

# Monitoring standing dead wood for carbon accounting in tropical savanna

R. J. Fensham

Queensland Herbarium, Environmental Protection Agency, Mt Coot-tha Road, Toowong, Qld 4066, Australia.  
Email: Rod.Fensham@epa.qld.gov.au

**Abstract.** Changes in the biomass of dead wood can be substantial in tropical woodland and should be included in complete carbon budgets. A recent estimate of biomass changes from long-term, permanent monitoring sites within the eucalypt woodlands of Queensland suggests that carbon accumulated in dead standing wood is similar to the amount of carbon accumulated within live woody biomass. However, this assessment did not include trees that were dead at the commencement of monitoring but have since been burnt or fallen over. Original data are presented from a permanent monitoring site, suggesting that on average 27% of dead standing biomass is consumed per low-intensity fire. The consumption of standing dead wood by fire, together with likely ‘fall-down rates’, could offset a substantial proportion of recently estimated dead standing wood carbon sink. The potential for using existing allometric equations to ascribe biomass to the components of dead standing wood (trunks, branches, bark) on permanent monitoring plots is reviewed. Some valid generalisations can be recognised but a lack of standardisation in allometric studies presents problems. It is suggested that data from survey, monitoring and experiments could drive worthwhile models of dead standing wood biomass. This pool will probably vary substantially through time and space, depending on climatic conditions, fire and termite activity.

## Introduction

Auditing changes in the biomass of woody vegetation is an important component of carbon budgets. Most studies have concentrated on live woody biomass where it can be derived from basal area by using harvest-based allometric equations (see Williams *et al.* 2005). The carbon stocks in dead standing wood may not be stable and may be substantial in some situations. In semi-arid savanna for example drought-induced dieback can result in sudden and substantial conversion of live tree to dead standing wood carbon (Viljoen 1995; Allen and Breshears 1998; Fensham *et al.* 2003). Fensham and Holman’s (1999) survey suggests that ~25% of live basal area was converted to dead standing wood during 5 years of intense drought over a large area of northern Queensland in the 1990s. The killing of trees by ringbarking or poisoning to enhance pasture growth is also common practice in some areas of Queensland. The fate of the dead standing wood resulting from natural dieback or tree killing is unclear, although in tropical savannas the frequency of burning is likely to be a critical factor. The replacement of Aboriginal burning with pastoral management may have resulted in both a lower frequency and intensity of burning in some environments (Russell-Smith *et al.* 2003). This will logically result in a larger pool of carbon bound within dead wood. Thus, dead standing wood is a potentially major pool of carbon in savannas, and methods to account for changes in this pool need to be developed. This paper, then, focuses on

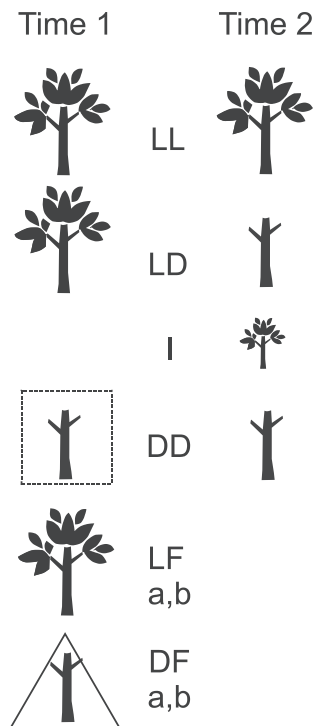
the fate of dead standing wood, commencing with a review of the only attempt to audit changes in this carbon pool in savanna. The data from permanent monitoring plots are reinterpreted in light of original data on consumption by fire and fall down rates affected by decomposition as reviewed from the literature. The potential for integrating existing allometry and permanent monitoring for a more robust assessment of dead standing wood is also reviewed and other suggestions for improving our understanding of these changes in savanna provided.

## A recent estimate of dead-wood carbon-biomass change

The only published attempt to audit changes in dead standing wood in tropical savanna is the recent study by Burrows *et al.* (2002) from Queensland, Australia. This was a component of estimates of carbon sequestration in relation to standing biomass in grazed eucalypt woodlands derived from a network of permanent monitoring sites (TRAPS; Back *et al.* 1999), including 30 long-term (average monitoring period = 14.14 years) and 27 short-term (average monitoring period = 2.05 years) sites. Standing dead wood was not measured at the initial monitoring, but the biomass of the initial standing dead was estimated from the biomass of the standing dead at the final monitoring by applying a negative decay function to allow for loss through decomposition during the course of monitoring. Despite biomass loss

from decomposition, positive values for standing dead wood change overall were obtained because of the addition of death of live trees to the final dead standing wood pool through the death of live trees (Fig. 1).

