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The effects of fuels treatments on soil carbon respiration in a Sierra Nevada pine plantation

Leda N. Kobziar[a,*], Scott L. Stephens[b]

[a] School of Forest Resources and Conservation/School of Natural Resources and Environment, Institute of Food and Agricultural Sciences, University of Florida, P.O. Box 110410, Gainesville, FL 32611-0410, USA

[b] Department of Environmental Science, Policy, and Managements, University of California, Berkeley, 137 Mulford Hall #3114, Berkeley, CA 94702-3114, USA

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Abstract

Fire-prone forests in the American west are presently slated for extensive fuels reduction treatments, yet the effect on soil CO_2 efflux rates, or soil respiration, has received little attention. This study utilizes the homogeneity of a Sierra Nevada ponderosa (*Pinus ponderosa* Dougl. ex P. & C. Laws)–Jeffrey pine (*Pinus jeffreyi*, Grev. & Balf.) plantation to investigate changes in soil respiration following mechanical shredding of understory vegetation, or mastication, in 2004; mastication coupled with prescribed burning in 2005; and burning alone also in 2005 as measured over the growing seasons from 2003 to 2005. Soil respiration, soil temperature and soil moisture were measured in two masticated stands which were burned the following year, and in one burned stand; the three of which were compared with two controls stands. Soil respiration response to treatments was detectable even though spatial variability within sites was high (coefficients of variation of 39–66%). Mastication produced short-term reductions in respiration rates, reduced soil moisture by 20%, and mitigated a year-to-year reduction in soil temperature evidenced by controls. Prescribed fire in masticated stands lowered soil respiration from 3.42 to 2.68 μmol m^{-2} s^{-1} while fire in the untreated stand raised rates from 3.41 to 3.83 μmol m^{-2} s^{-1}, although seasonal increases in control sites were greater than those in the untreated stand. Masticated then burned site soil moisture increased by 52% while soil temperature decreased over the span of the growing season. Microclimate variables were not consistently effective in explaining spatial trends. Exponential models using soil temperature and/or moisture to predict temporal trends in respiration were only significant in treated stands, suggesting that treatment implementation increased sensitivity to environmental factors. These results imply that fuels reduction practices in water-stressed forests may have important consequences for ecosystem carbon dynamics.

Keywords: Fire; Soil temperature; Soil moisture; Gas exchange; CO_2

1. Introduction

Forest soils contain more than 70% of the terrestrial world's soil carbon pool (Post et al., 1982), and thereby play a major role in global carbon cycles and their influence on climate. Yet little is known about how management practices, especially prescribed fire, affect forest soil carbon emissions. Soil surface CO_2 efflux rates, which include respiration from both autotrophic (root) and heterotrophic (soil macro- and microorganisms) sources, have been shown to account for up to 67% of total mean ecosystem respiration in a young ponderosa pine plantation in California (Xu et al., 2001). As the importance of sequestering carbon

* Corresponding author. Tel.: +1 352 846 0901.
E-mail address: lkobziar@ufl.edu (L.N. Kobziar).

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to potentially offset global climate warming increases, forest management goals may soon undergo a significant shift. Traditional, production-based paradigms may be replaced by a new ethic of ecosystem carbon budgeting. A parallel shift emphasizing restoration of fire processes in fire-prone ecosystems is already underway (Stephens and Moghaddas, 2005a; Stephens and Ruth, 2005). As wildfire suppression and mitigation costs throughout the western US continue to rise, land managers are challenged to act expediently to reduce hazardous fuels. Over 2 million hectares of forested lands are slated for extensive fuels reduction treatments in the Sierra Nevada alone over the next decade, including the implementation of prescribed fire, thinning, and shredding of understory vegetation, or mastication (USDA, 2002). There is a compelling need for understanding the relationship between fuels reduction prescriptions and the increasingly important issue of carbon flux in forested ecosystems.

Assuring the health and fire resilience in dense plantation forests is a relatively new challenge faced by natural resource managers. Although tree density in most northern California plantations is currently at an ‘‘acceptable’’ level, mortality risk from inter-tree competition, disease, insect infestation and fire will increase significantly over the next 10–20 years (Landram, 1996). High fire hazards are already present in and around many of these plantations (Stephens and Moghaddas, 2005b), linked to high success rates in replanting and dense post-fire understory growth, low summer fuel moisture, steep, mountainous terrain, frequent ignitions from lightning, and increased public recreation in National Forests. Such plantations, which cover 155,000 ha in the Sierra Nevada, have also been heralded for their capacity to sequester carbon. Yet the high fire hazards associated with their structural features may ultimately lead to a greater emission of CO_2.

How fuels reduction treatments affect soil respiration is closely linked to the role soil physical and chemical factors play in soil respiration. Fire is known to impact the organic matter content of the soil, along with nitrogen mineralization rates, total nitrogen, soil temperature and pH, and above- and below-ground species composition (DeBano, 1991). Low intensity burns, such as prescribed understory burns, are believed to increase pH and available nutrients in the soil (P, K, Ca, and Mg) through the deposition of ash (Hare, 1961), although there is much debate concerning the effect such fires have on soil organic and inorganic nitrogen, largely depending on the temporal scope of the study (DeLuca and Zouhar, 2000; Neary et al., 1999). A long-term study of forest management practices and their effects on soil C also showed that biomass harvesting in the southeastern US only caused short-term changes in soil C, with no lasting effect (Johnson and Curtis, 2001). The transformation of organic C into CO_2 during biomass burning, and into black carbon, has both short-term and longer-term implications for the lifetime of carbon stocks.

Some researchers suggest that increased growth rates following low-severity fires compensate for the carbon emissions lost during burns, resulting in a negligible net effect on ecosystem carbon and on the atmosphere (Crutzen and Goldammer, 1993). Thinning results in the physical removal of autotrophic respiration sources, and decreases in soil respiration following thinning have been attributed to the corresponding loss of root density (Tang et al., 2005).

Using prescribed fire as a fuels reduction technique may have additional consequences to the soil in the event of a wildfire. In a post-wildfire ponderosa pine (*Pinus ponderosa* Dougl. ex P. & C. Laws)–Douglas–fir (*Pseudotsuga menziesii* (Mirb.) Franco) forest, losses of labile C and N were lower and soil microbial communities were more resistant where prescribed burning treatments had been enacted previous to the wildfire (Choromanska and DeLuca, 2001). This may be considered an additional benefit of fuels reduction practices on soil carbon sequestration. Kaye and Hart (1998) examined whether prescribed fire and thinning aimed at restoring a ponderosa pine–bunchgrass ecosystem affected soil CO_2 efflux and some key soil characteristics. Using both tree removal and prescribed burning resulted in the highest rate of emissions of soil CO_2, although the increased rates were best explained by increases in soil temperature, and did not correlate with changes in soil N, P, OM, C/N ratio, or fine-root biomass (Kaye and Hart, 1998). Thinning resulted in elevated soil respiration rates, soil temperature, and soil moisture, while prescribed burning had no effect in an old-growth mixed-conifer forest in California (Concilio et al., 2005). Other studies of thinning or tree harvest have shown increases, decreases, and no change in soil respiration rates (Edwards and Ross-Todd, 1983; Weber, 1990; Toland and Zak, 1994; Tang et al., 2005). The discrepancy between treatment effects on soil respiration in these studies highlights the importance of documenting both pre- and post-treatment soil respiration trends, taking into account the importance of spatial and temporal variation in controls. It is also important to recognize that forest disturbances, whether anthropogenic or natural, can significantly impact forest microclimate. Thinning and/or prescribed burning effects on incoming solar radiation, forest structure, and ground cover can further influence soil CO_2 efflux by affecting soil moisture and soil temperature (Ma et al., 2004; Concilio et al., 2005; Tang et al., 2005).

This study addresses how soil respiration, soil moisture, and soil temperature are impacted by fuels reduction treatments in a pine plantation in the central Sierra Nevada of California. The fuels manipulations include (1) mastication of understory vegetation and thinning of small trees with slash distributed and left on-site, (2) mastication as above followed by prescribed understory burning, (3) prescribed understory burning alone, and (4) control. Also, soil temperature and soil moisture are analyzed for their roles in governing the temporal and spatial patterns in soil CO_2 efflux both between and within treatments. This investigation expands the limited understanding of some of the broader ecological ramifications of fuels management practices.

