Decay patterns and carbon density of standing dead trees in California mixed conifer forests

Stella J.M. Cousins [a,*], John J. Battles [a], John E. Sanders [a], Robert A. York [a,b]

[a] Department of Environmental Science, Policy, and Management, University of California Berkeley, Berkeley, CA 94720-3114, United States
[b] Center for Forestry, Blodgett Forest Research Station, University of California Berkeley, Georgetown, CA 95634, United States

ABSTRACT

Dead wood plays important structural and biogeochemical roles in forest ecosystem processes. Some aspects of woody debris dynamics have been carefully studied, but the decay patterns and carbon density of standing dead (SD) trees are only weakly characterized. Climbing forest mortality rates are also driving increases in the creation and abundance of SD trees. All forms of forest carbon accounting, from stand-level biomass calculations to dynamic earth systems models, are improved by a better understanding of SD tree physical and chemical traits. Using dimensional analysis, we described the patterns of density, carbon concentration, and net carbon density in decaying SD trees of six California mixed conifer species. As decay class advanced, trees showed a progressively lower density and a small increase in carbon concentration. Net carbon density of the most decayed SD trees was only 60% that of live trees. The key characteristics of SD trees that determine these patterns are species, surface to volume ratio, and relative position within the tree. The decay of SD trees and how deadwood biomass is estimated in large scale inventories also have repercussions in greenhouse gas accounting. When the measured changes in carbon density are applied to SD carbon stock estimates for California mixed conifer forests, the decay-adjusted estimates are 3.66–3.74 Tg (18%) lower than estimates that do not incorporate change due to decay.

© 2015 Elsevier B.V. All rights reserved.

1. Introduction

Tree death is a critical process in forest ecosystems. When trees die, the subsequent decay releases carbon to the atmosphere as well as an abundance of resources (e.g., nutrients and energy) into the forest ecosystem (Franklin et al., 1987; Harmon et al., 1986). Decomposition often begins while the tree is still standing (Boddy, 2001; Harmon et al., 1986; Whittaker et al., 1979). Standing dead (SD) trees store a significant amount of carbon and provide essential habitat for wildlife, but also present potential safety and fire hazards (Hilger et al., 2012; Keen, 1955; Knapp, 2015; Raphael and White, 1984). Thus, SD trees play an important role in the ecology and management of forests.

As a direct result of well-documented increases in forest morbidity and mortality, SD trees are becoming more important for forest carbon dynamics. Increasing climatic stress, chronic and widespread air pollution, and pest outbreaks have fueled forest die-offs worldwide, with broad implications for ecosystem structure and function (Allen et al., 2010; Anderegg et al., 2012;

Bytnerowicz et al., 2007; Hicke et al., 2013). In the North American West, increases in mortality are widely attributable to warming and increased water deficits (van Mantgem et al., 2009), often in combination with irruptions of bark beetles (Bentz et al., 2010; Ganey and Vojta, 2011). Across the western US from 1997–2010, bark beetles killed trees containing 2–24 teragrams (Tg) of carbon per year on over 5 million ha (Hicke et al., 2013; Meddens et al., 2012). Similarly, mountain pine beetle (*Dendroctonus ponderosae*) related mortality in British Columbia has caused forests there to become a net source of carbon, potentially for many decades (Kurz et al., 2008). Following a severe disturbance such as this, the majority of aboveground carbon stocks may be stored in SD trees (Hagemann et al., 2010). Fire suppression also contributes to increased abundance and decreased average size of dead trees (Stephens, 2004; Stephens and Moghaddas, 2005). Climate change projections and emissions trends indicate a future of exacerbated environmental stress both for California's forests (Battles et al., 2009; Moser et al., 2009; Panek et al., 2013) and forests throughout the western United States (Allen et al., 2010; IPCC, 2007). Additionally, high exposure to ozone pollution that contributes to tree stress and death – already the norm in parts of California – is expected in nearly 50% of global forests

---

within this century (Fowler et al., 1999; Panek et al., 2013). Elevated tree mortality will contribute to a growing deadwood carbon pool and could result in regional increases in GHG emissions (e.g., Kurz et al., 2008).

During the time a dead tree remains standing, a typical sequence of changes occurs leading to an overall reduction in tree size (Domke et al., 2011; Raphael and Morrison, 1987). Tree volume declines through loss of leaves, twigs, and branches, which fall to the forest floor and join the down woody material (DWM) pool (Aakala et al., 2008; Raphael and Morrison, 1987). Concurrent with these dimensional reductions are changes to the tree's physical and chemical properties caused by weathering, decomposition, and insect activity (Harmon et al., 1986; Domke et al., 2011). In many temperate species, wood and bark density decline with advancing decay, while the carbon content of tissues rises slightly. The net outcome is a decrease in total carbon density (Harmon et al., 2013, 2011). Size, tree species, and present decay condition are key determinants of wood decay rates (Harmon et al., 1986; Russell et al., 2014; Yatskov et al., 2003). Parts of the stem can also decay at different rates. For example, contact with soil can increase moisture content and facilitate access for decay organisms, accelerating decomposition near ground level compared to upper portions of the stem (Boddy, 2001; Graham, 1925; Harmon et al., 1986; Maser and Trappe, 1984). Gradients in the density of wood from the originating live tree also shape patterns of decay. For example, conifers typically have higher wood density and carbon concentrations at the base of the stem relative to the upper reaches (Bowyer et al., 2007). SD trees retain substantial necromass for decades (Keen, 1929; Hilger et al., 2012) and sometimes even over a century (Mast et al., 1999). Since SD trees can represent a significant carbon reservoir in many forests, explicitly accounting for SD tree carbon dynamics will refine estimates of forest carbon storage and flux, and improve models of decomposition (Kurz et al., 2009; Litton et al., 2007; Woodall et al., 2008).

