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Title: Earthworms, soil fertility and aggregate-associated soil organic matter dynamics in the Quesungual agroforestry system

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Abstract

Issues of food security, environmental degradation and global climate change underscore the need for the improved understanding of sustainable agricultural systems around the globe. The Quesungual slash-and-mulch agroforestry system (QSMAS) of western Honduras offers a promising alternative to traditional slash-and-burn (SB) agriculture for the mountainous tropical dry forest zones of Central America, but the overall influence of this system on soils is not fully understood. We examined earthworm populations, soil fertility and soil organic matter (SOM) dynamics under QSMAS and SB agriculture, with secondary forest (SF) as a reference. Both QSMAS and SB consisted of treatments with and without inorganic fertilizer (N-P-K) additions, resulting in five management treatments, each present on three replicate farms. Baseline soil samples (0-15 cm) were collected prior to forest clearing and establishment of QSMAS plots in 2003 and in SB and SF plots in 2005 to determine initial soil concentrations of C and N. Soils were sampled in 2006 and 2007 for bulk soil C and N and P availability, as well as for aggregate fractionation and determination of C and N within the different aggregate size fractions. Earthworm populations were assessed in July 2007. Earthworm numbers and biomass were higher under QSMAS than under SB (13.4 vs. 0.8 g fresh biomass m^{-2}; respectively). Significant interactions between cropping system and fertilization suggest that QSMAS increased the availability of added inorganic P, 3 times more under QSMAS than for SB. Comparisons with SF, indicated that both cropping systems resulted in a dramatic loss of C (average 5 g C kg^{-1} soil) since treatment implementation and that this loss was mainly associated with the disruption of C rich large macroaggregates (> 2000 µm). After taking into account baseline soil C differences between plots, no major differences in total SOM losses were found between QSMAS and SB management. However, earlier establishment of QSMAS plots suggests that the overall rate of C loss since treatment establishment was lower for QSMAS than for SB. Results from this study suggest that the Quesungual agroforestry system offers great potential to improve soil fertility and biological health in the region relative to traditional slash-and-burn agriculture.

Key words: agroforestry; carbon storage; earthworms; phosphorus availability; slash-and-burn agriculture; soil aggregates

Introduction

Traditional farming systems across much of Latin America have relied on shifting cultivation and the use of fire to prepare land for planting. Many of these farms occupy marginal soils that are prone to erosion and require long fallow periods to restore soil fertility (Szott *et al.*, 1999). Increasing demand for food production and the resultant intensification of these cropping systems has led to widespread soil degradation and declining productivity. To better understand this process, numerous studies have examined the influence of slash-and-burn (SB) agriculture (Giardina *et al.*, 2000; McGrath *et al.*, 2001) as well as improved management of fallow periods (Barrios *et al*., 1997; Phiri *et al*., 2001) on soil organic matter (SOM) and nutrient dynamics. Although burning can temporarily increase the availability of soil nutrients via the rapid mineralization of organic nutrient pools, additions of nutrient rich ash and changes in soil pH, fire can also give rise to long-term nutrient losses through increased erosion and leaching, as well as volatile losses during combustion (Kauffman *et al.*, 1995; Juo and Manu, 1996; Ellingson *et al.*, 2000; Sommer *et al.*, 2004). Relative to intact forest, shifting cultivation has also been shown to deplete soil C by reducing inputs of organic matter and accelerating the mineralization of SOM (Tiessen *et al.*, 1992; Roder *et al.*, 1997; McDonald *et al.*, 2002). This loss of soil C is critical, as SOM is fundamental to long-term soil productivity (Craswell and Lefroy, 2001) and remains highly pertinent to global C cycling and climate change issues (Lal, 2004).

Research on SOM dynamics in agroecosystems has focused extensively on how management influences the stabilization of soil C through effects on soil structure (Paustian *et al.*, 1997; Bronick and Lal, 2005). The physical protection of organic matter within soil aggregates is a key form of SOM stabilization (Tisdall and Oades, 1982) and is perhaps more sensitive to disturbance than other stabilization mechanisms (i.e., biochemical recalcitrance and the binding of organic matter to mineral surfaces). Although C within microaggregates (53-250 μm) is generally considered to be more stable and relevant for long-term SOM storage (Oades, 1984; Angers *et al.*, 1997), macroaggregates (>250 μm) often contain higher concentrations of C (Tisdall and Oades, 1982) and have been suggested to play a critical role in the formation of new microaggregates (Six *et al.*, 2000). Thus, agricultural management which affects macroaggregate dynamics could have far reaching implications for SOM stabilization and agroecosystem sustainability (Six *et al.*, 2000; Denef *et al.*, 2007). Although tillage produces perhaps the most obvious impacts of management on soil aggregates (Beare *et al.*, 1994; Six *et al.*, 1999), other management factors such as fertilization, burning and residue application can have important influences on soil structure and associated C storage as well (Mulumba and Lal, 2008; Fonte *et al.*, 2009). While the potential for SB agriculture to degrade soils is well understood, relatively few studies have specifically addressed effects on soil structure and the implications of this for SOM dynamics. Findings by García-Oliva *et al*. (1999) from the dry forests of Mexico, suggest that the clearing and burning of forest can significantly decrease aggregate stability and associated C storage in surface soils within a single growing season. Although Wairiu and Lal (2003) found less dramatic impacts on soil structure with shifting cultivation, they observed a decline in macroaggregate and whole soil C concentrations, as well as a decrease in large macroaggregates (>5mm) following burning and cultivation.

In light of the potential negative impacts of shifting cultivation on SOM and nutrient dynamics, particularly under a regime of shortening fallow cycles, the need for viable alternatives becomes evident and pertinent (Barrios *et al*., 2005). The present study focuses on the Quesungual slash-and-mulch agroforestry system (QSMAS) in the mountainous region of western Honduras, a promising alternative to traditional SB practices (Hellin *et al*., 1999;

Welchez *et al*., 2008). This system forgoes burning and instead relies on the maintenance of native trees in farmers' fields to reduce erosion on the steep hillsides that dominate the region. These remaining trees are regularly pruned, offering a source of mulch and supplemental nutrients to sustain crop yields for many more years relative to traditional management. Along with the stabilizing effect of trees and the improved residue handling, the lack of burning in the QSMAS may also benefit soil biota, fertility, and structure via numerous mechanisms. Soil fauna are key regulators of organic matter turnover and nutrient cycling and have been identified as major contributors to soil health and productivity (Wardle *et al.*, 1995; Brussaard *et al.*, 2006; Barrios, 2007). As sensitive indicators of management, earthworms are of particular relevance since they can greatly influence soil structure, nutrient availability and crop growth (Fragoso *et al.*, 1997; Blanchart *et al.*, 1999; Lavelle *et al.*, 2001). In addition to stimulating earthworm activity and providing nutrients, residue additions have also been suggested to improve the fertilizer use efficiency (Vanlauwe *et al.*, 2001) and increase nutrient retention and availability in soils (Nziguheba *et al.*, 1998). Finally, residue application can enhance soil structure and contribute to the maintenance of SOM (Mapa and Gunasena, 1995; Mulumba and Lal, 2008).

