Soil carbon heterogeneity in piñon–juniper woodland patches: Effect of woody plant variation on neighboring intercanopies is not detectable

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ABSTRACT

Soil carbon often varies significantly among vegetation patch types, but less known is how the size and species of plants in the tree canopy patches and the cover types of the intercanopy patches affect the carbon storage, and whether vegetation characteristics affect storage in adjacent patches. To assess this, we measured fine-fraction soil carbon in a semiarid woodland in New Mexico USA for canopy patches of two co-dominant woody species, *Pinus edulis* and *Juniperus monosperma* that were paired with inter-canopy patch locations covered by herbaceous grass (*Bouteloua gracilis*) or bare ground. Soil carbon at shallow depths was greater in canopy than intercanopy patches by a factor of 2 or more, whereas within intercanopy patches soil carbon in grass locations exceeded that in bare locations only after accounting for coarse-fraction carbon. Hypothesized differences among canopy patches associated with species or size were not detected (although some size-depth interactions consistent with expectations were detected), nor, importantly, were effects of species or size of woody plant on intercanopy soil carbon. The results are notable because where applicable they justify estimates of soil carbon inventories based on readily observable heterogeneity in above-ground plant cover without considering the size and species of the woody plants.

1. Introduction

Land management of the world's extensive drylands confronts a range of issues including productivity, forage and grazing, fuel wood, erosion, encroachment, and desertification, all of which depend on patterns and dynamics of soil carbon (Archer et al., 2001; Hibbard et al., 2001, Jackson et al., 2002; 2003; House et al., 2003; Huxman et al., 2005; Breshears, 2006). The expected changes in global climate will likely have consequences for land management requiring additional focus on soil carbon as a factor in local ecosystem dynamics and as a contributor to carbon management and related sequestration (Pacala et al., 2001; Schimel et al., 2001; Breshears and Allen, 2002; Houghton, 2003; Lal, 2004). Vegetation cover in these drylands is on average lower and has a much stronger local patch structure than that of regions of higher rainfall. Drylands such as the piñon-juniper woodlands of the U. S. southwest, viewed above ground, are a mosaic consisting of woody patches and intervening areas with varying degrees of herbaceous cover from continuous to virtually non-existent (bare) (Breshears, 2006). Because this vegetation mosaic is relatively stable over time, a corresponding surface soil mosaic is also present. Therefore, understanding carbon dynamics of systems like piñon–juniper woodlands requires assessing how the patch structure affects carbon storage. Although advances in estimating above ground carbon in dryland ecosystems is being facilitated through new remote sensing approaches (Asner and Heidebrecht, 2002; Harris et al., 2003), estimating below-ground soil carbon still requires relatively costly on-the-ground observations.

To better account for, monitor, and manage soil carbon, improved estimates of soil carbon are needed in a variety of dryland ecosystems. Because of the importance of remote sensing approaches, it would be particularly helpful if patterns of below-ground carbon could be related to readily observable patterns of the above-ground vegetation. The above-ground pattern that is the most obvious and frequently considered in studies of soil carbon in dryland ecosystems is the presence or absence of the canopy cover of woody plants (shrubs or trees). The surface soil horizons in canopy patches are often deeper than that in adjacent intercanopy patches, most likely as a result of the enhanced input of litter that occurs in canopy patches (Bates et al., 2002; Breshears, 2006).

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Among canopy patches, larger woody plants likely have influenced soils for longer periods and might be expected to have higher amounts of soil carbon than those associated with smaller woody plants. Further, differences among woody plants could be associated with species differences due to associated differences in litter quality, microclimate, and perhaps life expectancy and population turnover (Breshears et al., 1998; Lebron et al., 2007; Madsen et al., 2008). Within intercanopy patches, a smaller-scale dichotomy occurs between bare locations and those with and without herbaceous cover (considered here in the broader context in which grasses are herbaceous). For example, in semiarid grasslands soil carbon has been shown to differ beneath herbaceous grass patches vs. between them (Hook et al., 1991; Gill and Burke, 1999); this heterogeneity might also occur within the intercanopy patches of semiarid shrublands and woodlands. Importantly, woody plants could potentially affect soil carbon in neighboring intercanopy patches directly by extending their roots into adjacent gaps or indirectly by shading or other influences (Breshears 2006). If so, it would be expected that this influence would vary with the size and species of the woody plants. Assessing the influence of neighboring woody plants on soil carbon in intercanopy patches, however, has received little attention. Testing for the effects of woody plants on intercanopy soil carbon is particularly important because such effects could complicate approaches to estimate soil carbon inventories from amounts of different types of cover (e.g., woody plant, intercanopy herbaceous, intercanopy bare). Although many of the individual contrasts discussed have been quantified in various systems, studies that evaluate these hierarchical factors simultaneously are generally lacking, particularly tests for the effects of woody plants on neighboring intercanopy soil concentrations.

