-forests. Tree roots can occupy fissures up to several metres in bedrock and decompose there (Figure 11.c), creating soil and adding SOC, and contributing to translocation of water that may contain dissolved organic carbon from above [e.g., [296]]. The inclusion of this characteristic in carbon modelling, is suggested based on experimental evidence of likely forest root decomposition in weathered and fractured bedrock plus underground transportation of some of that resultant carbon as dissolved organic carbon [289]. Organic carbon in fractures in weathered bedrock, even granitic, is linked to root distribution. The rock surface in the fractures (to < 0.02 m thickness) can contain associated soil carbon [297]. The fraction of root mass in bedrock (and hence the soil carbon derived directly from root decomposition there) naturally depends on, for example, the depth of the A and B soil horizons, tree species, climate, tree age, water distribution, and bedrock type [298–300].



(**a**) (**b**)

(**c**)

Figure 11. (**a**) and (**b**): thin soils with large boulders that would prohibit typical soil sampling to more than about 0.2 m depth. The amount of total SOC may be similar to that for other *Eucalyptus regnans* and *E. obliqua* forests of equal long-term-average biomass in that climate, and would need to be included in carbon modelling of land use. Photographed during clearfell cable logging of primary forest in coupe WE008e, near Mt. Wedge, Tasmania. (**c**) Roots descending over 5 m into fractured bedrock in dry schlerophyll forest, South Australia, exposed during mining. When these trees die (apart from those exposed) some of the root carbon will form local SOC.

In a critique of a soil carbon GIS and modelling study by Dean and Wardell-Johnson [301], McIntosh, et al. [302] referred to three reports on soil organic carbon in northern Tasmania [303–305], as being available as points of reference for comparison for SOC down to 1 m depth, i.e., they purportedly formed benchmarks. These three reports will be examined here to portray aspects of SOC measurement appropriate for use in the calculation of land use effects on the carbon cycle and for comparison with carbon-cycle modelling.

The soil depths studied in Grant, et al. [303], for example, are only to between 0.8 and 1.2 m depth and many have a '+' sign on the last measurement. Additionally, there were some methodological issues because the experiments were not originally designed for spatial carbon assessment. They had been instigated to determine soil suitability for plantations [306]. Those methodological issues are explored here. Part of peer review is to make sure that standard scientific protocols have been followed. Science that gets published without peer review is called 'grey literature' and therefore using it as a basis for the next advance in science is viewed as meaning that the next step might possibly be more dubious than if using peer-reviewed science as a basis. The three 1995 Tasmanian reports [303–305] were such grey literature. Consequently, checks would have been appropriate on how applicable their data is to scientific assessment of spatial carbon accounting, under the scientific peer review process.

That '+' sign indicates that the soil profile continued deeper than was sampled, therefore, it is likely that there was more carbon to be tallied than that reported. Such a comparison against modelled SOC values is often not valid because the values derived from modelling, such as in Dean and Wardell-Johnson [301], which used CAR4D, include SOC to wherever it may have been translocated, such as down into the fractures of bedrock, or laterally offsite by groundwater or streams. Due to the limited data available, in modelling the carbon cycle in CAR4D, the entire soil profile (including fractured or semi-permeable bedrock) is treated as homogenous: undifferentiated laterally or depth-wise [301,307]. Some modelling software accounts for SOC to only 0.3 m, for highly calibrated sites or it uses very generalised profiles, or tweaks the carbon-compound half-lives or emission pathways to match SOC to 0.3 m, for example in FullCAM [[308]; S. H. Roxburgh, CSIRO, personal communnication, 2021].

McIntosh, et al. [302] referred to the modelled total of 685 Mg ha^{-1} for the case study #1 site in Dean and Wardell-Johnson [301], which was a *E. regnans*-dominated forest (introduced in [307]) as being:

'…more than twice the maximum measured soil C value under *E. regnans* available to these authors in 2010 which was 273 Mg.ha^{-1} in the previously mentioned Stronach profile [21].'

The SOC values in McIntosh, et al. [302] were only tallied to 1 m depth rather than for the full profile as in carbon modelling, which suggests that the 273 Mg ha^{-1} for the Stronach site in Grant, et al. [303] (the reference '[21]' cited in McIntosh, et al. [302]) had also been truncated to 1 m depth from its reported 305 Mg ha^{-1} to 1.2 m.

Values of SOC for the full profile can be estimated by fitting simple exponentials to reported empirical SOC data using equations of the form:

$$cumulative\_SOC = g\,[1 - \exp(bz)] \tag{2}$$

where z is depth in metres (negative below 0); Dean, et al. [92]. If z is lower than where SOC was measured, then the extrapolation relies on there being soil or rocks that can adsorb SOC occurring lower in the profile, or DOC or colloidal SOC being carried deeper or sideways. From soil sampling in McIntosh, et al. [302], the site of highest SOC had 304–326 Mg ha^{-1} to 1 m depth. Extrapolation using Equation 2 for this site gives ~498 Mg ha^{-1}. This value is closer than their declared benchmark of 273 Mg.ha^{-1}, to the temporal average of 685 Mg ha^{-1} for site #1 modelled in Dean and Wardell-Johnson [301], which was for the entire soil profile and any translocated SOC.

Additionally, site #1 in Dean and Wardell-Johnson [301] had above-average, long-term, live biomass, as shown later in this paper, and therefore could be expected to have above-average SOC, whereas the sites in Grant, et al. [303] are more likely to have been average. The live biomass for mature *E. regnans* stands can vary greatly, e.g., 262–647 → 697–1053 [57,77,92,309], and is possibly related to the 'site index' (forestry terminology for potential wood volume at a particular age). If this difference persists in particular locations over successive generations of *E. regnans* trees, then by the positive relationship between aboveground biomass and soil carbon [mentioned above], there should be an equivalent range of SOC between those stands. On a pro-rata basis the variation in biomass found in Dean, et al. [92] for stands of approximately equal maturity, corresponds to a variation in SOC of 184-to-455 Mg ha^{-1} (across the average of 330 Mg ha^{-1}).

Regarding suitable points of reference, there were other SOC data available back in 2010, from soil carbon accounting experiments in *E. regnans*-dominated forests: data from Victoria [258] had been used to calibrate the model 'CAR4D' [307]. For those sites, there was an estimated 1,300 to 3,000 Mg ha^{-1} to 10 to 20 m depths, and a measured average of 650 Mg ha^{-1} to 1 m depth [258]. The SOC in CAR4D was modelled as being less than at those Victorian sites, concomitant with the frequently shallower soils in the Styx Valley, Tasmania. Now that SOC data from experiments designed for carbon accounting are available for *E. regnans*-dominated forests in Tasmania [92,302,310], CAR4D can be recalibrated.

4.4.2. Difficult Locations: Coarser Components

Not only dimensions need to be considered comprehensively but also other parts of the soil. Many, though not all, experiments involving sampling of SOC in soil have not measured the carbon in firm particles greater than 2 mm width [290,311]. The reason for this may be because: (a) they were considered chemically unable to bind to organic carbon [290,312], (b) some researchers may do it simply to align with an established protocol, or (c) the harder fragments can be more difficult to grind in preparation for elemental analysis. But the practice can miss out on substantial portions of SOC, resulting in underestimations for some forests and possibly incorrect calibration of carbon dynamics models [290,312,313]. Significant amounts of organic carbon can be dissolved into stones or adsorbed on their weathering surfaces, with significant contribution to total SOC [314–316].

Where there has been forest fires or post-logging burns, there may be pieces of charcoal as well as coarse mineral fragments. Without any post-logging burn, Hopmans, et al. [317] found SOC to 0.3 m depth was 209 Mg.ha^{-1}: with and without post-logging burns the amount of SOC in charcoal and rocks >2 mm width was 42% and 29% of the total SOC, respectively [317]. In forests with occasional fire but no post logging burns Buma, et al. [318] found that the SOC to 0.1 m depth contained ~16% charcoal in the >2 mm fraction. Holub and Hatten [213] and Harrington, et al. [319] also sampled carbon in material >2 mm width but only up to 4.75 mm width.

Charcoal is more likely to be near the surface (unless in buried sediments) and coarse stony fragments are more likely to be closer to the bedrock [313]. It is logical to include such carbon pools in forest carbon accounting related to climate change. The organic carbon associated with charcoal and mineral fragments is inherently included in earlier modelling work [47,235,269] and was measured empirically in Dean, et al. [92]. Discarding and not counting the organic carbon in the >2 mm charcoal and rock fragments in the soil, could be one of the reasons for the lower SOC benchmark suggested by McIntosh, et al. [302].

Soil may also contain very old organic carbon that is not derived from the current or recent forest biomass, but from ‘fossil’ carbon in soil derived from sedimentary bedrock [320,321]. This could complicate modelling of forest carbon dynamics and measuring the effects of land-use-change. Organic matter found in rocks and soil during forest carbon accounting experiments originating from the forest should ideally be differentiated from that from shale, some sandstone and mudstone etc (such as the grey or black varieties). Radiocarbon dating, radiocarbon natural abundances ($\Delta^{14}C$), or carbon isotope ratios ($\delta^{13}C$) may allow such differentiation [322].

4.4.3. Soil Sampling Specific for SOC Accounting

Apart from just the depth aspect, it is worth considering those 1995 Tasmanian purportedly benchmark studies more closely in different dimensions, especially as there has been further developments in science now that spatial soil carbon accounting is a common goal. When considering points of reference (benchmarks) for comparisons, one must consider possible bias and error margins. One of the aspects that has progressed considerably in the last few decades to reduce bias and error margins is elemental analysis of soils [e.g., [323,324]]. The three 1995 Tasmania reports used a modified Walkley-Black (WB) method for carbon assay but without any mention of a correction factor:

‘These have been determined using the Walkley and Black colorimetric method (Rayment and Higginson 1992).’ [[325], p18], plus:

‘The procedure is that described by Rayment and Higginson (1992) using 0.5 M sodium dichromate ($Na_2Cr_2O_7.2H_2O$). Read absorbance using the “Cecil spectrophotometer CE 292”.

doi:10.20944/preprints202410.0555.v1

Determine organic carbon values from plotted standard curve of absorbance verses organic carbon (%)' [306]

The WB method may not measure all the organic carbon in the soil, depending on the chemical structure of the organic molecules, and on how they are bound in the soil, which in turn is dependent on for example, climate, soil type, parent rock type, tree species and land management [326–328]. And it may not measure the carbon in charcoal, as is necessary in forest carbon accounting. The WB method relies on oxidation of organic carbon, under moderate heat generated by the action of aqueous sulfuric acid, with accompanying reduction of Cr^{6+} to Cr^{3+}, but the oxidation of carbon may be incomplete. Which is why the USA Forest Service recommends that the method should not be used [329].The amount of oxidised carbon is determined by titration for Cr^{6+} or by colorimetry of the Cr^{3+}, or simply by measuring the amount of CO_2 released. The recommended correction factor, which is a multiplier, is usually near 1.3 but calibration for specific environments and land use may require multipliers nearer 1.7 [330]. For some Tasmanian plantations Wang, et al. [331] found that the WB method detected 97% of the carbon found by using an Elemental Analyzer and therefore no substantial multiplier was needed, but that may have been due to the younger, loosely bound carbon [332], and possibly as the plantations may have been sprayed with atrazine (which contains organic carbon and may not have been bound strongly to the soil). One modified-WB method uses heat to oxidise more of the carbon and thus requires a lower multiplier [327], but the heat method was not used in the 1995 Tasmania reports, though sodium dichromate dihydrate was used instead of potassium dichromate. Meersmans, et al. [327] recommend calibrating the WB method for different soils and land use. Any use of a multiplier in the three 1995 Tasmania reports, was not reported, and the same colorimeter absorbance vs carbon curve may've been used for all samples, thus rendering them quite imprecise for spatial carbon accounting.

Another increase in uncertainty occurs when the soil bulk density is either not calculated or not reported, in a method appropriate for soil carbon accounting. The three 1995 Tasmania reports that McIntosh, et al. [302] suggested should be points of reference, provided the concentration of carbon in the soil as wt% from the fine, sieved soil, but the reported bulk density was the weight of the whole sample, including stones and roots, divided by the volume of the entire sample [306]. To use such data for spatial SOC calculations requires recalculating the density of the fine soil fraction using any reported stone and root volumes and weights. However, the volume of stone and roots was reported within wide ranges for the different horizons in the soil types in the area, and not precisely for the specific sample from which carbon or bulk density were measured. For soil carbon accounting it is necessary to subtract from the weight and volume of the soil core, the weight and volumes (in quantitative terms for each sample) of any material in the core that isn't measured as part of the fine soil [333–335]. If the carbon in any of those components was derived from the forest biomass then it can be included later in the carbon accounting. Preferably, the bulk density and carbon concentration should be determined from the same sample [336]. Therefore, in those three 1995 Tasmanian reports, the volume and weight of soil within each sample volume could only be determined semi-quantitatively, and consequently with substantial error margins.

To illustrate the error margins introduced by that earlier methodology, a few examples from the three 1995 Tasmanian reports will be processed here. The amount of roots was given as, for example, 'many medium roots' in an A1 and a B1 horizon, 'abundant fine roots' in an A1 horizon and 'common coarse roots' in a B2t horizon. The stone content of the soil was given in more quantitative terms; for example: 2–10% granite (20-60 mm fragments) and 10–20% granite (60-200mm fragments) in a B1 horizon, 0–<2% basalt in an A1 horizon, and 20–50% sandstone in a B2 horizon. If one assumes the average densities for such rock types [337], and assumes that the bulk density was measured with the average amount of stone volume for a particular horizon (e.g., 35% sandstone for a horizon with 20–50% sandstone by volume) then the measured bulk density can be adjusted to give an effective one. However, insufficient detail was provided for a quantitative adjustment for the root volume.

By thus accounting for stone volume, the total soil carbon for four example soil types 'Cuckoo', 'Kapai', 'Stronach' and 'Maweena' which host wet-eucalypt forests, in Grant, et al. [303] reduces from 402, 344, 305, and 331 Mg ha^{-1} respectively (to the measured depths of 1.2, 1, 1.2 and 0.96 m respectively) to 180, 244, 297, and 112 Mg ha^{-1} respectively. I.e., reductions of between 2 and 66 %. Due to imprecision in stone content alone, the error margin for total SOC after adjusting for stone content, is up to ~±18%. It would be higher if including error margins in the rock densities, and if the root volumes were quantified in a similar style.

4.4.4. Comparing Carbon Stocks across Climates and Time Periods

In Australian forests, and indeed globally, SOC generally increases with rainfall (assuming other factors being equal) [338–340]. The largest concentration of Tasmanian rainforests is in the west and northwest of Tasmania, which is a higher rainfall zone (on average) than where the tall-open forests in Tasmania are [301,341]. Therefore, Tasmanian rainforest could be expected, on average, to have higher SOC values than the tall-open forest. The GIS analysis in Dean and Wardell-Johnson [301], based on national SOC and rainfall layers, and a Tasmania vegetation layer, shows this to be the case: 2218 and 1559 mm yr^{-1} and 369 and 271 Mg ha^{-1}, respectively for rainforest and tall-open forest. Likewise, the fraction of Tasmanian rainforest in close vicinity to tall-open forest in Tasmania is expected to have lower SOC than the average rainforest. That close proximity, lower-rainfall location (average 1241 mm yr^{-1}) is where McIntosh, et al. [302] measured SOC in the two forest types, and found an average value of 102 Mg ha^{-1} to -0.3 m depth for rainforest. When critiquing Dean and Wardell-Johnson [301] and comparing the two values for rainforest SOC, McIntosh, et al. [302] appeared to not take into account the difference in climate, and wrote:

'…the unreferenced 369 Mg.ha^{-1} of C at 0–30 cm depth quoted by Dean and Wardell-Johnson ([27], Table 1) for rainforest soils in Tasmania is over three times too high…'.

Many of the western Tasmanian rainforest soils in western Tasmania [which is where rainforests are concentrated] are organosols, with an average SOC of 450 Mg ha^{-1} to 0.47 m depth [342]. Assuming equal SOC distribution over that depth gives 287 Mg ha^{-1} to 0.3 m. Or alternatively, application of a simple exponential falloff with depth (Equation 2 above), gives 311 Mg ha^{-1} to 0.3 m depth. If the soil profiles and bedrock allowed SOC to venture much deeper than 0.3 m, then extrapolating to the full profile gives SOC as 1200 Mg ha^{-1}. The value of 369 Mg ha^{-1} in Dean and Wardell-Johnson [301] is 19% higher than the value of 311 Mg ha^{-1} found empirically by di Folco and Kirkpatrick [342]. Such a difference is reasonable, considering that the GIS data used was a nation-wide layer.

As described above, the mixed-forest and rainforest each have their own ideal geographical ranges, with some spatial overlap if fire history permits. On pedogenesis timescales, rainforest sites in close proximity to mixed-forest sites, are possibly often occupied by mixed-forest, and vice-versa. The overlap, in the absence of logging, ebbs and flows with the vagaries of fire [119]. There was still decomposing eucalypt debris in several of the rainforest plots studied by McIntosh, et al. [302] and some contained live eucalypts:

'While rainforest sites contained negligible quantities of live eucalypt boles (small eucalypts were encountered in just 3 rainforest sites), they contained significant quantities of eucalypt CWD (Table 2). Eucalypts contributed 27% of total CWD volume and 29% of total CWD C-mass in rainforests.' [201],

Thus, considering the timescales involved for change in SOC compared with those for biomass change, as described above, the experiment in McIntosh, et al. [302] was not really designed for their stated objective of differentiating between the SOC in mixed-forest and in rainforest. Moreover, these timescales may mean that such a differentiation in SOC of the ecotone region does not exist; it may only exist in the regions where rainforest and tall-open forest do not swap locations over time, and this would match with the data to-date, in the reports discussed above.

The blurred spatio-temporal boundary between mixed-forest and rainforest is in part because coarse woody debris and especially soil carbon, representative of either forest type, persists to a degree depending on its half-life. This material constitutes 'legacy carbon' [343,344] (Figure 12). Empirical studies attempting to contrast SOC stocks in rainforest and mixed-forest in close proximity, will thus be unproductive until the transition between the two is almost complete for all forest attributes, include soil.

Figure 12. Fallen *E. regnans* logs spanning Cliff Creek, Styx Valley, Tasmania. The creek is centred in a 200 m wide gully mapped as API-type rainforest. The logs are remnants of previous mixed-forest. This contrast typifies the blurred spatio-temporal boundary between mixed-forest and rainforest, being in-between from a carbon dynamics perspective. The gully is not rainforest from a carbon perspective, even though living eucalyptus trees are absent.

When rainforest and mixed-forest are in close proximity, and may interchange location over time, then it is appropriate to consider the effect on differences in biomass, rather than on SOC. For the purposes of carbon accounting for climate change effects, this should be considered over long time periods, to get the overall impact. A long-term, time-based average carbon stock, has equivalences to a landscape-level average at one point in time, if numerous instances of forests of different ages are present across the landscape. Here the claim by Moroni, et al. [201] that net carbon will decrease and stay that way if mixed-forest is not logged but allowed to mature and go through succession to pure rainforest is re-examined, but over a long enough time period to get representative, time-averaged carbon stock:

‘As Tasmania’s wet forests transition from mixed forest to rainforest they can be expected to lose more than half their total (live + dead, standing + downed) bole wood volume and biomass as smaller dimension rainforest trees replaced the larger eucalypts. … Certainly, setting aside Tasmanian wet eucalypt forest to store C will not deliver the usual long term C accumulation benefits common to forests elsewhere and maximizing landscape level C-stocks is likely to require periodic disturbance to maintain the C-dense eucalypts in the landscape.’ [201]

The ’periodic disturbance’ in that extract most likely implies fire and/or logging. Prior to industrialisation and without Aboriginal burning (though burning on the far east coast may have trickled effects across to the centre of the island), mixed-forests occupied large areas in Tasmania (such as in the logging concession areas) and therefore they do not need logging to exist. The Primary mixed-forest has been the foundation of export-scale commercial forestry in Tasmania for nearly a century because of its eucalypt content. Whereas extraction reconnaissance projects for rainforest areas, such as in northwest Tasmania, have found logging to be either uncommercial or to require low extraction fractions to maintain forest health [[345-347,348], p37-38]. Thus there are at least two possible reasons for the controversy over benefits of the two alternative forest types. Prior to industrialisation in the area of the logging concessions, there were sufficient periodic disturbances of perhaps once every several hundred years in the form of wildfire, e.g., 450 to 500 years [349,350] to maintain a dominance of mixed-forest. Higher-frequency disturbance at such intensity is not necessary.

More fire is inevitable in the future with anthropogenic climate change [351,352], so the chances of succession to rainforest, or maintenance of rainforest, are less likely than normal. Rainforest attrition is forecast with climate change [353], which will put neighbouring mixed-forests in Tasmania at risk (rather than the risk coming from replacement by rainforest). Rainforest species in mixed-forest have an intrinsic benefit in the face of climate change: they protect carbon in the wider landscape from fire [354]. CBS logging can create poor regeneration of the rainforest species and create stands more favourable to pulpwood than sawlog production (i.e., shorter wood-product half-life and therefore add to climate change) [355–357]. As the seed source for the rainforest understory becomes restricted spatially with fragmentation of mixed-forest [357], further logging will make it more-difficult to regain primary-forest carbon levels, which will also contribute to climate change. The importance of primary forests in climate change policy has been re-iterated, with policy initiatives broached for their conservation [358]. Limiting forest fragmentation and general anthropogenic disturbance is likely to help maintain existing mesic micro-environments that protect against drying-out under climate change and thus against fire. At least 50% of rural fires in Australia are of anthropogenic origin [e.g., [359,360]]. Eliminating that ignition source and managing combustible material near infrastructure, such as roadside grassland and farmland [361], would help reduce forest carbon losses under climate change.

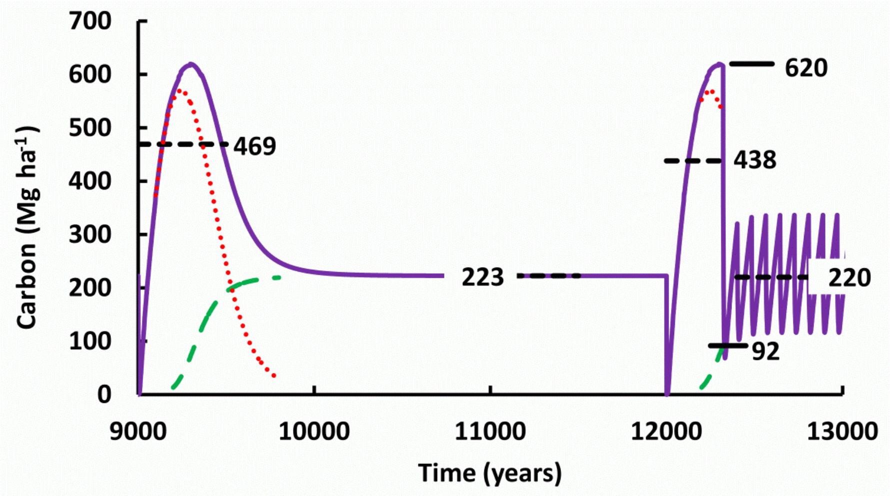
The difference in biomass accompanying the succession from mixed-forest to rainforest matches the typical case for forest succession, where the pioneer (coloniser) species is fast growing and has appreciable initial biomass [362,363]. Whereas Moroni, et al. [201] call it ‘unusual ecology’. In mixed-forest the coloniser species are eucalypts, e.g., *Eucalyptus regnans*. The issue of decline in biomass with forest succession was covered numerically and graphically in Dean, et al. [235], and it was mentioned that landscape-level carbon stocks in biomass can decline if: ‘the understorey, which dominates the later stages of succession, has a lesser biomass than the maximum for the *E. regnans* dominated forest’.

To illustrate possible dynamics of carbon in biomass of rainforest succession and to compare it against logging, two simulations were run using CAR4D (Figure 13), with two levels of understorey biomass: one very low and one higher but still only at around 20% of total stand biomass at the time of logging. Rainforest understorey can contribute 50% of total biomass prior to onset of substantial eucalypt senescence in mixed-forest [364]. Due to the low precision of understorey allometrics, no definitive quantitative result is shown in Figure 12, but even when including wood-products, it is not necessarily the case that harvesting cycles store more carbon than long-term rainforest.

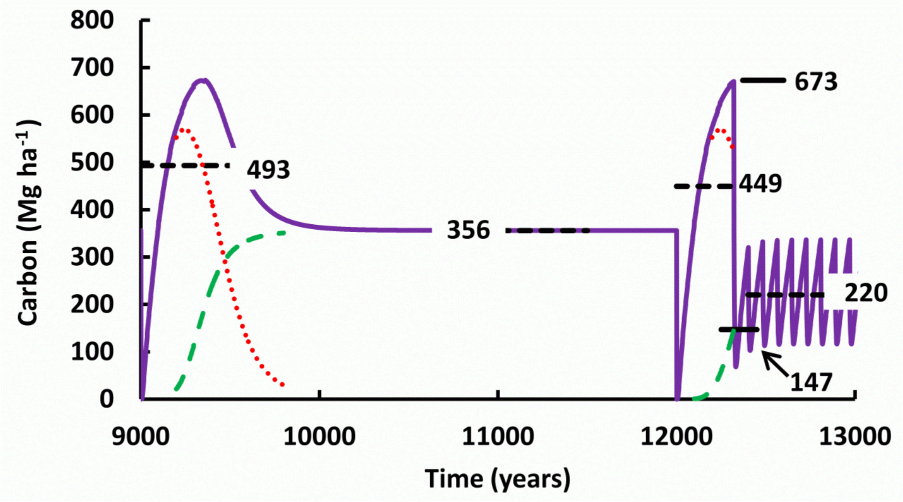
McIntosh, et al. [302] measured and compared standing biomass in the two forest types. They had a mixed-forest:rainforest biomass ratio of 1:0.45. The two scenarios in Figure 13 have ratios of 1:0.48 and 1:0.72. The first of those is close to that of McIntosh, et al. [302]. In both cases, the long-term rainforest average total carbon does not go below the long-term logging average carbon (where the carbon is summed across biomass and wood products). That is because rainforest total carbon doesn’t oscillate as much as that of a logged forest. The magnitudes in the frequent oscillations must be averaged over time, with an average that is lower than the peaks, which most likely correspond to the point at which they would be logged.

The long-term average C in biomass in rainforest can exceed the long-term eucalypt logging-cycle carbon (including wood-products) if it is a third or more of the biomass of mature wet-eucalypt primary forest (Figure 13.a). The C in biomass of rainforest understorey only needs to be about 15% of the C in biomass of primary mixed-forest at the time of logging, if the two forest types are to be equivalent in terms of C. However, it needs to be at least 20%, if upon succession it is to exceed the mixed-forest carbon (Figure 13.b). If comparing the long-term carbon stocks of pure rainforest and mixed-forest, then the long-term rainforest carbon only needs to be more than half of peak mixed-forest carbon at the time of logging of primary mixed-forest, if it is to substantially exceed long-term mixed-forest carbon (Figure 13.b).This is due to the more-frequent low carbon periods in mixed-forest than in rainforest. Note that this 50% requirement is very different to the 100% requirement implied in McIntosh, et al. [302] and Moroni, et al. [201]. Considering the error margins involved in modelling the carbon stocks of the two forest types and wood products, although Figure 13.b shows rainforest carbon exceeding logged mixed-forest carbon, in the long-term, it’s likely that it is not definitive at this stage: more accurate and precise data are needed.

Rather than considering the rainforest understorey biomass within mixed-forest as a representation of what the pure rainforest will be like after succession, there’s an additional possible change upon succession that will increase rainforest biomass. Gilbert [55] noted that rainforest biomass is slightly smaller in mixed-forest than in pure rainforest but added that it was difficult to find sites of equivalent productivity, from which a significant comparison could be made. Notably though, paired mixed-forest/rainforest sites in northwest Tasmania were found to have higher rainforest wood volumes in the rainforest members [365]. Thus, some of the eucalypt biomass may be replaced by rainforest biomass upon succession, which would make rainforest have a slightly higher advantage than indicated in Figure 13. Though another explanation for this increase in biomass for pure rainforest is that it could usually occupy higher site-index (more productive) sites than does mixed-forest.



(**a**)



(**b**)

Figure 13. Simulated aboveground carbon stocks of forest succession and logging scenarios including wood products, typical of Site-1 in Dean, et al. [47], for primary-forest and harvesting cycles (modelled using CAR4D). Important for comparisons: time-based averages of total carbon (including wood products)— dashed horizontal lines. (Solid curve=total carbon in biomass plus wood products; dashed green curve=understorey biomass; dotted red curve=*E. regnans* biomass, purple solid horizontal lines= total carbon stock at time of logging.) The rainforest understorey filled space between senescing eucalypts which succeeded to rainforest; afterwards intense wildfire with seeding from nearby *E. regnans*. New mixed-forest grows to age 320 years, then logged (parameters as in Dean, et al. [47]). Rainforest biomass was set to a maximum of ~1/3 (**a**) and ~1/2 (**b**) of peak of the mixed-forest— corresponding to 15% and 22% (respectively) of stand-level biomass at time of primary-forest logging. In (**a**) long-term rainforest biomass= ~half that of the time-averaged mixed-forest biomass and equal to that of the harvest cycles (including wood-products). In (**b**) long-term rainforest biomass= ~time-averaged mixed-forest biomass and greater than that of the harvesting cycles (including wood-products).

doi:10.20944/preprints202410.0555.v1

4.5. ΔSOC with Logging Burns and Disturbance by Machinery

Measuring the change in SOC accompanying land management is difficult, and for the forestry activity of logging even more so because of natural forest heterogeneity (the pedogenic patchwork, Stutz and Lang [279]) and the hodgepodge disturbances by logging machinery. The intense burn of logging debris ('residue') in CBS-type logging adds even more variability, but sufficient experimental evidence has been accumulating to infer definitive effects [366–372]. It must be noted, that for carbon-cycle modelling purposes such as determining values to include in computer software, the effects of the CBS activity are considered separately to the additions to SOC that occur afterwards from logging residue over longer time periods. The measured values of ΔSOC accompanying these relatively short-term interactions for different forest types range from -6% to -50%, but this is only for the upper soil surface, to a depth of no more than 0.15 m [239]. The burn appears to oxidise SOC to no more than ~-0.1 m [367], where there is no logging residue or roots penetrating the soil. The fraction of the total SOC pool emitted, depends on the depth distribution of SOC. That fraction can be estimated for *E. regnans*-dominated forests in two locations where the depth distribution of SOC has been reported. The equations for the profile in-between trees for locations in Tasmania and Victoria are given by:

$$cumulative\_SOC = 308.603[1 - \exp(1.44138z)] \tag{3}$$

$$cumulative\_SOC = 689.436[1 - \exp(1.62936z)] \tag{4}$$

where *cumulative_SOC* is the cumulative SOC in Mg·ha^{-1} to a certain depth *z*, in metres (negative below-ground). Equation 3 is for the Styx, Tyenna and Florentine valleys in Tasmania from Fig 9.b and Supporting Table S2 in Dean, et al. [92], and Equation 4 is for Toolangi in the Great dividing Range, Victoria from Figure 11 in Polglase, et al. [258].

Percentage drops in SOC for the upper soil surface are collated in Table 1 in Dean, et al. [239]. The second datum for Pennington, et al. [367] in that table needs to be corrected to ΔSOC=13.4% down to 0.1 m depth. The values for ΔSOC then range from -37% to 0.02 m depth to an average of -30% to 0.15 m depth. There are on average, greater emissions of SOC closer to the surface (R^2=0.67 for a straight line fit of ΔSOC versus depth). Applying Equations 3 and 4 to the 10 data points in that table, allows the fraction of total SOC residing near the surface to be calculated, which in turn allows calculation of the average ΔSOC from the felling disturbance and CBS burns, across the whole soil profile, as 3.6(2.5)% and 4.1(2.8)% respectively (standard deviations in brackets), i.e., ~4% loss of SOC from the soil profile due to the CBS burn.

The soil bulk density near the surface increases during logging (from disturbance by machinery), and again during the post-logging burn [367,373]. Pennington, et al. [367] found an increase in bulk density of 15% to 0.1 m depth due to the CBS burn alone. As a result of the common process of measuring to a fixed depth before and after logging activities, the increase in bulk density means that more mineral soil from deeper down is contained in the sample after logging. It too would have lost some SOC as its volume has decreased, which would increase the SOC loss for the whole profile to possibly more than 4%. The soil compaction and turbation indicate a need for additional care in experimental design and interpretation.

The estimated short-term drop in SOC from successive CBS logging events, used in modelling in CAR4D was 2.5% per cycle [47,301], which is conservative compared with the ~4% calculated above. A sensitivity modelling analysis showed that if the emission per cycle goes from 2.5% to 5% then the long term ΔSOC due to converting primary forest to logging cycles goes from -48% to -57% [47], which is a substantial difference. Therefore, both pinning down the effect better, and working on its future prevention, are important for climate change mitigation.

Previously in CAR4D the ΔSOC with each CBS event was erroneously set at -10% [307]. The error arose from applying a measured drop in the surface soil (0.2 m depth) found by Polglase, et al. [258] to the whole profile, instead of calculating its proportion of the whole profile, as done above here. Using the corrected CAR4D, the long-term (over 1.5 millennia) drop in SOC with conversion of a primary forest stand of *E. regnans* mixed-forest to logging cycles, case study #1 in Dean and Wardell-Johnson [301] was 47%. This was a correction of the earlier estimate of 87% in [307].

Absolute values of ΔSOC are important to carbon trading and determining the net effect on climate change from management. However, it is the percentage that is most important when comparing different land uses: it indicates what fraction of an ecosystem is being lost, and whether one land use constitutes an emission relative to another. The percentage loss of SOC with conversion of primary forest to logging cycles does not change if the initial stocks are lower: the long-term loss upon logging is linearly proportional to the pre-logging long-term average SOC stock [Figure 5a in [47]].

Without recalibrating CAR4D, a simple calculation can show the effect of the recently improved data on SOC stocks for *E. regnans*-dominated forests in Tasmania, on the absolute value of ΔSOC. If the SOC value of 330 Mg ha^{-1} from Dean, et al. [92], is used as the pre-logging SOC for site-1 in Dean, et al. [47] (which was also case study #1 in Dean and Wardell-Johnson [301]), then the long-term C emission upon conversion to logging cycles of average tall-open forests State-wide in Tasmania from 2010-to-2030 is 52(±26) Megatonnes of CO_2-e instead of the 66(±33) in Dean and Wardell-Johnson [301]. The change in SOC with logging for the low end— case study #2 in Dean and Wardell-Johnson [301] (a low biomass wet-sclerophyll forest)— remains as stated because the SOC stock was given simply as a typical fraction of biomass (and the change in biomass is unaltered).

5. Sustainability Interpretation for Corporate Forestry

The World Bank suggests that sustainability should be included in corporate social responsibility (CSR) but doesn't specify if that is environmental sustainability, and some companies interpret sustainability as meaning business viability [182,183]. Some corporations rely on the older targets and methods of corporate responsibility and issue sustainability rhetoric, against a background of legal and financial requirements which have not shifted markedly to reflect a changed meaning of CSR [[374], p17-18]. For example, when solutions for the global syndemic (the combination of the obesity, undernutrition and climate change pandemics) were reviewed, a change in management of CSR was recommended:

'Corporate social responsibility efforts, which are too often marketing exercises, need to evolve into a stronger accountability model, in which targets and performance criteria are independently specified, monitored, and publicly shared.' [184]

From the point of view of sustainability of the wood stock, the primary aim in the Tasmanian pulpwood concessions was to provide mill pulpwood and the second aim was a quota of sawmill timber. For scheduling log supply for the pulp mill, the rotation age was set at 80 years [52]. That period has passed but primary forest, both inside and outside of the concession area, are still a target of corporate forestry, and logging of primary forest vestiges within the concession area still occurs. Indeed, the impossibility of sustainability was recognised early on by industry (namely Australian Newsprint Mills, grantees of the Styx/Florentine concession), and there was no original claim to sustainability:

'Since the growing stock is dominated by a large quantity of overmature timber, the sustainable yield which could be cut from a "normal" succession of age classes on all sites has little relevance in determining the allowable cut for the present level of industry.' [52].

This is straightforward logic where clearfelling is used, but in some locations sustained yield is still claimed but redefined: the 'primary forest premium' referring to an accepted loss for secondary forest logging, such as 50% compared with the primary forest [375].

One practical step towards sustainability is to treat the resource as valuable. An example where this has not occurred is in the logging of large trees such as from mixed-forests in Tasmania: as early as the mid-1950s, while surveying forests in the Styx Valley for sawlog and pulpwood it was noticed that ~10% shattered on felling [376]. Even in contemporary times no large trees, or even sections of them, are lowered to the ground gently, thus wasting some potential sawlog timber. Also, a few large trees may be dynamited (with substantial timber shattering) if the feller cannot find a safe exit route after chain-sawing (e.g., coupe WE008e, pers. obs., 2004). The existence of cable logging and climbing arborists in Tasmania indicates the technology exists to keep logs intact.

Environmental sustainability is a widely accepted policy, and is often part of CSR [374,377]. 'Sustainability' is part of the latest name of the Tasmanian State forest agency: 'Sustainable Timber Tasmania'. Previous names were the 'Forestry Department', 'Forestry Commission' and 'Forestry Tasmania'. But the resource extraction method involving primary forests in the original pulpwood concession boundaries is fundamentally unchanged from the mid-20th century, except that for some logging, the CBS process has been more dispersed across a larger area via 'aggregate retention'. Sustained yield has not been achieved, for the reason described by industry in the last quote above. The name change does however fit in with the corporate concept of forestry, by promoting sustainability as part of market engineering [378].

'Aggregate retention', is essentially clearfell but with vestigial islands of forest [379,380]. The burn intensity is lower but the islands are sometimes burnt [379], which may not reduce the total carbon emissions. Without a reduction in the annual log quota, the dispersion of clearfell means an increased number of logging units and road area (increased forest fragmentation), which in turn means an increased area of forest subject to edge effects and escaped burns. Larger trees are more susceptible to fatalities at forest edges, where windspeeds are higher; the edges may have less SOC; and roadsides increase drying-out and grass cover, which can help the spread of fire [381–386].

With regards to sustainability of forest carbon levels, the Tasmanian Regional Forest Agreement (RFA) mentioned a need to manage the forests in a way that maintained or enhanced the carbon within the carbon cycle and that research priorities included the estimation of the impact of logging and fire on the carbon cycle [387]. The New South Wales (NSW, another State in Australia) RFA was more general by denoting environmentally sustainable logging as an objective that required long-term commitment. The NSW concept of sustainability included maintenance of forests' capacity for: 'formation of soil, energy flows and the carbon, nutrient and water cycles, fauna and flora communities and their interactions', biomass production, ecologically sustainable extraction levels, and minimisation of deleterious effects [388].

Overall, when including ecosystem ecology as part of sustainability, it has not yet been established in absolute terms by State-managed, corporate forestry in Australia [389]. For plantations in Queensland in 1997 procedures that were still being implemented included: minimum tillage to reduce soil erosion (and associated loss of organic compounds), and watercourse and riparian vegetation protection; and the effects of retention of logging residue on a rage of processes were still being investigated [390,391]. For native *Eucalyptus marginata* (jarrah)-dominated forests in Western Australia, as of 2004, more felling for timber had occurred than could be regrown, i.e., sustained yield had not been achieved for a significant period, larger trees had been targeted, with accompanying biodiversity loss [108].

Sustainability at the species level can also be considered. In contrast to the natural, spatial occurrence of the different eucalypt species of the Styx/Florentine pulpwood concession, logging focussed on *E. regnans*: shown by the areas logged of different species by 1976. From the 1950s to the 1970s, 63% of the oldgrowth area logged was of *E. regnans.* Initially, the Concession was only 19% *E. regnans* [by area], and 49% *E . obliqua*, and 23% *E. obliqua*) (data in ANM [52]). Therefore, the ratio of logged *E. regnans* versus availability in the concession was 3.4-to-1. The same ratios for the other species logged were: *E delegatensis*—0.82:1, *E obliqua*—0.5:1, and other sundry species—0.083:1. Tasmanian State forest was still being high-graded for *E. regnans* from 1999 to 2009 [47].

The preference for *E. regnans* was initially because *E. regnans* was the most suitable for pulping, having the least fraction of material to be initially discarded in primary processing [52,392].

'The species has proved particularly suitable for pulping and has been used to provide a good quality, easily bleachable soda pulp on the one hand and, on the other, a sulphate pulp which has been used for the manufacture of kraft paper. It also forms the basis of the Australian newsprint industry because it has been found to be the one eucalypt species which is most satisfactory for grinding.' [392].

From 1941 to 1957 the sourced timber was 90% oldgrowth *E. regnans*, until changes in the pulping process allowed the use of other mature eucalypt species, reducing the mature *E. regnans* requirement to 75%, and further changes in 1971 allowed the use of regrowth timber including some acacia, reducing mature *E. regnans* requirements to 50% by 1979 [52].

Analysis of data in ANM [52], shows that tree height, rather than species alone, was a factor in species selection in the Styx/Florentine logging concession. Stand descriptions therein refer to the API types (described in Section 2) for the eucalypts in the stands. For the three main eucalypt species present in the concession area, eucalypt category E1 was logged in preference to category E2 (Table 1). For example, the ratio of E1 to E2 in the area of oldgrowth *E. delegatensis* present was 1:7.0, but the ratio in the area logged was 1:1.4, i.e., less E2 was logged less than if logging areas across stands of E1 and E2 *E. delegatensis* had been selected randomly. This preference for logging the taller height categories means that the forests were high graded based on tree height. Data on the areas of remnant oldgrowth for the different species in 1976 in the Styx and Florentine Valleys [52], show that *E. regnans* is twice as likely to be in category E1 than E2, whereas *E. delegatensis* is 7 times more likely to be in E2 than in E1, and *E. obliqua* is 5 times more likely to be in E2 than in E1 (Table 1.(a)). Thus, there were two fronts of high-grading: species (owing to compatibility with the pulping process adopted) and height. Thus *E. regnans* has been depleted the most.

Table 1. Ratios of areas of different species relative to those in height category E1. Based on data in ANM [52]. Each species was normalised with respect to E1 separately and for parts 1.a and 1.b separately. **(a)** Oldgrowth as of 1976, and **(b)** oldgrowth logging from 1953–1976. Comparing 1.a and 1.b shows that the taller height categories were preferentially logged.

(**a**)

	E. reg	***E. del***	***E. obl***
E1	1	1	1
E2	0.53	7.0	5.4
E3	0.0051	2.9	5.2
E4	0.00023	0.11	0.21

(**b**)	***E. reg***	***E. del***	***E. obl***
E1	1	1	1
E2	0.18	1.4	2.7
E3	0.013	0.17	3.1
E4	0	0	0

In the later 1970s there was already possible evidence of over logging of *E. regnans*: *E. regnans* seed was scarce and some areas of previous *E. regnans* oldgrowth were re-seeded with other eucalypt species such as *E. delegatensis* [393]. This may have exacerbated the situation by converting areas to a different species. This process appeared to be still continuing in 2011. The non-*E. regnans* species are of lower carbon stocks (on average). The average amount of pulpwood per hectare for oldgrowth in the concession, as of 1976, for *E. delegatensis* and average *E. obliqua* stands was 15% and 29% less than for *E. regnans*, respectively [data in [52]]. Species, site productivity and provenance of seed, can influence biomass [394]. Therefore, with the historical high-grading of *E. regnans* having removed the taller trees and there being a seed shortage, remaining seed for manual coupe regeneration of *E. regnans* may come from areas where the genetic provenance is shorter.