We aimed to evaluate the influence of agroecosystem management on SOM dynamics, soil fertility and earthworm populations in tropical hillside farming systems. Specifically, we sought to assess the impacts of traditional SB management and QSMAS on soil aggregation, aggregate-associated C and N, P availability, and earthworm abundance, in comparison with secondary forest. We hypothesized that increased organic matter inputs and the lack of burning in the Quesungual system would reduce losses in soil aggregation, C storage, P availability and earthworm abundance relative to SB management.

Materials and methods

2.1 Study site and experimental design

This research was conducted at a field site in the Lempira Department of western Honduras, near the town of Candelaria (N 14°4', W 88°34'). This mountainous region is covered by a patchwork of sub-humid tropical forest interspersed with annual crops and pasture. Soils are generally shallow and rocky, with Entisols comprising the dominate soil type (Hellin *et al*., 1999; Welchez *et al*., 2008). The research plots had slopes of between 30 and 65% and were sandy clay loam in texture. At 400 m in elevation, mean monthly temperatures vary between 22 and 27 °C. Rainfall is bimodal and averages 1400 mm yr^{-1} with a distinct dry season from November to late April. Maize (*Zea Mays*; L.) is generally planted with the onset of the rains in May and typically harvested in October. Beans (*Phaseolus vulgaris*; L.) are commonly planted in these same fields one month prior to maize harvest.

Preliminary investigation on the Quesungual system by the International Center for Tropical Agriculture (CIAT) sought to evaluate the system by establishing experimental plots to monitor changes in soil properties and crop production over time. These plots were installed in 2003 by clearing uncultivated land (forest) located on three farms with similar soil types, slopes, and management histories (> 10 yrs fallow). New project funding obtained in 2004 provided for the establishment of additional land management treatments for comparison with QSMAS. Parcels of remaining forest on the three replicate farms were used for the installation of plots under traditional SB management, as well as for plots maintained under secondary forest (SF) to serve as a reference by which to evaluate the impacts of QSMAS and SB management.

By April of 2005, 10 x 10 m plots were in place with five combinations of management system x fertilizer treatments, each represented once per replicate farm. Treatments included:

QSMAS with fertilizer (QSMAS+F), QSMAS with no fertilizer added (QSMAS-F), SB with fertilizer (SB+F), SB with no fertilizer added (SB-F), and SF. In the QSMAS treatment, the forest was selectively thinned and pruned. Large woody debris was removed, while leaves and small branches were left on the soil surface to decompose. In the SB treatment, the secondary forest within the plots was slashed and burned, thus leaving the soil bare at planting. Starting in 2003 for the QSMAS plots and 2005 for plots under SB management, inputs of inorganic fertilizer in the +F treatments totaled 147 kg N ha^{-1} yr^{-1} and 106 kg P ha^{-1} yr^{-1} for both maize and the following crop of beans, according to typical application rates for farms in area. Unfertilized plots received no inorganic fertilizer inputs since study establishment. Pruning and residue management (mulching or burning) for the QSMAS and SB plots respectively, were conducted on an annual basis several weeks prior to the planting of maize. Disturbance in the plots was minimized, as crops were planted by direct seeding (i.e., no tillage) and animals were excluded from all plots for the duration of the study.

2.2 Soil sampling and preparation

Soil samples for aggregate fractionation (two samples per plot) were collected in two successive years during the dry season (April 2006, and April 2007) and once in the wet season (July 2007) by digging a small pit and cutting away an intact soil core (8 x 8 cm square) to a depth of 15 cm with a knife. These cores were kept cool until returned to the lab, at which point the field moist soils were passed through an 8mm sieve by gently breaking soil clods along natural planes of weakness and then air-dried for subsequent analyses. A subsample of the air-dried soil was passed through a 2mm sieve to remove large rocks and organic debris and used for determination of total C and N, as well as available P.

Archived soil samples from CIAT provided baseline values of total soil C and N prior to the establishment of experimental plots. These samples include soil taken from the QSMAS plots in July of 2003 (prior to forest disturbance) and soil from all plots in April 2005 (prior to establishment of the SB and SF plots). Nine soil samples were removed by grid sampling and composited to provide a single value for each plot. In the lab, these soils were air-dried and passed through a 2 mm sieve to remove rocks and debris in preparation for storage and eventual analysis for total C and N.

2.5 Earthworm sampling

Earthworms were sampled in all plots in July 2007, adjacent to the soil samples (two per plot), by rapid excavation and hand-sorting of pits (25 x 25 x 30 cm deep; Fonte et al. 2009). Earthworms were not sampled in the dry season due to low soil moisture and negligible activity. Subsequent to collection, soils below 30 cm in each pit were inspected for the presence of earthworm burrows to assess the potential for deeper dwelling species not captured by our sampling. Earthworms were returned to the lab, cleaned and weighed to determine fresh biomass (including gut contents). Sub-samples of the collected earthworms from each treatment were then preserved in 4% formalin and sent to the Kansas University Natural History Museum and Biodiversity Research Center for identification by Dr. Sam James.

2.3 Aggregate fractionation

Sub-samples of the 8mm sieved, air-dried soils were fractionated by wet-sieving based on methods outlined by Elliott (1986). Briefly, 50 g soil was submerged in deionized water for 5 min on a 2000 µm sieve for slaking. The sieve was then moved up and down (in and out of the water) 50 times in a circular motion over the course of 2 min. Material remaining on the top of the sieve was gently washed into a pre-weighed aluminum pan. Smaller soil particles passing through the 2000 µm sieve were then transferred to a 250 um sieve and the above process repeated with both a 250 um and a 53 um sieve, yielding four aggregate fractions: large macroaggregates (> 2 mm), small macroaggregates (250-2000 µm), microaggregates (53-250 µm) and silt and clay (<53 µm). All fractions were placed in the oven at 60 °C until dry and then weighed to determine the proportion of the whole soil in each fraction. Mean weight diameter (MWD) was calculated following van Bavel (1950), by summing the weighted proportions of each aggregate fraction, and used as an indicator of aggregate stability.

