

Nutrient emissions from prescribed fire in the Lake Tahoe Basin:
Implications from field and laboratory observations

P. Verburg¹, A. Shackelford¹, L.-W. A. Chen², D. Zhu², R. Susfalk³, B. Fitzgerald³

¹Division of Earth and Ecosystem Sciences

²Division of Atmospheric Sciences

³Division of Hydrologic Sciences

Desert Research Institute, Reno NV

2215 Raggio Parkway

Reno, NV 89512



Abstract

Prescribed fire is a common management practice for reducing excessive forest fuel loading to minimize the risk of wildfire. Prescribed fire may however adversely impact air and water quality by releasing nutrients from soils and vegetation upon combustion. The quantity and quality of nutrients released is dependent on fuel characteristics and environmental conditions. Fuel moisture is an important variable that can impact nutrient release since it affects combustion conditions. This study assesses the carbon (C) and nitrogen (N) release as affected by fuel moisture during a prescribed fire near Incline Village (NV) following mechanical thinning. The field component of this study involved a pre- and post-fire fuel inventory to estimate C and N losses under fall fuel moisture conditions. The laboratory component of the study further investigated effects of moisture on nutrient release and speciation. The laboratory study focused on the main gaseous and particulate C and N species that can affect air quality. Different fuel types were wetted followed by combustion in a custom designed combustion chamber that allowed for direct measurement of amounts and composition of nutrients released into the air. The moisture levels for soil, litter and duff ranged from 3 (air-dry) to 25% while moisture contents for vegetative materials ranged from 5 (air-dry) to 85%.

Results from the field study showed that total fuel reductions were close to 90% and C and N losses closely followed patterns in fuel mass reductions. Soil extractable ammonium (NH_4^+) increased immediately following fire, but we no clear trends were observed for extractable nitrate (NO_3^-). The laboratory combustion experiment showed that increasing fuel moisture caused increases in total particulates, including $\text{PM}_{2.5}$ and C and N species, and gaseous ammonia (NH_3) emissions for several fuel types. Nutrient emission factors were highest for litter and leaves. We calculated atmospheric emissions for a hypothetical moist burn combining

field and laboratory data. These calculations showed that under moist conditions particulate, and NH_3 , emissions per unit fuel consumed will increase but this may be partially offset by a lower amount of fuels consumed compared to dry conditions. When comparing our data with model predictions by CONSUME 3.0, our data suggested a higher impact of moisture on particulate emissions compared to CONSUME. Some of the discrepancies between our calculations and CONSUME may be related to the fact that our calculations were solely based on our laboratory emission factors whereas CONSUME takes into account weather and combustion condition (i.e. smoldering vs. flaming). However, CONSUME does not consider separate emission factors for the various fuel types and does not include N emissions.

The results from our study have potentially important management implications. Conducting fuel treatments when fuel moisture is low will likely maximize fuel consumption while minimizing air quality impacts. However, dry burns not only increase fuel consumption which will reduce C sequestration, they also favor conversion of fuels to CO_2 which is an important greenhouse gas. Although we did not specifically test this in our study, broadcast burns may lower C sequestration compared to for instance pile burns since broadcast burns can cause significant consumption of litter and duff compared to piles that have a much smaller footprint. As a result, while more C is being released under dry conditions, air quality impacts may be lower under these conditions suggesting a trade-off between short-term C sequestration and air quality management objectives. In addition, high consumption of litter and duff may negatively impact short and long-term nutrient availability by removing a pool of easily mineralizable N. Our emission factors can help to better inform existing models that estimate fuel reduction and emissions following prescribed fire such as CONSUME to allow for more accurate predictions of C and N emissions from prescribed fire. The fuel inventories prepared in

this study can be added to existing Fuel Characteristic Classification System fuelbeds and thus provide additional information on the variability in fuel loads in the Lake Tahoe basin.

Currently, limited data is available on spatial and seasonal variability in fuel moisture creating uncertainty in model parameterization to assess emission estimates. In addition, long-term impacts of prescribed fire on C sequestration and N availability are unclear as they likely depend on vegetation response.

Funding for this research was provided by the Bureau of Land Management through the sale of public lands as authorized by the Southern Nevada Public Land Management Act and Applied Research Initiative funds provided by the State of Nevada as administered through the Desert Research Institute.

1. Introduction

Fire is an integral part of forest and range ecosystems in the western US. For over a century, fire has been suppressed in an effort to protect livestock, timber, homes, and property. In the Lake Tahoe Basin, fire suppression has resulted in an increased presence of high-density, small diameter trees that are more susceptible to catastrophic wildfires. Especially in late successional forests fire suppression has led to rapid fuel accumulations (Kilgore and Heinselman 1990), leading to an elevated risk of wildfire (Page-Dumroese and Jurgensen 2006). In addition, these forests are less drought tolerant, have higher rates of bark beetle infestations, increased tree mortality, lower biological diversity, and a higher incidence of invasive species (Richter and Ralston 1982; Kaye et al. 1999; Harrod and Reichard 2002; Caldwell et al. 2002). Prescribed fire, often following mechanical thinning is a commonly used management practice in the Lake Tahoe Basin to reduce fuel loads. Currently used prescribed fire techniques include pile burning, and underburning to remove harvest residues following mechanical thinning. In

addition, broadcast burning can be conducted to remove understory fuel accumulation several years after initial thinning. With the use of prescribed fire and other fuel reduction treatments increasing, information on the risks, effectiveness, and environmental impacts of these treatments becomes increasingly important. Carbon (C) and nitrogen (N) emissions are of concern in prescribed fire management primarily due to their potential to impact air and water quality. Nitrogen can cause eutrophication of water bodies, while C emissions can impact accumulation of greenhouse gases and air quality through production of small particles that can impact human respiratory activity. Both elements have low volatilization temperatures and thus are very mobile during and following fire (Baird et al. 1999).

Lake Tahoe is especially sensitive to nutrient inputs environment due to its large lake surface (~500km²) to watershed (~800km²) ratio (Murphy 2006; Boardman 1959). This ratio and the nutrient-poor granitic soils in most of the surrounding watershed are responsible for the exceptional clarity of this ultra-oligotrophic lake (Gertler et al. 2006). Loss of lake clarity has been found to be inversely proportional to primary productivity ($r^2 = 0.74$). Total C fixation (or growth) by lake phytoplankton is at least partly correlated to nitrate (NO₃⁻) content ($r^2 = 0.266$, $p < 0.05$; Goldman 1988). Atmospheric deposition of total dissolved inorganic N has been found to be 19 times that of watershed stream loading (Jassby et al. 1994). The primary sources of atmospheric N within the Tahoe Basin are emissions from fire and automobiles (Caldwell et al. 2002). Out of basin sources of pollution originating from the Central Valley and the San Francisco Bay (Jassby et al. 1994; Gertler et al., 2006) also contribute to N loadings yet these are considered to be minor contributors compared to in-basin sources (Gertler et al. 2006). Given its potential impact on N loadings, it is critical to better understand the controls of N emissions from prescribed fire in the Lake Tahoe Basin.

In this study we focus on the effects of fuel moisture on nutrient emissions. Fuel moisture can vary depending on burn season with fuels typically being wetter during spring than during fall. Increasing fuel moisture is known to affect the type of combustion (flaming or smoldering) and the duration of heating (Hungerford et al. 1992) with wetter fuels causing more smoldering relative to flaming combustion. Flaming combustion leads more to complete oxidation converting C and N to CO₂ and NO_x, respectively, than smoldering combustion (Koppmann et al. 2005; Chen et al. 2007). Particles are generated from both flaming and smoldering phases but differ in size, morphology, and optical properties (Reid et al. 2005a; 2005b; Chen et al. 2006; Chakrabarty et al. 2006; McMeeking et al. 2009). Typically, high temperature flames produce black C soot agglomerates that absorb light, while lower temperature, smoldering burns yield particles that are whiter and more spherical in shape.

In addition to influencing atmospheric nutrient emissions, fuel moisture content can affect soil nutrient mobilization by affecting the soil temperature reached during fire (Aston and Gill 1976, Boring et al. 2004). Increased fuel moisture leads to lower maximum temperatures, fire intensity, fuel combustion and fire spread (Gillon et al. 1995, DeBano et al. 1998, Boring et al. 2004, Glass et al. 2008). Although effects of prescribed fire on soils are typically less severe than wildfires because of lower temperatures achieved, several studies show that prescribed fire can cause an increase in NH₄⁺ immediately following a fire due to a combination of pyrolytic SOM decomposition and microbial N mineralization (Johnson et al. 1998, Neary et al. 1999). Subsequently, the increased NH₄⁺ levels can stimulate nitrification resulting in increased NO₃⁻ concentrations (Murphy et al. 2006; DeBano et al 1998).

The main objective of this study was to measure both atmospheric and soil C and N losses from prescription fire as affected by fuel moisture content. The first component involved

a field study assessing fuel inventories including biomass and nutrient content prior to and after a broadcast fall burn. We also assessed potential impacts of the prescribed fire on streamwater quality by measuring streamwater sediment and nutrient concentrations in a stream adjacent to the area that was burned as part of this study. The second component consisted of a laboratory study addressing the effects of fuel moisture on fuel consumption and nutrient emissions. Laboratory experiments are useful for controlling and investigating individual parameters affecting emissions from biomass burning. The Fire Lab at Missoula Experiments (FLAME I and II) conducted in 2006 and 2007 investigated a wide range of biomass types and loads in a laboratory combustion setup (McMeeking et al. 2009). Fuels used during the FLAME study were either fresh or dried sufficiently to readily ignite. A systematic investigation of the fuel moisture effect on biomass burning was not part of FLAME I and II and is also rare in the literature. In this study, laboratory-controlled burning experiments were conducted with fuels prepared at different moisture levels to represent dry fall and moist spring conditions. This report documents changes in emission factors of C and N species as a function of fuel moisture content. Based on continuous and time-integrated measurements, it also examines the emission mechanisms that are essential for developing process-based emission models. Finally, the field and laboratory data were combined to estimate area-based atmospheric nutrient losses for a variety of fuel types. These estimates were then compared to estimates using the CONSUME 3.0 model (Prichard et al., 2005).

2. Methods

2.1. Field Study

We collected pre-burn fuel data in seven areas on the northwest side of Lake Tahoe designated for prescribed fire. However, only one of these sites was actually burned so we were only able to collect before and after burn data for one broadcast burn. The field site was located near Incline Village, NV. The plot that was burned (referred to as burn plot) is southwest-facing and located just to the west of Diamond Peak Ski Resort at 39° 14'N 119° 55'W, while the control plot is south-facing and located on the northwestern end of Incline Village at 39° 15'N 119° 57'W (Fig. 1). Initially, the plot scheduled to be burned was adjacent to the control plot. However, burning of the Diamond Peak plot had a higher priority and as a result our initial burn plot did not get burned. Average elevation of the burn plot is 2100 m while elevation of the control plot is 2056 m. Overstory vegetation in both plots is dominated by Jeffrey pine while understory vegetation is composed of green-leaf manzanita (*Arctostaphylos patula*), bitterbrush (*Purshia tridentata*), squaw carpet (*Ceanothus prostratus*), bush chinquapin (*Chrysolepis sempervirens*), and huckleberry oak (*Quercus vaccinifolia*). The climate in the area is characterized by warm dry summers and cold winters; most precipitation mostly as winter snow with infrequent summer storms. The average temperature of each of the four warmest months is greater than 10°C, and the average temperature of the coldest months is less than 3°C. The soils in the burn plot belong to the Cassenai series (mixed, frigid Dystric Xeropsamments), gravelly loamy coarse sand, 30 to 50% slope, and very stony. Soils in the control plot belong to the Jorge series (loamy-skeletal, isotic, frigid Andic Haploxeralfs) very cobbly fine sandy loam, 30 to 50% slopes (Soil Survey Staff, 2009).

Pre-burn sampling took place between 09/10/08 and 09/17/08, and post-burn sampling took place from 10/21/08 through 10/29/08. The burn was conducted by personnel at the North Lake Tahoe Fire Protection District on 10/17/08. Five precipitation events occurred in October

between pre-fire sampling and the burn (10/3; 1.19 cm, 10/4; 0.03 cm, 10/10; 0.0025 cm, 10/11; 0.046 cm and 10/12; 0.025 cm for a total of 1.98 cm; Western Regional Climate Center). The treatment plot was hand-thinned prior to ignition that involved felling some of the small diameter trees, and trees that were already dead. No fuels were mechanically removed from the site.

Both the burn plot (6.45 ha) and the control plot (6.20 ha) had twelve sample points each for pre-burn sampling. Twelve different points were established for post-burn analysis. Sampling points were randomly chosen using Hawth's Tools in ArcGIS (Beyer, 1994). The points were required to be at least 30 m apart, and at least 30 m from plot boundaries to minimize effects of plot boundaries and potential interference between sampling points as a result of establishment of transects (Fig. 2). Using the WGS 1984 projection, points were loaded onto Garmin GPSmap 60CSx units and located in the field with an accuracy of 10 m. The points were assigned a random compass direction and a 15.24 m (50 ft) sampling plane was generated following the Brown (1974) protocol used for the downed fuel inventories. After the burn, post fire sampling planes were located parallel to and 10 m away from pre-fire sampling planes.

Using the sampling planes described above, the planar-intersect technique as described in the "Handbook for Inventorying Downed Woody Debris" was implemented for litter, duff, and 1, 10, 100, and 1000 hour downed and dead fuels (Brown 1974). This methodology is widely used in research and also by the USDA Forest Service to conduct pre-burn fuels inventory for prescribed fires. The fuel classes are based on the average diameter of downed materials. One-hour corresponds with leaves, litter, and stick debris smaller than 0.64 cm (0.25"). Ten-hour fuels correspond with debris between 0.64 and 2.54 cm (0.25"- 1"), 100 hour fuels are between 2.54 and 7.62 cm (1 - 3") diameter, and 1000 hour fuels are 7.62 cm (3") and greater in diameter. Additionally, the forest floor was sampled to determine the density of litter and duff,

and to calculate duff and litter mass both before and after the burn. For this study “litter” includes the Oi horizon, containing recognizable only slightly decomposed plant parts, and “duff” includes the Oe and Oa horizons, consisting of intermediately decomposed finely fragmented residues as well as highly decomposed amorphous residues that do not contain recognizable tissue structures (Brady and Weil 2002). Litter and duff were sampled by collecting materials within a 25 cm diameter PVC ring. Depth of the layer was registered allowing for bulk density calculations. Mineral surface soil was collected from the A horizon between 0-5 cm depth.

Living shrub biomass was measured within a 1-m² PVC frame established at the end of each sampling plane. Any shrub more than 50% contained by the frame was tallied, measured, and sampled by destructive or non-destructive techniques depending on the shrub species. Allometric equations have been developed for manzanita, chinquapin, and bitterbrush, and for these species biomass was determined non-destructively (Susfalk 2008). Four bole measurements were recorded and averaged for use in the allometric equations. No allometric equations had been developed for squaw carpet and huckleberry oak and biomass was determined by destructive sampling. When these species were encountered in the vegetation plots, they were removed and dried at the lab to determine biomass of leaves and stems in accordance with the 1, 10, 100, and 1000 hour fuel classifications described above. Additional samples were collected for all standing shrub species to determine moisture content, total C and N concentrations, and for use in the laboratory combustion studies.

At the time of collection, samples were placed in ziplock bags and stored on ice before being brought back to the lab where they were refrigerated to prevent evaporation. The moisture content of all fuels was determined gravimetrically. Litter, duff, and vegetation were dried at

70°C for 24 hours, but the 100, and 1000 hour fuels remained in the oven for 48 hours until constant weight was reached. Soils were dried at 105°C for 24hr.

Extractable NH_4^+ and NO_3^- concentrations were determined both before and after the fire. Five grams of litter, duff, and soil samples were extracted using 50 mL 2M KCl solution following a 12-16 hour soak period. Prior to the extractions soil was sieved to 2 mm. Litter and duff were unprocessed to better represent actual field conditions but an effort was made to exclude mineral content by handpicking. Samples were extracted using a SampleTek Vacuum Extractor. Extractable NH_4^+ and NO_3^- were determined at the Desert Research Institute using a Lachat FIA 8000+ auto analyzer. Total C and N were determined for litter, duff, soil, downed woody debris of four size classes, and for leaves and stems of up to three size classes for each species of standing shrub. All samples were ground at the University of Nevada Reno to 1.0 mm. Total C and N concentrations were measured at the Soil Forage and Water Analytical Laboratory at Oklahoma State University using a LECO TruSpec CN analyzer.

In the field, we attempted to quantify the maximum burn temperature to allow for correlations between temperature and N release. Aluminum sheeting was dotted with twelve heat sensitive paints (Tempilaq) that permanently change color once a given temperature is reached. Temperatures range for the paint was 70°C to 1200°C. One sheet was placed at the ground surface at the beginning and end of each sampling transect for a total of 24 sheets per plot. Although this method has been used successfully in some studies (Hobbs et al. 1984, Graham 2005), we were not able to interpret the results consistently since the sheets were charred or otherwise unreadable similar to the experiences by Caldwell et al. (2002).

A water quality monitoring site was installed on First Creek at Dale Drive (39.255433°, -119.986604°, WGS84). The site was equipped with an in-stream turbidimeter (DTS-12, FTS

Inc, Victoria, BC, Canada), conductance and water temperature sensor (Campbell Scientific, Logan, UT), and pressure transducer (KPSI, Hampton, VA) to monitor stage. A datalogger (Campbell Scientific) collected data from these sensors every 10 minutes. An automated vacuum sampler collected discrete water samples when triggered by changes in turbidity using a modified version of the Turbidity Threshold Program (Lewis, 1996). Chemistry analyses were conducted at DRI's Water Chemistry Laboratory. Multiple samples collected by the autosampler on November 1, 2008 were composited into a flow-weighted event mean concentration (EMC) sample based on instantaneous flow measurements. The number of samples collected at this site was limited due to the lack of observed elevated turbidity and water chemistry concentrations.

2.2. Laboratory Study

Effects of fuel moisture on atmospheric emissions from litter, duff, soil, and leaves and stems of standing vegetation samples were determined by conducting laboratory burns. The litter and duff samples were created by homogenizing all pre-burn litter and duff samples from the field. Leaf and stem samples were created through the homogenization of materials collected from the burn plot. Separate burns were conducted for forest floor litter, duff, and leaves and stems of the three shrub species encountered in the burn plot. To prepare fuels for desired/different moisture contents, individual litter, duff, and soil samples from different locations were first homogenized and air dried for a week under relative humidity of 30%. Calculated amounts of water were added to these samples to achieve fuels with moisture content of 10 or 20% of dry mass. Shrub leaf and stem samples were also composited, and either air dried or soaked in water for 24 to 96 hours. The fuel moisture content, determined from small fractions of samples, increased over time and gradually reached saturation (Fig. 3). Dry, 24-hr

soaked, and 96-hr soaked biomass were used for the burning experiments. Table 1 shows the fuels and moisture levels examined in this study. The two wet moisture levels (II & III) simulated natural fuels of relatively high moisture contents. We did not measure fuel content in the field throughout the year and we were not able to find seasonal fuel moisture data for the Lake Tahoe basin. Burgan (1979) lists that live fuel moisture can range from 30 to >50% moisture depending on plant part and species so our moisture contents fell within that range. Two replicate burns were conducted for each fuel-moisture combination.

Figure 4 shows the combustion facility and measurement suite. Fuels were loaded on a uniformly-heated hotplate ($25 \times 25 \text{ cm}^2$) with adjustable temperatures up to 500°C . Fuels were weighed before and after the experiment. Fuel loads were 23 – 69 g per burn. The hotplate kept fuels warm throughout the experiment to simulate large scale burns where environmental temperatures could be much higher than those in the laboratory without a heater. The hotplate was located in a 60-cm diameter fire pit. Horizontal air flow was kept calm, and most of the smoke was vented through the chimney with an exhaust fan. Fuels were placed on the hotplate (maintained at 450°C) for ~30 seconds before starting to be ignited by an electric hot air gun (Loof Lighter). It often required continued application of hot air ($\sim 600^\circ\text{C}$) for an extensive period of time to ignite and sustain combustion for high moisture fuels. The use of hot air gun minimized interferences from igniter (e.g., propane torch) emissions.

The smoke sampling probe consisted of 5.08 cm diameter conductive tubing stretching from the top of chimney, about 2 m above the fuels, to a mixing plenum (Fig. 4). Plumes generally reached the ambient temperature in the plenum. Through two size-cut cyclones, $\text{PM}_{2.5}$ (particles with aerodynamic diameter $< 2.5 \mu\text{m}$) were sampled onto 1) Teflon filter (for $\text{PM}_{2.5}$ mass and elements) with calcium carbonate-impregnated cellulose backup filter (for sulfur

dioxide [SO₂]) and 2) quartz-fiber filter (for organic C [OC] and elemental C [EC] by the IMPROVE_A protocol (Chow et al. 2007a) and water-soluble ions) with citric acid-impregnated cellulose backup filter (for NH₃). The analytical methods for filter samples are the same as described in Chow (1995) and Chow et al. (2004). Least quantifiable limits (LQLs) were established with dynamic blanks through the same sampling channels but in the absence of combustion. Measurement precision is <10% for concentrations greater than 10 times the LQLs. Additionally, portions of quartz-fiber filters were submitted to the CHNS-O analyzer for quantifying total C (TC), N (TN), and hydrogen (TH) in PM_{2.5}. TC by CHNS-O analysis agrees with OC + EC by IMPROVE_A within ±8%.

An Electric Low Pressure Impactor (ELPI) was used to measure particle size and number concentrations in real time. The ELPI has 12 stages covering an aerodynamic size range of 10 nm to 10 μm. Size retrieval followed the algorithm developed by Marjamäki et al. (2005), which generally give accurate size distributions for particles smaller than 3 μm (Pagels et al. 2005). Through Teflon tubing from the plenum and downstream of a Teflon particle filter (Fig. 4), several gaseous species were continuously monitored by analyzers including a nondispersive infrared CO₂/H₂O analyzer (LI-840, LiCor Biosciences, Lincoln, NE), an extractive Fourier Transform Infrared (FTIR) Spectrometer (Midac, Costa Mesa, CA, for CO), and research-grade NO_x and NH₃ analyzers (TECO 42 and 17C, Thermo Scientific, Waltham, MA). These analyzers were calibrated against standards traceable to the National Institute of Standard and Technology before and after the experiments. The detection ranges (10-s average) were 0.001–10, 0.1–3000, and 1–10000 ppmv for the TECO, Li-Cor, and Midac analyzers, respectively. The precision of gas measurements are estimated to be <5% if they are within the instrumental detection range. All our experiments met the condition.

Since the experimental setup sampled only a small portion of smoke plumes, production efficiencies of gaseous and particulate species (per unit fuel consumption) were calculated using a C balance approach (Andreae and Merlet, 2001). The emission factor (EF) is defined as the ratio in concentration of a measured species over total C (GTC) released from a burn, which may be reported in gram per kilogram of C burned. Alternatively, it can be scaled to gram per kilogram of fuel consumed using the C content of the fuel. GTC includes C in CO₂, CO, and PM_{2.5}. Volatile organic compounds (VOCs) including CH₄ were not measured, though under most circumstances VOCs are minor contributors, compared to CO₂ and CO, in the C budget of emissions (< 5%). Natural variability of EFs, according to replicate burns, usually exceeds the EF uncertainties estimated from the analytical precision.

Particles from each individual burn were collected on a single set of Teflon and quartz-fiber filters that yielded burn-average chemical compositions. For consistency, all continuous measurements were baseline subtracted and integrated over the filter sampling period. NH₃ from the TECO 17C correlated reasonably with those of filters ($r = 0.83$), but the TECO 17C values were lower by an average of 21% and as much as 80%. This most possibly results from the loss of NH₃ in the Teflon sampling tubing and to the pre-analyzer particle filter (e.g., Mukhtar et al. 2003). Emission factor calculations were therefore based on filter measurements which deployed much shorter/larger diameter tubing from the plenum. CO₂, CO, and NO_x adsorption on Teflon surfaces are generally negligible.

2.3. Statistical Methods

To determine the effects of the burn on fuel load, total C, total N, or extractable N (NH₄⁺ and NO₃⁻ from litter, duff, and soil) in the field we conducted two-way Analysis of Variance

(ANOVA) using plot (control vs. burn plot) and time (pre- vs. post-burn) as main factors. Significant interactions between plot and time were interpreted as being caused by the prescribed fire. Following the ANOVA, we conducted unpaired t-tests to detect specific treatment effects for all measured parameters.

For each fuel type we conducted a linear regression between the emission factors for each volatilized compound and moisture content. These regressions were used to calculate emissions from the field fall burn using field moisture data. To estimate emissions for a moist burn, moisture content of all fall season fuels was tripled, representing spring moisture conditions as may occur during a relatively dry year (Burgan 1979, Chandler et al. 1983, Chuvieco et al. 2004). Linear regressions were also conducted to establish relationships between fuel moisture and extractable soil N.

For all statistical analyses, effects or correlations were considered significant when $p < 0.05$. For several of the regression analyses, a notable difference in emission factor was seen between the dry fuels (Moisture Class I) and wet fuels (Moisture Class II and III), without resulting in a significant linear relationship. To avoid overlooking these differences, increases or decreases are reported when regression analysis resulted in a 75% significance or greater ($p < 0.25$). We explored the use of non-linear regression analysis but this analysis showed similar patterns as the linear regression analysis, and was therefore not used. T-tests and linear regressions were performed using Microsoft Excel software, and ANOVA analysis was done using DataDesk 6.0 Software.

3. Results and Discussion

3.1. Field Study

3.1.1. Forest Floor Moisture

Moisture content was very low for organic and surface horizons in both control and burned plots. The ANOVA results reveal that differences between plots (burn plot vs. control plot) significantly affected moisture content of litter ($p < 0.0001$), duff ($p = 0.0025$), and soil ($p = 0.0129$) with moisture content in the control plot being higher than in the burn plot (Table 2; Fig. 5). The moisture content of litter, duff, and mineral soil significantly increased in the control plot due to precipitation events that occurred between the pre and post-fire sampling dates. However, the moisture content in the burn plot litter and duff layers significantly decreased while moisture content of mineral soils in the burn plot did not change between pre- and post-fire sampling dates (Figure 5). This pattern resulted in a significant site*time interaction for litter ($p < 0.001$) and duff ($p = 0.005$), with a trend towards significance for soils ($p = 0.058$) indicating that moisture content was affected by the burn treatment.

