8 Criteria and Indicators of Forest Soils used for Slash-and-Burn Agriculture and Alternative Land Uses in Indonesia

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FUNCTIONS OF FOREST SOILS

What are forest soils? Do they differ from agricultural soils? If so, what about agroforestry soils and the soils that are part of slash-and-burn agricultural systems involving a forest fallow? A substantial part of all land currently used for intensive agriculture, in the temperate zone as well as in the tropics, was once a forest soil and was either used for slash and burn agricultural systems (Steenberg, 1993), or was at least cleared from its forest cover by a slash-and-burn method. A substantial part of current forests has at some point in time been cleared for agriculture, and was allowed to revert to natural vegetation. Subsequent agricultural use of former forest soils is partly based on the inheritance of readily available nutrients and easily decomposable soil organic matter (SOM) pools, but also

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on the soil structure with its network of old tree root channels and on the soil biota in as far as they survived the forest clearing operations (Kooistra & Van Noordwijk, 1996). As long as this inheritance lasts, the agricultural soils will maintain a partial forest soil character. Charcoal formed from roots in the topsoil may stay in the exforest soil for a long time; surface-produced charcoal may wash down slope and be incorporated in sedimentation zones.

Forest soils are not fundamentally different from nonforest soils, but the concerns about long-term changes in forest soils may differ from those under more intensive agricultural management. Changes in soil quality can be considered from at least four perspectives:

1. Effects on the sustainability of current land use practices.
2. Direct effects on other environmental compartments via transfers of nutrients, sediment, water and gases.
3. Indirect effects on land use decisions elsewhere if current land use is nonsustainable.
4. Change in options of conversion to other future land uses.

Category 1 is probably the most obvious one, but Categories 2 to 4 also play a role in the debates on forest and forest-derived soils. Nonsustainability of current land use may be due to effects on the abiotic and/or biotic resource base for future production. It may involve, for example, changes in soil physical properties affecting the water balance, changes in the nutrient balance and soil organic matter content affecting plant nutrition, or changes in essential soil biota.

Examples of Category 2 are the much debated functions of forest soils in watershed protection and regulation of stream flows, and the more recently discovered function of forest soils in oxidizing atmospheric CH_4 and thus offsetting the emissions of (nearby) sources of this greenhouse gas.

An example of Category 3 is the concern about unsustainable forms of slash-and-burn agriculture that may lead to a continued hunger for new forest lands (Brady, 1996). Environmental arguments in favor of promoting alternatives to slash and burn are largely based on the values of the forests before conversion and thus on the deflection value of land use practices (Category 3). Deflection is defined as the area of forest **not** converted due to a certain intensified land use practice—it is based on the inverse of the number of people productively engaged (or employed) and satisfying their livelihood needs per ha of a land use system (averaged across the full production cycle). The difference between this value and that for a long-fallow rotation slash and burn system under local circumstances (say 25 ha per person, for 1 ha cultivated per year with a 50 yr fallow rotation and two persons per family) is the deflection value of the intensified land use practice. The counterpoint to the deflection argument, the likelihood that productive and sustainable alternatives to slash and burn may accelerate rather than slow down forest conversion is now discussed as the Pandora's box issue. Only a combination of intensified, sustainable production systems and adequate protection of the edges of the remaining forests will have the desired effect. A major alternative to slash-and-burn agriculture was developed in the beginning of the 20th century in Sumatra (Indonesia) in the form of jungle rubber systems, or rubber agroforests (Gouyon et al., 1993). Rubber trees were initially planted as an enriched fallow species to increase the value of the forest vegetation regrowing after an upland rice (*Oryza sativa* L.) crop. The result was a sustainable and productive land use system, where the value of the rubber rapidly became the dominant part of the system and the first year's food crops an aspect of secondary importance. The productivity of the system, however, attracted an inflow of people to the area and, in combination with logging operations that provide access and large-scale commercial plantations, led to a situation where little natural forest is left on the lowland peneplains of Sumatra (Tomich & Van Noordwijk, 1996; Van Noordwijk et al., 1995). The overali impact of alternative land use systems is thus based on a combination of the deflection value, the effectiveness of forest protection and the likelihood of migration, which in itself depends on the labor market in other sectors of the economy.

Category 4 is more difficult to assess, but comes into play in discussions about soil pollution: even if it is not detrimental for current land use, or leads to leakage to other environmental compartments, it may reduce future options. On agricultural lands, for example, accumulation of heavy metals such as Cu due to the use of large amounts of pig slurry may make the land unsuitable for grazing by sheep at any point in the near future or for growing vegetable crops. The use of persistent pesticides or pesticides that affect particularly sensitive soil organisms, may reduce the options for future land use conversions. If we restrict the discussion to situations where forest soils will remain under forest cover, we may still consider natural forest, some form of managed logging enterprise or plantation forest. Logging practices may affect the future value of the soil for natural forest in a different way from the value for tree plantations.

