

FINAL REPORT

Quantifying the effect of fuel size on charcoal formation during prescribed fire

Project ID: 13-3-01

Principle Investigator:

Dr. Matthew Hurteau
Department of Ecosystem Science and Management
Pennsylvania State University
State College, PA 16804
Phone: 814-865-7554
matthew.hurteau@psu.edu

Co-Principal Investigator:

Dr. Jason Kaye
Department of Ecosystem Science and Management
Pennsylvania State University
State College, PA 16804
Phone: 814-863-1614
ipk@psu.edu

Student Investigator:

Morgan Wiechmann
IGDP in Ecology
Pennsylvania State University
State College, PA 16804
Phone: 847-436-2431
morganwiechmann@gmail.com



Report prepared by Morgan Wiechmann

Abstract

Fire suppression, increasing temperature, and prolonged drought have resulted in increased wildfire frequency and severity in the western United States. Large and severe wildfires impact the carbon cycle both through direct emissions and reduced sequestration resulting from tree mortality. Forest thinning and prescribed burning reduce high-severity fire risk, but require removal and emissions of carbon. However, during each fire event not all biomass is emitted to the atmosphere. A fraction of the burning vegetation and soil organic matter is converted into charcoal, a stable carbon form. We hypothesized that charcoal carbon deposition from combusted coarse woody debris during prescribed burning would be greater than charcoal carbon deposition from non-combusted fine woody debris. We quantified post-treatment charcoal carbon formation in organic matter and the first 5 cm of the mineral soil from downed logs (> 30 cm diameter) that were present prior to treatment.

Our results indicate that there was no difference in the amount of charcoal carbon produced from combusted coarse woody debris and combusted fine woody debris. We also compared treatment effects on charcoal production and found that the burn-only, understory-thin and burn, and overstory-thin and burn treatments had significantly more charcoal carbon than the control. Charcoal carbon represented 0.19% of total ecosystem carbon, a relatively small fraction of total ecosystem carbon. However, charcoal carbon is long-lived and will likely continue to accumulate with repeated burning, leading to additional increases in long-term soil carbon storage. Given increasing efforts to reduce high-severity wildfire risk with thinning and burning, our results help improve our understanding of the effects these treatments have on ecosystem carbon flux by providing additional information of charcoal carbon formation.

Background and purpose

Anthropogenic climate change is projected to result in regional warming and drying in the western United States (Seager *et al.*, 2007; Solomon *et al.*, 2009; Seager and Vecchi, 2010). These changes in climate are projected to increase the frequency of large wildfires across the region (Pechony and Shindell, 2010; Westerling *et al.*, 2011). Additionally, 20th century fire suppression has increased tree density and the accumulation of forest floor biomass, making forests that evolved with a high-frequency, low-severity fire regime more prone to high-severity, stand-replacing fire events (Stephens and Ruth, 2005; Scholl and Taylor, 2010).

Fire emissions from these wildfire events represent a large and highly variable component of the United States carbon (C) budget (Wiedinmyer and Neff, 2007). Studies have shown that forests in the western United States can remain a net source of C for years to decades following a stand replacing wildfire event (Dore *et al.*, 2008; Meigs *et al.*, 2009). Since forests are valued globally for mitigating atmospheric carbon dioxide, concern about climate change has made mitigation of greenhouse gas emissions and increased C storage a priority for forest managers (Pan *et al.*, 2009).

Altering the forest structure and decreasing fuel loads through mechanical thinning and prescribed burning can increase forest resistance to stand-replacing wildfires (Agee and Skinner, 2005). By reducing the risk of stand-replacing wildfires forest managers may also be protecting against potential C losses, through reduced direct wildfire C emissions and increased

resistance to biome switching following the wildfire event (e.g. grasslands or shrublands) (Hurteau *et al.*, 2008; Hurteau and Brooks, 2011). This is assuming that the C losses associated with the management practices are less than the avoided C losses associated with the wildfire or alternate vegetation state (Hurteau *et al.*, 2008; Hurteau and Brooks, 2011).

Understanding the influence that prescribed fire has on C accumulation rates is essential for quantifying total ecosystem C flux to the atmosphere, and thus accurate C accounting. Various studies have investigated forest C losses from fire restoration management (Kaye *et al.*, 2005; Finkral and Evans, 2008; Hurteau and North, 2009; Stephens *et al.*, 2009; Hurteau and Brooks, 2011; Mitchell *et al.*, 2009), but few studies have investigated the potential long-term C storage resulting from prescribed fire. Fire contributes to long-term C storage by producing highly recalcitrant, pyrogenic charcoal, a C sink that has often been overlooked by fire scientists (DeLuca and Aplet, 2008; Donato *et al.*, 2009).

Pyrogenic charcoal is formed from the incomplete combustion of biomass, in the absence of oxygen. It is represented by a continuum of forms ranging from partly charred plant material to soot and graphite particles (Schmidt and Noack, 2000). Charcoal is C-rich and nitrogen-depleted, and has a highly aromatic molecular structure that contributes to increased resistance to microbial degradation, making charcoal a stable soil C component (Schmidt and Noack, 2000). Charcoal in forest soils has been documented to date more than 10,000 years back in time (Baldock and Smernik, 2002; Preston and Schmidt, 2006). This characteristically long mean residence time makes charcoal a significant contributor of global C sequestration in soils (Forbes *et al.*, 2006; Lehmann *et al.*, 2008; Preston, 2009).

Little direct field evidence has been collected to quantify soil charcoal content in forests that experience low-severity fire, such as prescribed burning. DeLuca and Aplet (2008) suggested that fuel reduction treatments that do not include prescribed burning may reduce soil charcoal content and thus, long-term C storage in mineral soils. A study in a Florida scrub-oak forest, found that a quarter of litter-fall was converted into charcoal following prescribed burning (Alexis *et al.*, 2006). In conifer forests located in the western US, potential combustible debris on the ground in the form of coarse woody debris (CWD) may provide a source for charcoal formation. In Sierra Nevada mixed-conifer forests CWD represents 3-8% of total ecosystem C, storing 12-27 Mg C ha⁻¹ (Hurteau and North, 2010; Wiechmann *et al.*, In Prep). Without directly quantifying potential charcoal C additions from combusted CWD, there may be an unaccounted long-term C storage pool. Furthermore, even if charcoal formation rates are relatively low, the long residence time of charcoal in soil may still have a substantial effect on estimated C flux from prescribed fire (DeLuca and Aplet, 2008).