2. Methods

2.1. Site description

The Stanislaus National Forest's Granite Project Area, where ponderosa and Jeffrey pine (*Pinus jeffreyi*, Grev. & Balf.) plantation forests offer a high degree of vegetative, topographic, and soil homogeneity, is an excellent site for evaluating such highly variable metrics as soil carbon respiration rates. Located in the Groveland Ranger District in the central Sierra Nevada (Fig. 1), nearly 2000 ha of ponderosa and Jeffrey pine plantation forest were planted to replace the second-growth mixed conifer stands that had been destroyed during the 1973 Granite Fire. This district is characterized by a high fire frequency, with approximately 70% of its forests and woodlands burned in large fires (>800 ha) since the 1970s. Fuels prescriptions in the Granite plantations are aimed at reducing potential fire behavior and competition between trees and understory vegetation, and increasing forest health and resistance to disturbances. Over 6500 ha of plantation and second-growth forest will be treated to reduce hazardous fuels in the overall scope of the Granite Project. The project area has been deemed a Demonstration Site by the US Department of the Interior/Department of Agriculture Joint Fire Science Program, because pine plantations are common throughout the nation as the most effective means of reforestation after fire or harvest.

The plantation forest units sampled in this study are all located within 10 km west of Cherry Lake, and include control units 184 and 150, Burn unit 132, and Mastication + Burn units 185 and 106. The units were chosen at random from structurally similar plantation stands stratified by the particular fuels reduction treatment assigned to them. Limitations on prescribed burning opportunities prevented the replication of the Burn treatment. Slopes in each of the five units are gentle, ranging from 3 to 15%, and all units face south or south to southeast. Elevations range from 1500 to 1800 m. While the area sampled within each unit was the same (400 m^2), total unit sizes ranged from 14 to 82 ha.

The Granite plantations are influenced by a Mediterranean climate where summer drought conditions are common. Total annual precipitation averages ~120 cm, largely comprised of snowfall (~80%), which can at times linger through June. Average summer and winter temperatures are 21 and 4 °C (based on 50 years of data; WRCC, 2006). The overall temporal scale of this study extended from June 2003 to November 2005. Excepting the unusual amount of precipitation in late October 2004 (33 cm versus a 50-year average of 6.5 cm), total precipitation before and during our sampling period was similar between the 3 years (Fig. 2). Soils are developed from either metasedimentary or granitic rock, and belong to the Fiddletown series (USDA, 1981). These Incepticols in the Pachic Xerumbrepts class are moderately deep to deep (50–100 cm) with a gravelly sandy loam texture in the upper horizons. The surface soil is approximately 50 cm deep and is slightly acid (pH 5.6–6.6; USDA, 1981). The subsoil is very gravelly sand loam, about 25 cm thick on average. Soils are generally dry from July to October (USDA, 1981).

Seedlings planted in the study units were germinated from seed sources within the local western slope Sierra Nevada mixed-conifer forest type, and included Jeffrey pine, sugar pine (*Pinus lambertiana* Dougl.), ponderosa pine, Douglas–fir, white fir (*Abies concolor* Gord. & Glend), incense-cedar (*Calocedrus decurrens* [Torr.] Floren.), and infrequent giant sequoia (*Sequoiadendron giganteum* (Lindl.) Buchh). Ponderosa and Jeffrey pine comprise more than 90% of the pre-mastication and over 95% of the post-mastication tree composition in all units. Unit 106 has slightly more Jeffrey than ponderosa pine, while all other units were stocked with more ponderosa pine. Infrequent California black oak (*Quercus kelloggii* Newb.), and dogwood (*Cornus nutallii* Audubon ex. Torr. and Gray) entered the plantations via natural seed dispersal mechanisms from

Fig. 1. Map of location of the Stanislaus National Forest in the central Sierra Nevada of California, USA.

proximate mixed-conifer forests, or via sprouting from established oak stumps. The understory is largely composed of whitethorn (*Ceanothus cordulatus* Kellogg), and greenleaf manzanita (*Arctostaphylos patula* E. Greene), with less abundant species including gayophytum (*Gayophytum diffusum* Torrey & A. Gray), Sierra current (*Ribes nevadense* Kellog.), Sierra gooseberry (*Ribes roezlii* Regel.) and bracken fern (*Pteridium aquilinum* (L.) Kuhn).

2.2. Mastication and burning treatments

Mastication in units 185 and 106 was completed by mid-June of 2004. All small trees (≤23 cm in diameter) were masticated and all resulting materials were distributed and left on site. Post-mastication treatment spacing of conifers was approximately 5 m, reducing density from 363 to 272 trees/ha in unit 185 and to 222 trees/ha in unit 106, although average basal areas were not significantly different between the two units (Table 1). Understory herbaceous and shrub vegetation was also masticated, along with diseased and suppressed trees. Prescribed burning was conducted in the two masticated units, and in unit 132, which had not been treated. All burns took place on 28 June 2005 between 10:00 a.m. and 11:00 p.m. using a combination of backing and strip-head firing techniques (Martin and Dell, 1978). Nearly 2 cm of rain fell on 17 June, which enabled prescribed burning during what would typically be within the summer drought and fire season. Desired environmental conditions for the burns were all met, including relative humidity between 25 and 65%, wind

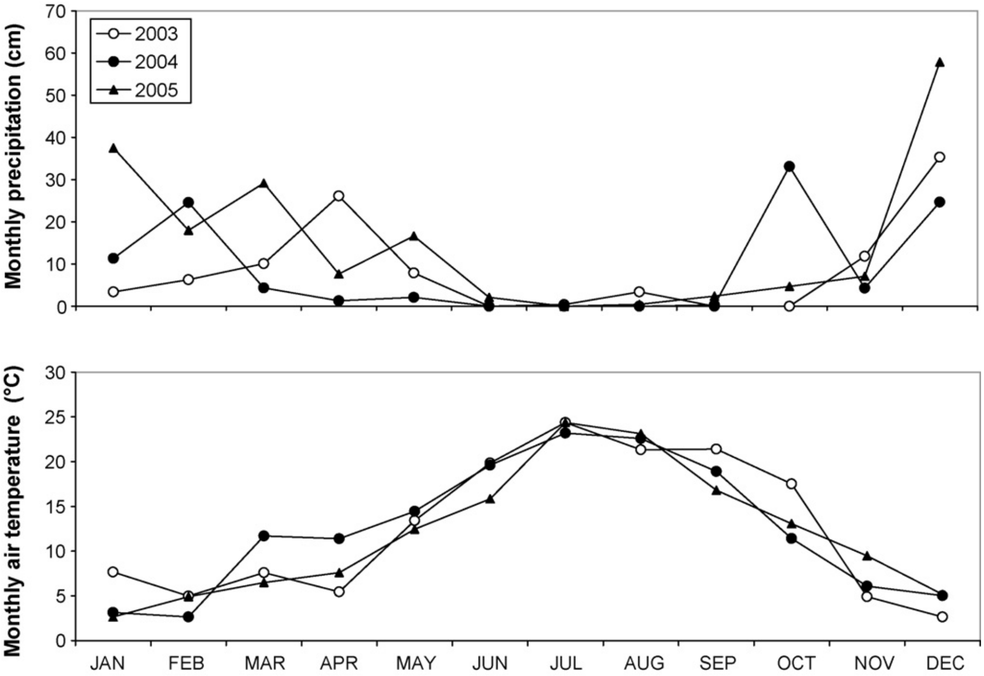

Fig. 2. Average monthly air temperature (°C) and total monthly precipitation (cm) over the 3-year study period in the Stanislaus National Forest plantations, collected at Mount Elizabeth Remote Access Weather Station (RAWS), CA, between years 1955 and 2005.

speed below 8 km/h, temperature between 0 and 24 °C, and 10 h fuel moisture between 7 and 15% throughout the day. Pre- and post-treatment forest unit characteristics are described in Table 1.

2.3. Soil respiration, temperature, and moisture measurements

Measurements were conducted between June and November from 2003 to 2005. Due to logistical constraints and access limitations during the winter, early spring, and late fall seasons, all five units were consistently measured each year between July and late October. Typically once each month, soil carbon respiration rates (SRR) in each unit was measured from early morning to evening in order to capture diurnal fluctuations. In 2004, measurements were taken during the last and first weeks of September and October, respectively.