To assess both heterogeneity and connectivity in soil carbon as related to vegetation patches, we focus here on semiarid piñon–juniper woodlands, which might be expected to have particularly pronounced spatial heterogeneity among the diverse drylands of the world. The co-dominant woody plants in piñon–juniper woodlands have a high foliar density, are evergreen, and have their lowest foliage close to the ground—three factors that increase heterogeneity (Breshears and Ludwig, in press; Villegas et al., in press), and that may also increase the potential for variation in size and species of neighboring woody plants to influence soil carbon concentrations in intercanopy soils. This fundamental issue about soil carbon heterogeneity and connectivity has important relevance for carbon storage and dynamics in extensive drylands. Piñon–juniper woodlands and savannas are some of the most extensive ecosystems in the western US (McPherson, 1997). These ecosystems can also be very dynamic due not only to grazing but also to major changes associated with fire (Romme et al., 2009) and drought-induced die-off (Breshears et al. 2005) and subsequent associated vegetation dynamics (e.g., Rich et al. 2008) and soil conditions (Classen et al. 2005). In short, accounting for existing soil carbon has emerged as an important land management issue and a necessary precursor to subsequent assessments of how soil carbon inventories might change in response to disturbance and climate change.

Our overall objective was to evaluate several hierarchical components of vegetation pattern that could affect soil carbon patterns and site carbon inventory in a highly heterogeneous piñon–juniper woodland. We tested for soil carbon differences between canopy patches of woody plants and intercanopy patches that separate them. Among canopy patches, we additionally tested for soil carbon differences between species—*Pinus edulis* and *Juniperus monosperma*—and as a function of woody plant size. Among intercanopy patches, we tested for soil carbon differences between herbaceous locations with *Bouteloua gracilis* and locations with bare soil. Our design specifically allowed us to test if size and/or species of

neighboring woody plants affected concentration of intercanopy soil carbon concentrations. On the basis of our findings, we discuss the implications of spatial heterogeneity in soil carbon as a function of vegetation cover and soil depth and how ecosystem soil carbon could be impacted by future disturbance events.

2. Materials and methods

2.1. Site description

The study was conducted on a mesa top at an elevation of 2140 m, in a piñon–juniper woodland dominated by *P. edulis* Engelm. and *J. monosperma* [Engelm.] Sarg.), located within Technical Area 51 of the Los Alamos National Laboratory in northern New Mexico (35° 50'N, 106° 16'W). The upper soil layer is classified as Hackroy sandy loam derived from rhyolitic Bandelier Tuff (Nyhan et al, 1978; clayey, mixed, superactive, mesic, shallow aridic Haplustalfs). The mean slope is 4–5% with aspects varying from southeast to southwest across our plots. The dominant herbaceous species in the intercanopy patches between trees is *B. gracilis* [H.B.K.] Lag. Tree canopy cover is close to 50% (Martens et al., 1997; Breshears, 1997b). Mean annual precipitation is approximately 40 cm, 40% of which comes as monsoonal rains from July to September. Vegetation, soils, and hydrology of the sites have been studied in detail at the site (Breshears, 2006, 2008 and references therein). The sampling for the study occurred during 2001, prior to a major drought-induced die-off (Breshears et al., 2005).

2.2. Sampling design

2.2.1. Plots

Ten rectangular, approximately 0.43 ha plots were located within piñon–juniper woodland at the site. Each plot was located at least 12 m from the edges of paved surfaces and 3 m from the edge of unpaved roads and previously manipulated experimental plots. Areas that had significant exposed bedrock (highly dissected soil surface) were not included in the sampling regime as soils were patchy, very shallow and often only resided in fissures in the bedrock, making representative sampling difficult or impossible.

