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Fire in Australian Savannas: from leaf to landscape

Running title: "Fire in Australian Savannas"

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ABSTRACT

Savanna ecosystems comprise 22% of the global terrestrial surface and 25% of Australia (almost 1.9 million km^2) and provide significant ecosystem services through carbon and water cycles and biodiversity. The current structure, composition and distribution of Australian savannas have co-evolved with fire, yet remain driven by the dynamics of the constraints of their bioclimatic niches. Fire in Australian savannas influences both the biophysical and biogeochemical processes at multiple scales from leaf to landscape. Here we present the latest estimates of Australian savanna biomass burning and how they contribute to global greenhouse gas budgets. We then review our understanding of the impacts of fire on ecosystem function and local surface water and heat balances, which in turn influence regional climate. We show how savanna fires are coupled to the global climate through the biogeochemical cycles and how altered fire regimes, resulting from changes in land management, invasive species or climate, can amplify or diminish the impacts of fire. We explore opportunities to reduce net anthropogenic greenhouse gas emissions through changes in savanna fire management. We suggest that climate change is likely to alter the structure and function of savannas through shifts in moisture availability and increases in atmospheric carbon dioxide (CO_2), in turn altering fire regimes with further feedbacks to climate.

INTRODUCTION

Tropical savanna ecosystems account for around 22% of the global land surface (Ramankutty & Foley, 1999). Annually, up to 75% of global tropical savanna landscapes are burned either by natural or anthropogenic fires (Hao *et al.*, 1990) and accordingly, 50% of the total biomass burned takes place in the savanna region (Hao & Liu, 1994). The wet-dry tropics of northern Australia include extensive areas of savanna vegetation, which occupy approximately 1.9 million km^2. This area accounts for 12% of the world's tropical savanna ecosystems, making this savanna biome of global significance. In this region, fire is arguably the greatest natural and anthropogenic environmental disturbance, with vast tracts burnt each year through lightning strikes and by pastoralists, aboriginal landholders and conservation managers (Russell-Smith *et al.*, 2003; Andersen *et al.*, 2005).

While these frequent savanna fires are extensive in area, they are of relatively low intensity when compared to the infrequent but intense fires of southern Australia (Williams *et al.*, 1998). Fire intensity is seasonal, with early dry season fires being of low intensity (< 1000 $kW\ m^{-1}$) which causes minimal canopy damage. As the dry season progresses, the fuel load accumulates and cures generating greater fire intensities. However, by the late dry season and pre-monsoonal period (August to October), fire intensity can be an order of magnitude greater (Williams *et al.*, 1998), with fires usually burning over very large fronts and tending to cause more damage (crown scorching over 90%). Such intense fires reduce foliage cover and blacken the soil (see typical example from Howard Springs - **Error! Reference source not found.**~~Figure 1~~).

The land surface is the interface for the exchange of radiation, heat, moisture, CO_2, and other trace gases with the atmosphere (**Error! Reference source not found.**~~Figure 2~~). Fire (Beringer *et al.*, 2003; Chambers *et al.*, 2005) and other disturbances (Hutley & Beringer, 2010; Hutley *et al.*, 2013) change the ecosystem characteristics such as structure, species composition and physiological function (Beringer *et al.*, 2011a). These changes result in altered **biophysics**, including energy partitioning (e.g. an enhanced sensible heat flux) and shifts in albedo (Beringer *et al.*, 2003; Jin & Roy, 2005). In addition, the aerodynamic properties of the ecosystem may change, affecting surface-atmosphere coupling. For example, a fire that causes a loss of canopy leaf area, with a subsequent reduction in canopy photosynthesis and evapotranspiration, greatly influences post-fire fluxes of water and carbon. Therefore, the influence of fire on ecosystem structure and function, and biophysical processes, have implications at a range of scales (**Error! Reference source not found.**~~Figure 2~~) (Bonan, 2008; Beringer *et al.*, 2011a).

Variability in ecosystem characteristics modifies surface–atmosphere exchanges, which in turn influence the overlying atmospheric boundary layer. At the local scale, enhanced sensible heat fluxes over patches of burnt landscape can induce and affect mesoscale circulation systems (Knowles, 1993). Variations in atmospheric heating rates above burnt and unburnt savanna generate horizontal pressure gradients which drive atmospheric motion at a range of scales. At the regional scale, savanna fires can have significant impacts on water, energy and CO_2 exchanges (e.g. Lynch & Wu, 2000) and as a result, are likely to have important feedbacks to the regional climate, including the atmosphere and hydrology. For example, spatial variability in ecosystem characteristics can generate contrasts that influence regional-scale climate systems such as the Australian monsoon (Lynch *et al.*, 2007). Previous work has focussed on the influence of land use and land cover change on these coupled dynamics (Evans *et al.*, 2011; Pielke *et al.*, 2011; Mahmood *et al.*, 2013). However, the extent and frequency of fires in Australia make this a crucial yet under represented research issue.

Ecosystems also interact with the earth system at local to global scales through **biogeochemical** cycling (C, N, P, etc.) (**Error! Reference source not found.**~~Figure 2~~). Ecosystems can be sinks or sources of CO_2 and other trace gases and can therefore enhance or diminish the overall greenhouse gas concentration in the atmosphere. Burning of savannas makes a positive contribution to global CO_2 concentrations through the emission of greenhouse gases, while the ensuing regrowth makes a negative contribution as CO_2 is assimilated from the atmosphere. The impact of fire on ecosystem productivity and non-CO_2 trace gases is not well understood (Beringer *et al.*, 2007), in particular the influence below-ground process such as termites and greenhouse gas emissions from soil. Furthermore, alterations in global greenhouse gas concentrations influences climate and the global circulation. Therefore, understanding the biogeochemical and biophysical processes of savanna ecosystems and the influence of fire (and other disturbances) is important for assessing interactions between climate, greenhouse gas budgets and water budgets from regional to global scales (Arneth *et al.*, 2010).

The objective of the paper is to review our understanding of the impacts of fire on biophysical and biogeochemical properties of Australian savannas at multiple scales, from leaf level physiology to regional climate. We use an earth system framework to elucidate the impact of fires in savannas on 1) emissions from biomass burning, 2) leaf to ecosystem carbon budgets, 3) long-term regional carbon budgets, 4) soil non-CO_2 greenhouse gas exchange, 4) energy and water cycles, 5) local climate and the atmospheric boundary layer, and 6) regional climate feedbacks. The focus of the paper is on biophysics and biogeochemistry rather than ecological drivers, that have already been documented in previous work, including ecological theory (Sankaran *et al.*, 2004), evolutionary ecology (Bowman *et al.*, 2010), phenology (Williams *et al.*, 1997), environmental drivers (Williams *et al.*, 1996), nutrient cycling (Holt & Coventry, 1990), plant demographics (Prior *et al.*, 2009; Midgley *et al.*, 2010) and fire (Williams *et al.*, 1999; Yates *et al.*, 2008; Murphy *et al.*, 2010). The spatial variability in savanna ecosystem characteristics has been previously documented by Hutley *et al.*, (2011) and a description of the spatial patterns and processes across the landscape is given in Beringer *et al.*, (2011a, 2011b). While acknowledging the scale of the Australian savanna ecosystems, we draw examples from the tropical savanna region of Northern Australia where we have sufficient information to assess many of the connections in an earth system framework.

