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Effects of prescribed fire frequency on wildfire emissions and carbon sequestration in a fire adapted ecosystem using a comprehensive carbon model.

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Abstract

Prescribed fire to reduce forest fuels has been routinely applied to reduce wildfire risk in many parts of the world. It has also been proposed that prescribed fire can be used to mitigate greenhouse gas (GHG) emissions. Although prescribed fire creates emissions, if the treatment also decreases the incidence of subsequent wildfires, it is possible for the net outcome to be an emissions decline. Previous studies have suggested prescribed fire, at the frequencies required to materially impact wildfire occurrence, generally leads to net emissions increases. A focus on emissions means any change in carbon storage within the ecosystem remains unaccounted for; because living, dead, and soil carbon pools are characterized by different residence times, a re-distribution of carbon amongst these pools may either reduce or increase long-term ecosystem carbon stores. A full ecosystem carbon model has been developed to investigate the implications of prescribed fire management on total Net Ecosystem Carbon Balance (NECB), inclusive of both emissions and carbon storage. Consistent with previous work, the results suggested limited potential for reducing net GHG emissions through applying prescribed fire, with higher emissions from prescribed fire approximately offset by lower emissions and avoided carbon losses from the subsequent reduction in wildfire frequency. For example, shortening the prescribed fire interval from 25 to 10 years resulted in a NECB sequestration that was typically less than ± 0.4 Mg C ha^{-1} yr^{-1}, or less than approximately 0.1% of the total ecosystem carbon storage. Hence, whilst there was limited opportunity for achieving emission abatement outcomes from changing prescribed fire management, there were no significant emission penalties for doing so. These results suggest land managers should be free to adopt prescribed fire regimes to target specific management outcomes, without significantly impacting net emissions or total ecosystem carbon storage over the long term.

Keywords: Temperate forests, fire severity, south-eastern Australia, Net Ecosystem Carbon Balance, pyrogenic carbon

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1. *Introduction*

Climate change is increasing the risk of wildfires in temperate biomes, with more extreme landscape drying combined with extreme weather events leading to increased occurrence of destructive megafires (Andela *et al.* 2017; IPCC 2019). If an increase in wildfire frequency in temperate biomes is sustained it may alter long-term equilibria between fire-induced CO_2 emissions and carbon (C) uptake by forests, potentially leading to increased total C emissions, and reduced ecosystem C storage (Seidl *et al.* 2017). For example Yang *et al.* (2015) showed that fires reduce the C sink of global terrestrial ecosystems by 0.57 Pg C yr^{-1}. This has implications for the terrestrial C cycle, as over the last several decades forests have acted as a persistent sink for anthropogenic CO_2 emitted to the atmosphere, with temperate forests accounting for around 30% of global C sinks in established forests (Pan *et al.* 2011). Climate change that alters the frequency and severity of wildfires may therefore threaten this sink capacity by altering C cycling dynamics to favour emissions and reduce C storage in the biosphere (Hoegh-Guldberg *et al.* 2018). Australian forest vegetation is dominated by fire-tolerant species of *Eucalyptus* and related genera (Gill and Catling 2002), that can persist on a site with fire return intervals as short as 6 to 9 years (Boer *et al.* 2009), and typically up to every 20 years or more (Murphy *et al.* 2013). Fire is integral to the ecology of these *Eucalyptus* forests with more than 550 000 ha year $^{-1}$ of temperate Australian forests burnt on average in wildfires (DoEE 2016). Over recent decades, areas burnt by wildfires have more than doubled and the number and severity of these wildfires has increased dramatically. For example, early in 2020 around 5.8 million hectares of mainly temperate forest were burnt in southern Australia, and many of the fires exceeded 100 000 ha and continued to burn for weeks (Boer *et al.* 2020). The magnitude of the greenhouse gas (GHG) emissions released in those fires remains understudied but preliminary estimates indicated up to 900 Million tonnes CO_2-e

(Australian Government 2020a), nearly double Australia's average annual emissions over the last 10 years, of 560 Million tonnes CO_2-e yr^{-1} (Australian Government 2020b). Smoke and air pollution from these megafires has raised concerns about short and long term health consequences in the affected populations (Vardoulakis *et al.* 2020), with levels reaching up to 10 times those deemed hazardous, and resulting in excess number of deaths and hospitalizations for cardiovascular and respiratory problems (Arriagada *et al.* 2020; Walter *et al.* 2020). Air pollution from the 2020 Australian fires affected not only Australia but also the Pacific region, New Zealand, and reaching as far as South America and Antarctica. Although the economic and other impacts of the 2019-2020 megafires in Australia are difficult to estimate, the cost of human and animal lives, mental health impacts, economic costs of lost tourism, and fire impacts on the agricultural sector are all undoubtedly significant. High severity wildfires also lead to soil and catchment erosion, affecting water quality (Edwards *et al.* 2015; Blake *et al.* 2020; Robinne *et al.* 2020). Although managing forest fuels with prescribed fires is expensive, the costs must be weighed up against the negative impacts of uncontrolled megafire. In response to the 2020 megafires, a review of fire management practices and calls for more widespread application of Indigenous burning practices have been raised (Russell-Smith *et al.* 2020).

Reducing GHG emissions through fire management in tropical savanna ecosystems is well-established (Cook *et al.* 2016). In temperate forests prescribed fire is routinely applied to reduce wildfire risk to protect lives and property (McCaw 2013). Because prescribed fire is widely applied in fire-adapted temperate forests to reduce wildfire impacts, the question arises as to whether it can be used to mitigate GHG emissions and maximize total ecosystem carbon (TEC), as climate change increases wildfire frequency (Liang *et al.* 2017). Although fuel reduction by prescribed fire itself generates emissions, if the treatment also decreases the intensity and extent of subsequent wildfires, it is possible for the net outcome over the long term to be a total emissions decline (Bradstock *et al.* 2012). In a study that specifically investigated the emissions implications of fire management in Australian temperate forests, Bradstock *et al.* (2012) concluded that the use of prescribed fire is unlikely to yield a net reduction in C emissions, as the emissions savings from wildfire mitigation were insufficient to overcome the significant emissions costs of applying the required prescribed fire treatments in these ecosystems. Other studies in temperate ecosystems have supported this conclusion (Hurteau *et al.* 2011; Williams *et al.* 2012; Campbell and Ager 2013; Restaino and Peterson 2013).

A potential limitation of previous work is that only a subset of the total forest C budget is typically considered within any given analysis, with the main focus often limited to emissions arising from the combustion of forest fuels (Bradstock *et al.* 2012). Some studies have extended the scope to include the dynamics of living biomass (Hurteau *et al.* 2011), though other studies have included the full ecosystem C balance (Krofcheck *et al.* 2017). The danger of incomplete accounting is that it could lead to erroneous conclusions. For example, demonstrating that prescribed fire could lead to a decline in total fire emissions over the long term does not necessarily imply a total net ecosystem GHG benefit, as a focus solely on emissions will miss changes in C storage associated with the re-distribution of C within the ecosystem between the living biomass, dead biomass and soil C pools. Because these pools are characterized by different residence times, such a re-distribution of C, with subsequent post-fire recovery through forest regrowth, has the potential to change total ecosystem C storage over time, that may either increase or decrease the trends in emissions. Studies that include the fire impacts on C dynamics in vegetation, debris and soil, over decadal timeframes, in addition to fire emissions, are therefore required to gain a complete understanding of the net ecosystem C implications of fire management (Wiedinmyer and Hurteau 2010).

