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Improving estimates of savanna burning emissions for greenhouse accounting in northern Australia: limitations, challenges, applications

Jeremy Russell-Smith[A,B,I], *Brett P. Murphy*[A,B], *C. P. (Mick) Meyer*[C,G], *Garry D. Cook*[A,D], *Stefan Maier*[E,F], *Andrew C. Edwards*[A,B,F,G], *Jon Schatz*[A,D] *and Peter Brocklehurst*[A,H]

[A]Tropical Savannas Management Cooperative Research Centre, Charles Darwin University, Darwin, NT 0909, Australia.
[B]Bushfires NT (Northern Territory Government), Winnellie, NT 0821, Australia.
[C]CSIRO Marine and Atmospheric Research, Aspendale, VIC 3195, Australia.
[D]CSIRO Sustainable Ecosystems, Winnellie, NT 0821, Australia.
[E]Satellite Remote Sensing Services, Western Australian Land Information Authority (Western Australia Government), Floreat, WA 6014, Australia.
[F]Charles Darwin University, Darwin, NT 0909, Australia.
[G]Bushfire Cooperative Research Centre, Melbourne, VIC 3002, Australia.
[H]Department of Natural Resources, Environment and the Arts (Northern Territory Government), Palmerston, NT 0831, Australia.
[I]Corresponding author. Email: jeremy.russell-smith@nt.gov.au

Abstract. Although biomass burning of savannas is recognised as a major global source of greenhouse gas emissions, quantification remains problematic with resulting regional emissions estimates often differing markedly. Here we undertake a critical assessment of Australia's National Greenhouse Gas Inventory (NGGI) savanna burning emissions methodology. We describe the methodology developed for, and results and associated uncertainties derived from, a landscape-scale emissions abatement project in fire-prone western Arnhem Land, northern Australia. The methodology incorporates (i) detailed fire history and vegetation structure and fuels type mapping derived from satellite imagery; (ii) field-based assessments of fuel load accumulation, burning efficiencies (patchiness, combustion efficiency, ash retention) and N : C composition; and (iii) application of standard, regionally derived emission factors. Importantly, this refined methodology differs from the NGGI by incorporation of fire seasonality and severity components, and substantial improvements in baseline data. We consider how the application of a fire management program aimed at shifting the seasonality of burning (from one currently dominated by extensive late dry season wildfires to one where strategic fire management is undertaken earlier in the year) can provide significant project-based emissions abatement. The approach has wider application to fire-prone savanna systems dominated by anthropogenic sources of ignition.

Additional keywords: Arnhem Land, burning efficiency, emission factors, fire mapping, fuel loads, National Greenhouse Gas Inventory, Northern Territory.

Introduction

It is widely recognised that biomass burning of tropical savanna biomes is a globally significant driver of CO_2 cycling and a source of the greenhouse gases CO_2, CH_4, N_2O, other chemically reactive atmospheric trace gases, and aerosol particles (Scholes and Andreae 2000; Kondo *et al*. 2003). However, accurate quantification of such emissions is problematic, being reliant on reliable estimation of various parameters including the spatial and temporal distribution of burning, appropriate fuel load estimates and gaseous emission factors for different fuel fractions and fire types. The application of different assumptions and approaches, particularly estimation of the extent of biomass burning and fuel load estimates, has resulted in various global or regional estimates of savanna biomass burning and derived emissions (Seiler and Crutzen 1980; Crutzen and Andreae 1990; Hao and Liu 1994; Scholes *et al*. 1996*a*, 1996*b*; Shirai *et al*. 2003; Streets *et al*. 2003; Liousse *et al*. 2004). Although there are considerable uncertainties in greenhouse gas (GHG) emissions estimates, the extent and contribution of biomass burning from Australian savannas is typically ranked second or third after Africa (e.g. Hao and Liu 1994; Carmona-Moreno *et al*. 2005).

Under the provisions of the Kyoto Protocol, participating Tier 1 (developed economy) countries are required, where pertinent, to account for emissions of GHGs (specifically CH_4,

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N_2O) from 'prescribed burning of savannas' (UNFCCC 1998: Article 3, Annex A). Although Australia only recently has ratified the Kyoto Protocol, Australia's National Greenhouse Gas Inventory (NGGI) reports on savanna burning emissions; typically, accountable GHG emissions annually contribute between 1 and 3% of Australia's NGGI (Meyer 2004; AGO 2007*a*). The Kyoto Protocol also establishes a framework for developing market-based instruments to address anthropogenic sources and sinks of GHG emissions (UNFCCC 1998: Article 6). Accountable GHG emissions from Australian savanna burning are predominantly associated with anthropogenic sources (Russell-Smith *et al*. 2007).

Contemporary fire regimes across the sparsely settled 1.9 million km^2 northern Australian tropical savannas region have significant implications for biodiversity and soil conservation, GHG emissions, pastoral production, and broader social issues (e.g. Dyer *et al*. 2001; Williams *et al*. 2002; Russell-Smith *et al*. 2003, 2007; Whitehead *et al*. 2003). Key issues include developing economically viable solutions for implementing ecologically sustainable landscape-scale fire management, particularly for vast, remote, biodiversity-rich regions where, concomitantly, few employment opportunities exist, especially for indigenous (Aboriginal) communities (Whitehead *et al*. 2003). In 2005, the first major program aimed at substantially reducing GHG emissions from savanna burning in northern Australia commenced over 24 000 km^2 of the Western Arnhem Land Fire Abatement (WALFA) project area. Although that project operates under the authority of Australian and Northern Territory Government approvals, it is essentially a 'voluntary' offset arrangement (as in Bayon *et al*. 2006) given current absence of formal, regulated market mechanisms in Australia.

In the present paper, we address ongoing development and regional refinement of the pyrogenic emissions assessment methodology established under guidelines developed by the Intergovernmental Panel on Climate Change – IPCC (1997). Those guidelines underpin current methodologies adopted both in Australia's NGGI accounting of savanna burning, as well as informing WALFA project emissions abatement assessments. We begin with description of the current NGGI methodology (AGO 2007*b*). We then describe details of remotely sensed- and field-based methodologies developed for and key results arising from ongoing biomass burning emissions assessments for the WALFA region. In particular, we address the implications of fire seasonality on biomass burning and resultant emissions, and provide an assessment of uncertainties associated with respective parameters. Our broad purpose is to identify limitations and challenges in current Australian approaches, especially with reference to extending the current WALFA methodology to other prospective GHG abatement projects in fire-prone northern Australia, and potentially to fire-prone savanna systems generally.

Savanna burning and Australia's NGGI methodology

Australia's NGGI accounts emissions for Sector 4E – Prescribed Burning of Savannas using a country-specific methodology (AGO 2007*b*). Inventory accounting of GHG emissions uses an essentially bottom-up approach in which the emissions from each source, or subsector, are estimated as the product of an activity and an emission factor. The emission from major sectors and regions is the aggregate of all the sources and subsectors. Parties to the United Nations Framework Convention on Climate Change (UNFCCC) compile their national GHG inventories using reporting categories defined in the Revised 1996 Guidelines for Greenhouse Gas Inventories (IPCC 1997) following the procedures outlined in the IPCC Good Practice Guidelines (IPCC 2000, 2003). The methodologies used for each subsector may be the IPCC default methodologies or verified country-specific methodologies.

In this accepted IPCC methodology, for each region j, the emission of a trace species i (E_{ij}) is determined from savanna fire activity, i.e. the mass of fuel pyrolysed (FP_j) and the emission factor for each trace species (EF_{ij}):

$$E_{ij} = FP_j \times EF_{ij} \tag{1}$$

The mass of fuel pyrolysed is the product of the area exposed to fire (A_j), the fuel load (FL) and the burning efficiency factor (BEF). BEF is defined as the mass of fuel that is exposed to fire that is pyrolysed. It is calculated from the mass of fuel (M_{fuel}) before combustion and the mass of ash and unburned fuel residue remaining after combustion (M_{ash}),

$$BEF = 1 - \frac{M_{ash}}{M_{fuel}}$$

As a first approximation, fuels in the NGGI are divided into two size classes: fine fuels comprising leaf litter, grass and twigs less than 6-mm diameter, and heavy fuels. In relatively intense fires, fine fuels typically burn completely (i.e. with very high efficiencies), whereas heavy fuels tend to burn with lower efficiency. If k is the fuel class, then:

$$FP_j = A_j \sum_k (FL_{jk} BEF_{jk})$$

The area exposed to fire is the area of the fire scar A_j^1 corrected for the patchiness of the fire (P_j), i.e.

$$A_j = A_j^1 P_j$$

The emission factors (EFs) can be defined either relative to fuel mass pyrolysed, or relative to the fuel elemental content. The NGGI methodology uses the latter. For carbon species, CH_4, CO and volatile organic compounds (VOC), EFs are expressed relative to fuel carbon, and the nitrogen species N_2O and NO_x are expressed relative to fuel nitrogen. Fuel carbon mass is determined from fuel mass by the fuel carbon content (CC_{jk}) while fuel nitrogen is derived from the fuel mass by the product of CC_{jk}, and the fuel nitrogen to carbon ratio NC_{jk}.

Combining these equations, for CH_4, CO and VOC, the emission E_{ij} is then:

$$E_{ij} = EF_{ij} A_j^1 P_j \sum_k (FL_{jk} BEF_{jk} CC_{jk}) M_i \tag{2}$$

and for N_2O and NO_x is:

$$E_{ij} = EF_{ij} A_j^1 P_j \sum_k (FL_{jk} BEF_{jk} CC_{jk} NC_{jk}) M_i \tag{3}$$

where M_i is the ratio of the molecular mass to the elemental mass for each species. By convention, the mass of NO_x is expressed as NO_2.