The purpose of this study was to quantify the effects of pre-fire fuel load and fuel type on charcoal formation resulting from different fuel reduction treatments that included prescribed burning in a mixed-conifer forest. We tested the following hypotheses: (1) charcoal C formation is greater from the combustion of CWD than when combustion of smaller fuels occurs, (2) all forest stands that were burned would contain more charcoal C than forest stands that were not burned, and (3) CWD in treatments that were thinned and burned would produce more charcoal C than treatments that were only burned. To improve estimates of C contained in charcoal resulting from prescribed fire, we developed linear regression models that could be used to predict charcoal C formation from combusted CWD. Lastly, using our empirical results we expanded on the forest C sequence provided by DeLuca and Aplet (2008) to make century

scale estimates of potential charcoal C storage in a Sierran mixed-conifer forest that was either restored to the historical fire regime or the fire regime was not restored and experienced two wildfires.

Study description and location

This study was conducted in an old-growth mixed-conifer forest within the Teakettle Experimental forest, a 1300 ha reserve located approximately 80 km east of Fresno, CA in the Sierra Nevada. The climate is characterized as Mediterranean, with average annual precipitation of 125 cm, falling almost exclusively as snow (North *et al.*, 2002). The elevation ranges from 1900-2600 m. This mixed-conifer forest ecosystem is dominated by five tree species: *Abies concolor*, *A. magnifica*, *Calocedrus decurrens*, *Pinus jeffreyi*, and *P. lambertiana*. The Teakettle experiment, located in the Experimental Forest, was established to examine the ecological effects of a range of structural manipulations and prescribed burning. The experiment utilized a full factorial design, crossing three levels of thinning (no thin, understory-thin, overstory-thin) with two levels of burning (no burn, prescribed fire). Three replicates of each treatment were established using four-hectare treatment units. The understory-thin treatment removed all trees 25-75 cm diameter at breast height (DBH). The understory-thin was initially designed to reduce impacts on California spotted owl (*Strix occidentalis occidentalis*) habitat, although the guidelines now have been primarily used to reduce stand-replacing fire risk in Sierra mixed-conifer forests. The overstory treatment removed all trees greater than 25 cm DBH, with the exception of 22 large diameter trees ha⁻¹. The thin and burn plots were mechanically treated in 2000 and burned in 2001. The thin-only plots were treated in 2001. Prescribed fires were implemented during fall 2001 (North *et al.*, 2002). Within each treatment unit, 9-49 sub-plots were established for sampling understory vegetation and surface fuels.

Data Collection

Prior to treatment implementation, all trees were mapped, tagged, and measured. Permanent 10 m² circular plots were established at either 9 or 49 (intensive) points within the four-hectare treatments units. In each subplot, vegetation, woody debris, and litter depth were measured (North *et al.*, 2002). Using a modified version of the Brown's planar intercept method fine woody debris (FWD) was quantified at the hectare level (Brown, 1974). Following treatment implementation, the experimental units were re-sampled following the same protocol. Additional measurements within the circular plots of the burn treatments included visually estimating percent ash and percent char on all coarse woody debris (CWD) falling within the circular plots (Wayman and North, 2007). Following treatment implementation, all CWD was mapped and measured (Innes *et al.*, 2006).

Field Sampling

We focused our sampling efforts on charcoal formation from CWD (logs \geq 30 cm diameter) combusted during the 2001 prescribed fire. Sampled logs were chosen from a pre-

existing database that included logs that were mapped prior to the prescribed burn treatment. We used this spatially explicit database of log measurements to identify individual logs that were within 15 m of the monumented grid points within treatments that included fire. From the candidate population, we randomly selected 100 individual logs (10 logs from each treatment unit that included burning and 10 from the control). We collected seven soil cores at each log that encompassed organic matter and the top 5 cm of mineral soil. We constrained the mineral soil depth to 5 cm in an effort to limit sampling to charcoal particles formed during the 2001 prescribed fire. Organic matter was removed and separated prior to collecting the mineral soil. Three cores were taken directly adjacent to the log using 50 cm spacing on the down slope side of the log. Four additional cores were collected using a transect that ran perpendicular to the log from the mid-point. Off-log cores were spaced at 5 cm, 15 cm, 30 cm, and 60 cm down slope of the log. Soil cores were collected using a 10.16 cm diameter metal core. In addition to soil core samples, we sampled char depth at three locations on each log following the methodology of *Donato et al. (2009)*. We used these measurements to quantify the amount of char mass on the log (*Donato et al., 2009*).

Lab Analysis

We used 1 mm and 2 mm sieves to isolate charcoal macro-particles and collected 1 mm and 2 mm charcoal pieces from 536 of the subsamples to obtain the C contribution of each size class. Because the 2 mm size fraction captured the majority of charcoal C (Table 3.1), we focused the remainder of our sampling efforts exclusively on charcoal particles > 2 mm. In addition, *Nocentini et al. (2010)* found that over half of all C in charcoal was contained in pieces > 2 mm in size. Following methods similar to *Brimmer (2006)* and *Alexis et al. (2006)*, charcoal separation was performed on white trays under supplemental light using the unaided eye for the 2 mm size fraction and magnifying lamp with an enlargement factor of 175% for the 1 mm size fraction.

After charcoal was picked from the organic matter or soil, the charcoal samples were dried at 65°C, weighed and ground. We used the EA 1110 CHNS-O (CE Instruments 1996) elemental analyzer to obtain the percent carbon and nitrogen in all samples. The CHNS(O) Analyzer determines the percentages of C, H, N, S & O of organic compounds, based on the principle of "Dumas method" which involves the complete and instantaneous oxidation of the sample by "flash combustion" (*Matejovic, 1996*). We further analyzed for percent hydrogen in the mineral soil charcoal samples using the same elemental analyzer. Oxygen content in the mineral soil was calculated by the difference (*Nocentini et al., 2010*).

