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Climate Change, Carbon Sequestration, and Wildfire Management in Sierran Mixed Conifer Forests

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I. Abstract

We measured the impacts of prescribed fire and small clear-cut tree harvesting on carbon dynamics in a mixed-conifer forest in the central Sierra Nevada. Soil CO_2 efflux, above ground tree biomass, annual tree radial growth, and ecosystem carbon stored as litter, fine root and in the mineral soil were measured in four treatment sites: an un-manipulated control, a prescribed fire site, and two harvested sites, in one of which the soil was mechanically ripped to reduce soil compaction, a common practice done on industrial forest lands in the Sierra Nevada. Biomass and radial tee growth was also measured in a thinned site and a thinned and burned site. The biomass was determined one year before treatment, in 2001 one year after, in 2003, and finally seven year after treatment in 2009.

Treatments affected pools and exchanges of carbon, reducing biomass stocks and the capacity of the forest to uptake carbon, but also to released C from the soil surface. Soil respiration was decreased by both fire and harvest (circa 20%), in part because the disturbances altered energy input and water availability. Prescribed fire and thinning reduced stand biomass, removing mostly small unselected (fire) or selected (thinning) trees. The thinning effects were concentrated in the first post-treatment year, whereas fire effects on tree radial growth showed responses between the first and the seventh years after fire. In the seven years post-disturbance interval, fire decreased tree density 30%, radial growth 12%, and stand radial growth 27%. Thinning reduced only 9% tree density, and increased tree radial growth 20% and stand radial growth 6%, compared to the control site. Effects of fire and thinning on biomass and biomass productivity were similar. After seven years, the biomass of both the fire and the thinned site reached pre-treatment levels. Excluding the first post-disturbance year, the thinned site stored more carbon than the control site, were radial growth has being declining over the same period. The effects of fire combined with thinning were higher than for the singe treatment, and growth, biomass and productivity were decreased between 30 and 40% compared to the control site.

II. Background and Purpose

Forest ecosystems constitute a major reservoir of the global terrestrial carbon (Houghton et al. 1990; Tans et al. 1990) and have the potential to sequester anthropogenic carbon emissions. Therefore, understanding carbon cycling in forest ecosystems is critical for estimating the future global carbon budget. In addition most forests are altered by centuries of human impact. For example, many forests throughout western North America are now prone to large, intense fires because of the last century fire suppression resulting in high fuel accumulation and plant density (Agee 1993, Hessburg et al., 2005). Removals of biomass by mechanical methods or by prescribed fire are practices aimed to reduce risks of high severity fires and restoring forests to their pre-settlement conditions. However mechanical thinning or prescribed fires are also disturbances for forest ecosystems. Disturbances usually reduce photosynthetic area and affect ecosystem carbon dynamics, often switching forests from sink to source of carbon (Amiro 2010, Dore 2012). Their effect on ecosystem carbon greatly varies with type, intensity, and frequency of silvicultural treatment applied. Even if treatments cause short-term carbon losses, it is necessary to compare these costs with the net long term costs of more intense, high intensity fires (Campbell 2011). In addition, it is necessary to consider benefits such as enhancement of forest resilience and resistance, increased biodiversity, improvement of hydrological benefits, and erosion protection. It is important to understand the effects of management practices on carbon balances in forest ecosystems to be able to quantify their long term costs, to be able to minimize carbon losses during and after treatments, and estimate the time necessary to recover the carbon lost from treatments.

Specifically, the objectives of the present study were to quantify the effects of the most commonly used fuels treatments and commercial harvesting methods used in mixed conifer forests in Sierra Nevada on ecosystem carbon pools and soil CO_2 fluxes. Mixed conifer forests are the most common industrial forest lands in the Sierra, are one of the prime water sources for California, and are critical habitat for many important species. We compared four different treatment types: prescribed fire, unmanipulated control, clear cut harvest units with activity fuels removed, and clear cut harvest units with and without activity fuels removed followed by mechanical soil ripping. For each treatment we estimated carbon stored in the main ecosystem carbon pools: tree biomass, litter, fine root and mineral soil. In addition we quantified how in this forest treatments affected carbon sequestration as aboveground tree growth and carbon release as soil CO_2 efflux.

III. Study Description and Location

The study was conducted at Blodgett Forest (38°54′N, 120°39′W), a University of California Research Station in the central Sierra Nevada near Georgetown, California. Blodgett is and dominated by mixed-conifer forests composed by sugar pine (*Pinus lambertiana*), ponderosa pine (*Pinus ponderosa*), white fir (*Abies concolor*), incense-cedar (*Calocedrus decurrens*), Douglas-fir (*Pseudotsuga menziesii*), and California black oak (*Quercus kelloggii*). Elevation ranges from 1100 to 1410 m. Total annual precipitation averages about 1600 mm, falling mainly between September and May. The average minimum daily temperature in January is 0.6 °C and the average maximum daily temperature in July is 28.3 °C (Xu and Qi, 2001).

The loamy-sandy soils are underlain by Mesozoic, granitic material and are predominantly classified as the Holland and Musick series (fine-loamy, mixed, semiactive, mesic Ultic Haploxeralfs; Olson and Helms 1996). The forest, an actively managed commercial timberland, has been repeatedly harvested and subjected to fire suppression for the last 90 years reflecting a management history common to many forests in California (Laudenslayer and Darr 1990). Fire was a common ecosystem process in Blodgett Forest before the policy of fire suppression began early in the 20th century. Between 1750 and 1900, the median composite fire interval at the 9–15 ha spatial scale was 4.7 years with a fire return interval range of 4–28 years (Stephens and Collins 2004).

