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Short-term responses of selected soil properties to clearing and cropping of *miombo* woodlands in central Zimbabwe

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ABSTRACT

Clearing and cultivation of indigenous woodlands for agriculture may be among the most important mechanisms of physical, chemical and biological land degradation in Zimbabwe, and southern Africa in general. The objective of the study was to determine the effects of clearing *miombo* woodland and converting the land to maize (*Zea mays* L.) cropping on selected soil properties on clay (Chromic luvisol) and loamy sand (Ferric acrisol) soils in central Zimbabwe. Soil samples were collected from undisturbed, cleared and cultivated woodlands after four cropping seasons and analyzed for soil organic C, total N and P, exchangeable bases, cation exchange capacity, infiltration rate, aggregate stability and microbial biomass C and N at 0–5, 6–10 and 11–20 cm depths. Results showed that clearing and conversion of *miombo* woodlands to croplands reduced soil nutrients, cation exchange capacity (range: 9.6–21.0 $cmol_c\ kg^{-1}$ in clay; 7.0–15.5 $cmol_c\ kg^{-1}$ in loamy sand), and microbial C (range: 0.06–0.54% in clay; 0.02–0.37% in loamy sand). The extent and nature of change was variable, depending on the soil type and depth. Clearing of trees and leaving soil surface covered with grass did not always translate to a significant decline in soil organic C after four seasons (range: 0.69–2.24% in clay; 0.24–1.43% in loamy sand), unless the clearing was followed by successive cultivation and cropping without N fertilization. The reduced soil quality under cultivation was attributed to a potential pulse in decomposition and mineralization processes caused by soil disturbance, followed by leaching of released nutrients to lower horizons. This could be aided by nutrient removal in crop parts during harvest, without adequate soil nutrient replenishment. Under the woodland ecosystem, litter-fall may help to maintain steady-state infiltration rate (range: 45–126 $cm\ h^{-1}$ in clay; 32–97 $cm\ h^{-1}$ in loamy sand) by protecting the soil surface from damage and ensuring the formation of stable aggregates which preserve pore continuity. It was recommended that when *miombo* woodlands are to be cleared, management decisions that reduce tillage intensity and maximize residue retention should be put into practice; otherwise the clearing is strongly discouraged.

1. Introduction

Miombo woodlands are an essential biome that supports a large number of fauna and flora and covers about 10% of the African land area that is equivalent to about 2.5–4 million km^2 (Goldberg, 2001; Nord, 2008). In most cases, and particularly in Zimbabwe, the woodlands are found on infertile soils in warm areas with a mean annual temperature of 18–23 °C (Campbell et al., 1996; Lulandala, 2007) and a relatively low mean rainfall amount of 600–800 mm (Goldberg, 2001). The soils are therefore relatively fragile in terms of susceptibility to degradation and would require careful management since reclamation of degraded soils is generally expensive and sometimes not fully achievable. These soils usually have low cation exchange capacity (CEC) and low contents of N and extractable P (Frost, 1996), making them unsuitable for permanent agriculture and requiring considerable vegetation cover to avoid erosion and degradation (Campbell et al., 2007). However, in practice, the woodlands have experienced massive deforestation (WWF, 2010) and clearing for crop production (FAO, 2010). It is critical to understand the effects of such land use change on soil properties and land degradation in order to provide evidence-based policy advice on land management.

Miombo tree communities are dominated by generas *Brachystegia*, *Julbernardia* and *Isoberlina* (*Fabaceae*, subfamily *Caesalpinioideae*) that are found in most countries of southern and central Africa, and being the dominant forest component of Angola, Zambia, Tanzania, Malawi, Mozambique and Zimbabwe (Malmer, 2007). The canopy coverage determines the density and height of

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the grass layer, and common canopy coverage is 50%, which enables grass to grow even up to 2 m in height (Nord, 2008). The *miombo* woodlands and their accumulation of leaf litter on soil surfaces have been shown to foster higher levels of organic matter in the surface soil, promoting better soil structure, improving soil fertility, and carbon storage in both soil and woody biomass (Walker and Desanker, 2004; Munishi et al., 2010; Timberlake and Chidumayo, 2011). A negative change in these physico-chemical properties may result in severe land degradation. The permanent vegetation therefore plays an important role in improving soil properties and processes that ensures a healthy ecosystem.

Globally, there are concerns about changes in land use and land cover due to the realization that land surface processes influence climate and these processes impact on ecosystem goods and services (Lambin et al., 2003). Sileshi et al. (2007) summarized the ecosystem services in the *miombo* eco-region of eastern and southern Africa into: provisioning services, such as food and energy; regulatory services, including climate and microclimate modification; and supporting services, such as biodiversity conservation and pollination in the *miombo* eco-region. Deforestation and conversion of the woodlands to agricultural land may affect climate processes in many ways related to their impact of reduced flora (Katsvanga et al., 2005) and fauna (Sileshi and Mafongoya, 2006). Mapanda et al. (2010) studied the effects of land cover and land use on soil emission of CO_2, CH_4 and N_2O across different ecosystems that included undisturbed, cleared and cultivated *miombo* woodlands in Zimbabwe. They indicated that land cover change was an important driver of greenhouse gas emissions. Their study also pointed to the need to look into how soil physical, chemical and biological properties have changed with the clearing and cultivation of *miombo* woodlands since these properties would normally influence microbial activities that result in greenhouse gas emissions.