2.2.2. Sampling units

Within each plot we identified trees of three size classes based on the lower, middle and upper thirds of the distribution of tree sizes for all trees greater than 1 m tall at our site (Martens et al., 1997). Corresponding small, medium, and large size classes for piñon were <14.6 cm diameter at 10 cm height, 14.6 – 25.1 cm, and >25.1 cm, respectively, and for juniper were <10 cm at 10 cm height, 10.3 – 17.6 cm, and >17.6 cm. Because *J. monosperma* often develops multiple stems, we used single stem equivalents (effective diameter) by applying the formula $ED = (\sum d_i^2)^{0.5}$ where d_i is the diameter at the base of the ith stem of a multiple-stemmed tree (Chojnacky, 1994). Our design yielded 6 trees per plot and 60 trees overall. Where possible, we chose isolated trees with no over-lapping canopy with any other tree. Where isolated trees were not available, we used the tallest tree (dominant) or a tree sharing maximum height of the cluster (co-dominant) with canopies immediately adjacent to a neighboring intercanopy patch. We made three exceptions to these rules in order to complete our sampling design. In one instance we chose a non-dominant tree at the edge of a cluster that shared < 20% canopy overlap with the cluster and was functionally isolated in a large intercanopy patch. In this case, we sampled on the side of the tree furthest from the cluster. In two plots, we were unable to locate a specific size/species combination. In these two instances, trees within 15 m of a plot that met all other criteria were substituted. Primary analyses were

performed with and without these trees and the results were not significantly different. Further, the block effect on "plot" in our statistical model was not significant in either case. We therefore have included the two "outsider" tree samples in all analyses based on these observations.

2.2.3. Transect orientation

For each tree, a sampling transect was established from the tree bole and extending into an adjacent intercanopy patch. The transect was positioned to minimize influence of other sampled trees for all canopy and intercanopy samples. For isolated trees, transect direction was chosen to maximize the distance from tree clusters while still providing for adequate grass and bare soil intercanopy samples. For trees within clusters, the transect direction was chosen based on two criteria. First, a 90° sector was centered on the chosen tree bole to maximize distance from other tree boles and provide a sufficiently open area to allow for a minimum of 2 m between canopies and associated intercanopy samples. Second, a specific transect direction was selected within this 90° sector that allowed for intercanopy sampling. Intercanopy samples included a grass patch with minimum diameter of 5 cm and a bare patch of at least 10 cm diameter, and both located at least 2 m from any canopy dripline in any direction (with a bias toward greater distances to non-sampled clusters). In all cases, seedlings in the understory beneath canopies were avoided as much as possible in setting up transects.

2.3. Soil sample collection and analysis

2.3.1. Sample collection

Three soil cores (one each for canopy, intercanopy grass, and intercanopy bare; each 3 cm diameter) were extracted at each chosen tree along a transect as defined above using an ESP+ soil corer (JMC, Newton, IA). Each core was taken to the depth of tuff bedrock (C horizon) or to coarse parent material (CR horizon), which in some cases was greater than 1 m and required a second core from the same hole. The C or CR horizon for each core (or pair of cores when two were taken) was identified in the lab, with soil depth designated as the top of the CR or C horizon. Each core was divided into intervals of 0–5 cm, 5–10 cm, and every 10 cm thereafter to the top of the CR or C horizon. The interface between litter and soil was determined as the point where litter was visually no longer incorporated into the soil matrix. To establish this, the litter was first removed by hand, remaining loose litter was blown away, then the litter/soil layer was lightly scraped off until only soil remained. With this approach, some litter may have remained buried below this somewhat arbitrary but consistently identified soil surface. The soil/litter mixture layer was usually less than 1 cm thick, and was mostly litter. This approach minimized litter-C contribution to soil-C from litter fall and allowed us to remove any effects of differential amount of litter from either tree species or from grass cover.

2.3.2. Sample analysis

A total of 1075 samples from depths 0–70 cm were sieved at 2 mm to remove stones and organic debris, hand ground using a mortar and pestle, and weighed. A second sieving at 0.018 mm was required to homogenize samples for total carbon analysis. Three size fractions were collected: whole soil ($<$2 mm fraction), coarse soil (0.018 mm – 2 mm), and fine soil ($<$0.018 mm).

We determined total soil carbon on approximately 0.75 g of each fine ($<$ 0.018 mm sieve) soil sample using a VarioMax C/N analyzer manufactured by Elementar (Hanau, Germany) by dry combustion as outlined elsewhere (Lal, 1997; Sollins et al., 1999; Akala and Lal, 2000). The instrument was standardized using glutamate with known carbon assay. Several empty crucibles and glutamate standards were analyzed throughout the run to ensure quality control. When anomalous results were detected in the analysis of standards or blank samples, new groups of samples were run.