BIOMASS BURNING EMISSIONS

Spatial and temporal patterns of fire emissions in north Australian savannas result from both strongly seasonal but annually reliable rainfall conditions which are conducive to frequent fires, and mostly anthropogenic ignitions. The frequency of fire occurrence across Australia is shown in **Error! Reference source not found.**~~Figure 3~~ and has been derived from 15 years (1997 - 2011) of AVHRR data (updated from (Maier & Russell-Smith, 2012)). Australian tropical savannas (outlined in red in **Error! Reference source not found.**~~Figure 3~~) experience a high occurrence of fires, with some regions exceeding one fire per annum. At the continental scale, fire frequency shows high correlation with total annual rainfall its seasonality, whilst at the regional scale savanna fires are more strongly influenced by anthropogenic ignition patterns (Russell-Smith *et al.*, 2007). When averaged annually, 18%

of Australia's 1.9 million km^2 of tropical savannas (**Error! Reference source not found.**~~Figure 3~~) were fire affected over the period 1997-2011. A total of 69% of fires occurred in the late dry season months (August-November) under relatively severe fire weather conditions (Russell-Smith *et al.*, 2013).

Globally, it is estimated that landscape and biomass fires cause CO_2 emissions of between 2 to 4 Pg C yr^{-1} (Bowman *et al.*, 2009). This is equivalent to around 20-40% of the 9.5±0.5 Pg C yr^{-1} emissions from fossil-fuel combustion in 2011 (Le Quéré *et al.*, 2012). Another study on global biomass burning by van der Werf *et al.* (2010) estimated CO_2-e emissions (including trace gases) between 2001–2009 at 2.5 Pg C yr^{-1} and that savannas contributed 44% (1.1 Pg C yr^{-1}) to global deforestation emissions. Using the Global Fire Emissions Database (GFED) v3.1 (Mu *et al.*, 2011) we calculate that Australian savanna fires contribute 10% (0.11 Pg C yr^{-1}) to this total (refer to Table 1) and account for 84% of Australia-wide CO_2-e emissions from biomass burning (**Error! Reference source not found.**~~Figure 4~~).

Despite these emissions the net annual CO_2 emissions from savanna fires are often regarded as CO_2 neutral on the assumption that wet (growing) season growth balances out emissions from the preceding burning season (Ciais *et al.*, 2011). However, such an assumption is not met where savanna carbon stocks are degraded under higher frequency and higher intensity fire regimes on decadal scales (Beringer *et al.*, 2007; Cook & Meyer, 2009). Moreover, savanna fires generate substantial emissions of the relatively long-lived greenhouse gases methane (CH_4) and nitrous oxide (N_2O) (Schulze *et al.*, 2009), which can ultimately react to produce tropospheric ozone (O_3), itself a significant global warming contributor (Finlayson-Pitts, 1997). However, using the GFED database we calculate CO_2-e emissions from non-CO_2 emissions as 0.55 Mt CO_2-e yr^{-1} (Table 1). In addition, savanna fires release black carbon aerosols, for example, Beringer *et al.* (1995), calculated that during 1992, savanna fires in the Northern Territory produced a large quantity (5.23 ± 0.37 x 10^9 g) of Total Particulate Matter less than 2.5 μm in diameter. These black carbon aerosols potentially have strong positive radiative forcing (Ramanathan and Carmichael, 2008) and may change the surface albedo of savanna areas thereby increasing solar energy absorption (Govaerts, 2002).

As required under the provisions of the Kyoto Protocol, Australia's National Greenhouse Gas Inventory (NGGI) applies a country-specific methodology for estimating accountable (i.e. CH_4 and N_2O) emissions from savanna burning. Applying that methodology (Department of Climate Change & Energy Efficiency, 2012), we estimate that mean non-CO_2 emissions from contemporary north Australian seasonal savanna fires were 9.2 Mt CO_2-e per annum between 1997 and 2011. Although the seasonal effects of CO_2 emissions on greenhouse budgets are poorly understood, Russell-Smith *et al.*, (2007) noted that, on average, CO_2 emissions from north Australian savanna fires contribute of the order of 200 Mt CO_2-e per annum, which is equivalent to a third of Australia's NGGI, but not accountable.

LEAF TO ECOSYSTEM CARBON BUDGETS

Leaf Carbon

Canopy performance is significantly altered following fire events. This is caused both by the reduction in functional leaf area due to senescence of scorched leaves, and by altered gas exchange characteristics of newly expanding leaves that flush to replace those killed by the fire. The new foliage that emerges in the weeks to months following a fire is not immediately photosynthetically competent (**Error! Reference source not found.**~~Figure 5~~). Thus, the overstorey trees must not only expend C resources in reconstructing new foliage, but additionally suffer an opportunity cost associated with the reduced net assimilation rate of new foliage during the reconstruction phase.

What might the C cost of reconstructing the canopy after a fire be? Using the following assumptions, an estimate can be produced: canopy scorch is 0.5 m^2 of foliage m^{-2} ground area, corresponding to about 80% of dry season canopy cover in the mesic savannas of northern Australia; specific leaf area is 5 m^2 kg^{-1}, typical for savanna eucalypts in northern Australia(Cernusak *et al.*, 2006, 2011); the C mass fraction of new foliage is 0.5; however, 1.25 g of C is required to produce 1 g of foliage C due to growth respiration costs. Under these assumptions, the C cost of replacing scorched foliage would be roughly 60 g C m^{-2} ground area, or about 5% of the annual gross primary productivity (GPP) of the overstorey.

The opportunity cost associated with reduced C assimilation during the reconstruction phase, when emerging foliage is not fully photosynthetically competent, is more difficult to quantify. It depends on the time courses of leaf expansion, the rates at which gains in photosynthetic capacity proceed as new leaves expand, and the water becoming available for transpiration. Negative to very low rates of net photosynthesis can persist in emerging eucalypt leaves until they have nearly fully expanded, although this will likely vary somewhat among species (Choinski Jr *et al.*, 2003). Beringer *et al.* (2007) suggested that the reduction in canopy photosynthesis could be similar in magnitude to the C cost of replacing burned foliage. Thus, the total cost to the C balance of overstorey savanna trees associated with dry season fires in northern Australian savannas could be on the order of 120 g C m^{-2} ground area, or about 10% of the annual GPP of the overstorey.