Under IPCC guidelines, the NGGI methodology is Tier 2, that is, disaggregated spatially, with country-specific emission factors and parameters. It contrasts with Tier 3 methodologies, which are spatially highly disaggregated with emissions estimated by using complex models. For savanna fires, for example in a Tier 3 methodology, fuels and GHG emissions would be calculated from a carbon cycle model that incorporates disturbance from fire and is driven by monthly varying inputs that include climate parameters, radiation, vegetation indices, fire areas and hotspots. Both approaches have their strengths. Tier 3 methodologies, when based on an appropriate model, can account for complex non-linear processes and interactions, and can provide estimates of emissions at fine spatial and temporal resolution. Calibration and verification of the models is achieved by comparison with detailed measurements of processes and outputs at a small number of key locations in the landscape. However, the cost of Tier 3 sophistication can be a loss of transparency, and difficulty and expense in sourcing accurate spatial input data.

Tier 2 methodologies are largely empirical. Their reliability and strength depend on the accuracy of the classification into strata that capture most of the system variance between strata, and minimise the variance within strata. The process of determining the parameter values across all strata, if complete, fully characterises the behaviour and accuracy of the methodology. However, empirical models are valid only within the domain for which they are calibrated and therefore it is important that the classification produces strata that are stable to external drivers such as changing climate, land management or other sources of disturbance. If this is achieved, then the parameter values for each stratum (i.e. FL_j, BEF_j, etc.) should also remain stable with overall system behaviour occurring through changes in activity (fire area) or the population or size of the individual strata (vegetation class, seasonality, etc.).

Tier 2 methodologies with the properties described above should be suitable for project-based GHG accounting for carbon offsets or carbon credits. In the WALFA project, the GHG emissions offset derives from increasing the residence time of carbon and nitrogen in the landscape by reducing the average fuel consumption per year, through altering the seasonality and extent of savanna fires by early season prescribed burning. To account for this, the regional stratification of the NGGI (*j*) is expanded to the vegetation classes used in the fire management planning; $j =$ *eucalypt closed forest*, *eucalypt open forest*, *eucalypt woodland*, *sandstone woodland*, *sandstone heath*, *closed forest* and *riparian*. Fuel load classes are expanded to $k =$ *fine*, *coarse*, *heavy*, *shrub fuels.* Two additional strata are required to account for seasonality in burning efficiency; season ($l =$ *early*, *late*) and fire severity ($m =$ *low*, *moderate* and *high*). A third stratum describing fire regime ($n =$ *years since last burned*) extends the accuracy of fuel loads through accounting for fuel accumulation with time.

Including these extensions in Eqns 2 and 3 produces, for CH_4, CO and VOC:

$$E_i = EF_i \sum_{jln} \left((A^1_{jln} P_l) \sum_{km} (FL_{jkn} BEF_{jkl} S_{lm} CC_{jk}) \right) M_i \quad (4)$$

and, for N_2O and NO_x:

$$E_i = EF_i \sum_{jln} \left((A^1_{jln} P_l) \sum_{km} (FL_{jkn} BEF_{jkl} S_{lm} CC_{jk} NC_{jk}) \right) M_i \quad (5)$$

and where S_{lm} is the fraction of fires of severity class *m* in season *l*.

WALFA greenhouse gas emissions assessment methodology

This section describes evaluation of algorithm parameter values, based on extensive field- and geographic information system (GIS)-based studies undertaken for the WALFA project over the past 5 years. Background information detailing the regional context of contemporary fire patterns in the fire-prone, biodiversity-rich 23 893-km^2 WALFA region, and putative impacts on fire-sensitive vegetation components, are described in a companion paper (Edwards and Russell-Smith 2009). That paper also provides details of applied vegetation structure and fire extent mapping methodologies, including validation of respective derived surfaces. Here, we focus on the broader methodology applied to, and substantive results arising from, ongoing assessment of GHG emissions derived from savanna burning in that same region.

The methodology and results presented here build substantially on preliminary work (Russell-Smith *et al.* 2004) that has informed development of Australia's current NGGI methodology for accounting of GHG emissions from savanna burning, especially with respect to regional parameter values for fuel loads and burning efficiencies (Meyer 2004; AGO 2007*b*). The current WALFA regional assessment methodology includes major refinements on the NGGI, *viz.* (1) finer-resolution fire extent mapping (with Landsat as opposed to Advanced Very High Resolution Radiometer (AVHRR) imagery), especially accounting for seasonal differences in fire characteristics and properties; (2) substantial further field assessment, enabling more robust parameterisation of habitat- and seasonal-specific relationships for different *FL* and *BEF* components; and (3) further detailed assessments of C and N concentrations of different fuel components. In particular, the consideration of seasonality issues in the WALFA methodology is an important addition in that it provides a further opportunity for accounting for GHG emissions abatement via strategic fire management in the early dry season (EDS; see below, and in Discussion), other than by simply reducing the annual mean extent of burning.

For simplicity of presentation, we describe the methods and results for individual major parameters as considered previously in Eqns 1–5. In subsequent sections, we (1) summarise and consolidate these individual observations; (2) compare recent results with those as given in Meyer (2004), Russell-Smith *et al.* (2004), and as incorporated into the current NGGI methodology (AGO 2007*b*); (3) identify key gaps and required further refinements; and (4) present a detailed statistical assessment of uncertainties associated with derived parameters.

Burnt area

The land area exposed to fire is the combination of the fire scar area, i.e. the area defined by the outer boundary of the scar and the degree of patchiness within the fire scar.

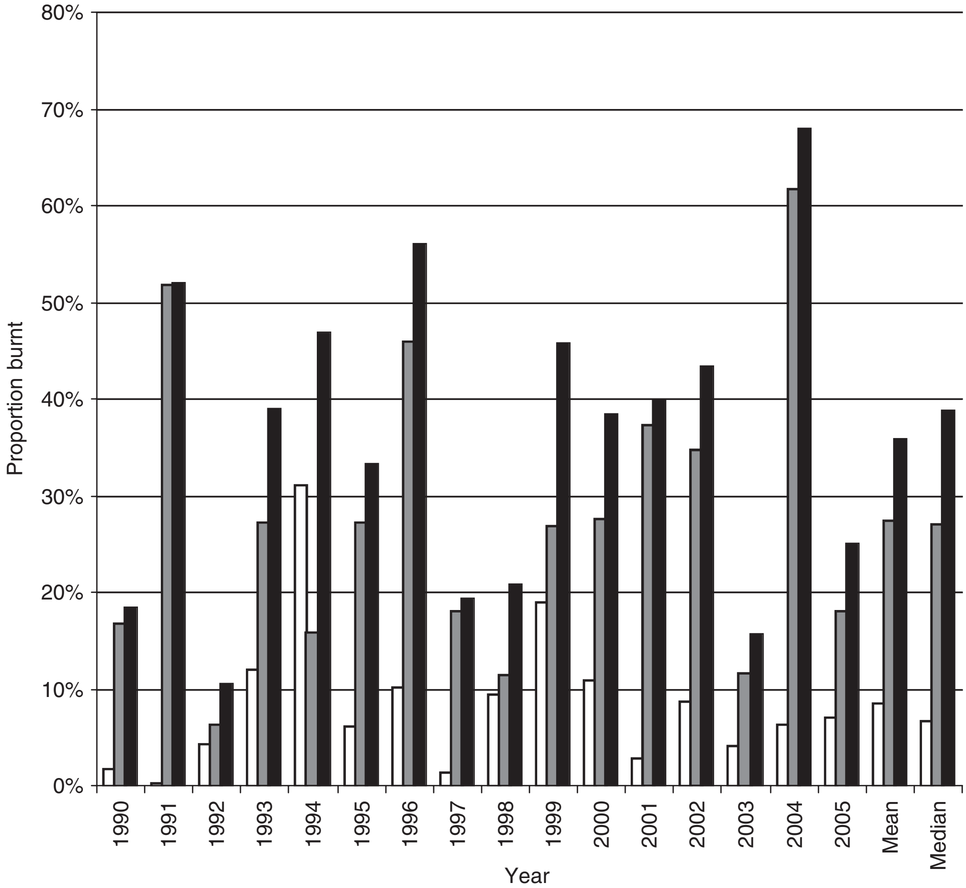

Fig. 1. Fire extent in the 23 893 km^2 Western Arnhem Land Fire Abatement (WALFA) region, by season, 1990–2004 (from Edwards and Russell-Smith 2009), where fire extent for early dry season = clear bars, late dry season = grey bars, annual = black bars. Mean and median values also shown.

Scar area

Validated fire history mapping for the WALFA area is available from 1990 to the present, mostly from Landsat imagery sampled three times through the ≥7 month dry (burning) season, as outlined in Edwards and Russell-Smith (2009). For convenience, but based also on indigenous fire management seasonality (Haynes 1985; Russell-Smith *et al.* 1997*a*), burnt area mapping for the period from April to the end of July is defined as occurring in the early dry season (EDS), and from August to December as the late dry season (LDS). Based on fire severity observations made from 178 plots over 10 years in adjacent Kakadu and Nitmiluk National Parks, fires in the EDS period are observed typically to be of low severity or intensity, whereas fires in the LDS are observed to be typically much more severe or intense (Russell-Smith and Edwards 2006). In the absence of fire severity mapping, currently the subject of a PhD program being undertaken by one of us (A. C. Edwards), we apply these fire severity observations proportionately by season to assess post-fire fuel consumption (see below).

Over the period 1990–2005, a mean of 35.7% per annum (p.a.) of the WALFA region was burnt (or, more properly, fire-affected), comprising 8.3% in EDS and 27.3% in LDS periods, respectively (Fig. 1).

