THESIS

CARBON AND NITROGEN ERODED FROM BURNED FORESTS IN THE WESTERN U.S.

Submitted by

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ABSTRACT

CARBON AND NITROGEN ERODED FROM BURNED FORESTS IN THE WESTERN U.S.

Post-wildfire landscapes and downstream aquatic resources are influenced by carbon (C) and nitrogen (N) losses from soil erosion. As opposed to soil erosion, rarely measured losses of sediment C and N may account for a substantial portion of fire impacts. We measured erosion of C and N following eight wildfires for four to six years in the western U.S and compared losses from untreated, burned hillslopes and small catchments with those from adjacent areas that received erosion mitigation treatments. Losses of C, N and sediment were greatest the first two years and declined in subsequent years. Cumulative losses from untreated, burned areas were 16 – 4,700 kg C/ha and 0.7 – 185 kg N/ha over the study period. Across wildfire locations, median sediment C and N concentrations ranged from 0.011 – 0.036 g N/kg and 0.23 – 0.98 g C/kg. Post-fire erosion control treatments reduced C, N and sediment losses by 65-75% compared to untreated areas and generally increased the concentrations of C and N in eroded material. The total C and N lost in post-fire erosion was < 20% of the estimated amount lost from organic and mineral soil layers during combustion and < 5% of the estimated amount remaining in mineral soils after combustion. In general, the N lost with soil erosion is unlikely to impair the productivity of recovering forests, but the eroded N may have consequences on downstream water quality and aquatic habitat.

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INTRODUCTION

Large increases in erosion following forest fires are a concern for burn site productivity, as well as downstream water quality and aquatic habitat (Moody, 2001). Erosion increases are common after wildfire due to the loss of runoff and sediment interception materials (canopy, litter, duff, and organic debris), as well as decreased infiltration rates resulting from fire induced changes to soil properties (Gresswell, 1999). Episodic, intense rainfall events, such as those common to climates in the western U.S., often produce flash floods and debris flows which can transport tons of sediment downstream in a matter of hours (Helvey, 1980; Neary, 2005). Increases in sediment production alter aquatic environments through physical mechanisms (Ewing, 1996; Kershner, 2003); less is known in regards to the chemical effect of nutrients associated with eroded sediments. Nitrogen (N) is a key element in regards to post-wildfire recovery, water quality and aquatic habitats. To investigate the environmental effect of post-wildfire sediment N transport it is necessary to first quantify the magnitude of sediment N loss following wildfire. Quantifying sediment carbon (C) transport is also important in relation to post-wildfire soil biotic processes and forest productivity (Pietikainen, 2000; Zackrisson, 1996). Previous studies have investigated post-wildfire sediment transport and the effect of wildfire on soil carbon and nutrients (Bormann, 2008; Grier, 1975; Johnson, 2012), but few studies have quantified post-wildfire sediment C and N transport.

The size and environmental influence of post-wildfire C and N transport due to erosion is difficult to predict due to the complexity of post-wildfire sediment transport mechanisms and post-fire soil nutrient concentrations. The erosive response following a wildfire is dependent upon sediment availability, size and severity of the burn, precipitation regime, terrain, and wildfire frequency (Moody, 2001). As a result of differences between these factors, estimates of post-wildfire sediment yields often vary across time and between fires by as much or more than two orders of magnitude (Debano, 1998; Groen, 2008; Riechers, 2008). Post-wildfire soil C and N concentrations may also differ within and between fire locations due to differences between pre-existing nutrient pools and the extent to which those pools are exposed during the fire (Caldwell, 2002; Dyrness, 1989; Hormann, 2011; Raison, 1985;). Given the variability associated with post-wildfire sediment C and N transport, reports associated with a single wildfire may not be relevant in alternate wildfire cases.

Erosion mitigation treatments are often applied to areas where post-wildfire runoff and sediment are a threat to life, property, water quality and/or aquatic biota. Sediment yields are typically greatest in high burn severity areas where the majority of vegetation and soil organic matter is consumed during the fire. Straw mulch, hydromulch, log erosion barriers and wood strand mulch are common treatments which may be applied throughout high risk areas to limit runoff and erosion. The effectiveness of these treatments at reducing post-wildfire erosion has been studied previously (Riechers, 2008; Groen 2008; Robichaud, 2013). However, their effect on post-wildfire sediment C and N concentrations and the associated losses of sediment C and N is unknown.

This study investigates the concentration and loss of C and N in eroded sediments following eight wildfires in the central and northern Rocky Mountains of the western U.S. The primary objectives of this study are to quantify post-wildfire sediment C and N losses and determine how C and N concentrations and losses differ across time, between locations, and as a result of applied erosion mitigation treatments.

METHODS

Eight wildfire sites previously established by the U.S.D.A. Forest Service (Robichuad, 2008; Robichaud, 2013a; Robichaud, 2013b) were selected for this study (Figure 1). Sites were located within areas of high burn severity. High burn severity was identified by the complete consumption of the organic layer across >70% of a given area (Parsons, 2010). Four of the fire sites were equipped with silt-fence barriers to measure erosion from hillslope plots (~20-250 m2 contributing area), while five sites contained catchment basins to quantify the sediment output from small, individual watersheds (~3-13 ha contributing area). The Hayman site was equipped with hillslope plots as well as two separate sets of watershed catchment basins. At all sites, both hillslope plots and watershed basins were constructed in adjacent locations to compare erosion rates between untreated, burned areas and areas where erosion mitigation treatments were applied (Table 1; Robichuad, 2008; Robichaud, 2013a; Robichaud, 2013b). At hillslope sites, each mitigation treatment was applied to 3-9 replicate plots which were constructed along the same hillslope as the untreated plots. For watershed sites, treatment comparisons were made using untreated and treated catchment basins located in adjacent watersheds with similar contributing areas.

Pre-fire overstory for the majority of sites was a mix of coniferous species (Table 2). Vegetative understory was a diverse mix of grass and shrub species. Soil texture varied between sites from sandy loam/loamy sand to silt loam. Average annual precipitation for individual sites ranged from 300 – 1380 mm. During the study, the majority of sites experienced 3-10 sediment-producing storm events; the Hayman site experienced > 17 sediment producing events during the study (Table 2).