3.1.2. Fuel Load

This Lake Tahoe Basin second growth Jeffrey pine forest had a pre-fire fuel load of $162.74 \text{ Mg ha}^{-1}$ in the burn plot, with almost half of that mass attributed to the litter and duff fuel types (81.59 Mg ha^{-1} ; Fig. 6). The overall fuel loss in this plot resulting from fire equaled $146.51 \text{ Mg ha}^{-1}$ (or 88% of the initial fuel load). Fuel loading in the control plot was much lower than the burn plot, and the differences between pre- and post-fire fuel loads were 22.82 Mg ha^{-1} (or a 32.6% difference of the initial fuel load). Before and after fuel inventories averaged 58.63 Mg ha^{-1} with 76% attributable to litter and duff. The differences between control and burned plot

fuel loading may have been caused by differences in site characteristics and sampling error associated with the fuel inventory method. Several sample points were located on rocky outcrops with little or no accumulated fuels, or beneath shrubs where litter and duff depth were thin relative to areas under tree cover. Actual values and percent loss for each of the fuel types are shown in Appendix 1. Appendix 2 provides values for four additional pre-burn fuel load estimates in the Incline Village, NV area. Patterns in C and N closely followed total fuel loads. Carbon loss in the burn plot was 70.1 Mg ha^{-1} or 94.6% (Fig. 7). This is slightly higher than the overall percentage of fuel consumption (88.1%). The change in total N in the burn plot was 0.92 Mg ha^{-1} (89.5%; Fig. 8) which is very close to the percentage of fuel consumption in the burn plot (88%). Differences in the percentage of C volatilized compared to N may reflect the higher volatilization temperature of N compared to C. The significant reduction in ecosystem N pools could potentially impact future ecosystem fertility since especially duff and litter N decreased substantially. Litter and duff represent an N pool that can rapidly decompose and provide N for plant growth. Consequently, long-term productivity could decrease following removal of these labile N pools.

Fuel loads in the burn plot were comparable to fuel loads observed in other studies conducted in the western US. For instance, a forest on the western side of Lake Tahoe Basin, had a fuel load of $141.72 \text{ Mg ha}^{-1}$, with litter and duff attributing 60% (84.70 Mg ha^{-1} ; Stephens et al., 2004). In experimental forests absent of logging activity for the last 70 years, fuel loading in the late fall was 177.0 , and 150.7 Mg ha^{-1} for ponderosa pine, and douglas fir forests in the Northern Sierra Nevada. The total mass of litter and duff fuels was 59% for the Ponderosa pine forest, and 84% for the douglas fir forest (Kauffman and Martin 1989). Old-growth mixed conifer forests in Sequoia National Park had an average fuel load of 191.6 Mg ha^{-1} of litter, duff,

and downed woody debris with just over half of the fuel being comprised of litter and duff (105.7 Mg ha⁻¹; Knapp 2005). Little and Ohmann (1986) determined biomass in western Oregon and western Washington clear-cut areas slated for prescription fire and in their study fuel totals for litter, duff, and downed woody debris ranged from 70.9 to 304.2 Mg ha⁻¹, with an average of 174.8 Mg ha⁻¹. On average, 60% of the mass was attributed to forest floor fuels (105.4 Mg ha⁻¹). Lastly, Page-Dumroese and Jurgensen (2006) conducted similar fuel inventories in different forest stands in the northwestern U.S. that have no evidence of thinning activities or fire history in the last 70 years. Western hemlock forests in Northern Idaho and Montana averaged 221.5 Mg ha⁻¹, ponderosa pine forests in central Idaho averaged 161.0 Mg ha⁻¹, and grand fir forests, also in central Idaho averaged 174.5 Mg ha⁻¹.

3.1.3. Extractable Soil Nitrogen and Streamwater Loading

Similar to other studies we found significant increases in extractable NH₄⁺ concentrations in the litter, duff, and soil layers following the fire in the burn plot (Fig. 9). The MANOVA results showed that time, plot, and their interaction significantly affected the NH₄⁺ concentration in litter, duff, and soils (p<0.05; Table 2). The concentration of NH₄⁺ decreased significantly in control plot litter, but remained unchanged in control plot duff and soil layers (Fig. 9). Although extractable NH₄⁺ concentration increased in the burn plot, the loss of litter and duff biomass lead to a slight decrease in total extractable NH₄⁺ in post fire litter and duff horizons when expressed on an area basis (Fig. 9). Extractable NH₄⁺ concentrations significantly increased in the burn plot mineral soil resulting in an increase in the total amount of extractable NH₄⁺ in burn plot soils assuming no significant change in soil mass. Control plot litter NH₄⁺ significantly decreased

potentially as a result of leaching caused by the rain events that occurred between pre- and post-fire inventories.

Extractable NO_3^- concentrations did not significantly change in the burn plot soils immediately following fire (Fig. 10). In general, NO_3^- concentrations were one to two orders of a magnitude lower than NH_4^+ concentrations. Surprisingly, extractable NO_3^- concentration significantly decreased between pre- and post-fire sampling in the control plot litter layer (Fig. 10) further suggesting that inorganic N leaching occurred. The interaction of site and time was significant in affecting NO_3^- concentration for the litter layer only, suggesting an effect of the fire. The ANOVA analysis found no other significant factors affecting extractable NO_3^- concentration (Table 2). When multiplied by soil mass the biggest increase in NO_3^- mass occurred in the control plot soil (Fig. 10). Although we did not see an immediate increase in soil NO_3^- as a result of the prescribed fire, on the longer term the increased soil NH_4^+ concentration could stimulate nitrification, impacting water quality (Kovacic et al. 1986, Riggen et al. 1994). However, elevated NH_4^+ and subsequent increases in NO_3^- in the surface mineral soil has also resulted in short-term gains in soil productivity (Johnson and Curtis 2001).

Continuous stage, water temperature, conductance, and turbidity data for water year 2008-2009 are presented in Figure 11. Overall, changes in stage in Third Creek were minor, ranging from about 0.05 m (0.16 ft) during base flow to a maximum of about 0.08 m (0.26 ft) during snowmelt. Elevated turbidity was observed during most rain or snowmelt events, however, the duration and magnitude of these were considered to be low relative to other nearby creeks in the North Lake Tahoe Area, such as Rosewood Creek (Fig. 12). Rosewood Creek is small tributary to Third Creek located almost entirely in the urbanized areas of Incline Village. Rosewood Creek is slightly larger than First Creek with an average annual discharge of 0.0085

$\text{m}^3 \text{ s}^{-1}$ (0.3 cfs). However, Rosewood Creek is known for its ability to generate high turbidity values relative to its small flows. Elevated turbidity values observed at First Creek were typically 70 to 90% lower than those observed during the same hydrologic event at Rosewood Creek.

The highest turbidity values of 66 NTU were observed during a thunderstorm on November 11, 2008 (Fig. 13). This event was the first hydrologic event after the North Lake Tahoe Fire Prevention District had thinned and subsequently burned 20 acres on October 10, 2008 along the east side of First Creek, north of Dale Drive (Personal Communication, F. Schafer, NTLFPD, 12/2008). The magnitude and duration of this elevated turbidity event appears consistent with other hydrologic events during water year 2009. The burning treatment did not have any short-term impact on suspended sediment transport within First Creek.

The Nevada Division of Environmental Protection (NDEP) has collected water quality samples at First Creek as part of their ambient monitoring program. Table 3 presents the two water chemistry samples collected as part of this study, the current water quality standard that NDEP lists for First Creek, and summary statistics based on NDEP's grab samples. During a rain event on 11/1/08, observed TN concentrations exceeded water quality standards and were higher than all historical observations collected by NDEP. This was likely due to NDEP's dataset being skewed towards baseflow conditions with little representation of samples collected during hydrologic events.

In summary, turbidity, stage, and electrical conductance were monitored in First Creek above Dale Drive during water year 2009. This water year included few rain events and below average to average snowpack. In-stream turbidity during a rain event one-month after a prescribed burn was consistent with an event prior to burning and with turbidity levels observed

during other hydrologic events. The First Creek watershed is dominated primarily by undeveloped land uses and secondarily by single-family residential land-uses. The low in-stream turbidity values observed were consistent with these land uses, compared to the nearby, urbanized Rosewood Creek. Low ambient concentrations suggest that this watershed holds excellent future potential to assess the impacts of fuels treatments, land-use changes, or increasing urban pressures on water quality.

Although in general fire causes NH_4^+ and NO_3^- concentrations to increase, responses to prescribed fire have been variable. For instance, at South Lake Tahoe Johnson et al. (2004), Murphy et al (2006b) and Miller et al (2006) found a clear increase in labile nutrients following the Gondola wildfire but little evidence of increased N availability following prescribed fire. In contrast, Chorover et al. (1994) found an increase in soil solution and streamwater NH_4^+ and NO_3^- concentrations following prescribed fire at a western Sierra Nevada site. Three years following the burn, streamwater NH_4^+ declined below pre-burn baselines, while NO_3^- remained above. In addition, Moghaddas and Stephens (2007) found a large increase in both NH_4^+ and NO_3^- in the mineral soil after prescribed fire on the west side of the Sierra Nevada that persisted up to 8 months after the fire. Stephens et al. (2004) also found clear increases in soil NH_4^+ , and NO_3^- content 3 weeks after a prescribed fire at Sugar Pine Point State Park located on the western side of Lake Tahoe. However, these increases did not result in significant changes in streamwater N. Moghaddas and Stephens (2007) ascribed large increases in soil NH_4^+ to the accumulation of fuels in the forest floor in response to several decades of fire suppression causing fire severity to be higher than expected. Similarly, large amounts of fuels present in slash piles can result in localized high-severity fires that can significantly impact soil chemical properties (e.g. Korb et al. 2004; Jonsson and Nihlgard 2004). Upon release, NH_4^+ and especially

ortho-P may be directly adsorbed to soil particles (albeit by different mechanisms) especially in volcanic soils having amorphous, highly reactive clay minerals whereas NO_3^- can readily leach (e.g. Belillas and Roda 1993). In addition, persistence of increased nutrient levels may depend on vegetation responses. If vigorous regrowth occurs, it is very likely that any available nutrients will be taken up, causing soil nutrient levels to decrease. In many areas of the Sierra Nevada, N fixing species can occupy burned areas, especially after wildfires. As a result, soil N levels may increase following fires due to increased inputs of fixed N (e.g. Johnson 1995). Our results fit within the range of responses observed in other studies but despite the increase in soil NH_4^+ concentrations watershed impacts of changes in soil nutrients appeared to have been small as evidenced by water quality data.

3.2. Laboratory Combustions

3.2.1. Fuel Consumption

The moisture content of fuels used in our laboratory burns is shown in Table 1. Dry and wet biomass fuels showed distinct fire behaviors. Dry fuels were ignited easily by the hot air and flames quickly spread over the fuel. Residues after the burn were small except the duff composite, for which only half of the dry mass was combusted. The duff sample likely contained substantial mineral material, as its C content (32%) was lower than the nominal value (i.e., 45 – 50%) for biomass (McMeeking et al. 2009). It took much longer for wet fuels to ignite when they were heated by the same hot air. Smoke was visible before ignition. Flaming periods were relatively short, followed by prolonged smoldering combustion. Wet fuels left more and variable amounts of unburnable residues than dry fuels. For soil, flames were absent throughout the experiments, and only 9 – 10% of the dry mass was consumed, regardless of moisture

content. In contrast to our initial hypothesis, in the laboratory burn increasing moisture content did not always lead to decreases in fuel consumption but for most fuel types, fuel consumption was lowest at the highest moisture level compared to the lowest moisture level (Table 4). Still, for litter and shrub stems fuel consumption was lowest at the intermediate moisture level. Gillon et al. (1995) found significant decreases in fuel consumption with increasing moisture level in laboratory burns. The laboratory conditions, and the method of ignition used in the study by Gillon et al. (1995) were more representative of prescribed fire under field conditions. Also, other studies comparing spring to fall burns reveal far greater fuel consumption for drier fall burns (Gill et al. 1978, Kauffman and Martin 1989, Boring et al. 2004, Knapp et al. 2005). The discrepancy between these field studies and our laboratory studies may be due to our laboratory burn environment, primarily related to the duration of heating. In the lab, the heat duration may have been long enough to evaporate moisture thus allowing for fuel consumption to continue. Under field conditions the flaming front generally passes after dry fuels are consumed and as fuels are increasingly drier, consumption increases (Knapp et al. 2005) and the flaming front becomes more intense (Alexander 1982). A better simulation of a woodland environment perhaps using mineral soil as a base and an ignition method similar to common prescribed fire methodology, such as fuel based strip ignition on one side of the sample burn material would allow for an improved analysis of the effect of moisture content on fuel consumption.

3.2.2. Time-Integrated Carbon Measurement

Combustion efficiency (CE), defined as the fraction of C emission in the form of CO₂, best indicates the relative importance of flaming (high CE) and smoldering (low CE) phases (Sinha et al. 2004; Janhäll et al. 2009). CE was > 0.9 for all dry fuels (except soil) and was < 0.9 for all wet fuels in this study (Table 5). In general, wet duff/leaves produce lower CEs than wet

litter/stems, though there are no apparent differences in CEs between the two wet moisture levels (II and III; Table 5, 6). Carbon monoxide is usually found to carry most of the non-CO₂ C from biomass burning (Yokelson et al. 1996; Andreae and Merlet 2001; McMeeking et al. 2009). For fuels with high moisture contents, however, particulate C emissions can approach or exceed the amount of released as CO emissions. The sum of OC and EC is shown (moisture level III, Table 5) to exceed 2 – 4 times C in CO for three stem burns and be 57 – 113% of C in CO for the rest of burns except soil.

Thermal/optical analysis indicates that most of the particulate C is in an organic form. The increasing emission factors for PM_{2.5} and organic C with fuel moisture content are clear for leaf combustion (and to a less degree for litter, duff, and stem; Fig. 14). This likely reflects an enhanced primary process converting plant material to organic C, since the experimental setup does not allow plumes to age extensively for secondary organic C formation. Soil contains little biomass, and the influence of soil moisture on OC emission is limited. Although only leaf and duff burns show EC EFs to decrease with increasing fuel moisture content, EC fraction in TC ranges from 0.01 to 0.68 with all the highest values found from burning dry biomass. This is consistent with EC being generated from flaming combustion that intensifies with dry fuels. From such burns, Chakrabarty et al. (2006) identified soot-based particles similar to those originating from diesel engines.

The TH/TC ratio tends to increase with the OC fraction in TC (Fig. 15), which corroborates the dominance of hydrocarbon and elemental C in the (operationally-defined) OC and EC, respectively. For 17 biomass burns in which OC/TC > 0.97 (i.e., low to no EC), the TH/TC ratio averages at 0.13 ± 0.02 . Considering H/C ratios of 0.167, 0.139, and 0.108 for carbohydrates [(CH₂O)_n], cellulose [(C₆H₁₀O₅)_n], and lignin [(C₁₀H₁₃O₃)_n], respectively, the

OC may just result from decomposition of cellulose, which is the most abundant organic polymer in biomass (Gani and Naruse 2007).

Particles from wet fuel combustion often appear yellowish to brownish on filters and in water extracts, in contrast to a black appearance from dry fuels (Andreae and Gelencser 2006). A two-wavelength (370 and 880 nm) optical transmissometer (OT21, Magee Scientific Co, Berkeley, CA) was used to examine Teflon filters for wavelength-dependent absorption of particles. The absorption exponent (AE) was calculated by

$$AE = -\frac{\ln(ATN_{370}) - \ln(ATN_{880})}{\ln(370) - \ln(880)} \quad (1)$$

where ATN is the filter attenuation used as a surrogate for light absorption (Moosmüller et al., 2009). The increasing AE towards higher OC/TC ratio (Fig. 15) shows that OC absorption is more skewed toward shorter wavelengths than EC absorption, a principle characteristic of brown C (BrC). Ultraviolet (UV) absorption by such BrC can be comparable to that by black C (Kirchstetter et al., 2004). Though levoglucosan ($C_6H_{10}O_5$) is regarded as the main product from cellulose decomposition during biomass burning and has been used as a marker for apportioning biomass burning contributions to ambient $PM_{2.5}$ (Simoneit et al. 1999; Rinehart et al. 2006; Chow et al. 2007b), it is colorless. Formation of BrC from biomass burning must involve not only thermal decomposition but also oxidation and/or other reactions in the plumes.

3.2.3. Time-Integrated Nitrogen Measurement

Lobert et al. (1990) suggested that NO_x accounts for most of fuel N detected in biomass burning plumes, followed by NH_3 , hydrogen cyanide (HCN), N_2O , and nitriles. This is the case for some dry biomass (i.e., litter and stems) burned in this study (Table 5). In other cases NH_3

carries the most N, while N_2O is below the detection limit and HCN not quantified. Our study found that wet fuel combustion caused substantial particulate N emissions, which was not previously addressed in Lobert et al. (1990). Only a small fraction of the particulate N can be attributable to water soluble NO_3^- and NH_4^+ . Other particulate N (OPN, particulate N less those in NO_3^- and NH_4^+) EFs increase with fuel moisture content (Fig. 14; Table 6) and are particularly high (> 3 g/kgC and up to 75% of N in NH_3) when burning wet duff and leaves (see moisture level II and III in Table 1). The fact that OPN and, to a lesser extent, NH_3 EFs are inversely correlated with CE for all fuel types, including soil, supports that they are primarily produced during smoldering combustion. This is not the case for NO_x .

It is possible that some N-containing gases, such as NH_3 , mixed with $PM_{2.5}$ in plumes and/or on filters and was measured as OPN. NH_3 adsorbed on quartz-fiber filters should be detected as NH_4^+ under the current procedure (i.e., water extraction/automated colorimetry) while adsorption of other gases seems minor compared to the amounts of OPN observed. In addition, OPN is highly correlated with OC ($r = 0.93$; slope = 0.022) for all biomass burns. These evidences suggest that most OPN forms, along with OC, from the decomposition or pyrolysis of plant material and is in the form of organic N.

The N balance is evaluated by scaling N/GTC ratios in the combustion products to those in the fuels (Figure 16). Better closures for wet fuels result, for the most part, from higher NH_3 and OPN emissions. Nearly 100% of N was recovered from the plumes of burning wet litter composite and bitterbrush leaves. In other burns, however, substantial N is missing. Besides HCN and nitriles, Lobert et al. (1990) suggested that up to 50% of biomass N could end up becoming molecular N (N_2), which cannot be detected in open fires due to the large N_2 background concentration in ambient air. Combustion experiments conducted in a closed

chamber supplied with helium/oxygen air showed dominant N₂ emissions from biomass burning, particularly during flaming combustion (Kuhlbusch et al. 1991). The mechanism of N₂ formation is unclear, but such a process is clearly weakened or prohibited under the relatively low-temperature smoldering phase, leaving N available for other species.

3.2.4. Time-Resolved Measurement

Continuous data provide additional insights into the process of biomass combustion. Temporally-resolved dry- and wet-fuel combustion emissions are compared in Figure 17 using manzanita leaves as an example. NH₃ concentrations in Figure 17 (from TECO 17C) may be biased low as the averaged NH₃ for these two burns are ~10% lower than those measured on filters. The time-resolved measurements are only discussed qualitatively.

Flaming phase for wet manzanita leaves, consistent with the period with highly elevated CO₂ concentrations, was rather brief (~1 min) relative to that of dry manzanita leaves (3 – 4 min), though it was in this period that most of the CO₂ and NO_x were emitted. High temperatures in the flames provide sufficient thermal energy to break up plant organic matter into small fragments, producing CH₄, VOCs, CO, NH₃, etc., which are subsequently oxidized to CO₂ and NO_x. Dry fuels generate more intense flames, and therefore CO₂ and NO_x dominate the C and N species (except N₂) in the smoke. From the end of flaming phase through the smoldering phase, thermal energy decreases and higher fractions of CO, VOCs, and NH₃ are commonly observed (e.g., Figure 17a and Chen et al. [2007]). For wet fuels, NH₃ could be the dominant N species throughout the burn (e.g., Figure 17b).

To investigate the emission of particle mass and the dominant size fraction, real-time ELPI particle size and number measurements were converted to particle volume concentration (μm³ cm⁻³) assuming spherical particles. The burn-averaged particle volume concentration

correlates well with PM_{2.5} mass measured on Teflon filters ($r = 0.90$, across all the samples) with a mass/volume slope of $1.01 \pm 0.08 \text{ g cm}^{-3}$.

For the two manzanita leaf burns in Figure 17, time-resolved mean particle volume (V_p in μm^3) was calculated from the ratio of particle volume and number concentrations. From V_p the particle volume mean (aerodynamic) diameter (D_v) was determined:

$$D_v = \left(\frac{6V_p}{\pi} \right)^{1/3} \quad (2)$$

Soot particles that mostly consist of EC are expected to form in the flames, and indeed peaks of particle volume concentration track those of CO₂ well in Figure 17a. D_v at the peaks is estimated to be $0.34 - 0.38 \mu\text{m}$. Larger D_v up to $0.56 \mu\text{m}$ are observed right before the flames start, which could be related to the initial breakdown of plant organic matter. These large particles, however, appear not to account for substantial PM_{2.5} volume or mass from the dry fuel combustion. Reid et al. (2005) and Guyon et al. (2003) reported typical values of D_v ranging from $0.25 - 0.3 \mu\text{m}$ for fresh smoke, but with some values as high as $0.5 \mu\text{m}$.

Figure 17b shows NH₃ emissions, in the absence of NO_x emission, during a prolonged heating/smoldering period before the wet manzanita leaves were ignited. Unlike Figure 17a, ELPI recorded intense particle emissions during this pre-flame period, and this explains the much higher OC and OPN *EFs* from the wet fuels. D_v that correspond to each of the pre-flame peaks range from $0.56 \mu\text{m}$ to $0.81 \mu\text{m}$, with the largest D_v occurring right before the flame starts. These particles possibly consist of polyaccharides and/or its derivatives from the decomposition of cellulose. There is a smaller ELPI peak corresponding with the major CO₂ hump (flaming phase). Particle size at this peak appears to be smaller, i.e., $D_v = 0.45 \mu\text{m}$, but is more consistent with soot particles observed in Figure 17a.

3.2.5. Soil Extractable Nitrogen

Immediately following the laboratory burns soil extractable NH_4^+ concentrations tended to decrease while NO_3^- concentrations tended to increase with increasing moisture levels (Fig. 18). Patterns were however not significant. The decrease in NH_4^+ concentrations may reflect lower burn temperatures while the increase in NO_3^- may have been due to nitrification. The limited number of replicates caused relationships not to be very strong. Still, patterns were consistent with a laboratory study of Glass et al. (2008) using a temperature-controlled muffle furnace.

3.3. Emission Predictions for Dry and Moist Burns

Using the laboratory burn data and the regression equations generated to determine the influence of moisture content on emissions, we calculated emissions from a relatively dry burn, and a hypothetical moist burn (Figure 19). For the calculation of moist emissions we assumed a similar amount of fuel reduction for the moist and dry burns. Because the fuel reduction under moist conditions is most likely lower than under dry conditions, the projected emissions should be considered as an upper limit. The CO_2 emissions were lower for fuels burned under moist conditions while CO emissions were higher. Still, decreases in CO_2 emissions were greater than the increases in CO emissions for all fuel types (Figures 19a, b) resulting in an overall decrease in gaseous C emissions. For all fuel types, emissions of NH_3 were higher under moist conditions compared to drier conditions (Figures 19c, d). The NO_x emissions were lower for duff and stems under moist conditions while they were similar for other fuel types. The sum of NH_3 and NO_x emissions was higher for moist conditions especially for duff. Total particulate emissions were

much higher with elevated moisture for all fuels, with twice as much particulate emission for the stem fuel type, and six times higher emissions for the litter fuel type under moist conditions (Figures 19e, f). As a result, even if fuel consumption was lower under moist than dry conditions, particulate emissions would still be expected to be higher than under dry conditions. Particulate elemental C emissions were negligible compared to organic C emissions, and were not consistently higher under moist conditions. Particulate organic C emissions were however consistently higher for moist conditions, with particulate emissions for the stem fuel type being twice as high, and emissions from litter being ten times higher under moist than under dry conditions (Figures 19g, h). Particulate organic N emissions were two to three times higher for all fuel types under moist conditions (Figures 19i, j). Particulate mineral N (NH_4^+ and NO_3^-) followed a similar pattern, with higher emissions for all fuel types under elevated moisture conditions. Increases were slightly higher for particulate NH_4^+ , than for particulate NO_3^- for all fuel types except stems, where the opposite was true (Figures 19k, l). Except for CO_2 and CO , emissions of particulate and total N and C showed a dramatic increase under moist compared to dry conditions even when we did not account for potentially smaller fuel consumption.