Large and small macro aggregates were then further separated according to Six *et al*. (2000). Oven-dried macroaggregates (6 g) were slaked in deionized water for 20 minutes. Following slaking, these samples were then shaken with 50 stainless steel bearings (4 mm diameter) on top of a submerged 250 µm mesh screen until all macroaggregates were broken. A continuous flow of water washed the small material (< 250 µm) through the mesh and helped avoid the further breakup of microaggregates released from the macroaggregates. Material passing through the 250 µm mesh was then wet-sieved (as described above) through a 53 µm sieve, thus generating three fractions for each macroaggregate size class: coarse sand and particulate organic matter (cPOM; >250 µm), microaggregates within macroaggregates (mM; 53-250 µm) and macroaggregate occluded silt and clay (Msc; <53 µm). Each of these fractions was transferred to pre-weighed aluminum pans and dried at 60°C before weighing.

2.4 Nutrient analyses and calculations of rate of SOM change

Sub-samples of bulk soil and all oven-dried soil fractions were ground and then analyzed for total C and N concentrations using a PDZ Europa Integra C-N isotope ratio mass spectrometer (Integra, Germany). Total C in all fractions was considered to be organic C, since no evidence for carbonates was found for this soil. Subsamples of the 2 mm sieved bulk soils collected on the three sampling dates in 2006 and 2007 were sent to the Agriculture and Natural Resources (ANR) Analytical Laboratory (http://groups.ucanr.org/danranlab/) at the University of California, Davis for determination of P availability (Olsen P) according to Olsen and Sommers (1982).

The rate of SOM loss in each treatment was calculated by dividing the overall change in total soil C, since treatment establishment (2003 for QSMAS and 2005 for SB and SF), by the number of years that each treatment had been in place.

2.6 Statistical analyses

Data from the two soil sub-samples from each plot were combined for each sampling date and average values were used for all statistical tests. Differences between the five treatments were analyzed using ANOVA with farms treated as blocks and considered a random variable. In order to account for system differences in initial SOM, analyses of total soil C and N were also conducted using ANCOVA, with baseline soil C and N levels used as covariates. Due to the unbalanced design of the experiment (i.e., no fertilization treatments for SF), the principal comparisons of interest in this study (cropped vs. secondary forest plots, fertilized vs. unfertilized, QSMAS vs. SB, and the cropping system x fertilizer interaction) were carried out using orthogonal contrasts. This approach allows for the direct comparison of the most meaningful treatment combinations, while best utilizing the available degrees of freedom. Thus, all differences reported below are based on orthogonal contrasts unless noted otherwise. Comparisons were conducted independently for each sampling date and natural log transformations were applied as needed to meet the assumptions of ANOVA. Simple linear regression was used to examine the relationships between soil structural indicators and total soil

C. Significance level for all statistical tests was set at P= 0.05, unless reported otherwise. All analyses were conducted using JMP 7.0 statistical software (SAS Institute Inc, 2007).

Results

3.1 Earthworm populations

Earthworm populations were significantly influenced by land management system. Contrasts suggested that total fresh biomass was significantly higher under QSMAS (13.4 g m^{-2}) than SB (0.8 g m^{-2}; Fig 1), with SF having intermediate values (6.2 g m^{-2}) and not significantly different from either treatment. Earthworm abundance followed the same trend with significantly lower population density under SB (11 individuals m^{-2}) than for QSMAS (92 individuals m^{-2}; Fig 1), while SF, with 37 individuals m^{-2}, was not statistically different from either treatment.

The earthworms identified from each treatment were dominated by *Pontoscolex corethrurus* (Müller). While this was the only species found in the SB and SF plots, three additional earthworm species were found under QSMAS management. These include an Anteoides sp. and two less certain classifications (likely in the Eodrilus and Dichogaster genera). Both the Anteoides and the Eodrilus sp. are likely new, previously unidentified species (Sam James, pers. comm.).

3.2 Phosphorus availability

Orthogonal contrasts revealed a significant effect of treatment on P availability (Olsen P) for both the April 2006 and July 2007 sampling dates (Fig. 2). Although the anticipated effect of fertilizer application on Olsen P was apparent for both the April 2006 and the July 2007 sampling dates, there were also significant cropping system x fertilizer interactions on both dates. This interaction indicated that P availability increased with fertilization 5.2 times more under QSMAS versus SB management for April 2006 and 1.9 times more for the July 2007 sampling date (Fig. 2). A similar, yet insignificant trend was apparent for the April 2007 sampling date.

3.3 Impacts on soil structure

Management effects on soil structure appeared to become more pronounced with time, as a greater number of significant differences were observed for the 2007 sampling dates than for samples taken in 2006 (Table 1). The season of sampling (dry vs. wet) may have also played role, as evidenced larger differences in MWD for July 2007 (wet season) than for April of 2006 and 2007 (dry season; Fig 3). In general, the most significant contrasts were observed between cropped versus uncropped (SF) systems, while the type of cropping system employed (QSMAS vs. SB) and fertilization yielded lesser effects (Table 1). The influence of forest conversion to agriculture on soil structure was most evident from differences in MWD between SF and the two cropping systems for both the April and July 2007 sampling dates. For example, in July 2007, orthogonal contrasts revealed that MWD was significantly higher for the SF treatment (2662 µm), than for the average of the four cropping treatments (1468 µm; Fig. 3). Contrasts also revealed that MWD was significantly higher for the SB treatments than the QSMAS plots for the same sampling date (1637 vs. 1298 µm).

The impacts of cropping on soil structure are also evident from changes in the distribution of soil among the different aggregate size classes (Table 1). The disturbance associated with forest clearing generally decreased the proportion of large macroaggregates and increase the percentage of all other size fractions. The type of cropping system in place also yielded significant differences during the final July 2007 sampling, where the proportion of large macroaggregates was lower and that of microaggregates higher in the QSMAS versus SB management (Table 1).

3.4 C and N dynamics

Soil under SF tended to have the highest concentration of total C relative to the two cropping systems, however; this difference was only marginally significant ($P = 0.076$) on the final July 2007 sampling date. Baseline soil samples indicate that SB plots started with higher soil C than QSMAS or SF plots at the time of treatment establishment. Thus, taking into account initial differences in SOM using ANCOVA, soil C and N under SF were found to be significantly higher than for the cropping systems for all sampling dates. The type of cropping system employed also yielded significant effects, as SB demonstrated greater soil C and N than the QSMAS plots for the July 2007 sampling (Table 2; N data not shown). However, this difference disappeared when initial soil C values were used as a covariate. When examining overall rates of SOM loss since 2005, differences between QSMAS and SB plots were not significant, with an average of 2.4 g C kg^{-1} soil lost each year. However, taking into account that the QSMAS plots were converted from forest nearly 2 yrs earlier than plots under SB management, the rate of loss for the QSMAS system since forest conversion (1.4 g C kg^{-1} yr^{-}1) was significantly less than that observed under SB (2.5 g C kg^{-1} yr^{-1}). Overall soil C loss under SF was small (0.5 g C kg^{-1} yr^{-1}) and significantly lower than under the two cropping systems (Fig. 4).