One terrestrial pool of C that is often overlooked is pyrogenic carbon (PyC). PyC includes a wide range of products including black C, soot, elemental C, char, charcoal, and biochar (Bird *et al.* 2015). Globally, fires can generate 116-385 Tg PyC per year, corresponding to 0.2–0.6% of the annual terrestrial net primary production (Santin *et al.* 2016). Studies have shown that about 3-5% of combustible biomass will be converted to PyC from a single fire (Worrall *et al.* 2013; Jenkins *et al.* 2016; Krishnaraj *et al.* 2016) and as much as 12% from consecutive fires (Volkova *et al.* 2021). Experimental results have also suggested the chemical composition of PyC from burning forest fuels can vary as a function of fire spread mode, and thus may be amenable to manipulation (Surawski *et al.* 2020). Despite its recognized importance as a long term C store (Swift 2001; Forbes *et al.* 2006), PyC is not explicitly considered as a pool in current global or regional C cycle models, in part due to their rudimentary representation of fire processes (Bowman *et al.* 2009; Volkova *et al.* 2018). Therefore, recognizing PyC in C models may present an opportunity for targeted management to increase this C sink (Lehmann *et al.* 2008). A major hurdle in determining the long-term C balance of forests under changing fire regimes is therefore the availability of suitable modelling tools that incorporate the impacts of fire on all above- and below-ground C stocks, including the production of PyC, whilst also tracking total emissions and other ecosystem fluxes such as decomposition of dead plant material, and soil respiration. Such ‘full-system’ modelling capability facilitates the calculation of the Net Ecosystem Carbon Balance (NECB) (Chapin *et al.* 2006), which is a numerical summary of the net balance between all ecosystem C gains (through e.g. net primary productivity) and all ecosystem C losses (through e.g. fire, decomposition, and soil respiration).

The aim of the study was to develop a comprehensive fire and ecosystem C budget model, calibrated to the forests of southeast Australia, and use it to: (i) explore the efficacy of different prescribed fire scenarios in reducing the frequency of high severity wildfires, across a range of background wildfire probabilities; (ii) explore the implications of different prescribed fire management scenarios on the total net ecosystem C balance (NECB), inclusive of living biomass, dead biomass, soil organic C, and PyC; and (iii) quantify the potential for prescribed fire management to mitigate wildfire emissions and associated changes in total ecosystem C storage.

2. Materials and Methods

2.1. Overview of the ecosystem carbon model

A novel forest ecosystem C cycling model was developed for this study in the Python programming language (Van Rossum and Drake 2009) using annual time steps and inputs (increments) of C, allocated from annual net primary productivity (NPP) to trees, shrubs and grasses; with each C pool assigned a longevity to quantify the residency time of C in the biosphere (Figure 1). The model was first calibrated to account for above- and below-ground C dynamics, before introducing fire. The introduction of fire events leads to a transfer (or conversion) of C from one pool to another, and also to GHG emissions arising from the combustion of organic matter. Contrasting fire regimes were included through varying the timing of fire events, and through varying the severity of the fire impacts as represented by the proportion of matter combusted from each C pool, and by varying rates of transfer between pools.

The effects of weather were included in the model indirectly, embedded within the empirical data for both prescribed fires (collected from burns during mild weather) and wildfires (collected during more extreme weather). High intensity crown wildfires were therefore assumed to occur under conditions different than prescribed burns (more details about fire regimes are given in Section 2.3).

The gaseous fire emissions species accounted for in the model were CO_2, N_2O, CH_4 and CO. The C model equations were based on first-order growth and decay dynamics that are commonly applied in models of the terrestrial C cycle (Goldewijk *et al.* 1994; Roxburgh *et al.* 2006). See Supplementary 1.1 for the details of the equations.

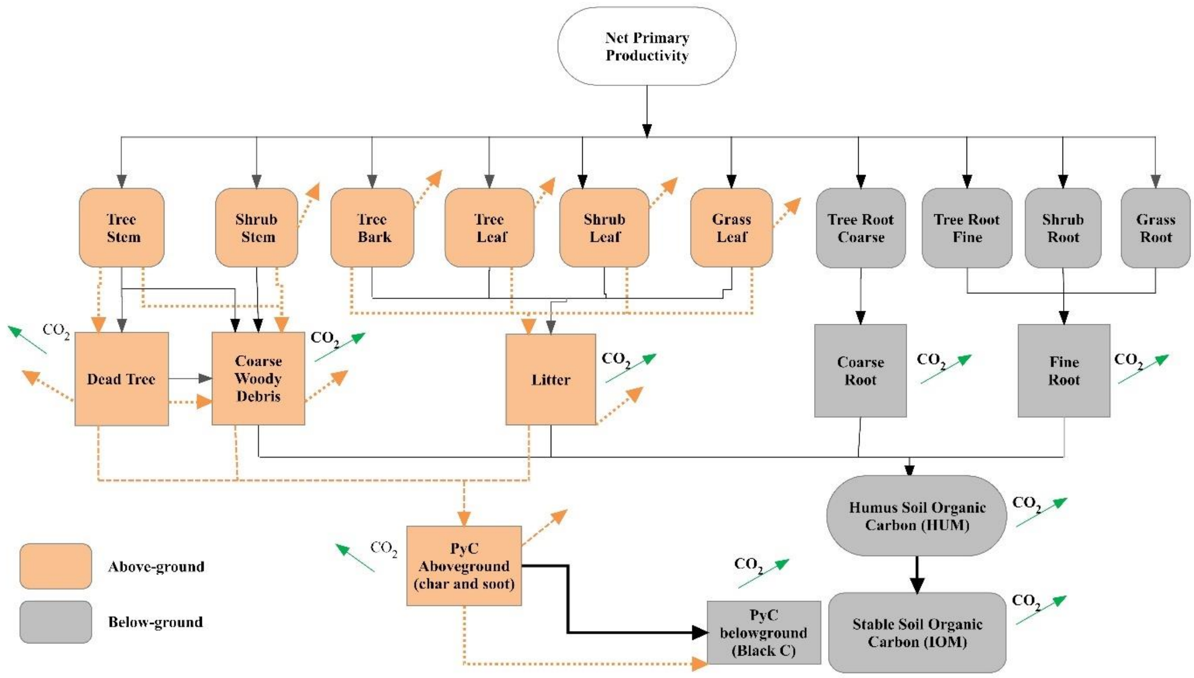


Figure 1. Model overview showing interactions between living biomass, dead organic matter and soil organic matter in response to fire. Green arrows of CO_2 are the decomposition from individual pools. Dashed lines are fire effects on C pools, dashed arrows up are fire emissions including CO_2 and non - CO_2.

2.2. Model calibration

Model parameter values were estimated from extensive empirical data collected from temperate Australian forests, located predominantly within the 'Foothills Mixed' forest class of

Victoria (Haverd *et al.* 2013; Volkova and Weston 2013; Volkova *et al.* 2014; Volkova and Weston 2015; Volkova *et al.* 2019b). Model calibration was achieved through iteratively adjusting parameter values such that model predictions of the living and debris C pools matched the field observations to within ±1 Mg C ha^{-1} (Table S1). To facilitate calibration the modelled C pools were assumed to be at steady state on the basis that: (i) the two C pools that take longest to equilibrate (tree biomass and SOC) are minimally impacted by fire in the forest types targeted in this study, and that the majority of sites on which the calibration data was collected had not been materially disturbed for at least 26 years (see 'Foothills mixed' references immediately above); (ii) the intention of the calibration was to capture the general dynamics of a fire-adapted temperate *Eucalyptus* forest, rather than to focus on any specific location, and (iii) to ensure the non-steady state model dynamics were realistic, the temporal trend in forest biomass accumulation was also calibrated against published above-ground biomass accumulation curves for the same forest type (Grierson *et al.* 1992) (Figure S1).

The primary driver of the terrestrial C cycle is NPP. Consistent with the modelled NPP for the study region, NPP was set to 7.5 Mg C ha^{-1} yr^{-1} (Haverd *et al.* 2013). For the remaining parameters, the living biomass pools were first calibrated to field data through modifying the allocation coefficients (α), longevities (L) and partitioning fractions (h). The dead biomass pools were then calibrated through modifying their longevity and partitioning fractions, followed by the soil organic C pools. In total, 65 parameters were calibrated for the living biomass, dead biomass and soil C pools (Table S2).

2.3 Fire module

During a fire event there can be instantaneous transfers of C between the pools, and to the atmosphere, as depicted by the dashed lines in Figure 1. These transfers are implemented in the model as fractional losses, and in a year in which a fire occurs, are applied following the growth calculations described above.