We compared our emission estimates with those produced by the CONSUME 3.0 model (Prichard et al., 2005). This model simulates fuel consumption and C emissions using routines describing different combustion phases (flaming vs. smoldering) based on fuel and environmental conditions. We used our fuel loadings and moisture conditions as input variables. CONSUME predicted slightly lower fuel consumption under more moist conditions. Since we assumed similar fuel consumption under dry and moist conditions we compared the relative rather than absolute emissions of gaseous vs. particulate C compounds. Overall, CONSUME showed a smaller impact of moisture on the relative contribution of particulate to gaseous

compounds compared to our laboratory studies especially for duff (Table 7). In both moisture scenarios, CONSUME predicted that the combustion phase for duff was dominated by smoldering (70% for both scenarios). For litter and vegetation, CONSUME does not include moisture contents causing dry and moist scenarios to yield the same results. While it may be tempting to directly compare emission estimates, our estimates and those produced by CONSUME were derived in very different ways. CONSUME uses fuel moisture as an input to determine combustion phase, yet it is one variable amongst many variables used to calculate fuel consumption and emissions. In addition, CONSUME assumes one ecosystem emission factor, i.e., it assumes that all fuel classes within a certain ecoregion and cover type have similar emission factors that are different for smoldering vs. flaming combustion. In our study we however showed that not all fuel classes have the same emission factor. The CONSUME model provides much more realistic fire characteristics since it includes many environmental factors known to impact fire behavior and combustion phases. In addition, CONSUME currently does not simulate N emissions. Consequently, our data could potentially be used to further refine emission calculations used in CONSUME by assigning different emission factors for the various fuel classes. Our data would also allow for inclusion of N emissions that can be important to assess nutrient losses from ecosystems and inputs into Lake Tahoe. Without having field emission data it is difficult to determine which approach provides more realistic emission predictions.

4. Conclusions and Recommendations

To study the moisture effect on biomass burning emission we conducted field fuel inventories prior to and after a fall prescribed fire, collected litter, duff, soil, and aboveground

shrub vegetation from an alpine forest, and subjected different fuel types to controlled laboratory combustion experiments at three moisture levels. The fuel loads encountered in the field were consistent with other studies and the prescribed fire resulted in a large fuel reduction of close to 90% of the initial fuel load. The prescribed fire did not affect water quality of a stream adjacent to the burned area, mostly because precipitation following the fire was too low to induce significant streamflow. The laboratory combustion experiments show that increased fuel moisture lowers the overall combustion efficiency, shortens the flaming phase, and prolongs smoldering period. Emission factors for CO, OC, NH₃, and OPN increase with the fuel moisture content; the effect is generally larger for plant leaves and duff materials than for stems, litter, or soil. For high moisture leaves, emitted particulate OC can be as much as 4 times C in CO and OPN can be up to 75% of N in NH₃.

In addition to air quality and nutrient losses, our findings could have implications for ecosystem C storage and greenhouse gas emissions. Fuel consumption is typically smaller under moist conditions limiting C loss. In addition, moist burns favor production of CO over CO₂ limiting contribution to greenhouse gas emissions. Still, this potentially positive impact on C sequestration would have to be weighed against release of other gases and particulates that could have direct human and ecosystem health impacts. Overall, it appears that burning under dry conditions would favor increased fuel consumption, more complete combustion and thus more release of CO₂ into the atmosphere thereby reducing C sequestration compared to moist conditions. When comparing broadcast burning with other types of fuel removal techniques such as pile burning or off-site burning, broadcast burning could potentially lead to higher C losses since our data showed that substantial amounts of litter and duff material were consumed during the burn. The surface soil impact of pile burning is likely to be smaller given the smaller

surface area. However, consumption of the piles may be more complete than would occur under broadcast burning so the consumption of litter/duff may be offset by a smaller consumption of large fuels.

This study confirms that fuel moisture content may be an important factor controlling the C and N partitioning in biomass burning emissions. Since high-moisture fuels lead to substantial NH_3 and OPN emissions that have relatively high deposition velocities, prescribed burning during wet seasons (e.g., spring in Lake Tahoe) would mitigate nutrient loss from long-range transport but increase the potential of nutrient deposition into the lake water. Our fuel inventories could potentially benefit ongoing efforts to characterize fuelbeds within the Lake Tahoe Basin such as the Fuel Characteristic Classification System fuelbeds (e.g. Ottmar et al., 2007). Our data would allow for more refinement of currently available fuel consumption/emission models such as CONSUME by allowing for assignment of separate moisture-dependent EFs for the various fuel types. In addition, these tools could be expanded by incorporating estimates of N emissions from prescribed fires.

5. References

- Alexander, M.E. 1982. Calculating and interpreting forest fire intensities. *Can. J. of Bot.* 60: 349-357.
- Andreae, M.O. and A. Gelencser. 2006. Black carbon or brown carbon? The nature of light-absorbing carbonaceous aerosols. *Atmos. Chem. Phys.* 6: 3131–3148.
- Andreae, M.O. and P. Merlet. 2001. Emission of trace gases and aerosols from biomass burning, *Glob. Biogeochem.Cycles.* 15: 955–966.

- Aston, A.R., and A.M. Gill. 1976. Coupled soil-moisture, heat and water-vapor transfers under simulated fire conditions. *Aus. J. of Soil Res.* 14: 55-66.
- Aufderheide, R., and W.G. Morris. 1949. Broadcast slash burning after a rain. *Fire Control Notes* 10: 1-6.
- Baird, M., D. Zabowski, and R.L. Everett. 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. *Plant Soil.* 209: 233-243.
- Belillas, C.M., and F. Roda. 1993. The effects of fire on water-quality, dissolved nutrient losses and the export of particulate matter from dry heathland catchments. *J. Hydrol.* 150: 1-17.
- Beyer, H. L. 2004. Hawth's Analysis Tools for ArcGIS. Available at <http://www.spatial ecology.com/htools>.
- Boardman, H.P. 1959. Some interesting and important facts about Lake Tahoe. Truckee Basin Water Committee and Nevada Cooperative Snow Surveys Committee of State Association of Soil Conservation Districts. Reno, NV.
- Boring, L.R., Hendricks, J.J., Wilson, C.A., and R.J. Mitchell. 2004. Season of burn and nutrient losses in a longleaf pine ecosystem. *Int. J. Wildland Fire.* 13: 443-453.
- Brady, N.C., and R.R. Weil. 2002. *The Nature and Properties of Soils.* Upper Saddle River, New Jersey, Prentice Hall.
- Brown, J.K. 1974. Handbook for inventorying downed woody material. USDA For. Serv. GTR INT-16. Intermountain Forest and Range Experiment Station. Ogden, Utah.
- Burgan, R.E. 1979. Estimating live fuel moisture for the 1978 national fire danger rating system. USDA Forest Service Research Paper INT-226. Intermountain Forest and Range Experiment Station. Ogden, Utah.

- Caldwell, T.G., D.W. Johnson, W.W. Miller, and R.G. Qualls. 2002. Forest floor carbon and nitrogen losses due to prescription fire. *Soil Sci. Soc. Am. J.* 66: 262-267.
- Chakrabarty, R.K., H. Moosmüller, M.A. Garro, W.P. Arnott, J. Walker, R.A. Susott, R.E. Babbitt, C.E. Wold, E.N. Lincoln, and W.M. Hao. 2006. Emissions from the laboratory combustion of wildland fuels: Particle morphology and size, *J. Geophys. Res.-Atmospheres*, 111, D7, D07204.
- Chandler, C., P. Cheney, P. Thomas, L. Trabaud, and D. Williams. 1983. *Fire in Forests Volume 1: Forest Fire Behavior and Effects*. New York, John Wiley & Sons, Inc.
- Chen, L.-W.A., H. Moosmüller, W.P. Arnott, J.C. Chow, J.G. Watson, R.A. Susott, R.E. Babbitt, C. Wold, E. Lincoln, and W.M. Hao. 2006. Particle emissions from laboratory combustion of wildland fuels: In situ optical and mass measurements, *Geophys. Res. Lett.*, 33, L04803, 1–4, doi:10.1029/2005JD006659.
- Chen, L.W., H. Moosmuller, P.W. Arnott, J.C. Chow, J.G. Watson, R.A. Susott, R.E. Babbitt, C.E. Wold, E.N. Lincoln, and W.M. Hao. 2007. Emissions from laboratory combustion of wildland fuels: Emission factors and source profiles. *Environ. Sci. Tech.* 41: 4317-4325.
- Chen, L.W.A. 2010. Moisture effects on carbon and nitrogen emission from burning of wildland biomass. *Atmos. Chem. Phys.* In Press.
- Chorover, J., P.M. Vitousek, D.A. Everson, A.M. Esperanza, and D. Turner. 1994. Solution chemistry profiles of mixed-conifer forests before and after fire. *Biogeochemistry*. 26: 115-144.
- Chow, J.C. 1995. Critical review: Measurement methods to determine compliance with ambient air quality standards for suspended particles. *J. Air Waste Manag. Assoc.* 45: 320–382.

- Chow, J.C., J.G. Watson, L.-W.A. Chen, M.C.O. Chang, N.F. Robinson, D. Trimble, and S.D. Kohl. 2007a. The IMPROVE_A temperature protocol for thermal/optical carbon analysis: Maintaining consistency with a long-term database. *J. Air Waste Manag. Assoc.* 57: 1014–1023.
- Chow, J.C., J.G. Watson, H.D. Kuhns, V. Etyemezian, D.H. Lowenthal, D.J. Crow, S.D. Kohl, J.P. Engelbrecht, and M.C. Green. 2004. Source profiles for industrial, mobile, and area sources in the Big Bend Regional Aerosol Visibility and Observational (BRAVO) Study. *Chemosphere.* 54: 185–208.
- Chow, J.C., J.G. Watson, D.H. Lowenthal, L.-W.A. Chen, B. Zielinska, L.R. Mazzoleni, and K.L. Magliano. 2007b. Evaluation of organic markers for chemical mass balance source apportionment at the Fresno supersite. *Atmos. Chem. Phys.* 7: 1741–2754.
- Chuvieco, E., Aguado, I., and A.P. Dimitrakopoulos. 2004. Conversion of fuel moisture content values to ignition potential for integrated fire danger assessment. *Can. J. For. Res.* 34: 2284-2293.
- DeBano, L.F., D.G. Neary, P.F. Ffolliott. 1998. *Fire's Effects on Ecosystems*. Toronto, Canada, John Wiley & Sons.
- Gani, A. and I. Naruse. 2007. Effect of cellulose and lignin content on pyrolysis and combustion characteristics for several types. *Renewable Energy.* 32: 649–661.
- Gill, A.M., W.S.W. Trollope, and D.A. Macarthur. 1978. Role of moisture in the flammability of natural fuels in the laboratory. *Aust. For. Res.* 8: 199-208.
- Gertler, A.W., T.A. Cahill, S. Cliff, and L. Tarnay. 2006. Local air pollutants threaten Lake Tahoe's clarity. *California Agriculture.* 60: 53-58.

- Gillon, D., V. Gomendy, C. Houssard, J. Marechal, and J.C. Valette. 1995. Combustion and nutrient losses during laboratory burns. *Int. J. Wildland Fire*. 5: 1-12.
- Glass, D.W., D.W. Johnson, R.R. Blank, and W.W. Miller. 2008. Factors affecting mineral nitrogen transformations by soil heating: A laboratory-simulated fire study. *Soil Science* 173: 387-400.
- Goldman, C.R. 1988. Primary productivity, nutrients, and transparency during the early onset of eutrophication in ultra-oligotrophic Lake Tahoe, California Nevada. *Limnol. Ocean.* 33: 1321-1333.
- Graham, J.B. 2005. Forest fuel and fire dynamics in mixed-oak forests of Southeastern Ohio. Masters Thesis. Dept of Environmental and Plant Biology, College of Arts and Science, Ohio University.
- Guyon, P., B. Graham, J. Beck, O. Boucher, E. Gerasopoulos, O.L. Mayol-Bracero, G.C. Roberts, P. Artaxo, and M.O. Andreae. 2003. Physical properties and concentrations of aerosol particles over the Amazon tropical forest during background and biomass burning conditions. *Atmos. Chem. Phys.* 3: 951–967.
- Harrod, R.J. and S. Reichard 2002. Fire and invasive species within the temperate and boreal coniferous forests of western north America. Pages 95-101 *in* K.E.M. Galley and T.P. Wilson eds. *Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species*. Fire Conference 2000: the First National Congress of Fire Ecology, Prevention, and Management. Misc. Pub. No. 11 Tall Timbers Research Station, Tallahassee, FL.
- Hobbs, R.J., J.E. Curral, and C.H. Gimingham. 1984. The use of ‘thermocolor’ pyrometers in the study of heath fire behavior. *J. Ecol.* 72: 241-250.

- Hungerford, R.D., M.G. Harrington, W.H. Frandsen, K.C. Ryan, and G.J. Niehoff. 1991. Influence of fire on factors that affect site productivity, *In* A. E. Harvey and L. F. Neuenschwander, (eds.) Proceedings of the management and productivity of western-Montane forest soils., Vol. General Technical Report INT-280, US Forest Service, Intermountain Research Station.
- IPCC: Climate Change. 2007. The physical science basis. Summary for policymakers. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, prepared by Intergovernmental Panel on Climate Change (IPCC), Geneva, Switzerland.
- Janhäll, S., M.O. Andreae, and U. Pöschl. 2009. Biomass burning aerosol emissions from vegetation fires: particle number and mass emission factors and size distributions. *Atmos. Chem. Phys. Discus.* 917183–17217.
- Jassby, A.D., J.E. Reuter, R.P. Axler, C.R. Goldman, and S.H. Hackley. 1994. Atmospheric deposition of nitrogen and phosphorous in the annual nutrient load of Lake Tahoe (California Nevada). *Water Resour. Res.* 30: 2207-2216.
- Johnson, D.W. 1995. Soil properties beneath Ceanothus and Pine stands in the eastern Sierra-Nevada. *Soil Sci. Soc. Am. J.* 59: 918-924.
- Johnson, D.W., Curtis, P.S., 2001. Effects of forest management on soil C and N storage: meta analysis. *For. Ecol. Manage.* 140: 227-238.
- Johnson, D. W., R.B. Susfalk, R.A. Dahlgren, and J.M. Klopatek. 1998. Fire is more important than water for nitrogen fluxes in semi-arid forests. *Environ. Sci. Pol.* 1: 79-86.
- Johnson, D.W., R.B. Susfalk, T.G. Caldwell, J.D. Murphy, W.W. Miller, and R.F. Walker. 2004. Fire effects on carbon and nitrogen budgets in forests. *Wat. Air Soil Pollut.* 4: 263-275.

- Jonsson, A.M., and B. Nihlgard. 2004. Slash pile burning at a Norway spruce clear-cut in southern Sweden. *Wat. Air Soil Pollut.* 158: 127-135.
- Kauffman, J.B. and R.E. Martin. 1989. Fire behavior, fuel consumption, and forest floor changes following prescribed understory fires in Sierra Nevada mixed conifer forests. *Can. J. For. Res.* 19: 455-462.
- Kaye, J. P., S.C. Hart, R.C. Cobb, and J.E. Stone. 1999. Water and nutrient outflow following the ecological restoration of a ponderosa pine - bunchgrass ecosystem. *Restor. Ecol.* 7: 252-261.
- Kilgore, B.M., and M.L. Heinselman. 1990. Fire in wilderness ecosystems. In: Hendee, J.C. Stankey, G.H., and R.C. Lucas (eds). *Wilderness management*. 2nd ed. Golden, CO: North American Press: 297–335.
- Kirchstetter, T.W., T. Novakov, and P.V. Hobbs. 2004. Evidence that the spectral dependence of light absorption by aerosols is affected by organic carbon. *J. Geophys. Res. -Atmospheres*, 109, D21, D21208, doi:10.1029/2004JD004999.
- Knapp, E.E., J.E. Keeley, E.A. Ballenger, and T.J. Brennan. 2005. Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. *For. Ecol. Manag.* 208:383-397.
- Koppmann, R., K. von Czapiewski, and J.S. Reid. 2005. A review of biomass burning emissions, part I: gaseous emissions of carbon monoxide, methane, volatile organic compounds, and nitrogen containing compounds. *Atmos. Chem. Phys. Discuss.* 510455–10516.
- Korb, J.E., N.C. Johnson, and W.W. Covington. 2004. Slash pile burning effects on soil biotic and chemical properties and plant establishment: recommendations for amelioration. *Ecol. Rest.* 12: 52-62.

- Kovacic, D. A., D.M. Swift, J.E. Ellis, and T.E. Hakonson. 1986. Immediate effects of prescribed burning on mineral soil nitrogen in ponderosa pine of New Mexico. *Soil Science*. 141: 71-76.
- Kuhlbusch, T.A., J.M. Lobert, P.J. Crutzen, and P. Warneck. 1991. Molecular nitrogen emissions from denitrification during biomass burning. *Nature*. 351: 135–137.
- Lewis, J. 1996. Turbidity-controlled suspended sediment sampling for runoff-event load estimation. *Water Resour. Res.* 26: 2069-2077.
- Little, S.N., and J.L. Ohmann. 1988. Estimating nitrogen lost from forest floor during prescribed fires in Douglas-fir western hemlock clearcuts. *Forest Science*. 34: 152-164.
- Lobert, J.M., D.H. Scharffe, W.M. Hao, and P.J. Crutzen. 1990. Importance of biomass burning in the atmospheric budgets of nitrogen-containing gases. *Nature*. 346: 552–554.
- Marjamäki, M., M. Lemmetty, and J. Keskinen. 2005. ELPI response and data reduction - I: Response functions, *Aerosol Sci. Technol.* 39: 575–582.
- McMeeking, G.R., S.M. Kreidenweis, S. Baker, C.M. Carrico, J.C. Chow, J.L. Collett Jr., W.M. Hao, A.S. Holden, T.W. Kirchstetter, W.C. Malm, H. Moosmüller, A.P. Sullivan, and C.E. Wold. 2009. Emissions of trace gases and aerosols during the open combustion of biomass in the laboratory. *J. Geophys. Res.-Atmospheres*. 114, D19210, doi:10.1029/2009JD011836.
- Miller, W.W., D.W. Johnson, T.M. Loupe, J.S. Sedinger, E.M. Carroll, J.D. Murphy, R.F. Walker, D. Glass. 2006. Nutrients flow from runoff at burned forest site in Lake Tahoe Basin. *Calif. Agricult.* 60: 65-71.
- Moghaddas, E.Y., and S.L. Stephens. 2007. Thinning, burning, and thin-burn fuel treatment effects on soil properties in a Sierra Nevada mixed-conifer forest. *For. Ecol. Manag.* 250: 156-166.

- Moosmüller, H., R.K. Chakrabarty, and W.P. Arnott. 2009. Aerosol light absorption and its measurement: A review. *J. Quant. Spectrosc. Radiat. Transfer.* 110:844–878.
- Mukhtar, S., A.J. Rose, S.C. Capareda, C.N. Boriack, R.E. Lacey, B.W. Shaw, and C.B. Parnell Jr. 2003. Assessment of ammonia adsorption onto Teflon and LDPE tubing used in pollutant stream conveyance, *Agricultural Engineering International: the CIGR Journal of Scientific Research and Development*, Manuscript BC 03 012, Vol. V, Paper 29, 1–13.
- Murphy, J.D., D.W. Johnson, W.W. Miller, R.F. Walker, E.F. Carroll, and R.R. Blank. 2006. Wildfire effects on soil nutrients and leaching in a Tahoe Basin watershed. *J. Environ. Qual.* 35: 479-489.
- Neary, D. G., C.C. Klopatek, L.F. DeBano, and P.F. Ffolliott. 1999. Fire effects on belowground sustainability: a review and synthesis. *For. Ecol. Manag.* 122: 51-71.
- Nevada Division of Environmental Protection. State of Nevada Surface Water Monitoring Network at First Creek. URL: <http://ndep.nv.gov/bwqp/1b.html> [accessed: July 17, 2010].
- Ottmar, R.D.; Sandberg, D.V.; Riccardi, C.L.; Prichard, S.J. 2007. An overview of the Fuel Characteristic Classification System – quantifying, classifying, and creating fuelbeds for resource planning. *Can. J. of Forest Res.* 37: 2383-2393.
- Page-Dumroese, D.S., and M.F. Jurgensen. 2006. Soil carbon and nitrogen pools in mid to late successional forest stands of the northwestern United States: potential impact of fire. *Can. J. For. Res.* 36: 2270-2284.
- Pagels, J., A. Gudmundsson, E. Gustavsson, L. Asking, and M. Bohgard. 2005. Evaluation of aerodynamic particle sizer and electrical low-pressure impactor for unimodal and bimodal mass-weighted size distributions. *Aerosol. Sci. Techn.* 39: 871-887.

- Pathway (2007). Pathway, 2007 Technical Supplement for Water Quality. URL: <http://tiims.org/Data-Repository/Documents/Lake-Tahoe-Basin/Science-and-Reporting/Data-Synthesis,-Reporting,-and-Management/Management/Pathway/Evaluation-Report/Technical/Pathway-2007-Evaluation-Technical-Report---Water-Q.aspx> [accessed: July 13, 2010].
- Prichard, S.J., Ottmar, R.D., Anderson, G.K., 2005. Consume 3.0 User's Guide. United States Department of Agriculture Forest Service, Pacific Northwest Research Station, Seattle, WA.
- Raison, R. J., P.K. Khanna, and P.V. Woods. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. *Can. J. For. Res.* 15: 132-140.
- Reid, J.S., T.F. Eck, S.A. Christopher, R. Koppmann, O. Dubovik, D.P. Eleuterio, B.N. Holben, E.A. Reid, and J. Zhang. 2005a. A review of biomass burning emissions part III: intensive optical properties of biomass burning particles. *Atmos. Chem. Phys.* 5: 827–849.
- Reid, J.S., R. Koppmann, T.F. Eck, and D.P. Eleuterio. 2005b. A review of biomass burning emissions, part II: Intensive physical properties of biomass burning particles. *Atmos. Chem. Phys.* 5: 799–825.
- Richter, D. D., and C.W. Ralston. 1982. Prescribed fire: effects on water quality and forest nutrient cycling. *Science*. 215: 661-663.
- Riggen, P.J., R.N. Lockwood, P.M. Jacks, and C.G. Colver. 1994. Effects of fire severity on nitrite mobilization in watersheds subject to chronic atmospheric deposition. *Environ. Sci. Technol.* 28: 369-375.

- Rinehart, L.R., E.M. Fujita, J.C. Chow, K.L. Magliano, and B. Zielinska. 2006. Spatial distribution of PM_{2.5} associated organic compounds in central California, *Atmos. Environ.* 40: 290–303.
- Simoneit, B.R.T., J.J. Schauer, C.G. Nolte, D.R. Oros, V.O. Elias, M.P. Fraser, W.F. Rogge, and G.R. Cass. 1999. Levoglucosan, a tracer for cellulose in biomass burning and atmospheric particles. *Atmos. Environ.* 33: 173–182.
- Sinha, P., P.V. Hobbs, R.J. Yokelson, D.R. Blake, S. Gao, and T.W. Kirchstetter. 2004. Emissions from miombo woodland and dambo grassland savanna fires. *J. Geophys. Res.-Atmospheres*. 109, D11, D11305, doi:10.1029/2004JD004521.
- Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. U.S. General Soil Map (STATSGO2) for Nevada. Available online at <http://soildatamart.nrcs.usda.gov> accessed 11/18/2009.
- Stephens, S.L., T. Meixner, M. Poth, B. McGurk, and D. Payne. 2004. Prescribed fire, soils, and stream water chemistry in a watershed in the Lake Tahoe Basin, California. *Int. J. Wildland Fire*. 13: 27-35.
- Susfalk R.B. 2008. Personal Communication. Allometric Equations for Sierra Shrubs. Desert Research Institute. Reno, NV.
- Western Regional Climate Center. 2009. Historical Precipitation Data for Incline Village, NV. <http://www.wrcc.dri.edu/cgi-bin/rawMAIN.pl?nvinc>. Accessed 11/12/2009.
- Yokelson, R.J., D.W.T. Griffith, and D.E. Ward. 1996. Open-path Fourier transform infrared studies of large-scale laboratory biomass fires. *J. Geophys. Res.* 101: 21067–21080.

Table 1. Moisture content of fuels burned in the laboratory combustion studies.

Material	Moisture Class I	Moisture Class II	Moisture Class III
Litter	5.3 ^a	14.4	22.0
Duff	7.2	16.3	23.9
Soil	2.7	11.8	19.4
Squaw Carpet Leaves	7.0	47.9	65.5
Squaw Carpet Stems	6.9	44.3	57.4
Manzanita Leaves	5.1	40.0	59.9
Manzanita Stems	5.3	38.6	52.2
Bitterbrush Leaves	6.7	72.9	84.2
Bitterbrush Stems	6.0	39.2	56.9

^a: moisture content in percent

Table 2. ANOVA results for moisture, extractable NH_4^+ and NO_3^- from litter, duff, and soil (0-5 cm) horizons resulting from changes in time, plot differences, or time-plot interactions.

		Time	Plot	Time * Plot
Litter	Moisture	ns ^a	***	***
	NH_4^+	***	***	***
	NO_3^-	ns	ns	**
Duff	Moisture	ns	**	**
	NH_4^+	**	***	**
	NO_3^-	ns	ns	ns
Soil	Moisture	(*)	*	(*)
	NH_4^+	*	*	*
	NO_3^-	ns	ns	ns

^a: ns: $p > 0.10$, (*): $p < 0.10$, *: $p < 0.05$, **: $p < 0.01$, ***: $p < 0.001$

Table 3. Water quality for selected water samples from First Creek.

First Creek @ Dale Drive	NO3-N	NH3-N	OPO4-P	DP	TP (mg/l)	TKN	DKN	TN	TSS	pH	TU (NTU)	EC (µmhos/cm)
Grab Sample: 8/29/08 *	0.015	0.008	0.01	0.01	0.023	0.13	0.06	0.122	10.0	7.6	--	--
EMC: 11/1/08 **	<.001	0.007	0.043	0.06	0.139	0.72	0.20	0.713	111	7.2	--	--
NDEP Standard ***	--	--	--	--	0.6	--	--	0.3	4	7.0-8.1	--	--
NDEP Grab Samples (1999-2008; n=40)												
Mean	0.01	0.1	--	0.01	0.02	0.20	--	0.18	6.73	7.8	2.9	86
Min	0	0.1	--	0.00	0.00	0.10	--	0.00	0.00	7.2	0.9	53
Max	0.05	0.1	--	0.02	0.09	0.50	--	0.56	15.00	8.1	5.0	128
Stdev	0.0176	--	--	0.01	0.01	0.08	--	0.13	3.78	0.3	1.1	15

*: Grab sample taken during baseflow conditions.

** : Event Mean Concentration (EMC) during a rain event.

***: (Pathway, 2007).

Table 4. Fuel consumption of fuels burned in the laboratory combustion studies.