Patchiness

Not all the area within a fire scar is burnt; however, optical satellite sensors typically lack sufficient resolution to detect this fine scale, i.e. within-pixel patchiness. Patchiness is measured using ground surveys, and in regional savannas is related both to seasonality and severity (Williams *et al.* 1998; Price *et al.* 2003; Russell-Smith and Edwards 2006). For present purposes, we consider separate estimates of fine-scale patchiness for the EDS and LDS, based on data assembled for the Arnhem Plateau as follows: (1) the transect study of Price *et al.* (2003); (2) more recent post-fire assessments made both on assessment plots, and along transects adapting the methodology of Price *et al.* (2003). Price *et al.* (2003) demonstrated that EDS fires are significantly more patchy than LDS fires, and the more recent post-fire assessments support this (Table 1).

Vegetation–fuel type

Validated vegetation–fuels mapping was undertaken for the WALFA region based on five vegetation structure classes (Closed forest – 571 km^2, Open forest – 6809 km^2, Woodland – 6175 km^2, Sandstone woodland – 5024 km^2, Sandstone heath – 5314 km^2), applying the methodology as outlined in Edwards

Table 1. Fine-scale patchiness of fires on the Arnhem Land plateau in early and late dry seasons

Patchiness is defined here as the average proportion of Landsat pixels identified as burnt that are actually burnt based on ground-truthing assessments

Season	Source	Replicates	Patchiness (% burnt)	Weighted mean
Early	Price *et al.* (2003)	1300	74.9	70.9
	Present study	669	63.3	
Late	Price *et al.* (2003)	556	84.6	88.9
	Present study	280	97.6	

and Russell-Smith (2009). This mapping surface is used as the basis for stratifying (1) derived fire history parameters (e.g. fire frequency, time-since-last-burnt) for application in biomass burning calculations; and (2) informing sampling efforts for assessment of fuel component accumulation with time-since-last-burnt. For present purposes, the relatively small Closed forest class is combined with Open Forest.

Fuel load accumulation

Fuel load accumulation of four fuel components (fine fuels, coarse fuels, heavy fuels, shrubs) has, to date, been assessed at 72 sites, in most cases each with three replicate plots ($n = 219$), spread widely throughout the WALFA study area. Sites were stratified by vegetation–fuel type and time-since-last burnt. At each plot, a permanent 100-m transect, of variable lateral dimension depending on tree (≥5 cm diameter at breast height, DBH) density, was established. Originally, it was intended to treat each site as an individual observation. However, it became readily apparent that, given the large plot size and scattered distribution of replicate plots, 'replicate' plots exhibited both substantial between- and within-plot heterogeneity with respect to recent fire history attributes. As such, for fuel accumulation purposes, we have treated plots as discrete observations. In other instances, however (e.g. carbon and nitrogen sampling of fuel components), results are reported by site – comprising the bulking of samples from plot 'replicates'.

Fuels sampling at each plot was undertaken in the middle of the year (dry season) as follows:

Fine fuels. Sampling undertaken consistently at five equidistant 1 × 1 m quadrats. Fine fuel (<0.6 cm diameter) samples were collected separately for grass and litter components, weighed in the field using a digital balance, and subsequently corrected for oven-dry weight based on one subsample for grass and litter, respectively. For presentation purposes, litter and grass components are aggregated and described here as 'fine fuels'.

Coarse fuels. Defined as sticks, etc. (≥0.6–<5 cm diameter), sampling undertaken consistently at 10 equidistant 1 × 1 quadrats, corrected for dry weight.

Heavy fuels. Sampling undertaken consistently in a 5 × 100-m swath, recording the length, diameter and hollowness of all fuel sections >5 cm diameter. Assuming each piece was cylindrical in shape, we estimated total volume of heavy litter and total mass assuming a specific gravity of 0.995 t m^{-3} (approximating that of eucalypt wood; Eamus *et al.* 2000).

Shrubs. Live shrubs were counted by species, and dead shrubs cumulatively, consistently in five 1 × 20-m sections, in each of four height classes: <0.5 m; 0.5–1.0 m; 1–2 m; and >2–<5 cm DBH. To estimate mean live shrub mass for each plot, we harvested and weighed the aboveground parts of at least three individuals of 107 live shrub samples (representing 46 common species) per respective height class. Species were allocated to one or more vegetation types in which they commonly occur and, for each vegetation type, mean shrub mass per height class was calculated. For 37 samples from 21 common species, we also measured the ratio of leaf mass to stem mass, and a mean value calculated for each height class. These values were used to estimate the mean mass of dead shrubs.

Simple linear regression was used to examine relationships between accumulation of fuel load components (i.e. grass, litter, coarse fuels, heavy fuels, shrubs) with time-since-fire, stratified by vegetation type. A natural log-transformation was applied both to fuel load response (given strong positive skews for most observations), as well as to time-since-fire (given apparent non-linearity). In those cases where significant relationships ($P < 0.05$) were not observed between fuel load components with time-since-fire, we assumed that fuel load was best described by a simple mean, calculated using natural log-transformed values. Based on a previous assessment of fine fuels (Russell-Smith *et al.* 1998), fuel loads for Closed forest were assumed to be the same as Open forest.

Fine fuel loads were positively related to time-since-fire in all vegetation types, whereas coarse and heavy fuels, and shrub fuel loads were only related to time-since-fire in Sandstone woodland (Fig. 2). Given the general paucity of fine fuel samples >4 years (Fig. 2), we have also included in Fig. 2 an additional 72 fine fuel samples from adjacent Kakadu National Park, sampled in a comparable manner with fine fuels data reported here (Russell-Smith *et al.* 1998). These additional samples are particularly pertinent to Sandstone heath vegetation, but provide some increased temporal sampling coverage also to Open forest and Sandstone woodland vegetation types.

A significant seasonal feature not addressed in sampled fine fuel accumulation data concerns additional inputs of leaf and twig litter in the LDS. For example, 5 years of annual observations from an experimental fire treatment concerning lowland woodland in Kakadu National Park indicated that mean fine fuel loads were 1.7 t ha^{-1}, or 53%, greater in LDS treatments owing to substantial litter fall from trees during the dry season (Cook 2003; Williams *et al.* 2003). This has a significant bearing on seasonally available fuel loads and, as considered later, provides a major opportunity for substantially reducing GHG emissions through strategic EDS savanna burning. To account for seasonal variation in fine fuel loads, we have corrected LDS fine fuel loads by 1.7 t ha^{-1}, relative to the fuel loads predicted on the basis of our collected data (shown in Fig. 2).

Burning efficiency factors

We measured the proportion of each fuel component consumed (i.e. either volatilised or converted to ash), referred to as fuel consumption, following exposure to fires of varying severity where fuel loads had been measured the previous day. Although some very fine ash may have been lost in this process, it was

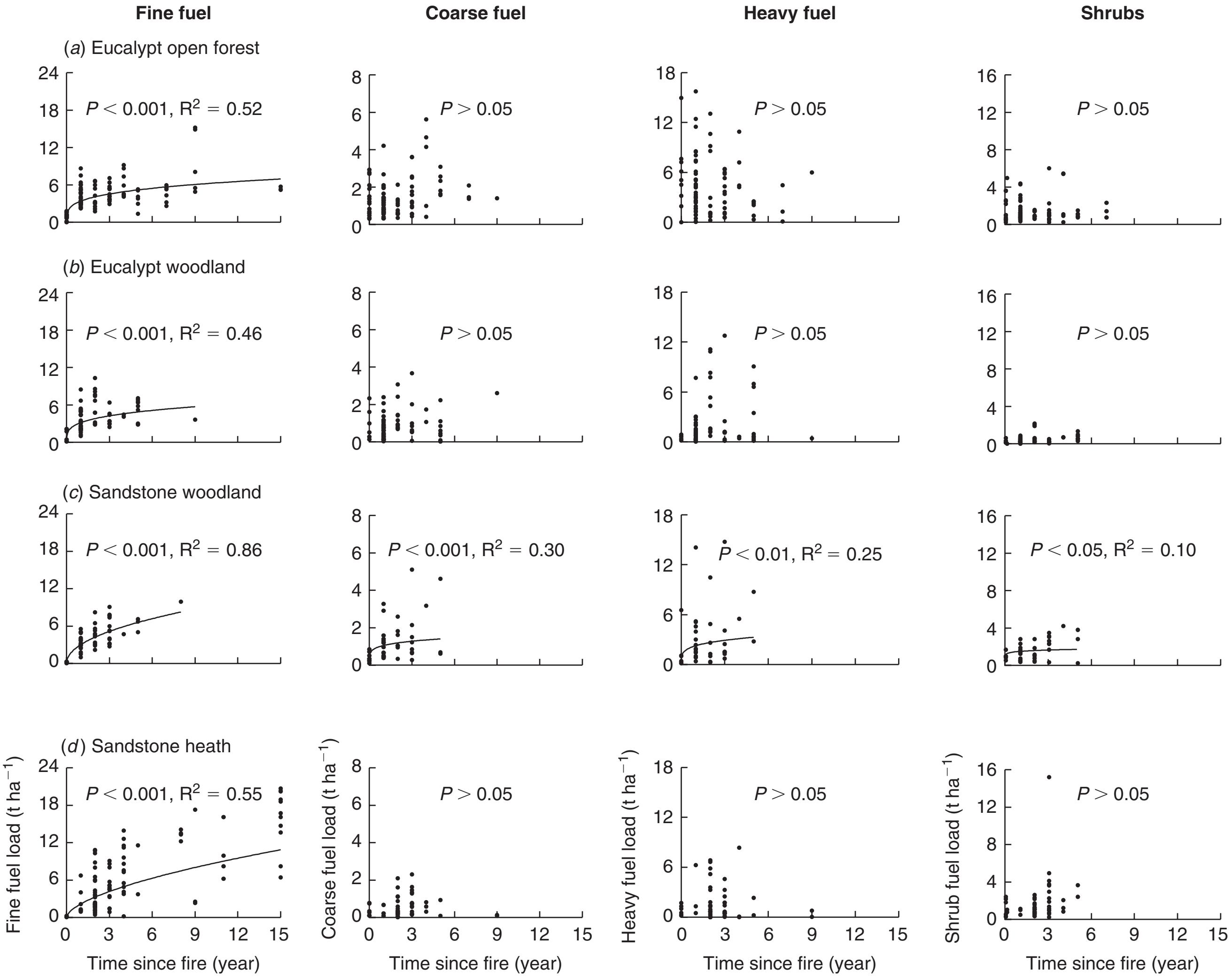

Fig. 2. Relationship between accumulation of respective fuel components and time-since-fire, in respective vegetation types: (*a*) eucalypt open forest; (*b*) eucalypt woodland; (*c*) sandstone woodland; and (*d*) sandstone heath. The regression lines, *P*-values and R^2 values refer to the results of linear regressions of fuel load (natural log-transformed) *v.* time since fire (natural log-transformed). Refer text for additional details.