Determining the contribution of dead standing wood to rates of change in of the total carbon pool requires analysis of the data presented for the initial and final carbon pools in table 1 of Burrows *et al.* (2002). By using this data, increases in carbon stocks in dead standing wood are approximated here as 40.1 and 57.5% of the total increase in carbon stocks estimated by Burrows *et al.* (2002) from the long-term and short-term permanent monitoring sites, respectively (Table 1). These substantial contributions compare with the proportion of dead standing wood in the standing biomass pools when they were accurately accounted for at  $t_2$  of 7.1 and 11.0% for the short- and long-term sites, respectively (Table 1). The total average annual carbon increment calculated by Burrows *et al.* (2002) is very



**Fig. 1.** Representation of the biomass pools necessary for a balanced analysis of standing live and dead wood carbon pools in woodland trees between Time 1 and Time 2. LL, trees that are live at the Time 1 and Time 2; LD, trees that are live at Time 1, and dead standing wood at Time 2; I, trees not present at Time 1 that have developed as ingrowth by Time 2; DD, dead standing wood at Time 1 that is also dead standing wood at Time 2; LF, trees that were live at Time 1 but were not standing at Time 2 (a, burnt; b, fallen); DF, dead standing wood at Time 1 not standing at Time 2 (a, burnt; b, fallen). The dotted box identifies the portion of the pool that was not measured but inferred by using a decay function from those DD trees measured at Time 2. The triangle identifies a pool of trees that were omitted from the analysis of Burrows *et al.* (2002).

sensitive to the derivation of substantial accumulation of dead standing wood biomass at the monitoring sites over the course of monitoring. This has been approximated in Table 1 as  $\sim 0.16 \text{ t C ha}^{-1} \text{ year}^{-1}$  and  $0.45 \text{ t C ha}^{-1} \text{ year}^{-1}$  for long- and short-term sites, respectively.

It is suggested here that there is a missing standing dead wood pool in the Burrows *et al.* (2002) estimate. The dead standing wood present at the initial measurement that has been burnt subsequently, or fallen to the ground, before the final measurement (DF, Fig. 1), appears not to have been included. This imbalance will result in underestimates of the biomass of standing dead wood at the initial measurement, and therefore overestimates of the rates of change in carbon pools over the course of monitoring. Thus, there is a need to improve estimates of this important pool of carbon, which this paper aims to address.

### Dead standing wood lost to fire and decay

To assess the potential effect of fire on the DF pool in Fig. 1, data from a 1.56-ha permanent monitoring site situated in *Eucalyptus* savanna in northern Queensland were analysed. The site and monitoring methodology are described in Fensham *et al.* (2003) and Fensham and Butler (2004). Stags were mapped, the diameter at breast height measured and permanently tagged. The monitoring site was established in 1996 after a severe drought between 1992 and 1996 resulted in considerable drought-induced mortality of the dominant eucalypt trees (mostly *Eucalyptus xanthoclada*, a close relative of *E. crebra*). In 1996, there were 45 recently dead individuals ( $>3 \text{ m}$  high) in the plot constituting  $27.01 \text{ t C ha}^{-1}$ . The biomass of these recently dead trees was calculated by using the allometric equations provided in Burrows *et al.* (2000), assuming 1-year's decomposition according to the decay function provided in Burrows *et al.* (2002) identifying a loss of 9.9% of dry weight. This coincides with the biomass of leaves, buds and  $\sim 12\%$  of the branches relative to total above-ground dry weight as indicated for *E. crebra* woodland (table 3 in Burrows *et al.* 2002). The loss of standing dead-wood biomass was then assessed following four low-intensity fires between 1996 and 2000 (Bryam fire-line intensity less than  $900 \text{ kW m}^{-1}$  in all cases). Monitoring of live plants was conducted after each fire but dead standing wood was not accurately recorded until after the fourth fire in 2001. All stags lost from the dead standing wood pool between 1996 and 2001 were known to have been incinerated.

The biomass of the dead standing wood that was present in 1996 after four fires was  $7.65 \text{ t C ha}^{-1}$  (i.e. assuming no further decomposition after the first year). Thus, the proportion burnt by the individual fires can be calculated as 27.1% of the dead standing wood biomass on the site before each of the individual fires (i.e.  $27.01 - (0.271 \times 27.01) = 19.70 - (0.271 \times 19.70) = 14.37 - (0.271 \times 14.37) = 10.48 - (0.271 \times 10.48) = 7.65$ ).

Table 1. Analysis providing approximations of the contribution of dead standing wood to total carbon-stock change rates from initial and final carbon pools presented in table 1, Burrows *et al.* (2002)

Average annual carbon increment calculated as average final C biomass – average C initial biomass/average time period, and estimated for all sites by using averages weighted according to site numbers (long-term sites = 30, short-term sites = 27). Analyses are only approximate because the average annual carbon-change rate is not derived for each site and then averaged across sites as per the Burrows *et al.* (2002) analysis. The approximations presented in this table seem reasonably robust for the long-term sites, compared with the available data in Burrows *et al.* (2002) (total average annual carbon increment 0.39 t C ha<sup>-1</sup> year<sup>-1</sup> compared with 0.37 t C ha<sup>-1</sup> year<sup>-1</sup>), but less robust for the short-term sites (total average annual carbon increment 0.78 t C ha<sup>-1</sup> year<sup>-1</sup> compared with 0.70 t C ha<sup>-1</sup> year<sup>-1</sup>)

