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Soil property changes over a 120-yr chronosequence from forest to agriculture in western Kenya

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Abstract. Much of the native forest in the highlands of western Kenya has been converted to agricultural land in order to feed the growing population, and more land is being cleared. In tropical Africa, this land use change results in progressive soil degradation, as the period of cultivation increases. Both rates and variation in infiltration, soil carbon concentration and other soil parameters are influenced by management within agricultural systems, but they have rarely been well documented in East Africa. We constructed a chronosequence for an area of western Kenya, using two native forest sites and six fields that had been converted to agriculture for up to 119 yr.

We assessed changes in infiltrability (the steady-state infiltration rate), bulk density, proportion of macro- and microaggregates in soil, soil C and N concentrations, as well as the isotopic signature of soil C (δ^{13}C), along the 119-yr chronosequence of conversion from natural forest to agriculture. Infiltration, soil C and N decreased within 40 yr after conversion, while bulk density increased. Median infiltration rates fell to about 15 % of the initial values in the forest, and C and N concentrations dropped to around 60 %, whilst the bulk density increased by 50 %. Despite high spatial variability, these parameters have correlated well with time since conversion and with each other.

1 Introduction

Soil infiltrability (defined as the infiltration rate when water at atmospheric pressure is made freely available at the soil surface) (Hillel, 1971) and soil carbon (C), the major part of soil organic matter (SOM), are interrelated parameters that largely determine agricultural productivity. A high infiltrability enables water to enter the soil to become available for plant uptake and allows ground water recharge. It also reduces the risk of erosion. Infiltrability is a reflection of soil structure and texture (Cresswell et al., 1992), soil biological activity (Mando, 1997; Leonard et al., 2004), soil aggregation (LeBissonnais and Arrouays, 1997) and SOM content (Franzluebbers, 2002). Soil C enhances biological activity and thereby promotes nutrient retention and cycling.

Biological activity enhances soil aggregation (Jastrow et al., 1998), aeration, water holding capacity and infiltrability. Soil bulk density is often correlated with both soil C and with infiltrability (Mbagwu, 1997; Mariscal et al., 2007; Arvidsson, 1998). Hence, these three parameters (infiltrability, soil C and soil bulk density) are particularly suitable for studying changes in soil fertility and production capacity (Doran and Parkin, 1994).

The means and variances of infiltrability, soil C and other soil parameters are influenced by land use management. Whilst high spatial variability in these parameters may obscure statistical significance in experiments, it may also provide insights into spatial and temporal processes. To avoid or reduce the risk for large-scale surface runoff and erosion, the occurrence of infiltrability values that are higher than the prevailing rainfall intensities may be more important than the average infiltration rate for an area. Spatial variability is usually very high for infiltrability (Jetten et al., 1993; Mbagwu, 1997; Van de Genachte et al., 1996; Williams and Bonell, 1988). For example, Lal (1996) reported that infiltrability varied by a factor of two within 1 m.

Chronosequences, constructed from sites at different stages and durations of succession from forest to agriculture (i.e. time for space substitution) (Walker et al., 2010), are commonly used to describe and study changes in soils and soil degradation (Kimetu et al., 2008; Lemenih et al., 2005a, b; Awiti et al., 2008). Losses of 50 % to 80 % in soil C and N have been reported from sites that have been under agriculture for 20–50 yr (Solomon et al., 2007; Lemenih et al., 2005 a, b). Despite many studies on infiltrability in relation to land use in the tropics, there have been relatively few studies involving longer-term (>10–15 yr) effects of converting forests to permanent agriculture. Some authors have studied changes up to 10 yr after manual and mechanical clearing in Africa (e.g. Lal, 1989, 1996), and shifting cultivation in Asia (e.g. Toky and Ramakrishnan, 1981). Giertz and Diekkruger (2003) and Giertz et al. (2005) compared agricultural areas with natural vegetation in Benin (West Africa), but the time since conversion was unknown. Results from these studies show that natural forests have higher infiltrability and soil carbon content than cultivated lands. For example, Yimer et al. (2008) reported 70 % lower infiltrability under barley cultivation (15 yr) and in grazing land, compared to forest, in Ethiopia. Omuto (2008) reported a 25 % decrease in infiltrability 10 yr after conversion from a semi-arid shrubland to agriculture. For a true chronosequence (i.e. not a time for space substitution) from forest to pasture land in Latin America, Zimmermann at al. (2006, 2010) reported decreases in infiltrability of 90 %. This study uses a time for space approach for a chronosequence of 120 yr for conversion from forest to agricultural land in Nandi, Kenya.