Expanding eucalypt leaves also tend to have lower water use efficiency than fully expanded leaves. This is caused by relatively high intercellular CO_2 concentrations in expanding leaves associated with low photosynthetic capacity and high respiration rates (Cernusak *et al.*, 2009). At the ecosystem scale, canopy transpiration has been observed to recover to pre-fire rates faster than canopy C assimilation (Beringer *et al.*, 2007), likely associated with the trajectory in intercellular CO_2 concentrations as expanding leaves develop.

Canopy carbon

Savannas represent a large fraction of the total tropical vegetation biomass and are highly responsive to their local environments. The rate of canopy carbon uptake at seasonal and shorter timescales is strongly controlled by local environmental drivers such as soil moisture or rainfall (Eamus *et al.*, 1999; Cook & Heerdegen, 2001; Kanniah *et al.*, 2013b), nutrient availability (Sankaran *et al.*, 2005), solar radiation (Kanniah *et al.*, 2010a, 2012, 2013a) and fire (Beringer *et al.*, 2007). Fire affects the radiative balance of the ecosystem immediately due to combustion of the grass-dominated understorey vegetation and blackening of the surface.

Low intensity fires (<1000 kW m^{-1}) at mesic savanna site (Howard Springs) caused minimal canopy damage with a low impact on the surface energy balance and only a slight increase in Bowen ratio. However, moderate fires (1000-5000 kW m^{-1}) resulted in complete canopy scorch and almost total defoliation in the weeks following (see Figure 1). Consequently, canopy transpiration was reduced and energy partitioning altered. Combined with reduced ecosystem transpiration, this resulted in a reduced carbon uptake at the site (**Error! Reference source not found.**~~Figure 6~~). After fire, the Bowen ratio was found to increase greatly due to large increases in sensible heat fluxes. These changes in surface energy exchange following fire, when applied at the landscape scale, may have important impacts on climate through local changes in circulation patterns and changes in regional heating, precipitation and monsoon circulation (Beringer *et al.*, 2003) (see later Sections).

Aerosols generated from savanna burning have been found to significantly affect the direct and diffuse components of solar radiation as well as its spectral composition (Eck, 2003; Kanniah *et al.*, 2010a), which could feedback to affect the canopy GPP of savannas (Kanniah *et al.*, 2012). For example, Kanniah *et al.* (2010b) found that smoke aerosols and humidity haze produced varying aerosol optical depths (0.1 to 0.4) which enhanced the fraction of diffuse radiation from 11 to 22% and resulted in a change in NEE.

Long term regional savanna NPP and NEP

In this section we assess components of the long-term Net Primary Productivity (NPP) and Net Ecosystem Production (NEP) in the regional savanna carbon budget, using a land surface model. We define NEP as GPP minus ecosystem respiration (Re), in the absence of disturbance. As described in Haverd *et al.* (2013b, 2013a), components of NPP and NEP were derived using the BIOS2 modelling environment, constrained by multiple observation types, and forced using remotely-sensed estimates of Leaf Area Index (LAI) and meteorology from the Bureau of Meteorology's Australian Water Availability Project data set (BoM AWAP) (Jones *et al.*, 2009). BIOS2 is a fine-spatial-resolution (0.05^o) offline modelling environment, including a modification of the Community Atmosphere Biosphere Land Exchange (CABLE) land surface scheme (Wang *et al.*, 2011) incorporating the Soil–Litter–Iso soil model (Haverd & Cuntz, 2010) and the Carnegie-Ames-Stanford Approach with Carbon-Nitrogen-Phosphorus (CASA-CNP) biogeochemical model (Wang *et al.*, 2010). This scheme is used in the Australian Community Climate and Earth System Simulator

(ACCESS). BIOS2 parameters are constrained and predictions are evaluated using multiple observation sets from across the Australian continent, including streamflow from 416 gauged catchments, eddy flux data (CO_2 and H_2O) from 12 OzFlux sites, litterfall data, and data on soil, litter and biomass carbon pools.

Error! Reference source not found.~~Figure 7~~ (i to iii) shows the annual time series of precipitation, NPP and NEP (1911-2011) for Australian savannas. The savanna region defined in the study Haverd *et al.* (2013a) is characterised by high interannual variability (IAV) in precipitation (640 ± 137 mm yr^{-1}, 1σ), which contributes to the high IAV in NPP (3.79 ± 0.62 t C ha^{-1} yr^{-1}, 1σ) and NEP (0.109 ± 0.36 t C ha^{-1} yr^{-1}, 1σ) and is consistent with previously understood drivers of savannas carbon fluxes exchanges (Kanniah *et al.*, 2010a). For comparison, the gross C-CO_2 emissions (t C) from biomass burning are equivalent to around 9% of the savanna NPP. **Error! Reference source not found.**~~Figure 7~~ (iv) shows the same time series, presented as 10-yr running means, converted to percentile rank. This reveals a strong correlation between precipitation and NPP at the decadal time-scale. The percentile rank time series of decadally-averaged NEP can deviate significantly from that of NPP (e.g. around 1984 and 2000), corresponding to periods of high heterotrophic respiration following long periods of biospheric carbon accumulation.

The average NEP trend for the 1990-2011 period for the Australian savanna biome was significantly positive (0.135 ± 0.055 t C ha^{-1} yr^{-1}, 1σ), slightly higher than the Australian continental average value for the same period (0.117 ± 0.036 t C ha^{-1} yr^{-1}, 1σ), and largely attributable to the CO_2 fertilisation effect (i.e. the positive response of NPP to rising CO_2) (Haverd *et al.*, 2013b). The NEP values, by definition, exclude the influence of disturbances such as fire, which are discussed in the following section.

Biome net ecosystem carbon balance (NECB) and net biome production (NBP).

NEP is defined above as the difference between GPP and Re (Schulze *et al.*, 2000; Chapin *et al.*, 2006). However, many other processes are also involved in the terrestrial carbon balance, including fire, methane fluxes, dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) losses to rivers, volatile organic carbon emissions (VOC), erosion, disturbances (e.g. insect outbreaks) and land-use change. In most terrestrial ecosystems, carbon is accumulated fairly steadily over time by plant growth, but lost in relatively infrequent, large emission events associated with episodic disturbance. The resulting net accumulation is described by the Net Ecosystem Carbon Balance (NECB), defined by Chapin *et al.* (2006) as NEP less these episodic carbon losses from additional natural and anthropogenic disturbances. Thus, NECB is representative of longer-term ecosystem productivity, and better represents a system that experiences frequent disturbance. We follow the convention that NEP and NECB are negative for carbon losses from the ecosystem to the atmosphere. The Net Biome Productivity (NBP) is the NECB extrapolated to larger spatial scales (Chapin *et al.*, 2006).