(i) typically impractical to measure ash residue immediately following fire treatments given more pressing fire management priorities, and (ii) anyway, it is likely that some fine ash may have redistributed from surrounding burnt areas. Regardless, these errors are likely to be minimal compared with the sampling process (dustpan and brush, vacuum cleaner). The severity of 42 imposed fire treatments was assessed on the basis of leaf-scorch height, following the methodology of Russell-Smith and Edwards (2006), where: low-severity fires = leaf scorch height <2 m, or fires patchy with <20% of ground cover remaining unburnt; moderate severity = leaf scorch height >2 m, but upper canopy unscorched; high severity = upper canopy scorched.

Burning efficiency is defined as the mass of fuel exposed to fire that is pyrolysed. For fine fuels, which can be sampled using small quadrats, *BEF* is usually determined directly from the mass of fuel (M_{fuel}) before combustion and the mass of ash and unburnt fuel residue remaining after combustion (M_{ash}), i.e. $BEF = 1 - M_{ash}/M_{fuel}$. However, generally this is impractical for coarse and heavy fuel because large area quadrats would be required to ensure homogeneous samples. In the present study, an alternative approach was applied.

In the case of fine fuels and shrubs, we compared pre-fire and post-fire measurements of fuel load to calculate fuel consumption. Post-fire fuel loads were assessed using an identical methodology to pre-fire fuel loads. In the case of coarse and heavy fuels, mean estimates of consumption, based on the proportion of individual sticks and logs consumed by the fire (as evidenced from ash trails), were made along 100 × 1-m transects. This approach does not allow for residual ash and therefore, unless subsequently corrected, fuel load consumption tends to overestimate *BEF*.

We compared fuel consumption by fires of varying severity using analysis of variance. Given that fuel consumption was strictly bounded by zero and one, we applied a logit-transformation before analysis (Crawley 2002). For each fuel component, fire severity had a significant effect on fuel consumption, being consistently greater in fires of high *v.* low severity (Fig. 3). Using the respective EDS and LDS frequencies

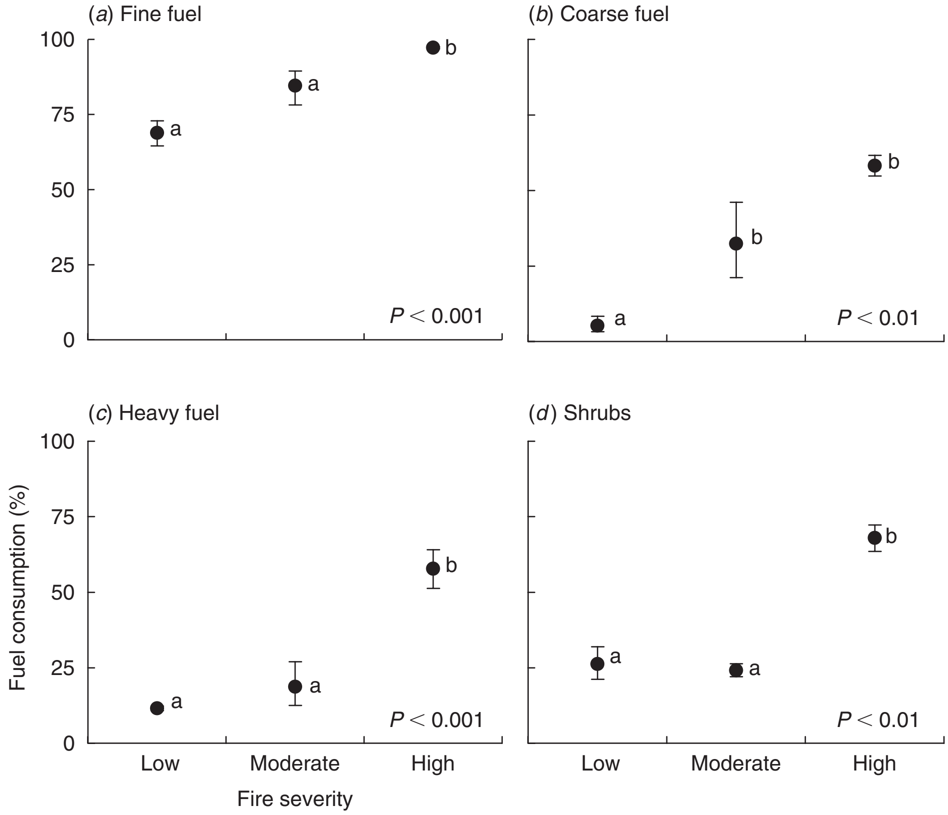

Fig. 3. The mean proportion of each fuel component consumed (fuel consumption) in relation to fire severity. Each P-value refers to the results of an analysis of variance. The means marked by the same letter are not significantly different ($P < 0.05$, Tukey HSD (Honestly Significant Differences) test). The error bars represent ± 1 standard error.

Table 2. Consumption rates for each fuel component for low-, moderate- and high-severity fires
The frequency of low-, moderate- and high-severity fires in the early and late dry seasons (from Russell-Smith and Edwards 2006) is used to derive estimated fuel consumption rates for each season. Refer text for details

Fire severity	Frequency (%)		Fuel consumption (%)			
	Early dry season	Late dry season	Fine fuel	Coarse fuels	Heavy fuels	Shrubs
Low	72	19	69	5	12	26
Moderate	20	47	85	32	19	24
High	8	34	97	58	58	68
Early dry season			74	15	17	29
Late dry season			86	36	31	39

of low-, moderate- and high-severity fires as reported in Russell-Smith and Edwards (2006), we derived estimates of fuel consumption for each fuel component for each season (Table 2).

Of the fuel that was consumed in each fire, we assumed that some was pyrolysed (volatilised) and the remainder was converted to ash. Some of the ash remained *in situ*, and some was entrained within the smoke plume. To estimate the proportion of fuel that was pyrolysed rather than just consumed, we required an estimate of the proportion of consumed fuel that was converted to ash.

At seven sites where we estimated the proportion of fuel consumed, we also measured the post-fire abundance of ash. Dry-weight estimates of ash were undertaken in five 20×20-cm quadrats per plot, sampled within 1 day of fire treatments. Using the mass balance approach of Fearnside *et al.* (1993), we estimated the proportion of consumed fuel (including all fuel components) that was converted to ash and remained *in situ*. To use this approach, it was necessary to estimate the carbon content of ash ($35.1 \pm 4.9\%$ (s.e.m.), $n = 7$) and of each fuel component (see carbon content below). The mean figure for *in situ* ash conversion was $11.6 \pm 2.3\%$ (s.e.m.) of consumed fuel. Given that

Table 3. Carbon content and nitrogen to carbon (N : C) ratios measured for vegetation components (*a*). These values were used to estimate values for the fuel components (*b*)

Standard errors are indicated for the means for each vegetation component. Fine fuels were calculated assuming an average litter composition of 46% non-grassy leaves, 39% twigs and 15% grass. Shrub-type fuel was calculated assuming an average litter composition of 69% twigs and 31% non-grassy leaves

Component	Carbon content (%)	N : C ratio	*n*
(*a*) Vegetation (measured values)			
Grass	43.9 ± 0.3	0.0087 ± 0.0002	68
Non-grassy leaves	50.6 ± 0.3	0.0122 ± 0.0003	66
Twigs	50.1 ± 0.2	0.0081 ± 0.0004	38
(*b*) Fuel (estimated values)			
Grass	43.9	0.0087	
Fine fuels	49.4	0.0101	
Coarse and heavy fuels	50.1	0.0081	
Shrubs	50.2	0.0093	

we could not determine which fuel components contributed to the ash, we assumed that the *in situ* ash conversion rate was the same for all fuel components.

Fuel consumption (Table 2) was converted to *BEF* by reducing each by the average fraction of fuel remaining after combustion as ash (11.6%). In the absence of more comprehensive data, we applied this rate equally to fires of low, moderate and high severity.

Carbon and nitrogen content of fuel components

We measured carbon content and N : C ratio of standing grasses from 68 sites and non-grassy leaf litter from 66 sites. Measurements were also made for twigs collected from 38 sites. Using these measured values, we derived estimates of carbon content and N : C ratio for each fuel component (Table 3). To do this, we made the following assumptions: (1) fine litter had a composition equivalent to 46% non-grassy leaves, 39% twigs and 15% grass (based on mean litter composition at 50 sites); (2) coarse and heavy fuels were equivalent to twigs; and (3) shrubs had a composition equivalent to 69% twigs and 31% non-grassy leaves (based on mean leaf : stem ratio for 37 individual shrub samples).

