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Fire regimes and carbon in Australian vegetation

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Abstract

Fires regularly affect many of the world's terrestrial ecosystems, and, as a result, fires mediate the exchange of greenhouse gases (GHG) between the land and the atmosphere at a global scale and affect the capacity of terrestrial ecosystems to store carbon (Bowman et al. 2009). Variations in fire -regimes can therefore potentially affect the global, regional and local carbon balance and, potentially, climate change itself (Bonan 2008). Here we examine how variation in fire regimes (Gill 1975; Bradstock et al. 2002) will potentially affect carbon in fire-prone Australian ecosystems via interactions with the stocks and transfers of carbon that are inherent to all terrestrial ecosystems. There are two key reasons why an appreciation of fire regimes is needed to comprehend the fate of terrestrial carbon. First, the status of terrestrial carbon over time will be a function of the balance between losses (emissions) from individual fires (of differing type, season and intensity), which occur as a result of immediate combustion as well as mortality and longerterm decomposition of dead biomass, and carbon that accumulates during regeneration in the intervals between fires. The length of the interval between fires will determine the amount of biomass that accumulates. Second, fire regimes influence the composition and structure of ecosystems and key processes such as plant mortality and recruitment. Hence, alternative trajectories of vegetation composition and structure that result from differing fire regimes will affect carbon dynamics. We explore these themes and summarise the dynamic aspects of carbon stocks and transfers in relation to fire, present conceptual models of carbon dynamics and fire regimes, and review how variation in fire regimes may affect overall storage potential as a function of fireinduced losses and post-fire uptake in two widespread Australian vegetation types. We then appraise future trends under global change and the likely potential for managing fire regimes for carbon 'benefits', especially with respect to emissions.

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Fire regimes and carbon in Australian vegetation

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Abstract

Fire regimes affect carbon stocks in terrestrial ecosystems as well as transfers of carbon within these ecosystems and to the atmosphere. We present conceptual models of the way that fire regimes affect losses and recovery of carbon and the potential for ecosystems to function as carbon sinks. We illustrate aspects of these processes in two important but contrasting biomes, the tropical savannas of northern Australia, and the temperate eucalypt forests of south-eastern Australia. Both ecosystems function as sinks most of the time. However, sink strength is sensitive to interactions between climatic variability and fire regime. For these two biomes, variation in fire regime is a significant driver of sink/source dynamics. Global change may potentially diminish the carbon storage capacity of these ecosystems. Development of a national perspective on the future fate of terrestrial carbon will require further insights into the nature of fire and vegetation change. The potential for managing fire regimes for carbon benefits (reduced emissions, enhanced storage) is greater in the savannas than in the temperate forests, because of different fuel dynamics, and a greater efficacy of prescribed burning in mitigating area burnt in the savannas. Assessment of future fire management for carbon benefits will depend on further elucidation of these fundamental ecological attributes, scope of potential co-benefits, the development of carbon trading markets and fulsome accounting protocols.

Introduction

Fires regularly affect many of the world's terrestrial ecosystems, and, as a result, fires mediate the exchange of greenhouse gases (GHG) between the land and the atmosphere at a global scale and affect the capacity of terrestrial ecosystems to store carbon (Bowman *et al.* 2009). Variations in fire regimes can therefore potentially affect the global, regional and local carbon balance and, potentially, climate change itself (Bonan 2008). Here we examine how variation in fire regimes (Gill 1975; Bradstock *et al.* 2002) will potentially affect carbon in fire-prone Australian ecosystems via interactions with the stocks and transfers of carbon that are inherent to all terrestrial ecosystems.

There are two key reasons why an appreciation of fire regimes is needed to comprehend the fate of terrestrial carbon. First, the status of terrestrial carbon over time will be a function of the balance between losses (emissions) from individual fires (of differing type, season and intensity), which occur as a result of immediate combustion as well as mortality and longer-term decomposition of dead biomass, and carbon that accumulates during regeneration in the intervals between fires. The length of the interval between fires will determine the amount of biomass that accumulates. Second, fire regimes influence the composition and structure of ecosystems and key processes such as plant mortality and recruitment. Hence alternative trajectories of vegetation composition and structure that result from differing fire regimes will affect carbon dynamics.

We explore these themes and summarise the dynamic aspects of carbon stocks and transfers in relation to fire, present conceptual models of carbon dynamics and fire regimes, review how variation in fire regimes may affect overall storage potential as a function of fire-induced losses and post-fire uptake in two widespread Australian vegetation types. We then appraise future trends under global change and the likely potential for managing fire regimes for carbon ‘benefits’, especially with respect to emissions.

Fire regimes and terrestrial carbon

Carbon transfers and stocks: consequences of fire regimes

There are three basic transfers of carbon between and within the terrestrial biosphere and the atmosphere that are mediated by fire. These are:

1. Transfer of carbon from the biosphere to the atmosphere. This may occur via immediate emissions from individual fires, or increases in rates of decomposition and heterotrophic respiration following fire.

2. Transfer of carbon between terrestrial pools. This includes transfers from living to dead biomass and charcoal; changes in plant population structure due to mortality and recruitment; soil organic matter transferred as dissolved organic carbon and particulates; soil erosion and transport into waterways.

3. Uptake of carbon from the atmosphere by post-fire regeneration of vegetation.

Emissions of greenhouse gases from biomass burning are significant; they were estimated to be ca. 3% of global GHG emissions in 2004 (Van der Werf *et al.* 2006). Terrestrial stocks of carbon consist of standing living and dead, above ground biomass of the over storey, understorey and ground layer; the litter layer; below ground biomass stocks; and soil organic carbon. In Australia, above ground carbon stocks may reach 800 t C ha^{-1} for tall, wet *Eucalyptus regnans* forests (Keith *et al.* 2009a). In other eucalypt forests in south-eastern Australia, estimates range from 150 – 800 t C ha^{-1} at sites investigated by Keith at al (2010), and 25-450 t C ha^{-1} at sites investigated by Grierson *et al.* (1992). In the jarrah (*E. marginata*) forests of southern Western Australia, estimates range from 80-200 t C ha^{-1} (Pekin *et al.*

2009). Comparative figures for the tropical savannas are up to ca. 80 t C ha^{-1} (Chen *et al.* 2003), and Cook *et al.* (2005) found a range from 12 to 58 t C ha^{-1} depending on the soil depth. Similar ranges were reported for Northern Territory savannas using the National Carbon Accounting Toolbox (Law and Garnett (2011).