Parameter	Long-term sites (Average $(t_2 - t_1) = 14.14$ years)			Short-term sites (Average $(t_2 - t_1) = 2.05$ years)			All sites Average annual carbon increment (t C ha <sup>-1</sup> year <sup>-1</sup> )
	$t_1$ (t C ha <sup>-1</sup> )	$t_2$ (t C ha <sup>-1</sup> )	Average annual carbon increment (t C ha <sup>-1</sup> year <sup>-1</sup> )	$t_1$ (t C ha <sup>-1</sup> )	$t_2$ (t C ha <sup>-1</sup> )	Average annual carbon increment (t C ha <sup>-1</sup> year <sup>-1</sup> )	
Total live	37.28	40.61	0.24	33.65	34.33	0.33	0.28
Total dead	0.88	3.11	0.16	3.33	4.25	0.45	0.30
Total	38.16	43.72	0.39	36.98	38.58	0.78	0.58
Average percentage dead (Total dead/total)	2.3	7.1	40.1	9.0	11.0	57.5	51.3

As suggested earlier, the exclusion of dead standing wood that has been burnt or fallen to the ground before the final measurement (DF, Fig. 1) will result in an overestimation of dead standing wood change rates. It is possible to use the data from the long-term TRAPS monitoring sites and available estimates of fire frequency, and estimates of the rates of dead standing wood lost to fire and decay to provide a coarse indication of the magnitude of this overestimation. This analysis is not intended to be a definitive description of the fate of dead wood in woodlands across Queensland or within the sites covered by the TRAPS network, which would require far more comprehensive data (see below). However, an indicative analysis, designed to improve the estimates of both pools and rates of change of standing dead wood, is warranted. Such an analysis is presented here and commences with the assumption that there is no net change in dead standing wood, i.e. that the initial unmeasured standing dead wood is identical to the final measured standing dead wood averaged across sites ( $3.11 \text{ t C ha}^{-1}$ ).

A component of the initial dead-wood pool will remain at the final measurement (DD in Fig. 1). Burrows *et al.* (2002) estimated this component as  $0.88 \text{ t C ha}^{-1}$  by using decay functions applied to the DD component as it was measured at the final monitoring. A second unmeasured component of the initial dead standing wood pool is removed if the plots are subject to fires (DFa, Fig. 1). Fensham and Fairfax (2003) developed a land-management history from a land-manager survey, which suggested that the average fire frequency for uncleared vegetation in central Queensland was about one fire per decade in the 1990s decade and that the proportion of high intensity (resulting in tree crown scorch) and low intensity fires was about equal. The estimate of fire frequency was based on 76 respondents describing the fire frequency for randomly located points they managed within  $\sim 7$  million hectares of central Queensland. The remotely sensed fire mapping for Queensland presented in Russell-Smith *et al.* (2003) suggests a decreasing trend in fire frequency from north to south.

Assuming the fire frequency from the land-manager survey in central Queensland is indicative for the long-term TRAPS sites and the loss of dead standing wood occurs at the rate quantified above for the northern Queensland monitoring site (see above), an estimate of the DFa pool (Fig. 1) can be made. The average number of fires during the average period of the long-term TRAPS sites (14.14 years) is 1.4 fires, representing a loss of  $1 \times 27.1\% + [(0.4 \times 27.1\%) \times (100 - 27.1\%)] = 35.0\%$  of the standing dead-wood biomass that would on average be consumed by fires. With the assumption of no net change in dead standing wood, loss of carbon through combustion in fires over 14.14 years is estimated as  $1.09 \text{ t C ha}^{-1}$  (35% of  $3.11 \text{ t C ha}^{-1}$ ).

Some dead standing wood will fall to the ground during the course of monitoring (DFb in Fig. 1). There is no available data to estimate the size of this component in relation to

tropical savanna, but some data on stag 'fall-down rates' are available from long-term studies in other environments. These data are presented as annual rates of tree loss. In arid Australian woodlands, the fall-down rate for *Myoporum platycarpum* stags is 4.7% per annum (reworked from Fig. 1 in Sinclair 2004). In North American aspen-dominated boreal forest, stag fall-down rate is 9–20%, with no differences detected in relation to stem size (Lee 1998). In arid tropical woodlands in Kenya, the annual decomposition rate of dead wood biomass in the ground layer was 8.8%, with almost all decomposition conducted by termites (Buxton 1981). Xylophagous termites are particularly active in some Australian tropical woodlands (Fox and Clark 1972) and are likely to have a major influence on fall-down rates in these environments.

For this coarse analysis, the carbon biomass component not accounted for by DD and DFa is estimated as  $1.97 \text{ t C ha}^{-1}$  [ $3.11 - (0.88 + 1.09)$ ]. For no net change in standing dead wood this component of initial standing dead wood must fall to the ground (DFb in Fig. 1) at an annual rate of  $1.97/14.14 = 0.14 \text{ t C ha}^{-1} \text{ year}^{-1}$ . This represents an annual rate of loss from the dead standing wood pool attributed to stag fall-down of 4.5% ( $0.14/3.11$ ). The data from other environments (see above) suggest that an annual fall-down rate of 4% or greater for dead standing wood biomass loss in Australian tropical woodlands could be expected. Although there is considerable uncertainty in this hypothetical modelling exercise, it does demonstrate that with consumption of dead standing wood by fire and loss through decomposition, the putative dead standing wood carbon sink in the estimates from the long-term TRAPS sites (Burrows *et al.* 2002) could be substantially overestimated. The dead standing wood contributes  $\sim 40\%$  to the total carbon increment. Thus, the omission of the DF component (Fig. 1) suggests that the magnitude of the total carbon sink presented by Burrows *et al.* (2002) for grazed woodlands of Queensland is also over-estimated.