Silt fences and catchments were setup within 2-3 months after each wildfire and sediments were collected for a period of four to six years. Eroded sediments were collected from catchments during the spring and fall. On occasion, sediments were also collected after summer sediment-producing storms. Prior to sediment excavation, noticeable organic material, such as treatment particles, leaves, and woody debris were removed. All sediments collected during the study, including those collected in the spring following snow melt, were included in the analysis. Sediment deposits weighing <1000 kg were removed with buckets and weighed on-site (0.2 kg increment, 50 kg max). For sediment deposit weighing >1000 kg, sediments were removed with mechanical equipment and an estimate of the sediment deposit weight was made based on the total sediment volume and the sediment bulk density (Core method; Blake and Hartge, 1986) During the removal process, subsamples of sediment for laboratory analysis were taken from each filled bucket or mechanical equipment load.

Sediment samples were dried for 24 hours at 100 °C to determine gravimetric water content and the total dry weight of the sediment deposit. All composite samples were homogenized and passed through a 2 mm mesh sieve prior to the C and N analysis. Sediment carbon and nitrogen content was determined by dry combustion (Nelson et al., 1996) on a Leco TruSpec C & N determinator (St. Joseph, MI, USA). Total dry sediment weight for each plot and catchment was divided by contributing area to calculate the hillslope or watershed sediment yield. For watershed catchments, the C and N yield from each cleanout was determined by the product of the sediment yield and the determined C and N concentration of the composited sediment subsamples. For hillslope scale sites, the average sediment yield from replicate plots of the same treatment type was multiplied by the C and N content of the composited subsamples taken during the plot cleanouts.

Sediment C and N concentrations and sediment, C, and N yields were compared over time, across treatment types, and between sampling scales (hillslope or watershed) in a generalized linear mixed-effects model. Treatment Type, Post-wildfire Year, and Catchment Type were used as fixed effects within the model. Site was used as the random effect. Catchment*Treatment Type within Site was also used as a random effect to account for variation between plots with different catchment and treatment combinations. The model incorporated a repeated measures structure on the residuals to account for measurements taken from the same plots over time. A lognormal distribution, suitable for data that cannot be < 0, was used in the model for yield comparisons. The model used a beta distribution for the comparison of C and N concentrations. Yield and concentration differences were compared using the least squares mean estimates with a Tukey-Kramer adjustment to account for uneven sample sizes. Significance for this study was defined as $p \leq 0.05$.

RESULTS

Sediment yields from untreated areas were typically greatest during the first two post-wildfire years and declined substantially in subsequent years (Figure 2). Erosion occurring after the second post-wildfire year typically accounted for <10% of the cumulative sediment yield during the 4-6 year study period. Across sites, the median sediment yield through the first post-wildfire year was 1.9 Mg/ha with a range of 0.1 – 64.3 Mg/ha. The median cumulative sediment yield over the entire study period was 3.8 Mg/ha with a range of 2 – 125 Mg/ha.

The median sediment C and N loss among sites during the study was 9.9 kg N/ha and 160 kg C/ha. Across all sites, total losses ranged from 0.7 – 185 kg N/ha and 16 – 4,700 kg C/ha. Losses occurring through the end of the first post-wildfire year ranged from 0.03 – 128 kg N/ha and 0.7 – 3,300 kg C/ha (Figure 5). Yields measured at the Hayman sites were far greater than yields observed in other locations. In the first two post-wildfire years, the untreated Hayman hillslope and watershed sites lost an average of 62 Mg/ha of sediment which contained 150 kg N/ha and 3,700 kg C/ha. Across study sites, no significant difference between carbon and nitrogen yields was found between sediments collected from hillslope and watershed scale sites ($p = 0.69$).

The median C and N concentration for sediments collected from untreated sites during the study spanned from 0.011 to 0.036 g N/kg and 0.23 to 0.98 g C/kg. (Figure 3). Sediments from hillslope sites had significantly greater carbon and nitrogen contents than those from watershed scale sites ($p < 0.04$). The median nitrogen concentration for sediments collected from hillslope and watershed scale sites was 0.034 g N/kg and 0.025 g N/kg respectively. The median carbon concentration for sediments from hillslope and watershed scale sites was 0.81 g C/kg and 0.50 g C/kg respectively. Sediment C and N concentrations tended to decline as sediment yields increased (Figure 4). For sediment deposits weighing > 4 Mg/ha, C and N concentrations were typically less than 0.5 g C/kg and 0.015 g N/kg. For sediment deposits which weighed < 1 Mg/ha, carbon and nitrogen contents were highly variable. With the exception of the Fridley site, the C and N concentration of eroded sediments did not significantly change throughout the four to six year study period ($p = 0.33$). Following a high intensity storm event during the first post-wildfire year at Fridley, the C and N content of collected sediments declined by approximately 70% for the remainder of the study period.

A large portion of the carbon and nitrogen loss at each site often resulted from a few, high intensity storm events. The average 10-minute rainfall intensity (I10) for sediment producing storm events ranged from 23.0 – 39.7 mm hr-1 (Sites with < 3 sediment producing events were not included). Single storm events at the Fridley, Cannon and Hayman (hillslope) sites with I10's > 55 mm h-1 produced 98%, 95%, and 76% of the total sediment, carbon, and nitrogen yield for each respective site during the 4-6 year study period. At the Cannon site, the total sediment yield was < 0.14 Mg/ha before a storm event with a 100-year return interval (I10 = 134 mm h-1) occurred in the fourth post-wildfire year and produced nearly 10 Mg/ha of sediment. At Fridley, a 5-year storm event with an I10 of 55 mm/hr occurred in the first post-wildfire year and yielded approximately 21 Mg/ha of sediment. In the three years following, the Fridley site only yielded an additional 0.3 Mg/ha of sediment. High intensity rainfall events were far more frequent at the Hayman monitoring sites which coincides with the increased sediment production recorded at those sites.

The effectiveness of mitigation treatments at reducing erosion varied between sites. During the 4-6 year study period log erosion barriers, straw mulch and wood strands reduced cumulative yields in comparison to control areas by an average of 69%, 67%, and 74%

respectively (Figure 6). Hydromulch was the least effective treatment and only had a noticeable effect on sediments yields at one of the three sites in which it was applied. Compared to untreated plots, the C and N concentration of eroded sediments was significantly higher ($p < 0.001$) for plots treated with log erosion barriers, straw mulch and wood strands (Figure 7). The C and N concentration of sediments collected from plots treated with hydromulch was not significantly different from untreated plots.