NECB and NBP are important variables to quantify for Australian savannas as these ecosystems are subjected to disturbance processes that range in temporal scales from days to months (herbivory including termites and large grazing animals), annual to decadal (fire) and for coastal and sub-coastal savannas, decadal to century time scales, via impacts from extreme storm events and cyclones (Cook & Goyens, 2008; Hutley *et al.*, 2013). Disturbance plays a fundamental role in savanna dynamics and is important for maintaining tree and grass coexistence. Fire frequency, herbivory and climatic variability are drivers of tree recruitment and growth, with high levels of disturbance resulting in demographic bottlenecks that constrain the growth and recruitment of woody components, resulting in grass persistence or even dominance (House *et al.*, 2003; Bond, 2008; Lehmann *et al.*, 2011). Grazing cattle are a major source of disturbance in savanna systems (although stocking rates in Australia are minimal) since they consume grass, reduce fuel for fire and create ecological space for shrub and tree invasion (Hill *et al.*, 2005). In the absence of disturbance, particularly fire, savannas tend to become more woody, although canopy closure may be limited by rainfall or herbivory (Murphy & Bowman, 2012).

Savannas are well adapted to fire disturbance with dominant savanna tree species such as *Eucalyptus* and *Corymbia* investing in thick, protective bark. This appears to be a trait that is just as important as fast growth in height or diameter that enables juvenile trees to survive repeated fire (Lawes *et al.*, 2011). While mature trees can survive fire, repeated damage can result in the exhaustion of internal carbohydrate reserves, which can be significantly lower in roots and stems of burnt trees, when compared to unburnt individuals (Hutley, Lawes, unpublished data).

To estimate savanna NBP, direct observations of NEP as well as fire derived carbon losses are required at the stand scale (flux towers) through to the regional and biome scales via remote sensing and modelling. There are only few estimates of savanna NECB and NBP available

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Table 2~~(Table 2~~). A number of savanna stand based studies have been conducted in high rainfall (>1400 mm annual rainfall) coastal sites across the Northern Territory of Australia. We have previously estimated NECB at a long-term eddy covariance flux tower at Howard Springs using 5 years of data from 2001 to 2005 (Beringer *et al.*, 2007). Fire had direct impacts through GHG emissions but also had indirect effects through the loss of productivity due to reduced functional leaf area index and the carbon costs of rebuilding the canopy. The impact of fire on the latent energy exchange of the canopy was evident for 40 days while foliage regrew; however, the carbon balance took approximately 70 days to recover. Annual NEP without a fire event at Howard Springs was estimated to be +4.3 t C ha^{-1} yr^{-1} within a range of +3.5 to +5.1 t C ha^{-1} yr^{-1}. We previously calculated the average annual indirect fire effect to be -0.7 t C ha^{-1} yr^{-1} using a neural network model approach and estimated average emissions of fine and coarse fuels as -1.6 t C ha^{-1} yr^{-1}. This allowed us to calculate a NECB of +2.0 t C ha^{-1} yr^{-1}. We then partitioned this remaining sink and suggest that most of this can be accounted for by woody increment growth (+1.2 t C ha^{-1} yr^{-1}) and shrub encroachment (+0.5 t C ha^{-1} yr^{-1}).

The Howard Springs site is only representative of the mesic savannas (> 1200 mm annual rainfall) whereas there is much variation in savanna structure and composition (Hutley *et al.*, 2011) across the range of subtropical Australian savannas that extend to the 500–600 mm rainfall isohyet (Fox *et al.*, 2001). Cook *et al.* (2005) examined savanna tree growth and fire at high rainfall sites in Kakadu National Park and with biennial, low intensity fires, tree growth contributed approximately 0.5 t C ha^{-1} to NBP of these savannas. At a broader spatial scale, Williams *et al.* (2004) used flux tower derived estimates of NEP and applied this to a savanna region of western Arnhemland in the Northern Territory where the spatial and temporal dynamics of the fire regime are well understood and fuel (carbon) consumption and

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emission well quantified (Russell-Smith *et al.*, 2009), providing a robust estimate of NBP at a regional scale (32, 484 km^2). In this exercise, the range of observed NEP and the range of burnt areas were varied to estimate NBP under a variety of scenarios and this Arnhemland region ranged from being a source (-0.67 t C ha^{-1} y^{-1}) to a weak sink. At a savanna biome scale (1.91 x 10^8 ha), a similar result was obtained by Barrett (2011) using a modelling approach, whereby a wide range of NEP and NBP values were simulated. Data-assimilation methods have also been used including the Vegetation and Soil carbon Transfer (VAST) model (Barrett, 2002) where GPP, NPP, NEP were estimated over a 20 year period. Satellite derived estimates of burnt area and carbon emissions were then used to estimate NBP across the north Australian savanna biome with NBP ranging from near 0 to a weak sink of 0.2 t C ha^{-1} yr^{-1}. Tropical savanna occurs across a 1000 mm rainfall range in north Australia and this biome scale estimate is significantly lower than the high rainfall plot based estimates at Howard Springs for example, because low productivity semi-arid savannas were included in this modelling study. This value of NBP is close to the global savanna NEP estimate of 0.14 t C ha^{-1} yr^{-1} from Grace *et al.* (2006).

None of these estimates of NEBC and NBP includes losses of dissolved or particulate carbon nor losses from volatile organic carbon emissions which are required for a complete assessment of carbon cycling in this ecosystem at all spatial scales. At high rainfall sites (>1200 mm) with high NPP (~10 t C ha^{-1} y^{-1}) and frequent fire the carbon residence times are short, of the order of 5 years or less (Chen *et al.*, 2003; Barrett, 2011). At the Howard Springs flux site (Beringer *et al.*, 2007; Hutley *et al.*, 2011), where wet season runoff is around 410 mm y^{-1} and aquifer recharge about 200 mm y^{-1} (Cook *et al.* 1998), the export of carbon via these pathways could be significant.

