Carbon stocks and patterns in native shrub communities of Senegal's Peanut Basin

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ARTICLE INFO

ABSTRACT

Accurate and reliable estimates of carbon (C) storage in landscapes are critical to the development of effective policies and strategies to mitigate atmospheric and climate change. Carbon stocks of two native woody shrub (*Guiera senegalensis* J.F. Gmel and *Piliostigma reticulatum* (DC.) Hochst) communities and associated soils within Senegal's Peanut Basin were determined and the spatial structure of soil C quantified. These shrubs are of interest because they dominate semiarid sub-Sahalien Africa and commonly coexist with row crops but have been largely overlooked as a key vegetative component of this landscape. Peak-season shrub biomass C was measured in forty-five 0.81 ha plots at 8 locations using allometric relationships along with soil sampling (0 to 40 cm depth) and analysis for organic C and bulk density. Soil samples to a depth of 20 cm were taken every 2 m in 24×20 m grids and every 0.5 m in four nested 3 m×3 m grids containing at least one shrub or tree canopy, and geostatistical techniques were then used to quantify scale and degree of soil C spatial dependence. Estimates of peak-season biomass C ranged from 0.9 Mg C ha^{-1} to 1.4 Mg C ha^{-1} with an overall mean of 1.12 Mg C ha^{-1} (SEM = 0.079) in the *G. senegalensis* sites and from 1.3 to 2.0 Mg C ha^{-1} (mean = 1.57 Mg C ha^{-1}; SEM = 0.18) in the *P. reticulatum* communities. The overall mean of SOC to 40 cm was 17 and 17.2 Mg C ha^{-1} respectively, at the *G. senegalensis* and *P. reticulatum* sites with 57% of that C residing in the top 20 cm. Semivariograms of soil C showed moderate spatial dependence and spatial autocorrelation at distances of less than 0.56 and 1.34 m at the *G. senegalensis* and *P. reticulatum* sites, respectively. Comparison across the different grids showed that the presence of shrub canopies at either site had much closer relationship to soil C levels than trees.

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1. Introduction

Assessment and improved understanding of total system carbon (C) stock and its individual components in Senegal's Old Peanut Basin is important for understanding biogeochemical processes in this ecosystem, improving soils for crop productivity, and practical strategies to sequester C in soils. Carbon lost from the Parkland agroforestry systems in this basin significantly contributes to atmospheric change, particularly increases in carbon dioxide concentrations (Houghton et al., 1993). Therefore a quantification of C stocks for different land management systems allows for better estimates of these C losses to the atmosphere as land degradation patterns are compared over time (Lal, 2002). Besides the global goal of mitigating elevated atmospheric carbon dioxide, sequestering C in soils would be of interest to land managers by improving soil properties (Woomer et al., 1997). These benefits could include increased land productivity,

better yields and also contribute to improved overall soil quality and health, which in turn can help buffer these inherently fragile ecosystems (Bationo and Mokwunye, 1991; Brouwer and Bouma, 1997) from abiotic stresses (Elliot et al., 1993; Woomer et al., 1994; Murage et al., 2000).

The policies and scientific research/actions concerned with C cycling depend on accurate information about spatial distribution of C in vegetative and soil components of terrestrial ecosystems. The Kyoto Protocol (1997) presents an internationally negotiated framework for guiding these policies. Lal et al. (1999), and Lal (2002) argue that Article 3.3 and Article 3.4 of the protocol provides rationale for the importance of managing drylands to sequester C via two key mechanisms: restoration of desertified lands (Lal et al., 1998a,b) and the promotion of perennial woody biomass (Manley et al., 1995).

The Peanut Basin of Senegal is located on the Sahel's north/south vegetative gradient between the sparsely wooded grasslands of the north and tree-dominated ecosystems to the south. It is characterized by intensively cultivated Parkland systems (Freeman, 1992) comprising mainly of randomly dispersed trees and woody shrubs in farmers' fields (Weber and Major, 1984). Depending on geological substrate,

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temperature and rainfall amounts, the woody shrub component covers the entire landscape in some areas and normalized difference vegetation indices (NDVI) show it is the single largest vegetation component in this region (A. Lufafa pers. comm., 2005; M. Dièye pers. comm., 2005). *Guiera senegalensis* J.F. Gmel and *Piliostigma reticulatum* (DC.) Hochst dominate this shrub species. In farmers' fields these shrubs are normally pruned back to the soil surface and the residue is burned in the spring of every year prior to cultivation but the shrubs will continue to grow if left uncut. Consequently, because of their dominance in the landscape and coexistence with cropping activities, these shrubs have the potential to be a significant source or sink for C within the global cycle, depending on land use and management.

A number of studies of C stocks in the Peanut Basin have been done that have had a range of estimates for the vegetative component (Woomer, 1993; Bationo et al., 1998; Manlay, 2000; Bationo and Buerkert, 2001; Batjes, 2001; Manley et al., 2002; Liu et al., 2004; Tschakert et al., 2004; Woomer et al., 2004a,b). However, these studies have had various levels of resolution and generally neglected the importance of the shrub component. We hypothesized that the shrubs, because of their abundance, are more important than the tree component or other sources of C such as animal manure in regulating the C stocks of the Peanut Basin. The objective of this study was to quantify shrub biomass C and evaluate the relative influence of the shrubs and trees on the spatial dependence of soil C levels.