The influence of management on SOM storage in this study seemed to largely correspond with effects on soil structure. This was evidenced by significant correlations between MWD and bulk soil C concentrations on all sampling dates ($P \leq 0.028$, $R^2 \geq 0.31$). Additionally, more C and N was stored in large macroaggregates under SF than under SB or QSMAS on all sampling dates. This was most evident on the July 2007 sampling date, where large macroaggregates under SF held 8.84 g C kg^{-1} soil versus an average of 3.52 g C kg^{-1} soil in the two cropping systems (Table 2). The increases in the amount C and N stored in large macroaggregates under SF corresponded with higher C and N storage in all of the large macroaggregate components (cPOM, mM and Msc) in SF relative to QSMAS or SB for all sampling dates (Table 2). As indicated by changes in soil structure, C and N stored within free microaggregates and the free silt and clay fraction were relatively lower under SF compared to the cropping systems for both the April and July 2007 samples (Table 2). Patterns of N storage within the aggregate fractions (data not shown) closely resembled the trends seen for C, as would be expected due to the close association between these two elements.

Discussion

4.1 Management impacts on soil fauna

Although earthworm populations reported in this study are relatively low, they fall within the range of reported abundance for other studies under similar climatic conditions in the region (Fragoso and Lavelle, 1992; Oritz-Ceballos and Fragoso, 2004; Huerta *et al.*, 2006). The comparatively high abundance of earthworms observed in the QSMAS system (Fig. 1) likely results from a number of factors. The large quantity of high quality litter applied in this system as well as tree root inputs likely serve as key food sources for soil fauna, while both the presence of a tree canopy and the mulch layer act to regulate fluctuations in temperature and moisture (Oritz-Ceballos and Fragoso, 2004; Sanchez De Leon *et al.*, 2006; Sileshi and Mafongoya, 2007). Slash-and-burn management, in contrast, leaves few trees intact and essentially eliminates the mulch layer. Soil heating, due to burning, may also contribute to earthworm mortality in these shallow soils. Although not significant, QSMAS generally displayed greater earthworm populations than soils under SF (Fig. 1). Soils under SF experience greater shade cover and continuous mulch layer compared to QSMAS management; however, litter inputs under SF are likely of lower quality since they consist mainly of senesced leaves (Fonte and Schowalter, 2004) and may provide reduced nutritional value for earthworms. Furthermore, litter inputs under SF may include proportionally more inputs from tree species with low quality litter, as these tend to be selected against by farmers during forest conversion to QSMAS (Ordonez Barragan, 2004).

The dominance of *P. corethrurus* in our samples was not surprising given that this species is a highly successful colonizer and is widely distributed throughout the humid tropics, particularly in disturbed ecosystems (Lavelle *et al.*, 1987). Although diversity cannot be accurately evaluated since all specimens were not preserved for identification, the sub-sample of identified earthworms suggests that the QSMAS may support a higher diversity of earthworms; four species total were found in under QSMAS compared to only one (*P. corethrurus*) in the SB and SF plots. Several studies have reported higher earthworm diversity in less disturbed sites (Brown *et al.*, 2004; Sanchez De Leon *et al.*, 2006), which explains the low apparent diversity in the SB plots. The low diversity observed under SF may be due the lower overall populations supported by this system (resulting from lower nutritional value of inputs) or simply an artifact of the limited sampling in this study.

4.2 Cropping system and phosphorus availability

As a frequent limiting nutrient throughout much of the tropics, P (along with N) has been identified as an important determinant of crop yields in this region (Ordonez Barragan, 2004). As expected, fertilizer additions (composed of inorganic N and P) significantly increased the availability of P in soils (Fig. 2). However, the significant interaction between cropping system and fertilizer application observed in this study indicates that within plots receiving fertilizer additions, P was more available under QSMAS compared to SB management. Although inputs of organic P (as applied residues) may contribute to P availability in the QSMAS system, there was no difference between unfertilized QSMAS vs. SB plots (Fig. 2), thus suggesting that the greater increase in available P with fertilization in the QSMAS system was due to other factors. Fertilization may have instigated a higher rate of P mineralization from residues in the QSMAS plots, or alternatively, the added residues may have helped to maintain the added inorganic P in a more available state. This is consistent with findings by Phiri *et al*. (2001) who showed that slash and mulch management of improved fallow agroforestry species can increase both the readily available and the readily mineralizable P in P-fixing Andean soils. Further, findings by Nziguheba *et al*.(1998) suggested that organic byproducts of high quality residues, specifically *Tithonia diversifolia* (Hemsley), can compete with P for sorption sites in the soil and therefore increase the availability of added triple super phosphate. Another possibility is that increased surface runoff and erosion under SB management as reported by Rivera Peña (2008) may have promoted the loss of the relatively soluble, inorganic P that was added to these plots. Regardless of the mechanism responsible, the higher P availability in the QSMAS system indicated here (Fig. 2) likely contributes to its success and continued adoption throughout the region.

4.3 Influences on soil structure

In this study the greatest impacts on aggregate stability and SOM storage were associated with the conversion from native forest to cropping systems. A number of studies have reported the degradation of soil structure following forest conversion to agricultural lands (Garcia-Oliva *et al.*, 1999; Spaccini *et al.*, 2001; Ashagrie *et al.*, 2007). Although reduced aggregation following cultivation is often attributed to the mechanical disturbance associated with tillage (Beare *et al.*, 1994; Six *et al.*, 1999), the cropping systems in this study experienced only a minimum of such disturbance, as farmers typically sow maize and apply fertilizer into small holes (2-5 cm deep) made with a machete and no other form of soil displacement occurs. Thus, most of the impacts of forest conversion on soil structure are likely attributable to other mechanisms such as changes in organic matter inputs, altered microclimate or loss of top soil due to erosion. The application of residues has been shown to improve aggregation in agricultural systems (Mapa and Gunasena, 1995; Mulumba and Lal, 2008; Gentile *et al.*, 2009) and continual inputs in the SF plots may help maintain a high level of aggregation in this system. However, large quantities of residue inputs in the QSMAS system did not appear to improve soil structure compared to SB for any of the sampling dates (Fig. 3). This suggests other factors (in addition to residue inputs) may be responsible for the reduced aggregation in these systems. For example, both cropping systems were associated with a dramatic loss of shade cover due to heavy pruning and tree removal, which likely leads to more extreme diurnal fluctuations in soil temperature and moisture. Increased drying and wetting of soils associated with the loss of shade cover could account for some of the observed decrease in aggregate stability in the two cropping systems (Denef *et al.*, 2001). Finally, although Rivera Peña (2008) found significantly reduced erosion under QSMAS compared to SB, soil losses were still greater than that observed under native vegetation. Thus differences in aggregation between cropped and SF soils may be at least partly related to the loss of SOM rich top soil in both systems.