Data analysis

To test the hypothesis that more charcoal is produced from CWD than FWD, we used a linear regression analysis to compare the charcoal weight (g) of samples collected at the different distances. Charcoal in organic matter and charcoal in mineral soil were analyzed separately. We detected no significant differences between charcoal mass (g) collected adjacent to or at varying distances from the logs within each treatment unit. This finding was

consistent for both organic matter and mineral soil. Since there was no effect of distance from log, all samples were used to determine charcoal C production at individual logs.

A nested analysis of variance (ANOVA) was used to compare treatment effects on charcoal C production at logs. Charcoal C samples from each log were averaged and converted to an areal scale. C concentrations were scaled to an areal basis (g m^2) using the volume and depth of the soil corer, data on log length, the perpendicular transect length (60 cm) and previously derived bulk density (BD) measurements at Teakettle experimental forest (mineral BD: 0.95 g cm^{-3} , organic matter: 0.1 g cm^{-3}). Cores were treated as subsamples with the sampling unit being the log ($n=30$ for treatments that burned, $n=10$ for un-burned treatments). Data were log-transformed to meet all assumptions for the ANOVA. We used Tukey's HSD post hoc analysis to determine if there were significant ($p < 0.05$) differences between treatments.

To estimate charcoal production in each treatment unit we obtained information on log length for all logs in the treatment units using the pre-treatment log dataset. Using measured values for each log within each treatment unit, we estimated charcoal C produced at mapped logs in each treatment unit to obtain an estimate of total C production in each 4 hectare treatment unit ($\text{Mg charcoal C ha}^{-1}$). We again used ANOVA to draw comparisons of charcoal C production among the different treatments at the hectare scale. Based on this estimate we were able to compare charcoal C amount with different C pools (e.g. total soil, live tree, etc) on a per hectare basis. Again, data were log-transformed to meet the ANOVA assumptions of normality and equal variance and Tukey's HSD post hoc analysis was used to determine if there were significant ($p < 0.05$) differences between treatments.

To quantify the effect of prescribed fire on charcoal formation, we used linear regression models in an information theoretic framework (Burnham and Anderson, 2002). The variables included in the analysis were pre-treatment fine woody debris (Mg C ha^{-1}); pre-treatment litter depth (cm), 2002 percent coarse woody debris charred, 2002 percent ash cover, and 2013 char mass on sampled CWD (Mg C). From these predictor variables we developed a candidate set of models that included charcoal C (g C m^2) as the response variable. We ranked all possible model combinations for each treatment using the Akaike Information Criterion for small samples (AIC_c) (Burnham and Anderson, 2002). We chose the best model (lowest AIC_c value) to estimate charcoal C as a function of the fire effect predictors for each treatment (burn-only, understory-thin and burn, and overstory-thin and burn). Data were log-transformed to meet the assumptions of normality and equal variance.

200-year forest C sequence

Following DeLuca and Aplet (2008) we created a hypothetical scenario to estimate the accumulation of aboveground biomass (live trees and snags), CWD, charcoal C, and fire emissions extended 200 years into the future. Using previous data collected at the Teakettle experimental forest, the empirical results of this study, and additional assumptions gathered from the literature, we projected forest C stocks in an old-growth mixed-conifer forest under two hypothetical scenarios where the forest was thinned from below, then subject to (1) prescribed fire, followed by 9 additional prescribed burns over a 200 year period, or (2) no prescribed fire, followed by two high-severity fires over a 200-year period.

In the first forest C sequence (understory thin and burn followed by 9 prescribed fires at 20-year intervals) aboveground biomass was calculated using site-specific growth rates ($\text{Mg C ha}^{-1} \text{ yr}^{-1}$) for the 10-years following the initial understory thin and burn and the 9 repeated prescribed fires. Wiechmann *et al.* (In Prep) quantified a 10-year growth release of $1.8 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ following the initial thin and burn and an aboveground biomass accumulation rate of $3.9 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. For this scenario we assumed the accumulation rate reported by Wiechmann *et al.* (In Prep) for the 10-years following each prescribed burn over the 200 year sequence. During the second decade following each burn, aboveground biomass was assumed to accumulate at $0.5 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ (Houghton *et al.*, 2000; Law *et al.*, 2001; Hicke *et al.*, 2006). CWD was assumed to increase $1.95 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ during the 10-years following the initial thin and burn, and $0.27 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ during the second decade after each repeated burn based on the thin and burn and burn only treatment results presented by Wiechmann *et al.* (In Prep). Annually, 1.55% of CWD was assumed to decompose during years that did not follow treatments (Harmon *et al.*, 1987). Prescribed fires were assumed to consume 6.8% of the aboveground biomass, and 6.1% of CWD (North *et al.*, 2009). Charcoal C content was $0.66 \text{ Mg C ha}^{-1}$ following the initial understory-thin and burn treatment (Table 3, understory-thin and burn). Additional charcoal C inputs from the repeated prescribed fire were assumed to be $0.68 \text{ Mg C ha}^{-1}$ per fire (Table 3, burn-only). We assumed that half of the charcoal was consumed in each subsequent fire following the initial understory-thin and burn following DeLuca and Aplet (2008).

In the second sequence (understory-thin only, followed by two high-severity wildfires), we projected aboveground biomass from site-specific simulations from Hurteau and North (2009). CWD C was projected from results of Wiechmann *et al.* (In Prep). Values and assumptions from the literature and empirical results from this study were used to predict charcoal C. Hurteau and North (2009) simulated one mid-century wildfire occurring over a 100 year period after an initial understory-thin treatment; we expanded this to include a second mid-century wildfire. CWD accumulated 0 Mg C ha^{-1} 10 years following the understory thin treatment (understory thin in Wiechmann *et al.* (In Prep)). Equivalent to the first forest sequence, a background decay rate of $1.55\% \text{ yr}^{-1}$ was assumed for CWD (Harmon *et al.*, 1987). Fifty percent of CWD was assumed to be consumed by wildfires, with a 5-fold increase in CWD 10-years following the wildfire event (Dore *et al.*, 2008). Before the wildfire, background charcoal C was assumed to be $0.03 \text{ Mg C ha}^{-1}$ (Table 3, control). Additional charcoal inputs from the wildfires was assumed to be 2% of CWD per fire (DeLuca and Aplet, 2008), with half of the charcoal being consumed during each fire event (DeLuca and Aplet, 2008).