The objective of this study was to determine the effect of disturbing *miombo* woodlands by clearing and cultivation on selected soil chemical, physical and biological properties of two contrasting soils in central Zimbabwe. Literature on converting *miombo* woodlands to agricultural land in developing countries has pointed to a number of causes that include high population in relation to available land, high dependency on non-oil primary products, and market and policy failures (Chipika and Kowero, 2000; Barbier, 2004). It was hypothesized that these conversions would result in deterioration of soil chemical, physical and biological properties, thus impacting negatively on ecosystem health. This was based on the assumption that when a soil is cultivated, it often loses 20–40% of its carbon (C) and the largest part of this loss occurs within the first 5 years (Davidson and Ackerman, 1993; Williams et al., 2008).

2. Materials and methods

2.1. Study sites

The study was conducted at the University of Zimbabwe Farm (UZ-Farm) located about 15 km north of Harare (31°00′48″ E; 17°42′24″ S), and the Grasslands Research Station (GR-Station) located about 67 km east of Harare (31°29′00″ E; 18°10′14″ S). The two experimental sites were the same sites on which a four-season (2006/2007–2009/2010) greenhouse gas emissions study, partly reported by Mapanda et al. (2010) and also Mapanda et al. (2012), was done. A soil and vegetation study had been carried out before selection of areas with uniform vegetation and similar soil type at each site. The woodland area at the UZ-Farm was about 100 ha and mostly occupied by mixed *Brachystegia spiciformis* and *Julbernardia globiflora* tree species, while the woodland at the GR-Station covered about 2200 ha with similar tree species, in addition to *Terminalia sericea* and *Burkea africana*, in association with *Combretum* and *Acacia* species in other areas. The *Hyparrhenia* species were the most dominant grass species co-existing with the trees at both sites. The red clay soil at UZ-Farm is classified as Chromic luvisol (FAO, 1988) derived from dolerite, while the brown loamy-sand at GR-Station is a Ferric acrisol (FAO, 1988) derived from granite (Nyamapfene, 1991). Other soil and site characteristics are given in Table 1.

Table 1
General characteristics of the study sites including climatic data covering the period 2005–2010, soil properties (mean ± standard deviation) in the 0–0.1 m depth and vegetation status of woodlands in 2006 (Mapanda et al., 2010).

Characteristic	UZ-Farm	GR-Station
Altitude (m above sea level)	1505	1637
Mean annual rainfall (mm yr^{-1})	748 ± 176	829 ± 79
Mean maximum temperature (°C)	26.1 ± 0.7	24.4 ± 0.3
Mean minimum temperature (°C)	12.4 ± 0.4	10.5 ± 0.2
Soil parent material	Dolerite	Granite
Slope (%)	2–3	<2
Soil pH (in water)	5.8 ± 0.6	5.2 ± 0.7
Bulk density (Mg m^{-3})	1.49 ± 0.04	1.81 ± 0.06
Soil organic C (%)	1.66 ± 0.55	0.95 ± 0.06
Clay content (%)	51 ± 0	11 ± 0.7
Cation exchange capacity (cmol$_c$ kg^{-1})	9.5 ± 1.5	4.2 ± 0.1
Tree density (ha^{-1})	2604 ± 429	4896 ± 729
[a]Shrub density (ha^{-1})	2500 ± 977	3854 ± 1310
[b]DBH range (and median) (m)	0.05–0.27 (0.10)	0.04–0.15 (0.08)

[a] shrub is woody vegetation with height of <2 m.
[b] DBH is tree diameter at breast height.

2.2. Experimental treatment and management

The treatments were first introduced at the two experimental sites in October 2006 as described by Mapanda et al. (2012). Four treatments were introduced, each on a plot measuring 4 m × 6 m arranged in a randomized complete block design with four replicates. The plot sizes were a fair compromise to minimize excessive clearing of the woodlands, and they were generally within the range recommended by Walker et al. (2012) as a reasonable balance of effort and precision for the tree sizes found at the two sites (Table 1). At such plot sizes variability in soil and vegetation properties was also minimum before clearing. The treatments were: (1) undisturbed woodland, (2) cleared woodland, without cultivation and fertilization, (3) cleared woodland with cultivation, and without N-fertilization (maize-cropped), and (4) cleared woodland with cultivation, with fertilization (maize-cropped, 120 kg N ha^{-1}).

The clearing of tree stands from just above-ground was carried out once in October 2006 (about two weeks before the onset of the cropping season) by hand using axes, and cultivation was undertaken manually and annually using hand picks to achieve a plough depth of about 0.15–0.20 m in the 2006/2007, 2007/2008, 2008/2009 and 2009/2010 cropping seasons. Maize (*Zea mays* L.) was sown each season, and NH_4NO_3 (34.5% N) was applied annually at a rate of 120 kg N ha^{-1} (for treatment 4), while annual basal dressings of P (30 kg ha^{-1}, as single super phosphate) and K (30 kg ha^{-1}, as muriate of potash) were applied before sowing on all plots under maize, including those that did not receive N fertiliser (i.e. treatment 3).