DISCUSSION

Storms occurring within the first two post-wildfire years were responsible for the bulk of sediment C and N loss during the 4-6 year study period. The frequency of sediment producing storm events and the associated yields tended to be far less after the initial 1-2 year period. At seven of nine sites, losses from the first two post-wildfire years accounted for greater than 90% of the total C and N loss at each site. On average, 55% of the total N losses and 60% of the total C losses were a result of a single storm event which occurred in the first two post-wildfire years. Previous studies have shown a similar decline in sediment transport following the initial one to two year recovery period (Amaranthus, 1989; Groen, 2008; Helvey, 1985; Robichaud, 2006). These results highlight the impact of individual storm events and the importance of rapid protection for burned soils and downstream resources.

Across all sites, both the cumulative and first year C and N losses varied by more than two orders of magnitude. Previous studies have reported sediment N losses similar to those observed during this study (Baird, 1999; Bormann, 2008; Helvey, 1985). A large portion of the variability in the amount of C and N loss can be attributed to differences between sediment yields, as C and N concentrations did not differ substantially between sites. In previous studies, storm characteristics such as size, frequency and intensity have been shown to affect erosion rates (Robichaud, 2013a; Robichaud, 2013b). The large C and N losses observed at the Hayman site can be attributed to the combination of highly erodible soils, slow vegetation recovery, and a propensity for high intensity rainfall events (Robichaud, 2013b).

High intensity storms pose a threat to burned areas well beyond the first few years of fire recovery. As evidenced in this study by the 100-year storm event (I10 = 135 mm hr-1) in the fifth post-wildfire year at the Cannon site which lead to C and N losses approximately 100 times greater than those from any other storm which occurred during the study period. High intensity rainfall events of the magnitude recorded at Cannon, Hayman, and Fridley are not frequent in the western U.S., but their occurrence at multiple sites during this study exemplifies their potential to drastically increase erosion rates during both the initial and later stages of burn site recovery. Despite considerable differences between site locations, sediment C and N concentrations were similar between sites with relatively similar contributing areas. The C and N concentrations of A-horizon soils are largely unaffected by wildfires as oxidation temperatures are rarely reached in the A horizon (Johnson, 2001; Nave, 2011). During high severity fires, organic material is removed by ignition and convection; the resulting surface soil resembles soils found in the mid to low A-horizon of unburned forests (Bormann, 2008). The removal of organic material is likely to produce post-fire surface soil C and N concentrations that are less variable between sites than would be expected in organic rich, pre-fire forest soils. Sediments collected from watershed sites tended to have lower C and N concentrations than sediments collected from hillslope sites. By visible observation, watershed sites showed increased rill and channel erosion in comparison to hillslope sites; which likely increased the transport of nutrient-poor material from lower soil horizons.

Cumulative sediment losses of C and N in this study were < 25% of previously reported pyrogenic C and N losses from organic and mineral soil layers during forest fires (Binkley, 2012; Bormann, 2008; Grier, 1975; Johnson, 2007). Total losses were estimated to equal < 5% of the C and N remaining in post-wildfire soils (Baird, 1999; Johnson, 2007). Baird et. al previously estimated that post-wildfire sediment N losses reduced A-horizon soil N capital by ~1-2% following post-wildfire sediment N losses of 14-22 kg N/ha . Sediment N losses from Hayman exceeded those reported by Baird et. al. by 5-10 fold, which indicates that post-fire N capital reduction due to sediment loss may exceed 2% in highly erodible areas. Nonetheless, the post-wildfire soil C and N lost due to erosion is unlikely to be detrimental to post-wildfire recovery processes due to the relative size of remaining soil C and N pools. Although the effects of sediment C and N loss on burned areas are minimal in comparison to pyrogenic losses, C and N inputs of the magnitude reported in this study may alter carbon and nutrient dynamics in downstream ecosystems. Current research pertaining to the effects of post-wildfire sediment C and N inputs on aquatic systems is limited due to difficulties involved with determining the biologic availability and ultimate fate of sediment carbon and nutrients.

As the size and frequency of wildfires have increased, a large amount of interest has been placed on the effect of wildfires on global carbon budgets. Erosion and corresponding soil degradation has been shown to increase the rate of gaseous emissions from terrestrial systems (Lal, 2003). Estimates of carbon emissions during a wildfire vary significantly due to difference in fuels as well as fire size and severity. Following a wildfire study conducted in the Rocky mountain region of Oregon, gaseous carbon emissions from coniferous forest landscapes were estimated to be 25 Mg C/ha (Meigs, 2009). The average amount of carbon transported from the sites in this study was 0.4 Mg C/ha. Estimating a 20% carbon emission rate from sediment mineralization (Lal, 2003), the atmospheric carbon inputs from post-wildfire soil degradation would equal 0.3% (80 kg C/ha) of the pyrogenic emission losses. Based on the relative size of potential sediment based GHG emissions in comparison to pyrogenic emissions, the impact of post-wildfire erosion on global carbon budgets is likely to be minimal.

The use of erosion mitigation treatments continues to rise as wildfires become more prevalent along urban interface regions of the western U.S. Due to secondary fire effects, such as flooding, erosion and debris flows; severely burned areas that are left untreated pose a threat to nearby property and public safety (Stewart 2003). Three of the four erosion mitigation treatments investigated in this study reduced sediment, C, and N yields by > 65%. In a prior study involving the Hayman, Myrtle Creek, Hot Creek, and School sites (Robichaud, 2013A) no significant difference in vegetative cover was found between untreated areas and the applied treatment types (Hydromulch, straw mulch, and wood straw mulch) during the first four post-fire years (with the exception of the first post-fire year at the Myrtle Creek site). The lack of a difference between vegetative cover for the treated and untreated areas suggests that the reduction in sediment C and N losses have little to no effect on vegetation recovery rates.

Table 1: Wildfire name and start date, National Forest and State, location and elevation, annual precipitation, catchment scale and contributing area, slope, number of sediment producing rainfall events, 10-minute rainfall intensity from sediment producing events, and erosion mitigation treatment types for each study site.