Differences between 2004 and current parameters

Compared with the original 2004 methodology developed for the WALFA project region (Meyer 2004; Russell-Smith *et al.* 2004), revised parameters that have a considerable effect on estimated CO_2 equivalent (CO_2-e) emissions substantially include various fuel load estimates and, to a much lesser extent, pyrolysis efficiencies (Table 4). Fine fuel load estimates have decreased slightly in the EDS, but increased markedly for the LDS given additional leaf litter inputs. Other fuel load components have also generally declined in both seasons, with the exception of a marked decrease in heavy fuels; for a typical year (average of 2000–04), the 2004 methodology estimate of heavy fuel load is five times the current estimate. As noted in Table 4, this is due to the use of a different calculation method, especially the taking into account of the hollowness of most woody debris due to termite activity. We note, nevertheless, that woody debris is likely to be highly spatially variable owing to site productivity (i.e. related to stand basal area), past severity of fire regimes, and especially the influence of cyclonic and localised very strong wind events. In combination, these factors afford significant challenges for modelling.

Current estimates of pyrolysis efficiency have both increased and decreased, depending on the fuel component (Table 4). Importantly, however, LDS pyrolysis efficiencies for fine fuels, the source of the majority of CO_2-e emissions, have decreased substantially (from an assumed 100% in 2004 estimates, to 86% as measured; Table 1). The differences between the 2004 methodology and current parameters have a considerable effect on emissions estimates. For a typical year (average of 2000–04), estimates of CO_2-e emissions using the current parameters are 83% of those using the earlier 2004 parameters (Table 4; Fig. 4).

Uncertainty in emissions

Good practice guidance for GHG inventory preparation (IPCC 1997, 2006) requires that emission estimates include an assessment of their uncertainties. The determination of parameter values provides sufficient information on parameter uncertainty to support such an uncertainty analysis. The uncertainty analysis follows the approach used in the Australian NGGI. This is a Monte Carlo analysis in which activity and input parameter values are replaced by probability density functions (PDFs) and the probability distributions of the outputs following the Monte Carlo simulation. The 95% uncertainty ranges are defined by the 2.5 and 97.5 percentiles, which avoids any assumptions about the symmetry of the distributions.

The uncertainty model was based on algorithms described in Eqns 4 and 5. The PDFs are listed in Table 5. Mean fire severity, burning efficiency, carbon content, N : C ratio, and CH_4 and N_2O emission factors were fitted by normal distributions. Seasonal fire extent areas were fitted by log-normal distributions. Fuel loads were calculated using the regressions shown in Fig. 2 with the parameter values replaced by the appropriate normal distributions. Patchiness was not adequately described by either normal or log-normal distributions. Late season patchiness varied from 0.9 to 1.0, whereas the frequency distribution of early season patchiness was distributed between 0 and 1, increasing from a minimum at 0.3 to a maximum at 1 (Fig. 5). In the uncertainty analysis, both distributions were approximated by triangular PDFs. For the present study, all parameters were assumed to be independent; however, it is likely that correlations do exist between both parameters and strata. In future studies, these should be investigated, quantified and included in the analysis.

The total CO_2-e emissions and their uncertainties are presented in Table 6. Most of the emission (~65%) comes from LDS combustion of fine fuels, predominantly in eucalypt and sandstone woodland, with a smaller contribution (11%) from EDS fine fuels. The upper and lower 95% uncertainty limits for emissions from fine fuels are a factor of ~0.3 of the mean emission. However, this increases to a factor of ± 2.5 in the coarse and heavy fuels in the early fire season. Across vegetation classes,

Table 4. Summary of parameters used in emission estimates, based on data available from Russell-Smith *et al.* (2004), and current methodology as presented here

FLA, fuel load accumulation; *BEF*, burning efficiency factor. Average fuel loads (2000–04) and estimated emissions for the study area are also shown

Parameters and variables	2004	Current	Comments and refinements
(*a*) FLA			
Average fuel load (t ha^{-1})			
Early dry season			• Further fuels sampling at longer-unburnt sites (>5 years),
Fine fuels	4.60	4.16	especially for Open forest, Woodland, Sandstone woodland
Coarse fuels	1.12	0.69	• Further fuels sampling at sites both in early and late dry seasons,
Heavy fuels	7.72	1.35	especially to refine leaf litter inputs
Shrubs	0.83	0.62	• Given lack of relationship for both coarse and heavy fuel
Late dry season			components with time-since-burnt, explore relationships with
Fine fuels	4.40	5.95	vegetation density parameters (e.g. basal area, Foliage
Coarse fuels	0.99	0.71	Projective Cover)
Heavy fuels	8.29	1.45	• Large difference between heavy fuel estimates in 2004 and
Shrubs	0.75	0.84	current values due to different calculation method
Fire patchiness (proportion of area burnt)			
Early dry season	71%	71%	• Further sampling of fire patchiness, particularly in LDS
Late dry season	84%	89%	• Modelling of patchiness with rockiness and other terrain features, and fire severity
(*b*) *BEF*			
Pyrolysis efficiency (proportion of fuel pyrolysed)			
Early dry season			• Further post-fire combustion sampling effort with respect
Fine fuels	70%	74%	to fire severity
Coarse litter	3%	15%	• Ultimately, application of fire severity as derived from
Heavy litter	5%	17%	remote sensing to entire study region, rather than use
Shrubs	45%	29%	of seasonality as surrogate
Late dry season			
Fine fuels	100%	86%	
Coarse fuels	25%	36%	
Heavy fuels	25%	31%	
Shrubs	75%	39%	
Residual ash (proportion of consumed biomass)			
Entrained ash	6%	–	• Further assess post-fire ash deposition and ash entrainment,
Ash remaining *in situ*	–	11.6%	especially with respect to fire severity
(*c*) Carbon and nitrogen content			
Fuel carbon content	46% (for all fuels)		• Adequate sampling undertaken
Fine fuels		49%	
Coarse and heavy fuels		50%	
Shrubs		50%	
Fuel N : C ratio	0.0106 (for all fuels)		• Adequate sampling undertaken
Fine fuels		0.0101	
Coarse and heavy fuels		0.0081	
Shrubs		0.0093	
(*d*) Estimated emissions (t)			
CH_4	11 126	9772	• Assess relationships between emission factors, fire severity
N_2O	303	216	and fuel moisture (seasonality)
CO_2 equivalent	327 354	272 040	• Refine emissions assessment in light of undertaking further work on above parameters

uncertainty factors range from ±0.5 to ±0.7. Overall, these are substantially lower than the factor of ±2 for the savanna woodlands and arid grasslands reported in the Australian NGGI for 2006 (AGO 2007*a*).

The sensitivity of total CO_2-e emissions to individual parameters is determined from a multiple regression between inputs (parameters and activities) and the total emissions. The analysis, presented in Fig. 6, describes the relative change in total emissions caused by variation in the input parameters. This analysis shows that emissions are highly dependent on the heavy fuel loads, the early and late season fire patchiness, and the burning efficiencies, particularly of the fine and heavy fuels. Notably, uncertainty in fire area has little leverage on the total emissions, because it is assumed that fire scar area estimates between strata are independent, and consequently their variability tends to average out.

The total emission also appears relatively insensitive to uncertainty in emission factor (in this case for CH_4). However, this may be a consequence of a simplification in the inventory model, due to lack of data, which assigns an annual mean emission factor to

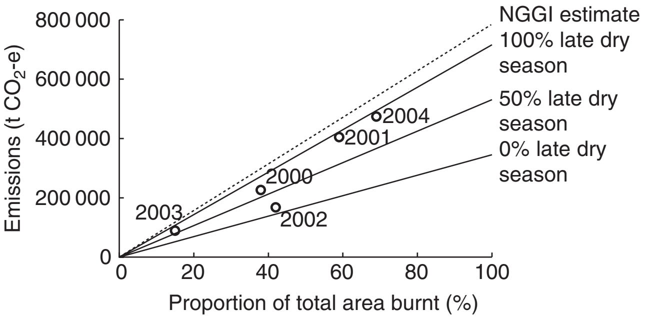

Fig. 4. Estimated CO_2 equivalent (CO_2-e) emissions from the Western Arnhem Land Fire Abatement (WALFA) study area in each year in the period 2000–04. The lines indicate the relationship between emissions and the proportion of the total area burnt, and the proportion of burning taking place during the late dry season. Emissions estimated using the current National Greenhouse Gas Inventory (NGGI) methodology (AGO 2007*b*) are also shown.

all strata. Emission factors probably vary with season, fuel class and fire severity and therefore, when stratified emission data become available, it is likely that greater sensitivity to emission factor will become apparent.

Discussion

Emissions estimates and uncertainties

The current study indicates that in a typical year (average of 2000–04), GHG emissions attributable to savanna burning from the WALFA project area were 272 Gg (Table 4) CO_2-e, with a 95% confidence range of $\sim\pm100$ Gg CO_2-e (Table 6). The confidence limit is based solely on the uncertainties of the parameter estimates, and it is assumed the inventory structure (i.e. the emissions model) is sound. This is a characteristic of all bottom-up inventories that are not constrained by independent measurements of emissions or emission surrogates such as concentrations of trace species such as CO or particulate matter. The reliability of the emissions estimates relies on the accuracy of the inventory model and the determination of the model parameters and the input (i.e. activity) data. Reliability is emerging as a significant issue among the growing number of bottom-up global and regional spatial inventories of biomass combustion emissions. For example, for the Australian region, the emissions estimates of Horowitz *et al*. (2003), Shirai *et al*. (2003), Hoelzemann *et al*. (2004), Ito and Penner (2004) and van der Werf *et al*. (2006) differ by up to a factor of 6, particularly owing to differences in estimation of fire area and fuel load (Kasischke and Penner 2004). Each of these estimates is considered by their authors to be reliable and the question, therefore, is: where do the differences arise? From our uncertainty assessment, the WALFA domain emissions should be accurate at the 95% confidence level to within a factor of 30–35% of the mean. Uncertainties associated with WALFA parameter estimates, and their broader applicability to other (especially Australian) savanna regions, are considered below.

