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Properties of Advanced Weathering-Stage Soils in Tropical Forests and Pastures

Jagdish Krishnaswamy* and Daniel D. Richter

ABSTRACT

Effects of tropical forest conversion to nonforest uses greatly affects low-activity soils with substantial variable charge, but quantitative changes are poorly studied. We evaluated forest conversion effects on Ultisols and Oxisols by establishing a spatial comparison study on alluvial fans and terraces in the Rio General Valley of Costa Rica. Soils from pairs of forest and pastures on similar local landforms were sampled to evaluate differences in physical and chemical properties to a depth of 120 cm. In the upper 30 cm, soils under forests had lower bulk densities than under pastures (0.81 vs. 1.04 g cm⁻³), and soil C under pasture was less than that under forest by about 15.9 ± 10.4 Mg ha⁻¹. Compared with forest soils, those under pastures had a greater soil pH_{H2O} throughout the upper 120 cm (0.44 ± 0.10 pH units). Soil effective cation-exchange capacity (ECEC) was 39 ± 19.9 kmol_c ha⁻¹ greater under pastures than under forests, a difference in ECEC that was mainly associated with an additional 19.3 ± 17.7 kmol_c ha⁻¹ of exchangeable Ca. Forest clearing and burning followed by conversion reduced soil C moderately but apparently increased the ability of mineral soils to retain exchangeable-cationic nutrients because of increased soil pH. Water-stability of aggregates was less in pastures than under forests (36.5% less for aggregates >2 mm). Reduced soil C was statistically associated with changes in the percentage of water-stable aggregates >0.25 mm ($R^2 = 0.34$, $P < 0.05$). Our understanding of the effects of land uses on advanced weathering-stage soils would benefit from monitoring studies that directly observe temporal soil changes caused by management.

VAST AREAS throughout the tropics have lost their forest cover to pastures, ~25 000 km² yr⁻¹ in recent decades. The conversion is especially prominent in Central and South America (William and Turner, 1994). Forest conversion to nonforest uses has significant effects on biogeochemical and physical properties of soils (Buol, 1994). Resultant soil erosion, loss of water-stable aggregates, loss of nutrients, and change in soil C storage, all have implications for the sustainability of land use (Pimental et al., 1995; Richter and Markewitz, 2001) and future climate change (Schlesinger, 1997).

Many areas currently under conversion are dominated by soils with variable charge. Many of these are in an advanced state of weathering as soils proceed through developmental stages (Jackson and Sherman, 1953). This advanced stage of weathering is characterized by Ultisols and Oxisols and the commonly associated soil minerals kaolinite, gibbsite, and hydrous Fe-

oxides. Such soils are found on over 8 million km² in Central and South America alone (Richter and Babbar, 1991). Variable charge affects many of the biogeochemical and hydrologic properties of advanced weathering stage soils (El-Swaify and Dangler, 1977; Sollins et al., 1988). This includes nutrient retention, ion exchange, aggregation, erodibility, drainage, aeration, and infiltration.

The total surface charge density (σ_T) and total charge (\varnothing_T) of variably charged soil systems can be represented by:

$$\sigma_T = \sigma_C + K_1(\mu T)^{1/2} \sinh[1.15z (\text{PZC} - \text{pH})],$$

$$\sigma_T \gg \sigma_C$$

$$\varnothing_T = K_2 S \sigma_T \quad [1]$$

where subscripts T and C refer to total and constant charge, K_1 and K_2 are constants, μ equals soil solution electrolyte concentration, T represents the absolute soil system temperature, z is the counter-ion valence of the dominant ion in the soil solution, PZC equals the soil pH at the point of zero charge, pH refers to the pH of the soil system, and S represents the specific surface area (modified from Uehara and Gillman, 1980; Sollins et al. 1988). Although forest conversion to pasture can potentially change T , μ , z , PZC, pH, and S , the dominant effect is on μ , pH, and PZC. Losses of soil organic matter (SOM) tend to increase PZC, but this may be countered by an increase in pH. The final effects will depend on the relative changes in these two factors.

Intra-aggregate bonds are formed and maintained by a variety of mechanisms in which functional OH groups on metal oxides and SOM are key participants (Sposito, 1989). Stability of the soil aggregates can be affected by changes in SOM and pH effects on net charge, positive or negative (El-Swaify and Dangler, 1977; Suarez et al., 1984; Chorover and Sposito, 1995). Soil organic matter also plays a role in nonelectrostatic intra-aggregate forces (Bartoli et al., 1992b). Larger aggregates are protected by roots and fungal hyphae (Oades, 1984). Thus, SOM dynamics and changes in surface charge because of pH are the most important determinants of aggregate stability in soils undergoing forest conversion in the tropics.

Advanced weathering-stage soils are well represented by many soils of the Terraba basin (the *Zona Sur*) in southern Costa Rica. Under forest cover such soils are characterized by stable aggregation, rapid drainage, and resistance to erosion (Sollins et al., 1988; Richter and

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Babbar, 1991). Forest conversion and subsequent agricultural land uses typically eliminate the protective litter layer, reduce the organic matter content of the mineral soil, and alter the soil electro-chemical environment. Any disturbance such as the conversion of permanent, deeply rooted forest cover to pasture have complex biogeochemical effects on soils and the ecosystem. However, changes in the biogeochemistry of variable charge soils are poorly documented. Current knowledge of the chemical processes of cation and anion retention, soil surface chemistry and soil aggregate stability of advanced weathering-stage soils at the landscape scale is rather rudimentary. Relatively little quantitative data are available on aggregate stability of highly weathered soils such as Oxisols and Ultisols and their response to forest and to nonforest uses (Castro and Logan, 1991; Bartoli et al., 1992a).

This paper compares chemical and physical properties of forests and pastures on acidic, advanced weathering-stage soils dominated by variable charge properties in the Rio General Valley of Costa Rica. The specific objectives were to (i) quantify the differences in soil chemical and physical properties, specifically SOM and pH, between forests and pastures; and (ii) assess the secondary effects of differences in organic matter and pH on chemical properties such as cation exchange and physical properties such as bulk density and water-stable aggregates.