2.3.3. Supplemental analyses related to inventory

Our primary focus was on spatial variation in soil carbon. However, to also enable simple extrapolations to soil carbon inventory, we collected two types of supplemental data. We analyzed an additional 200 samples of coarse fraction soils (0.018 mm $<$ particle size $<$ 2 mm) that had been saved from our initial sieving to test the assumption that C% from the coarse soil fraction was the same as that from the fine soil fraction. In three cases, 0–5 cm canopy, 5–10 cm canopy, and 0–5 cm grass, the carbon concentration for the whole soil was significantly different when based on the fine soil alone than when the coarse and fine soils were used together. Regression equations were generated for these three relationships. Using the resulting regression equations, corrected whole soil C% values were calculated.

To evaluate the mass of carbon stored in soils on a constant volume/equivalent mass basis (Ellert et al., 2001), we measured bulk density for the three major patch types and for a subset of depth intervals. Samples were collected using a 10 cm diameter ring driven into the soil to a specified depth within each of the three vegetation patch types, with greater replication at shallow depths (42 total at 0 to 4 or 5 cm; 12 total at 5 to 10 and at 10 to 22.7 cm; and 4 at 22.7 to 33.5 cm). The bulk density estimates were corrected for soil moisture content (USDA, 1999). Regressions of bulk density against mid-points of depth showed a significant trend for increasing bulk density for "canopy" (bulk density (g/cc) = 0.951 + 0.125 ln(depth in cm), $p < 0.0007$) but no significant relationships for grass and bare ground ($p > 0.18$ and $p > 0.38$, respectively). We estimated the total carbon content by depth using the percent carbon data and data collected on the bulk density. Using these site-specific estimates of bulk density, we calculated the carbon inventory for each location by multiplying the whole soil C concentrations by the corresponding bulk density estimates and layer depth. To calculate the total carbon inventory, the overall mean of bulk density from all samples was used for grass (mean = 1.33 g/cc, s.e. = 0.023) and bare (mean = 1.30, s.e. = 0.023). For the canopy soils the values from the regression line were used to 30 cm, beyond this the value for the 20–30 cm depth class was used. Finally, we estimated total soil carbon inventory for our site by multiplying previous estimates (Breshears, 2008) of percent cover for each patch type by the patch storage estimate of g C/m². This value is reported in MgC/ha to be more representative of the size of our study area.

2.4. Statistical analysis

We used the Proc Mixed model in SAS to analyze variation in the fine soil fraction considering main effects for soil depth, type of cover (tree canopy, intercanopy grass, or intercanopy bare), tree size, and tree species; we also evaluated two-way interactions and, where data were sufficient to support it, three-way interactions. Plots and cores taken from the same transect within those plots were treated as random variables. A repeated measures analysis was used to account for autocorrelation among multiple samples within each soil core. When a test resulted in multiple comparisons, a Tukey-Kramer adjusted p-value is reported. All means reported here are least squares mean estimates generated by Proc Mixed during model fitting. We used $p < 0.05$ for significance.

In addition to an analysis of the full dataset, subgroups based on a single class of samples were developed and models were fit

separately for each major subgroup (one depth, one species, one size class, one cover type). Our primary results have been divided into "full dataset" results and "subgroup" results to reflect this. Significant terms in each of the various models are discussed in the appropriate section. Spatial autocorrelation that was likely to be present between samples at different depths in the same soil core, between cores taken from the same tree, and between cores taken from the same plot, were all accounted for in our model.

3. Results

3.1. Spatial patterns of soil carbon

Soil carbon concentrations for the fine fraction were relatively low and ranged between 0.35 and 2.0%. Overall soil carbon patterns averaged across all categories of cover varied with depth, as expected, decreasing by about 50% within the top 30 cm (Fig. 1a). When averaged across depth, soil carbon in canopy patches was significantly greater than in either the grass or bare intercanopy patch types, and soil carbon in grass intercanopy locations was significantly greater than bare intercanopy locations (Fig. 1b). More specifically, soil carbon in canopy locations was greater than grass locations for 0–10 cm and was greater than bare locations for 0–30 cm (Fig. 1b). However, within canopy patches there were not overall differences between species—*J. monosperma* vs. *P. edulis* (Fig. 1c). Nor were there overall differences among canopies of different sizes, when aggregated across species (Fig. 1d).