2. Materials and methods

2.1. Study area characteristics

The study area is the Peanut Basin in Senegal, West Africa. Located east of Dakar, the area center lies approximately at 16°W, 14.7°N with a spatial coverage of ~44,000 km^2. The climate is semiarid, with on average more than 85% of precipitation falling between August and October. Mean annual precipitation is approximately 540 mm, skewed towards the south and with high variability from year to year (Dacosta, 1989). The mean annual minimum and maximum temperatures are 20 °C and 34 °C, respectively, with a marked seasonal variation. Geological substrates in the area include mainly aeolian deposits of Harmattan wind sand (Herrmann, 1996) of Quaternary age over sedimentary rocks of Cretaceous to Miocene age (Monciardini, 1966), and highly eroded colluvial–alluvial ferruginous sediments derived from paleosols (Neogene) and Precambrian bedrock (Renaud, 1961; Michel, 1973). Basin soils are sandy, classifying as Psamments (FAO: Arenosols or Regosols) and Calcids (FAO: Calcisols) according to Soil Survey Staff (2003) and fall broadly into two indigenous types, i.e. Dior and Deck (Badiane et al., 2000). Basin vegetation is shrubland with scattered trees (Diouf and Lambin, 2001).

2.2. Carbon stock estimation

2.2.1. Biomass C estimation

The peak-season standing biomass of shrubs in farmers' fields occurs in late spring just before the shrubs are pruned back to prepare for the summer cropping season. Thus we assessed shrub C stocks at this time of year. The procedure was to develop allometric equations that use easily measured shrub properties to estimate biomass. In turn, plot scale inventories of shrub densities in combination with allometric equations were done to estimate shrub biomass.

2.2.1.1. Allometric equations. The allometric equations were developed based on 75 shrubs (49 *G. senegalensis* and 26 *P. reticulatum*) sampled from eleven sites that captured the north–south rainfall gradient in the Peanut Basin. Measurements of maximum height (*maxht*), mean crown diameter (*mcd*), mean diameter of the shrub base (*mbd*), and total number of stems (*stems*) were recorded for individual shrubs at their peak growth in March (Breman and Kessler, 1995; Ker, 1995). The shrubs were clipped and roots were excavated to measure biomass in the above- and belowground components. Samples (leaves, stems and roots) of the biomass components were oven dried at 105 °C for 48 h to determine biomass on a dry weight basis. The data set obtained was randomly split to create an independent set of 12 *G. senegalensis* and 6 *P. reticulatum* samples that was used for validation of the biomass predictive models. Linear (log $Y = \beta_o + \beta_1 * X_1$), logarithmic ($Y = \beta_o + \beta_1 * \log X$), exponential (log $Y = \log \beta_o + X^{\beta 1}$), log–log (log $Y = \log \beta_o + \beta_1 * \log X$), and quadratic ($Y = \beta_o + \beta_1 * X + \beta_1 * X^2$) regression models ($Y =$ dry weight of biomass in grams, $X_1 \ldots X_n$ are the respective explanatory variables in each model e.g. maxht, mcd, mbd) were used as the pool of independent variables to build optimized models. Optimal equations were selected based on adjusted R^2 values and independent variables that maximized the significance (*p*-value) of the regression coefficients. The validation data set was used to evaluate the predictive capacity of the regression estimators (Neter et al., 1996) and to select the final equations.

2.2.1.2. Shrub biomass inventories. Shrub biomass inventories were performed in 90×90 m plots that were originally designed to derive relationships between remotely-sensed shrub abundance and landscape-level biomass C stocks (Lufafa et al., 2008). Eight sites (6 for *G. senegalensis* and 2 for *P. reticulatum*) with varying number of plot replicates (more plots at sites with great variation in shrub density) were selected for the inventory. The location, elevation and number of replicates at each of the sites are provided in Table 1. Measurements of biomass predictive variables (as adduced from the allometric equations) were recorded on all shrubs encountered in each plot and used to assign biomass to each shrub. The proportion of C in all biomass pools was assumed to be 47% of the biomass dry weight (Skog and Nicholson, 1998).

2.2.2. Soil C estimates

Total soil organic C (Mg ha^{-1}) to 40 cm depth in the plots was calculated from measurements of C concentrations (g C kg^{-1}) of the 0–20 and 20–40 cm soil layers and soil bulk density at 15 and 30 cm depths. In each of the 90×90 m plots, 15 sampling points were located along three transects positioned approximately 30 m apart. Soils were collected from the 15 sampling points with four sub-samples at each point bulked and mixed to obtain a composite sample. Bulk density was measured as described by Okalebo et al. (2002) at 15 and 30 cm depths. Samples for soil C were air dried and analyzed for total C by combustion on a LECO C-144 C analyzer (LECO Inc., St. Joseph, Michigan). No attempts were made to correct for carbonate as near-surface soil horizons in the study area are predominantly acidic (pH<7) (Tschakert et al., 2004).

Table 1
Location, elevation, number of plot replicates and average number of shrubs at the biomass inventory sites

Shrub species	Location, latitude/longitude	Elevation (m)	Replicates	Shrubs/ha[a]
G. senegalensis	Keur Asanulo N14.78, W16.74	34.3	3	275 (59.7)
	Keur Mandiemba N14.75, W16.67	46.7	4	409 (35.7)
	Keur Matar Aram N14.77, W16.86	50.5	7	239 (74.6)
	Keur Ibra Fall N14.75, W16.76	25.1	2	312 (162.0)
	Ndiagne N14.76, W16.77	22.5	5	407 (36.0)
	Thilla Ounte N14.79, W16.68	27.3	3	228 (42.0)
P. reticulatum	Sikatrou N13.98, W15.99	24.0	8	134 (20.9)
	Sanguel N14.03, W16.04	23.5	13	288 (22.4)

[a] Standard error of the mean in parentheses.