Fire extent mapping. We consider this generally highly reliable for the WALFA region given the application of multi-temporal annual mapping based on fine-scale Landsat imagery (from 1990–present) and associated independent validation (Edwards and Russell-Smith 2009). However, apart from several regional fire histories derived from Landsat imagery, primarily for savanna conservation properties (e.g. Kakadu National Park), consistent (semi-automated) fire extent mapping at the Australian savanna-wide scale is available only from coarse-resolution National Oceanic and Atmospheric Administration (NOAA)-AVHRR imagery since 1997 (Craig *et al*. 2002), and more recently from MODIS (Moderate Resolution Imaging Spectroradiometer). An assessment of the accuracy of fire extent mapping in northern Australian savannas based on comparison with mapping from Landsat and associated field validation found that, apart from one Landsat scene, fire-mapping from AVHRR consistently under-estimated the 'true' extent of burning by as much as 10–20% (Yates and Russell-Smith 2002). Australia's NGGI currently uses the AVHRR-derived fire mapping product as the basis for calculations.

For regional GHG emission abatement project purposes, however, reliable estimation of seasonal fire extent needs to be based preferably on multi-temporal annual fire mapping using fine resolution imagery, or at least calibration of fire mapping from coarser-scale imagery against fine-scale imagery spatio-temporal subsets. For the latter, it is a requirement that the error of the coarse-scale mapping is predictable. This is difficult to achieve with the manual or semi-automated mapping currently used operationally. It requires the use of fully automated mapping that is independent of a human operator. Luckily, these fully automated mapping systems are becoming available (e.g. Roy *et al*. 2005). Some of these algorithms also provide confidence estimates, which would be important inputs for accuracy assessment. Beyond this, there is the potential to develop algorithms to estimate the fraction of a pixel affected by fire. So far, development of algorithms for mapping fire extent has focussed on binary ('hard') classification of each pixel using usually only the direction of change of reflectance. By using quantitative values of change of reflectance, there is the possibility to distinguish subclasses (at least classes like 'partly burnt' and 'completely burnt').

Vegetation structure–fuel type mapping. We also consider the fuel type cover to be reliable for the WALFA region based on robust digital canopy cover classification (1-ha scale) of Landsat imagery and other spatial data sources, and associated independent validation (Edwards and Russell-Smith 2009). By contrast, consistent vegetation mapping for Australian savannas is available at 1 : 2 000 000 (Fox *et al*. 2001), whereas the NGGI uses an 'agro-ecological' regionalisation comprising five zones (Meyer 2004; AGO 2007*b*). For regional GHG emission abatement project purposes, however, reliable vegetation–fuels mapping needs to be based on validated fine-scale vegetation structural mapping.

Fuel loads. Fine fuel load accumulation for different vegetation structural types in the WALFA study area was found to occur at broadly similar rates and in similar quantities to those reported in other comparable regional studies (Russell-Smith *et al*. 1998; Cook 2003). Given the frequency of fire in the WALFA area, mean fine fuels (4.16 t ha^{-1}; Table 4) were well below equilibrium (long-unburnt) levels observed in above studies. Although not addressed specifically by field sampling in the present study, it is widely observed that accumulation of fine fuels in savannas is substantially greater in the LDS,

Table 5. Probability density functions (PDFs) used in the Uncertainty Analysis

FL, fuel load; *BEF*, burning efficiency factor; EF, emission factor; EOF, eucalypt open forest; EW, eucalypt woodland; SH, sandstone heath; SW, sandstone woodland; a0 and a1 are the first two respective parameters of expression describing fuel accumulation in Fig. 2, see text for further details. CV, coefficient of variation; s.e., standard error of the mean

Parameter		Unit	PDF	Min.	Max.	Most likely	CV or s.e.	
Area		ha	Lognormal	–	–	Annual activity	0.1	CV
Patchiness	Early		Triangular	0	1	1		
	Late		Triangular	0.8	1	1		
Fuel load	Fine-EOF	a0	Normal			1.22	0.06	s.e.
$\ln(FL) = a0 + a1 \times t$	Fine-EW	a0	Normal			1.15	0.07	s.e.
	Fine-SH	a0	Normal			0.77	0.10	s.e.
	Fine-SW	a0	Normal			1.11	0.06	s.e.
	Shrub-SW	a0	Normal			0.36	0.11	s.e.
	Fine-EOF	a1	Normal			0.26	0.03	s.e.
	Fine-EW	a1	Normal			0.27	0.04	s.e.
	Fine-SH	a1	Normal			0.60	0.06	s.e.
	Fine-SW	a1	Normal			0.48	0.03	s.e.
	Shrub-SW	a1	Normal			0.09	0.04	s.e.
Mass	Coarse-EOF	$t\ ha^{-1}$	Log-normal			1.43	0.07	CV
	Coarse-EW	$t\ ha^{-1}$	Log-normal			0.90	0.11	CV
	Coarse-SH	$t\ ha^{-1}$	Log-normal			0.58	0.13	CV
	Coarse-SW	$t\ ha^{-1}$	Log-normal			1.23	0.14	CV
	Heavy-EOF	$t\ ha^{-1}$	Log-normal			4.81	0.11	CV
	Heavy-EW	$t\ ha^{-1}$	Log-normal			2.18	0.12	CV
	Heavy-SH	$t\ ha^{-1}$	Log-normal			1.68	0.14	CV
	Heavy-SW	$t\ ha^{-1}$	Log-normal			3.42	0.15	CV
	Shrub-EOF	$t\ ha^{-1}$	Log-normal			1.46	0.76	CV
	Shrub-EW	$t\ ha^{-1}$	Log-normal			0.49	2.28	CV
	Shrub-SH	$t\ ha^{-1}$	Log-normal			1.77	0.64	CV
BEF								
Low severity	Fine		Logistic			0.69	0.06	CV
	Coarse		Logistic			0.06	0.49	CV
	Heavy		Logistic			0.12	0.07	CV
	Shrub		Logistic			0.27	0.20	CV
Moderate severity	Fine		Logistic			0.84	0.07	CV
	Coarse		Logistic			0.34	0.36	CV
	Heavy		Logistic			0.20	0.38	CV
	Shrub		Logistic			0.24	0.09	CV
High severity	Fine		Logistic			0.97	0.003	CV
	Coarse		Logistic			0.58	0.06	CV
	Heavy		Logistic			0.58	0.11	CV
	Shrub		Logistic			0.68	0.06	CV
Fire severity distribution								
Early season	Low		Truncated normal	0	1	0.722	0.20	CV
	Moderate		Truncated normal	0	1	0.198	0.20	CV
	High		Truncated normal	0	1	0.079	0.20	CV
Late season	Low		Truncated normal	0	1	0.187	0.20	CV
	Moderate		Truncated normal	0	1	0.473	0.20	CV
	High		Truncated normal	0	1	0.341	0.20	CV
Carbon content	Coarse		Normal	0	1	0.49	0.02	CV
	Coarse		Normal	0	1	0.50	0.02	CV
	Heavy		Normal	0	1	0.50	0.02	CV
	Shrub		Normal	0	1	0.50	0.02	CV
N : C ratio	Coarse		Normal	0		0.0096	0.1	CV
	Coarse		Normal	0		0.0081	0.1	CV
	Heavy		Normal	0		0.0081	0.1	CV
	Shrub		Normal	0		0.0093	0.1	CV
EF	CH_4		Normal	0		0.0035	0.02	CV
	N_2O		Normal	0		0.0076	0.02	CV

associated with increased litter (leaf, twig) drop as the dry season progresses (e.g. Hopkins 1966; Williams *et al*. 1998; Cook 2003; Hennenberg *et al*. 2006). Cook (2003) has shown that the amount of litter drop is proportional to stand basal area in regional savannas. Kauffman *et al*. (1994) also note the diminishing contribution of grassy fuels to fine fuel loads along a gradient of increasing woody biomass in Brazil. We contend that estimation of fine fuel loads in different regional vegetation structure types, and under different environmental settings (e.g. rainfall and soil conditions) can be described satisfactorily by simple relationships. As noted previously, fine fuels provide the major contribution to emissions from savanna burning in the WALFA region, and presumably for Australian savannas generally.

Conversely, statistically significant relationships describing accumulation of other fuel components (coarse, heavy, shrub fuels) were not apparent in our assembled data with the exception of Sandstone woodland, and then at much reduced explanatory power relative to fine fuels (Fig. 2). The overall contribution of coarse and heavy fuels combined ($\sim 2\,t\,ha^{-1}$; Table 4) is higher than that reported generally in African and South American studies (Kauffman *et al*. 1994; Shea *et al*. 1996; Barbosa and Fearnside 2005), but substantially less than the $4.4\,t\,ha^{-1}$ reported by Rose (2006) from an assessment of heavy fuel accumulation in Australian eucalypt savanna open-forest. This latter study also found no clear relationship between heavy fuel accumulation with time since fire. Rather, other observations would suggest that both dead standing stems and heavy ground fuels are more likely to accumulate in response to severe fires or fire regimes (Williams *et al*. 1998; Cook *et al*. 2005), as well as (probably highly significant) effects of localised and subregional strong windy events (e.g. pre-monsoon storms, cyclones). GHG emissions from burning of tropical forests are, conversely, far more dependent on combustion of heavy fuels (Pereira *et al*. 1999).

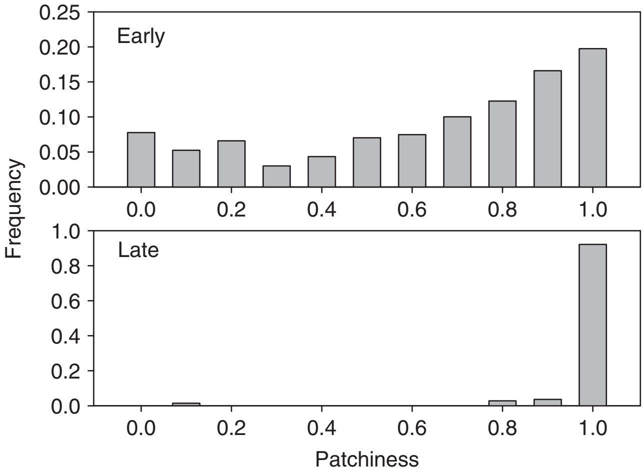

Fig. 5. Frequency distribution of fire patchiness during early and late dry seasons.