Carbon storage in terrestrial ecosystems will depend on how fire regimes affect the balance between carbon uptake and loss. The intensity and frequency of fires will mediate the balance between losses and accessions of carbon. High intensity fires inherently result in greater combustion of biomass compared with fires of low intensity. Changes to post-fire micro-climates may be greater after high intensity fires due to high losses of cover (Hobbs and Atkins 1988), and charcoal may also increase with increasing fire intensity. Processes such as mortality and regeneration may vary according to fire intensity. For example, mortality of eucalypts in Australian forests is a positive function of fire intensity (Gill 1997; Williams *et al.* 2003; Vivian *et al.* 2008). High intensity fires may therefore create an initial pool of dead wood (boles and branches) and may predispose surviving trees to collapse through creation of basal injuries (Inions *et al.* 1989). Plant recruitment in many fire-prone systems is mediated by fire intensity. In sclerophyllous forests and shrublands in Australia, many plant species possess germination cues (e.g. heat, smoke and light) that result in a positive relationship between regeneration and fire intensity (Bradstock 2008). Losses of carbon via erosion and run-off will tend to be greater after high intensity fires (Doerr *et al.* 2006).

Length of the inter-fire interval will determine not only the amount of biomass that is potentially available to burn, but also the ongoing balance between uptake by photosynthesis and losses via herbivory, respiration and the resultant sequestration potential. These processes will in turn be affected by vegetation composition and structure. There is a nexus between length of inter-fire interval and community composition in many fire-prone vegetation types in Australia (Enright *et al.* this volume; Keith this volume) and elsewhere (Bond & van

Wilgen 1996; Pausas *et al.* 2004; Keeley *et al.* 2009) which can potentially affect overall productivity. For example in Australian sclerophyllous vegetation, obligate seeding trees and shrubs may decline in response to frequent fire (Enright *et al.*; Gill this volume). On the other hand replacement of a sclerophyllous or grassy understorey by 'mesic' woody species may occur if fire is absent for long periods of time (Lunt *et al.* this volume).

Spatial variation in fire regime components as well as underlying patterns of factors that affect plant productivity (e.g. terrain, soils and weather) will determine landscape and regional scale dynamics of carbon (Kashian 2006). An over-riding feature of the way in which fire affects carbon is the temporal asymmetry of losses and uptake. Fire can release vast quantities of GHGs to the atmosphere in hours, minutes and days. In contrast, ecosystem uptake of carbon during the intervals between fires (years, decades or centuries) is comparatively slow. Thus vegetation-fire-carbon interactions conform to a "slow in-rapid out" paradigm (Körner 2003).

The rates of these processes can influence perceptions of the role of fire in mediating carbon dynamics. Thus after large fires, for example, public discussion may focus on emissions with little regard given to uptake of carbon in the aftermath. To understand fire and carbon, due regard must be given to the time scales and processes over which vegetation dynamics and fire regimes operate.

Conceptual models of effects of fire regimes on carbon

In vegetation subject to recurrent fire, carbon stocks will follow a fluctuating trajectory characterised by episodes of short, sharp decline (e.g. combustion and other immediate post-fire losses) followed by attenuated recovery. Above-ground plant biomass pools initially grow rapidly following disturbance before reaching a quasi steady state in some forests and shrublands following fire (Specht and Specht 1999). This trend has been well documented for the litter pool in eucalypt forests due to their pivotal status as surface fuel (Raison *et al.* 1983), but other components of biomass have similar responses (e.g. Specht and Specht 1999). Such a pattern reflects changes in the relative contributions of growth, mortality and decomposition. This basic model (Fig. 1) can be used to explore the way that differing fire regimes may alter the dynamics of carbon and the size of stocks and emissions. There is limited evidence that soil carbon stocks may follow a similar trajectory (e.g. Ryan *et al.* 2008).

The carbon stock (C) will be determined by time since last fire in a simple accumulation model (Fig. 1). In Fig. 1a C_T is the carbon stock pre-fire. Fire causes rapid emissions (an amount C_e) and an immediate post-fire carbon stock of C_0 , i.e.

$$C_e = C_T - C_0 \qquad (1)$$

In this simple case, the loss of carbon during a fire (C_e), and further early post fire losses (up to time T_s) is exactly matched by subsequent gains as recovery of vegetation progresses up to time T_f, where carbon stocks again are equivalent to C_T . Hence T_f defines the 'carbon recovery time' - the time taken to recover losses of C from the previous fire event, which will vary as a function of the ecosystem type and fire severity.

Net Ecosystem Productivity (NEP, the rate of carbon fixed in the system due to Net Primary Productivity, NPP, minus carbon returned to the atmosphere due to heterotrophic respiration; Chapin *et al*. 2006) becomes positive at time T_s and remains strongly positive until time T_f (Fig. 1a). Average NEP in the period from the time of fire up to T_f ($\overline{NEP}_{T0\text{-}Tf}$) is sufficient to offset all fire-related losses, despite fluctuations at differing times after fire. Net Biome Productivity (NBP) is NEP minus carbon returned to the atmosphere due to disturbances such as fire (Chapin *et al.* 2006) averaged over the length of the fire cycle. NBP will be zero if:

$$\text{NBP} = (\overline{NEP}_{\text{T0-Tf}} * \text{T}_f) - \text{C}_e = 0 \qquad (2)$$

At differing times after fire, over the course of the fire cycle (e.g. at times $T_f + x$, or $T_f - x$).carbon stock may be higher or lower, as indicated by C_1 and C_2, respectively (Fig. 1a). In the first case (C_1) NBP will be positive because [$\overline{NEP}_{T0\text{-}Tf} * (T_f + x)$] > C_e.. In contrast, in the second case (C_2) NBP will be negative because [$\overline{NEP}_{T0\text{-}Tf} * (T_f - x)$] < C_e. Further fires of identical intensity and carbon loss (C_e) occurring at fixed intervals of T_f, $T_f + x$, or $T_f - x$, will respectively result in zero, positive (sink) or negative (source) trends in carbon storage over time, assuming that the post-fire trajectory of NEP ($\overline{NEP}_{T0\text{-}Tf}$) remains constant (i.e. all fires irrespective of length of inter-fire interval) over the course of the fire cycle.

The consequences of differing levels of consumption of biomass and carbon loss from fire are illustrated in Fig. 1b. Here, we assume that the post-fire trajectory of recovery is the same as in Fig. 1a; that is, our null hypothesis is that average NEP ($\overline{NEP}_{T0\text{-}Tf}$) is constant. Other trajectories are possible, for example if average NEP is lower following high intensity fires compared with low intensity fires.

If fire intensity and carbon loss are reduced (point C_0^- in Fig. 1b) NBP becomes positive if length of inter-fire interval (in this case T_f) remains constant. A shorter time (T_f - a) is required to recover fire-related carbon losses (i.e. C_T; NBP = zero). A lengthening of the interval between fire ($T_f + x$) will result in the ecosystem functioning as a net carbon sink (i.e. positive NBP), if fire intensity and carbon loss is lower than under the original scenario (i.e. C_3 in Fig. 1b > C_1 in Fig. 1a). By contrast, if fire intensity and carbon emissions increase, $\overline{NEP}_{T0\text{-}Tf}$ will be insufficient to replenish stored carbon even if the interval between fires is lengthened (i.e. post-fire stock is C_4; Fig 1b); this will result in the ecosystem becoming a net source of carbon (i.e. negative NBP; $C_4 < C_T$).