Because soil temperature and moisture often vary in relation to aspect and topography (Xu et al., 1997), all areas sampled within units had similar topographic

Table 1
Forest characteristics during each treatment stage (year) in five Stanislaus National Forest plantation units, CA

Unit	Treatment	Year	Trees (ha)	BA (m^2/ha)	Average DBH (cm)	Average height (m)	Average Ht to LC (m)	Canopy cover (%)	Dominant tree
185	None	2003	363	25.67	27.75	11.96	2.00	57.14	PIPO
	Mastication	2004	272	28.95	35.54	14.84	3.25	71.43	PIPO
	Burn	2005	272	29.78	37.26	15.41	5.28	66.00	PIPO
106	None	2003	363	22.37	26.74	12.03	2.58	35.71	PIJE/PIPO
	Mastication	2004	222	20.61	33.97	14.82	3.73	39.30	PIJE/PIPO
	Burn	2005	222	22.29	35.34	14.80	7.45	39.30	PIJE/PIPO
132	None	2003	368	28.54	30.84	14.47	3.86	53.57	PIPO
	Burn	2005	368	28.54	35.02	17.31	9.30	54.10	PIPO
184	Control	2003	236	12.25	24.68	9.93	1.51	14.29	PIPO
150	Control	2003	550	26.63	23.17	11.34	3.73	75.00	PIPO

Note. DBH is diameter at 1.4 m height, Ht. to LC is height to live crown, BA is basal area, PIPO is ponderosa pine, PIJE is Jeffrey pine.

attributes. In each of the five units, nine soil CO_2 efflux sampling points were established and their locations permanently marked to ensure that the location was identical following the treatments ($N = 45$ total). The nine sampling points were arranged on a 3×3 matrix spaced 10 m apart with a randomized starting point. At each point, a 4.4 cm tall soil collar with a diameter of 11 cm was inserted ~1.0 cm into the soil. Site disturbance was limited, as no alterations to the O horizons or soil within the soil collar were made. The Li-Cor 6400-09 soil chamber coupled with a Li-Cor 6400 photosynthesis system (Li-Cor, Lincoln, NE) was used to measure CO_2 emissions. Ambient CO_2 levels were set for the Licor every hour. Soil temperature was measured at 10 cm depths within 10 cm of each sampling point using a temperature probe connected to the Li-Cor photosynthesis unit. These depths have been shown to provide soil temperatures that are closely related to variation in soil carbon respiration rates (Xu and Qi, 2001). In each stand between the hours of approximately 07:00 and 17:00, nine soil plots were measured hourly for SRR and soil temperature (T_s). All five units were measured within ~5 days of each other, with no precipitation or significant changes in air temperature between days. A single daily measure of soil moisture was conducted. Percent soil moisture (M_s) on a dry weight basis was assessed by extracting soil cores adjacent to the soil collars, then oven-drying the samples for 48 h at 105 °C.

2.4. Data analysis

All analyses were based on average daily values of SRR, T_s, and M_s for each sampling point ($N = 72$ measurements per day each month in each unit). The effects of treatments on mean soil respiration rates were evaluated for both stages of forest fuels manipulation. Stage 1 included the mastication of understory vegetation and trees less than 23 cm in diameter in two units (106 and 185). During stage 2, these two units, along with 132, were burned (Table 1). The hypothesis that treatments affected mean SRR, T_s, and M_s was evaluated between stage 0 (pre-treatment) and stage 1, stages 1 and 2, and between stages 0 and 2. Because treatments were implemented each year, the three stages correspond to the 3 years of sampling (2003–2005). During each stage's growing season, sampling was conducted consistently between July and October. This allowed for the detection of any pulses of response to the treatments, which might otherwise be undetectable in the yearly mean values. The JMP IN 5.1statistical software (SAS Institute, 2003) was used in all analyses.

Because the data was unbalanced (two controls and Mastication + Burn units, one Burn unit), a mean difference approach followed by an F-test ($P < 0.05$) was used to test for treatment effects on SRR, T_s, and M_s. The mean difference approach coupled with a between treatments one-way ANOVA corresponds to a simple repeated measures analysis testing whether changes in values between the treatment stages differed between the treatments and the controls. Yearly average SRR, T_s, and M_s values for each stage and unit were subtracted from averages from the other stages. For example, to address the question of whether SRR was affected by burning, the differences between SRR from stages 1 to 2 in burned units were compared with the same differences in control units. An effect was concluded to have occurred if the stage 1–2 mean differences were significantly greater or less than in the treatments versus the control units. This approach has been used successfully in other studies of the ecological effects of fuels reduction treatments (Beche et al., 2005; Apigian et al., 2006).

Spatial variability in SRR, T_s, and M_s was examined by comparing means and standard deviations derived from nine sampling points for each stand (totaling 45 points for each month) before and after each treatment. Values were calculated by sampling month, and were also assessed over the growing season of each treatment stage. Coefficients of variation (CV) were statistically compared (t-test, $P < 0.05$). Treatment effects on spatial variability (as represented by CV values) were tested against the controls using the means difference approach as described above.

As T_s and M_s have been shown to influence SRR in other studies, the predictive abilities of both linear and nonlinear (natural Log of SRR) models on SRR in each treatment type and for each treatment stage were explored. Exponential [1], power [2], and linear [3] functions were fitted to SRR data for stage 0, 1, and 2 mean monthly values of T_s and M_s as follows:

$$\mathrm{SRR} = \beta 0\, \mathrm{e}^{\beta 1 T_s} \tag{1}$$

$$\mathrm{SRR} = \beta_0 T_s^{\beta 1} \quad \text{or} \quad \beta_0 M_s^{\beta 1} \tag{2}$$

$$\mathrm{SRR} = \beta_0 T_s + \beta_1 \quad \text{or} \quad \beta_0 M_s + \beta_1 \tag{3}$$

where β_0 and β_1 are coefficients estimated through regression analysis. Interaction terms including M_s and T_s were also tested. From Eq. (1), the β_1 coefficient is used in the following equation to produce the Q_{10} value, or the multiplier for SRR given an increase of 10° in soil temperature (Lundegardh, 1927):

$$Q_{10} = \mathrm{e}^{10\beta_1} \tag{4}$$

After normality in the population distributions for each value was established, a multiple regression procedure was employed to evaluate predictive models that combined the effects of T_s and M_s on both spatial and temporal variation in SRR in each year and treatment type. In each model type, variables were tested for significant relationships ($P < 0.05$; $R^2 > 0.45$) with SRR as well as log-transformed SRR using Pearson's product–moment correlation analysis in each treatment type. All models were based on mean monthly values for each treatment type and year.

3. Results

Mean pre-treatment soil carbon was 4.07, 4.75, and 4.18% in control, Mastication + Burn, and Burn units, respectively (combustion gas analyzer method for total nitrogen and total carbon; Method 972.43; AOAC, 1997). Mean percentage soil nitrogen composition was 0.19, 0.22, and 0.17 in control, Mastication + Burn, and Burn units, respectively. Both soil carbon and soil nitrogen did not differ significantly between the treatment types (one-way ANOVA, $P > 0.05$).

Soil CO_2 efflux ranged from 2.37 to 4.55 (μmol m^{-2} s^{-1}), with the highest mean annual growing season rates occurring in 2003 for each treatment type (Table 2). Soil temperatures were generally higher on average in 2003 and 2005 (Table 2). Soil moisture content corresponded with precipitation events (Figs. 2 and 3) and decreased with increasing soil temperatures in 2003 ($R^2 = 0.24$), 2004 ($R^2 = 0.25$), and 2005 ($R^2 = 0.44$), ($P < 0.01$). Soil moistures did not exceed 25% of the weight of soil samples on average. Air temperature and T_s generally followed the same seasonal pattern of increasing in the mid-summer months and decreasing after October (Figs. 2 and 3). Although October 2004 was marked by an unusually high precipitation event (Fig. 2), all sampling was completed before the rain fell. Soil moisture metrics for the September/October 2004 sampling month, therefore, were not reflective of the precipitation (Fig. 3).

3.1. Analysis of treatment effects

Although some results show measurements taken in November 2004 and 2005, all comparisons between variables in different years were based on data collected from the same months (July–October of each comparison with stage 1 values). When all treated units were pooled together into a single "disturbed" group, absolute mean differences between 2003 and 2005 were greater for all variables than between 2004 and 2005 (Fig. 4). Absolute mean differences in control units' soil temperature were greater for both 0–2 and 1–2 stage comparisons than in disturbed units. From stages 1–2, mean differences in SRR, T_s, and M_s were lower in the disturbed than in control units (Fig. 4b). The opposite was true for SRR and M_s when 2003–2005 values were compared (Fig. 4a).