Site Description

The study area is mainly located on several alluvial fans on the lower west-facing slopes of the Cordillera de Talamanca (Fig. 1) in the Terraba basin of Costa Rica. The Cordillera are composed of uplifted marine sedimentary and volcanic rocks with local intrusions of quartz diorite and granodiorite. At the foot of the Cordillera are extensive alluvial fans and terraces drained by braided networks of streams. Uplift of the mountain ranges relative to the incising streams cause the higher abandoned terraces to have the oldest soils. These alluvial fans and surfaces are up to 35 000 yr in age, based on ^{14}C of buried wood and logs (Kesel and Spicer, 1985). The soils on these terraces and fans in the Rio General Valley range from Entisols and Inceptisols on recent deposits to Alfisols, Ultisols, and Oxisols on older, more stable surfaces. These sedimentary deposits are composed of materials that have experienced several weathering cycles at higher elevations prior to their present depositional locations. Low activity kaolinite, goethite, gibbsite, and oxides of Fe and Al are common (Kesel and Spicer, 1985). The soils on older terraces belong to Suborders Udox, Ustox, Udults, and Ustults with great groups being Hapludox, Acrudox, Haplustox, and Acrustox.

The average annual rainfall in the Rio General Valley is 3139 (± 419) mm, and the most pronounced dry-season months are December to March (each month with <100 mm). The soil moisture regime is predominantly Udic and Ustic, with a 3- to 5-mo dry season. Average monthly temperature in the valley is 24.7 (± 0.96) °C.

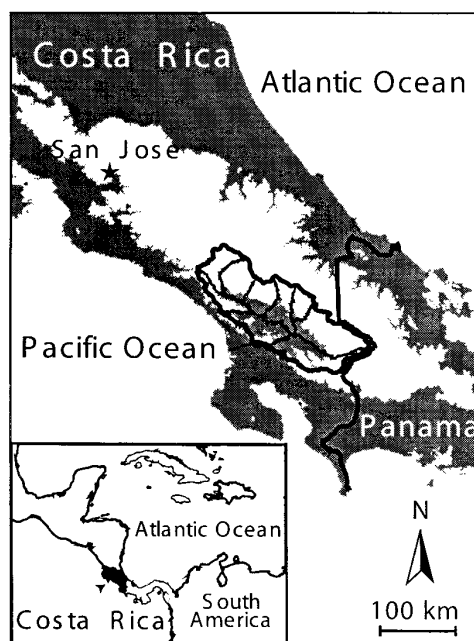


Fig. 1. Location of Rio General valley in the Terraba basin of Costa Rica.

Although the valley has been inhabited by Native Americans for millennia, since about 1950, forest cover in the Terraba basin has been reduced by >80% in the 3500 km² outside the high-elevation forest reserves. Fire-maintained savannah woodlands occur in areas with a longer dry season (Kesel and Spicer, 1985). Vegetation prior to about 1950 was tropical moist forest much of which is present only as relicts in the main valley, alluvial fans, and river terraces (Skutch, 1971). The region's dominant current land use is cattle (*Bos taurus*) grazing. Pastures now occupy about 2000 km² of the basin.

MATERIALS AND METHODS

Following a general geomorphological survey, interfluvies on the alluvial fans and terraces were identified that contained forest and pasture ecosystems. We used a soil-comparative approach that is widely used for studies of long-term ecosystem change (over decades to millennia), an approach also known as a space-for-time substitution or chronosequence. With this approach, spatially distinct systems are selected and taken to represent temporal stages of ecosystem evolution. The assumptions inherent in the method are not trivial (Pickett, 1989; Richter and Markewitz, 2001). Although the comparative approach is widely used for examining soil and ecosystem change through time, site selection is a critical step, since space and time are entirely (albeit purposefully) confounded. Sources of spatial variation must be minimal compared with effects attributable to time.

The constraints of space-for-time-substitution were considered in the field selection of blocks and plots. The five interfluvies were treated as experimental blocks and 20 by 20 m plots installed within forests and pastures of each block. Farmers may well have selected certain types of areas to clear, but we attempted to select soils on the same or similar landforms

and all sites within blocks had no visual indications that they had been different from the forest sites prior to clearing.

Forests had closed canopies and no obvious evidence of recent human disturbance. Most forest conversion to pasture took place between 1950 and 1980 based on land-use history, maps, satellite imagery, and local information (Krishnaswamy, 1999). When forests were cleared, wood was used for construction timber, furniture, and fuel, but much or most was probably burned and decomposed on site. Pastures have dense grass cover, scattered trees, and are often subject to intense cattle grazing. Chemical inputs however have been minimal.

Additional soils were also sampled in a tree plantation, two additional pastures, and three intensive agricultural sites (one in sugar-cane [*Saccharum* L.] and two in pineapple [*Ananas* P. Mill.] plantations) in the same region to enlarge the data set used to evaluate general soil properties and relationships between parameters. These sites were all from upland interfluvies, and were within a few kilometers of the other sites. The slopes of all the sites were all <10%.

Soil Sampling

Bulk density was estimated with a 60-mm slide-hammer sampler (Blake, 1983) from three replicates in each soil plot in forests and pastures. In forests and pastures, bulk-density was estimated in the 0- to 10- and 10- to 30-cm layers. Undisturbed subsamples of soil were carefully collected by breaking aggregates along planes of weakness when field moist to give clods less than about 40-mm diam. These were air-dried for subsequent aggregate analyses, and carefully packed for shipment to the soils lab at Duke University.

Samples for other chemical parameters were collected with a 110-mm diam. soil auger. Samples came from the corners of the 20-m square plot at four depths: 0 to 10, 10 to 30, 30 to 70, and 70 to 120 cm. In one plot out of the 16, several depths >30 cm were not sampled because of the presence of stones. Soil samples were air-dried and passed through a 2-mm screen prior to chemical and textural analysis. Soils were air-dried on site, carefully packed for shipment, imported to the USA on a USDA permit that allowed no entry-treatment of the samples.