2.3. Soil C spatial structure

The goal of this study was to quantify and determine the relative influence of the shrubs and trees on soil C spatial patterns. Across the landscape, shrubs are interspersed among different tree species and we hypothesized that tree type could be an influence on soil C. As a preface to this study, soil C under canopies of the most common trees coexisting with the shrubs was determined. Results (not presented) showed no statistical difference in soil C due to tree type, thus enabling our study to be representative of any tree/shrub community in our study area.

Two sites, one each for *G. senegalensis* and *P. reticulatum* were selected for this study. A 24×20 m grid (selected to capture at least two trees and interspersed shrubs) was established at both shrub study sites and points sampled every 2 m (for a total of 143 samples). Within this major grid, samples were taken at a finer scale of 0.5 m within four, small nested 3 m×3 m grids (198 fine scale samples, for a total of 339 samples for the entire site). Each of the smaller grids contained at least one shrub or a tree canopy. Soil samples were taken to a depth of 20 cm and analyzed for total C.

Geostatistical analyses (Goovaerts, 1999) were conducted on the data to quantify the patterns of spatial variation in soil C across the "shrub-tree" sites. Semivariograms (Isaaks and Srivastava, 1990; Atteia et al., 1994; Wackernagel, 1994; Gotway and Hergert, 1997) were first calculated (in Splus) using all 339 points at a site to provide an unbiased description of the scale and pattern of soil C spatial variation (Oliver and Webster, 1986). Semivariograms were recalculated to examine the effect of the shrubs and trees on soil C patterning. First, the 98 data points from the two small grids around the shrubs were deleted and semivariograms calculated on the remaining data; then, the 98 points from the two small grids around the trees were deleted and semivariograms recalculated.

The semivariance, $\gamma(h)$, is calculated for each specific lag distance h as:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} \left[z(x_i) - z(x_i + h)^2 \right] \tag{1}$$

where $N(h)$ is the number of pairs of points separated by distances h, $z(x_i)$ is the measured sample value at point x_i, and $z(x_i + h)$ is the sample value at point $x_i + h$. The minimum pair distance used was 0.5 m and the maximum was 12 m (roughly half the maximum distance available from the data).

Least squares estimates of the parameters $\theta = (C_0, C_0 + C_1, a)$, where C_0 is the nugget (variance that is not spatially dependent), $C_0 + C_1$ is the sill (semivariance corresponding to the range), and a is the range (scale of spatial autocorrelation), were obtained by fitting the sample variogram $\hat{\gamma}(h)$ to the variogram model $\gamma(h; \theta)$. The least squares estimator for θ is obtained by finding the $\hat{\theta}$ that minimizes the *ad hoc* criteria:

$$D(\theta) = \sum_{o}^{h_0} n(h) \left\{ \hat{\gamma}(h) - \gamma(h; \theta) \right\}^2 \tag{2}$$

where $n(h)$ is the number of observations used in computing the sample variogram when using weighted least squares, and h_0 is the maximum distance used in fitting the variogram model.

Depending on the best reduced sum of squares, semivariograms were fit to linear, spherical or exponential models (Cressie, 1985; McBratney and Webster, 1986; Zhang et al., 1992). Adequacy of fitted models was tested using cross-validation techniques (Zhang et al., 1992) where the mean error between measured and kriging estimated values, the correlation coefficient between measured and estimated values and the reduced kriging variance are used to select the appropriate variogram model. Parameters for the selected model were used to evaluate the magnitude of spatial dependence and the scale of spatial autocorrelation for the variate (Robertson, 1987; Robertson et al., 1988; Jackson and Caldwell, 1993a).

3. Results and discussion

3.1. C stock estimation

3.1.1. Allometric relationships

Studies elsewhere have found log–log and quadratic equations most useful for predicting biomass for a number of shrub species (Bentley et al., 1970; Rittenhouse and Snew, 1977; Bryant and Kothmann, 1979). In this study, log–linear relationships produced the highest coefficients of determination with random residuals and significant two-sided *p*-values ($p > 0.05$). Table 2 is a summary of the regression coefficients and the corresponding mean square error of prediction (MSEP) for the selected models. The fit of maxht and mcd on *G. senegalensis* aboveground biomass yielded an r^2 value of 0.90, while the same fit yielded a coefficient of determination of 0.87 for *P. reticulatum* aboveground biomass estimates. The best fit models for shrub belowground biomass were those that incorporated mbd for *G. senegalensis* ($r^2 = 0.69$) and mcd for *P. reticulatum* ($r^2 = 0.81$). Mean crown diameter (mcd) was equally the best predictor of *P. reticulatum* total biomass ($r^2 = 0.83$) while for *G. senegalensis*, total biomass was best predicted by a combination of mcd and mbd ($r^2 = 0.80$).

Significant correlations between observed and predicted biomass were observed and these relationships are plotted in Fig. 1. While the regressions of observed and predicted biomass are significant, there are differences between the regression line and unity (1:1 relationship).

3.1.2. Biomass C stocks

Total biomass C stock at the *G. senegalensis* sites ranged between 0.93 and 1.40 Mg C ha^{-1} (Table 3) with an overall mean of 1.12 Mg C ha^{-1} (SEM±0.079). Approximately 82% of this C was belowground with only 18% allocated aboveground in leaves and stems. The belowground fraction constitutes 86% of total (mean=1.57 Mg C ha^{-1}) biomass C stock at the *P. reticulatum* sites, whereas a relatively smaller proportion (0.15 Mg C ha^{-1}) resides in the aboveground fractions. Significant correlation (where belowground biomass C=2.2418*aboveground C stock+ 0.4592; *r*=0.89) was observed between averaged aboveground and belowground biomass C stocks across the *G. senegalensis* sites. Derivation of these relationships was not possible for the *P. reticulatum* sites because of sample-size limitations, but regressions of individual plot above- and belowground biomass C stocks were significant (where belowground biomass C=10.72*aboveground C stock−0.099; *r*=0.94).

