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ARTICLE

A 6 year longitudinal study of post-fire woody carbon dynamics in California's forests

Bianca N.I. Eskelson, Vicente J. Monleon, and Jeremy S. Fried

Abstract: We examined the dynamics of aboveground forest woody carbon pools — live trees, standing dead trees, and down wood — during the first 6 years following wildfire across a wide range of conditions, which are characteristic of California forest fires. From repeated measurements of the same plots, we estimated change in woody carbon pools as a function of crown fire severity as indicated by a post-fire index, years since fire, pre-fire woody carbon, forest type group (hardwood vs. softwood), elevation, and climate attributes. Our analysis relied on 130 U.S. national forest inventory plots measured before and 1 year after fire, with one additional remeasurement within 6 years after fire. There was no evidence of net change in total wood carbon, defined for this study as the wood in standing trees larger than 12.7 cm diameter at breast height and down wood larger than 7.6 cm in diameter, over the post-fire period in any of the three severity classes. Stands that burned at low severity exhibited considerable shifts from live to standing dead and down wood pools. In stands that burned at moderate severity, live wood decreased significantly whereas no net change was detected in standing dead or down wood. High severity fire burning resulted in movement from standing dead to down wood pools. Our results suggest that the carbon trajectories for stand-replacing fires may not be appropriate for the majority of California's forest area that burned at low to moderate severities.

Key words: longitudinal analysis, post-fire dynamics, forest stand recovery, disturbance, Forest Inventory and Analysis.

Résumé : Nous avons étudié la dynamique des réservoirs de carbone ligneux de la partie aérienne des forêts (les arbres vivants, les arbres morts sur pied et au sol) au cours des six premières années après un feu pour une large gamme de conditions caractéristiques des feux de forêt de la Californie. À partir de mesures répétées des mêmes placettes, nous avons estimé les changements dans les réservoirs de carbone ligneux en fonction de la sévérité des dommages à la cime des arbres selon un indice après feu, du nombre d'années après le feu, du carbone ligneux avant le feu, du groupe de types forestiers (feuillus ou résineux), de l'altitude et des caractéristiques du climat. Nos analyses ont porté sur 130 placettes de l'inventaire national états-unien (FIA) mesurées avant et un an après le feu de même qu'une autre fois au cours des six premières années après le feu. Il n'y a pas eu d'indices de changement net du carbone ligneux total après feu, correspondant dans cette étude au bois des arbres sur pied ayant un diamètre à hauteur de poitrine plus grand que 12,7 cm et des arbres au sol ayant un diamètre supérieur à 7,6 cm, dans aucune des trois classes de sévérité du feu. Les peuplements ayant subi un feu de sévérité faible ont connu des changements considérables dans les réservoirs de carbone ligneux, passant d'une dominance d'arbres vivants à une dominance d'arbres morts sur pied et au sol. Dans les peuplements brûlés par un feu de sévérité modérée, les arbres vivants ont diminué significativement alors qu'aucun changement net n'a été détecté chez les arbres morts sur pied ou au sol. Le feu de sévérité élevée a provoqué un déplacement de réservoir ligneux des arbres morts sur pied vers les arbres au sol. Nos résultats indiquent que les trajectoires de carbone typiques des feux qui entraînent le remplacement des peuplements peuvent ne pas être appropriées pour les cas plus communs de peuplements subissant des feux d'intensités faible à modérée. [Traduit par la Rédaction]

Mots-clés : analyse longitudinale, dynamique après feu, rétablissement des peuplements forestiers, perturbation, programme d'analyse et d'inventaire forestiers (FIA).

Introduction

Natural disturbances such as wildfire are increasing in frequency, severity, and extent across forested ecosystems of the western U.S. (Westerling et al. 2006; Miller et al. 2009; Stephens 2005; Dennison et al. 2014). Wildfire disturbances introduce sharp discontinuities in a forest's carbon cycle, typically resulting in net carbon emissions to the atmosphere, which continue for several years to several decades (Goetz et al. 2012). Understanding postdisturbance recovery dynamics is essential if we are to include disturbance impacts in models of forest carbon budgets (Liu et al. 2011) and formulate policies that can adapt management decisions to address future increases in fire frequency, severity, and extent. Carbon pool dynamics following wildfire represent a critical gap in our understanding of postdisturbance carbon dynamics (Goetz et al. 2012), which could be improved by long-term monitoring (Yocom Kent et al. 2015).

Wildfire generates immediate greenhouse gas emissions and initiates a sequence of rapid and gradual transfers from live to dead carbon pools and from dead carbon pools to the atmosphere. These changes can continue for decades but will eventually be offset by new growth from regeneration and succession. Carbon emissions have been reported to continue 10 years after a stand-

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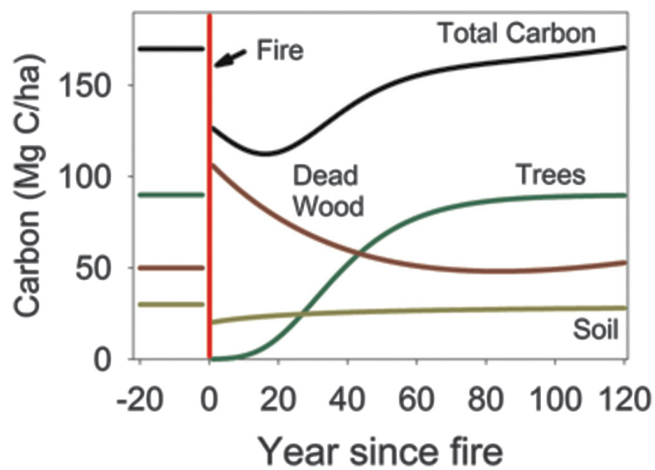
replacing fire (Dore et al. 2008) and were found to be three times as large as immediate emissions in 13 years after the fire (Auclair and Carter 1993). Snags convert rapidly to down wood pools in less than 8 years for nearly all but very large snags (Ritchie et al. 2013), and down wood dynamics strongly depend on pre-fire stand structure, fire severity, and post-fire treatments (Brown et al. 2003). These and similar studies focus on dead wood as habitat elements in relation to post-fire logging or on direct carbon flux measurements. Yocom Kent et al. (2015) present one of few studies that track carbon dynamics by live and dead pools and show how the post-fire carbon trajectories are related to fire severity.

The best way to study post-fire dynamics is to follow measurements of live and dead carbon pools over time. However, such empirical data are quite rare. Current knowledge and assumptions about recovery trajectories are typically derived from chronosequences of selected stands (e.g., Seedre et al. 2014; Kashian et al. 2013; Amiro et al. 2010; Bond-Lamberty et al. 2004). Chronosequences make the crucial assumption that the only relevant difference among sites is years since fire (Johnson and Miyanishi 2008); however, other factors such as pre-fire conditions, fire behavior, landscape context, or climatic variation following the fire may influence pos-tfire ecosystem dynamics. The understandings derived from such studies are encapsulated in conceptual diagrams of post-fire disturbance dynamics by carbon pool (e.g., Fig. 1). Such conceptual diagrams suggest a quasideterministic relationship and may not reflect the range of variability that occurs across large landscapes. Furthermore, these conceptual models typically assume stand-replacing disturbances, and the domain to which they apply frequently remains unclear. Although chronosequences may be the only feasible approach to study temporal dynamics over very long periods, direct observations over time such as long-term studies of revisited plots provide the most reliable evidence about temporal changes (Walker et al. 2010) and should be used to validate chronosequence studies when possible (Johnson and Miyanishi 2008). In addition, inferences from long-term plot studies can be generalized to the population that the plots represent, if the plots are a probability sample from a well-defined population.