### Allometry and ground-based monitoring

What are the options for improving the carbon accounting of dead standing wood in tropical savanna? It seems likely that permanent monitoring plots such as the TRAPS sites in Queensland (Burrows *et al.* 2002), and those at Kakadu National Park (Edwards *et al.* 2003) will make a significant contribution. However, dead standing wood is not necessarily included in routine monitoring. Furthermore a record of stag size as conventionally measured by basal area provides only a poor indication of biomass as it decomposes and disintegrates. Dead standing wood at permanent monitoring sites could be more accurately monitored by using allometry that allows for the segregation of stag components (e.g. trunk, bark and branches). Australian allometric studies published in the mainstream literature that allow for the assessment of separate stag components are summarised in Table 2. These allometric equations were

**Table 2.** Summary of published allometric equations for predicting biomass of standing dead wood components

Tree size range is determined from a graphical representation in Chen (2002) for the allometric equations from O'Grady *et al.* (2000). Diameter at breast height (1.3 m) is converted to diameter at 0.3 m by using  $1.21x + 0.317$  from Fensham *et al.* (2002). *Eremophila mitchellii* is measured at ground level in Harrington (1979), although regressions here are applied for diameter at 30-cm height. Reference to Chen (2002) reveals errors in the table 3 of O'Grady *et al.* (2000). Corrected power functions are *Erythrophleum chlorostachys*, branch:  $y = 0.0002x^{4.52}$ ,  $r^2 = 0.94$ ; *Eucalyptus bleeseri*, bark:  $y = 0.013x^{2.50}$ ,  $r^2 = 0.97$ ; *E. miniata*, Total:  $0.09x^{2.51}$ ,  $r^2 = 0.97$ ; *E. tetradonta*, Total:  $0.001x^{3.30}$ ,  $r^2 = 0.94$

Species	Reference	Range of diameter at 0.3-m height (cm) or height (m)	N
<i>Acacia aneura</i>	Burrows (1976)	0.6–27	32–65
<i>Eucalyptus signata</i>	Westman and Rogers (1977)	28–77	19–23
<i>Eucalyptus umbra</i>	Westman and Rogers (1977)	24–68	11–16
<i>Banksia aemula</i>	Westman and Rogers (1977)	24–54	10
<i>Eremophila mitchellii</i>	Harrington (1979)	0.6–5.2 (height)	18
<i>Geijera parviflora</i>	Harrington (1979)	3.9–7.0 (height)	9
<i>Acacia harpophylla</i>	Scanlan (1991)	0.5–5.0 (height)	29
<i>Eucalyptus crebra</i>	Burrows <i>et al.</i> (2000)	3–64	16–20
<i>Eucalyptus melanophloia</i>	Burrows <i>et al.</i> (2000)	2–53	17–20
<i>Eucalyptus populnea</i>	Burrows <i>et al.</i> (2000)	1–76	20–22
<i>Eucalyptus miniata</i>	O'Grady <i>et al.</i> (2000)	8–30	8
<i>Eucalyptus tetradonta</i>	O'Grady <i>et al.</i> (2000)	8–27	8
<i>Eucalyptus bleeseri</i>	O'Grady <i>et al.</i> (2000)	9–27	8
<i>Eucalyptus porrecta</i>	O'Grady <i>et al.</i> (2000)	9–30	8
<i>Terminalia ferdinandiana</i>	O'Grady <i>et al.</i> (2000)	5–18	8
<i>Erythrophleum chlorostachys</i>	O'Grady <i>et al.</i> (2000)	8–18	8

applied to a range of stem-diameter classes within the range of the sample from which they were derived. To correct for biases resulting from back-transforming regression procedures applied to log-transformed data, the approximate correction of adding half the antilog of the residual standard deviation to the predicted intercept (Baskerville 1972) was applied where residual standard deviation was less than 0.7 (see Burrows *et al.* 2000).

The proportion of tree components shows a wide range of values for different tree species. A major component of this variation is undoubtedly related to methodological issues rather than to actual differences among species. The studies represent variable sample size and range (Table 2). For example, the sample of Burrows *et al.* (2000) was much larger and included a broader range of tree sizes than the study of O'Grady *et al.* (2000).

For nearly all species, the proportion of stem to total above-ground biomass exhibits a negative exponential increase (Fig. 2a). The most obvious exception is *Banksia aemula*, which has a shrub form and very large woody fruits. Stem biomass for most species is more than 90% of above-ground biomass. The exceptions are the smaller trees and shrubs *Acacia harpophylla*, *Acacia aneura*, *Banksia aemula*, *Eremophila mitchellii* and *Geijera parviflora*. The variation in the proportion of stem to total plant biomass will partly be a function of the condition of the canopy at the time of sampling and will vary greatly immediately after leaf flush and during drought.