To characterize carbon dynamics of silvicultural treatments commonly used in mixed conifer forests, this study compares four different treatment types installed in three compartments: prescribed fire (FIRE) ; un-manipulated control (CTRL); and clear cut harvest with and without mechanical soil ripping ($HARV_{RIP}$, $HARV_{NO-RIP}$). The compartments are subject to the same climatic and edaphic conditions because of their close proximity (less than 10 km apart). The FIRE and CTRL sites were part of the Fire and Fire Surrogate Study (FFS), a study started in 2000 to analyze the effect of fuel treatments on vegetation structure and other ecosystems elements at 13 locations across US (Stephens and Moghaddas 2005, McIver et al., 2012). The FIRE site was burned initially in the fall of 2002 and burned a 2^{nd} time in the fall of 2009. Four small clear cut harvest areas (each between 2000 and 7000 m^2) were installed in summer 2010. In these areas all trees were removed and the residual material was piled and burned. In two of the four units, soils were mechanically ripped, a common post-harvesting practice used in the region to prepare the soil for tree planting.

Above tree biomass and annual tree radial increment

In the Blodgett forest, during the Fire and Fire Surrogate Study (FFS) four treatments were implemented: an un-manipulated control, a prescribed fire only, a mechanical thinning only, and a prescribed fire plus mechanical thinning (fire plus thinning). Each treatment had three replicates, compartments with areas of between 12 and 28 ha. In each compartment, every tree diameter (at breast height), species and plant status was measured in 20, 0.04 ha circular plots distributed on a systematic grid (240 plots total). Only trees with diameter larger than 5 cm were considered in this study. Measurements of the same plots were repeated the year before treatments, in 2001, the first year after treatments, in 2003, and the seventh year after treatments, in 2009. In 2010, in each of the three replicates of each treatment, 30-60 trees of each of the five main species, distributed over a gradient of size classes, were cored to quantify annual radial bole increment from circa 1960 to 2009. Data were used to calculate the average annual radial increment of each main species over five size classes, based on the tree diameter at breast height (diameter below 35 cm, 35 to 55 cm, 55 to 75, 75 to 95 cm and above 95 cm) from 7 years before to 7 years after treatments. If a size class had no sampled radial increments, its increment was estimated as the average of the two adjacent classes (same species, year and unit), or as the increment of the closest class (in case of missing smallest or biggest classes). The annual radial increment was scaled up from tree to stand level as the sum of the bole increment of each tree present in the stand. This stand bole increment includes both, the effect of treatments on the increment of single trees and the effect of treatments on tree density.

The annual increment of the sampled trees was used to calculate annually the diameter of every tree between 1995 and 2009, and, using allometry, the stand biomass for the year. For the period 1996 to 2001 diameters were back-calculated from all trees recorded in 2001; for the period 2002 to 2005 diameters were calculated from all trees recorded in 2003; and for the period 2006-2009 from the trees recorded in 2009. The difference in stand biomass between two consecutive years was used to express the stand annual aboveground tree productivity This methodology had high accuracy than relying on measurements from dbh tapes. Tree biomass was calculated using allometric equations provided in Jenkins et al. (2004). Aboveground and fine root biomass was converted to carbon assuming a carbon concentration of 48% (Penman et al. 2003).

In 2010, after the harvest, in the logged areas, 1 year old saplings were planted on a 2.4 x 2.4 m grid. To calculate sapling biomass and analyze the effects of mechanical soil ripping on the new generation of trees, during summer 2012, diameter and height of saplings where measured on 4, 115 to 190 m long transects per unit, running between the farthest edges of the units. Both, the $HARV_{RIP}$ and $HARV_{NO\_RIP}$ treatments, had two replicates, each with between 240 and 400 trees measured. 85 saplings were cut, and each tree was divided in its woody and leaves component. The relationship between diameter and dry weight of leaves, wood, and whole plant was determined separately and was used to calculate the carbon stored as tree biomass in the young plantation. The resulting equations were $b_L = 1.5305\ dm^{2.1416}$ ($R^2 = 0.73$) for leaves, $b_s = 0.9741\ dm^{2.2691}$ ($R^2 =$

0.87) for stems, and $b_P = 2.5502\ dm^{2.1834}$ ($R^2 = 0.82$) for the whole plant, where b_L, b_S and b_P was the dry weight of leaves, stem and whole plant respectively, and *dm* was the diameter at the base of the plant.

The quantify effects of treatments the BACI approach was used (Stewart-Oaten & Bence, 2001), where post-disturbances differences between a treatment and control were corrected for pre-existing differences between the same sites. In case more than a year in the pre or post disturbance periods was available, as of the annual radial stem increment, differences were averaged over a seven year pre and post treatments period. If only post-treatment data were available, we calculated the effect of treatments as the difference between CTRL and treatment sites.

Soil CO_2 efflux

Soil CO_2 efflux measurements were taken using two different techniques: the chamber technique and the profile technique. With the chamber technique we periodically measured soil CO_2 efflux in multiple locations at each experimental site. We used the profile technique to monitor soil CO_2 production and efflux continuously at each site. Because of the high costs and complexity of the profile measurement systems, we installed only one profile per treatment site. We feel this approach complemented and validated each other and were able to quantify both spatial and temporal variability in soil CO_2 efflux.

a. Chambers

We measured soil CO_2 efflux using a Li-6000 and later a Li-6400 with the soil chamber attachment (Li-Cor, Lincoln, USA) every two weeks at 110 locations over a two-day period during snow free periods, starting in June 2011 and continued until end of 2012. In spring 2012, the Li-6000 and Li-6400 were inter-compared in the field obtaining very good results ($r^2 = 0.95$) and 2011 data were corrected for the 15 % difference found between the two instruments. Measurements were restricted to the interval from 9:00 to 17:00 hours to limit the effect of daily fluctuations and the order of the sites and plots changed randomly, always measuring the UND and FIRE site and the $HARV_{RIP}$ and $HARV_{NO\_RIP}$ sites in the same day. During measurements the chamber was positioned on 10-cm diameter PVC soil collars, installed 1 cm into the soil to avoid soil disturbance during measurements and to repeat measurements of the same locations. Soil CO_2 efflux was calculated from the change of CO_2 concentration over time and averaged for two cycles over a 10 ppm range encompassing the ambient CO_2 concentration. Soil water content (SWC, measured at 0 – 5 cm depth using a HH2 and ML2x, DeltaT devices, Cambridge, UK) and soil temperature (Ts, measured at 10 cm depth using a 6000-09TC, Licor, Lincoln, USA) were measured nearby each soil CO_2 efflux collar.

