Vegetation influence on soil quality in a highly degraded tropical soil[1,2]

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ABSTRACT

The influence of various plant species, two leguminous trees (*Andira inermes* and *Albizia procera*), two leguminous covercrops (*Arachis glabrata* and *Centrosema acutifolium*), and two grasses (*Brachiaria humidicola* and *Hemarthria altissima*), on the soil microbial biomass and abiotic parameters, was evaluated in a highly eroded tropical soil of the Corozal series (clayey, mixed isohyperthermic Aquic Haplohumults). Soil samples were taken monthly at two depths (0- to 5- and 5- to 15-cm) from September 1999 to July 2000. Microbial biomass carbon (MBC), microbial biomass nitrogen (MBN), potentially mineralizable N, extractable N, soil organic matter (SOM), and total organic nitrogen (TON) were significantly greater under grasses. Microbial pools and activities were generally higher at the 0- to 5-cm depth. Soil respiration was significantly affected by plant species and date of sampling; in general, soils under grasses had the highest values. The mean proportions of microbial biomass comprising total organic C and N were 2.8 and 1.4%, respectively, in vegetated soils; higher values for C were observed in bare soil. There was a decrease in the mineralizable C proportion of MBC (respiratory quotient) with increasing MBC values. The lowest respiratory quotients were observed for soils under grasses. In this study, soil ecosystem health appears to benefit from vegetation, with soils under grasses exhibiting improved stability due to higher SOM, TON, biologically active C and N pools and lower relative C losses.

Key words: microbial biomass, soil quality, tropical soils, soil restoration

RESUMEN

Influencia de la vegetación sobre la calidad de un suelo tropical degradado

Se evaluó la influencia de diferentes materiales vegetativos [dos leguminosas arbóreas (*Andira inermes* y *Albizia procera*), dos leguminosas cobertoras (*Arachis glabrata* y *Centrosema acutifolium*) y dos gramíneas (*Brachiaria humidicola* y *Hemarthria altissima*)] sobre la biomasa micro-

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biana y parámetros abióticos en un suelo altamente erodado de la serie Corozal (arcilloso, mixto isohipertérmico Aquic Haplohumults). Mensualmente se tomaron muestras de suelos a dos profundidades (0- a 5- y 5- a 15-cm) por un período de once meses. Los valores de la biomasa microbiana de carbono (BMC), la biomasa microbiana de nitrógeno (BMN), N extraíble, N total y la materia orgánica del suelo fueron mayores ($P < 0.05$) en suelos bajo gramíneas como grupo. La reserva microbiana de C y N y su actividad fue mayor a la profundidad de 0- a 5-cm. La respiración microbiana fue afectada significativamente por las especies y el tiempo, siendo los valores mayores en suelo bajo gramíneas. La biomasa microbiana de C y N fue un 2.8 y 1.4% de la totalidad del C y N orgánico del suelo, con valores mayores para C en suelos desnudos (control). El cociente de respiración (C mineralizable/BMC) disminuyó al aumentar la biomasa microbiana y los valores más bajos se obtuvieron en suelos bajo gramíneas. En este estudio, las propiedades que describen la salud del ecosistema del suelo exhiben una posible mejoría debido a la vegetación, con suelos bajo gramíneas manifestando mayor estabilidad debido a mayor contenido de materia orgánica, nitrógeno total, reservas biológicamente activas y pérdidas de C relativamente menores.

Palabras clave: biomasa microbiana, calidad del suelo, suelos tropicales, restauración

INTRODUCTION

Changes in land use have resulted in the rampant removal of overstory vegetation from highly weathered tropical soils (WRI, 2000). The most obvious signs of soil degradation following these activities are increases in soil erosion rates, exposing subsoil horizons which inherently contain less soil organic matter (SOM). The SOM is of fundamental importance for maintaining the fertility and sustainability of tropical soils dominated by low activity clays because of its role in nutrient supply and retention, and because SOM provides much of the soil's cation exchange capacity (Snyder et al., 1993). The soil microbial biomass is an important constituent of SOM which can regulate nutrient supply. Thus, activities which enhance the quality and quantity of organic matter and its components, following degradation, are important (Greenland et al., 1992).