3.1.1. Stages 0–1: mastication effects on SRR, T_s, and M_s

Analysis of pre–post-treatment annual growing season means for 2003–2004 indicated no significant overall mastication effect on SRR in Mastication + Burn units (Table 3). When the means difference analysis was applied to monthly means, significant impacts were detected between (stages 0 and 1) July and August (Table 4). This was not the case for the later-season pairs of months, or between 2003 annual means and each month in 2004, suggesting that mastication

Table 2
Mean (±S.E.) soil respiration rate (SRR), soil temperature (T_s), and soil moisture (M_s) for each year and three different fuels reduction treatments in the Stanislaus National Forest pine plantations, CA

Year	Treatment type	SRR (μmol m^{-2} s^{-1})	T_s (°C)	M_s (%)
2003 (pre-treatment)	Control	3.46 (0.35) a	18.94 (0.26) a	10.91 (0.52) a
	Mastication + Burn	4.54 (0.54) a	16.69 (0.21) b	12.00 (0.37) a
	Burn	4.55 (0.44) a	18.83 (0.31) a	8.30 (0.56) b
2004 (post-mastication)	Control	2.37 (0.22) a	13.40 (1.99) a	10.89 (1.72) a
	Mastication + Burn	3.42 (0.45) a	12.79 (0.25) a	9.78 (0.42) ab
	Burn	3.41 (0.56) a	13.77 (0.36) a	8.24 (0.29) b
2005 (post-burning)	Control	3.25 (0.32) a	14.09 (0.22) a	11.47 (0.81) a
	Mastication + Burn	2.68 (0.41) a	12.26 (0.22) b	15.10 (0.79) b
	Burn	3.83 (0.58) a	14.94 (0.30) a	8.46 (0.36) c

Note. Significant differences, denoted by different letters for each column within each year, were only evaluated within years and do not represent treatment effects (Tukey HSD test; Tukey 1953).

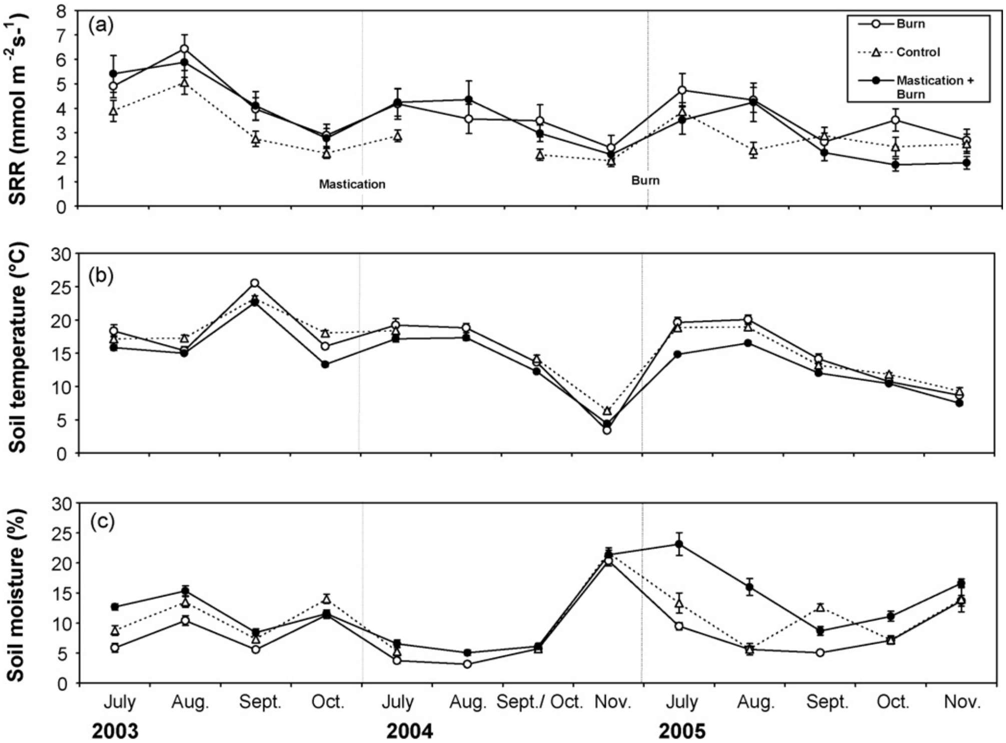

Fig. 3. Seasonal variation in soil carbon respiration (a), soil temperature (b), and soil moisture (c) following mastication in June 2004 and prescribed burning in June 2005 in Stanislaus National Forest pine plantation units. Each dot is the monthly grand mean for the treatment type ($N = 18$ in Mastication + Burn and Control, $N = 9$ in Burn).

resulted in a short-term reduction of SRR when compared with control units (Table 4). Mastication resulted in a significant reduction of annual growing season mean T_s and M_s when compared with controls, but the mean difference in T_s between 2003 and 2004 was less in masticated units than in controls (3.56 °C versus 5.54 °C; Table 3). These differences in T_s were evident on a paired month-by-month basis for September–October, but not in July or August, indicating a lag in response. Conversely, soil moisture was impacted by mastication immediately following the treatment (in July and August), but not in the latter months (Table 4). The Burn unit was not treated during 2004.

3.1.2. Stages 1–2: burning effects on SRR, T_s, and M_s

When compared with inter-annual changes in growing season SRR in control units, prescribed burning resulted in a reduction (Table 5) of annual mean SRR in Mastication + Burn units from 3.42 to 2.68 μmol m^{-2} s^{-1} between stages 1 and 2 (Table 2). Control unit SRR increased between the 2 years from 2.37 to 3.25 μmol m^{-2} s^{-1}, and the increase in annual mean Burn unit SRR after prescribed burning was significantly lower than that in the controls (0.18 μmol m^{-2} s^{-1} versus 0.97 μmol m^{-2} s^{-1}; Table 5). The fire had no significant impact on T_s or M_s in the Burn unit, in contrast to the Mastication + Burn units where mean annual growing season soil

Table 3
Mean differences (95% confidence interval) between stages 0 and 1 (2003–2004) response variables and results of one-way ANOVA for control and Mastication + Burn units

Response	Control	Mastication + Burn	Mastication effect[a]	F	P
SRR (μmol m^{-2} s^{-1})	1.10 (0.34)	1.12 (0.45)	None	0.01	0.98
T_s (°C)	5.54 (0.61)	3.56 (0.25)	Reduction	6.29	0.02
M_s (%)	0.01 (0.78)	2.23 (1.09)	Reduction	12.2	<0.01

[a] Mastication effect was concluded if masticated site mean fell outside the confidence interval of the control units, and ANOVA was significant ($P < 0.05$). The Burn unit was not treated during stage 1.

Table 4
Month-by-month analysis of treatment effects depicting findings of significance in mean differences between treatments and controls, between paired months from each year (i.e. July 2003 vs. July 2004)

Year–year	Month	Mastication + Burn SRR/T_s/M_s	Burn SRR/T_s/M_s[a]
2003–2004	July	−/o/−	n/a
	August	−/o/−	n/a
	September	o/−/o	n/a
	October	o/−/o	n/a
2004–2005	July	−/−/+	o/o/+
	August	o/o/+	+/+/o
	September/October	−/−/+	−/−/+
	November	−/o/−	o/−/o
2003–2005	July	−/−/o	o/o/+
	August	o/+/o	o/+/−
	September	−/o/+	−/o/−
	October	−/−/o	o/o/−

Variables include soil respiration (SRR), temperature (T_s), and moisture (M_s). Significance of compared mean differences was evaluated using a one-way ANOVA ($P < 0.05$). Increases, decreases, and no change between the treatment stages are designated with +, −, and o symbols, respectively.
[a] Burn unit was not treated in 2004.

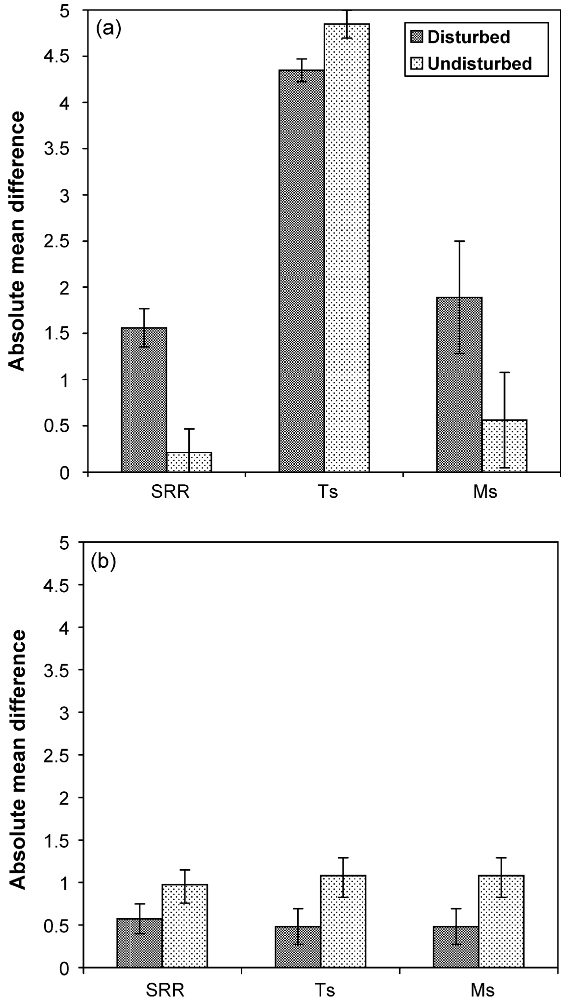

Fig. 4. Absolute mean differences between pooled disturbed and undisturbed units in soil respiration (SRR), soil temperature (T_s), and soil moisture (M_s). Values are shown for 2003–2005 (a) and 2004–2005 (b). Bars represent standard errors of the means.

temperatures decreased by 4% and soil moisture increased by 52% of the mean values (Tables 2 and 5). Soil moisture and temperature both increased between stages 1 and 2 in the control units (Table 5).