Measurements in the FIRE and CTRL treatments were collected on 29 different locations. In the harvested area, we measured soil CO_2 efflux in 20 different locations for both the $HARV_{RIP}$ and $HARV_{NO\_RIP}$ treatments. In the CTRL and FIRE sites, the plots were scattered over a total area of circa 150 m x 200 m and at the HARV sites over an area of circa 150 m x 400m.

b. Soil CO_2 profile

We used solid-state CO_2 sensors (CARBOCAP model GMM 220, range 0–1%, Vaisala, Helsinki, Finland) installed at 2, 8, 16, and 24 cm soil depths at each site. The CO_2 sensors had a range of 0–10,000 ppm. Soil temperature was measured with thermocouples at the same depth used for the CO_2 sensors, and soil water content (ECHO, Decagon, Pullman, WA) was measured at two depths, 8 and 16 cm.

Temperature and pressure corrected CO_2 concentration from the sensors were used to calculate soil respiration (expressed in μ mol m^{-2} s^{-1}) based on Fick's first law of diffusion $Rs = -Ds\, dC/dz$ (Tang et al. 2005). *D*s is the diffusion coefficient in the soil and *C* is CO_2 concentration at depth *z*. *Ds* is calculated by the product of the CO_2 diffusion coefficient in the free air, *Da*, and the gas tortuosity factor, ξ.

ξ was modeled using a site specific empirical specific model (as in Yassal et al., 2005) based on its relationship with the soil air filled porosity (ε), that was determined by the soil SWC, bulk density, and particle density for the mineral soil. ξ and it's relationship with ε was determined in the lab for samples collected in the area adjacent to the profiles (two replicates per site, at 3 depths: 0-5 cm, 10-15 and 15-20 cm, 24 samples total). The air above the 5 cm diameter, 5 cm long aluminum cylinder containing the soil sample was sealed in a 100 ml volume, and initially flushed with nitrogen. Air was able to diffuse from the bottom of the cylinder trough the soil. We measured the increase of oxygen versus time in the volume above the soil (using a SO-200 Apogee, Logan, USA) for 3-5 minutes after 2 minutes necessary to reach a linear, constant rate. For each site and depth measurements were repeated for four different water content status (96 total measurements). Soil water content was determined from the weight of the sample determined after each diffusion measurement, and the dry weight (102 °C for 24 hours and until constant weight) obtained, as well as the bulk density, for each soil sample at the end of all measurements. The resulting equation was $\xi = 0.0139e^{6.2889*\varepsilon}$ ($r^2 = 0.7$) and no difference were found among soils of different sites and depths (Fig 1).

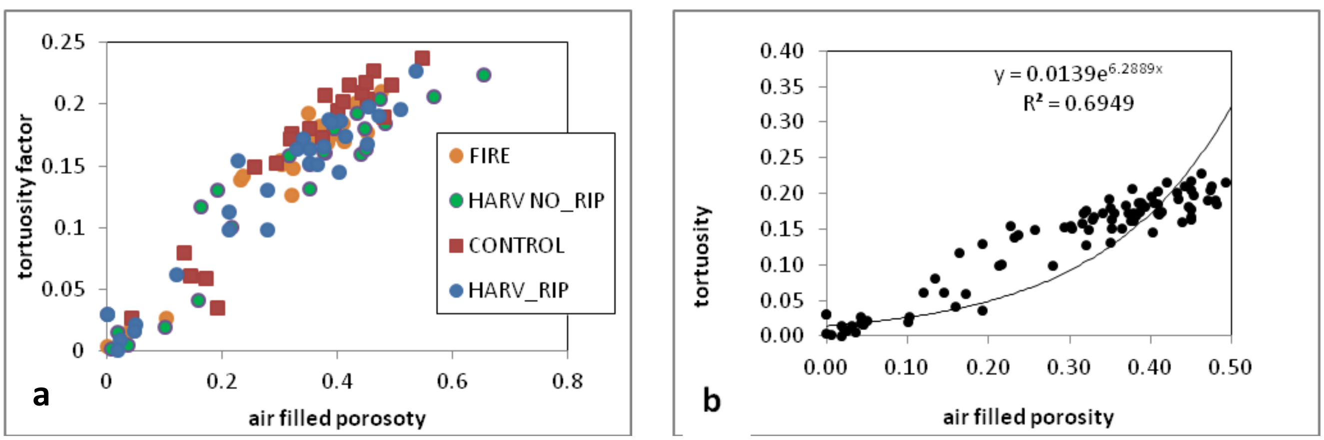

Fig 1: *Result of the empirically determined tortuosity factor. a) Comparison of results at the control and fire and harvest treatment sites. b) Exponential model fitted to experimental data.*

In winter 2012 sensors were calibrated against known CO_2 concentration in the lab. Because we installed only one profile at each treatment area, whereas the chambers covered a much larger area (29 samples for FIRE and CTRL, 20 samples for the HARV sites), at each treatment site the profile data were adjusted to match the chamber averages for 2011 and 2012 (Fig. 2), obtaining a flux monitored in continuously and representing the whole treatment area. Profile measurements analyzed were conducted in 2011 and 2012.