Soil Analysis

Soil samples were analyzed for pH (2:1 in water and 0.01 M CaCl₂), 1 M KCl-exchangeable acidity and Al using NaF titration (Thomas, 1982). Cations (Na, K, Ca, and Mg) were extracted with 1 M ammonium acetate buffered at pH 7 and analyzed by atomic absorption spectrophotometry. Effective cation-exchange capacity was estimated from the sum of extracted base cations and KCl-acidity, and base saturation (BS) from the quotient of the sum of exchangeable base cations and ECEC. A subset of samples were analyzed for cation-exchange capacity (CEC) at pH 8.2 using a BaCl₂ extraction and triethanolamine buffer (Mehlich, 1938; Thomas, 1982). Total C and N were determined using Dumas dry combustion (Perkin Elmer CHN/S analyzer, Perkin Elmer, Norwalk, CT) on pulverized samples. Texture was determined using the pipette-gravimetric method.

Throughout the chemical analyses at least 10% of the samples were randomly replicated for quality monitoring. The differences between replicates was usually <5% (>90% of samples) and was never >10%.

Three clayey (>50%) samples from 70 to 120 cm with lowest ECEC (<0.5 cmol_c kg⁻¹) were chosen for mineralogical analyses. The clay fraction (<2 mm) was obtained by Calgon dispersion and was treated with Mg acetate before oriented X-ray

diffraction (Whittig, 1983). The diffractograms (intensity vs. 2θ) were plotted to detect peaks.

Air-dried samples for aggregate analyses were carefully sieved for 30 s using a 5-mm sieve to collect aggregates retained on a 1-mm sieve positioned below the 5-mm sieve. Aggregate stability was measured using a modified single-sieve method (Kemper and Rosenau, 1986; Beare and Bruce, 1993) with 1-h shaking at 39 strokes per min in water. Sieve sizes were 0.25, 1, and 2 mm. Sand, roots, and stones remaining on the sieve after shaking were determined after NaOH (0.05 M) treatment to disperse aggregates. Two replicates were performed for each sieve size. Tests with larger aggregates (>1 and >2 mm) were performed on aggregate samples pooled across all five blocks, whereas the 1- to 5-mm aggregates were tested using the 0.25-mm sieves separately for each block. In addition, about 50 g of air-dry aggregates from the pooled sample were used for estimating the distribution of aggregate sizes by using a nest of sieves (5, 4, 2.8, 2, and 1 mm) and gentle shaking for 10 s.

Contents and changes in contents of soil constituents for pastures were estimated using the equal mass method to correct for bulk density changes following compaction (Veldkamp, 1994).

Box and whisker plots displaying lower and upper quartiles and median, were utilized to compare land uses. This method has the advantage of estimating central tendencies better when the number of replications is small (≤5), and in addition gives an idea of the spread about the central tendency. Wherever appropriate, 95% confidence intervals about the median were computed. These approximate intervals are a function of the quartiles and are little affected by outliers (McGill et al., 1978). This was supplemented with ANOVA (Kleinbaum et al., 1988). Contrasts between land uses were analyzed by the LSD multiple comparison technique. The results were compared with other techniques (Duncan, Tukey and Waller) to ensure that results were not sensitive to choice of method. It is to be noted that whereas the ANOVA accounts for block differences if present while detecting land-use treatment effects, the box and whisker plots indicates relative effects of land-use across the landscape. Unless reported otherwise, all estimates given are medians with 95% confidence intervals if they are positive.

RESULTS AND DISCUSSION

Forest Soil properties

The undisturbed soils under forest on the terraces and fans are generally acidic (Table 1), and in an advanced stage of weathering (Kesel and Spicer, 1985). The median percentage of clay ranges from 46% in superficial layers to over 58% at 120 cm. The median ECEC is very low (0.86 cmol_c kg⁻¹ at 70–120 cm) and pH_{H2O}

Table 1. Soil properties† under forest in the Rio General Valley, Costa Rica.

Depth	pH _{CaCl2} ‡	pH _{H2O} §	ECEC¶	BS#	Clay	C
cm	cmol _c kg ⁻¹			%		
0–10	3.86	4.35	4.95	43	46	5.09
10–30	4.03	4.34	2.72	12	53	2.63
30–70	4.06	4.46	1.58	13	56	1.19
70–120	4.20	4.65	0.86	21	58.5	0.74

† Values reported are median value from five replicate blocks.

‡ 2:1 in 0.05 mol L⁻¹ CaCl₂.

§ 2:1 in H₂O.

¶ Effective cation-exchange capacity.

Base saturation.

exceeded $\text{pH}_{\text{CaCl}_2}$ in all of the samples. Base saturation of ECEC decreases with depth but is only 43% in the upper 10 cm, the exchange complex being dominated by exchangeable acidity, mainly Al.

Mineralogical analyses of <2-mm fraction from the deeper soil layers (70–120 cm) of the alluvial fans confirmed the presence of kaolinite, gibbsite, and goethite. Peaks were detected at and near 12.35 and 20.3, and 24.9 for kaolinite, 18.3 and 20.3 for gibbsite and 21.2 for goethite. The pH increased with depth, indicating the commonly observed tendency to attain values closer to the PZC (Sposito, 1989) of a mixture of kaolinitic-oxidic minerals in the absence of organic matter, a thermodynamically more stable state.

The contribution of clay minerals to ECEC is remarkably small in these soils, $1.48 \text{ cmol}_c \text{ kg}^{-1} \text{ clay}$ at 70 to 120 cm, another indication of the advanced state of weathering. The mineralogical and chemical properties described above confirms to the so-called *mineralogical monotony* (i.e., the domination of Fe oxides and kaolinite) of many Ultisols and Oxisols (Schwertmann and Herbillon, 1992).

The soils are well aggregated and friable. Bulk density of soils under forest (Fig. 2) were 0.72 ± 0.148 (0–10 cm) and 0.98 ± 0.113 (10–30 cm). Water-stable aggregates >0.25 mm were 94.5% in the upper 30 cm.

The forest soils are rich in C, storing $79.5 \pm 11.69 \text{ Mg ha}^{-1}$ in the upper 30 cm, and concentrations ranging from 5.1% at 0 to 10 cm to 0.74% at 70 to 120 cm (Fig. 3). This reflects the protective role of oxide coatings in soil aggregates that inhibit decomposition of organic matter.