Data collected by the U.S. Forest Inventory and Analysis (FIA) program on inventory plots burned in wildfires in California provide a unique opportunity to follow and describe post-fire dynamics by carbon pool. The FIA program measured a subset of plots 1 year after they were burned by wildfire (post-fire) and again at the standard remeasurement cycle (post-post-fire), resulting in repeated plot measurements. Mean trends of actual dynamics in the first few years following wildfires in California forests can be estimated using these data, thus avoiding reliance on chronosequences. Our objectives were to (*i*) estimate the rate of net change in three woody carbon pools over time in different fire severity classes and (*ii*) describe the variability in pre- and post-fire trajectories within fire severity classes.

Our detailed analyses provide insights into post-fire dynamics by carbon pool formulated by forest type group and fire severity classes. Our analyses further present substantial variation among stand trajectories by carbon pool within fire severity classes. We expect that our findings will (*i*) be broadly useful to forest managers, for example, those charged with providing carbon storage in the forest as an environmental service, as well as those seeking to manage dead wood habitat elements; (*ii*) replace assumptions and models with empirical evidence concerning the fate of carbon in burned stands in debates over the climate benefits of fuel management (e.g., North and Hurteau 2011; Harmon 2013); and (*iii*) improve our ability to include post-fire carbon dynamics in simulation studies of current and future carbon balances, which may further inform existing pre-fire (e.g., hazard reduction) and post-fire (e.g., salvage and rehabilitation) guidelines for dead wood management (e.g., Brown et al. 2003).

Fig. 1. Conceptual carbon recovery of conifer forests after stand-replacing wildfire (reprint of fig. 3 in Ryan et al. (2010), which was based on lodgepole pine forests that burned in Yellowstone National Park, USA).



Materials and methods

Forest Inventory and Analysis plots and Fire Effects and Recovery Study

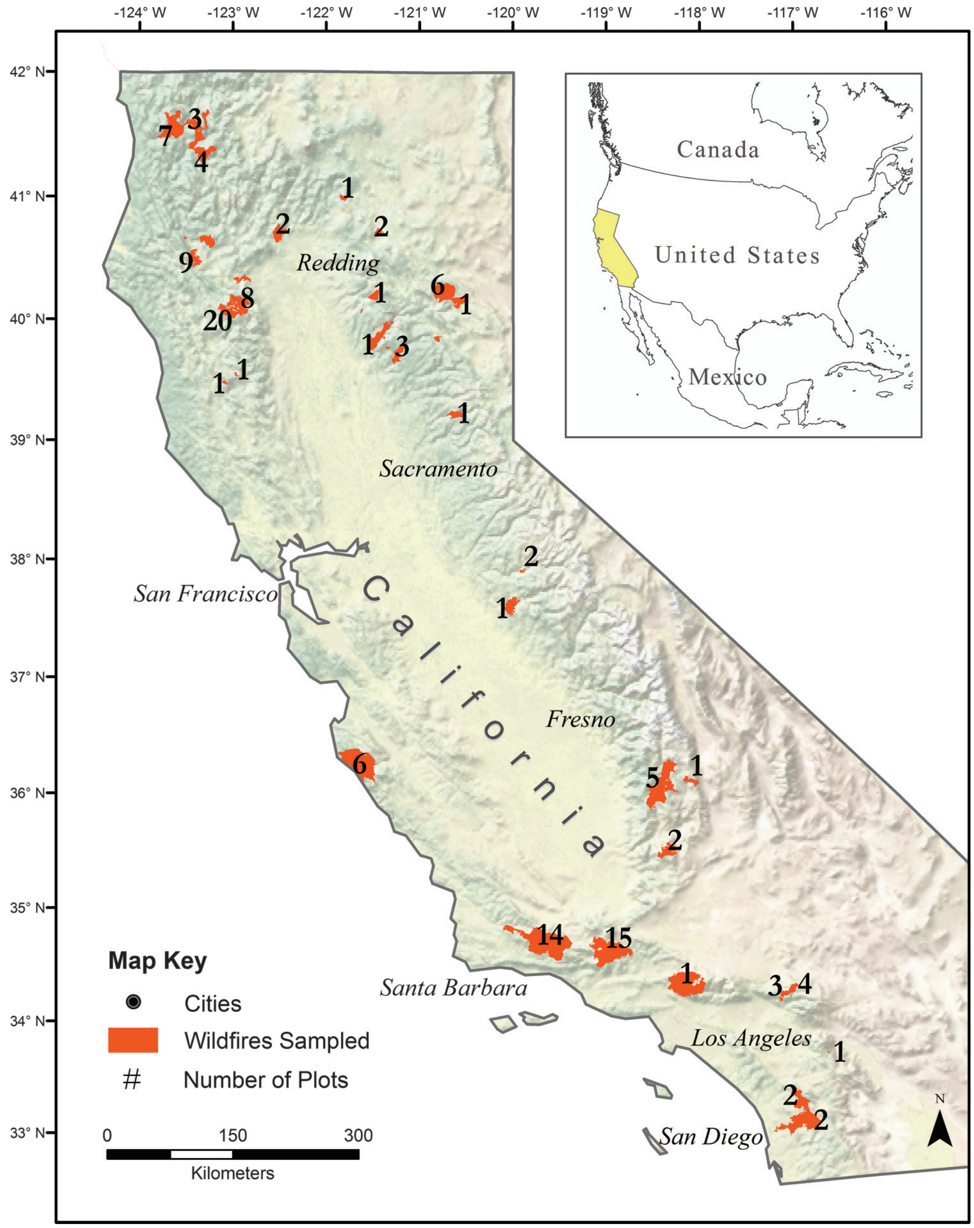
All data used in this analysis were collected by the FIA program administered by the USDA Forest Service's Pacific Northwest Research (PNW) Station and by the Vegetation Mapping and Inventory Program of USDA Forest Service Region 5 (R5). The national FIA sampling design consists of a spatially balanced sample of one field plot every 24 km² (Bechtold and Patterson 2005). Each year, a spatially balanced subsample of 10% of the FIA plots is visited, resulting in a remeasurement cycle of 10 years. The plots in the R5 sample use identical field protocols but are a spatial intensification of the FIA sample (commonly twice the number of FIA plots) in selected forest types mapped by processing remote sensing imagery (USDA Forest Service Region 5 2016).

Some of the plots in this network are burned through by wildfires every year. In 2003, the FIA program initiated a Fire Effects and Recovery Study (FERS) to obtain post-fire field data to assess the impacts of large wildfires on forest conditions, evaluate relationships between pre-fire forest structure and post-fire conditions, and monitor the pace and form of post-fire recovery. Post-fire FERS remeasurements were implemented on FIA and R5 plots within fire perimeters, primarily on public lands and, as recommended by Jain et al. (2010), within 1 year following a fire. For the post-fire FERS remeasurements, standard FIA measurements were supplemented with additional metrics (Jain et al. 2010) that provide the basis for calculating a post-fire index summarizing what a fire leaves behind with respect to tree canopy (Jain and Graham 2007).