CRITERIA AND INDICATORS

The Santiago declaration for conservation and maintenance of soil and water resources (Ramakrishna & Davidson, 1998, this publication) lists seven indicators. For a number of indicators such as erosion, loss of soil organic matter, and soil compaction (a, d, e) it is not directly clear how to evaluate them, but they probably fall under Category 1 and relate to future on-site soil productivity. The other indicators (b, c, f, and g) involve elements of Category 2. None of the indicators involves Categories 3 and 4 (unless soil erosion is seen from the perspective of reducing future conversion options). This may not be surprising, as Category 3 is based on human choices at the landscape level and beyond, and thus is hard to attribute to a particular plot. Category 4 also is more difficult to evaluate, as it concerns options rather than realities.

All the indicators refer to changes from historical values rather than absolute standards. This is understandable from the wide variation in any indicator values in undisturbed forest systems across the globe, but it makes is difficult to evaluate the indicators for any particular site, as reliable historic data are an exception rather than rule. The best we may expect to do is have parameter ranges from historical or presently undisturbed sites as reference and note the gross deviations.

In this contribution to the debate on such indicators we draw on the initial results of the Alternatives to Slash and Burn project in Indonesia for a review of measurable indicators and historic data. Criteria for evaluating the impacts of land use on (former) forest soils (Table 8–1) can be grouped by soil function, focusing on the sustainability of land use practices (Category 1), and on externalities or effects on environmental functions of forest soils (Category 2). The measurables for these various functions, however, show a considerable degree of overlap. Many of them are linked with the maintenance of surface mulch and soil organic matter.

Indicators can be discussed at two levels: easily observable phenomena that can be used in rapid assessments, but which are quick and dirty, and real measurables, for which standardized protocols can be made and interpretation schemes for the values obtained. Qualitative field level indicators may be sufficient for monitoring on-site changes by (forest) farmers or other land users. To them the presence of a surface litter layer and clear forest streams may be enough to evaluate the system they work with. Yet, such simple indicators are not sufficient for legally binding commitments that may lead to law-suits. The latter require rigorous laboratory procedures. Even with such procedures, the interpretation of data may not be unequivocal as absolute reference values are lacking for many of the parameters. For example, a debate on how often landslides occur in natural forest landscapes can cast doubt on any data on sediment loads of rivers after forest conversion.

In the rest of this chapter, we will focus on two proposed indicators of the soil organic matter (SOM) content, as this relates to both on-site and off-site functions of forest soils. The SOM content of exforest soils is of crucial importance to the productivity of subsequent crops (Nye & Greenland, 1960; Palm et al., 1996). We will discuss the C-saturation deficit and C_{org} fractionation as two approaches to make changes in soil organic matter more readily quantifiable. For both of these indicators we will give a scientific rationale, a suggested protocol for assessment and review the data available for interpreting results.

CARBON SATURATION DEFICIT

Rationale

Forest soils may lose a considerable part of their soil organic matter content after changes in management or conversion to other land use. Yet, the variation in soil organic matter content between different sites and soils is so large, that it is not easy to find a proper point of reference, to judge whether specific values are lower than would be expected under undisturbed forest conditions. On the basis of Hassink and Whitmore (1997), we propose to use a dimensionless C saturation deficit , C_{sat}, as the difference between the current C_{org} content and a reference content, $C_{org,\,ref}$, which is supposed to indicate saturation of the C protection capacity as approximated in an undisturbed forest condition.

$$C_{sat} = (C_{org,\,ref} - C_{org}) \,/\, C_{org,\,ref} = \; - (\, C_{org} \,/\, C_{org,\,ref}\,) \qquad [1]$$

Table 8–1. Criteria and indicators for evaluating the impacts of land use on (previous) forest soils in the Alternatives to Slash-and-Burn project.

Soil functions/criteria	Indicators (field level—qualitative)	Measurables (quantitative)
	Maintain on-site productivity	
Maintaining soil as a matrix of reasonable structure (soil capital)	Erosion: absence of gullies, presence of riparian filter strips and other sedimentation zones, soil cover by surface litter or understory vegetation Compaction: pocket penetrometer Structure: 'spade test', root pattern	Net soil loss = internal soil loss—internal sedimentation Percentage soil cover, integrated over the year (or over annual rainfall) Bulk density of topsoil Soil macroporosity and H_2O infiltration rates
Water balance: buffering water between supply as precipitation and demand for transpiration	Soil cover and absence of gullies as indicator or infiltration	Water infiltration versus run-off Soil water retention Effective rooting depth
Nutrient balance: buffering nutrients between supply from inside and outside the system and demands for uptake (soil nutrient capital)	Annual exports of P and cations as fraction of total and 'available' stock Financial value of net nutrient exports as fraction of potential replacement costs in fertilizer	Changes in stocks of plant available nutrients Changes in mineralization potential or size of organic matter pools C-saturation deficit (see text; Eq. [1]) Limiting-nutrient trials
Maintaining essential soil biota, such as mycorrhizal fungi and *Rhizobium* (soil biological capital)	Sporocarps (mushrooms) for ectomycorrhizal species Signs of 'ecosystem engineers' among the soil fauna: earthworms, termites	Spore counts for V.A. mycorrhiza Mycorrhizal infection and nodulation in roots in the field and in 'trap crops' in the lab
Tolerable levels of pests and diseases	Weed flora Pest outbreaks, pesticide use	Soil seed bank of weedy species Population dynamics of potential pests and their natural enemies
	Landscape/global level	
Providing regular, high quality water	Stream flow response time to rain storms Turbidity of streams	Stream flow amounts and variability Sediment load of streams
Air filter: mitigating net emission of greenhouse gasses	Aboveground C stocks in biomass and necromass	Absence of agro-chemicals in water Soil C stocks relative to soil C saturation deficit Net emissions of N_2O and CH_4
Biodiversity reservoirs: allowing recolonization of depleted neighboring landscape units, and germplasm collection for ex-situ exploitation	Diversity of aboveground vegetation, based on 'plant functional attributes' or PFA diversity (Gillison and Carpenter, 1994)	Diversity of soil biota in selected 'indicator' groups