On a 2004–2005 paired monthly basis, Mastication + Burn SRR decreased from stages 1 to 2 in all months except August (Table 4). Detectable changes in Mastication + Burn T_s were intermittent through the 2005 sampling season, while increases in M_s were significant for all paired months except November (Table 4). Although overall changes between growing season annual mean T_s and M_s from stages 1 to 2 in the Burn unit were not significantly different from the controls (Table 5), increases were evident in M_s between the 2 years during the months of July and September/October (Table 4). The effects of burning on Burn unit soil temperatures were not detectable in the first post-burn month of sampling. In August, soil temperatures in this unit increased, then decreased for the following months when compared with the controls (Table 4).

Table 5
Mean differences (95% confidence interval) between stages 1 and 2 (2004–2005) response variables and results of one-way ANOVA for control and treatment units

Response	Control	Mastication + Burn	Burn effect[a]	F	P	Burn	Burn effect[a]	F	P
SRR (μmol m^{-2} s^{-1})	−0.97 (0.50)	0.74 (0.33)	Reduction	42.5	<0.01	−0.18 (0.32)	Increase	5.13	0.03
T_s (°C)	−1.08 (0.48)	0.53 (0.32)	Reduction	64.21	<0.01	−0.88 (0.38)	None	0.34	0.57
M_s (%)	−0.58 (1.09)	−5.32 (1.64)	Increase	25.82	<0.01	−0.47 (0.02)	None	1.75	0.20

[a] Burn effect was concluded if burned site mean difference fell outside the confidence interval of the control units, and ANOVA was significant ($P < 0.05$).

Table 6
Mean differences (95% confidence interval) between stages 0 and 2 (2003–2005) response variables and results of one-way ANOVA for control and treatment units

Response	Control	Mastication + Burn	Combined treatment effect[a]	F	P	Burn	Burn effect[a]	F	P
SRR (μmol m^{-2} s^{-1})	0.21 (0.53)	1.86 (0.43)	Reduction	19.64	<0.01	0.97 (0.43)	Reduction	3.96	0.05
T_s (°C)	4.84 (0.33)	4.43 (0.31)	Reduction	3.83	0.05	4.06 (0.63)	Reduction	6.16	0.02
M_s (%)	−0.56 (1.09)	−3.09 (1.56)	Increase	7.93	<0.01	−0.52 (1.05)	None	1.83	0.19

[a] Treatment/burn effect was concluded if burned site mean difference fell outside the confidence interval of the control units, and ANOVA was significant ($P < 0.05$).

3.1.3. Stages 0–2: mastication + burning effects on SRR, T_s, and M_s

Overall treatment effects resulting from mastication and burning in Mastication + Burn units were similar between stages 0 and 2 and between stages 1 and 2 (Tables 5 and 6). Soil carbon respiration rates were reduced more in Mastication + Burn units than in controls, while reductions in T_s were less than in controls (Table 6). Increases in M_s between stages 1 and 2 in Mastication + Burn units (5.32) were greater than between stages 0 and 2 (3.09), and both were significantly greater than in controls. In the same units, annual mean T_s reductions between pre-treatment and burned stages were greater than reductions between the masticated and burned stage (Tables 5 and 6).

From pre-treatment to post-burning, both control and Burn unit SRR decreased, with a significantly greater reduction in Burn units (Table 6). This contrasts with increases in control and Burn unit SRR between stages 1 and 2, as well as decreases in SRR in Mastication + - Burn units (Table 5). Although the fire's influence on T_s in the Burn unit was not significant between stages 1 and 2, when compared with 2003 mean values, post-fire T_s decreased less than in controls (Table 6). In both stages 1–2 and stages 0–2, fire had no significant impact on soil moisture in the Burn units. When paired months were compared between 2003 and 2005, burning effects on SRR were indicated in September only, and in all months except August in Burn and Mastication + Burn units, respectively (Table 4). Temperature differences between paired months in Mastication + Burn units were variable, while increases in T_s were evident between paired August months in both Mastication + - Burn and Burn units (Table 4).

3.2. Relationship between soil temperature, water, and respiration

In each treatment type, spatial variations in SRR were poorly explained by soil temperature using any of the three models (Eqs. (1)–(3); $\alpha = 0.05$). Even when all measurements were divided into low and high soil moisture categories, the relationship between spatial variance in SRR and T_s was not significant. Over each of the growing season sampling years, soil moisture rarely exceeded 20% (Fig. 3), and differences in SRR and T_s relationships between soil moisture categories of greater or less than 10% were not detectable in any of the units. Some influence of T_s on SRR spatial variance across all units was indicated at the end of the growing seasons of 2004 and 2005, but not in 2003: relationships were weak for all years (Fig. 5). The

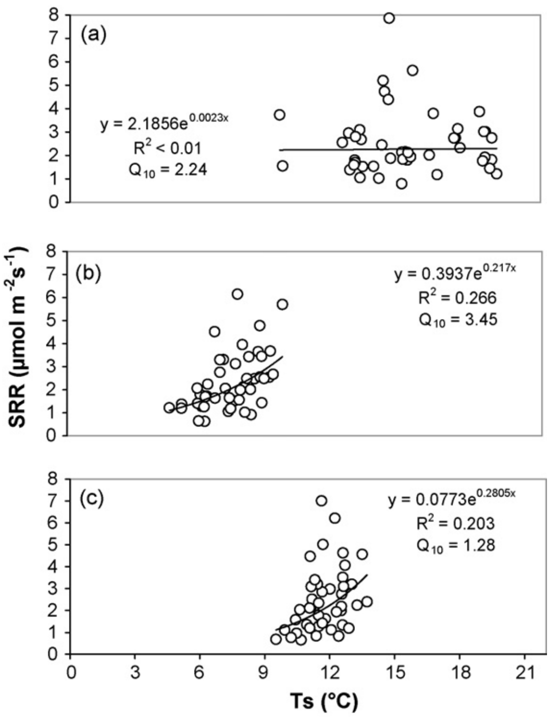

Fig. 5. Exponential regression of soil carbon respiration rates (SRR) with soil temperature (T_s) in all treatment types for the months of October in (a) 2003, and October through November in (b) 2004; and (c) 2005. Each circle represents one soil plot averaged over October and November in the Stainslaus National Forest pine plantation stands ($N = 45$).

earlier onset of fall season precipitation in 2004 and 2005 contrasts with the lack of precipitation until late November 2003, when higher air and soil temperatures were also recorded (Fig. 2).

Spatial trends in soil moisture were significantly (F-test, $P < 0.05$) but weakly related to SRR in control units in 2003 ($R^2 = 0.33$ for all equations). The linear relationship between SRR and M_s in 2004 control units was also weakly positive ($R^2 = 0.28$; $P < 0.05$), while Eq. (2) produced a slightly better fit ($R^2 = 0.33$). No model produced a significant fit for the other treatment types or years. Multiple regression analyses of each SRR value and its corresponding T_s and M_s metrics did not produce significant models for any year or over the entire study period. However, in 2003, spatial variation in control unit SRR was somewhat explained by the combined effects of T_s and M_s in the multiple regression equation ($R^2 = 0.34$; $P = 0.04$). Log-transformations of SRR did not improve model fits.

The exponential Eq. (1) resulted in the best fit for temporal variability in SRR based on fluctuations in T_s over time. Regression analysis of modeled versus actual SRR was significant for Burn only and Mastication + Burn units in 2004, and for post-fire Burn units in 2005 (Table 7). Temporal trends in T_s explained over 90% of SRR variability in 2004 Burn and Mastication + Burn units, and 63% in the Burn unit in 2005. Although the exponential model provided the best fit for control units when compared with Eqs. (2) and (3), regression analysis showed that T_s did not explain a significant amount of temporal variation in SRR at any stage in controls (Table 7).