Material	Moisture Class I	Moisture Class II	Moisture Class III
Litter	92.2 ^a	72.1	79.5
Duff	52.2	47.3	44.8
Soil	8.5	9.5	8.6
Shrub Leaves	88.1	76.7	72.5
Shrub Stems	90.6	70.0	86.7

^a: fuel consumption in percent

1 Table 5. Time-integrated combustion efficiency (CE) and emission factors^a for major C and N species as well as PM_{2.5} from controlled
 2 burning experiments.

Plant Species		Downed Material			Aboveground Shrub					
		Composite			Bitterbrush		Manzanita		Squaw Carpet	
Parameter\Fuel Type		Litter	Duff	Soil	Leaves	Stems	Leaves	Stems	Leaves	Stems
Moisture Level I (Dry)	Carbon % ^b	50%	32%	3%	52%	48%	49%	48%	47%	50%
	Burned % ^c	92%	52%	9%	92%	81%	90%	98%	83%	93%
	CE	0.94 ± 0.01	0.92 ± 0.02	0.86 ± 0.02	0.93 ± 0.01	0.94 ± 0.00	0.91 ± 0.01	0.94 ± 0.00	0.92 ± 0.01	0.92 ± 0.00
	CO	126.2 ± 12.7	140.6 ± 27.3	297.3 ± 48.2	112.6 ± 30.3	127.4 ± 8.8	113.2 ± 43.5	128.4 ± 6.7	144.5 ± 22.3	162.1 ± 8.9
	OC ^e	2.5 ± 0.2	9.3 ± 6.7	14.9 ± 4.4	9.1 ± 1.8	6.4 ± 2.0	21.2 ± 7.9	5.5 ± 0.6	11.3 ± 0.2	4.7 ± 0.1
	EC ^e	4.6 ± 0.3	5.7 ± 0.1	0.7 ± 0.2	15.4 ± 2.7	2.9 ± 0.3	15.9 ± 3.9	2.0 ± 0.4	8.1 ± 0.8	2.5 ± 0.5
	PM _{2.5} ^e	8.1 ± 0.9	18.3 ± 10.2	N.D.	27.1 ± 5.8	13.0 ± 3.9	47.5 ± 14.1	8.7 ± 0.4	25.3 ± 3.2	14.5 ± 1.1
	NO _x	5.9 ± 0.3	4.9 ± 0.9	0.9 ± 0.2	9.1 ± 2.8	9.1 ± 1.0	7.1 ± 3.0	5.4 ± 1.0	7.7 ± 1.5	10.5 ± 0.8
	NH ₃ ^e	1.3 ± 1.1	3.7 ± 2.4	11.5 ± 6.1	7.7 ± 2.4	3.4 ± 0.6	4.0 ± 1.6	1.9 ± 0.8	6.8 ± 1.6	5.6 ± 0.1
	NO ₃ ^{-e}	0.04 ± 0.01	0.03 ± 0.02	N.D.	0.05 ± 0.01	0.12 ± 0.00	0.16 ± 0.04	0.12 ± 0.00	0.07 ± 0.00	0.09 ± 0.02
NH ₄ ^{+e}	0.05 ± 0.01	0.06 ± 0.03	N.D.	0.04 ± 0.00	0.04 ± 0.01	0.04 ± 0.00	0.03 ± 0.00	0.05 ± 0.00	0.08 ± 0.00	
OPN ^e	0.24 ± 0.19	0.44 ± 0.35	0.31 ± 0.23	0.40 ± 0.11	0.31 ± 0.12	0.47 ± 0.21	0.09 ± 0.03	0.45 ± 0.04	0.28 ± 0.01	
Moisture Level II	Moisture ^d	10%	10%	10%	73%	39%	40%	39%	48%	44%
	Burned % ^c	72%	47%	10%	88%	66%	73%	53%	69%	92%
	CE	0.79 ± 0.06	0.70 ± 0.07	0.80 ± 0.06	0.88 ± 0.01	0.82 ± 0.06	0.72 ± 0.03	0.86 ± 0.11	0.67 ± 0.09	0.86 ± 0.06
	CO	253.2 ± 59.8	444.9 ± 68.1	431.4 ± 143.8	113.2 ± 42.4	261.3 ± 44.0	97.5 ± 76.7	107.1 ± 2.4	77.7 ± 57.7	108.1 ± 96.0
	OC ^e	99.8 ± 34.9	110.9 ± 41.5	17.5 ± 1.0	66.1 ± 9.9	66.6 ± 36.1	236.0 ± 62.4	62.8 ± 64.1	292.2 ± 115.1	92.6 ± 18.4
	EC ^e	5.2 ± 0.3	1.0 ± 0.1	1.5 ± 0.5	7.3 ± 4.0	3.7 ± 0.7	5.6 ± 1.1	2.3 ± 1.2	8.5 ± 3.2	2.6 ± 0.3
	PM _{2.5} ^e	160.5 ± 56.2	143.9 ± 62.2	N.D.	96.9 ± 18.3	119.3 ± 73.7	214.9 ± 84.1	94.9 ± 91.3	270.8 ± 68.4	109.6 ± 32.7
	NO _x	6.4 ± 1.5	6.7 ± 1.4	2.8 ± 0.8	9.9 ± 4.0	13.0 ± 3.1	2.2 ± 2.9	2.9 ± 1.2	4.0 ± 5.4	10.7 ± 7.9
	NH ₃ ^e	4.3 ± 1.7	26.7 ± 14.0	24.7 ± 6.6	10.2 ± 1.5	10.7 ± 0.9	10.0 ± 0.5	3.8 ± 0.5	11.9 ± 4.1	11.0 ± 5.4
	NO ₃ ^{-e}	0.17 ± 0.00	0.13 ± 0.01	N.D.	0.23 ± 0.02	0.20 ± 0.02	0.08 ± 0.03	0.08 ± 0.01	0.23 ± 0.04	0.20 ± 0.01
NH ₄ ^{+e}	0.24 ± 0.04	0.15 ± 0.01	N.D.	0.10 ± 0.00	0.03 ± 0.02	0.31 ± 0.09	0.05 ± 0.06	0.28 ± 0.11	0.06 ± 0.01	
OPN ^e	1.62 ± 0.59	3.57 ± 1.50	0.99 ± 0.26	2.03 ± 0.05	1.39 ± 0.68	4.4 ± 1.42	1.07 ± 1.33	7.70 ± 3.28	2.17 ± 0.46	
Moisture Level III	Moisture ^d	20%	20%	20%	84%	57%	60%	52%	66%	57%
	Burned % ^c	80%	45%	9%	86%	78%	68%	92%	65%	90%
	CE	0.74 ± 0.10	0.69 ± 0.07	0.82 ± 0.01	0.51 ± 0.07	0.84 ± 0.04	0.62 ± 0.09	0.87 ± 0.00	0.72 ± 0.01	0.88 ± 0.08
	CO	313.6 ± 105.8	407.6 ± 29.2	357.7 ± 15.9	215.6 ± 31.1	216.8 ± 60.4	134.2 ± 43.5	188.5 ± 25.1	142.6 ± 22.3	133.1 ± 102.6
	OC ^e	116.6 ± 48.8	131.1 ± 59.2	22.1 ± 0.8	393.6 ± 54.1	67.8 ± 11.1	315.3 ± 111.1	43.1 ± 5.8	210.4 ± 20.2	61.5 ± 39.7
	EC ^e	7.6 ± 4.9	1.1 ± 0.2	2.3 ± 0.9	5.3 ± 0.5	3.6 ± 0.1	7.7 ± 1.3	2.6 ± 1.0	4.8 ± 0.0	2.7 ± 0.7
	PM _{2.5} ^e	179.9 ± 54.7	208.9 ± 56.2	N.D.	768.7 ± 43.2	99.5 ± 4.4	493.8 ± 231.9	61.5 ± 3.8	387.2 ± 31.1	129.1 ± 43.5
	NO _x	8.4 ± 2.5	3.8 ± 0.2	5.4 ± 0.1	9.6 ± 2.0	7.5 ± 0.5	2.7 ± 2.0	4.7 ± 0.5	6.4 ± 2.2	9.1 ± 2.3
	NH ₃ ^e	5.3 ± 0.4	17.7 ± 5.9	24.5 ± 18.0	19.1 ± 1.4	10.9 ± 1.8	10.1 ± 0.4	3.8 ± 0.0	9.5 ± 0.8	9.7 ± 4.9
	NO ₃ ^{-e}	0.20 ± 0.05	0.09 ± 0.03	N.D.	0.28 ± 0.11	0.18 ± 0.02	0.12 ± 0.00	0.11 ± 0.01	0.14 ± 0.03	0.13 ± 0.08
NH ₄ ^{+e}	0.30 ± 0.11	0.15 ± 0.06	N.D.	0.32 ± 0.11	0.06 ± 0.01	0.42 ± 0.14	0.07 ± 0.01	0.18 ± 0.00	0.05 ± 0.03	
OPN ^e	1.59 ± 0.56	3.36 ± 1.40	0.60 ± 0.07	11.83 ± 0.53	1.59 ± 0.39	4.32 ± 1.52	0.66 ± 0.04	4.43 ± 0.36	1.33 ± 0.76	

3

^a In g/kgC (g per kilogram of carbon burned). Values are based on the average and standard deviation of two replicate burns. N.D. indicates the species are below detection limit. Emission factors with higher variability (average/standard deviation < 2) are marked in red.

^b Percentage of carbon in dry fuels.

^c Combined percentage of dry fuel burned.

^d Fuel moisture content in percentage.

^e Filter measurements

^f Moisture content < 5%.

Table 6. Significance of relationships between fuel moisture and gaseous and particulate C and N emissions for fuels burned in the laboratory combustion experiments.

Compound	Leaves	Stems	Litter	Duff	Soil
<u>Gaseous</u>					
CO ₂	** - ^a	* -	(*) -	(*) -	
CO	+	+	(*) +	(*) +	
NO _x			+		** +
NH ₃	** +	* +	(*) +		
<u>Particulate</u>					
PM 2.5	***+	**+	*+	*+	
Organic C	**+	**+	(*)+	(*)+	
Elemental C	**-	+		*-	
Organic N	**+	*+	(*)+	+	
NO ₃ ⁻	*+	(*)+	*+	+	
NH ₄ ⁺	*+		*+	+	

^a: (*): p<0.1, *: p<0.05, **: p<0.01, ***: p<0.001. The '+' or '-' sign indicates an increase or decrease in the emission factor with increasing moisture content. A '+' or '-' sign without an associated p-value indicates a trend towards a significant linear relationship at the 75% level.

Table 7. Example of comparison of emissions between this study and CONSUME 3.0 model results for duff.

	This study		CONSUME	
Moisture	9.3%	27.9%	9.3%	27.9%
CO ₂	95.9 ^a	88.7	91.8	91.8
CO	3.6	7.4	7.5	7.5
PM _{2.5}	0.5	3.8	0.7	0.7

^a: emissions as a percentage of the sum of CO₂, CO and PM_{2.5}

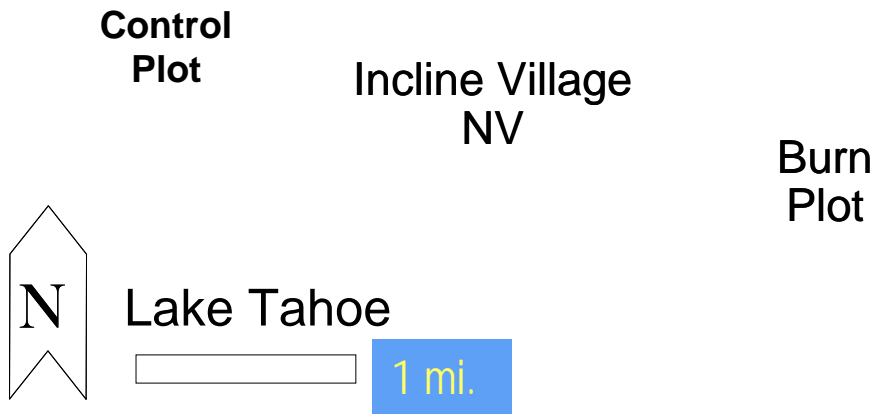


Figure 1. Locations of control and burned plot

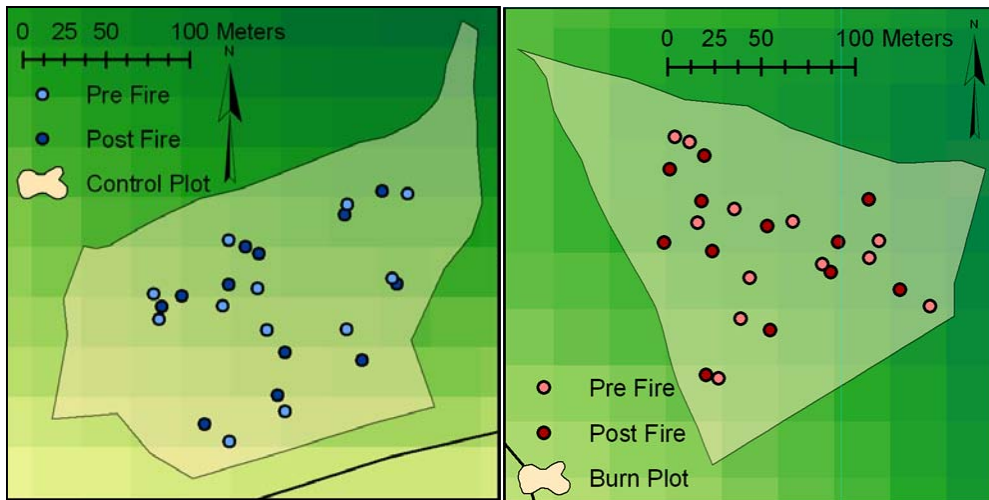


Figure 2. Sample locations in control (left) and burned plot (right)

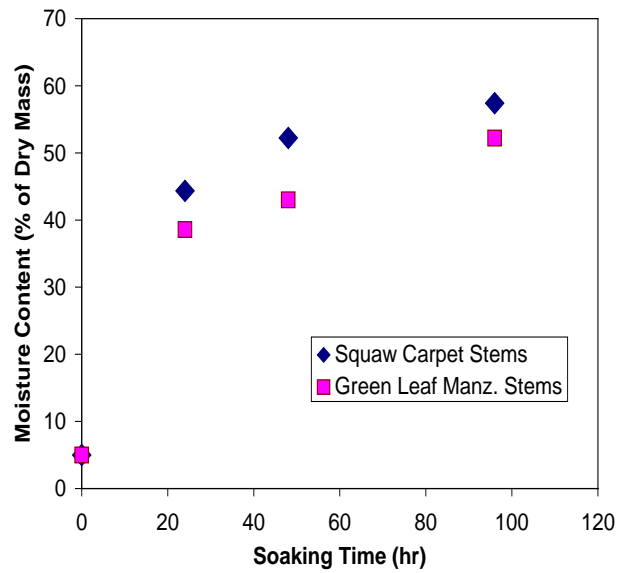


Figure 3. Examples of fuel moisture content, starting air dried (0 hr), as a function of time soaking in water. Excess water was drained before the measurement of moisture content.

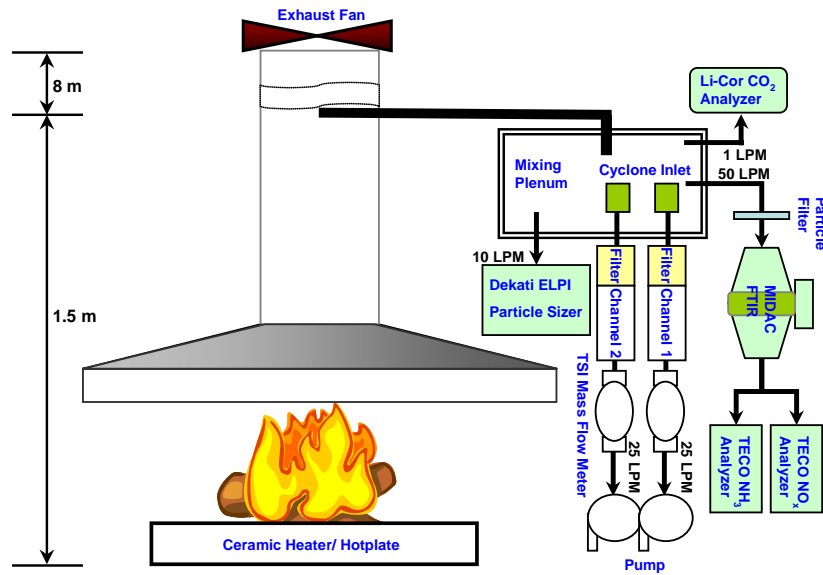


Figure 4. Schematic of the controlled burning experiment configurations.

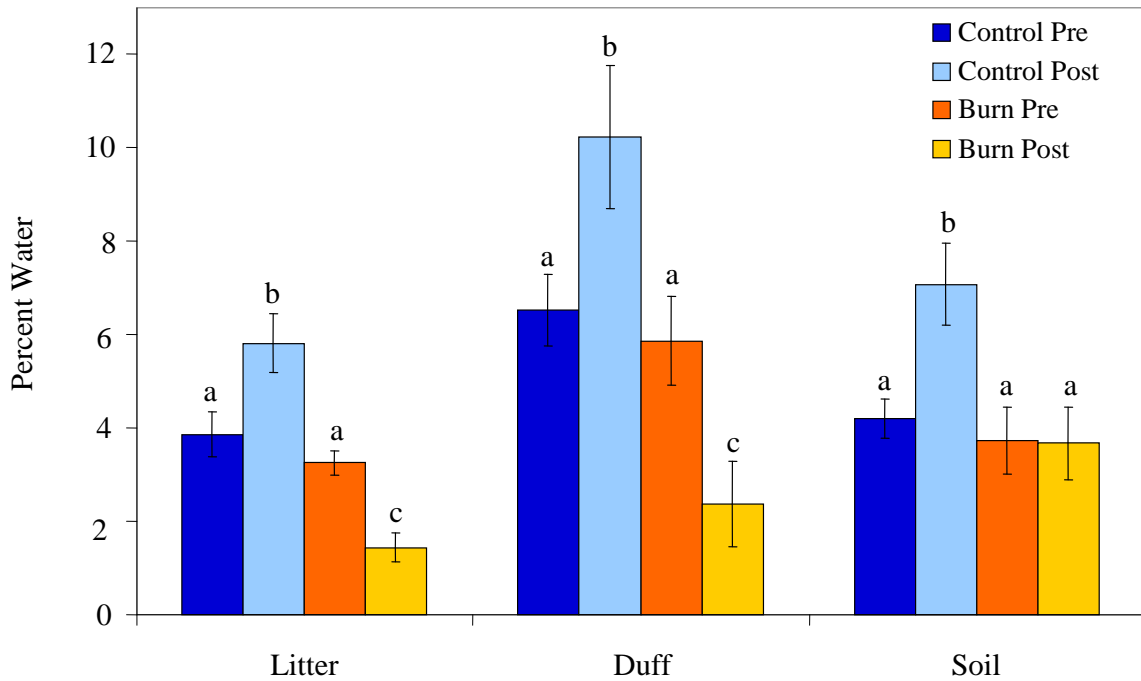


Figure 5. Average pre- and post-fire moisture content of forest floor and mineral soil in control and burned plots. Error bars represent one standard error. Different letters indicate significant differences within a group ($p < 0.05$).

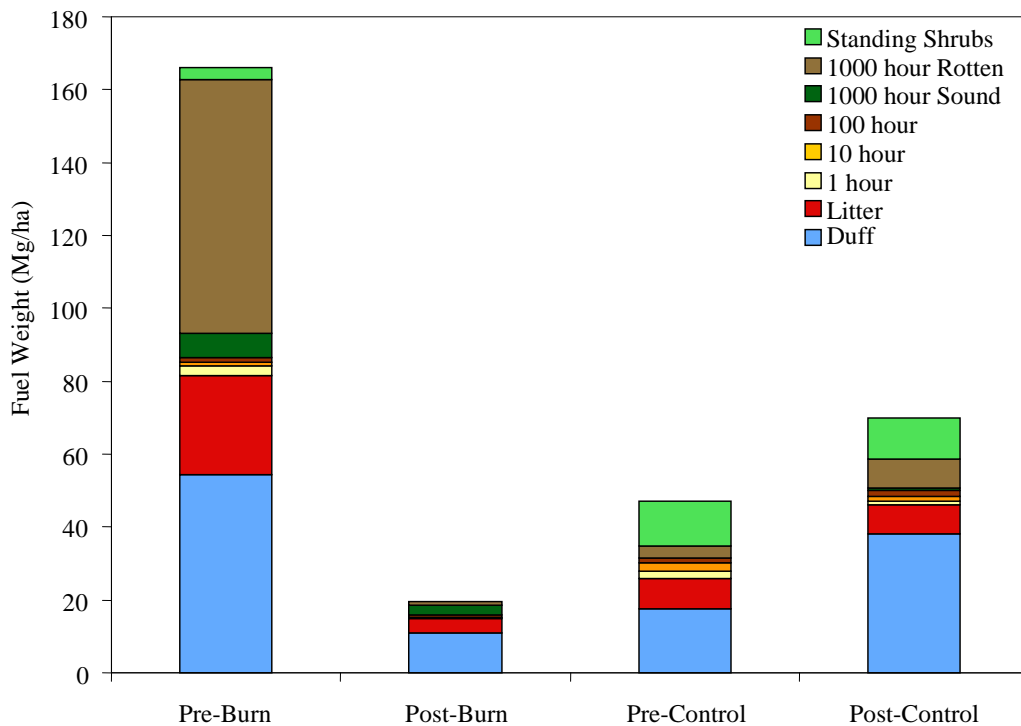


Figure 6. Fuel load inventory in the burn and control plots, before and after prescribed fire in the burn plot.

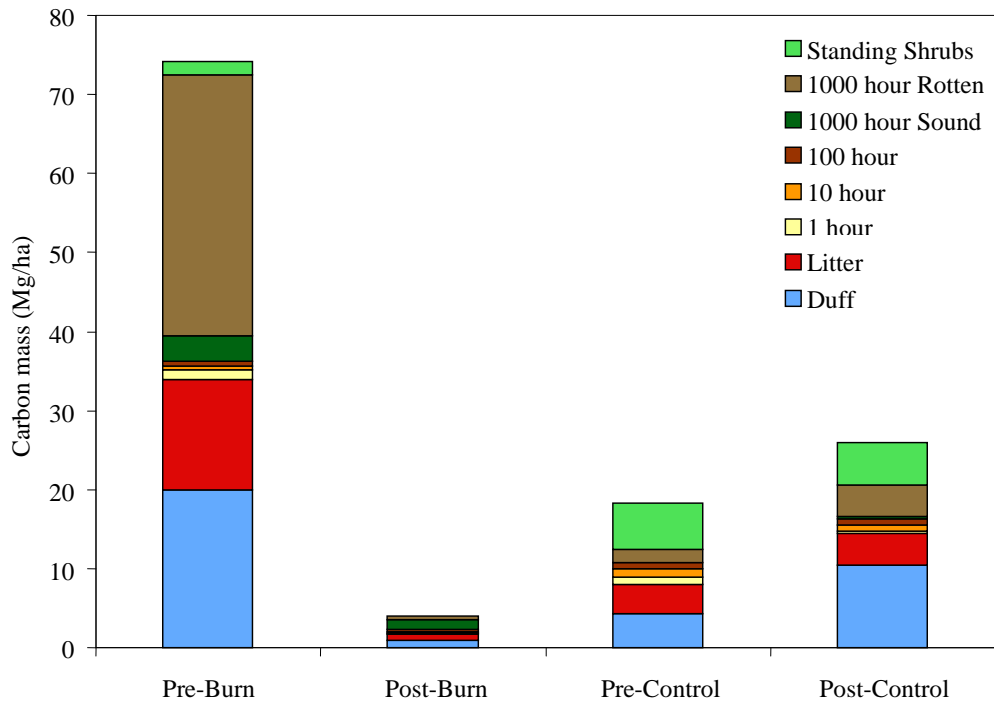


Figure 7. Carbon mass in the burn and control plot, before and after prescribed fire in the burn plot.

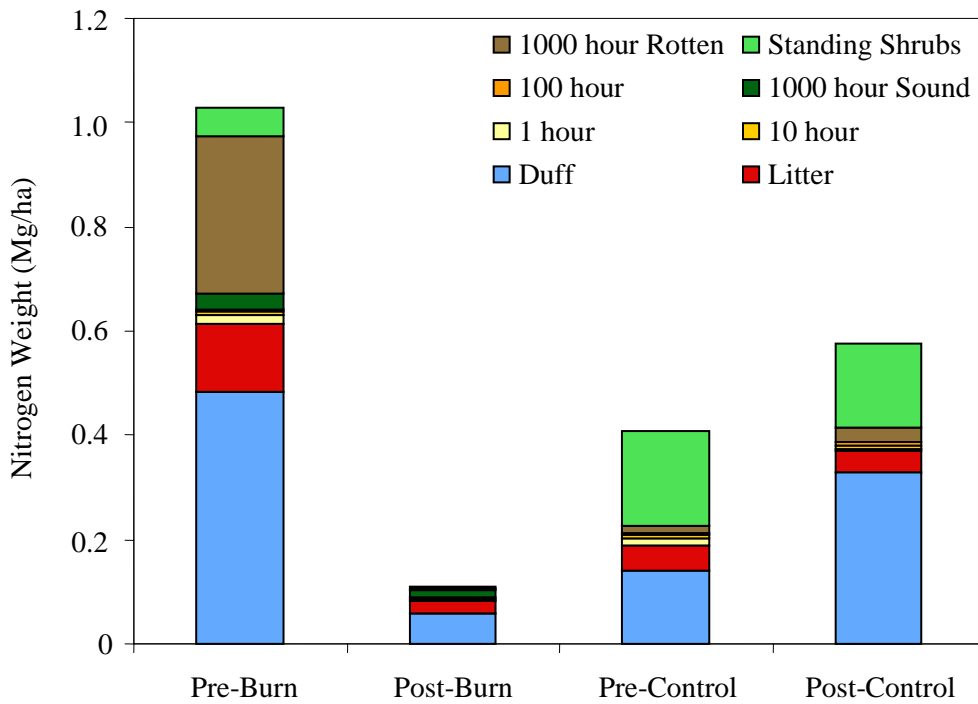


Figure 8. Mass of N in the burn and control plots before and after prescribed fire in the burn plot.