There was no statistical difference in above or belowground biomass C either within site or across site for *G. senegalensis*. Mean aboveground biomass C stocks were statistically similar within- but different across the two *P. reticulatum* sites (LSD=0.061). No statistical difference was observed in belowground biomass C either within or across these two sites. Comparison across shrub type reveals a statistical difference between belowground biomass C (*p*-value=0.038) and

Table 2
Selected model coefficients of determination (R^2), MSEPs and Pearson correlation coefficients (r) for observed and predicted biomass

Shrub species	Biomass model	Regression		Prediction	
		MSE	R^2	MSEP	r
G. senegalensis	LogAbgm=4.39+(0.0056*maxht)+ (0.011*mcd)	0.08	0.90	0.03	0.98
	LogBgbm=7.41+(0.015*mbd)	0.16	0.69	3.64	0.88
	LogTbm=7.14+(0.0054*mcd)+ (0.0083*mbd)	0.10	0.80	2.99	0.91
P. reticulatum	LogAbgm=4.20+(0.013*maxht)+ (0.0059*mcd)	0.07	0.87	0.29	0.99
	LogBgbm=7.14+(0.0099*mcd)	0.11	0.81	2.60	0.99
	LogTbm=7.31+(0.0098*mcd)	0.09	0.83	2.09	0.99

Abgm = Aboveground dry weight biomass; Bgbm = Belowground dry weight biomass; Tbm = Total dry weight biomass; maxht = shrub maximum height; mbd = mean shrub basal diameter; mcd = mean shrub crown diameter.
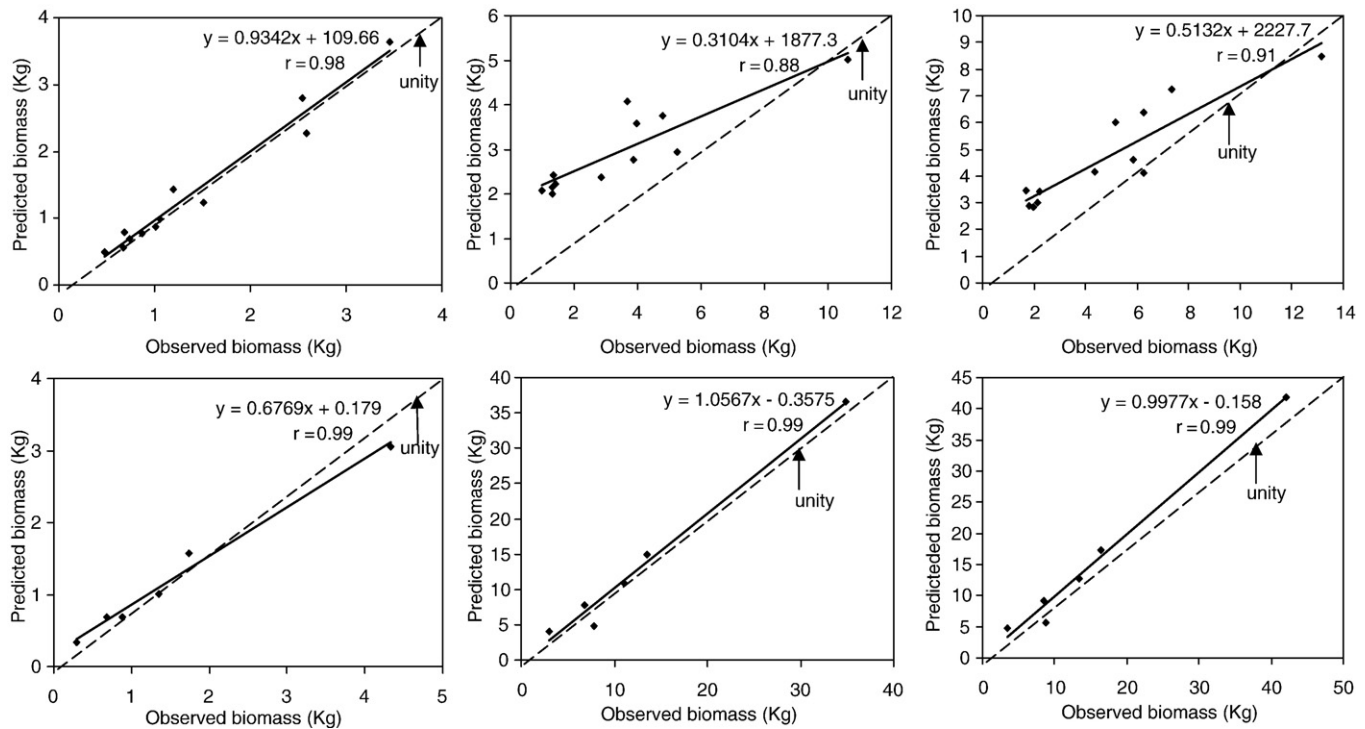
Fig. 1. Unity plots of observed and predicted biomass. Top left to right: aboveground, belowground and total *G. senegalensis* biomass. Bottom left to right: aboveground, belowground and total *P. reticulatum* biomass.

no differences between aboveground biomass C stock for the two shrub species. Number of shrubs significantly correlated ($p<0.001$) with C stocks in all *P. reticulatum* biomass fractions (Table 4) and warrants consideration as a surrogate indicator of C stocks. There were no significant relationships (data not presented) between *G. senegalensis* shrub numbers and any of the biomass C fractions, probably reflecting our tenuous ability to isolate single shrub entities without excavation.