In addition, the role and fate of soil black carbon (charcoal) is largely unknown. The production of black carbon for savanna and grassland fires are generally < 3% but there is considerable uncertainty (Grace *et al.*, 2006). One of only a few estimates comes from southern Africa where, Kuhlbusch *et al.* (1996) found that 0.6–1.5% of the exposed biomass carbon was converted to black carbon. The fate of black carbon is poorly understood and some of it could be either blown away, washed away, oxidised in proceeding fires or buried. In Australia, estimates of buried black carbon from the Australian National Soil Archive from the savanna regions around Darwin were 11.9% median (min 0% to max 58.6%) (Proportion (%) of black C of the soil organic carbon pool) (Lehmann *et al.*, 2008). Black carbon has been show in modelling studies to reduce the temperature sensitivity and increase the turnover time of carbon in soils (Lehmann *et al.*, 2008), which will not have an impact on long term NBP. However, transport or oxidation to non-CO_2 gasses will not be accounted for in typical measurements of savanna NEP using Eddy Covariance flux towers and therefore black carbon production, transport and consumption should be quantified to establish a robust NBP for savannas.

Fire frequency, mortality and recruitment: experimental results from northern Australia

Experimental assessments of the impacts of fire frequency and fire intensity on tree and shrub mortality have been undertaken since the early 1990’s. Thus Williams *et al.,* (1999) demonstrate that total live-stem basal area can increase marginally in both control (unburnt) and early dry season burnt plots in open *Eucalyptus* woodland in northern Australia. In contrast, substantial declines (declines of 27 %) were observed over the 4 year experimental study in plots receiving late season burns. A linear decline in tree survival with increasing fire intensity was observed (Williams et al. 1999). A similar trend (of increasing loss of stem basal area and survival with late compared to early fires) was also reported by Prior *et al*., (2010) and Murphy *et al.,* (2014). The intensity and frequency of fires also impacts on savanna structure. Frequently burned sites tend to have annual grasses dominating but the absence of burning is associated with a decrease in annual grass cover and either an increase or decrease in perennial grass cover (Russell-Smith *et al.,* (2003). Long-term exclusion of fire (23 years) results in a significant change land cover: grass cover declines, tree stem density increases and there is an increase in the density of rainforest tree species (Russell-Smith *et al.,* 2003, Scott *et al.,* 2012). The transition from open eucalypt woodland to a taller, more closed forest having a smaller dominance of euculypt species is associated with reduced fire frequency because of the lack of fuel (cured grass). The re-introduction of fire for 3 years had no impact on the overall composition of the grass layer and woody species but reduced the mid-story stem density and a reduction in canopy cover (Scott *et al.,* 2012). Longer-term changes in savanna structure can be inferred from the analyses of Prior et al., (2010 who showed that cooler early dry season fires result in the largest loss of living stems in small trees, whilst late fires have the largest impact on intermediate (3 – 5m tall) and very tall (> 20 m) trees. Furthermore, recruitment rates were reduced by fire and that a recruitment bottleneck occurs in response to fire because of the differential effects of fire on small, medium and tall trees (Prior *et al.,* 2010). Increased stem density and canopy cover are associated with increased GPP and NPP globally and therefore increased savanna tree standing biomass and canopy cover will be reflected in changes in savanna carbon and water budgets at local and regional scales.

Climate change, woody thickening and drought impacts on savannas

Woody thickening, (an increase in standing biomass of woody species) is a global phenomenon most commonly observed in arid and semi-arid regions including savannas and shrublands (Bowman et al. 2001, Scott et al. 2009, Witt et al. 2009). Impacts of woody thickening on biogeochemical cycling (McCulley *et al*., 2004) and C stocks (Gifford and Howden, 2001; Burrows *et al*., 2002) is regionally substantial and globally significant (Pacala *et al*., 2001; Foley *et al*., 2005). Thus the conversion of grasslands to closed woodlands could represent an increase in terrestrial sink strength of 94.3 PgC (Scholes and Hall 1996). The causes of woody thickening include changes in land-use practice, including changes in fire regime and grazing pressure (Scholes and Archer 1997, Archer *et al*., 1995; van Langevelde *et al*., 2003) and cessation of clearing and abandonment of formerly managed cropland (Schimel *et al*., 2001; Gifford and Howden, 2001; Jackson *et al*., 2002). More recently there has an increasing awareness of potential roles for climate and changes in atmospheric CO_2 concentration in driving woody thickening (Fensham *et al*., 2005; Berry and Roderick, 2006; Davis *et al*., 2007; Sankaran *et al*., 2008). Most recently, the results of a modelling study using a highly detailed mechanistic soil-plant-atmosphere model tested the hypothesis that increased atmospheric CO_2 concentrations, with or without photosynthetic acclimation, can increase GPP and that this, along with associated changes in soil moisture content arising from reduced stomatal conductance, can explain woody thickening (Macinnis-Ng *et al.,* 2010). They concluded that a large increase in LAI can be supported through the interaction of (a) increased GPP; (b) decreased stomatal conductance; and (c) increased soil moisture content in water limited environments, even with significant increases in atmospheric VPD. A positive feedback between increased LAI and increased GPP was apparent in their

Drought-induced tree mortality is occurring across all forested continents and is predicted to increase in the 21st century (Allen *et al.,* 2010, Dai 2011). The scale (temporal and spatial) of drought and mortality are very large, with regional-scale die-off recorded in Southern Europe, north and south America and Australia. Drought is associated with increased temperatures because of an increase in the partitioning of incoming solar radiation to sensible heat fluxes, arising from a decline in partitioning to latent heat fluxes. As a consequence of both effects (increased mortality; increased temperatures) the potential for more frequent and more severe fires increases because of the increase in fuel load, faster curing times and a drying of the landscape overall. Increased fire frequency and intensity reduces standing biomass (C stock) and can reduce landscape NPP over the short- and long-term. Changes in fire regime can also cause shifts in biome composition (see above), thereby significantly changing biome function (C uptake; C storage). Drought and increases in temperature and VPD to supra-optimal values are associated with reduced NPP (Eamus *et al.*, 2013) and changes in WUE (Eamus 1991) whilst increased mortality is associated with reduced NPP in the short-to-medium term. Whilst increased temperatures have been linked to drought as causing increased rates of mortality, recent analyses suggest that increased VPD associated with drought may be more important than previously realised, in driving mortality (Breshears *et al.*, 2013, Eamus *et al.*, 2013).

Soil greenhouse gas (GHG) exchange

Soils in savanna ecosystems contribute to the production and consumption of the greenhouse gases (GHG) CO_2, CH_4 and N_2O via soil microbial processes and subterranean termite activity. In addition to direct GHG emissions during biomass combustion (Section 2), fire may also affect soil-atmosphere exchange of GHGs in the long-term (Castaldi *et al.*, 2006, 2010) by altering soil C inputs, nutrient inputs, surface microbial activity, surface moisture and temperature. However, the effect of fire upon savanna soil-atmosphere exchange of CO_2,

CH_4, N_2O, and other trace gases is unclear and often contradictory (Anderson & Poth, 1998; Pinto, 2002).