The variable lack of response exhibited by shrub fuel accumulation with time is also likely to reflect very substantial intersite variability in historical processes (Russell-Smith *et al*. 1998). In the absence of simple predictive relationships describing accumulation in these fuel components, we have assumed that, at large spatial scales, arithmetic means are the most useful descriptors.

Burning efficiency factors. In Australian savanna studies, fire patchiness is recognised to reflect aspects of seasonality, fire severity and rockiness (Price *et al*. 2003; Williams *et al*. 2003) and, with further data collection (see Table 4), is potentially readily modelled. More limited and uncertain data are available, however, for post-fire assessments of fuel consumption and aspects associated with ash entrainment and retention.

In the present study, consumption of different fuel components was found to be dependent on fire severity. Based on the observed distribution of mild, moderate and severe fires throughout the year, we estimated mean consumption of fine fuels to be 74% in the EDS and 86% in the LDS, and 29 and 39% for heavy fuels in respective seasons (Table 2). Studies elsewhere report consumption of savanna grassland and woodland fine fuels ranging generally between ~90 and 100% (Kauffman *et al*. 1994; Shea *et al*. 1996; Ito and Penner 2004; Barbosa and Fearnside 2005), although Hoffa *et al*. (1999) note that, in Zambian grassland, pyrolysis increases substantially as the dry season progresses (from ~50% in early June to 90% in August). Greater variability is evident in consumption of heavy fuels; for example, with reported values ranging from 9 to 31% in studies presented by Kauffman *et al*. (1994), Shea *et al*. (1996), Ito and Penner (2004), Barbosa and Fearnside (2005), Rose (2006). Clearly, for modelling purposes, the effects of fire severity and

Table 6. The 95% confidence ranges for total CO_2 equivalent (CO_2-e) emissions for 2000–04, for respective fuel components

Note that the range in the total uncertainty estimate is not symmetrical around the mean emissions estimate of 272 Gg CO_2-e (as presented in Table 4), given various transformations for determining Probability Distribution Functions as described in Table 5. CV, coefficient of variation

Vegetation class	Emission (Gg CO_2-e)									
	Early Season				Late Season				Total	
	Fine	Coarse	Heavy	Shrub	Fine	Coarse	Heavy	Shrub		CV
Eucalypt Open Forest	0.6–3.9	0.0–0.6	0.0–0.1	0.0–1.1	10–19	0–4	0–15	0–7	**16–38**	0.25
Eucalypt Woodland	5.8–40.8	0.1–5.1	0.0–0.0	0.3–8.6	65–138	2–22	3–60	1–16	**108–228**	0.20
Sandstone Heath	0.9–6.5	0.0–0.5	0.0–0.0	0.0–0.9	27–57	0–6	0–19	0–26	**38–88**	0.23
Sandstone Woodland	0.7–5.1	0.0–0.4	0.0–0.0	0.1–0.6	39–72	2–6	4–11	4–8	**56–96**	0.13
Uncertainty	**8.0–56.2**	**0.2–5.8**	**0.0–0.1**	**0.5–9.8**	**142–284**	**6–31**	**11–85**	**8–45**	**225–424**	
CV	0.4	0.8	1.2	0.7	0.2	0.5	0.7	0.5	0.16	

seasonal variation in fuel moisture need to be more consistently described.

We did not directly measure the production of entrained ash during a fire, i.e. ash transported away from the site of a fire within a smoke plume. However, the results of Cook (1994) suggest that during lowland woodland fires in Kakadu National Park, entrained ash represents between ~3 and 9% of consumed biomass. However, ash that remains *in situ* following a fire will comprise at least some entrained ash that has resettled from surrounding areas. Although we do not account for entrained ash specifically in the current study, it is consistent with the methodology that has been used to calculate pyrolysis efficiency elsewhere; namely, entrained ash is generally not accounted for (e.g. Kauffman *et al*. 1994; McNaughton *et al*. 1998). A further important factor not accounted for is the effect of fire severity on the production of ash, given that severe fires are likely to result in more efficient volatilisation of consumed biomass (Stronach and McNaughton 1989). For present purposes, we have assumed that residual ash encompasses entrained ash redeposited from nearby sources. With these qualifications in mind, we note that the *in situ* ash conversion rate calculated here (11.6%; Table 4) is generally consistent with studies from Kakadu National Park (7%: Cook 1994) and other tropical savanna and forest ecosystems, which typically range from ~3% (e.g. Kauffman *et al*. 2002; Barbosa and Fearnside 2005) to 11% (e.g. Kauffman *et al*. 1994). Clearly, issues concerning ash production and entrainment require further research.

Emission factors. Although most above parameters have been evaluated for all or most strata, the EFs are taken from the Australian NGGI methodology. These are annual averages measured in Kakadu National Park by Hurst *et al*. (1994*a*, 1994*b*) using grab samples of air collected in flasks either at ground level or from aircraft. Although they sampled both early and late dry season fires, the variability between samples collected within fires was greater than the differences between fires and they were unable to draw any conclusions about seasonal variation in EFs. This is a standard problem when sampling heterogeneous plumes. A second series of airborne and ground-based measurements made in 1999 during the late fire season by Shirai *et al*. (2003) closely support the results of Hurst *et al*. (1994*a*, 1994*b*). The Australian CO EFs (~0.05 to 0.08) are slightly higher compared with 0.05 for other savanna regions (Andreae and Merlet 2001), and indicate a small bias towards mixed and smouldering combustion, probably associated with the higher fraction of tree-leaf litter in the fine fuels than occurs in the grass-dominated savannas on other continents. The Australian savanna CH_4 emission factors, however, are similar to those reported from southern Africa but higher than South America (Andreae and Merlet 2001; Shirai *et al*. 2003).

The CO emission factor is closely related to Modified Combustion Efficiency ($MCE = [CO_2]/([CO] + [CO_2])$), which is strongly correlated with the EF for trace species (Hao *et al*. 1996; Ward *et al*. 1996). MCE varies with fuel class; grasses tend to have high MCE (~0.95), whereas forest fuels have lower MCE (~0.9: Ward *et al*. 1996; Andreae and Merlet 2001) and therefore the fuel class mixture is likely to be a determinant of CH_4 emission as is fuel moisture. The relationship between MCE and EF was exploited by Korontzi *et al*. (2003, 2004) to explore the seasonality of emissions from fires in southern Africa. These authors found early season fires in southern Africa tend to have lower MCE. Hence methane emission is likely to be disproportionately higher in the early fire season than indicated by fire area; in fact, it was suggested that the bulk of CH_4 emissions might come from early season fires (Korontzi 2005). In contrast, NO_x emissions are positively correlated with MCE and therefore are likely to be disproportionately weighted towards late season fires. The relationship between MCE and N_2O emissions is less clear although there is some indication that it is weighted to high MCE. This might also apply in Australia; however, given that the more severe fires of the LDS are more likely

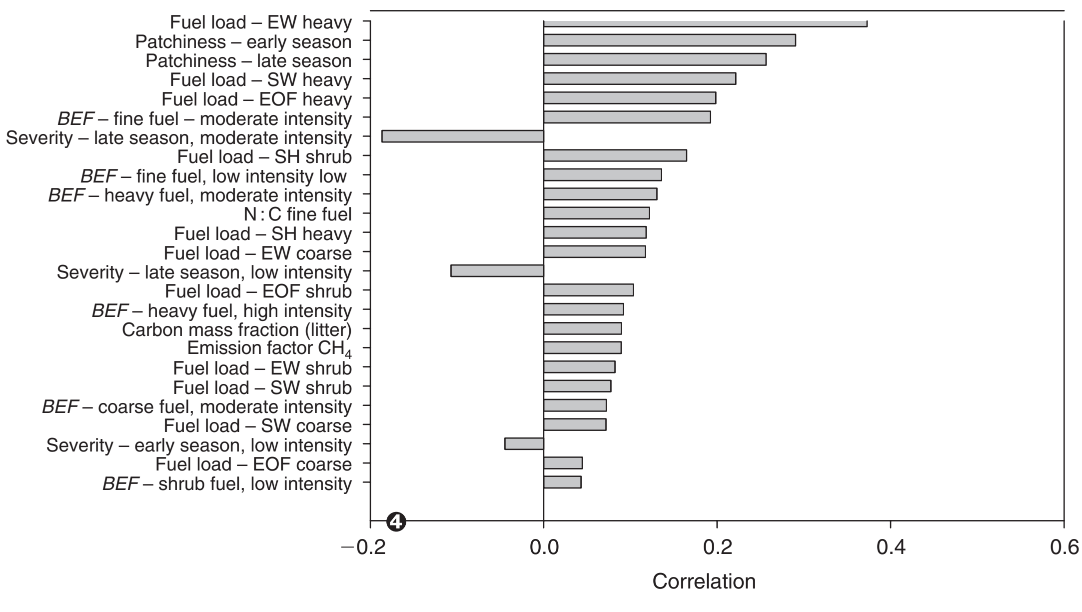

Fig. 6. The sensitivity of total greenhouse gas emissions (CO_2 equivalent, CO_2-e) to input parameters.

to burn shrubs, coarse and heavy fuels, which will burn with a greater proportion of smouldering combustion (Andreae and Merlet 2001), the EFs of these reduced species are likely to be greater. Hurst *et al.* (1994*a*) found that smouldering combustion in north Australian savannas produced substantially more CO and CH_4 than flaming combustion.