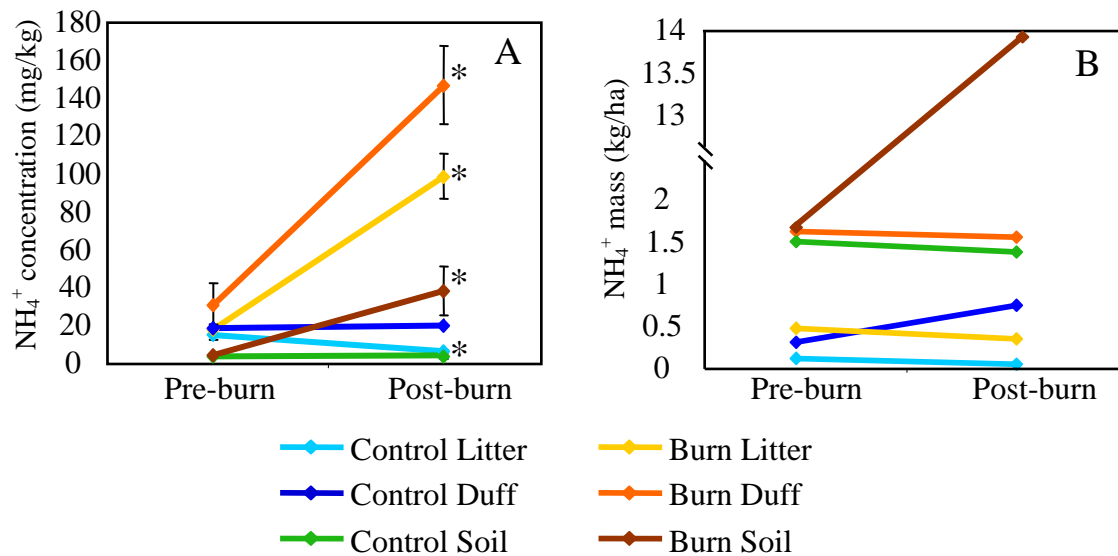


Figure 9. Changes in extractable NH_4^+ concentration (A) and content (B) in the control and burn plots. Error bars represent one standard error. Significant changes over time are noted by an asterisk (* $p < 0.05$).

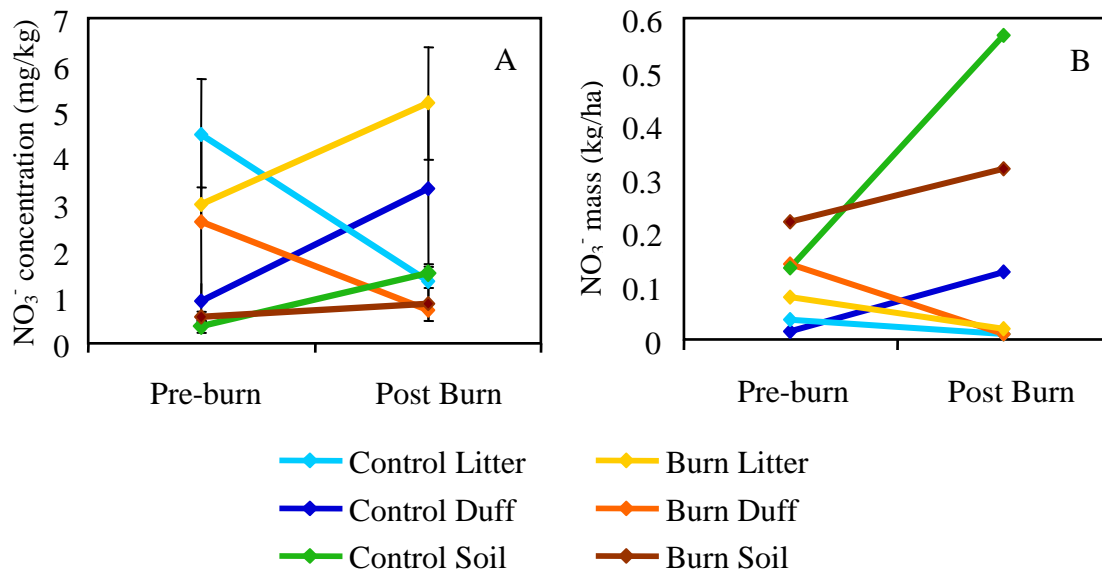


Figure 10. Changes in extractable NO_3^- concentration (A) and content (B) in the control and burn plots. Error bars represent one standard error.

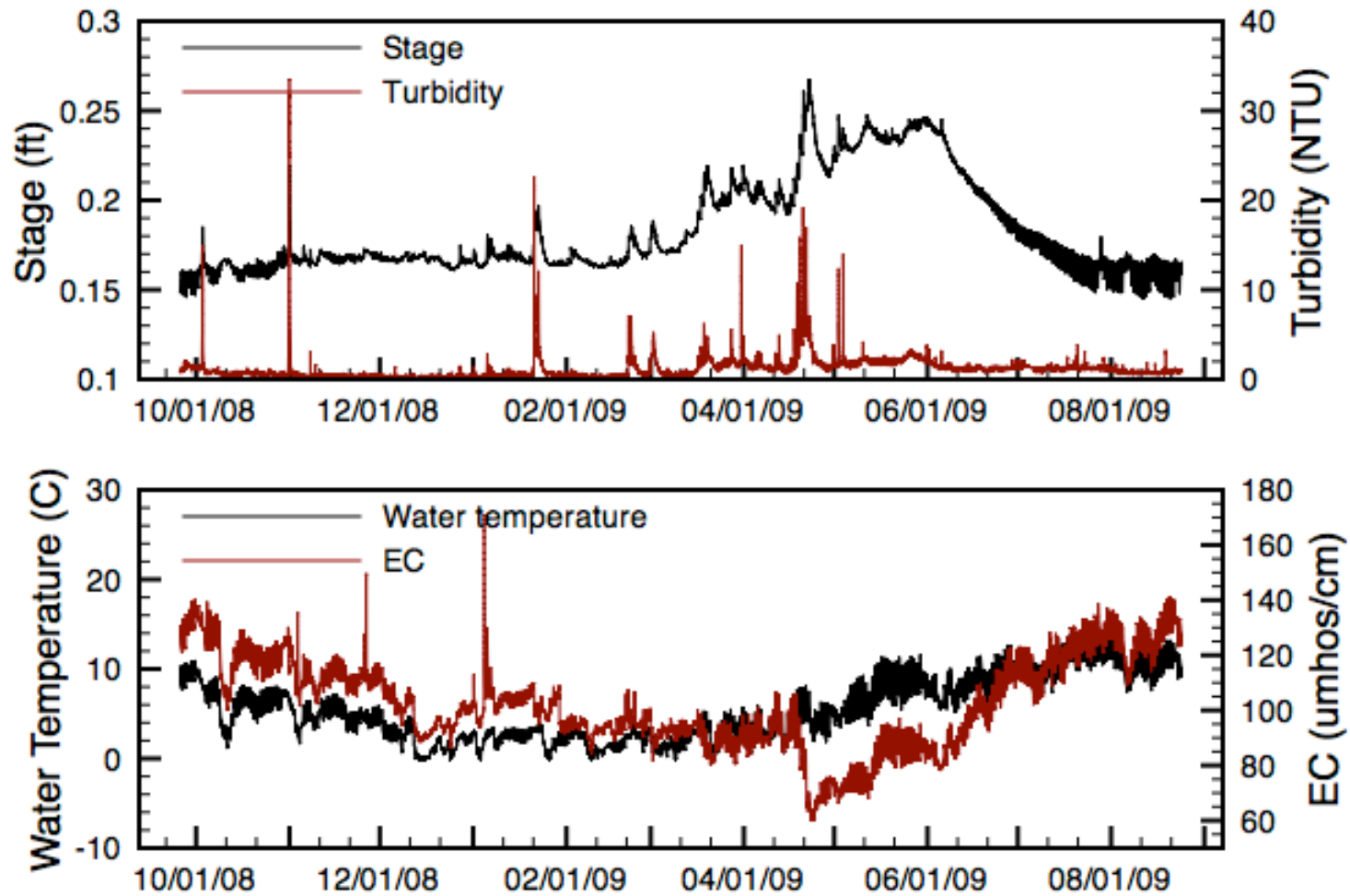


Figure 11. Stage, turbidity, water temperature, and electrical conductance measured at First Creek during WY2009. This graph is comprised of 15 min data using a 2-hr moving average.

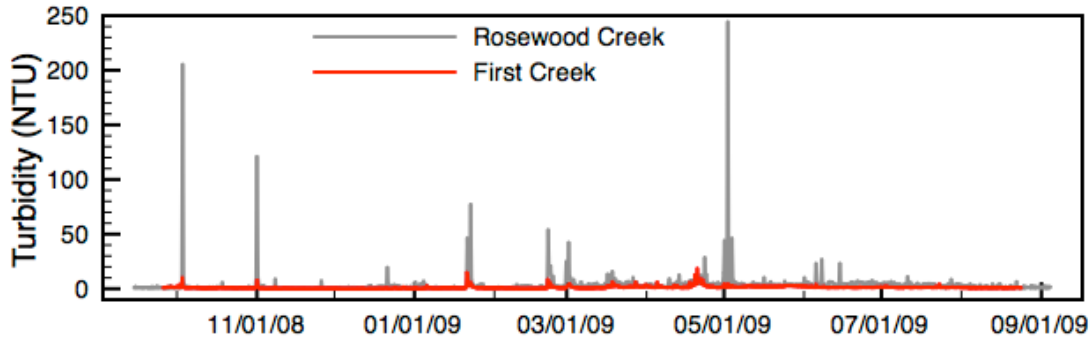


Figure 12. Comparison of turbidity between First Creek and nearby Rosewood Creek for water year 2008-2009. Rosewood Creek data courtesy of R. Susfalk (DRI).

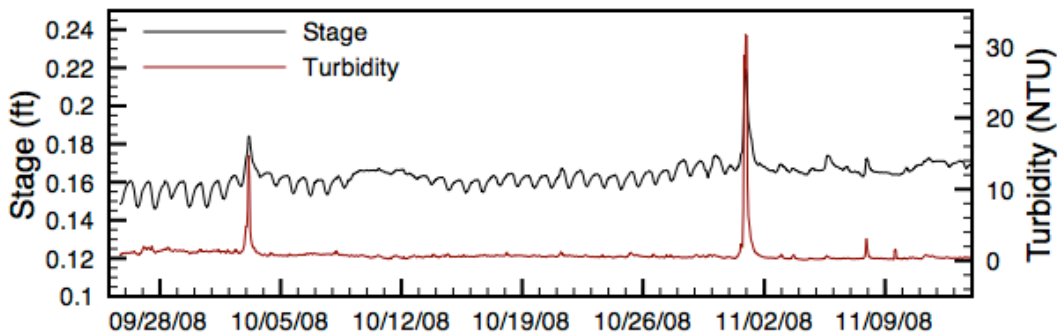


Figure 13. Stage and turbidity between 9/26/08 and 11/15/08. The controlled burn occurred on 10/10/2008. This graph is comprised of 15 min data using a 2-hr moving average, therefore the short-term peak values appear attenuated.

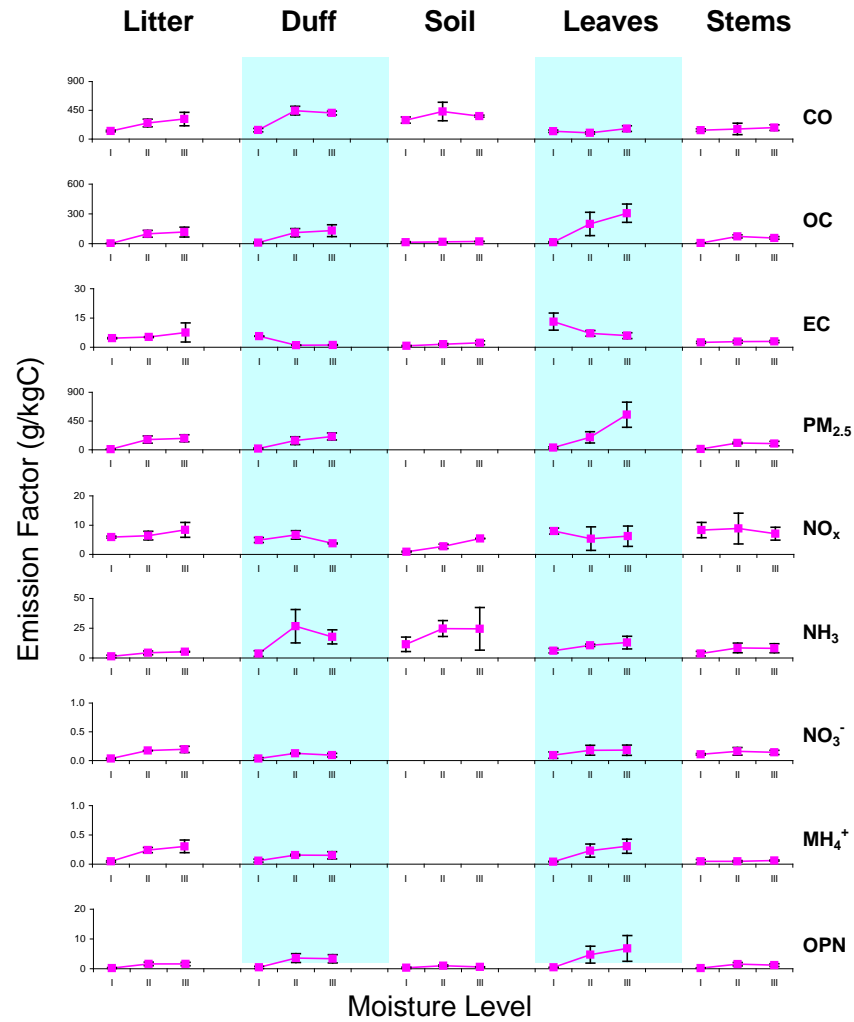


Figure 14. Emission factors, by fuel type, as a function of fuel moisture level, based on data presented in Table 5 (all the leaf and stem data have been combined).

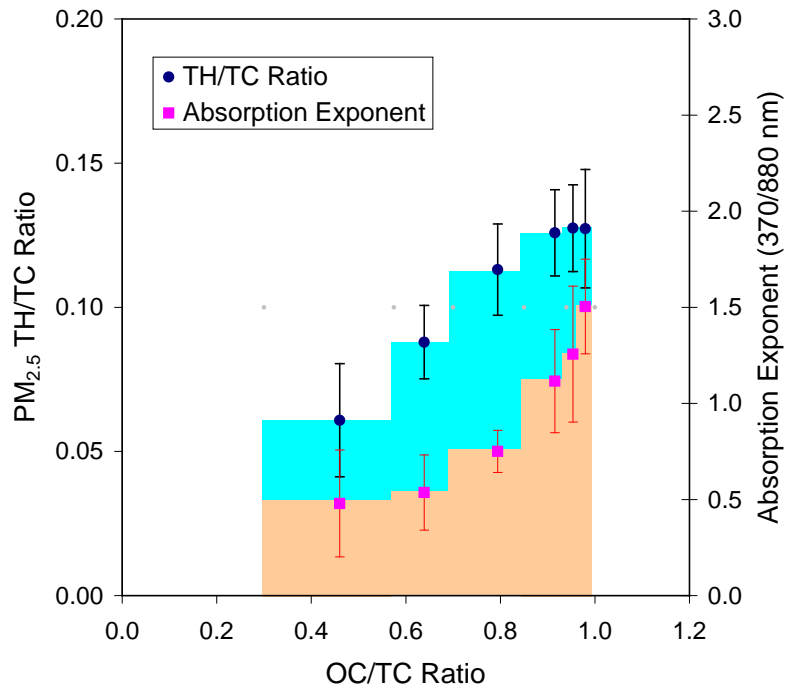


Figure 15. Conditional means of TH/TC ratio and absorption exponent of biomass-burning $PM_{2.5}$ as a function of OC fraction in TC. The symbols correspond to 6 groups with OC/TC ratio of 1) 0.3 – 0.575, 2) 0.575 – 0.7, 3) 0.7 – 0.85, 4) 0.85 – 0.94, 5) 0.94 – 0.97, and 6) 0.97 – 1. Soil combustion samples were excluded from this analysis.

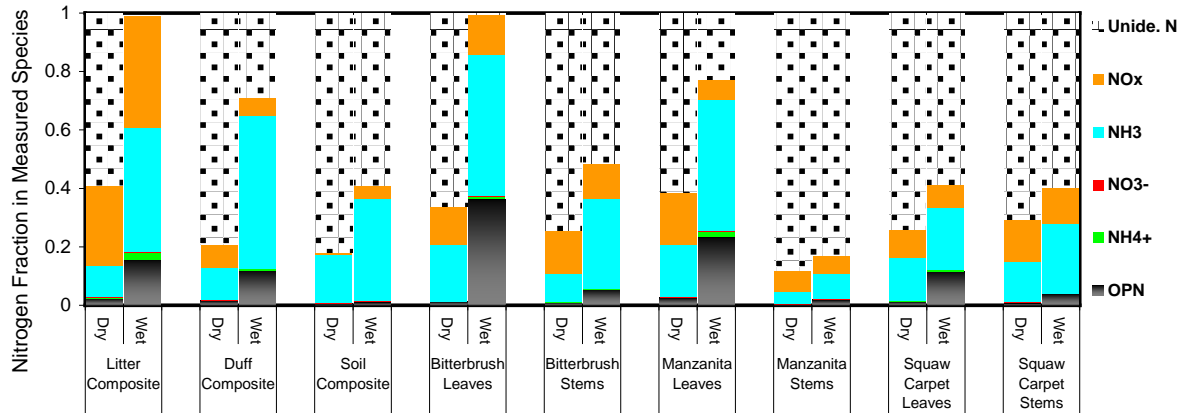
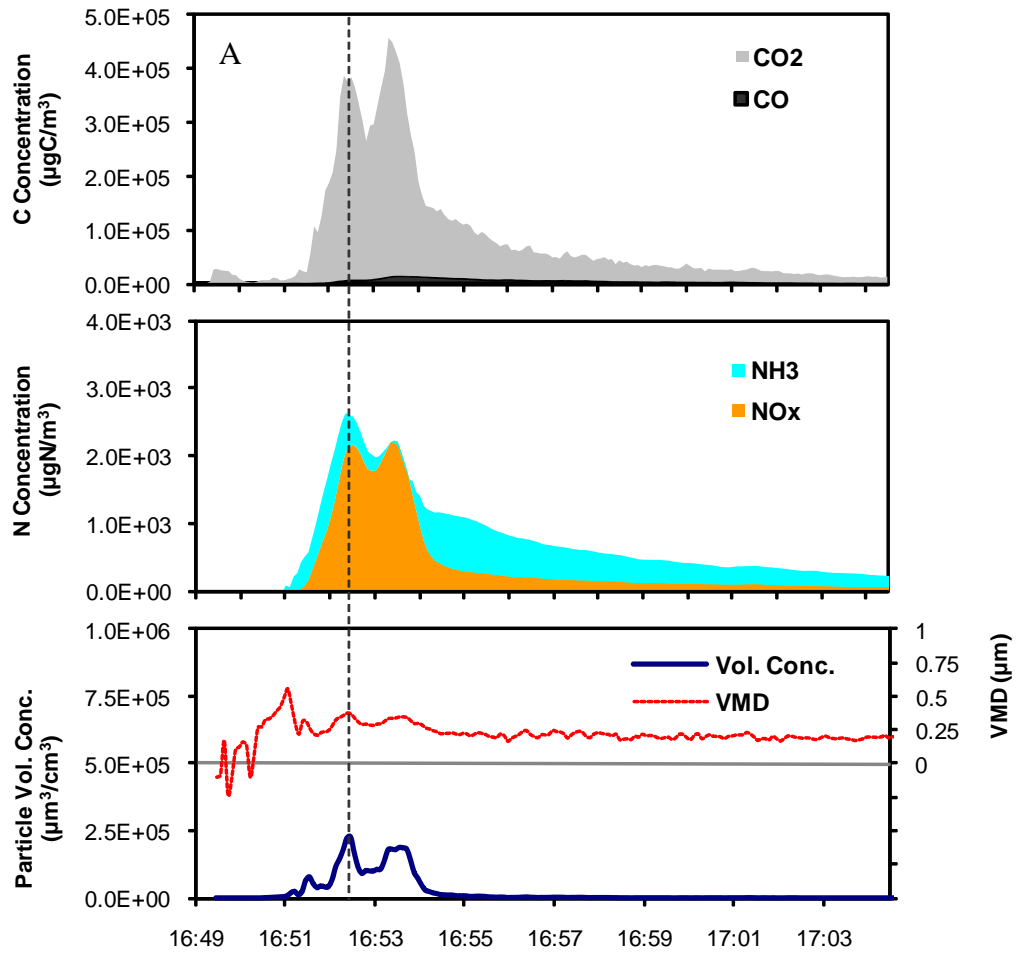


Figure 16. Nitrogen balance with respect to fuel N content, determined from N/C ratios in smoke plumes and in fuels. Unidentified N (Unide. N) is the fuel N that is not accounted for by measured species. Dry and wet fuels correspond to the Moisture Level I and III in Table 1.



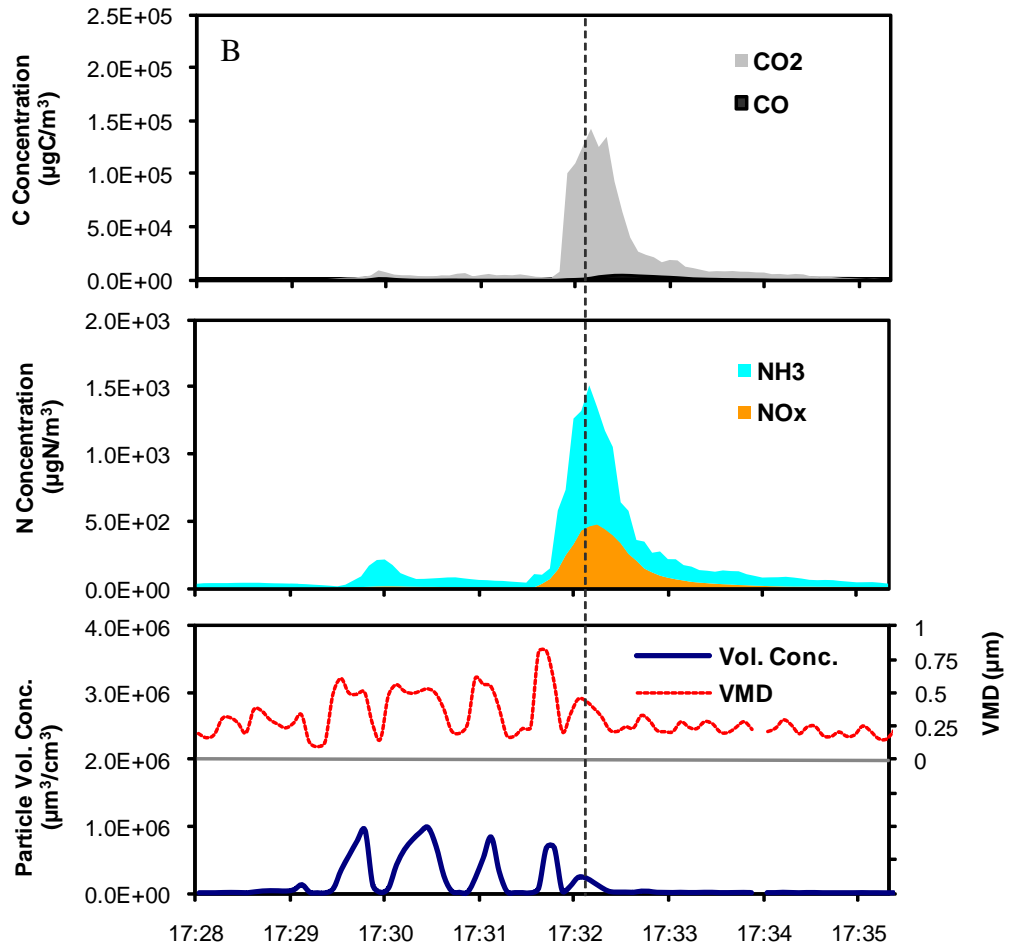


Figure 17. Time series of gaseous (stacked area) and particulate concentrations during the burn of (A) dry and (B) wet manzanita leaves. Particle size is represented by the volume mean diameter (VMD, i.e., D_v).