The main question now is how to derive a site-specific value of $C_{org,\ ref}$. In trying to construct such a point of reference, the differences between undisturbed forest soils from different sites should be linked to easily identifiable site (climate) or soil parameters. Within a given climatic zone, commonly measured soil parameters are preferred that can act as proxy for known processes leading to build-up (or breakdown) of soil organic matter. As soil biological properties are not routinely measured, chemical and physical factors should be sought that are associated with protection of soil organic matter from microbial breakdown. The two main candidates are soil texture and soil pH.

Differences in C_{org} concentrations between clay and sandy soils have long been noted and are usually attributed to various degrees of physical protection of organic matter against microbial attack. The CENTURY model, for example, links the decomposition constant for various soil organic matter pools directly to the clay content of the soil (Parton et al., 1994a). Matus (1994) found, however, no difference in the decomposition rates of ^{14}C-labeled crop residues between clay and sandy soils. Similarly, Hassink (1994, 1997) found that texture is not a good predictor of N mineralization on grassland soils. Apparently, once the larger C protection capacity of clay soils is saturated, the turnover rate of new inputs is similar to that in soils with a lower protection capacity. The C-saturation deficit, as defined above, appears to be a good predictor of net N mineralization of grassland soils, with $C_{org,\ ref}$ derived by regression analysis of C_{org} on texture for forest soils (of the Netherlands, in the application by Hassink, 1997). The scaling used for C_{org} in C_{sat} suggests that the C_{org} content on degraded soils or in subsoil (with a high C saturation deficit) may rise relatively quickly when organic inputs are applied, while a sustained increases in C_{org} under higher C input regimes (for example caused by elevated atmospheric CO_2) is not likely. The physical protection is not absolute, and especially tillage is an effective way of exposing fresh ped surfaces to microbial attack.

Apart from texture, soil pH may be used in establishing $C_{org,\ ref}$ for tropical forest soils. A negative relation between soil pH and C_{org} was established for forest soils of Sumatra (Indonesia) in the 1930s by Hardon (1936) for the pH range of 3.5 to 5.5, with a possible upward trend at higher pH. Although no clear mechanism for pH related C transformations is known, and models such as CENTURY do not include pH as a soil factor, the relationship found by Hardon merits further exploration.

Protocol

To establish the reference value $C_{org,\ ref}$ for a given climatic zone, a large data set is needed, e.g., derived from soil surveys, in which C_{org} is recorded as well as current land use (natural forest and other categories), soil texture, pH, elevation (as proxy for temperature) and preferably soil type (by a locally relevant classification system).

It is desirable that historical data are included in the data set, as long as the soil analysis methods have not essentially been modified. Historical soil data can be used to check any long term trends in the C_{org} of forest soils.

In a multiple regression analysis the various soil and site factors can be tested in their ability to predict (or account for) local variations in C_{org} of forest soils.

It may be necessary to use a logarithmic transformation of C_{org} to obtain the homogenous residuals on which regression analysis is normally based. Data transformation was needed as the C_{org} of soils did not have a symmetric normal distribution reflecting multiple mechanisms and an imperfect soil classification scheme (e.g.. shallow layers of peat do not classify a soil as histosol, yet cause a very high C_{org} in the topsoil). Back transformation of the fitted values will lead to wide confidence intervals, again reflecting the variability of the data base used.

Example and Data Interpretation

Van Noordwijk et al. (1997) analyzed soil data for Sumatra obtained in the 1980s in the context of the LREP (Land Resources Evaluation and Planning Project) project. Approximately 2800 profile data were found with complete records of soil type (soil taxonomy), land use, texture, and C_{org}. Five broad soil groups emerged from the analysis with significant between- group differences in C_{org}:

Histosols (peat) covering about 10% of Sumatra, but which may contain >90% of all C stored in Sumatran soils,

Andisols and wetland soils (Aquic groups) both contain about 10% (w/w) of C_{org}. On the Andisols C is intimately bound to clay complexes, while in wetland soils, the C is partially protected from decomposition by anaerobic conditions,

Among the remaining mineral soil types, two groups could be identified: relatively fertile upland soils (mainly Inceptisols) and the Oxisols plus Ultisols, with an average C_{org} content of 3.8 and 3.2%, respectively. The differences between all groups were statistically significant in a *t*-test.