Although the high aggregate stability observed under SF was expected, differences between the two cropping systems are contrary to what was anticipated. We hypothesized that higher residue inputs and enhanced protection of the soil surface would lead to improved aggregation in under QSMAS relative to SB. However, this trend was not apparent on any of the sampling dates (Fig 3), and the reverse appeared true for the final, wet-season sampling in July of 2007. We speculate that much of this discrepancy results from differences in baseline SOM content between the different experimental plots. Given that decay products from SOM can greatly contribute to aggregation (Tisdall and Oades, 1982) and that a clear positive relationship exists between MWD and soil C content, it seems that higher SOM level at the start of the experiment in the SB plots would likely be associated with improved soil structure.

4.4 Implications for SOM

Baseline values for total soil C and N allowed for a valuable assessment of treatment effects on SOM. These comparisons revealed a considerable impact of cropping (vs. SF) on SOM storage. This result agrees with a number of studies that have documented soil C loss following conversion of forest to agriculture (Garcia-Oliva *et al.*, 1999; McDonald *et al.*, 2002; Wairiu and Lal, 2003). Of particular relevance are the findings of McDonald *et al*. (2002) who observed large losses of SOM following forest conversion to both agroforestry and traditional agricultural systems.

The effects of management on SOM in this study largely corresponded with the changes observed for soil structure. Treatment differences in C and N storage among the various aggregate fractions essentially mirrored treatment effects on the distribution of soil among the different aggregate size classes (see Tables 1 and 2). This suggests that changes in SOM storage were mainly due to the redistribution of soil among the different aggregate fractions and less so to enrichment or depletion of C and N within these fractions. Specifically, alterations to SOM appeared to be driven largely by the proportion of large macroaggregates in soils, a fraction which tended to be enriched in C and N relative to smaller aggregate size classes. Large macroaggregates thus appear to be a sensitive and highly relevant fraction for measuring changes in SOM related to management associated disturbance in these soils (Tables 1 and 2).

According to the principles of aggregate hierarchy, organic matter content increases with aggregate size class, since larger aggregates are themselves composed of smaller aggregates and organic binding agents which hold them together (Tisdall and Oades, 1982). Aggregates in this study displayed such a trend (data not shown), thus indicating that particulate organic matter that was once protected within large macroaggregates under forest was released following the break-up of this fraction (due to forest conversion) and subsequently lost to decay. This idea is further supported by a loss of cPOM-C and other large macroaggregate components in large macroaggregates with conversion of forest to agriculture. The mM fraction, in particular, has been put forth as a sensitive indicator of disturbance driven changes in SOM (Six *et al.*, 2000; Denef *et al.*, 2007). While the Msc fraction of large macroaggregates appeared to be the most sensitive indicator of SOM loss, with a 65% decrease of C contained in this fraction (or a loss of 1.6 C g kg^{-1} soil on the July 2007 sampling date), the mM fraction demonstrated the greatest total loss of large macroaggregate C observed in this study. In July 2007, the two cropping systems on average contained 3.0 g C kg^{-1} soil less in this fraction than did soils under the native forest (Table 2), thus highlighting the potential for forest conversion to significantly impact long-term SOM storage (Six *et al*. 2000). Previous studies have shown similar changes in macroaggregate-associated SOM between soils under native vegetation vs. cultivated land (Six *et al.*, 2002; Denef *et al.*, 2007; Tan *et al.*, 2007). It should be noted that our findings may not be entirely applicable to other tropical regions, where organic matter stabilization is driven more by interactions with clay minerals than by occlusion within soil aggregates (Zotarelli *et al.*, 2005; Lehmann *et al.*, 2007).

The fact that disturbance associated with forest conversion occurred nearly two years earlier in the QSMAS plots complicates the interpretation of comparisons conducted in this study somewhat, but still allows for meaningful conclusions about the long-term dynamics of these systems. We note that SB practices are typically carried out for only 2-3 yrs before fields must be abandoned to fallow. Thus, our study which follows soil development for more than 2 yrs under SB provides highly relevant information about the longer term consequences of SB management. At the same time, plots under QSMAS have been shown to remain productive for a much longer time (5-10 yrs). Thus evaluation of QSMAS later into the development of this system (3-4 yrs) is perhaps more relevant for understanding how this system impacts long-term fertility and SOM. Baseline soil values helped account for the apparently higher levels of SOM under SB versus QSMAS management and provided critical insight as to how total soil C and N changed over time in the different treatments. Interestingly, SOM in the QSMAS plots remained relatively stable from the time of forest conversion, in July 2003, to the time of complete study implementation in April 2005 (data not shown). This finding suggests that the impacts of forest conversion are slower to materialize under QSMAS management compared to SB. We speculate that high levels of soil C were maintained during this time interval due to the relatively large quantity of organic inputs associated with forest conversion. Under QSMAS the application of all residues (from both pruned and completely removed trees) represents an input of biomass that is likely several times greater than what normally falls to the soil surface under native forest in a single season. Although this large input of residues under QSMAS appears to counterbalance the impacts of cropping on SOM in the short-term, litter inputs in subsequent years decreased markedly with continued tree mortality and inadequate promotion of new saplings. This decline in organic matter inputs apparently led to the eventual depletion of SOM in the QSMAS plots (evident from the 2007 samples). Regardless, QSMAS did suggest slightly lower (albeit non-significant) rates of soil C loss than plots under SB between 2005 and 2007. Furthermore, when the full time period of cropping is taken into consideration (4 vs. 2 yrs for QSMAS and SB; respectively), QSMAS management appears to delay soil C losses substantially compared to SB (Fig 4), where the vast majority of C inputs are lost to combustion upon forest conversion.

5. Conclusions

Identifying sustainable land management practices is critical to addressing long-term issues of food security, environmental degradation and global climate change. This study evaluated the impacts of shifting cultivation and agroforestry, a widespread form of land management throughout the tropics, on soil fauna, fertility and SOM dynamics. Our findings suggest that conversion of forest to agriculture can have large impacts on SOM storage through effects on aggregate associated C and N, with the disruption of organic matter rich large macroaggregates being of particular relevance. Results also indicate that the Quesungual agroforestry system has great potential to improve soil biological health and the efficacy of inorganic fertilizers applied throughout the region. Although the effects on long-term SOM dynamics is less clear, at least an initial delay of soil C losses in the Quesungual system may represent an improvement over traditional practices. In the search for truly sustainable land management options for small farmers, results from this study indicate that although the Quesungual may not be perfect, it offers a greatly improved option for communities in mountainous tropical dry forest zones throughout the region.

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Table 1: Distribution of soil aggregate fractions under different land management and fertilizer treatment combinations (SF = secondary forest, QSMAS = Quesungual slash-and-mulch agroforestry system, SB = slash-and-burn agriculture, +F = fertilized, -F = unfertilized) for 3 sampling times at a field site in western Honduras.