The proportion of the total plant represented by trunk shows a greater range than for stems, with a contrasting form

of negative exponential decrease (Fig. 2b). This may be at least partly a problem of definition, as 'trunk' (see Fig. 2 for our assumptions) is not precisely defined by any of the studies. In the O'Grady *et al.* (2000) study, the eucalypts showed a wide range of responses despite having generally similar growth forms. This may be a product of the limited range of the sample that included virtually no trees larger than 30 cm at 0.3-m height (Table 2). This limitation may be particularly acute for *Erythrophleum chlorostachys*, a tree with similar growth form to the *Eucalyptus* species included in the sample, but with a maximum tree size in the sample of 18-cm girth at 0.3-m height (Table 2). The sample range is more comparable between Burrows *et al.* (2000) and Westman and Rogers (1977), although there seems to be more differences between the studies than between the *Eucalyptus* species (Fig. 2), probably an artefact of defining 'trunks' at sampling.

Issues of definition should not provide a major discrepancy for the bark proportions. However, the high proportional values of bark for *Acacia harpophylla* probably represent a methodological artefact, given its substantially higher values than for other species (Fig. 2c). These other species include 'ironbark' *E. crebra* and *E. melanophloia*, with a bark morphology suggesting a greater proportion of bark biomass than for *Acacia harpophylla*.

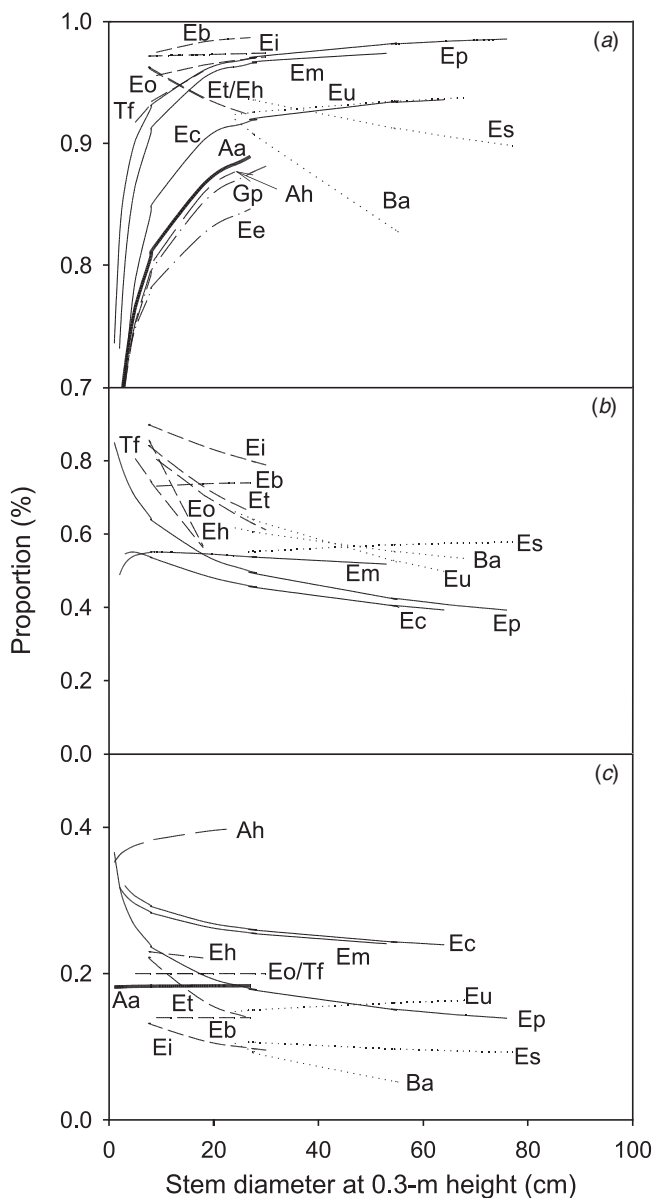
Some generalisations in relation to the proportion of bark on trees larger than 10-cm girth at 30-cm height can be provided from Fig. 2c. The proportion of bark to stem biomass is high for 'ironbark' *Eucalyptus crebra* and *E. melanophloia* (mostly  $0.25 \pm 0.02\%$ ); intermediate

for 'box' *E. populnea* and 'stringybark' *E. porrecta*, *E. tetradonta*, *E. umbra* (mostly  $0.16 \pm 0.04\%$ ); and low for smooth-barked *E. bleeseri* and *E. signata* (mostly  $0.11 \pm 0.03\%$ ). The shape of the curves is usually decreasing negative exponential such that small trees have higher values than these expressed averages and ranges (Fig. 2c).