2.3. Soil sampling and preparation of samples

Soil sampling for determination of soil physical, chemical and biological properties in the experimental plots was conducted once during the fifth season (2010/2011) in which no cropping was done. The sampling was done in mid-January when the soil was relatively moist and friable. Mini-pits were dug to a depth of up to 25 cm at

three randomly selected locations in each plot using a shovel. The mini-pits enabled visual inspection of the strata, allowed more accurate measurement of soil depths and reduced soil disturbance by retaining the structural integrity of the soil. Soil samples were collected from one wall of each pit at three depths (0–5 cm, 6–10 cm, and 11–20 cm), starting with the bottom depth to avoid cross contamination. Each collected sample was placed in a labelled plastic bag and the samples were taken to the laboratory where each sample was thoroughly mixed and allowed to air dry. Part of the soil to be used for the determination of aggregate stability, nutrients (N, P and K) and cation exchange capacity was sieved through a 2 mm sieve, while the remaining soil was ground to pass through a 0.5 mm sieve for determination of soil organic C, and microbial biomass C and N. The prepared soil samples were stored in clearly labelled khaki bags at room temperature, while the soil fraction for microbial biomass C and N were stored in a refrigerator at 4 °C.

2.4. Analysis of soil samples for chemical properties

Soil samples were analyzed for soil organic C, nutrients (total N, P and K) and cation exchange capacity using the methods described by Okalebo et al. (2002). Soil organic C was extracted using concentrated H_2SO_4 and $K_2Cr_2O_7$ with external heating, and determined by titrating with $Fe(NH_4)_2(SO_4)_2{\cdot}6H_2O$. Nutrients (total N, P and K) were extracted using concentrated H_2SO_4 and H_2O_2 with external heating. Total N was determined using the semi-micro Kjeldahl method (Bremner and Mulvaney, 1982), while total P was determined colorimetrically using the UV visible spectrophotometer after generating colour using ascorbic acid. Total K was determined using the atomic emission spectrophotometer. Cation exchange capacity and the exchangeable bases were determined by extraction using ammonium acetate as described by Anderson and Ingram (1993). Exchangeable K and Na were measured by flame photometry, and Ca and Mg by atomic absorption spectrophotometry.

2.5. Analysis of soils for infiltration rate and aggregate stability

Steady-state infiltration rate was measured in situ using the double ring infiltrometer as described by Anderson and Ingrams (1993). Metal rings (inner ring, 32 cm diameter, outer ring 54 cm diameter, height 40 cm) were driven vertically into the soil in each plot for about 15 cm using a hammer with the smaller ring centred in the larger ring. The soil surface was not cleaned i.e. was maintained in its natural state while the double ring was driven into the soil. Both cylinders were filled with water to a height of 14 cm. The time taken for the head in the inner ring to fall by one centimetre was recorded allowing the head to fall to 4 cm. When the head fell to 4 cm it was refilled to 14 cm continuing to note the time taken for the head to fall by 1 cm. The water level in the inner and outer ring was always maintained at the same height. When the time taken for the water level to fall was constant the soil steady state infiltration was assumed to have been achieved and the experiment stopped. The results were fitted to the Kostiakov equation (Hillel, 1982) shown in Eq. (1).

$$F = at^{-b} \tag{1}$$

where F is the infiltration rate in cm h^{-1}; t is the time from start to end of the experiment in hours; a is the constant representing the cumulative infiltration after time t of infiltration, and b is the constant that gives the relative importance of time of infiltration.

Water stable aggregates were measured on soil samples using the Yoder apparatus adapted from Kemper and Rosenau (1986). Four grams of air dried soil were passed through a 2 mm sieve and placed on 0.2 mm sieves. The sieves were immersed in deionised for 30 min. After 30 min the sieves were removed and placed on the holders of the wet sieving machine (Yoder apparatus) and the machine was switched on, lowering and raising the nest of sieves through a distance of 1.3 cm at a rate of 35 cycles per minute for six minutes, following Barthes and Roose (2002). After six minutes the machine was switched off and the soil aggregates on the 0.2 mm sieve were washed into pre-weighed containers and oven dried at 105 °C for 24 h. After oven drying containers with soil the aggregates were reweighed to determine the weight of the aggregates and the container. After weighing the soil, aggregates were sieved, using 0.2 mm sieves, into dispersive 0.05 M NaOH solution for 30 min using the same Yoder apparatus, washed into pre-weighed containers oven dried at 105°c for 24 h and weighed to determine the weight of coarse sand (CS > 0.2 mm). The results were expressed as Macro-aggregate index (*Ima*) defined by Eq. (2), i.e. 1000-fold the weight ratio of water-stable macro-aggregates to oven-dried initial sample minus coarse sands:

$$Ima = \frac{1000(F > 0.2 - \mathrm{CS})}{g\mathrm{DM} - \mathrm{CS}} \tag{2}$$

where: DM = dry matter content of the sample; CS = coarse sand (>0.2 mm); g = mass of sample used, and $F > 0.2$ = fraction of soil on 0.2 mm sieve after sieving.

2.6. Analysis of soil samples for microbial biomass C and N

Soil microbial biomass C and N were determined as the difference in extractable C and N between the fumigated and unfumigated soil (Eqs. (3) and (4)). The chloroform fumigation extraction method adapted from Howarth and Paul (1994) was used to release microbial cell material from microbial biomass. In this method, 50 ml beaker containing 5 g fresh soil samples and a 100 ml beaker with 50 ml alcohol-free chloroform were placed in vacuum desiccators. Another desiccator was maintained without chloroform and was kept under dark conditions for 72 h at room temperature. The fumigated desiccators were then evacuated using a vacuum pump until the chloroform rapidly boiled. Soil samples were transferred to 250 ml conical flask and the fumigated and unfumigated soils were analyzed for extracted C using the Walkley-Black method (Nelson and Sommers, 1996), and extractable N using the semi-micro Kjeldal method (Bremner and Mulvaney, 1982; Okalebo et al., 2002).