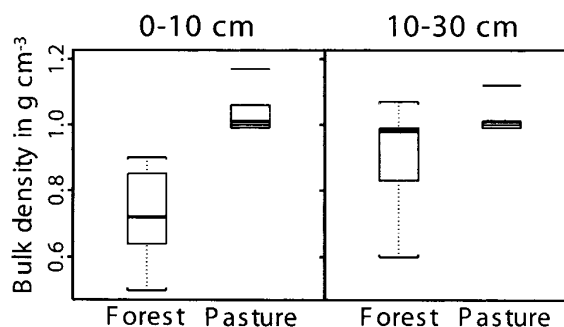


Fig. 2. Soil bulk density under forests and pastures. Box and whiskers plot with median, lower and upper quartiles, and outliers.

Forest Versus Pasture Comparisons

Texture and Bulk Density

A decrease in clay content because of selective erosion is one of the changes known to result from forest conversion (Lal, 1977; Fearnside and Barbosa, 1998). However, it should be noted that the traditional textural analysis of soils of the humid tropics is known to be affected by the aggregation of clay particles that resists clay dispersion (Igwe et al., 1995; Galvao and Schulze, 1996; Kauffman et al., 1998). This may mask the detection of relative clay depletion if the forest soil particles are more strongly aggregated. There is no evidence of any selective erosion of clay or other textural changes after forest conversion. In fact, soil texture in the upper

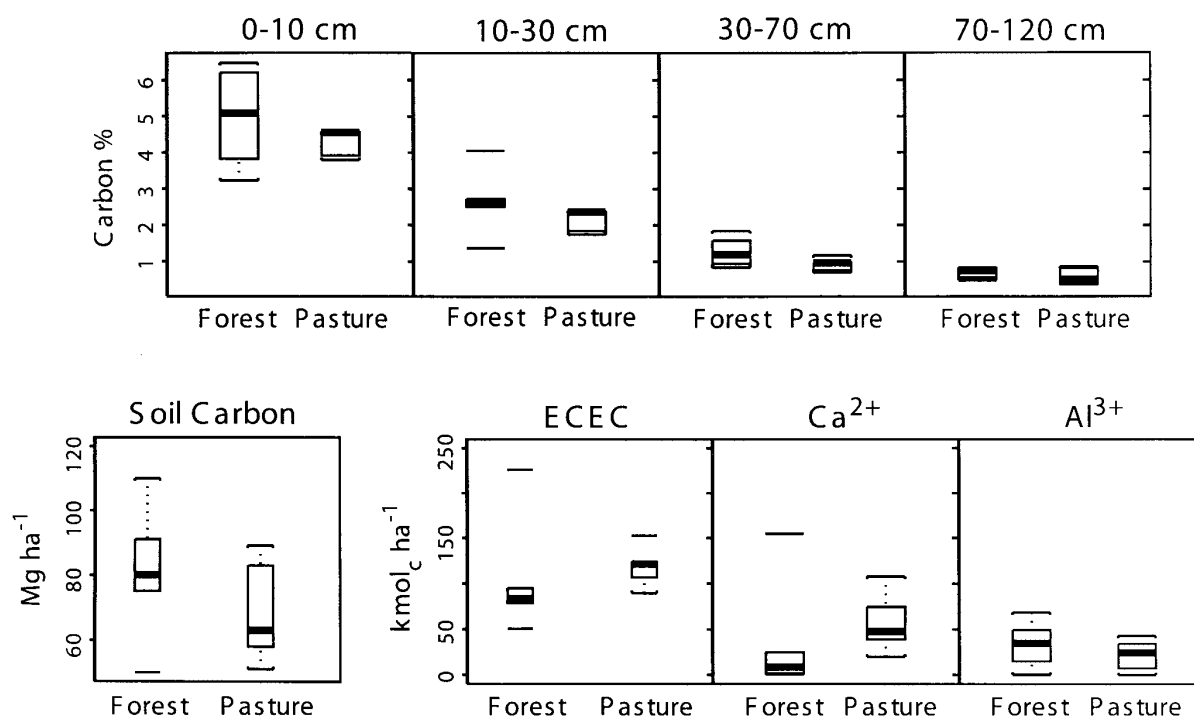


Fig. 3. Soil C concentration in forests and pastures in the upper 120 cm and contents of C and exchangeable cations under forests and pastures in the upper 30 cm. Box and whiskers plot with median, lower and upper quartiles, and outliers.

30 cm is remarkably similar for forests (39% clay at 0–10 cm, 52% at 10–30 cm) and pastures (39% clay at 0–10 cm and 53% at 10–30 cm), with means being statistically ($P < 0.1$) indistinguishable. This is also evidence in support of our view that the differences in inherited properties are not large.

The bulk density of pasture soils was higher than those of forest soils (Fig. 2). Conversion to pasture has apparently increased bulk densities from 0.72 ± 0.148 to 1.01 ± 0.042 g cm⁻³ in the upper 10 cm and from 0.98 ± 0.113 to 1.01 ± 0.007 g cm⁻³ for the 10- to 30-cm depth. Means were significantly different ($P < 0.05$) for the 0- to 10-cm depth. The greater variability of bulk density in pasture soils than under forest (Fig. 2) may be associated with variable ages of pastures and cumulative compaction because of cattle. Despite the change, the overall bulk density in pastures (1.0 g cm⁻³) is still relatively low, and assuming a particle density of 2.5 g cm⁻³, the estimated pore space would be ~60%. Thus, in the event of management practices destroying the aggrega-

tion drastically such as in cultivation, there would appear to be great potential for pore space to collapse.

Organic Carbon and pH

The concentration of soil C was numerically greater under forest than pasture throughout the upper 120 cm (Fig. 3). The median concentration difference ranged from 10.6% at 0 to 10 cm to 30.3% at 70 to 120 cm. However, means were not significantly different except at 30 to 70 cm ($P < 0.1$). Median C contents under forests and pastures in the upper 30 cm were 79.5 ± 11.7 and 63.5 ± 17.9 Mg ha⁻¹, respectively. Overall, there seems to be a small to moderate loss of soil C and this is attributed to the inputs from grass roots, manure, and fires in the pastures.