Understory thin and burn C releases were quantified from the site-specific fire emissions, $23.4 \text{ Mg C ha}^{-1}$ (North *et al.*, 2009). For this study we concentrated on emissions that were biologically related (fire emissions) and did not include C releases from other sources such as milling waste, equipment use, long-lived wood products, and transportation to the mill. The repeated prescribed burn scenario emitted $14.8 \text{ Mg C ha}^{-1}$ per fire event (North *et al.*, 2009). Wildfire emissions were assumed to be $25.8 \text{ Mg C ha}^{-1}$ per fire, based on fire emissions from the understory thin simulation from Hurteau and North (2009).

Findings

CWD and charcoal C production

We hypothesized that charcoal produced from CWD would be greater than charcoal produced from fine fuels; however no significant differences were detected between charcoal C adjacent to the logs and sampled perpendicular to the logs. There are two potential causes of this finding, 1) either there is in fact no difference in charcoal production between the different fuel size classes or 2) the result is an artifact of our sampling design and we did not sample at a great enough distance from the logs to exclude the influence of charcoal produced from CWD. Charcoal pieces that were formed from the combustion of CWD and shed from the sampled log may have been transported down slope of the log either by gravity or erosion.

Treatment effect on charcoal C production

To estimate the impact fuel reduction treatments have on charcoal C formation as a function of CWD availability, we scaled the amount of charcoal formed from the combustion of logs to an areal extent (Fig. 2). All treatments that included burning, in both soil layers, had significantly more C than the control (control, organic matter: 3.7 g C ha^{-1} , mineral soil: 30.3 g C ha^{-1}) (Fig. 2). The only statistical differences detected between treatments that were burned were in organic matter charcoal C, where charcoal C formed in the overstory-thin and burn treatment (93.7 g C ha^{-1}) was less than the burn only and understory-thin and burn treatments ($352.2 \text{ g C ha}^{-1}$, $242.8 \text{ g C ha}^{-1}$, respectively) (Fig. 2). Overall, more charcoal C was contained in the mineral soil layer than in the organic matter, suggesting that there is vertical movement of charcoal C down the soil profile, increasing the probability charcoal will remain sequestered and reside on site for a longer duration (Fig. 2).

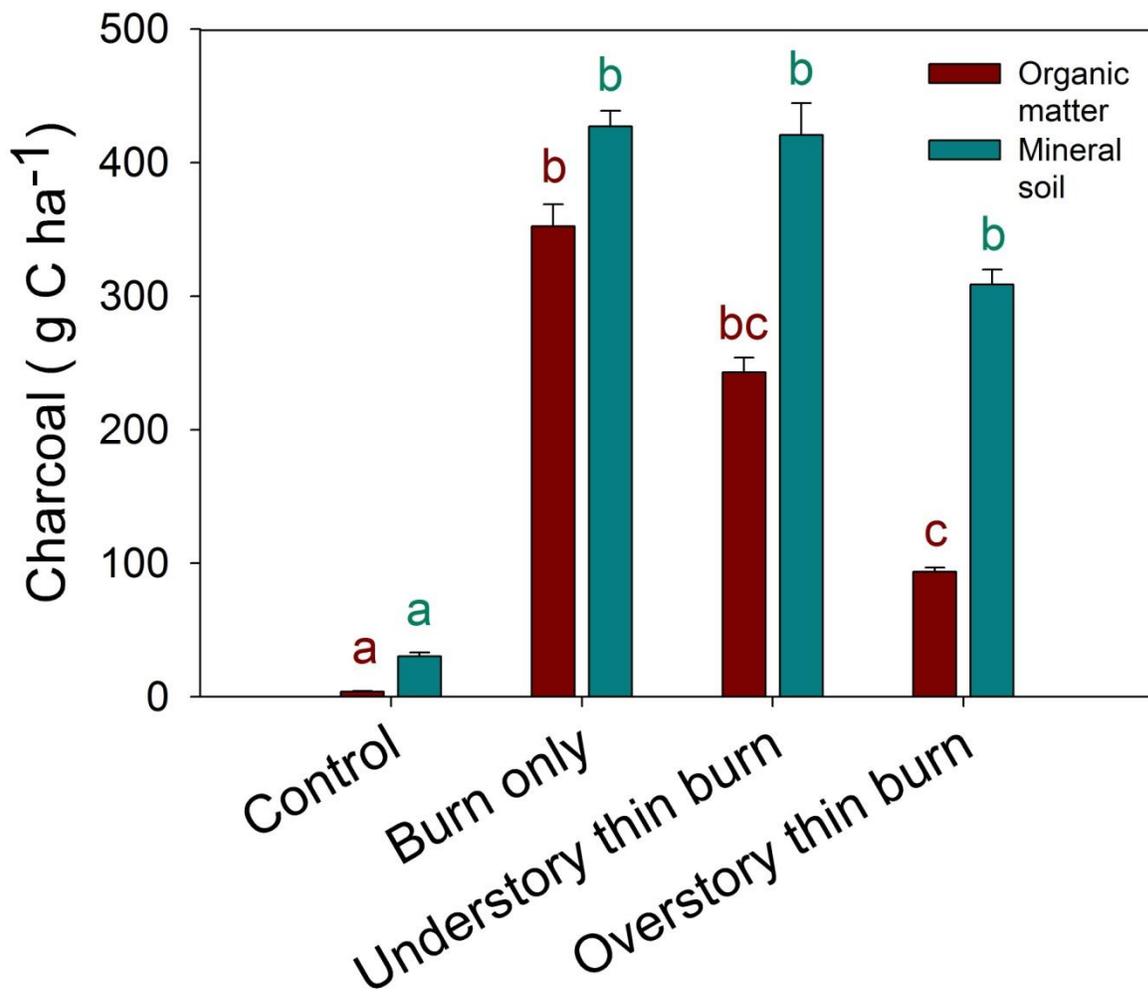


Figure 1. Using the pre-fire log dataset, we estimated the mean charcoal C (g C ha⁻¹) produced in each treatment unit that included prescribed burning and the control for charcoal found in organic matter (red) and mineral soil (blue). Carbon stock comparisons were made across treatments for charcoal in the same soil profile. Same colored bars with different letters are significantly different ($p \leq 0.05$) ($n=3$). Charcoal C was log transformed to meet ANOVA assumptions.