In contrast to Fig. 1b, the model in Fig. 1c illustrates an example where post-fire recovery of carbon is fundamentally altered (i.e. lower average NEP), reflecting changes to vegetation growth patterns. Thus the degree of reduction in maximum storage capacity and overall loss of carbon will be an inverse function of the length of the fire cycle (e.g. C_5 and C_6; Fig. 1c). Such a scenario could occur due to effects of an inherent change in the fire cycle on vegetation structure and composition or *ex situ* environmental effects such as a changing climate via, for example, its impact on productivity.

In reality fire intensity and interval are not fixed. Thus trends in NBP may vary through time as events vary and regimes fluctuate, according to changes in management, climatic drivers and corresponding responses of vegetation. Thus the overall outcome of a fire regime on carbon dynamics (average Net Biome Productivity, $\overline{NBP}$) will be reflected in the long-term balance between annual average gains ($\overline{NEP}$) and average annual losses from fires ($\overline{Ce}$) over multiple fire cycles:

$$\overline{NBP} = \overline{NEP} - \overline{Ce} \quad (3)$$

Negative values of $\overline{NBP}$ will result from a net loss (i.e. the vegetation acting as an ongoing source of carbon transfer to the atmosphere) whereas positive values will indicate an ongoing terrestrial sink. The latter case is illustrated in Fig. 1d, where varying fire intensities and intervals and post-fire trajectories of recovery result in a net gain of carbon.

The task of accounting for the way in which carbon responds to the gamut of processes inherent to a long term fire regime is daunting, but obviously pivotal to understanding the fate of terrestrial carbon under differing fire regimes. Diverse influences (e.g. management and climate change) have the potential to alter carbon through alteration of fire regime components and therefore vegetation responses Knowledge of these attributes is needed not only to understand the fate of carbon and biosphere/atmosphere interactions in flammable ecosystems, but also to clarify the extent to which we can manipulate fire regimes to increase the storage potential of terrestrial ecosystems and achieve a concurrent reduction in GHG emissions.

Fire regimes and carbon storage potential

The overall effect of variation in fire regimes on potential for carbon storage will reflect the balance between consumption of biomass during individual fires and resultant losses, and fixation of carbon in biomass and soils in the intervals between fires, over the course of the fire cycle (Fig. 1). Changes to fire regimes will potentially alter the rates of both consumption and fixation, and hence the long term storage potential. In the following section, we illustrate these dynamic processes via two case studies – the tropical savannas of northern Australia and the temperate eucalypt forests of SE Australia. For both biomes, there are detailed measures of consumption, in addition to long-term measures of NEP. This enables evaluation of long-term sink strength (NBP) under current fire regimes but also the consequences of changes to regimes. In particular, which of the models of alternative fire regimes (Fig. 1) is likely to apply?

Tropical savannas

Tropical savannas cover an area of ca. 2M km^2 in Australia and account for ca. 30% of Australia's terrestrial carbon stocks (Williams *et al.* 2004). They are subject to fire, primarily during the dry season, on an annual-decadal basis (Andersen *et al.* 2003; Fensham this volume), Fuel consumption and resultant carbon loss depend on the seasonal timing of fires. Beringer *et al.* (2007) estimated that 1.5 to 3 t C ha^{-1}were consumed and emitted by annual dry season fires in mesic savanna (*Eucalyptus miniata –E. tetrodonta* dominated). Cook *et al.* (2005) estimated rates of carbon transfers from the trees to be 1.6 to 2.9 t C ha^{-1} yr^{-1} in annually burnt savannas. In the interval between fires, Beringer *et al.* (2007) estimated that NEP was 3.5-5 t C ha^{-1} yr^{-1} over a five year period using measurements from eddy covariance flux towers.

Based on these measurements, Beringer *et al.* (2007) estimated NBP to be a net sink of ca. 2 t C ha^{-1} yr^{-1} in their mesic savanna site. Values for NBP of a similar magnitude (a sink of 1-2 t C ha^{-1} yr^{-1}) were reported by Chen *et al.* (2003) at the same site using a mass-balance approach, and by Williams *et al.* (2004) for this vegetation type across Arnhem Land**.** Murphy *et al.* (2010) in an assessment of 136 vegetation monitoring sites across different savanna vegetation types occurring along a substantial rainfall gradient, showed that frequent and/or intense fires may reduce increment growth in savanna trees, thus reducing NPP. However, across all landscape types and fire regimes represented by this set of monitoring plots, NBP of the tree stratum was effectively zero, as carbon gains in some sites were offset by carbon losses in others (Murphy *et al.* 2009). In a modelling exercise for the whole of the savanna biome Barrett (2011) found evidence of a very weak sink at a decadal time scale over the whole savanna biome within Australia, with annual NBP varying from sink to source depending on variation in annual rainfall.

Liedloff and Cook (2011) addressed this question of spatial and temporal variability in the various estimates of NBP in the mesic savannas, by testing the influence of long-term interactions between rainfall and fire regime on carbon dynamics. The FLAMES simulation model showed that these savannas were sinks in 60 to 85 % of years depending on fire frequency, with the greatest proportion of sink years occurring in simulations where the fire frequency was about one in ten years. The source strength, during the 15-40 % of years when the savannas were a source, ranged from about 1.5 t C ha^{-1} yr^{-1} for a fire frequency of one fire in ten years, through to less than 0.5 t C ha^{-1} yr^{-1} for annul fires. The modelled system was a very weak sink of less than 0.1 t C ha^{-1} yr^{-1} for most fire frequencies. During years when the modelled savanna was a sink, its sink strength was about 0.7 t C ha^{-1} yr^{-1} with a fire frequency of seven in ten years, becoming closer to zero with increasing or decreasing frequencies; i.e. there is an optimal fire regime for NBP. Richards *et al.* (2011) also found that for mesic savannas in northern Australia there was an optimal fire regime for soil carbon storage. This was a frequency of low intensity fires of one every five years, but it could take 100 years to achieve a detectable impact from changing the regime.

These examples show that the carbon source-sink relationships in savannas are sensitive to climate variability, and that the long-term storage potential is sensitive to variation in intensity/season and frequency of fire. Thus, estimates of NBP in the savannas depend on when in the fire cycle they are being measured. The carbon dynamics of savannas is consistent with the models illustrated in Fig. 1c. A reduction in fire intensity may result in a greater amount of stored carbon irrespective of fire frequency. Due to the relatively high frequency of current fires, decreases in frequency may result in an increase in stored carbon. Under dry conditions, carbon may be lost without a change in fire regime due to a reduction in NEP in the manner shown in Fig. 1c.