Wildfire [Fire Start Date]	**National Forest [State]**	**Location (°N, °W) [Elevation (m)]**	**Annual Precip. (mm)**	**Catchment Scale [Contributing Area]****	**Mean Slope [Min/ Max] (%)**	**Sediment Producing Rainfall Events**	**Average I_{10} ± Std. Dev. ($mm\ hr^{-1}$) [Study Period Max I10]**	**Applied Erosion Mitigation Treatment(s)**
Valley Complex [July 31, 2000]	Bitterroot [Montana]	45.91°, 114.03° [1725]	400	Watershed [3.6^{C}, 2.8^{L} Ha]	46 [12 / 86]	7	25.7 ± 18.2 [59]	Log erosion barriers
Fridley [August 19, 2001]	Gallatin [Montana]	45.51°, 110.78° [1940]	700	Watershed [13.3^{C}, 11.8^{L} Ha]	43 [0 / 77]	8	31.1 ± 17.3 [55]	Log erosion barriers
Hayman [June 8, 2002]	Pike & San Isabel [Colorado]	39.18°, 105.36° [2387]	400	Watershed [3.0^{C1}, 3.1^{L1}, 4.6^{C2}, 5.2^{H2}, 3.3^{S2} Ha]	32 [7 / 65]	17	36.7 ± 15.9 [65]	Log erosion barriers, Hydromulch, Straw Mulch
				Hillslope [33.3 m^2]	20, 40***	23	28.5 ± 16.4 [72]	Straw, Wood Strands
Cannon [June 15, 2002]	Humboldt-Toiyabe [California]	38.45°, 119.47° [2230]	300	Watershed [12.6^{C}, 10.9^{L} Ha]	38 [1.8 / 69]	3	57.7 ± 66.8 [134]	Log erosion barriers
Kraft Springs [August 31, 2002]	Custer [Montana]	45.41°, 104.12° [1190]	360	Watershed [2.8^{C} Ha]	28 [3.5 / 51]	4	36.1 ± 17.8 [61]	Straw Mulch
Hot Creek [July 23, 2003]	Boise [Idaho]	43.76°, 115.22° [2310]	1100	Hillslope [72 m^2]	48	8	24.4 ± 9.8 [38]	Straw Mulch
Myrtle Creek [July 19, 2003]	Idaho Panhandle [Idaho]	48.72°, 116.46° [1857]	790	Hillslope [285 m^2]	44	6	39.7 ± 19.5 [59]	Hydromulch, Straw Mulch
School [August 5, 2005]	Umatilla [Washington]	46.22°, 117.66° [1686]	1380	Hillslope [212 m^2]	67	9	23.0 ± 18.7 [35]	Hydromulch, Straw Mulch, Wood Strands

* Rainfall events producing < 20 kg/ha of sediment are not included

** Separate, paired watersheds used for each mitigation treatment (Control^{C}, Log Erosion Barriers^{L}, Hydromulch^{H}, Straw^{S}, Wood strands^{W})

*** Two different hillslopes were used for sediment catchments at the Hayman monitoring site

Table 2: Soil classification(s), texture class, dominant vegetative overstory and understory by study site.

Study Site	Soil Classification(s)	Soil Texture Class	Dominant Overstory	Dominant Understory
Valley Complex	Sandy skeletal, mixed, frigid Haplustepts	Gravelly loam	Douglas fir; Ponderosa pine	Mallow ninebark; Pinegrass
Fridley	Loamy skeletal, mixed, typic Agriborolls; Loamy, skeletal, mixed, Mollis Eutroboralfs; Rock outcrop;	Gravelly loam	Douglas fir; Lodgepole pine	Snowberry; Idaho fecue
Hayman	Sandy skeletal, mixed, frigid shallow typic Ustorthents; Sandy skeletal, micaceous, shallow Typic Cryorthents	Gravelly coarse sand	Ponderosa pine; Douglas Fir	Common juniper; Kinninkinick
Cannon	Loamy-skeletal, mixed, superactive, frigid Pachic Haploxerolls	Gravelly silt loam	Black pine; Pinyon juniper	Sage; Bitterbrush
Kraft Springs	Coarse loamy, mixed, superactive, frigid Typic Caliustepts; Coarse-loamy, mixed, superactive, frigid Typic Haplustolls	Fine sandy loam	Whitebark pine; Logepole pine	Bluebunch wheatgrass; Sun sedge
Hot Creek	Loamy skeletal mixed Typic Cryorthent, Lithic Cryorthent, Entic Haploboroll, Typic Xerochrept, Typic Xerorthent, Typic Haplumbrep	Sandy loam	Douglas fir; Subalpine fir	Geyer's sedge; Grouse whortleberry
Myrtle Creek	Andic Dystrudept, Typic Udivitrands, Vitrandic Dystroxerepts	Ashy sand	Douglas fir; Ponderosa pine	Ninebark; Dwarf huckleberry
School	Loamy skeletal, isotic, frigid Vitrandic Argixerolls; Loamy skeletal, mixed, superactive, frigid Lithic Haploxerolls	Ashy loamy sand	Douglas fir; Grand fir	Bluebunch wheatgrass; Pinegrass