The fire component of the model was calibrated to simulate two types of disturbances: prescribed fire (PF) and wildfire (WF). PF is assumed to be a low intensity fire, with limited physical damage to fire tolerant eucalypts (McArthur and Cheney 2015). WF is assumed to be a crown fire of high intensity, leading to the loss of all leaves (through combustion and litterfall), and causing death of some mature trees (McArthur and Cheney 2015). Such crown fires are assumed to be beyond control by direct suppression, and can only occur when litter loads reach above 10-12 Mg ha^{-1} (or 80% of the steady state) (Raison *et al.* 1986). Recognizing that wildfires can occur at fuel loads as low as 3 Mg ha^{-1} (Burrows 1999) and can recur almost immediately after prescribed fires (see Volkova *et al.* 2014), they have minimal long-term impacts on the forest C budget (Volkova *et al.* 2014), low intensity wildfires were not considered in the model scenarios. Differences between PF and WF were represented within the model as fire-type specific differences in the combustion losses from the affected C pools, and fire-type specific differences in the re-distribution of C among the C pools (Figure 1; Table S2).

To fully describe the impacts of fire on ecosystem C balance two pyrogenic C pools are included in the model, that are created during fire events as pyrogenic residues from the combustion of organic matter. The 'char and soot' pool includes the pyrogenic residues that remain visible, i.e., above-ground, for periods up to weeks or years (C_{char}). The 'Black C' pool represents the accumulation of some of the char and soot into the soil matrix, potentially in more stabilised forms (C_{black_C}), Supplementary 1.2.

The decomposition rate of PyC in the absence of fire was also estimated as described for other pools (Table S2). The parameters specifying the loss and redistribution of C due to both

WF and PF were based on empirical data (Volkova and Weston 2013; Volkova *et al.* 2014; Volkova and Weston 2015; Jenkins *et al.* 2016; Krishnaraj *et al.* 2016).

2.4 Emission estimates

GHG emissions from fires were estimated following the IPCC guidelines for GHG inventories (IPCC 2006), see Supplementary 1.3. Emissions were estimated individually for fine and coarse fuels. Fine fuel loss (FL_f) was estimated as the sum of combustion losses from tree bark, tree foliage, grass leaves, shrub leaves and litter fuels. Coarse fuel loss (FL_c) was estimated as the sum of combustion losses from shrub stem, dead standing trees, coarse woody debris (CWD) and char, see Supplementary 1.3. Parameters for the emission estimates were extracted for the relevant Australian temperate forests: gas specific emission factors (EF) from Volkova *et al.* (2019a) and C and nitrogen fractions from Volkova and Weston (2015), (Supplementary, Table S2).

2.5 Predicted impact of changes in fire management practices on Net Ecosystem Carbon Balance (NECB) and total ecosystem carbon (TEC)

Total Ecosystem Carbon (TEC) was calculated as the sum of all above- and below-ground living and dead organic C pools, soil organic C, and PyC. NECB was calculated as the difference between the annual total input of C (NPP), and the sum of all annual C losses through decomposition, soil respiration, and combustion from fire (Figure 1). Therefore the NECB summarises the rate at which C is either being lost (NECB < 0) or is being sequestered (NECB > 0) in the forest, and where under steady state conditions NECB = 0.0. Prior to analysis the model was first checked to ensure the integrity of the C mass balance by running the model for 100000 years and confirming the long-term average NECB converged to a value of 0.0, with fire included and with fire excluded.

2.6 Model scenarios

Two scenarios were explored: 'steady-state' and 'dynamic'. The steady state analysis provides an overview of the long-term (or equilibrium) outcomes of the model (see explanation in the next section), and therefore by definition assumes NECB=0.0. In contrast the dynamic analysis explores annual changes in emissions, C storage and NECB over time, following an assumed change in fire management; the dynamic analysis is therefore non-equilibrium. For both scenarios, the mechanism for potential abatement of wildfire by prescribed fires within the area targeted for treatment was the primary focus of study. Although a landscape level abatement is not explicitly included in the modelling, the potential for this mechanism to contribute additional abatement in these forest types is considered briefly in the *Discussion*.

2.6.1 Steady-state scenarios

For steady-state analyses a fully factorial experiment was modelled, varying simultaneously the annual probability of wildfire, and the interval between prescribed fires. The potential for wildfire occurrence was included in the model as a random variable, but the actual modelled occurrence of wildfire depended on there being sufficient standing fuel loads (11 Mg ha^{-1}). The removal of standing fuel by prescribed fire below this threshold provides the mechanism within the model for fuel management to reduce the occurrence of high intensity crown WF, with the subsequent flow-on effects to C storage and GHG emissions.

Simulations were conducted at 17 different wildfire probabilities (P_{WF}), where P_{WF} = (0, 0.005, 0.01, 0.015, 0.02, 0.025, 0.03, 0.035, 0.04, 0.045, 0.05, 0.055, 0.06, 0.065, 0.07, 0.075, 0.08); and 30 different prescribed fire intervals (I_{PF}), where I_{PF} = ('No PF', 50, 48, 46, 44, 42, 40, 38, 36, 34, 32, 30, 28, 26, 24, 22, 20, 18, 16, 15, 14, 13, 12, 11, 10, 9, 8, 7, 6, 5). This gave 510 possible combinations of P_{WF} and I_{PF}, with the combination P_{WF} = 0.0 and I_{PF} = 'No

PF' corresponding to the hypothetical situation where all fires are excluded. The wildfire probabilities can be interpreted as a surrogate for changing climate with regard to fire regime, ranging from historical conditions (low P_{WF} = 0.01-0.03, corresponding to approximately 1.0 – 2.5 wildfires per 100 years), to a climate with more frequent extended dry spells (P_{WF} = 0.08, corresponding to approximately 5 wildfires per 100 years).

For each combination of P_{WF} and I_{PF} the model was run until steady-state TEC was achieved. Preliminary analyses indicated this required running each combination for 10 000 years, with the final 2 000 years retained for analysis. Because of the marked annual variability in model predictions arising from the stochastic WF events, the average annual emissions and C stocks across the final 2 000 years of each run were calculated to provide an indication of the long-term implications of each combination of P_{WF} and I_{PF}. A Monte Carlo approach was employed to calculate the average (and standard deviation) outcome of each combination across 2 500 replicate model runs.

2.6.2 Dynamic scenarios

The steady state analysis provides an overview of the long-term average expected outcomes across a broad parameter space of P_{WF} and I_{PF}. However, under the steady state analysis NECB is by definition 0.0, and thus changes in both emissions and C storage under non-equilibrium conditions, as might be observed in reality, are not captured by this analysis. To investigate the dynamic behavior of the model in greater detail two separate runs for each contrasting wildfire probability were selected: P_{WF} = 0.01 and P_{WF} = 0.08 (equating to approximately 1 and 5 wildfires per century respectively); with a reduction of prescribed fire return interval of 25 years (currently applied in southern Australia) down to 10 years. For each wildfire scenario a step change to the prescribed fire interval was made during the model run, and the subsequent dynamic responses investigated.

Analyses were run for 10 000 years, with a management change from I_{PF} = 25 years to I_{PF} = 10 years implemented at year 9 000. Such a long period (9 000 years) was selected to allow the model to stabilize, prior to application of the management change. Annual results were saved for 100 years either side of the management change. Average estimates of NECB and TEC were based on 250 000 Monte Carlo iterations. The greater number of Monte Carlo iterations in this analysis compared to the steady state analysis reflects a need to detect relatively small changes in NECB following management change, relative to both TEC and to the annual variability in model predictions.