We evaluated variation in soil carbon for the fine fraction as a function of cover, depth, size, and species (as well as tree transect and plot) and the interactions among these factors (Appendix Table 1). Cover and depth were both significant, and additionally had a significant interaction, whereas neither size nor species did. Further, there was significant variation associated with samples taken at the same tree (i.e., corresponding canopy, grass, and bare soil cores). Variation among plots was not significant. Although grass and bare locations were significantly different when considered overall, and the magnitude of this difference appears greatest at shallow depths, we were unable to detect depth-specific differences among these two patch types, likely due to the reduced power associated with multiple comparisons. Cover type was also strongly significant as a predictor in most subgroup evaluations. However, cover by itself was not significant at any depth below 40 cm.

Contrary to our expectation that soil carbon concentration would be greater below larger trees, tree size did not have a significant effect on soil carbon by itself or in any overall inter-action with factors of depth or cover (Appendix Table 1), although there were interactions for small, medium, and large piñons, as well as for small junipers (Fig. 2). Similarly, species of tree (piñon vs. juniper) did not significantly affect overall soil carbon concentration, although there was a significant species by depth interaction for small size class samples, with small junipers having greater soil carbon concentrations than small piñons at shallow but not deeper depths (Fig. 2).

For a subset of measurements, we compared our estimates of soil carbon that were for the fine soil fraction to the whole soil carbon fraction. The whole soil fraction was significantly greater than the fine soil fraction in canopy patches at the upper two depths and in grass patches at the upper depth, as shown by the following results:
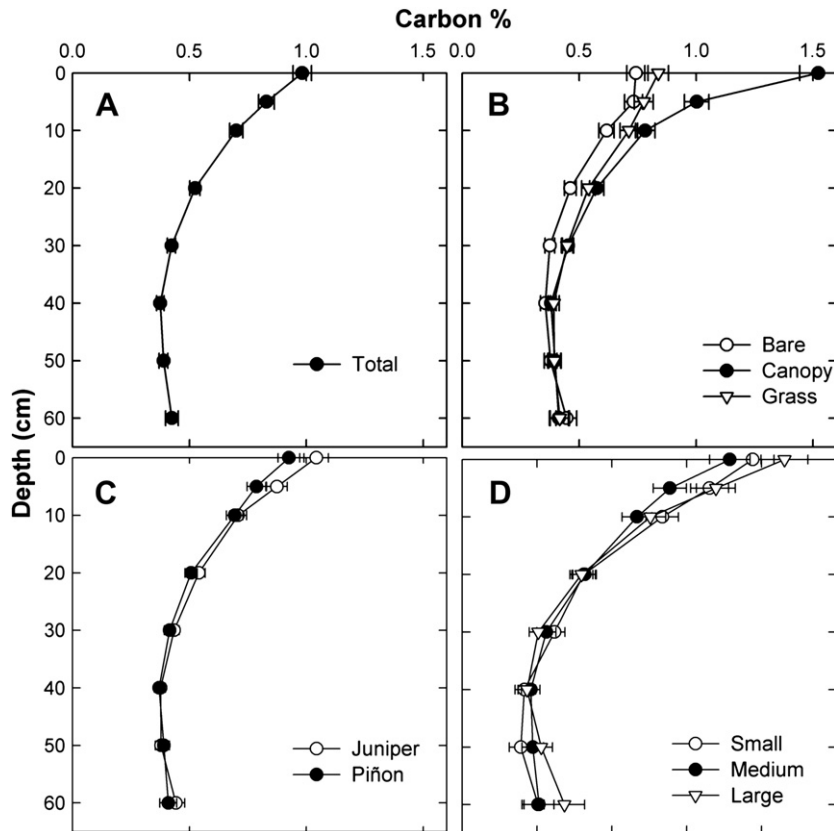