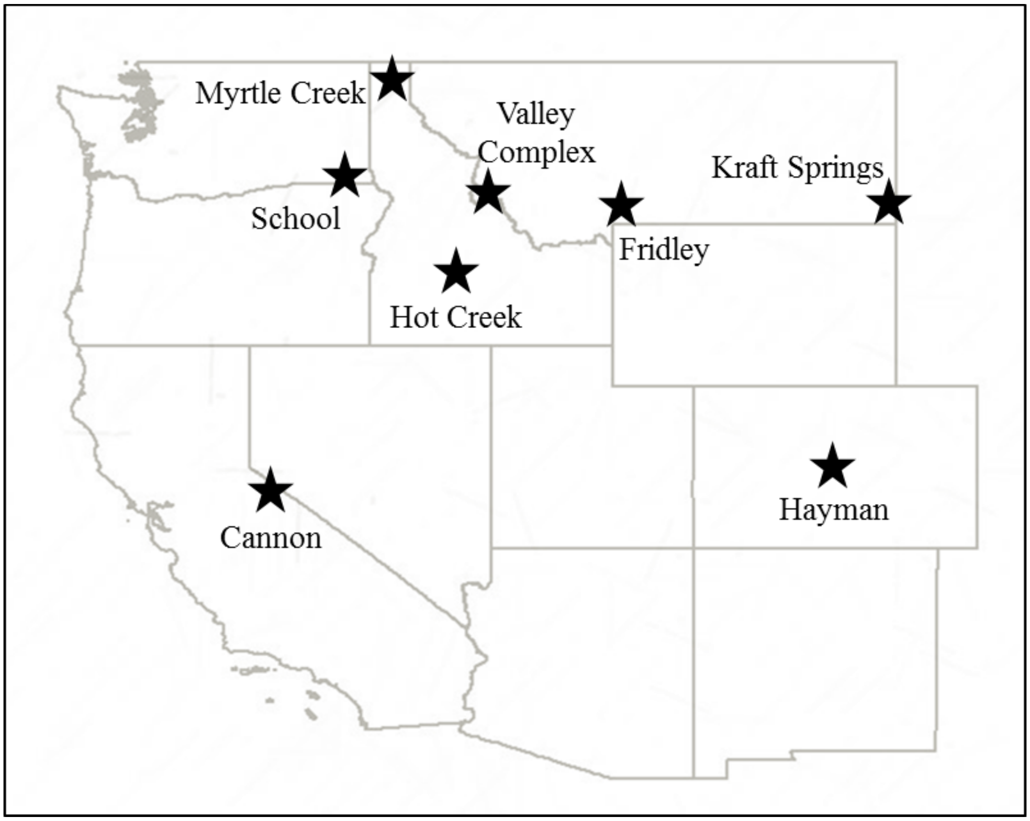

Figure 1: Location of wildfire study sites in the Western U.S.

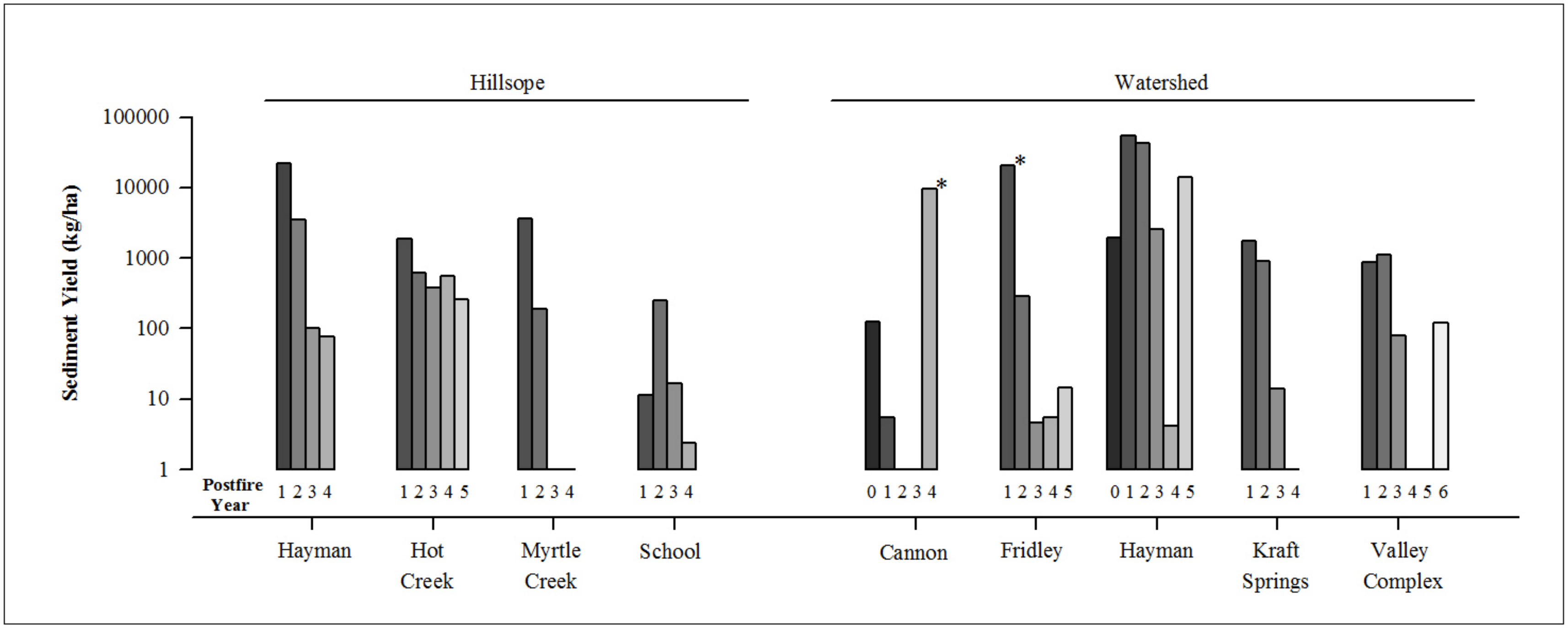

Figure 2: Average, annual sediment loss from post-wildfire erosion at untreated sites with hillslope scale plots and/or watershed scale catchments. Sediment losses were measured for a period of four to six years. The calendar year of the fire is denoted as year zero.

*Sediment yield approximated due to a single event yield which exceeded catchment capacity.