Plot selection

Post-fire FERS visits were undertaken after years during which either one or more large fires occurred (2002, 2003, 2006, 2007, 2009) or the total area burned statewide was substantial (2008). For this analysis, we included FIA plots in public ownership and R5 plots that had at least three field visits, namely a pre-fire visit, the FERS post-fire visit, and one or more regularly scheduled visits following the post-fire visit (post-post-fire visit), so that post-fire carbon dynamics could be examined. We only selected FIA and R5 plots where at least 25% of the plot was classified as forestland, where forestland is defined as areas ≥ 0.4 ha that are capable of maintaining at least 10% tree stocking. Fire does not change whether an area is defined as forestland, as long as it does not render the site incapable of growing trees. These selection criteria

Fig. 2. Map of California fires and fire complexes included in this study. The number of Forest Inventory and Analysis (FIA) plots is indicated next to the fire boundaries.



resulted in 130 plots for which at least three field visits had been completed by the end of the 2014 field season (Fig. 2) and for which post-post-fire visit data was collected 1–6 years after the fire. The plots used in this analysis were sampled within 32 wildfires ranging in size from 312 to 97 000 ha (Table 1).

The majority of the plots in the sample (62%) were covered by softwood forest types, with California mixed conifer being the most common (66%). Most (84%) of the hardwood forest plots fell within one of the western oak forest types, with the Canyon live oak type alone accounting for 55%. Based on vegetation composition and trees cored to estimate site index, about 36% of the plots were classified as one of the two least productive site quality classes, and these plots typically supported hardwood forest types (60%). The remaining 64% of the plots were attributed as moderate to high productivity and mostly supported softwood forest types (75%).

Field measurements

Each FIA plot consists of four points within a 1 ha circle. At each point, live and dead trees were measured at two nested circular subplots of different size. Live and dead trees with diameter between 12.7 and 61 cm were tallied in 7.32 m radius subplots (total area, 672.45 m^2). Trees with diameter greater than or equal to 61 cm were tallied in 17.95 m radius subplots (total area, 4050 m^2). Pieces of down wood with diameter greater than 7.6 cm and length greater than 0.9 m were sampled using two 17.95 m radial transects per point (total length for each plot, 143.6 m). Only trees and pieces of wood that fell in forestland were included in the sample. Details of the measurement protocols are available in the FIA field manuals (USDA Forest Service 2009).

A six-class crown post-fire index (PFI) describes a stand in terms of the relative abundance of unburned (green), scorched (brown), and burned (black) tree crowns based on slight refinement of a six-class system developed by Jain and Graham (2007). For each tree that

Table 1. California fires included in this study.

Fire name	Fire year	Fire area (ha)	No. of FIA plots	Elevation (m)
American River Complex	2008	4 488	1	1954
Antelope Complex	2007	9 037	1	1671
Apache	2008	312	1	1975
BTU Lightning Complex	2008	21 731	1	447
Basin Complex	2008	66 209	6	453–1304
Bear Wallow Complex	2008	5 312	1	1471
Butler 2	2007	5 559	4	2206–2306
Canyon Complex	2008	13 306	3	1201–1532
Chalk	2008	6 583	1	723
Clover	2008	6 389	1	2300
Cub Complex	2008	5 960	1	1808
Day	2006	65 484	15	1355–2035
Hat Creek Complex	2009	3 785	2	1885
Iron Complex (six fires)	2008	42 948	8	857–1356
Lime Complex (four fires)	2008	25 871	8	1093–1488
McNally	2002	60 490	5	1733–2537
Moonlight	2007	26 288	6	1405–1946
North Mountain	2008	1 200	2	1581
Panther	2008	18 007	3	1428
Piute	2008	15 113	2	1599–1709
Poomacha	2007	19 996	2	1123–1419
SHU Lightning	2009	11 465	2	743–1132
Siskiyou Complex	2008	26 662	7	454–1329
Slide	2007	5 168	3	2186
Soda Complex	2008	1 231	1	1172
Station	2009	65 086	1	1358
Telegraph	2008	13 793	1	788
Ukonom Complex (three fires)	2008	24 451	4	661–850
Upperlake 2008 Lightning	2008	763	1	1670
Witch	2007	65 588	2	638–1025
Yolla Bolly (six fires)	2008	34 836	20	1435–2116
Zaca	2007	97 270	14	924–1868

Note: The number of Forest Inventory and Analysis (FIA) plots used in this study is the number that were part of the Fire Effects and Recovery Study. The elevation data are the elevations of FIA plots located within fire boundary and included in this study; the range of elevations are listed for fires that contained multiple FIA plots.

was alive at the pre-fire visit, field crews at the post-fire visit recorded an ocular estimate of the percentage of the pre-fire compacted live crown length that was unburned, scorched, or burned. In essence, they visualized "compacted" sections of crown in each category (e.g., combine separate unburned sections as one continuous length, then divide by total compacted crown length to obtain a percent value). In this study, we collapsed the six canopy PFI classes from Jain and Graham (2007) into three classes of crown fire severity to describe each stand: (1) low, a plurality of trees contain primarily green crown and no trees contain any black crown (n = 66) (classes 1 and 2); (2) moderate, a plurality of trees contain brown or black crown but at least some also contain green crown (n = 37) (class 3); and (3) high, no trees contain green crown (n = 27) (classes 4, 5, and 6). These three classes (low, moderate, and high) indicate fire severity evident in the forest canopy.

Biomass and carbon calculations

Aboveground live tree wood biomass (Mg) was obtained from the FIA database. The biomass is estimated using regional biomass and volume equations and includes the biomass of stem, bark, and live branches for trees with diameter at breast height (DBH, 1.3 m) greater than 12.7 cm (Zhou and Hemstrom 2010). Wood volume of snags is often estimated using allometric equations developed for live trees. This can be problematic as snags often have broken tops, their shape may deviate from that of a live tree due to decay, and the volume equations to estimate wood volume are based on outside-bark diameter measurements while most snags have lost their bark. For downed logs, Fraver et al. (2007) showed that the mean volumes estimated using the formulae for the frustum of a paraboloid and a cone provided the best estimate of the total log volume. We adopted the approach and calculated dead wood biomass for snags with DBH greater than 12.7 cm as follows. If the top was intact, we computed snag volume as the mean of the volume of a cone and a paraboloid using measured DBH and height (Fraver et al. 2007). If the top was missing, we first estimated the total height, including the missing piece, using height–diameter equations for live trees developed by Barrett (2006). From the estimated total height, measured actual height, and DBH, we estimated the top diameter assuming a cone and a paraboloid shape. Then we estimated snag volume as the mean of the volume of a frustum of a cone and a frustum of a paraboloid, excluding the missing top. Snag volumes were then converted to biomass using species-specific wood density and decay-class reduction factors (Harmon et al. 2008, 2011). Live wood and standing dead wood biomass were converted to elemental carbon (Mg) by applying a factor of 0.5 and then expanded to generate a carbon density ($Mg \cdot ha^{-1}$), via division by total sample area of the four subplots. Down wood biomass for pieces with diameter greater than 7.6 cm was calculated based on line intersect sampling theory (see eq. 4 in Table 3.1 in Woodall and Monleon (2008)) with species-specific wood density and decay-class reduction factors (Harmon et al. 2008, 2011) and then converted to carbon density ($Mg \cdot ha^{-1}$).