Vance et al. (1987):

$$\text{Microbial C} = (\mathrm{EC}_f - \mathrm{EC}_u) \times 2.64 \tag{3}$$

Brookes et al. (1985):

$$\text{Microbial N} = \frac{\mathrm{EN}_f - \mathrm{EN}_u}{0.54} \tag{4}$$

where: EC_f and EC_u = extractable C of fumigated and unfumigated soils, respectively; EN_f and EN_u = extractable N of fumigated and unfumigated soils, respectively.

According to Howarth and Paul (1994), the fumigation extraction method poses no risk of NH_4^+ immobilization and denitrification activity, and there is low interference from non-microbial labile C and N substrates that can be used during incubation as in other methods.

2.7. Data analysis

Data sets of measured soil chemical, physical and biological properties were subjected to two way analysis of variance using GenStat 8.1 statistical package to analyze differences between treatments, while the least significant difference (LSD) at $P = 0.05$ was used in separation of significantly different treatments. Linear

regression analysis was done to establish if there was any relationship between soil organic C, aggregate stability and steady-state infiltration rate.

3. Results

3.1. Soil chemical properties

Soil organic C ranged from 0.69 to 2.24% in the UZ-Farm clay soil and from 0.24 to 1.43% in the GR-Station loamy sand soil (Fig. 1). Compared with the undisturbed woodland the cultivation of cleared woodland significantly ($P < 0.05$) reduced soil organic C at all depths by 16–55% (mean 36%) in the clay soil and 23–73% (mean 50%) in the loamy sand soil. This change was more distinct on plots that were cultivated for four cropping seasons without N fertiliser application. On the contrary, the plots that were cleared once in the 2006/2007 cropping season had the highest soil organic C within the first two depths in the clay soil after four seasons. The general observation on these plots was that of a more vigorous growth of numerous suckers from the tree-stumps compared with the undisturbed woodland plots. By the fourth season it was mainly one or two suckers that were dominating and growing taller than the others. A similar trend was observed on the loamy sand soil but the vegetation proliferation was less than that observed on the clay soil. In the loamy sand, there was however, no considerable change in soil organic C due to clearing, except at the 6–10 cm depth that showed a 36% decline in soil organic C relative to the undisturbed woodland plots.

The total concentrations of the major plant nutrients, N, P and K, in the woodland and cropped plots after four cropping seasons are shown in Fig. 2. The overall trend in total N (0.07–0.23% in the UZ-Farm clay soils; 0.05–0.19% in the GR-station loamy sand soils) was that of a decline in total N for cleared plots and plots that were cleared and cultivated without N application, especially at the lowest depth of 11–20 cm (Fig. 2a and b). The application of NH_4NO_3 resulted in a decline in total N in the clay soil compared to no N application, except in the 6–10 cm depth. This 6–10 cm depth corresponded with the depth of fertiliser incorporation at planting, and also resulted in the buildup of soil N in the loamy sand soil (Fig. 2b). Total *P* ranged from 0.02 to 0.16% in UZ-Farm clay soils and from 0.02 to 0.08% in the GR-Station loamy sand soils, and significant treatment effects ($P < 0.05$) were noted in both soils and at all depths. The least *P* concentration in clay soil was found in cleared woodland plots while the undisturbed woodland plots had the highest *P* concentrations in the 0–5 cm and 6–10 cm depths (Fig. 2c). In the loamy sand soils, there was no distinct trend across the sampled depths but the highest *P* concentrations were found among the cultivated plots on which 30 kg P ha^{-1} $season^{-1}$ were applied (Fig. 2d). Total K ranged from 0.41 to 1.09% in the UZ-Farm clay soils and from 0.25 to 1.30% in the GR-Station loamy sand soils (Fig. 2e and f). No distinct trend could be established in K concentration with treatment or soil depth in the clay soil. However, in the loamy sand soil cropped plots had the least K concentrations of not more than 0.52% in all soil depths, despite having received 30 kg K ha^{-1} $season^{-1}$ for four cropping seasons.

The total exchangeable bases (TEB) were 7.4–20.1 $cmol_c$ kg^{-1} at the UZ-Farm, and 7.0–15.5 $cmol_c$ kg^{-1} at the GR-Station, while the cation exchange capacities (CECs) were 9.6–21.0 $cmol_c$ kg^{-1} at the UZ-Farm, and 9.3–18.7 $cmol_c$ kg^{-1} at the GR-Station (Fig. 3). The clearing and cultivation of woodlands significantly ($P < 0.05$) reduced both total exchangeable bases and cation exchange capacity. The cation exchange capacity decreased at all depths by 15–54% (mean 38%) in the clay soil and 14–50% (mean 32%) in the loamy sand soil. These rates of decline in cation exchange capacity closely correspond to the rate of soil organic C decline in the clay soil, but were relatively less responsive to the rate of soil organic C decline in the loamy sand soil.