Temperate forests

Temperate forests cover several hundred thousand km^2 in southern Australia (Commonwealth of Australia 2007). These forests may be subject to fires of varying intensity at intervals from years to centuries depending on forest type (Gill this volume). Thus fire regimes in temperate forests differ from those in the savannas even though both ecosystems are largely dominated by eucalypts.

Carbon losses from fires are poorly studied in these forests. Nonetheless some estimates are available based on estimates derived for recent fires. An old growth Mountain Ash (*E. regnans*) forest site at Wallaby Creek, in the Central Highlands of Victoria was burnt by a high intensity fire in February 2009. The fire at this site consumed the entire leaf canopy and supporting branches and much of the understorey, though the dominant carbon pool (the mature tree stems) remained intact. Estimates of the emissions from this event (based on known allometry and measures of understorey standing crop) were approximately 50-70 t C ha^{-1} (Beringer, Hutley *et al.* unpublished data). Such an estimate represents the upper extreme likely in eucalypt forests as this particular forest type contains some of the highest carbon stocks in the world (Keith *et al.* 2009a) and is of limited extent.

The 2003 and 2006-07 fires in Victoria burnt approximately 2.5 million ha. Based on inventory figures for the Australian Greenhouse Gas Inventory (Emissions Inventory team 2010), total emissions from both fires were 28.3 Mt of carbon, or approximately 11.1 t C per ha. Most of the forests and woodlands burned in these fires are less productive than the wet Mountain Ash forests. Such estimates will therefore typify the range that can be expected across temperate eucalypt forests in general and accord with other estimates of fuel consumption based on process-based models (Bradstock *et al.* in review).

Estimates of NEP in unburnt conditions are so far restricted to two sites in moist forests. At a high altitude site near Tumbarumba in southern New South Wales a sink of 4.5 to 6.7 t C $ha^{-1}yr^{-1}$ was measured under average rainfall using flux tower and biometric techniques (Leuning *et al.* 2008, Keith *et al.* 2009b, van Gorsel *et al.* 2009). NEP declined during drought, ranging from a net source of 0.1 t C $ha^{-1}yr^{-1}$ to a sink of 2.4 t C $ha^{-1}yr^{-1}$ (Fig 2). The reduction in sink potential was driven primarily by a decline in biomass increment as a consequence of drought and insect damage. The forest was a sink for 5 out of 6 years, which included periods of average and below-average rainfall.

NEP has also been estimated using eddy covariance methods in the Mountain Ash forest described above. Despite below-average rainfall, the site was a sink of approximately 3.7 t C ha^{-1} y^{-1} (Kilinc *et al.* 2010). The understorey contributed ~20% to NEP. This estimate was confirmed by Sillett *et al.* (2010) who measured bole and branch increment.

These data can be used to estimate carbon recovery time (T_f in Fig. 1a). Assuming NEP is 3 t C ha^{-1} y^{-1}, then about 4 years is needed to recover the losses estimated for the 2003 and 2006/7 Victorian fires. This is well within the range of return intervals of unplanned fires in temperate open eucalypt forests in south-eastern Australia (20-25 years; Bradstock 2008; Price and Bradstock 2011). For the special case of Mountain Ash forests, about 20 years is required to recover losses from high intensity fire (i.e. assuming losses of 70 t C ha^{-1} and NEP of 3.7 t C ha^{-1} y^{-1}; see above). This is also well below both the usual return intervals of fires in tall open ash forests in SE Australia (50-200 years; Mackey *et al.* 2002). These estimates suggest that over the course of the fire cycle, losses from unplanned, high intensity fires can be recovered in the intervals between fires, and that under current fire regimes, over the course of the fire cycle, temperate Australian eucalypt forests act as net sinks. A broadly similar conclusion was reached for sub-alpine, conifer forests of the Rocky Mountains, USA (Ryan *et al.* 2008).

In both these examples, there may be considerable spatial heterogeneity in fuel consumption during major fires. Moreover, the average values of NEP used above to derive estimates of carbon recovery time may not take into account the situation soon after fire, when NEP may be considerably lower (see Fig. 1). Other perturbations to NEP through droughts or wet periods may alter these averages further. More work is required to measure the full trajectory of NEP in these forests, particularly in the critical early post-fire period.

Not withstanding these variations, it appears that the current average fire cycle in eucalypt forests is sufficiently long to accommodate large variations in fire intensity and NEP, which may potentially increase the estimates of carbon recovery time proposed above. Therefore these forests may behave in the manner outlined in Fig. 1a, where the average inter-fire interval is in excess of the carbon recovery time. Despite occasional high intensity fires, such forests therefore appear to function as robust sinks under contemporary fire regimes. Other factors, including management, have the potential to alter the sink strength and storage capacity of these forests. In particular, changes to NEP under a drying climate may diminish the capacity of these forests to recover carbon after fire, as in Fig. 1c, thereby lengthening the carbon recovery time. Less is known about the potential for changes in vegetation structure and composition to alter sink potential as a result of differing fire regimes.

Consequences of global change

Over coming decades, climate change and other factors (e.g. human populations, invasive species) will affect fire regimes and, by implication, a wide range of land management values including ecosystem carbon. How will changes to fire regimes and vegetation structure and composition affect the status of terrestrial carbon in Australian ecosystems? What are the wider implications of such changes? Can they be practically mitigated?

Changes to fire regimes in Australia are likely to vary in direction and magnitude among ecosystems in response to climate change (Williams *et al.* 2009b; Bradstock 2010; Cary *et al.* this volume). The fate of carbon in Australian ecosystems will be a function of the interactive effects of these changes to moisture and fire regimes. As described, a diminution of NEP due to reduced moisture has the potential to reduce the rate at which carbon is recovered following fire. Such a change, in concert with changes to fire regimes and other factors that influence vegetation composition, will affect carbon dynamics.

The evidence presented in the case studies of tropical savanna and temperate forests suggests that these ecosystems function as sinks of varying magnitude most of the time under current fire regimes. In savannas the situation is delicately poised because current fire regimes are just sufficient for carbon recovery (Fig. 1a, b). Thus a decline in NEP or an increase in the proportion of more severe late dry season fires is likely to reduce the sequestration potential of savannas and may be sufficient to shift NBP in these ecosystems from a weak sink to a source. Invasion by exotic grasses (Setterfield *et al.* 2010) may reinforce this trend by elevating fire intensity and carbon losses as well as promoting structural change through thinning of the tree canopy. The interaction between cyclones and fire may also be pivotal. For example, substantial carbon losses over the long-term are likely as a consequence of the extensive Cyclone Monica, which caused substantial tree death over 7000 km^2 of Arnhem Land (Cook and Goyens 2009); these reserves of carbon may be at risk from the annual-biennial fires which affect the savannas. Possible increases in the intensity of future cyclones under climate change (Knutson *et al.* 2010) may exacerbate such losses. Given the magnitude of tropical savannas in Australia, such changes would be significant for national carbon accounts.