It should be noted that, to date, all analyses of seasonality rely on derived relationships between a greenness index that describes the moisture content of the fuel and MCE; there are currently no comprehensive measurements of the seasonality of emissions composition. However, it is likely that interaction between the mix of grass, forest litter, coarse and heavy fuels, and fuel moisture on MCE is expected to produce EF seasonality that is regionally specific. In Australian savanna, fuels tend to be fully cured before burning commences in the EDS and therefore the strong seasonality reported by Korontzi (2005) probably does not occur to the same extent in Australia. However, quantifying the seasonality of emission factors is obviously a priority for improving emissions accounting.

Bottom-up and top-down approaches

Although verification of a bottom-up inventory model structure and parameters lends confidence to the accuracy of model predictions, independent validation is required to prove them. Validation of emissions at regional or larger scales is a major challenge, seldom undertaken, and, to date, has given mixed results. A study of emissions in northern Australia used an emission model coupled with a dispersion model to calculate regional hourly concentrations of particulate matter $\leq 2.5\,\mu m$ in size (PM2.5) produced by biomass burning throughout 2004, and found good agreement between observed and predicted concentrations in Darwin, and observed and predicted aerosol optical depth (AOD) at two locations (Luhar *et al.* 2008; Meyer *et al.* 2008). A similar approach involving coupling the Global Fire Emissions Database (GFED) model (Giglio *et al.* 2006) to a Lagrangian dispersion model was used by Saarikoski *et al.* (2007) to evaluate the contribution of several fire events in northern Europe to local air quality with partial success. However, studies using CO concentrations from MOPPITT (Measurement of Pollution in the Troposphere sensor on NASAs TERRA satellite) to estimate biomass burning emissions in southern Africa using atmospheric inversion techniques (Arellano *et al.* 2004; Petron *et al.* 2004) concluded that bottom-up inventories substantially underestimate total emissions. These studies also found substantial differences in the seasonality of emissions from the inventory predictions.

What then are the options for improving the accuracy of emissions accounting? Looking from the bottom up, there is an increasing requirement to define the dependence of GHG emissions on fuel class, fuel properties and seasonality. This can be achieved to a degree through field measurement, but there are alternatives. Increasingly, data assimilation techniques are being used to constrain weather model predictions with observations both at the land surface and from satellites. Similar approaches are used to constrain parameters in land surface models (e.g. Wang and Barrett 2003; Barrett *et al.* 2005). Data that could be used to constrain emissions models include: satellite measurements of fire severity (Smith and Wooster 2005; Smith *et al.* 2005); vegetation curing index; fuel moisture estimates (e.g. Hao and Qu 2007; Verbesselt *et al.* 2007); and both surface and satellite observations of aerosol and CO concentration.

Fire severity can be used directly to predict aerosol and gaseous carbon and nitrogen emission rates while curing and greenness indices, as well as fuel moisture estimates, can be used to estimate MCE and hence EF for a wide range of trace species (Hao *et al.* 1996; Korontzi *et al.* 2004). There is evidence that fuel type (cured dead grasses *v.* green alive shrubs or trees) can be derived from satellite remote sensing (Maier, in press). Finally, measured concentration fields from satellite or ground-based approaches can be used to constrain both emission estimates and model parameters.

Satellite-based thermal measurements of active fires (fire radiative power) provide another independent estimate of fuel combustion rates (Wooster *et al.* 2005). These can be used for calibration or validation of the other approaches, although the method still needs to be tested for fires in Australia.

The WALFA offset model

The underlying premise of WALFA as a GHG emissions offset project is that substantial annual emissions abatement can be achieved, and quantified, through the implementation of strategic prescribed fire management and associated monitoring and accounting. As described previously, the vast WALFA project area is a remote, rugged, biodiversity-rich, today mostly unpopulated landscape where, between 1990 and 2005, an average of 36% p.a. of the region was burnt, predominantly by LDS wildfires. It is the contention of project partners that reinstatement of strategic fire management applying both traditional knowledge and using contemporary practices (e.g. aerial prescribed burning) and tools (e.g. GIS for project planning, implementation and monitoring purposes) can (1) substantially reduce GHG emissions associated with wildfires; (2) help address chronic contemporary fire regime impacts on biodiversity values; and (3) provide culturally appropriate employment opportunities for regional indigenous communities.

Achieving significant GHG emissions abatement requires first the undertaking of a prescribed burning program that is strategic in its implementation to reduce the current annual extent of wildfire. To that end, project partners have commenced critical assessment of using landscape features (watercourses, terrain, tracks) and aerial burning to deliver more effective management (Price *et al.* 2007). Edwards *et al.* (2003) document the reduction in area burnt using a more strategic management approach in the adjoining 20 000 km^2 Kakadu National Park, from a mean of 45% for 1980–95 compared with 40% for the period 1996–2000.

Associated substantial emissions reductions can be achieved also through implementing a fire management program that focusses on delivering strategic burning in the EDS. As detailed in the present study, burning in the EDS provides increased patchiness and reduced fuel consumption associated with typically less severe fires. Overall, our data allow us to estimate that emissions from early season fires typically emit 48% of the emissions of late season fires per unit area (compare lines for 0% and 100% LDS fires in Fig. 4). An assessment of the fire history of Kakadu National Park, 1980–94 (covering the 15-year period after its creation), found a pronounced shift from a fire regime dominated by LDS fires until the mid-1980s, to

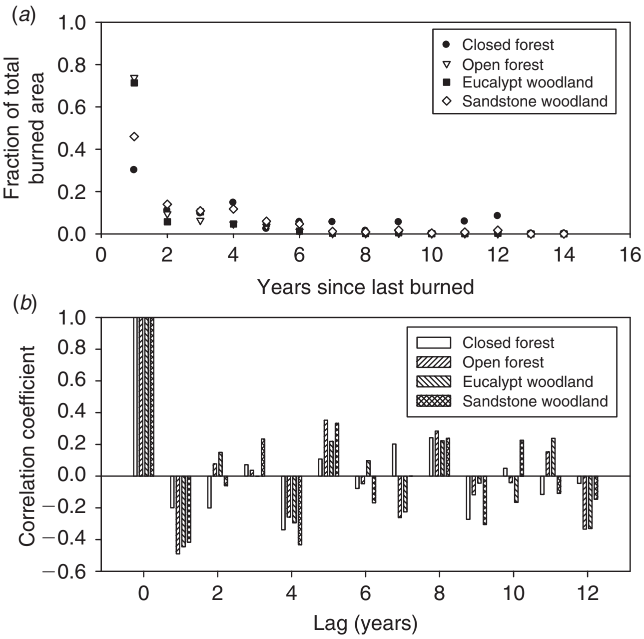

Fig. 7. Analysis of fire regimes in the Western Arnhem Land Fire Abatement (WALFA) domain: (*a*) fire return interval for WALFA vegetation classes; (*b*) the autocorrelation functions of annual fire area in each of the vegetation classes.

one dominated by EDS fires subsequently (Russell-Smith *et al.* 1997*b*). In the more recent fire history assessment undertaken by Edwards *et al.* (2003; see above) for Kakadu, the 5% reduction achieved in annual burning the second 5-year period compared with the first 15 years was attributable predominantly to reduction in late season fires. The Kakadu example illustrates that, with adequate resources, it is feasible to implement a more benign, GHG-friendly fire regime.

The final issue that we canvass here concerns determining the length of the pre-project (business-as-usual) emission baseline against which WALFA (and other similar savanna fire abatement projects) need to measure their abatement outcomes. This is particularly pertinent given that current IPCC guidelines (IPCC 1997, 2000) do not specify a temporal baseline horizon. If the annual baseline emission has substantial interannual variability, then ideally the sources of the variability, typically climatic drivers, should be determined and their impacts quantified, so that their effects can be separated from the management impacts. Although at regional scale, the impacts of climate may be apparent (e.g. Meyer 2004), at fine spatial scale, climate impacts are often swamped by other sources of variability and therefore cannot easily be quantified (Russell-Smith *et al.* 2004). However, climate trends typically occur on a timescale of decades, and therefore shorter-term interannual variation can often be treated as random noise around a mean. The issue to resolve is the length of an appropriate time window that averages out within fire-cycle variation, without removing slow trends due to climate shifts.

The fire regimes in the vegetation strata of the WALFA domain can be characterised by their fire return frequency. Fig. 7*a* shows the distribution of fire return frequency in the area burned in 2005. In all cases, the median return period is 13 years or less, and in all classes more than 85% of the area was reburnt less than 10 year previously. In Eucalypt woodland, Sandstone woodland and Open forest, the median return period is closer to 2 years. The short return period is reflected in the autocorrelation functions of the fire history (Fig. 7*b*), which show negative correlations at lags of 1, 4, 7, etc., years, which are presumably consistent with the time required to rebuild the fuel loads in the absence of prescribed strategic fire management. An averaging period of at least 10 years (i.e. covering several fuel accumulation–fire cycles) is required to define an emissions baseline that is not compromised by natural variability. The established WALFA emissions baseline covers the pre-project decadal period, 1995–2004.

Conclusion

The regional GHG emissions inventory methodology presented here is shown to be generally robust, delivering emissions estimates congruent with those of Australia's NGGI. The regional methodology is enhanced by the explicit incorporation of terms for fire seasonality and severity. Addressing reduction in estimate uncertainties requires further work, especially on coarse and heavy fuels, and seasonality and severity components of burning efficiency (patchiness, ash retention and entrainment) and EFs. More generally, emission estimates derived from this project-based, bottom-up approach need to be qualified and constrained with respect to independent top-down assessments, for example, taking advantage of measurements of CO and aerosol particulates derived from satellite-based sensors coupled with dispersion modelling. With regionally specific adaptations of emissions inventory parameters, the developed methodology has significant potential for emissions accounting of savanna burning offset and abatement projects in other regions. Such opportunities would appear to exist especially in situations where there is a need to implement more strategic burning to reduce anthropogenically dominated, LDS emissions.

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