In general, the C_{org} content decreases from primary forest, to secondary forest to areas used for tree crops and a slash-and-burn series, comprising food crops, shrub fallow land, and *Imperata* (alang-alang, cogon, or speargrass) grasslands. On the major upland soils, the difference in C_{org} content between land use types is about 0.5% C. At an average bulk density of 1.25 g cm^{-3}, this represents 10 Mg ha^{-1} for a 15-cm top soil layer. Changes in deeper layers may be expected to be less, and the total change is probably less than twice the change estimated from the top layer only. On the Andisols and the wetland soils, larger differences in C_{org} content are observed between land use types, but the smaller number of observations makes comparisons less certain. Potentially, land use effects on C_{org} may be more pronounced on these soils as management reduces the protection of C_{org} when Andisols are tilled and wetland soils drained.

A comparison can be made (Fig. 8–1) with the analysis made in the 1930s of a large data set obtained by Hardon (1936) from Lampung on the southernmost corner of the island. Lampung was then under transformation from forest to agricultural land, a change that today has been virtually completed. For nearly all land use categories, Hardon's data fell within the more recent data for the Inceptisols and the Oxisols plus Ultisols. There is no indication of increase in soil

C storage under forests in the 50-yr time span during which atmospheric CO_2 concentration increased by 20% in this period, from 0.029 to 0.035%. Hardon's average topsoil content over all land uses (3.53 %) in Lampung is close to the average of 3.46% for these soil groups (i.e., excluding volcanic, wetland, and peat soils) for the whole of Sumatra in the 1980s. The sampling frequency of land use types in this type of survey, however, may not be a true reflection of land use change and its effect on soil C storage.

The data set for the 1980s (Fig. 8–2) confirms the relation between soil pH and C_{org} established in the 1930s by Hardon (1936). The combined data show that the lowest C_{org} content can be expected in the pH range of 5.0 to 6.0. Below a pH of 5.0 reduced biological activity may slow down the breakdown of organic matter. Interestingly, most agricultural research recommends lime applications to the pH range of 5.0 to 6.0; this may stimulate breakdown of organic matter and thus contribute to crop nutrition, but possibly at the costs of maintaining the soil organic matter content. Effects of soil pH on C_{org} might be related to a shift from a dominantly bacterial to dominantly fungal soil foodweb.

In a multiple regression analysis, soil texture, pH, soil type, and land use were investigated (Van Noordwijk et al., 1997). The quantitative factors: pH, clay, and silt, had a slope that differs significantly ($P < 0.001$) from zero. The relative weighing factors for clay and silt are 1.4 and 1.0, respectively. The regression coefficient for elevation ($P < 0.01$) and for slope ($P < 0.05$) also were significantly different from zero. In this regression, the effects of elevation are studied separately from the elevational distribution of soil groups. They indicate a positive effect on C_{org} of lower temperatures. Compared with the average contents per soil type and land use, the C_{org} content will decrease 15% per unit

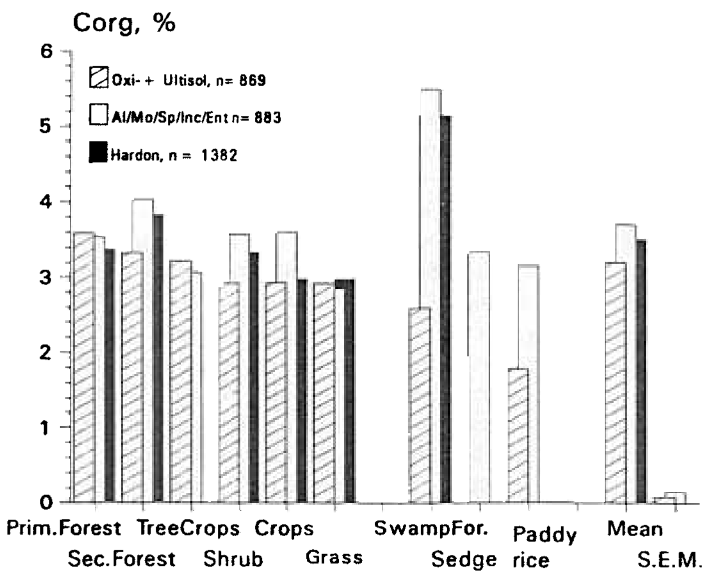

Fig. 8–1. Comparison of average C_{org} content of topsoil for upland soils in the early 1980s with data of Hardon (1936) for Lampung (Van Noordwijk et al., 1997).

increase in pH, increase 1 and 0.7% increase in clay and silt content, respectively, increase by 4% per 100 m increase in elevation and decrease by 0.3% increase in slope. From the regression analysis we propose the following equation for $C_{org,ref}$ in the upper 15 cm of mineral soil from Sumatra under forest cover:

$$C_{org,ref} = \exp(1.256 + 0.00994 \times Clay\% + 0.00699 \times Silt\%$$
$$- 0.156 \times pH + 0.000427 \times Elevation + 0.834 \text{ (if soil is Andisol)}$$
$$+ 0.363 \text{ (for swamp forest on wetland soils)}$$

The linearized effect of pH is not very satisfactory in view of the increase of C_{org} above pH 6, but adding a quadratic term for pH to the regression equation did not results in a significant reduction of the residual variance.