Employing these allometric equations during ground-based monitoring would require that stags be assessed in the field on the basis of the presence of stems (branches and trunk), trunk only and the presence or absence of bark on these components. Some of these components cannot be directly determined by using the allometric relations presented in Fig. 2, but can be derived. For example in the case of *E. populnea*, trunk without bark can be approximated as

trunk with bark  $\times$  bark/(bark + wood). Stags will rarely fit neatly into component classes. Thus, a stag with a trunk and the major branches intact could be inaccurately categorised as having either a trunk only or as having both trunk and branches. The use of these coarse categories requires a judgement that best approximates the condition of a stag during a continuous process of disintegration. Finer levels of component categories could be used, such as minor branches intact, major branches intact, full trunk intact, half trunk intact, and these subdivided biomass categories could be estimated from the component allometrics with uncertain accuracy.

The process of decomposition also distorts allometric interpretations of dead wood and live trees. Brown (2002) suggested that decomposition classes need to be



**Fig. 2.** Diameter at 30-cm height relative to proportions of tree components (a) stem (trunk and branches) relative to total dry weight, (b) trunk relative to total dry weight and (c) bark relative to bark and wood dry weight for a range of tree species. The definition of stem is used here in the sense of Burrows *et al.* (2000) and differently than it is in other studies, e.g. stems do not include branches in Westman and Rogers (1977) and O'Grady *et al.* (2000) and include trunk and branches >2-cm diameter in Scanlan (1991). Species are grouped by the study from which the allometric equations were derived (solid thin line, Burrows *et al.* (2002); long dash, Scanlan (1991); medium dash, O'Grady *et al.* (2000); dotted, Westman and Rogers (1977); dash dot, Harrington (1979); thicker solid, Burrows (1976)). Relationships are shown for the range of basal-area values included in the studies (approximated from tree heights given by Scanlan 1991 and Harrington 1979). Individual species are identified as *Acacia aneura* (Aa), *A. harpophylla* (Ah), *Banksia aemula* (Ba), *Eremophila mitchelli* (Ee), *Erythrophleum chlorostachys* (Eh), *Eucalyptus bleeseri* (Eb), *E. crebra* (Ec), *E. melanophloia* (Em), *E. miniata* (Ei), *E. populnea* (Ep), *E. porrecta* (Eo), *E. signata* (Es), *E. tetradonta* (Et), *E. umbra* (Eu), *Geijera parviflora* (Gp) and *Terminalia ferdinandiana* (Tf). Proportions are calculated such that the numerator is determined from the sum of constituent parts (as defined by the terminology of the individual studies) as follows: *stem*: Burrows (1976), stem/stem + phyllode; Westman and Rogers (1977), live stem wood + live stem bark/live stem wood + live stem bark + live branch wood and bark + terminal twigs and leaves; Harrington (1979), wood/wood + leaf; Scanlan (1991), stem total + branch/stem total + branch + leaf; Burrows *et al.* (2000), stem/stem + capsule + capsules + leaf; O'Grady *et al.* (2000), stem + branch/stem + branch + leaf; *trunk*: Westman and Rogers (1977), live stem wood + live stem bark/live stem wood + live stem bark + live branch wood and bark + terminal twigs and leaves; Scanlan (1991), stem total/stem total + branch + leaf; Burrows *et al.* (2000), trunk/trunk + branches + capsules + leaf; O'Grady *et al.* (2000), stem/stem + branch + leaf; *bark*: Burrows (1976), bark/bark + wood; Westman and Rogers (1977), live stem bark/live stem wood + live stem bark; Scanlan (1991), stem bark/stem bark + stem wood; Burrows *et al.* (2000), bark/bark + wood; O'Grady *et al.* (2000), bark/bark + wood.

superimposed over determinations of dead wood volume to assist with accurate determinations of biomass. In tropical savanna the primary cause of dead-wood decomposition is by termites that typically cause hollowing. The problems of tree hollows and other uncertainties for allometry are highlighted by the meta-analysis of Williams *et al.* (2005). These authors also demonstrate that tree height in addition to basal area is an important variable for accurate biomass predictions and that this combination substitutes the need for segregating many individual Australian tree species. It seems likely that adequate allometric equations for stags may also be expediently developed from basal area and height data, probably in combination with some measure of tree hollowing. If the substantial investment in permanent monitoring is to provide worthwhile information for determining the trends in the dead standing wood biomass of Australian savanna, it will be essential to develop these allometric relations for stags. In the meantime, the basal area, height and some indication of the degree of hollowing of stags should be included in the data-collection procedures for monitoring sites. Procedures for assessing hollowing have largely been developed within studies of habitat for arboreal fauna and generally require description of entrance sizes in relation to tree size (e.g. Taylor and Chisholm 2005).

### Modelling standing dead wood

An alternative approach to direct inventory could seek to understand the factors that affect the production and decomposition of both standing and fallen dead wood. Monitoring and field experiments could be designed to address the following questions:

- (1) what are the patterns and environmental determinants of tree death;
- (2) what are the spatial patterns of fires;
- (3) how does fire influence the fate of dead standing wood; and
- (4) what are the temporal and spatial patterns of decomposition in relation to dead standing wood?