Temperate eucalypt forests are predicted to be strongly affected by warmer and drier conditions in late 21st century with fires becoming more frequent and intense (Williams *et al.* 2009; Cary *et al.* this volume). Such effects will tend to increase losses of carbon and, in concert with reduced NEP, resulting in a diminution of stored carbon. In contrast to savannas, the current fire cycle in temperate forests is well in excess of the minimum that is needed to ensure carbon recovery (see above). Thus increases in fire-related losses of carbon and reduced uptake between fires may reduce NBP but may not be sufficient to alter the status of these forests as net sinks of terrestrial carbon over multiple fire cycles. Support for this hypothesis is provided by King *et al.* (in press) who used process-based models of fire regimes, climate change and carbon dynamics in temperate eucalypt forests in SE Australia. Under projections of 2070 climate, modelled increases in area burned and fire intensity reduced the carbon store , but did not alter emissions, uptake (and hence NBP) to a level where these forests became a net carbon source (King *et al.* in press).That is NBP was potentially reduced, but remained positive (i.e between CT and C1 in Fig 1a; between Ct and C3 in Fig 1b).   Australian forests may remain as net sinks in the future, but with diminished capacity.

The carbon sequestration capacity of Australian forests under a regime of recurrent fire is partly a function of the strong resprouting capability of the overstorey eucalypts (and other closely related genera) and recovery from fire does not involve complete regrowth of burnt tree stems. In particular, resprouting from above-ground epicormic buds enables stems of many tree species to survive fires of wide ranging intensity (Gill 1997). A notable exception, are the "ash-type" forests found in cool, moist environments of SE Australia, which are dominated by obligate seeder tree species (e.g. *E. regnans*, *E.delegatensis*; Mackey *et al.* 2002; Gill this volume). These forests have exceptional carbon storage capacity (see above), but may be vulnerable to an increase in fire frequency due to their lengthy juvenile period (15 to 20 years). While the carbon storage capacity of these forests could be diminished by the loss of these dominant species, eventual replacement by resprouting eucalypt species may moderate long-term losses, but these species would have a lower carbon carrying capacity.

Such buffering against future climate change effects may be more limited in other ecosystems dominated by non-sprouting woody species (e.g. Mediterranean climate heaths and shrublands; arid *Acacia* dominated woodlands; Enright et al this volume; Nano *et al.* this volume). For example, thinning, contraction or elimination of *Acacia* stands in arid woodlands through changes to fire regimes may result in replacement by grasses and therefore loss of a significant pool of above-ground carbon. This may be nationally significant, given their vast extent. There are large uncertainties involved in predicting the future of fire in the dry interior of the continent. Fire activity may diminish in response to climate change over significant parts of arid and semi-arid lands (Bradstock 2010; Cary *et al.* this volume). This could lessen the likelihood of loss of carbon via altered fire regimes, though other effects of climate change on key processes (e.g. plant recruitment and growth) may diminish productivity. As with savannas, invasive grasses may play a key role in determining fire regimes, vegetation change and the carbon status of the arid interior (Nano *et al.* this volume)

These examples indicate that future climates and fire regimes have the potential to shift significant quantities of carbon from current terrestrial storages to the atmosphere. While it is possible that some of the adverse changes to NEP, outlined above, may be offset by enhanced productivity of woody vegetation under elevated atmospheric carbon dioxide (Berry and Roderick 2006), fire has the potential to exacerbate changes to the atmosphere wrought by combustion of fossil fuels. Therefore, much of the immediate focus for management of landscape fires concerns mitigation of GHG emissions.

Transfers to the atmosphere: Emissions from vegetation fires

Emissions species

Fires produce a variety of emissions, the most common of which are carbon dioxide (CO_2) and water vapour, H_2O (Andreae and Merlet 2001; Cook and Meyer 2009; Russell-Smith *et al.* 2009 a,b). About 90% of the carbon emissions are released as CO_2 and CO, with the remaining emitted as methane (CH_4), non methane hydocarbons, oxygenated volatile organic carbons (OVOCs), and particulate matter. Other important GHG species released are nitrous oxide (N_2O) and other oxides of nitrogen. The main emission species that contribute to global warming are CO_2, CH_4 and N_2O; all differ in their longevity in the atmosphere and their global warming potential. Relative to CO_2 , and over a 100 year period, the global warming potential of CH_4 is 21 and N_2O is 310 (Cook and Meyer 2009). Flaming combustion produces relatively more CO_2 and less CH_4 than smouldering combustion. In contrast, N_2O is of intermediate oxidation state and its production appears to vary with flame temperature rather than oxygen supply. Further work needs to be done to understand variation in production of the nitrogenous gases (Cook and Meyer 2009).

Global and national emissions

Emissions from landscape fires were estimated to be ca. 3% of global GHG emissions in 2004 (Van der Werf *et al.* 2006). Australia contributes about 6-8% of the world's emissions from fire (Cook and Meyer 2009). On an annual basis, the majority of these emissions are from savanna fires in northern Australia (5-15 Mt CO_2-e per annum; Russell-Smith *et al.* 2009a). Such fires contribute about 3% of Australia's emissions, and may constitute 40% of the emissions of some jurisdictions such as the Northern Territory. Significant emissions also result from periodic large fires in the temperate forests of southern Australia (Bradstock and Williams 2009; Emissions Inventory Team 2010).

Under the Kyoto Protocol of 1997, carbon emissions from forests are only partially accounted for. Non-CO_2 species are counted – principally methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O) and other oxides of nitrogen (NO_x). CO_2 produced from vegetation fires is not taken into account (Cook and Meyer 2009). This convention reflects important assumptions about post-fire recovery of vegetation and the nature of fire regimes that warrant further scrutiny, particularly if accounting rules in international protocols are adjusted to accurately account for the potential impacts of altered disturbance regimes and land management practices (i.e. Fig 1b,c).

Managing fire regimes for carbon benefits

Global tree biomass in many of the world's forests and savannas is postulated to be below its climatic or "green world" optimum because of recurrent fires (Bond *et al.* 2005), indicating that variation in fire regimes is a critical determinant of carbon storage and emissions from terrestrial ecosystems. The models outlined (Fig. 1) indicate the potential for changes in fire regimes to alter the status of carbon in fire-prone ecosystems. The prospect of significant losses of carbon to the atmosphere via altered fire regimes under future global change heightens the challenge. Can such losses be reduced or negated? Can fire regimes be managed to abate or offset emissions derived from combustion of fossil fuels through reduced emissions and enhanced storage of carbon? Such beneficial outcomes will depend on interactions between biophysical and socio-economic factors and the degree to which fire regimes are sensitive to management measures.