As average C-saturation for nonforest land uses, the data set suggests 91% for land with upland crops, 83% for land under tree crops of various types, 85% for *Imperata* grasslands and 81% for young secondary vegetation (shrub). C-saturation decreases with –2.6% per 10% slope. The small reductions for cropped land may come as a surprise. In Sumatra, however, there is little permanently cropped land and most land that is recorded as cropped in a survey was cleared from forest fallows. Also, much of the crop lands is under a no-till system after slash-and-burn land clearing.

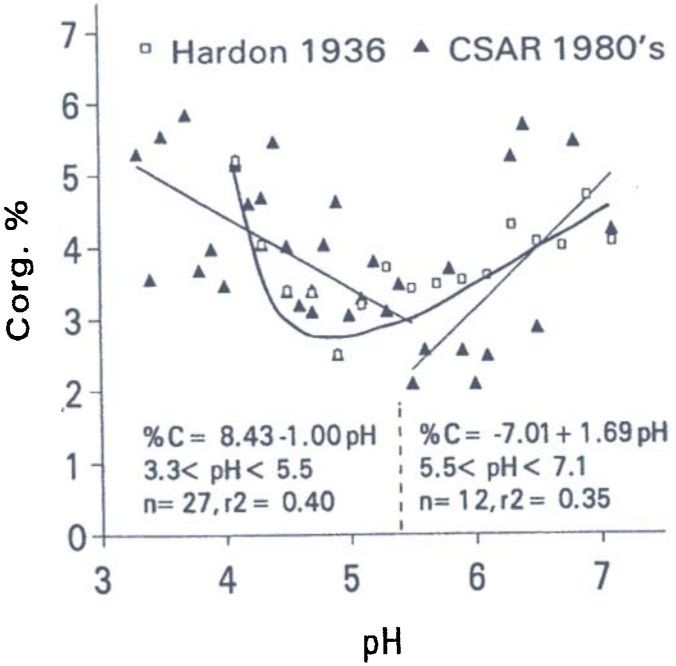

Fig. 8–2. Relation between soil pH and C_{org} for forest soils in Sumatra; the squares refer to data of Hardon (1936), the triangles to recent data (Van Noordwijk et al., 1997).

The concept of a C saturation index may be used in other conditions, but the specific parameters of a regression equation underlying $C_{org,ref}$ are likely to differ, and other parameters may have to be included in the regression equation. The use of models such as the CENTURY model to predict what a site-specific $C_{org,ref}$ would be may be resorted to if data are absent. One should realize, however, that these models may still miss important relationships (such as the effect of soil pH) and may not be universally applicable (Gijsman et al., 1996).

CARBON FRACTIONATION

Rationale

Total soil organic matter content, and its derivatives such as the C saturation index may not be very sensitive indicators, as they change relatively slowly under different management regimes. Measurement of total soil-C is adequate for evaluating C-stocks in the soil, but not for studying soil-C dynamics and functions of soil organic matter over the short term, as only a small part of the total C is responding rapidly to management. Development and testing of improved soil management practices would be easier if more sensitive indicators were available. Models of soil organic matter, such as the CENTURY model (Parton et al., 1994a,b) assume a number of pools of different turnover rates, but the measurement of such pools is still an area of active research and method improvement.

Suitability of fractionation methods should be based on:

1. Simple procedures that can be well standardized and lead to a small lab error,
2. Sensitive discrimination of land use practices with different qualities and quantities of organic inputs,
3. Pools with a different turnover time,
4. Pools that are significant indicators for soil functions such as N and P mineralization.

A large body of research (Ellert & Gregorich, 1995; Blair et al., 1997; Heal et al., 1997) is concerned with differentiating soil C-pools and fresh organic inputs on the basis of their chemical characteristics such as response to mild oxidation agents. But apart from the nature of the organic pools, their nurture or environment during the decomposition process also plays a role, and separating nature and nurture effects is complicated. Physical fractionation methods based on size and physical density of soil fractions in a certain size range (Christensen, 1992; Cambardella & Elliot, 1992; McColl & Gressel, 1995) may allow the distinction of pools with different degrees of physical protection from decomposers. During decomposition plant litter, with an initial physical density around 1.0 g cm^{-3} becomes more intimately associated with mineral particles with a physical density of around 2.5 g cm^{-3}. The total specific weight of a fraction will thus reflect the amount of clay associated with it, and may be related with the turnover time of the fraction. We will here focus on just one of the available methods for physical fractionation and indicate the types of evidence on its value and shortfalls, but it is important to stress that this is an area of ongoing debate and method improvement, Standardization is valuable in that it allows inter-lab comparisons, but it also may be an obstacle to progress. Interpretation of results of any fractionation procedure should be based on local context.