Q_{10} values indicated that temperature sensitivity was lower in 2004 than in 2003, and even lower following prescribed burning in the treated stands (Table 7). When SRR was modeled across all years, soil temperatures in Mastication + Burn units explained 57% of the temporal variability in SRR (Table 7). Less than half of the variability in SRR in all stands over the entire sampling period was explained by soil temperature.

Regardless of the year or treatment, variation in M_s over time was ineffective in explaining temporal variation in SRR ($P > 0.10$ for Eqs. (1)–(3)). Multiple regression analysis incorporating the effects of both T_s and M_s produced a single significant model for temporal SRR trends; in Mastication + Burn units over all years ($R^2 = 0.54$; $P = 0.02$). Again, Log-transformations did not improve model performance.

3.3. Spatial variability between treatment units and treatments

Spatial variability within and between the treatment types was evaluated using the coefficient of variation (CV, %), which was calculated both for yearly mean values and monthly values of SRR, T_s, and M_s. In control, Burn, and Mastication + Burn treatments, yearly SRR CVs ranged from 39 to 42%, 31 to 49%, and 51 to 66% (Fig. 6A). Pre-treatment spatial variability in control SRR was significantly lower than that in the Mastication + Burn units (Fig. 6A). The same comparison did not yield detectable differences within 2004, even though CVs increased in Mastication + Burn sites (paired t-test, $\alpha = 0.05$). Unlike post-fire variability

Table 7
Effectiveness of temporal variations in soil temperature (T_s) in explaining the variance of soil respiration (SRR) over 3 years in each treatment type in the Stanislaus National Forest pine plantations

Year	Treatment type	Q_{10}	Eq. (1)	$R^{2\,a}$	P^a
2003	Control	5.38	$SRR = 9.3554\ e^{-0.0553 T_s}$	0.18	0.51
	Burn	4.92	$SRR = 5.6453\ e^{-0.0137 T_s}$	0.04	0.71
	Mastication + Burn	4.08	$SRR = 3.6978\ e^{0.0099 T_s}$	0.01	0.98
2004	Control	2.02	$SRR = 1.4433\ e^{0.0337 T_s}$	0.83	0.26
	Burn	2.97	$SRR = 2.1853\ e^{0.0308 T_s}$	0.92	0.02
	Mastication + Burn	2.8	$SRR = 1.6048\ e^{0.0559 T_s}$	0.98	<0.01
2005	Control	2.54	$SRR = 2.1270\ e^{0.0178 T_s}$	0.14	0.49
	Burn	2.87	$SRR = 1.8931\ e^{0.0416 T_s}$	0.63	0.01
	Mastication + Burn	1.95	$SRR = 0.6587\ e^{0.1088 T_s}$	0.87	0.08
All years	Control	2.50	$SRR = 1.8479\ e^{0.026 T_s}$	0.18	0.27
	Burn	3.11	$SRR = 2.3115\ e^{0.0298 T_s}$	0.36	0.09
	Mastication + Burn	2.56	$SRR = 1.2724\ e^{0.0675 T_s}$	0.57	0.01
All years	All treatments	2.83	$SRR = 1.9033\ e^{0.0348 T_s}$	0.44	0.04

[a] Significance is reported for regression of modeled SRR against actual SRR, while goodness-of-fit is reported for model fit to the data.

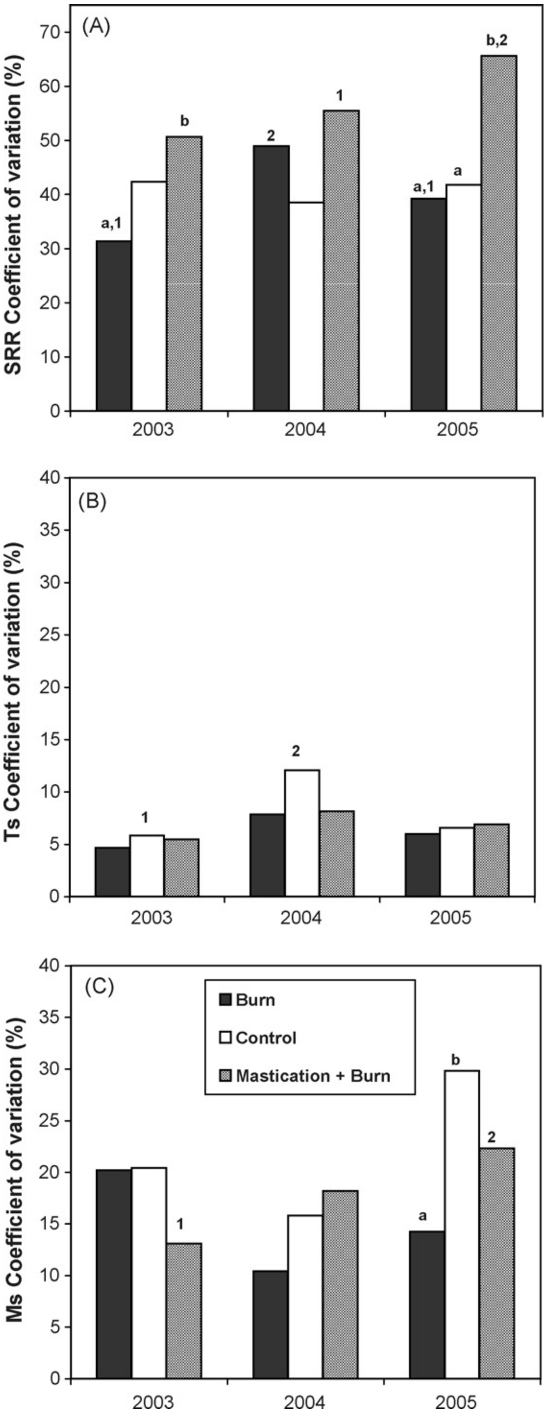

Fig. 6. Coefficient of variation (%) for soil respiration rate (A), soil temperature (B), and soil moisture (C) for 2003–2005. Each bar represents the CV for yearly means for each treatment type. Significant differences between treatment types within each year are noted with different lower-case letters, while differences between years within each treatment type are noted with different numbers (paired t-test; $P < 0.05$).

in SRR, pre-fire variation did not differ between the two treatment types.

Means difference analysis of CVs between 2004 and 2005 showed that increases in SRR spatial variability in Mastication + Burn units were greater than those in control units. Burning in masticated stands increased CV, while prescribed fire in the Burn unit decreased spatial variability in SRR (Fig. 6A). Although CV increased significantly in the Burn unit between stages 0 and 1, means difference analysis indicated that this change was not significantly different from that in the control units. Soil temperature variability in control units increased between stages 0 and 1, while changes between other treatment types and years were not evident (Fig. 6B). In Mastication + Burn units, mastication and prescribed burning together increased spatial variability in soil moisture, whereas burning in the masticated units did not have an impact (Fig. 6C).

When monthly CVs were plotted over the sampling seasons, Mastication + Burn CV in SRR was generally higher than in the other two stands, except from September to November 2004, when Burn unit CVs were higher (Fig. 7a). Trends did not appear to follow any clear seasonal pattern. Spatial variability in T_s peaked in treatment units in November 2004 following the late-October rainfall, and in July 2004 for M_s (Fig. 7b and c). Regression of SRR, T_s, and M_s CVs against each other yielded mixed results. In 2003, higher CVs in M_s corresponded with lower CVs in T_s in Mastication + Burn units (F-test of model fit; $P < 0.05$). In 2004, SRR CVs increased with lower M_s CVs for both Burn and control units ($P < 0.05$).

4. Discussion

The range in annual growing season mean soil respiration for all units and years in this study (2.37–4.55 μmol m^{-2} s^{-1}) is similar to that reported in ponderosa pine plantations in the Sierra Nevada by Xu and Qi (2001; 2.43–6.03 μmol m^{-2} s^{-1}), and in central Oregon by Law et al. (1999; 1.0–6.5 μmol m^{-2} s^{-1}). The lower upper extents are reflective of the limited sampling season in this study, as the highest SRR in the other reports were documented in the early spring, when the Granite plantations were not consistently accessible due to snow cover.