The net rates of C additions and losses determine the amount of SOM at any one time (Lal et al., 1998). Accretion following vegetation removal can include amounts of litter fall from on site vegetative regrowth or off site transport by wind and water, root exudates, and dead root material. Losses from the system include C leaching, erosion, and oxidation. The overall process of humification of organic materials can be influenced by internal factors within the soil plant system, such as the quantity and quality of the organic matter added, clay content, clay type, drainage, soil nutritional status and acidity, and external factors such as temperature, rainfall, and tillage.

Historically, chemical properties have been used as measures of soil fertility (Havlin et al., 1999), yet nutrient supply alone does not ade-

quately describe the state or direction of soil degradation. Indicators of soil quality hold much promise for ascertaining the overall functioning of the soil ecosystem, all of which is important from the perspective of biological productivity, environmental quality, and plant and animal health (Doran and Parkin, 1994). Although SOM is an important soil quality indicator, the response of SOM to changes in vegetation cover and management occurs very slowly; many years may be required to measure changes from perturbation (Gupta et al., 1994; Henrot and Robertson, 1994). There is growing evidence that soil biological pools and biochemical processes may hold potential as early sensitive indicators of soil ecological stress during restoration (Powlson et al., 1987; Sparling, 1992; Barkle et al., 2000). Because of the complex dynamics of soil ecosystems, no single property is satisfactory for the study of the microbial biomass and its activity, all of which is fundamental for maintaining soil quality. A possible solution is to combine the information offered by several parameters.

Because of the role of soils in greenhouse emissions (Henront and Robertson, 1994) much interest has recently been generated in regard to the effects of vegetation removal on changes in soil properties. However, less information has been published on changes in soil quality of degraded soils following restoration with varying vegetative materials. Ramos-Santana et al. (2000) initiated a study of combinations of tropical grasses, turf legumes, shrubs, and tree species for their adaptation to highly degraded humid tropical soils. On the basis of that study, we hypothesized that a series of soil biological parameters could serve as potentially sensitive indicators of ecosystem health and response to changes in land use. The objective of this experiment was to test that hypothesis by using several of the experimental plots of Ramos-Santana et al. (2000).

MATERIALS AND METHODS

The research was conducted at the Agricultural Experiment Station in Corozal, Puerto Rico. The soil is a clayey, mixed, isohyperthermic Aquic Haplohumults (Lugo-López et al., 1995). The field where the plots were established had slopes and slope lengths of approximately 35% and 30 m, respectively. The vegetation from the field was clear-cut and removed from the field on 3 October 1995. The soil had obvious signs of degradation; visible down slope gullies due to water erosion were apparent throughout the field. Meteorological information, which included daily air temperature and precipitation 30 days prior to each sampling date, was collected from a nearby weather station. Mean daily maximum temperatures ranged from 27.2 to 33.9 °C, and mean daily

minimum temperatures ranged from 16.7 to 25.5 °C. Mean precipitation ranged from 14.0 mm (July 2000) to 399 mm (December 1999).

Two leguminous trees, Moca (*Andira inermes* C.W. Wright DC.), and Acacia (*Albizia procera* Roxb. Benth.); two leguminous shrubs, forage peanut (*Arachis glabrata* Benth., also known as rhizoma perennial peanut) and Centrosema (*Centrosema acutifolium* Benth.); and two forage grasses, Brachiaria (*Brachiaria humidicola* Schweickt.) and Limpograss (*Hemarthria altissima* Poir.) were selected among forty-seven species that were established in April 1996. Plant material was planted from plugs previously propagated in the greenhouse.

The experimental arrangement was a randomized complete block with seven treatments (six plant species and an unplanted bare soil as control), with three replications and two sampling depths. The area for each treatment was a plot (1.82 × 1.52 m) that was surrounded by 15-cm-high polyethylene edging installed to a depth of 6 cm to exclude run-on and divert runoff to a collector point for measurement (Ramos-Santana et al., 2000).

Beginning in August 1999, soil samples were collected at monthly intervals at two depths (0- to 5-, and 5- to 15-cm) from within each plot. Composite soil samples were obtained from ten and five subsamples of the 0- to 5- and 5- to 15-cm depth intervals, respectively. Upon transport to the laboratory, soils were passed through a 6-mm sieve and stored at 5 °C until analysis. Soil moisture was quantified gravimetrically for each depth interval and sampling depth; soil bulk density was quantified from known soil volume and oven-dried (105 °C) soil mass measurements.