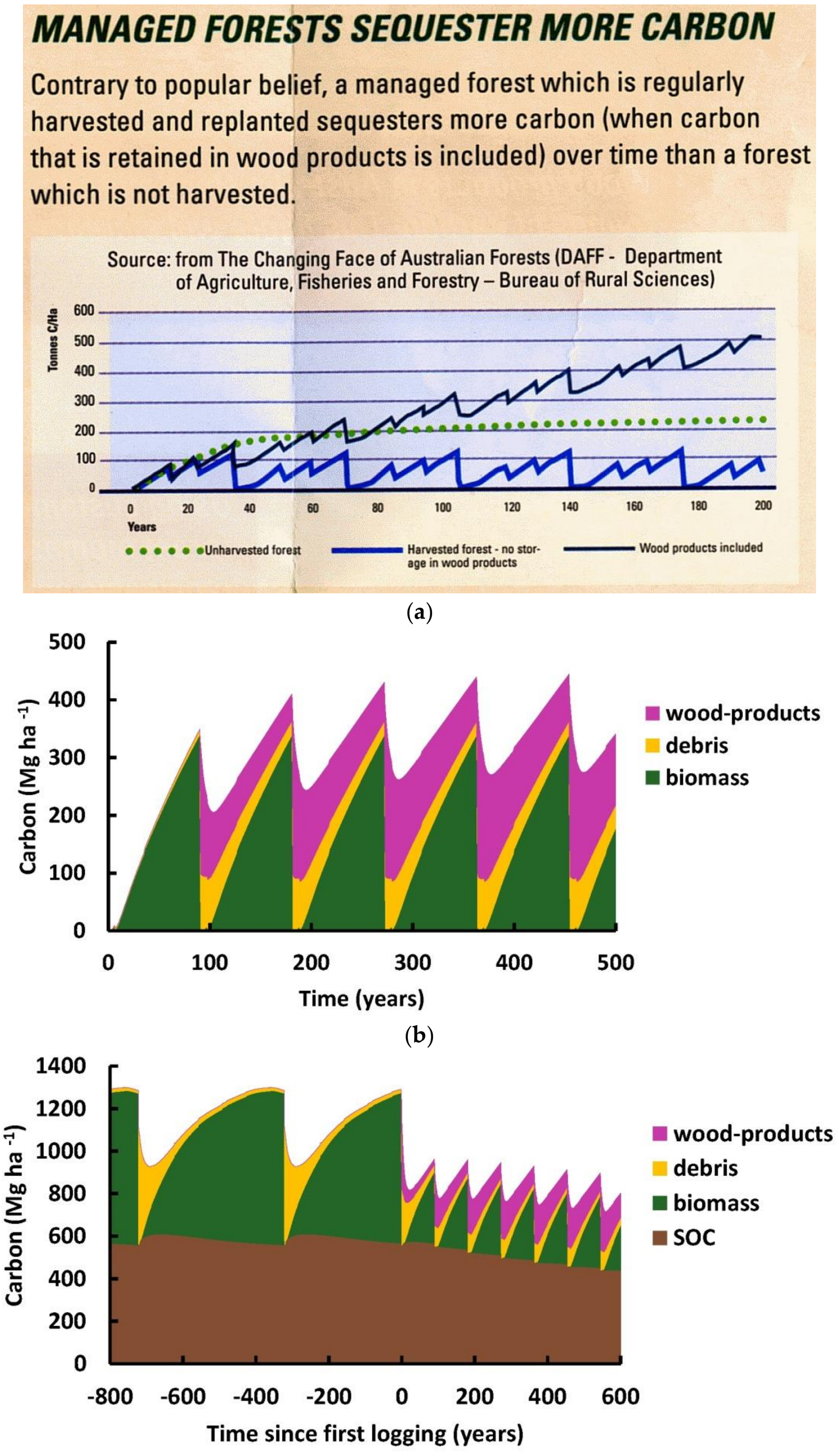
Taller trees have been targeted globally, despite their disproportionately greater contribution to climate-change-mitigation and ecosystem processes, and despite their decline due to climate change [395–398]. This would gradually reduce local carbon storage, in a similar way to the reduction in animal body or targeted component size, due to people hunting larger individuals [399,400].

5.1. Applying a Missing Baseline

When developing and publishing the results of carbon accounting models, the system boundaries for material flow must be acknowledged, without which unfactored emissions or double accounting may occur [4,401]. This lack of clarity may lead to knowledge gaps and publication of erroneous information [4]. A baseline (reference point, a benchmark) of 0 Mg ha^{-1} for carbon stock (i.e., omitting legacy carbon such as soil carbon and logging residue— a blank slate) is often portrayed in carbon-oriented promotional material for forestry [e.g., [402], Figure 10]. In Figure 14.a promotional material from Forest & Wood Products Australia (a forestry representative body) is shown with a baseline of 0 Mg ha^{-1}, i.e., as though forestry management starts on a blank slate. In this review paper, the process was simulated using CAR4D. Parameters in CAR4D were generously set to a logging-cycle length of 90 years and with wood-product half-lives of 80 years for sawlog and 4 years for pulpwood. These half-lives include those of mill residues, some of which are routinely incinerated within a year of logging. Also, these longer half-lives are a proxy for some of the wood-products contributing to SOC in landfill, while noting that the methane produced there (which has a higher global warming potential than CO_2) may counterbalance the longevity of that pool [[403,404], p98-102]. Emissions from local and international freighting of the wood-products, and thinning between full harvests, were omitted from the simulations. The same simulation was run again but with the benchmark being the original primary forest biomass and SOC instead of starting with them at 0 Mg ha^{-1} (Figure 14.c and Fig 14.b respectively). The largest difference in results between the two simulations is due to the neglect of the legacy SOC of the primary forest, in the corporate forestry version. The overall trend in carbon stock was down instead of up, when the legacy carbon was included. A zero or unnaturally low baseline for forest SOC is also used in some scientific calculations for assessment of forest bioenergy climate change effects, though it is often disclosed as such [e.g., [405,406]]. Ignoring the legacy SOC would be more appropriate for plantation establishment on long-cleared land or in some of the forests in Europe considered by [407], where logging has occurred for many centuries, but not for primary forest conversion to logging cycles. Therefore, the promotional material with a benchmark of 0 Mg ha^{-1} could be an example of generic marketing.

The numerical output from CAR4D for the simulations showed that for recovery of total carbon stock to that of the time-averaged, original primary-forest, then wood-product half-lives would have to be increased to 400 and 20 years for sawlog and pulpwood respectively, including for mill residues. It would be difficult to achieve such half-lives. But if they could be achieved, then the recovery would take ~1260 years. During that time, the drop in SOC would be counterbalanced by carbon in the wood products. Meanwhile however, emissions due to the steadily decreasing SOC would contribute to climate change and consequently also to climate-change positive feedback [globally].

Similarly, life-cycle-analyses (LCA) that claim emission offsets for burning wood waste [from logging, i.e., ‘forest residue’ or ‘mill residue’] for energy require accounting of the C balance of logging, and comparison with other energy sources such as hydroelectricity, wind and solar power. Comparable system boundaries in the LCA must be used for any comparison with substitutable materials [408]. The emissions in processing and delivering the wood-product must be considered [409], just as they are in determining the C-footprint of other industrial products. LCA will include: diesel usage during logging, in haulage [e.g., [410]] and in international freighting; mill energy usage for virgin- and recycled-products; and NOx emissions. Such additional emissions could be high in the current market situation, as wood-product movement occurs from Tasmania to Japan, then to China and other countries, and then sometimes back to China for recycling [411], though that latter move has been limited by some countries in recent years.



MANAGED FORESTS SEQUESTER MORE CARBON

Contrary to popular belief, a managed forest which is regularly harvested and replanted sequesters more carbon (when carbon that is retained in wood products is included) over time than a forest which is not harvested.

Source: from The Changing Face of Australian Forests (DAFF - Department of Agriculture, Fisheries and Forestry – Bureau of Rural Sciences)

(a)

(b)

(c)

Figure 14. (**a**) Pamphlet advertisement by Forest & Wood Products Australia, using graph from Ximenes, et al. [[402] Figure 10] and not from DAFF & BRS as stated in the advertisement—emphasises the temporarily increasing wood-product pool— giving the illusion of increases *ad infinitum*. It ignores the primary forest legacy carbon. (**b**) and (**c**): output from CAR4D for Site-1 in Dean, et al. [47], drawn in 'stacked-area' format: **(b)** ignoring the primary forest carbon (i.e., zero baseline), and (**c**) not ignoring primary forest carbon. Style (b) would be more appropriate to use for plantations on long-cleared land or where logging has occurred for several centuries, but not for the LUC of primary forest conversion to logging cycles.

Forestry is not the only land use where legacy SOC from the previous forest makes a large difference to a carbon footprint. A major difference was found when accounting for legacy SOC when previously forested land was used for grain crops [412]. Those authors considered that inclusion to aid realism, though requiring more work and providing location-dependent outcomes for the land managers. That difference coincides with considering where along the time axis in Figure 9, a particular logging cycle may be; giving a different future drop in SOC depending on when the accounting begins.

5.2. New Regulation Regarding Sustainability

The most logical way to address the lack of scientific knowledge about the environmental impact of intensive forest logging is firstly to acknowledge it and secondly use the precautionary principle. In this vein there is a recent step towards an intent of sustainability by the 'Standards Reference Committee SRC AS/NZS 4708' in a proposed update to forest practices standards. It includes the precautionary principle applied to cultural, economic, environmental (including the carbon cycle) and social sustainability [413]. The proposal was for Australian and New Zealand forestry, by a committee comprised of forest industry representatives, academic institutions, government agencies and unions. It applies to all threats of serious or irreversible environmental damage (in their definition). However, soil carbon is not mentioned specifically, and the emphasis for soil sustainability appears to be on avoiding erosion. With respect to carbon the proposed standard, states:

'The average carbon stock on the defined forest unit shall be maintained or increased over successive harvest cycles.'

and one of the allowable conditions for forest conversion is that it:

'does not destroy areas of significantly high carbon stock; … '

The first condition suggests that there might be a method of forest conversion to logging cycles that is contrary to what normally occurs (compare with Figure 9 and Figure 14.b). The second condition most likely refers to areas where the land is converted to plantation, orchards or long-term non-forest uses. Additionally, as corporate forestry claims sustainability [e.g., [414]] but rarely measures a site's soil carbon prior to logging or in the long-term, then a management activity is unlikely to forecast emission of an area's carbon. Nevertheless, this latter condition would be a significant statement towards forest carbon sustainability if: (a) applied to conversion of primary forests to logging cycles, (b) 'destroy' is interpreted as either rearranged or only partially eliminated, and (c) a list of forest stands with high carbon stock can be created. This latter condition links across to the second part of this review paper where such a list is considered.

5.3. Terminology Hinging on Sustainability

A subtle forestry industry public relations tool is renaming activities to give the indication of more-sustainable operations, such as calling the burning of waste from timber-mills 're-use' or 'recycling' [e.g., [415]]. Other popular name changes are to label forests simply as biomass; the felled or still-standing timber that is not hauled out, as 'residue'; coarse woody-debris and dead hollow trees as 'fuel'; and the word 'harvesting' in place of 'logging' for primary forests (Figure 15) [416]. The original meaning of harvest was to collect from a crop that one has sown. This renaming occurs elsewhere in society too, such as harvesting rainwater and wild seaweed, but rarely to fossil fuels. The change over time can be seen in the journal Australian Forestry, published by Taylor & Francis. Early on 'harvesting' was used only for plantations, which matched with its dictionary definition (Figure 15). One of the first uses of it for primary forests was in the pulp and paper industry c1960, during the controversy over whether or not to leave Melbourne's water catchment as primary forest, and the resolution was for more investment in a public relations campaign, in research, and in logging [417–419]. The number of articles in Australian Forestry including the words 'logging' and 'harvesting' has grown over time (Figure 15), indicating more attention to that part of resource extraction but the growth rate for the use of the word 'harvesting' has nearly doubled that for 'logging' (0.79(0.05) and 0.34(0.06) respectively; standard errors in brackets). The two growth rates

doi:10.20944/preprints202410.0555.v1

are significantly different: the probability of them being the same is P= 3.6×10^{-8}, in a Student's t-test distribution. Their usage within a scientific paper is not mutually exclusive however, for example there are still logging roads and log trucks, not harvesting roads and harvest trucks, but, for example, selective-logging is now often termed selective-harvesting, in primary forests.

The terminology towards sustainability has been more embellished, where it is often claimed that wood-products sequester carbon (and therefore logging helps prevent climate change) [420–426]. Whereas as the carbon sequestration, from gas to solid, was actually performed by the trees.

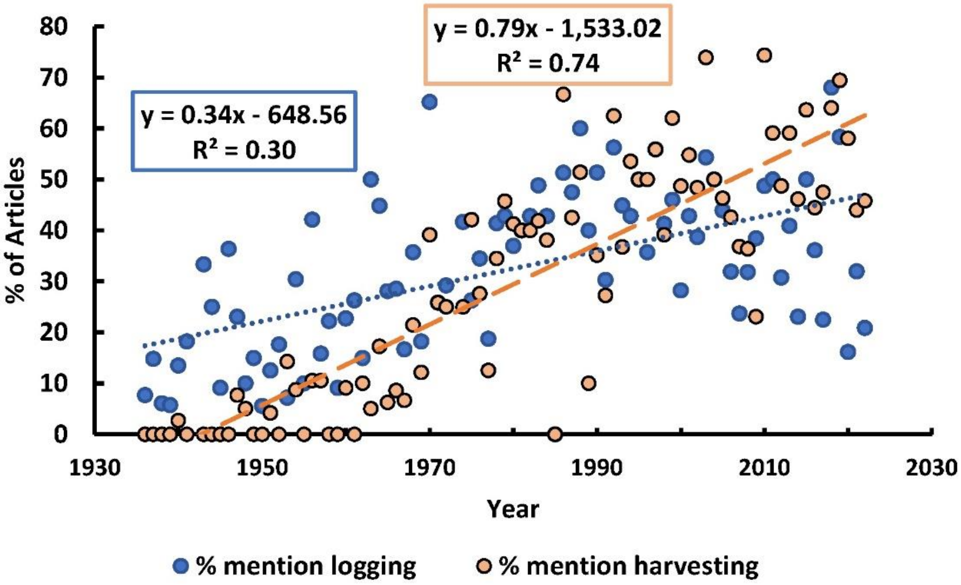


Figure 8. Change in use of the words 'logging' and 'harvesting' from 1936 to 2022, in the journal Australian Forestry. Data were obtained by using the search function on the journal home page and GoogleScholar©. The word 'harvesting' has increased ~twice as much as the word 'logging', since the journal started in 1936. Initially 'harvesting' was exclusively for plantations, then it was increasingly for logging primary forests.

5.4. Product Substitution for Sustainability

Using timber instead of some other building construction materials, including timber from primary forests, has been suggested as being more sustainable, as it has lower carbon emissions—the 'substitution effect' in considering the carbon footprint of logging— but most of these comparisons have not noted products with lower emissions than timber [e.g., [427,428]]. For example, the products most frequently compared against timber are standard concrete (using ordinary Portland cement), steel and aluminium. Whereas lower-carbon-footprint forms of concrete, which can also have superior structural properties (for example), appear to not be mentioned in those substitution comparisons [429–432]. Additionally, when comparing building products, the recycling and durability of steel and aluminium (which reduces their carbon footprint over time) does not appear to be included. Another criticism has been that temporal characteristics of substitution are often not compared in a practical sense [429]. In the 1970s, in several annual reports, the Tasmanian State forest agency mentioned that sales of sawn eucalypt timber were adversely affected by, for example: concrete slabs for housing, particle board, plywood, low-cost imported timbers, low rates of housing construction, and *Pinus radiata* from interstate [e.g., [433]]. This suggests that a trade perspective could still be involved to some degree in some of the comparisons.

An aspect that has frequently been disregarded when comparing the carbon footprint of timber and concrete building materials is that concrete slowly re-sequesters carbon by carbonising (reverting back to calcium carbonate by reacting with CO_2 in the air, in the presence of moisture), and it can be recycled to some degree [434–439]. Conversely, structural timber can only release carbon to the atmosphere, it cannot sequester carbon. A more-recent comparison did include concrete carbonisation (including the peak carbonisation upon building demolition) and steel recycling but still found less emissions when using timber. There was still a point to be considered though, because they had assumed substituting the burning of fossil fuels by burning forestry residue and wood from demolitions [406] but did not compare against using renewable energy such as solar power had been used (instead of burning wood), not even by the year 2216, when newer technologies will be generating base-load power [440]. More comprehensive comparisons are necessary before sustainability by product substitution can be claimed, at least for countries such as Australia with ample renewable energy and relatively recent primary forest conversion.

Science relevant to climate-change-mitigation got entangled in corporate strategy in the 1970s, when a forest industry PR organisation in the USA promoted the phrase ‘trees are the renewable resource’. That increased public approval of the industry from 34% to 55% [441]. Forest & Wood Products Australia recently promoted the trade marked brand ‘The Ultimate Renewable’ for both plantation and native forest timber under the one umbrella, with social media and online free TV advertisements [442]. The only detail on renewability and sustainability offered on The Ultimate Renewable website is: ‘…native forest managers renew their forests through natural or artificial reseeding’, and readers are referred to check forest certification: all State-level logging agencies are certified to some degree [443,444]. Part of the proof of renewability lies in the scientific data on life-cycle assessment of the products, which Hansen and Juslin [176] state should be provided by corporations as part of their environmental responsibility. This would lead back to the requirement for more scientific detail.

A recent review of the dependence of global warming potential of building products found that the details of the life-cycle assessment that were included, such as through modelling parameters, made a significant difference to the balance [445]. The ISO 14067:2018 standard for life-cycle analysis mentions that land use effects on soil carbon should be included, though not necessarily so for the original forest conversion. A time span of 90% of the time taken for SOC to reach equilibrium after land-use-change, is recommended for assessing the upper soil profile under agricultural crops. Modelling shows that this approach makes a difference to the findings of life-cycle assessment of products [446,447]. Following this ISO 14067:2018 recommendation and using the relaxation time (mentioned earlier), then the modelling duration should be at least ~1,000(±500) years, for the full soil profile for forest conversion from primary to secondary forests with intensive management (when at least 40% of biomass is removed per logging cycle).

5.5. Sustainability with Regards to Soil Carbon

Stewart [448] theorised that forestry can partake in climate change mitigation, through, for example, maintaining optimum tree cover, extending wood-product half-life, and not burning logging residue; and he mentioned that whole-tree harvesting would deplete soil carbon. Harmon, et al. [269] mathematically verified these postulates, with forecasts that the conversion of primary forests to logging cycles decreases the long-term C stock, even when including the carbon in wood products. Later modelling confirmed the concerns first voiced in the late 1970s: long-term, intensive logging, especially when accompanied by slash burning or removals for bioenergy, depletes forest SOC, and nitrogen and phosphorus in the soil [47,239,240,449–453].

Just before that safeguard cautioning in the 1970s, the Australian Government published an environmental appraisal of the woodchip (pulpwood) industry, including for carbon emissions from soil:

‘Some concern has been expressed about the environmental effect of increased carbon dioxide levels in the atmosphere as a result of increased soil respiration associated with clear felling. However, this effect is negligible both in relation to the magnitude of the grand cycles of carbon dioxide and oxygen in and across the ocean surfaces, and by comparison to the similar effect of the large-scale burning of fossil fuels in the past century.’ [106]

This appraisal missed the issue that it is not so much the change in soil ‘respiration’ but the long-term-average reduced input of carbon to the soil from biomass (Figure 9).

Sustainability for industries using biological systems was first mentioned in forestry, in the 18th century, describing sustained timber yield over consecutive logging cycles [454]. In the 1960s the concept was expanded by the US Forest Service to include 'all the benefits from the land' and some foresters concurred that sustainability had to incorporate more forest attributes than wood volume alone [455]. In 1992 the Australian government concluded that sustainability for forestry must include 'keeping options open for the future and adopting a precautionary approach to potentially irreversible consequences of particular uses.' However, in the early 1990s there was little evidence of Australian forestry interpreting the meaning of sustainability to include the land's soil carbon stock, though that must be part of true sustainability [456,457]. Consequently, it was considered that claims of ecological sustainability in Australia 'should be viewed with considerable scepticism' [456].

Monitoring of soil carbon by the forest industry is not currently a legal requirement, only a suggestion. It was not until the end of the 21st century that soil carbon was measured in the context of carbon accounts for corporate forestry. A review of eight technical standards for ascribing the carbon footprint of wood products used for building [458] found that five did not mention soil and the others included it only if there was a land use change, and one of these (ISO 14067:2013) mentioned ongoing research into change in soil carbon. Conversion of primary forest to secondary forest, is often not regarded as a land use change. Additionally, there is low precision on values for forest SOC stocks, and consequently more uncertainty for ΔSOC, including the modelling of the impact of climate change feedback on it [459,460].

One protocol has a more comprehensive consideration of sustainability: SNV-REDD+ [461]. It requires measurement and comparison of SOC but the time period is not mentioned and the maximum depth of analysis is only the IPCC's 0.3 m [280]. Reference is made to the IPCC 2006 standards [462], which were intended primarily for national-level accounting, where the default time period for SOC change is only 20 years— much shorter than relaxation time for the full soil profile.

For calculating the carbon footprint of bioenergy, incorporation of SOC dynamics has been recommended [463–465]. Modelling ΔSOC for bioenergy, from agricultural crops, using longer time spans (e.g., up to 200 years) and more depth of the soil profile (down to 1 m) made a 'huge impact' on the results [466]. A global forecast for using bioenergy from forest biomass to replace fossil fuels, which included SOC to 1 m depth and typical SOC half-lives, found that the global pool of SOC would significantly decline and that there was negligible climate benefit [453]. Alternatively, secondary forests and plantations are sometimes compared against a benchmark of 0 Mg ha^{-1}, with consequently no negative effect on SOC mentioned [402,467,468]. Parameterisation of soil carbon models from measurement of SOC taken while legacy carbon is present, but incorrectly attributed, can induce inaccurate forecasts by: suggesting low carbon stocks of the pre-disturbance forest, short half-lives of the slow SOC pool, and too-high SOC values in the disturbed forest [469].

To date, knowledge on the fate of soil carbon upon conversion of primary forests to logging cycles has not been reflected in policy, but there are indications of a slow and partial recognition of its importance. In a review of which pools are measured in which financial carbon accounting protocols, the fate of soil carbon is either listed as ongoing research or assumed to be zero change [458]. Early in the 21st century, for the routine updates of one national forest inventory (namely Sweden), it was recognised that more information on the existing soil carbon stock was needed so that any change could be included in reports [470]. Around the same time, in a review on the sustainability of logging, with a focus on building materials, attention to the fate of soil carbon is recommended:

'For harvested wood products (HWPs) there are additional considerations regarding the impacts associated with forestry operations, which may include issues associated with land use change, as well as impacts upon soil carbon and the ecosystem services that forests provide, in addition to harvestable timber. … Because up to twice as much wood material is removed from the forest when timber is used in buildings, it is important to include the fate of these wood by-products in any study (i.e., is this material used for energy, is a proportion left in the forest as soil improvement?). If surplus wood is left in the forest after harvesting operations, it is essential to consider the fate of this material with respect to GHG emissions.' [471]

Although the science of reported climate change effects has not been examined in this paper, it is relevant to mention one item here. Climate change is forecast to increase fire frequency in southern Tasmania [352]. Any carbon storage that was increased earlier due to the direct effect of higher atmospheric CO_2 concentration [e.g., [472]] would be lost by the future fires. Repeated burning is correlated with reducing a forest to a savannah or shrubland state, and reducing a savannah to grassland [473–476]. This is the trend that one can expect as climate change progresses. Soil nutrient losses are higher with more intense or higher frequency fire, and feedback leads to a more fire-prone ecosystem and reduced soil nutrients [477]. Over several millennia the process can dramatically alter the soil type, for example by reducing clay content by eluviation [477], and a higher proportion of more labile SOC [478]. This will reduce site-quality and therefore long-term C storage, which in turn constitutes positive feedback to climate change [e.g., [479,480]]. Although the 30% decline in SOC stock in Tasmanian mixed-forests and rainforests was modelled by correlation with climate changes occurring by 2100 [301], in that modelling SOC was a proxy for biomass C, assuming a positive correlation [238,239], with the decline based on the biomass's response to climate change. Owing to the long relaxation time for SOC (described above), the time for the forecast decline in SOC [301], would most likely be manifested long after the year 2100, and not within a century as reported there. It is the biomass that could decline by 30% by 2100 and the SOC would follow but much more slowly.

5.6. Bioenergy and Sustainability

Bioenergy is one of the few topics in the forestry realm where there could be said to be a true debate, because there is an ample amount of scientific literature representing the for and against arguments, and the topic is discussed between politicians [481–483]. Nevertheless, regardless of the against arguments, the trade in forest carbon for bioenergy is already well underway in many jurisdictions.

The claim of sustainability when burning forests for energy relies on timber regrowing and reabsorbing emitted greenhouse gasses [278], whereas burnt fossil fuels don't become fossil fuels again as readily. However, Ter-Mikaelian, et al. [484] explains how the claim that this generally produces climate-change-mitigation, is an oversimplification. Some obvious contraindications are:

(1) Any substitution for fossil fuels may, at least in part, instead be substituting for renewable energy such as solar, wind-power or hydroelectricity. (Which is a point often ignored, e.g., Gustavsson, et al. [406].)

(2) The effect of logging primary forest on soil carbon stock has often been discounted, though it is sometimes acknowledged that primary forests should not be used for bioenergy production [e.g., [485]].

(3) The time required for forest regrowth is longer than currently required for climate change mitigation. Instead, algae have a growth duration (hours to days) that is more relevant to climate change mitigation than is that of trees (years to decades, to centuries) [486].

(4) If forest bioenergy greatly decreases energy prices then some people may increase their energy consumption.

(5) As described above, sustained yield from conversion of primary forest to secondary logging cycles, cannot occur. Similarly therefore, most of the burnt biomass is not recoverable.

(6) Product substitution that provides avoided emissions is sometimes mistakenly treated as carbon removal from the atmosphere [487].

If the balance being considered is only that between the emissions from burning fossil fuels and those from burning forest biomass, the review by Giuntoli, et al. [488] found that the balance depends on details, such as the market response to increasing wood price. The replacement ratio may not be 1:1 for various reasons [488]. There are also other industrially driven factors such as increased logging due to bioenergy profits, logging intensification, average stand-age reduction, increased logging frequency, change in lumber use towards fuelwood, and change in species planted [488]. Regarding 'change in lumber use' for example, Giuntoli, et al. [488] noted that logging residues such as bark and stumps, may contain impurities and therefore not be of suitable quality for fuel pellets, and so pulpwood will be burnt instead. The authors of that review also found that the support for pro-bioenergy from forests relied on better use of forests and increasing forest area, but the evidence for these is weak. 'In general, our review finds that all studies that project a large role of forest bioenergy in climate change mitigation rely on too optimistic assumptions, at times even unrealistic.'

In paper mills, ventures to increase wood particle recovery, decrease toxic effluent, and increase fossil-fuel substitution, have often led to either equivalent or increased GHG emissions [e.g., [489]]. Notably, from a climate-change-mitigation perspective, Mathieu, et al. [490] found that it was better to burn waste paper than to place it in landfill, though that burning was used to substitute for fossil fuels, and substitution using cleaner energy was not assessed.

Stewart [448] discussed the pros and cons of bioenergy from forests in terms of fossil fuel substitution but renewable energy such as wind or solar was not as popular in the 1970s so he may not have been aware that bioenergy might inadvertently be substituting for those too. He recommended that the bioenergy feedstock come from used rather than fresh, timber and paper. If the recommendations of Stewart [448] had been implemented or empirically investigated, then the forest industry would have been able to mitigate climate change more successfully.

Since the short-cycle eucalyptus pulpwood output from tropical countries began to dominate the global pulpwood market, the demand for pulpwood from Australia has declined, and in its place, proponents of the forest industries and the Australian government have considered using native forest residues for bioenergy, and have even referred to it as 'green' hydrogen [491–495]. Whereas actual 'green' hydrogen can be produced by desalination of seawater using solar or wind power [e.g., [496]].

The greenness of green hydrogen depends on the carbon footprint of its production, and if derived from biomass then it depends on either: (a) pyrolysis or gasification of the biomass [497–499], or (b) the electricity derived from biomass burnt to hydrolyse water to make the hydrogen. Forest residues from native forests in Australia include non-sawlog biomass, such as non-target tree species in clearfell logging sites (such as rainforest species), pulpwood, sawmill offcuts, sawdust and pulp-mill residues.

Since the era of Stewart [448], the Australian forest industry has supported intensive forest usage:

'Popular concern in some quarters that timber harvest or other professionally acceptable forest management practices may seriously deplete the forest soils of nutrients has no scientific basis. The harvest of trees and even limbs at infrequent intervals removes relatively low amounts of soil nutrients per hectare on an annual basis- far less than the suburbanite removes by his annual raking and disposal of leaves. Nutrients are replaced in forests that have been cut by decomposition of material that remains and through other natural processes. Only the most remote likelihood exists that soils of native forests would be significantly depleted by normal levels of timber harvest (possibly excepting sandy soils of very low nutrient status and where phosphate fertilizer application is already coming into routine use). Further, numerous scientific observations by soil specialists lend no support to assertions that forest soils may be irrevocably depleted of nutrients by clearcut harvests at reasonable intervals. In such a remote case that soil deficiency did occur, it could readily be corrected by application of fertilizer as is routinely done in modern agriculture.' [106]

The Australian government has most-recently announced that burning native forest residues for bioenergy will not be considered renewable energy, but that curtailing of emissions could be reversed by an alternative government, or by direct gasification of forest residues [500]. Diverse opportunities exist for energy production for humans without having to burn trees or fossil fuels, such as hydrogen production by solar-powered catalysis of water splitting, or with more efficient energy transduction: from genetically modified micro-algae [501–503].

Advocates of bioenergy also appear to rely on the claim that forests are 'sustainably managed'. However, that characteristic is usually assumed rather than proven, for example:

'Holistic assessments show that forests managed according to sustainable forest management principles and practices (around one billion hectares globally, of which over 420 million hectares are certified; UNECE FAO, 2019) can contribute to climate change mitigation by providing bioenergy and other forest products that replace GHG-intensive materials and fossil fuels, and by storing carbon in the forest and in long-lived forest products.' [504]

In that review of approaches to bioenergy assessment, entitled 'Applying a science-based systems perspective…', Cowie, et al. [504] set out to clarify the 'significant role that bioenergy can play in displacing fossil fuels'. They cited a review by Achat, et al. [505]. That review found that forest soil carbon declines when removing harvest residues for bioenergy. However, Cowie, et al. [504] did not refer to that mention of reduced soil carbon, but to its mention of the potential drop in forest productivity with excessive residue removal.

Examination of UNECE [506], which was cited in Cowie, et al. [504] for the >420 million hectares 'certified' as sustainably managed, showed that the certification of 424 Mha is by the Forest Stewardship Council (FSC) and the Programme for Endorsement of Forest Certification (PEFC). Such certification schemes recommend conservation of general soil attributes (e.g., by avoiding erosion and compaction) but do not require maintenance of soil carbon stocks [507]. Therefore in the review by Cowie, et al. [504] there was no evidence of sustainability of soil carbon stocks, although they claimed general sustainability over very large areas, and therefore on their logic, they also claimed that climate change mitigation via bioenergy was practicable from those areas.

The reliance on certification schemes is echoed in Aguilar, et al. [481] where the Sustainable Biomass Program is mentioned as a certification system relying in turn on systems such as FSC and PEFC. They studied sustainability but over only 12 years. Prudently, they emphasised this empirical limitation of only a 12-year condition analysis (i.e., too short a duration for representative and measurable ΔSOC).

It was difficult to trace the one billion sustainably managed hectares mentioned in Cowie et al. (2021)'s, in UNECE [506] but there was a 1.7 billion hectares. This was forests in the 'UNECE region' that is part of the UN's Sustainable Development Goal for 2030, which is designed to be part of the intended 'circular economy' and part of the UN's aims to address climate change and to 'regenerate and sustainably manage natural resources'. This area increased from 1.7 billion hectares in 2020 to 2.1 billion hectares in 2020 (i.e., 43 and 54% of the world's forests respectively) [508,509]. However, Siry, et al. [509], in reference to the original 1.7 billion hectares, refer to the management intensity and protection as 'moot' (i.e., questionable) because there wasn't enough data to suggest good management. The mere existence of forestry plans does not mean sustainability, and indeed the area of actual sustainable management globally (a subset of the area with management plans) could not be assessed in 2010 as there was insufficient data on definitions, criteria and assessment methods [510]. The management of the 2.1 billion hectares simply refers to land under nationally- or community-approved forest management plans that last for at least 5 years [511]. It's likely that the sustainability part of those plans depends on the blank slate (benchmark of 0 Mg ha^{-1}) concept for SOC, or that they don't consider SOC. But the initial starting conditions must be considered when assessing the carbon balance of forest management [512]. Thus the large area of sustainably managed forest, in Cowie, et al. [504], does not exist. That removes one of the bases for their logic in claiming that sustainable forest use exists, from which bioenergy can be extracted.

The emphasis on sustainable management is echoed in UNECE [506]:

'The EU's revised Renewable Energy Directive (RED II, 2018/2001/EU) entered into force in December 2018 (European Commission, 2019). …Specific to forest biomass, RED II notes that biofuels, bioliquids and biomass fuels produced from forest biomass should minimize the risk of unsustainable practices. …To ensure the sustainable harvesting of biomass, RED II requires the regeneration of harvested areas, special attention for areas designated for protective purposes, the conservation of biodiversity, and the tracking of carbon stocks. Thus, primary-sourced forest biomass should be harvested following sustainable forest management principles implemented through national laws or best management practices at the level of sourcing areas. Operators should take appropriate steps to minimize the risk of using unsustainable forest biomass for the production of bioenergy. … Wood is a cost effective and potentially renewable source of energy, which can supply a big share of global heat if the natural resource base is sustainably managed, including the environmental and social dimensions.' [506]

The references to claims of actual sustainability appear to be circular in that the UNECE cite Cowie, et al. [504] and vice-versa:

‘The sustainability of wood-pellet production in the United States southeast destined for the EU and the United Kingdom continues to be debated in public media and other forums (e.g., Popkin, 2021; Hodgson, 2021). Science-based reports assessing the integrity of carbon pools from forests used to procure fibre for pelletization suggest that carbon stocks are not being negatively affected, and new demand could contribute to the growth and regrowth of wood fibres (Aguilar et al., 2020; Cowie et al., 2021).’ [513]

The UNECE reflects on different regional criteria regarding sustainability of forest management and lists different criteria for inspiration [514]. Among them are the Montréal Process and the Forest Europe criteria. The 2015 version of the Montréal Process indicators separate soil conservation (‘resource protection’) into Criterion 4, and carbon conservation [515] into Criterion 5, both of which are qualitative. Criterion 5 merely reiterates the forest industry initiatives, though more mildly by using the word ‘may’, that wood products may be more sustainable than ‘manufacturing products that have significant carbon footprints’ and that forest biomass may ‘offset the need to burn fossil fuels’ [515]. The demand is that the contribution to global carbon cycles is retained [516]. It does however, state that the criteria will be periodically reviewed to reflect advances in knowledge.

The Montréal Process Technical notes provide more detail. For soil in Criterion 4 each country’s report is merely obliged to summarise how they met best management practices and their efforts to monitor compliance [516]. For carbon in Criterion 5 forest managers are referred to the IPCC and UNFCC for guidance on assessment [516]. For both criteria countries are told that ‘Useful data may be obtained from government, university, industry, and research organisation sources.’. This again seems circular for assessing industry’s carbon footprint. It fundamentally relies on accepted standards of practice and in these there is no indication of a requirement to measure or model soil carbon over a timescale appropriate to its half-life or its depth distribution. Likewise, the Forest Europe criteria reflect the requirement that the contribution to the carbon cycle be maintained:

‘Criterion 1: Maintenance and Appropriate Enhancement of Forest Resources and their Contribution to Global Carbon Cycles… 1.4 Forest Carbon. Carbon stocks and carbon stock changes in forest biomass, forest soils and in harvested wood products. …

Criterion 2: Maintenance of Forest Ecosystem Health and Vitality… 2.2 Soil condition. Chemical soil properties (pH, CEC, C/N, organic C, base saturation) on forest and other wooded land related to soil acidity and eutrophication, classified by main soil types’ [514]

The paucity of ΔSOC measurement in relation to forestry and bioenergy, and the consequent absence of a thorough check on SOC sustainability, has had a major effect on outcomes. There is hope for a check on soil carbon sustainability, if the Forest Europe criteria are enforced over long-term effects: measurement or modelling of soil carbon stocks over appropriate time scales may then be undertaken.

5.6.1. Carbon Modelling Example

One of the few models of the bioenergy carbon footprint that includes soil organic carbon and decomposition of root and aboveground biomass was in an assessment for a power station in New South Wales (NSW) in Australia [517]. The question was whether biomass should be sourced from native forests for bioenergy. The author submitted the report as an independent expert witness while working for the Department of Primary Industries [518]. That submission formed a counterbalance to those from the government’s Environmental Protection Authority and the local Council. Cowie [517] claims that the burning of forest biomass produces less greenhouse gasses compared with burning fossil fuels and will cause net carbon sequestration over 80 years. The claim is stated to rely on ‘sustainably managed forests’ and science:

‘Switching from coal to sustainably-harvested woody biomass as an energy source reduces atmospheric CO2 over time scales relevant to climate stabilisation. … Sustainable forest management ensures that annual biomass removals do not exceed annual forest growth. The forest carbon stock is therefore stable; the same quantity of CO2 is released as is sequestered by the forest each year, so there is no net transfer of carbon from the forest to the atmosphere.' Cowie [517]

The majority of the biomass (70%) is to come from 'plantation and native forest harvest residues' [517]. That native forest biomass is from two sources: wood that normally would be pulpwood from native forests, and trees from 'land clearing and other approved activities', 44% and 56% by weight, respectively [519]. The single largest component (56%), land clearing, is usually deforestation for livestock farming. The second largest component (44%) is from native forest logging (public and private forests, 60% and 40% respectively, HRL Technology Group [519]). Cowie [517] modelled the carbon accounts for one logging cycle of tall open native forest from 2020 to 2100 using the computer software, FullCAM [308,520].

In the modelling by Cowie (2021) her Figure 2, shows the soil carbon drop by a net 20% during the 100 year logging cycle (from 45 Mg ha^{-1} to 36 Mg ha^{-1}), and it does not recover, regardless of whether or not biomass is extracted for bioenergy. Therefore, although not shown in her report, after two rotations it would be lowered further, and so on. Considering the timescales taken to recover SOC (Figure 10 above), the proposed logging and burning, according to the modelling of Cowie [517], will create a long-term carbon debt. However Cowie [517] overtly states that the only requirement for sustainability is that biomass is replenished. Thus, the SOC loss appears to be ignored.

The biomass stock, as modelled in Cowie [517], recovers after about 75 to 80 years, but that is greater than the usual logging-cycle-length for that region of 50 to 60 years [521]. In the normal scenario of integrated harvesting 'the crown, stump, bark, leaves, small branches etc. are left in forest for biodiversity and forest health' [519]. This would not occur under bioenergy extraction and therefore, 'forest health' may suffer too, which would be a second reduction in sustainability.

6. Benchmarks for Conservation Directives

There are sometimes rebuttals between authors of science whose publications have findings that support more-conservative use of nature instead of prevalent logging methods and vice-versa, for example: Poynter and Ryan [522] vs. Keith, et al. [523] and Lindenmayer and Sato [524]; Dean [525] vs. Moroni, et al. [526]; and Sterman, et al. [527] vs. Prisley, et al. [528]. In this section I analyse detail to check for validity, and highlight sources of likely discord in a recent pro-conservation paper by Sanger and Ferrari [529], related to climate change mitigation and biodiversity conservation. Hearsay suggests that paper may have been subject to controversy, although it is not yet reported in the scientific literature. The paper broached, for the first time, a carbon stock competition in Tasmania and claimed to win it, at a particular site:

'The current study represents the highest measured carbon stock for a forest in Tasmania (Table 1), with measurements well over previous studies.'

In Sanger and Ferrari [529] the authors also claimed to have undertaken the most comprehensive study of carbon stock assessment in Tasmania to date.

This paragraph summarises some of their study's pitfalls, with details provided below. For example, it did not compare their tally with existing reports of total carbon for other sites in Tasmania as the authors said that to the best of their knowledge they did not exist. To gauge earlier work on Tasmania's forests, their study did compare their biomass-only tally with that of some other studies, but some were missed. More importantly from a science perspective, comparisons were not placed on a level footing. Also, incorrect statements were made about which carbon pools were studied in some reports. They undertook some measurements in more detail, by tree climbing to measure the girth higher up the trunk and on branches for some of the trees in their study plot, rather than relying as heavily on taper formulas and allometric equations as in many other studies. However, their method of girth measurement that was published in public internet media, showed unnecessary flaws compared with the standard technique. Additionally, selection of some of the allometric equations they used appears unfounded, and their soil carbon analysis was not comprehensive compared with other contemporary measures, which is contrary to one of their claims. Overall, some standard scientific protocols were not followed, which contributed to their study not being an example of robust science, nor of the best available science. To provide a carbon comparison on a level footing, detail is provided below of recalculation of the carbon stocks represented in Sanger and Ferrari [529] and several earlier studies. Another reason for presenting such a carbon comparison here, is that the process in itself, provides valuable, general insights into some aspects of the use of science for forest carbon accounting more generally.

6.1. Measurement of Carbon Stocks by Using Proxies

Studying the sums or changes in the carbon stocks of Tasmanian forests, has only occurred during the last forty years, roughly, and sometimes as a biproduct of a different investigation [e.g., [368]]. Therefore, actual carbon data are scarce. Different attributes of nature pique scientific interest at different times, depending on society's needs, such as for resource extraction, atmospheric amelioration (as currently for climate change mitigation), drinking water runoff, or interest in a specific animal species. But for whatever forest attribute is in vogue, there are usually other data that can be transformed or used as a proxy. For example Johnson [530] noted that data on the change in soil carbon with forest management was often in reports that focussed on other nutrients. For carbon in trees, a long-standing proxy for carbon is merchantable timber volume, to which a 'biomass expansion factor' (BEF) is applied to include the carbon in the unmerchantable, aboveground parts of a tree [531]. This method misses some carbon in other parts of the forest stand however, if some species felled during logging contribute to forest biomass but are unmerchantable (such as some rainforest species amongst eucalyptus species), or if their merchantability varies with market demand. It also has a high error margin unless the BEF is tailored to each stand, which is rarely done.

In their carbon account of Tasmanian wet-sclerophyll forests, Moroni and Lewis [532] used a biomass expansion factor of 1.46, and added soil carbon values from similar soils elsewhere. That BEF value came from Snowdon, et al. [533], where it was derived by reassessment of earlier reported data for Australian native forests in general [e.g., [534]]. Many of the allometric ratios presented in the review by Grierson et al. [534] were based on the whole of the stem and not just on the merchantable portion, so the BEFs should be higher than theirs if based on the merchantable part only. To account for that difference, in their assessment of the C stock for the whole of State forest in Tasmania, MBAC [535] multiplied by an expansion factor of 1.25, to represent the wood in the entire stem, before applying the 1.46 expansion factor to get to the whole of the aboveground mass of the tree, and then a third expansion factor of 1.29 to account for unlogged trees in the forest stand. However, most of the data in reports such as Grierson et al. [534], were from quite young trees, e.g., less than 100 years old and so the BEFs for mature forests should be lower [536]. This is because relatively more biomass is in the trunk for mature forest trees and therefore more of the total is merchantable (unless senescence is well established). Consequently, Snowdon et al. [533] used a lower limit of 25 years for their data collation for BEF. Possibly these two biases of whole-stem and mature-age cancel out to some degree. The average BEF given in Grierson et al. [534] for the most common eucalyptus species in the tall wet-sclerophyll forests of Tasmania, namely *E. regnans*, *E. obliqua* and *E. delegatensis*, was 1.32 but the average age was only 43 years. Limiting this to stands older than 24 years gives an average BEF of 1.28 (and the average age of those trees as 61 years), but that is based on entire stem volume and not just the merchantable part of the stem. This is lower than the 1.46 used in Moroni and Lewis [532], but there's insufficient information to give a definitive adjustment to it. For tall-open eucalyptus forests, in the Farm Forestry Toolbox, a computer program designed for tree growers and merchants in Tasmania [537], for expanding from whole stem biomass to aboveground biomass, the default BEF is 1.313. Alternatively, based on weights from destructive sampling of small, Tasmanian *E. obliqua* (average DBH of 0.683 m for 44 trees) Ximenes, et al. [[538] Table 4 & Table 10] reported data yielding BEFs of 1.43 and 1.58, depending on the merchantability of stem wood in the crown for pulpwood. They also showed that in that particular stand, the BEF might increase for larger trees. BEFs can range from 1.25 for a high-quality stand with sawlog and pulpwood extraction to 5.0 for a low-quality stand with sawlog usage only [[533], p xiv]. Therefore, the value of 1.46 might be sufficiently conservative and appropriate to use for average, larger trees. In general however, the use of BEFs will introduce large error margins for carbon accounting unless empirically derived for a particular forest stand, as in Ximenes et al. [538]. Even for different species within a stand, the BEF can vary significantly [539].