A detailed study on the effect of fire on soil based GHG emissions at the Howard Springs savanna woodland was conducted by Livesley *et al.* (2011) over a 16 month period from October 2007 to January 2009. Soil GHG fluxes were measured at high temporal resolution before and after an experimental fire using a field-based gas chromatograph connected to automated chambers. Monthly manual chamber measurements on unburnt and experimentally burnt plots provided greater replication and spatial coverage. There was no apparent impact of fire upon soil CO_2 emissions following either of the two experimental burns, one in 2007 (data not shown) and one in 2008 (**Error! Reference source not found.**~~Figure 9~~). The savanna soil generally acted as a CH_4 sink, and this did not change after fire. However, relatively large CH_4 emissions were observed in a short 24 hour period directly following the fire as the ash bed smouldered. The fire treatments had no impact on the negligible soil N_2O exchange rates.

The moderate intensity of these savanna fires at Howard Springs did not alter soil properties enough to change the biogeochemical processes involved in the production and consumption of soil GHGs. Similar results were observed in South American (Pinto, 2002) and South African (Zepp *et al.*, 1996) savannas systems where no significant effect of fire on soil respiration flux was detected. Significant changes in soil respiration can only be expected after prolonged fire treatments (annual burn or long term fire prevention) (Pinto, 2002). However, a separate study in Australia's Northern Territory demonstrated that frequent fires (annually) can potentially lower soil respiration in the wet season following a fire (Richards *et al.*, 2012), as reduced overstorey C led to reduced belowground C inputs and consequently reduced soil respiration.

Soil CH_4 oxidation (uptake) activity is often greatest between 10 and 20 cm down the soil profile (Potter *et al.*, 1996), which is beyond the thermal impact of a low or medium intensity fires. This may explain why fire had no apparent effect on soil CH_4 fluxes in the Howard Springs (Livesley *et al.*, 2011), even though there was a significant decrease in soil surface moisture levels (0-5 cm) and a significant increase in surface temperature levels in the fire treatments. In South American and South African savannas, soil CH_4 oxidation was similarly unresponsive to fire events (Cofman Anderson & Poth (1998), Zepp *et al.*, (1996)) although the mechanisms involved were unknown (Castaldi *et al.*, 2006). The temporary absence of termite activity after fire may lead to a net increase in soil CH_4 uptake, as microbial oxidation is no longer offset by termite CH_4 emissions (Poth *et al.*, 1995). Alternatively, CH_4 uptake may increase after fire as soil diffusivity increases after surface organic material has combusted. Soil diffusivity is one of the main controllers of soil CH_4 uptake (Smith *et al.*, 2003; von Fischer *et al.*, 2009; Stiehl-Braun *et al.*, 2011), limiting the amount of CH_4 and oxygen that can reach the methanotrophic bacteria.

Forest or woodland fires often lead to an increase in soil NO_3^- and NH_4^+ (Attiwill & Adams, 1993) which may provide a substrate for nitrification or denitrification processes and thus increased N_2O emissions. Livesley *et al.* (2011) observed an increase in NH_4^+ but no change in NO_3^- after Howard Springs savanna fires but soil N_2O fluxes remained negligible (±1.0 µg N m^{-2} h^{-1}) suggesting tight N cycling in these savannas (Bustamante *et al.*, 2006). Many savanna fire studies have measured an increase in soil inorganic N but no discernible increase in N_2O flux (Levine *et al.*, 1996; Anderson & Poth, 1998; Pinto, 2002; Andersson, 2003).

The research suggests that fire in savanna systems can potentially impact soil respiration (decrease) as well as soil CH_4 uptake (increase). The specific circumstances under which these impacts can be observed are not apparent and the mechanisms involved are unclear and should be subject to further research. There is reasonable evidence to suggest that fire has no, or very little, impact upon savanna soil N_2O emissions.

ENERGY AND WATER BALANCES

The previous sections have provided an assessment of the effects of fire on the carbon balance and GHG exchanges from leaf to biome scales. However, there are also significant impacts on the radiation and energy balance of savannas following fire. When flying above Australia's tropical savanna late in the dry season, the visible broad spatial extent of blackened landscape supports the hypothesis of implications for atmospheric circulation at a range of scales. We have previously measured radiation, energy and carbon exchanges over unburned and burned (both before and after low and moderate intensity fires) open woodland savanna at Howard Springs, Australia (Beringer *et al.*, 2003). Fire affected the radiation balance immediately following fire through the consumption of the grass-dominated understorey and blackening of the surface. Albedo was halved following fire (from 0.12 to 0.07 and from 0.11 to 0.06 for the moderate and low intensity sites respectively), but the recovery of albedo was dependent on the initial fire intensity. The low intensity fire caused little canopy damage with little impact on the surface energy balance and only a slight increase in Bowen ratio. However, the moderate fire resulted in a comprehensive canopy scorch and almost complete leaf drop in the days and weeks following fire. The shutdown of most leaves within the canopy reduced transpiration and altered energy partitioning markedly, with much less energy partitioned into the evaporative heat fluxes and much more into sensible heating of the atmosphere (**Error! Reference source not found.**~~Figure 10~~). Leaf death and shedding also resulted in a cessation of ecosystem carbon uptake and the savanna turned from a sink to a source of carbon to the atmosphere because of the continued ecosystem respiration (previous section). Post-fire, the Bowen ratio increased greatly due to large increases in sensible heat fluxes. These changes in surface energy exchange following fire, when applied at the landscape scale, may have impacts on climate through local changes in circulation patterns and changes in regional heating, precipitation and monsoon circulation.

FIRE, LOCAL CLIMATE AND BOUNDARY LAYER PROCESSES

As shown above, fire scars can radically alter the surface energy budget of tropical savanna by reducing surface albedo, increasing available energy for partitioning into sensible and latent heat fluxes, as well as by increasing ground heat flux. Changes such as these can alter atmospheric heating rates and boundary-layer conditions, which can ultimately feedback to affect the local and regional climate. We have previously measured radiative and energy fluxes and boundary layer profiles over burnt and un-burnt tropical savanna near Howard Springs (Wendt *et al.*, 2007). At the burnt site a moderate intensity fire, estimated between 1,000 - 3,500 kW m^{-1}, initially affected the land surface by removing all understorey vegetation, charring and blackening the ground surface, scorching the overstorey canopy and reducing the albedo (Wendt *et al.*, 2007). Tethered balloon measurements showed that, despite the presence of pre-monsoonal rain events occurring during the measurement period, the lower boundary layer over the burnt site was up to 2°C warmer than that over the un-burnt site during the middle of the day and this warming extended to at least 500 m above the surface (**Error! Reference source not found.**~~Figure 11~~). This increase in boundary-layer heating, when applied to fire scars at the landscape scale, can have the ability to form or alter local mesoscale circulations and ultimately create a feedback to regional heating and precipitation patterns that may affect larger-scale processes such as the Australian monsoon (Pielke *et al.*, 2011). For example, a similar effect has been observed across the Western Australian 'Bunny Fence' where native vegetation contrasts adjacent agricultural fields, which generated altered boundary-layers and modified precipitation patterns (Lyons *et al.*, 1993; Evans *et al.*, 2011).