In continental US and Canada forests, SD trees form 5% to 35% of aboveground forest biomass (Aakala et al., 2008; Vanderwel et al., 2008; Woodall et al., 2008). Conifer forests in the Western US are typically at the upper end of this range, with field-based estimates ranging from 2.4 to 7.2 Mg carbon ha^{-1} (Woodall et al., 2012, 2013). Standing dead carbon in California mixed conifer forests specifically is greater still, with an average of 9.5 Mg carbon ha^{-1} and a total of 20.5 Tg carbon in SD trees (Battles et al., 2014; Gonzalez et al., 2010). SD trees form one of five forest sector carbon pools included in the U.S. National Greenhouse Gas Inventory (NGHGI) (Aalde et al., 2006; EPA, 2015). The NGHGI is in turn used for reporting to international bodies including the Intergovernmental Panel on Climate Change and United Nations (IPCC, 2006; United Nations, 1992). There is also growing demand for accurate accounting of GHG emissions at state and regional levels. California's Global Warming Solutions Act, which requires inventory of storage and emissions of GHG by forests, is one example (State of California, 2006).

Though SD trees are essential to forest processes, incorporation of their physical and chemical transformations into forest inventory and carbon accounting is a relatively recent development (Domke et al., 2013; Fahey et al., 2005; Heath et al., 2011; O'Connell et al., 2012; Woodall, 2012). As previously implemented, the NGHGI and dependent statewide inventories handled SD trees by applying live wood and bark properties to gradually decreasing tree volumes (Battles et al., 2014; Woudenberg et al., 2010; Woodall et al., 2011). The most recent NGHGI reports, based on results from the US Forest Service Forest Inventory and Analysis program (FIA), now also include adjustments for wood decay in SD trees in addition to estimates of whole tree volume loss (O'Connell et al., 2014; Russell et al., 2015; EPA, 2015). Biomass and GHG estimates employ a suite of ratios that are used to adjust

the density of SD trees according their species and decay class (1: intact – 5: advanced decay) (O'Connell et al., 2014; USFS, 2010). These density reduction ratios rely on the demonstrated correspondence between decay rates of DWM and SD trees. The SD adjustments in use for major mixed conifer species originate from the SD to down dead ratios for softwoods (classes 1–3) and the SD to down dead ratios for all inventoried species (classes 4–5) (Harmon et al., 2011, 2008). Comparison of DWM and SD density suggests that patterns of density and carbon loss in SD trees differ from those of DWM; for mixed conifer species, sampled SD densities are not yet available (Harmon et al., 2011, 2013). Finally, a typical large-scale inventory sets carbon concentration of SD trees at 50% of total biomass (Birdsey, 1992; Fahey et al., 2005; Woodall et al., 2011). However, this assumption underestimates SD tree carbon concentration by 5–10%, as carbon density increases with advancing decay in both hardwood and softwood species groups, including California mixed conifer species (Harmon et al., 2013). Continued improvement in forest carbon estimates requires further quantification of the changes characteristic of SD trees: not only diminishing size, but also distinctly altered physical and chemical properties (Heath et al., 2011; Smith et al., 2003; Woodall et al., 2012).

The goal of this study was to understand the patterns and processes of *in situ* decay of SD trees in the mixed conifer forests of California. To achieve this goal, we first measured the decline in carbon density (g cm^{-3}) of SD trees along a decay class gradient. We specifically accounted for the contributions of changes in wood density and carbon concentration that occur during decay. We then asked whether patterns in carbon density with advancing decay varied by species, tree size, or vertical position along the tree bole. To scale up the results from our site-based research and to gauge their relevance to carbon management, we used the same tree inventory and decay classification used by the FIA program. Based on previous studies of decay patterns in SD trees (Harmon et al., 2011), our hypothesis was that bole density would decline with decay class. On the basis of carbon measurements from DWM and some SD trees (Harmon et al., 2013), we expected carbon concentration to rise slightly with decay class, but net carbon density to decline. We also anticipated that density losses in SD trees would vary by tree size and species. Given the more rapid decay of woody material in contact with the soil (Maser and Trappe, 1984), the base of SD trees was expected to decay faster than the upper bole. Finally, in an application of these measurements, we developed decay-adjusted estimates of SD carbon stock for California mixed conifer forests and compared the outcomes of the live:dead decay ratios from this study with those used by the FIA program. Taking the patterns above into account, we predicted that implementing decay adjustments in biomass inventory would lead to lower estimates of carbon stock in SD trees than alternative unadjusted approaches.

2. Methods

2.1. Study areas

Dimensional analysis of SD trees was carried out in mixed conifer forests at Blodgett Forest Research Station (BFRS) and Sequoia National Park. BFRS is situated on the western slope of the Sierra Nevada near Georgetown, California (38°52′N; 120°40′W; Fig. 1). Six native tree species are commonly found in mixtures of varying proportions: white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens*), coast Douglas-fir (*Pseudotsuga menziesii var. menziesii*), sugar pine (*Pinus lambertiana*), ponderosa pine (*Pinus ponderosa*), and California black oak (*Quercus kelloggii*). The elevation of study sites ranges from 1220 to 1350 m. Annual precipitation at BFRS