3.2. Steady-state infiltration rate and aggregate stability

Converting the *miombo* woodlands to croplands significantly ($P < 0.05$) reduced the steady-state infiltration rate of both the UZ-Farm clay soil and the GR-Station loamy sand soil, after four cropping seasons (Fig. 4). The steady state infiltration rates (F_{final}) ranged from 45 to 126 cm h^{-1} in the UZ-Farm clay soil and from 32 to 97 cm h^{-1} in the GR-Station loamy sand. While the clearing of trees alone did not show any considerable effect on the F_{final} in the loamy sand soil relative to the undisturbed woodland (Fig. 4b), it resulted in a 33% decline in the F_{final} in clay soil. Clearing and cultivation of plots resulted in 57–80% decline in the F_{final} on both soils, and the highest decline was observed on cropped plots that did not receive N fertiliser at the UZ-Farm clay soil. The effect of N fertiliser application on the F_{final}, relative to none application, was however insignificant on the GR-Station loamy sand soil.

The water-stable macro-aggregation indices (*Ima*) ranged from 565 to 913 (mean 784) at the UZ-Farm and from 373–874 (mean 690) at the GR-Station (Fig. 5). There was a positive correlation ($P < 0.05$) between aggregate stability and infiltration rate on both clay soil ($r = 0.5$) and loamy sand soil ($r = 0.6$). Aggregate stability significantly decreased, by 9–38%, with both clearing and cropping without N fertiliser application in the three sampled depths at the UZ-Farm. At this site, cropped plots with N fertiliser application had macroaggregation indices of similar magnitudes to those on the undisturbed woodlands, except at the 0–5 cm depth in which

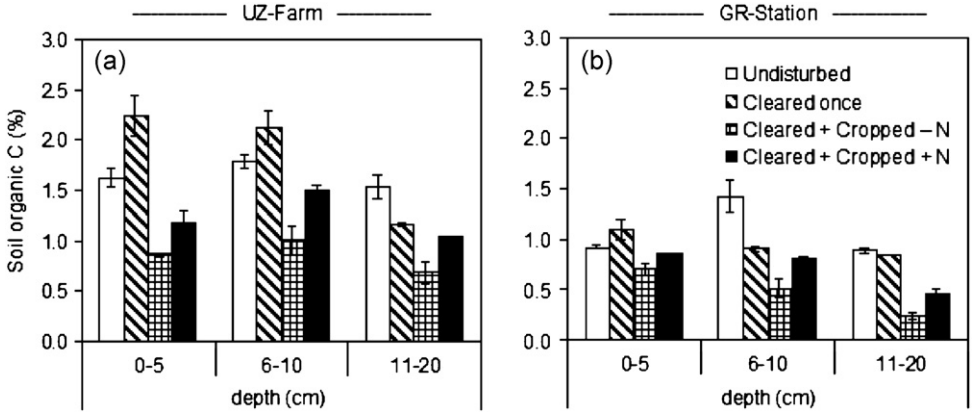

Fig. 1. Soil organic C at three soil depths from the undisturbed and cleared woodlands, as well as the cleared and cropped land with (+N) and without (−N) N fertilization at the UZ-Farm (a) and GR-Station (b). Error bars denote standard errors of means.