Soil samples were split in two portions, one of which was either air-dried or humidified to attain 40% w/w moisture for microbial biomass and soil respiration measurements, and the other was air-dried for chemical measurements. This moisture level is the expected field capacity for this soil (Snyder et al., 1993), and ensured that differences among microbial measurements were not due to changes in the instantaneous field soil moisture level. Thus, all microbial measurements represent values under optimal soil water content.

Soil pH was determined by using a glass electrode immersed in the supernatant of a 1:2 soil:water mixture after a 2-h shaking period. Soil organic matter (SOM) was quantified by using the dichromate oxidation technique (Nelson and Sommers, 1982) with a conversion factor from organic C to SOM of 2.24. Extractable P was determined by the Bray 1 method (Bray and Kurtz, 1945). Extractable bases were exchanged with a 1*M* NH_4OAc (pH 7) solution followed by quantification by atomic absorption spectrometry. Soil acidity was quantified by 1*M* KCl extraction of exchangeable H^+ and Al^{+3} followed by titration with standardized 0.1*M* HCl solution (Thomas, 1982). Soil effective cation exchange capac-

ity (ECEC) was determined by cation summation of bases (Ca, Mg, K, Na) and acidity. Total organic N (TON) was determined by Kjeldhal digestion followed by the quantification of N in the distillate by titration using boric acid as indicator (Bremner and Mulvaney, 1982).

Inorganic N was determined by extracting 10 g soil (field moist) with 40 ml of 1 *M* KCl for 1 h, centrifuging ($280 \times g$), and measuring NH_4^+-N and NO_3^--N in the supernatant, using an autoanalyzer (Keeney and Nelson, 1982). A modification of the 7-d anaerobic incubation technique of Myrold (1987) was used as an index of the potentially mineralizable N (PMN) in the soils. Soil (10 g field moist) was incubated with 15 ml distilled water in capped 50-ml centrifuge tubes at 35° C for seven days. Mineralizable N was expressed as the increase in NH_4^+ concentration over the 7-d anaerobic incubation.

Microbial biomass carbon (MBC) and microbial biomass nitrogen (MBN) were quantified by the fumigation extraction method (Vance et al., 1987). A soil sample (40 g fresh wt) was split into two equal portions, one of which was fumigated with ethanol-free chloroform in a glass desiccator; the other was incubated without fumigation at the same moisture content, time period and temperature. At 48 h, the chloroform was removed by suction, and soils were extracted with 100 ml of 0.5*M* K_2SO_4. For determination of microbial C flush, dissolved organic C in the filtered extract was quantified in a UV-Vis spectrophotometer (Shimadzu Corp., Kyoto, Japan)[6] at 280 nm (Ladd and Amato, 1989; Nunan et al., 1998). Calibration curves were prepared with potassium hydrogen phthalate in the range of 0 to 200 mg/kg C. For determination of microbial N flush, the total N content of the extract was determined by a modified Kjeldhal procedure (Sparling and West, 1988). Briefly, the extract (25 ml) was acidified with 0.25 ml H_2SO_4 and heated to 110 to 120 °C until 1 to 2 ml of the solution remained. After cooling, 3 ml of conc. H_2SO_4 was added; a $K_2SO_4/CuSO_3$ catalyst was added, and the solution heated for 3 h at 340 °C. After cooling, the NH_4^+-N content in the solution was distilled and titrated with 0.05 N H_2SO_4 using 4% boric acid-solution as indicator. Soil MBC and MBN were estimated as the flush (extractable C and N from chloroform-fumigated samples minus C and N extracted from nonfumigated samples) divided by a K_c and K_n factor, respectively. A K_c of 0.33 was used to calculate MBC (Sparling et al., 1990; Sparling and West, 1988), and a K_n of 0.54 was used to calculate MBN (Brookes et al., 1985; Joergensen, 1996).

[6]Trade names in this publication are used only to provide specific information. Mention of a trade name does not constitute a warranty of equipment or materials by the Agricultural Experiment Station of the University of Puerto Rico, nor is this mention a statement of preference over other equipment or materials.