4.1. Treatment effects

Even in the relatively homogenous plantation forest sites, the response of soil respiration rates to fuels reduction treatments was complex. Despite the narrow

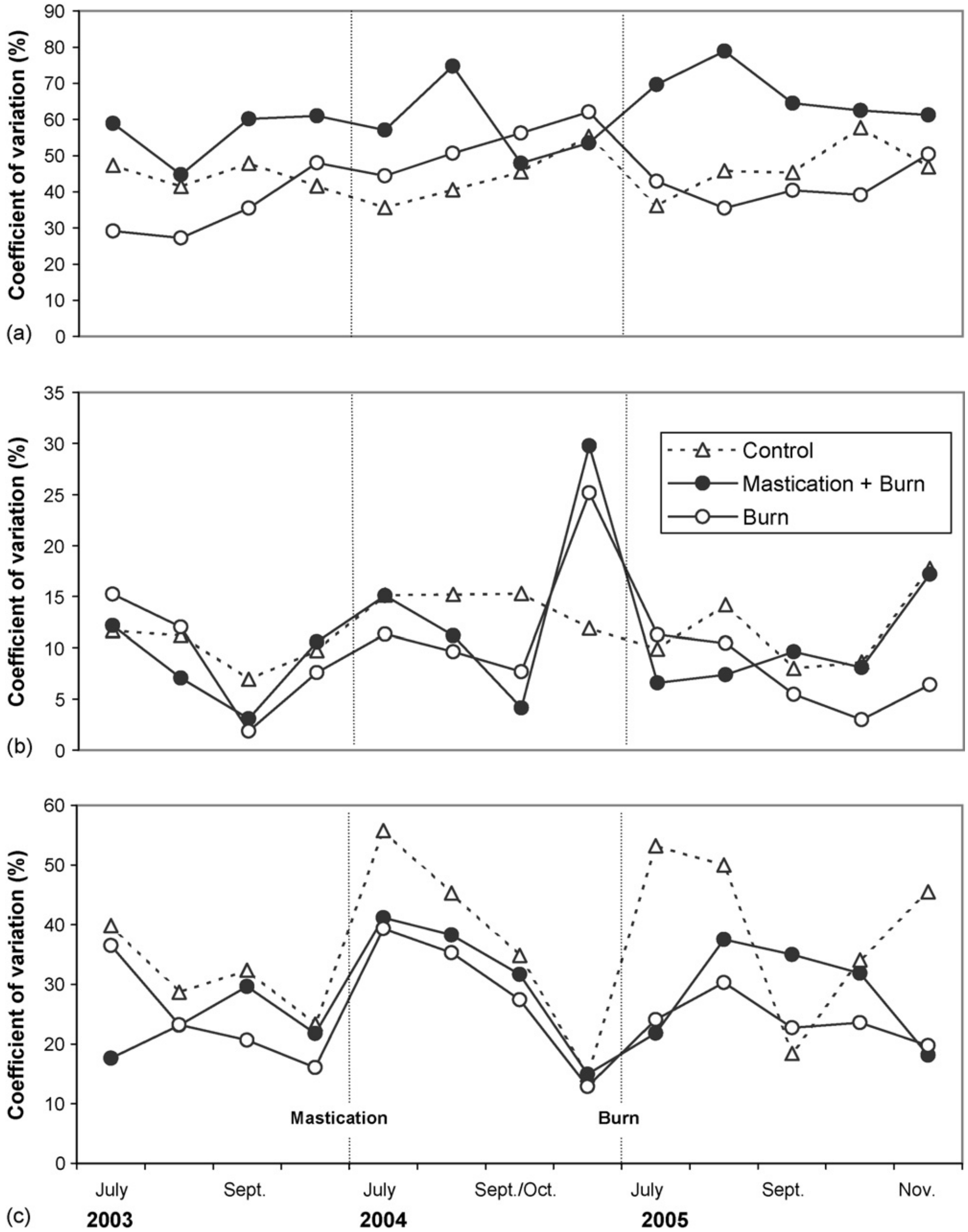

Fig. 7. Seasonal trends in spatial variability over 3 years in soil respiration (a), temperature (b), and moisture (c) levels in control and treatment stands in the Stanislaus National Forest, CA. Mastication + Burn units were masticated in 2004 and burned in 2005. The Burn unit was also burned in 2005 and was untreated during 2003–2004.

range of SRR and the high degree of spatial variability over the three seasons of sampling, overall differences between disturbed and undisturbed sites were detectable, if not consistent between treatment types. Although pooled mean differences in SRR, T_s, and M_s appeared greater in undisturbed than disturbed sites between 2004 and 2005, this result is more likely a product of the contrasting individual responses of Mastication + Burn and Burn units to the prescribed fire. Results also indicate that the combination of mastication and burning had a different impact on CO_2 efflux patterns than did mastication or burning alone. Furthermore, burning masticated sites affected SRR, T_s, and M_s differently than burning non-masticated sites.

Although mastication did not impact mean annual SRR, differences between paired months indicated a pulse of reduction in respiration rates. This short-term response is likely due to the reduction in root respiration resulting from the shredding of vegetative material and the reduction of live root contribution to SRR. The finding of no net yearly change in SRR may be the result of a differential mastication impact on heterotrophic and autotrophic sources. Although other studies addressing the effects of forest mastication on soil respiration are not available for comparisons, conflicting results from thinning experiments lend insight into this finding. Logging slash has been found to promote microbial soil activity (Sohlenius, 1982), and deeper litter layers have been linked to higher SRR in thinned forest stands (Concilio et al., 2005), suggesting that mastication of understory vegetation and small trees would increase heterotrophic CO_2 efflux as slash provides fresh, highly nutritive food sources for microorganisms. On the other hand, short-term decreases in SRR in another Sierra Nevada ponderosa pine plantation following thinning were attributed to a reduction in root density and biomass, or a reduction of the autotrophic contribution to SRR (Tang et al., 2005). Root respiration alone in the pine plantation has been shown to account for 47% of total SRR (Xu et al., 2001). Although increases in photosynthetic rates in the remaining trees can eventually lead to higher root respiration rates (Peterson et al., 1997), significant reductions in live root biomass after mastication can be expected to result in some lowering of autotrophic respiration (Tang et al., 2005). Ketling et al. (1998) found marked decreases in soil respiration rates within 1 month of root severing, while 2 years after thinning Concilio et al. (2005) reported increases in SRR. It is plausible that in the Granite plantation, decreased autotrophic and increased heterotrophic respiration combined to result in no net annual mastication effect on SRR. Changes in soil temperature following mastication were more straightforward. In the masticated units, inter-annual (2003–2004) decreases in T_s as indicated by the controls were most likely mitigated by the addition of insulating slash residues (Gordon et al., 1987).

While there were parallel increases in all variables in burn and control units between stages 1 and 2, SRR and T_s in Mastication + Burn sites decreased as a result of burning. That the increase in burn unit SRR was lower than in the controls suggests that prescribed fire may have mitigated a seasonal increase in SRR between the years in this unit. Lower soil respiration rates in high versus low-intensity prescribed burns in Mediterranean maquis terrain resulted in decreased soil microbial activity (D'Ascoli et al., 2005), and reduced post-fire heterotrophic SRR was attributed to temperature-induced loss of moisture in the humus layer (Pietikainen and Fritze, 1993). Authors studying wildfire effects on CO_2 flux in aspen (*Populus tremuloides* Michx.) and spruce (*Picea* spp.) stands also found that soil temperatures were warmer and surface layer moisture potential was reduced, resulting in decreased SRR (Weber, 1990; O'Neill et al., 2002). Alternatively, Wuthrich et al. (2002), found that in a higher intensity prescribed burn, microbial biomass decreased slightly, even though soil respiration increased and remained high for months after the fire. The reason that SRR may increase after burning is that even moderately intense fires generate a large pool of easily decomposable compounds from dead plant cells, which are then respired by soil microorganisms (Wuthrich et al., 2002).

In this study, however, mastication followed by prescribed fire resulted not only in a distinct reduction of mean annual SRR and soil temperature, but also in a marked increase in soil moisture. This trend was also documented when compared with pre-treatment moisture values in Mastication + Burn units. Fire's effect on soil moisture is a product of multiple hydrologic processes (DeBano, 1991). Studies have documented increases (Haase, 1986; Ma et al., 2004), reductions (Ma et al., 2004), and no change (Ryan and Covington, 1986) in soil moisture after fires (Hart et al., 2005). In the masticated and burned Granite plantation units, the increase in M_s was most likely a product of reduced evapotranspiration losses, decreases in litter and duff layer interception of water, and some reduced canopy interception losses. The prescribed burns resulted in widespread scorching of tree boles and canopy foliage, as well as significant consumption of the litter layers. Yet these effects also apply to the Burn unit, where SRR was not reduced following the fire and there was no change in M_s. Although the Burn unit was not treated prior to the fire, understory vegetation characteristics were similar to the masticated Mastication + Burn stands, which had more extensive understory vegetation prior to mastication. The post-fire reduction in litter depth and cover were greater in the Mastication + Burn stands than in the Burn stand, and as measures of fire-induced tree injuries were significantly higher in the Burn unit, it is more likely that a differential effect of the burns on heterotrophic, rather than autotrophic respiration, is responsible for the different treatment effects. Interestingly, similar results were reported for an old-growth forest in the Sierra Nevada, where M_s was significantly increased after thinning combined with burning, but decreased after burning (Ma et al., 2004). In the Granite plantation, lack of replication in the Burn only treatment may have significantly confounded these results.