Protocol

A fractionation procedure developed by Meijboom et al. (1995) on the basis of colloidal silica suspensions (Ludox) leads to a light, intermediate, and heavy fraction (Hassink, 1995). This method has been tested in a number of sites in the tropics.

The procedure involves the following steps:

1. Dry sieve the soil through a 2-mm mesh sieve to remove roots and coarse litter particles,
2. Rewet 500 g of soil and leave it for 24 h to equilibrate,
3. Wash the samples on a 150-μm sieve under a gentle stream of water; a 250-μm sieve can be placed on top of the finer one to avoid clogging of the sieve; fine aggregates may be crushed on the coarser sieve during the washing; the silt and clay sized particles passing through the 150-μm sieve are discarded,
4. Collect material from both sieves and separate the coarse mineral sand particles from fractions that contain organic material by decantation in swirling water; this procedure needs further standardization; the mineral fraction is discarded, but may have to be checked on an organic C content while testing the method,
5. The remaining sand-sized fractions are separated into three fractions, by sequential immersion into silica suspensions (Ludox) of two physical densities: 1.13 and 1.3 g cm^{-3}. In the method description by Meijboom et al. a density of 1.37 g cm^{-3}, was used, but its viscosity may cause problems and a suspension of 1.3 g cm^{-3} is preferred (Hairiah et al., 1995); the three fractions are indicated as light (floating on 1.13 g cm^{-3}), intermediate (floating on a 1.3 g cm^{-3} suspension, but not on 1.13 g cm^{-3}), and heavy (not floating on either). The material coming to the surface on a given suspension within a specified time is scooped off, rinsed and dried. Its total C and N content can be measured by conventional means.

For further studies the silt and clay fraction may be collected and yield additional information (Hassink, 1997). Full standardization of the method and comparisons of results between laboratories have not yet been achieved and details of the methods (use of soil dispersion agents prior to washing, mesh size of the sieves used, density used to separate intermediate and heavy fractions) may differ between the various publications. Results on the second, third, and fourth criterion for a fractionation scheme are promising, however.

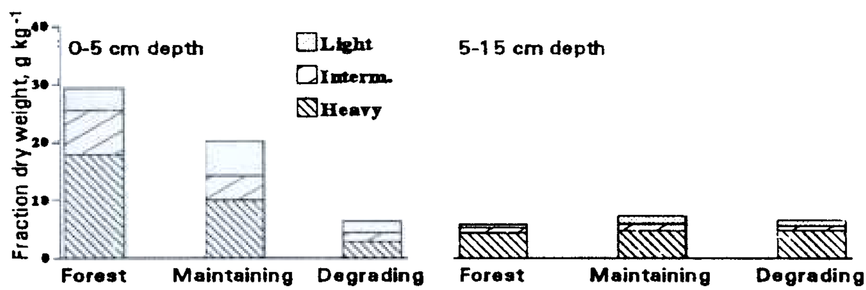

Fig. 8–3. Soil organic matter fractions, based on the Ludox size–density fractionation scheme (Meijboom et al., 1995), for three groups of land use practices in the North Lampung benchmark area of the Alternatives to Slash and Burn project in Indonesia (Hairiah et al., 1995).

Data Interpretation: Indicator Value of Ludox Fractions

In a survey of the organic matter content of topsoil in the N. Lampung benchmark area of the Alternatives to Slash and Burn project in Sumatra (Indonesia; Fig. 8–3), three groups of land use practices could be differentiated on the basis of Ludox fractions of the top 5 cm of mineral soil:

1. Forest (remnants of logged-over primary and various types of secondary forest)
2. SOM-maintaining practices: woodlots, forest plantations established with slash-and-burn land clearing, home gardens, and unburnt *Imperata* grasslands,
3. Degrading lands: burnt *Imperata*, sugar cane plantations with annual burning of residues and forest plantations established with bulldozer land clearing.

For the second category of land use systems the sum of the Ludox fraction (g kg^{-1}) in the top 5 cm of soil may still decrease by about 20 to 30% from the forest level. Under degrading situations, the data suggested that 8 to 10 yr after opening the forest, the sum of the Ludox fraction decreased by 70 to 80%. In the 5- to 15-cm depth layer, however, the converted forest sites exceeded the forest. Total content of the Ludox fractions (in g kg^{-1} of soil) for this second layer is only 20 to 50% of that in the top 5 cm. In the 5- to 15-cm soil layer the heavy fraction is dominant compared with the light and intermediate fraction in dry weight. In as far as the sum of the fractions can be used as indicator, rather than the fractions per se, the procedure could be simplified. Evidence so far is not unequivocal. In a study of long term soil fertility experiments in Kenya (Kapkiyaga, 1996, personal communication), however, results for the simpler particulate organic matter (POM) method (Anderson & Ingram, 1993) based on size only were not improved by a subsequent density fractionation. The POM method, however, uses 50 μm as smallest sieve size and the Ludox method as defined by Meijboom et al. (1995) uses 150 μm.