An early eucalypt sawlog inventory (1922) for tall forests in Tasmania measures 32 transects covering 124 ha of the Florentine Valley [540]. But that dataset is difficult to compare with more-contemporary surveys as the BEFs cannot be estimated without knowing the historical merchantability criteria for eucalyptus sawlogs. The more-contemporary BEF that was applied by Moroni and Lewis [532], caters for both sawlog and pulpwood being extracted.

After that 1920s sawlog inventory, in the pulpwood era records were kept by Australian Newsprint Mills during logging of the Styx/Florentine pulpwood concession [541]. In the midst of that activity, a detailed inventory of small plots in the Styx, Florentine and Tyenna valleys, in terms of tree species, sizes and ages, was undertaken to show related forest types on sites of different productivity and in different stages of ecological succession from mixed-forest to rainforest [55]. In that exposition there were three plots of mixed-forest and two of rainforest. Those data included girths of trees and were therefore amenable to carbon inventory, because allometric equations based on DBH can be applied, without relying on an expansion factor. The highest biomass representatives were selected from each of the two main forest types, to provide a comparison with total C found in Sanger and Ferrari [529] [where it was claimed they had the highest carbon reported in Tasmania], and to provide another datum for the mixed-forest vs. rainforest comparison (detailed above). The mixed-forest stand was 0.149 ha by Road 10 in the Florentine Valley and dominated by even-aged *E. regnans*, older than 300 years. The rainforest was ‘transect A’, 0.116 ha by Road 7 in the Florentine Valley and some of its trees were older than 500 years. A range of allometric equations was tested to see what difference they made to the totals (Table 2). The largest rainforest trees in the two stands were assigned 25% senescence, regardless of whether or not the allometric equations already had senescence incorporated (to ensure conservative carbon stocks). To get 20% and 15% senescence in the eucalypts, the stand age in Eq. 8 of Dean, et al. [307] was set at 317.49 years and then 299.575 years, respectively. In the final comparison with Sanger and Ferrari [529], the eucalypts were assigned 15% senescence, because although there was no crown loss, the crowns were ‘stag headed’, which indicated some demise compared with being in their prime. Although not stated in Gilbert [55], his description suggests the API phototype for the mixed-forest plot would be E1c.M+, and that the rainforest plot was M+.

Table 2. Carbon stocks (Mg ha^{-1}) to 4 significant figures, of live biomass [including roots, root/shoot ratio= 0.15]. Based on data in Gilbert [55] for: (**a**) mixed-forest by Road 10, Florentine Valley; and (**b**) rainforest tree species in the same mixed-forest on the first row, and on the second row: rainforest by Road 7, Florentine Valley. Different allometric equations for understorey go from left to right across the table: ‘und’ in (a) and (b); and go from top to bottom down the table for *Eucalyptus regnans*: ‘Er’ in (a). Allometric equations were sought from the following publications: ‘D2020’= Dean, et al. [92] with the root C corrected as mentioned in the main text, Eq. 7; ‘D2003/2006’= Dean, et al. [307] or Dean and Roxburgh [88] which are equal to within 4 significant figures for this forest stand; ‘D2003’= Dean, et al. [307]; ‘D2011’= Dean, et al. [542] with half of the sums corresponding to incorrectly adjusted general rainforest tree equation in Keith, et al. [124] to get temperate rainforest trees (my error, not theirs); ‘D2011 corr.’= Dean, et al. [542] corrected; ‘K2000 temperate’= temperate rainforest from Keith, et al. [124]; ‘K2000 general’= general rainforest from Keith, et al. [124]; ‘$\text{new}_{\text{rainforest\_C}}$’= new rainforest allometric equation introduced in this text (Eq. 11).

(**a**)

und. *Er*	K2000 general	K2000 temperate	D2003	D2011	D2011 corr.	$\text{new}_{\text{rainforest\_C}}$
D2020	1414	1629	1464	1272	1398	1263
D2003/2006 20% senescence	1305	1520	1355	1163	1289	1154
D2003/2006 15% senescence	1352	1567	1402	1210	1336	1201

(**b**)

rainforest species in forest type	K2000 general	K2000 temperate	D2003	D2011	D2011 corr.	new$_{\text{rainforest\_C}}$
mixed-forest	220.0	306.9	240.2	162.5	213.5	158.9
rainforest	608.3	849.1	699.2	453.8	599.4	451.4

The different allometric equations used to produce Table 2 are explained in the next section. The root/shoot ratio used was 0.15, throughout, as Gilbert [55] noted that the trees were mature.

6.2. Suitable Allometric Equations for Carbon in Tree Biomass

Root mass is rarely measured directly, as it is usually more difficult for people to measure things below-ground than aboveground, similar to the difficulty with measuring higher up a tree (section 5.3). Consequently, tallying carbon often relies on assumed root:shoot ratios. For a given environment, as a tree matures, this ratio usually decreases. Throughout many carbon assessments in Tasmania a root:shoot ratio of 0.25:1 has been assumed, though the lower one of 0.15:1 was modelled for mature trees [307], and the even lower one of 0.136:1 is the default for tall-open eucalyptus forests in the Farm Forestry Toolbox [537]. The value of 0.15:1 is an average of relevant literature values [67]. In tall-open forest, through a complex empirical and modelling process for one individual mature understorey tree a root:shoot ratio of 0.18 (±0.05):1 was found [67]. The data ranges suggest that in mixed-forests, 0.15:1 could be a suitable root:shoot ratio for both mature eucalypts and understorey rainforest trees. Therefore, it has been adopted in this report and, for the purposes of comparing carbon stocks, we convert other people's reported values to that figure, where possible.

The allometric equations for *Eucalyptus regnans*, from Dean, et al. [307] and Dean and Roxburgh [88], only produced a difference in stand-level C stocks for living biomass at the 5th significant figure, (e.g., 1201.2 and 1201.4 respectively, for the 15% eucalypt senescence scenarios with the new rainforest equation, Eq. 11) and therefore they were combined into one row in the table. The equation for *Eucalyptus regnans* in Dean, et al. [92] was specific to their study site in that the upper limit was in terms of the maximum live tree size observed and its degree of senescence. However, in production of that equation I had made the mistake of multiplying the root biomass twice by 0.47 instead of once, to get carbon. Therefore, the carbon stock reported in Dean, et al. [92] was a little lower than it should have been, and it requires the following multiplier before use:

$$(1 + 0.15) \div (1 + (0.47 \times 0.15)) = 1.07426436 \qquad (5)$$

and the corrected allometric equation in full becomes:

$$Eregnans_C = 1.07428436 \times 3.394142DBH^2\{1/[1 + \exp(5.081129DBH - 27.68206)]\} \qquad (6)$$

$$= 3.646206DBH^2\{1/[1 + \exp(5.081129DBH - 27.68206)]\} \qquad (7)$$

where *Eregnans_C* is mass of carbon for the whole tree in Mg and *DBH* is in m.

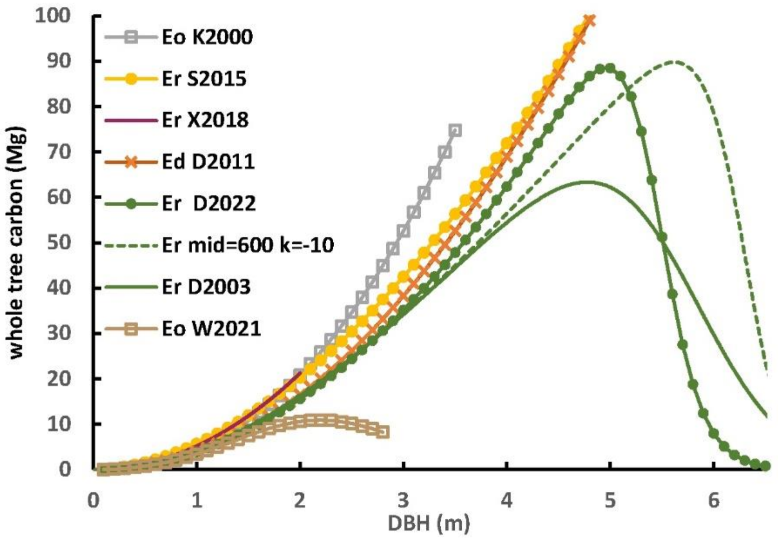
The allometric equations for *Eucalyptus regnans* in Dean, et al. [307] and Dean and Roxburgh [88] differ only for stem volume, with more data being used to derive the equation in the newer report. The older equation [307] is:

$$V = Vol\_max \times \left(1 - \left\{1\Big/\left[1 + \left(DBH + 0.01\big/_{DBH\_mid}\right)^k\right]\right\}\right) \qquad (8)$$

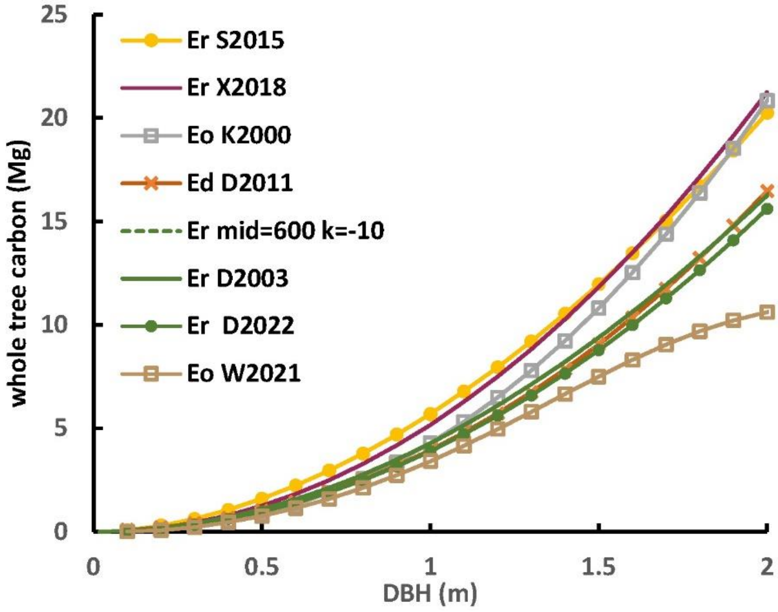
where *Vol_max*= 380 m^3, *DBH_mid*= 4.3 m, *k*= 2.57, *V* is the stem volume in m^3 and DBH is in m, and the newer one [88] is :

$$V = Vol\_max \times \left(1 - \left\{1\Big/\left[1 + \left(DBH\big/_{9.2}\right)^2\right]\right\}\right) \qquad (9)$$

where Vol_max= 1100 m^3, V is the stem volume in m^3 and DBH is in m. In both cases the equations for the other tree components and a user-adjustable senescence in the form of a two-parameter sigmoid function, are given in Dean, et al. [307], with the influence of growth parameters for different sites explained in Dean, et al. [235].



(**a**)



(**b**)

Figure 9. Comparison of different eucalyptus allometric equations, shown at two scales, from (**a**) larger to (**b**) smaller trees. For the equations that didn't include root mass, it was added with a root:shoot ratio of 0.15:1. Equations' species and references: 'Ed 2011'= *Eucalyptus delegatensis* [542] but with the root portion corrected, as described in the main text; 'Eo K2000 temperature'= *E. obliqua* [124]; 'Eo W2021'= *E. obliqua* [543]; 'Er D2003'= *E. regnans* [307], with m= 400 years and k= -6; 'Er S2015'= *E. regnans* [77]; 'Er X2018'= *E. regnans* [544]; 'Er D2020'= *E. regnans* [92] with the root C corrected as mentioned in the main text, Eq. 7; 'Er mid= 600, k= -6'= *E. regnans* [88] with an older-age but faster senescence than for Er D2003, to show flexibility of that equation.

In Figure 9 it can be seen that the two allometric equations for *E. obliqua* diverge more sharply than do the other equations. The higher one, Eo K2000, was the average from four different environments. The highest biomass stand of those four was from an atypical poorer site [124], but noted as having much less disturbance than the other three stands: it made the average allometric trend to higher values. The lower curve for *E. obliqua* in Figure 9, [543], was from one site with a history of relatively frequent fire [350], being in the Huon district, in southern Tasmania, which had a longer history of agriculture, logging and urbanisation than did the Styx Valley in south-central Tasmania [545]. Indeed, the final dip in that lower-yielding equation was based on one tree alone, of DBH 2.74 m, with the next-smallest tree having a DBH of 2.02 m which is near the peak in the curve

[546]. That allometric equation was therefore mostly based on the stem taper of non-senesced trees whereas the larger trees had lost their crowns and thus weren't represented by it (Tim Wardlaw, Sustainable Timber Tasmania, pers. comm., 2024). This is an example of where equations developed from a small number of data, and therefore possibly containing a couple of extremes, may produce equations with limited applicability, and potentially low-accuracy results if applied beyond their bounds, whereas a greater range of input data is more likely to yield a more generally applicable equation.

Possible error margins from using allometric equations can be inferred from the range of trends in Figure 9(b). For example, for a young, mature *E. obliqua* tree of a relatively small diameter of DBH 1.5 m, the allometric equation of Keith, et al. [124] gives the carbon mass of 10.806 Mg whereas that of Wardlaw [543] gives 7.500 Mg. If one couldn't choose between the two equations and had to select the average, then the error margin [about the mean] is ±1.65 Mg, which is a relative error of 19%. If calculating a site's carbon by adding up the carbon in each tree, for perhaps 20 such trees per hectare, then that relative error is transferred directly to the total carbon content. For larger trees, usually fewer data go into construction of the allometric equation and therefore its uncertainty is higher. The error margin is also higher for larger trees due to estimating the degree of senescence, or when devising the allometric equation-- how much senescence was incorporated into it for which DBH? Error margins also get added from such things as measurement errors in the plot size, land slope and DBH measurement. The discrepancy between the Er D2023 and Er D2020 allometric equations for *E. regnans* (in Figure 9) and that of Sillett, et al. [77] is nearly as great as for that between the *E. obliqua* equations, but that may be because Sillett, et al. [77] didn't include internal hollows as a tree ages (for those that have them). The allometric equation from Ximenes, et al. [544] for aboveground biomass of *E. regnans* was based on direct weighing and therefore included carbon loss accompanying senescence, but it was only for trees with DBH ≤2 metres, and no internal hollows were noticed (F. Ximines, Dept. of Primary Industries, New South Wales, pers. comm. 2024). That equation yields carbon content close to that of Sillett, et al. [77] for young *E. regnans* trees, and makes the equations Er D2023 and Er D2020 look conservative.

For commercial forestry work over a larger area in Tasmania, an equation for *E. obliqua* stem volume (the merchantable part of the tree) was developed by State forestry, but it was based on both DBH and tree height and represented the stem taper (which was integrated to give volume) [547]. In typical commercial forestry it is important to quantify the stem, rather than the branches, because the branches are not sold. The DBH values of specimens used to calibrate that taper model ranged up to 3 m, which means it could be useful in carbon accounting. Tree height is not a reliable variable for the more-mature eucalypt trees for several reasons: (a) they may lose different amounts of the crown, from senescence, (b) short, stout trees, growing in more open conditions will have developed extra branch mass whereas trees in stands that have not thinned much will accrue greater height, (d) trees that have only recently lost their crowns have not yet invested in the extra branch mass typical of short, stout trees, (e) extra complexity will be added as a stand thins with old age, neighbouring trees may (or may not) have fallen and so the extra light reaching the tree allows a large branch mass to develop, without the tree increasing height but with increase in DBH. One outcome of the range of circumstances is that trees of larger DBH (i.e., *E. regnans* of 5–8 m) are often shorter [80].

Thus, for useful allometric equations, a large range of data are needed including for the large trees experiencing a range of factors, rather than relying on extrapolation. Data on an extensive range of small, medium and large *E. regnans* trees in the Styx, Florentine and Arve valleys of Tasmania and in various historical records, provided the input for the carbon accounting software CAR4D, which was used for forecasting, spatial analysis, and analysis of industry and fire effects, over long time periods [88,307].

The allometric equation for *E. regnans* from Sillett, et al. [77] keeps increasing for higher DBH, because it doesn't account for eventual decline with advanced age, such as internal stem-wood decomposition (hollows) from typical senescence or fire. The allometric equation for *E. delegatensis* from Dean, et al. [542] initially also keeps increasing within the range of DBHs for which it was applied but it is of sigmoidal form and therefore reaches an asymptote. It was partly constructed from an allometric based on data restricted to DBH of 3.5 m from Keith, et al. [124] for *E. obliqua*. But other contributing parts for that equation for *E. delegatensis* included an allometric equation with senescence (for *E. regnans*), so it will be constrained. Also, senescence was subtracted on a tree-by-tree basis before tallying the stand totals in Dean, et al. [542]. Therefore, the forest carbon stocks would not have been overestimated.

An appraisal of allometric equations for the rainforest species in mixed-forest in Tasmania is necessary. Destructive sampling for specific species could give more accurate data for the rainforest understorey than volumes based on stem taper functions, and such sampling has been undertaken as part of commercial inventory. The first species-specific allometric equations for the rainforest understorey species in mixed-forest were developed by the Forestry Commission [of Tasmania] from rainforest in the South Arthur River area of northwest Tasmania [347]. Although they didn't specify the exact range of tree sizes examined, their plots included mature rainforest across sites of different productivities. The maximum heights for myrtle trees on the more productive, well drained, sites was typically around 37 m but could be up to 45 m [347,548]. Their equations give the volume of wood in stems under the bark as a function of DBH and height, but they did not publish a formula for tree height. Such volumes were processed as part of this review to yield aboveground carbon. Height was parameterised as a function of DBH from trees on a 1 ha plot at State forestry's research plot at Warra in southern Tasmania. The commonly used expansion factor of 1.46 [533] was used, and a basic density of 0.5 Mg m^{-3} [543]. Individual tree data in that dataset from Warra were reported as part of Australia's Terrestrial Ecosystem Research Network (TERN) project [546], but there the allometric equation for myrtle volume used was that for a different species— New Zealand mountain beech (*Nothofagus solandri var. cliffortioides*) [549], and the basic density used to convert to carbon was 0.58 Mg m^{-3}. The reason for choosing a different species in the TERN report was not stated and it gave stem volumes (and hence tree carbon) that were only 76.5% as high as those for Tasmanian myrtle, from Walker and Candy [347]. The TERN data from Warra were re-processed but using the allometric equation for myrtle volume from Walker and Candy [347]. The DBH range was notably restricted at the Warra site. For example, the maximum DBH and heights of myrtle were 0.448 m and 29.4 m respectively, which are smaller than for those observed in the Styx and Florentine valleys, and the earlier work in NW Tasmania. Therefore, any comparison between the relevant allometric equations for the different sites can only be done at the low end of the size spectrum. But the TERN data do offer a potentially more accurate benchmark in that range against which to compare any non-species-specific rainforest understorey allometric equations. The height as a function of DBH equation from the Warra data in Wardlaw [543] was developed from a much more restricted dataset than the Walker and Candy [347] equation for volume. Therefore, to the Warra dataset of 100 points for myrtle, a range of other DBHs and heights for myrtle were added to derive a new equation for height as a function of DBH, for myrtle trees: 2 data points from the literature [548], 8 from my own historical data collection Tasmania-wide, and 6 from citizen science by Wilderness Society volunteers ([92,542,550]). These extra data had a maximum DBH of 2.39 m, and a maximum height 48 m. The new equation was constrained to match the curvature of the Wardlaw [543] equation for myrtle trees at low DBH by including in a mathematical regression, the 79 DBH-height data point pairs reported in Wardlaw [546] where the height had been calculated from their equation rather than measured in the field. The new equation, of Chapman-Richards growth-function format, was refined using Labfit [551]:

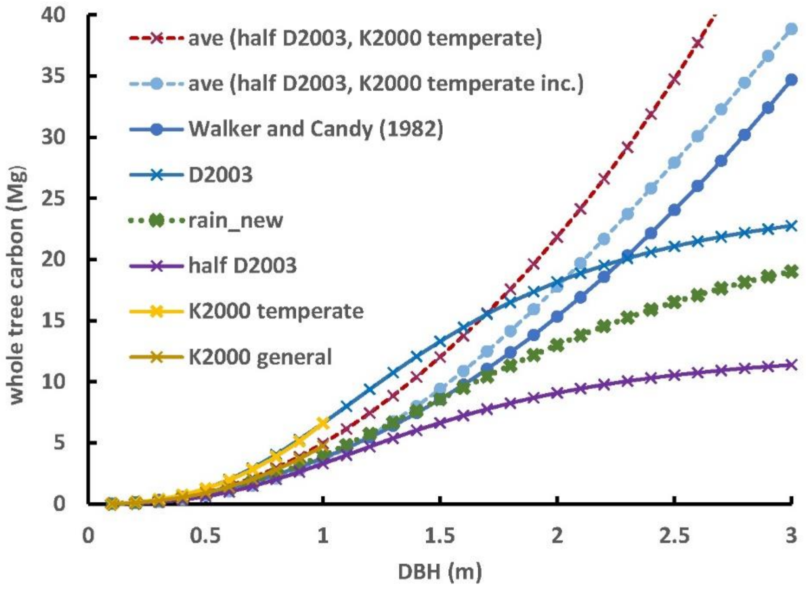
$$height = a \times e^{-\left({}^{(2b-1.25)}/_{DBH}\right)} \tag{10}$$

where height is the tree height in m, DBH is in cm, a= 35.9377 (SD 0.8331) m, P(t)<0.001, b= 5.4261 (SD 0.2688) cm P(t)<0.001, $adjR^2$= 0.75, Df= 117 (though less in reality due to using some calculated heights from Wardlaw [546]), and P< 3 x 10^{-8}. Note that this function has essentially reached its asymptote when DBH values are as low as 1.5 m, and therefore it could reasonably be used for DBH values larger than the 2.39 m in the data set. This equation was used to get DBH-height pairs for input to the Walker and Candy [347] equation for myrtle stem volume, combined with the expansion factor 1.46

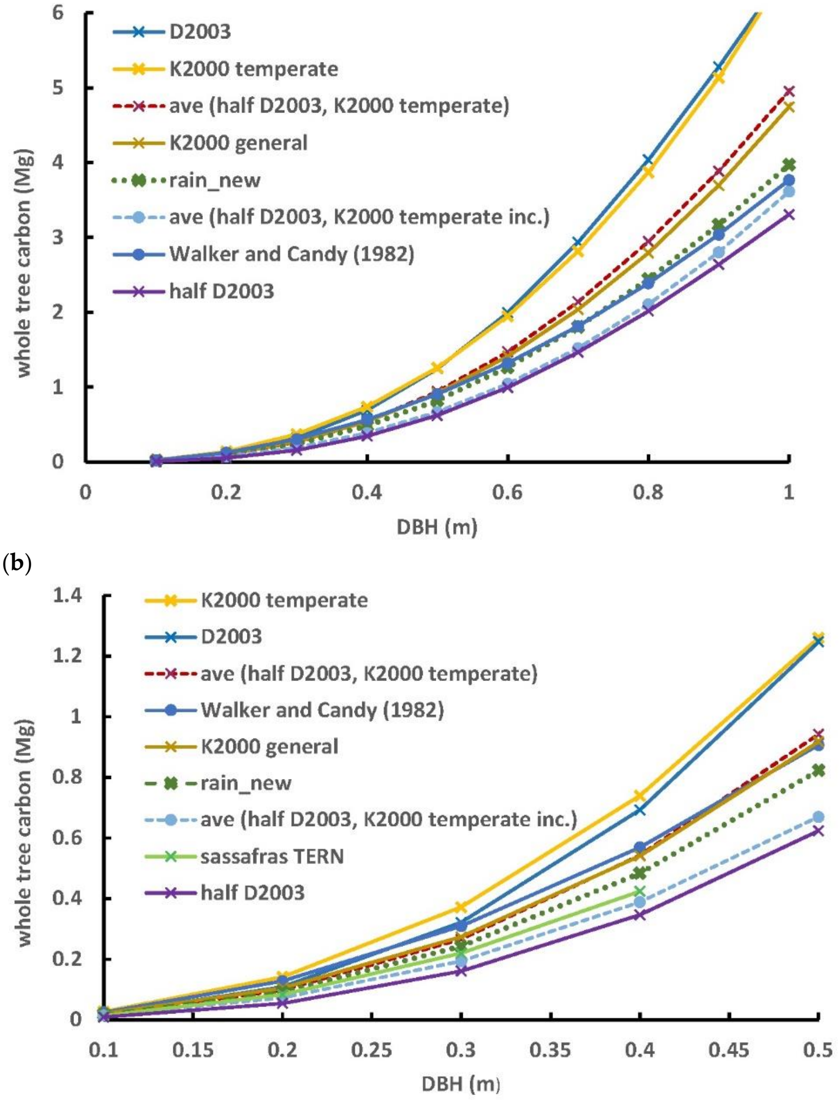
and density 580 Kg m^{-3} [546], and a root/shoot ratio of 0.15, to extend the allometric comparison process for tree carbon of rainforest understorey trees (Figure 10).

Non-species specific (generic) allometric equations for the biomass of Australian rainforest trees were published in Keith, et al. [124] and given in terms of an adjustment to a global standard for rainforests for two different regions in Australia: sub-tropical and temperate. My error in Dean, et al. [542] was in subtracting the adjustment to get from global rainforest to Australian temperate rainforest, rather than adding it. The mathematical form of those equations in Keith, et al. [124] was log-log and although an upper limit on DBH wasn't stated, the formula was only graphed for trees with DBHs between 0.1 and 1.0 m. For wider trees, the biomass would possibly have increased unrealistically due to the exponential form of the equation. Therefore, two different approaches were subsequently used. Firstly, in Dean, et al. [307], an allometric equation was developed which gave biomasses matching those for temperate rainforest understorey from Keith, et al. [124] up to 1 m DBH (Figure 10(b)), but then instead of increasing exponentially for higher DBHs it was made to approach an asymptote by making it a sigmoid function (Figure 10(a)). Secondly, in Dean, et al. [542], the allometric from Keith, et al. [124] was averaged with the allometric equation for rainforest trees of Dean, et al. [307] (which was halved to ensure conservative values). Although the second method still would increase exponentially, the combination gave reasonable values for DBHs up to about 3 m (Figure 10(a)). Note that the maximum DBH for understorey rainforest trees in the data in Gilbert [55] was about 2 m, which means that carbon stocks in Table 2 that use the Keith, et al. [124] equations are most likely too high, which leaves the $\text{new}_{\text{rainforest\_C}}$ as being more likely to be applicable.

For rainforest understorey trees there is an adjustment necessary, to a published allometric equation, due to an error I had made in the formulation of the allometric equation in Dean, et al. [542], which was pointed out to me by Barrie May (pers. comm., CO_2 Australia Ltd., 2012) during his calculations for a report on Tasmanian carbon stocks [550]. My error was to subtract the adjustment to get from global rainforest to Australian temperate rainforest in Keith, et al. [124], rather than add it. The effect was that the carbon stocks reported in Dean, et al. [542] were below what they should have been. I corrected that error for the comparison of different C studies here and the corresponding allometric equation is labelled 'D2011 corr.' in Table 2. The error has most impact near DBH 0.5 m [and is less for other values of DBH]. If that erroneous process was applied to the stands in Gilbert [55], it would have made an ~9.5% reduction in the stand totals for mixed-forest, and an ~24% reduction for both the rainforest understorey and the pure rainforest (Table 2).



(**a**)



(**b**)

(**c**)

Figure 10. Comparison of different rainforest understorey allometric equations, shown at different scales, from (**a**) larger to, (**b**) medium, to (**c**) smaller trees. For the equations that didn't include root mass, it was added with a root:shoot ratio of 0.15:1. 'K2000 temperate'= temperate rainforest [124]; 'K2000 general'= general rainforest [124]; 'K200 temperate inc.'= incorrect adjustment to the general rainforest tree equation in Keith, et al. [124] to get rainforest trees (my error, not theirs); 'D2003'= [307]; 'half D2003'= D2003 divided by 2; 'ave'= average of D2003 and K2000 temperate; 'rain_new'= Eq. 11, new rainforest allometric equation introduced here; 'Walker & Candy (1982)'= [347] using height derived from Eq. 10 introduced here, density of 0.58 Mg m^{-3}, and expansion factor of 1.46; 'sassafras TERN'= sassafras [546].

The Walker and Candy [347] equation is another that forecasts exponentially increasing volumes with DBH (Figure 10.(a)) and therefore it too cannot be used for the larger-DBH myrtle trees. The myrtles with larger DBHs that I have observed in mixed-forest in Tasmania (such as over 2 m) usually have increased hollow volume, both above and belowground, representative of senescence, and therefore their C content approaches an asymptote. To represent the additional scientific knowledge gained over the last 20 years, shown in Figure 10, the allometric equation for C in rainforest understorey trees from Dean, et al. [307] was remodelled to give a new equation:

$$new_{rainforest\_C} = 1.15 \times 0.5 \times 45\left(1 - \left\{1/\left[1 + \left(\left(DBH + 0.01\right)/2\right)^{2.5}\right]\right\}\right) \quad (11)$$

where $new_{rainforest\_C}$ is the carbon in the whole understorey rainforest tree in Mg and DBH is in m. The multiplier '1.15' represents the root/shoot ratio, and the '0.5' multiplier represents carbon being half of the biomass. This equation can be applied to all DBH ranges of rainforest trees measured in mixed-forests for typical carbon inventories in Tasmania. The carbon stock derived by this equation is much lower than that from the temperate rainforest understory allometric equation from Keith, et al. [124], and only slightly higher than that from Walker and Candy [347] plus the expansion factor, and it is therefore conservative. It will eventually approach the same asymptote as in Dean, et al. [307] but only for higher DBH values than observed. At low DBH (< 0.5 m) it gives less carbon than does the Walker and Candy [347] process but more carbon than does the equation for sassafras from Wardlaw [546]. But that latter equation assumes a cone above 1.3 m and it is therefore quite approximate, and sassafras is relatively shade tolerant so has a high branch volume and therefore its corresponding expansion factor is probably higher than the 1.46 used in that work.

As with the graphs of the allometric equations for eucalypts (Figure 9) it is worth gauging the precision (or error margins) of allometric equations for the understorey species from Figure 10. For example, for a tree of DBH 0.5 m: there's a variation between the equations of ~33%. The difference between curves gives an indication of the possible error margins that can be assigned to the total rainforest understorey carbon in a mixed-forest or rainforest study site. The wide error margin is another reason why the $new_{rainforest\_C}$ allometric equation was chosen for use rather than that of Keith, et al. [124] in tallying carbon stocks for the comparison process. In this way, the total, even if still with a high error margin, could not be viewed as excessive, and indeed for trees of DBH 1 m, it is in the lower half of the spread of carbon values.

6.3. Inventories of Carbon at the Landscape Scale

A major, publicly available inventory of the primary tall-eucalypt forests in Tasmania was by ANM [52], to describe the forests in terms of merchandise, namely the pulpwood and possibly sawlog merchantable volumes in the Styx/Tyenna/Florentine pulpwood concession area. Some details from this assessment were given in Section 2 above. As pulpwood uses more of the stem of a mature eucalypt tree than does sawlog alone, then a BEF can be applied to process the merchantable volume data of ANM [52] into carbon in live biomass of the forest stand. Appendix 4 in ANM [52] gives yields [of pulpwood plus sawlogs] in terms of Mg per hectare for oldgrowth forests in the concession, based on 'recent production records' from 'various contractors' over the period 1976 to 1977.

The yields were grouped under the API types (see section 2), grouped by eucalypt height and canopy coverage. The yields reported were for greenwood as delivered to the pulp mill and were not for dry biomass. This is confirmed by the density they found when comparing volumes with weights at the mill: 1.057 Mg m^{-3}, which they used for calculating spatial yield tables and annual extraction rates [52 p10 & 21]. Therefore, to get those yields of greenwood mass to dry biomass (before dividing by 2 to get carbon) they must be multiplied by the basic density of *E. regnans* (which was the main timber logged from which the yields were obtained) and divided by 1.057. The basic density of 0.5124 Mg m^{-3} [307] was used here, being for *E. regnans*, as it was the main timber logged from which the yields were obtained [52]. The usual expansion factor of 1.46 was applied, to get whole aboveground mass and a factor 1.15 to include roots. The product was then multiplied by 1.29 to include the trees in the coupe that were not logged, such as those considered unmerchantable, or on stream banks, or killed in escaped regeneration burns [535] (to give column 5 in Table 3). This may account to some degree for rainforest trees which not taken off-site but just felled or burnt (not all species were pulped). The values thus derived are in Table 3, where the phototypes were also grouped to match the grouping used in a Tasmania-wide forest C assessment by Moroni, et al. [552]: column 5 in Table 3.

If the reported yields in ANM [52] are processed differently, there is a strong similarity in C stocks for some API-types with those reported by Moroni, et al. [552]. If it is assumed that ANM's reported yields were for dry timber (not green timber), and the carbon in roots is not added, and no 1.29 non-merchantable-tree-factor is applied, then one obtains the values in column 6 of Table 3. Numeric values from Moroni, et al. [552] were not available directly but were read off their graphs, with an error of about $^{+}$/-10 Mg ha^{-1} [553] and are given in column 7 in Table 3. By comparing columns 6 and 7 it appears that values of C stock in live biomass for E1a and b, E1c and d, and E2c and d in Moroni, et al. [552], are identical [within error margins] to the C in aboveground live biomass derived from the reported yields in ANM [52] but assuming they were for dry biomass and not adding root mass. However, Moroni, et al. [552] describe their method as:

'Total standing tree volume (standing gross bole volume) was estimated from sample plot data collated in 2007 from >3500 permanent and temporary inventory plots. … Standing-tree bole volumes are converted to forest C mass as described below. Bole volumes of all standing living and dead trees of all species were multiplied by a basic density of 500 kg m^{-3} [31] allowing biomass to be estimated. Total above-ground live-tree biomass (bole, branches and foliage) was estimated from live bole biomass by multiplying by an expansion factor of 1.46 [32]. Root biomass was included by multiplying total above-ground biomass by 1.25.'

It is not possible to tell if the similarity for those three averaged API types (columns 6 and 7 in Table 3) is coincidence or not. But if they were calculated from the same data set then the carbon values in Moroni, et al. [552] are missing the carbon in roots and are thus too low. Similarly, the carbon values may also be too low because it's likely that some rainforest species such as myrtle and sassafras, or some portion of them, may not have been tallied in the ANM inventories, as only eucalypts and *Acacia dealbata* (silver wattle) were pulped in that era [52]. But, conversely, if the carbon values in Moroni, et al. [552] were calculated from that ANM [52] data set then they will be too high as the original data were for green wood (including moisture). Regardless, the values for the different API types in Table 3 are averages over hundreds of hectares and thus unlikely to represent local peaks such as in Sanger and Ferrari [529].

Table 3. Oldgrowth merchantable timber yields (pulpwood + sawlog) from the Styx/Florentine concession [52], converted to spatial C density, and compared with Tasmanian forest carbon stocks from Moroni, et al. [552]. AG= aboveground, PA= API type averaged. Units Mg ha^{-1} = tonnes per hectare.

API type	ANM [52] green yield (Mg ha^{-1})	yield x 1.46 x 0.5 = AG C (Mg ha^{-1})	PA	ANM [52] yield, expanded C with roots (Mg ha^{-1})	PA AG C assuming dry yield (Mg ha^{-1})	Moroni, et al. [552 Figure 3] C in biomass (Mg ha^{-1})
E1a	718	524.1	E1a, E1b	340	473	470
E1b	578	421.9				
E1c	699	510.2	E1c, E1d	232	322	319
E1d	183	133.7				
E2a	514	394.8	E2a, E2b	310	432	285
E2b	641	467.95				
E2c	361	263.5	E2c, E2d	175	244	244
E2d	307	224.1				

Forest stands with peaks in carbon content may not be reported as high volumes in logging records as the trees may not be merchantable, e.g., due to strong senescence when pulpwood is not marketable. In the pulpwood era one of the highest merchantable volumes reported was 150,000 super feet [hoppus] per acre [96,541]. This was converted to C in Mg ha^{-1}, using a value for logs of 0.0030045 to convert super feet to m^3 [554], converting to per hectare, and using the same multiplication factors as for column 5 in Table 3, giving: 618 Mg ha^{-1} of carbon in live biomass (including roots). Two example stands with this C stock had areas of 0.81 ha and 11 ha, of predominantly 100 to 150–160 year old *E. regnans* mixed-forest with API-type E1a* (i.e., eucalypts over 76 m high and 70–100% eucalyptus crown cover— high stand density), on steep slopes, in coupe L.38 in Lords block, Florentine valley [541]. They were clearfell logged then burnt (CBS), from 1959 to 1962, so will not reach the same carbon content until the year ~2120, and only if unlogged and if climate change allows a similarly productive and fire-free climate there until then. The stands were

doi:10.20944/preprints202410.0555.v1

considered 'young' mixed-forest as the rainforest understorey, although up to ~30–37 m high, was not well developed, but would have developed broader, denser crowns, as the eucalypts underwent self-thinning [and if they had not been logged] [555]. This carbon stock is nearly double the average for the E1a API-type and the logging record refers to the site as 'extremely productive'. It demonstrates that the broader forest averages, even within one API-type, cannot compete with smaller sites for high carbon stocks.

Now that some of the variety in allometric equations and expansion factors has been shown, it is an appropriate point to describe what must be compared when gauging contenders for the status of highest carbon stocks in Tasmania, as claimed by Sanger and Ferrari [529] for their study plot. The C stock in live biomass (above and below ground) from their Table 2 is 916 Mg ha^{-1}. The error margin on that value can be partly gleaned from the standard deviation between the data on their 4 transects in their Table B2: ~50% of the mean. No error margin is mentioned for the C in their individual trees with DBH >2.5 m, so it would have to be estimated from the imprecision in allometric equations, such as +/- 20%, from the derivation of the equations shown above in this section. There is also an accrued error from imprecision in each measurement made in the forest (section 2). Overall, as a rough guide, one could reasonably assume +/- 25% error, (229 Mg ha^{-1}), which makes the reported C in live biomass including roots, in Sanger and Ferrari [529] between 687 and 1145 Mg ha^{-1}. Therefore, as a rough guide for comparison with other reports, one must check if the C in live plant biomass in that other data is at least: (a) 687 Mg ha^{-1} if the root:shoot ratio used is 0.25:1; or (b) 632 Mg ha^{-1} if the root:shoot ratio used is 0.15:1; or (c) 549 Mg ha^{-1} if only aboveground C in live plant biomass is reported. Where possible, the root:shoot ratio in other data were converted to 0.15:1, to put all data sets on the same footing. There will also be error margins on the other data, so any overlap could be from two directions. If there is overlap then the two data points are indistinguishable. In reports where dead biomass and soil carbon are also reported, they can provide more detail and possibly more certainty in the comparison. One must also consider the area of forest measured, as it is easier to get a high carbon value in a small, non-randomly located plot (e.g., a 0.05 ha study plot centred around a non-senesced tree of DBH 5 m), than in a larger area such as 5,000 ha, which by its nature, must be more medium. In this, comparing size of study areas could however lead to an unintentional bias, because Sanger and Ferrari [529] stated that their study area was not randomly located but selected because 'it had the highest density of large trees that were safe for climbing'.

6.4. The First Direct Carbon Assessments in Tasmanian Tall Forests

The first, intentionally carbon-oriented assessment of temperate mixed-forests in Tasmania was by Dean, et al. [307] with a mixed empirical-simulation study. Some useful milestones in forest carbon studies prior to that were Olson [556] and Harmon, et al. [269] in the USA, and Grierson, et al. [557] and Polglase, et al. [258] in Victoria, Australia. The latter two were in similar forests to that in Tasmania [307], and related data for such forests were reported earlier [e.g., 55,80,83,558]. Dean, et al. [307] parameterised the allometric equations for *E. regnans* and understorey species with their own data plus historical data from both Victoria and Tasmania. The high-end data were from Tasmania as the larger trees and older forest stands had long-since been logged out of Victoria. Differences in *E. regnans* between the two States were accounted for in parameterisation, such as the fact that they generally grow slower in Tasmania but last longer (Adrian Goodwin, Forestry Tasmania, pers. comm. 2002). Dean, et al. [307] measured and modelled live biomass (including for example, non-circularity of stems and internal decomposition from senescence), and modelled dead biomass (fallen and standing) and soil carbon [including any translocated SOC]. However, Sanger and Ferrari [529] appear to have not known of that earlier work (even though they cited data from a conference paper which drawn from the original paper), nor of a later report by Moroni and Lewis [532] which also included all the same main carbon pools:

"To the authors' knowledge, no studies from Tasmania have estimated the carbon from the whole forest ecosystem, including soil carbon", Sanger and Ferrari [529].

Additionally, Sanger and Ferrari [529] state that root biomass was not included in the values given in Dean and Roxburgh [88], although it was explicitly included. A standard of science writing is to correctly represent earlier work when mentioning it. Errors are part of humanity but the collection of citing errors in Sanger and Ferrari [529] becomes an issue if the statements about earlier work are used as part of a foundation of further work by others, through the 'amplification' effect [559].

The values for most of the different carbon pools were not stated explicitly in Dean, et al. [307] but, as done with some other studies, they can be read off graphs, in this case their Figure 3.4. The specific growth and decomposition in that study were parameterised to simulate those observed in a 20 ha, *E. regnans*-dominated stand with API-type 'E2d.M-' in logging coupe SX004C in the Styx Valley, Tasmania. Notably, the growth simulation had the standing biomass in *E. regnans* peaking near 215 years of age while the understorey biomass was still low, which matches with the relative portions measured earlier in mixed-forest [55]. Their graph also provided the carbon stock values for when the stand age reached 321 years, which was when the coupe was logged. The carbon stocks for those two ages are listed in Table 4(a).

The soil organic carbon (SOC) pool in that study was for the full soil profile (including any soil carbon that was translocated vertically or horizontally); therefore values will be higher than those measured to a specific depth. Moreover, it is not directly comparable with the SOC measured in Sanger and Ferrari [529] which was to only 0.3 m depth— that would require knowledge of the rate of change in SOC with depth. It can be misleading to compare the carbon stock of different soil types based only on the upper soil profile, to ≤ 0.3 m because the steepness of the falloff of SOC with depth can vary substantially between soil types. A soil with higher SOC above 0.3 m may have less SOC for the full profile than a soil with less above 0.3 m [e.g., Figure 2 in 560]. Considering the different uncertainties, to derive a value for SOC so that the total carbon stocks can be compared, a direct method is to treat the value to 0.3 m depth as a fraction of the total SOC. In Polglase, et al. [258] it was only ~25% of the total, and in Dean, et al. [92] it was <33% of the total. As a first approximation, for the SOC for Dean, et al. [307] to only -0.3 m, it will be assumed to be 30% of the total SOC derived from CAR4D: 179 Mg ha^{-1} (Table 4). Some further adjustment may be needed because in Dean, et al. [307] the SOC was higher than that of other Tasmanian forest studies because (as stated above) it was based on values from *E. regnans* forests in Victoria [258]. The half-life of SOC in the model CAR4D would need to be decreased to bring that total SOC back down near typical Tasmanian values to 0.3 m depth. In the absence of that recalibration of CAR4D, when making a comparison with Sanger and Ferrari [529], it is possible to simply instead adopt a value from elsewhere: the value for SOC to -0.3 m as in Dean, et al. [92]: 108 Mg ha^{-1}, which will be introduced below.

In CAR4D trees died gradually as part of self-thinning, and once completely dead they fell and were in the CWD carbon pool. Therefore, there was no separate carbon pool for stags, for comparison with studies where one is measured.

For another comparison of data with Sanger and Ferrari [529], listed in Table 4(b) is carbon in live biomass for a small, 0.341 ha patch of *E. obliqua*-dominated ~450 year old oldgrowth within logging coupe WR005D within the State forestry's Warra Long-Term Ecological Research site from Dean and Roxburgh [88]. For both sites the forest age was determined by tree ring counts from celery-top pine (*Phyllocladus aspleniifolius*) (Kathryn Allen, dendrochronologist, pers. comm., 2002, 2004; method given in Allen, et al. [561]).

