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ORIGINAL PAPER

Changes in soil chemistry associated with the establishment of forest gardens on eroded, acidified grassland soils in Sri Lanka

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Abstract Topsoil properties were determined in forest gardens established about 20 years ago on eroded grassland soils (abandoned tea lands) in the wet zone of the Sri Lankan highlands. They were compared with adjacent, eroded grasslands (abandoned tea lands) on strongly weathered soils vs soils at earlier stages of pedogenic development in a two-way analysis of variance. Soil pH in forest gardens was, on average, 6.1, nearly one unit higher than in the adjacent grasslands. In the garden soils, the cation exchange capacity (CEC measured at pH 4.8) was nearly double, exchangeable calcium concentrations five times and exchangeable magnesium three times as high as in the grasslands soils. Total soil N content was found to be nearly 40% higher in the gardens. Topsoil gravel contents in the gardens were less than half as high as in the grasslands. The increases in exchangeable bases and N in gardens, relative to grasslands, were attributed to increased nutrient retention and acquisition. Higher retention was partly due to the higher $CEC_{pH4.8}$, and probably to reduced erosion and increased, continuous fine root density in the garden topsoils. Higher field CEC in gardens was likely to result from generally higher C contents and from the reversal of acidification, presumably caused by base accumulation and decomposition processes. Our results suggest that forest garden establishment on degraded grasslands can lead to accumulation of mobile nutrients in the topsoil, probably due to increased nutrient retention, subsoil uptake and litter input exceeding nutrient uptake by the standing biomass.

Keywords Soil acidity · Exchangeable bases · CEC at low pH · Multistrata agroforests · Tropical tree fallows

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Introduction

Soil erosion by water and chemical degradation are the most widespread and severe causes of productivity declines on sloping agricultural land in South and Southeast Asia (Wood et al. 2000). On the slopes in the wet zone of the Sri Lankan highlands, tea plantations have been re-established for many decades. Krishnarajah (1985) estimated an average annual soil loss of 40 t ha^{-1} from these tea plantations on steep and very steep slopes. Base cation leaching, volatilisation and ash dispersal (Bruijnzeel 1998) and the widespread use of ammonium sulphate fertilizer contributed to soil acidification. Declining productivity and weed invasion resulted in the abandonment of 40,000 ha of tea plantations in Sri Lanka by the late 1980s (Botschek et al. 1998). After invasive grasses or other plants forced production to be abandoned, fire has been reported to delay succession and biomass accumulation, even in a number of humid tropical areas of Sri Lanka where spontaneous fires are rare (Garrity et al. 1997). Within the tea producing region South of Kandy, Amarasinghe and Pemadasa (1982) reported continued signs of erosion in the 'humid zone dry patana grasslands', some of which were dominated by *Cymbopogon nardus* L. Rendle and *Pennisetum polystachyon* L. (Schult.).

While attempts at re-cultivating degraded grasslands with monocultures of a number of crops failed, mixtures of crops and useful fallow species adapted to the local site conditions appear to be promising alternatives (Liebman and Dyck 1993; de Foresta and Michon 1997; Macdicken et al. 1997). However, the distribution and socio-economic importance of diverse, tree-dominated homegardens has been widely underestimated throughout the tropics (Nair 2001). Already during the late 1970s many smallholders converted former tea lands that had become grasslands into mixed home gardens, some of which turned into permanent multi-species tree plantations. The stature and structure of such forest gardens have been described as most similar to secondary forests (Perera and Rajapakse 1991; Ewel 1999). However, substantial amounts of a wide range of products, including fruits, timber, fuelwood, etc., are exported from the forest garden ecosystem for home consumption and sale (Nuberg et al. 1994). This implies that large amounts of nutrients are taken up from the soil by the growing forest garden biomass during its establishment, some of which are not returned to the soil. The effects of tree-dominated polycultures on soil properties have been described as elusive (Nair 2001). Due to the long time it takes for chemical changes in the mineral soil to be detectable and due to associated methodological complications, the nature and extent of the effects have not yet been determined. The aim of this study was to investigate changes in topsoil chemical properties associated with the establishment of forest gardens on eroded, acidified grassland soils in the wet zone of Sri Lanka.