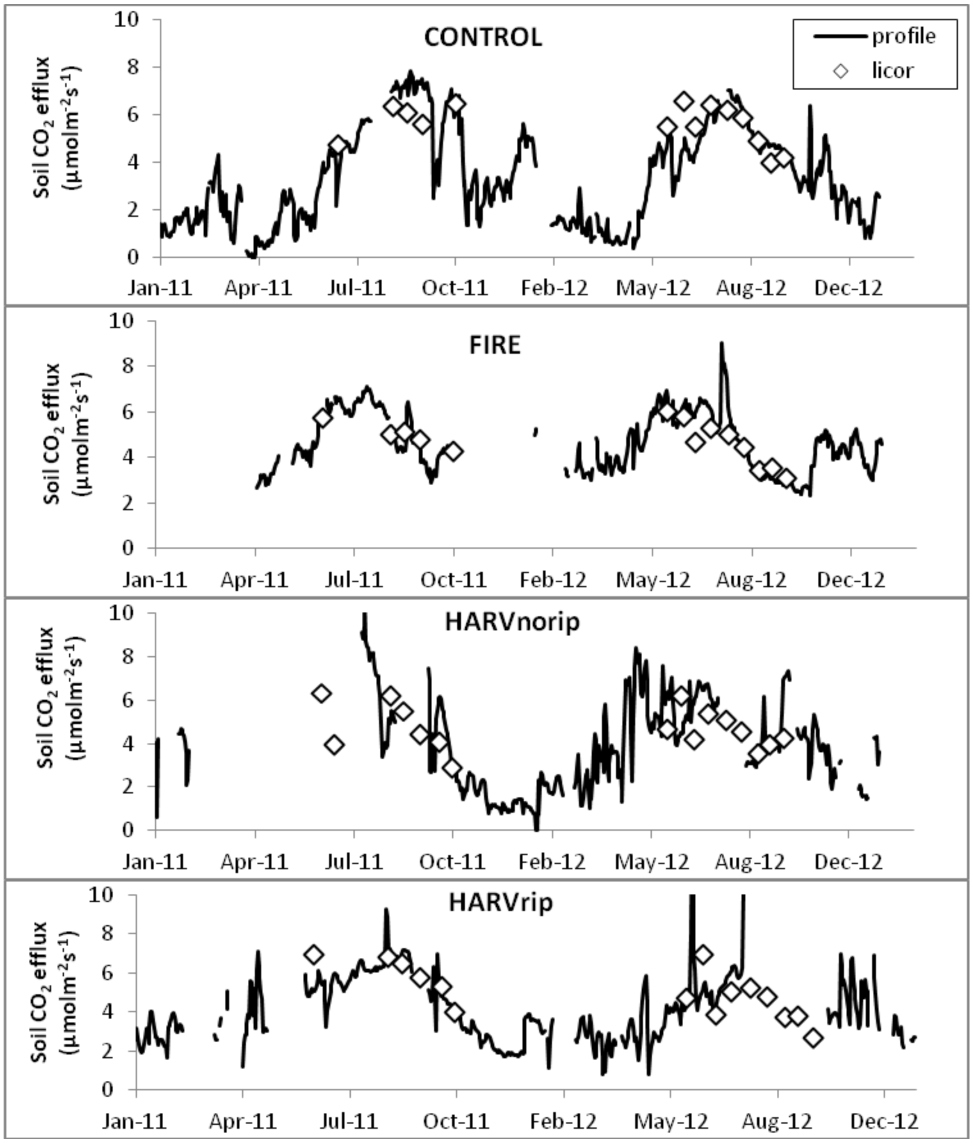

Fig 2: *Seasonal trend of soil CO_2 efflux measured with profiles in continuous (black line) and with chambers periodically on 20-29 locations per site during 2011 and 2012.*

<u>*Soil carbon pools and characteristics*</u>

Fine root biomass was measured in summer 2012. Samples of 5 cm diameter were collected at two depths (0-15 cm and 15-30 cm) on 10 locations adjacent to randomly selected soil CO_2 efflux collars at each of the four sites.

Fine roots were hand-picked and separated in < 2mm, and 2-5 mm diameter classes, without distinction between live and dead roots. Dry weight was determined after drying samples at 70 °C until constant weight.

Litter was collected inside each 10 cm diameter soil CO_2 efflux collar (27 plots at the FIRE and CTRL sites, 18 plots at both the HARV sites) at the end of 2012. Dried litter (70 °C for 4 days) was divided into leaves and woody components.

Soil bulk density and carbon content

Bulk density of 0 to 5 cm mineral soil layer was determined on 5 cm diameter, 5 cm high samples collected inside each soil CO_2 efflux collar (27 plots at the FIRE and CTRL sites, 18 plots at both the HARV sites) after removing the litter. Bulk density of the deeper 5 to15 cm soil layer was determined on 8 locations (5 cm diameter, 10 cm high samples), adjacent to randomly selected soil CO_2 efflux collars at each site.

To estimate carbon content of the 0-15 cm mineral soil layer, at each site soil was sampled separately over two depths (0-5 cm and 5-15 cm), over a 5 cm diameter area, in 18 locations adjacent to randomly selected soil CO_2 efflux collars. Samples were air dried, sieved (2mm), grinded and their carbon content was determined using an elemental analyzer.

IV. Key Findings

A. *Effect of fire and harvest on forest biomass and radial tree growth*

Data on forest structure (Fig. 3) shows conditions at the FIRE and CTRL sites were similar prior to treatments. The small clearcuts removed almost all aboveground biomass except for down wood that was not consumed in the burn piles. The prescribed fire treatment in 2002 decreased strongly tree density the first year after the fire. However tree density decreased additionally between 2003 and 2009, both for the fire only and the fire plus thinning, compared to a more stable condition at the thinned site (Fig. 3 and Table 1).

Carbon					N trees			
	PRE	POST 1	POST 7	Effect	PRE	POST 1	POST 7	Effect
	2001	2003	2009	PRE-POST7	2001	2003	2009	PRE-POST7
Treatment	t C ha^{-1}	t C ha^{-1}	t C ha^{-1}	%	N ha^{-1}	N ha^{-1}	N ha^{-1}	%
Control	173 (±18)	180 (±17)	203 (±19)	+29	590 (±46)	581 (±43)	558 (±34)	-6
Burned	157 (±8)	157 (±10)	161 (±11)	-12	476 (±35)	393 (±38)	282 (±41)	-30
Thinned	176 (±4)	151 (±4)	177 (±3)	-15	415 (±60)	357 (±10)	344 (±10)	-9
Thin +Bur	185 (±12)	140 (±12)	147 (±14)	-36	523 (±22)	233 (±3)	221 (±7)	-49