Table 4. Carbon stocks (Mg ha^{-1}) to 3 significant figures, live biomass includes roots (root/shoot ratio of 0.15) from: (**a**) from Figure 3.4 in Dean, et al. [307] from measurements and modelling in computer program CAR4D. Values read from the published figure are to +/-15 Mg ha^{-1}, and (**b**) from [Table 1. in 88] a 450 year old oldgrowth *E. obliqua*-dominated stand.

(**a**).

E. regnans age (years)	E. regnans	Understorey	Standing live	Dead trees, CWD & litter	Plant biomass (alive & dead)	SOC to -0.3 m	Total
215	553	70.8	624	20.3	644	197	1310

321	438	343	781	15.5	796	197	1450

(**b**)

E. obliqua	Understorey	Standing live
733	22.5	755

In that first attempt at simulating change in C stocks with time in mixed-forest [307], although the modelled understorey matched observations during the mixed forest stage, in the succession to rainforest stage it eventually surpassed the carbon content of the *E. regnans*, by an incorrect extrapolation, e.g., after 350 years reaching ~850 Mg ha^{-1} vs. 533 Mg ha^{-1}, respectively. This contravenes earlier evidence [55], and consequently the understorey biomass was halved in subsequent use of that model. The mixed-forest vs. rainforest comparison was performed by several other authors [201,302,550], all showing that a mixed-forest has peak in biomass carbon that surpasses the peak rainforest carbon, as in Figure 13 above. These different data sets, are useful to review here because they provide further examples of C stocks to compare with Sanger and Ferrari [529].

The next publicly reported assessment of carbon stock for a specific location in Tasmanian forests was for a commercial enterprise, but for carbon credits rather than timber [562], and no SOC was tallied. That forest was in a less productive area than the central pulpwood concessions, and the location had already been selectively logged. Therefore biomass was lower, and it was not a contender with Sanger and Ferrari [529] for high carbon stocks. Nevertheless, both aboveground live and dead biomass (including coarse woody debris) were measured, and reported in terms of carbon, so it qualified for entrance and could have been in the list of existing carbon studies in Tasmania by the latter authors. Several other carbon credit projects followed but they were also on low productivity land (and therefore relatively low carbon stocks) compared with the central pulpwood area in State forests [e.g., 563].

The next carbon study to be published for a specific location in Tasmanian State forests was for 7.656 ha of a *E. delegatensis*-dominated mixed-forest within the planned logging coupe, FO044A in the Florentine valley, [542]. The API-type of the forest was E1 and E2 with 1–90 % eucalypt tree canopy coverage (Table 5). The understorey was mostly tall myrtle rainforest (M+) but shorter or just scrub in places. There was a wide range of biomass spatially across the area. Aboveground plant biomass, dead and alive, was measured (including assignment of 25% senescence (loss) for the eucalyptus trees) and reported in terms of carbon stock: 622(180) Mg ha^{-1} (standard deviation amongst the different plots in brackets). As mentioned above, allometric equations were developed for *E. delegatensis* and rainforest understorey in that work. Following further development of the equation for rainforest understorey in the present work (Eq. 11), updated carbon stocks for that location are presented here (Table 5). The new total for aboveground carbon (living plus dead plant biomass) is only 3% higher than published earlier. The average plot size was ~0.77 ha, which is large enough to represent a forest stand, and therefore the carbon contents are listed separately for each plot, as some could be contenders for locations of high carbon.

Table 5. Carbon stocks to 3 significant figures, of live and dead plant biomass [including roots with root:shoot = 0.15:1, and 25% senescence for the eucalyptus trees] derived from data collected in Dean, et al. [542] and using the updated allometric equations described above. The value in brackets for the site average carbon is the standard deviation from the plots, and the values in brackets for the individual plots are standard deviations accompanying the derivation of allometric equations used in the calculations— both ~28%. In the API-types: 'co'=cut over (earlier selective logging), and 'S'= scrub.

plot #	phototype	projected area (ha)	C in live biomass (Mg ha^{-1})	C dead plant biomass (Mg ha^{-1})	C in live and dead plant biomass (Mg ha^{-1})
1	coEc1.M+	0.283	677	180	857 (240)
2	coEc1.M+	0.283	328	198	526 (147)
3	coEc1.M+	0.283	386	138	524 (147)

4	coEc1.M+	0.283	652	293	944 (264)
5	E2b.S	1.13	564	112	676 (189)
6	E2f.M+	1.12	327	145	472 (132)
7	E2b.S	1.12	410	88	498 (140)
8	coEc1.M+	1.12	643	232	875 (245)
9	E2d.M-	1.12	451	108	559 (157)
10	coEc1.M+	0.908	344	171	515 (144)
site		7.66	478(142)	166(62)	645 (180)

6.5. Government-Commissioned Carbon Assessments of Specific Forest Types

For the part of their consultancy that compared carbon in mix-forest and rainforest, May, et al. [550] reported three main separate data collections, the first two spanned the Styx, Florentine and Tyenna Valleys. The first included measurement of live and dead plant biomass. The second study additionally included measurement of SOC to 0.45 m depth, so the carbon pools measured were mainly the same as those in Sanger and Ferrari [529], and indeed sites were chosen in both studies that were viewed as having high carbon stocks. In the second one, live and dead biomass were reported in terms of volume, which were converted to mass of C here. Unfortunately though, the second study was discontinued and didn't reach the peer review stage. In neither of these first two studies were the allometric equations used for eucalyptus species published. In both studies, their mixed-forest and rainforest plots were purportedly environmentally equivalent pairs. The mixed-forest and rainforest areas in such proximity are related by fire spread. Fire is less likely (and thereby less likely to convert rainforest to mixed-forest) in damper spots (such as due to shade or topographic water flow), or spots sheltered from the main wind direction on typical days of extreme fire danger. Whereas, only if the fire spread was random, could the two components in any one pair be genuinely environmentally equivalent. Additionally, some plots delineated as rainforest still contained live and/or dead eucalypts, so were still at the intermediate phase between mixed-forest and rainforest.

The third study was based on aboveground live and dead plant biomass data donated by The Wilderness Society and additionally included data from northwest Tasmania. In these three studies in May, et al. [550] there was no mention of tree roots and therefore it assumed in the present work that root carbon was not included in their tallying.

In the first study in May, et al. [550] there were sixteen each of mixed-forest and rainforest 0.1257 ha plots (totalling 2.01 ha for each forest type). The mixed-forest plots were E1a, E1b, E2a or E2b API-types and randomly positioned. Those categories were written in May, et al. [550] in terms of forest categories FC1 and FC3, with the translation to APIs given in Moroni, et al. [552].) Therein, field data were converted to spatial carbon stocks using the Farm Forestry Toolbox [537]. The allometric equations for *E. regnans* and *E. delgatensis* in that software are listed as confidential, being from Forestry Tasmania, and they were not publicly available. Therefore, they cannot be compared with other equations as in Figures 9 or 10. A taper equation was available for *E. obliqua* [564], but taper equations alone cannot account for branches or internal trunk hollows. Allometric equations for rainforest understorey were developed by May, et al. [550] but not published. The carbon contents provided in May, et al. [550] are therefore only listed here verbatim. After ~50 years of commercial logging in the area since Gilbert [55], it seems unlikely that the carbon contents of the semi-randomly positioned plots across such a large area could be examples of peak carbon. Indeed, the eucalypts measured were below median size for the region, with the highest DBH being 2.6 m, and the tallest eucalypt being 55 m high. i.e., small-to-medium in size for mature eucalypts. Therefore it cannot be seen how any of their plots could be of the stated E1 type as the average mature eucalypt height for that category must be between 55 and 76 m [52,94]. Nevertheless, in their processed data, although the average carbon mass found in live plant biomass of 373 Mg ha^{-1} is no contender against Sanger and Ferrari [529] for the highest reported carbon in Tasmania, the mass at individual plots in Figure 110 of May, et al. [550] showed four likely candidates. Data was extracted from their Figure 110 to an accuracy of about ±10 Mg ha^{-1}, and root carbon was added (root:shoot ratio= 0.15:1). Study-area-averages for dead biomass were also added: standing trees (with root carbon)= 120.75 Mg ha^{-1}, and

CWD= 23 Mg ha^{-1} (Table 6). The API-types of the forests were oldgrowth E1a, E1b, E2a and E2b, with a range of understorey types but the data points could not be matched to API-types using their Figure 110.

In the second study in May, et al. [550], the intention was to have a mixed-forest plot measured at each of 5 locations and 6 rainforest plots measured across 2 locations. But it was only possible to measure and report on two sites. Each plot had a maximum size of 0.1256 ha. Stem volume was calculated using unpublished allomeric equations from Forestry Tasmania, and converted to mass using a basic density of 0.5 Mg m^{-3}, and scaled up to whole, aboveground tree carbon using the BEF of 1.46. That mass was halved to get aboveground carbon in live plant biomass (Table 6). Standing dead timber for this second study was not reported separately. The CWD was also reported in terms of volume. This was converted to mass of carbon, assuming an average density of 0.335 Mg m^{-3} as in Grove, et al. [565] and in the State-level section of May, et al. [550]: giving 100(±38) Mg ha^{-1}. Soil carbon was reported down to 0.45 m depth rather than 0.3 m depth: 89(±58) Mg ha^{-1} in the mixed-forest, which, if assuming the same proportional decline with depth as for the in-between-tree zone as in Dean, et al. [92], gives 67(±43) Mg ha^{-1}, to -0.3 m.

The third study in May, et al. [550], for three locations in Tasmania, partly overlapped with Dean, et al. [542] and Dean, et al. [92] in the Florentine and Styx Valleys respectively, in that most of the unprocessed data came from the same source. In the Styx Valley though, only half the data had been collected in time for May, et al. [550] [compared with what was used in Dean, et al. [92]] and the site-specific allometric equation hadn't been developed. The allometric equations used in the third study of May, et al. [550] were from Dean, et al. [542] except that for understorey trees the temperate rainforest equation of Keith, et al. [124] was used. For the Florentine Valley component, the results from Dean, et al. [542] were given to May, et al. [550] for them to report and so that area won't be re-listed here. The results for the northwest Tasmania region within the third study are not contenders with for a high carbon area and so won't be further mentioned. That leaves only the Styx Valley results from the third study to portray here. To put that rainforest carbon on a level playing field with the other studies (i.e., $\text{new}_{\text{rainforest\_C}}$ from Eq. 11), a pro-rata correction was applied to the results from the erroneous allometric equation used, as suggested from Table 2. (a correction of -48.2%). It is unknown how much senescence they modelled for the eucalyptus trees in the Styx Valley data so their values are reported verbatim here (Table 6(c)).

Table 6. Carbon stocks (Mg ha^{-1}) in aboveground carbon pools from May, et al. [550] to 4 significant figures, of live and dead plant biomass, with added roots (root/shoot ratio of 0.15) and using the updated allometric. The values in brackets for the site average carbon pools are the standard deviations amongst the six plots. (a) live above ground derived from their Figure 110 (plus roots) and for totals: with carbon in dead biomass. (**a**) First study, spanning the Styx, Tyenna and Florentine Valleys. For the site averages the figures in brackets are the standard deviations from the 16 plots. (**b**) Second study, one site in each of the Styx and Florentine valleys. (**c**) Third study, Styx Valley.

(**a**)

plot #	C in live biomass	Total (live plus dead above- and below-ground biomass)
1	808.7	952.5
2	732.4	876.2
3	668.4	812.2
4	648.4	792.1
5	557.5	701.2
6	534.0	677.8
7	408.1	551.8
8	364.3	508.1
9	357.1	500.8
10	338.9	482.7

11	255.2	398.9
12	247.5	391.2
13	244.5	388.3
14	236.9	380.6
15	226.0	369.7
16	213.2	357.0
average	429(201)	573(201)

(**b**)

location	PI type	C in live biomass	Total (live plus dead above- and below-ground biomass)
Styx Valley	E1c.M+ or E1d.M+	636	736
Florentine Valley	E1a.M+ or E1b.M+	523	623

(**c**)

C in live biomass	C in dead biomass	Total (live plus dead above- and below-ground biomass)
780	196	977

6.6. The State Forest Agency's Carbon Assessments of Specific Forest Types

Chronologically, the next report on carbon stocks in Tasmania was by Moroni and Lewis [532], with values that could be contenders with Sanger and Ferrari [529] for high carbon. These wet-sclerophyll forests studied were in the Huon district, southern Tasmania, with API-types E2c and/or E2d (the lower category in Table 3) and they were mature *E. obliqua*—dominated [532]. Live and dead aboveground plant biomass were measured for six 0.2 ha plots. The fire history was not stated. Biomass for standing live trees was calculated from unpublished allometric equations for bole [merchantable stem] volume, and the BEF applied was 1.46. Root carbon was estimated with a root:shoot ratio of 0.25:1. Soil carbon was reported to a possibly variable depth depending on the particular soil type, as it was retrieved from values for purportedly similar soils in the north of Tasmania, that were described in the three 1995 Tasmanian reports mentioned above [303–305]. In Moroni and Lewis [532] the SOC amounts were compared with those modelled by the program Fullcam [566,567], which only models to 0.3 m depth, so it may be that the original empirical values were trimmed to that level.

The individual carbon pools for the six plots were not tabulated but there were graphs of total C, debris C, and live biomass: data were extracted from those graphs with an accuracy of ~+/- 4 Mg ha^{-1}, and after reaggregation, the reconstructed site averages only differed from those reported by +/- 3 Mg ha^{-1} (meaning there was a minimal transcription error). From these data, and the fact that each plot had been attributed the same SOC of 179 Mg ha^{-1}, it was possible to derive the values for the other C pools for each plot. From this vantage point it was possible to change the root:shoot ratio from 0.25:1 to 0.15:1, for comparison with the other reported values in this section (Table 7). This lowered the site average total C by 6%. Notably, the site average live plant biomass is similar to that in Table 4(a) for the E2d.M- site in Dean, et al. [307]. The site average of 634(187) Mg ha^{-1} is approximately two standard deviations higher than the value of 244 Mg ha^{-1} for the State-wide E2c,E2d average in Moroni, et al. [552].

doi:10.20944/preprints202410.0555.v1

Table 7. Carbon stocks (Mg ha^{-1}) to 4 significant figures, live and dead plant biomass [including roots, root/shoot ratio= 0.15] derived from figures in Moroni and Lewis [532] and using the updated allometric. The values in brackets for the site average carbon pools are the standard deviations amongst the six plots. Note that the 'total' C includes their value for SOC of 179 Mg ha^{-1} down to an unstated depth, possibly to -0.3 m.

plot #	C in live biomass	Debris	total C
1	397.2	24.70	600.9
2	508.2	47.43	734.7
3	556.1	126.9	862.0
4	709.6	23.46	912.0
5	711.9	65.97	956.8
6	926.2	21.71	1127
site	634(187)	52(41)	866(182)

6.7. A comprehensive Carbon Assessment Including More Soil Carbon Detail

The next scientific publication on carbon stocks for tall eucalypt forest of Tasmania reported live and dead biomass and soil carbon for plots in the Styx Valley and partly in the Tyenna Valley [92]. Sanger and Ferrari [529] stated that they only reported soil carbon. The site averages for the different pools were reported but for the carbon in living trees, plot-level values were shown in their Figure 12(a) [92]. There are two plots which looked in that figure like possible contenders for having the highest carbon stocks in Tasmania.

Dean, et al. [92] calculated SOC to the full depth of where it transcended down through the soil, rocks and interstitial spaces, by extrapolation of the upper changes in SOC with depth. Additionally, SOC was measured separately close to the larger eucalyptus tree trunks, both above and below ground, and for the soil (lignomor) in the softer, more decomposed logs. The overall value was 330 Mg ha^{-1} to 90% of the maximum SOC (or 367 Mg ha^{-1} in total). The error margin, in terms of standard deviation, on that value can be estimated as 25% (82.5 Mg ha^{-1}), from the various measurements and formulations that went into its derivation. For the purposes of comparison with Sanger and Ferrari [529], the soil organic carbon which had been allocated from half of the soft-log mass in Dean, et al. [92], was reattributed to the general CWD pool. For comparison with the value of 149 (SD 103) Mg ha^{-1} from Sanger and Ferrari [529], the value of SOC that will be used from Dean, et al. [92] to 0.3 m depth, will be that for ground away from trees, it is that derived from data in Dietrich [310]: 108(SD 10) Mg ha^{-1}. This is notably less than a third of the total SOC.

A site-specific allometric equation for *E. regnans*, which included senescence, was developed in Dean, et al. [92] (see section 6.2). Some of the plots, with lower overall live biomass, contained some *E. delegatensis* and *E. obliqua* trees, and the corresponding equations mentioned above (section 6.2) were used to calculate C in biomass for those trees. Here, we have recalculated the understorey biomass using Eq. 11. Values are summarised in Table 8 but without the SOC pool.

Table 8. Carbon stocks (Mg ha^{-1}) to 3 significant figures, of live and dead plant biomass [including roots, root/shoot ratio= 0.15] and soil carbon, derived from Dean, et al. [92]. The value in brackets for the site average carbon is the standard deviation from the 10 plots. Some plots straddled two API-types: both are shown.

plot #	phototype	projected area (ha)	live biomass	dead biomass	total biomass
1	M- & E1c.M+	1.11	474.1	45.92	520.0
2	E2b.M-	1.11	298.5	187.3	485.7
3	E1c.M+	1.11	256.8	150.4	407.2
4	E1c.M+	1.13	547.9	90.43	638.4
5	E1c.M+	1.13	678.3	304.5	982.7
6	E2c.M-	1.11	399.4	64.61	464.0
8	E1c.M+	1.11	649.6	133.2	792.7

9	E1c.M+ & M-.E1f	1.12	500.1	16.04	516.2
10	M- & E1c.M+	1.12	409.8	109.1	518.9
11	E1c.M+	1.11	402.9	83.31	486.2
site		11.16	462(137)	119(83)	580(176)

6.8. Placing the Pro-Conservation Report on a Level Playing Field

Where possible, for the purpose of comparing carbon stocks, the root:shoot ratio of 0.15:1 has been applied to other data collections mentioned in this section. Consequently, that must also be done to the data reported in Sanger and Ferrari [529]. Also for making an equitable comparison, some of the other factors used in that report must be adjusted: (a) account for ground slope, (b) use the new rainforest allometric equation, and (c) use species specific wood densities and allometric equations for the eucalyptus trees where possible.

There are firstly some possibly larger matters to address though in Sanger and Ferrari [529] regarding what appears to be their particular adoption of standard scientific processes, for example (a) citing of earlier work, (b) error margins; (c) accounting for senescence in the larger trees, especially trunk hollows; (d) some missing information; and (e) comparing results with those in earlier work. Those will be addressed these here. The issue of incorrect citing of earlier literature was discussed above and will only be mentioned again with respect to tree hollows etc.

When citing the equations used from Dean, et al. [542], except in one instance, Sanger and Ferrari [529] transcribed them verbatim, even copying errors in the original work (some of which had been corrected in a corrigendum). For example, Eq3 in Dean, et al. [542] for the temperate rainforest understorey aboveground biomass:

$$\begin{aligned} Temp\_rain\_AGB &= 0.001\exp \\ &\times ((2.5667\ln(\mathrm{DBH}) + 8.9133)) \end{aligned}$$

was copied as:

"U_{AGB} = 0.001exp x (2.5667 ln (DBH) + 8.9133)", Sanger and Ferrari [529],

where the multiplication sign, 'X', after 'exp' is incorrect mathematical syntax. And the '0.01' in Eq. 2 in Dean, et al. [542] was transcribed verbatim but had been corrected to 0.1 in the corrigendum [568]. The corrigendum was uncited so it may have been inadvertently missed. The exception to their verbatim transcriptions was the allometric equation for *E. delegatensis*, namely Eq. 2 in Dean, et al. [542]:

$$\begin{aligned} Edel\_AGB &= 1612.4 \\ &\times \left(1 - \left(\frac{1}{1 + ((\mathrm{DBH} + 0.01)/12.714)^{2.2283}}\right)\right) \end{aligned}$$

which was transcribed as:

'E_{AGB} = 1612.4 x (1- (1/ 1 + ((DBH + 0.01)/12.714)$^{2.2283}$)', [529].

The 'del' is missing in the transcription. Notably Sanger and Ferrari [529] describe it as being an 'averaging allometric equation' for 'eucalypt trees', whereas in Dean, et al. [542] it is specifically described as being designed for the aboveground biomass of *E. delegatensis* trees. An allometric equation developed for one species has been erroneously stated as generic for all eucalyptus species. That would mislead readers about the earlier publication and might result in other people copying that error for their own calculations further afield, while citing the original publication (the amplification effect). A possible reason for the error is that Sanger and Ferrari [529] wanted an equation that would suit any eucalypt species in their study plot. Such a generic equation was already available, at least for trees of DBH ≤ 1 m [124], and it could have been adopted and modified for senescence in trees with larger DBH values, following for example Dean, et al. [92], possibly using individual tree data [124,543,544].

doi:10.20944/preprints202410.0555.v1

Some information missing from Sanger and Ferrari [529] is the species names for the trees measured and modelled. This is important in deciding parameters for carbon accounting. Only the eucalypt species present in the larger 98 ha surrounding their ~2 ha study plot are mentioned. One could assume the species in the study plot are the same as in the larger 98 ha, but in Sanger and Ferrari [529] their choice of allometric equations suggests a subset. In the surrounding 98 ha of forest there were *E. regnans*, *E. obliqua* and *E. globulus*, with no mention of *E. delegatensis* being present (though they used an allometric equation developed for that species), whereas in the public media, the first author, Sanger, stated:

'What makes it really special ... is that there are giant trees from four different species of eucalypt tree,…' [569]

There are strong links between the publication, Sanger and Ferrari [529], an internet Facebook page and a commercial internet site run by a company called the 'The Tree Projects' [570,571]. On that commercial site *E. delegatensis* is mentioned as being in the larger forest stand, and only *E. obliqua* , *E. regnans* and *E. globulus* are featured on their Facebook site [571]. This is helpful but some uncertainty remains as to the species present in their study plot.

There could be a difference in species between their plot and the surrounds because: (a) they only measured 2% (by area) of the 'Grove of Giants' (their name for the 100 ha total), and (b) the average height of the trees with DBH over 2.5 m that they measured was 56 m (maximum height 72 m, and maximum DBH 5.12 m) whereas the surrounds have some trees near 80 m tall and a maximum DBH > 6 m. Thus, maybe a higher proportion of the trees in their plot were *E. obliqua* or their plot was more senesced than the surrounds. But this cannot be determined, especially as it can't be determined whether the latitude and longitude provided for their plot refers to the centre, or a corner, or just somewhere in the larger 100 ha expanse. The URL web data link for 'the data that support the findings' provided in Sanger and Ferrari [529] does not lead to a web page, and when contacted, the journal's administration didn't provide an alternative URL.

Typically in scientific papers on forest carbon, the tree species are mentioned in the Introduction or Methods sections. An even more useful place to mention them in this case [529], would have had been in their Table B1, where the individual DBHs were listed for trees with DBH >2.5 m. As the species in the understorey were not mentioned, they cannot provide a clue about the fire history and thus the likely eucalypt assemblage. However, the company's website provides geographic coordinates for the walking trail in the 'Grove of Giants' and overlaying that with the State forest agency's API-type map of 1984, gives the API types as E1b.S.ER and E1c.S. The 'S' indicates scrub (understorey < 15 m) and the 'ER' indicates young (regenerated) eucalypts present with the oldgrowth trees. Photos of trees being measured on the company's Facebook page supports these API types, and a range of eucalyptus and understorey species [571]. In a traverse along the recommended walking trail in the 'Grove of Giants' by J. Kirkpatrick (University of Tasmania, pers. comm., 2024) myrtle, sassafras, celery-top pine and Tasmanian Laurel (*Anopterus glandulosus*) were seen in the understorey, making it young rainforest understorey, rather than scrub. It had possibly begun to mature, since the State forestry's aerial photography. In summary, the relative concentration of different eucalypt species in the study plot of Sanger and Ferrari [529] cannot be determined from any of the information publicly available. Therefore, the error margin on the study plot's carbon content could be increased further.

Their choice of mixing different allometric equations for different parts of their data processing of the eucalyptus trees in Sanger and Ferrari [529], namely for *E. regnans* from Sillett, et al. [572] and *E. delegatensis* from Dean, et al. [542], seems unnecessary, and could possibly lead to some bias, causing inaccuracy in the carbon stocks. For small eucalyptus trees in southern Tasmania (where the 'Grove of Giants' is) Bowling [573] found that for small regrowth eucalyptus trees (mostly with DBH < 0.64 m) the volume (for a given DBH and height) does not vary much between species, though some trends in deviation from the norm were noticed. With more data for each species, some definitive trends appear. For example, even for trees of the same DBH (1 m) and height (58 m), the wood volume in the stem, from integrating under their taper equation curves (Figure 11), for *E. obliqua* [574], *E. delegatensis* [574], and *E. regnans* [88], is 13.3, 12.5 and 12.0 m^3 respectively— a spread of about 10%.

doi:10.20944/preprints202410.0555.v1

The taper equation for *E. obliqua* mentioned above by Goodwin [547], gave a volume of 12.8 m^3, and could have been graphed in Figure 11 but was not because in the mid-section it was too close to the curve for *E. delegatensis*. (For people who might use these equations, I just mention here that there's a typographical error in Goodwin [547] which could cause confusion: the equations for α_2 and α_3 in the parabolic part of the taper, are actually for α_3 and α_2, respectively.)

Furthermore, Ilic [575] gave the basic densities of *E. obliqua*, *E. delegatensis* and *E. regnans* as 580, 524 and 485 Kg m^{-3}, respectively. This reinforces the sequence of decreasing mass that was indicated by the taper equations for equal-dimensioned trees in that species sequence. When normalised to the mass for *E. obliqua*, the mass ratios for that sequence are: 1: 0.850: 0.756. This gives a value to the imprecision (error margin) associated with not identifying the species— possibly up to about ±12 %. Wood density is an important contributor to carbon calculations when applying an allometric equation that is otherwise intended to be generic across species [576].

Additionally, *E. regnans* are commonly known to be, on average, the more-buttressed of the ash-type eucalypts, which could induce a potential extra error margin if mixing allometric equations between species. There is minimal information available for allometric equations of non-plantation *E. globulus*, (which are likely to be in the study plot of Sanger and Ferrari [529]) and it is considered an area of necessary research, but young specimens have an average basic density near 600 Kg m^{-3} [577].

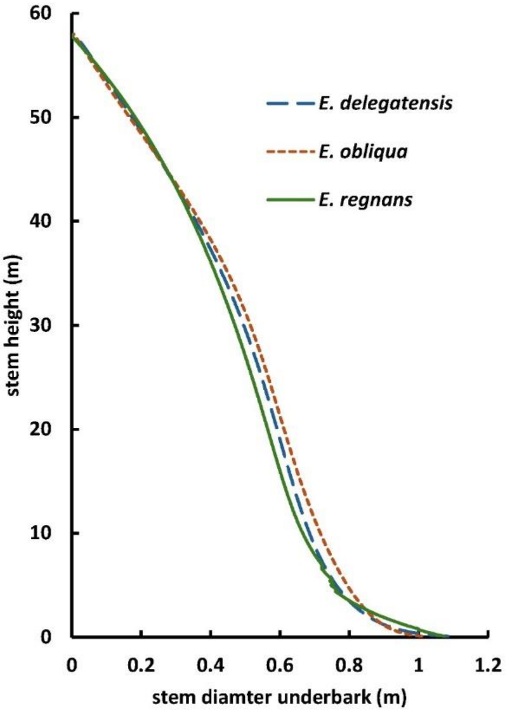


Figure 11. Different stem shapes of some eucalyptus species with the same DBH (1 m) and height (58 m), based on taper equations, showing difference in underbark wood volume, which yield a volume range of ~10%. The roughness for *E. regnans* near the top of the buttress region is because its taper equation was for over-bark shape, and the bark becomes thinner there.

The benefit for Sanger and Ferrari [529] from choosing an allometric equation for *E. delegatensis* for their smaller trees is that it gives a medium volume. But if their trees were mostly *E. obliqua* or *E. globulus*, then they may have underestimated the volumes, or overestimated them if they were *E. regnans*.

Accounting for senescence in older trees is important when determining carbon stocks [307,309,578]. Sanger and Ferrari [529] state that they used an equation for *E. regnans*, from Sillett, et al. [572] to calculate missing carbon due to hollows etc.:

doi:10.20944/preprints202410.0555.v1

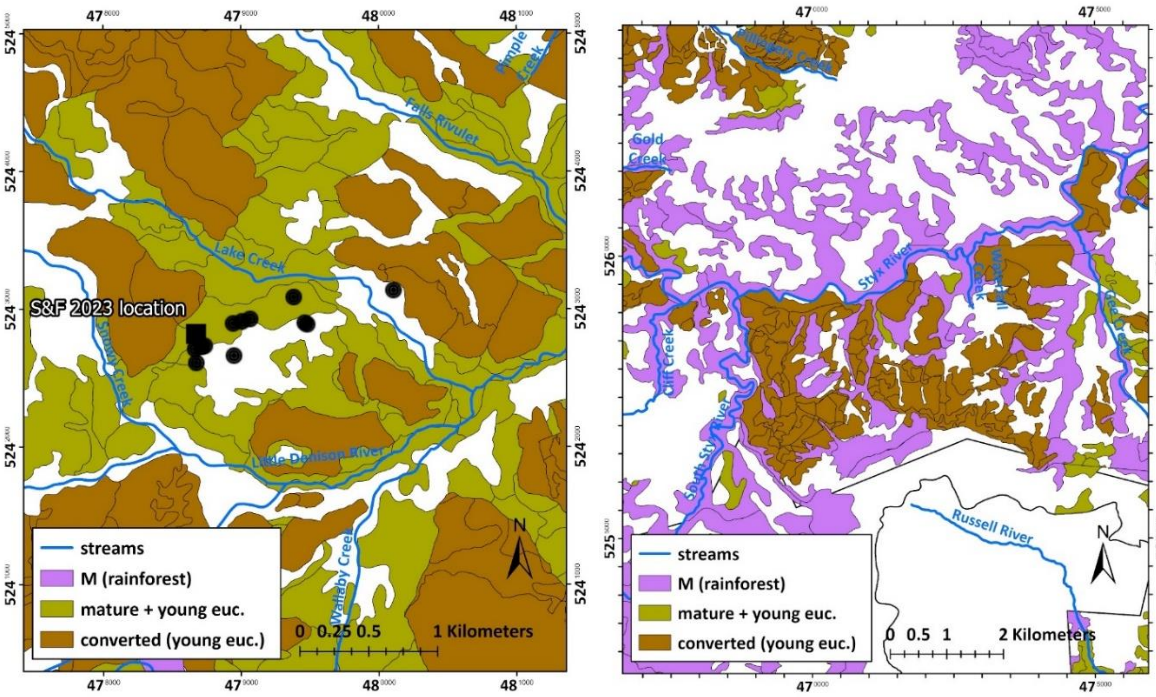
‘Internal decay was factored by equations to predict occurrence and volume of decay related to tree size was also derived from Sillett et al. (2010).’

However, in ‘Sillett et al. (2010)’ it is explicitly mentioned that internal decay was not accounted for, which was confirmed with its lead author: ‘Our published numbers for EURE do not account for hollows or decay’ (Steve Sillett, Cal. Poly. Humboldt, pers. com. 2023). Consequently, to put carbon stocks on a level playing field for comparison purposes, some carbon must be subtracted from that apportioned to the larger trees in Sanger and Ferrari [529]. There is no reason to believe that the eucalypts in the plot of Sanger and Ferrari [529] had less hollows than those with which they are being compared here, such as in the Styx and Florentine Valleys. Both senescence and fire, and their combination, will increase the likelihood of lost carbon from the tree trunks. Therefore, it is shown here, how at least as much fire has occurred in the locality of the ‘Grove of Giants’, as in the Styx and Florentine Valleys.

Fire can increase hollows beyond that from normal senescence, especially in the buttress region or an existing hollow, because of turbulent flow. Turbulent flow, will in places, increase the speed of passing air and therefore delivers more oxygen, which pushes the exothermic reaction forward, creating a hotter, larger and longer burn. For a tree trunk this increases loss of carbon compared with wood decomposition alone. Such a fire is shown in Figure 12 after a light, prescribed burn of surrounding buffel grass in central Australia.

Figure 12. Intense fire due to turbulent flow in tree hollow River Red gum (*Eucalyptus camaldulensis*), following a light, prescribed burn of feral agricultural grass (*Cenchrus ciliaris*) in Todd River, central Australia. Carbon was still being emitted the following day after the grass had long-since self-extinguished. Similar release of carbon and hollow enlargement could occur for ground-level forest fires.

Strong evidence of fire in the region of the ‘Grove of Giants’ is that there are young eucalypts in the stand (the ‘ER’ in the API type) and that the understorey is not mature. Additionally, in the nearby Warra ecological research site, which also has a history of recent fire, the mature eucalypts were noted to be ‘almost invariably hollow, or at least heavily decayed at their centre’ [350]. Comparing maps for the ‘Grove of Giants’ region and the Styx Valley show that the creek lines have quite different vegetation: no rainforest gullies surrounding the ‘Grove of Giants’ whereas they prevail in the Styx Valley. In the former the gullies contain eucalypts <110 years old and mature eucalypts, possibly constituting wet-eucalypt forest rather than rainforest (Figure 13).



(**a**) (**b**)

Figure 13. Indicators from vegetation of more frequent fire near the study area of (**a**) Sanger and Ferrari [529] with mature and young eucalypts in the creek lines (black square= study plot), compared with (**b**) the central Styx Valley with rainforest in the creek lines. Hence probably more hollow development in the mature eucalypts in (a). Blue lines= creeks and rivers. Vegetation categories based on API types from State forest agency maps.

Several of the trees in the photos of the 'Grove of Giants' showed indicators for deep basal cavities [570,571]. In three dimensions they are clearer. For example, a 3D model based on aerial LiDAR of one of the largest *E. globulus* trees in the larger 100 ha surrounding their study plot, shows two deep basal fissures and higher up a deep stem indentation, which could lead to an internal basal hollow [579].

One of the largest *E. regnans* trees measured in Tasmania in the last 20 years, the 'El Grande' tree (which I measured to have a DBH of 6.19 m), had a stem volume of 406 m^3 but had internal stem hollows, which may have reduced that wood volume by up to 40%, down to 244 m^3 [77]. The measured volume for the *E. globulus* tree [mentioned in the last paragraph], when not accounting for stem hollows was 325 m^3 [569]. But this could be much lower if the internal losses are similar to that in the El Grande tree. In Sanger and Ferrari [529] crown loss is already taken into account for the trees with DBH >2.5 m as the calculations include tree height. Dean, et al. [542] subtracted 25% from the eucalypt mass to account for internal hollows. Combining the two influences, it is reasonable to subtract 15% from the values in Sanger and Ferrari [529] from the trees with DBH >2.5 m, to place their plot's total carbon on a level playing field for comparison purposes. Although this may sound like maligning of the larger trees, it is only to prevent overestimation of their carbon stock, and it does not detract from their numerous other values such as wind attenuation, carbon storage and cycling, local climate moderation and unique ecological contributions.

The variation in hollow occurrence described in section 2 suggests that in the absence of definitive detail, one must allow for a balanced and adjustable degree of hollowing. Other than assuming an average internal decomposition which can be adjusted for location [307], other methods are destructive sampling [124] [but then the tree dies] and coring. The latter method was used by Sillett, et al. [580] for individual Coast redwood (Sequoia sempervirens) and for stand-level carbon accounting they modelled the shape of trunk internal decay for trees with DBH > 2 m, through reductions in wood density..

Ground slope was not mentioned in Sanger and Ferrari [529]. Slope doesn't usually make much difference to soil organic carbon calculation [other than depth calculation on steep slopes] because the cores or pits are dug vertically, and the extrapolation to a hectare is done assuming the surface of the collected sample is horizontal. On very steep slopes however, one may not be digging only downwards but, from the soil's perspective, sideways too, and therefore measuring more of the topsoil, giving higher carbon stocks than due. Sloping ground makes a difference to the biomass calculation, as the projected plot area is reduced, which increases the calculated carbon in biomass per hectare, by the secant of the cosine of the slope. Using GoogleEarth Pro©, the typical slope in the 'Grove of Giants' was found to be about 20% or 11°, which increases carbon in biomass by about 2%, from the values reported in Sanger and Ferrari [529]. The projected plot area becomes 1.914 ha rather than the original 1.95 ha. The ground slope from a cross-section of the 3D model from LiDAR data of the large *E. globulus* [581] was 9(±1)°,which would increase carbon in biomass by 1.2%. In Sanger and Ferrari [529], there was initially some uncertainty about the plot that they had laid out, because it was twice described as 100 x 130 m, once as 150 x 130, twice as 19,500 m^3, and once as 2 ha. On balance that is 4:2 in favour of 150 x 130 m, which was used in the recalculations here, before adjusting for slope.

In each measurement of trunk diameter there is an error margin when reading a distance off the tape. This type of error margin is usually taken as ± two tenths of the minimum graduation mark on the measuring device, but it also depends on a user's experience level. People who routinely measure tree diameters have acquired habits that reduce error in placement of the tape before any reading is taken, such as sliding the tape to-and-fro a few times to make sure it has achieved the minimum local girth. Other basics include making sure the tape is perpendicular to the axis of the trunk or branch, and that it is not over any local swelling such as from a burl or branch development, but rather below or above it (or both, and then the average taken). Such information was passed on to the citizen science crew for the group measurements made in some earlier work [e.g., 92,542]. From photographs in public media on the internet however, it appears to have not been passed on to the arborists deployed for data collection in Sanger and Ferrari [529]. There were instances of tapes over substantial amounts of shed bark from higher up the tree, tapes not level for trunk and DBH measurements, tape over part of a burl, and a tape over a large branch-collar [569,571]. These all add apparent wood volume, rather than simply increasing the error margins. Measurements were taken off some photos in the public media that showed incorrect measurement technique in citizen science data collection in the 'Grove of Giants' [569,571 19-December-2022, 19-January-2023, 23-February-2023]. The angle from perpendicular [to the trunk axis] on diameter measurements was between 5 and 9°, giving an extra trunk volume (locally) of ~0.4 and 1.6% respectively. The extra diameter over the bark strands from higher up the trunk was ~5%, which translates directly to the same percentage of extra volume. The placing of the tape over the branch collar gave an extra volume (locally) of ~21%. It cannot be established if the examples provided in public media were exceptions or the typical procedure adopted. A rough estimate from the non-perpendicular diameter tapes may be, overall, conservatively in the order of 1%, which is insubstantial. If it was as high as 5% on average then that would be a significant factor when comparing carbon stocks. Most humans are outside of their normal 2D oecumene (as discussed in section 3) when climbing tall trees and this makes measurement more difficult, but practice of standard measurement techniques can help overcome difficulties and ensure less error in measurement. Regardless of the errors in measurement, due to the uniqueness of tree climbing, allowing public access to the data (by activating the mentioned data repository link in their paper) would have benefited science by allowing improvement to existing allometric equations.

Having described several studies and summarised their data in the tables above, they can now be compared with the carbon stocks derived by Sanger and Ferrari [529], but only after adjustment of the latter for common allometric equations, wood density, senescence; and ground slope.

To summarise the differences between the reassessed and original calculations in Sanger and Ferrari [529], the multiplication factors are explained here. To convert from my earlier erroneous understorey allometric that they used, to the newly developed one here, without knowledge of their individual DBH measurements, I applied the ratio from Table 2(b) for the mixed-forest site assessment of Gilbert [55]: 158.9/162.5 = 0.97785 (which decreases biomass). The root:shoot ratio of 0.15 was used rather than the 0.25 in Sanger and Ferrari [529] (which decreases biomass). A wood density of 512.4 Kg m^{-3} was used rather than their 520 Kg m^{-3}. I applied an 11° ground slope to the wood measurements (which increases live and dead biomass). And I applied a 15% senescence to eucalyptus trees with DBH > 2.5 m if their height was less than the typical 75 m (assuming, in the absence of other information, that the trees are *E. regnans*) and some senescence had already been deducted by measurement of reduced height (another reduction of live plant biomass). The results are shown in Table 9.

It is worth noting the effect of different amounts of senescence in the large trees. Assuming no senescence in the larger trees, other than that originally measured (by reduced crown mass), increases the C in study-wide live biomass by 9.9 % from 742.2 to 815.5 Mg ha^{-1}, whereas assuming 20% senescence in the larger trees decreases C in study-wide live biomass by only 3.3%, to 717.8 Mg ha^{-1}. This gives another indication of likely error margins.

The error margins for the biomass values from Sanger and Ferrari [529] were earlier described as ±25%, and the corresponding absolute values for the adjusted data are shown in Table 9. The error margins assigned to their values for CWD and soil organic carbon to 0.3 m depth [in that table] were the standard error (standard deviation divided by square root of sample size (namely 4)) found experimentally. Those standard deviations could be reduced by collecting more data. But to decrease the error margins in the biomass carbon values, more intensive experimental procedures would be needed, and if still using allometric equations, then more experiments to get relevant ones, such as for mature *E. globulus* (if present).

Table 9. Carbon stocks in different pools, from Sanger and Ferrari [529] but adjusted as described in the main text, to enable comparison on a level footing with other studies in Tasmanian. The error margins for live plant biomass are ±25%, and the error margins for coarse woody debris (standing and fallen, CWD) and soil organic carbon (SOC) to 0.3 m depth, were one standard error of their empirically determined variability.