Materials and methods

Approach and site selection

Topsoil properties of forest gardens established about 20 years ago on eroded grasslands (former tea lands) were compared with adjacent eroded grasslands, which had also been under tea production for many years. It was assumed that the forest garden sites, at the time of conversion from grassland, were characterised by an ecological and management history of the soil and vegetation, similar to those of the adjacent grassland. This was validated by historical records and farmer's interviews with a semi-directive and a semi-structured component to minimise error (Huntington 2000). Recognition of the state of soil degradation at the time of forest garden establishment was based on vegetation cover and composition, residual surface gravel accumulation, soil structure and consistence, as well as on the color of the topsoil (FitzPatrick 1992; Stocking and Clark 1999). The duration of the grassland stage exceeded 5 years at forest garden as well as grassland sites. Although no mineral fertilizer has been applied at any site since tea production had been abandoned, most forest garden soils were amended with variable amounts of animal manure. Animal manure dry matter inputs were estimated (Table 1), based on number of

Table 1 Variability in major forest garden management factors potentially influencing nutrient stocks and/or fluxes

Site	Soil type	Age (years)[1,2]	Slash treatment[2,3]	Manure application[2]						Basal area (m^2 ha^{-1})	
				Source	Period (years)	Total input (kg ha^{-1})[4]				At sampling	Exported as timber[1,2]
						DM	N	P	K		
1L	RBL	20	Mulched unburned	Dairy	Initial 0.02	136	7	1	3	24	17
2L	RBL	22	Mulched unburned	Poultry	Last 12	6,090	488	150	149	21	11
3L	RBL	20	Mulched unburned	Dairy	Initial 3	4,244	223	39	108	33	0
1P	RYP	22	Burned	Dairy	Initial 0.02	272	14	2	7	12	50
2P	RYP	23	No data	None	None	0	0	0	0	19	No data

RBL Reddish-brown latosolic soils, *RYP* red-yellow podzolic soils
[1] Since forest garden establishment
[2] Elicited in semi-directive/semi-structured interviews with farmers
[3] Treatment of slashed vegetation before forest garden establishment
[4] Estimated total animal manure dry matter (*DM*), N, P, K contents since forest garden establishment, based on number of animals, manure proportion used for sampling plot area, duration of application, manure source (all from farmers interviews[2]) and means from input–output model from Powers and van Horn (2001)

animals the manure was derived from, the proportion of total manure used for sampling plot area, duration of application and manure source. Estimations of nutrient contents applied in manure were based on means from input–output model from Powers and van Horn (2001). Nutrient export from the forest garden ecosystem as harvested products was also highly variable, including timber, fuelwood, fruit and other plant products for home consumption and sale. Basal area estimates of timber removal (Table 1) were based on harvested log numbers and sizes as elicited in the semi-directive/semi-structured interviews.

Within the mid-country wet agro-ecological region WM_2 (Panabokke 1996) in Kandy District in the central highlands of Sri Lanka (80°32′–80°37′E; 7°07′–7°18′N), five replicate pairs of gardens and adjacent grasslands were selected on different upper slopes between 540 and 750 m above sea level (m.a.s.l.). Annual rainfall in the study area averages around 2,500 mm. Adjacent sites were located on the same underlying solid-rock geology (GSMB 1996). Slope angles varied between 28 and 70%, not exceeding 8% difference between adjacent paired sites. Three pairs of sites were selected on the relatively young, freely draining reddish brown latosolic (RBL) soils, which contain primary minerals in the topsoil and are derived from basic and intermediate rocks, with a very well developed structure. They largely correspond to rhodic nitisols (Panabokke 1996; FAO–ISRIC–ISSS 1998). The other two pairs of sites were selected on the freely draining red-yellow podzolic (RYP) soils, which are characterised by clay translocation and a low silica-to-sesquioxide ratio. They are strongly acid soils developed from non-basic Precambrian crystalline metamorphic rocks and correspond largely to haplic acrisols and alisols (Panabokke 1996; FAO–ISRIC–ISSS 1998).