Table 1: *Carbon stored as aboveground biomass and tree density for control, burned, thinned and thinned and burned (Thin+Bur) sites the year before the treatment (PRE, 2001), first year after treatment (POST1, 2003) and seventh year after treatment (POST 7, 2009). Data represent the average of 3 replicates where trees > 5 cm diameter were measured in 20, 0.40 ha plots. Effects of treatments on carbon were expressed as relative difference between the year before and the 7th year after treatments.*

Tree radial increments decreased after fire (-12%), increased slightly when coupled with thinning (5%), whereas it increased 20% after thinning alone (Fig. 4a). For each species (*Pinus lambertian, pinus ponderosa*, *Abies concolor*, *Calocedrus decurrens*, *Pseudotsuga menziesii*) burning decreased the annual radial bole increment between 7 and 20%, thinning increased radial increment between 13 and 30%, and thinning plus burning had a mixed effect, increasing *Abies concolor* radial increment 30% , but decreasing *Pseudotsuga menziesi* radial increment 13%. If we scale to the stand level, because of the decreased tree density at the treated sites, (Fig. 4 b), the before-after analysis showed radial increment decreased 27% after burning, increased 6% after thinning, and decreased 7% after combined fire plus thinning. All treatments reduced stand aboveground productivity (as difference between annual biomass of two consecutive years), however the fire plus thinning site had the lower post-treatment average productivity of 6 t C ha^{-1} (2003-2009), with a treatment effect of -44% (Fig. 4 c).

The control site, despite the higher tree densities, is still accumulating carbon. However the radial growth decreased from 2.95 to 2.58 mm yr^{-1} over a seven year period before and after the treatment year of 2002. Carbon stored as tree biomass increased from 174 to 204 t C ha^{-1} at the CTRL site between 2001 and 2009, was reduced circa 15% by fire and thinning and 36% by fire plus thinning (Table 1). All active treatments decreased tree density, however the decrease was highest at the fire plus thinning treatment (Table 1).

In the first year after treatment the FIRE site lost 4 t C ha^{-1}, compared to the 29 t C ha^{-1} at the thinned site and 48 t C ha^{-1}at the fire plus thinning site (Table 1). However between the first and the seventh post-treatment year, the fire and the fire plus thinning sites only accumulated circa 15 t C ha^{-1}, compared to 59 t C ha^{-1} accumulated by the thinned site, and 55 t C ha^{-1} of the control site. Both the fire and the thinning treatments recovered to the pre-treatment biomass levels in our 7 years study period (Table 1).

At the HARV sites (Table 2), soil ripping didn't affect the planted saplings, at least in the short term. No differences were found between $HARV_{RIP}$ and $HARV_{NO-RIP}$ in the % of living trees, height, diameter and thus carbon, that two years after treatment, was still only around 2 g C m^{-2}.

	% Live	**height cm**	**diameter cm**	**leaves g C m^{-2}**	**wood g C m^{-2}**	**tot carbon g C m^{-2}**
no rip	87%	47.8	3.4	1.41	1.06	2.48 (±0.22)
rip	89%	48.3	3.3	1.25	0.94	2.20 (±0.22)

Table 2: *Characteristics of saplings planted at the HARV sites, with and without soil ripping. Results (± standard error) represent the average of two replicates for each treatment.*

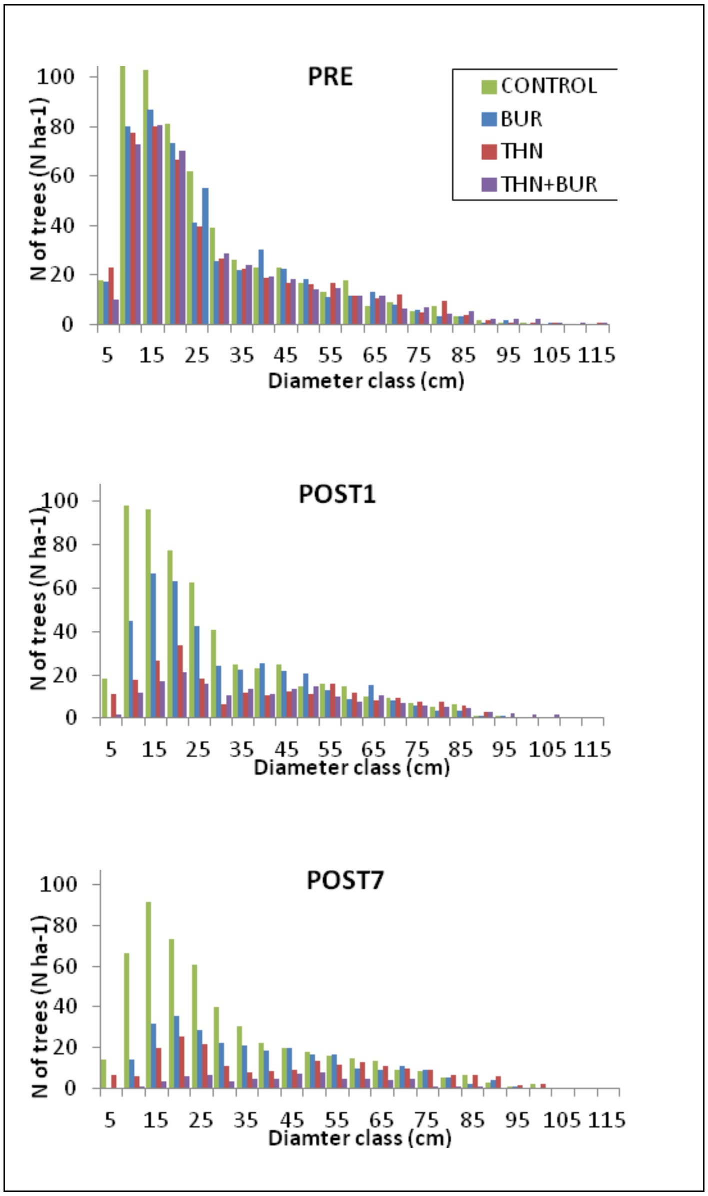

Fig. 3*: Distribution of the trees (> 5 cm diameter) over 5 cm classes in 2001, the year before the treatment (PRE); in 2003 (POST 1), the first year after treatments; in 2009 (POST 7), the seventh year after treatments. Each class is labeled with the lower interval limit.*