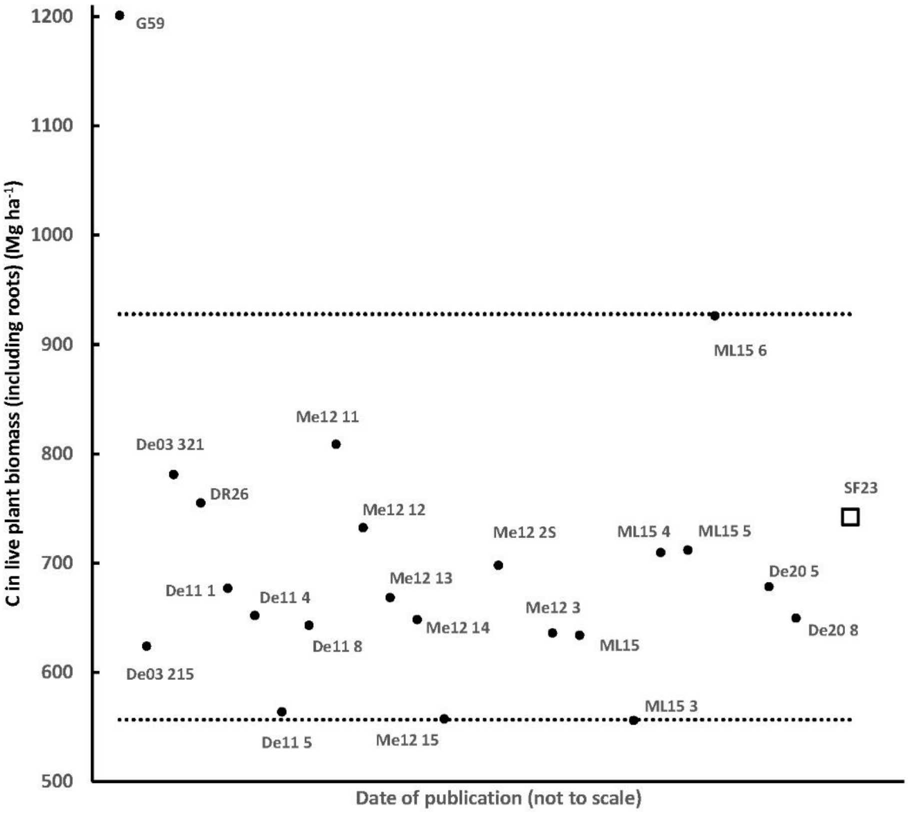
Carbon pool	C (Mg ha^{-1})	lower error margin	upper error margin
aboveground in trees > 2.5 m DBH	369.6	277.2	462.1
aboveground in trees from 1 to 2.5 m DBH	120.4	90.27	150.4
aboveground in understorey trees	155.4	116.5	194.2
roots	96.81	72.61	121.0
CWD	252.6	205.1	299.4
SOC to -0.3 m	149	88.5	209
live aboveground plant biomass	645.4	484.0	806.7
live plant biomass	742.2	556.7	927.8
total (plant biomass, plus SOC to -0.3 m)	1144	850.2	1436

6.9. Comparison of the Carbon Stocks from Different Reports

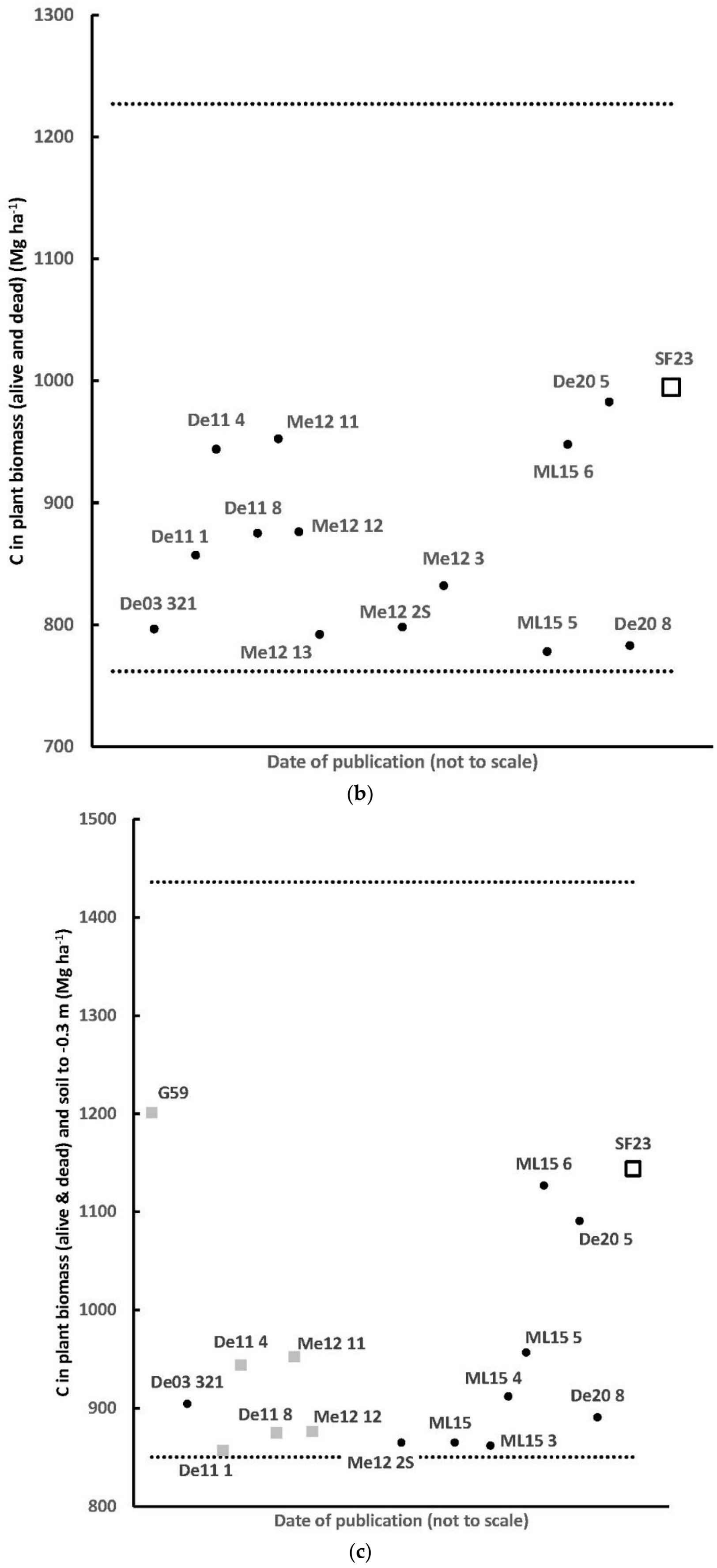
In Figure 14, the first comparison of the carbon stocks in the study plot of Sanger and Ferrari [529] with other studies, is of living plant biomass, then studies that included other pools too are shown, such as dead plant biomass and SOC. For the graphs, the tally from Sanger and Ferrari [529] was set as the benchmark to compare others against, as they claimed to have the most carbon reported in Tasmania. For ease of visualisation in the graphs error margins have only been applied to that tally from Sanger and Ferrari [529], but further on the influence of equivalent error margins on all the data being compared, is examined. There are of course error margins associated with the other reports, some of which may be as high or higher than those in Sanger and Ferrari [529]. The graphs are simpler this first way though, when there are many data points, and therefore they are easier to interpret.

The data points from reports that are centred below the lower error margin of Sanger and Ferrari [529]) are not shown in the graphs and in the table, in order to be conservative with showing possible competitors. This is also partly because, with science at its current level of development and field sampling so limited, forest-based carbon data for the tall eucalypt forests are generally too imprecise to warrant closer interrogation.

Although it is generally not permitted in scientific journal publications to show both graphs and tables of the same data, it is done so here because they provide different information. The graphs provide a useful visualisation of the distribution of values whereas Table 10 is useful for providing data for further work, such as in future comparisons. Often when retrieving data for this paper (and earlier ones) it has been necessary to use the software Datagrabber [582] to extract information from graphs, a process which adds its own error margins, in addition to those in the original data. Thus, in the interests of science both types are provided here, and it is suggested that it be considered as a scientific standard.



(**a**)



(**b**)

(**c**)

Figure 14. Comparison of carbon stocks in different pools, from Sanger and Ferrari [529] with those from other studies. The horizontal dashed lines are the error margins for Sanger and Ferrari [529], from Table 9. The code name is shown in column 4 of Table 10. In **(c)** the points that didn't have SOC

measurements adding to their total C, but still got within the error margins of Sanger and Ferrari [529] with all its pools, are shown as grey squares. Point G59 did not have debris or SOC measured, but still was within the margins in all figures.

Table 10. Summary of carbon stocks, to 3 significant figures, for different pools that are within the error margins of those in Sanger and Ferrari [529] and therefore indistinguishable from them; plus projected area and aerial photo interpretation (API) type. Note that the area may not necessarily be the maximum area, merely a sample, except for Dean, et al. [307] where the 20 ha is mostly modelled but with an empirical base for the whole stand.

report & study	Projected area (ha)	plot name	code	API type	live plant biomass (Mg ha^{-1})	live+dead plant (Mg ha^{-1})	live+dead plant, +soil (Mg ha^{-1})
[55]	0.149	Road 10	G59	E1c.M+	1200		
[307]	20	SX004C 215 years	De03 215	E2d.M-	624	644	752
[307]	20	SX004C 321 years	De03 321	E2d.M-	781	797	905
[88]	0.341	WR005D	DR26	E2c.S	755		
[542]	0.283	1	De11 1	coE1c.M+	677	857	857
[542]	0.283	4	De11 4	coE1c.M+	652	944	
[542]	1.130	5	De11 5	E2b.S	564	676	
[542]	1.120	8	De11 8	coE1c.M+	643	875	
[550] S1	0.126	1	Me12 11	E1a or E1b, E2a, E2b	809	953	
[550] S1	0.126	2	Me12 12	E1a or E1b, E2a, E2b	732	876	
[550] S1	0.126	3	Me12 13	E1a or E1b, E2a, E2b	668	792	
[550] S1	0.126	4	Me12 14	E1a or E1b, E2a, E2b	648	701	
[550] S1	0.126	5	Me12 15	E1a or E1b, E2a, E2b	558	678	
[550] S2	0.126	Styx Valley	Me12 2S	E2c.M+ & E2d.M+	698	798	865
[550] S3	5.59	Styx Valley	Me12 3	E1c.M+, E2b.M- & E2c.M-	636	832	
[532]	1.2		ML15	E2c and/or E2d	634	686	865
[532]	0.2	3	ML15 3	E2c and/or E2d	556	683	862
[532]	0.2	4	ML15 4	E2c and/or E2d	710	733	912

[532]	0.2	5	ML15 5	E2c and/or E2d	712	778	957
[532]	0.2	6	ML15 6	E2c and/or E2d	926	948	1130
[92]	1.13	5	De20 5	E1c.M+	678	983	1090
[92]	1.13	8	De20 8	E1c.M+	650	783	891
[92]	1.13	9	De20 9	E1c.M+ & M-.E1f	500	516	624
[529]	1.914		SF23	E1b.S.ER and/or E1c.S	742	995	1140

In Figure 14 the error margins are allocated only to the benchmark datum, namely the tally in Sanger and Ferrari [529]. The error margin for the biomass pool (alive and dead) was set at ±25%. If the same error had been set for the total carbon at their site then the contenders would only have to reach [100% minus 25% which is] 75% of the total, to be indistinguishable from it. If the benchmark and all the contenders have the same error margins as each other, then what percentage of the benchmark do the contenders have to reach to be indistinguishable from that benchmark? The answer, which is the error margins on the benchmark and on the test reports, is not 12.5% (25 divided by 2). It is given by the following equation:

$$em\_perc = 100\left\{{(100 - test\_perc)}\Big/{(100 + test\_perc)}\right\} \tag{12}$$

where *em_perc* is the error margin on both the benchmark and the test report (the contender) as a percentage (equal on both) and test_perc is what percentage of the benchmark datum, the test report datum has to reach to be indistinguishable from the benchmark. If the report datum plus the em_perc error margin surpasses the benchmark datum minus the em_perc error margin, then the benchmark and test are indistinguishable. Also, if one subtracts em_perc from the benchmark datum and adds it to the test report datum, and if the two results are equal then the two datums are indistinguishable.

From Equation 12, the value equivalent to ±25% error on the benchmark Sanger and Ferrari [529] but none on the test reports, is 2500/175= ±14.28571% error margin on the benchmark and on the test reports. All the reports mentioned in this section were tested, by assigning that error margin to both them and to the benchmark, and the result was the same as in Table 10 and Figure 14. As people are used to the base 10 system and multiples of it, for the sake of making visualisation easier, the ±14.28571% can be approximated to ±15%. It then should become obvious to people who routinely work in forest carbon accounting, that this error margin (15% on all reports, or ±25% on only the benchmark) is conservative. For example, consider the errors, starting with running a tape measure through a forest on a hillside, placing a tape at 1.3 m aboveground around the buttress of mature trees, reading the tape, and the errors in allometric equations (such as can be inferred from Figures 9 and 10) and errors in estimating the degree of senescence, in trees that are afflicted.

If one knows the error margin on both the test report and benchmark and wants to know what fraction of the benchmark a test report has to reach to be comparable with it, then that is given by:

$$test\_perc = {(100 - em\_perc)}\Big/{\left\{1 + {em\_perc}/{100}\right\}} \tag{13}$$

where *em_perc* and *test_perc* are as in Eq. 12. For example, if the error margin on both reports is 20% then the test report datum only has to be 80/1.2 = 66.666 (reoccurring) % of the benchmark datum to be indistinguishable from it. The two latter equations incidentally represent an arc of a circle of radius $\sqrt{(150^2+150^2)}$ and the various possibilities are best shown in Figure 15.

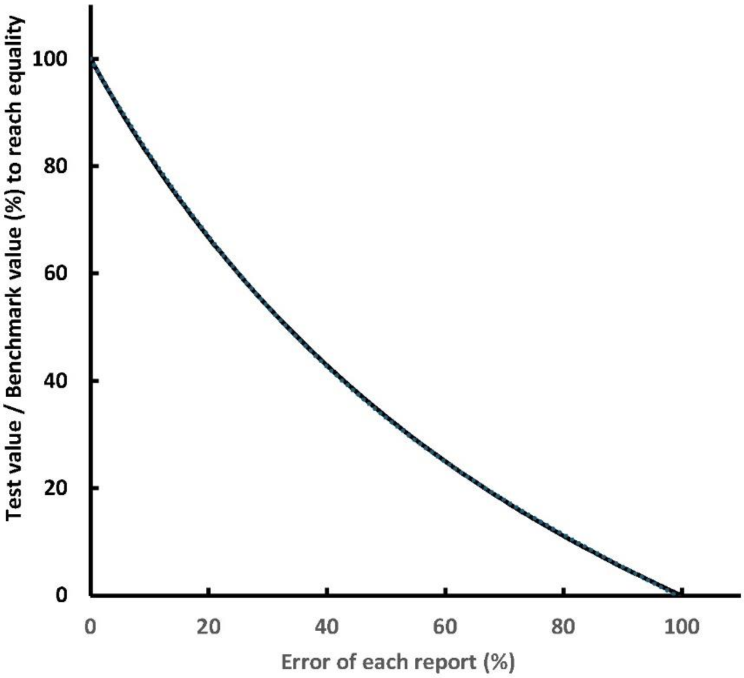


Figure 15. Relationship between the fraction of the benchmark report datum that a test report has to reach (%), and the error margins on both reports (%) for the two reports, for them to be indistinguishable. This simplifies comparisons of various reports if one can assume equal error margins for them. The curve is given mathematically in Equations 12 and 13.

The error margins for these data types, as mentioned in section 2, are partly genuine error margins, in the physics sense, and partly indicative of natural variation in biology. Most often in biology the former type are not shown, only the latter, which accrues as a range of values due to unmeasured influences. Other places that genuine measurement errors are not mentioned are, for example, in financial matters, such as a domestic electricity bill, because then one could apply an equivalent margin when paying it. In biology the natural variation is usually assumed to be much larger than the measurement error and it is the only one cited, usually in the form of a standard deviation or standard error, as in the soil carbon value in Sanger and Ferrari [529]. That method assumes that sufficient samples have been acquired to include all relevant effects, and that similar effects operate elsewhere. However, when making comparisons, such as of carbon stocks, then stating an indicative error margin is much more important, the measurement error should be included too. Therefore, the overall ~±15% error margin that is used here (or the ±25% on only the benchmark) is conservative.

The SOC in different studies was measured to different depths. To provide a comparison it was therefore necessary to trim the depth of soil being considered to the minimum, which was -0.3 m in Sanger and Ferrari [529] and possibly also in Moroni and Lewis [532]. Although Sanger and Ferrari [529] claimed to be the most comprehensive carbon study of Tasmanian forests they did not measure as deeply as did some others, didn’t include SOC in moist stones and only had four sample points, but they did understand that latter shortcoming. Their soil carbon section was better than some of the earlier reports in that it used elemental analysis for carbon rather than the Walkley-Black method and it did include carbon in large charcoal particles. In future work, if for different locations it becomes routine to examine SOC further down or laterally, and significantly different amounts of SOC are there or have been translocated by water, then the competition would have to be rerun and it may provide better discrimination.

Regarding the competition declared by Sanger and Ferrari [529] of the highest carbon stocks in Tasmania— Figure 14 shows that they are no longer the clear winners, but approximately equal with several other sites. Additionally, if SOC and CWD had been measured for the site in Gilbert [55], then it may have the most carbon. The large error margins and natural variability in the current state of carbon accounting science for mature forests mean that many different sites are indistinguishable (Figure 14), a situation that was also noted in forest carbon comparisons nearly 20 years ago [88], though detail has improved since then.

One must also consider the size of the area sampled and its randomness of selection. The 20 ha for the site in Dean, et al. [307] is the largest in Table 9, but that was mostly a simulation study, so it can be disregarded. Several carbon studies didn't reach the contenders table, perhaps because of their larger size. That leaves, from Table 9, the area of 5.59 ha for the Styx Valley part of Study 3 in May, et al. [550] as the only study site larger than the 1.914 ha of Sanger and Ferrari [529]. But the allometric equations used in the former study may not be comparable, e.g., the high-yielding temperate rainforest equation of Keith, et al. [124] may have been used. Notably, some of the same raw data were used in both May, et al. [550] and Dean, et al. [92], but the tally was lower in the latter study, which also included a larger area.

The 0.149 ha site in the Florentine Valley of Gilbert [55], measured before almost the entire valley was logged, has arguably the highest carbon stock to date for a Tasmania forest, but a relatively small area. However, it was possibly selected because, rather than being an extreme case, it was representative of high biomass forests that existed prior to logging. Logging records or the Styx and Florentine Valleys show that typical forest stands of specific height cohort and with a high wood volume were typically each from 1 to 15 ha in size [541]. The average size of the stands of the same forest type, E1c.M+ as measured in Gilbert [55], from the aerial phototype mapping of Forestry Tasmania, was ~21 ha (standard deviation 29 ha). These figures give a likely indication for the extent of the stand in Gilbert [55]. Nevertheless, neither that study nor Sanger and Ferrari [529] had selected their study plots totally at random within a forest type. Random site selection is partly interchangeable with studying a larger area, and so the Study 3 in May, et al. [550] with its larger area comes into consideration again, but it still raises scepticism for the reasons mentioned earlier. Overall then, the benchmark and contenders in Table 9 and Figure 14 remain indiscernible, except possibly for the larger area in Sanger and Ferrari [529], i.e., unless the size of their plot was also non-random, not just its location. If the size was not random, then for a more level comparison the area limits of forest stands sampled in the other studies in Table 9 would need to be determined. On balance, questions that consequently arise are: how much of the 'Grove of Giants' is represented by the ~1.9% of it sampled; is the majority of it much lower C, equivalent or higher? and similarly for the surrounds of other sites in Table 9.

A pertinent question from a carbon conservation perspective is: where else in Tasmania are there high carbon forests? Table 10 indicates that it's likely to be in forests with either E1 or E2 phototype and with at least a 20% crown cover of eucalypts. For example, there is an area of forest similar in size and appearance, as seen from GoogleEarth ©, similar to the 'Grove of Giants', ~3.2 km to the north. Further afield, using an API-type map of 2012 from the Tasmanian State forestry agency, there was ~42,400 ha of either mature E1 or E2 with eucalypt crown coverage of at least 50%, left standing within Tasmania's original southern central pulpwood concession boundaries. After the logging and the WHA extension of the late 1980s there was ~32,700 ha left in State forest and after further logging and the 2013 WHA extension there was ~13,300 ha left in State forest to be logged. From GIS analysis of the 2012 data, logging appears to have been concentrated in the Florentine, Tyenna and Styx Valleys and the largest contiguous stands of primary, mature E1 and E2 forest that now remain, are in at the northern and southern extremities of the original pulpwood concession, such as near the 'Grove of Giants', a little to north in the Russell River area, further south in the Kermandie catchment and north of the main Florentine catchment near Wayatinah. As the conservation activities for the 'Grove of Giants' stand show, conservation effort appears to have switched from the centre of the Florentine and Styx pulpwood concession since some of that central part of it formed part of the WHA extension of 2013. Its focus has recently intensified on the large individual trees, perhaps because both the general public and the State forest agency have revealed a sensitivity on that topic, and because the large trees are probably emblems of contrast with current human civilisation. They are current levers to gain conservation areas— but smaller and smaller examples become icons as the larger ones fall to various effects of the inexorable human consumption of nature. How then to most expediently identify the prominent stands? Apart from simply walking through Tasmania's forests, LIDAR is currently a common remote sensing method for determining tree height and this could be combined with stand density (the forestry term for the number of trees per hectare) to find likely stands of high carbon stock, but it would need ground-truthing to determine the degree of hollow formation in the trees. Height alone may not be sufficient though for high carbon mass and biodiversity. Stand density will play a part in carbon accumulation but may detract from biodiversity. For example the highest concentration of the tallest trees was in the Andromeda block in the Styx Valley Kostoglou [87], but their diameters were below average, which indicated a high stand density even when mature— they may have germinated from an atypically intense burn which limited early regeneration of the rainforest understorey and later growth of large hemi-epiphytes. When viewing sites with potentially high carbon mass, there will be a trade-off between stand density (including old and recent fire effects), senescence (which may be accompanied by regeneration to for mixed-aged stands and more carbon, as in Keith, et al. [309]) and basal diameter (which can allow dominance of the carbon tally from a few individuals), and it's likely that site contention can only be solved through thorough measurement.

6.10. The Proclamation of Giant Trees and HIGH carbon

There may be a few reasons for trying to find a specific forest site with a high carbon mass or 'high carbon', such as: (a) to describe the nature in a particular region, (b) to help calibrate and parameterise a carbon forecasting model for a particular forest type as a function of age, (c) to add to the database of the global carbon distribution, (d) to alert and motivate members of the public who are interested in conserving such sites, (e) to describe the limits on how much carbon stock can be sustained overtime and for how long, and (f) it is likely to be a target of corporate forestry as merchantable biomass is more concentrated and it is therefore of public interest to calculate the anthropogenic carbon emissions.

From comments in Sanger and Ferrari [529] and by the TallTreeProjects in public media [570,571] it can be inferred that they align most strongly with option (d). An example is the frequent use of a strong adjective 'ultra-large':

'Ultra-large trees (>2.5 m DBH) were measured throughout the entire plot and smaller trees, understory and coarse woody debris were measured within four transects within the plot following methodology adapted from Keith et al. (2014; Figure 1).' [529]

Also, in their Table B2 they refer to smaller trees as: 'Large trees (1–2.5 m DBH)'. The maximum dimensions of some typical eucalypts in such forests were given in section 2. The maximum tree dimensions in Sanger and Ferrari [529] are a DBH of 5.12 m, a height of 72 m, and a volume of 254 m^3—not 'ultra-large', but near the median in diameter and average height, if *E. regnans*. In Sanger and Ferrari [529] any tree with a DBH >2.6 m is called 'ultra-large'. That prompts the question as to what genuinely large trees could be termed. Trees in tall open forests in Tasmania were first publicly called giants when their forests were allocated to newspaper production: the *E. regnans*—dominated forests of the Styx/Tyenna/Florentine pulpwood concession: 'Giant trees will build new industry. Newsprint from Tasmanian forests' [100]. But that was in a newspaper (not a scientific paper), where titles are designed to attract readership, for financial competitiveness. The terminology 'ultra' may have been for public relations purposes, to create a vision of something unprecedented and extraordinary, to prompt support for conservation. The commercial attention to the high organic mass of the site has consequently moved from developing products from it [through logging and milling] and inadvertently breaking down its molecules into smaller ones such as atmospheric greenhouse gasses, to commercial tourism with the molecular structure in its current configuration.

Use of the internet in forest-use controversies is not new however, but the ease of publication of videos and photographs is more recent. In the early 1990s a public 'listserver' could be used, which was the precursor of the modern-day Facebook discussion group. Listservers could not be accessed on mobile phones at the time, only via computer, and without digital photographs and video data on that publication medium, there was much less public viewing and consequently less public interest.

Some other tools used in Sanger and Ferrari [529] are also revealing. The name 'Grove of Giants' may be an adaptation of the 'Grove of Titans', which is a stand of Coastal Redwoods (*Sequoia sempervirens*) in northern California with a high mass of carbon that has been the subject of extensive scientific study and is part of Jedediah Smith Redwoods State Park. It was named in 1998 and initially studied for canopy ecology then later for carbon content and the number of people visiting it has increased significantly via internet social media [580,583,584]. The redwoods form more of a true grove however, by the definition of the word, as the forest floor is much barer than in the 'Grove of Giants'. There are other similarities between the tall trees of Tasmania and the redwoods of the USA: they've both had tree carbon studied by tree climbing and they've both been the focus of corporate forestry for timber extraction [100,572,585 p106]. The competitive declaration for a high carbon stock is also not new but appears to have been initiated by Keith, et al. [309] for *E. regnans*—dominated forests in Victoria (Australia), with the title 'Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests', which can be compared with the title "The Grove of Giants: Tasmanias' most carbon-dense forest" from Sanger and Ferrari [529]. That global claim for Victorian *E. regnans*, similarly didn't include error margins in their comparison, and the carbon stock was possibly surpassed by *S. sempervirens* forests in several locations, depending on error margins [580]. Error margins have been included in some carbon comparisons [e.g., 586 Figure 4, 587 Table 3].

The area surrounding the study plot of Sanger and Ferrari [529] has larger trees, e.g., a *E. globulus* of diameter 5.35 m and height ~80 m [588]. Advancement of conservation however didn't require measurement of carbon in that larger area— the strong public media presence induced the State forest agency to reserve the surrounding area from logging [589]. But there is likely to be [carbon] leakage— the State forestry agency is likely to log primary forest elsewhere instead, with less media attention, to get that same timber volume, if it still has the same annual quota to fulfil. A media report says that this did indeed occur [589].

7. Conclusions

Several possible flaws in science were examined above and corrections provided, including: mis-citing scientific literature, inadequate literature review, not recognising relevant spatial dimensions and time, and inappropriate or missing benchmarks. The peer review process for scientific publications is only part of quality control. Added to that is a background of science developed for specific purposes, but with purposes that are in the process of change as human society expands and depletes parts of the biosphere. These three items (flaws, history and changing purpose) combine to create a struggle for science. Although it is always disheartening to realise a flaw in one's published work, in the longer term it is to the benefit of the reputation of science to note flaws in scientific studies and to provide corrections. It must be done for both industry-promoting science and for conservation-promoting science, as in this review. This approach can help to assure the public and other scientists that extra quality control is being performed, above the peer review process. This in turn should allow more confidence by the public in findings of science, such as when it reveals trends not physically observed within a typical human lifetime, for example the many different contributions to anthropogenic climate change and its acceleration. This in turn should permit more climate change mitigation activities.

There has so far been insufficient awareness of the value of nature to prevent its continuing decline [590]. For appreciation of humanity's impact on forests so far, and the ongoing legacy of anthropogenic climate change that current and future generations will attend to [590], it is necessary for the public to recognise the dimensions and composition of forests that we have changed, and what vestiges of them remain. For appropriate recognition of that legacy, overplaying of remaining tree size or overshadowing what has been removed before, must be avoided and corrected, as indeed must underestimation of carbon stocks by not counting parts that are difficult to measure. Better accounting can help appropriate recognition, which could in turn help avoid premature forest conversion to greenhouse gasses. All the dimensions, current and past, need to be viewed upon one time axis, otherwise each successive generation of people may believe that they are operating without an undercurrent (they may assume the 0 Mg ha^{-1} benchmark) and that the vestigial primary forests can provide them with resources similar what their predecessors used. This belief can be promoted in part by the term sustainability, which as shown in this paper, has not been achieved for industrial use of Australian forests. Instead, the legacy which constitutes the background must be visible and acknowledged, to show the situation in a comprehensive light, and this will in turn show that many extractions from nature, are cumulative rather than unique.

The uncertainties and error margins examined above mean that numerous improvements are needed in scientific data to help improve recognition and interpretation of nature, such as for climate change forecasts and climate change mitigation. For example, more data on the physical and chemical routes and precision for carbon fluxes associated with decomposing trees and wood products (including, for example, recycled paper), and the translocation of SOC vertically and horizontally via water flow. The current work showed that more effort is required to increase knowledge about soil carbon stocks and change, which concurs with Makarieva, et al. [591]. More applied research could also be invested in recycling of wood products and determining their LCA, including building materials and paper, to reduce demand for forest conversion [592]. Some change to science is also needed, to ensure that an extra layer of quality control is enacted for some topics, such as the forest carbon cycle. This could be in part, through increased deposition of data at the publication stage, as is done in some other fields. Possibly also, society in general can add to the available data by recording basic data for all medium to large trees. This is done to a degree already in some places, such as for urban street trees. Forest agencies can partake by, for example, recording data on all trees felled during logging, merchantable or not.

Some comment on climate change mitigation and forestry is appropriate, in the context of this paper. If less area of primary forest is to be converted to secondary forest logging cycles, and the corresponding carbon emission is to be reduced, then in the short-term either the timber volume quota must be addressed, or swapped to plantation-grown timber [135 p213-214], and supplemented with recycled timber. A different solution appears from a more comprehensive viewpoint: the continually increasing global human population within a limited area that is currently shared with other species [593], suggests that there is no reason to assume that other plants and animals can provide the raw materials necessary for humans into the future. Therefore, a longer-term source of materials such as timber, from other than forests or plantations must be sought, such as synthetic timber. This is underway and although partly adopted, is currently more at the research than the production stage. It is often based on recycled plastics rather than on copying the molecular structure of wood, although the meso-scale (> nanometre) has been copied, with promising results [e.g., 594,595,596]. (With plastic-like products one must of course have a 100% non-discard policy, due to their effects on other species.) A related product, with a similar precipitating cause, is synthetic food such as cultured protein, for human consumption, but that is also still under development [597]. This swap of resource inputs, may span both 'degrowth' and 'green growth' concepts [598], as it maintains productivity and supply. More applied research could also be invested in recycling of wood products and determining their LCA, including building materials and paper, to reduce demand for forest conversion [592]. Makarieva, et al. [591] recommend not using 'intact forests' for timber production but instead for climate control, as they still have that innate capacity, which is provided through balances between ecosystem components. They recommend that timber be sourced from areas where the genetics of the original ecosystem are unavailable to recreate the ecosystem within reasonable timeframes, e.g., plantations and other cropland, as such locations have already lost their climate control ability. These alternative routes will be part of the mix to accompany improvements in science.

Funding: This research received no external funding.

Acknowledgments: The author would like to thank: Richard Doyle for help interpreting the older soil surveys; Andrew Norton for discussions on bioenergy; Jamie Kirkpatrick for comments on an earlier version of the manuscript and on approaches to science; Forestry Tasmania for access to forests, documents and data; Fabio Ximenes and Stephen Sillett for discussions on their work; and the University of Tasmania for access to software and their library.

Data Availability Statement: Any raw data supporting the conclusions of this article will be made available by the author on request, except GIS data which was from a commercial source.

Conflicts of Interest: The author declares no conflicts of interest.

References

1. UNFCCC. Nationally determined contributions under the Paris Agreement. In Proceedings of the Conference of the Parties serving as the meeting of the Parties to the Paris Agreement. Third session, Glasgow, Scotland, 2021; p. 42.
2. Stoddard, I.; Anderson, K.; Capstick, S.; Carton, W.; Depledge, J.; Facer, K.; Gough, C.; Hache, F.; Hoolohan, C.; Hultman, M.; et al. Three decades of climate mitigation: Why haven't we bent the global emissions curve? *Annual Review of Environment and Resources* **2021**, *46*, 653-689, doi:10.1146/annurev-environ-012220-011104.
3. Jamieson, D. *Reason in a Dark Time. Why the Struggle Against Climate Change Failed.*; Oxford University Press: New York, USA., 2014; p. 288.
4. Kennelly, C.; Berners-Lee, M.; Hewitt, C.N. Hybrid life-cycle assessment for robust, best-practice carbon accounting. *Journal of Cleaner Production* **2019**, *208*, 35-43, doi:10.1016/j.jclepro.2018.09.231.
5. Barredo, J.I.; Brailescu, C.; Teller, A.; Sabatini, F.M.; Mauri, A.; Janouskova, K. *Mapping and assessment of primary and old-growth forests in Europe*; Publications Office of the European Union: Luxembourg, 2021; Volume EUR 30661 EN, p. 43.
6. Pernica, M. Forestry students and workers and their attitudes towards the environment. *Journal of Forest Science* **2004**, *50*, 47-54, doi:10.17221/4600-JFS.
7. Kirkpatrick, J.B. The politics of the media and ecological ethics. In *Ecology for Everyone. Communicating ecology to scientists, the public and the politicians*, Wills, R., Hobbs, R., Eds.; Surrey Beatty & Sons: Sydney, 1998; pp. 36-41.
8. Higgins, P.A.T.; Chan, K.M.A.; Porder, S. Bridge over a philosophical divide. *Evidence & Policy* **2006**, *2*, 249-255, doi:10.1332/174426406777068920.
9. Donald, R.; Graves, L. Making them pay: Comparing how environmental facts matter in two accountability contexts. *New Media & Society* **2024**, *14614448241279250*, doi:10.1177/14614448241279250.
10. Kozik, R.; Pawlicka, A.; Pawlicki, M.; Choraś, M.; Mazurczyk, W.; Cabaj, K. A meta-analysis of state-of-the-art automated fake news detection methods. *IEEE Transactions on Computational Social Systems* **2024**, *11*, 5219-5229, doi:10.1109/TCSS.2023.3296627.
11. Cook, J.; Oreskes, N.; Doran, P.T.; Anderegg, W.R.L.; Verheggen, B.; Maibach, E.W.; Carlton, J.S.; Lewandowsky, S.; Skuce, A.G.; Green, S.A.; et al. Consensus on consensus: a synthesis of consensus estimates on human-caused global warming. *Environmental Research Letters* **2016**, *11*, 1-7, doi:10.1088/1748-9326/11/4/048002.
12. Dunlap, R.E. Climate change skepticism and denial: an introduction. *American Behvioural Scientist* **2013**, *57*, 691-698, doi:10.1177/0002764213477097.
13. Oreskes, N.; Conway, E.M. *Merchants of Doubt: How a Handful of Scientists Obscured the Truth on Issues from Tobacco Smoke to Global Warming*; Bloomsbury Publishing: London, 2010; p. 274.
14. Anderegg, W.R.L.; Trugman, A.T.; Badgley, G.; Anderson, C.M.; Bartuska, A.; Ciais, P.; Cullenward, D.; Field, C.B.; Freeman, J.; Goetz, S.J.; et al. Climate-driven risks to the climate mitigation potential of forests. *Science* **2020**, *368*, 1-9, doi:10.1126/science.aaz7005.
15. Friedlingstein, P.; Dufresne, J.-L.; Cox, P.M.; Rayner, P. How positive is the feedback between climate change and the carbon cycle? *Tellus B* **2003**, *55*, 692-700.
16. Kovalyuk, B.P.; Skorenkyy, Y.L.; Nikiforov, Y.M. *Physics Laboratory*; Ternopil Ivan Puluj National Technical University: Ternopil, Ukraine, 2016; p. 130.
17. Bruce, D.; Schumacher, F.X. *Forest Mensuration*; McGraw Hill: New York, 1942.
18. Halsey, L.G. The reign of the p-value is over: what alternative analyses could we employ to fill the power vacuum? *Biology Letters* **2019**, *15*, 1-8, doi:10.1098/rsbl.2019.0174.
19. Leo, G.D.; Sardanelli, F. Statistical significance: p value, 0.05 threshold, and applications to radiomics—reasons for a conservative approach. *European Radiology Experimental* **2020**, *4*, 1-8, doi:10.1186/s41747-020-0145-y.
20. Gibson, C.C.; Ostrom, E.; Ahn, T.K. The concept of scale and the human dimensions of global change: a survey. *Ecological Economics* **2000**, *32*, 217-239, doi:10.1016/S0921-8009(99)00092-0.
21. Cash, D.W.; Adger, W.N.; Berkes, F.; Garden, P.; Lebel, L.; Olsson5, P.; Pritchard, L.; Young, O. Scale and cross-scale dynamics: governance and information in a multilevel world. *Ecology and Society* **2006**, *11*, 1-12.
22. Newell, J.P.; Vos, R.O. Accounting for forest carbon pool dynamics in product carbon footprints: Challenges and opportunities. *Environmental Impact Assessment Review* **2012**, *37*, 23-36, doi:10.1016/j.eiar.2012.03.005.

23. Neff, M.W.; Corley, E.A. 35 years and 160,000 articles: a bibliometric exploration of the evolution of ecology. *Scientometrics* **2009**, *80*, 657-682, doi:10.1007/s11192-008-2099-3.
24. Seppelt, R.; Dormann, C.F.; Eppink, F.V.; Lautenbach, S.; Schmidt, S. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. *Journal of Applied Ecology* **2011**, *48*, 630-636, doi:10.1111/j.1365-2664.2010.01952.x.
25. Lautenbach, S.; Mupepele, A.-C.; Dormann, C.F.; Lee, H.; Schmidt, S.; Scholte, S.S.K.; Seppelt, R.; Teeffelen, A.J.A.v.; Verhagen, W.; Volk, M. Blind spots in ecosystem services research and challenges for implementation. *Regional Environmental Change* **2019**, *19*, 2151-2172, doi:10.1007/s10113-018-1457-9.
26. NRMjobs; SST. *Advert ID: 20015822. Sustainable Timber Tasmania. Carbon & Ecosystem Services Analyst.*; NRMjobs: Aldgate, South Australia, 2023.
27. Rohlf, D.J. Scientists, agendas, and litigation: a response to Peery et al. *Frontiers in Ecology and the Environment* **2019**, *17*, 555-556, doi:10.1002/fee.2130.
28. Gutiérrez, R.; Jones, G.; Redpath, S.M.; Franklin, A.B.; Simberloff, D.; Turner, M.G.; Radeloff, V.C.; White, G.C.; Peery, M.Z. Reinforcing the concept of agenda-driven science: a response to Rohlf. *Frontiers in Ecology and the Environment* **2019**, *17*, 556-557, doi:10.1002/fee.2131.
29. Noon, B.R.; Murphy, D.D. CASE STUDY 3 Management of the Spotted Owl: The Interaction of Science, Policy, Politics, and Litigation. In *Principles of Conservation Biology*, Carroll, G.K.M.C.R., Ed.; Sinauer Associates, Inc. : Sunderland, Massachusetts, 1994; pp. 380-388.
30. Wilhere, G.F.; Maguire, L.A.; Scott, M.; Rachlow, J.L.; Goble, D.D.; Svancara, L.K. Conflation of values and science: response to Noss et al. *Conservation Biology* **2012**, *26*, 943–944, doi:10.1111/j.1523-1739.2012.01900.x.
31. Noss, R.F.; Dobson, A.P.; Baldwin, R.; Beier, P.; Davis, C.R.; Dellasala, D.A.; Francis, J.; Locke, H.; Nowak, K.; Lopez, R.; et al. Bolder thinking for conservation. *Conservation Biology* **2012**, *26*, 1-4, doi:10.1111/j.1523-1739.2011.01738.x.
32. Budaev, S.; Jørgensen, C.; Mangel, M.; Eliassen, S.; Giske, J. Decision-making from the animal perspective: Bridging ecology and subjective cognition. *Frontiers in Ecology and Evolution* **2019**, *7*, 1-14, doi:10.3389/fevo.2019.00164.
33. Waisbord, S. Intertwining science journalism with (post)development. *Journalism Studies* **2024**, *25*, 575-582, doi:10.1080/1461670X.2023.2201862.
34. McKay, R.T.; Dennett, D.C. The evolution of misbelief. *Behavioural and Brain Sciences* **2009**, *32*, 493-561, doi:10.1017/S0140525X09990975.
35. Bacon, F. *Novum Organum Scientarium*; Wirceburgi: Würzburg, Germany, 1779 edition, 1620.
36. Bolles, E.B. *Galileo's Commandment. An Anthology of Great Science Writing*; Little Brown and Company: London, 1999; p. 485.
37. Fortunato, S.; Bergstrom, C.T.; Börner, K.; Evans, J.A.; Helbing, D.; Milojević, S.; Petersen, A.M.; Radicchi, F.; Sinatra, R.; Uzzi, B.; et al. Science of science. *Science* **2018**, *359*, 1-7, doi:10.1126/science.aao0185.
38. Beck, C. Quality control in science. Available online: https://www.mpg.de/12138624/quality-control-in-science (accessed on 19-September-2024).
39. National Health and Medical Research Council, A.R.C., Universities Australia,. *Australian Code for the Responsible Conduct of Research 2018*; Commonwealth of Australia: Canberra, Australia, 2018; p. 7.
40. Elsevier. How to conduct a review. Available online: https://www.elsevier.com/en-au/reviewer/how-to-review (accessed on
41. National Academies of Sciences Engineering and Medicine. *Reproducibility and Replicability in Science*; The National Academies Press: Washington, D.C., 2019; p. 257.
42. Kusumoto, F.M.; Bittl, J.A.; Creager, M.A.; Dauerman, H.L.; Lala, A.; McDermott, M.M.; Turco, J.V.; Taqueti, V.R.; Fuster, V. Challenges and controversies in peer review. *Journal of The American College of Cardiology* **2023**, *82*, 2054-2062, doi:10.1016/j.jacc.2023.08.056.
43. Eckert, M. *Safeguarding peer review to ensure quality at scale*; Frontiers Media SA, Switzerland: Lausanne, Switzerland, 2024.
44. Webster, H. *History of the Modern World*; D. C. Heath & Co.: Boston, USA, 1930; p. 881.
45. Fraser, G. *The New Physics for the Twenty-First Century*; Cambridge University Press: Cambridge, England, 2006; p. 548.
46. Bacon, F. *Essays Civil and Moral*; Cassell: London, 1905; p. 192.
47. Dean, C.; Wardell-Johnson, G.W.; Kirkpatrick, J.B. Are there any circumstances in which logging primary wet-eucalypt forest will not add to the global carbon burden? *Agricultural and Forest Meteorology* **2012**, *161*, 156-169, doi:10.1016/j.agrformet.2012.03.021.
48. Lewin, D.W. *The Eucalypti Hardwood Timbers of Tasmania, and the Tasmanian Ornamental and Softwood Timbers. A Descriptive Treatise on their Commerical Uses, with Reports on their Durability.* ; Gray Brothers: Hobart, Tasmania, 1906; p. 139.
49. Wood, L.J.; Kirkpatrick, J.B. The allocation of rights to public forests in Tasmania: a geographical critique. *Applied Geography* **1984**, *4*, 215-234, doi:10.1016/0143-6228(84)90034-1.
50. Gray, H.R. State forests of the Empire: Tasmania. *Empire Forestry Journal* **1939**, *18*, 202-211.