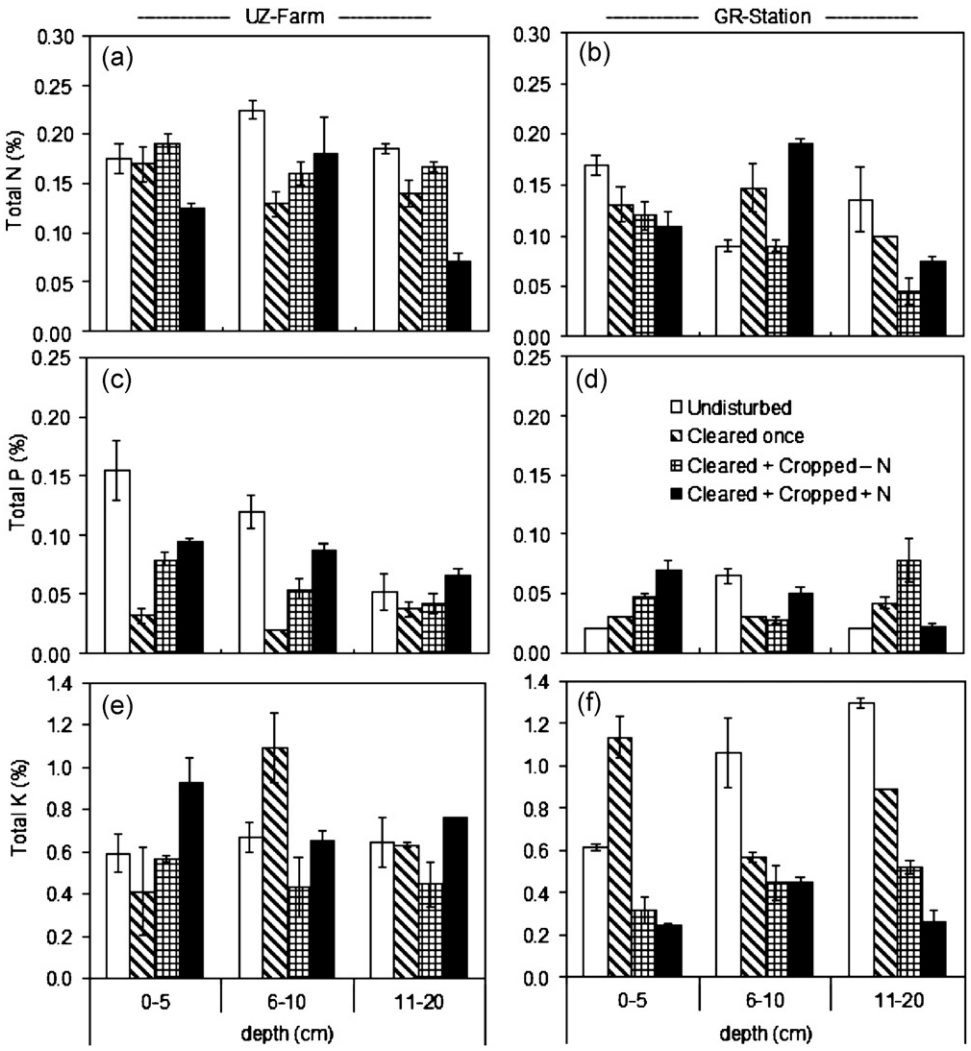

Fig. 2. Total N (a and b), P (c and d) and K (e and f) in soils from the undisturbed and cleared woodlands, as well as the cleared and cropped land with (+N) and without (−N) N fertilization at the UZ-Farm and GR-Station at three depths. Error bars denote standard errors of means.

the index decreased by 17%. On the contrary, the highest aggregate stability in the GR-Station loamy sand soil was found on plots that were cleared once, without cultivation (Fig. 5b), and this is the same treatment that did not show significant response on infiltration rate (Fig. 4b). Cropped plots without N fertiliser application had the highest decline in aggregate stability (change of 27–51%) relative to the undisturbed woodland plots in all sampled depths at the GR-Station. There was a positive correlation ($P < 0.05$) between aggregate stability and soil organic C on both clay soil ($r = 0.61$) and loamy sand soil ($r = 0.51$).

3.3. Soil microbial biomass C and N

Soil microbial C (0.06–0.54% at the UZ-Farm; 0.02–0.37% at the GR-Station) and microbial N (0.02–0.06% at the UZ-Farm; 0.02–0.05% at the GR-Station) differed significantly ($P < 0.05$) across treatments, and this was largely consistent at all the soil depths (Fig. 6). The undisturbed woodland plots had highest soil microbial C and N than the cleared or the cleared and cropped plots. Soil microbial C decreased by 50–89% in the clay soils and by 38–92% in the loamy sand soil after four cropping seasons following the clearing and cultivation of the woodlands. However, soil microbial N decreased by a lesser magnitude (17–67% in the clay soil; 0–50% in the loamy sand soil) following the same woodland disturbances.

4. Discussion

The hypothesis that the clearing of woodlands and maize-cropping would result in deterioration of soil chemical, physical and biological properties relative to the undisturbed woodlands was supported. However, the extent and nature of change in soil quality was variable and depended on the soil type and sampling depth of the soil. Results showed that the clearing of trees without cultivation does not always translate to a decline in soil organic C after four seasons, unless the clearing is followed by successive cultivation and cropping, and worse cropping without N fertilization. The indication by Davidson and Ackerman (1993) and Williams et al. (2008), that when the soil is cultivated, it often loses 20–40% of its organic C mostly within the first 5 years was also supported by the results of this study (Fig. 1). Munishi et al. (2010) investigated the role of *miombo* woodlands as C sinks in Tanzania, and noted soils as having much potential for C storage, while the decrease in woodland density due to human utilization contributed to the lowering of C stocks. In this study, a vigorous growth of numerous suckers from the tree-stumps was observed after clearing, and by the fourth season the density of shrubs in cleared plots was relatively higher due to these suckers. This could explain why soil organic C was considerably higher on these plots, which may not be possible when the suckers are removed.