Fig. 1. Soil carbon (fine size fraction) profiles: A. overall, B. by vegetation patch type C. by species for canopy patches, and D. by tree size for canopy patches. Error bars represent 1 standard error. The overall dataset included 1075 samples. For A there are 180 replicates for each depth class to 30 cm. The number of replicates below this depth could be less because of variations in total soil depth. For B-D, the number of replicates to 30 cm is 60, 90, and 60 respectively, with smaller numbers below 30 cm because of shallower soils.
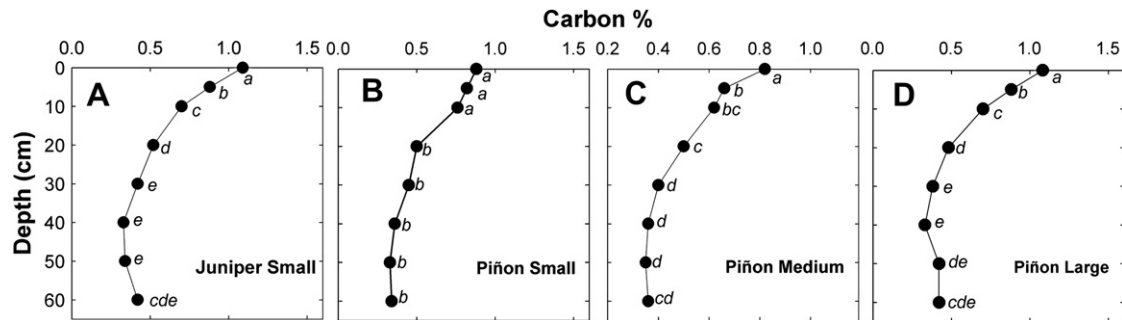
Fig. 2. Soil C (fine size fraction) profiles for subsets with significant species by depth interactions. The overall dataset included 1075 samples. For each group the number of replicates to 30 cm is 10, with smaller numbers below 30 cm because of shallower soils. Different letters within the same panel indicate significant differences a $p < 0.05$.

Canopy (0–5 cm): Coarse C% $= 0.54 + 1.66$ * Fine C% ($p = 0.006$, $r^2 = 0.72$);

Canopy (5–10 cm): Coarse C% $= 0.32 + 1.10$ * Fine C% ($p = 0.006$, $r^2 = 0.62$); and

Grass (0–5 cm): Coarse C% $= -0.20 + 1.74$ * Fine C% ($p < 0.001$, $r^2 = 0.92$).

The estimates of carbon storage for the top 10 cm by cover type that corrected to account for the coarse fraction, where needed, were 17.9 MgC/ha for tree canopy, 11.3 for grass, and 9.7 for bare (Table 1). Because the upper soil layers beneath the tree canopies had lower bulk densities, the contrast between the types for the total fine carbon content is less than the contrast in percent carbon. However, more fine-fraction carbon is stored beneath the tree canopies. Based on previous estimates, our site mosaic has approximately 50% canopy cover, 17% grass cover and 33% bare soils (including soil biological crusts; Breshears et al. 1997b; Breshears, 2008). Because we did not detect differential effects of woody plants on intercanopy soil carbon concentrations, we simply multiplied the cover proportions by the patch-based soil C estimates to estimate an approximate total site inventory of 33.0 Mg Soil C/ha.

4. Discussion

4.1. Spatial heterogeneity in soil carbon related to vegetation cover

4.1.1. Plant cover effects on soil carbon

We found substantial variation in soil carbon between the canopy patches of woody plants and within intercanopy locations

Table 1
Carbon inventory based on corrected estimates for whole soil within three vegetation patch types (standard errors in parentheses, propagated based on errors associated with fine soil samples and with bulk density samples).

	Vegetation Patch		
	Intercanopy Bare	Intercanopy Grass	Canopy
Carbon inventory (g/m²)			
0–5 cm	490 (22.5)	622 (23.3)	1098 (160.1)
5–10 cm	484 (22.2)	505 (23.2)	698 (92.0)
10–20 cm	822 (30.9)	920 (42.2)	1006 (129.2)
20–30 cm	610 (18.4)	700 (26.2)	785 (98.6)
30–40 cm	490 (14.8)	583 (17.6)	609 (18.3)
40–50 cm	477 (14.4)	505 (15.2)	515 (15.5)
50–60 cm	504 (18.9)	505 (19.0)	542 (20.3)
60–70 cm	596 (27.4)	544 (25.0)	555 (25.5)
Cumulative inventory (g/m²; 0–50cm)	2882	3214	3613

that separate them, consistent with other studies (Davenport et al., 1996; Hysell and Grier, 1996; Conant et al., 1998; Kramer and Green, 2000; Bates et al., 2002; Shukla et al., 2006; Selmants and Hart, 2008). At our site, carbon storage under tree patches was as much as ~25% greater than bare soil and up to ~175% greater in the top 5 cm. We also expected to find more carbon in soils under grasses than in bare soils as did others (Hook et al., 1991; Hook and Burke, 2000). We did not initially detect differences between grass and bare intercanopy locations, but were able to do so after accounting for coarse soil fraction. The finding of significant differences between tree canopy and grass as well as between grass and bare soils is unique to our study and suggests that at least three patch types are necessary to quantify the soil carbon storage in our system.