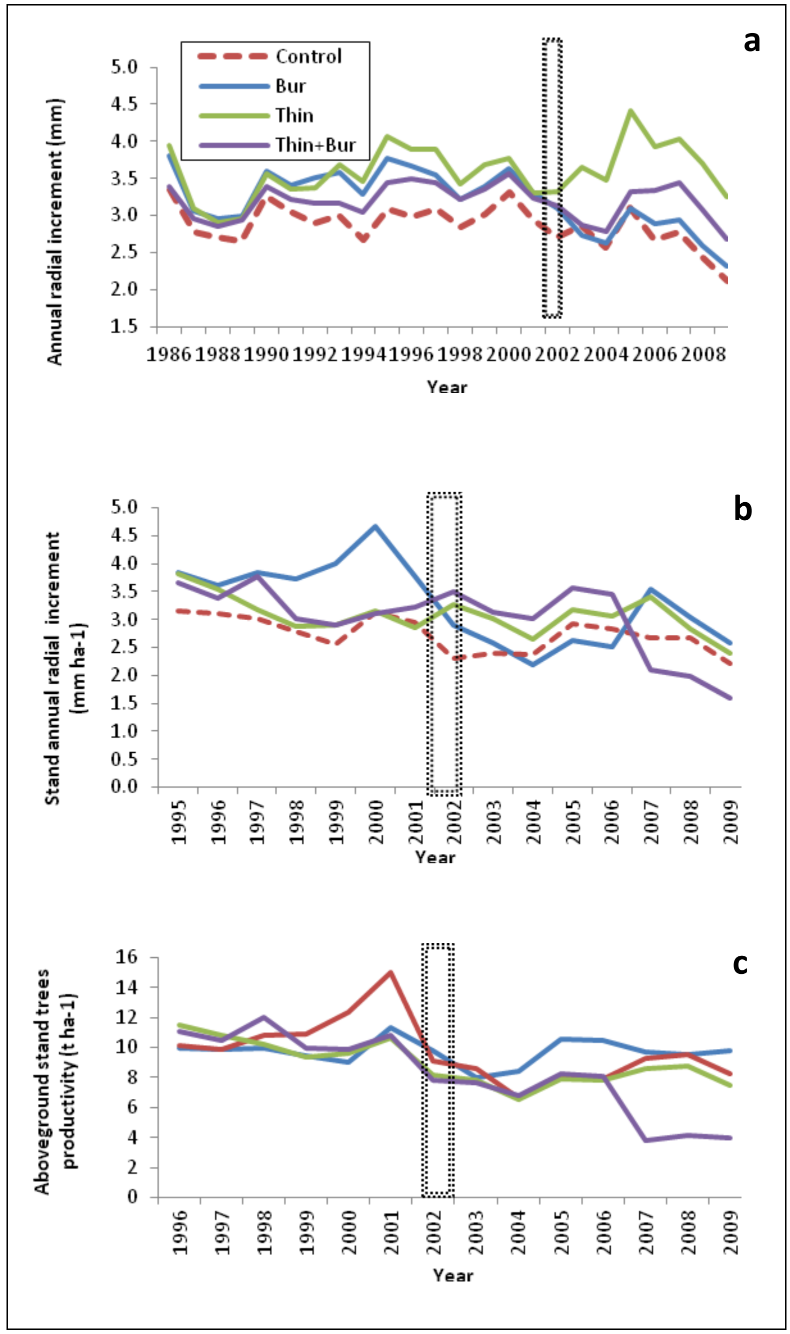

Fig. 4*: A) Trees radius annual increment for the control, burned site (Burn), thinned site (Thin), and burned plus thinned site (Thin+Bur), from 1986 to 2009. The rectangle shows the treatment year 2002. Data are the averages of B) Stand annual radial increment. The stand annual increment is the results of both, the effect of treatments on trees increment (A) and the effect of treatments on tree density. C) The difference in stand biomass between two consecutive years was used to express the stand annual aboveground tree productivity.*

B. Effect of fire and harvest on soil, fine roots and litter

Soil carbon between in the first 15 cm in the mineral layer was 8% lower at the FIRE site (Table 3), and 13% lower at the $HARV_{RIP}$ (both in the 0-5 cm and 5-15 cm) site compared to the CTRL site. Fine roots were unchanged after the fire, but reduced 60-70% after the harvest, with or without soil ripping. Also the litter layer was reduced in all treatment sites compared to the CTRL, around 40% by fire and harvest without ripping, and 80% by the harvest with ripping. In general, all treatments reduced the carbon stored at the soil surface and in the mineral soil, however the harvest with mechanical soil ripping had the greatest impact, reducing carbon in every measured mineral carbon pool. If we consider the soil related carbon pools (mineral soil, fine roots and litter) carbon stored seven years after the fire was 19% lower, and two years after the harvest was up to 26% lower in the ripped sites when compared to the CTRL site.

Pool	**(gC m^{-2})**	**Control**		**FIRE**		**HARV$_{NO-RIP}$**		**HARV$_{RIP}$**
Soil	**0-5 cm**	2435 (± *183)*	***	2453 (± *162)*		2866 (± *304)*	*	2321 (± *222)*
	5-15 cm	1953 (± *280)*		1593 (± *172)*		1979 (± *721)*	***	1512 (± *250)*
	Total 0-15 cm	4388 (± *269)*	**	4046 (± *228)*		4845 (± *605*	*	3833 (± *303)*
Fine root	**0-30 cm**	517 (± *162)*		583 (± *203)*	***	143 (± *66)*	***	191 (± *137)*
Litter	**-**	702 (± *75)*	***	383 (± *41)*	*	408 (± *91)*	***	126 (± *37)*

Table 3: *Carbon pool in the control and in the prescribed burned (FIRE), and harvested site, with and without mechanical soil ripping ($HARV_{RIP}$, $HARV_{NO-RIP}$) in 2012. Asterisks denote different level of significance in the comparison of carbon content (%) the control with the treated site (* 0.05; ** 0.01, and *** <0.001).*

C. Effect of fire and harvest on microclimate

Both fire and harvest reduced the canopy cover of the stands, with a consequent increase in the amount of energy reaching the ground and a decrease in the water used by vegetation through transpiration. Soil temperature (Ts) was 30% higher in the harvest site ($p < 0.001$) and 16% higher in the fire site ($p = 0.001$; Fig.