To determine the relative contribution of charcoal to total ecosystem C within treatments, we compared the results from this research to a previous study. Wiechmann *et al.* (In Prep) found that total ecosystem C among treatments ranged from 201-441 Mg C ha⁻¹, with live tree C (78-287 Mg C ha⁻¹) having the largest stock. Of the treatments that included prescribed burning total ecosystem C ranged from 201-398 Mg C ha⁻¹, with live tree C (78-259

Mg C ha⁻¹) having the largest stock in burn treatments (Wiechmann *et al.*, In Prep). Charcoal contributed 0.40-0.68% of total ecosystem C storage, a relatively small amount (Table 1).

Table 1. The 2013 charcoal C stock size and the C stock size (Mg C ha⁻¹) of different pools 10-years post-treatment (2011) for the control and three burn treatments. Values in parentheses are standard errors from mean.

	Control	Burn only	Understory thin burn	Overstory thin burn
Live tree	286.5 (34.4)	259.1 (14.5)	169.9 (3.87)	78.4 (19.1)
Snag	45.6 (10.1)	45.2 (1.26)	48.1 (8.36)	28.0 (2.50)
CWD	15.4 (1.99)	12.1 (4.44)	26.6 (6.35)	17.1 (3.59)
FWD	12.7 (1.45)	10.2 (0.78)	7.03 (2.79)	8.19 (0.89)
Litter and duff	5.20 (0.83)	4.90 (0.93)	4.83 (1.47)	2.91 (0.53)
Soil	75.2 (10.3)	67.3 (4.71)	84.8 (5.80)	66.7 (1.45)
Shrub	0.02 (0.00)	0.02 (0.00)	0.04 (0.01)	0.07 (0.01)
Charcoal	0.03 (0.003)	0.68 (0.02)	0.66 (0.03)	0.40 (0.01)
Total	440.7 (38.6)	398.9 (34.9)	331.3 (21.9)	201.3 (11.9)
Charcoal % of total	0.007%	0.17%	0.20%	0.20%

Predicting charcoal C

Of the candidate set of predictors, across treatments and soil layers, log char was a very influential predictor of charcoal C production from combusted CWD of the candidate set of predictor variables (Table 2 and Table 3). In several cases, models with log char as the predictor were ranked as the second best model, based on AIC_c values (Table 2 and Table 3). However, in these cases AIC_c values were within two of the best model, indicating equal support. Log char explained between 18 and 35% of the charcoal C variance in the organic matter layer and 29-49% of the variance in the mineral soil layer (Table 2 and Table 3). While charring on logs is a contributing factor to charcoal production, other predictor variables, such as those relating to fire behavior, may improve the ability to estimate charcoal production from prescribed burning.

Table 2. Top two models for each treatment with the smallest AIC_c values for modeling charcoal C (g C m²) production in organic matter. Δ AIC_c is the difference in AIC values from the model with the lowest AIC_c.

Treatment	Predictor(s)	AIC _c	Δ AIC _c	R ²
Burn only	Fine woody debris (Mg C ha ⁻¹)	5.50	0	0.136
	Log char (Mg C)	5.55	0.018	0.184
	Fine woody debris (Mg C ha ⁻¹) + percent ash (%)	9.81	4.30	0.137
Understory thin and burn	Fine woody debris (Mg C ha ⁻¹)	5.84	0	0.09
	Log char (Mg C)	6.01	0.176	0.257
Overstory thin and burn	Log char (Mg C)	6.71	0	0.354
	CWD charred (%) + log char (Mg C)	11.03	4.32	0.365

Table 3. Top two models for each treatment with the smallest AIC_c values for modeling charcoal C (g C m²) production in the first 5cm of the mineral soil. Δ AIC_c is the difference in AIC values from the model with the lowest AIC_c.

Treatment	Predictor(s)	AIC _c	Δ AIC _c	R ²
Burn only	CWD charred (%)	6.06	0	0.161
	Log char (Mg C)	6.49	0.04	0.487
Understory thin and burn	Log char (Mg C)	6.20	0	0.449
	CWD charred (%) + percent ash (%)	10.1	3.89	0.116
Overstory thin and burn	Percent ash (%)	8.6	0	0.247
	Log char (Mg C)	8.63	0.05	0.289

Implications for forest management and century-scale C storage

We provide a 200-year comparison of future development of two forests to illustrate the long-term potential of charcoal C storage from different restoration practices (Fig. 3.4). In this 200-year hypothetical forest sequence, adapted from DeLuca and Aplet (2008), a forest that was thinned once and experienced two severe wildfires stored less total C (379 Mg C ha⁻¹) than a forest system that was fire-maintained by prescribed surface fires (437 Mg C ha⁻¹) (Fig. 3.4). Total C emissions were greater in the fire-maintained system (156 Mg C ha⁻¹), than the forest stand that was not burned with prescribed fire (52 Mg C ha⁻¹ released) (Fig. 3.4). The repeated prescribed burn sequence accumulated approximately 1.4 times as much live tree and snag biomass (418 Mg C ha⁻¹) than the wildfire sequence (299 Mg C ha⁻¹) (Fig. 3.4). C contribution from aboveground biomass (snags and live trees) in the understory thin and burn treatment (418 Mg C ha⁻¹) greatly increased total C storage (Fig. 3.4). Long-term C storage in the form of charcoal was 2.8 times greater in the forest stand that was restored (3.72 Mg C ha⁻¹) than in the wildfire sequence (1.34 Mg C ha⁻¹) (Fig 3.4). There was an additional 2.72 Mg charcoal C ha⁻¹ produced in the restored sequence than in the wildfire forest sequence (Fig. 3.4). At year 2200, charcoal C storage was equivalent to 2.4% of emissions in the fire

maintained forest sequence; similarly charcoal C storage was equivalent to 2.6% of total emissions in the stand-replacing fire sequence.