4.1.2. Tree species and size effects on soil carbon

Piñon and juniper species differ in litter quality (Murphy et al., 1998) and rooting patterns (Williams and Ehleringer, 2000), (Breshears et al., 1997a), both of which affect soil carbon storage. Previous measurements at our site hinted that soil carbon concentrations under junipers might be greater than that under piñons (Davenport et al., 1996). Several other studies documented differences in soil carbon concentrations between piñon and juniper species (Klopatek et al., 1994; Debano and Klopatek, 1988; Conant et al., 2000). However, we did not find the expected differences with species and, in addition, we found no evidence of increasing soil carbon storage with tree size under piñon–juniper canopies. It may be that decomposition substantially lags litter production or that the stature, productivity or age of *P. edulis* (maximum age ~220 y; Davenport et al., 1996) and *J. monosperma* on our site are not sufficient to generate detectable below ground storage differences related to tree-size; our trees, for example, are smaller than those evaluated by Klemmedson and Tiedmann (2000). Our finding of heterogeneity associated with vegetation patch type between canopy and intercanopy locations but not with tree size or tree species overall suggests that soil carbon concentrations are relatively similar across canopy patches.

We also expected that tree size and species would affect neighboring intercanopy soil samples—that there would be patch-scale connectivity as well as heterogeneity (Breshears 2006). Indeed, this was a key knowledge gap we were addressing. However, we did not detect overall effects of tree size and species on neighboring intercanopy soils. We do know that there is connectivity in water use between canopy and intercanopy patches through use of shallow intercanopy soil water by woody plants (Breshears et al., 1997a). In addition, bole size was previously found to be positively correlated with soil water depletion at 1–2 m from the tree bole for junipers, whereas no difference was found for piñons (Breshears et al., 1997a), leading to our expectation that

species of neighboring tree would influence soil carbon as well as soil water uptake. Our ability to detect such a relationship was diminished by the lack of variation associated with tree size and tree species among canopy patches. Consequently, our lack of ability to detect the effect of patch connectivity on intercanopy soil carbon concentrations is most likely applicable to other locations where soil carbon concentrations do not vary greatly among canopy patches. Conversely, effects of patch-scale connectivity on soil carbon concentrations might be detectable and potentially important where soil carbon concentration does vary substantially with tree size and species among canopy patches.

Note that we found a larger difference between surface soils (top 5 cm) beneath canopy and bare ground when the carbon concentrations of the coarse (between 0.018 and 2 mm) fractions were included in the carbon analysis, indicating that litter-rich shallow soils below grass and tree canopies preferentially contained coarser materials. Our initial method of sieving down to very fine particle size fractions to gain homogeneous samples underestimated C in these shallow soils. By introducing the C held in the coarse fractions of the soil back into our analyses, we found statistically significant and potentially biologically important differences between patch types in the upper few centimeters of soil that were previously undetectable. More generally this finding highlights that methods that do not incorporate the carbon concentration in both the coarse fraction and the fine fraction may underestimate carbon concentration and inventory in near surface soils impacted by litter and fine root inputs. Such corrections could be important when trying to detect small changes in carbon storage or when estimating carbon storage over large areas.

4.2. Baseline soil carbon and implications for disturbance and management

Determining functional units regarding key ecosystem properties can aide in scientific prediction and management, as highlighted in other studies of piñon–juniper woodlands (Wilcox and Breshears, 1996; Reid et al., 1999; Wilcox et al., 2003a, 2003b; Lebron et al., 2007; Madsen et al., 2008). Our results suggest that there are three functional units for soil carbon in the semiarid woodland that we studied: canopy patches of woody plants, and, within intercanopy patches, herbaceous and bare locations. The largest soil carbon differences were between canopy patches and either intercanopy herbaceous or intercanopy bare locations. Importantly, hypothesized differences among canopy patches associated with species or size were not detected (although some size-depth interactions consistent with expectations were detected), nor were effects of species or size of woody plant on intercanopy soil carbon. The results indicate that intercanopy soil carbon was relatively insensitive to size and species of woody plant and to canopy-intercanopy patch connectivity via woody-plant roots. Therefore, site soil carbon concentrations related largely to canopy-intercanopy heterogeneity, with effects of canopy-intercanopy connectivity not detectable based on soil carbon concetrations alone (Breshears, 2006). The lack of canopy-intercanopy heterogeneity is significant because it enables estimates of soil carbon inventories based primarily on amount of woody plant canopy coverage.