Soil microbial respiration was quantified by incubating soil (10 g fresh wt) in a sealed 125-ml flask containing 6 ml of 0.1N NaOH. After a 7-d incubation the NaOH was titrated with standardized HCl by using an automatic titrator (Orion EA 960, Thermo Electron Company, USA). Values of soil respiration were expressed as mg C/kg (dry wt)/wk.

The data were subjected to analysis of variance using GLM procedure (SAS Institute, 1996). Data were analyzed as a randomized complete block in split-plot design. Whole plots corresponded to the six plant species and the control; the sub-plots corresponded to the date of sampling; and the split-split plots corresponded to the sampling depth. The class variable for time was not included in the analysis for extractable bases, pH, acidity, extractable Al, ECEC, or PMN since these were analyzed once. Assumptions of normality and variance homogeneity were corroborated by using Shapiro-Wilks and Brown-Forsythe tests, respectively. Multiple comparisons were performed with a protected Fisher's LSD with a significance level of 0.05 for volumetric water content. Single degree of freedom contrasts were used to make comparisons among means of groups of plants (legume trees, legume shrubs, and grasses) and bare soil (control). Correlation analysis was performed by using the CORR procedure of SAS.

RESULTS AND DISCUSSION

Interactions were not significant for soil physical (bulk density) and chemical properties (SOM, exchangeable bases, exchangeable acidity, and ECEC) evaluated, except for soil volumetric water content. Soil water content was generally higher at the 5- to 15-cm depth than at the 0- to 5-cm depth intervals for all species and the control ($P < 0.05$) except for that of the two forage species. In those two species, soil water content was similar at the two depths (data not shown). There was a general trend for exchangeable bases to decrease and for exchangeable Al to increase with depth from 0- to 5- to 5- to 15-cm depth (data not shown). Extractable soil phosphorus (P) was undetectable (detection limit 0.05 mg P/kg) at the beginning and at the end of the experiment for all plots evaluated. Contrast comparisons of groups showed that bare soil had significantly ($P < 0.05$) higher acidity and exchangeable Al than soils with legumes and grasses, both alone and combined (Table 1).

In the study by Ramos-Santana et al. (2001), the plants that best adapted to the soil conditions in terms of aerial coverage and biomass production were the grasses (*H. altissima* and *B. humidicola*), followed by *A. procera*. Intermediate growth was observed for *A. glabrata*, whereas very poor growth and coverage were observed for *C. acutifolium* and for *A. inermes*. The soil protection provided by the grasses

TABLE 1. *Vegetation effects on soil physical and chemical properties. Values for each individual species and control are the means averaged across depth and time.*

Species	Group	Volumetric water content	pH	Exchangeable bases	Exchangeable acidity	Exchangeable Al	Effective cation exchange capacity
		$cm^3/100\ cm^3$		----------	$cmol_c/kg$	----------	
A. procera	Legume trees	41.14 c[1]	4.66	3.03	4.15	3.92	7.18
A. inermis		41.26 bc	4.46	1.23	6.99	6.64	8.23
B. humidicola	Grasses	45.97 ab	4.72	3.55	3.91	3.63	7.46
H. altissima		47.26 a	4.56	2.52	4.48	4.28	6.99
A. glabrata	Legume shrubs	40.95 c	4.50	1.62	5.80	5.50	7.42
C. acutifolium		40.77 c	4.46	1.75	5.60	5.31	7.36
Control	Bare soil	40.11 c	4.45	1.35	6.25	6.95	7.59
F-test		*	NS	NS	NS	NS	NS
				Contrasts			
Vegetation vs. bare soil		ND[2]	ND	ND	5.31[3]	5.18	NS[4]
					6.25	6.95	
Legumes vs. grasses		ND	ND	ND	5.64	5.34	NS
					4.20	3.96	
Grasses vs. bare soil		ND	ND	ND	4.20	3.96	NS
					6.25	6.95	
Legumes vs. bare soil		ND	ND	ND	5.64	5.34	NS
					6.25	6.95	

[1]Means with different letters within a column are significantly different at $P < 0.05$ using Fisher's protected LSD.
[2]ND denotes contrast was not determined.
[3]Contrast means are significantly different at $P < 0.05$.
[4]NS denotes non-significance at $P < 0.05$.

may have contributed to the observed trend for higher soil pH, greater percentage base saturation, and lower exchangeable Al because of presumably lower nutrient losses due to less leaching and runoff. Further proof of the excellent aerial coverage provided by the grasses is that during the course of the study sediment yields from runoff were 0, 621, and 2,525 g/plot/year for the grasses, *A. glabrata*, and *A. procera*, respectively (Ramos-Santana et al., 2000).