3. Results

3.1 Steady State Scenarios

3.1.1 Total Ecosystem Carbon and Emissions in the absence of prescribed fire

In the absence of any fire (i.e., neither prescribed nor wildfire) TEC is predicted to be at the theoretical maximum of 438 Mg C ha^{-1}, with 41% of C being stored aboveground and 59% belowground (Table 1). As the probability of wildfires increased, the TEC decreased such that at P_{WF} = 0.08, TEC is reduced by 21% to 347.6 Mg C ha^{-1}. The reduction in TEC was due to reductions in the aboveground living (37%), aboveground dead (32%), belowground living and dead (17%) and soil (14%) C pools. Stocks of PyC increased with increased probability of wildfire from approximately 4 Mg C ha^{-1} at P_{WF} = 0.01 to above 10 Mg C ha^{-1} at P_{WF} = 0.08. Emissions from wildfires for the most frequent wildfire probability (P_{WF} = 0.08) averaged 1.72 Mg C ha^{-1} yr^{-1} over 100 years, or approximately 37 Mg C ha^{-1} per wildfire event (Table 1).

Table 1. Modelled estimates of emission and total ecosystem carbon and its components for wildfire probabilities ranging from 0 to 0.08 per 100 years, in the absence of prescribed fire.

Probability of wildfire (P_{WF})	Number of wildfires	Emission Mg C ha^{-1} yr^{-1}	Emission per wildfire, Mg C ha^{-1}	Carbon pools, Mg C ha^{-1}						
				AGC living	AGC dead	BGC	PyC		Soil	TEC
							Char	Black C		
				(a)	*(b)*	*(c)*	*(d)*	*(e)*	*(f)*	*(a+b+c+d+e+f)*
0	none	none	none	132.9±0	44.9±0	43.9±0	none	none	216.5±0	438.2±0
0.01	0.9±0.1	0.43±0.05	47.3±1.1	119.9±1.4	41.0±0.4	42.0±0.2	0.33±0.03	3.57±0.32	208.5±0.9	415.4±2.6
0.02	1.7±0.1	0.76±0.06	44.4±0.9	110.3±1.6	38.2±0.5	40.7±0.2	0.51±0.03	5.70±0.31	202.5±1.0	397.9±3.0
0.03	2.4±0.2	1.01±0.05	42.2±0.5	103.1±1.5	36.1±0.4	39.6±0.2	0.62±0.02	7.08±0.25	198.1±0.9	384.5±2.9
0.04	3.0±0.2	1.21±0.05	40.7±0.8	97.6±1.5	34.4±0.4	38.7±0.2	0.69±0.02	8.03±0.22	194.6±0.9	374.1±2.8
0.06	3.9±0.2	1.50±0.05	38.4±0.7	89.6±1.2	32.1±0.4	37.5±0.2	0.77±0.01	9.22±0.16	189.5±0.8	358.6±2.4
0.08	4.7±0.2	1.72±0.04	36.8±0.6	84.0±1.1	30.5±0.3	36.6±0.2	0.80±0.01	9.94±0.13	186.0±0.7	347.8±2.2

Values are the mean of 1,000 iterations; ± are the standard deviation of the mean. Where AGC living is the sum of tree stem, tree bark, tree leaf, grass leaf, shrub leaf, shrub stem; AGC dead is the sum of dead trees, CWD, litter, char; BGC is the sum of tree fine and coarse roots, grass roots, shrub roots, coarse and fine root fuels; PyC is the pyrogenic carbon, consisting of aboveground (Char) and belowground (Black C); Soil is the sum of HUM and IOM fractions, TEC is the total ecosystem carbon.

3.1.2 Impact of prescribed fire on wildfire frequency

Prescribed fire at short return intervals of 5 to 7 years was the most effective at reducing the frequency of high intensity wildfires for all wildfire probability scenarios, with all wildfire predicted to be eliminated at prescribed fire intervals of 5 and 6 years. Increasing the intervals between prescribed fires from 7 to 15 years doubled the expected number of wildfires at lower P_{WF} and tripled the expected number at higher P_{WF} (Figure 2).

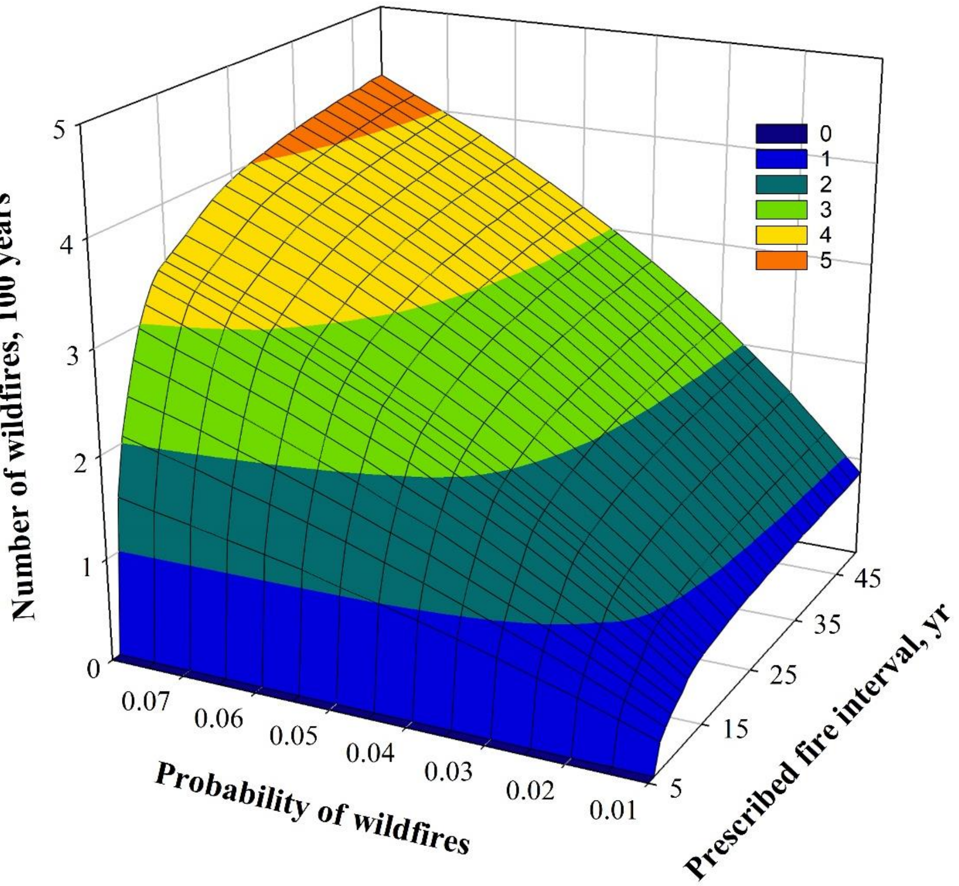
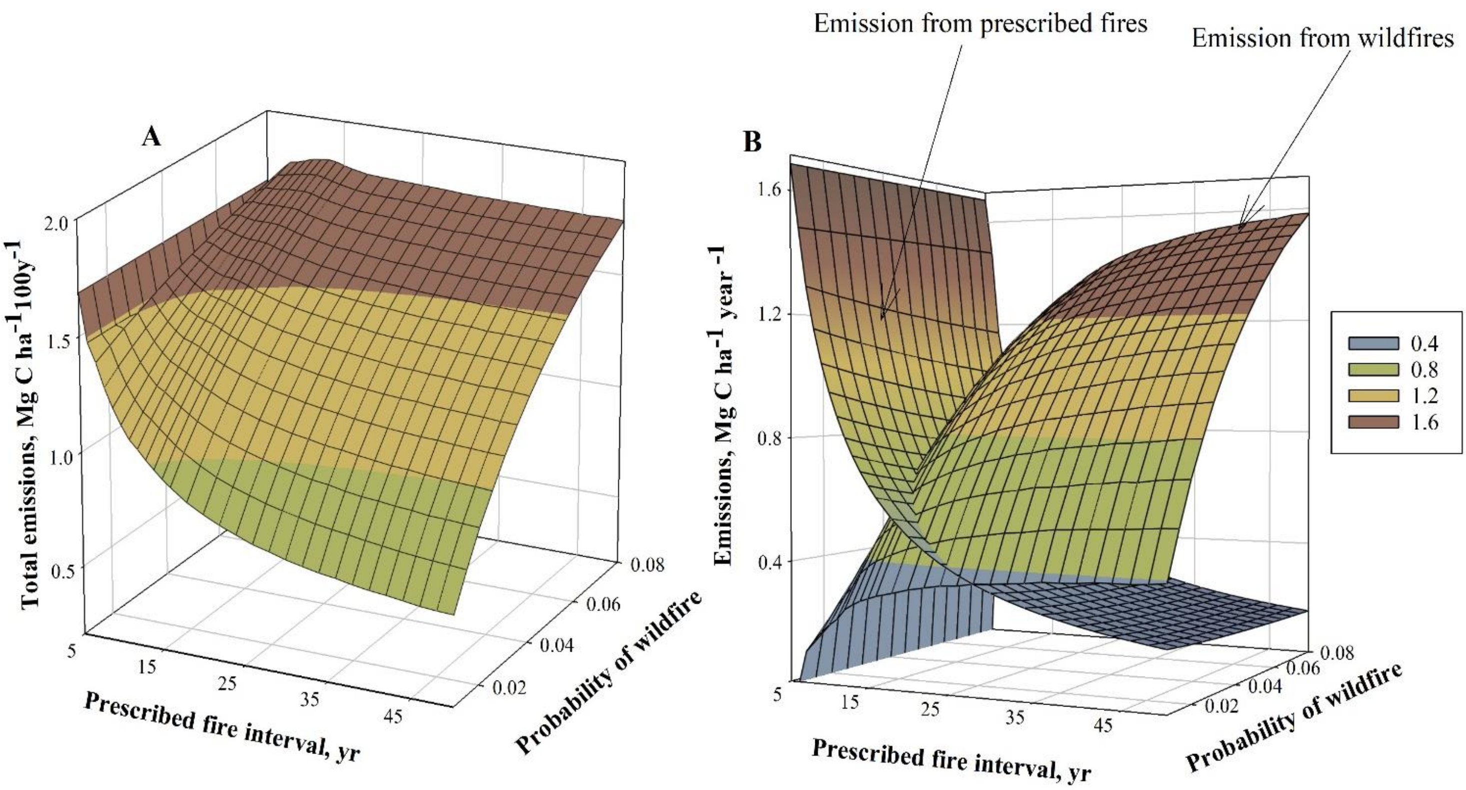