Recovery after abandonment of agriculture has also been studied, e.g. Giambelluca (2002) presented a time series of recovery of soil conductivity with up to 30 yr fallow after long-term agriculture in Vietnam. There are a handful of other studies (cf. Ilstedt et al., 2007) documenting recovery after afforestation (e.g. Bonell et al., 2010; Chirwa et al., 2003; Mapa, 1995) and agroforestry practices (e.g. Hulugalle and Ndi, 1993, 1994; Salako and Kirchhof, 2003). The effect of grazing and subsequent recovery by natural succession has received relatively much attention in South and Latin America (e.g. Spaans et al., 1990; Zimmermann et al., 2006, 2010; Zimmermann and Elsenbeer, 2009; Hassler et al., 2011; Martinez and Zinck, 2004), but less in Africa (but see e.g. Savadogo et al., 2007; Abdelkadir and Yimer, 2011). In Southeast Asia, the effect of using heavy machinery for wood extraction in logging operations has been studied (e.g. Kamaruzaman et al., 1987; Kamaruzaman, 1996; Huang et al., 1996; Malmer and Grip, 1990).

Much of the native forest in the highlands of western Kenya has been converted to agricultural land, and some is still undergoing conversion, in order to feed the growing population. In tropical Africa, this land use change is commonly reported to result in progressive soil degradation that increases with the duration of cultivation (Juo et al., 1995; Lemenih et al., 2005a, b; Lal, 1996). Soil degradation, defined as the loss of actual or potential productivity or utility as a result of natural or anthropogenic factors (Lal, 1993) and mediated through interrelated physical, chemical and biological processes, threatens agricultural sustainability. Among the physical processes, deterioration of soil structure is of particular importance, since it leads to accelerated erosion. Erosion is recognized as one of the major symptoms of soil degradation that results in important on- and off-site costs (Pimentel et al., 1995). A reduction in soil organic carbon is also a key soil degradation process, especially in the low input agriculture typical of much of the tropics, where soil productivity is very dependent on SOM (Kapkiyai et al., 1998; Ouedraogo et al., 2001; Tiessen et al., 1994). Global reviews by both Murty et al. (2002) and Don et al. (2011) showed an average decrease in soil carbon after conversion of forests to cropland of around 30 %. Murty et al. (2002) did not find a similar general trend when converting forests to pasture, whilst Don et al. (2011) did find such a decline (12 % decline in soil C).

The aim of this study is to assess changes in soil parameters and their variability at sites that have been under agricultural cultivation for different times in Kenya, with the objective of identifying the time course of these changes.

2 Material and methods

2.1 Site description

The study was undertaken in the South Nandi forest (00°06′ N, 35°00′ E) and in cultivated plots adjacent to it near Kaptumo, Nandi district, western Kenya (Fig. 1). South Nandi forest is one of the largest remaining fragments of the original Kakamega Forest, and is one of the last remnants of virgin tropical rainforest in this area. Since 1895, parts of the forest have been cleared and converted to agriculture. The altitudes of the sites range between 1850 and 2040 m above sea level. Mean annual temperature is about 19.6 °C, with mean annual precipitation of 2000 mm, distributed across two rainy seasons (from March to June, and from September to November) (Solomon et al., 2007; Kimetu et al., 2008). The soils of South Nandi forest are well-drained, deep and dark to reddish brown with friable clay and a thick humic top layer, principally developed on biotite gneiss parent material. They are classified as humic nitosols (FAO-UNESCO, 1997). The natural vegetation is composed of Guineo-Congolian tropical rainforest species, including *Aningeria altissima, Milicia excelsa, Antiaris toxicaria* and *Chrysophyllum albidum*, and montane forest species, including *Olea capensis* and *Croton megalocarpus* (Solomon et al., 2007). The agricultural landscape is dominated by a mixture of maize fields and grass-covered grazing land. At this altitude and latitude, almost all grasses are C_4 (Tieszen et al., 1979). There are also tea fields. Sampling was done in late March 2009, i.e. well into the rainy season, and the maize was 10 cm to 1.5 m high in the different fields depending on planting time (higher if the farmer planted early in the season).