The effects of vegetating a bare soil is apparent, as bare soil had significantly lower SOM and TON than groups of legumes, grasses, and both combined (Table 2). Legumes in general had lower SOM and TON than grasses. The SOM and TON concentrations in the beginning and at the end of the experiment were similar within species and groups of species (contrasts). Conversion of forests of the humid tropics into pastures and agricultural fields is expected to produce in the long term a decline in SOM and soil fertility. In a study in Costa Rica, the largest decrease (about 50%) in SOM occurred between six and 15 months following removal of 20-yr-old secondary vegetation of ferns, grasses, and scattered stands of trees, after which time the rates of SOM decrease

TABLE 2. *Vegetation effects on soil organic matter, total organic N and extractable N pools. Values for each individual species and control are the means averaged across depth and time.*

Species	Group	Soil organic matter	Total organic N	Extractable N
		g/kg	g/kg	mg N/kg
A. procera	Legume trees	15.2	1.6	3.45
A. inermis		9.6	1.2	3.18
B. humidicola	Grasses	15.8	1.7	5.33
H. altissima		17.9	1.7	4.10
A. glabrata	Legume shrubs	9.3	1.2	3.53
C. acutifolium		15.0	1.4	4.54
Control	Bare soil	6.0	1.2	3.66
F-test		NS[1]	NS	NS
		Contrasts		
Vegetation vs. bare soil		12.7[2]	1.45	NS
		6.0	1.21	
Legumes vs. grasses		12.3	1.36	3.68
		16.9	1.76	4.72
Grasses vs. bare soil		16.9	1.76	4.72
		6.0	1.21	3.66
Legumes vs. bare soil		12.3	1.36	NS
		6.0	1.21	

[1]NS denotes non significance at $P < 0.05$.
[2]Contrast means are significantly different at $P < 0.05$.

were lower (Henrot and Robertson, 1994). Since our site was an eroded hillside with presumably low SOM and TON, and about two years had elapsed after removal and replanting of differing vegetative materials in our study, low rates of SOM and TON decrease were observed.

Differences in soil extractable N (NH_4^+-N + NO_3^--N) were not observed between individual species, and legumes did not significantly increase extractable N beyond bare soil (Table 2). Extractable N was significantly higher ($P < 0.05$) in grasses than in legumes or in bare soil. Most of the extractable N was in the NO_3^- form, as soil NH_4^+ was in almost all cases undetectable. Since the presence of NH_4^+-N and NO_3^--N in soil is a function of N microbial mineralization-immobilization rates, competition by plants and microorganisms, leaching, denitrification and volatilization, it is difficult to ascertain the possible reasons for the fluctuations as a function of the observed species × time, and time × depth interactions. Nevertheless, there is a robust population of nitrifying microorganisms in these highly degraded acid and low organic matter soils, all of which was unexpected (Paul and Clark, 1989). The low amounts of extractable N detected suggest that in these highly degraded soils both plant materials and soil microorganisms are N-limited during parts of the year and are thus competing for a limited resource. Under these conditions, it has been shown that plants compete with soil microorganisms better for NO_3^- than for NH_4^+ (Schimel et al., 1989); thus nitrification may be an important mechanism for providing non-legume plants with much of their N nutritional requirements.

Microbial biomass C and N were significantly influenced by time × depth interactions, (Figures 1a and 1b). There may be a complex set of factors influencing MBC and MBN values with respect to depth and time; no significant correlations were observed with soil water contents. Maximum air temperatures fluctuated only 6 °C throughout the experiment. Although a decrease in soil water content was detected from January through April, values were still within the range of what is considered favorable for microbial biomass growth and survival (Paul and Clark, 1989). Soils under grasses had significantly higher MBC and MBN than those under legumes and bare soil (Table 3). Seasonal fluctuations between sampling events may have precluded the detection of statistical differences among individual species.