Soil pH was lower under forest than under pasture throughout the upper 120 cm (Fig. 4). Means were significantly different at 10 to 30 and 30 to 70 cm ($P < 0.1$), and at 70 to 120 cm. ($P < 0.05$). The conversion from

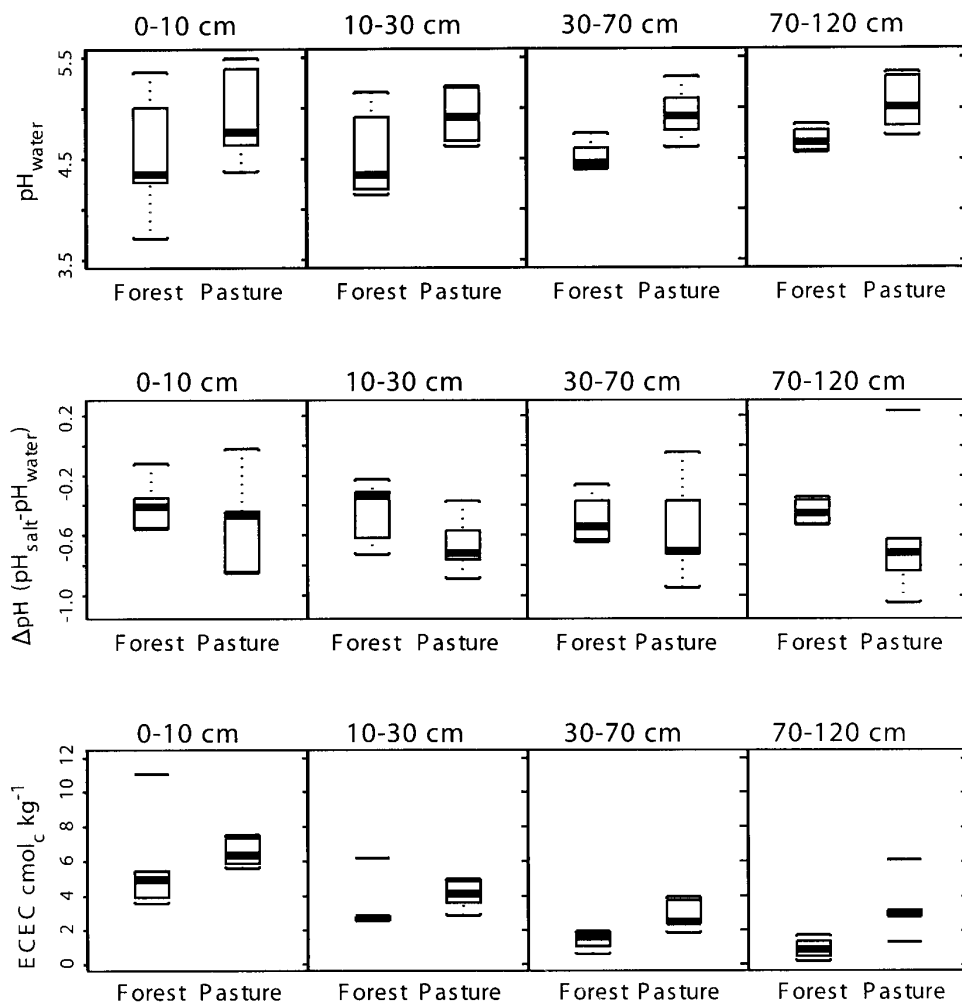


Fig. 4. Soil pH, $\Delta pH (pH_{salt} - pH_{H_2O})$, and effective cation-exchange capacity under forests and pastures. Box and whiskers plot with median, lower and upper quartiles, and outliers.

forest to pasture has apparently elevated soil $\text{pH}_{\text{H}_2\text{O}}$ by 0.44 ± 0.102 pH units in the upper 120 cm. Effective BS was correlated with pH, based on the larger data pooled across all depths ($R^2 = 0.60$, $P < 0.0001$, $n = 60$). Median base saturation on a content basis (30 cm) was 73.9% in pastures and 22.8% in forests. However, means were not statistically different although effective BS tended to be higher in pastures at all depths.

Elevated pH can persist in unlimed tropical pastures after conversion for long periods (decades) depending on the management regime (Ghuman and Lal, 1991; Veldkamp, 1994; Dam et al., 1997). In addition to the major initial liming effect of the incorporation of burnt forest biomass, the return of base-rich animal manure and ash inputs from frequent fires in pastures may be maintaining higher pH levels and BS in converted pastures. Cattle manure inputs from grazing cattle have been noted to enhance Ca^{2+} , pH, and thus ECEC in weathered soils (Pichot, 1971; Asio et al., 1998).

Soil Cation-Exchange Capacity

Differences in ECEC between soils under forests and pastures are evident throughout the upper 120 cm (Fig. 3 and 4). Mean ECEC of pastures were significantly higher ($P < 0.05$) than those in forests at 30- to 70-cm and 70- to 120-cm depths. The difference in ECEC in soils of pastures from those of forests is an estimated 1.41 ± 0.102 $\text{cmol}_c \text{ kg}^{-1}$ in the upper 120 cm, an increase of 34%.

Another index of net charge apart from ECEC is ΔpH ($\text{pH}_{\text{CaCl}_2} - \text{pH}_{\text{H}_2\text{O}}$) (Mendonca and Rowell, 1996). In general, ECEC was negatively correlated with this index of effective charge, at depth >30 cm ($R^2 = 0.23$, $n = 28$, $P < 0.02$). The trends in ΔpH with land-use are similar to that for ECEC (Fig. 4) which hints at a real change in soil charge.

The increase in cation-storage capacity apparently because of forest conversion to pasture is estimated to be 39.0 ± 19.9 $\text{kmol}_c \text{ ha}^{-1}$ in the upper 30 cm (Fig. 3). Although the ANOVA results indicated no significant differences between pasture and forest, this was attributed to the effects of one outlier block (Fig. 3). The median cation-storage capacity without considering this block are 114.3 ± 19.32 $\text{kmol}_c \text{ ha}^{-1}$ in pastures and 81.2 ± 19.6 $\text{kmol}_c \text{ ha}^{-1}$ in forests, an increment of ~ 33 $\text{kmol}_c \text{ ha}^{-1}$. Mean ECEC in pastures (110 $\text{kmol}_c \text{ ha}^{-1}$) was also significantly different ($P < 0.05$) from forests (77 $\text{kmol}_c \text{ ha}^{-1}$) when this block was omitted.