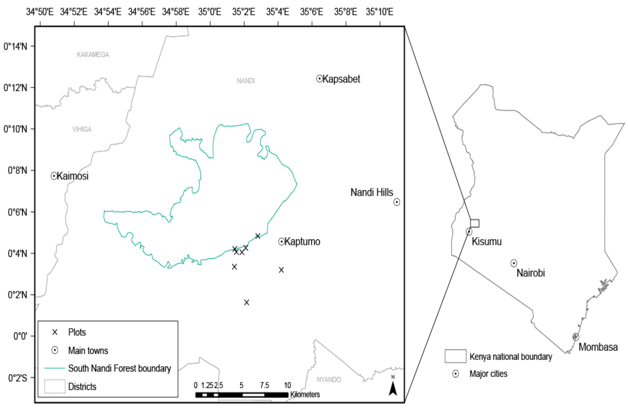

Fig. 1. Map of Kenya showing the study area and the study plots near Kaptumo, Nandi District.

2.2 Study design

The study was conducted on eight different plots with an average size of 40 m^2: two of them within existing natural forest (0 yr since conversion) and six in maize (*Zea mays* L.) fields. The six maize plots had been in agricultural production after conversion for 39 (two plots), 57, 69 or 119 (two plots) yr, as reported by the farmers. Plots were randomly placed in farmers fields and in forest sites. Thus, the chronosequence ranging from 0 to 119 yr was established using the eight study plots (Table 1). The sampling sites were all within a 25 km^2 area (Fig. 1). The two closest fields or sites were 300 m apart.

Land use since deforestation has been alternating between maize cultivation and grass (C_4) pastures. Crop yield estimates were obtained by interviewing farmers and recording their verbal estimates of yield as bags (approx. 90 kg) per acre (0.4 ha) ($n = 5$).

Surface soil bulk density was determined from four undisturbed samples per plot. The samples were collected using a cylindrical sampler 5 cm long and 5 cm in diameter. Dry bulk density was calculated by dividing the mass after oven drying at 70 °C by the volume of the core. Four samples for soil carbon, nitrogen and aggregate analyses were taken from the topsoil in each plot. Subsamples of the soil were analyzed for %C, δ^{13}C and %N on an elemental analyzer – isotope ratio mass spectrometer (EA-IRMS) that was linked to an element analyzer (Carlo Erba CHN1110). For soil particle analysis, samples were wet-sieved through successively smaller mesh sizes. For each sieve, samples were shaken for 2 min (25 strokes/minute) with an amplitude of 3 cm. Mesh sizes were 2 mm, 0.25 mm and 0.053 mm, which collected large macroaggregates (>2 mm), small macroaggregates (0.25–2 mm) and microaggregates (0.053–0.25 mm), respectively. The fraction that passed through all sieves, i.e. <0.053 mm, was silt and clay. Previous work by Kimetu et al. (2008) at the same site concluded that textural differences between the sites were small, albeit on a small sample of top soil ($n = 2$), and that those small differences were due to length of agricultural cultivation rather than intrinsic soil property differences. Soil texture analyses were not done in this study, but it is assumed that the sites had a similar soil texture. The clay-silt content is given in Table 1.

Infiltrability (Hillel, 1971) was measured using the double-ring infiltrometer method (Bouwer, 1986) at six points per plot. The inner and the outer rings had a diameter of 20 and 30 cm, respectively. Water was carefully poured into the inner and outer ring to avoid disturbing the soil surface, maintaining a head of 3–4 cm during the measurements. The purpose of using a double-ring infiltrometer is for the space between the two rings to prevent edge effects due to non-vertical flow affecting vertical infiltration from the inner ring (Bouwer, 1986). The advantages of using the double-ring method are its simplicity, relatively low cost and common use (Teixeira et al., 2003).

The water level in the inner ring was measured every 3 min for one 15 min period each hour, giving 5 infiltration readings per ring every hour. Measurements continued until a constant infiltration rate was reached, or after four hours from the time measurements started.