At soil moisture levels below ~20%, increases in soil moisture have been linked to higher SRR (Tang et al., 2005). That the distinct increase in soil moisture in Mastication + Burn units did not result in increased SRR suggests that either microbial activity or microbial biomass, or both, were impacted by the burning. Microbial activity and the resulting CO_2 respiration is linked to soil moisture, generally increasing with moisture availability (Brady and Weil, 1999). Significantly higher M_s should correspond with higher heterotrophic SRR. A potential conclusion, therefore, is that the prescribed fire significantly reduced the abundance or activity of heterotrophic organisms in masticated units. Busse et al. (2005), showed that when post-mastication residue depths of 7.5 cm or greater were experimentally burned in California pine plantations, maximum surface soil temperatures reached 600 °C in dry soils (~4% volumetric moisture content) and over 350 °C in wet (25% moisture) soils. Heating to lethal plant thresholds (~60 °C) up to 10 cm in soil depth lasted for 7 h in dry soils (Busse et al., 2005). Soil moistures in the Granite plantation masticated units were less than 10% just prior to burning, and fire decreased litter and duff depths from an initial average of 8.9 to 3.75 cm. The high degree of residue consumption along with the lower range of soil moisture suggests that soil temperatures in the masticated units' O horizons exceeded the lethal limit for fine roots as well as most soil microorganisms (Hart et al., 2005).

4.2. Roles of soil temperature and moisture in dictating SRR trends

Many efforts have been made to explain fluctuations in forest soil respiration using biophysical factors; most frequently soil temperature and soil moisture (Law et al., 1999; Xu and Qi, 2001; Ma et al., 2005; Tang et al., 2005). The type of relationship these factors have with SRR and each other depends on the moisture status of the ecosystem in question. In summer drought, Mediterranean-type climate zones, SRR relationships to T_s are stronger and positive when M_s is higher and weaker and/or negative at lower M_s levels (Xu and Qi, 2001; Concilio et al., 2005; Ma et al., 2005; Tang et al., 2005). In this study, soil temperature and moisture variables were generally ineffective in explaining spatial variation in SRR, although relationships were stronger in the latter months of each sampling period, when soil moistures were higher. That a positive early Fall season trend between spatial differences in SRR and T_s was found only in 2004 and 2005 suggests that the forest treatments may have influenced the sensitivity of soil respiration to T_s. Also, Q_{10} values depicting spatial relationships between SRR and T_s were highest for these months following mastication.

The seasonal relationships between T_s and temporal variation in SRR were stronger than the spatial relationships, a finding corroborated in a similar ponderosa pine plantation by Tang et al. (2005), and Xu and Qi (2001). In notable contrast to the control sites, the exponential relationships relating SRR and T_s over time produced significant predictive models in masticated and masticated and burned units. Although Tang et al. (2005), found that Q_{10} did not vary in response to thinning in a ponderosa pine plantation, this work showed that temperature sensitivity in masticated sites was higher than in controls, but that a decrease in sensitivity followed prescribed fire in both Burn and masticated units. This reduction was especially evident in Mastication + Burn units, where soil moistures increased between 2004 and 2005. Other work has also shown that soil respiration had a reduced sensitivity to soil temperature at higher soil moisture levels (Xu and Qi, 2001). In an Alaskan spruce forest, burning also resulted in a decrease in soil respiration sensitivity to changes in soil temperature (O'Neill et al., 2002).

4.3. Spatial variability

Coefficients of variation in the Granite plantations were slightly higher (39–66%) than in other pine plantations. Xu and Qi (2001) reported a 30% CV, while Fang et al. (1998) found variation to be around 55% in a slash pine plantation. In other ecosystems, the highest coefficients of variation in soil CO_2 efflux are reported for summer months when soil moisture is lowest (Boone et al., 1998; Buchmann, 2000). The relatively low soil moistures encountered throughout this study's sampling seasons help explain the high degree of variation in soil respiration rates relative to other studies which documented soil respiration over entire years. Yet the analysis of CVs shortly after precipitation in November 2004 shows that, in the Granite plantation, variability in SRR and T_s increased immediately after a rain event. This increase may be short-term, as the response is likely more reflective of the significant drop in soil temperatures during the November post-rain sampling iteration. The Xu and Qi (2001) assertion that SRR is more sensitive to soil temperature under higher moisture conditions is corroborated by this observation.

In another Sierra Nevada ponderosa pine plantation, Tang et al. (2005) found that thinning resulted in decreased spatial variation in SRR and T_s. This decrease was attributed to an increase in the spatial homogeneity of the forest canopy, as remaining trees were more evenly distributed across the site and more uniformly sized. In the Granite plantation, spatial heterogeneity in the forest canopy following mastication was also decreased due to the removal of smaller trees. The objective of the mastication treatment was to create a relatively open, even-spaced plantation stand, where competition from understory vegetation was significantly decreased, if only for the short-term. The main understory constituent in these stands, whitethorn, sprouts readily from intact roots following top-kill.

Changes in the spatial homogeneity of SRR in this study were only detectable following prescribed fire, when variability increased in Mastication + Burn units but decreased in the Burn unit. This result reflects observations that the understory fire in masticated units burned less consistently than in the Burn unit. Although more-consistent overstory spacing after mastication increased stand homogeneity, the contribution of slash residuals to the forest floor was uneven, presumably leading to microscale differences in fire behavior and its effects on SRR in Mastication + Burn units. The spatial coefficient of variation in litter depth in Mastication + Burn units after the fire was 1.03 in contrast to 0.77 in the Burn only unit.

In a Sierra Nevada mixed-conifer forest, experimental burning where duff moisture content was between 30 and 120% resulted in strong spatial variation in duff consumption in relation to distance from tree bases, in contrast to burning when duff moisture was below 30% (Hille and Stephens, 2005). In the Granite plantations, duff moisture contents in the Burn unit averaged 19%, much lower than the 75% in the Mastication + Burn unit. As documented by Hille and Stephens (2005), duff moisture was lower closer to tree bases due to overstory interception. In the Granite plantations, owing to the decreased tree and shrub cover, rainfall 1 week prior to burning may have wetted the forest floor more in the masticated stands than in the untreated stands, resulting in the higher duff moisture levels and a higher spatial variability in forest floor consumption. This variability in consumption may have translated to increased variability in SRR and M_s in the Mastication + Burn units and not in the Burn unit, where lower duff moisture content would have corresponded to more-consistent duff consumption (Brown et al., 1985).

Concilio et al. (2005) suggested that where litter depth variability was lower following fire, SRR homogeneity was increased. The converse would be consistent with the increased spatial homogeneity of SRR in Burn units following prescribed fire in this study. The fact that variability in both SRR and M_s in Mastication + Burn units increased after the fire supports the hypothesis that fire behavior was patchy within the masticated sites. In the Burn unit, it is possible that fire effectively mitigated any pre-burn heterogeneity in one or more of the drivers of SRR, such as microbial activity in the forest floor or fine roots in the surface soil.

5. Conclusion

The effects of fuels reduction treatments on soil CO_2 differ by treatment type, and are markedly influenced by compounding treatments. Mastication followed by prescribed fire reduced mean soil respiration, while respiration rates in the untreated burned site increased after fire. Mastication alone had no net impact on soil respiration, although soil temperature and moisture were reduced. This study has begun to address the impacts of mastication, prescribed fire, and the combination thereof on soil respiration and its most commonly measured environmental drivers. A more complete understanding requires longer-term monitoring, which can continue to untangle the complexities of treatment effects through analyses of changes in both spatial and temporal variability of SRR and its drivers. Although soil moisture and temperature are used in large-scale estimates of global patterns of soil carbon efflux (i.e. Raich and Potter, 1995), this study suggests that forest management practices can have significant impacts on the relationships between these variables. Attempts to model carbon cycles in disturbance-prone forests should address the influence of both anthropogenic and natural disturbances on soil carbon efflux, especially in fire-prone forests where fuels reduction prescriptions await widespread implementation. That significant post-treatment changes were detected in SRR implies that larger-scale fuels reduction manipulations could have important consequences to ecosystem carbon dynamics.

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