To determine the amount of biomass that is pruned upon reclearing (when the regrowth is dense enough to impede crop growth), we selected and pruned shrubs in demarcated plots at peak season and left them to regrow in line with farmers' practice. These were then clipped again at a time coincident with farmers pruning and the amount of biomass was measured. Table 5 shows the average aboveground biomass regrowth for the shrubs with average values of 12% and 22% as a proportion of peak peak-season biomass respec-

tively, for *G. senegalensis* and *P. reticulatum*. We did not quantify surface litter C because movement of this component by wind across the landscape made it difficult to trace its source with reasonable accuracy. However, work conducted elsewhere in Senegal (Manlay et al., 2002; Woomer et al., 2004b) shows that this may be a relatively large C pool ranging from 0.493–0.980 Mg C ha^{-1}.

3.1.3. Soil C stocks

The 0–40 cm soil profile in the *G. senegalensis* sites contained an average of 17 ($\pm$0.528) Mg C ha^{-1}, 58.3% of which resides in the 0–20 cm (Table 3). Despite ranging from 8.37 to 12.03 Mg C ha^{-1} and 6.63 to 7.58 Mg C ha^{-1} at the 0–20 cm and 20–40 cm depth respectively, there were no statistical differences in SOC across the *G. senegalensis* sites at these depths. There was more variability observed in the 0–20 cm depth (average CV of 57.3% across the sites) as compared to the 20–40 cm depth (average CV of 39.4%) and this could be due to organic matter "hotspots" from animal manure or the "islands of fertility" phenomena (Vinton and Burke, 1995; Schlesinger et al., 1996; Kelly and Burke, 1997). Similar trends in variability were observed in the *P. reticulatum* sites with a CV of 55.8% at the 0–20 cm depth compared to 48% at the 20–40 cm depth. There were statistical differences in soil C between the two *P. reticulatum* sites, with more soil C at the site with lower biomass C. We postulate that this could be a reflection of historical land use and management which we did not capture in the study.

Significant correlations ($r=0.87$; p-value$=0.05$) were observed between SOC at the two depths in the *P. reticulatum* sites, whereas no

Table 3
Peak-season biomass, soil and system C stocks in native shrubs in Senegal's Peanut Basin

Shrub type/location	Biomass C		Soil C		Total C
	Aboveground	Belowground	0–20 cm	20–40 cm	
	Mg ha^{-1}				
G. senegalensis					
Keur Asanulo ($n=3$)	0.184 (0.079)	0.842 (0.393)	9.86 (23.61)	6.63 (7.12)	17.516
Keur Mandiemba ($n=4$)	0.181 (0.089)	0.861 (0.367)	9.50 (28.14)	6.99 (18.08)	17.532
Keur Matar Aram ($n=7$)	0.244 (0.353)	1.131 (1.853)	8.37 (17.89)	6.84 (8.12)	16.585
Keur Ibra Fall ($n=2$)	0.309 (0.192)	1.094 (0.278)	12.03 (30.59)	7.05 (9.88)	20.483
Ndiagne ($n=5$)	0.217 (0.249)	0.925 (0.473)	9.95 (32.34)	7.25 (16.25)	18.342
Thilla Ounte ($n=3$)	0.150 (0.017)	0.783 (0.473)	9.95 (43.48)	7.58 (30.39)	18.463
LSD	n.s.	n.s.	n.s	n.s.	
P. reticulatum					
Sikatrou ($n=8$)	0.107 (0.219)	1.108 (2.384)	11.27 (68.56)	9.03 (60.27)	21.52
Sanguel ($n=13$)	0.173 (0.179)	1.757 (1.969)	8.07 (28.05)	6.02 (11.31)	16.02
LSD	0.061	n.s.	1.320	0.987	

Ranges within each site are in parentheses.

Table 4
Regression relationships between shrub numbers and *P. reticulatum* biomass C stocks as Mg ha^{-1}

Dependent variable	Regression equation	r	p
Aboveground biomass C	$Y=0.0312+0.0005*$ number of shrubs	0.75	<0.001
Belowground biomass C	$Y=0.293+0.0053*$ number of shrubs	0.74	<0.001
Total biomass C	$Y=0.334+0.0062*$ number of shrubs	0.75	<0.001

Table 5
Peak-season and regrowth biomass for *G. senegalensis* and *P. reticulatum* at selected sites in Senegal's Peanut Basin

Shrub type	Peak-season biomass	Regrowth biomass	Regrowth as a percentage of peak-season biomass
	kg		
G. senegalensis			
Plot 1	12.31	1.32	10.75
Plot 2	7.43	1.51	20.33
Plot 3	8.91	0.78	8.78
Plot 4	11.83	0.80	6.74
Overall mean	10.12 (0.21)[a]	1.10 (0.035)	11.65
P. reticulatum			
Plot 1	13.97	3.37	24.12
Plot 2	8.96	3.12	34.82
Plot 3	13.46	1.94	14.41
Plot 4	18.19	2.39	13.14
Overall mean	13.65 (1.89)	2.71 (0.33)	21.62

[a] Standard error in parentheses.

correlations existed between SOC at the two depths in the *G. senegalensis* sites probably reflecting differences in C sequestration mechanisms in the two shrub communities. Carbon input and sequestration in the *P. reticulatum* sites is mostly driven by vegetation decomposition processes, whereas stochastic processes due to wind movement and dust entrapment (Elkins et al., 1986; Garner and Steinberger, 1989; Coppinger et al., 1991) could be an additional influence on C sequestration in the drier *G. senegalensis* sites.