Table 8–2. Decomposition constants for forest soil organic matter (C_{org}) and for the Light, Intermediate and Heavy fraction of macro-organic matter obtained with the Ludox method based on an analysis of $\Delta^{13}C$ of soil organic matter 1–10 years after conversion of forest to sugarcane in Lampung (Indonesia) (Hairiah et al., 1995).

Fraction	Decomposition constant, k (yr^{-1})	Standard error of estimated k (S.E.)	Percentage of variance accounted for (R^2)
	0.194	0.026	91.3
	0.185	0.049	72.6
	0.142	0.013	96.1
	0.168	0.024	90.3
	0.082	0.029	57.8

Turnover Time of Ludox Fractions

A direct assessment of the turnover of these Ludox fractions of the original forest soils was obtained from a chronosequence of sites where forest had been converted to sugarcane in the past 10 yr (Hairiah et al., 1995). Analysis of the stable C isotope ratio $^{12}C/^{13}C$ of the Ludox fractions allowed distinction of the organic matter in the three fractions derived from the forest vegetation (a C3 photosynthetic pathway) and from the sugar cane (with a C4 photosynthetic pathway). From these time series decomposition constants could be derived (Table 8–2), for the total C_{org} pool as well as for the various fractions. These decomposition parameters, however, are a net effect of transformations between pools and decomposition (release of CO_2); current data do not allow a full separation of inter-pool conversions. On the basis of the apparent turnover time the light and intermediate fraction can be clearly distinguished from the heavy fraction and the total C_{org} pool, and the regression lines for the decay of the various fractions where better defined (larger R^2 value) than for C_{org}); however, the differences in turnover rate between the fractions are smaller than one might expect. Ten years after forest conversion 25, 40, and 60% of the light, intermediate, and heavy fractions, respectively, still has a forest C signature. These results may be the best indication so far of the gain in information if physical fractionation is taken beyond size as criterion. We do not yet have, however, a satisfactory parametrization for the whole decomposition cascade and inter-pool transformations in the soil (Matus, 1994).

Functional Significance of Ludox Fractions

Barrios et al. (1996, 1997) tested the Ludox method in the analysis of sequential agroforestry systems in Kenya and Zambia and found that the light and intermediate fractions obtained with this method appear to be the most important ones for the N mineralization in the first year after the fallow. When the various Ludox fractions obtained from a range of land use practices in Lampung (Sumatra) with different qualities and quantities of organic inputs were incubated, the light fraction appeared to have the highest specific P mineralization rate (Fig. 8–4).

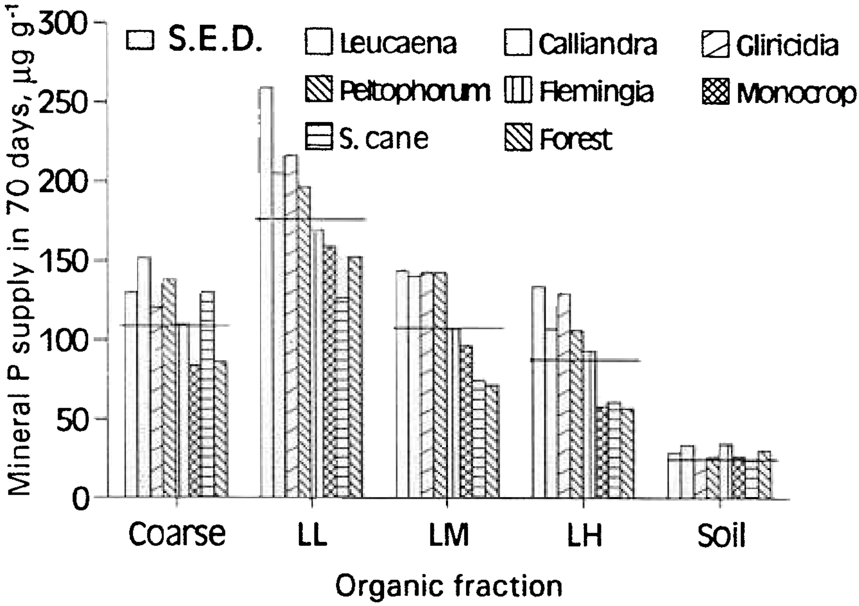

Fig. 8–4. Phosphorus mineralization during a 70-d incubation of organic matter fractions obtained from a range of land use practices with different qualities and quantities of organic inputs in North Lampung.

Conclusion on Ludox Method

The Ludox fractionation method gives more sensitive indicators for studying C dynamics than total soil C_{org}, especially for the 0- to 5-cm depth. Interlaboratory variability in methods, valuable as it may be in the method development stage, makes it difficult to use the method for any legally binding purposes. It is best seen as a research tool as yet. Other approaches, such as the flotation method used by Vanlauwe et al. (1996) should be further compared with the Ludox method.