Figure 2. Number of wildfires at ranging prescribed fire intervals (I_{PF}) and wildfire probabilities (P_{WF}).

3.1.3 Impact of prescribed fire on emissions

Total combined emissions from both prescribed and wild-fires increased relative to the no prescribed fire scenario for all P_{WF} (Figure 3A). Emissions were highest at short-prescribed fire intervals (7 years), ranging from 1.4 Mg C ha^{-1} yr^{-1} at P_W = 0.01 to 1.8 Mg C ha^{-1} yr^{-1} at P_{WF} = 0.08. Although total emissions generally increased with decreasing interval of prescribed fire for all wildfire probabilities, the relationship with P_{WF} was nonlinear, such that at P_{WF} = 0.08 total emissions remained steady at approximately 1.8 Mg C ha^{-1} yr^{-1}, irrespective of prescribed fire interval (Figure 3A), albeit with a slight increase between PF intervals of 7 and 15 years. Separating the prescribed and wildfire emissions shows the compensating nature of the emissions profiles between the two sources (Figure 3B), with increasing emissions from shorter prescribed fire intervals being partially offset by decreasing wildfire emissions through reductions in wildfire frequency.

The difference in emissions between model runs with (E_{+PF}) and without (E_{-PF}) prescribed fire summarises the net emissions cost/benefit of prescribed fire management (Figure 3C). For prescribed fire intervals less than 7 years high intensity wildfire is excluded through the continuous removal of available fuels (such as fine litter and grasses). At all prescribed fire intervals greater than or equal to 7 years the total emissions with prescribed fire were greater than the scenario where prescribed fire was not included, although the increase in total emissions was relatively minor, and never exceeding 1.0 Mg C ha^{-1} yr^{-1} on average. The only scenario indicating a net decline in emissions with the introduction of prescribed fire was with I_{PF} interval of 6 years for the highest wildfire probability of P_{WF} = 0.08 (Figure 3C, solid line).