3.2. Soil C spatial structure

Variability of soil C across the 24×20 m grid and the small grids around the trees was higher at the *P. reticulatum* site, whereas variability around the shrubs was higher at the *G. senegalensis* site (Table 6). Mean total C was consistently higher under trees at both sites and lower under the shrubs.

Semivariograms for all the 339 points at either site (Fig. 2 and Table 7) showed a moderate spatial dependence among sampling locations with a proportion of structural variance (C) to total estimated variance (sill; $C+C_0$) of 53% and 55% respectively, for *G. senegalensis* and *P. reticulatum* sites. The range of spatial autocorrelation was shorter for *G. senegalensis* (within 0.56 m) as compared to *P. reticulatum* (within 1.34 m) although semivariograms were essentially not flat beyond these distances, probably indicating patchiness or repeating soil C patterns. Excluding the 98 data points from the small grid around the shrubs produced variograms shown in Fig. 3. The shapes of the resulting semivariograms are unchanged but there are changes in the structural and total variance, reduced spatial dependence (from 53 to 51% at the *G. senegalensis* site and 55% to 52% at the *P. reticulatum* site), and reduction in the scale of autocorrelation (Table 7). Removal of the 98 data points from the small grids around trees in the plots had the opposite effect (Fig. 4), changing variogram

Table 6
Descriptive statistics for the study grids of C structure at the two sites

Statistic	*G. senegalensis*			*P. reticulatum*		
	24×20 m grid	Tree-grid	Shrub-grid	24×20 m grid	Tree-grid	Shrub-grid
μ	0.55 (0.017)[a]	0.72 (0.035)	0.47 (0.016)	0.58 (0.031)	0.75 (0.046)	0.46 (0.012)
σ	0.20	0.35	0.16	0.36	0.46	0.12
Range	1.13	1.99	0.83	2.15	2.63	0.57
CV (%)	36.4	48.6	34.0	62.1	61.3	26.1
n	143	98	98	143	98	98

Shown are means (μ), standard deviations (σ), range and sample size (n) for the soil C measurements.
Tree-grid and shrub-grid refer to the small 3×3 m grids around the tree and shrub, respectively.
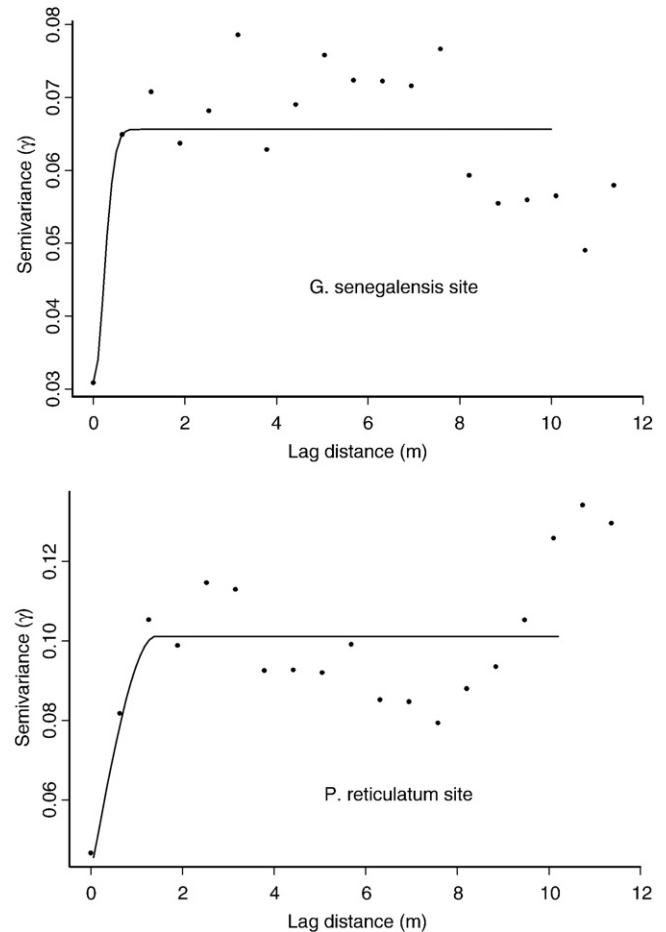[a] Standard error of the mean in parentheses.

Fig. 2. Semivariograms of soil C based on 339 samples from the 480 m^2 plot. Semivariograms stratify calculated variances by the distance (lag) separating each pair of points. Parameter values for the solid line models for each variogram are presented in Table 7.

shapes, increasing the scale of autocorrelation (to ranges that corresponded to half the distance between shrub grids) and spatial dependence to 56 and 94% at the *G. senegalensis* and *P. reticulatum* sites, respectively (Table 7).

Individual plants have been shown to influence soil properties in various ecosystems, including forest trees (Boerner and Koslowsky, 1989), dune grasslands, and tussock grasses (Schlesinger et al., 1990,

Table 7
Summary of model parameters fitted through each of the semivariograms in Figs. 2 through 4

Semivariogram points	Model	Nugget (C_0)	Sill (C_0+C)	Range (m)	$C/(C_0+C)$
G339	Gaussian	0.0308	0.066	0.56	0.53
P339	Spherical	0.0462	0.102	1.34	0.55
Gshrub-grid	Spherical	0.017	0.038	7.46	0.56
Pshrub-grid	Spherical	0.0044	0.079	6.67	0.94
Gtree-grid	Spherical	0.0418	0.085	0.068	0.51
Ptree-grid	Spherical	0.0716	0.150	0.77	0.52