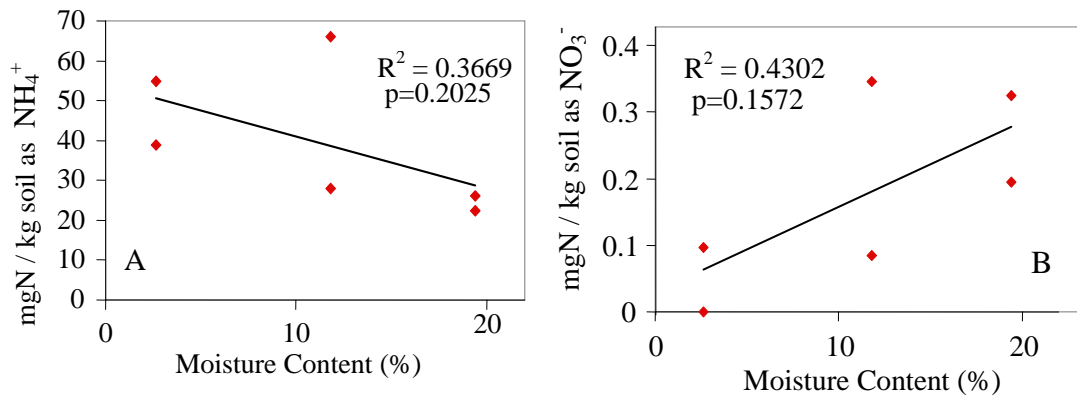
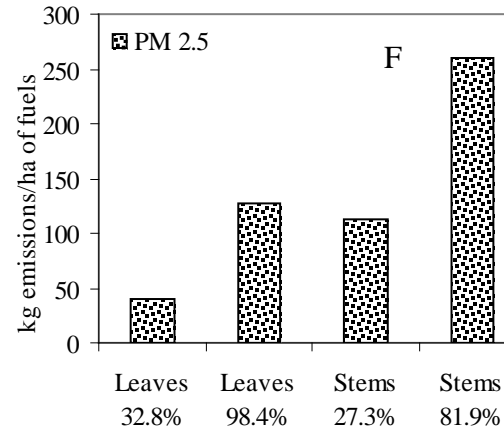
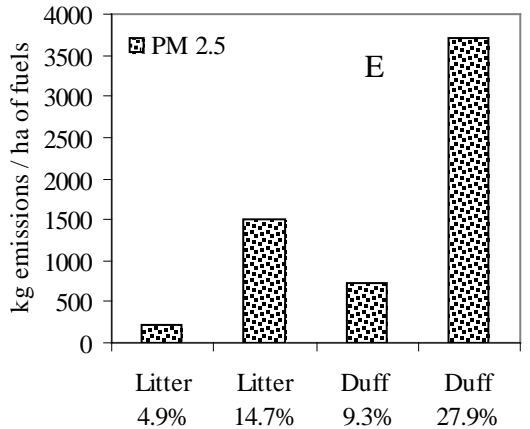
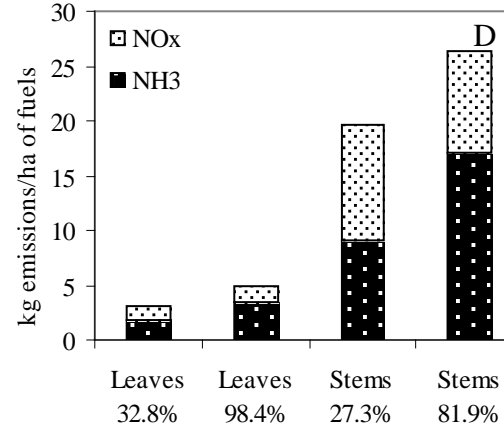
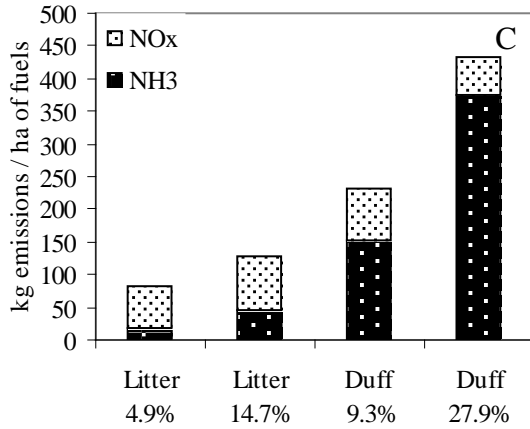
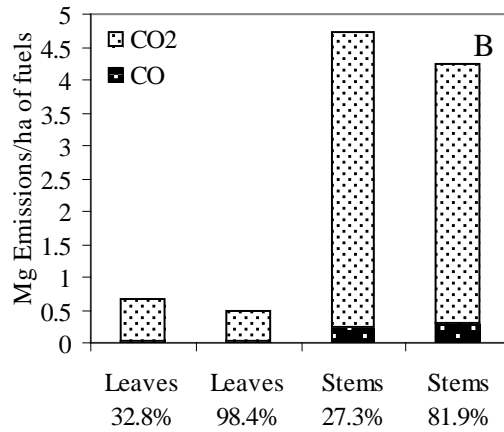
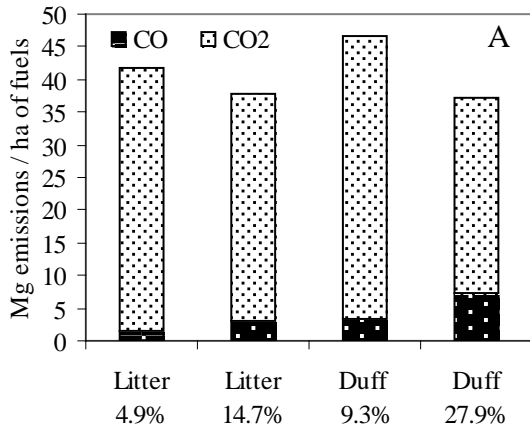


Figure 18. Soil extractable NH_4^+ (A) and NO_3^- (B) concentrations after being heated in the laboratory.



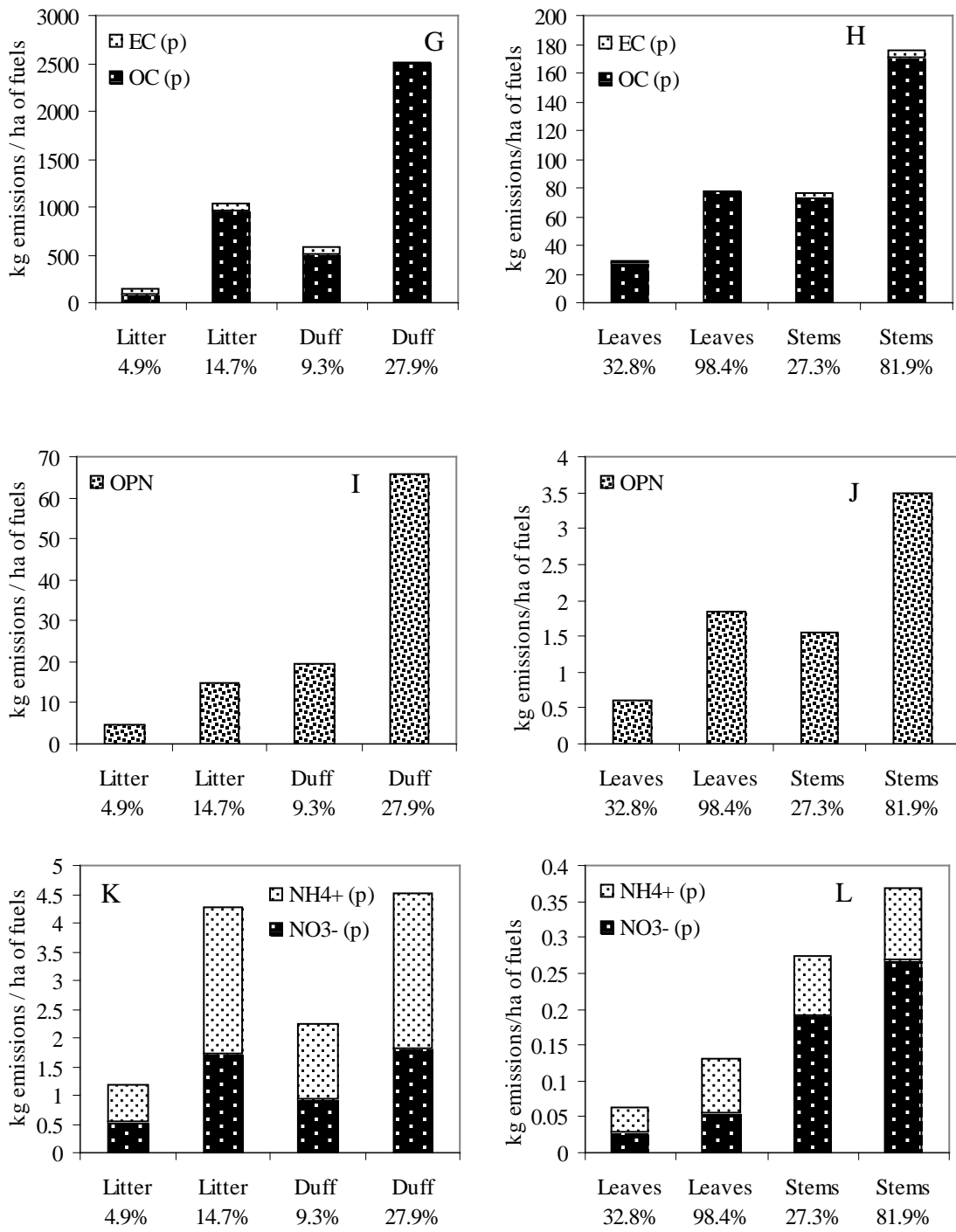


Figure 19. Atmospheric emissions of the most abundant C and N species for the dry fall conditions, and at triple the moisture content to simulate a moist spring burn.

Appendix 1

Fuel load values (Mg/ha) before and after fire in the burn plot and differences occurring between the pre- and post- sampling dates are located in Tables 1.1 and 1.2. Table 1.1 corresponds to the burn plot, and Table 1.2 corresponds to the control plot. The values listed in Table 1.1a and 1.2a were collected before the burn, tables 1.1b and 1.2b were collected after the burn, or date of treatment, and Tables 1.1c and 1.2c display changes in fuel load. A positive (+) sign indicates increases as they were observed in the control plot. All other values should be interpreted as either the amount of mass, the amount of volatilization (Table 1.1c), or the change in mass (Table 1.2c). The percentages provided in parentheses indicate the percentage of consumption (Tables 1.1c and 1.2c). Values are provided from all major fuel types. Saplings and pole trees were not encountered in the vegetation plot analysis, and are considered to contribute less to overall emissions than any individual species of standing shrub. Pre-burn biomass measurements were taken at different locations than post-burn measurements, and calculations in the Brown (1974) protocol essentially averaged fuel load measurements. Thus, fuel load loss can be estimated, but changes in biomass cannot be determined as “significant” or “insignificant”.

Table 1.1a. Biomass, C, and N
plot pre-treatment

Burn Plot Pre	Fuel Load Mg/ha	Carbon Mg/ha	Nitrogen Mg/ha
Duff	54.34	20.00	0.48
Litter	27.25	13.94	0.13
Downed Debris	81.15	38.52	0.36
Standing Shrubs	3.49	1.67	0.05
Total	166.23	74.13	1.03

Table 1.1b. Biomass, C, and N
in the burn plot post-treatment

Burn Plot Post	Fuel Load Mg/ha	Carbon Mg/ha	Nitrogen Mg/ha
Duff	11.07	0.94	0.06
Litter	3.72	0.73	0.03
Downed Debris	4.76	2.27	0.02
Standing Shrubs	0.17	0.08	0.00
Total	19.72	4.02	0.11

Table 1.1c. Changes in biomass, C and N in the burn
in the burn plot resulting from fire

Burn Plot Change	Fuel Load Mg/ha (%)	Carbon Mg/ha (%)	Nitrogen Mg/ha (%)
Duff	43.27 (79.62)	19.06 (95.28)	0.42 (87.87)
Litter	23.53 (86.34)	13.21 (94.80)	0.11 (80.64)
Downed Debris	76.39 (94.14)	36.25 (94.10)	0.34 (93.97)
Standing Shrubs	3.32 (95.27)	1.59 (95.39)	0.05 (95.15)
Total	146.51 (88.14)	70.12 (94.58)	0.92 (89.46)

Table 1.2a. Biomass, C, and N
in the control plot pre-burn

Control Plot Pre	Fuel Load Mg/ha	Carbon Mg/ha	Nitrogen Mg/ha
Duff	17.64	4.31	0.14
Litter	8.32	3.70	0.05
Downed Debris	8.96	4.39	0.04
Standing Shrubs	12.30	5.84	0.18
Total	47.22	18.25	0.40

Table 1.2b. Biomass, C, and N
in the control plot post-burn

Control Plot Post	Fuel Load Mg/ha	Carbon Mg/ha	Nitrogen Mg/ha
Duff	38.07	10.39	0.33
Litter	7.98	3.97	0.04
Downed Debris	12.64	6.21	0.05
Standing Shrubs	11.35	5.38	0.16
Total	70.04	25.95	0.57

Table 1.2c. Changes in Fuel Load, C and N
between the pre- and post- sampling dates

Control Plot Changes	Biomass Mg/ha (%)	Carbon Mg/ha (%)	Nitrogen Mg/ha (%)
Duff	+20.43 (+115.82)	+6.08 (+141.07)	+0.19 (+135.71)
Litter	0.34 (-4.09)	+0.27 (+7.30)	-0.01 (-20.00)
Downed Debris	+3.68 (+41.07)	+1.82 (+41.46)	+0.01 (+25.00)
Standing Shrubs	0.95 (-7.72)	+0.46 (-7.88)	-0.02 (-11.11)
Amount Lost	+22.82 (+48.33)	+7.70 (+42.19)	+0.17 (+42.5)

Appendix 2. Fuel loading for additional plots slated for prescription fire
as of Spring 2010.

	1 st Creek Mg/ha	2 nd Creek Mg/ha	Tyner North Mg/ha	Tyner South Mg/ha
Woody Debris	89.31	10.69	15.77	17.36
Standing Shrubs	0.74	0.20	6.37	4.46
Duff	76.18	143.83	35.24	18.22
Litter	8.70	11.95	7.01	7.01
Total	174.93	166.67	64.39	47.06

Appendix 3. Relationships between atmospheric emissions and moisture content for various fuels.

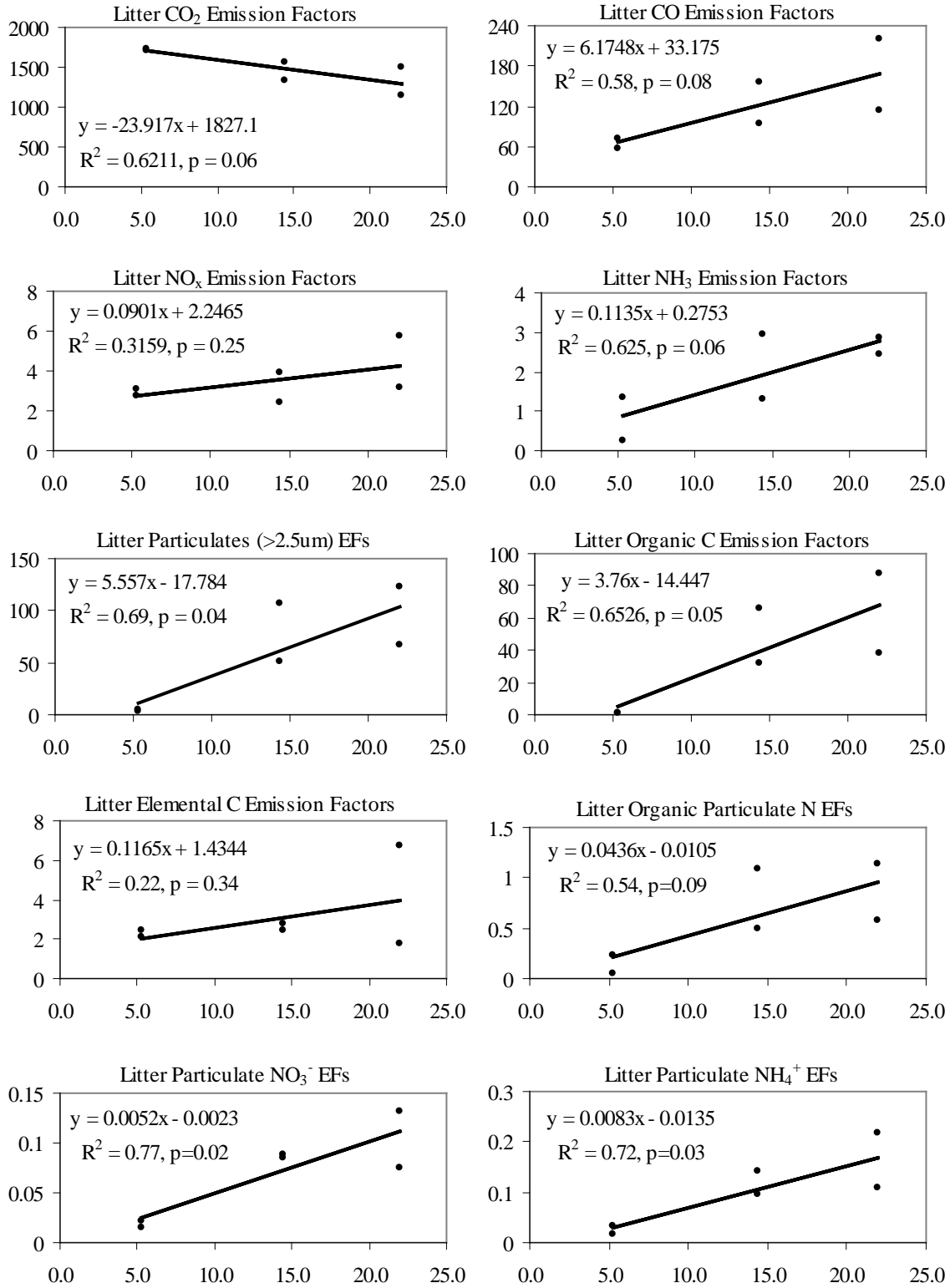


Figure A. Litter Emission Factors. Moisture content in percent is shown on the x-axis, and g emission per kg fuel is shown on the y-axis.

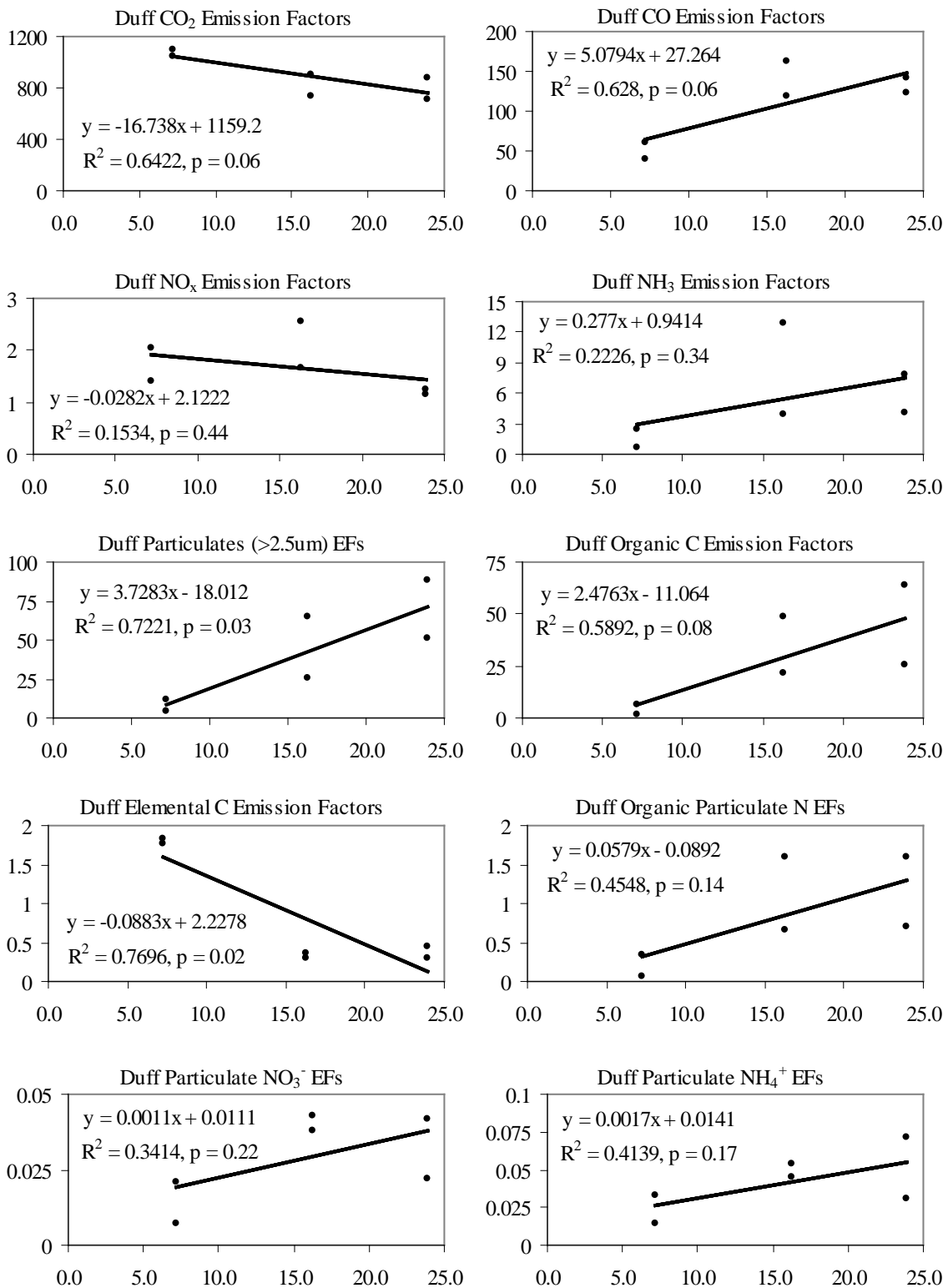


Figure B: Duff Emission Factors. Moisture content in percent is shown on the x-axis, and g emission per kg fuel is shown on the y-axis.

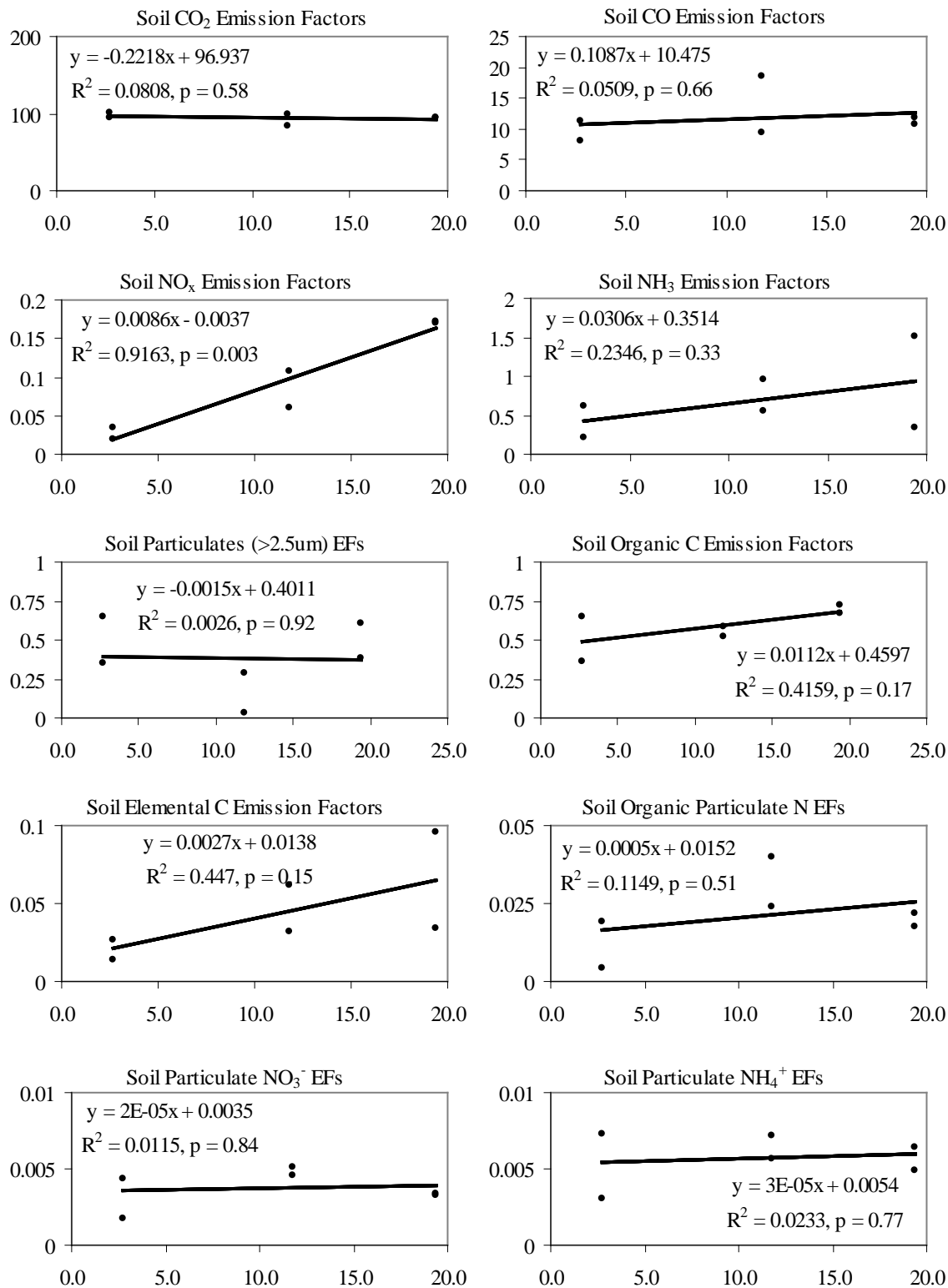


Figure C: Soil Emission Factors. Moisture content in percent is shown on the x-axis, and g emission per kg fuel is shown on the y-axis.