Ancillary variables

Several potentially relevant ancillary variables were either recorded on each plot or assigned to each plot via geographic overlay. Elevation (m) was derived from a digital elevation model with 10 m resolution. Selected climate variables were derived from mean monthly and annual conditions over the most recent three decades, provided in 800 m resolution PRISM layers (Daly et al. 2008): mean annual temperature (°C), mean annual precipitation

Table 2. Annual post-fire rate of change (percent per year for years 1–6 after the fire) by woody carbon pool, crown fire severity class, and forest type group (HW vs. SW), after accounting for the effects of pre-fire carbon.

	Crown fire severity class		
Carbon pool	Low (n = 66) HW: n = 21; SW: n = 45	Moderate (n = 37) HW: n = 12; SW: n = 25	High (n = 27) HW: n = 16; SW: n = 11
All wood	−0.2 (−1.5, 1.1)	−1.5 (−3.3, 0.4)	−3.2 (−6.7, 0.4)
HW	+0.2 (−2.2, 2.7)	−1.7 (−5.8, 2.6)	−1.3 (−5.1, 3.0)
SW	−0.4 (−1.8, 1.1)	−1.3 (−3.3, 0.7)	−5.4 (−10.9, 0.4)
Live wood	−2.5 (−4.2, −0.6)	−3.1 (−5.7, −0.5)	+7.9 (−5.0, 22.6)
HW	−1.9 (−5.6, 2.0)	−2.1 (−7.7, 3.8)	+14.5 (12.5, 16.6)
SW	−2.8 (−4.5, −1.0)	−3.6 (−6.0, −1.2)	+1.8 (−9.8, 14.9)
Standing dead wood	+16.0 (8.3, 24.2)	+0.5 (−4.0, 5.2)	−7.3 (−10.9, −3.5)
HW	+17.0 (−6.7, 46.7)	−4.2 (−9.2, 1.0)	−4.2 (−7.8, −0.3)
SW	+15.8 (9.7, 22.3)	+1.2 (−3.9, 6.6)	−11.0 (−16.5, −5.1)
Down wood	+13.0 (5.2, 21.3)	+9.0 (−5.3, 25.5)	+31.7 (9.5, 58.4)
HW	+14.1 (2.7, 26.8)	+11.4 (−12.3, 41.5)	+26.8 (−7.0, 72.8)
SW	+11.8 (1.1, 23.6)	+8.4 (−8.1, 27.9)	+36.0 (7.9, 71.4)

Note: 95% confidence limits are provided in parentheses. If a confidence interval includes 0, the estimate is not different from 0 at the 0.05 significance level. HW, hardwoods; SW, softwoods.

(mm), December minimum temperature (°C), and August maximum temperature (°C).

Pre-fire woody carbon (live + dead wood carbon) was calculated for each plot using the same biomass and carbon calculations as described above. Each plot was classified according to forest type group (hardwood or softwood, based on the plurality of stocking) and fire severity class (low, moderate, and high).

Analysis

The response variables for this study were total woody carbon and woody carbon of live trees, snags, and large down wood. All four variables are positive, highly skewed, and have many zero values and nonconstant variance. To deal with those issues, we estimated plot carbon as a function of time using multiplicative models, fitted using Poisson pseudo maximum likelihood (Santos Silva and Tenreyro 2006). We included random plot and fire effects to account for the correlation among measurements from the same plot and fire, respectively. However, once the plot random effect was included, the variance component for the fire effect was very small and was not statistically significant in the analysis, so it was not considered any further.

We modelled post-fire carbon as a function of the number of years since fire, with fire severity and forest type group as class variables. We included the two- and three-way interactions among those variables to test whether the post-fire carbon trends varied with fire severity and forest type group and to estimate separate trends for those factors. Because post-fire carbon is strongly correlated with pre-fire carbon, we included the latter as a covariate in all models. We also included additional covariates (PRISM climatic variables, elevation, and site productivity) to account for their effect on plot carbon.

The models were fitted in SAS PROC GLIMMIX (SAS Institute Inc. 2011). Multiple parameters were tested simultaneously using contrasts. We followed the advice of Santos Silva and Tenreyro (2006) and used a robust estimator of the covariance to fully take into account the heteroskedasticity in the model (option empirical in SAS PROC GLIMMIX).

Results

Post-fire rates of change by carbon pool and fire severity

Of the ancillary variables, pre-fire woody carbon was highly significant in all models ($p < 0.0001$). The coefficient was always positive, indicating that the post-fire carbon was highly and positively correlated with the pre-fire carbon. After including pre-fire woody carbon, fire severity and forest type group, neither elevation nor any of the PRISM climate variables were statistically significant and were excluded from the models. Forest type group and its interaction with year since fire and fire severity was only significant for standing dead wood carbon, indicating that the post-fire carbon trends did not differ between hardwood and softwood forests for most carbon pools. Nevertheless, we reported separate estimates for hardwood and softwood stands by fire severity class for all carbon pools, as well as a combined estimate across forest types (Table 2).

After controlling for pre-fire live wood carbon, post-fire trajectories of live wood carbon did not vary significantly by crown fire severity ($p = 0.26$), resulting in an overall decrease of 2.6% per year (95% CI: 1.1%, 4.1%) across fire severity classes. However, because stands that burned at high severity contain essentially no live wood by definition (Table 3) and therefore cannot lose any live wood, including those stands in estimates of loss in live wood carbon could be inappropriate. Thus, we report estimates of the annual rate of change for the three severity classes separately (Table 2). Live wood carbon decreased significantly in stands that burned at low (−2.5% per year) and moderate (−3.1% per year) severity and remained unchanged where severity was high. The significant decrease in live wood carbon where crown fire severity was low and moderate suggests not only delayed mortality for trees that were recorded as live at the post-fire visit, but also that this delayed mortality more than offsets any potentially increased growth rates on surviving trees. The period of post-fire analysis (≤6 years) is too short for regeneration to contribute any discernable increase in live wood. The effect of forest type group was not significant ($p = 0.54$), indicating that a common estimate for hardwood and softwood stands is sufficient. Yet, separate estimates by forest type and severity class indicate that there is no change in live wood carbon in hardwood stands that burned at low and moderate severities and a significant increase in live wood carbon in hardwood stands that burned at high severity (Table 2).

The post-fire trajectories for standing dead wood were statistically different among stands that burned at low, moderate, and high crown fire severities ($p < 0.0001$). Standing dead wood carbon increased by 16% annually where severity was low and decreased by 7.3% annually where severity was high (Table 2; Fig. 3, center). No significant change in standing dead wood carbon was detected where severity was moderate ($p = 0.85$; Table 2; Fig. 3, center). The increase in snag carbon where severity was low results from the delayed mortality of trees that were recorded as live at the post-fire visit. Stands that burned at high severity see no new influx of snags, because by definition, they contained few or no live trees at the post-fire visit that could die and become snags (Table 3). Until new live trees become established, such stands can only lose car-

Table 3. Mean pre- and post-fire carbon (Mg·ha^{-1}) by pool, crown fire severity class, forest type group (HW vs. SW).