The steady-state infiltrability (Hillel, 2004) and sorptivity constants (defining the soil capacity to absorb water when the water flow is influenced by a matric pressure gradient) were estimated by means of curve-fitting using Philip's equation (Philip, 1957), by minimizing the squared residuals using the solver tool in Microsoft Excel.

According to Philip's equation,

$$I = st^{0.5} + i_c t$$

where I (mm) is the cumulative infiltration at time, t (h), for infiltration; s (mm h$^{-0.5}$) is the term defined as sorptivity; i_c (mm h^{-1}) is the term defined as transmissivity or steady-state infiltrability.

2.3 Statistical analyses

Correlations between parameters (infiltrability, bulk density, %C, %N, macro- and microaggregates, and yield) were calculated using Spearman's non-parametric rank correlation (PASW 18). Due to the non-normal distribution and unequal variances of the data, tests of significance for differences between median values were conducted using Mood's median test. We used the Levene's test to test for differences in variation between the forest (year 0) and the agricultural (year 39–119) sites (Minitab 16).

3 Results

3.1 Steady-state infiltrability

Infiltrability was significantly higher ($p < 0.05$) in the forest (year 0) compared with any of the agricultural fields.

Table 1. Sample site information.

Location	Elevation (m a.s.l)	Slope	Aspect	Site age (yr)	Field size (ha)	Silt + clay (%)
Chepkongony	2026	4.5°	South	0		2.0
Chepkongony	2026	4°	East	0		0.9
Chepkongony	2019	14°	East	39	0.2	2.6
Chepkongony	2036	10°	South	39	0.4	6.5
Chepkongony	1956	5°	Northeast	57	1.2	5.1
Chepkongony	1990	11.5°	South	69	0.3	8.7
Kaptumo	1877	3°	East	119	0.8	9.0
Koyo	1919	11.5°	North	119	0.6	8.2

Infiltrability was significantly lower in the sites that had been under agriculture for 39 yr than in the forest, but significantly higher than in fields that had been under agriculture for 57–119 yr ($p < 0.05$). In the forest, the median infiltrability was 342 mm h^{-1}. In the plots that had been converted to agriculture more than 57 yr ago, most (83 %) of the infiltrability values were below 100 mm $h^{-1}$and 17 % of the sampling points had rates below 50 mm h^{-1}. Infiltrability was below 100 mm h^{-1} for all sampling points in the plots that had been converted 119 yr ago (Table 2, Fig. 2). Thus, infiltrability has decreased with time since conversion, particularly during the first 57 yr (Fig. 2). The median infiltrability after 119 yr of cultivation was only around 15 % of the median infiltrability in the forest. The coefficient of variation (CV = stdev/mean) was 35–80 % within plots. Variance was significantly higher ($p < 0.05$) in the forest (year 0) than in the agricultural plots, with infiltration rates ranging from 37 mm h^{-1} to 859 mm h^{-1} in the forest, whilst after 119 yr of cultivation, the range was between 18 mm h^{-1} and 78 mm h^{-1} at the plots that were converted 119 yr ago (Fig. 2).

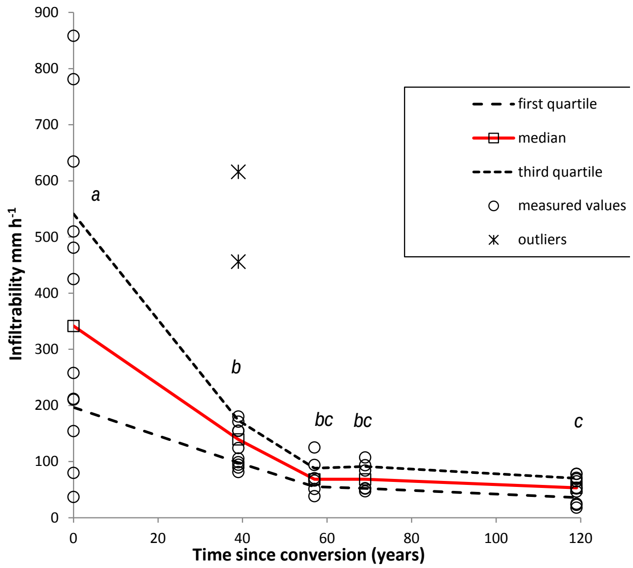

Fig. 2. Infiltrability rates for the plots along the chronosequence as a function of years since conversion to agriculture. Letters in italics represent significance. Years since conversion not followed by the same letter are significantly different ($p < 0.05$, Mood's median test).