Plant recording, soil sampling and analysis

Within 15 by 15-m plots, plant species and girth at breast height (GBH) of all stems above 5-cm GBH were recorded for the calculation of basal area. Between July and early September 2000, soil cores were taken within these plots at ten random points to a depth of 10 cm of the A horizon after removing the litter layer. For soil chemical analyses, bulked samples were sieved to 2 mm and cone-quartered at all stages of sub-sampling. All analyses were carried out on duplicate sub-samples. Soil pH was determined on field moist soil in de-ionised water (1:2.5) on the day of sampling. Oven dry soil (Anderson and Ingram 1993) was used for all other soil chemical analyses. For determination of the cation exchange properties, soil samples were saturated with 1 M ammonium acetate (Anderson and Ingram 1993) at pH 4.8 in leaching tubes fitted with non-absorbent cotton wool. After washing with 80% ethanol, the exchangeable ammonium was displaced by 1 M sodium chloride at pH 2.5 (Anderson and Ingram 1993). Ammonium in the leachate was measured by flow injection colorimetry (TECATOR 50-10, Foss, UK). Exchangeable base cations and aluminium were determined in the ammonium acetate leachate, using an atomic absorption spectrophotometer (Analyst 100, Perkin Elmer). Total C and N contents were determined using an NCS-Analyzer (NA 1500, FISONs Instruments). This involves ignition by flash combustion of ground soil samples in pure oxygen at 1,700°C and direct injection into a gas chromatograph coupled with a thermal conductivity detector. Total P was determined on ground soil samples by digestion in concentrated sulphuric acid and hydrogen peroxide for 2 h at 365°C (Anderson and Ingram 1993). Organic P was measured as the difference in total P content between ignited (at 500°C) and un-ignited samples extracted in 1 M sulphuric acid (Anderson and Ingram 1993). Phosphate in all extracts was determined by flow injection colorimetry using the molybdate/stannous chloride method (Allen 1989), with separate working standards made up in the same solutions as used for each extraction.

The light fraction of soil organic matter, i.e. mineral-free macro-organic matter with a density <1 g cm^{-3} (Anderson and Ingram 1993), was separated from the heavy fraction by shaking end-to-end of duplicate field moist soil samples in tap water on the day of sampling. The suspension was allowed to settle for 30 min before decanting through a 200-μm sieve. The separation procedure was repeated three times. The material retained in the sieve was washed off with tap water onto a filter paper for oven drying for 24 h at 105°C and subsequent weighing.

For water holding capacity (WHC), soil porosity and soil respiration rate, ten undisturbed soil cores were taken in polyvinyl chloride tubes of 4.3-cm diameter to 10-cm depth at five random points within the plot. WHC was determined by placing one of the two soil cores from each point on Whatman filter paper no. 42 in a Büchner funnel and submerging it in water. Subsequently, the soil was left to drain until it ceased dripping for 2 min. The water content at this stage was expressed as percentage of soil oven dry weight, which was regarded as WHC (method modified from Ilstedt et al. 2000). Soil total porosity was calculated from soil dry bulk density for non-stony soils (Anderson and Ingram 1993). For determination of soil respiration rate, the second set of undisturbed soil cores taken from the five sampling points was brought to 60% WHC after 2 days of refrigeration. The cores were placed in 1-l preserving jars fitted with injection ports sealed with rubber stoppers and left open for a 12-h equilibration in the dark at 30±2°C. After the preserving jars remained closed for further 12 h, gas samples of 0.5 ml were taken and injected into a standardised gas chromatograph (Shiadzu GC-9 AM) to measure CO_2 content (method adapted from Amador et al. 2000).

Statistical analysis

A two-way analysis of variance (ANOVA) for land use and soil type was carried out on soil chemical parameters with the statistical software package Minitab 13.1. For WHC, soil total porosity and soil respiration rate, the statistical package SPSS 10.07 was employed. Anderson–Darling normality tests were carried out on residuals from the balanced ANOVA. Bartlett's tests for equality of variances were carried out on absolute values. The absolute values of not normally distributed data or data with unequal variances were transformed by the Box–Cox procedure (MINITAB 2000).