	Crown fire severity classes		
Carbon pool	Low (n = 66)	Moderate (n = 37)	High (n = 27)
All wood			
Pre-fire			
Total	123 (87)	87 (74)	48 (40)
HW	87 (63)	77 (63)	33 (25)
SW	140 (92)	92 (80)	69 (49)
Post-fire			
Total	119 (83)	73 (65)	30 (26)
HW	85 (64)	63 (63)	20 (17)
SW	135 (87)	78 (66)	44 (32)
Live wood			
Pre-fire			
Total	108 (80)	75 (70)	39 (34)
HW	77 (62)	66 (61)	28 (24)
SW	122 (84)	80 (75)	55 (42)
Post-fire			
Total	105 (75)	47 (56)	1 (3)
HW	78 (61)	45 (57)	1 (3)
SW	117 (79)	48 (56)	1 (3)
Standing dead wood			
Pre-fire			
Total	6 (8)	5 (8)	2 (4)
HW	4 (8)	7 (11)	1 (1)
SW	7 (9)	5 (6)	5 (5)
Post-fire			
Total	9 (12)	22 (28)	27 (25)
HW	3 (6)	15 (19)	18 (16)
SW	12 (13)	26 (31)	41 (31)
Down wood			
Pre-fire			
Total	10 (9)	6 (6)	6 (8)
HW	6 (6)	4 (7)	4 (6)
SW	11 (9)	7 (6)	10 (8)
Post-fire			
Total	6 (8)	4 (6)	2 (2)
HW	4 (5)	3 (4)	1 (2)
SW	6 (8)	4 (7)	2 (3)

Note: Standard deviations are provided in parentheses. HW, hardwoods; SW, softwoods.

bon from this pool as snags decay, fall, and become down wood. The overall effect of forest type group was significant (p = 0.01). Estimates are provided by forest type group (Table 2), indicating no change in standing dead wood carbon in hardwood stands that burned at low severity. In stands that burned at high severity, standing dead wood carbon decreases at a slower rate in hardwood stands (4.2% per year) than in softwood stands (11% per year).

Down wood carbon increased at an annual rate of 13.3% (95% confidence interval, 6.4%, 20.7%) in the first 6 years after fire (p = 0.0001), after accounting for pre-fire carbon and fire severity. The observed increase in down wood carbon suggests that, at least in the first few years after fire, the decay of down wood is slower than the accumulation due to snag fall in all three severity classes. Although neither fire severity (p = 0.85) nor forest type group were significant in the model (p = 0.75) and a single estimate is sufficient, separate estimates by forest type group and by crown fire severity are provided for completeness (Table 2).

Evidence was insufficient to conclude that the carbon for all wood pools combined changes during the initial, 6 year post-fire recovery period (p = 0.14) after accounting for fire severity and pre-fire carbon (Table 2; Fig. 4, black line). Although not statistically significant at the 0.05 level, there was a decline in stand carbon with the magnitude of decline increasing as fire severity increases (Table 2). There was no evidence of an effect of forest type group on all wood carbon, after accounting for pre-fire carbon (p = 0.33).

Variability within fire severity classes

There was considerable variability in the size of the three woody carbon pools across all crown fire severities, both before and after fire (Figs. 3 and 4). Pre-fire carbon was highly correlated with post-fire carbon, with observed pre- and post-fire carbon being highest in stands that burned at low severity and lowest in stands that burned at high severity. Stands that burned at high severity were typically hardwood forests on low-productivity lands. Although 33% of the plots in hardwood forest types burned at high severity, only 14% of the plots in softwood forest types did so. Thirty-eight percent of the low-productivity plots burned at high severity, but only 11% of the moderate- and high-productivity plots burned at high severity. Because variability tends to increase with pool size, both pre- and post-fire woody carbon was more variable among plots that burned at low or moderate severities than among plots that burned with high fire severity (Fig. 3; Table 3).

During the initial period of post-fire recovery, there is remarkable diversity in sign and magnitude of the annualized change of live and dead wood for stands in all crown fire severity classes (Fig. 5). For live wood carbon, the variability in annualized change, expressed by the standard deviation (SD), was larger for stands that burned at low (SD = 6.9 Mg·ha^{-1}·year^{-1}) and moderate (SD = 6.2 Mg·ha^{-1}·year^{-1}) severities than in stands that burned at high severity (SD = 0.4 Mg·ha^{-1}·year^{-1}) (Fig. 5A). The live wood pool for low severity stands had many steep, negative slopes that were mirrored by positive slopes in the standing dead wood carbon pool for low severity stands (Fig. 5). A similar but less pronounced pattern was observed for stands that burned at moderate severity. For standing dead wood, the variability in annualized change was similar across all three burn severity classes with standard deviation ranging from 3.7 Mg·ha^{-1}·year^{-1} for both low severity and moderate severity stands to 3.2 Mg·ha^{-1}·year^{-1} for high severity stands (Fig. 5B). The variability in annualized change of down wood was highest in stands that burned at low severity (SD = 4.1 Mg·ha^{-1}·year^{-1}), followed by stands that burned at moderate severity (SD = 2.2 Mg·ha^{-1}·year^{-1}) and high severity (SD = 1.2 Mg·ha^{-1}·year^{-1}) (Fig. 5C).

Discussion

The use of national FIA data has the advantage of providing a spatially balanced probability sample of the area affected by the majority of the fires in California over a period of time, instead of a subjectively selected sample from one fire or a few fires. This sampling design greatly increases the inferential strength of the results for the fires included in the study and, to the extent that the 2002–2009 fires in this sample are representative of fires in California forests, to forest wildfires across the state. In addition, because the data come from a panelized inventory with permanent plots, pre-fire measurements are available. Pre-fire stand characteristics greatly influence fire effects, so including them in the models facilitates controlling for their influence on post-fire dynamics. In particular, plot measurements allow calculation of pre-fire carbon directly, a variable that is highly correlated with post-fire carbon. Including pre-fire carbon in the models increases the precision of the estimators and, more importantly, allows for inference conditional on the value of pre-fire carbon. Thus, the time trends estimated in this study are for stands with the same level of pre-fire carbon.

With pre-fire woody carbon and fire severity as explanatory variables in the model, neither broad forest type groups nor climatic variables contribute any additional information for the all wood, live wood, and down wood models. The standing dead wood carbon model, for which forest type group was significant, was an

Fig. 3. Observed trajectories (gray lines) and mean change (bold black lines) between post-fire and post-post-fire visits by carbon pool and crown fire severity classes (low, moderate, and high). Mean change by forest type group for standing dead wood carbon (bold dashed, softwoods; bold dots, hardwoods).