3.2 Bulk density

Median soil bulk density was 52 % lower for the plots that had been under agriculture for 39 yr than for the plots in the forest ($p < 0.05$) and ranged from from 0.62 g cm^{-3} in the forest to 0.94 g cm^{-3} in cultivated fields (Fig. 3a). There were no significant differences in BD for any of the agricultural sites (39–119 yr). The CV for BD data was 5–25 %.

3.3 Soil C, N and δ^{13}C

Soil C and N content were significantly ($p < 0.05$) lower in the cultivated fields (median 3.8 and 0.4 % respectively) compared to the forest (6.2 and 0.7 % respectively) (Fig. 3b and c). Median δ^{13}C was significantly lower in the C_3 dominated forest (−26 ‰), than in the cultivated fields, which all had similar values around −18 ‰ ($p < 0.05$) (Fig. 3c). The CV for C and N data ranged from 5–25 %. Variance was significantly higher ($p < 0.05$) for N in the forest.

3.4 Soil aggregates

The median percentage large macroaggregates (>2 mm) was significantly higher ($p < 0.05$) in the forest (37.3 %) than for the sites that had been converted 57 to 119 yr ago (2.5–11.7 %) (Fig. 3e). There were no changes in small macroaggregates (0.25–2 mm; data not shown), but there was an increase in median values of microaggregates (0.053–0.25 mm) with time since conversion: 6.9 %in microaggregates in the forest, which was significantly lower ($p < 0.05$) than in cultivated land (25.7–39.2 %) 57 to 119 yr after conversion (Fig. 3f). Variance was significantly lower ($p < 0.05$) for microaggregates in the forest.