The nugget is the y-intercept of the graph, the sill is the semivariogram value (y value) where each graph becomes a plateau, the range is the distance (x value) where the plateau begins and $C/(C_0+C)$ is the degree of spatial dependence, a ratio of the structural to population variance.
G339 and P339 represent semivariograms constructed based on all the 339 samples from the 24 m×20 m plot at the *G. senegalensis* and *P. reticulatum* sites, respectively. Gshrub-grid and Pshrub refer to semivariograms constructed based on the 143 points that comprise the 24 m×20 m grid plus the samples from the two 3 m×3 m grids around the shrubs. Gtree-grid and Ptree-grid refer to semivariograms constructed based on the 143 points that comprise the 24 m×20 m grid plus the samples from the two 3 m×3 m grids around the trees.
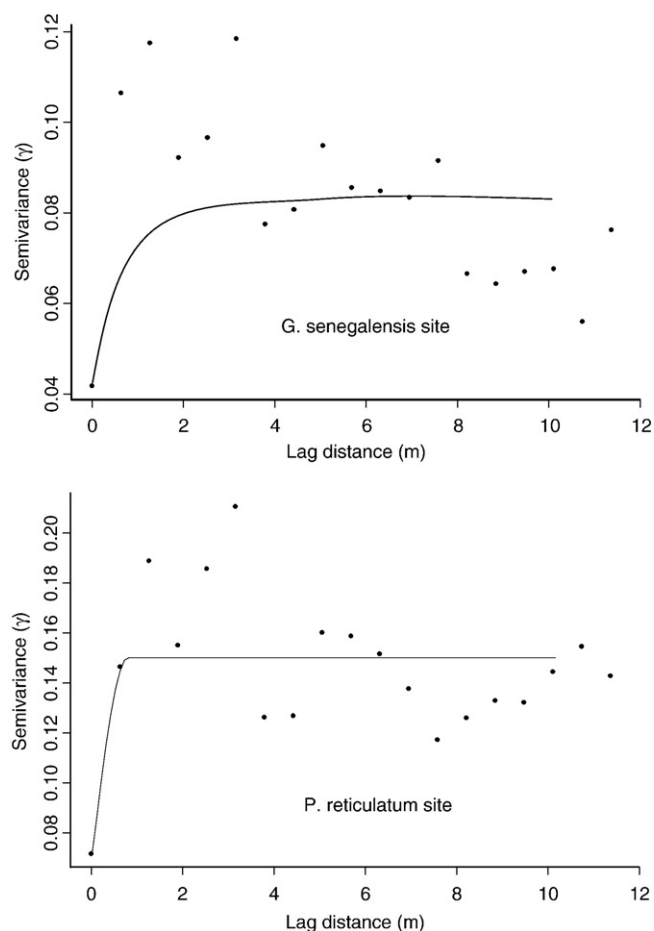
Fig. 3. Separate semivariograms for soil C with the small-grid data around the shrubs removed. Semivariograms were calculated on the 143 points that comprise the 24 m×20 m grid plus the samples from the two 3 m×3 m grids around the trees.

1996). According to Hook et al. (1991), spatial variability in primary production and redistribution of surface soil are the two major processes implicated in generating the heterogeneity of soil properties associated with plant cover in dry areas. The spatial patterns of primary production directly affect the spatial pattern of C through plant litter inputs (Muneto et al., 2001) and decomposition of belowground biomass. There were increases in spatial dependence (relative to the 339 point grid) of soil C at both sites when the 98 points for the small-grid data around the trees were removed from the semivariograms and decreases in spatial dependence when the 98 points for the small-grid data around the shrubs were removed.

These changes in spatial dependence imply that the importance of plant species (shrub or tree) overshadows that of plant presence in structuring soil C patterns at the two sites. At either site, the shrub component has a stronger influence on soil C structure than do the trees although this influence was more pronounced at the *P. reticulatum* site (spatial dependence of 94%). The differences in shrub type influence on soil C patterning could be attributed to differences in biomass productivity, rainfall amounts and surface soil redistribution processes at either site.

The average above- and belowground biomass for *G. senegalensis* were about 0.46 and 2.0 Mg ha^{-1}, and those of *P. reticulatum* were about 0.29 and 3.04 Mg ha^{-1}, respectively (Table 5; dividing C stock values by 0.47 to convert to biomass). The greater total primary production of *P. reticulatum* may lead to higher accumulation of C and hence its stronger influence on soil C variability. On the other hand, a proportion of variability in C measured at the *G. senegalensis* site is most likely due to accumulation of fine materials from the capture

of windblown materials by the shrub canopy (Sterk et al., 1996), hence lessening the direct influence of the shrubs on C patterning. The higher rainfall amounts and low wind speed in the *P. reticulatum* site curtail this possibility and allow the accumulation and stabilization of fallen litter in areas proximal to the shrub. In addition, the long lifespan of *P. reticulatum* means the species can occupy a place for a longer time than the relatively short-lived *G. senegalensis*, and this may amplify the effect of *P. reticulatum* on the local soil properties.

In this study we hypothesized that at the landscape level, shrubs of the parkland systems are greater controls on systemic C stocks than is the tree component. Decades of research have shown that soil variability increases with the area measured (Beckett and Webster, 1971; Palmer, 1990; Nolin et al., 1996). However, in the absence of large-scale gradients in topography, soil depth and parent material (Jackson and Caldwell, 1993b) as is the case in our general study area, overall landscape C variability may not be different than our study grids. This is because within-field variance often does not vary significantly with the size of the field (Beckett and Webster, 1971; Saldana et al., 1998; Conant and Paustian, 2002). Indeed, Ferrari and Vermeulen (1995) demonstrated that pooled soil samples from fields 0.33 to 2.5 ha in size had similar coefficients of variation lending credence to our assertion that at the landscape level we should expect the same C variability and controls in soil C patterning.