Organic Matter, pH, and Effective Cation-Exchange Capacity

A regression model of ECEC as response with the only the interaction of pH and C as a covariate was significant ($\text{ECEC} = 1.30 + 0.18 (\text{pH}_{\text{H}_2\text{O}} \times \text{C})$, $R^2 = 0.40$, $P < 0.0001$). The simulated effect of pH on ECEC was analyzed graphically. The pK_a of carboxylic groups in organic matter is ~ 4 (Thomas and Hargrove, 1984). The data were split into two groups based on CaCl_2 pH: one pH 4 and below, the other above pH 4.0. Above pH 4, increases in soil C are associated with increases in

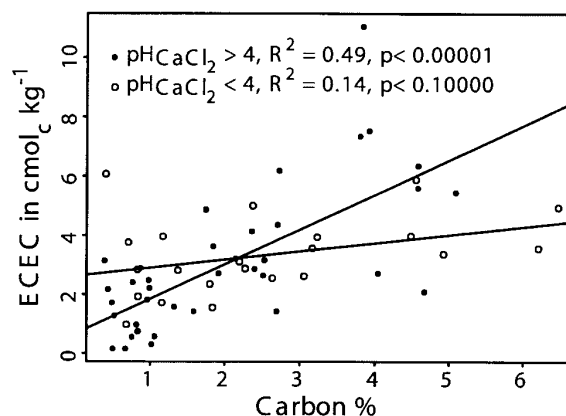


Fig. 5. Effective cation exchange capacity (ECEC) contribution of soil organic matter for soil samples with measured pH (2:1 in CaCl_2) below and above four.

ECEC, whereas below, there is a weak or no response (Fig. 5). At elevated pH deprotonation of organic functional groups is enhanced. Above a soil $\text{pH}_{\text{CaCl}_2}$ of about 4, the contribution of SOM to pH-dependent variable charge is well expressed (Fig. 5).

Cation-Exchange Capacity at pH 8.2 and Soil Organic Matter

The role of SOM as a source of potential negative charge is further illustrated (Fig. 6) by the relationship of CEC at pH 8.2 and C. At pH 8.2 and high concentration of extractant, all the inorganic mineral soil hydroxyls and organic carboxyl functional groups would be deprotonated. The regression model however indicates that the contribution of mineral components is not substantial and the major source of variable charge is SOM. The regression model with soil C explains 97% of the variability in CEC at pH 8.2. Based on the regression,

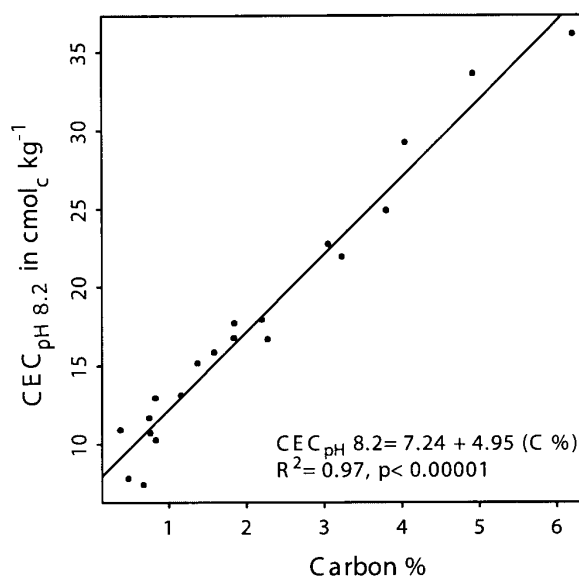


Fig. 6. Cation-exchange capacity at pH 8.2 as a function of soil C.

the contribution of organic matter to soil CEC in these soils is estimated to be about $248 \text{ cmol}_c \text{ kg}^{-1}$ of organic matter, which falls in the typical range for humus.

Effective Charge: Organic Matter and Mineral Components

Based on intercepts in the regression models presented, it is estimated that only 10 to 30% of the potential charge can be attributed to mineral components and the bulk (>70%) is dependant on the SOM and its response to pH.

Effects of Land Use on Effective Charge

Changes in ECEC because of land-use in variable charge soils have been estimated in varied ecosystems (Alegre et al., 1988; Ghuman and Lal, 1991; Richter et al., 1994; Koutika et al., 1997; Menzies and Gillman, 1997). The presence of low-activity soil minerals with low constant charge makes the ECEC of the General Valley soils strongly linked to SOM and its response to change. The results presented indicate that although there is a small to moderate reduction in organic matter, soils under pastures have higher negative charge than forests. In addition, the pH in pasture soils is higher than that in forests. The regression models presented earlier clearly demonstrate the role of enhanced pH on variable charge derived mainly from SOM. The apparent increase in pH after conversion suggests that pH-dependent charge is responsible for the increase in ECEC.

To test whether this mechanism (effects of pH on variable charge) could explain the higher ECEC in pasture soils as compared with forests, ECEC was plotted against the percentage of C separately for forests and pastures (Fig. 7). The ECEC–soil C relationship appears to be shifted upwards for pastures. Intercepts were significantly different at $P < 0.05$. This change is primarily attributed to the pH effects on SOM acid functional groups in addition to enhancing the charge of mineral components. In pastures, the impact of moderate reduction in SOM on ECEC (lesser number of potential ex-

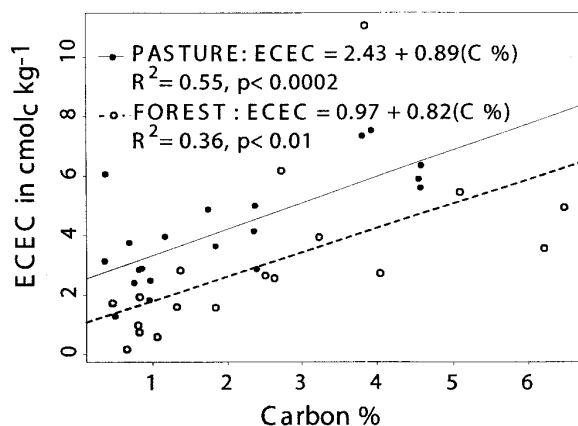


Fig. 7. Comparisons of forests and pastures with respect to the relationship between effective cation-exchange capacity and soil C.

change sites) has been more than compensated by an elevation in pH, which enhances deprotonation of the acidic functional hydroxyls and the inorganic mineral hydroxyls. This mechanism has been proposed for highly weathered soils elsewhere (Mendonca and Rowell, 1996).