	Large macros	Small macros	Micros	Silt & clay	cPOM[a]	mM[b]	Msc[c]	cPOM	mM	Msc
	>2000 µm	250-2000 µm	53-250 µm	<53 µm	>250 µm	53-250 µm	<53 µm	>250 µm	53-250 µm	<53 µm
	% of whole soil				% of large macros			% of small macros		
Treatment					**April 2006**					
SF	46.4	35.7	12.6	5.3	19.7	51.6	28.7	30.1	50.8	19.2
QSMAS +F	36.1	40.0	17.5	6.4	18.6	57.7	23.7	29.0	52.6	18.4
QSMAS -F	32.8	42.5	17.9	6.8	22.1	54.5	23.3	34.8	48.3	16.9
SB +F	32.9	43.6	17.0	6.5	24.3	55.4	20.4	28.0	54.4	17.7
SB -F	35.8	38.5	17.6	8.0	19.0	53.3	27.6	27.8	52.4	19.8
	A						A F I			
Treatment					**April 2007**					
SF	53.6	35.3	8.2	3.0	25.6	53.0	21.3	29.0	51.4	19.6
QSMAS +F	27.2	49.6	18.3	4.8	19.7	60.6	19.7	24.2	58.4	17.4
QSMAS -F	30.7	46.0	18.4	4.9	21.0	58.8	20.1	26.9	55.3	17.8
SB +F	33.5	45.1	16.7	4.7	17.7	62.8	19.5	20.4	61.6	18.0
SB -F	36.3	45.5	13.6	4.6	24.6	54.9	20.5	23.6	58.2	18.1
	A	A	A	A		A F			A	
Treatment					**July 2007**					
SF	43.6	41.1	12.1	3.2	23.2	57.0	19.9	25.3	56.6	18.1
QSMAS +F	13.6	52.9	26.7	6.8	22.5	59.4	18.1	35.1	49.1	15.7
QSMAS -F	13.4	50.0	29.5	7.1	28.3	54.5	17.2	39.0	44.0	17.0
SB +F	16.8	52.2	24.7	6.3	26.0	57.5	16.4	28.6	55.5	15.9
SB -F	25.2	45.5	23.1	6.2	19.4	61.2	19.3	28.8	53.9	17.3
	A S	A	A S	A				S	A S	

[a] Coarse sand and particulate organic matter
[b] Microaggregates within macroaggregates
[c] Macroaggregate occluded silt and clay

Orthogonal contrasts significant at the P = 0.05 level
A Agricultural disturbance (QSMAS and SB) vs. SF
F Fertilization effect: +F vs. -F
S System effect: QSMAS vs. SB
I Interaction effect: Fertilizer x system interaction [(QSMAS +F) – (QSMAS -F) vs. (SB +F) – (SB -F)]

Table 2: Distribution of soil C among aggregate fractions under different land management and fertilizer treatment combinations (SF = secondary forest, QSMAS = Quesungual slash-and-mulch agroforestry system, SB = slash-and-burn agriculture, +F = fertilized, -F = unfertilized) for 3 sampling times at a field site in western Honduras.

	Large macros	**Small macros**	**Micros**	**Silt & clay**	**Large macros**			**Small macros**		
					cPOM[a]	**mM**[b]	**Msc**[c]	**cPOM**	**mM**	**Msc**
	>2000 µm	250-2000 µm	53-250 µm	<53 µm	>250 µm	53-250 µm	<53 µm	>250 µm	53-250 µm	<53 µm
	g C kg^{-1} soil				**g C kg^{-1} soil**			**g C kg^{-1} soil**		
Treatment					**April 2006**					
SF	9.93	6.94	2.28	1.24	1.24	4.73	3.58	1.47	3.88	1.62
QSMAS +F	8.03	6.69	2.45	1.25	0.98	4.47	2.26	1.14	3.94	1.60
QSMAS -F	6.49	6.64	2.53	1.40	0.83	3.72	1.96	1.30	3.89	1.49
SB +F	7.91	9.14	3.28	1.76	0.97	4.65	2.06	1.33	5.90	2.26
SB -F	7.07	6.53	2.35	1.52	0.62	3.70	2.54	1.02	3.86	1.71
	A				A	A	A			
Treatment					**April 2007**					
SF	10.78	6.38	1.47	0.78	1.81	5.87	2.88	1.24	3.71	1.71
QSMAS +F	6.09	7.23	2.22	0.95	0.76	3.81	1.33	0.77	4.84	1.65
QSMAS -F	5.74	5.80	2.07	0.91	0.72	3.54	1.32	0.64	3.86	1.51
SB +F	6.59	6.86	2.17	1.00	0.82	4.45	1.57	0.60	4.83	1.71
SB -F	6.74	6.73	1.84	0.94	0.75	4.11	1.81	0.69	4.53	1.77
	A		A	A	A	A	A	A		
Treatment					**July 2007**					
SF	8.84	7.03	1.82	0.63	1.12	5.25	2.48	1.09	4.17	1.62
QSMAS +F	3.10	8.15	3.42	1.13	0.49	1.94	0.71	1.27	4.91	1.68
QSMAS -F	2.59	6.28	3.21	1.04	0.37	1.59	0.63	1.03	3.42	1.55
SB +F	3.24	9.43	3.94	1.28	0.66	1.87	0.73	1.48	5.74	1.91
SB -F	5.16	7.13	3.25	1.14	0.77	3.18	1.26	1.19	4.33	1.62
	A	F	A	A	A	A S	A S		F	

[a] Coarse sand and particulate organic matter (>250 µm)
[b] Microaggregates within macroaggregates (53-250 µm)
[c] Macroaggregate occluded silt and clay (<53 µm)

Orthogonal contrasts significant at the P = 0.05 level
A Agricultural disturbance (QSMAS and SB) vs. SF
F Fertilization effect: +F vs. -F
S System effect: QSMAS vs. SB

Figure Captions

Figures 1a & b: Earthworm populations (biomass and count) in top 30 cm of soil from replicated field trial with different land management and fertilizer treatments (SF = secondary forest, QSMAS = Quesungual slash-and-mulch agroforestry system, SB = slash-and-burn agriculture, +F = fertilized, -F = unfertilized) sampled in July 2007 in western Honduras. Error bars represent the standard error around each treatment mean.

Figure 2: Phosphorus availability (Olsen P) in surface soils (0-15 cm) from replicated field trial with different land management and fertilizer treatments (SF = secondary forest, QSMAS = Quesungual slash-and-mulch agroforestry system, SB = slash-and-burn agriculture, +F = fertilized, -F = unfertilized) sampled in July 2007 in western Honduras. Error bars represent the standard error around each treatment mean.