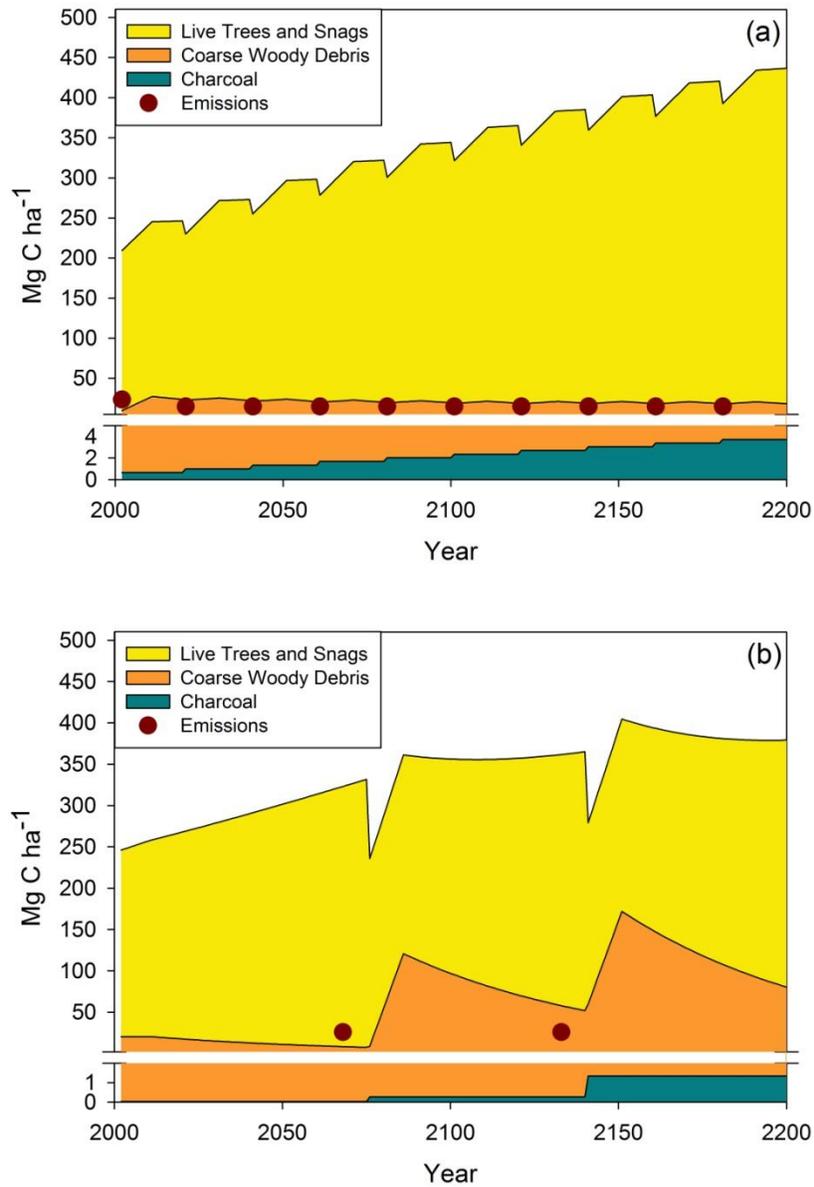


Figure 2. Carbon stocks and emissions from a hypothetical 200-year forest sequences. The two hypothetical sequences: (a) an initial understory-thin followed by repeated prescribed burning at 20-year intervals, and (b) an initial understory-thin followed by two severe wildfires. Note the different y-axis scales below the break.

Management Implications

Our estimates of C accumulation in charcoal with repeated burning suggest that a simple mass-balance approach to estimating emissions from fire may yield an overestimate of emissions and an underestimate of post-fire total ecosystem C (Fig. 2). Similar to DeLuca and Aplet (2008), our findings provide further information to land managers to more accurately quantify C stocks on the landscape. Further understanding of the effects of restoring natural fire regimes on charcoal C will improve our ability to quantify the effects of restoration treatments on forest C dynamics.

From our model results (C dynamics under scenarios with and without repeated burning) restoring a natural fire regime in these historically frequent-fire forests could result in a much larger fraction of total ecosystem C stored in this recalcitrant C form in comparison to a scenario that only includes burning by wildfire (Fig. 2). The increased C storage from prescribed burning, as opposed to not implementing prescribed burning (Fig. 1), provides additional information for fuel-reduction treatment decision making to reduce forest fire risk and stabilize or increase C sequestration in the forest system.

Charcoal C is an expensive and time-consuming measurement for forest managers to obtain. Our linear models help estimate charcoal C accumulation as a function of more attainable data. We developed a candidate set of models from models that included charcoal as the response and all possible combinations of % cover of ash, mineral soil, organic cover, and mean scorch height of adjacent trees as predictor variables. The results of our regression analysis found that log char mass explained 18-35% of the variation in organic matter charcoal and 28-48% of the variation in mineral soil charcoal (Table 2 and Table 3).

Relationship to other recent findings and ongoing work

DeLuca and Aplet (2008) suggested that fuel reduction treatments that do not include prescribed burning may reduce soil charcoal content and thus, long-term C storage in mineral soils. Using our empirical results we expanded on the forest C sequence provided by DeLuca and Aplet (2008) to make century scale estimates of potential charcoal C storage in a Sierran mixed-conifer forest that was either restored to the historical fire regime or the fire regime was not restored and experienced two wildfires. Our findings were similar to those of DeLuca and Aplet (2008). Our estimates of C accumulation in charcoal with repeated burning suggest that a simple mass-balance approach to estimating emissions from fire may yield an overestimate of emissions and an underestimate of post-fire total ecosystem C.

In a study of charcoal content in the Sierra Nevada Mountains, Mackenzie *et al.* (2008) found little to no spatial auto-correlation in charcoal content across the forest. We rejected our hypothesis that soil charcoal content would vary as a function of proximity to coarse woody debris. These two findings in combination suggest that both coarse and fine woody debris may be contributing to charcoal production as a function of mass per unit area. This finding warrants additional investigation because if there is no differential effect of fuel size on charcoal formation, our treatment-level estimates of charcoal production are an underestimate of the actual charcoal produced. MacKenzie *et al.* (2008) found charcoal C content in

Sierran mixed-conifer forests to range from 2.0-4.5 Mg C ha⁻¹ in the surface 6 cm of mineral soil, and exhibited a constant charcoal C amount with increased soil depth up to 60 cm. Our study quantified considerably less charcoal C in the same forest type (0.03-0.68 Mg C ha⁻¹, Table 1). One possible explanation for the discrepancy in results could be methodological. We used visual separation to quantify charcoal and focused on macro-particles, while MacKenzie *et al.* (2008) used a chemical extraction method that was not constrained to macro-particles.