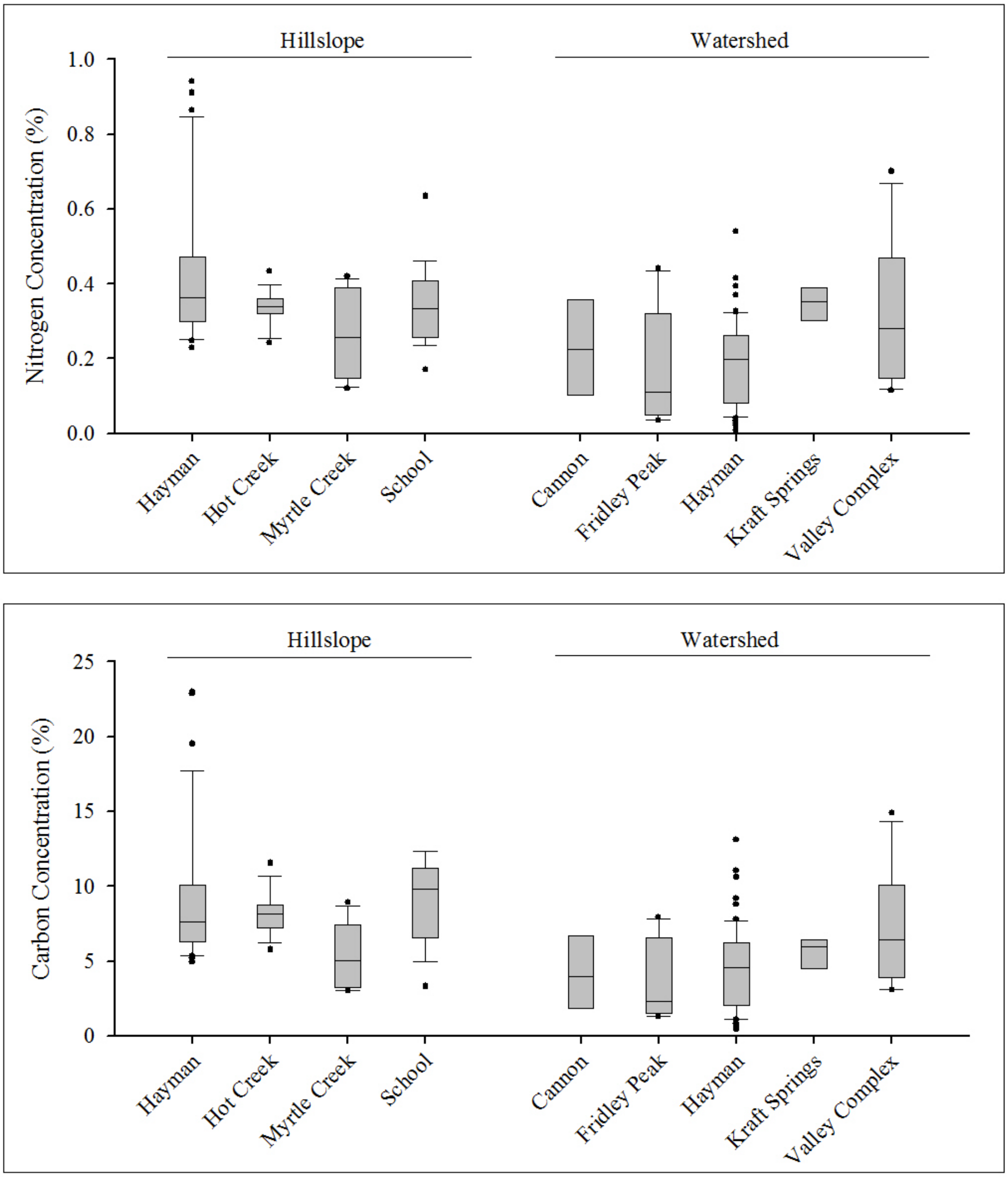

Figure 3: Boxplots of post-wildfire sediment carbon and nitrogen concentration from hillslope plot and watershed catchment sites. Boxplots include data from eroded sediments collected over the four to six year study period from all untreated and treated plots and catchments. Concentration of sediment from hillslope plots and watershed catchments are significantly different.

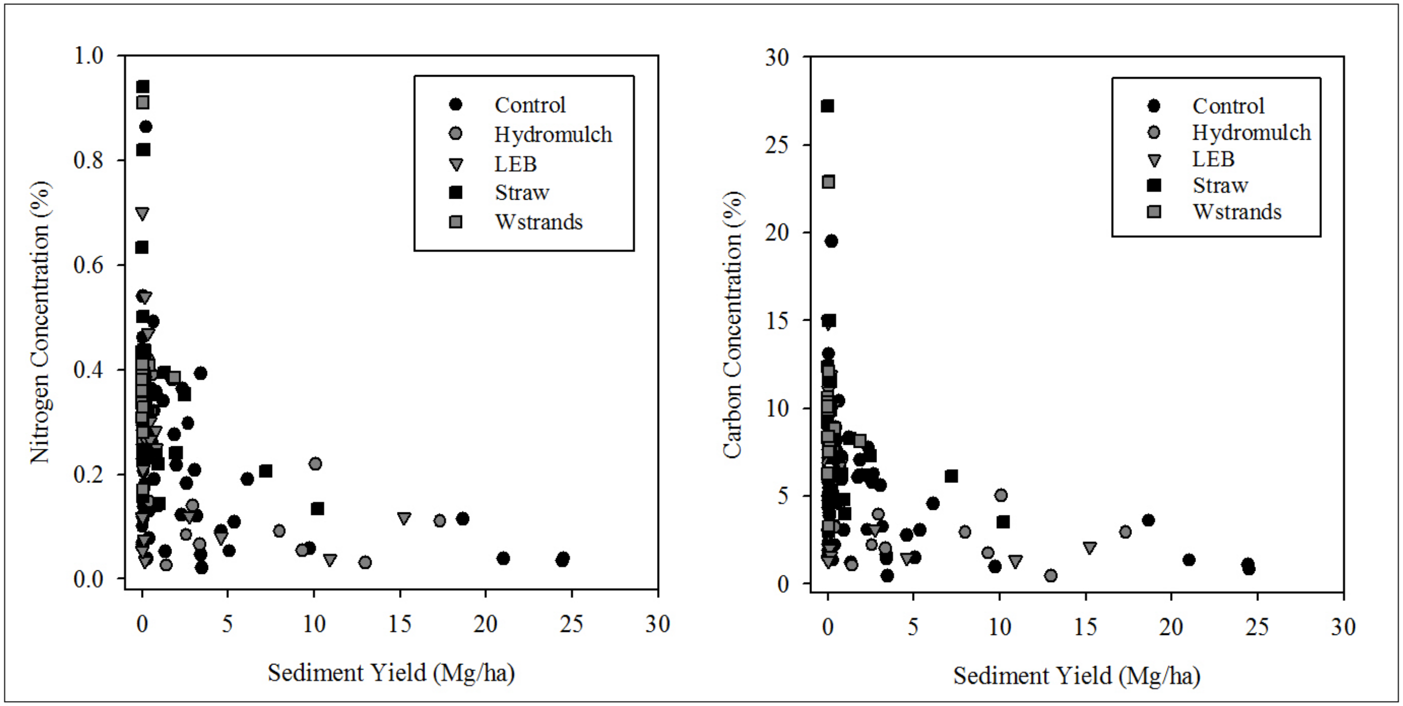

Figure 4: Relationship between the magnitude of post-wildfire sediment yield and sediment carbon and nitrogen concentration. Data includes all sediment yield and concentration data from the eight study sites monitored over a four to six year period. Individual data points are shown by corresponding type of erosion mitigation treatment applied to the contributing area of the plot or catchment.