The precise size of the microbial biomass pool is difficult to predict because of temporal variability, differences among techniques, and conversion factors (K_c and K_n) (Martens, 1995; Jenkinson et al., 2004). Also because of the strong influence of soil management on the soil MBC and MBN pools, it is difficult to compare our values with those of other studies in similar soils. For example, MBC in an oxic Humitropept and

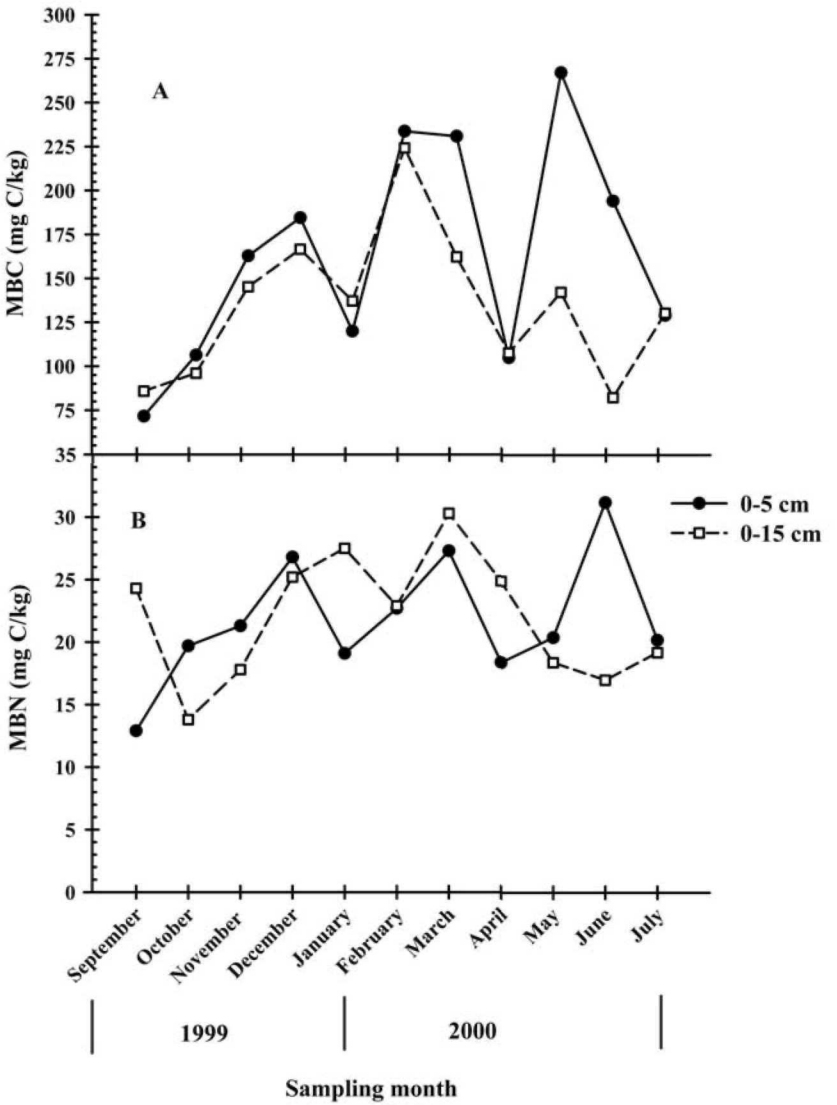

FIGURE 1. Variation of soil microbial biomass C (MBC) (A) and microbial biomass N (MBN) (B) with time at two depths. Each value is the mean of all vegetative species and the control.

in a fluventic Dystrandept was approximately 2,000 and 1,300 mg C/kg, respectively (Henrot and Robertson, 1994). In tropical forest soils of Brazil, Costa Rica, Panama, and Hawaii, MBN ranged from nine to 265 mg N/kg (Vitousek and Matson, 1988). The MBC and MBN values determined in this study appear to be in the low range of values determined in other studies for tropical soils.