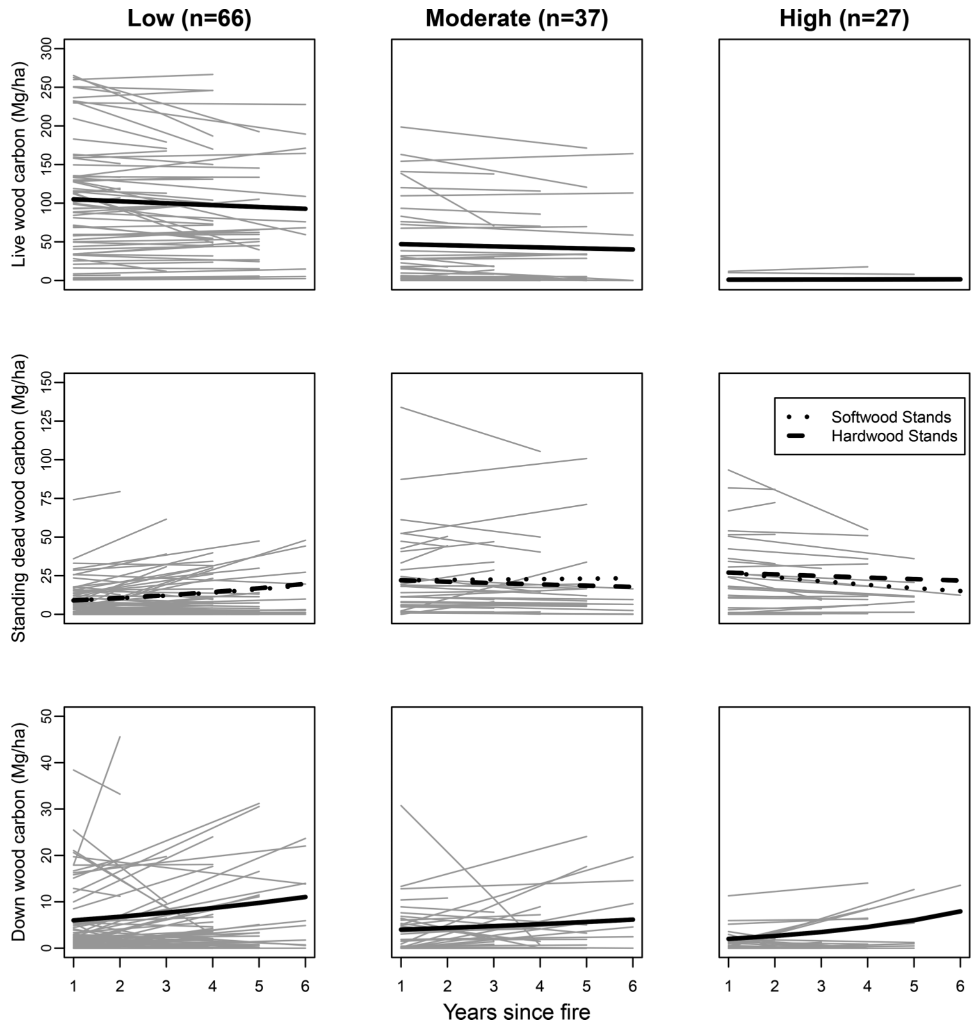


Fig. 4. Observed all wood carbon trajectories between post-fire and post-post-fire visit (solid gray lines, years 1 to 6) for the following three fire severities: (*a*), low; (*b*), moderate; (*c*), high. The red vertical line represents the fire. The pre-fire to post-fire carbon measurements are connected by gray, dashed lines to facilitate following the carbon trajectories of individual plots. Pre-fire measurements happened between 1 and 7 years before the fire. Shown are the mean pre-fire carbon (dots) and post-fire carbon (lines) by carbon pool, as well as all wood (black), live wood in (dark green), standing dead wood (dark brown), and down wood (light brown).

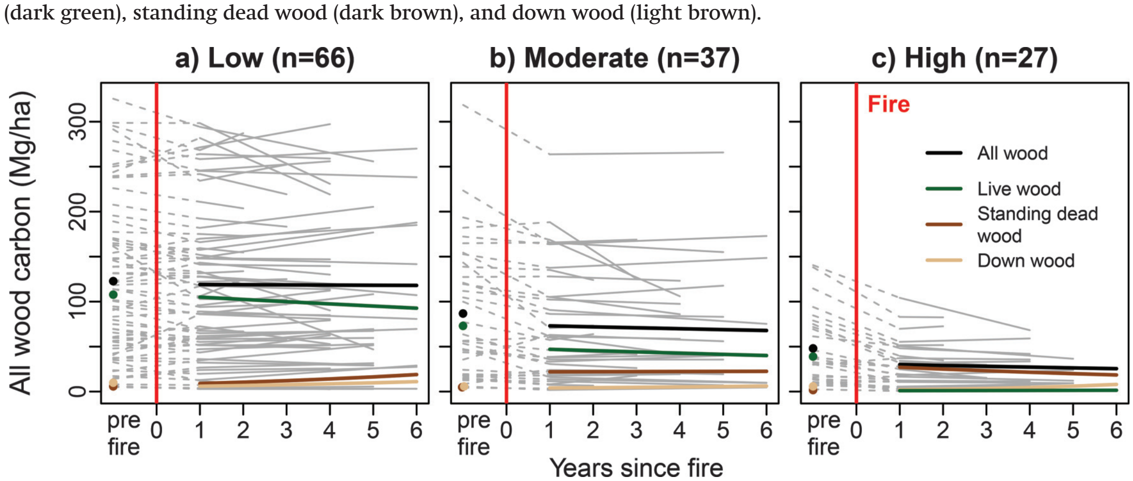
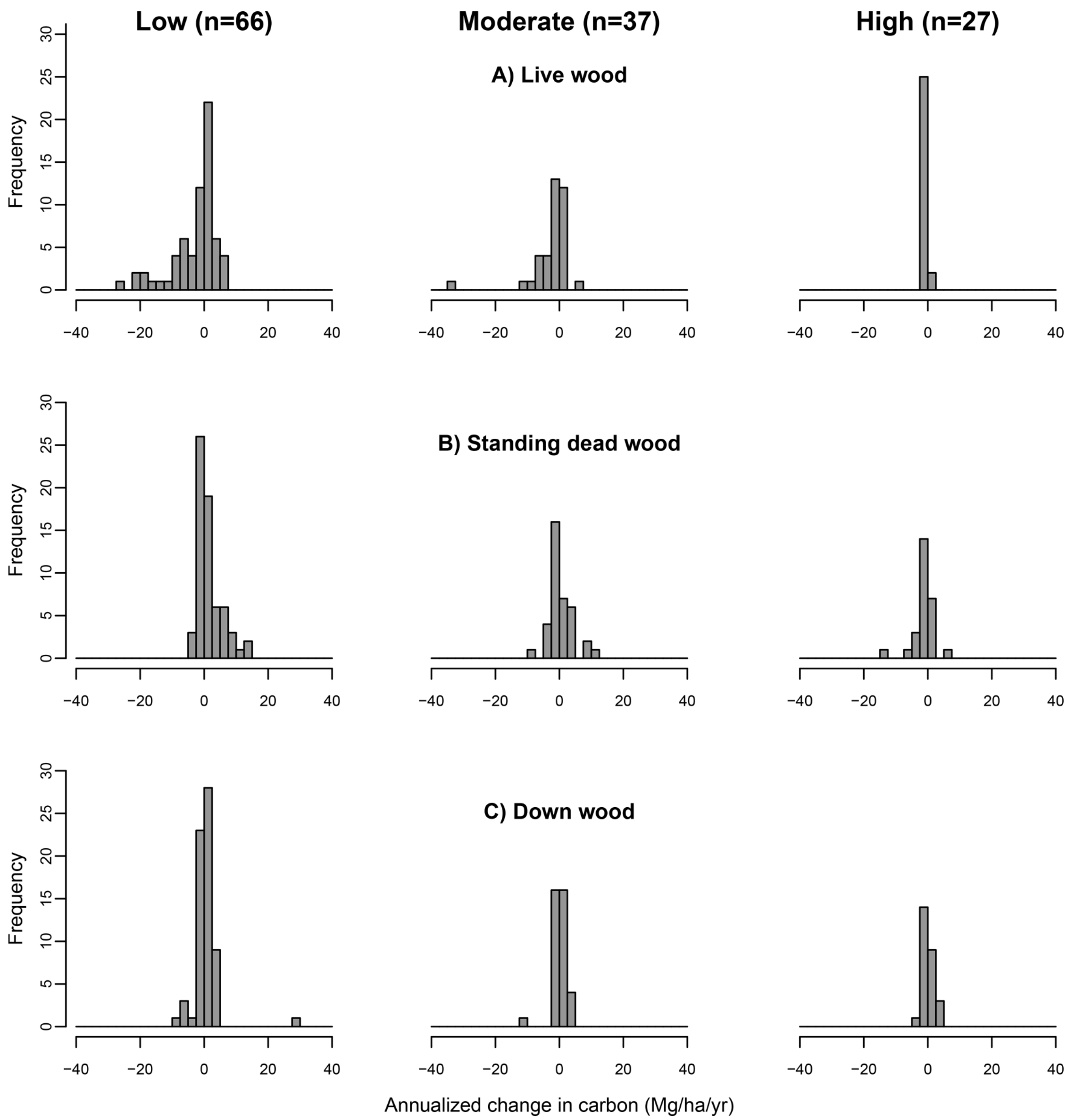