The answers to these questions may allow for the development of models that describe the fate of dead standing wood in relation to ecological processes that are easier to monitor and map than the dead wood itself. Fensham and Holman's (1999) survey of drought-induced dieback and the experimental studies (Williams *et al.* 1999) of tree mortality in relation to savanna fires from northern Australia are both highly relevant to the first question. However, reliable data on fire frequency across Queensland are lacking. The NOAA-AVHRR imagery collated by Russell-Smith *et al.* (2003) greatly underestimates the spatial extent of fires in Queensland (P. Thompson, pers. comm.). Distortions are related to the distance from the receiver in Perth, its coarse 1.1-km<sup>2</sup> spatial resolution and more universal

problems imposed by cloud cover and rapid recovery of herbaceous vegetation that also beset mapping fire scars from Landsat imagery (Bowman *et al.* 2003).

The second question is also informed by some existing studies. Paudyal *et al.* (1997) demonstrated that they were able to provide a spatial representation of the dieback event documented by Fensham and Holman (1999) for northern Queensland. The data from both Fensham and Holman (1999) and Williams *et al.* (1999) can be incorporated into models such as FLAMES (Liedloff *et al.* 2001) describing savanna dynamism. Further development and calibration of models with real data is required to provide meaningful predictions of tree mortality in northern Australia.

This study has provided some data on the role of fire for influencing the fate of dead standing wood. This information should be augmented from other regions in Australia and similar information developed to describe the fate of fallen dead wood. Far less is known of the temporal and spatial patterns of decomposition. Such studies should target environments representing the spectrum of climate regimes, vegetation types and levels of xylophagous termite activity in Australian savanna.

With a relatively simple set of information it may be possible to build meaningful models of dead standing wood, given generalised maps of live basal area, climate fluctuations and fire regimes, all of which may soon be available or could be derived from existing vegetation maps (e.g. Fox *et al.* 2000) or remotely sensed vegetation cover (e.g. Paudyal *et al.* 1997), climate-modelling surfaces (e.g. Jeffrey *et al.* 2001) and fire-history mapping (e.g. Russell-Smith *et al.* 2003). These models could be validated with field data and may be refined to provide a reasonably accurate audit of changes within the potentially sizeable pool of dead-wood biomass in northern Australia.

### Acknowledgments

Russell Fairfax and Don Butler are thanked for assistance with data collection and comments on the manuscript. Rod Rogers, Madonna Hoffman provided feedback to ensure that the application of allometrics was appropriate. Gordon Guymmer, John Neldner, Russell Fairfax and Bruce Wilson also provided valuable comments on the manuscript as it developed.

### References

- Allen CD, Breshears DD (1998) Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences of the United States of America* **95**, 14 839–14 842. doi: 10.1073/pnas.95.25.14839
- Back PV, Burrows WH, Hoffmann MB (1999) TRAPS: a method for monitoring the dynamics of trees and shrubs in rangelands. *Proceedings of the VIIth International Rangeland Congress* **2**, 742–744.