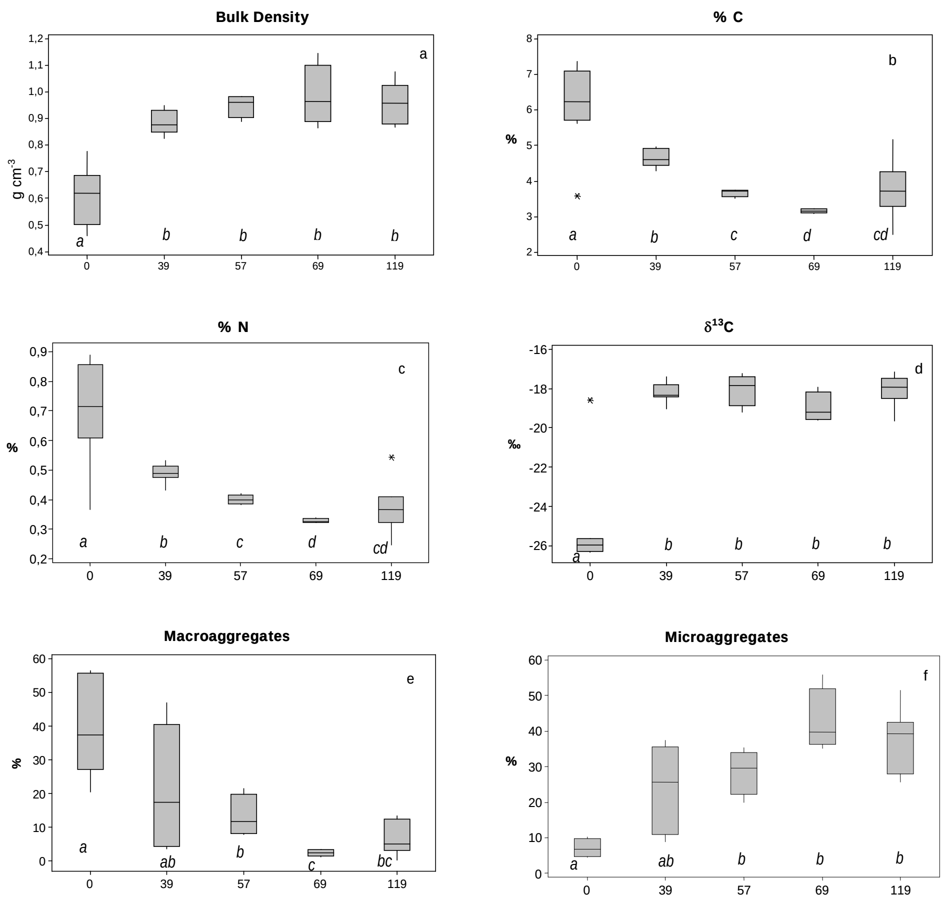

Fig. 3. Boxplots of soil parameters. **(a)** Bulk density, **(b)** soil carbon (%C), **(c)** soil N (%N), **(d)** δ^{13}C, **(e)** macroaggregates, **(f)** microaggregates. X-axis scale is time since conversion. * represents outliers. Box plot shows median, i.e. line in grey box, the first (below median line) and third (above median line) quartiles. Whiskers extend to the maximum or minimum data point within 1.5 box heights from the top or bottom of the box. Letters in italics below bars represent significance. Years since conversion not followed by the same letter are significantly different ($p < 0.05$, Mood's median test).

3.5 Correlations

All parameters, except ^{13}C, correlated well ($p = <0.05$) with time since conversion (Table 2). Infiltrability was negatively correlated ($p = <0.05$) with age, bulk density; the % microaggregates were positively correlated ($p = <0.05$) with %C, %N and large macroaggregates and weakly with yield ($p = <0.1$). Infiltrability, bulk density, C and N were correlated strongly with each other and with time since conversion ($p < 0.05$ or $p < 0.01$) (Table 3).

4 Discussion

In this study we report a gradual decline in infiltrability, soil C and N concentrations after conversion of forest to agriculture in a 119 yr chronosequence. Median infiltrability was reduced to 40 % of the forest rates (342 mm h^{-1}) in the plots that had been under cultivation for 39 yr (140 mm h^{-1}) and to 15 % in the plots that had been under cultivation for 119 yr (46 mm h^{-1}). Yimer et al. (2008) reported similar high infiltrability values from Ethiopian forests (around 450 mm h^{-1}) but a faster decline, to 25 % after 15 yr of crop cultivation or grazing land (around 120 mm h^{-1}).

The time resolution of our data is not optimal to evaluate the initial process of soil degradation, i.e. the youngest agricultural plot was converted 39 yr ago. Much, or even most, of the differences measured between year 0 and 39 might have occurred during the first few years, as shown in other studies. In Ibadan, Nigeria, infiltrability decreased from 1560 mm h^{-1} in the forest to 398 mm h^{-1} after three years of traditional farming (Lal, 1996). Giertz and Diekkruger (2003) and Giertz et al. (2005) compared agricultural areas of unknown age with natural vegetation of dry forests and Savannah in Benin, West Africa, and found that

Table 2. Steady-state infiltrability (mm h^{-1}) for sites of different times since conversion to agriculture, represented by the median rate and % of readings above and below particular rates. Q3–Q1 is the interquartile range covering 50 % of the values.

Years since conversion	Median infiltrability (mm h^{-1})	Q3–Q1	% < 50 mm h^{-1}	% < 100 mm h^{-1}
0	342	435	8	17
39	140	82	0	33
57	69	54	17	83
69	69	46	17	83
119	46	47	36	100

average infiltrability and variability in the agricultural areas were about 30 % of the infiltrability and the variation in forest.

Giambelluca (2002) found a recovery in soil conductivity after fallowing long-term agricultural land in Vietnam from 20 to 60 mm h^{-1} after less than 19 yr and to 80 mm h^{-1} after 30 yr. This is consistent with the 2–5 fold increase found in a meta-analysis (Ilstedt et al., 2007) of studies of recovery after afforestation (e.g. Bonell et al., 2010; Chirwa et al., 2003; Mapa, 1995) and agroforestry practices (e.g. Hulugalle and Ndi, 1994, 1993; Salako and Kirchhof, 2003).