REGIONAL CLIMATE FEEDBACKS

The potential for feedbacks between changes in land surface properties following fire (as described above) and the regional climate has been investigated by Görgen *et al.* (2006). A fire-regrowth scheme was implemented in the soil-canopy component of the Conformal-Cubic Atmosphere Model (C-CAM, (McGregor & Dix, 2001)) with a grid resolution of 65 km over Australia. Surface properties were modified in the model by the fire intensity and its spatial and temporal extent (timing of the event, and the length of the regrowth period) were mapped using the Advanced Very High Resolution Radiometer (AVHRR) satellite. Albedo, roughness length, LAI, and fractional vegetation coverage were the biophysical properties used to simulate the impacts of these factors on surface energy balances and flux partitioning.

In an initial experiment, we simulated a large-scale and high intensity fire in the late dry-season with a long regrowth period extending well into the wet season (Görgen *et al.*, 2006). On average, the fire caused a change of about 150 $W\ m^{-2}$ in net radiation, which led to increased soil temperature, larger turbulent fluxes, higher mixing ratios in the boundary layer, increased wind speeds, and an increased boundary layer height. Moisture availability was the limiting factor in convective precipitation responses. Precipitation increases of around 15% were statistically significant during the pre-monsoon season (**Error! Reference source not found.**~~Figure 12~~a). These changes were associated with an intensification of the Pilbara heat low (**Error! Reference source not found.**~~Figure 12~~b) and the potential for increased lateral inflow of moist oceanic air.

The implications of these findings for Australian monsoon precipitation and circulation were further investigated using a factorial experimental design to fully characterise the response of the system across a realistic range of fire and regrowth characteristics in 90 independent experiments (Abramson *et al.*, 2006; Lynch *et al.*, 2007). It was found that the total area receiving monsoon precipitation could increase over northern Australia by up to 30% in the presence of large, high intensity fires late in the dry season. Indeed, the timing of the fire accounted for 58% of the variance in monsoon precipitation, followed by the area (18%) and the intensity (15%). Fires above 40% of the maximum possible size that occur late in the dry season have a strong positive impact on monsoonal circulation as quantified by the Australian Monsoon Index (**Error! Reference source not found.**~~Figure 13~~). Furthermore, due to an increase in moisture convergence the uplift intensity, rather than the moisture availability, controlled the precipitation variability. Late dry season fires of high intensity can significantly affect the Australian Monsoon Index from an average of 0.03 m s^{-1} to 0.43 m s^{-1} (**Error! Reference source not found.**~~Figure 13~~ - horizontal axis). Finally, and perhaps most significant, these findings clearly indicated the dependency of the impacts on key thresholds in intensity and area in the savanna fire regime.

CONCLUSIONS

This paper has demonstrated that the modification of the savanna land surface via fire influences rest of earth system via biophysical and biogeochemical cycles with feedbacks to regional and global climate. The future status of savannas worldwide remains uncertain as they are particularly threatened by land use change and disturbance (fire, cyclones, grazing, invasive species). We suggest that a priority for future research is to understand how these disturbance agents affect biophysical and biogeochemical cycling and how these may interact with climate change in the future. We suggest that climate change will confound projections to quantify how savannas may respond to future environmental perturbations. For example, reductions in rainfall would have consequences for grass and tree biomass (and hence fuel load) with a feedback to fire intensity (reduction), which may reduce the impact of fire and therefore alter biophysical and biogeochemical processes in the short term. Longer term shifts in fire frequency and intensity will have a flow on effect upon savanna demography, thereby altering savanna structure and function that in turn modifies land surface properties such as LAI, surface roughness and flux partitioning. This then has the potential to feedback from local to global climate both mechanically (*local*) and through secondary effects driven by regional circulation changes.

Additional research priorities include the need to quantify additional processes to obtain a comprehensive Net Biome Productivity, such as non-CO_2 fluxes, dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) losses to rivers, volatile organic carbon emissions (VOC), black carbon, and sediment transport (Randerson *et al.,* 2002). Although this paper focussed on the ecosystem scale, it is expected that grass and trees will respond differently to future environmental change and disturbance that we may see shifts in the tree:grass ratio due to changes in rainfall and increasing atmospheric CO_2. Increasing temperature and changing growing season length will also modify fire regimes that will subsequently feedback to further alter tree:grass dynamics. Therefore an understanding of the complex suite of feedbacks is required using a process based approach to better understand potential responses. Despite the importance for the earth system and human well-being, savannas represent a gap in earth observations and it is a challenge for remote sensing and modelling communities to better capture these ecosystems in global climate and vegetation models.

The challenge will be to manage our savanna ecosystems to provide ecosystem services in the face of changing fire regimes and the potential interaction between climate change, invasive species and land management. Currently "our capacity to manage fire remains imperfect and may become more difficult in the future as climate change alters fire regimes. This risk is difficult to assess, however, because fires are still poorly represented in global models" (Bowman *et al.*, 2009, page 481). The solution requires an earth system science framework to provide a holistic understanding of linkages between physical and human systems to provide tools for assessment, management and policy.

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Table 1. Annual emissions from the Australian savanna biome calculated using the Global Fire Emissions Database (GFED) v3.1 (Mu *et al.*, 2011).

Species	Mean
All	107 x 1E12 g C per year
CO_2	370 x 1E12 g CO_2 per year
CH_4	50 x 1E10 g CH_4 per year
N_2O	472 x 1E8 g N_2O per year
NO_x	476 x 1E9 g NO_x per year
CO	138 x 1E11 g CO per year

Table 2. NBP and NECB estimates for Australian savannas. A positive value denotes a net carbon sink.