Figure 3: Aggregate stability (mean weight diameter) in surface soils (0-15 cm) from replicated field trial with different land management and fertilizer treatments (SF = secondary forest, QSMAS = Quesungual slash-and-mulch agroforestry system, SB = slash-and-burn agriculture, +F = fertilized, -F = unfertilized) sampled in July 2007 in western Honduras. Error bars represent the standard error around each treatment mean.

Figure 4: Rate of loss of soil carbon in surface soils (0-15 cm) from a replicated field trial with different land management and fertilizer treatments since conversion of forest to agriculture (SF = secondary forest, QSMAS = Quesungual slash-and-mulch agroforestry system, SB = slash-and-burn agriculture, +F = fertilized, -F = unfertilized). Rates for SF and SB represent losses between April 2005 and July 2007, while those for QSMAS extend from July 2003 to July 2007. Error bars represent the standard error around each treatment mean.

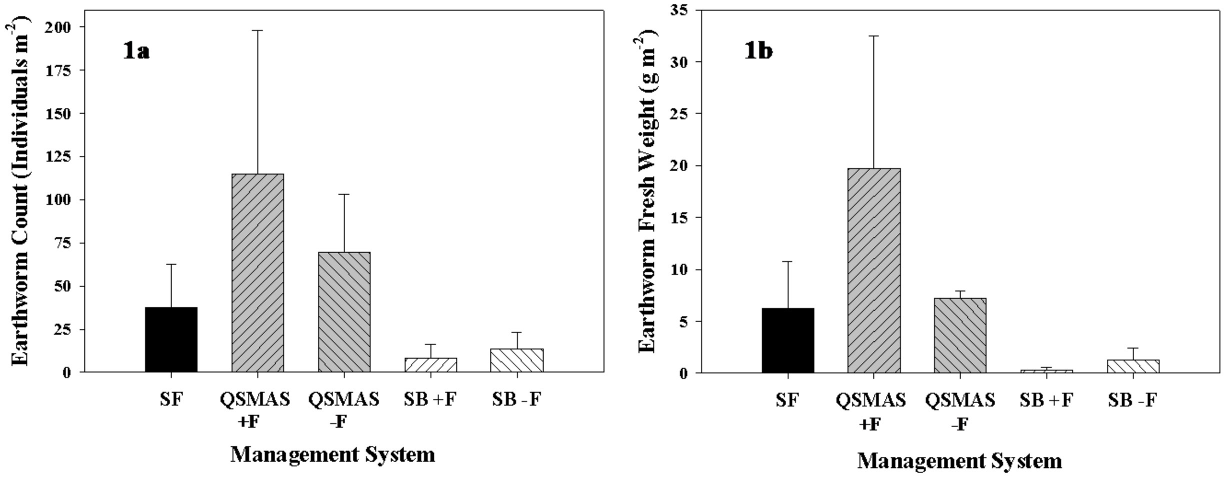
1a
Earthworm Count (Individuals m-2)
0
25
50
75
100
125
150
175
200
SF
QSMAS +F
QSMAS -F
SB +F
SB -F
Management System
1b
Earthworm Fresh Weight (g m-2)
0
5
10
15
20
25
30
35
SF
QSMAS +F
QSMAS -F
SB +F
SB -F
Management System

Figure 1

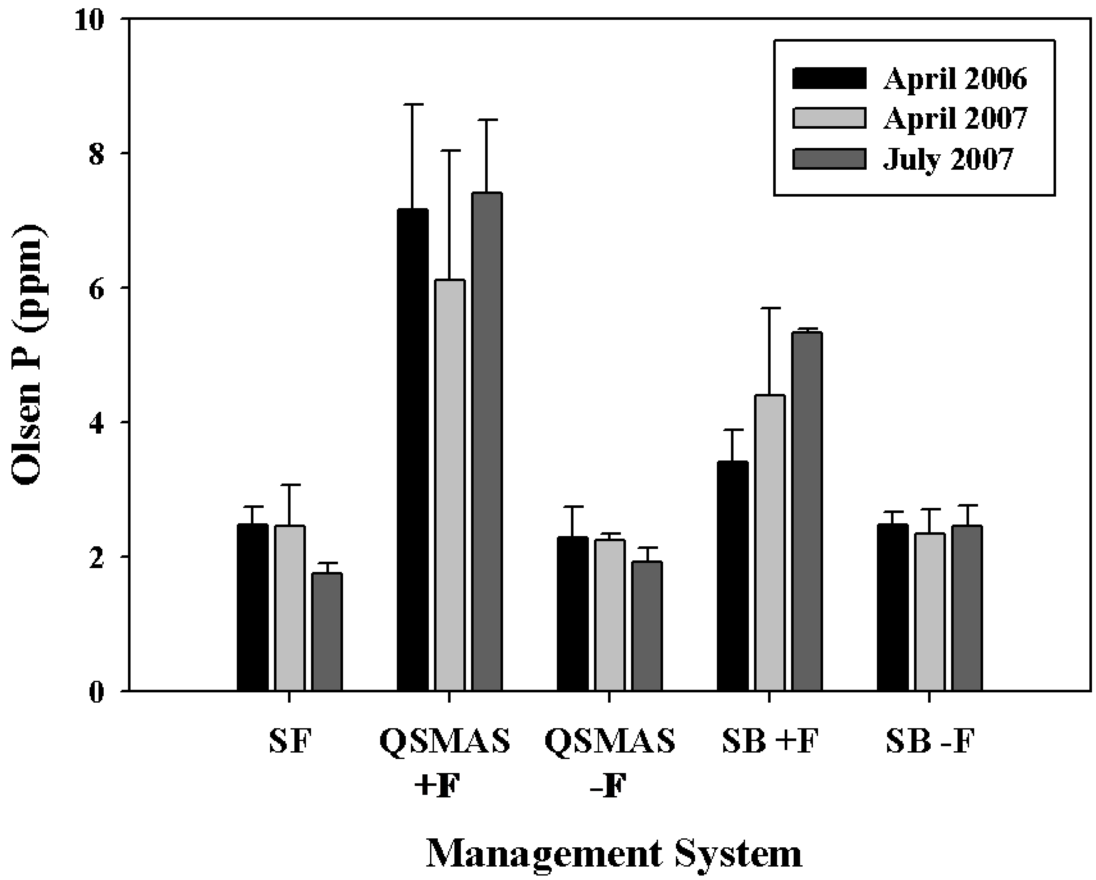
Olsen P (ppm)
0
2
4
6
8
10
April 2006
April 2007
July 2007
SF
QSMAS +F
QSMAS -F
SB +F
SB -F
Management System