51. Helms, A.D. A giant eucalypt (*Eucalyptus regnans*) Nichols Spur, Junee, Derwent Valley, Tasmania. *Australian Forestry* **1945**, *9*, 24-28.
52. ANM. *Working Plan For Concession Area Of Australian Newsprint Mills Limited In The Derwent Valley*; Australian Newsprint Mills Limited, Boyer: Boyer, Tasmania, 1979; p. 45.
53. Specht, R.L. Vegetation. In *The Australian Environment'*, 4th ed.; Leeper, G.W., Ed.; CSIRO and Melbourne University Press: Melbourne, 1970.
54. Kirkpatrick, J.B.; Peacock, R.J.; Cullen, P.J.; Neyland, M.G. *The Wet Eucalypt Forests of Tasmania*; Tasmanian Conservation Trust Hobart: Hobart, 1988.
55. Gilbert, J.M. Forest succession in the Florentine Valley, Tasmania. *papers and Proceedings of the Royal Society of Tasmania* **1959**, *93*, 129-151.
56. Kirkpatrick, J.B.; DellaSala, D.A. Temperate Rainforests of Australasia. In *Temperate and Boreal Rainforests of the World: Ecology and Conservation*, DellaSala, D.A., Ed.; Island Press: Washington DC, 2011; pp. 195-212.
57. Fedrigo, M.; Kasel, S.; Bennett, L.T.; Roxburgh, S.H.; Nitschke, C.R. Carbon stocks in temperate forests of south-eastern Australia reflect large tree distribution and edaphic conditions. *Forest Ecology and Management* **2014**, *334*, 129-143, doi:10.1016/j.foreco.2014.08.025.
58. Petrie, A.H.K.; Jarrett, P.H.; Patton, R.T. The vegetation of the Blacks' Spur region: A study in the ecology of some Australian mountain eucalyptus forests: I. The mature plant communities. *Journal of Ecology* **1929**, *17*, 223-248, doi:10.2307/2256042.
59. Ash, J. The location and stability of rainforest boundaries in north-eastern Queensland, Australia. *Journal of Biogeography* **1988**, *15*, 619-630, doi:10.2307/2845440.
60. Zhuang, P.; Gao, X.-M. The concept of the Rainy Zone of West China and its significane to the biodiveristy conservation of China. *Biodiversity Science* **2002**, *10*, 339-344.
61. Barni, P.E.; Manzi, A.O.; Condé, T.M.; Barbosa, R.I.; Fearnside, P.M. Spatial distribution of forest biomass in Brazil's state of Roraima, northern Amazonia. *Forest Ecology and Management* **2016**, *377*, 170-181, doi:10.1016/j.foreco.2016.07.010.
62. Asselin, H.; Payette, S. Late Holocene opening of the forest tundra landscape in northern Québec, Canada. *Global Ecology and Biogeography* **2005**, *14*, 307–313, doi:10.1111/j.1466-822x.2005.00157.x.
63. Marques, E.Q.; Marimon-Junior, B.H.; Marimon, B.S.; Matricardi, E.A.T.; Mews, H.A.; Colli, G.R. Redefining the Cerrado–Amazonia transition: implications for conservation. *Biodiversity and Conservation* **2020**, *29*, 1501-1517, doi:10.1007/s10531-019-01720-z.
64. Clarke, P.J.; Lawes, M.J.; Midgley, J.J.; Lamont, B.B.; Ojeda, F.; Burrows, G.E.; Enright, N.J.; Knox, K.J.E. Resprouting as a key functional trait: how buds, protection and resources drive persistence after fire. *New Phytologist* **2013**, *197*, 19-35, doi:10.1111/nph.12001.
65. Casson, P.B. The forests of western Tasmania. *Australian Forestry* **1952**, *16*, 71-86, doi:10.1080/00049158.1952.10675293.
66. Dixon, D. A taxonomic revision of the Australian Ficus species in the section Malvanthera (Ficus subg. Urostigma: Moraceae). *Telopea* **2003**, *10*, 125-153, doi:10.7751/telopea20035611.
67. Dean, C.; Kirkpatrick, J.B.; Osborn, J.; Doyle, R.B.; Fitzgerald, N.B.; Roxburgh, S.H. Novel 3D-geometry and models of the lower regions of large trees for use in carbon accounting of primary forests. *AoB Plants* **2018**, *10*, 1-19, doi:10.1093/aobpla/ply015.
68. Furlaud, J.M.; Prior, L.D.; Williamson, G.J.; Bowman, D.M.J.S. Fire risk and severity decline with stand development in Tasmanian giant *Eucalyptus* forest. *Forest Ecology and Management* **2021**, *502*, 1-17, doi:10.1016/j.foreco.2021.119724.
69. Kirkpatrick, J.B.; Brown, M.J. A numerical analysis of Tasmanian higher plant endemism. *Botanical Journal of the Linnean Society* **1984**, *88*, 165-183, doi:10.1111/j.1095-8339.1984.tb01569.x.
70. Frost, E.E.; Levia Jr., D.F. Hydrologic variation of stemflow yield across co-occurringdominant canopy trees of varying mortality. *Ecohydrology* **2014**, *7*, 760-770, doi:10.1002/eco.1397.
71. Levia Jr., D.F.; Frost, E.F. A review and evaluation of stemflow literature in the hydrologic and biogeochemical cycles of forested and agricultural ecosystems. *Journal of Hydrology* **2003**, *274*, 1-29.
72. Levia Jr., D.F.; Van Stan, I.I.J.T.; Inamdar, S.P.; Jarvis, M.T.; Mitchell, M.J.; Mage, S.M.; Scheick, C.E.; McHale, P.J. Stemflow and dissolved organic carbon cycling: temporal variability in concentration, flux, and UV-Vis spectral metrics in a temperate broadleaved deciduous forest in the eastern United States. *Canadian Journal of Forest Research* **2012**, *42*, 207-216, doi:10.1139/x11-173.
73. Van Stan II, J.T.; Siegert, C.M.; Levia Jr., D.F.; Scheick, C.E. Effects of wind-driven rainfall on stemflow generation between codominant tree species with differing crown characteristics. *Agricultural and Forest Meteorology* **2011**, *151*, 1277-1286, doi:10.1016/j.agrformet.2011.05.008.
74. Oyarzún, C.E.; Godoy, R.; Staelens, J.; Donoso, P.J.; Verhoest, N.E.C. Seasonal and annual throughfall and stemflow in Andean temperate rainforests. *Hydrological Processes* **2011**, *25*, 623-633, doi:10.1002/hyp.7850.
75. Levia, D.F.; Germer, S. A review of stemflow generation dynamics and stemflow-environment interactions in forests and shrublands. *Reviews of Geophysics* **2015**, *53*, 673-714, doi:10.1002/2015RG000479.

76. Terra, Marcela d.Castro Nunes S.; de Mello, C.R.; de Mello, J.M.; de Oliveira, Vinícius A.; Nunes, Matheus H.; Silva, Vinícius O.; Rodrigues, André F.; Alves, Geovane J. Stemflow in a neotropical forest remnant: vegetative determinants, spatial distribution and correlation with soil moisture. *Trees* **2018**, *32*, 1-13, doi:10.1007/s00468-017-1634-3.
77. Sillett, S.C.; Van Pelt, R.; Kramer, R.D.; Carroll, A.L.; Koch, G.W. Biomass and growth potential of *Eucalyptus regnans* up to 100 m tall. *Forest Ecology and Management* **2015**, *348*, 78-91, doi:10.1016/j.foreco.2015.03.046.
78. Maiden, J.H. *Forestry Handbook. Part II. Some of the Principal Commercial Trees of New South Wales*; Forest Department of New South Wales: Sydney, 1917; p. 224.
79. McIntosh, D. National Register of Big Trees. Australia's Champion Trees. **2018**.
80. Ashton, D.H. The root and shoot development of *Eucalyptus regnans* F. Muell. *Australian Journal of Botany* **1975**, *23*, 867-887, doi:10.1071/BT9750867.
81. Herrmann, W. Vulnerability of Tasmanian giant trees. *Australian Forestry* **2006**, *69*, 285–298, doi:10.1080/00049158.2006.10676249.
82. Williams, J.L.; Lindenmayer, D.; Mifsud, B. The largest trees in Australia. *Austral Ecology* **2023**, *48*, 653-671, doi:10.1111/aec.13292.
83. Galbraith, A.V. *Mountain Ash (Eucalyptus regnans F. van Mueller): a General Treatise on its Silviculture, Management, and Utilization*; Government Printer: Melbourne, 1937; p. 51.
84. Hickey, J.E.; Kostoglou, P.; Sargison, G.J. Tasmania's tallest trees. *Tasforests* **2000**, *12*, 105-122.
85. Ferguson, K.V.M. Some Statistics of Timber Yields from Virgin Stands of White Mountain Ash. *Australian Forestry* **1948**, *12*, 13-15, doi:10.1080/00049158.1948.10675264.
86. Law, G. *Distant affinities. The challenges facing temperate forests in World Heritage Areas (USA, Slovakia and Japan). Report to the Winston Churchill Memorial Trust of Australia, December 2011*; 2010; p. 67.
87. Kostoglou, P. *A survey of ultra tall eucalypts in southern Tasmania. A report to Forestry Tasmania*; Forestry Tasmania: Hobart, Australia, 2000; p. 22.
88. Dean, C.; Roxburgh, S.H. Improving visualisation of mature, high-carbon-sequestering forests. *Forest Biometry, Modelling and Information Sciences* **2006**, *1*, 48-69.
89. Julin, K.R.; Farr, W.A. *Stem Fluting of Western Hemlock in Southeast Alaska*; U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: Portland, Oregon, 1989; p. 8.
90. Dryden, J. Palamon and Arcite. In *Fables Ancient and Modern*; 1700.
91. Rodgers, J. *The English Woodland*; Batsford: London, 1941; p. 132.
92. Dean, C.; Kirkpatrick, J.B.; Doyle, R.B.; Osborn, J.; Fitzgerald, N.B.; Roxburgh, S.H. The overlooked soil carbon under large, old trees. *Geoderma* **2020**, *376*, 1-20, doi:10.1016/j.geoderma.2020.114541.
93. Green, R.N.; Trowbridge, R.L.; Klinka, K. Towards a Taxonomic Classification of Humus Forms. *Forest Science, Monograph* **1993**, *29*, 1-49.
94. Stone, M.G. Forest-type mapping by photointerpretation: A multi-purpose base for Tasmania's forest management. *Tasforests* **1998**, *10*, 15-32.
95. Hickey, J.; Davis, S.; Wardman, R.; Harris, J. How much rainforest is in Tasmania? A better answer to a difficult question. *Tasforests* **1993**, *5*, 13-24.
96. ANM. The forests of the Concession. *Newsprint Log, Australian Newsprint Mills Pty. Ltd.* **1947**, *2*, 21-25.
97. Irby, I.G. *Report of the Forestry Department for the Year Ended 30th June, 1922*; Forestry Department: Hobart, Tasmania, 1922; Volume 27.
98. Quarmby, D. Old forests and Tasmania's early national parks movement. In Proceedings of the Australia's Ever-changing Forests V. Proceedings of the Fifth National Conference on Australian Forest History, Hobart, Tasmania, 2002; pp. 197-208.
99. The Mercury. Newsprint Industry. Area of Timber Supply for Papr Making. *The Mercury* 1937, p. 9.
100. The Mercury. Giant trees will build new industry. Newsprint from Tasmanian forests. *The Mercury* Friday, 23 September 1938 1938, p. 8.
101. ANM. The forests of the concession. *Newsprint Log (Australian Newsprint Mills Pty. Ltd., Tasmania)* **1959**, *15*, 21-25.
102. Dean, C.; Horn, E. A hard, high-carbon, lignomor with conchoidal fracture: cunnite, from mature myrtle beech (*Nothofaus cunninghamii* (Hook. f.) Oerst.). *Geoderma* **2019**, *In Press*, doi:10.1016/j.geoderma.2018.07.006.
103. Noordeloos, M.; Gates, G.M. Preliminary studies in the genus Entoloma in Tasmania – II. *Cryptogamie, Mycologie* **2009**, *30*, 107-140.
104. Mesibov, R. The millipede genus Tasmaniosoma Verhoeff, 1936 (Diplopoda, Polydesmida, Dalodesmidae) from Tasmania, Australia, with descriptions of 18 new species. *ZooKeys* **2010**, *41*, 31–80, doi:10.3897/zookeys.41.420.
105. Eberhard, S.; Giachino, P.M. Tasmanian Trechinae and Psydrinae (Coleoptera, Carabidae): a taxonomic and biogeographic synthesis, with description of new species and evaluation of the impact of Quaternary

climate changes on evolution of the subterranean fauna. *Subterranean Biology* **2011**, *9*, 1-72, doi:10.3897/subtbiol.9.2516.

106. Crane, W.J.B. Natural Environmental Aspects of Forest Management. In *Economic and Environmental Aspects of the Export Hardwood Woodchip Industry*, Cromer, D.A.N., Lamb, V.J.E.I.D., McArthur, A.G., Welsey, D., Girdlestone, J.N., Eds.; Australian Government: Canberra, Australia, 1975; Volume 2, pp. 43-76.
107. Bauhus, J. Rooting Patterns of Old-Growth Forests: is Aboveground Structural and Functional Diversity Mirrored Belowground? In *Old-Growth Forests. Function, Fate and Value*, Wirth, C., Gleixner, G., Heimann, M., Eds.; Ecological Studies; Springer-Verlag: Berlin Heidelberg, 2009; pp. 211-229.
108. Calver, M.; Wardell-Johnson, G.W. Sustained Unsustainability? An evaluation of evidence for a history of overcutting in the Jarrah forests of Western Australia and its consequences for fauna conservation. In *Conservation of Australia's Forest Fauna*, 2nd ed.; Lunney, D., Ed.; Royal Zoological Society of NSW: Sydney, Australia, 2004; pp. 94-114.
109. Lindenmayer, D.B.; Cunningham, R.B.; Tanton, M.T.; Nix, H.A.; Smith, A.P. The conservation of arboreal marsupials in the montane ash forests of the central highlands of victoria, south-east Australia: III. The habitat requirements of leadbeater's possum *Gymnobelideus leadbeateri* and models of the diversity and abundance of arboreal marsupials. *Biological Conservation* **1991**, *56*, 295-315.
110. Galbraith, A.V. The disastrous forest fires of January 1939, in Victoria. *Empire Forestry Journal* **1939**, *18*, 10-18.
111. Law, B.S. The ecology of bats in south-east Australian forests and potential impacts of forestry practices: a review. *Pacific Conservation Biology* **1996**, *2*, 363-374.
112. Newell, G.R. Responses of Lumholtz's tree-kangaroo (*Dendrolagus lumholtzi*) to loss of habitat within a tropical rainforest fragment. *Biological Conservation* **1999**, *91*, 181-189.
113. Kirkpatrick, J.B. Big history of land clearance and deforestation. *Journal of Big History* **2024**, *7*, 1-18, doi:10.22339/jbh.v7i3.7301.
114. Kirkpatrick, J.B. The growthist/greenie ritual conflict and the nature of the argument— the example of the woodchip export EIS. In Proceedings of the Institute of Australian Geographers' Conference, Perth, Australia, 10-18 May 1986, 1987; pp. 312-318.
115. Dargavel, J. Views and perspectives: Why does Australia have 'forest wars'? *International Review of Environmental History* **2018**, *4*, 33-51, doi:10.22459/IREH.04.01.2018.04.
116. Florence, R.G. Resource use and biological and environmental planning for eucalypt forest management. *Australian Forestry* **1972**, *36*, 147-153.
117. Fedotova, A. Forestry experimental stations: Russian proposals of the 1870s. *Centaurus* **2014**, *56*, 254-274, doi:10.1111/1600-0498.12070.
118. Dargavel, J. Integration within the model: A new era for Australian forest planning. *Impact Assesment* **1990**, *8*, 101-118, doi:10.1080/07349165.1990.9726031.
119. Cremer, K.W. Eucalypts in rain forest. *Australian Forestry* **1960**, *24*, 120-126.
120. Cremer, K.W. Studies on problems of regeneration of Eucalyptus regnans in Tasmania. University of Tasmania, 1966.
121. Frankcombe, D.W. The regeneration burn. *Appita* **1966**, *19*, 127-132.
122. Paul, E.A.; Robertson, G.P. Ecology and the agricultural sciences: a false dichotomy. *Ecology* **1989**, *70*, 1594-1597, doi:10.2307/1938091.
123. Puettmann, K.J.; Wilson, S.M.; Baker, S.C.; Donoso, P.J.; Drössler, L.; Amente, G.; Harvey, B.D.; Knoke, T.; Lu, Y.; Nocentini, S.; et al. Silvicultural alternatives to conventional even-aged forest management - what limits global adoption? *Forest Ecosystems* **2015**, *2*, 1-16, doi:10.1186/s40663-015-0031-x.
124. Keith, H.; Barrett, D.; Keenan, R. *Review of allometric relationships for estimating woody biomass for New South Wales, the Australian Capital Territory, Victoria, Tasmania and South Australia*; Australian Greenhouse Office: 2000; p. 119.
125. Polinko, A.; Coupland, K. Paradigm shifts in forestry and forest research: a bibliometric analysis. *Canadian Journal of Forest Research* **2020**, *51*, 127-135, doi:10.1139/cjfr-2020-0311.
126. Kirkpatrick, J.B.; Brown, M.J. Planning for Species Conservation. In *Nature Conservation: Cost Effective Biological Surveys and Data Analysis*, Margules, C.R., Austin, M.P., Eds.; CSIRO: Canberra, Australia, 1991; pp. 83-89.
127. Dumbrell, A.J.; Clark, E.J.; Frost, G.A.; Randell, T.E.; Pitchford, J.W.; Hill, J.K. Changes in species diversity following habitat disturbance are dependent on spatial scale: theoretical and empirical evidence. *Journal of Applied Ecology* **2008**, *45*, 1531-1539.
128. Elton, C.S. The structure of invertebrate populations inside neotropical rain forest. *Journal of Animal Ecology* **1973**, *42*, 55-104, doi:10.2307/3406.
129. Specht, R.L.; Specht, A. Species richness of sclerophyll (heathy) plant communities in australia the influence of overstorey cover. *Australian Journal of Botany* **1989**, *37*, 337-350, doi:10.1071/BT9890337.
130. Mittelbach, G.G. A matter of time for tropical diversity. *Nature* **2017**, *550*, 51-52, doi:10.1038/nature24142.

131. Hu, Y.; Kelly, L.T.; Gillespie, G.R.; Jessop, T.S. Lizard responses to forest fire and timber harvesting: Complementary insights from species and community approaches. *Forest Ecology and Management* **2016**, *379*, 206-215, doi:10.1016/j.foreco.2016.07.040.
132. Perry, D.R.; Williams, J. The tropical rain forest canopy: A method providing total access. *Biotropica* **1981**, *13*, 283-285, doi:10.2307/2387806.
133. Basset, Y.; Novotny, V.; Miller, S.E.; Pyle, R. Quantifying biodiversity: Experience with parataxonomists and digital photography in Papua New Guinea and Guyana. *BioScience* **2000**, *50*, 899-908.
134. Carr, A.; Zeale, M.R.K.; Weatherall, A.; Froidevaux, J.S.P.; Jones, G. Ground-based and LiDAR-derived measurements reveal scale-dependent selection of roost characteristics by the rare tree-dwelling bat *Barbastella barbastellus*. *Forest Ecology and Management* **2018**, *417*, 237-246, doi:10.1016/j.foreco.2018.02.041.
135. Lindenmayer, D.B. *The Forest Wars*; Allen & Unwin: Crows Nest, New South Wales, Australia, 2024; p. 267.
136. Macher, T.-H.; Schütz, R.; Hörren, T.; Beermann, A.J.; Leese, F. It's raining species: Rainwash eDNA metabarcoding as a minimally invasive method to assess tree canopy invertebrate diversity. *Environmental DNA* **2023**, *5*, 3-11, doi:10.1002/edn3.372.
137. Blakely, T.J.; Harding, J.S.; Didham, R.K. Distinctive aquatic assemblages in water-filled tree holes: a novel component of freshwater biodiversity in New Zealand temperate rainforests. *Insect Conservation and Diversity* **2012**, *5*, 202-212, doi:10.1111/j.1752-4598.2011.00155.x.
138. Bar-Ness, Y.D.; Kirkpatrick, J.B.; McQuillan, P.B. Age and distance effects on the canopy arthropod composition of old-growth and 100-year-old Eucalyptus obliqua trees. *Forest Ecology and Management* **2006**, *226*, 290-298, doi:10.1016/j.foreco.2006.01.041.
139. Ishii, H.T.; Tanabe, S.-i.; Hiura, T. Exploring the relationships among canopy structure, stand productivity, and biodiversity of temperate forest ecosystems. *Forest Science* **2004**, *50*, 342-355, doi:10.1093/forestscience/50.3.342.
140. Dalagnol, R.; Wagner, F.H.; Emilio, T.; Streher, A.S.; Galvão, L.S.; Ometto, J.P.H.B.; Aragão, L.E.O.C. Canopy palm cover across the Brazilian Amazon forests mapped with airborne LiDAR data and deep learning. *Remote Sensing in Ecology and Conservation* **2022**, *8*, 601-614, doi:10.1002/rse2.264.
141. Gibbons, P.; Lindenmayer, D.B.; Barry, S.C.; Tanton, M.T. Hollow selection by vertebrate fauna in forests of southeastern Australia and implications for forest management. *Biological Conservation* **2002**, *103*, 1–12, doi:10.1016/S0006-3207(01)00109-4.
142. Pedley, S.M.; Oxbrough, A.; Martin, R.D.; Irwin, S.; Kelly, T.C.; O'Hallora, J. Can ground-based assessments of forest biodiversity reflect the biological condition of canopy assemblages? *Forest Ecology and Management* **2016**, *359*, 190-198, doi:10.1016/j.foreco.2015.10.006.
143. Straw, N.A.; Williams, D.T.; Fielding, N.J.; Jukes, M.; Connolly, T.; Forster, J. The influence of forest management on the abundance and diversity of hoverflies in commercial plantations of Sitka spruce: the importance of sampling in the canopy. *Forest Ecology and Management* **2017**, *406*, 95-111, doi:10.1016/j.foreco.2017.06.010.
144. Whitworth, A.; Braunholtz, L.D.; Huarcaya, R.P.; MacLeod, R.; Beirne, C. Out on a limb: arboreal camera traps as an emerging methodology for inventorying elusive rainforest mammals. *Tropical Conservation Science* **2016**, *9*, 675-698.
145. Whitworth, A.; Beirne, C.; Huarcaya, R.P.; Whittaker, L.; Rojas, S.J.S.; Tobler, M.W.; MacLeod, R. Human disturbance impacts on rainforest mammals are most notable in the canopy, especially for larger‐bodied species. *Biodiveristy Research* **2019**, *25*, 1166‐1178, doi:10.1111/ddi.12930.
146. Dumbrell, A.J.; Hill, J.K. Impacts of selective logging on canopy and ground assemblages of tropical forest butterflies: Implications for sampling. *Biological Conservation* **2005**, *125*, 123-131, doi:10.1016/j.biocon.2005.02.016.
147. Malcolm, J.R. Comparative abundances of neotropical small mammals by trap height. *Journal of Mammalogy* **1991**, *72*, 188-192, doi:10.2307/1381995.
148. Corcoran, E.; Denman, S.; Hamilton, G. Evaluating new technology for biodiversity monitoring: Are drone surveys biased? *Ecology and Evolution* **2021**, *11*, 6649-6656, doi:10.1002/ece3.7518.
149. Delabre, I.; Boyd, E.; Brockhaus, M.; Carton, W.; Krause, T.; Newell, P.; Wong, G.Y.; Zelli, F. Unearthing the myths of global sustainable forest governance. *Global Sustainability* **2020**, *3*, 1-10, doi:10.1017/sus.2020.11.
150. Hughes, J.D. How the ancients viewed deforestation. *Journal of Field Archeology* **1983**, *10*, 435-441.
151. Kile, G.A.; Nambiar, E.K.S.; Brown, A.G. The rise and fall of research and development for the forest industry in Australia. *Australian Forestry* **2014**, *77*, 142-152, doi:10.1080/00049158.2014.990867.
152. Bernays, E.L. The theroy and practice of public relations: a résumé. In *The Engineering of Consent*, 1st ed.; Bernays, E.L., Ed.; University of Oklahoma Press: Norman, Oklahoma, 1955; pp. 3-4.
153. Krien, A. *Into the Woods. The Battle for Tasmania's Forests*; Black inc.: Melbourne, 2012; p. 316.
154. Kirkpatrick, J.B. The political ecology of biogeography. *Journal of Biogeography* **2000**, *27*, 45-48, doi:10.1046/j.1365-2699.2000.00372.x.
155. Clarkson, R.B. *Tumult in the Mountains: Lumbering in West Virginia 1770–1920.*; McClain Printing Company: Parson, West Virginia, 1964.

156. Ajani, J. *The Forest Wars*; Melbourne University Press: Melbourne, 2007; p. 362.
157. Medimorec, S.; Pennycook, G. The language of denial: text analysis reveals differences in language use between climate change proponents and skeptics. *Climatic Change* **2015**, 1-9, doi:10.1007/s10584-015-1475-2.
158. Hanson, D.; White, R. Rhetorics of environmental routinisation in one Australian company's annual reports. *Australian Journal of Social Issues* **2003**, *38*, 477-493.
159. McCright, A.M.; Dunlap, R.E. Anti-reflexivity: The American Conservative Movement's Success in Undermining Climate Science and Policy. *Theory, Culture & Society* **2010**, *27*, 100-133, doi:10.1177/0263276409356001.
160. Sneddon, J.N.; Soutar, G.N.; Lee, J.A. Exploring wool apparel consumers' ethical concerns and preferences. *Journal of Fashion Marketing and Management* **2014**, *18*, 169-186, doi:10.1108/JFMM-03-2013-0039.
161. Bayldon, W.; Narishetty, S.; Rose, G.D.; Rothwell, J.; Mills, P.C. Effects of eight vehicles on transdermal lidocaine penetration in sheep skin *in vitro*. *Journal of Veterinary Pharmacology and Therapeutics* **2013**, *37*, 151-160, doi:10.1111/jvp.12073.
162. Vidot, A.; Conifer, D. Investigation launched into claims Australian cattle were slaughtered with sledgehammer in Vietnam. **2015**.
163. Winter, C.; Poole, L. Animals Australia allege camel cruelty while live export company loads ship in Adelaide. **2014**.
164. Fuller, K.; Razaghi, T. *Scholarship to honour the life of Glen Turner*; Australian Broadcasting Corporation: 13-February-2015 2015.
165. Woodburn, J. *Moree shooting: Farmer continued firing as environmental officer tried to flee, court told*; Australian Broadcasting Corporation: 06-August-2014 2014.
166. Bradshaw, C.J.A.; Ehrlich, P.R. *Killing the Koala and Poisoning the Prairie. Australia, America, and the Environment*; The University of Chicago Press: Chicago, 2015; p. 235.
167. INTERPOL. *Global Forestry Enforcement. Strengthening Law Enforcement Cooperation Against Forestry Crime*; Lyon, France, 2019; p. 11.
168. INTERPOL. *Forestry crime: targeting the most lucrative of environmental crimes*; Lyon, France, 14-December-2020 2020.
169. Halaj, D.; Báliková, K.; Brodrechtová, Y. The perception of an image of the state forest enterprise by general public in chosen region of the Slovak Republic. *Journal of Forest Science* **2022**, *68*, 26-34.
170. Poynter, M. Timber shortage decades in the making, but being worsened by 'save-the-forests' political ideology. Available online: https://www.onlineopinion.com.au/view.asp?article=21548 (accessed on
171. Ajani, J. *Is using native forests for energy really carbon-neutral?*; The Conversation, 22-December-2011 2011.
172. Bennett, L.T.; Adams, M.A. Ecological effects of harvesting in Victoria's native forests: quantification of research outputs. *Australian Forestry* **2004**, *67*, 212-221, doi:10.1080/00049158.2004.10674937.
173. Czech, B.; Krausman, P.R.; Borkhataria, R. Social construction, political power, and the allocation of benefits to endangered species. *Conservation Biology* **1998**, *12*, 1103–1112.
174. Heath, R.L. Corporate Issues Management: Theoretical Underpinnings and Research Foundations. In *Public Relations Research Annual*, Grunig, J.E., Grunig, L.A., Eds.; Lawrence Erllbaum Associates, Inc.: Hillsdale, New Jersey, 1990; Volume 2, pp. 29-65.
175. Jaworski, B.; Kohli, A.K.; Sahay, A. Market-driven versus driving markets *Journal of the Academy of Marketing Science* **2000**, *28*, 45-54.
176. Hansen, E.; Juslin, H. Marketing of forest products in a changing world. *New Zealand Journal of Forestry Science* **2005**, *35*, 190-204.
177. Saltelli, A. Why science's crisis should not become a political battling ground. *Futures* **2018**, *104*, 85-90, doi:10.1016/j.futures.2018.07.006.
178. Li, N.; Puumalainen, K.; Toppinen, A. Managerial perceptions of corporate social and financial performance in the global forest industry. *The International Forestry Review* **2014**, *16*, 319-338.
179. Latapí Agudelo, M.A.; Jóhannsdóttir, L.; Davídsdóttir, B. A literature review of the history and evolution of corporate social responsibility. *International Journal of Corporate Social Responsibility* **2019**, *4*, 1-23, doi:10.1186/s40991-018-0039-y.
180. Fernow, B.E. *What is Forestry*; Government Printing Office: Washington, D.C., 1891; p. 52.
181. Knuchel, H. *Planning and Control in the Managed Forest*; Oliver & Boyd: Edinburgh, 1953.
182. World Bank Institute. Public Policy for Corporate Social Responsibility. e-conference, July 7-25, 2003.
183. Blowfield, M. Corporate Social Responsibility: reinventing the meaning of development? *International Affairs* **2005**, *81*, 515-524.
184. Swinburn, B.A.; Kraak, V.I.; Allender, S.; Atkins, V.J.; Baker, P.I.; Bogard, J.R.; Brinsden, H.; Alejandro Calvillo; Schutter, O.D.; Devarajan, R.; et al. The global syndemic of obesity, undernutrition, and climate change: The Lancet Commission report. *The Lancet* **2019**, *393*, 791-846, doi:10.1016/S0140-6736(18)32822-8.
185. Wilson, R. Marketing of forestry and its products: a NSW perspective. *Australian Forestry* **2006**, *69*, 161-166.
186. COAG. *National Climate Change and Commercial Forestry Action Plan 2009–2012, Council of Australian Governments, Canberra, Australia.*; Canberra, Australia., 2009; p. 24.

187. The CIE. *Forest and Wood Products Australia Generic Marketing Promotions, Evaluation. Discussion Paper*; The Centre for International Economics: Canberra and Sydney, 2015; p. 69.
188. Michaels, D. *Doubt is Their Product. How Industry's Assault on Science Threatens Your Health*; Oxford University Press: New York, 2008; p. 371.
189. Hobson, K.; Niemeyer, S. "What sceptics believe": The effects of information and deliberation on climate change scepticism. *Public Understanding of Science* **2013**, *22*, 396-412, doi:10.1177/0963662511430459.
190. McDonald, R.I.; b, H.Y.C.; Newell, B.R. Personal experience and the 'psychological distance' of climate change: An integrative review. *Journal of Environmental Psychology* **2015**, *44*, 109-118, doi:10.1016/j.jenvp.2015.10.003

191. Pant, V.K.; Singh, R. A review of social forestry practices in ancient India (neolithic time to 6th Century CE). *Utkal Historical Research Journal* **2021**, *34*, 57-66.
192. Nipunage, D.S.; Kulkarni, D.K. Deo-rahati: an ancient concept of biodiversity conservation. *Asian Agri-History* **2010**, *14*, 185-196.
193. Lentz, D.L.; Hockaday, B. Tikal timbers and temples: ancient Maya agroforestry and the end of time. *Journal of Archaeological Science* **2009**, *36*, 1342–1353, doi:10.1016/j.jas.2009.01.020.
194. Evans, J. The History of Tree Planting and Planted Forests. In *Planted Forests. Uses, Impacts & Sustainability*, Evans, J., Ed.; Food and Agriculture Oranisation of the United Nations: Wallingford, U.K., 2009; p. 213.
195. Hughes, J.D. Ancient deforestation revisited. *Journal of the History of Biology* **2011**, *44*, 43-57, doi:10.1007/s10739-010-9247-3.
196. Brown, J.E. *A Practical Treatise on Tree Culture in South Australia*; Spiller, South Australian Government: Adelaide, South Australia, 1881; p. 116.
197. Yang, J.; Liu, L.; Roberts, H.; Liu, Z.; Song, L.; Zhang, P. Holocene flood records and human impacts implied from the pollen evidence in the Daming area, North China Plain. *Frontiers in Earth Science* **2024**, *12*, 1-12, doi:10.3389/feart.2024.1349195.
198. Short, B. Conservation, class and custom: lifespace and conflict in a nineteenth-century forest environment. *Rural History* **1990**, *10*, 127-154.
199. Pádua, J.A. 'Annihilating Natural Productions': nature's economy, colonial crisis and the origins of Brazilian political environmentalism (1786-1810). *Environment and History* **2000**, *6*, 255-287.
200. Van Name, W.G. Anti-conservation propaganda. *Science* **1925**, *61*, 415-416.
201. Moroni, M.T.; Musk, R.; Wardlaw, T.J. Forest succession where trees become smaller and wood carbon stocks reduce. *Forest Ecology and Management* **2017**, *393*, 74-80, doi:10.1016/j.foreco.2017.02.051.
202. Baxter, T.I. Whither forests of the Tasmanian Wilderness World Heritage Area? *Australian Environment Review* **2014**, *29*, 112-117.
203. IFA IFG. *Submission to the Victorian Regional Forest Agreements Major Event Review Following the 2019-20 Bushfires*; The Institute of Foresters of Australia and Australian Forest Growers: Melbourne, Australia, 2021; p. 20.
204. Australian Forest Products Association. Forest industries applaud Coalition election commitments to drive innovation and modern manufacturing timber jobs. News, Media Release 14-April-2022. Available online: https://ausfpa.com.au/forest-industries-applaud-coalition-election-commitments-to-drive-innovation-and-modern-manufacturing-timber-jobs/ (accessed on 17-November-2022).
205. Abbott, D. A new shared vision for Australia's forests. Media Release. 17 March 2021. Available online: https://www.forestry.org.au/media-release-a-new-shared-vision-for-australias-forests/ (accessed on 13-May-2023).
206. Jackson, W.; Freeman, M.; Freeman, B.; Parry-Husbands, H. Reshaping forest management in Australia to provide nature-based solutions to global challenges. *Australian Forestry* **2021**, *84*, 50-58, doi:10.1080/00049158.2021.1894383.
207. Norton, T.W. Conserving biological diversity in Australia's temperate eucalypt forests. *Forest Ecology and Management* **1996**, *85*, 21-33, doi:10.1016/S0378-1127(96)03747-4.
208. Kiernan, K. Conservation, timter and perceived values at Mt Field, Tasmania. In Proceedings of the Australia's Ever-changing Forests V. Proceedings of the Fifth National Conference on Australian Forest History, Hobart, Tasmania, 2002; pp. 209-227.
209. Kirkpatrick, J.B. Conservation Worrier. DE Press Inc.: Hobart, Tasmania, 2017; pp. 133-142.
210. Hitchcock, P. Independent Verification Group Forest Conservation Report 5A Verification of the Heritage Value of ENGO- Proposed Reserves. **2012**, 308.
211. Knoepp, J.D.; Adams, M.B.; Harrison, R.; West, L.; Laseter, S.H.; Markewitz, D.; Richter, D.D.; Jr., M.A.C. History of forest soils knowledge and research. In *Global Change and Forest Soils. Cultivating Stewardship of a Finite Natural Resource*, Busse, M., Giardina, C.P., Morris, D.M., Page-Dumroese, D.S., Eds.; Developments in Soil Science; Elsevier: 2019; Volume 36, pp. 43-55.
212. Grand, S.; Lavkulich, L.M. Effects of Forest Harvest on Soil Carbon and Related Variables in Canadian Spodosols. *Soil Science Society of America Journal* **2012**, *76*, 1816, doi:10.2136/sssaj2012.0103.

213. Holub, S.M.; Hatten, J.A. Soil carbon storage in Douglas-fir forests of western Oregon and washington before and after modern timber harvesting practices. *Soil Science Society of America Journal* **2019**, *83*, S175-S186, doi:10.2136/sssaj2018.09.0354.
214. Dietzen, C.A.; Marques, E.R.G.; James, J.N.; Bernardi, R.H.A.; Holub, S.M.; Harrison, R.B. Response of deep soil carbon pools to forest management in a highly productive Andisol. *Soil Science Society of America Journal* **2017**, *81*, 970–978, doi:10.2136/sssaj2016.09.0305.
215. Andrieux, B.; Beguin, J.; Bergeron, Y.; Grondin, P.; Paré, D. Drivers of postfire soil organic carbon accumulation in the boreal forest. *Global Change Biology* **2018**, *2018*, 1-20, doi:10.1111/gcb.14365.
216. Li, J.; Pei, J.; Liu, J.; Wu, J.; Li, B.; Fang, C.; Nie, M. Spatiotemporal variability of fire effects on soil carbon and nitrogen: A global meta-analysis. *Global Change Biology* **2021**, *27*, 4196–4206, doi:10.1111/gcb.15742.
217. Gao, F.; Cui, X.; Sang, Y.; Song, J. Changes in soil organic carbon and total nitrogen as affected by primary forest conversion. *Forest Ecology and Management* **2020**, *463*, 1-12, doi:10.1016/j.foreco.2020.118013.
218. James, J.; Page-Dumroese, D.; Busse, M.; Palik, B.; Zhang, J.; Eaton, B.; Slesak, R.; Tirocke, J.; Kwon, H. Effects of forest harvesting and biomass removal on soil carbon and nitrogen: Two complementary meta-analyses. *Forest Ecology and Management* **2021**, *485*, 1-19, doi:10.1016/j.foreco.2021.118935.
219. Hibbard, K.A.; Schimel, D.S.; Archer, S.; Ojima, D.S.; Parton, W. Grassland to woodland transitions: integrating changes in landscape structure and biogeochemistry. *Ecological Applications* **2003**, *13*, 911-926.
220. Jandl, R.; Lindner, M.; Vesterdal, L.; Bauwens, B.; Baritz, R.; Hagedorn, F.; Johnson, D.W.; Minkkinen, K.; Byrne, K.A. How strongly can forest management influence soil carbon sequestration? *Geoderma* **2007**, *137*, 253-268, doi:10.1016/j.geoderma.2006.09.003.
221. Laub, M.; Demyan, M.S.; Nkwain, Y.F.; Blagodatsky, S.; Kätterer, T.; Piepho, H.-P.; Cadisch, G. DRIFTS band areas as measured pool size proxy to reduce parameter uncertainty in soil organic matter models. *Biogeosciences* **2020**, *17*, 1393-1413, doi:10.5194/bg-17-1393-2020.
222. Chen, J.; Cui, H.-Y.; Jia, B.; Gang, S.; Li, Y.; Li, F.-C.; Mou, X.M.; Li, X.G. Soil sampling depth matters in assessing the impact of shrubification on soil organic carbon storage in grazed alpine meadows. *Geoderma* **2022**, *426*, doi:10.1016/j.geoderma.2022.116119.
223. Balesdent, J.; Basile-Doelsch, I.; Chadoeuf, J.; Cornu, S.; Derrien, D.; Fekiacova, Z.; Hatté, C. Atmosphere–soil carbon transfer as a function of soil depth. *Nature* **2018**, *559*, 559-602, doi:10.1038/s41586-018-0328-3.
224. Callesen, I.; Harrison, R.; Stupak, I.; Hatten, J.; Raulund-Rasmussen, K.; Boyle, J.; Clarke, N.; Zabowski, D. Carbon storage and nutrient mobilization from soil minerals by deep roots and rhizospheres. *Forest Ecology and Management* **2016**, *359*, 322-331, doi:10.1016/j.foreco.2015.08.019.
225. Krueger, I.; Schulz, C.; Borken, W. Stocks and dynamics of soil organic carbon and coarse woody debris in three managed and unmanaged temperate forests. *European Journal of Forest Research* **2017**, *136*, 123-137, doi:10.1007/s10342-016-1013-4.
226. Fontaine, S.; Barot, S.; Barré, P.; Bdioui, N.; Mary, B.; Rumpel, C. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature* **2007**, *450*, 277-280, doi:10.1038/nature06275.
227. Wäldchen, J.; Schulze, E.-D.; Schöning, I.; Schrumpf, M.; Sierra, C. The influence of changes in forest management over the past 200years on present soil organic carbon stocks. *Forest Ecology and Management* **2013**, *289*, 243-254, doi:10.1016/j.foreco.2012.10.014.
228. Lucas-Borja, M.E.; R. Ortega; I. Miralles; P. A. Plaza-Álvarez; J. González-Romero; E. Peña-Molina; Moya, D.; D. A. Zema; J. W. Wagenbrenner; J. de las Heras. Effects of wildfire and logging on soil functionality in the short-term in *Pinus halepensis* M. forests. *European Journal of Forest Research* **2020**, *139*, 935-945, doi:10.1007/s10342-020-01296-2.
229. Parton, W.J.; Stewart, J.W.B.; Cole, C.V. Dynamics of C, N, P and S in grassland soils: a model. *Biogeochemistry* **1988**, *5*, 109-131.
230. Dean, C.; Roxburgh, S.H.; Harper, R.J.; Eldridge, D.J.; Watson, I.W.; Wardell-Johnson, G.W. Accounting for space and time in soil carbon dynamics in timbered rangelands. *Ecological Engineering* **2012**, *38*, 51-64, doi:10.1016/j.ecoleng.2011.10.008.
231. Olson, J.S. Forest Studies. In *Health Physics Division Annual Progress Report for Period Ending July 31, 1959*, 15th ed.; ORNL, Ed.ORNL, Ed.; Health and Safety TID-4500; U. S. Atomic Energy Commission: Washington D.C. , 1959; Volume ORNL-2806, pp. 37-54.
232. Peltoniemi, M.; Thürig, E.; Ogle, S.; Palosuo, T.; Schrumpf, M.; Wutzler, T.; Butterbach-Bahl, K.; Chertov, O.; Komarov, A.; Mikhailov, A.; et al. Models in country scale carbon accounting of forest soils. *Silva Fennica* **2007**, *41*, 575-602, doi:10.14214/sf.290.
233. Hudiburg, T.W.; Higuera, P.E.; Hicke, J.A. Fire-regime variability impacts forest carbon dynamics for centuries to millennia. *Biogeosciences* **2017**, *14*, 3873-3882, doi:10.5194/bg-14-3873-2017.
234. Xia; Luo; Wang; Weng; Hararuk. A semi-analytical solution to accelerate spin-up of a coupled carbon and nitrogen land model to steady state. *Geoscientific Model Development* **2012**, *5*, 1259-1271, doi:10.5194/gmd-5-1259-2012.

235. Dean, C.; Roxburgh, S.H.; Mackey, B.G. Forecasting landscape-level carbon sequestration using gridded, spatially adjusted tree growth. *Forest Ecology and Management* **2004**, *194*, 109-129, doi:10.1016/j.foreco.2004.02.013.
236. Solomon, S.; Plattner, G.-K.; Knutti, R.; Friedlingstein, P. Irreversible climate change due to carbon dioxide emissions. *Proceedings of the National Academy of Sciences* **2019**, *106*, 1704-1709, doi:10.1073/pnas.0812721106.
237. Frölicher, T.L.; Joos, F. Reversible and irreversible impacts of greenhouse gas emissions in multi-century projections with the NCAR global coupled carbon cycle-climate model. *Climate Dynamics* **2010**, *35*, 1439-1459, doi:10.1007/s00382-009-0727-0.
238. Augusto, L.; Boča, A. Tree functional traits, forest biomass, and tree species diversity interact with site properties to drive forest soil carbon. *Nature Communications* **2022**, *13*, 1-12, doi:10.1038/s41467-022-28748-0.
239. Dean, C.; Kirkpatrick, J.B.; Friedland, A.J. Conventional intensive logging promotes loss of organic carbon from the mineral soil. *Global Change Biology* **2017**, *23*, 1-11, doi:10.1111/gcb.13387.
240. Kershaw, J.A., Jr.; Oliver, C.D.; Hinckley, T.M. Effect of harvest of old growth Douglas-fir stands and subsequent management on carbon dioxide levels in the atmosphere. *Journal of Sustainable Forestry* **1993**, *1*, 61-77.
241. Johnson, K.; Scatena, F.N.; Pan, Y. Short- and long-term responses of total soil organic carbon to harvesting in a northern hardwood forest. *Forest Ecology and Management* **2010**, *259*, 1262-1267.
242. Liski, J.; Ilvesniemi, H.; Mäkelä, A.; Starr, M. Model analysis of the effects of soil age, fires and harvesting on the carbon storage of boreal forest soils. *European Journal of Soil Science* **1998**, *49*, 407-416, doi:10.1046/j.1365-2389.1998.4930407.x.
243. Jandl, N.; Jandl, R.; Schindlbacher, A. Future management options for cembran pine forests close to the alpine timberline. *Annals of Forest Science* **2018**, *75*, 1-12, doi:10.1007/s13595-018-0760-4.
244. Leuschner, C.; Feldmann, E.; Pichler, V.; Glatthorn, J.; Hertel, D. Forest management impact on soil organic carbon: A paired-plot study in primeval and managed European beech forests. *Forest Ecology and Management* **2022**, *512*, 1-9, doi:10.1016/j.foreco.2022.120163.
245. Mayer, M.; Prescott, C.E.; Abaker, W.E.A.; Augusto, L.; Cécillon, L.; Ferreira, G.W.D.; James, J.; Jandl, R.; Katzensteiner, K.; Laclau, J.-P.; et al. Tamm Review: Influence of forest management activities on soil organic carbon stocks: A knowledge synthesis. *Forest Ecology and Management* **2020**, *466*, 1-25, doi:10.1016/j.foreco.2020.118127.
246. Peckham, S.D.; Gower, S.T. Simulated long-term effects of harvest and biomass residue removal on soil carbon and nitrogen content and productivity for two Upper Great Lakes forest ecosystems. *Global Change Biology Bioenergy* **2011**, *3*, 135–147, doi:10.1111/j.1757-1707.2010.01067.x.
247. Wang, F.; Mladenoff, D.J.; Forrester, J.A.; Blanco, J.A.; Scheller, R.M.; Peckham, S.D.; Keough, C.; Lucash, M.S.; Gower, S.T. Multimodel simulations of forest harvesting effects on long-term productivity and CN cycling in aspen forests. *Ecological Applications* **2014**, *24*, 1374–1389.
248. Seely, B.; Welham, C.; Kimmins, H. Carbon sequestration in a boreal forest ecosystem: results from the ecosystem simulation model, FORECAST. *Forest Ecology and Management* **2002**, *169*, 123–135.
249. Cardoso, P.; Borges, P.A.V.; Carvalho, J.C.; Rigal, F.; Gabriel, R.; Cascalho, J.; Correia, L. Automated discovery of relationships, models and principles in ecology. *bioRxiv* **2015**, *8*, doi:10.1101/027839.
250. Schmidt, M.; Lipson, H. Distilling free-form natural laws from experimental data. *Science* **2009**, *324*, 81-85, doi:10.1126/science.1165893.
251. Poeplau, C.; Don, A.; Vesterdal, L.; Leifeld, J.; Wesemael, B.V.; Schumacher, J.; Gensior, A. Temporal dynamics of soil organic carbon after land-use change in the temperate zone – carbon response functions as a model approach. *Global Change Biology* **2011**, *17*, 2415-2427, doi:10.1111/j.1365-2486.2011.02408.x.
252. Beheshti, A.; Raiesi, F.; Golchin, A. Soil properties, C fractions and their dynamics in land use conversion from native forests to croplands in northern Iran. *Agriculture, Ecosystems and Environment* **2012**, *148*, 121–133, doi:10.1016/j.agee.2011.12.001.
253. Huon, S.; Rouw, A.d.; Bonté, P.; Robain, H.; Valentin, C.; Lefèvre, I.; Girardin, C.; Troquer, Y.L.; Podwojewski, P.; Sengtaheuanghoung, O. Long-term soil carbon loss and accumulation in a catchment following the conversion of forest to arable land in northern Laos. *Agriculture, Ecosystems and Environment* **2013**, *169*, 43-57, doi:10.1016/j.agee.2013.02.007.
254. Wei, X.; Shao, M.; Gale, W.J.; Zhang, X.; Li, L. Dynamics of aggregate-associated organic carbon following conversion of forest to cropland. *Soil Biology & Biochemistry* **2013**, *57*, 876-883, doi:10.1016/j.soilbio.2012.10.020.
255. Ferré, C.; Comolli, R.; Leip, A.; Seufert, G. Forest conversion to poplar plantation in a Lombardy floodplain (Italy): effects on soil organic carbon stock. *Biogeosciences Discussions* **2014**, *11*, 9601-9627, doi:10.5194/bgd-11-9601-2014.
256. Vario, C.L.; Neurath, R.A.; Friedland, A.J. Response of Mineral Soil Carbon to Clear-Cutting in a Northern Hardwood Forest. *Soil Science Society of America Journal* **2014**, *78*, 309, doi:10.2136/sssaj2013.06.0226.