In this study, *Charcoal and Total Carbon in Soils from Foothills Shrublands to Subalpine Forests in the Colorado Front Range*, Licata and Sanford (2012) quantified charcoal and total C in soils across an elevation gradient in the Colorado Front Range. They reported that the most important influences on soil charcoal C formation are available biomass sources and the fire regime. Also, lignin-based surface fuels (woody debris) and deep duff layers were two large charcoal C sources that resulted in relatively higher charcoal C per fire (Licata and Sanford, 2012). In our linear models, used to predict charcoal C formation, that included litter depth were not among the best models to predict charcoal C in this study, pre-fire litter depth was, on average, greatest in the burn only treatments (1.75 cm) followed by the understory-thin and burn (1.3 cm) and overstory-thin and burn litter depths (0.58 cm). Instead, the results of our regression analysis found that log char mass explained 18-35% of the variation in organic matter charcoal and 28-48% of the variation in mineral soil charcoal (Table 2 and Table 3). As a result of our findings on treatment effects on charcoal production, we hypothesize that predictor variables relating to fire behavior will improve the ability to estimate charcoal production from more easily obtained measurements. Fire intensity measurements such as fire-line intensity may help better predict charcoal formation; oxygen availability, and fire duration have also been cited as possible causes of differences in charcoal production (Carvalho *et al.*, 2011). In addition, fire temperature may also be an important charcoal formation variable, because of its direct effect on charcoal structure and reflectivity (Cerdeira, 2010).

Future work needed

The conclusions of this study have given rise to additional questions that need to be tested to build on our understanding of charcoal C formation and long-term C storage. Below are areas of research to build on our current understanding of this topic:

1. Laboratory and *in situ* research to improve our understanding of the relationship between fuel size and fire characteristics and the influence on charcoal formation rates. Ideally this work would include accurate measures of fire duration, temperature, and proportion of smoldering combustion, among others.

2. In this research project we found that more charcoal C was contained in the mineral soil layer than the organic matter, suggesting that there is vertical movement of charcoal C down the soil profile. We hypothesized that the movement of charcoal vertically down the soil profile could be attributed to leaching of particles carried downward through suspension (lessivage) or bioturbation (reworking of soils and sediments by animals or plants). However, this has not been tested. Future research involving the movement of charcoal through the soil profile would provide information useful for modeling belowground C dynamics.

4. Additional work is needed to quantify the production of charcoal in wildfires and consumed during subsequent prescribed burns. These data would provide empirical data that would improve total ecosystem C modeling efforts and emissions estimates from burning.

Deliverables

Proposed	Delivered	Status
Masters' Thesis	Thesis defended in June, submitted in July	Complete July 2014
Scientific meeting presentation	Wiechmann, M.L., M.D. Hurteau. The effect of thinning and burning on charcoal formation and carbon storage in a mixed-conifer forest, Sierra Nevada, California. 2014 meeting of the Ecological Society of America.	Complete Aug 2014
Peer-reviewed paper	One paper in preparation for submission to Journal of Geophysical Research-Biogeosciences	In prep
Research Brief	Research brief for managers summarizing results of this project, available on www.hurteaulab.org	Complete
Final Report	Final report and metadata will be delivered to Joint Fire Science Program at completion of project	Complete Sep 2014

References

- Agee, J.K., Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. For. Ecol. Manage. 211, 83-96.
- Alexis, M.A., Rasse, D.P., Rumpel, C., Bardoux, G., Péchot, N., Schmalzer, P., Drake, B., Mariotti, A., 2006. Fire impact on C and N losses and charcoal production in a scrub oak ecosystem. Biogeochemistry 82, 201-216.
- Baldock, J.A., Smernik, R.J., 2002. Chemical composition and bioavailability of thermally altered *Pinus resinosa* (Red pine) wood. Organic Geochemistry 33, 1093-1109.
- Brimmer, R.J., 2006. Charcoal Quantity and Adsorptive Activity in Ponderosa Pine Ecosystems of Western Montana. In. University of Montana.
- Brown, J.K., 1974. Handbook for inventorying downed woody material.
- Burnham, K.P., Anderson, D.R., 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer.
- Cerda, A., 2010. Fire effects on soils and restoration strategies. CRC Press.
- DeLuca, T.H., Aplet, G.H., 2008. Charcoal and carbon storage in forest soils of the Rocky Mountain west. Frontiers in Ecology and the Environment 6, 18-24.
- Donato, D.C., Campbell, J.L., Fontaine, J.B., Law, B.E., 2009. Quantifying char in postfire woody detritus inventories. Fire Ecology 5, 104-115.
- Dore, S., Kolb, T.E., Montes-Helu, M., Sullivan, B.W., Winslow, W.D., Hart, S.C., Kaye, J.P., Koch, G.W., Hungate, B.A., 2008. Long-term impact of a stand-replacing fire on ecosystem CO₂ exchange of a ponderosa pine forest. Glob. Change Biol. 14, 1801-1820.