Results and discussion

Plant community

Forest garden plots contained, on average, 15 species of tree crops of a wide range of ages and with differential heights at maturity. The most common canopy species were jackfruit (*Artocarpus heterophyllus* Lam.), mango (*Mangifera indica* L.), areca nut (*Areca catechu* L.), coconut (*Cocos nucifera* L.), avocado (*Persea americana* Mill.), fragrant champaca (*Michelia champaca* L.), fishtail palm (*Caryota urens* L.), mahogany (*Swietenia mahogani* L.) and rubber (*Hevea brasiliensis* Müll.-Arg.). In canopy gaps, mother-of-cacao (*Gliricidia sepium* [Jacq.] Steud.), pepper (*Piper nigrum* L.), coffee (*Coffea* sp.) and the devil tree (*Alstonia macrophylla* Wall. ex G. Don.) were the most common species. Total tree basal area of stems >5 cm GBH in forest garden plots averaged at 21.6 m^2 ha^{-1} (±SE=3.4), similar to natural tropical secondary rainforests of the same age (Peña-Claros 2003). Adjacent grassland plots were dominated by *C. nardus* L. Rendle and/or *Pennisetum polystachyon* L. (Schult.) (75–100% of total vegetation cover), with some dense patches of the fern *Dicranopteris linearis* (Burm. f.) at one site. A few other herbs and woody perennials were present at low densities in all grassland plots, including relict tea plants (*Camellia sinensis* [L.] Kuntze.), resulting in an average total tree basal area of 0.004 m^2 ha^{-1} (±SE=0.002). This confirms that the above-ground plant biomass of the forest gardens is several magnitudes greater than that of the grasslands.

Due to their diverse biological and management characteristics, forest garden systems have inherently variable nutrient input and output pathways, for instance, including manure additions, fecal deposits by frequent visits of monkeys and birds, which also feed on forest garden biomass, removal of forest garden biomass as products for human use. Neither of the nutrient input or output pathways estimated quantitatively in this study, i.e. of manure nutrient additions, nor of timber removal (Table 1), were correlated with any of the measured response variables, suggesting that these factors are unlikely to be determining factors for soil properties evaluated here.

Reversal of acidification

In forest gardens, the soil pH was, on average, 6.1, i.e. nearly one unit higher than in the adjacent grassland soils (Table 2), by far exceeding the variation between the soil

Table 2 Means of topsoil properties in two adjacent land use types ±SE of means in a wet-tropical region in Sri Lanka

	Forest gardens (n=5)	Grasslands (n=5)	p
pH (1:2.5 in H_2O)	6.06±0.11	5.16±0.11	*
$CEC_{pH4.8}$ ($mmol_c$ kg^{-1})	80.7±13.0	47.7±9.8	*
TEB ($mmol_c$ kg^{-1})	41.9±10.0	9.6±2.7	*
Base saturation (%)	50.2±6.4	18.9±2.3	**
Exch. Ca/Al ratio	2.30±0.83	0.26±0.10	***
Exch. Ca ($mmol_c$ kg^{-1})	29.72±8.25	5.18±1.44	**
Exch. Mg ($mmol_c$ kg^{-1})	8.22±1.84	2.44±0.77	*
Exch. K ($mmol_c$ kg^{-1})	3.23±0.88	1.39±0.57	
Exch. Na ($mmol_c$ kg^{-1})	0.76±0.10	0.61±0.08	
Exch. Al ($mmol_c$ kg^{-1})	18.37±5.19	23.06±4.47	
Soil total C (%)	2.44±0.40	1.44±0.08	
Soil total N (%)	0.226±0.029	0.143±0.008	*
SOM light fraction (g kg^{-1})	1.59±0.47	1.12±0.26	
Soil total P (ppm)	607±113	435±24	
Soil organic P (ppm)	160±21	124±6	
Acid-extr. P (ppm)	76.0±24.8	35.3±5.2	
Gravel content (mass %)	16.5±2.5	37.3±8.7	*
Soil WHC (%)	39.6±4.5	32.6±3.6	
Soil total porosity (%)	54.94±2.90	54.05±1.61	
CO_2 evolution (μg kg^{-1} h^{-1})	4.80±0.89	4.62±1.41	

Data were obtained from two soil types of contrasting inherent fertility, and pooled within each land use type

$CEC_{pH4.8}$ Cation exchange capacity at pH4.8; *TEB* total exchangeable bases; *SOM* soil organic matter

*, **, and *** indicate significant differences (p<0.05, p<0.01, and p<0.001, respectively)

types (Table 3). Noble et al. (2000) reported nearly one unit lower pH values in grasslands acidified by 30 years of intensive use, relative to nearby undisturbed dipterocarp forest in Thailand, which they assumed to be similar to the un-degraded soil before pasture establishment. A reversal of such acidification appears to be occurring within 20 years of forest garden cultivation in Sri Lanka, probably partly due to base accumulation (Nakano and Syahbuddin 1989). Other mechanisms that may have contributed to a reversal of soil acidification might be associated with decomposition processes of organic inputs, including, for instance, decarboxylation of organic acids, ammonification of residue N, oxidation of organic residues and/or adsorption of organic acids to Al and Fe hydrous oxides resulting in a release of hydroxyl ions (Haynes and Mokolobate 2001).