Soil microbial respiration was significantly higher under grasses than under legumes and bare soil (Table 3). No significant differences were found between legumes and bare soil. This finding suggests that soils under grasses have higher potentially mineralizable C and N pools because of the increased above- and below-ground biomass production and subsequent decomposition in soil, and production of root exudates (Groffman et al., 2001; O'Donnel et al., 2001). Higher soil respiration ($P < 0.05$) was observed at the surface 0- to 5-cm than at 5- to 15-cm depth, because of stratification of aboveground biomass litter fall.

The potentially mineralizable nitrogen (PMN) pool represents a biologically active amount of N originating from various components of the SOM that could be mineralized by indigenous microorganisms dur-

TABLE 3. *Vegetation effects on soil microbial parameters. Values for each individual species and control are the means averaged across depth and time.*

Species	Group	MBC	MBN	MBC/MBN	PMN	Microbial respiration
		- - - - mg/kg - - - -			mg N/kg	mg CO_2-C/kg
A. procera	Legume trees	140.2	23.17	13.4	21.4	131.9
A. inermis		131.3	15.88	11.0	23.5	115.1
B. humidicola	Grasses	185.0	24.24	10.7	34.7	152.0
H. altissima		193.9	26.93	6.92	28.7	148.2
A. glabrata	Legume shrubs	111.5	20.81	7.68	20.4	134.6
C. acutifolium		148.6	21.66	8.74	25.8	125.7
Control	Bare soil	135.0	20.43	11.3	41.7	119.4
F-test		NS[1]	NS	NS	NS	*
			Contrasts			
Vegetation vs. bare soil		NS	NS	ND[2]	28.0	132.4
					41.7	119.4
Legumes vs. grasses		132.9[3]	20.4	ND	22.8	126.8
		189.5	25.6		31.7	150.1
Grasses vs. bare soil		189.5	25.6	ND	31.7	150.1
		135.0	20.4		41.7	119.4
Legumes vs. bare soil		NS	NS	ND	22.8	NS
					41.7	

[1]NS denotes non significance at $P < 0.05$.
[2]ND denotes contrast was not determined.
[3]Contrast means are significantly different at $P < 0.05$.

ing a growing season in temperate areas (Stanford and Smith, 1972). This PMN pool likely includes the bulk of the active-nonbiomass N and some fraction of the microbial N at the time of sampling (Duxbury and Nkambule, 1994). In this experiment, we expected the PMN pool to be a more sensitive indicator of changes in soil quality than the bulk SOM and TON, because it would represent N compartmentalization into pools that differ in their susceptibility to biological decomposition. Bare soil had higher PMN values than soils with legumes and grasses (Table 3). The higher values observed for the bare soil are probably due to N released from biologically active pools as a result of the disruption of organic N during sieving (Duxbury and Nkambule, 1994). No plants were growing in this soil and it was not physically disturbed for at least three years other than by hand weed control. In contrast, the higher PMN pools of soils under grass as compared to those under legumes are probably due to the accretion of N into biologically active N pools as a result of greater amounts of organic C and N input to the system. These active pools should be physically protected by larger soil aggregate size classes, which in these soils are hypothesized to occur in soils under grasses rather than in bare soil (Duxbury et al., 1989).

Microbial biomass C was significantly correlated to microbial respiration ($r = 0.67$, $P < 0.05$) and SOM ($r = 0.54$, $P< 0.05$); MBN was significantly correlated to TON ($r = 0.80$, $P < 0.05$). This correlation was expected as MBC and MBN presumably are the biologically active components of SOM and TON, respectively. A significant linear correlation between MBN and MBC ($r = 0.72$, $P < 0.05$) suggests that synchronization occurs in the turnover of C and N of these soils. The C:N ratio of the microbial biomass ranged from 6.9 to 13.4 (Table 3). Increasing C:N ratios of the microbial biomass are indicative of a shift in microbial populations from bacterial to fungal dominated (Paul and Clark, 1989). A narrower microbial biomass C:N ratio may also be indicative of a faster C and N turnover potential (Rice and García, 1994).