5b, Fig. 6) compared to the CTRL site. Similarly, soil water content (SWC) was 29% higher at the harvest site ($p = 0.02$) and 11 % higher at the fire site ($p = 0.02$; Fig. 5c and Fig. 6). As expected, both differences in Ts and SWC were highest at the harvest sites where all vegetation was removed. The fact that SWC was higher at both the FIRE and HARV sites than at the CTRL site suggests that the decrease in transpiration due to the reduced leaf area was stronger than the increase in evaporation due to the increase in ground level radiation. In general, the harvest site had the most extreme microenvironment conditions (Ts and SWC), while the CTRL site the least extreme of all sites.

An agreement between the effects of disturbances on microclimate between the high spatial resolution chamber method and the high temporal resolution profile method (data not shown) confirms that the profile locations we selected were representative of the larger scale treatment areas.

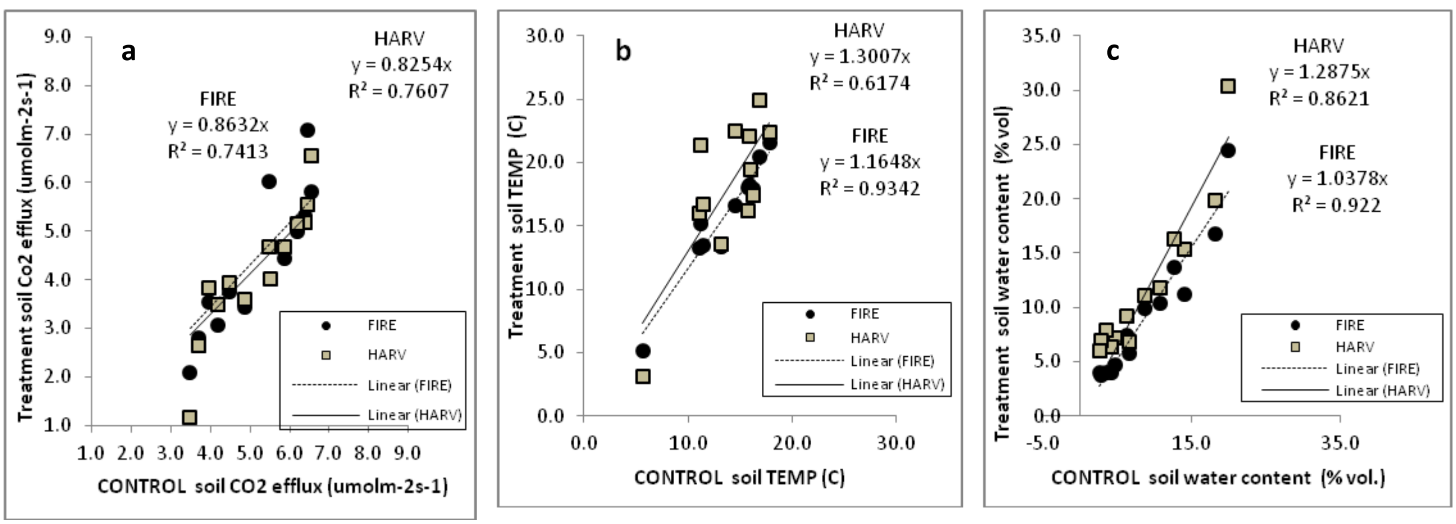

Fig. 5: *Comparison of soil respiration (a), soil temperature (b) and soil water content (c) between fire and harvest sites and the undisturbed, control site. Each symbol represents the average of the 20-29 plots measured in 2011 and 2012. The harvest data are the average of the areas where the soil was subject to ripping and areas where ripping was not applied. Slope and r^2 of the linear regression are also shown.*

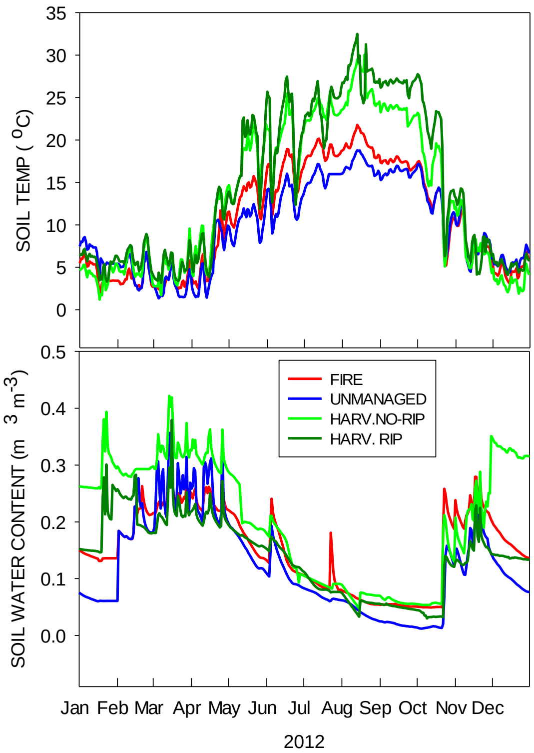

Fig 6: *Continuous measurements of soil temperature (10 cm) and soil water content (8 cm) in 2012 from profiles installed at the fire harvest and the unmanaged control site*

D. Effect of fire and harvest on soil CO_2 efflux

Soil CO_2 efflux was affected by treatment ($p = 0.03$). Soil CO_2 efflux was different at the FIRE ($p = 0.01$) and HARV site (average of $HARV_{RIP}$ and $HARV_{NO\text{-}RIP}$; $p = 0.02$) compared to the CTRL site. Soil CO_2 efflux was not different between the $HARV_{RIP}$ and $HARV_{NO\text{-}RIP}$ sites ($p = 0.5$). Soil CO_2 fluxes measured using the chamber technique between 2011 and 2012 shows both fire and harvest treatments decreased soil CO_2 efflux by 14% and 18%, respectively, compared to the CTRL site (Fig. 5a).