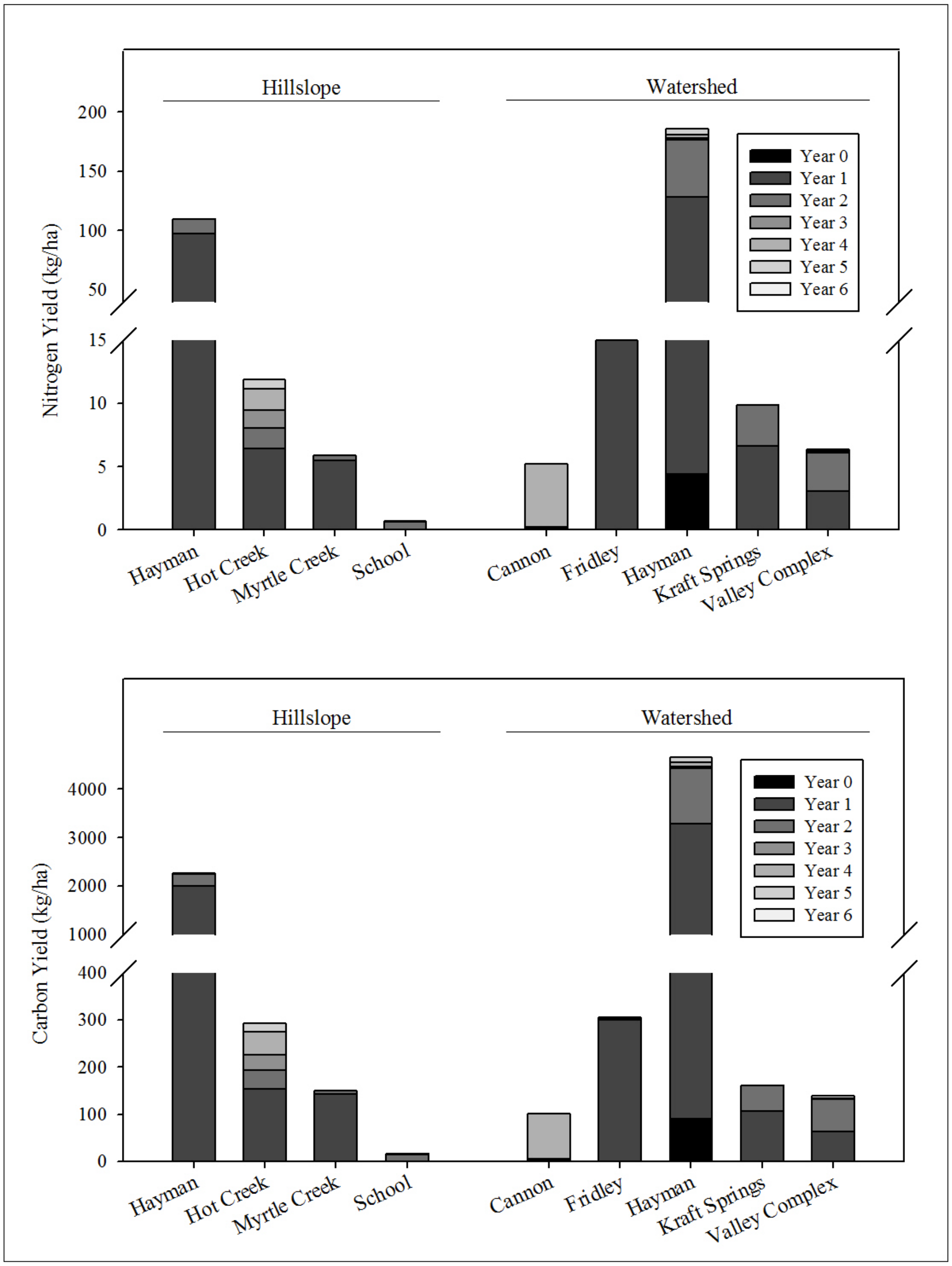

Figure 5: Average, annual carbon and nitrogen yields resulting from post-wildfire erosion at untreated sites equipped with hillslope scale plots and/or watershed scale catchments. Study periods ranged from four to six years across sites. Year zero pertains to the calendar year of the wildfire. *Yield approximated due to a single storm event which produced sediment beyond catchment capacity.