Management of fire regimes to provide carbon 'benefits' of this kind is conceptually straightforward. Fuel reduction, via mechanical thinning or prescribed fire, results in a short-term transfer of carbon to the atmosphere, but can also decrease the intensity or extent of subsequent wildfires (Narayan *et al.* 2007; Hurteau *et al.* 2008), thus leading to lower carbon emissions over the long-term. For example, Wiedinmyer and Hurteau (2010) predict that CO_2 emissions from drier forests in the western United States could be reduced by as much as a 60% if prescribed fires are directly substituted for wildfires.

In the tropical savannas of the Northern Territory, an active fire management program – the West Arnhem Land Fire Abatement Program (WALFA) is using prescribed burning to limit the size and intensity of dry season fires over an area of 30,000 km^2, delivering an emissions savings of at least 0.1 Mt CO_2-e per annum (Russell-Smith *et al.* 2009b). This management program has its basis in the emissions accounting protocols outlined above. It is assumed that CO_2 produced by fires in tropical savannas during the dry season is fully re-absorbed by plant growth in the subsequent wet season (Cook and Meyer 2009; consistent with the concept of carbon recovery time, C_T, Fig. 1a). Nonetheless, a reduction in fire frequency reduces emissions of accountable greenhouse gases (CH_4 and N_2O) because savanna fires emit substantially more of these GHGs than occurs through natural decomposition pathways of the litter (Cook and Meyer 2009). The quantum of emissions is therefore a function of fire frequency, which is annual to decadal (see Fig. 12.3 in Cook and Meyer 2009). The season of fire, through its effects on fire intensity, also affects emissions. Lower emissions of accountable GHGs are produced from early dry season fires than late dry season fires. Fire management therefore potentially reduces emissions through reductions in area burned (hence reduced fire frequency) and intensity.

The prospect of managing fire to reduce emissions and enhance carbon storage more generally in ecosystems is therefore attractive as a measure to mitigate climate change. This prospect requires detailed assessment. The concept of using treatment of fuel (e.g. prescribed fire) in this manner involves an 'expenditure' of carbon in order to 'save' a greater amount through reduced wildfire emissions. However, as argued by Mitchell *et al.* (2009) and Bradstock and Williams (2009) long-term carbon benefits depend on both the ecosystem in question, and its current fire regime. Benefits cannot accrue if the cycle of fuel treatment reduces the mean total of carbon stored in the ecosystem by an amount that is greater than the amount that is saved. Mitchell *et al.* (2009) demonstrate that, for forests of the Pacific north west of the USA, the prospects for using prescribed burning to achieve carbon benefits depend very much on forest type - as determined primarily by bio-climate, with the potential to achieve a carbon benefit being higher in drier forests compared with wet forests.

The potential for prescribed burning to deliver emissions mitigation in differing ecosystems will depend on a combination of the 'prescribed burning efficacy', or 'leverage' (Fig. 3; see Loehle 2004) on the one hand, and the difference in intensity/fuel consumption between unplanned and planned fire on the other (Bradstock and Williams 2009). For the temperate eucalypt forest of SW and SE Australia, leverage appears to be ca. 0.25-0.33 (Boer *et al.* 2009; Price and Bradstock 2011). That is, to reduce area burnt by unplanned fire by 1 ha, 3 or 4 ha of prescribed fire is required. In contrast, in the savannas, the leverage is closer to unity (i.e. each hectare of prescribed fire tends to displace one hectare of unplanned fire; Cook and Meyer 2009, Gill *et al.* 2000; Price, Russell-Smith *et al.* unpublished data). If, as in savannas, the return on the prescribed burning investment is relatively high (i.e. a leverage of 1 or greater), then a net benefit (i.e. an emission reduction) will accrue via a reduction in fire intensity and fuel consumption (Russell-Smith *et al.* 2009b), even if area burnt remains the same. The benefit will increase further if average fire intensity of prescribed fires is less than that of unplanned fires. Given the documented reduction in intensity of unplanned fires caused by prescribed burning in northern Australia savannas, management for mitigation of carbon emissions and enhancement of storage potential is feasible (Russell-Smith et al 2009b; Cook and Meyer 2009; Bradstock and Williams 2009).

For leverage levels of 0.25-0.3 the prospects for a net emissions benefit of prescribed burning in temperate eucalypt forests are more constrained. Such leverage potentially results in a net increase in the overall area burned (i.e. additive effect of prescribed and unplanned fires). Thus a net ‘benefit’ will only accrue when there is a large difference in the intensity and fuel consumption of prescribed and unplanned fires. Prescribed burning reduces fire intensity in temperate forests for some years after treatment (Fernandes and Bothello 2003; Bradstock *et al.* 2010), but the magnitude and longevity of such effects may vary among landscapes. Differences in average intensity of planned and unplanned fires may not be large enough to yield a decisive carbon benefit in eucalypt forests given observed levels of leverage (Bradstock and Williams 2009). The notion that a carbon benefit may be more readily achieved by management where fires are frequent and low-or mixed severity (e.g. savannas) as opposed to systems with less frequent, high intensity fires (e.g. temperate forests) is consistent with the conclusion of Mitchell *et al.* (2009).

The mitigation model developed for savannas is a multiple co-benefits model: managing for emissions may also enhance storage potential (albeit weakly), deliver biodiversity benefits, and enhance the economies and livelihoods of local Aboriginal people (Williams *et al.* 2009a; Cook *et al.* this volume). Such management efficacy and synergy would need to be demonstrated for the WALFA-type model of emissions mitigation to be applied in other major biomes in Australia. This will be a fertile area of inter-disciplinary research and development for all of the fire-prone biomes of Australia.

Initiatives to manage fire regimes for carbon benefits must be judged in a global context. At present the world's forests appear to be a net carbon sink (Grace *et al.* 2006; Fahey *et al.* 2010), despite considerable emissions from recurrent fires. The terrestrial biosphere is a net sink for about 40% of 6.4 Gt C emitted globally each year, from the combustion of fossil fuels (IPCC 2006). The scale of emissions from vegetation fires is significant, nationally and globally (circa. 5 to 20% of anthropogenic GHGs – see above). Alteration of this quantum is important but does not constitute a significant abatement panacea for global GHG emissions (Arneth *et al.* 2010). Similarly, attempts to mitigate any future fire-driven trend toward loss of carbon currently stored in terrestrial pools, may be locally significant but will be dwarfed by the magnitude of the problem of fossil fuel emissions.

Despite this context, fire management for carbon benefits will remain an important prospect. A contribution can be made in some circumstances. The WALFA model, with its suite of co-benefits, rather than a singular reliance on mitigation of carbon emissions, indicates the way forward. The potential for generation of income through carbon-based trading schemes is tantalising for chronically under-resourced land managers. It is therefore imperative to understand where such a prospect can be fulfilled. While the global benefits may be relatively small, the local benefits are potentially large.