The trends with depth and land-use for ΔpH are somewhat similar to the trends in ECEC discussed above (Fig. 4). This suggests that forest conversion to pasture has changed the nature and magnitude of the exchange complex with an increase in negative charge and a decrease in positive charge.

A part of the differences in ECEC, especially at >70-cm depth, could be attributed to differences in inherited properties between pastures and forests, since certain types of areas may have been cleared preferentially. However, there is no evidence that this factor played a major role in the current study. Ionic strength differences because of land-use change could also affect effective charge, however the methods used in this study cannot explicitly account for this. We however attribute the major part of the observed differences in measures of effective charge (ECEC and ΔpH) to effects of land-use change and subsequent management on variable charge properties especially in the upper 30 cm.

Exchangeable Cations

Additional ECEC sites at the elevated pH under pastures have retained an increment of $19.3 \pm 17.68 \text{ kmol}_c \text{ ha}^{-1}$ exchangeable Ca (see also Fig. 3). Pasture soils also tended to have higher levels of exchangeable K ($10.82 \text{ kmol}_c \text{ ha}^{-1}$) than forests ($3.33 \text{ kmol}_c \text{ ha}^{-1}$). If one block was left out, soils under pasture had significantly higher overall mean levels compared with pastures ($P < 0.1$). Pasture tended to have less exchangeable Al than forests in the upper 30 cm (Fig. 3). In general, nutrient cations have replaced some of the exchangeable Al and acidity on cation-exchange sites. Calcium saturation of exchange sites on a content basis increased from 23 to 47% ($P < 0.06$) in the upper 30 cm.

The generation of additional ECEC in pastures has retained Ca from the burnt forest biomass and subsequent grass-fire dynamics. A compilation of 20 studies of humid tropical forests world-wide (Ewel et al., 1981; Alegre et al., 1988; Nykvist, 1998) indicates that Ca^{2+} in tropical forest biomass and burnt ash is $62.6 \pm 46.85_{\text{sd}} \text{ kmol}_c \text{ ha}^{-1}$. Thus, the difference in exchangeable ECEC ($39 \text{ kmol}_c \text{ ha}^{-1}$) in forest and pastures would potentially

Table 2. Size distribution† of air-dried aggregates in forests and pastures.

Sieve size mm	0–10 cm		10–30 cm	
	Forest	Pasture	Forest	Pasture
	%			
5	2	2	3	2
4	9	11	10	10
2.8	28	28	28	28
2	30	29	32	29
1	31	30	27	31

† 50 g of aggregates were selected for each land-use pooled across five replicate blocks and sieved using a nest of sieves.

be able to retain a substantial part of the Ca from forest biomass alone.

Although nutrient cations associated with ECEC may have been derived from the ash of forest biomass, and subsequent cattle manure inputs, there are other potential sources as well. Nutrient retention may well be efficient in the grass pastures, and nutrients taken up by roots at depths >1 m in the soil have often been underestimated (Markewitz and Davidson, 1997). These inputs of nutrient cations would be retained by extensive grass rooting and enhanced ECEC of near surface soils.

Aggregate Size and Stability

The size distribution of the air-dry aggregates (1–5 mm) is generally similar for pastures and forests throughout the 0- to 30-cm soil depth (Table 2). Nearly 90% by weight of the 1- to 5-mm aggregates have effective diameters of 1 to 3 mm under pastures and forests. Any observed land-use differences in water stability of aggregates cannot be attributed to initial differences in aggregate size or soil texture.

Water-stable aggregates >0.25 mm are greater under forest than pasture (Fig. 8). Conversion to pasture has decreased water-stable aggregates from 93 ± 2.82 to $90 \pm 7.04\%$ in the upper 10 cm and from 96 ± 4 to $77 \pm 5.63\%$ for the 10- to 30-cm depth. Arithmetic means were significantly different ($P < 0.1$) for the 0- to 10-cm depth. The variability of stability in the pasture

aggregates is also much higher than those of forest soil aggregates in the surface 0 to 10 cm (Fig. 8) as observed for bulk density (Fig. 2).

The relative roles of SOM and net charge (ΔpH , $\text{pH}_{\text{CaCl}_2} - \text{pH}_{\text{H}_2\text{O}}$) was investigated in explaining the observed variability in water-stable aggregates >0.25 mm. Of these two, soil C emerged as a stronger covariate to aggregate stability (Fig. 8). A nonlinear fit explained 34% of the variability in percentage of water-stable aggregates >0.25 mm ($P < 0.05$), whereas only 19% was attributed to ΔpH ($P < 0.06$).