Fig. 5. Histograms of annualized post-fire change in carbon for each plot by carbon pool ((A), live wood; (B), standing dead wood; (C), down wood) and crown fire severity classes (low, moderate, and high).



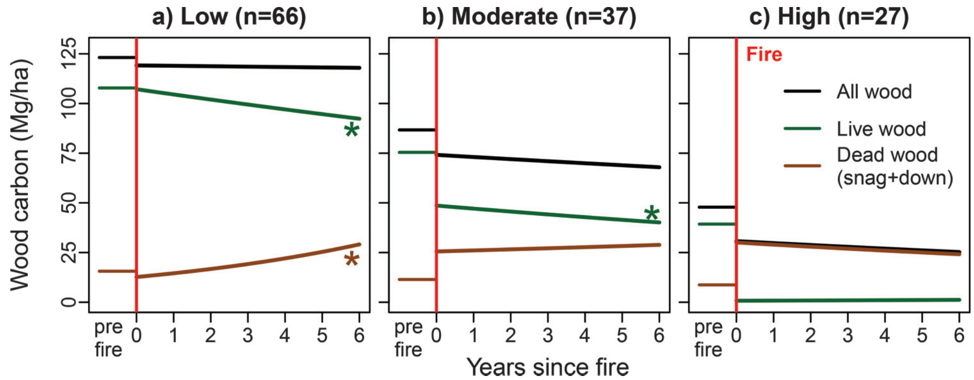
exception. Therefore, forest type group and climatic variables can be seen as surrogates for pre-fire carbon and are highly correlated with variables already included in the model. For example, California hardwood forests (mostly live oaks) tend to grow on less productive sites with comparatively higher temperatures, accumulate relatively low levels of aboveground carbon, and burn with high severity. It is important to note that creating broad forest type groups (hardwoods vs. softwoods) does not mean that there is no variability within these two forest type groups. However, the softwood–hardwood grouping may represent the strongest difference among stand types and species traits in terms of tree architecture, stand structure, and ability to resprout. We did not detect statistically significant differences between the two broad forest type groups after accounting for pre-fire woody carbon and fire severity in the models. It seems unlikely that differences for forest types within the broad softwood–hardwood groups could be detected.

Post-fire rates of change by carbon pool and fire severity

The post-fire carbon dynamics (1–6 years after fire) for plots that burned at high severity comport reasonably well with the first few years of the conceptual diagram provided by Ryan et al. (2010), depicting the post-fire carbon dynamics after stand-replacing fires in conifer stands (Fig. 6*c*). However, in all cases, the magnitudes of the rates of change of the carbon pools observed in this study are smaller. This may, to some degree, be due to the fact that we only looked at woody carbon pools, whereas the conceptual diagram presented by Ryan et al. (2010) may include foliage of live trees, as well as small trees and small down wood pieces that were excluded from our study. Although not statistically significant, the observed overall decline in woody carbon in the first 6 years following the fire in stands that burned at high severity is also consistent with post-fire total carbon decline observed for moderate and high severity fires in the eastern Cascades in Oregon, USA, which have been reported to act as net carbon sources 4–5 years after fire (Meigs et al. 2009).

However, plots that burned at low severity, which were far more numerous, show a very different pattern, with no evident decline in total wood carbon (Fig. 6*a*). This may be due to the difference in sign of the carbon changes in the different woody carbon pools that compensated each other (i.e., decreases in live

Fig. 6. Estimated mean post-fire dynamics for California forests' carbon pools following (*a*) low, (*b*) moderate, and (*c*) high severity fires. Significant trends are indicated by an asterisk.



wood but increases in dead wood). In the initial period of post-fire recovery, there were significant increases in both standing and down dead wood for stands that burned at low severity, resulting in a significant increase of dead wood carbon (standing dead + down wood carbon) (Fig. 6*a*). This suggests a trend opposite of that depicted in Ryan et al. (2010), where dead wood (standing and down dead wood combined) decrease in the years immediately following a stand-replacing fire. It is important to notice that, as presented in the literature, stand-replacing fire is typically assumed to imply a 100% reduction in live woody carbon, whereas our results in low severity and moderate severity stands show that live woody carbon decline due to fire only amounted to about 3% and 37%, on average, respectively (Figs. 6*a* and 6*b*). Although conceptual diagrams of carbon pool dynamics after stand-replacing fires have been published (e.g., Ryan et al. 2010), the results of this study suggest a need for such diagrams to represent the dynamics following low severity and moderate severity fires, which, according to this FIA sample, represent the majority of forest area burned by wildfire in California. In the first few years after fire, these diagrams would show a significant decrease in live woody carbon from delayed mortality in both low severity and moderate severity stands (Figs. 6*a* and 6*b*), an increase in dead wood carbon for stands that burned at low severity (Fig. 6*a*), and no change in dead wood carbon for stands that burned at moderate severity (Fig. 6i).