The soil carbon differences that we detected associated with plant cover are particularly pronounced at shallow depths. This pattern of a high degree of heterogeneity at shallow depths is associated with plant cover and is not unexpected but may be particularly important to consider in light of growing interest to better account for and manage carbon. Many if not most management actions affect shallow soils and associated carbon dynamics.

For example, tree thinning to reduce fire risk and to increase herbaceous production (Brockway et al. 2002) could compact shallow soils, alter herbaceous growth, and/or reduce litter inputs, all of which could in turn influence rates of soil carbon accumulation particularly at shallow depths. Notably, piñon–juniper ecosystems can be quite dynamic in response to disturbances such as fire, drought-induced plant mortality, and increases in soil erosion, all of which could alter shallow soil carbon inventories (Breshears and Allen, 2002; Allen, 2007; Romme et al., 2009). Increases in temperature projected to be concurrent with increased frequency of drought portends future increases in fire and drought-induced plant mortality and associated increases in wind and water erosion (Adams et al., 2009; Field et al., in press). Our results, then, are potentially important in the context of land management in that they enable estimates of soil carbon concentrations based on readily observed patterns of above-ground vegetation and facilitate evaluations of how alternate management and disturbance regimes that affect shallow soils might change site soil carbon concentrations.

In summary, our results highlight two scales of heterogeneity in soil carbon that could be important for other widespread semiarid ecosystems: substantial heterogeneity between canopy vs. intercanopy patches, and minor heterogeneity between herbaceous vs. bare locations within intercanopy locations. Notably, soil carbon patterns are apparently related to heterogeneity associated with cover but not with connectivity between canopy and intercanopy patches, whereby variation in size and species of neighboring trees could influence intercanopy soil carbon concentrations. These functional units likely affect soil carbon responses to climate change and land use. Given the high degree of soil carbon heterogeneity with vegetation patches, the relatively high concentrations of soil carbon at shallow depths, and the potential for disturbance and/or land management to impact shallow soil carbon, we argue that management in dryland ecosystems—such as that associated with forage, fuel wood, fire management, erosion control, and carbon management—requires more explicit consideration of the patchy structure of soil carbon within these systems.

Appendix

Analysis of variance (ANOVA) results for fine fraction soil carbon (%). *P*-values are reported for all fitted subgroup models (single depth, single patch type, single size, or single tree species) and the full dataset. Weakly suggestive *p*-value in the next to last reduction of the model is indicated by * or ** and corresponding *p*-value is listed in the notes. N/A indicates where the model did not return a value. A total of 1075 samples were used in the analysis. Numbers of replicates per cell for the depth analysis was 180 across all other factors (cover, size and species) for the surface depth classes (0–30 cm) but was less than this for the greater depths because of variations in the depth to bedrock.

	Subgroup	Cell (Block)	Tree (Block)	Depth	Cover	Size	Species	Cover by depth	Cover by size	Size by depth	Species by depth	Notes
Depth	0–5 cm	0.1565	0.0033	–	<.0001	–	–	–	–	–	–	–
	5 - 10 cm	0.1199	0.0004	–	<.0001	*	–	–	–	–	–	*.021
	10 –20 cm	0.136	0.0003	–	<.0001	0.4251	–	–	0.0304	–	–	–
	20 - 30 cm	0.078	0.0004	–	<.0001	*	–	–	**	–	–	*0.0940,**0.0816
	30 - 40 cm	N/A	0.0012	–	<.0012	–	–	–	–	–	–	–
	40 - 50 cm	No predictor was significant										
	50 - 60 cm	No predictor was significant										
	60 - 70 cm	N/A	0.0239		0.314	0.8362	–	–	0.0105	–	–	–
Cover	Bare	0.3116		<.0001	–	–	–	–	–	–	–	–
	Grass	0.1283		<.0001	–	–	–	–	–	–	–	–
	Canopy	0.187		<.0001	–	–	–	–	–	–	–	–
Tree size	Small	N/A	0.0087	<.0001	0.0412	–	0.9188	0.0215	–	–	0.0261	–
	Medium	0.2214	0.1306	<.0001	<.0001	–	–	<.0001	–	–	–	–
	Large	0.2576	0.1284	<.0001	0.0287	–	–	<.0001	–	–	–	–
Tree species	Piñon	0.4626	0.1518	<.0001	<.0001	0.4489	–	<.0001	0.0018	0.0018	–	–
	Juniper	0.0691	0.0223	<.0001	0.0017	–	–	<.0001	–	–	–	–
	Full dataset	**0.1281**	**0.0013**	**<.0001**	**<.0001**	–	–	**<.0001**	–	–	–	–

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