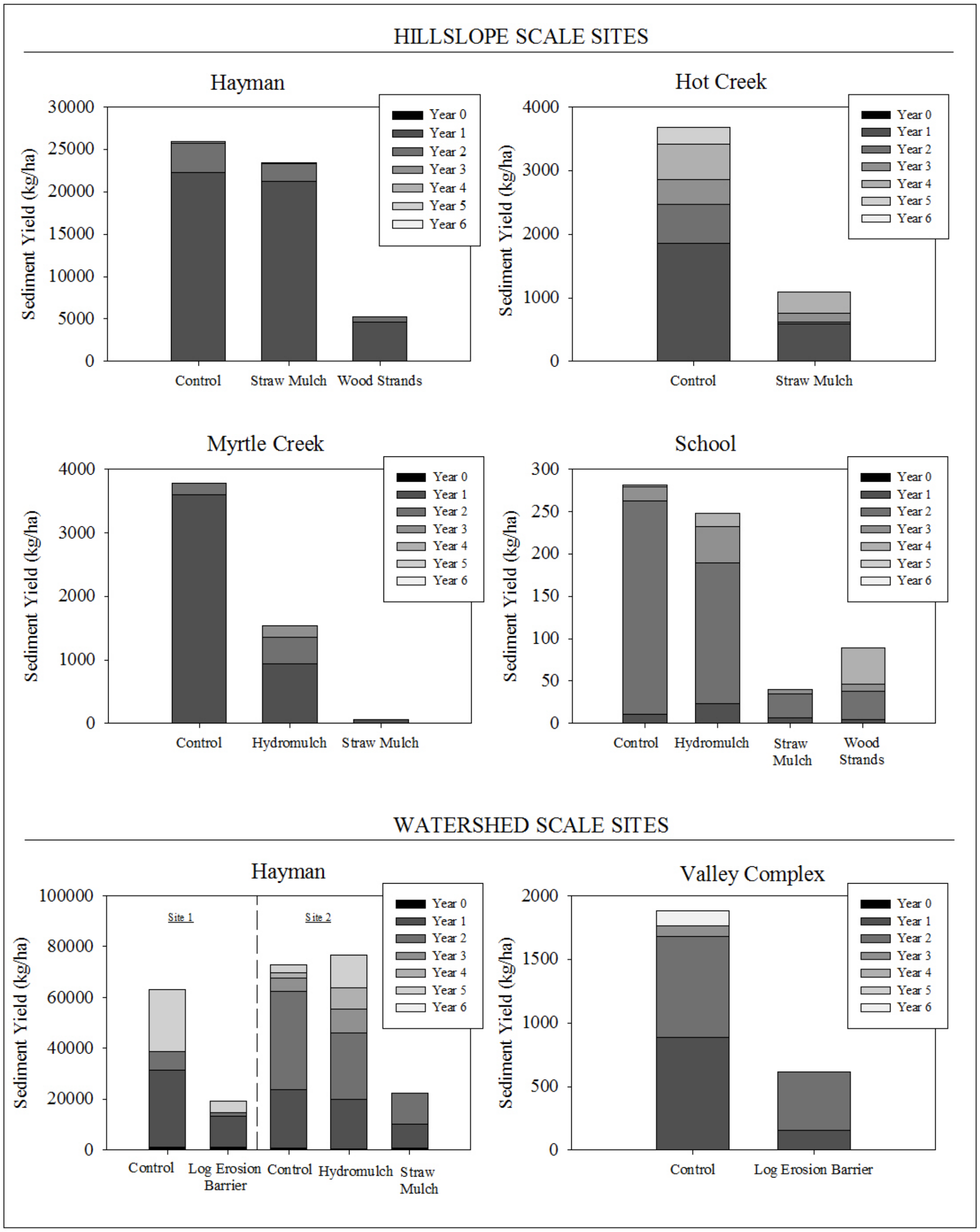

Figure 6: Comparison within study sites of sediment yields for untreated, control areas and areas treated with hydromulch, log erosion barriers, straw mulch, or wood straw mulch. Yields are shown as the sum of annual, average sediment yields measured for each hillslope plot or watershed catchment. Treatment comparisons for the Cannon and Fridley wildfire sites are not shown.

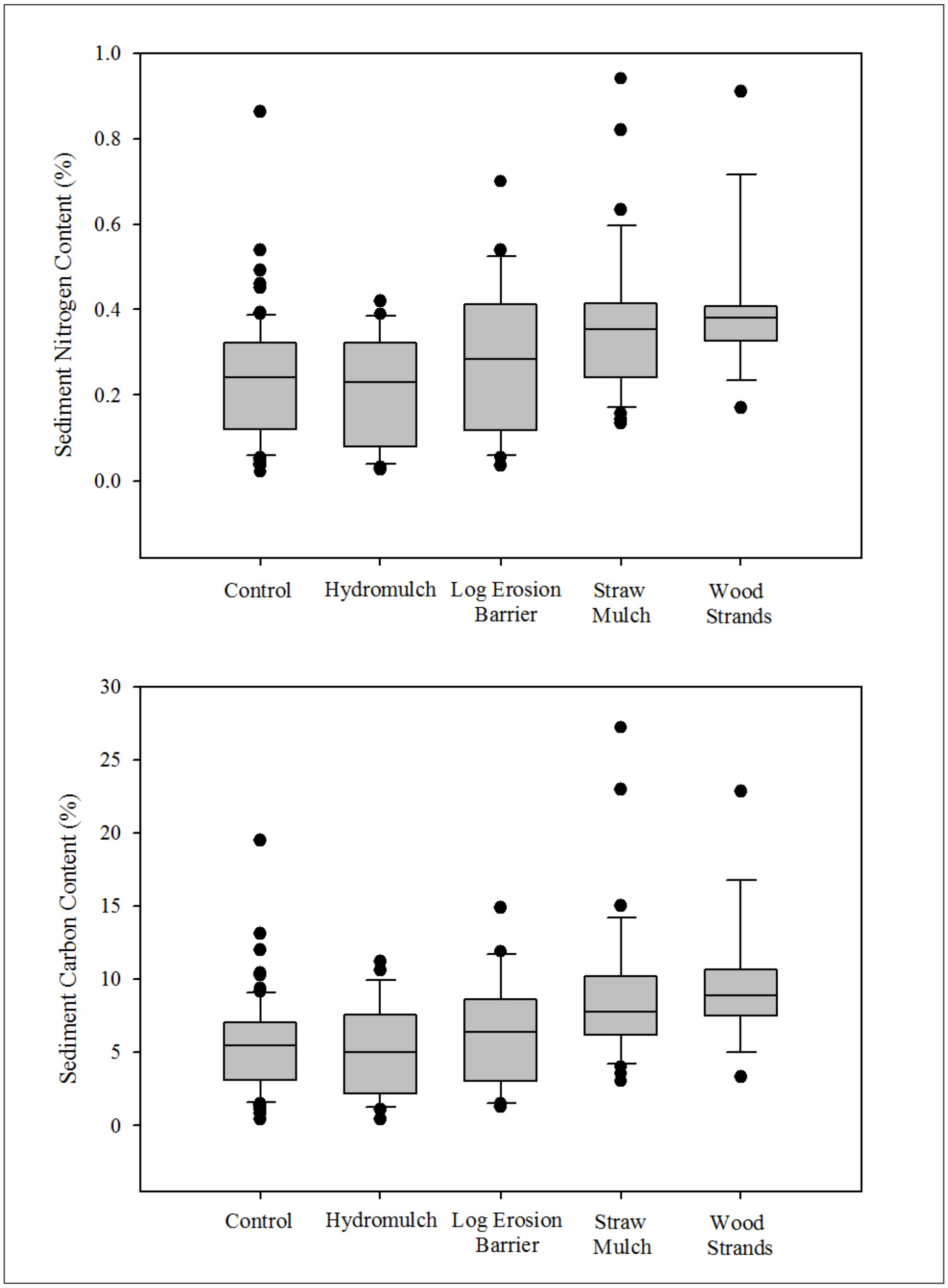

Figure 7. Boxplots of carbon and nitrogen concentration of eroded sediments collected from hillslope plots and watershed catchments with differing treatment types. Data includes sediment carbon and nitrogen concentration measured over the duration of the study period at all study sites.

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