In the aggregate-size range 1 to 5 mm, land-use differences between forests and pastures are detectable for all three sieve sizes: 0.25, 1, and 2 mm (Fig. 9). This is significant because air-drying may suppress land-management effects on aggregate stability of highly weath-

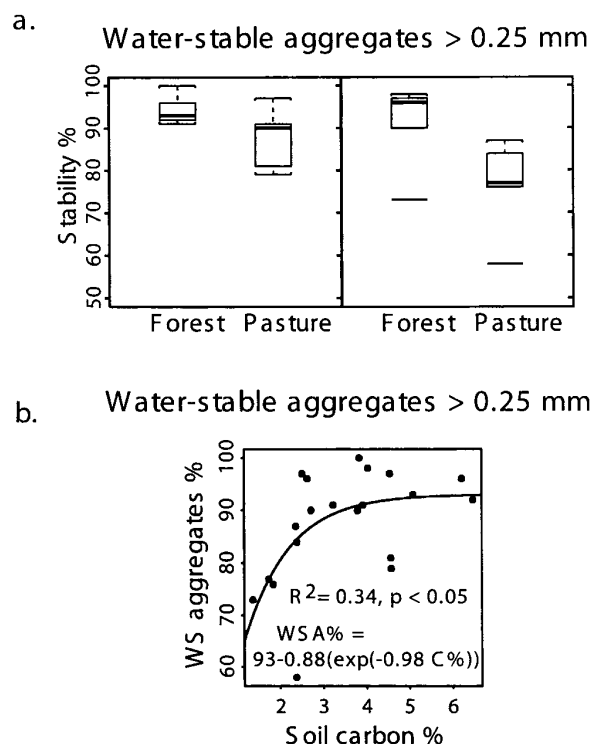


Fig. 8. Soil water-stable aggregates and land use: (a) Water-stable aggregates >0.25 mm under forests and pastures. Box and whiskers plot with median, lower and upper quartiles, and outliers and (b) water-stable aggregates >0.25 mm as a function of soil C.

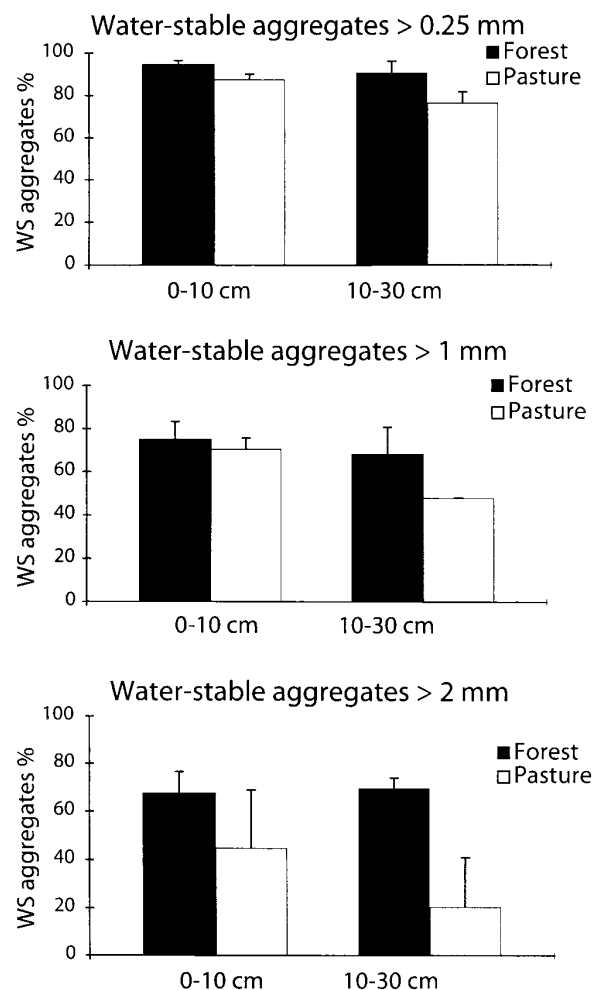


Fig. 9. Water-stable aggregates >0.25, 1, and 2 mm in forest and pasture soils. Means of five blocks (two laboratory replicates each) were used for the >0.25-mm plot, whereas means for the >1- and >2-mm aggregates were derived from a pooled sample across all five blocks. The error bar for >0.25-mm aggregates is based on five blocks, whereas the standard deviation for water-stable aggregates >1- and 2-mm are based on the two laboratory replicates.

ered soils (Kemper and Rosenau, 1986; Beare and Bruce, 1993). Larger soil aggregates (e.g., >2 mm) appeared to be much less water stable under pastures than under forest (Fig. 9).

Aggregate stability of larger aggregates is significantly less than that of smaller size aggregates (Fig. 9). The overall percentage of water-stable aggregates >0.25 mm in 0 to 30 cm of forest soils (94.5_{median}%) indicates high stability under undisturbed conditions. Thus, the most water-stable aggregate size class is close to 0.25 mm. Hypothetically, this indicates that smaller aggregates (0.1–1 mm) are stabilized by processes which are an expression of the weathering history and protective role of Fe and Al oxides. On the other hand, larger aggregates (1–5 mm) may be stabilized by organic matter and related factors. Although aggregate stability of cultivated soils was not measured in this study, the large losses of organic matter as inferred from comparisons with adjacent forest (from 6.21–2.52% at 0–10 cm and from 2.63–2.27% at 10–30 cm) would be expected to destabilize soil aggregates even more than in pastures. Long-term studies of the stability response of aggregates to soil management regimes are a critical need for evaluating management sustainability.

CONCLUSION

The conversion of forest to pastures and subsequent management have significantly changed soil properties in the Terraba basin in Costa Rica up to a depth of 120 cm. Soil pH, ECEC, and exchangeable cations have increased with small decreases in C. Aluminum and acidity have been replaced by Ca²⁺ as the dominant cation on exchange sites in surface soils in pastures up to 30-cm depth. The changes are related to the inputs of cations following incorporation of burnt forest biomass, fire, cattle grazing, and changes in variable-charge properties of these soils. This study illustrates that actual local impact of forest conversion on soil properties is dependent on how the subsequent management regime affects the charge properties of the remaining organic and inorganic constituents. Our results also indicate the potential of these soils to retain soil fertility after forest clearing if management practices address the maintenance of organic matter. The increased sensitivity of cleared soil to changes in aggregate stability emphasize that vegetative cover and organic matter inputs are critically important in managing variable-charge soils.

This study, like many other ecological studies, has been based on the space-for-time substitution or chronosequence approach (Pickett, 1989). Results from such studies need to be cautiously interpreted. Although they produce useful knowledge and insight, they are no substitute for direct observations of ecosystems undergoing change (Richter and Markewitz, 2001). Ecological scientists need an efficient network of soil-ecosystem experiments that will quantify soil and ecosystem change over time scales of decades. A global network of efficiently run field experiments is urgently needed across a range of soil textures, clay mineralogies, plant species, climates, and management regimes.

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