257. Lei, L.; Xiao, W.; Zeng, L.; Frey, B.; Huang, Z.; Zhu, J.; Cheng, R.; Li, M.-H. Effects of thinning intensity and understory removal on soil microbial community in *Pinus massoniana* plantations of subtropical China. *Applied Soil Ecology* **2021**, *167*, 1-10, doi:j.apsoil.2021.104055.
258. Polglase, P.; Adams, M.; Attiwill, P.M. *Measurement and Modelling of Carbon Storage in a Chronosequence of Mountain Ash Forests: Implications for Regional and Global Carbon Budgets*; Report commissioned by the State Electricity Commission, Victoria: Melbourne, Australia, 1994; p. 61.
259. He, J.; Dai, Q.; Xu, F.; Peng, X.; Yan, Y. Variability in carbon stocks across a chronosequence of masson pine plantations and the trade-off between plant and soil systems. *Forests* **2021**, *12*, 1-15, doi:10.3390/f12101342.
260. Law, B.E.; Sun, O.J.; Campbell, J.; S. Van Tuyl; Thornton, P.E. Changes in carbon storage and fluxes in a chronosequence of ponderosa pine. *Global Change Biology* **2003**, *9*, 510-524, doi:10.1046/j.1365-2486.2003.00624.x.
261. Diochon, A.; Kellman, L.; Beltrami, H. Looking deeper: An investigation of soil carbon losses following harvesting from a managed northeastern red spruce (*Picea rubens* Sarg.) forest chronosequence. *Forest Ecology and Management* **2009**, *257*, 413-420, doi:10.1016/j.foreco.2008.09.015.
262. Martin, J.L.; Gower, S.T.; Plaut, J.; Holmes, B. Carbon pools in a boreal mixedwood logging chronosequence. *Global Change Biology* **2005**, *11*, 1883-1894, doi:10.1111/j.1365-2486.2005.01019.x.
263. Squires, A. Parks and Woodland in Medieval Leicestershire 1086-1530. In *Leicestershire Landscapes*, Bowman, P., Liddle, P., Eds.; Leicestershire Museums Archaeological Fieldwork Group Monographs; No. 1; Leicestershire Museums Archaeological Fieldwork Group: University of Leicester, England, 2004; pp. 141-153.
264. Mendel, L.C.; Kirkpatrick, J.B. Historical progress of biodiversity conservation in the protected-area system of Tasmania, Australia. *Conservation Biology* **2002**, *16*, 1520–1529, doi:10.1046/j.1523-1739.2002.01155.x.
265. Scott, J.M.; Davis, F.W.; Mcghie, R.G.; Wright, R.G.; Groves, C.; Estes, J. Nature reserves: Do they capture the full range of America's biological diversity? *Ecological Issues in Conservation* **2001**, *11*, 999–1007, doi:10.1890/1051-0761(2001)011[0999:NRDTCT]2.0.CO;2.
266. Kayanja, F.I.B.; Byarugaba, D. Disappearing forests of Uganda: The way forward. *Current Science* **2001**, *81*, 936-947.
267. Christophel, D.; Spengler, S.; Schmidt, B.; Ewald, J.; Prietzel, J. Customary selective harvesting has considerably decreased organic carbon and nitrogen stocks in forest soils of the Bavarian Limestone Alps. *Forest Ecology and Management* **2013**, *305*, 167-176, doi:10.1016/j.foreco.2013.05.054.
268. Gifford, R.M.; Roderick, M.L. Soil carbon stocks and bulk density: spatial or cumulative mass coordinates as a basis of expression? *Global Change Biology* **2003**, *9*, 1507–1514, doi:10.1046/j.1529-8817.2003.00677.x.
269. Harmon, M.E.; Ferrel, W.K.; Franklin, J.F. Effects on carbon storage of conversion of old-growth forests to young forests. *Science* **1990**, *247*, 699-702.
270. Sathre, R.; O'Connor, J. Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science and Policy* **2010**, *13*, 104-114, doi:10.1016/j.envsci.2009.12.005.
271. Lindenmayer, D.B.; Laurance, W.F. A history of hubris – Cautionary lessons in ecologically sustainable forest management. *Biological Conservation* **2012**, *151*, 11-16.
272. Diamond, J.M. Biogeographic kinetics: estimation of relaxation times for avifaunas of southwest Pacific Islands. *PNAS* **1972**, *69*, 3199-3203.
273. Kuussaari, M.; Bommarco, R.; Heikkinen, R.K.; Helm, A.; Krauss, J.; Lindborg, R.; Öckinger, E.; Pärtel, M.; Pino, J.; Rodà, F.; et al. Extinction debt: a challenge for biodiversity conservation. *Trends in Ecology ad Evolution* **2009**, *24*, 564-571, doi:10.1016/j.tree.2009.04.011.
274. MPIGA. *Australia's State of the Forests Report. Five-yearly report 2008. Montreal Process Implementation Group for Australia and National Forest Inventory Steering Committee.*; Bureau of Rural Sciences: Canberra, 2008; p. 250.
275. Thompson, D.; Connell, M. *Issues in Assessing the Sustainability of Private Native Forestry*; RIRDC Publication No. 09/031; Canberra, 2009; p. 40.
276. Downham, R.; Gavran, M. *Australian plantation statistics 2019 update*; Australian Bureau of Agricultural and Resource Economics and Sciences: Canberra, 2019; p. 15.
277. Brander, M.; Ascui, F.; Scott, V.; Tett, S. Carbon accounting for negative emissions technologies. *Climate Policy* **2021**, *21*, 699-717, doi:10.1080/14693062.2021.1878009.
278. Laurijssen, J.; Marsidi, M.; Westenbroek, A.; Worrell, E.; Faaij, A. Paper and biomass for energy? The impact of paper recycling on energy and CO2 emissions. *Resources, Conservation & Recycling* **2010**, *54*, 1208-1218, doi:10.1016/j.resconrec.2010.03.016.
279. Stutz, K.P.; Lang, F. Forest ecosystems create pedogenic patchworks through woody debris, trees, and disturbance. *Geoderma* **2023**, *429*, 1-12, doi:10.1016/j.geoderma.2022.116246.
280. IPCC. *2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4 Agriculture, Forestry and Other Land Use*; National Greenhouse Gas Inventories Programme, IGES: Japan, 2006.
281. FAO. *Measuring and modelling soil carbon stocks and stock changes in livestock production systems: Guidelines for assessment. Version 1.*; Food and Agriculture Organisation of the Uniten Nations: Rome, Italy, 2019; p. 145.

282. Sochacki, S.J.; Ritson, P.; Brand, B.; Harper, R.J.; Dell, B. Accuracy of tree root biomass sampling methodologies for carbon mitigation projects. *Ecological Engineering* **2017**, *98*, 264-274, doi:10.1016/j.ecoleng.2016.11.004.
283. Salisbury, E.J. Stratification and hydrogen-ion concentration of the soil in relation to leaching and plant succession with special reference to woodlands. *Journal of Ecology* **1922**, *9*, 220-240.
284. Marsden, C.; Nouvellon, Y.; Laclau, J.-P.; Corbeels, M.; McMurtrie, R.E.; Stape, J.L.; Epron, D.; Maire, G.l. Modifying the G'DAY process-based model to simulate the spatial variability of Eucalyptus plantation growth on deep tropical soils. *Forest Ecology and Management* **2013**, *301*, 112-128, doi:10.1016/j.foreco.2012.10.039.
285. Rui, G.; Yike, W.; Ning, A.; Changhai, L.; Qiaoyu, Z.; Fangfang, Q. Characteristics and inflluencing factors of deep soil orgnaic carbon in characteristic economic forests in Northern Shaanxi. *Journal of Forest and Environment* **2021**, *41*, 464-470, doi:10.13324/j.cnki.jfcf.2021.05.003.
286. Wade, A.M.; Richter, D.D.; Medjibe, V.P.; Bacon, A.R.; Paul R. Heine; White, L.J.T.; Poulsen, J.R. Estimates and determinants of stocks of deep soil carbon in Gabon, Central Africa. *Geoderma* **2019**, *341*, 236-248, doi:doi.org/10.1016/j.geoderma.2019.01.004.
287. Jia, X.; Wu, H.; Shao, M.a.; Huang, L.; Wei, X.; Wang, Y.; Zhu, Y. Re-evaluation of organic carbon pool from land surface down to bedrock on China's Loess Plateau. *Agriculture, Ecosystems and Environment* **2020**, *293*, doi:10.1016/j.agee.2020.106842.
288. Germon, A.; Laclau, J.-P.; Robin, A.; Jourdan, C. Tamm Review: Deep fine roots in forest ecosystems: Why dig deeper? *Forest Ecology and Management* **2020**, *466*, 1-16, doi:10.1016/j.foreco.2020.118135.
289. Tune, A.K.; Druhan, J.L.; Wang, J.; Bennett, P.C.; Rempe, D.M. Carbon dioxide production in bedrock beneath soils substantially contributes to forest carbon cycling. *Journal of Geophysical Research: Biogeosciences* **2020**, *I25*, 1-13, doi:10.1029/2020JG005795.
290. Gross, C.D.; Harrison, R.B. The case for digging deeper: Soil organic carbon storage, dynamics, and controls in our changing world. *Soil Systems* **2019**, *3*, 1-24, doi:10.3390/soilsystems3020028.
291. Nepstad, D.C.; Carvalho, C.R.d.; Davidson, E.A.; Jlpp, P.H.; Lefebvre, P.A.; Negrelros, G.H.; Sllva, E.D.d.; Stone, T.A.; Trumbore, S.E.; Vieira, S. The role of deep roots in the hydrological and carbon cycles of Amazonian forests and pastures. *Nature* **1994**, *372*, 666-669, doi:10.1038/372666a0.
292. Smith, P.; Soussana, J.F.; Angers, D.; Schipper, L.; Chenu, C.; Rasse, D.P.; Batjes, N.H.; Egmond, F.v.; McNeill, S.; Kuhnert, M.; et al. How to measure, report and verify soil carbon change to realize the potential of soil carbon sequestration for atmospheric greenhouse gas removal. *Global Change Biology* **2019**, *26*, 219-241, doi:10.1111/gcb.14815.
293. Strey, S.; Boy, J.; Strey, R.; Welpelo, A.; Schönenberg, R.; Schumann, C.; Guggenberger, G. Digging deeper: The value of deep soil carbon for potential REDD+ projects in tropical forest communities in Amazonia. *Erdkunde* **2017**, *71*, 231-239.
294. Archer, N.A.L.; Quinton, J.N.; Hess, T.M. Below-ground relationships of soil texture, roots and hydraulic conductivity in two-phase mosaic vegetation in South-east Spain. *Journal of Arid Environments* **2002**, *52*, 535–553, doi:10.1006/jare.2002.1011.
295. Jackson, R.B.; Banner, J.L.; Jobbágy, E.G.; Pockman, W.T.; Wall, D.H. Ecosystem carbon loss with woody plant invasion of grasslands. *Nature* **2002**, *418*, 623-626.
296. Shaler, N.S. *The Origin and Nature of Soils*; U.S. Geological Survey: Washington, D.C., 1892; Volume 4, part I.
297. Frazier, C.S.; Graham, R.C. Pedogenic transformation of fractured granitic bedrock, southern California. *Soil Science Society America Journal* **2000**, *64*, 2057–2069.
298. Nie, Y.; Chen, H.; Ding, Y.; Yang, J.; Wang, K. Comparison of rooting strategies to explore rock fractures for shallow soil-adapted tree species with contrasting aboveground growth rates: a greenhouse microcosm experiment. *Frontiers in Plant Science* **2017**, *8*, 1-11, doi:10.3389/fpls.2017.01651.
299. Rodríguez-Robles, U.; Arredondo, T.; Huber-Sannwald, E.; Ramos-Leal, J.A.; Yépez, E.A. Technical note: Application of geophysical tools for tree root studies in forest ecosystems in complex soils. *Biogeosciences* **2017**, *14*, 5343–5357, doi:10.5194/bg-14-5343-2017.
300. Sehhati, M.T.; Sepehr, A.; Ekhtesasi, M.R.; Goudie, A. The eco-geomorphological roles of rocky deep crevices for water supply on arid zone mountain slopes (case study: Mehriz–Yazd, Iran). *Environmental Earth Sciences* **2015**, *74*, 493–504, doi:10.1007/s12665-015-4056-9.
301. Dean, C.; Wardell-Johnson, G. Old-growth forests, carbon and climate change: functions and management for tall open-forests in two hotspots of temperate Australia. *Plant Biosystems* **2010**, *144*, 180-193, doi:10.1080/11263500903560751.
302. McIntosh, P.D.; Hardcastle, J.L.; Klöffel, T.; Moroni, M.; Santini, T.C. Can carbon sequestration in tasmanian "wet" eucalypt forests be used to mitigate climate change? Forest succession, the buffering effects of soils, and landscape processes must be taken into account. *International Journal of Forestry Research* **2020**, *2020*, 1-16, doi:10.1155/2020/6509659.

303. Grant, J.C.; Laffan, M.D.; Hill, R.C. *Soils of Tasmanian State Forests 2. Forester Sheet. Soils Bulletin 2*; Forestry Tasmania: Hobart, Australia, 1995; p. 247.
304. Hill, R.B.; Laffan, M.D.; Grant, J.C. *Soils of Tasmanian State Forests 3. Forth Sheet. Soils Bulletin 3*; Forestry Tasmania: Hobart, Australia, 1995; p. 317.
305. Laffan, M.D.; Grant, J.C.; Hill, R.C. *Soils of Tasmanian State Forests 1. Pipers Sheet. Soils Bulletin 1*; Forestry Tasmania: Hobart, Australia, 1995; p. 272.
306. Herbert, A.; Hill, M.D.L.R.C.; Grant, J.C. *Laboratory procedures for soil analysis and preparation of plant materials, Soils Technical Report No. 2*, Second ed.; Forestry Tasmania: Hobart, Australia, 1995; p. 22.
307. Dean, C.; Roxburgh, S.H.; Mackey, B.G. Growth Modelling of *Eucalyptus regnans* for Carbon Accounting at the Landscape Scale. In *Modelling Forest Systems*, Amaro, A., Reed, D., Soares, P., Eds.; CABI Publishing: Wallingford, Oxford, U.K., 2003; pp. 27-39 + plates.
308. Paul, K.I.; Roxburgh, S.H. *FullCAM: building capability via data informed parameters. Accurately tracking forest carbon from biomass to soil. Report for Department of the Environment and Energy.*; CSIRO Agriculture: Canberra, Australia, 2017; p. 65.
309. Keith, H.; Mackey, B.G.; Lindenmayer, D.B. Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proceedings of the National Academy of Sciences of the United States of America* **2009**, *106*, 11635-11640, doi:10.1073/pnas.0901970106.
310. Dietrich, P. Carbon stocks in coarse woody debris and soil in the mixed forests and the rainforests in southern Tasmania. MSc, Technische Universität, Dresden, Germany, 2012.
311. Batjes, N.H. Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science* **1996**, *47*, 151-163.
312. G. Corti; Ugolini, F.C.; Agnelli, A.; Certini, G.; Cuniglio, R.; Berna, F.; Sanjurjo, M.J.F. The soil skeleton, a forgotten pool of carbon and nitrogen in soil. *European Journal of Soil Science* **2002**, *53*, 283-298, doi:10.1046/j.1365-2389.2002.00442.x.
313. Rivard, R.; Kimpe, C.R.D. Propriétés de quelques sols riches en graviers dans la région de Québec. *Canadian Journal of Soil Science* **1980**, *60*, 263-273, doi:10.4141/cjss80-029.
314. Harrison, R.B.; Adams, A.B.; Licata, C.; Flaming, B.; Wagoner, G.L.; Carpenter, P.; Vance, E.D. Quantifying Deep-Soil and Coarse-Soil Fractions: Avoiding Sampling Bias. *Soil Science Society America Journal* **2003**, *67*, 1602–1606, doi:10.2136/sssaj2003.1602.
315. Zabowski, D.; Whitney, N.; Gurung, J.; Hatten, J. Total soil carbon in the coarse fraction and at depth. *Forest Science* **2011**, *57*, 11-18.
316. Basile-Doelsch, I.; Balesdent, J.; Rose, J. Are interactions between organic compounds and nanoscale weathering minerals the key drivers of carbon storage in soils? *Environmental Science and Technology* **2015**, *49*, 3997-3998, doi:10.1021/acs.est.5b00650.
317. Hopmans, P.; Bauhus, J.; Khanna, P.; Weston, C. Carbon and nitrogen in forest soils: Potential indicators for sustainable management of eucalypt forests in south-eastern Australia. *Forest Ecology and Management* **2005**, *220*, 75-87, doi:10.1016/j.foreco.2005.08.006.
318. Buma, B.; Poore, R.E.; Wessman, C.A. Disturbances, their interactions, and cumulative effects on carbon and charcoal stocks in a forested ecosystem. *Ecosystems* **2014**, *17*, 947–959, doi:10.1007/s10021-014-9770-8.
319. Harrington, A.; Holub, S.M.; Bauer, C.; Ashley, E.; Steel. The distribution of tree roots in Douglas-fir forests in the Pacific Northwest in relation to depth, space, coarse organic matter and mineral fragments. *Northwest Science* **2017**, *91*, 326-343, doi:10.3955/046.091.0403.
320. Keller, C.K.; Bacon, D.H. Soil respiration and georespiration distinguished by transport analyses of vadose CO2, 13CO2, and 14CO2. *Global Biogeochemical Cycles* **1998**, *12*, 361-372, doi:10.1029/98GB00742.
321. Blair, N.E.; Leithold, E.L.; Ford, S.T.; Peeler, K.A.; Holmes, J.C.; Perkey, D.W. The persistence of memory: The fate of ancient sedimentary organic carbon in a modern sedimentary system. *Geochimica et Cosmochimica Acta* **2003**, *67*, 63-73, doi:10.1016/S0016-7037(02)01043-8.
322. Ishikawa, N.F.; Tayasu, I.; Yamane, M.; Yokoyama, Y.; Sakai, S.; Ohkouchi, N. Sources of dissolved inorganic carbon in two small streams with different bedrock geology: Insights from carbon isotopes. *Radiocarbon* **2015**, *57*, 439–448, doi:10.2458/azu_rc.57.18348.
323. Qi, M.; Chen, S.; Wei, Y.; Zhou, H.; Zhang, S.; Wang, M.; Zheng, J.; Rossel, R.A.V.; Chang, J.; Shi, Z.; et al. Using visible-near infrared spectroscopy to estimate whole-profile soil organic carbon and its fractions. *Soil & Environmental Health* **2024**, *2*, 1-9, doi:10.1016/j.seh.2024.100100.
324. Shamrikova, E.V.; Vanchikova, E.V.; Kyzyurova, E.V.; Zhangurova, E.V. Methods for measuring organic carbon content in carbonate-containing soils: A review. *Eurasian Soil Science, 2024, Vol. 57, No. 3, pp. 380–394* **2024**, *57*, 380–394, doi:10.1134/S1064229323603104.
325. Grant, J.C.; Laffan, M.D.; Hill, R.B.; Neilsen, W.A. *Forest Soils of Tasmania. A Handbook for Identification and Management*; Forestry Tasmania: Hobart, Australia, 1995; p. 189.
326. De Vos, B.; Lettens, S.; Muys, B.; Deckers, J.A. Walkley–Black analysis of forest soil organic carbon: recovery, limitations and uncertainty. *Soil Use and Management* **2007**, *23*, 221–229, doi:10.1111/j.1475-2743.2007.00084.x.

327. Meersmans, J.; Van Wesemael, B.; Van Molle, M. Determining soil organic carbon for agricultural soils: a comparison between the Walkley & Black and the dry combustion methods (north Belgium). *Soil Use and Management* **2009**, *25*, 346–353, doi:10.1111/j.1475-2743.2009.00242.x.
328. Fernandes, R.B.A.; Afonso de Carvalho Junior, I.; Silva Ribeiro Junior, E.; Mendonça, E.d.S. Comparison of different methods for the determination of total organic carbon and humic substances in Brazilian soils. *Revista Ceres* **2015**, *62*, 496-501, doi:10.1590/0034-737X201562050011.
329. Pearson, T.R.H.; Brown, S.L.; Birdsey, R.A. *Measurement Guidelines for the Sequestration of Forest Carbon. General Technical Report NRS-18*; Forest Service, United States Department of Agriculture: Pennsylvania, USA, 2007; p. 44.
330. Nelson, D.W.; Sommers, L.E. Total Carbon, Organic Carbon and Organic Matter. In *Methods of Soil Analysis. Part 3. Chemical Methods-SSSA Book Series no. 5*, Sparks, D.L., Page, A.L., Helmke, P.A., Loeppert, R.H., Soltanpour, P.N., Tabatabai, M.A., Johnston, C.T., M.E., S., Eds.; Soil Sceince Society of Amerrica and American Society of Agronomy: Madison, Wisconsin, 1996; pp. 961-1010.
331. Wang, X.J.; Smethurst, P.J.; Herbert, A.M. Relationships between three measures of organic matter or carbon in soils of eucalypt plantations in Tasmania. *Australian Journal of Soil Research* **1996**, *34*, 545-553.
332. Olayinka, A.; Adebayo, A.; Amusan, A. Evaluation of organic carbon oxidation efficiencies of a modified wet combustion and Walkley ‐ Black procedures in Nigerian soils. *Communications in Soil Science and Plant Analysis* **1998**, *29*, 2749-2756, doi:10.1080/00103629809370149.
333. McKenzie, N.; Ryan, P.; Fogarty, P.; Wood, J. *Sampling, measurement and analytical protocols for carbon estimation in soil, litter and course woody debris. Technical Report No. 14, National Carbon Accounting System*; Commonwealth of Australia: Canberra, 2000; p. 52.
334. Poeplau, C.; Vos, C.; Don, A. Soil organic carbon stocks are systematically overestimated by misuse of the parameters bulk density and rock fragment content. *Soil* **2017**, *3*, 61-66, doi:10.5194/soil-3-61-2017.
335. Mehler, K.; Schöning, I.; Berli, M. The importance of rock fragment density for the calculation of soil bulk density and soil organic carbon stocks. *Soil Science Society of America Journal* **2014**, *78*, 1186-1191, doi:10.2136/sssaj2013.11.0480.
336. Conen, F.; Zerva, A.; Arrouays, D.; Jolivet, C.; Jarvis, P.; Grace, J.; Mencuccini, M. The carbon balance of forest soils: detectability of changes in soil carbon stocks in temperate and boreal forests. In *The Carbon Balance of Forest Biomes*, Griffiths, H., Jarvis, P., Eds.; Garland Science/BIOS Scientific Publishers: Southampton, U.K., 2004; Volume 9, pp. 233–247.
337. Seigel, H.O. *A Guide to High Precision Land Gravimetric Surveys*; Scintrex Limited: Concord, Ontario, Canada, 1995; p. 132.
338. Bennett, L.T.; Hinko-Najera, N.; Aponte, C.; Nitschke, C.R.; Fairman, T.A.; Fedrigo, M.; Kasel, S. Refining benchmarks for soil organic carbon in Australia's temperate forests. *Geoderma* **2020**, *368*, doi:10.1016/j.geoderma.2020.114246.
339. Post, W.M.; Emanuel, W.R.; Zinket, P.J.; Stangenberger, A.G. Soil carbon pools and world life zones. *Nature* **1982**, *298*, 156-159, doi:10.1038/298156a0.
340. Jobággy, G.J.; Jackson, R.B. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* **2002**, *10*, 423-436, doi:10.2307/2641104.
341. Australian Forestry Council. FORWOOD: report of the Forestry and Wood-based Industries Development Conference 1974. In Proceedings of the Forestry and Wood-based Industries Development Conference, Canberra, Australia, 1974.
342. di Folco, M.-B.; Kirkpatrick, J.B. Organic soils provide evidence of spatial variation in human-induced vegetation change following European occupation of Tasmania. *Journal of Biogeography* **2013**, *40*, 197–205, doi:10.1111/j.1365-2699.2012.02779.x.
343. Harmon, M.E. Carbon sequestration in forests. Addressing the scale question. *Journal of Forestry* **2001**, *99*, 24-29.
344. Harmon, M.E. Woody Detritus Mass and its Contribution to Carbon Dynamics of Old-Growth Forests: the Temporal Context. In *Old-growth Forests: Function, Fate and Value. Ecological studies*, C., W., G., G., M., H., Eds.; Springer-Verlag: New York, 2009; pp. 159-190.
345. Mesibov, R. *Review of the Deep Red Myrtle Resource in Tasmania*; Ferguson, I.S., Ed.; Forestry Tasmania: Hobart, Tasmania, Australia, 2002; p. 56.
346. Elliott, H.J.; Hickey, J.E.; Jennings, S.M. Effects of selective logging and regeneration treatments on mortality of retained trees in Tasmanian cool temperate rainforest. *Australian Forestry* **2005**, *68*, 274-280, doi:10.1080/00049158.2005.10674977.
347. Walker, B.B.; Candy, S. Investigating methods of assessing rainforest in Tasmania. In Proceedings of the Tasmanian Rainforests. Recent Research Results, Hobart, Tasmania, 1982; pp. 7-34.
348. Duncan, F. *Tasmania's Vegetation and its Response to Forest Operations. Working Paper 6. Background information additional to the Environmental Impact Statement on Tasmanian Woodchip Exports Beyond 1988. EIS Study Group.*; Tasmanian Government Printer: Hobart, Tasmania, 1985; p. 127.

349. Wood, S.W.; Hua, Q.; Allen, K.J.; Bowman, D.J.M.S. Age and growth of a fire prone Tasmanian temperate old-growth forest stand dominated by *Eucalyptus regnans*, the world's tallest angiosperm. *Forest Ecology and Management* **2010**, *260*, 438-447, doi:10.1016/j.foreco.2010.04.037.
350. Hickey, J.E.; Su, W.; Rowe, P.; Brown, M.J.; Edwards, L. Fire history of the tall wet eucalypt forests of the Warra ecological research site, Tasmania. *Australian Forestry* **1999**, *62*, 66-71, doi:10.1080/00049158.1999.10674765.
351. Bowman, D.M.J.S.; Kolden, C.A.; Abatzoglou, J.T.; Johnston, F.H.; Werf, G.R.v.d.; Flannigan, M. Vegetation fires in the Anthropocene. *Nature Reviews Earth & Environment* **2020**, *1*, 500-515, doi:10.1038/s43017-020-0085-3.
352. Fox-Hughes, P.; Harris, R.; Lee, G.; Grose, M.; Bindoff, N. Future fire danger climatology for Tasmania, Australia, using a dynamically downscaled regional climate model. *International Journal of Wildland Fire* **2014**, *23*, 309–321, doi:10.1071/WF13126.
353. Riddington, M. Victoria’s rainforests and the potential impacts of a changing climate. *The Victorian Naturalist* **2014**, *131*, 209-218.
354. Lindenmayer, D.B.; Hobbs, R.J.; Likens, G.E.; Krebs, C.J.; Banks, S.C. Newly discovered landscape traps produce regime shifts in wet forests. *PNAS* **2011**, *108*, 15887-15891, doi:10.1073/pnas.1110245108.
355. Taplin, R.E.; Tighe, P.J.; Hill, A.H.; Hoystead, P.A.; McCuaig, A. Temperate rainforest species and forest regeneration in the Hastings forest region of the southern forests, Tasmania. In *The Rainforest Legacy. Australian National Rainforest Study. Special Australian Heritage Publication*, Werren, G., Kershaw, P., Eds.; Australian Government Publishing Service: Canberra, 1991; Volume 7(3), pp. 151-167, 301.
356. Jordan, G.J.; Patmore, C.; Duncan, F.; Luttrell, S.D. The effects of fire intensity on the regeneration of mixed forest tree species in the Clear Hill/ Mount Wedge area. *Tasforests* **1992**, *4*, 25-38.
357. Tabor, J.; McElhinny, C.; Hickey, J.; Wood, J. Colonisation of clearfelled coupes by rainforest tree species from mature mixed forest edges, Tasmania, Australia *Forest Ecology and Management* **2007**, *240*, 13-23.
358. Mackey, B.; DellaSala, D.A.; Kormos, C.; Lindenmayer, D.; Kumpel, N.; Zimmerman, B.; Hugh, S.; Young, V.; Foley, S.; Arsenis, K.; et al. Policy options for the world’s primary forests in multilateral environmental agreements. *Conservation Letters* **2015**, *8*, 139–147, doi:10.1111/conl.12120.
359. Forestry Tasmania. *2000 Annual Report. Forestry Tasmania*; Forestry Tasmania: Hobart, Tasmania, Australia, 2001; p. 62.
360. Bryant, C. *Understanding bushfire trends in deliberately lit vegetation fires in Australia*; Government, A., Ed.; Australian Institute of Criminology, Australian Government: Canberra, 2008; Volume 27, p. 618.
361. Enright, N.J.; Fontaine, J.B. Climate change and the management of fire-prone vegetation in southwest and southeast Australia. *Geographical Research* **2013**, *52*, 34-44, doi:10.1111/1745-5871.12026.
362. Hall, G.M.J.; Hollinger, D.Y. Simulating New Zealand forest dynamics with a generalized temperate forest gap model. *Ecological Applications* **2000**, *10*, 115-130.
363. Hiltner, U.; Huth, A.; Fischer, R. Importance of succession in estimating biomass loss: Combining remote sensing and individual-based forest models. *Biogeosciences. Discussions* **2020**, *preprint*, doi:10.5194/bg-2020-264.
364. Green, G. *Esperance 74D (EP074D) Logging Coupe Inventory*; Timber Workers for Forests: Hobart, Tasmania., 2002.
365. FORTECH. *Study No. 2 Rainforest in the Northwest of Tasmania*; Forestry Technical Services Pty Ltd: Canberra, ACT, 1982; p. 153.
366. Rab, M.A. Soil physical and hydrological properties following logging and slash burning in the *Eucalyptus regnans* forest in southeastern Australia. *Forest Ecology and Management* **1996**, *84*, 159-176.
367. Pennington, P.I.; Laffan, M.; Lewis, R.; Otahal, P. Assessing the long-term impacts of forest harvesting and high intensity broadcast burning on soil properties at the Warra LTER Site. *Tasforests* **2001**, *13*, 291-298.
368. Ellis, R.C.; Graley, A.M. Gains and losses in soil nutrients associated with harvesting and burning eucalypt rainforest. *Plant and Soil* **1983**, *74*, 437-450.
369. Tomkins, I.B.; Kellas, J.D.; Tolhurst, K.C.; Oswin, D.A. Effects of fire intensity on soil chemistry in a eucalypt forest. *Australian Journal of Soil Research* **1991**, *29*, 25-47.
370. Antos, J.A.; Halpern, C.B.; Miller, R.E.; Cromack, K., Jr.; Halaj, M.G. *Temporal and Spatial Changes in Soil Carbon and Nitrogen After Clearcutting and Burning of an Old-Growth Douglas-Fir Forest. Research Paper PNW-RP-552*; Forest Service, United States Department of Agriculture: Portland, Oregon, USA, 2003; p. 28.
371. Jiménez Esquilín, A.E.J.; Stromberger, M.E.; Massman, W.J.; Frank, J.M.; Sheppard, W.D. Microbial community structure and activity in a Colorado Rocky Mountain forest soil scarred by slash pile burning. *Soil Biology and Biochemistry* **2007**, *39*, 1111-1120.
372. Bormann, B.T.; Homann, P.S.; Darbyshire, R.L.; Morrissette, B.A. Intense forest wildfire sharply reduces mineral soil C and N: the first direct evidence. *Canadian Journal of Forest Research* **2008**, *38*, 2771-2783, doi:10.1139/X08-136.
373. Rab, M.A. Changes in physical properties of a soil associated with logging of *Eucalyptus regnans* forest in southeastern Australia. *Forest Ecology and Management* **1994**, *70*, 215.

374. Camilleri, Mark A. *Corporate Sustainability, Social Responsibility and Environmental Management. An Introduction to Theory and Practice with Case Studies*; Springer Nature: Cham, Switzerland, 2017; p. 195.
375. Putz, F.E.; Romero, C.; Sist, P.; Schwartz, G.; Thompson, I.; Roopsind, A.; Ruslandi, R.; Medjibe, V.; Ellis, P. Sustained timber yield claims, considerations, and tradeoffs for selectively logged forests. *PNAS Nexus* **2022**, *1*, 1-7, doi:10.1093/pnasnexus/pgac102.
376. Crane, A.H. Assessment of A.N.M. Concession-Upper Styx Area, Assessment of A.N.M. mature forest 1956/57. Report from Chief Commissioner, Forestry Commission Tasmania. **1957**, 8.
377. Bansal, P.; Song, H.-C. Similar but not the same: Differentiating corporate sustainability from corporate responsibility. *Academy of Management Annals* **2017**, *11*, 105-149, doi:10.5465/annals.2015.0095.
378. Han, X.; Hansen, E. From a production orientation to a stakeholder orientation: the evolution of marketing sophistication in private, multi-site U.S. sawmills. In Proceedings of the Proceedings of the 55th International Convention of Society of Wood Science and Technology, Beijin, China, 27-31-August-2012, 2012; pp. 1-10.
379. Chuter, R. *Feasibility of burning debris from wet eucalypt forests harvested to an aggregated retention prescription*; Forestry Tasmania: Hobart, Tasmania, Australia, 2007; p. 31.
380. Neyland, M.; Hickey, J.; Beadle, C.; Bauhus, J.; Davidson, N.; Edwards, L. An examination of stocking and early growth in the Warra silvicultural systems trial confirms the importance of a burnt seedbed for vigorous regeneration in Eucalyptus obliqua forest. *Forest Ecology and Management* **2009**, *258*, 481-494.
381. Shen, C.; Ma, L.; Hu, J.; Huang, L.; Chen, Y.; Guan, D. Soil carbon storage and its determinants in forest fragments of differentiated patch size. *Forests* **2019**, *10*, 1-13, doi:10.3390/f10111044.
382. Bae, J.; Ryu, Y. The magnitude and causes of edge effects on soil organic carbon stocks within and across urban to rural forest patches. *Landscape and Urban Planning* **2021**, *215*, 1-11, doi:10.1016/j.landurbplan.2021.104223.
383. Zeng, H.; Peltola, H.; Talkkari, A.; Venäläinen, A.; Strandman, H.; Kellomäki, S.; Wang, K. Influence of clear-cutting on the risk of wind damage at forest edges. *Forest Ecology and Management* **2004**, *2003*, 77-88, doi:10.1016/j.foreco.2004.07.057.
384. Dupont, S.; Brunet, Y. Impact of forest edge shape on tree stability: a large-eddy simulation study. *Forestry* **2008**, *81*, 299-315, doi:10.1093/forestry/cpn006.
385. Franklin, J.F.; Forman, R.T.T. Creating landscape patterns by forest cutting: Ecological consequences and principles. *Landscape Ecology* **1987**, *1*, 5-18, doi:10.1007/BF02275261.
386. Narayanaraj, G.; Wimberly, M.C. Influences of forest roads on the spatial patterns of human- and lightning-caused wildfire ignitions. *Applied Geography* **2012**, *32*, 878-888, doi:10.1016/j.apgeog.2011.09.004.
387. Commonwealth of Australia & TAS. *Consolidated Tasmanian Regional Forest Agreement*; Commonwealth of Australia and Tasmanian governments,: 1997; p. 84.
388. Commonwealth of Australia & NSW. *Regional Forest Agreement for Southern New South Wales*; Commonwealth of Australia and the State of New South Wales: 2001; p. 110.
389. Brown, A.J. Collaborative governance versus constitutional politics: decision rules for sustainability from Australia's South East Queensland forest agreement. *Environmental Science & Policy* **2002**, *5*, 19-32, doi:10.1016/S1462-9011(02)00022-9.
390. Costantini, A.; Dunn, G.M.; Grimmett, J.L. Towards sustainable management of forest plantations in south-east Queensland. II: Protecting soil and water values during second rotation Pinus plantation management. *Australian Forestry* **1997**, *60*, 226-232, doi:10.1080/00049158.1997.10676148.
391. Costantini, A.; Grimmett, J.L.; Dunn, G.M. Towards sustainable management of forest plantations in south-east Queensland. I: Logging and understorey residue management between rotations in steep country *Araucaria cunninghamii* plantations. *Australian Forestry* **1997**, *60*, 218-225, doi:10.1080/00049158.1997.10676147.
392. ANM. The properties of Australian Timbers. Mountain Ash. *Newsprint Log, Australian Newsprint Mills Pty. Ltd.* **1947**, *2*, 7-9.
393. Forestry Commission. *Silviculture Manual*; Forestry Commission of Tasmania: Hobart, Tasmania, 1978; p. 124.
394. Spencer, B.; Mazanec, R.; Abadi, A.; Gibberd, M.; Zerihun, A. Flowering phenology in a *Eucalyptus loxophleba* seed orchard, heritability and genetic correlation with biomass production and cineole: breeding strategy implications. *Nature Scientific Reports* **2020**, *10*, 1-16, doi:10.1038/s41598-020-72346-3.
395. Enquist, B.J.; Abraham, A.J.; Harfoot, M.B.J.; Malhi, Y.; Doughty, C.E. The megabiota are disproportionately important for biosphere functioning *Nature Communications* **2020**, *11*, 1-11, doi:10.1038/s41467-020-14369-y.
396. Ali, A.; Wang, L.-Q. Big-sized trees and forest functioning: Current knowledge and future perspectives. *Ecological Indicators* **2021**, *127*, doi:10.1016/j.ecolind.2021.107760.
397. Lindenmayer, D.B.; Laurance, W.F.; Franklin, J.F. Global decline in large old trees. *Science* **2012**, *1305*, doi:10.1126/science.1231070.
398. Lindenmayer, D.B.; Laurance, W.F. The ecology, distribution, conservation and management of large old trees. *Biological Reviews* **2016**, *92*, doi:10.1111/brv.12290.

399. Allendorf, F.W.; Hard, J.J. Human-induced evolution caused by unnatural selection through harvest of wild animals. *PNAS* **2009**, *106*, 9987–9994, doi:10.1073/pnas.0901069106.
400. Coltman, D.W.; O'Donoghue, P.; Jorgenson, J.T.; Hogg, J.T.; Strobeck, C.; Festa-Bianchet, M. Undesirable evolutionary consequences of trophy hunting. *Nature* **2003**, *426*, 655-658, doi:10.1038/nature02177.
401. Strømman, A.H.; Peters, G.P.; Hertwich, E.G. Approaches to correct for double counting in tiered hybrid life cycle inventories. *Journal of Cleaner Production* **2009**, *17*, 35-43, doi:10.1016/j.jclepro.2018.09.231.
402. Ximenes, F.A.; Robinson, M.; Wright, B. *Forests, wood and Australia's carbon balance*; Forest and Wood Products Research and Development Corporation, C.R.C.f.G.A., Ed.; Forest and Wood Products Research and Development Corporation (Australian Government): Melbourne, 2008; p. 20.
403. Pingoud, K.; Perälä, A.-L.; Pussinen, A. Carbon dynamics in wood products. *Mitigation and Adaptation Strategies for Global Change* **2001**, *6*, 91-111.
404. Dean, C. Land Use and Carbon Dynamics in Woody Ecosystems. PhD, Univeristy of Tasmania, Hobart, Tasmania, 2017.
405. Ågren, G.I.; Hyvönen, R. Changes in carbon stores in Swedish forest soils due to increased biomass harvest and increased temperatures analysed with a semi-empirical model. *Forest Ecology and Management* **2003**, *174*, 25-37, doi:10.1016/S0378-1127(02)00025-7.
406. Gustavsson, L.; Nguyen, T.; Sathre, R.; Tettey, U.Y.A. Climate effects of forestry and substitution of concrete buildings and fossil energy. *Renewable and Sustainable Energy Reviews* **2021**, *136*, 1-15, doi:10.1016/j.rser.2020.110435.
407. Schulze, E.D.; Bouriaud, O.; Irslinger, R.; Valentini, R. The role of wood harvest from sustainably managed forests in the carbon cycle. *Annals of Forest Science* **2022**, *79*, 1-13, doi:10.1186/s13595-022-01127-x.
408. Horne, R.; Grant, T.; Verghese, K. *Life Cycle Assessment. Principles, Practice and Prospects*; CSIRO Publishing: Collingwood, Victoria, Australia, 2009; p. 175.
409. Ingerson, A. Carbon storage potential of harvested wood: summary and policy implications. *Mitigation, Adaptation Strategies for Global Change* **2011**, *16*, 307-323.
410. May, B.; England, J.R.; Raison, R.J.; Paul, K.I. Cradle-to-gate inventory of wood production from Australian softwood plantations and native hardwood forests: Embodied energy, water use and other inputs. *Forest Ecology and Management* **2012**, *264*, 37-50.
411. Stafford, B. *Environmental Aspects of China's Papermkaing Fiber Supply*, Forest Trends: Hobart, Australia, 2007.
412. Joensuu, K.; Rimhanen, K.; Heusala, H.; Saarinen, M.; Usva, K.; Leinonen, I.; Palosuo, T. Challenges in using soil carbon modelling in LCA of agricultural products—the devil is in the detail. *The International Journal of Life Cycle Assessment* **2021**, *26*, 1764–1778, doi:10.1007/s11367-021-01967-1.
413. Responsible Wood. *AS/NZS 4708:202X. Sustainable forest management—requirements. Stage: Public Comment Draft*; Responsoble Wood: New Farm, Queensland, 2020; p. 84.
414. Tasmanian Timber Promotions Board. *Tasmanian Timber. Tasmanian Oak*; Launceston, Tasmania, Australia, 2023.
415. Rosen, K.; Lovett, S. Statement of Enviornmental Effects. Bombala Sawmill Modification 3. Dongwha Australia. **2018**, 229.
416. Schultz, B. Language and the natural environment. In *The Ecolinguistics Reader: Language, Ecology and Environment*, Fill, A., Mülhäusler, P., Eds.; Continuum: London, 2001; pp. 109-114.
417. Leslie, A.J. logging of catchments-- the Victorian problem. *Australian Forestry* **1962**, *26*, 45-49, doi:10.1080/00049158.1962.10675909.
418. IFA. The utilization of timber resources in the watersheds of Victoria. Tendered by W. G. Chandler for the Victorian Division of the Australian Institute of Foresters. *Australian Forestry* **1959**, *23*, 110-113, doi:10.1080/00049158.1959.10675875.
419. Leslie, A.J. The pulp and paper industry in relation to state forestry in Australia. *Australian Forestry* **1954**, *18*, 50-57, doi:10.1080/00049158.1954.10674076.
420. Stanturf, J.A.; Jones, S.B.; Ticknor, W.D. Managing Industrial Forestland in a Changing Society. *Journal of Forestry* **1993**, *91*, 6-11.
421. Hetemäki, L.; Hänninen, R. Outlook for Finland's Forest Industry Production and Wood Consumption for 2015 and 2020. **2009**, 11.
422. Tardif, P.; O'Connor, J. Selling Wood Products to the Green Building Market. **2013**, *2.0*, 83.
423. Reid, H.; Huq, S.; Inkinen, A.; MacGregor, J.; Macqueen, D.; Mayers, J.; Murray, L.; Tipper, R. Using Wood Products to Mitigate Climate Change: A Review of Evidence and Key Issues for Sustainable Development. **2004**, 90.
424. Pingoud, K.; Ekholm, T.; Sievänen, R.; Huuskonen, S.; Hynynen, J. Trade-offs between forest carbon stocks and harvests in a steady state – a multi-criteria analysis. *Journal of Environmental Management 210, pp. 96–103, 2018* **2017**, *210*, 96–103, doi:10.1016/j.jenvman.2017.12.076.
425. Deheza, M.; N'Goran, C.; Bellassen, V. What Incentives To Climate Change Mitigation Through Harvested Wood Products In The Current French Policy Framework? (Summary). **2014**, *Climate Report No 47*.