In comparison to the effects of non-mechanized agriculture, the effects of grazing seem more severe and rapid (e.g. Spaans et al., 1990; Zimmermann et al., 2006, 2010; Zimmermann and Elsenbeer, 2009; Hassler et al., 2011; Martinez and Zinck, 2004). For example, Zimmerman et al. (2006) measured 10-fold higher infiltrabilities in forests and other tree-covered systems than in nearby pasture land, and Martínez and Zinck (2004) found infiltrability of 2 mm h^{-1} in pastures and 143 mm h^{-1} in forests on fine textured soils. However, in a 12-yr grazing exclusion experiment in dry forests in Burkina Faso, infiltrability decreased with grazing intensity from 83 without grazing to 50 mm h^{-1} in intensively grazed plots (Savadogo et al., 2007), indicating that the very large reductions seen in more humid areas are perhaps less drastic in dry forests, maybe partly because pretreatment infiltrability rates were already low.

For infiltrability and C and N concentrations, we detect differences also after 39 yr, implying that degradation processes continued. It also indicates that these parameters are suitable to detect soil changes late in the process.

A decrease in variability, especially for infiltrability, followed the decreases in average and median values. High variability in infiltrability is commonly reported (Bamutaze et al., 2010; Lal, 1996). Similar values to those reported here for the CV of infiltrability (33–81 %) were obtained by Van de Genachte et al. (1996) in a rain forest in Guyana. Infiltrability is not difficult to measure, but, due to the large variation, it needs many replicate samples. The variation itself provides valuable information. In the forest, there were locations with infiltration capacity as low as in the agricultural fields. However, there were also many locations with high infiltrability, whilst such locations were largely missing in agricultural fields, especially in the plots that had been converted more than 57 ago (Fig. 2).

Tropical rains are often very intensive; in the area at least 10 % of the annual rainfall (about 12 times per year) has intensities >50 mm h^{-1} for periods of 15 min. Occasionally (about once per year), rainfall intensities reach >100 mm h^{-1} (Moore, 1979). These events represent 1 % of annual rainfall. In the cultivated area that was converted 119 yr ago, 36 % of the randomly sampled points had infiltrability below 50 mm h^{-1} and all (100 %) had infiltrability below 100 mm h^{-1} (Table 1). As there are very few agroforestry structures that could act as "funnels" for infiltration (high infiltrability areas), there is considerable risk for surface runoff and erosion, and hence less water is available for transpiration (Stroosnijder, 2009). Erosion is known to reduce crop yield via a reduction in effective rooting depth, loss of plant nutrients and soil organic carbon, loss of land area, and direct damage to seedlings (Lal, 1998). Decreased infiltration may also lead to less groundwater recharge if the decrease is higher than the difference in evapotranspiration between the systems (Bruijnzeel, 2004; Malmer et al., 2010).

From a landscape management perspective, one implication of the results of this study may be that wooded structures, e.g. tree lines or shelterbelts along contours (Ellis et al., 2006; Stroosnijder, 2009), woodlots or other agroforestry elements, need to be included in the agricultural system (for both restoring degraded landscapes and new agricultural areas), at a scale large enough to create enough high infiltration locations to reduce the risk for runoff and erosion at both the farm and landscape scale. Trees/agroforestry in the landscape can improve infiltrability (Nyamadzawo et al., 2007, 2008; Stroosnijder, 2009; Ilstedt et al., 2007) and will also enable more C sequestration or smaller losses of C (Nair et al., 2009; Nyberg and Hogberg, 1995; Stahl et al., 2002; Albrecht and Kandji, 2003). The appropriate scale of these agroforestry components will depend on rainfall distribution, soil type, slope, agroforestry type and agricultural practices. However, further research is needed on this subject, particularly on how runoff and infiltration at the landscape scale are affected by the interaction between the spatial variability in rainfall distribution and infiltrability.

Soil bulk density has increased significantly within 40 yr since conversion to agriculture. Soil bulk density is easy and cheap to measure, and thus appropriate for general descriptions of long-term soil changes. However, it also depends on the soil texture. Infiltrability (and C and N) shows changes in the later part of the chronosequence more clearly than bulk density (and aggregate size and δ^{13}C), i.e. infiltrability was significantly lower ($p < 0.05$) after 119 yr than after 39 since conversion (Figs. 2 and 3a).