Conclusions

Carbon dynamics, as with other ecological phenomena and processes, must be understood in the context of fire regimes in Australian landscapes. Fires determine emissions while intervals between fires determine the magnitude of recovery via uptake during regeneration. The balance between these processes governs long-term storage potential. The vegetation of both the tropical savannas of northern Australia and the temperate forests of southern Australia appear to be net carbon sinks, under current fire regimes, but less is known about other ecosystems, particularly the vast arid interior. Sinks strength is sensitive to interactions between climatic variability and variation in fire regimes. In Australia, current knowledge suggests the best prospects for deriving carbon benefits from fire management come from mitigating emissions in the tropical savannas, where the return on management investment appears to be greater and more secure than that in temperate forests. In practice, the potential for regional and community benefits from managing fires for carbon benefits will be greatest where co-benefits (biodiversity, employment) accrue as well as carbon benefits. This is a fertile area for future research, as carbon accounting systems (e.g. Brack and Richards 2002) evolve towards 'full-carbon' accounting systems, which include CO_2 and disturbance

A more fulsome understanding of carbon, including CO_2 fluxes in relation to variation in fire regime, is needed if fire management effects on carbon stocks and fluxes are to be included in any future full carbon accounting schema. Management of fire regimes at landscape scales will invariably involve trade-offs between different landscape values. Hence knowledge of the interactions between vegetation dynamics, biodiversity, fire regimes, carbon stocks and carbon transfers across the landscape and the corresponding economics will become increasingly important in a carbon constrained world.

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Fig. 1 Models of post-fire change over time (T, x-axis) in stocks of terrestrial carbon (C; left hand Y-axis). The response curve represents changes in stocks as vegetation recovers following fire (initial carbon stock C_0 immediately after fire). The slope of the line at a point in time represents Net Ecosystem Productivity (NEP).

a) Response over the course of one fire cycle, where a single fire consumes vegetation and emits carbon, followed by a period of regeneration during which carbon is fixed. The carbon stock at the time of fire is CT. The quantum of emissions due to the fire (i.e. during combustion and smouldering) is C_e; emissions may continue to reduce carbon stocks post-fire until time Ts. The time to recover the carbon lost due to fire is T_f. At (T_f + x) the stock (C_1) exceeds the pre-fire stock; i.e. there has been a net gain in stored carbon relative to the stock at the time of the fire. As a result Net Biome Productivity (NBP; right hand Y-axis) over the course of the fire cycle is positive. At a shorter post-fire interval (T_f - x), storage (C_2) has not yet compensated for carbon losses, resulting in a net decrease in carbon relative to T0, or negative NBP. (Here we follow the convention that both NEP and NBP are expressed in terms of accumulation of carbon in the ecosystem, not the atmosphere; positive values represent accumulation in the ecosystem, negative values losses from the ecosystem).

b) Consequences of alternative levels of consumption and loss during fires (i.e. differing fire intensities). Differing levels of fire-related loss are represented by C_0 - (lower carbon loss by fire) and C_0 + (higher carbon loss by fire). Alternative carbon recovery time (C_T) under lower carbon loss scenario is indicated by T_f – a. A higher level of net carbon gain (C_3) after lower carbon loss in the fire occurs at T_f + x, compared with a). By contrast, a net decrease in stored carbon (C_4) occurs at the same time if fire-related losses are higher. Here we assume post-response (NEP) remains unchanged relative to a).

c) Effects of a reduction in the rate of post-fire recovery of carbon (i.e. reduced NEP; dashed line) compared with the scenario in a). At all time intervals after fire, there is a net decline in stored carbon occurs (e.g. C_5, C_6), relative to the original trajectory of recovery.

d) Change in carbon stocks over multiple fire cycles. Here, fire intensity and intervals between fires vary. In this case, post-fire trajectories of recovery relative to emissions result in a net gain of carbon, as indicated by the line, i.e. NBP is positive over the course of multiple fire cycles.

Fig 1a.

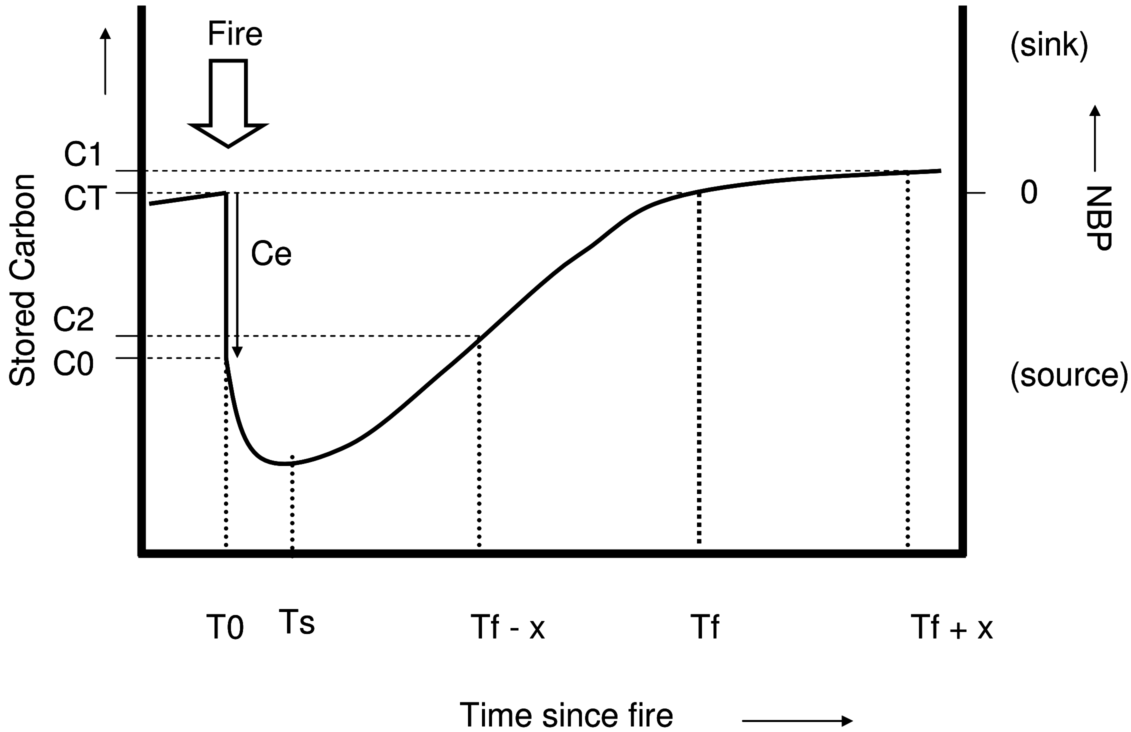
Fire
(sink)
C1
CT
Ce
C2
C0
Stored Carbon
0
NBP
(source)
T0
Ts
Tf - x
Tf
Tf + x
Time since fire