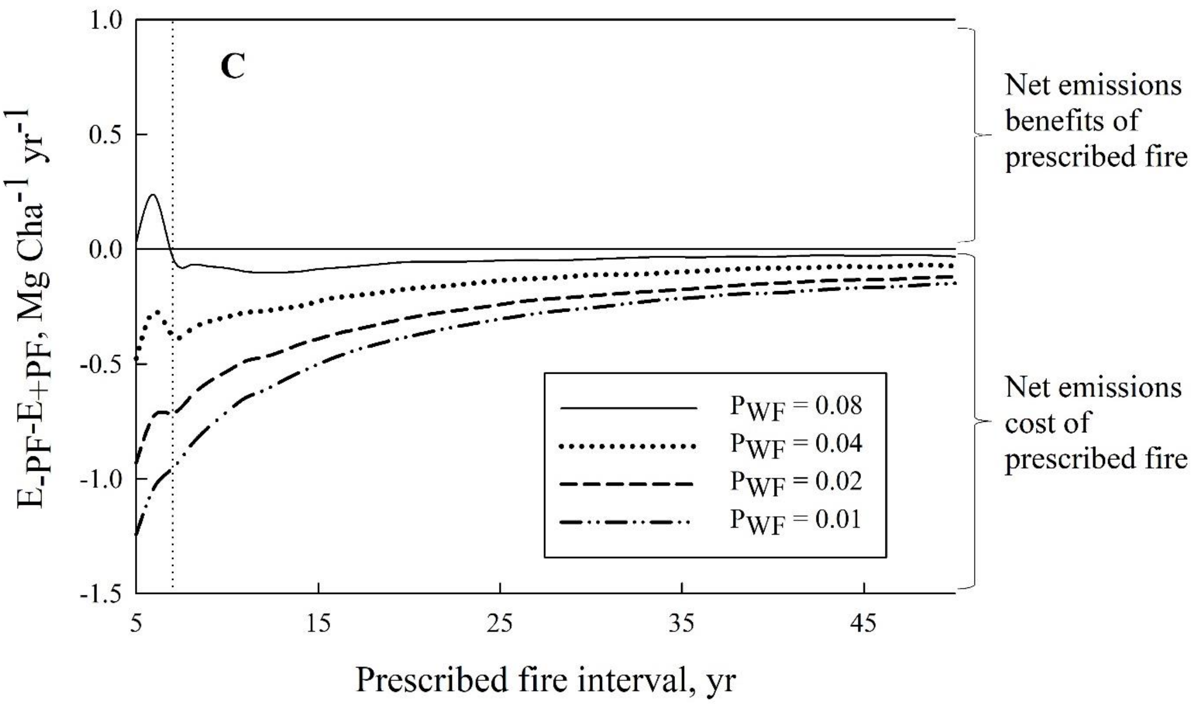


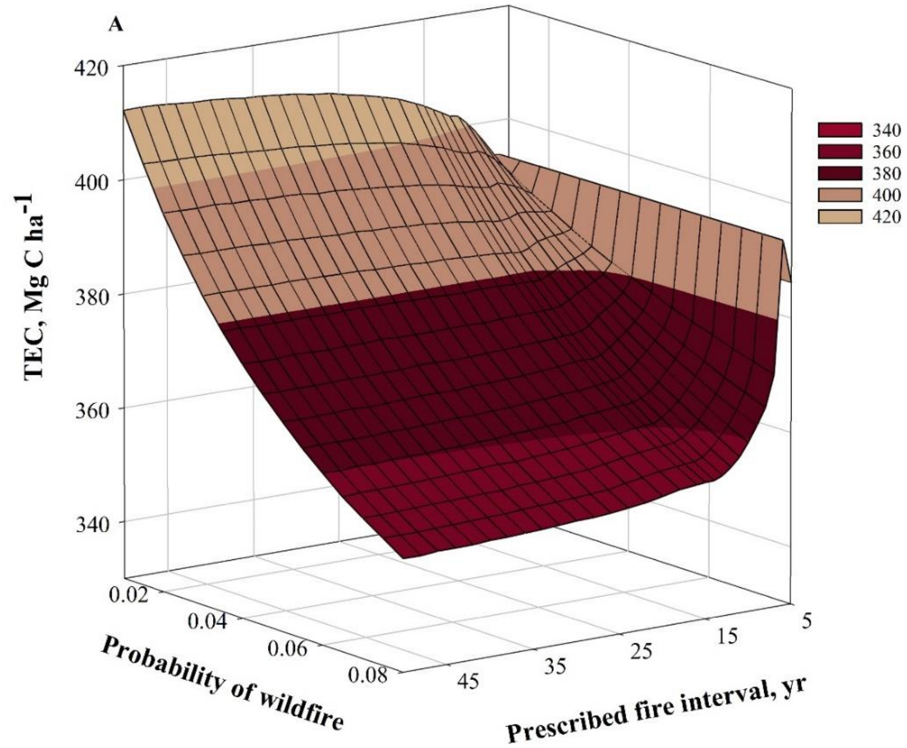
Figure 3 A) Total GHG emissions from wild- and prescribed- fires, and B) Emissions by fire type across a range of wildfire probabilities (P_{WF}) and prescribed fire intervals (I_{PF}). C) Change in long-term annual average fire emissions due to introduction of prescribed fire. E_{-PF} are emissions from wildfire without prescribed fire. E_{+PF} are combined emissions from both prescribed fire and wildfire. The dashed vertical line shows the limit below which all high intensity crown wildfires are excluded through the application of prescribed fire management.

3.1.4 Impact of prescribed fire on total ecosystem carbon (TEC)

Prescribed fire intervals less than or equal to 6 years reduce TEC to approximately 390 Mg C ha^{-1}, a reduction of approximately 50 Mg C ha^{-1} from the ‘no fire’ scenario. At this prescribed fire frequency litter cannot accumulate to the 80% of equilibrium threshold value that is recorded to be sufficient to carry a high intensity wildfire, therefore the TEC at all wildfire probability scenarios at these PF intervals are the same (Figure 4A). As prescribed fire interval increases, fuels accumulate to the point that allow high intensity wildfires to occur. For each probability of wildfire scenario, the outcome with decreasing prescribed fire interval varies. Under a regime of frequent wildfire (P_W = 0.08) a decrease in the prescribed fire interval led to an increase in TEC from 351 Mg C ha^{-1} at one prescribed fire every 50 years, to 373 Mg C ha^{-1} at one prescribed fire every 7 years. Approximately 60% of that increase was in the living biomass pool, with the majority of the remainder in PyC and SOC. In contrast, under a regime of infrequent wildfire (P_W = 0.01), decreasing the prescribed fire interval led to a decrease in TEC from 413 Mg C ha^{-1} at one prescribed fire every 50 years, to 396 Mg C ha^{-1} at one prescribed fire every 7 years (Figure 4A).

The main factor contributing to greater TEC at more frequent prescribed fire scenarios of 5-10 years at P_{WF} > 0.03 was via a reduced impact of high intensity wildfires on living biomass (above- and below-ground) and fuels (AGC dead and BGC, Figure 5). Those pools declined with increasing interval of prescribed fires stabilising at 25 years and beyond. More frequent but lower intensity prescribed fires stimulated production of PyC contributing to a spike in soil at about 8% of SOC at 5-yearly interval of prescribed fire. PyC declined to 2% of SOC if fires were infrequent, due to decomposition and lower input of PyC from fires (Figure 5, see Supplementary Figure S2 for absolute values).

The difference in TEC between model runs with prescribed fire (TEC_{+PF}) and in the absence of prescribed fire (TEC_{-PF}) summarises the net long-term average cost/benefit of prescribed fire management on ecosystem C (Figure 4B). Whether the introduction of prescribed fire leads to a net decrease or increase in TEC depends upon the probability of wildfire, with net C storage gains accruing with values of P_{WF} greater than approximately 0.03, and net C losses with values of P_{WF} less than approximately 0.03. Although the analysis indicated potential C benefit through introducing prescribed fire for some wildfire scenarios, this result represents a difference between two long-term average expectations. In practice, the timescale over which such changes might occur is of critical importance, as this will determine the magnitude of C gain (or loss) that might be observable over the yearly-to-decadal timeframes that are relevant to management decision making. This was explored via the dynamic scenarios.



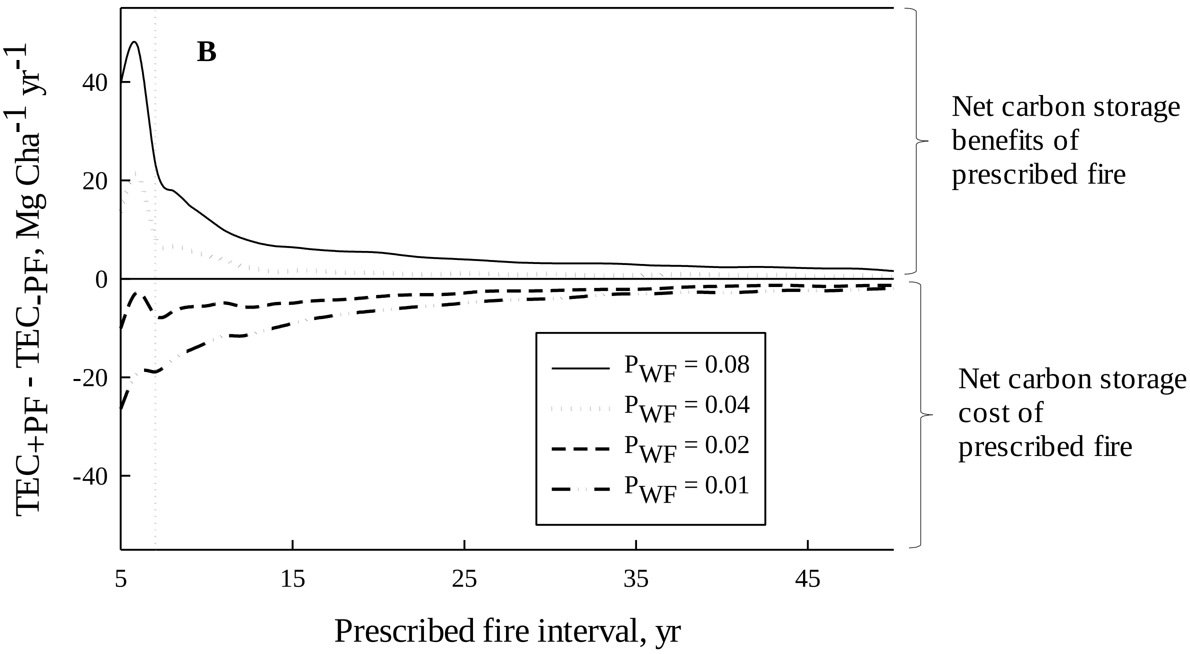


Figure 4. Results from a steady-state scenario showing long-term average in A) Total Eco-system Carbon (TEC) at a range of prescribed fire intervals (I_{PF}) and wildfire probabilities (P_{WF}). B) Change in TEC due to introduction of prescribed fire. TEC_{-PF} is the total ecosystem carbon without prescribed fire. TEC_{+PF} is the total ecosystem carbon with prescribed fire. The dashed vertical line shows the limit below which all wildfires are excluded through the application of prescribed fire management.