Source	**Method**	**Spatial scale**	**Temporal scale**	**NBP*** **(t C ha^{-1} y^{-1})**
Barrett *et al.*, (2011)	Model (VAST1.2)	Biome (NBP)	3 y	+0.016 – +0.20
Beringer *et al.* (2007)	Flux data	Stand (NECB)	5 y	+2.0
Hutley *et al*. (2005)	Flux data	Stand (NECB)	2 y	+1.54
Williams *et al.* (2004)	Spatial extrapolation	Region (NBP)	Annual mean	+2.12 – -0.67
Chen *et al.* (2003)	Inventory	Stand (NECB)	Annual mean	+1.1

Table 3. Annual sums are in units kg ha^{-1} yr^{-1}; Global Warming Potentials (GWP) are 100 year time horizon from Forster *et al.* (2007). All carbon dioxide equivalent estimates are in kg CO2-e ha^{-1} y^{-1}. Summarised from (Livesley *et al.*, 2011).

	CO_2 flux		**N_2O flux**		**CH_4 flux**	
	Unburnt	**Burnt**	**Unburnt**	**Burnt**	**Unburnt**	**Burnt**
Annual sums	44,385	52,876	−15.30	12.6	−1,611.7	−1,643.7
GWP	1	1	298	298	25	25
CO_2-e	44,385	52,876	−4.5	3.7	−40.3	−41.1

FIGURE LEGENDS

Figure 1. Unburnt and burnt savanna at Howard Springs (12°29′39.12″S 131°09′09″E) a typical Eucalypt open forest. Dominant overstorey species *Eucalyptus miniata* and *Eucalyptus tetrodonta* with a sorghum tall grass understorey.

Figure 2. The important linkages between the land surface and the earth system. Modification of ecosystems through fire will influence on ecosystem properties (structure, composition, and function) and then through biophysical and biogeochemical feedbacks at multiple scales from local boundary layer to global climate. The important exchanges are listed using black boldface text. Reprinted from Beringer *et al.* (2011a), page 1467.

Figure 3. Fire frequency of large fire events (generally > 4 km^2) in Australia for the period 1997-2011 derived from AVHRR burnt area mapping. Frequencies range from less than 0.1 pa (dark blue) to 1 pa (dark red). Areas in white have not been marked as burnt during the mapping period. Tropical savannas are outlined in red (updated from Maier & Russell-Smith (2012), page 84).

Figure 4. Mean annual fire emissions CO_2-e 2003-2009 for Australia (gC m^{-2} yr^{-1}). Reprinted from Mu *et al.* (2011), page 11.

Figure 5. Net photosynthesis plotted against stomatal conductance under high light conditions for expanding and mature foliage of *Eucalyptus miniata* and *Eucalyptus tetrodonta*. Data are combined for both burned and unburned trees. Expanding foliage typically showed net carbon losses to the atmosphere, even though incident photon flux density exceeded 1000 μmol photons m^{-2} s^{-1}. Reprinted from Cernusak *et al.* (2006), page 640.

Figure 6. Changes to net ecosystem productivity (NEP) following annual fire using a case study during 2003. The savanna changes from a sink to source after fire and remains a source for approximately 70 days despite the canopy being rebuilt and evapotranspiration returning to pre-fire levels after 40 days. The difference between the observed tower fluxes and the neural network (NN) model estimates of the fire free condition gives an estimate of the indirect impact of fire (the loss of canopy productivity and the cost of rebuilding the canopy)

on the canopy. The integrated effect of fire is to reduce NEP on average (2001-2006 events) by 0.7 t C ha^{-1}. Reprinted from Beringer *et al.* (2007), page 1000.

Figure 7. Annual time-series of regionally-averaged savanna (i) precipitation; (ii) NPP and (iii) NEP (NPP – heterotrophic respiration in the absence of disturbance). Shading represents 1-sigma uncertainties on the mean, and includes contributions from parameter uncertainties and forcing uncertainties, as evaluated in Haverd *et al.* (2013). (iv) Percentile rank time series of 10-y averaged precipitation, NPP and NEP. Each point is the percentile rank of the variable (precipitation, NPP or NEP), averaged over a 10-y window, centered at the time on the x-axis. A point having a percentile rank of 100% means that all other points in the time series have lower values.

Figure 8. Summary of the carbon stocks and fluxes from Howard Springs during 2001 to 2006. Net ecosystem production (NEP) is large but the frequent fire disturbance reduces the long-term storage through the direct emissions from coarse and fine fuels as well as an in direct loss of productivity, which gives a net biome productivity (NBP) of 2.0 t C ha^{-1} yr^{-1}. This carbon is accounted for in storage in woody material in shrubs and overstorey trees (values from Beringer *et al.* (2007) and Chen *et al.* (2003)).

Figure 9. Soil CO_2 (Panel A) CH_4 (Panel B) and N_2O (Panel C) exchange recorded with an automated measuring system from August 2008 to December 2009 at Howard Springs, NT. Panel D shows soil water content, soil temperature and rainfall in the same time period. Measurements were made in an area of unburnt savanna woodland and an area that received a controlled burn on 27 August 2008 (Livesley et al., 2011). Fire did not show a significant effect on soil CO_2, CH_4 and N_2O exchange other than an immediate spike in CH_4 emissions in the 24 hour period after the controlled burn.

Figure 10. Comparison of a) daily total evapotranspiration [ET] and b) daily total sensible heat flux for the moderate intensity fire site and for an unburnt control. The fire occurred on day 218 and the data are shown for days five through 10 following the fire. Reprinted from Beringer *et al.* (2003), page 336.

Figure 11. Mean atmospheric temperature profiles from 13 days of simultaneous soundings from the burnt and unburnt sites (between days 247 and 276, 2005). Profiles show mean (±

standard error) temperature data for each 25 m layer. The burnt site profile is shown as a red line and the unburnt site profile as a blue line. The dashed lines show the average for the nocturnal sounding (0100) and the solid lines show the day-time sounding at (1200). Data from Wendt *et al.* (2007).

Figure 12. Average (1979 to 1999) impacts of changed surface conditions on regional meteorological conditions in a C-CAM run with 80% burned area and fire intensity vs. control run without fires. a) Altered November pre-monsoon circulation patterns at 850 hPa. b) Statistically significant (dotted) precipitation increase [mm d^{-1}], October-November-December (OND) and December-January-February (DJF). Reprinted from Görgen *et al.* (2006), page 10 and 11.

Figure 13. Simulated differences between the fire scenario and the reference simulation of the response of the Australian Monsoon Index (AUSMI) [m s^{-1}] to variation in fire intensity and burned area. AUSMI is obtained by averaging the daily mean 850 hPa zonal wind speed from the equator to 10^{o}S and from 120^{o}E to 150^{o}E, the zone of monsoonal reversal of the flow in and out of Australia. The crosses indicate the population of the simulation space consisting only experiments with a timing of the fire-event later than the Julian day 250 (7 September). The contoured response metric is then Gaussian low pass filtered linear interpolations based on Delauney triangulations. Reprinted from Lynch *et al.* (2007), page 4.