The distribution of CO_2 concentrations and production at different depths differed among sites. Harvest caused important changes to the CO_2 concentration profile of the top 24 cm of soil, in both dry and wet conditions. CO2 concentration at both $HARV_{RIP}$ and $HARV_{NO\text{-}RIP}$ sites were higher than at the fire and unmanaged site, especially at deeper depths (16 and 24 cm).

Both, fire and harvest decreased annual soil respiration compared to the unmanaged, CTRL site. The annual emission was decreased even if soil temperature and water content in the disturbed sites were more favorable to soil respiration.

Disturbance increased spatial variability of soil CO_2 efflux. Coefficient of variability increased from an annual average of 32% at the control site to 37% at the burned site, and 49% to 51% at the harvested sites (without and with soil ripping, respectively), mirroring post-disturbance increases in spatial variability of soil temperature and soil water content. Because of the post-harvest increase in spatial variability, the ability to detect differences from other sites became lower, and the number of samples needed to obtain a value representative of the full population mean (within a 10% range) increased by 50%, from 60 to 120 samples. This sample size is, in our case as often in other studies, too large to be sustained on regular basis at several sites through a growing season. Initially we hypothesized that the highly complex, undisturbed forest with different tree species, sizes and ages, and numerous carbon pools, needed intense sampling compared to the simplified bare soil resulting from the clear cut harvest. However, soil CO_2 efflux of the ecosystem where natural complexity was reduced artificially by disturbance was more spatially heterogeneous and difficult to assess. These findings need to be considered when quantifying effects of management or treatments on soil CO_2 efflux, and the eventual lack of an effect could be due to the lack of an appropriate sampling size.

V. Management Implications

Treatments affected carbon dynamics of the mixed conifer forest in the Sierra Nevada. Every measured pool of carbon, such as the aboveground tree biomass and carbon in and above the soil, was altered when the forest was subject to fire or harvest. Treatments also affected carbon exchange between pools, decreasing fluxes of carbon stored annually as tree biomass or released from soil, in part because the reduction of the tree cover affected the main drivers of ecosystem processes, energy and water. However treatments differed in their effects. Fire only partially opened the dense, closed forest canopy cover, partially changing microclimate and fine roots but reducing soil carbon and litter. After 7 years from the event, fire decreased biomass and tree radial growth 12%, and decreased circa 30% stand radial growth and the aboveground tree productivity. Tree density was still decreasing between the first and the seventh post-treatment year, whereas number of trees was stable at the thinned site. Thinning had a small impact on the tree density, and a positive impact on tree radial growth (a 20% increase, 6% if scaled up to stand level), compared to the fire site where damages to the surviving trees caused a decrease in single trees and whole stand radial growth. Thus fire damages persisted over almost a decade, whereas when fuel and biomass are reduced mechanically, negative impacts were concentrated in the first year after the treatment. Between 2003 and 2009, the thinned forest accumulated more carbon than the control forest (59 t C ha^{-1} compared to 54 t C ha^{-1}). However, over the first seven post- treatment years, thinning decreased productivity of aboveground biomass 22% compared to the control site, and similarly to the fire site (-28%). Both the thinned and the burned site reached in a seven year period the pre-treatment biomass level. For the thinned and burned site the recovery period was longer than the observed 7 years interval. However, all active treatments significantly reduced fire hazards when compared to controls (Stephens et al. 2012).

In comparison with the disturbance due to fire and thinning, the clear cut harvest had the strongest effect on carbon pools and fluxes. Carbon stored in and above the ground and tree biomass was reduced 95%, compared to a net loss of 20% for the fire site. Seven years after the clear cut harvest, stand productivity was close to zero, and soil fluxes were reduced in a similar way by both fire and harvest (20% circa). Ripping the soil didn’t have a positive effect on saplings growth, but it increased the impacts of harvest on carbon dynamics.

In conclusion, post-disturbance carbon dynamics and recovery time varied with intensity and type of treatments, and thus management silvicultural practices can be selected to minimize carbon losses and alteration of natural processes on ecosystems while increasing ecosystem carbon uptake, resistance, and resilience to high severity fire and future climate related stresses.

VI. Relationship to Other Recent Findings and Ongoing Work in This Topic

Research by Matt Hurteau and Malcolm North continues in this area. Both have found similar results in the short term but we are the first to report results from the medium term in terms of time since treatments.

VII. Future Work Needed

The biggest challenge is information from fuel treatments that have been installed a decade or longer. Many studies report on initial results (1-2 years) but far fewer at 10-20 years. Another critical area is the effects of multiple treatments on the same area versus only one treatment. Our results do report on multiple treatments on the same area (2 prescribed fire in the same area).

VIII. Deliverables Cross-Walk

Proposed	Delivered	Status
Field trips	2010, Fire and Fire Surrogate Field Trip, Blodgett Forest Research Station, Georgetown, California. 2011 and 2012. Field trip of California Energy Commission	Completed

	staff.	
Conferences and presentations	2011, Blodgett Forest Research Workshop, University of California, Blodgett Forest Research Station, Georgetown, California 2011, ESPM Lunch Seminar Series, UC Berkeley 2011, Poster title: Effects of Fire and Harvest on Soil Respiration in a mixed-conifer forest, American Geophysical Union Conference, San Francisco, California Webinar given to the California Fire Science Consortium about the results from this study.	Completed
Publications	Peer reviewed journal papers expected. Work from this project contributed to a California Fire Sciences Consortium synthesis document on the effects of fuels treatments on forest carbon.	Completed, see publication list below.

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Picture 1. Prescribed fire following mechanical fuel treatment at UC Blodgett Forest in the Sierra Nevada, California.

Picture 2. Results after 2nd prescribed fire at UC Blodgett Forest in the Sierra Nevada, California.

Picture 3. First entry prescribed fire at UC Blodgett Forest in the Sierra Nevada, California.