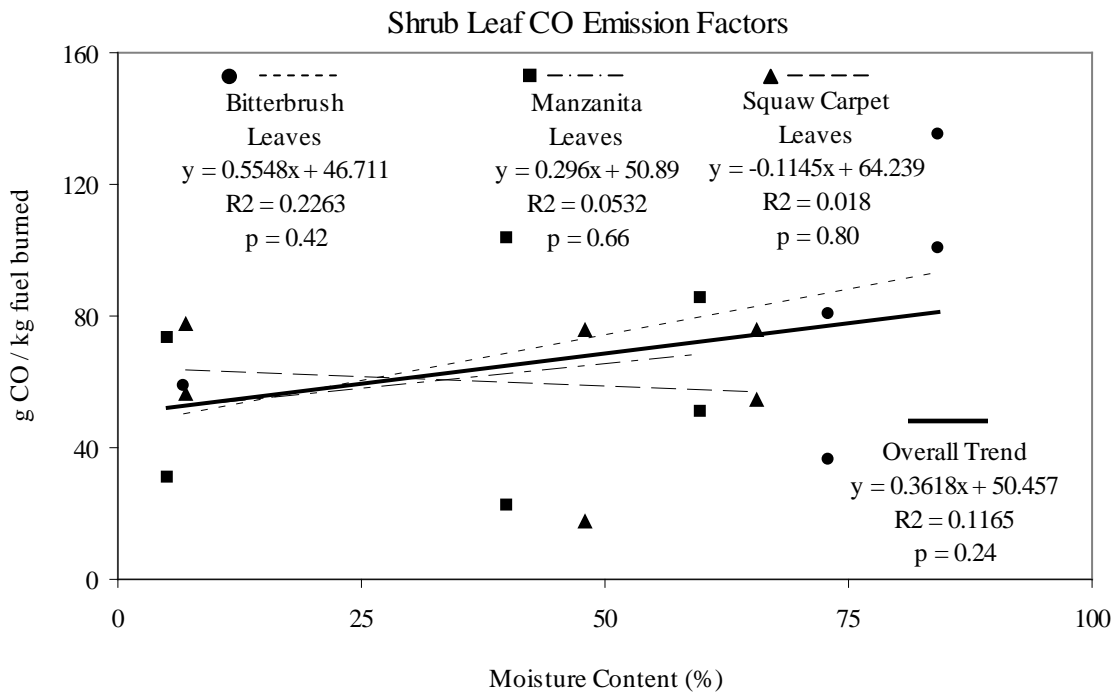
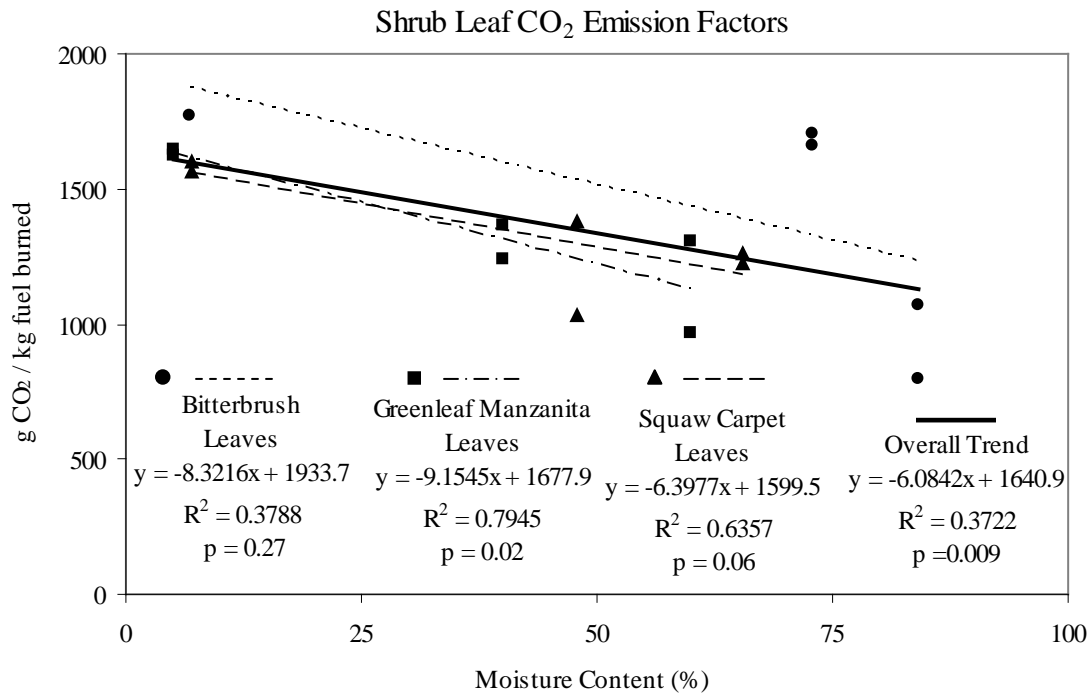


Figure D. CO₂ and CO emission factors generated from the burning of shrub leaves. The linear relationship between moisture content and emission factors is provided for each species and for shrub leaves overall.

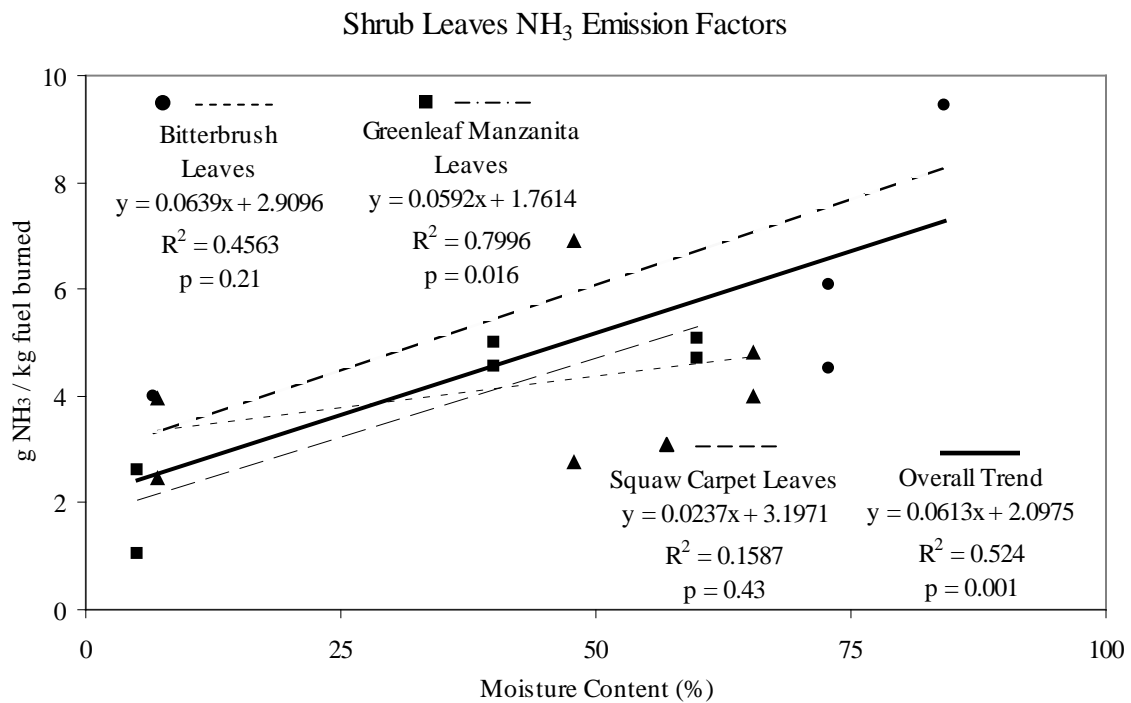
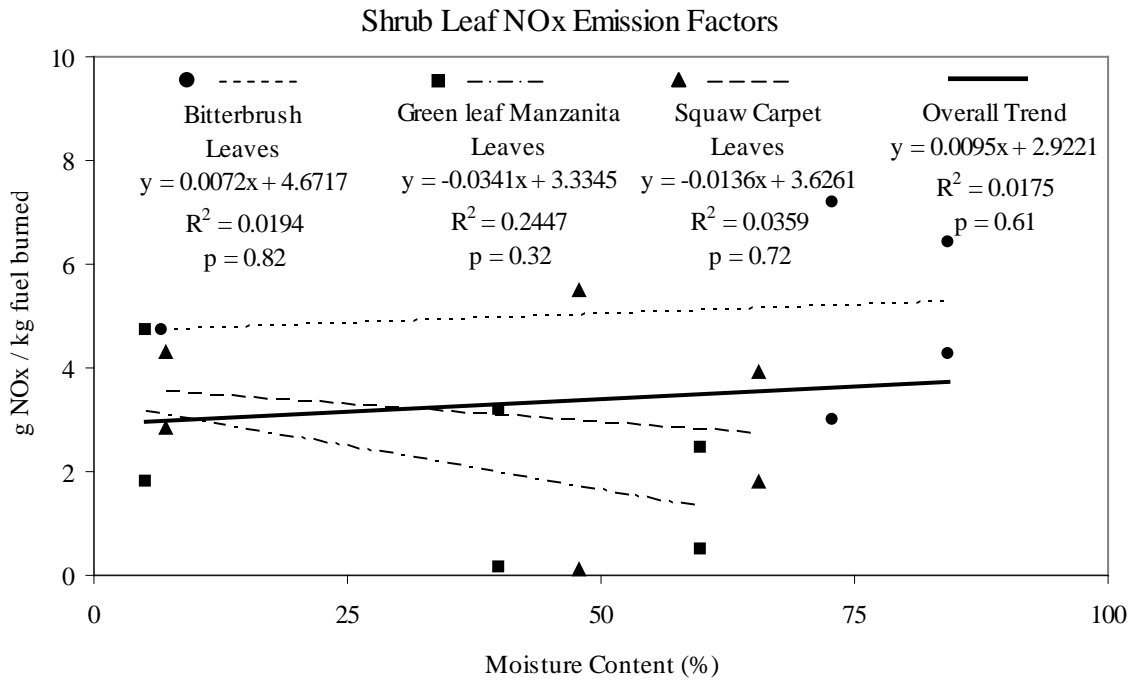


Figure E. NO_x and NH₃ emission factors generated from the burning of shrub leaves. The linear relationship between moisture content and emission factors is provided for each species and for shrub leaves overall.

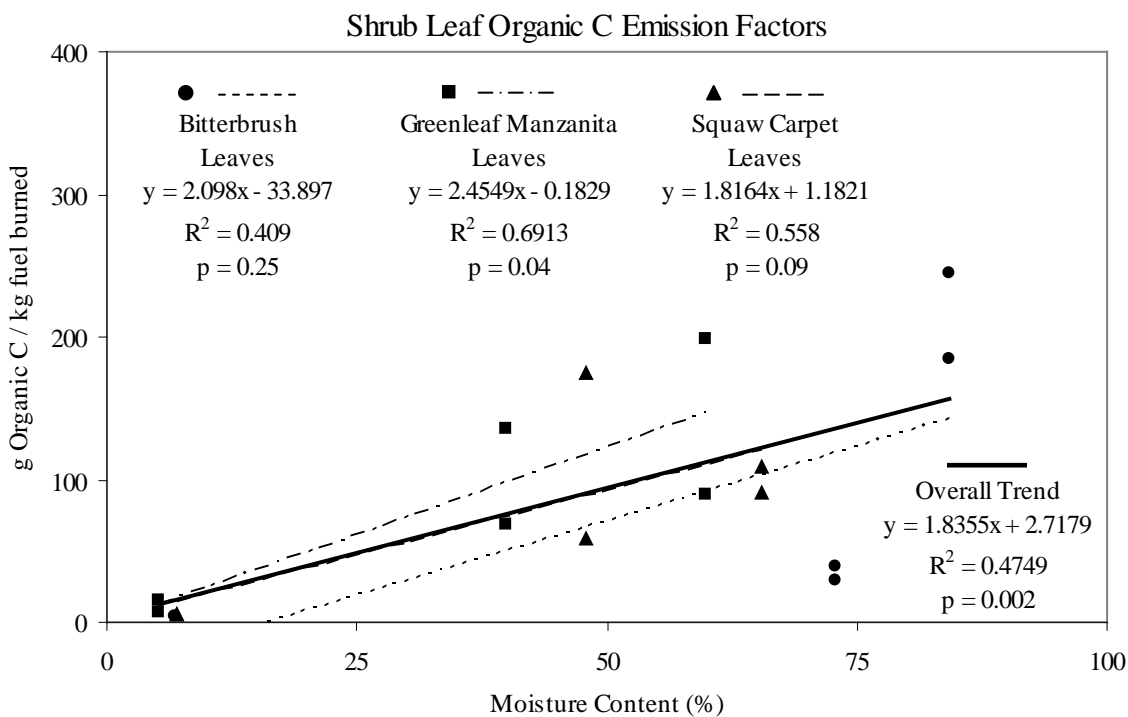
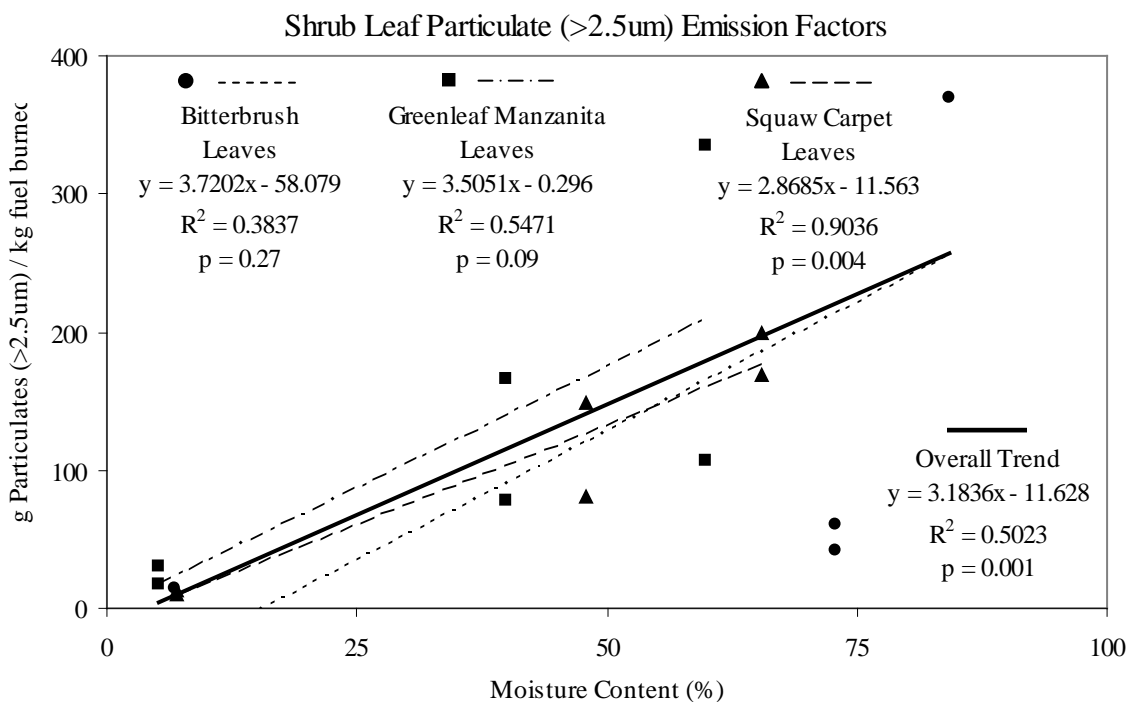


Figure F. Total particulates (>2.5um) and particulate organic C emission factors generated from the burning of shrub leaves. The linear relationship between moisture content and emission factors is provided for each species and for shrub leaves overall.

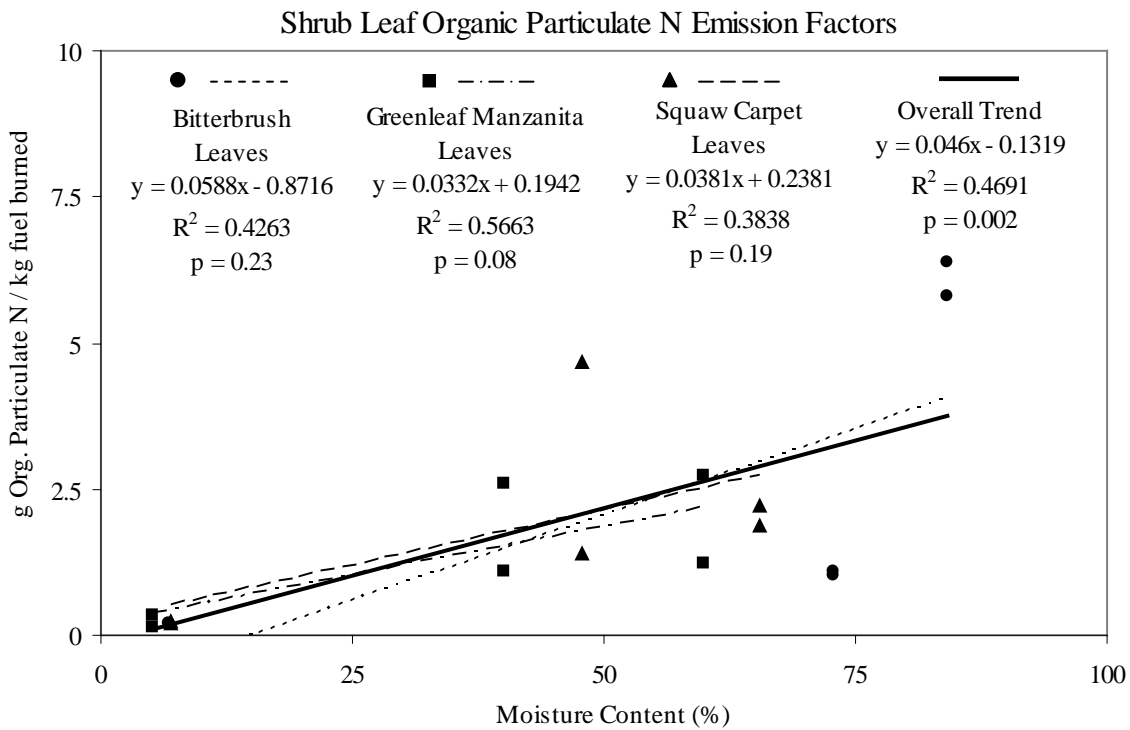
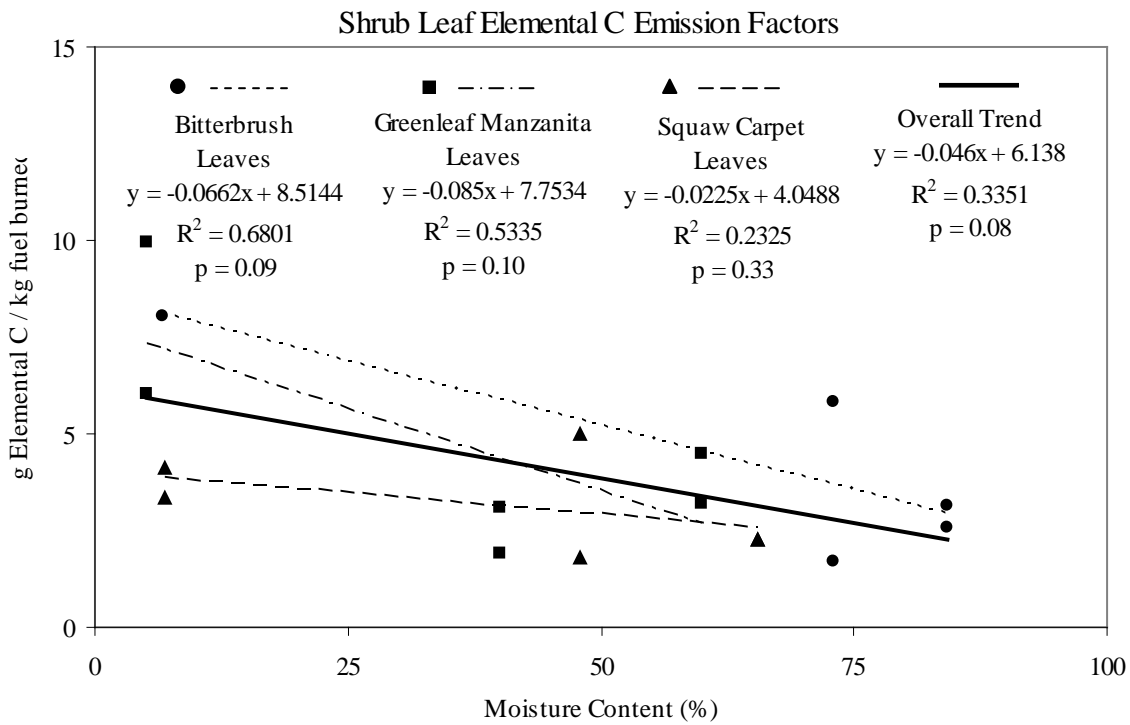


Figure G. Particulate elemental C and organic particulate N emission factors generated from the burning of shrub leaves. The linear relationship between moisture content and emission factors is provided for each species and for shrub leaves overall.

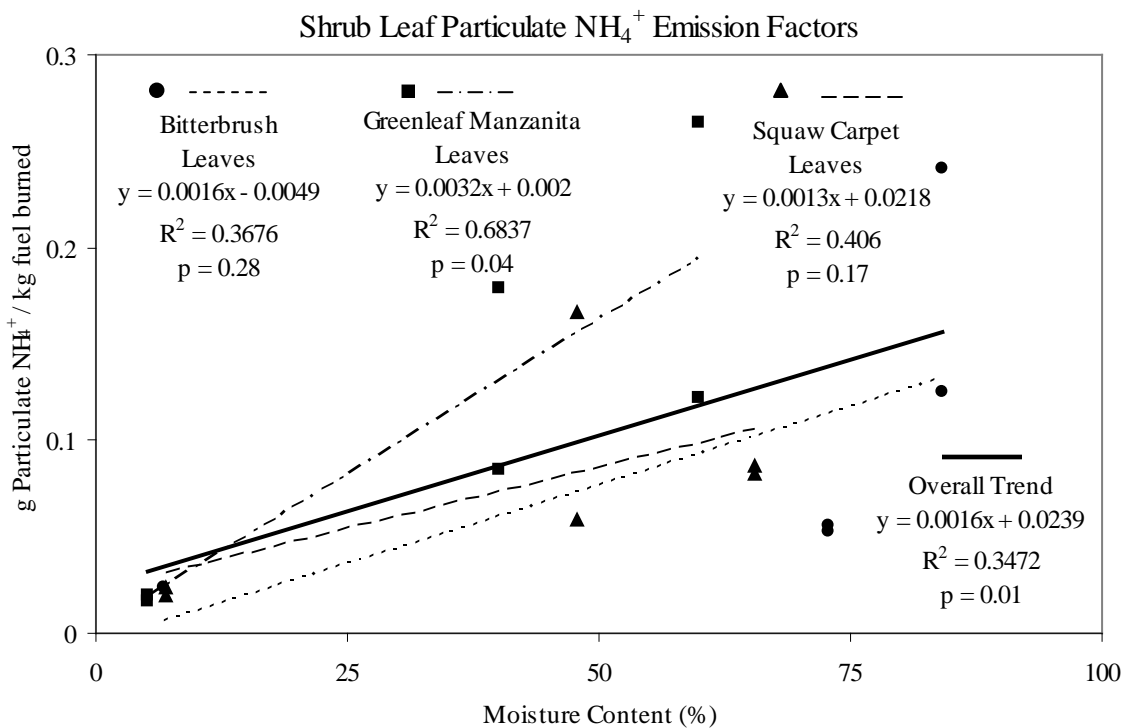
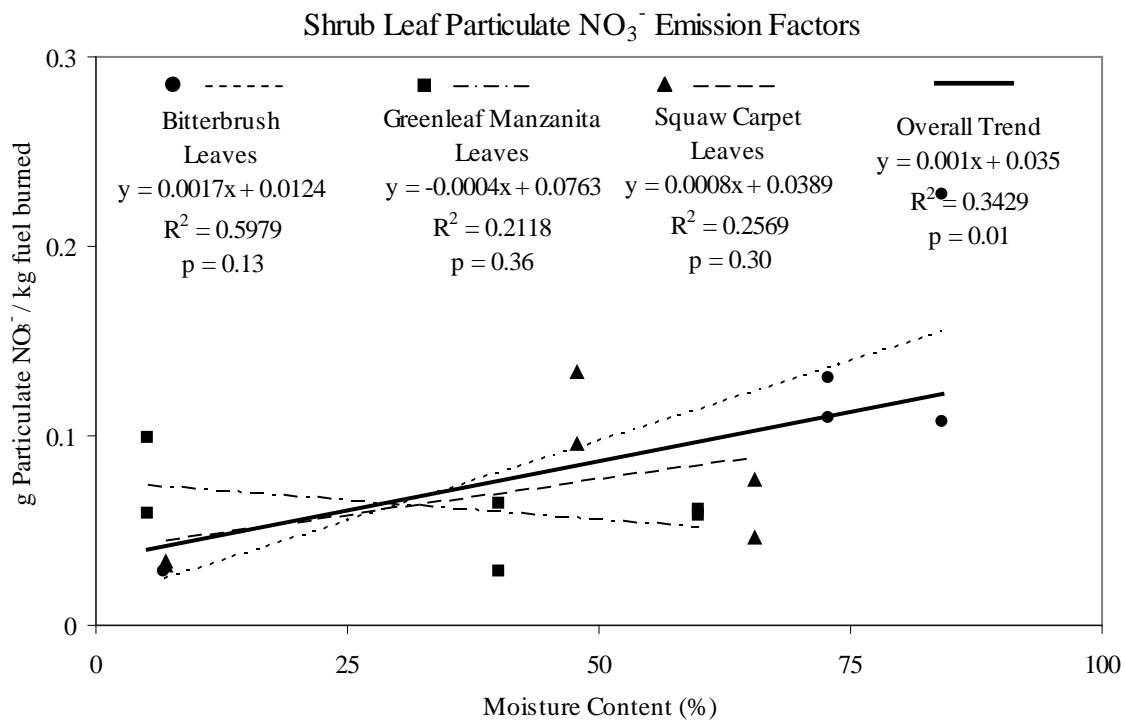


Figure H. Particulate NH₄⁺ and particulate NO₃⁻ emission factors generated from the burning of shrub leaves. The linear relationship between moisture content and emission factors is provided for each species and for shrub leaves overall.