Soil C was quite high in the forest and decreased by about 35 % with cultivation, averaged over the six agricultural sites (39–119 yr). δ^{13}C increased after forest conversion

Table 3. Correlations between soil and site parameters, including time since conversion to agriculture. i_c = steady-state infiltrability; BD = soil bulk density; Macro-aggr.% = % large macroaggregates (>2 mm) in soil; Micro-aggr.% = % microaggregates (0.053–0.25 mm) in soil. Spearman rank correlation for plot medians (top value) and p-values (bottom value), $n = 8$ for all parameters, except yield where $n = 5$.

	i_c	BD	%C	%N	Macro-aggr.%	Micro-aggr.%	Yield
Age	−0.982	0.788	−0.849	−0.849	−0.727	0.752	−0.889
p-value	<0.001	0.020	0.008	0.008	0.041	0.032	0.044
i_c		−0.810	0.833	0.857	0.714	−0.714	0.866
p-value		0.015	0.010	0.007	0.047	0.047	0.058
BD			−0.929	−0.881			−0.866
p-value			0.001	0.004			0.058
%C				0.976	0.690	−0.643	0.866
p-value				<0.001	0.058	0.086	0.058
%N					0.786	−0.714	0.866
p-value					0.021	0.047	0.058
Macro-aggr. %						−0.929	
p-value							0.001

to agriculture based on C_4 plants (maize and grass for grazing) from – 26 ‰ to −18 ‰, suggesting that the initial degradation of forest C occurs rapidly (within the first 39 yr), but that subsequent decomposition of forest C is very slow. The low variability in δ^{13}C and the stable δ^{13}C values in the plots that had been converted more than 39 yr ago indicate that varying between crop or grazing and tea on the same fields is not common. Had there been regular use of C_3 plants, e.g. trees or tea, in the agricultural practice, lower values and larger variation would have been seen in δ^{13}C. This is also supported by the farmers not mentioning it, but discussing the alternation of maize fields with grazing land (C_4 grasses). This suggests that some 40–50 % of the soil C is still of C_3 origin after 120 yr. Others have reported similar patterns of forest-derived C, for example in Ethiopia and in Brazil (Lemenih et al., 2005b; Lisboa et al., 2009).

The percentage of large macroaggregates in soil decreased with time since conversion, probably as a result of soil tillage and decomposition. Microaggregates increased with time since conversion, probably reflecting macroaggregate breakdown (Wright and Inglett, 2009). Microaggregates are relatively inert, with reported mean residence times (MRT) of 222 and 498 yr (Liao et al., 2006; Lisboa et al., 2009). Inflow to this soil pool is reported to be faster than outflow; this fact was supported by our δ^{13}C data, which also indicated a pool of relatively inert soil C (40–50 % of soil C). The slightly higher variation in soil C, δ^{13}C and in macroaggregates at the sites that were converted 119 yr ago might indicate a difference in management, e.g. inclusion of trees in fallows or tea during some periods, in the oldest fields.

There were very strong correlations between the time since conversion and infiltrability, bulk density, C% and N%, as well as between these four parameters, showing that they are well-suited for describing soil changes in chronosequences from forest to agricultural land (Table 2). These four parameters are furthermore cheap and easy to measure and, except for infiltrability, commonly used in agricultural soil research. In spite of admittedly weak data ($n = 5$ and farmer estimates of crop yield), the yield correlated with time since conversion and correlated weakly ($p = 0.058$) with the first four soil parameters (Table 2). The reported yields and their decline with time since conversion are similar to measured yield data from the same area (Ngoze et al., 2008).

5 Conclusions

Infiltrability, soil C and N concentrations were lower in the agriculture plots than in the forest plots. Bulk density was higher in agricultural fields than in the forest. Median infiltrability values at the plots that were converted to agriculture 119 yr ago were about 15 % of the values in the forest plots. The variation in infiltrability and %N was higher in the forest than in the agricultural fields. About half of the soil carbon in the forest was lost during cultivation. Despite this, 50 % of the remaining carbon consisted of organic material originating from the forest. Whilst most changes appear to occur within 40 yr, some soil parameters (infiltrability, C% and N%) appear to decline also after more than 40 yr of agriculture.

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