4. Perspectives

One approach that has been used to offset significant portions of previous C emissions is increasing woody vegetation density
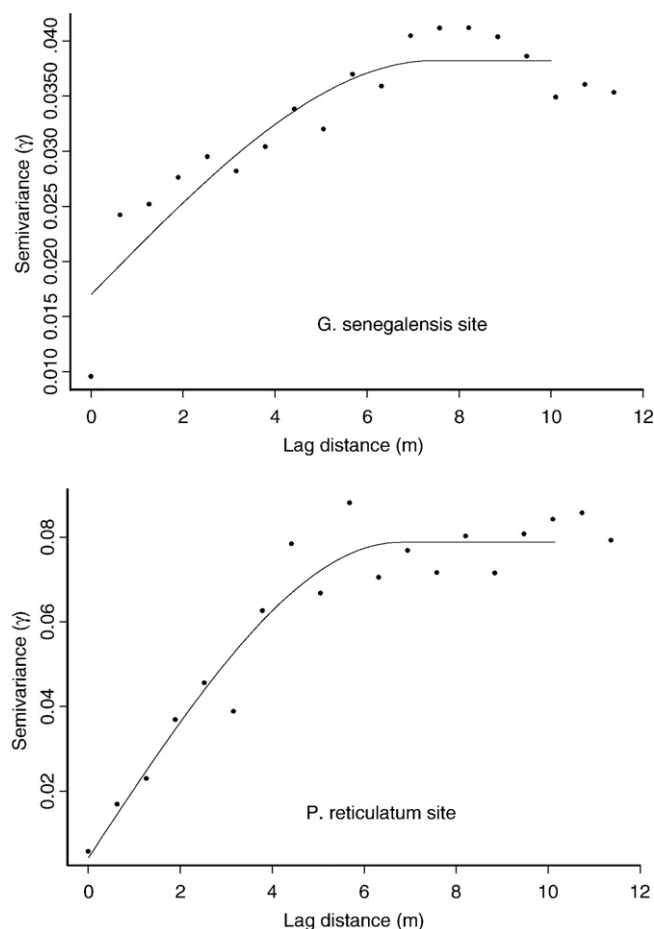

Fig. 4. Separate semivariograms for soil C with the small-grid data around the trees removed. Semivariograms were calculated on the 143 points that comprise the 24 m×20 m grid plus the samples from the two 3 m×3 m grids around the shrubs.

(Houghton et al., 1999; Van Auken, 2000; Pacala et al., 2001; Lal, 2002) and these shrub systems could be a C repository. However, as several authors argue (e.g. Covington et al., 1994; Lal, 1995; Lal et al., 1998a,b), depending on management, this strategy could have potential negative outcomes triggering large C losses, hence exacerbating climate change. Our study is a case in point, indicating a precarious balance between potential gains and rapid losses in C depending on management. If "business as usual" with annual pruning and burning continues in the basin, these systems represent on average a loss of 0.21 and 0.14 Mg C ha^{-1} yr^{-1} in cultivated soils for *G. senegalensis* and *P. reticulatum* sites, respectively. Conversely, using percent regrowth estimates in Table 5 shows that only 0.024 Mg C ha^{-1} (*G. senegalensis*) and 0.03 Mg C ha^{-1} (*P. reticulatum*) is returned to the fields at the second clearing when farmers cannot burn the biomass. This represents a modest 10 and 18% of total annual aboveground biomass C respectively, for *G. senegalensis* and *P. reticulatum*. Lufafa et al. (in press), showed the potential distribution of the shrubs across the Peanut Basin to have approximate areal coverages of 2.34×10^6 and 9.14×10^5 ha, respectively, for *G. senegalensis* and *P. reticulatum*. Tottrup and Rasmussen (2004), report an average cultivation density of 49% in sections of the Peanut Basin for 1999. Using this rather conservative cultivation density estimate and areal coverage of the shrubs reveals that these systems annually lose about 3.51×10^5 Mg of biomass C, an equivalent of 2.11×10^5 Mg of CO_2 per year (Brady and Weil, 1999).

With 98.4% of the land area in the basin under cultivation (Ba et al., 2000), crop residues represent a potential source of C. The biggest percentage of the cultivated land however, is cropped to pearl millet (51.1%) and peanut (38.2%) (Ba et al., 2000) which produce 1.0–2.0 and 0.7–1.0 Mg crop residue ha^{-1}, respectively (Badiane et al., 2000). All of the peanut residue and 50–100% of millet residue is removed from fields and largely used for animal fodder (Badiane et al., 2000) leaving only animal manure as a source of organic inputs. Badiane et al. (2000) determined in field surveys that on average 0.5 Mg animal manure ha^{-1} is applied only to millet fields and since the standard crop rotation is peanut–millet, this results in annual input of 0.25 Mg animal manure ha^{-1}. Furthermore, since only about half of the land in a typical village is under millet, averaging across all fields results in long-term inputs of 0.12 Mg manure ha^{-1} yr^{-1} or 0.048 Mg manure–C ha^{-1} yr^{-1}. Although trees practically contain more biomass C than do the shrubs (Tschakert et al., 2004; Woomer et al., 2004b), the overall fraction of tree C applied to soils is only the leaf material whereas in theory, the entire shrub aboveground biomass could be returned to soils upon pruning. Consequently, our results showed that shrubs are the dominant source of C to soils in the Peanut Basin.

Acknowledgments

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