OTHER INDICATORS OF ON SITE PRODUCTIVITY

For most of the indicators and measurables of soil qualities to be used for on-site productivity [Table 8–1, well-established methods and protocols exist (Anderson & Ingram, 1993; Kooistra & Van Noordwijk, 1996; Hall, 1996]. For rapid assessments, the thickness of the litter layer (rather than presence–absence of surface litter) can be used as indicator, although it should be evaluated with knowledge of the turnover rate and seasonality of litterfall. Spatial variability in litter deposition rates in rain forests may be higher than variability in turnover rates. Burghouts (1993) measured total aboveground fine litter production of 11 Mg ha^{-1} yr^{-1} for a Dipterocarp forest on an acid soil in Sabah. Litter mass on the forest floor was highly variable and varied from a layer of one or two leaves thick to a well-developed layer, 10- to 15-cm thick. He showed that local patches with a thick litter layer were not caused by a slower decomposition rate, but by differences in litter deposition rates. The expected lifetime of surface litter differs between climatic zones and depends on the chemical and physical quality of the litter as well as on soil macrofauna. If surface litter accumulates in plantation forests whereas it decomposed more rapidly in the mixed natural forest the plantation replaced, a further analysis is needed. Perhaps important groups of the soil fauna have not been able to adjust to the change in litter regime.

Assessments of biodiversity for belowground organisms are scarce, especially in the humid tropics. As rapid assessment method for aboveground plant diversity (Gillison & Carpenter, 1994) are available, a first approach is via a presumed relationship between above and belowground diversity in a range of forest-derived land use systems. Measurements of soil biodiversity in the Alternatives to Slash and Burn project include an assessment of surface litter macrofauna, macrofauna in the top 30 cm, general microbial properties of the topsoil, most probable number (MPN) counts of *Rhizobium* and P-solubilizing bacteria, mycorrhizal spore counts and mycorrhizal and *Rhizobium* infection of selected trap crops. Initial results showed large effects of land use on litter fauna, but comparatively small effects on organisms in the mineral soil. Belowground biodiversity indicators apparently respond more slowly to a change in forest cover than their aboveground counterparts. A further refinement of the indicators may be needed, however.

There is remarkably little detailed evidence that agricultural land use in its various degrees of intensity results in a loss of biodiversity in soil (Giller et al., 1997). A rare exception is that of the changes in earthworm populations on conversion of tropical rainforest to pasture in the Amazon Basin, where a single species survives (Fragoso et al., 1997) that leads to soil compaction due to its massive surface casting activity. In this example a reduction in diversity is coupled to and presumably responsible for a loss of function and agricultural productivity. In other cases soils can be mistreated to a remarkable extent and yet crops continue to support crop yields close to their theoretical maximum. Thus the interpretation of soil biological properties as sustainability indicators will require more background data than currently available.

INDICATORS OF LANDSCAPE OR GLOBAL FUNCTIONS OF FOREST SOILS

Functions of forest soils in modulating the water balance downstream are well-known; however, extrapolation between climatic zones of these functions may have lead to an overestimate of these roles in the humid tropics: if rainfall is so high that the whole soil system remains near saturation (with real or perched watertables near the surface), a fairly rapid transmission of rain storms to areas downstream can be expected, either by surface or subsurface flow of water, even under full forest cover. Recent developments in landscape level modeling of runoff and infiltration, indicate a scale dependence of effects of forests on the water balance and indicate that location of forest cover in a watershed is at least as important as the percentage forest cover *per se* (De Ridder et al., 1996).

The air filter function of forest soils and their slash-and-burn derivatives has been mostly discussed from a C storage point of view (Tinker et al., 1996). Opportunities for further C storage are generally limited in topsoils, as the fairly small C saturation deficits indicate. Potentially larger opportunities exist in the subsoil, as those layers may be well below their C-saturation.

Surveys of net CH_4 and N_2O emissions as part of the Alternatives to Slash and Burn project indicated that forest soils can act as a CH_4 sink of considerable strength. According to initial measurements in Sumatra (Van Noordwijk et al., 1995), 1 ha of forest may off-set the CH_4 emissions of about 20 ha of rice paddy. Direct CH_4 exchange, however, may be especially relevant where slash-and-burn land clearing is practiced in a forested landscape mosaic. Smouldering fires release substantial amounts of CH_4, but not all of this will reach the atmosphere if there is enough under-used CH_4 oxidation capacity nearby. The CH_4 oxidation function of forest soils appears to remain intact in forest-like conditions such as found in the rubber agroforests, but it is diminished in more intensive land use types. Further research on this topic may aim to distinguish which soil factor (bulk density as an indicator of macropositiy and hence CH_4 diffusion into soils, specific C fractions or N pools that may modulate the activity of methanotrophe bacteria, or the population size of this group) is responsible for differences between soils of different land use categories.

N_2O emissions appear to be linked to mineral N concentrations. The initial surveys indicated higher emissions from forest soils than for some of the degraded environments in *Imperata* grasslands. Net CH_4 and N_2O emissions can be combined with the changes in the C balance to estimate the net radiative forcing effect of various land uses.

Overall, the trajectory from common sense, simple indicators to protocols that are sufficiently tight to survive courtroom tests and become part of legally binding commitments, is extremely arduous. Whereas standardization of methods is necessary to make cross site and across-time comparisons, development and fine tuning of new methods also is required to keep up with increasing demands from society at large to guarantee that public functions of forest soils will be maintained in the future.

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