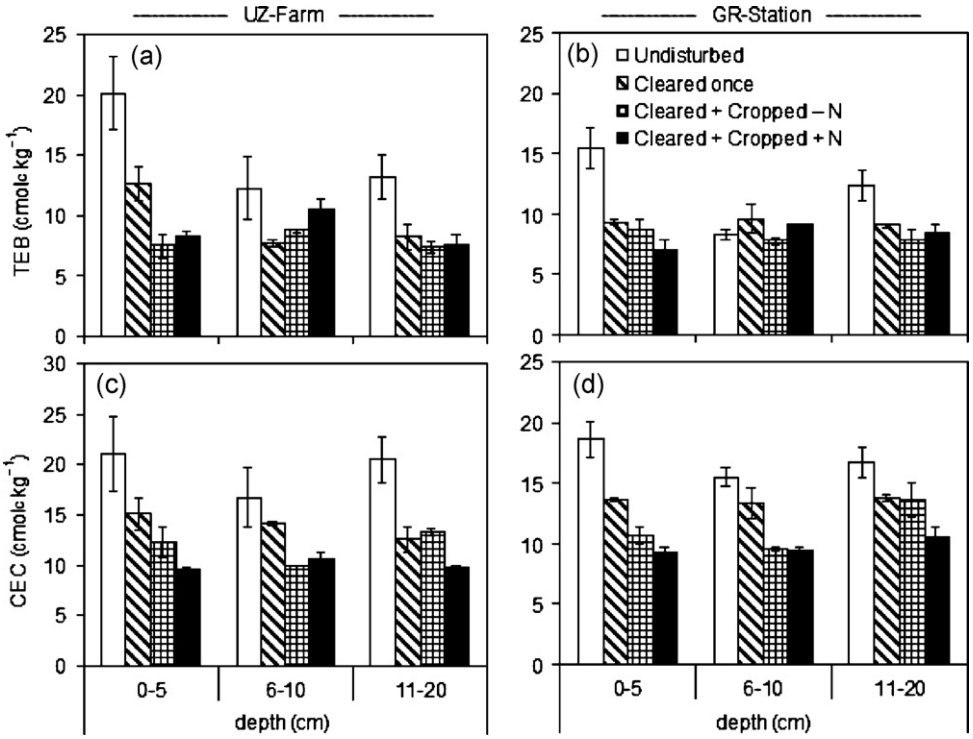

Fig. 3. Total exchangeable bases (a and b) and cation exchange capacity (CEC) (c and d) of soils from the undisturbed and cleared woodlands, and from cleared and cropped land with (+N) and without (−N) N fertilization at the two sites. Error bars denote standard errors of means.

The amounts of total N and P, and the exchangeable bases overall displayed a downward trend following clearing and also clearing and cultivation (Figs. 2 and 3), and this has been widely observed (Lindo and Visser, 2003; Wick et al., 2005) that after deforestation and land cover change from woodland to cropland, mineralization increases in soil and released nutrients would become more available for microbial transformations and plant uptake, unless they are rapidly lost through leaching (Fukuzawa et al., 2006). According to Baker et al. (2007) the conversion of natural ecosystems to croplands involves other changes that may have far more impact on soil organic C than just mechanical disturbance of the soil that increase aeration. They argued that these changes, including solar radiation on bare fields influence soil temperatures, combined with reduced litter-fall would result in reduced soil organic C, hence reduced nutrient availability from mineralization. Baker et al. (2007), however lamented the shallow sampling employed in most studies as source of bias, indicating that studies that have involved deeper sampling (beyond the 0.3 m depths) generally show no C sequestration advantage from reduced mechanical disturbance of soil.

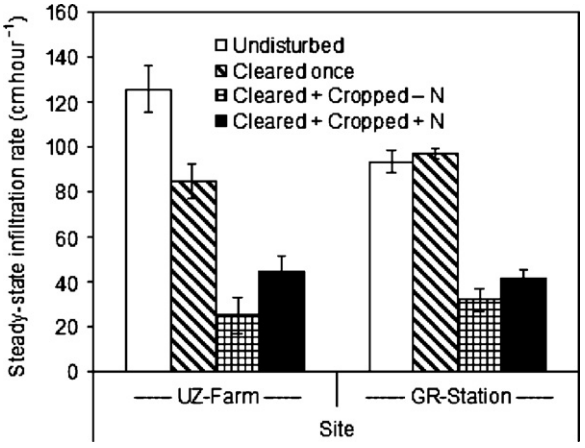

Fig. 4. The rates of water infiltration into the soils from the undisturbed and cleared woodlands, as well as the cleared and cropped land with (+N) and without (−N) N fertilization at the UZ-Farm (a) and GR-Station (b). Error bars denote standard errors of means.

Results showed a positive concurrence between the change in soil chemical properties (soil organic C, N, P, exchangeable bases and cation exchange capacity), and the change in soil physical properties (infiltration rate and aggregate stability). The macro-aggregation indices obtained from the cleared and cultivated plots in this study were relatively higher compared with those obtained from regular crop land in other studies (Barthes and Roose, 2002; Molumeli et al., 2008). The study by Barthes and Roose (2002) gave the indices in the range of 51 to 540 for the 0–10 cm depth from a range of soil with clay content of 5–45% under different climatic conditions that included sub-tropical. In their study lower macro-aggregation indices considerably favoured higher run-off and soil loss. Infiltration rate decreased with clearance and subsequent cropping of *miombo* woodlands, and this was in agreement with findings by Nord (2008). The relatively low steady state infiltration rate of GR-Station plots as compared to the UZ Farm could be partly attributed to the higher bulk density of the GR Station plots. The highest infiltration rates in the woodland plots can be attributed to litter layers that maintain macroporosity which increases the infiltration capacity (Van Noordwijk et al., 2003).

Microbial biomass C and N decreased significantly with tillage disturbance and lower organic matter inputs after cultivation as reported from other studies (Gosai et al., 2010). The rate of organic C input from plant biomass is considered as the dominant factor controlling the amount of microbial biomass in the soil from both managed ecosystems (Kallenbach and Grandy, 2011) and natural ecosystems (Jin et al., 2010). Research by Mapanda et al. (2010) and Mapanda et al. (2012) on the same site revealed that there was a decrease in soil moisture, and an increase in temperature, on the cultivated plots relative to the undisturbed woodland plots. Singh