- Baskerville GL (1972) Use of logarithmic regression in the estimation of plant biomass. *Canadian Journal of Forestry* **2**, 49–53.
- Bowman DMJS, Zhang Y, Walsh A, Williams RJ (2003) Experimental comparison of four remote sensing techniques to map tropical savanna fire-scars using Landsat-TM imagery. *International Journal of Wildland Fire* **12**, 341–348. doi: 10.1071/WF03030
- Brown S (2002) Measuring carbon in forests: current status and future challenges. *Environmental Pollution* **116**, 363–372. doi: 10.1016/S0269-7491(01)00212-3
- Burrows WH (1976) Aspects of nutrient cycling in semi-arid mallee and mulga communities. PhD Thesis, Australian National University, Canberra.
- Burrows WH, Hoffman MB, Compton JF, Back PV, Tait LJ (2000) Allometric relationships and community biomass estimates for some dominant eucalypts in central Queensland woodlands. *Australian Journal of Botany* **48**, 707–714. doi: 10.1071/BT99066
- Burrows WH, Henry BK, Back PV, Hoffman MB, Tait LJ, Anderson ER, Menke N, Danaher T, Carter JO, McKeon GM (2002) Growth and carbon stock change in eucalypt woodlands in northeast Australia: ecological and greenhouse sink implications. *Global Change Biology* **8**, 769–784. doi: 10.1046/j.1365-2486.2002.00515.x
- Buxton RD (1981) Termites and the turnover of dead wood in an arid tropical environment. *Oecologia* **51**, 379–384. doi: 10.1007/BF00540909
- Chen X (2002) Carbon balance of a eucalypt open forest savanna of northern Australia. PhD Thesis, Northern Territory University, Darwin.
- Edwards A, Kennett R, Price O, Russell-Smith J, Spiers G, Woinarski J (2003) Monitoring the impacts of fire regimes on vegetation in northern Australia: an example from Kakadu National Park. *International Journal of Wildland Fire* **12**, 427–440. doi: 10.1071/WF03031
- Fensham RJ, Butler DW (2004) The spatial pattern of dry rainforest colonising unburnt *Eucalyptus* savanna. *Austral Ecology* **29**, 121–128. doi: 10.1111/j.1442-9993.2003.01318.x
- Fensham RJ, Fairfax RJ (2003) A land management history for central Queensland, Australia as determined from landholder questionnaire and other sources. *Journal of Environmental Management* **68**, 409–420. doi: 10.1016/S0301-4797(03)00110-5
- Fensham RJ, Holman JE (1999) Temporal and spatial patterns in drought-related tree dieback in Australian savanna. *Journal of Applied Ecology* **36**, 1035–1050. doi: 10.1046/j.1365-2664.1999.00460.x
- Fensham RJ, Fairfax RJ, Holman JE, Whitehead PJ (2002) Quantitative assessment of vegetation structural attributes from aerial photography. *International Journal of Remote Sensing* **23**, 2293–2317. doi: 10.1080/01431160110106050
- Fensham RJ, Fairfax RJ, Bowman DMJS, Butler DW (2003) Effects of fire and drought in a tropical eucalypt savanna colonised by rain forest. *Journal of Biogeography* **30**, 1405–1414. doi: 10.1046/j.1365-2699.2003.00934.x
- Fox ID, Neldner VJ, Bannink PJ, Wilson BA, Brocklehurst PS, Clark MJ, Dickinson KJM, Beard JS, Hopkins AJM, Beeston GR, Harvey JM, Thompson EJ, Ryan T, Thompson SL, Butler DW, Cartan H, Addicott EP, Wilson GW, Bailey L, Cumming RJ, Johnson DC, Schmeider M, Stephens K (2000) A vegetation map for northern Australia. 1 : 2 000 000 scale. Queensland Herbarium, Environmental Protection Agency, Brisbane.
- Fox RE, Clark NB (1972) The incidence of termites in eucalypts of the Darwin area. *Australian Forest Research* **5**, 29–36.
- Harrington GN (1979) Estimation of above ground biomass of trees and shrubs in a *Eucalyptus populnea* F.Muell. woodland by regression of mass on trunk diameter and plant height. *Australian Journal of Botany* **27**, 135–143.
- Jeffrey SJ, Carter JO, Moodie KB, Beswick AR (2001) Using spatial interpolation to construct a comprehensive archive of Australian climate data. *Journal of Environmental Modelling and Software* **16**, 309–330. doi: 10.1016/S1364-8152(01)00008-1
- Lee P (1998) Dynamics of snags in aspen-dominated midboreal forests. *Forest Ecology and Management* **105**, 263–272. doi: 10.1016/S0378-1127(97)00286-7
- Liedloff AC, Coughenour MB, Ludwig JA, Dyer RM (2001) Modelling the trade-off between fire and grazing in a tropical savanna landscape, northern Australia. *Environment International* **27**, 173–180. doi: 10.1016/S0160-4120(01)00079-4
- O'Grady AP, Chen X, Eamus D, Hutley LB (2000) Composition, leaf area index and standing biomass of eucalypt forests near Darwin in the Northern Territory, Australia. *Australian Journal of Botany* **48**, 629–638.
- Paudyal D, Kuhnell C, Danaher TJ (1997) Detecting change in woody vegetation in Queensland using Landsat TM imagery. North Australian remote sensing and GIS conference proceedings, Cairns, Australia, April 1997.
- Russell-Smith J, Yates C, Edwards A, Allan GE, Cook GD, Cooke P, Craig R, Heath B, Smith R (2003) Contemporary fire regimes of northern Australia, 1997–2001: change since Aboriginal occupancy, challenges for sustainable management. *International Journal of Wildland Fire* **12**, 283–297. doi: 10.1071/WF03015
- Scanlan JC (1991) Woody overstorey and herbaceous understorey biomass in *Acacia harpophylla* (brigalow) woodlands. *Australian Journal of Ecology* **16**, 521–529.
- Sinclair R (2004) Persistence of dead trees and fallen timber in the arid zone: 76 years of data from the T.G.B. Osborn Vegetation Reserve, Koonamore, South Australia. *The Rangeland Journal* **26**, 111–122. doi: 10.1071/RJ04008
- Taylor RJ, Chisholm RA (2005) The occurrence of hollows in eucalypts and ironwood *Erythrophleum chlorostachys* in the Gulf region of the Northern Territory and its implications for timber harvesting. *Pacific Conservation Biology* **11**, 57–63.
- Viljoen AJ (1995) The influence of the 1991/92 drought on the woody vegetation of the Kruger National Park. *Koedoe* **38**, 85–97.
- Westman WE, Rogers RW (1977) Biomass and structure of a subtropical eucalypt forest, North Stradbroke Island. *Australian Journal of Botany* **25**, 171–191.
- Williams RJ, Cook GD, Gill AM, Moore PHR (1999) Fire regime, fire intensity and tree survival in a tropical savanna in northern Australia. *Australian Journal of Ecology* **24**, 50–59. doi: 10.1046/j.1442-9993.1999.00946.x
- Williams RJ, Zerihun A, Montagu K, Hoffman M, Hutley LB, Chen XC (2005) Allometry for estimating aboveground tree biomass in tropical and subtropical eucalypt woodlands: towards general predictive equations. *Australian Journal of Botany* **53**, 607–619.

Manuscript received 29 July 2004, accepted 8 September 2005