Exchangeable base accumulation

The sum of exchangeable bases in the garden soils was more than four times as high as in the grassland soils (Table 2). This was mostly due to the soil concentrations of exchangeable calcium, which were more than five times higher, but also to the concentrations of exchangeable magnesium, which were more than three times higher in the gardens than in the grasslands. Calcium occupied 37 and 11%, and magnesium occupied 10 and 5% of the cation exchange complex in the garden and grassland soils, respectively. The difference in exchangeable calcium between gardens and grasslands was significant (ANOVA of Main Effects of land use: $p<0.01$), although RBL soils showed three times as high levels as RYP soils (Table 3). Nakano and Syahbuddin (1989) suggested that exchangeable bases recover with increasing vegetation biomass if the fallow is allowed to proceed beyond the ‘bush’ stage and if erosion and leaching is checked by a continuous vegetation cover. Their 8-year-old bush fallow plot with 1.2 kg m^{-2} (±0.37 S.E.) above-ground biomass in West Sumatra had only accumulated 66% more exchangeable bases in the topsoil than in a nearby degraded grassland with bracken. This suggests that the 20-year-old forest gardens of this study accumulated, on average, more bases in the soil per year than the natural fallow in West Sumatra. The degree of base retention in the forest garden soil before the ‘bush’ stage with incomplete vegetation cover might well have been as low as in the 8-year-old natural fallow in West Sumatra. However, even 20-year-old forest gardens were characterised by gaps in the vegetation cover due to tree removal, some of which are exported from the ecosystem as timber and firewood (Table 1). In addition to preventing leaching and erosion, deep rooting garden trees may have recovered calcium and magnesium, which had been leached during acidification into the clay-rich argic horizon in the subsoil, and transferred it through litter and root turnover to the topsoil. The higher rate of base accumulation since the garden establishment on RBL soils as compared to RYP soils is presumably mainly due to higher rates of base release from mineral weathering and recovery from higher-activity clays in the subsoils (Panabokke 1996). The ratio of exchangeable calcium-to-aluminium in forest garden soils was more than seven times as high as in the grasslands

Table 3 Means of topsoil properties in two soil types ±SE of means in a wet-tropical region in Sri Lanka

	Reddish-brown latosolic soils (n=6)	Red-yellow podzolic soils (n=4)	p
pH (1:2.5 in H_2O)	5.70±0.19	5.47±0.33	*
$CEC_{pH4.8}$ (mmol$_c$ kg^{-1})	81.7±9.4	38.0±8.2	**
TEB (mmol$_c$ kg^{-1})	34.8±10.4	12.2±5.1	*
Base saturation (%)	38.1±8.1	29.3±10.1	
Exch. Ca / Al ratio	1.53±0.77	0.92±0.69	
Exch. Ca (mmol$_c$ kg^{-1})	24.09±8.41	7.48±3.22	*
Exch. Mg (mmol$_c$ kg^{-1})	7.10±1.83	2.68±1.17	
Exch. K (mmol$_c$ kg^{-1})	2.88±0.74	1.46±0.88	
Exch. Na (mmol$_c$ kg^{-1})	0.73±0.10	0.63±0.07	
Exch. Al (mmol$_c$ kg^{-1})	23.06±4.68	17.19±4.58	
Soil total C (%)	2.22±0.33	1.53±0.34	
Soil total N (%)	0.206±0.026	0.152±0.027	
SOM light fraction (g kg^{-1})	1.40±0.40	1.29±0.35	
Soil total P (ppm)	610±82	388±41	
Soil organic P (ppm)	160±16	116±5	
Acid-extr. P (ppm)	66.2±22.0	39.9±8.5	
Gravel content (mass %)	21.1±4.5	35.7±11.5	
Soil WHC (%)	40.0±2.8	30.3±5.2	
Soil total porosity (%)	56.8±1.6	51.0±2.3	
CO_2 evolution (µg kg^{-1} h^{-1})	6.33±0.71	2.28±0.28	*

Data were obtained from forest gardens and grasslands, and pooled within each soil type

$CEC_{pH4.8}$ cation exchange capacity at pH4.8; *TEB* total exchangeable bases; *SOM* soil organic matter

* and ** indicate significant differences ($p<0.05$ and $p<0.001$, respectively)

(Table 2). Additionally, the higher soil pH suggests that aluminium toxicity (Kinraide 1998) is unlikely in these forest gardens.