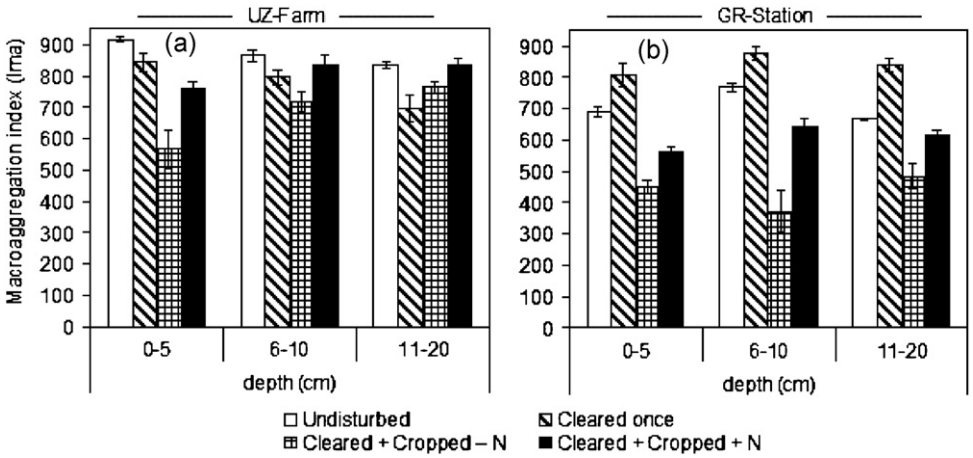

Fig. 5. Water-stable macroaggregation indices (*Ima*) of soils from the undisturbed and cleared woodlands, as well as the cleared and cropped land with (+N) and without (−N) N fertilization at the UZ-Farm (a) and GR-Station (b). Error bars denote standard errors of means.

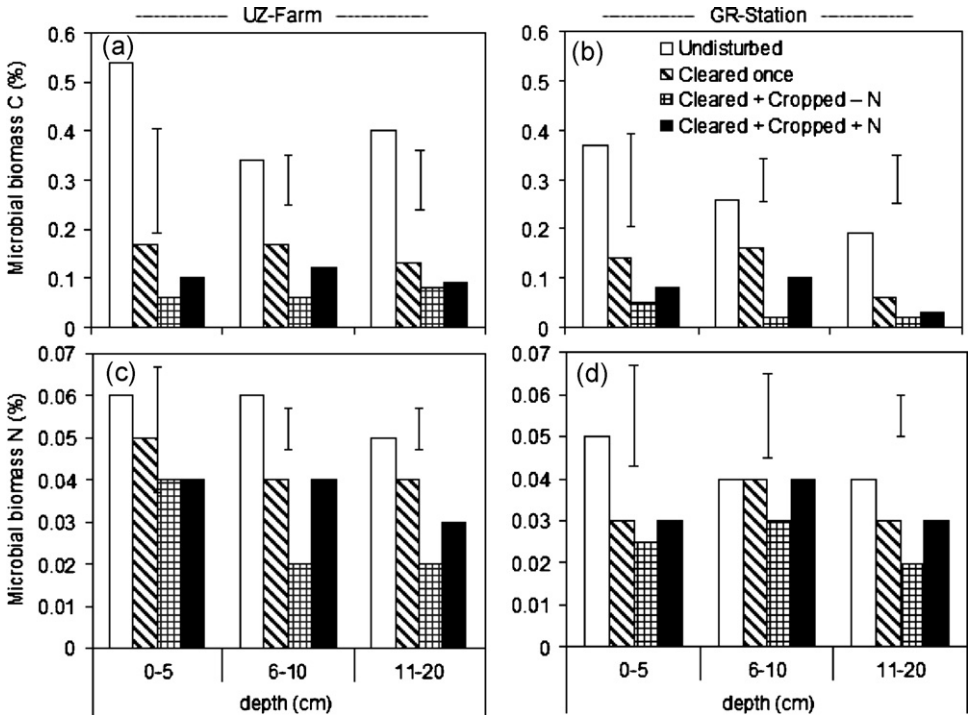

Fig. 6. Microbial biomass C (a and b) and microbial biomass N (b and c) in soils from the undisturbed and cleared woodlands, as well as the cleared and cropped land with (+N) and without (−N) N fertilization at the two sites at three depths. Error bars denote least significant difference (lsd).

et al. (2007) reported that soil microbial biomass is affected by soil moisture and soil temperature.

Cultivation reduces fauna and microbial biomass, particularly fungal biomass responsible for the formation of more binding agents, such as extra-cellular polysaccharides, and the development of hyphal networks enmeshing particles and favouring aggregate stability (Six et al., 2006). Continuous tillage and crop production resulted in reduction in water-stable aggregates because aggregates are routinely disrupted by tillage which release physically protected organic C (Roscoe and Buurman, 2003; Zotarelli et al., 2007). The magnitude of reduction in soil microbial biomass C and N following disturbances was similar to that found by Gosai et al. (2010), who also indicated that tillage causes significant spatial variation in microbial C and N content in the topsoil more than that of the bottom layer. Microbial biomass responds quickly to changes in soil management and is often used as an indicator of soil quality with lower soil microbial biomass C and N existing in soils disturbed by cultivation and receiving low organic matter input (Wang et al., 2012).

5. Conclusion

Clearing and conversion of *miombo* woodlands to croplands have considerable implications on soil chemical, physical and biological prosperities, which are site and soil specific in Zimbabwe. The extent and nature of change in soil quality is variable and also depends on the parameter measured and the soil depth considered. Clearing of trees without cultivation does not always translate to a decline in soil organic C after four seasons, unless the clearing is followed by successive cultivation and cropping without N fertilization. Under the woodland ecosystem litter-fall may help to maintain water infiltration by protecting the soil surface from damage and ensuring the formation of water-stable aggregates which preserve pore continuity. It may be

recommended that when *miombo* woodlands are to be cleared, management decisions that reduce tillage intensity should be put into practice, otherwise clearing is strongly discouraged.

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