426. Fernholz, K. Sustainable Forestry: Renewable Materials Storing Carbon. In Proceedings of the Building Sustainably: From Forestland Management to Carbon-Positive, Healthy Buildings, online, 20-October-2020, 2020; p. 13.
427. Naqvi, M.; Yan, J.; Dahlquist, E. Black liquor gasification integrated in pulp and paper mills: A critical review. *Bioresource Technology* **2010**, *101*, 8001–8015.
428. Lippke, B.; Oneil, E.; Harrison, R.; Skog, K.; Gustavsson, L.; Sathre, R. Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. *Carbon Management* **2011**, *2*, 303-333.
429. Harmon, M.E. Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. *Environmental Research Letters* **2019**, *14*, 1-8, doi:10.1088/1748-9326/ab1e95.
430. Guo, P.; Meng, W.; Du, J.; Stevenson, L.; Han, B.; Bao, Y. Lightweight ultra-high-performance concrete (UHPC) with expanded glass aggregate: Development, characterization, and life-cycle assessment. *Construction and Building Materials* **2023**, *371*, 1-14, doi:10.1016/j.conbuildmat.2023.130441.
431. Kulasuriya, C.; Vimonsatit, V.; Dias, W.P.S. Performance based energy, ecological and financial costs of a sustainable alternative cement. *Journal of Cleaner Production* **2021**, *287*, 1-8, doi:10.1016/j.jclepro.2020.125035.
432. Khalil, A.; Wang, X.; Celik, K. 3D printable magnesium oxide concrete: towards sustainable modern architecture. *Additive Manufacturing* **2020**, *33*, 1-9, doi:10.1016/j.addma.2020.101145.
433. Forestry Commission. Timber Production and Marketing. In *Report for year 1978-1979*; State Government of Tasmania: Hobart, Tasmania, 1979; pp. 10-11.
434. Villagrán-Zaccardi, Y.A.; Marsh, A.T.M.; Sosa, M.E.; Zega, C.J.; Belie, N.D.; Bernal, S.A. Complete re-utilization of waste concretes–Valorisation pathways and research needs. *Resources, Conservation & Recycling* **2022**, *177*, 1-16, doi:10.1016/j.resconrec.2021.105955.
435. Shah, I.H.; Miller, S.A.; Jiang, D.; Myers, R.J. Cement substitution with secondary materials can reduce annual global CO_2 emissions by up to 1.3 gigatons. *Nature Communications* **2022**, *13*, 1-11, doi:10.1038/s41467-022-33289-7.
436. Kjellsen, K.O.; Guimaraes, M.; Nilsson, Å. *The CO_2 Balance of Concrete in a Life Cycle Perspective. Nordic Innovation Centre Project NI-project 03018*; Danish Technological Institute: 2005.
437. Cao, Z.; Myers, R.J.; Lupton, R.C.; Duan, H.; Sacchi, R.; Zhou, N.; Miller, T.R.; Cullen, J.M.; Ge, Q.; Liu, G. The sponge effect and carbon emission mitigation potentials of the global cement cycle. *Nature Communications* **2020**, *11*, 1-9, doi:10.1038/s41467-020-17583-w.
438. Verbeck, G.J. *Carbonation of Hydrated Portland Cement. ASTM Special Technical Publication 205.*; Research and Development Laboratories of the Portland Cement Association: Chicago, Illinois, 1958; p. 20.
439. Xi, F.; Davis, S.J.; Ciais, P.; Crawford-Brown, D.; Guan, D.; Pade, C.; Shi, T.; Syddall, M.; Lv, J.; Ji, L.; et al. Substantial global carbon uptake by cement carbonation. *Nature Geoscience* **2016**, *9*, 880-883, doi:10.1038/ngeo2840.
440. Giuliani, U.; Grazian, S.; Alotto, P.; Agostini, M.; Bustreo; Zollino, G. Nuclear Fusion impact on the requirements of power infrastructure assets in a decarbonized electricity system. *Fusion Engineering and Design* **2023**, *192*, 1-8, doi:10.1016/j.fusengdes.2023.113554.
441. Heath, R.L. A rhetorical approach to zones of meaning and organizational prerogatives. *Public Relations Review* **1993**, *19*, 141-155.
442. FWPA. *About*; Forest & Wood Products Australia: Melbourne, Australia, 2023; p. 3.
443. FWPA. *FAQs*; Forest & Wood Products Australia: Melbourne, Australia, 2023; p. 3.
444. Wood, R. *Find Certified*; Responsible Wood: Kedron, Queensland, Australia, 2023.
445. Morris, F.; Allen, S.; Hawkins, W. On the embodied carbon of structural timber versus steel, and the influence of LCA methodology. *Building and Environment* **2021**, *206*, 1-10, doi:10.1016/j.buildenv.2021.108285.
446. Bessou, C.; Benoist, A.; Tailleur, A.; Godard, C.; Gac, A.; Cour, J.L.d.l.; Boissy, J.; Mischler, P. Accounting for land use contribution to climate change in agricultural LCA: Which methods? Which impacts? In Proceedings of the Global food challenges towards sustainable consumption and production, Bangkok, Thailand, 17-19-October-2018, 2018; pp. 1-4.
447. Seventer, M.; Luo, Z.; Eady, S.; Grant, T. Including long-term soil organic carbon changes in life cycle assessment of agricultural products. *The International Journal of Life Cycle Assessment* **2020**, *25*, 1231–1241, doi:10.1007/s11367-019-01660-4.
448. Stewart, P.J. Forestry for carbon dioxide fixation. *The Commonwealth Forestry Review* **1978**, *57*, 263-266.
449. Raison, R.J.; Khanna, P.K.; Crane, W.J.B. Effects of intensified harvesting on rates of nitrogen and phosphorus removal from *Pinus radiata* and eucalyptus forests in Australia and New Zealand. *New Zealand Journal of Forestry Science* **1982**, *12*, 394-403.
450. Dewar, R.C.; McMurtrie, R.E. Sustainable stemwood yield in relation to the nitrogen balance of forest plantations: a model analysis. *Tree Physiology* **1996**, *16*, 173-182.
451. Donald, D.G.M. Silviculture and yield. *South African Forestry Journal* **1987**, *141*, 45-52, doi:10.1080/00382167.1987.9630260.

452. Forsius, F.; Akujärvi, A.; Mattsson, T.; Holmberg, M.; Punttila, P.; Posch, M.; Liski, J.; Repo, A.; Virkkala, R.; Vihervaara, P. Modelling impacts of forest bioenergy use on ecosystem sustainability: Lammi LTER region, southern Finland. *Ecological Indicators* **2015**, *65*, 66-75, doi:10.1016/j.ecolind.2015.11.032.
453. Ahamer, G. Why biomass fuels are principally not carbon neutral. *Energies* **2022**, *15*, 1-39, doi:10.3390/en15249619.
454. Kuhlman, T.; Farrington, J. What is sustainability? *Sustainability* **2010**, *2*, 3436-3448, doi:10.3390/su2113436.
455. Smith, J.H.G. An economic view suggests that the concept of sustained yield should have gone out with the crosscut saw. *The Forestry Chronicle* **1969**, *45*, 167-171, doi:10.5558/tfc45167-3.
456. Doley, D. Session Review. Forests and the global environment. *Australian Forestry* **1993**, *56*, 317-326, doi:10.1080/00049158.1993.10674621.
457. Vance, E.D. Agricultural site productivity: principles derived from long-term experiments and their implications for intensively managed forests. *Forest Ecology and Management* **2000**, *138*, 369-396, doi:10.1016/S0378-1127(00)00425-4.
458. Tellnes, L.G.F.; Ganne-Chedeville, C.; Dias, A.; Dolezal, F.; Hill, C.A.S.; Zea Escamilla, E. Comparative assessment for biogenic carbon accounting methods in carbon footprint of products: a review study for construction materials based on forest products. . *iForest : Biogeosciences and Forestry* **2017**, *10*, 815-823, doi:10.3832/ifor2386-010.
459. Hararuk, O.; Xia, J.; Yiqi Luo1. Evaluation and improvement of a global land model against soil carbon data using a Bayesian Markov chain Monte Carlo method. *Journal of Geophysical Research: Biogeosciences* **2014**, *119*, 403-417, doi:doi:10.1002/2013JG002535.
460. Metsaranta, J.M.; Shaw, C.H.; Kurz, W.A.; Boisvenue, C.; Morken, S. Uncertainty of inventory-based estimates of the carbon dynamics of Canada's managed forest (1990–2014). *Canadian Journal of Forest Research* **2017**, *47*, 1082–1094, doi:10.1139/cjfr-2017-0088.
461. Subedi, B.P.; Pandey, S.S.; Pandey, A.; Eak Bahadur Rana; Bhattarai, S.; Banskota, T.R.; Charmakar, S.; Tamrakar, R. *Forest Carbon Stock Measurement: guidelines for measuring carbon stocks in community-managed forests*, Second ed.; Asia Network for Sustainable Agriculture and Bioresources; Federation of Community Forest Users, Nepal; International Centre for Integrated Mountain Development; Norwegian Agency for Development Cooperation: Kathmandu, Nepal, 2011; p. 67.
462. IPCC. *2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4 Agriculture, Forestry and Other Land Use, Chapter 2 Generic Methodologies Applicable to Multiple Landuse Categories*; National Greenhouse Gas Inventories Programme, IGES: Japan, 2006; p. 59.
463. Liu, W.; Yu, Z.; Xie, X.; Gadow, K.v.; Peng, C. A critical analysis of the carbon neutrality assumption in life cycle assessment of forest bioenergy systems. *Environmental Reviews* **2018**, *26*, 93–101, doi:10.1139/er-2017-0060.
464. Fan, Y.V.; Klemeš, J.J.; Ko, C.H. Bioenergy carbon emissions footprint considering the biogenic carbon and secondary effects. *International Journal of Energy Research* **2021**, *45*, 283-296, doi:10.1002/er.5409.
465. Albers, A.; Collet, P.; Benoist, A.; Hélias, A. Back to the future: dynamic full carbon accounting applied to prospective bioenergy scenarios. *International Journal of Life Cycle Assessment* **2020**, *25*, 1242-1258, doi:10.1007/s11367-019-01695-7.
466. Petersen, B.M.; Knudsen, M.T.; Hermansen, J.E.; Halberg, N. An approach to include soil carbon changes in life cycle assessments. *Journal of Cleaner Production* **2013**, *52*, 217-224, doi:10.1016/j.jclepro.2013.03.007.
467. Paul, K.I.; Polglase, P.J.; Richards, G.P. Predicted change in soil carbon following afforestation or reforestation, and analysis of controlling factors by linking a C accounting model (CAMFor) to models of forest growth (3PG), litter decomposition (GENDEC) and soil C turnover (RothC). *Forest Ecology and Management* **2003**, *177*, 485-501.
468. Cowie, A.L.; Smith, P.; Johnson, D. Does soil carbon loss in biomass production systems negate the greenhouse benefits of bioenergy? *Mitigation and Adaptation Strategies for Global Change* **2006**, *11*, 979-1002, doi:10.1007/s11027-006-9030-0.
469. Wutzler, T.; Reichstein, M. Soils apart from equilibrium – consequences for soil carbon balance modelling. *Biogeosciences* **2007**, *4*, 125-136, doi:10.5194/bg-4-125-2007.
470. Fridman, J.; Holm, S.; Nilsson, M.; Nilsson, P.; Ringvall, A.H.; Ståhl, G. Adapting National Forest Inventories to changing requirements – the case of the Swedish National Forest Inventory at the turn of the 20th century. *Silva Fennica* **2014**, *48*, 1-29, doi:10.14214/sf.1095.
471. Hill, C.A.S. The Environmental Consequences Concerning the Use of Timber in the Built Environment. *Frontiers in Built Environment* **2019**, *5*, 1-10, doi:10.3389/fbuil.2019.00129.
472. Berry, S.L.; Roderick, M.L. Changing Australian vegetation from 1788 to 1988: effects of CO_2 and land-use change. *Australian Journal of Botany* **2006**, *54*, 325-338.
473. Banfai, D.S.; Bowman, D.J.M. Dynamics of a savanna-forest mosaic in the Australian monsoon tropics inferred from stand structures and historical aerial photography. *Australian Journal of Botany* **2005**, *53*, 185-194.

474. Bond, W.J.; Woodward, F.I.; Midgley, G.F. The global distribution of ecosystems in a world without fire. *New Phytologist* **2005**, *165*, 525-538.
475. Cloudsley-Thompson, J.L. Desert expansion and the adaptive problems of the inhabitants. In *Environmental Physiology of Desert Animals*, Hadley, N.F., Ed.; Dowden, Hutchinson and Ross: Stroudsburg, Pennsylvania, 1975; pp. 255-268.
476. Specht, R.L. Changes in the Eucalypt Forest of Australia as a Result of Human Disturbance. In *The Earth in Transition: Patterns and Processes of Biotic Impoverishment*, Woodwell, G.M., Ed.; Cambridge Univesity Press: New York, 1990; pp. 177-197.
477. McIntosh, P.D.; Laffan, M.D.; Hewitt, A.E. The role of fire and nutrient loss in the genesis of the forest soils of Tasmania and southern New Zealand. . *Forest Ecology and Management* **2005**, *220*, 185-215.
478. Gill, R.A. Influence of 90 years of protection from grazing on plant and soil processes in the subalpine of the Wasatch Plateau USA. *Rangeland Ecology and Management* **2007**, *60*, 88-98.
479. Gough, C.M.; Vogel, C.S.; Harrold, K.H.; George, K.; Curtis, P.S. The legacy of harvest and fire on ecosystem carbon storage in a northern temperate forest. *Global Change Biology* **2007**, *13*, 1935-1949, doi:10.1111/j.1365-2486.2007.01406.x.
480. Williams, R.J.; Bradstock, R.A.; Cary, G.J.; Enright, N.J.; Gill, A.M.; Liedloff, A.C.; Lucas, C.; Whelan, R.J.; Andersen, A.A.; Bowman, D.J.M.S.; et al. The impact of climate change on fire regimes and biodiversity in Australia—a preliminary assessment. **2009**, 257.
481. Aguilar, F.X.; Mirzaee, A.; McGarvey, R.G.; Shifey, S.R.; Burtraw, D. Expansion of US wood pellet industry points to positive trends but the need for continued monitoring. *Nature Scientific Reports* **2020**, *10*, 1-17, doi:10.1038/s41598-020-75403-z.
482. Mai-Moulin, T. *Sustainable bioenergy pathways. Conditions to move forward*; Copernicus Institute of Sustainable Development, Utrecht University: Utrecht, The Netherlands, 2020; p. 279.
483. Johnson, F.X.; Roman, M. *Biofuels Sustainability Criteria. Relevant issues to the proposed Directive on the promotion of the use of energy from renewable sources*; European Parliament: Bruxelles, 2008; p. 33.
484. Ter-Mikaelian, M.T.; Colombo, S.J.; Chen, J. The burning question: Does forest bioenergy reduce carbon emissions? A review of common misconceptions about forest carbon accounting. *Journal of Forestry* **2015**, *113*, 57-68, doi:10.5849/jof.14-016.
485. Giuntoli, J.; Barredo, J.I.; Avitabile, V.; Camia, A.; Cazzaniga, N.E.; Grassi, G.; Jasinevičius, G.; Jonsson, R.; Marelli, L.; Robert, N.; et al. The quest for sustainable forest bioenergy: win-win solutions for climate and biodiversity. *Renewable and Sustainable Energy Reviews* **2022**, *159*, 1-20, doi:10.1016/j.rser.2022.112180.
486. Benemann, J.R. CO_2 mitigation with microalgae systems. *Energy Conservation and Management* **1997**, *38*, S475-S479, doi:10.1016/S0196-8904(96)00313-5.
487. Terlouw, T.; Bauer, C.; Rosa, L.; Mazzotti, M. Life cycle assessment of carbon dioxide removal technologies: a critical review. *Energy & Environmental Science* **2021**, *14*, 1701-1721, doi:10.1039/d0ee03757e.
488. Giuntoli, J.; Searle, S.; Jonsson, R.; Agostini, A.; Robert, N.; Amaducci, S.; Marelli, L.; Camia, A. Carbon accounting of bioenergy and forest management nexus. A reality-check of modeling assumptions and expectations. *Renewable and Sustainable Energy Reviews* **2020**, *134*, 1-21, doi:10.1016/j.rser.2020.110368.
489. Menzes, G.B.; Golkar, T.; Moo-Young, H.H. Pulp and paper mill effluents management. *Water Environment Research* **2011**, *83*, 1525-1531.
490. Mathieu, F.; François, N.; Nicolas, R.; Frédéric, M. Quantifying the impact of forest management on the carbon balance of the forest-wood product chain: A case study applied to even-aged oak stands in France. *Forest Ecology and Management* **2012**, *279*, 176-188.
491. Milani, D.; Kiani, A.; McNaughton, R. Renewable-powered hydrogen economy from Australia's perspective. *International Journal of Hydrogen Energy* **2020**, *45*, 24125-24145, doi:10.1016/j.ijhydene.2020.06.041.
492. AECOM; ARNEA. *Australia's Off-grid Clean Energy Market. Research Paper*; Canberra, Australia, 2014; p. 133.
493. SFM. Bioenergy: How the Timber Industry is Contributing to Growth in Renewables. Available online: https://sustainableforestmanagement.com.au/growth-in-renewable-bioenergy/ (accessed on 07-October-2024).
494. Bioenergy Australia. About Bioenergy. Available online: https://www.bioenergyaustralia.org.au/about/ (accessed on 07-October-2024).
495. Aalde, H.; Gonzalez, P.; Gytarsky, M.; Krug, T.; Kurz, W.A.; Ogle, S.; Raison, J.; Schoene, D.; Ravindranath, N.H.; Carle, J.B.; et al. IPCC Guidelines for National Greenhouse Gas Inventories. Chapter 4, Forest Land. Institute for Global Environmental Strategies: Hayama, Japan, 2006; pp. 4.1-4.83.
496. Wang, C.; Walsh, S.D.C.; Weng, Z.; Haynes, M.W.; Summerfield, D.; Feitz, A. Green steel: Synergies between the Australian iron ore industry and the production of green hydrogen. *International journal of hydrogen energy* **2023**, *48*, 32277-32293, doi:10.1016/j.ijhydene.2023.05.041.
497. Brown, D.; Rowe, A.; Wild, P. Techno-economic comparisons of hydrogen and synthetic fuel production using forest residue feedstock. *International Journal of Hydrogen Energy* **2014**, *39*, 12551-12562, doi:10.1016/j.ijhydene.2014.06.100.

498. Balat, H.; Kırtay, E. Hydrogen from biomass-- Present scenario and future prospects. *International Journal of Hydrogen Energy* **2010**, *35*, 7416-7426, doi:10.1016/j.ijhydene.2010.04.137.
499. Nurdiawati, A.; Urban, F. Decarbonising the refinery sector: A socio-technical analysis of advanced biofuels, green hydrogen and carbon capture and storage developments in Sweden. *Energy Research and Social Science* **2022**, *84*, 1-16, doi:10.1016/j.erss.2021.102358.
500. Australian Government. *Native Forest Wood Waste removed from Renewable Energy Target, Media Release, 16-December-2022* The Honourable Chris Bowen MP, Minister for Climate Change and Energy: Canberra, 2022.
501. Ardo, F.M.; Lim, J.W.; Ramli, A.; Lam, M.K.; Kiatkittipong, W.; Abdelfattah, E.A.; Shahid, M.K.; Usman, A.; Wongsakulphasatch, S.; Sahrin, N.T. A review in redressing challenges to produce sustainable hydrogen from microalgae for aviation industry. *Fuel* **2022**, *330*, 1-11, doi:10.1016/j.fuel.2022.125646.
502. Gupta, A.; Likozar, B.z.; Jana, R.; Chanu, W.C.; Singh, M.K. A review of hydrogen production processes by photocatalytic water splitting e From atomistic catalysis design to optimal reactor engineering. *International Journal of Hydrogen Energy* **2022**, *47*, 33282-33307, doi:10.1016/j.ijhydene.2022.07.210.
503. Rosero-Chasoy, G.; Rodríguez-Jasso, R.M.; Aguilar, C.N.; Buitrón, G.; Chairez, I.; Ruiz, H.A. Green hydrogen production: a critical review. In *Green Approach to Alternative Fuel for a Sustainable Future*, Shah, M.p., Ed.; Elsevier: Cambridge, MA, USA, 2023; pp. 381-390.
504. Cowie, A.L.; Berndes, G.; Bentsen, N.S.; Brandão, M.; Cherubini, F.; Egnell, G.; George, B.; Gustavsson, L.; Hanewinkel, M.; Harris, Z.M.; et al. Applying a science-based systems perspective to dispel misconceptions about climate effects of forest bioenergy. *Global Change Biology - Bioenergy* **2021**, *13*, 1210-1231, doi:10.1111/gcbb.12844.
505. Achat, D.L.; Deleuze, C.; Landmann, G.; Pousse, N.; Ranger, J.; Augusto, L. Quantifying consequences of removing harvesting residues on forest soils and tree growth – A meta-analysis. *Forest Ecology and Management* **2015**, *348*, 124–141, doi:10.1016/j.foreco.2015.03.042.
506. UNECE. *Forest Products Annual Market Review 2018-2019*; UNECE and FAO: Geneva, 2019.
507. Forest Stewardship Council. Guidance for Demonstrating Ecosystem Services Impacts. FSC-GUI-30-006 V1-1 EN. **2021**.
508. FAO; UNEP. *The State of the World's Forests 2020. Forests, biodiversity and people.*; Food and Agriculture Organization of the United Nations: Rome, 2020; p. 214.
509. Siry, J.P.; Cubbage, F.W.; Ahmed, M.R. Sustainable forest management: global trends and opportunities. *Forest Policy and Economics* **2005**, *7*, 551-561, doi:10.1016/j.forpol.2003.09.003.
510. FAO. *Global Forest Resources Assessment 2010. Main Report. FAO Forestry Paper 163*; Food and Agriculture Organization of the United Nations: Rome, 2010.
511. FAO. Forest Resources Assessment 2000. Food and Agriculture Organization of the United Nations: University of Michigan, 2001; p. 479.
512. Krankina, O.N.; Harmon, M.E. The impact of intensive forest management on carbon stores in forest ecosystems. *World Resource Review* **1994**, *6*, 161-177.
513. UNECE. *Forest Products Annual Market Review 2020-2021*; United Nations Economic Commission for Europe: Geneva, 2021; p. 86.
514. Linser, S.; O'Hara, P. *Guidelines for the Development of a Criteria and Indicator Set for Sustainable Forest Management. Geneva Timber and Forest Discussion Paper 73*; Lugov, D.C.A., Ed.; United Nations and Food and Agriculture Organization of the United Nations: Geneva, Switzerland, 2019; p. 91.
515. MPWG. *The Montréal Process. Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests*; Research Institute of Forestry, Chinese Academy of Forestry, Beijing, China: Beijing, China, 2015; p. 31.
516. MPWG. *Montréal Process Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests. Technical Notes on Implementation of the Montréal Process Criteria and Indicators. Criteria 1-7. Third Edition, June 2009. Revised July 2014.*; 2014; p. 101.
517. Cowie, A.L. *Expert report on Climate Change and Ecologically Sustainable Development matters Land and Environment Court Proceedings 2021/00128111*; Department of Primary Industries, New South Wales: New South Wales, Australia, 2021; p. 35.
518. NSW Parliament. Legislative Council. 2019-20-21. First Session of the Fifty-Seventh Parliament. Questions and Answers. No 597. **2021**, *Friday 1 October 2021*, 19.
519. HRL Technology Group. *Redbank Power Station QA/QC, Supply Chain and Material Handling. Addendum Report*; HRL Technology Group Pty Ltd: Mulgrave, VIC, Australia, 2021; p. 45.
520. Richards, G.P. *The FullCAM Carbon Accounting Model: Development, Calibration and Implementation for the National Carbon Accounting System. NCAS Technical Report No. 28. Parts 1 and 2.*; Australian Greenhouse Office: Canberra, Australia, 2001; p. 61.
521. Ximenes, F.; Coburn, R.; McLean, M.; Samuel, J.; Cameron, N.; Law, B.; Threllfall, C.; Wright, K.; Macintosh, S. *North Coast Residues. A project undertaken as part of the 2023 North Coast Forestry Project*; NSW Department of Primary Industries: Sydney, Australia, 2017; p. 66.

522. Poynter, M.; Ryan, M. Leadbeater's possum and Victoria's Central Highlands' forests: flawed science and environmental activism as drivers of forest management change. *Australian Foresty* **2018**, *81*, 250-272, doi:10.1080/00049158.2018.1537455.
523. Keith, H.; Lindenmayer, D.; Mackey, B.; Blair, D.; Carter, L.; McBurney, L.; Okada, S.; Konishi-Nagano, T. Managing temperate forests for carbon storage: impacts of logging versus forest protection on carbon stocks. *Ecosphere* **2014**, *5*, art75, doi:10.1890/es14-00051.1.
524. Lindenmayer, D.B.; Sato, C. Hidden collapse is driven by fire and logging in a socioecological forest ecosystem. *PNAS* **2018**, *115*, 5181-5186, doi:10.1073/pnas.1721738115.
525. Dean, C. Creative Carbon Accounting—A Reply to "TheWood, the Trees, or the Forest? Carbon in Trees in Tasmanian State Forest: A Response to Comments". *International Journal of Forestry Research* **2012**, *2012*, Article ID 762080, doi:10.1155/2012/762080.
526. Moroni, M.T.; Kelley, T.H.; McLarin, M.L.; Read, S.M. The Wood, the Trees, or the Forest? Carbon in Trees in Tasmanian State Forest: A Response to Comments. *International Journal of Forestry Research* **2012**, *2012*, 1-6, doi:10.1155/2012/848502.
527. Sterman, J.D.; Siegel, L.; Rooney-Varga, J.N. Reply to comment on 'Does replacing coal with wood lower CO2 emissions? Dynamic lifecycle analysis of wood bioenergy'. *Environmental Research Letters* **2018**, *13*, 1-8, doi:10.1088/1748-9326/aaf354.
528. Prisley, S.P.; Gaudreault, C.; Lamers, P.; Stewart, W.; Miner, R.; Junginger, H.M.; Oneil, E.; Malmsheimer, R.; Volk, T.A. Comment on 'Does replacing coal with wood lower CO_2 emissions? Dynamic lifecycle analysis of wood bioenergy'. *Environmental Research Letters* **2018**, *13*, doi:10.1088/1748-9326/aaf203.
529. Sanger, J.C.; Ferrari, A. The Grove of Giants: Tasmania's most carbon-dense forest. *Austral Ecology* **2023**, *48*, 1245-1251, doi:10.1111/aec.13407.
530. Johnson, D.W. Effects of forest management on soil carbon storage. *Water, Air and Soil Pollution* **1992**, *64*, 83-120.
531. Brown, S.; Lugo, A.E. Biomass of tropical forests: A new estimate based on forest volumes. *Science* **1984**, *223*, 1290-1293.
532. Moroni, M.T.; Lewis, T. *Comparing measured and modelled eucalypt forest carbon stocks. Technical Report 01/2015*; Forestry Tasmania: Hobart, Australia, 2015; p. 25.
533. Snowdon, P.; Eamus, D.; Gibbons, P.; Khanna, P.K.; Keith, H.; Raison, R.J.; Kirschbaum, M.U.F. *Synthesis of Allometrics, Review of Root Biomass and Design of Future Woody Biomass Sampling Strategies. Techincal Report No. 17.*; Australian Greenhouse Office, Commonwealth of Australia: Canberra, Australia, 2000; Volume 17.
534. Grierson, P.F.; Adams, M.A.; Attiwill, P.M. Estimates of Carbon Storage in the above-ground biomass of Victoria's Forests. *Australian Journal of Botany* **1992**, *40*, 631-640, doi:10.1071/BT9920631.
535. MBAC. *Forestry Tasmania's Carbon Sequestration Position*; MBAC Consulting Group Pty. Ltd.: Melbourne, Australia, 2007; p. 43.
536. Brown, S.; Gillespie, A.J.R.; Lugo, A.E. Biomass estimation methods for tropical forests with applications to forest inventory data. *Forest Science* **1989**, *35*, 881-902, doi:10.1093/forestscience/35.4.881.
537. PFT; Goodwin, A.N. *Farm Forestry Toolbox Version 5.3.8b*; Private Forests Tasmania: Hobart, Tasmania, 2016.
538. Ximenes, F.A.; Gardner, W.D.; Kathuria, A. Proportion of above-ground biomass in commercial logs and residues following the harvest of five commercial forest species in Australia. *Forest Ecology and Management* **2008**, *256*, 335-346.
539. Setiahadi, R. Comparison of individual tree aboveground biomass estimation in community forests using allometric equation and expansion factor in Magetan, East Java, Indonesia. *Biodiversitas* **2021**, *22*, 3899-3909.
540. Lawrence, A.G.S. Assessment of Florentine Valley. In *Annual Report*, Irby, I.G., Ed.; Forestry Department: Hobart, Tasmania, 1922; Volume 27, pp. 14-18 + maps.
541. ANM. *Australian Newsprint Mills. Unpublished logging coupe registers for the Florentine, Tyenna and Styx Valleys. Forestry Tasmania Library, Hobart, Tasmania*; c1960.
542. Dean, C.; Fitzgerald, N.B.; Wardell-Johnson, G.W. Pre-logging carbon accounts in old-growth forests, via allometry: An example of mixed-forest in Tasmania, Australia. *Plant Biosystems* **2011**, *146*, 223-236, doi:10.1080/11263504.2011.638332.
543. Wardlaw, T. Measuring a fire. The story of the January 2019 fire told from measurements at the Warra supersite, Tasmania. *Fire* **2021**, *4*, 1-18, doi:10.3390/fire4020015.
544. Ximenes, F.A.; Kathuria, A.; McLean, M.; Coburn, R.; Sargeant, D.; Ryan, M.; Williams, J.; Boer, K.; Mo, M. Carbon in mature native forests in australia: The role of direct weighing in the derivation of allometric equations. *Forests* **2018**, *9*, 1-26, doi:10.3390/f9020060.
545. Dargavel, J.B. The Development of the Tasmanian Wood Industries: a Radical Analysis. Australian National University, Canberra, Australia, 1982.
546. Wardlaw, T. *Warra Tall Eucalypt Stem Diameter, Height and Aboveground Woody Biomass Data. Version 1.0. Terrestrial Ecosystem Research Network. (Dataset).* TERN: 2022.

547. Goodwin, A.N. A taper equation for *Eucalyptus obliqua*. In Proceedings of the Integrating Forest Information Over Space and Time. IUFRO S4.01 Mensuration, Growth and Yield and S4.02 Forest Resource Inventory, Canberra, Australia, 13-17-January-1992, 1992; pp. 454-467.
548. APPM. Evaluation of South Arthur River Myrtle with Decay as a Pulpwood Resource. Unpublished report by Australian Pulp and Paper Mills, Burnie, Tasmania, deposited with the Forestry Commission. **1980**, 46, plus photos, maps and data sheets.
549. Harcombe, P.A.; Allen, R.B.; Wardle, J.A.; Pratt, K.H. Spatial and temporal patterns in stand structure, biomass, growth, and mortality in a monospecific Nothofagus solandrivar. cliffortioides (Hook, f.) Poole Forest in New Zealand. *Journal of Sustainable Forestry* **1997**, *63*, 313-345, doi:10.1300/J091v06n03_06.
550. May, B.; Bulinski, J.; Goodwin, A.; Macleod, S. *Tasmanian Forest Carbon Study. Report was commissioned by the State Government through the Tasmanian Climate Change Office.*; CO2 Australia Limited: 2012; p. 372.
551. da Silva, W.; da Silva, C.P. *LABFit Curve Fitting Software. v7.2.48*; Brazil, 2015.
552. Moroni, M.T.; Kelley, T.H.; McLarin, M.L. Carbon in Trees in Tasmanian State Forest. *International Journal of Forestry Research* **2010**, *2010*, 1-13, doi:10.1155/2010/690462.
553. Dean, C. Comment on 'Carbon in Trees in Tasmanian State Forest'. *International Journal of Forestry Research* **2011**, *2011*, 1-10, doi:10.1155/2011/212361.
554. Wickett, H.C. The measurent of logs and methods of stating their volumes. In Proceedings of the Australian Timber Industry Stabilisation Conference, Karunda, Queensland, 26-30 September 1960, 1960; p. 6.
555. Gilbert, J.M. Regeneration of the Eucalypts. PhD, University of Tasmania, Hobart, Tasmania, Australia, 1958.
556. Olson, J.S. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* **1963**, *44*, 322-331.
557. Grierson, P.F.; Adams, M.A.; Attiwill, P.M. *Carbon Storage in Soil and in Forest Products*; School of Botany, University of Melbourne: Parkville, Victoria, Australia, 1991; p. 42.
558. Ashton, D.H. The development of even-aged Stands of *Eucalyptus regnans* F. Muell. in central Victoria. *Australian Journal of Botany* **1976**, *24*, 397-414.
559. Ngatuvai, M.; Autrey, C.; McKenny, M.; Elkbuli, A. Significance and implications of accurate and proper citations in clinical research studies. *Annals of Medicine and Surgery* **2021**, *72*, 1-2, doi:10.1016/j.amsu.2021.102841.
560. Dincă, L.C.; Dincă, M.; Vasile, D.; Spârchez, G.; Holonec, L. Calculating organic carbon stock from forest soils. *Notulae Botanicae Horti Agrobotanici Cluj-Napoca* **2015**, *43*, 568-575, doi:10.15835/nbha43210041.
561. Allen, K.; Francey, R.; Michael, K.; Nunez, M. A structural time series approach to the reconstruction of Tasmanian maximum temperatures. *Environmental Modelling & Software* **1999**, *14*, 261–274, doi:10.1016/S1364-8152(98)00078-4.
562. REDD Forests Pty. Ltd. *Reducing Carbon Emissions by Protecting a Native Forest in Tasmania*; : Sydney, New South Wales, Australia, 2009; p. 95.
563. Forests Alive Pty. Ltd. *New Leaf Carbon Project*; Sydney, New South Wales, Australia, 2013; p. 143.
564. Goodwin, A.N. A taper equation for *Eucalyptus obliqua*. In Proceedings of the IUFRO Conference (S 4.01, Mensuration, Growth and Yield) on Integrating Forest Informatoin Over Space and Time, Canberra, Australia, 13-17 January 1992, 1992; pp. 454-467.
565. Grove, S.J.; Stamm, L.; Barry, C. Log decomposition rates in Tasmanian *Eucalyptus obliqua* determined using an indirect chronosequence approach. *Forest Ecology and Management* **2009**, *258*, 389-397, doi:10.1016/j.foreco.2008.11.006.
566. Brack, C.L.; Richards, G.; Waterworth, R. Integrated and comprehensive estimation of greenhouse gas emissions from land systems. *Sustainable Science* **2006**, *1*, 91-106, doi:10.1007/s11625-006-0005-6.
567. Waterworth, R.M.; Richards, G.P. Implementing Australian forest management practices into a full carbon accounting model. *Forest Ecology and Management* **2008**, *255*, 2434-2443.
568. Dean, C.; Fitzgerald, N.B.; Wardell-Johnson, G.W. Errata. Corrigendum. *Plant Biosystems* **2012**, *146*, 487, doi:10.1080/11263504.2012.701492.
569. Abbott, S. *Photographing 'world's biggest blue gum' to shine light on 500 yo giant in logging coupe*; Hobart, Australia, 04-November-2022 2023.
570. Sanger, J.; Pearce, S. The Tree Projects. Available online: https://www.thetreeprojects.com/ (accessed on 07-October-2024).
571. Sanger, J.; Pearce, S. The Tree Projects Facebook page. Available online: https://www.facebook.com/thetreeprojects (accessed on 07-October-2024).
572. Sillett, S.C.; Van Pelt, R.; Koch, G.W.; Ambrose, A.R.; Carroll, A.L.; Antoine, M.E.; Mifsud, B.M. Increasing wood production through old age in tall trees. *Forest Ecology and Management* **2010**, *259*, 976-994, doi:10.1016/j.foreco.2009.12.003.
573. Bowling, P.J. A volume table for regrowth of some Tasmanian eucalypts. *Australian Forestry* **1951**, *15*, 87-95, doi:10.1080/00049158.1951.10675804.

574. Bi, H. Trigonometric variable-form taper equations for Australian eucalypts. *Forest Science* **2000**, *46*, 397-409, doi:10.1093/forestscience/46.3.397.
575. Ilic, J. Woods of eucalyptus — part1. Distinguishing three species from the ash group (*E. regnans*, *E. delegatensis* and *E. obliqua*). *IAWA Journal* **1997**, *18*, 27-36.
576. Chave, J.; Réjou-Méchain, M.; Búrquez, A.; Chidumayo, E.; Colgan, M.S.; Duque, W.B.C.D.A.; Eid, T.; Fearnside, P.M.; Goodman, R.C.; Henry, M.; et al. Improved allometric models to estimate the aboveground biomass of tropical trees. *Global Change Biology* **2014**, *20*, 3177-3190, doi:10.1111/gcb.12629.
577. Ilic, J.; Boland, D.; McDonald, M.; Downes, G.; Blakemore, P. *Woody Density Phase 1 — State of Knowledge. Technical Report No. 18*; Australian Greenhouse Office, Commonwealth of Australia: Canberra, 2000; p. 218.
578. Brack, C. Not everything that can be counted counts. *Australian Forestry* **2014**, *77*, 75-77, doi:10.1080/00049158.2014.908484.
579. Calders, K.; D'hont, B.; Hambrecht, L.; Lucieer, A.; Pearce, S. *Lathamus Keep, Tasmania - 3D model by Q-ForestLab (Ghent University) (@CAVElab_UGent)*; Universiteit Gent: Ghent, Belgium, 2024.
580. Sillett, S.C.; Pelt, R.V.; Carroll, A.L.; Jim Campbell-Spickler; Antoine, M.E. Aboveground biomass dynamics and growth efficiency of *Sequoia sempervirens* forests. *Forest Ecology and Management* **2020**, *458*, 1-43, doi:10.1016/j.foreco.2019.117740.
581. Calders, K.; D'hont, B.; Hambrecht, L.; Lucieer, A.; Pearce, S. *High Res Slice of Lathamus Keep, Tasmania - 3D model by Q-ForestLab (Ghent University) (@CAVElab_UGent)*; Universiteit Gent: Ghent, Belgium, 2024.
582. Roxburgh, S.H. DataGrabber. Software Downloads. Available online: http://www.steverox.info/software_downloads.htm (accessed on 9 November 2010).
583. Sillett, S.C. Tree crown structure and vascular epiphyte distribution in *Sequoia sempervirens* rain forest canopies. *Selbyana* **1999**, *20*, 76-97.
584. Voigt, C. Impacts of Social Trails around Old-Growth Redwood Trees in Redwood National and State Parks. Humboldt State University, Arcata, California, 2016.
585. Erwin, K. *Fragile Majesty. The Battle for North America's Last Great Forest*; The Mountaineers: Seattle, WA, USA, 1989; p. 272.
586. Saatchi, S.S.; Harris, N.L.; Brown, S.; Lefsky, M.; Mitchard, E.T.A.; Zutta, W.S.B.R.; Buermann, W.; Lewis, S.L.; Hagen, S.; Petrova, S.; et al. Benchmark map of forest carbon stocks in tropical regions across three continents. *PNAS* **2011**, *108*, 9899-9904, doi:10.1073/pnas.1019576108.
587. Kurtz, B.C.; Almeida, T.M.H.d.; Coelho, M.A.N.; Deccache, L.S.J.; Tortorelli, R.M.; Gonzaga, D.R.; Madureira, L.K.; Guedes-Oliveira, R.; Barros, C.F.; Siqueira, M.F.d. Quantifying the carbon stocks in urban trees: The Rio de Janeiro Botanical Garden as an important tropical carbon sink. *Journal of Zoological and Botanical Gardens* **2024**, *5*, 579-589, doi:10.3390/jzbg5040039.
588. TWS. *Victoria's Giant Trees. Tall and Big Tree news: From Victoria, Tasmania and occasionally other places!*; 2023.
589. Holmes, A. *Tasmania expands native logging harvest area as other states wind back*; Australian Broadcasting Corporation: Hobart, Tasmania, Australia, 04-July-2023 2023.
590. Hayward, M.W.; Meyer, N.F.V.; Balkenhol, N.; Beranek, C.T.; Bugir, C.K.; Bushell, K.V.; Callen, A.; Dickman, A.J.; Griffin, A.S.; Haswell, P.M.; et al. Intergenerational inequity: Stealing the joy and benefits of nature from our children. *Frontiers in Ecology and Evolution* **2022**, *10*, 1-6, doi:10.3389/fevo.2022.830830.
591. Makarieva, A.M.; Nefiodov, A.V.; Morozov, V.E.; Aleynikov, A.A.; Vasilov, R.G. Science in the vanguard of rethinking the role of forests in the third millennium: comments on the draft concept of the federal law "Forest Code of the Russian Federation". *Forest Science Issues* **2020**, *3*, 1-19, doi:10.31509/2658-607x-2020-3-3-1-25.
592. Yu, B.; Fingrut, A. Sustainable building design (SBD) with reclaimed wood library constructed in collaboration with 3D scanning technology in the UK. *Resources, Conservation & Recycling* **2022**, *186*, 1-12, doi:10.1016/j.resconrec.2022.106566.
593. Hayward, M.W. Problems with the World's Ecosystems: the Future and Attempts to Mitigate Decline. In *The Living Planet: the State of the World's Wildlife*, Maclean, N., Ed.; Cambridge University Press: Cambridge, England, 2023; pp. 349-367.
594. Kamite, M. Compounding method for high quality building materials using fine-powder ligneous refuse technology. *Transactions of the Materials Research Society of Japan* **2001**, *26*, 817-820.
595. Santos, G.; Esmizadeh, E.; Riahinezhad, M. Recycling construction, renovation, and demolition plastic waste: Review of the status quo, challenges and opportunities. *Journal of Polymers and the Environment* **2024**, *32*, 479–509, doi:10.1007/s10924-023-02982-z.
596. Tao, J.; Tahmasebi, P.; Kader, M.A.; Feng, D.; Sahimi, M.; Evans, P.D.; Saadatfar, M. Wood biomimetics: Capturing and simulating the mesoscale complexity of willow using cross-correlation reconstruction algorithm and 3D printing. *Materials & Design* **2023**, *228*, 1-11, doi:10.1016/j.matdes.2023.111812.
597. Wang, Y.; Zhao, J.; Jiang, L.; Zhang, L.; Raghavan, V.; Wang, J. A comprehensive review on novel synthetic foods: Potential risk factors, detection strategies, and processing technologies. *Comprehenisve Reviews in Food Science and Food Safety* **2024**, *23*, 1-32, doi:10.1111/1541-4337.13371.

598. Komatsu, H.; Rappleye, J. Well-being in times of decline: The feasibility and effectiveness of degrowth for sustainability. *Journal of Future Studies* **2024**, *29*, 59-72, doi:10.6531/JFS.202409_29(1).0005.