The cation exchange complex

The cation exchange capacity ($CEC_{pH4.8}$) in the garden soils of 81 $mmol_c\ kg^{-1}$ was about 70% higher than in the grassland soils (Table 2). This difference was significant ($p < 0.05$), in spite of the overwhelming evidence for double as high levels of $CEC_{pH4.8}$ in RBL soils compared to RYP soils (Table 3). The measured $CEC_{pH4.8}$ of the more acid samples, i.e. the grassland soils, was presumably realistic, because their fresh soil pH ranged from 4.8 to 5.4. Norton et al. (1999) stated that the effective CEC increases at a low rate up to around pH 5 and at a higher rate above pH 5. Therefore, it is likely that the samples with higher pH, i.e. the garden soils (pH between 5.7 and 6.4), had a higher effective field CEC than the measured $CEC_{pH4.8}$, due to the high proportion of pH-dependent negative charges in strongly weathered tropical soils (Menzies and Gillman 1997). Such an underestimation of the field CEC might be further exacerbated in forest garden soils by steeper CEC gradients across the relevant pH range, associated with a higher contribution of organic matter to CEC (Noble et al. 2000).

Reported decreases in the CEC associated with soil degradation vary widely, but may reach one third of the initial CEC in undisturbed forest, which has been attributed largely to soil organic matter losses (Noble et al. 2000) and erosion of topsoil clay (Norton et al. 1999). In slow growing, open post-agricultural fallows continued losses in topsoil CEC may occur, associated with the decomposition of organic matter (Sirois et al. 1998). The restoration of CEC during fallow succession is presumably limited by biomass accumulation and associated organic matter additions by colonizing trees (Aweto 1981). The high planting densities and biomass accumulation common in forest gardens, therefore, allow high potential rates of CEC recovery.

Soil carbon and nutrient concentrations

In this study, total soil C levels in the gardens tended to be, on average, 69% higher than in the grasslands (p=0.06). The only forest garden site with lower C levels than the adjacent grassland was also the only site with a coarse topsoil texture (loamy sand). In contrast, the adjacent grassland had a finer texture (silty clay loam), more similar to all other sites, and would, therefore, inherently maintain more stabilised organic C (see Ladd et al. 1996), regardless of present land use.

In slash-and-burn agriculture with short cultivation periods of annuals and sufficiently long subsequent secondary forest fallow, soil C tends to remain relatively constant throughout the cycle of clearing, cropping and secondary forest re-growth (Kotto-Same et al. 1997). However, soil C contents are typically reduced by more than 40% (Detwiler 1986), after prolonged maintenance of low-biomass agricultural or post-agricultural ecosystems with low organic matter additions to the soil, including annual cropping, low-density or low-stature perennial plantations, such as tea, or frequently burned grasslands without external nutrient inputs. Studies reviewed by McGrath et al. (2001) suggest that, after agricultural production is abandoned, soil organic matter (SOM) restoration during natural forest re-growth may take 20 years or more until biomass accumulation translates through sufficiently large rates of root turnover and litter build-up into subsequent accrual of SOM. Tian et al. (2001) found trends of higher SOM already after 6 years in significantly more productive, planted leguminous tree fallows, compared to natural woody fallows, and significant increases in SOM by at least 30% relative to continuous maize and cassava cropping. In the present study, farmers reported high biomass productivity of *G. sepium*, the most common leguminous tree species planted during the first years of forest garden establishment in the study region. The trend of increased SOM associated with forest garden establishment is likely to be due to higher rates of root and litter inputs and soil organic matter accumulation than in grasslands, which are subjected to fire in most dry seasons.