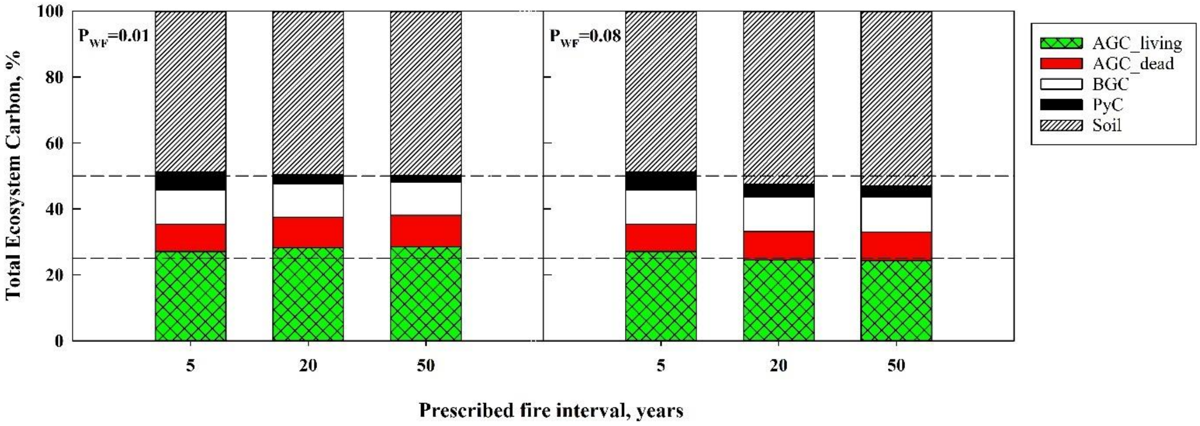


Figure 5. Contribution of individual C pools to the Total Ecosystem C (TEC) (expressed as a %) at low (P_{WF} =0.01) and high (P_{WF}=0.08) wildfire probabilities at a range of prescribed fire intervals. Dashed lines indicate 25% and 50%.

3.2. Dynamic scenarios

The results from the steady-state analysis showed decreasing the prescribed fire interval is expected to lead to a gain in ecosystem C storage when high intensity wildfires are frequent, and to a loss in C storage when wildfires are infrequent (Figure 4B). A critical question is, therefore, the timeframe over which such changes in ecosystem C would be expected to occur - this was investigated in the dynamic scenarios.

Over 100 years following the change in prescribed fire interval from 25 to 10 years, TEC decreased on average by approximately 10 Mg C ha^{-1} at P_{WF} = 0.01 and increased by 8 Mg C ha^{-1} at P_{WF} = 0.08 (Figure 6 A). The net annual change in the total C balance, summarised as NECB (which integrates both changes in C storage as well as the fire emissions), shows these changes are negligible, with average annual losses over the 100-year period of -0.09 Mg C ha^{-1} yr^{-1} for the P_{WF} = 0.01 scenario, and average annual sequestration of 0.04 Mg C ha^{-1} yr^{-1} for the P_{WF} = 0.08 scenario (Figure 6C). These annual changes correspond to less than 0.03% of the total ecosystem C stock, and would not feasibly be able to be measured, especially given they represent average outcomes over 250 000 replicate model runs. Indeed, examination of the individual model runs illustrates the temporal variability that might be observed in reality (Figure 6B), with each individual run representing a unique history of prescribed fire and wildfire. Such inter-annual variability would, in practice, mask any underlying expected trends in average C storage and/or emissions.

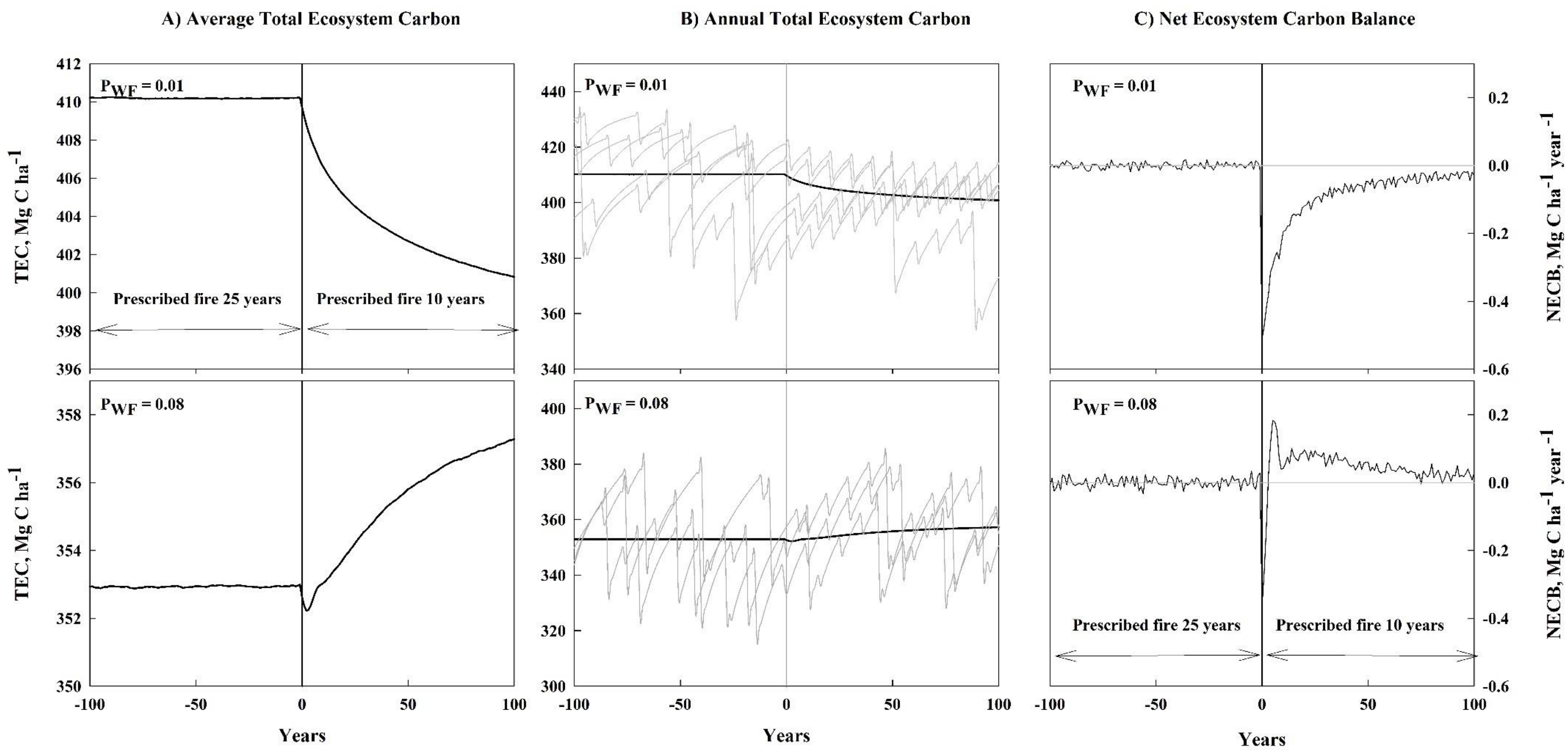


Figure 6. Results from the dynamic scenario, where the model is run to steady state with a prescribed fire interval of 25 years, with a change (at year 0) to a prescribed fire interval of 10 years. (a) Total Ecosystem Carbon (TEC) averaged over 250,000 replicate model runs. (b) Average TEC (black line) shown with five representative individual model runs (grey lines), to illustrate the between-replicate variability and temporal

variability in TEC. (c) Net Ecosystem Carbon Balance, averaged over 250,000 replicate model runs. NECB summarises the temporal patterns of net annual carbon loss (for P_{WF} *= 0.01), and carbon gain (for* P_{WF} *= 0.08), integrating both changes in carbon stock, and emissions.*

4. *Discussion*

In this study we present a full C balance ecosystem model that accounts for the effect of fires on above- and below-ground C pools and PyC, using a fire adapted ecosystem as a case study. In all but one case, the scenarios presented in this study indicated that prescribed fires in combination with high intensity crown wildfires release more emissions than in the absence of fire management, confirming previous findings for Australian temperate forests reported by Bradstock *et al.* (2012). Yet, a holistic approach in assessing the impact of fires on total ecosystem C balance revealed a more complex story.