Figure 2

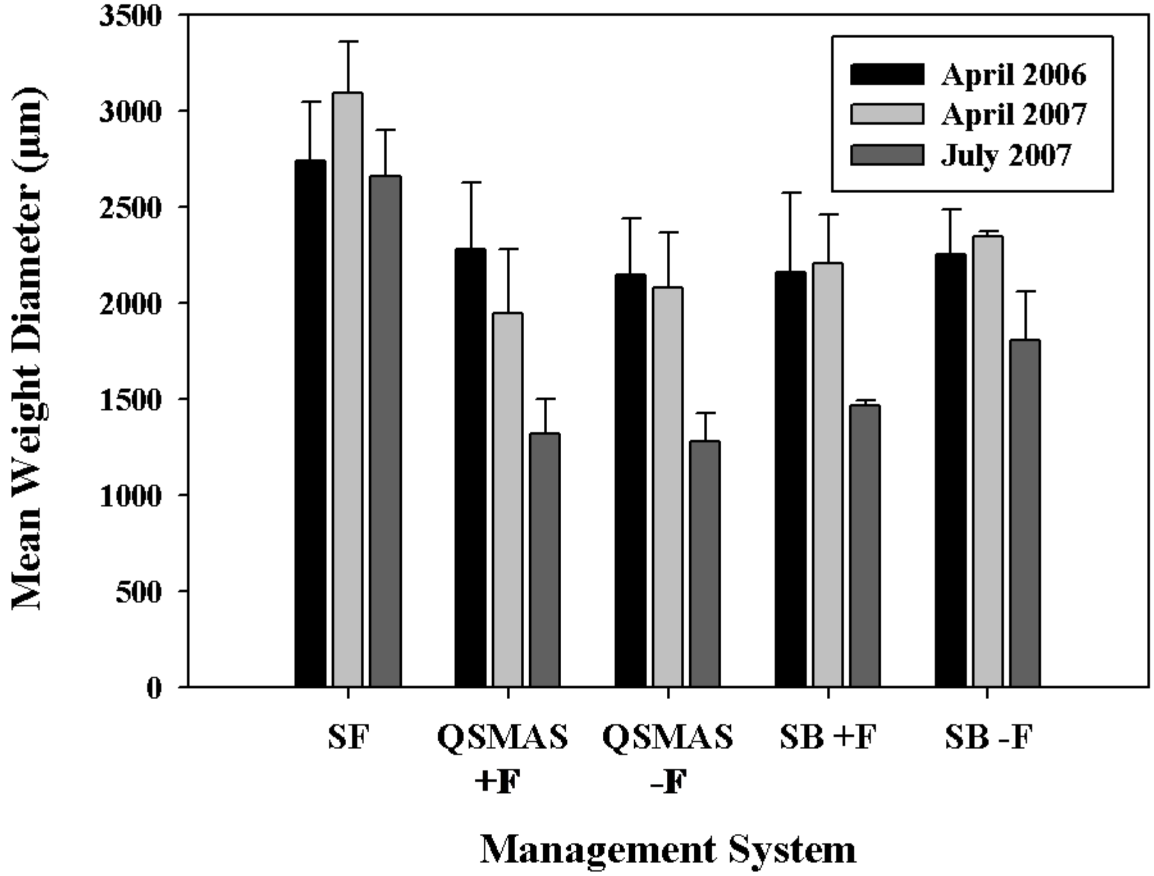
Mean Weight Diameter (µm)
3500
3000
2500
2000
1500
1000
500
0
April 2006
April 2007
July 2007
SF
QSMAS +F
QSMAS -F
SB +F
SB -F
Management System

Figure 3

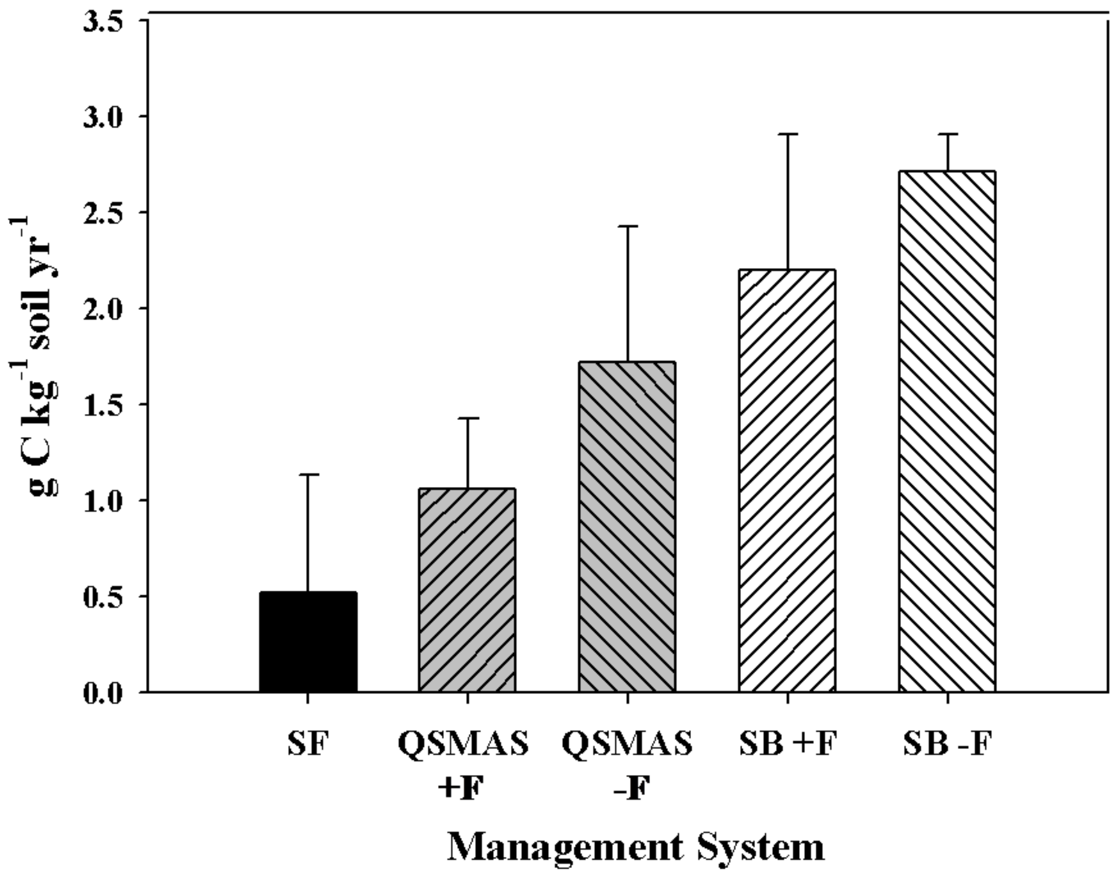
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
g C kg-1 soil yr-1
SF
QSMAS +F
QSMAS -F
SB +F
SB -F
Management System

Figure 4