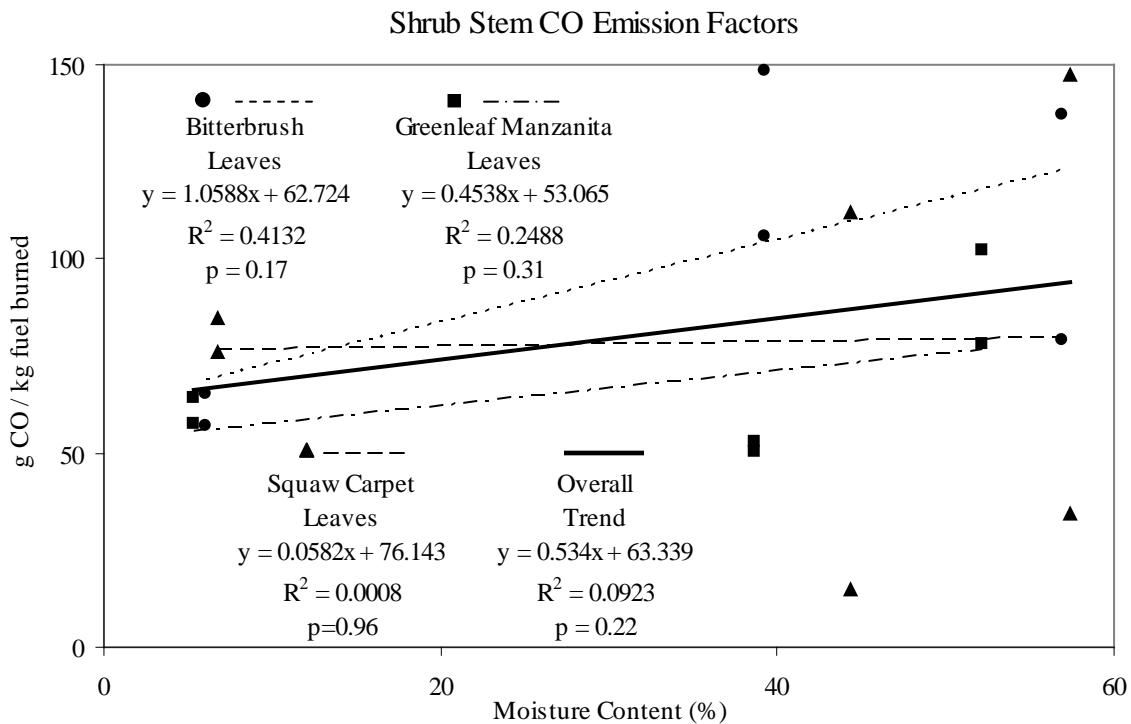
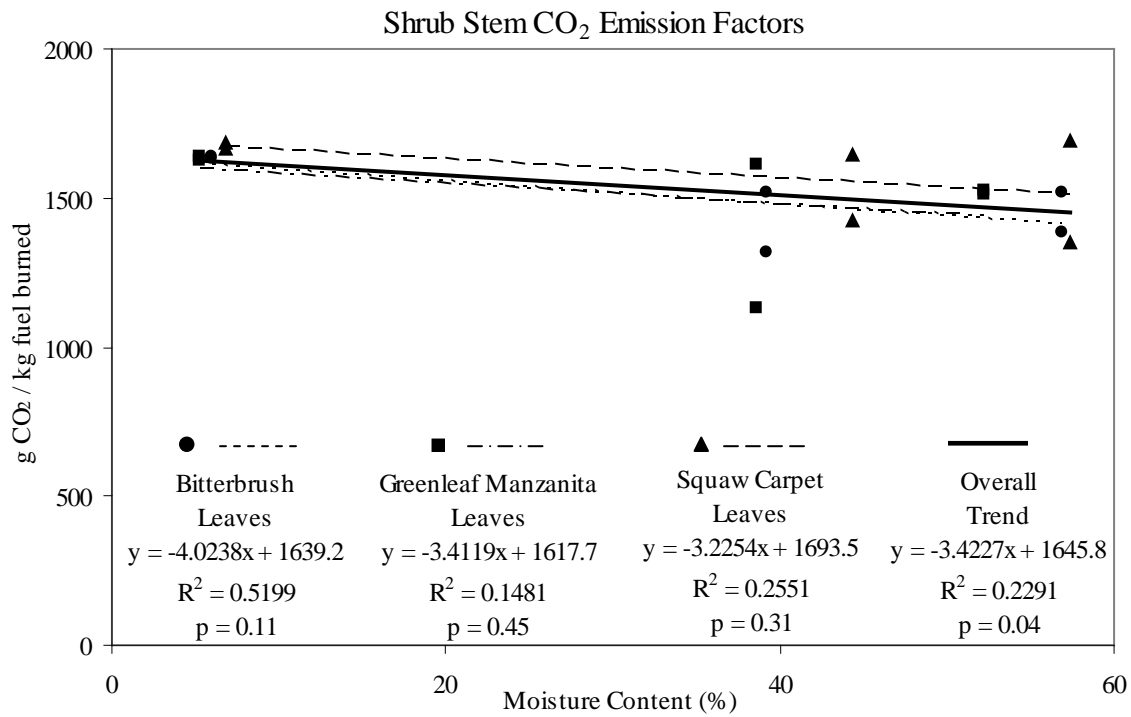


Figure I.CO₂ and CO emission factors generated from the burning of shrub stems. The linear relationship between moisture content and emission factors is provided for each species and for shrub stems overall.

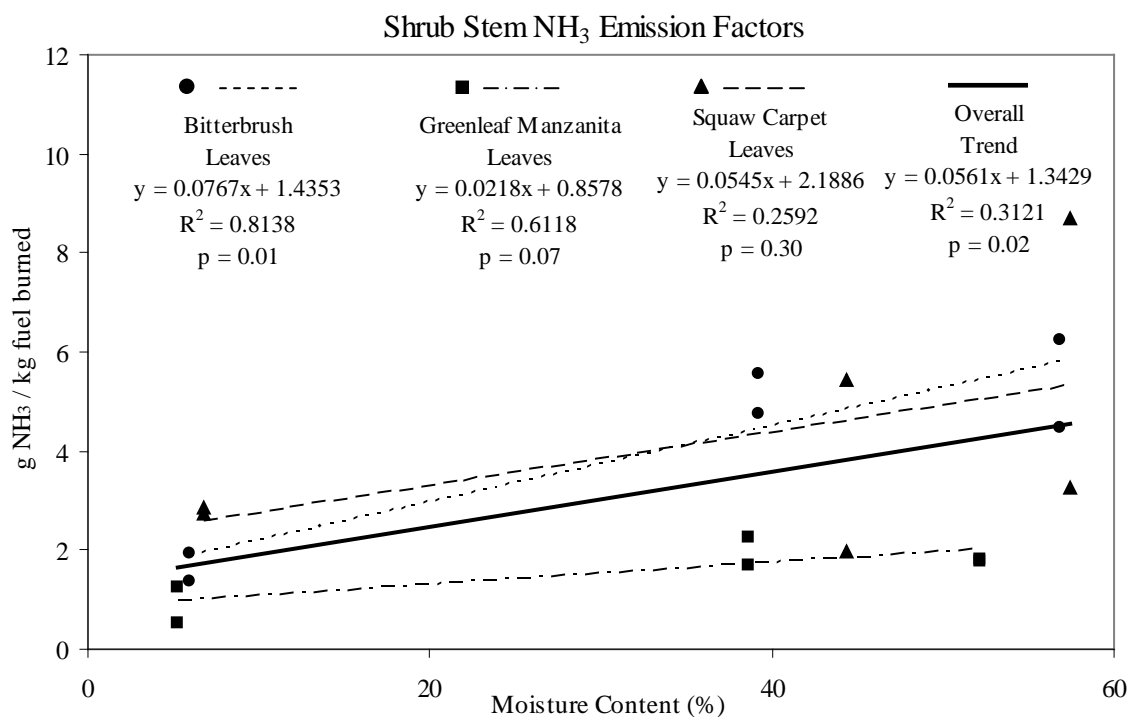
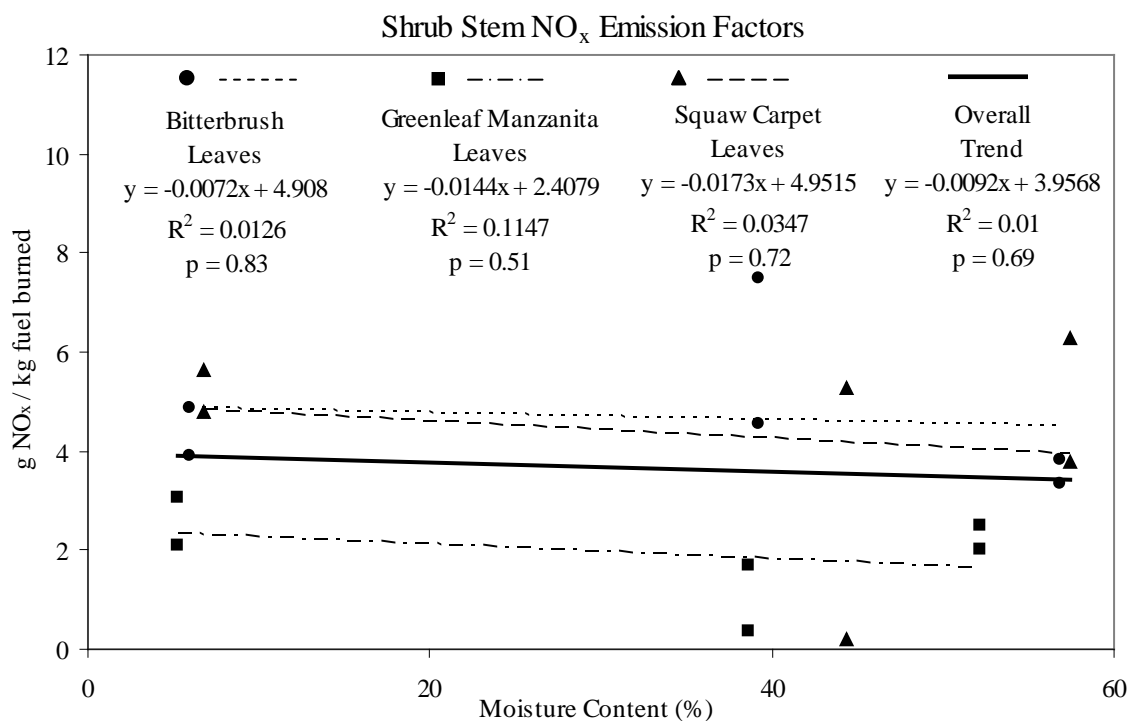


Figure J. NO_x and NH₃ emission factors generated from the burning of shrub stems. The linear relationship between moisture content and emission factors is provided for each species and for shrub stems overall.

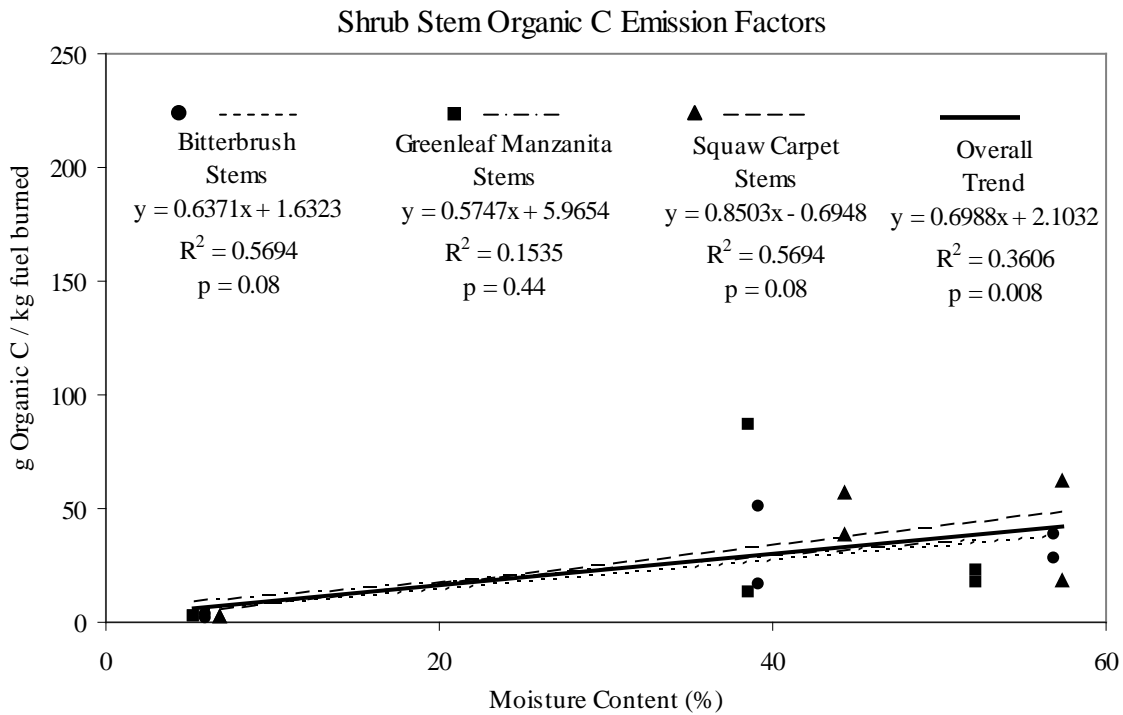
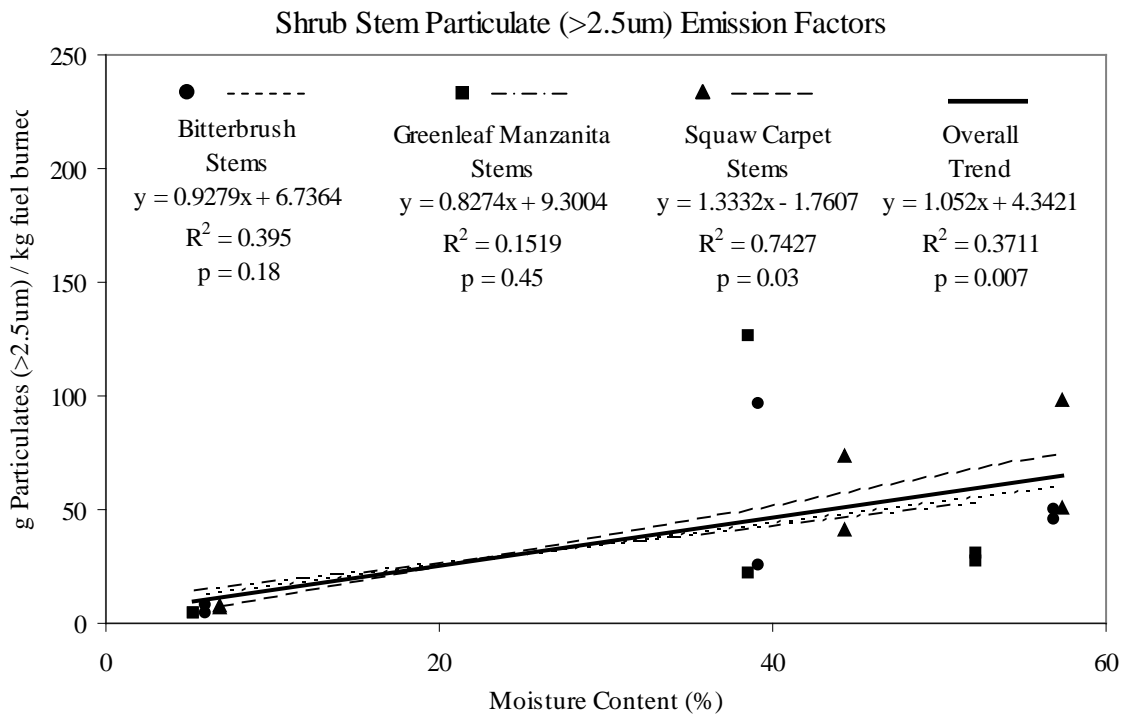


Figure K. Total Particulates (>2.5um), and particulate organic C emission factors generated from the burning of shrub stems. The linear relationship between moisture content and emission factors is provided for each species and for shrub stems overall.

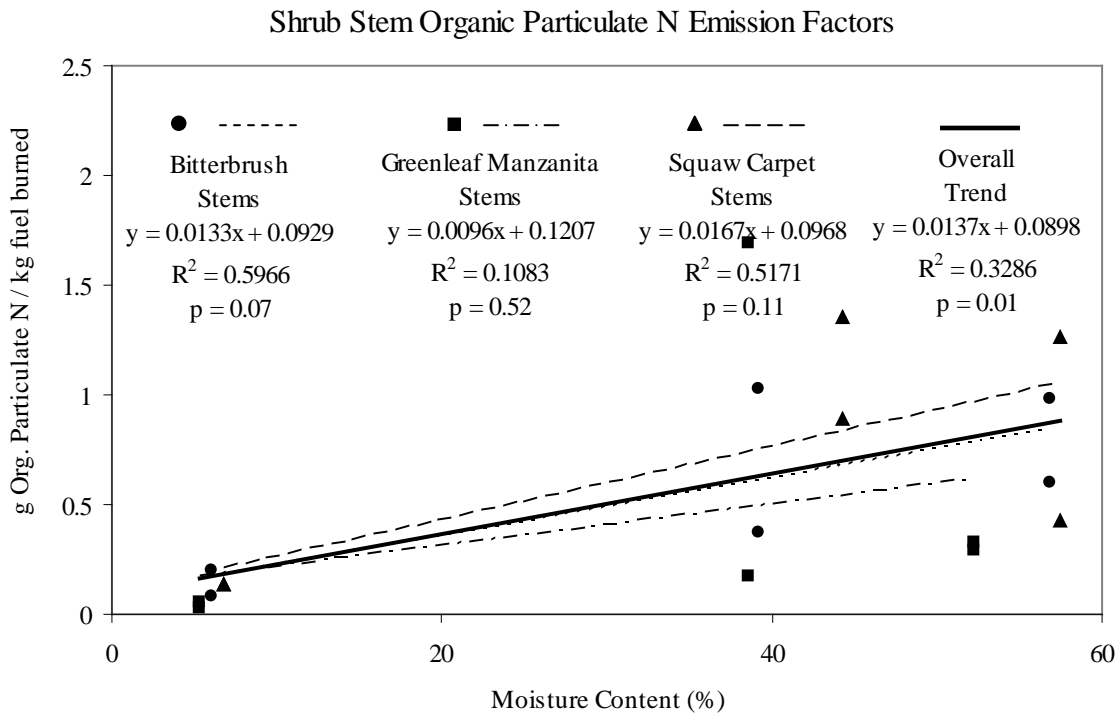
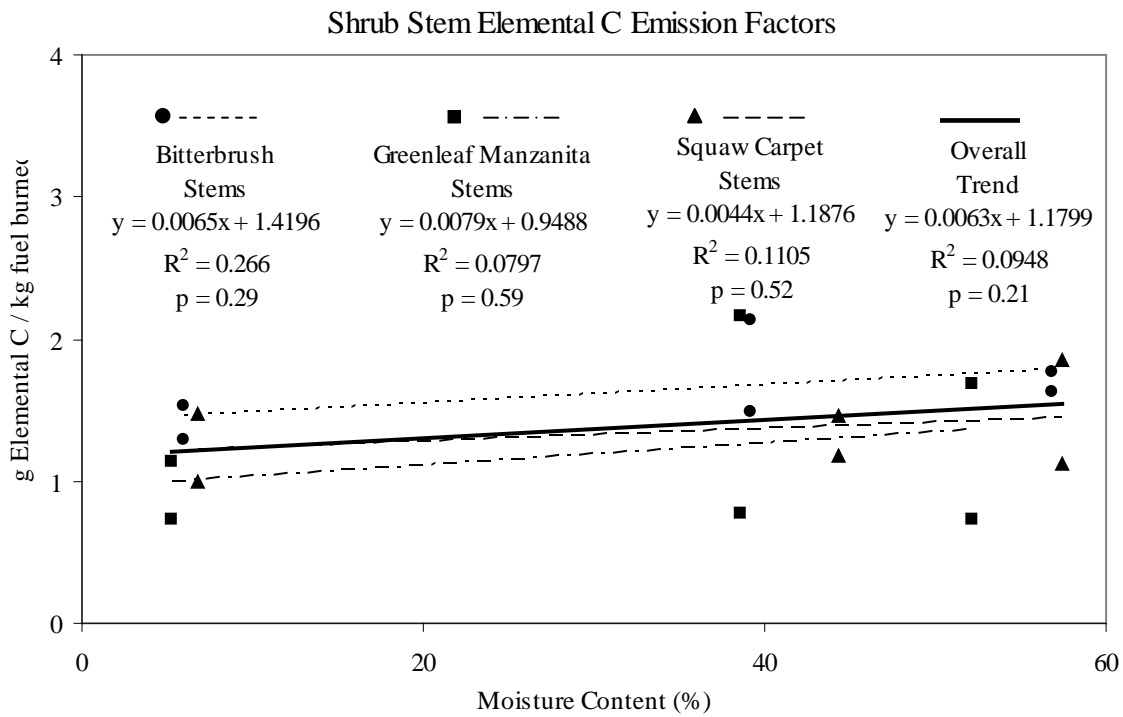


Figure L. Particulate elemental C and organic particulate N emission factors generated from the burning of shrub stems. The linear relationship between moisture content and emission factors is provided for each species and for shrub stems overall.

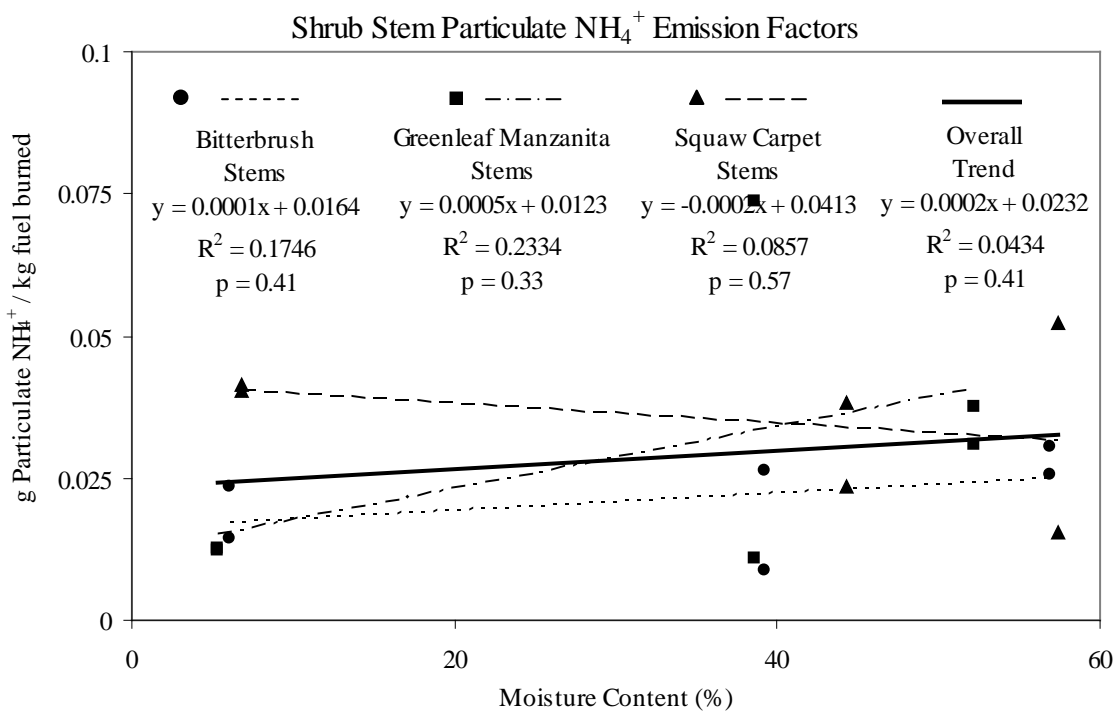
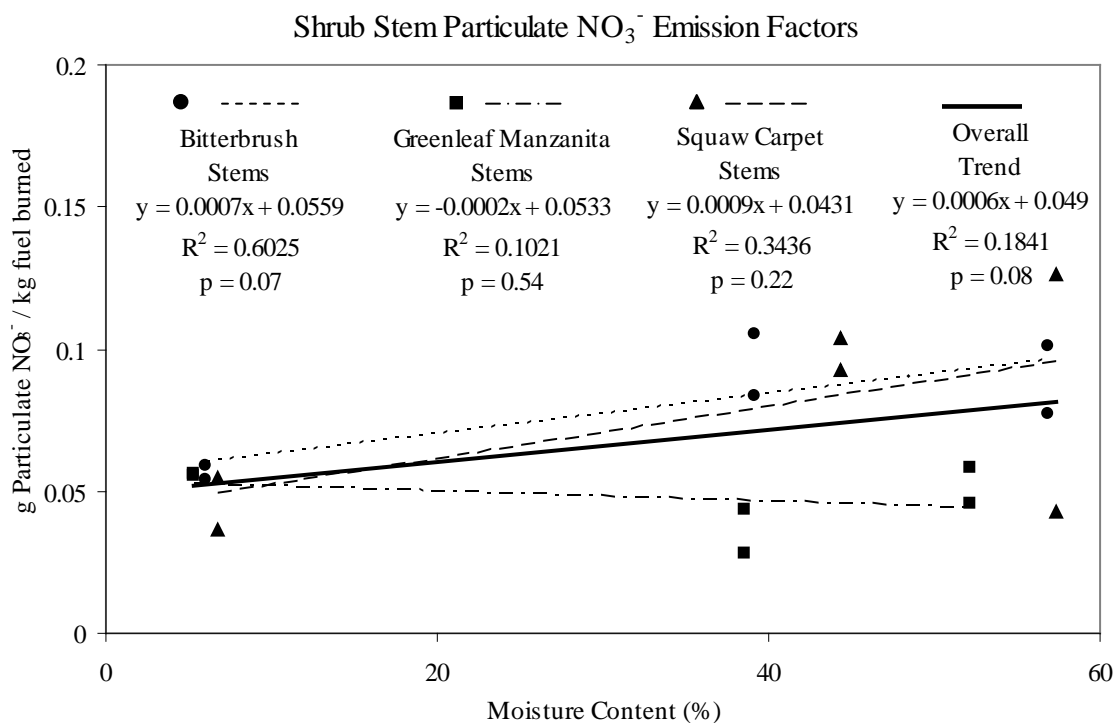


Figure M. Particulate NH₄⁺ and particulate NO₃⁻ emission factors generated from the burning of shrub stems. The linear relationship between moisture content and emission factors is provided for each species and for shrub stems overall.