Prescribed fires, applied at short return intervals, can lead to either an increase in TEC or to a reduction of TEC depending on the probability of occurrence of high intensity wildfire in the absence of prescribed fire. Prescribed fires applied on a 5- to 15-year cycle were successful at reducing the frequency of high intensity wildfires and also at increasing TEC for scenarios of two or more wildfires per 100 years, saving up to 10 Mg C ha^{-1} over the long term. Short prescribed fire intervals, of between five to ten years, have been reported as most effective at reducing wildfire hazard, fire severity and assisting fire suppression (McCarthy and Tolhurst 1998; Boer *et al.* 2009). Yet if occurrence of high intensity wildfire is less than two per 100 years, application of prescribed fires would both increase the emissions and decrease TEC, losing about 5 Mg C ha^{-1} over the long term compared to the no prescribed fire scenario. However, when the 100-year timescale over which these changes are expected to occur is considered, the annual average sequestration or loss is negligible, amounting to less than 0.03% of the total standing C store. Such small changes to the C balance would not be amenable to validation by direct measurement, particularly against the background of inter-annual variability in ecosystem C resulting from the combination of prescribed fire and wildfire.

We should also note that changes in weather expected under climate change may not only increase the occurrence of wildfire, but also potentially make it more difficult to conduct safe prescribed fires (Mitchell *et al.* 2014). We have selected a minimum prescribed fire return interval of 5 years; although this is shorter than most operational fire intervals, scenarios for 3 year return intervals have been investigated in these forests (Bennett *et al.* 2014). Repeated frequent prescribed fires applied before fuels fully re-accumulate can be possible for both ecosystem C and strategic asset protection.

One of the main factors contributing to the maintenance of TEC at more frequent prescribed fire scenarios was lower living biomass losses and short-term retention of coarse fuels such as CWD and dead trees, which not only benefit C storage but also provide habitat and protection for fauna. It has been estimated nearly 1 billion animals died in the 2020 Australian megafires, killing an estimated 30% of the entire koala population, listed as an endangered species (Lam *et al.* 2020). Studies confirm the lower presence of animal species in post-wildfire environments compared to unburnt areas several years after high severity fires due to the reduced availability of habitat components and a reduction in habitat structural components such as logs (Lawes *et al.* 2015; Chia *et al.* 2016). Application of frequent, low intensity burning has been shown to benefit wildlife diversity in many ecosystems of the US (Mitchell *et al.* 2014; Darracq *et al.* 2016). Yet repeated prescribed fires of short intervals in urban landscapes may conversely result in simplification of forest structure that benefit predators or exotic animals as well as reduce species diversity and promote invasion of exotic plants (Gill and Williams 1996). The implications of prescribed fire management on ecosystem values other than C, such as biodiversity, must be assessed with care.

While humus and inorganic fractions of SOC in our study increased with an increase in fire interval, similar to findings of Sawyer *et al.* (2018), fire-stimulated production of char contributed to an increase of PyC in soil (about 8% of SOC at 5 year intervals of prescribed fires) and therefore increased total SOC. Black C declined to 2% if fires were infrequent, due to decomposition losses and reduced input of Black C from fires, a pattern observed in chrono-sequence field studies (Alexis *et al.* 2012; Hobley *et al.* 2017). We applied conservative estimates for the production of PyC in fires, and our modelling estimates were within the values reported in the literature (see Bird *et al.* 2015). We note a major knowledge gap about the fate of PyC in fires, with only a handful of the studies reporting values for temperate Australian forests (Jenkins *et al.* 2016; Krishnaraj *et al.* 2016; Hobley *et al.* 2017).

The simulated wildfire emissions presented in this study equate to about a 25% to 27% loss of aboveground living and dead biomass for each high intensity wildfire event; this magnitude of emissions is consistent with emissions reported from field studies not drawn on to build and calibrate our model (see Adams 2013; Bennett *et al.* 2014). A spatial aspect unable to be included in the 'point-based' modelling presented here was the potential to achieve additional abatement via a landscape-scale mechanism, whereby prescribed fire treated land acts as a barrier, protecting adjacent untreated land that would otherwise have burnt in wildfire. A full analysis of this spatial mechanism would require embedding the current model within a spatial framework, inclusive of an appropriate fire spread algorithm (Sullivan 2009). However, based on existing data an approximate calculation can be made. It has been reported that the spatial 'leverage' afforded by prescribed fire in southeast Australia is in the order of 0.25 to 0.33 ha of land saved from wildfire for every ha of land treated by prescribed fire (e.g., Boer *et al.* 2009). Based on the results from Figure 6 (for $P_{WF} = 0.01$), with a prescribed fire interval of 25 years the TEC is approximately 410 Mg ha^{-1}, with a change to a prescribed fire interval of 10 years resulting in an expected loss of 8 Mg ha^{-1}. If through fire management an adjacent area is protected from wildfire, it would be expected to recover from the current 410 Mg ha^{-1} to 438 Mg ha^{-1} (the expected maximum C storage in the absence of fire, Table 1), a gain of 28 Mg ha^{-1}. Assuming a spatial leverage of 0.25, then for every 1 ha of forest treated, 0.25 ha is protected; and therefore in this example the landscape mitigation would be 28 Mg ha^{-1} x 0.25 = 7 Mg ha^{-1}. Even under optimal conditions, assuming maximum regrowth potential in the protected forest, adding the potential landscape abatement (7 Mg ha^{-1}) is still insufficient to overcome the *in-situ* loss of 8 Mg C ha^{-1}. Whilst this is a simplistic analysis, it does provide an initial indication of the landscape abatement potential, which could be refined with more detailed modelling.

Further areas of research should include embedding the current model within a spatial framework, additional experimental fires to fill the gaps in knowledge of C re-distribution and PyC accumulation, expanding modelling to other forest types and addressing the impact of fire frequency and severity on processes maintaining NPP including alteration of microbial communities regulating decomposition processes (Semenova-Nelsen *et al.* 2019; Hopkins *et al.* 2020). While our model was calibrated and validated with empirical data, changes in climate and fire regimes will lead to reduced accuracy of the model relative to the empirical data that can be expected from future fires. This could be addressed through replacing the simple empirical functions used here for growth (and fire behavior) with process-based sub-models that are sensitive to temporally varying climate data.

Wildfire frequency for Australian temperate forests is predicted to increase as the climate continues to change (Dutta *et al.* 2016; Sharples *et al.* 2016). Under these circumstances the ecosystem C model developed here suggests that prescribed fire is a land management option that has minimal impacts on the total forest C balance – whilst reducing the risk of wildfire.

The results of this study should inform fire science beyond temperate Australian forests. The general patterns predicted in this study confirm conclusions from many empirical and modelling studies, that increasing use of prescribed fire reduces the intensity and extent of wildfires. This conclusion applies more to ecosystems in Mediterranean climates where high intensity crown wildfires are much more severe than low intensity prescribed fires – as found in the North American chaparral and scrub ecosystems, than to more mesophytic ecosystems as found in central Europe and eastern United States (see Hunter and Robles 2020). Similar to other studies (e.g., Vilén and Fernandes 2011; Allen *et al.* 2013) we observed a decrease in total emissions only at high frequency of wildfires. The observed positive effect of prescribed fires on TEC is difficult to compare with other studies, as we considered all C pools, including PyC (Fig 5), while other studies differ in C pools usually included in the analyses (see Hunter and Robles 2020).

5. *Conclusion*

The full ecosystem C model presented here provides a basis to identify gaps in knowledge and address them with further field studies/empirical evidence to reduce uncertainty in predicting the impact of fires, including fire applied in forest management, on GHG emissions and total ecosystem C. The inclusion of both fire emissions as well as longer-term changes to the ecosystem carbon pools in the modelling suggested the net loss of ecosystem carbon arising from increasing the frequency of prescribed fire is approximately offset by lower emissions and avoided carbon losses from the subsequent reduction in wildfires. Although the results did not indicate potential to achieve emission abatement from the application of prescribed fire in temperate *Eucalypt* forests, nor did they indicate there was a significant GHG cost to manipulating prescribed fire intervals for other management purposes. These results suggest land management authorities should be free to develop suitable prescribed fire regimes, without significantly impacting net GHG emissions or total ecosystem C over the long term.

Authors contribution

LV, SHR, CJW conceived and designed the study including the model concept. LV has developed the model code with inputs and guidance from SHR. LV has written the first draft of the manuscript with the inputs from all authors. All authors participated in the model parameterisation, data analysis and writing of the manuscript.

Competing interests

Authors declare no competing interests.

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