- Finkral, A.J., Evans, A.M., 2008. The effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. *Forest Ecology and Management* 255, 2743-2750.
- Forbes, M.S., Raison, R.J., Skjemstad, J.O., 2006. Formation, transformation and transport of black carbon (charcoal) in terrestrial and aquatic ecosystems. *Sci Total Environ* 370, 190-206.
- Harmon, M.E., Cromack Jr., K., Smith, B.G., 1987. Coarse woody debris in mixed-conifer forests, Sequoia National Park, California. *Canadian Journal of Forest Research* 17, 1265-1272.
- Hicke, J.A., Logan, J.A., Powell, J., Ojima, D.S., 2006. Changing temperatures influence suitability for modeled mountain pine beetle (*Dendroctonus ponderosae*) outbreaks in the western United States. *Journal of Geophysical Research* 111, G02019.
- Houghton, R.A., Hackler, J.L., Lawrence, K.T., 2000. Changes in terrestrial carbon storage in the United States. 2: the role of fire and fire management. *Global Ecology and Biogeography* 9, 145-170.
- Hurteau, M., North, M., 2009. Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. *Frontiers in Ecology and the Environment* 7, 409-414.
- Hurteau, M.D., Brooks, M.L., 2011. Short- and Long-term Effects of Fire on Carbon in US Dry Temperate Forest Systems. *Bioscience* 61, 139-146.
- Hurteau, M.D., Koch, G.W., Hungate, B.A., 2008. Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. *Front. Ecol. Environ.* 6, 493-498.
- Hurteau, M.D., North, M., 2010. Carbon recovery rates following different wildfire risk mitigation treatments. *Forest Ecology and Management* 260, 930-937.
- Innes, J.C., North, M.P., Williamson, N., 2006. Effect of thinning and prescribed fire restoration treatments on woody debris and snag dynamics in a Sierran old-growth, mixed-conifer forest. *Can. J. For. Res.-Rev. Can. Rech. For.* 36, 3183-3193.
- Kaye, J.P., Hart, S.C., Fule, P.Z., Covington, W.W., Moore, M.M., Kaye, M.W., 2005. Initial carbon, nitrogen, and phosphorous fluxes following ponderosa pine restoration treatments. *Ecological Applications* 15, 1581-1593.
- Law, B.E., Thronton, P.E., Irvine, J., Anthoni, P.M., Van Tuyl, S., 2001. Carbon storage and fluxes in ponderosa pine forests at different developmental stages. *Global Change Biology* 7, 755-777.
- Lehmann, J., Skjemstad, J., Sohi, S., Carter, J., Barson, M., Falloon, P., Coleman, K., Woodbury, P., Krull, E., 2008. Australian climate-carbon cycle feedback reduced by soil black carbon. *Nat. Geosci.* 1, 832-835.
- Licata, C., Sanford, R., 2012. Charcoal and Total Carbon in Soils from Foothills Shrublands to Subalpine Forests in the Colorado Front Range. *Forests* 3, 944-958.
- MacKenzie, M.D., McIntire, E.J.B., Quideau, S.A., Graham, R.C., 2008. Charcoal Distribution Affects Carbon and Nitrogen Contents in Forest Soils of California. *Soil Science Society of America Journal* 72, 1774.
- Matejovic, I., 1996. The application of Dumas method for the determination of carbon, nitrogen and sulphur in plant samples. *Rostlinna Vyroba-UZPI (Czech Republic)*.

- Meigs, G.W., Donato, D.C., Campbell, J.L., Martin, J.G., Law, B.E., 2009. Forest fire impacts on carbon uptake, storage, and emission: the role of burn severity in the Eastern Cascades, Oregon. *Ecosystems* 12, 1246-1267.
- Nocentini, C., Certini, G., Knicker, H., Francioso, O., Rumpel, C., 2010. Nature and reactivity of charcoal produced and added to soil during wildfire are particle-size dependent. *Organic Geochemistry* 41, 682-689.
- North, M., Hurteau, M., Innes, J., 2009. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. *Ecological Applications* 19, 1385-1396.
- North, M., Oakley, B., Chen, J., Erickson, H., Gray, A., Izzo, A., Johnson, D., Ma, S., Marra, J., Meyer, M., Purcell, K., Rambo, T., Rizzo, D., Roath, B., Schowalter, T., 2002. Vegetation and ecological characteristics of mixed-conifer and red fir forests at the Teakettle Experimental Forest. In: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Gen. Tech. Rep. PSW-GTR-186 Albany, CA.
- Pan, Y., Birdsey, R., Hom, J., McCullough, K., 2009. Separating effects of changes in atmospheric composition, climate and land-use on carbon sequestration of U.S. Mid-Atlantic temperate forests. *Forest Ecology and Management* 259, 151-164.
- Pechony, O., Shindell, D.T., 2010. Driving forces of global wildfires over the past millennium and the forthcoming century. *Proceedings of the National Academy of Sciences of the United States of America* 107, 19167-19170.
- Pingree, M.R.A., Homann, P.S., Morrisette, B., Darbyshire, R., 2012. Long and Short-Term Effects of Fire on Soil Charcoal of a Conifer Forest in Southwest Oregon. *Forests* 3, 353-369.
- Preston, C., Schmidt, M., 2006. Black (pyrogenic) carbon: a synthesis of current knowledge and uncertainties with special consideration of boreal regions. *Biogeosciences* 3, 397-420.
- Preston, C.M., 2009. Fire's black legacy. *Nat. Geosci.* 2, 674-675.
- Schmidt, M.W.I., Noack, A.G., 2000. Black carbon in soils and sediments: Analysis, distribution, implications, and current challenges. *Global Biogeochemical Cycles* 14, 777-793.
- Scholl, A.E., Taylor, A.H., 2010. Fire regimes, forest change, and self-reorganization in an old-growth mixed-conifer forest, Yosemite National Park, USA. *Ecological Applications* 20, 362-380.
- Seager, R., Ting, M., Held, I., Kushnir, Y., Lu, J., Vecchi, G., Huang, H.-P., Harnik, N., Leetmaa, A., Lau, N.-C., 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316, 1181-1184.
- Seager, R., Vecchi, G.A., 2010. Greenhouse warming and the 21st century hydroclimate of southwestern North America. *Proceedings of the National Academy of Sciences* 107, 21277-21282.
- Solomon, S., Plattner, G.-K., Knutti, R., Friedlingstein, P., 2009. Irreversible climate change due to carbon dioxide emissions. *Proceedings of the national academy of sciences* 106, 1704-1709.
- Stephens, S.L., Moghaddas, J.J., Hartsough, B.R., Moghaddas, E.E.Y., Clinton, N.E., 2009. Fuel treatment effects on stand-level carbon pools, treatment-related emissions, and fire risk in a Sierra Nevada mixed-conifer forest. *Canadian Journal of Forest Research* 39, 1538-1547.

- Stephens, S.L., Ruth, L.W., 2005. Federal forest-fire policy in the United States. *Ecological applications* 15, 532-542.
- Wayman, R.B., North, M., 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. *For. Ecol. Manage.* 239, 32-44.
- Westerling, A.L., Turner, M.G., Smithwick, E.A., Romme, W.H., Ryan, M.G., 2011. Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. *Proceedings of the National Academy of Sciences* 108, 13165-13170.
- Wiechmann, M.L., Hurteau, M.D., North, M.P., Koch, G.W., Jerabkova, L., In prep. Mitigating climate change and restoring process: An analysis of post-treatment carbon dynamics in a mixed-conifer forest.
- Wiedinmyer, C., Neff, J.C., 2007. Estimates of CO₂. *Carbon Balance and Management* 2, 10.