Fig. 1b

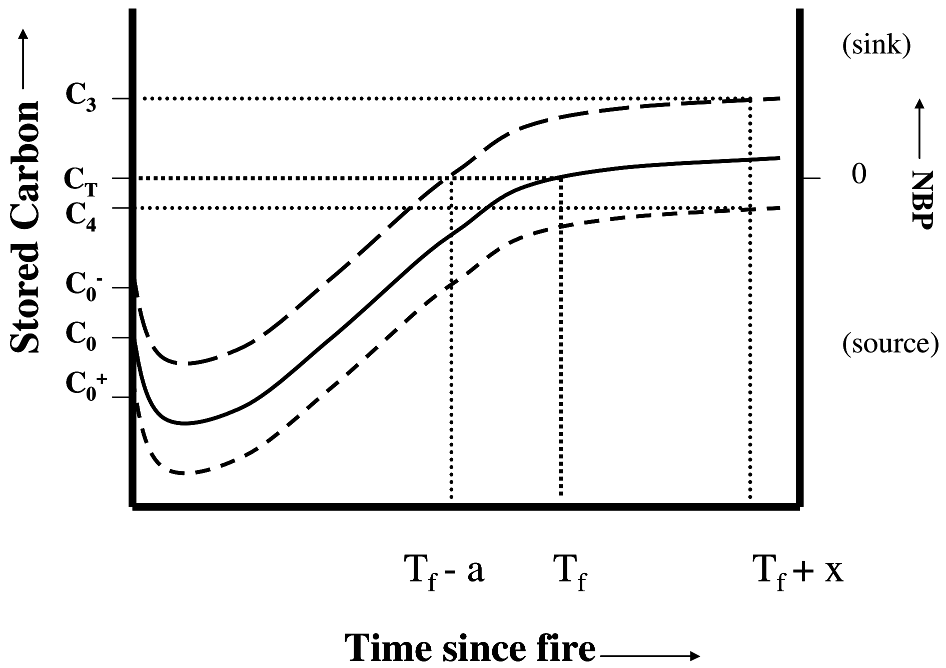
(sink)
C_3
C_T
C_4
C_0^-
C_0
C_0^+
Stored Carbon
0
NBP
(source)
T_f - a
T_f
T_f + x
Time since fire

Fig. 1c

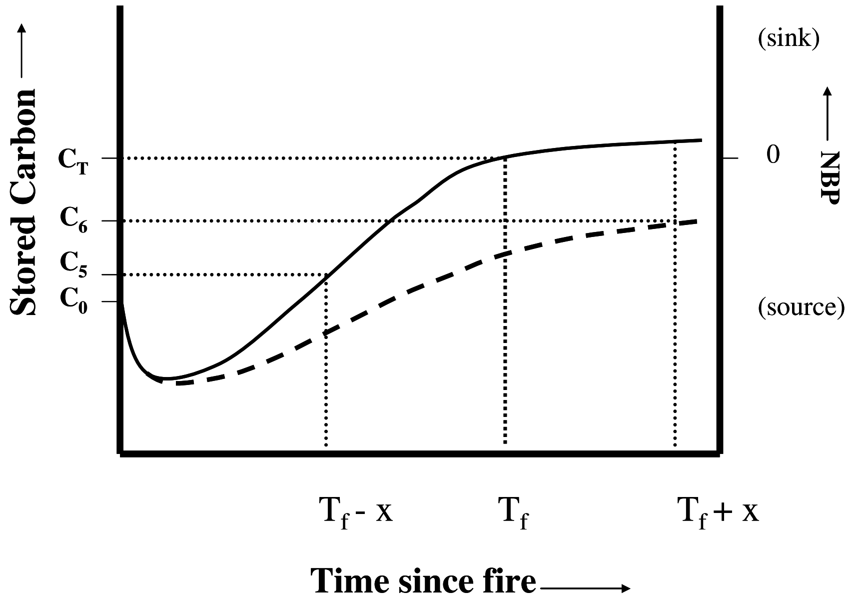
Stored Carbon
C_T
C_6
C_5
C_0
(sink)
0
NBP
(source)
T_f - x
T_f
T_f + x
Time since fire

Fig. 1d

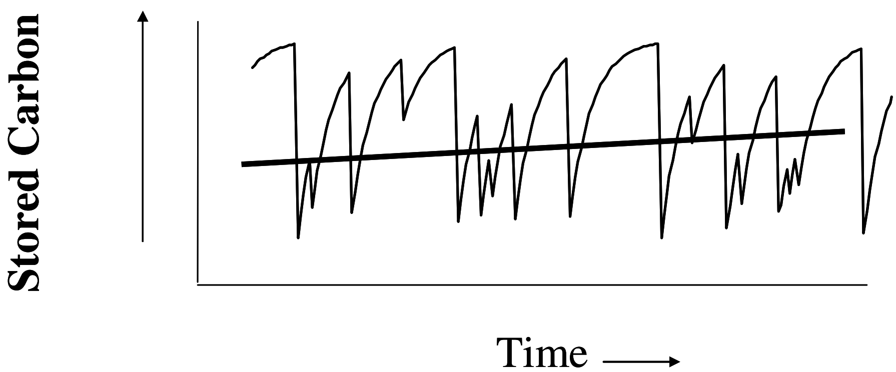
Stored Carbon
Time

Fig. 2: The inter-annual variability in NEP in a temperate eucalypt forest, Tumbarumba, SE Australia. Positive values indicate a net sink of CO_2; negative values a source of CO_2. Redrawn after Keith *et al.* (2009b).

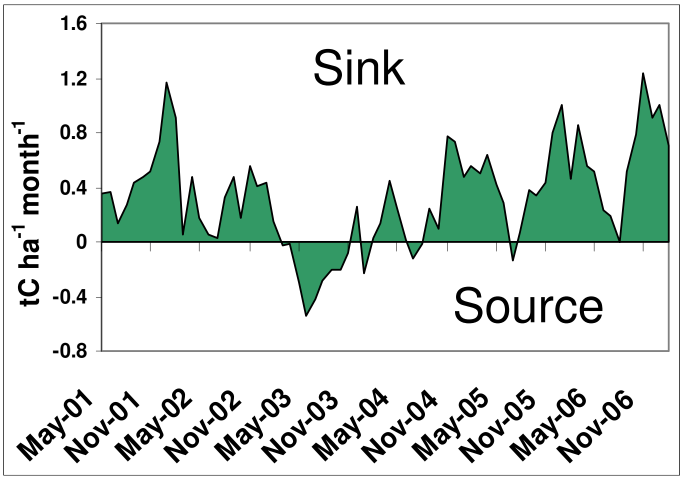

Fig. 3. Trends in the rate of unplanned fires across a landscape at a point in the landscape as a function of varying rate of prescribed burning (% area treated per annum), i.e. 'leverage'. Leverage values in the range from 0.17 to 1.0 are shown. Smaller values of L (i.e. a shallower slope) represent lower prescribed fire efficacy. For a leverage value of 0.25, mitigation of 1 unit of unplanned fire will require the application of 4 units of planned fire.