Biomass C and N in vegetative plots (excluding bare soil) at both 0- to 5- and 5- to 15-cm depths comprised on average 2.8 and 1.4% of the soil organic C and TON, respectively. Biomass C and N in bare soil only, at both 0- to 5- and 5- to 15-cm depths comprised 5.0 and 1.5% of the soil total organic C (TOC) and TON, respectively. There was no clear trend in the MBC/TOC and MBN/TON ratios with regard to specific vegetative materials. The greater proportion of active C pools in unvegetated soils suggests faster soil organic carbon turnover (Rice and García, 1994). Since in unvegetated soils these pools tend to be smaller, these greater proportions of labile C and N could be lost from the soil-plant system. There was no significant correlation between MBC or

MBN and acidity, all of which suggests that soil acidity (which was primarily due to free Al^{+3} in solution) was not a limiting factor for the growth and maintenance of microbial pools in these soils.

Soil respiration values quantified in this study are expected to overestimate those occurring under field conditions because of the effects of disturbance and water addition such as rainfall. Nevertheless, the ratio of soil respiration to BMC, which is termed the respiratory quotient, was calculated to evaluate the possible stability of BMC pools in soils (Anderson and Domsch, 1990; Turco et al., 1994; Priha and Smolander, 1994). This quotient can be a good indicator of the effects of environmental influences on the microbial population, with lower values indicative of more stable or mature systems. The respiratory quotient decreased with increases in MBC, all of which suggests increasing stability in soil C as microbial pools increase (Figure 2). Lowest respiratory quotients are observed for soils under grasses (*B. humidicola* and *H. altissima*). The breakpoint in the two lines suggests that mineralizable C in the soils is more stable at MBC values > 128 mg/kg

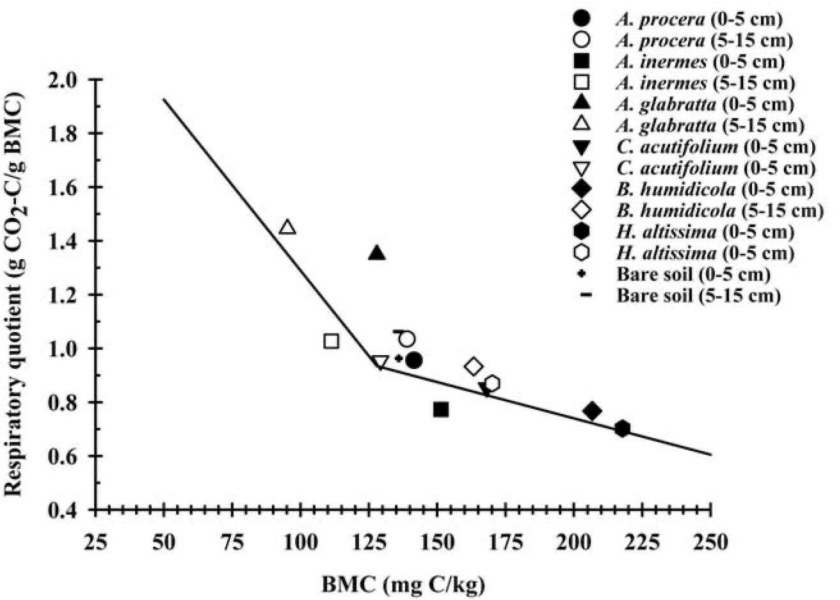

FIGURE 2. Relationships between respiratory quotient (RQ) and microbial biomass C (MBC) with differing vegetative species and bare soil at two depths. Each point is the mean of three replicates. The regression line is RQ = -0.0127*MBC + 2.56 for MBC < 128; and RQ = -0.0027*MBC + 1.28 for MBC >128; $R^2 = 0.85$, n = 41.

(using the fumigation extraction technique). Increasing MBC values may be of benefit for increasing C storage in the soils and for reducing CO_2 emissions to the atmosphere.

This study evaluated the dynamic nature of microbial biomass and ancillary properties which can serve as soil quality indicators of change as a result of vegetating a highly degraded tropical acid soil. Soils under differing vegetative materials receive inputs from aboveground and belowground primary production which influences SOM pools and their fractions. In addition to the effects of vegetation, the parameters studied and their relationships appear to be influenced by environmental factors (moisture, depth, and time after disturbance). A better understanding of microbial processes is needed in order to use bio-indicators to monitor changes in soil quality as a result of soil management practices.

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