Fire severity and years since fire (1 and 8 years after fire) were also important factors influencing carbon pools for the 2002 Rodeo–Chediski Fire in Arizona (Yocom Kent et al. 2015). There, live carbon increased in low severity sites but declined in high severity sites, whereas we found a net decrease in low severity and moderate severity stands and no change in live wood carbon in high severity stands. Differences in findings may be at least partly attributable to differences in severity definitions with field-based severity used in this study versus remotely sensed severity used by Yocom Kent et al. (2015). In this study, damage to trees, reflected in delayed mortality, resulted in a decrease in live wood carbon for the years following fire in low and moderate severity stands. However, we were unable to identify specific causes for the delayed mortality (e.g., insects or post-fire drought). It may be that the low severity sites from the 2002 Rodeo–Chediski Fire in Arizona experienced less delayed mortality than our low severity and moderate severity stands and that the remaining live trees experienced increased growth rates after the low severity fire, which resulted in an increase in live woody carbon in the first few years after fire. Although not significant, stands that burned with high severity, especially those dominated by hardwood, exhibited an increase in live woody carbon. This may be due to trees identified during the post-fire visit as dead by outward appearance that had actually survived and to sprouting of apparently dead hardwood trees. Yocom Kent et al. (2015) reported declines in standing dead carbon in both low severity and high severity sites, whereas we observed an increase in standing dead carbon in low severity sites, likely owing to delayed mortality observed at our study sites. This difference may be attributed to the fact that Yocom Kent et al. (2015) collected data 2 years after fire, whereas this study was based on data typically collected within 6 months to 1 year after fire. Similar to the findings presented in this study, Yocom Kent et al. (2015) reported a higher increase in down wood accumulation in high severity sites than in low severity sites.

Yocom Kent et al. (2015) make the case for assessing post-fire carbon dynamics over longer periods of time using long-term monitoring of plots. This study offers an example for accomplishing this by capitalizing on a national forest inventory system with supplemental post-fire plot visits. As these plots continue to be assessed, the database will accumulate additional post-post-fire visits enlarging the sample available for this kind of analysis and enabling analysis of longer recovery periods than the 6 years considered here. This unique data set can provide highly resolved (e.g., by carbon pool, forest type, tree size) empirical information about post-fire carbon dynamics from a large, regional sample and serves as a foundation for monitoring long-term post-fire recovery in west coast forest ecosystems. At this point however, chronosequence studies are the only effective way to extend the time series of post-fire dynamics beyond the currently available empirical data.

This study summarized both a general pattern of carbon pool fluxes and mean annual rates of net change in the first 6 years after fire. Each carbon pool has gains and losses. For example, the live wood pool has large carbon losses due to delayed mortality, but it also has carbon gains due to ingrowth of young trees and possibly increased growth rates of the surviving trees. Because we only looked at net changes in the live woody carbon pool, we could only infer that delayed mortality outweighed any gains in live tree carbon due to growth. In future work, we plan to track the outcomes for individual trees to gain insights into the subtleties underlying the observed net carbon dynamics by pool.

Variability within fire severity classes

Most existing studies describe mean post-fire trends in carbon pools. The results of this study highlight that there is considerable variability in post-fire carbon trajectories at the observed scale, even within fire severity classes. This variability may be as important as the mean trend. The observed variability applied to both the carbon stocks, as well as the changes in the post-fire environment. The observed mean trend indicates that there was no change or relatively small change in total wood carbon. Yet, individual plots showed both increases and decreases in total wood carbon or in some of the carbon pools, and those observed changes could be quite large in magnitude.

We found that post-fire woody carbon was more variable in stands that burned with low and moderate severities, likely due to greater variability in mortality, compared with stands that burned at high severity, where mortality was close to 100%. Based on the definition of these burn-severity classes (Jain and Graham 2007), we expected comparatively low post-fire mortality and greater potential for post-fire growth in the live tree carbon pool for plots that were dominated by residual green crowns. On the other hand, we expected greater amounts of delayed mortality and little potential for rapid rebuilding of live tree carbon stores on plots where only a few trees with green crowns remained. In most stands that burned at high severity, all trees died, so live wood carbon is always close to zero in these forests. These stands also contain little down wood, because as less productive forests, they contained less down wood before the fire or because high fire severity led to greater depletion of down wood stocks. Substantial variability among low, moderate, and high fire severities has been previously reported (Schoennagel et al. 2004). The observed higher variability in stands that burned at low and moderate severities is given by the PFI definition of Jain and Graham (2007). A more nuanced analysis that retains all PFI levels has the potential to reduce the variability within PFI classes and may be feasible as the FERS database grows.

The higher variability in annualized change in stands that burned at low or moderate severities may be attributed to the fact that these stands contained more woody carbon before they burned. However, another reason may be the delayed mortality that occurred in some of those stands following the post-fire visit, generating losses in live wood carbon and gains in standing dead wood carbon.

Kashian et al. (2013) pointed out that fire severity and pre-fire structure may impact patterns of post-fire carbon accumulation that cannot be captured by chronosequences. This analysis of remeasured FIA plot data, which accounts for pre-fire carbon and fire severity, is invaluable for assessing post-fire carbon trajectories. Loehman et al. (2014) concluded that we lack information on long-term ecosystem responses to disturbance such as delayed mortality, recovery time, regeneration trajectories, and land cover changes. The observed high variability in carbon trajectories by pool and fire severity class in this study suggests that much remains to be learned about post-fire recovery, especially where fire severity is low to moderate.

Potential future work

Our analysis focused on standing live and dead trees with DBH greater than 12.7 cm and down wood pieces with diameter greater than 7.6 cm. Therefore, carbon dynamics of saplings, seedlings, woody shrubs and the smallest size classes of down wood were not included in our study. However, both regeneration and fine wood can represent a substantial woody carbon pool that may change quickly in the first years after fire. As the FERS database grows and future analyses are expanded to include a larger post-fire time frame and with a recent FIA protocol change to include dead saplings, there is potential to include these carbon pools, along with duff and litter, in future analyses.

Conclusions

The empirical information developed in this study from a spatially balanced sample significantly advances our understanding of carbon dynamics after wildfire in California, USA. We report post-fire woody carbon dynamics based on 32 wildfires, which account for about 42% of the area burned in large (>2000 ha) wildland fires that occurred in California between 2002 and 2009 and a much larger share of fires that burned in forestland, given that less than half of burned area in California is forested. There was no evidence of net annual woody carbon change in the first 6 years following wildfire in any of the three crown fire severity classes, where woody carbon is defined as woody carbon of standing trees with DBH greater than 12.7 cm and carbon of down wood pieces with diameter greater than 7.6 cm. Stands with larger amounts of pre-fire woody carbon tended to burn less severely and resulted in more variable post-fire trajectories by carbon pool than stands with less pre-fire woody carbon. The majority of stands in our study burned at low or moderate severity, suggesting that the majority of forest stands in California burn at less than high severity. Mixed severity fire, in which fire perimeters contain a heterogeneous mix of patches of various sizes that burn at low, moderate, or high severity, is typical of the mixed conifer forests that cover much of California and the northern western United States (Jain et al. 2012). Conceptual post-fire carbon trajectories postulated for stand-replacing, high severity fires are not a good match for representing fire outcomes in forests that are more prone to low or moderate severity fires. Improving our understanding of post-fire carbon trajectories in forest stands that burn at low and moderate severity will help us appropriately model and simulate post-fire carbon recovery dynamics in the majority of California's forest area that burns at these levels of severity.

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