Total soil N levels in the gardens were 58% higher than in the grasslands, presumably associated with soil organic matter accumulation. However, there was more evidence for significant soil N than C accumulation during forest garden establishment. This was probably partly due to higher rates of inputs through N fixation in tree legumes, especially *G. sepium*, which had been widely planted in young forest gardens or gaps. Another source of N, presumably, was non-symbiotic fixation, e.g. by epiphytic cyanobacteria in the tree canopy (Kahindi et al. 1997). If in these conditions on-site N retention and rapid accumulation is to occur over several years, as was observed in the forest gardens in this study, increases in potentially mineralisable N (Tian et al. 2001) have to be accompanied by a synchronised uptake by sinks such as a continuous “fine-root safety net” of rapidly growing plant biomass. Otherwise, N pools that might be rapidly building up in legume-rich tree fallows would be easily leached. None of the measured P pools showed significant differences, presumably due to large amounts of P immobilisation in forest garden biomass. Forest re-growth after agricultural abandonment is commonly P-limited in strongly weathered soils and relies heavily upon mineral-associated P for decades (Tiessen 1998).

In addition to the increased cation retentiveness associated with the larger exchange complex, the more mobile nutrients including base cations and especially nitrate were probably more effectively retained by the higher degree of permanence and spatial continuity (Berish and Ewel 1988) of the forest garden root systems. By contrast, most grassland root systems in the study area are likely to be more intermittent, ephemeral, as well as spatially more patchy than forest garden root systems. Furthermore, the more continuous surface litter layer and tree roots in the gardens presumably resulted in higher rates of water infiltration and resistance against soil erosion by water.

Soil physical and biological properties

Gravimetric gravel contents in the garden topsoils were less than half of those encountered in the grasslands, which were characterised by patchy residual surface gravel accumulation, or ‘armor layers’ (Stocking and Murnaghan 2001), related to micro-topography. Müller-Dombois and Perera (1971) already observed concentrations of gravel at or close to the soil surface in grasslands of the Uva basin in the seasonally dry zone of the central highlands, which they attributed to severe erosion. About their grassland plots in the wet zone South of Kandy, they reported signs of erosion but noted the absence of gravel-rich layers at the soil surface. However, some forest garden farmers interviewed in this study pointed out that they recalled a lot of gravel at the soil surface before they started planting trees around 1980. High rates of organic matter input and horizontal and vertical mixing of the topsoil that took place since the forest garden establishment are likely causes of the observed dilution of a gravel-rich layer at the soil surface.

The high degree of within-site and between-site variability in grasslands and, particularly, in forest gardens resulted in non-significant differences in other soil physical and biological parameters measured, including soil organic matter light fraction (Anderson and Ingram 1993), WHC, soil porosity and CO_2 evolution (Table 2). However, soil respiration rate in RBL soils was, on average, more than 2.5 times as high as in RYP soils ($p<0.05$; Table 3). This could be related to a difference in the microbial community composition between the two soil types, for instance, the bacteria-to-fungus ratio as found by Orchard et al. (1992) in two silt loam soils of widely different fertility status in New Zealand. A different composition and higher activity of the microbial community in RBL soils might be promoted by the higher pH, CEC, base cation and macronutrient content (Singh and Gupta 1977), compared to RYP soils.

Conclusions

We compared topsoil properties of degraded grasslands (former tea lands) to those of forest gardens established presumably on the same type of land 20 years ago. In forest gardens, we found soil pH, cation exchange capacity, exchangeable calcium and magnesium concentrations and N content significantly higher than in degraded grasslands, exceeding the inherent differences between soil types.

In contrast to the results from the present study and previous research cited herein, tropical grasslands have also been reported to have higher soil pH, C and nutrient stocks than nearby undisturbed rainforests; hence, changes associated with land use conversion remain largely unpredictable (McGrath et al. 2001). Some of these contradictions in results may be due to substantial variation in vegetation cover, inputs of roots and litter, exports of produce, previous state of soil degradation, as well as pedology and topography. The forest gardens examined in this study were dense and diverse, presumably with high rates of organic matter inputs and low harvest indices, established on steep, eroded and acidified upland soils. Our results suggest that the restoration of exchangeable base reserves and the reversal of acidification associated with forest garden establishment exceeded the inherent differences reported for soil types of contrasting pH and chemical fertility. The lack of a significant increase in soil phosphorus and potassium was attributed to high rates of uptake to build up the large forest garden biomass. The clear accumulation of calcium, magnesium and N in the forest garden topsoil indicates that these elements are less likely to be limiting than phosphorus and possibly potassium after forest gardens are established on eroded and acidified soils on the upper slopes in the wet zone of Sri Lanka.

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