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Soil erosion in the humid tropics: A systematic quantitative review

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ABSTRACT

Healthy soils provide a wide range of ecosystem services. But soil erosion (one component of land degradation) jeopardizes the sustainable delivery of these services worldwide, and particularly in the humid tropics where erosion potential is high due to heavy rainfall. The Millennium Ecosystem Assessment pointed out the role of poor land-use and management choices in increasing land degradation. We hypothesized that land use has a limited influence on soil erosion provided vegetation cover is developed enough or good management practices are implemented. We systematically reviewed the literature to study how soil and vegetation management influence soil erosion control in the humid tropics. More than 3600 measurements of soil loss from 55 references covering 21 countries were compiled. Quantitative analysis of the collected data revealed that soil erosion in the humid tropics is dramatically concentrated in space (over landscape elements of bare soil) and time (e.g. during crop rotation). No land use is erosion-prone per se, but creation of bare soil elements in the landscape through particular land uses and other human activities (e.g. skid trails and logging roads) should be avoided as much as possible. Implementation of sound practices of soil and vegetation management (e.g. contour planting, no-till farming and use of vegetative buffer strips) can reduce erosion by up to 99%. With limited financial and technical means, natural resource managers and policy makers can therefore help decrease soil loss at a large scale by promoting wise management of highly erosion-prone landscape elements and enhancing the use of low-erosion-inducing practices.

1. Introduction

The ecosystem service of soil erosion control, for the delivery of which vegetation cover plays an important role, has been degrading worldwide (Millennium Ecosystem Assessment, 2005). As this regulating service is lost, soil formation can no longer compensate for soil loss due to an increase in erosion, which depletes soil resources and the ecosystem services they support (Lal, 2003; Morgan, 2005). The Millennium Ecosystem Assessment (2005) identified unwise land-use choices and harmful crop or soil management practices as the major drivers of increasing soil erosion. Soil erosion has multiple on- and off-site consequences such as decreasing crop yields, increasing atmospheric CO_2 concentration, decreasing water quality (turbidity and particle-born pollutants), sedimentation of reservoirs, and disturbed hydrological regimes such as increased flood risk due to riverbed filling and stream plugging (Chomitz and Kumari, 1998; Lal, 2003; Millennium Ecosystem Assessment, 2005; Morgan, 2005; Locatelli et al., 2011).

Research on factors influencing soil loss has resulted in widely used models, such as the RUSLE (revised universal soil loss equation). This model was built from plot data of experiments carried out in the United States and predicts soil loss from climatic (rainfall erosivity), edaphic (soil erodibility) and topographic (slope length and slope steepness) factors, as well as soil and vegetation management practices (Wischmeier and Smith, 1978; Renard et al., 1997). Management of soil and vegetation has long been recognized as the most efficient and effective way to influence the extent of soil loss, and therefore soil erosion control (Goujon, 1968).

The humid tropics are rich in carbon and biodiversity and attract major attention because of the rapid loss of rainforests (Strassburg et al., 2010; Saatchi et al., 2011; Tropek et al., 2014). Because of the large amount and high intensity of

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rainfall in the humid tropics, soil erosion can potentially reach dramatic levels in this region (El-Swaify et al., 1982; Lal, 1990). Tropical ecosystems with healthy soils can support multiple ecosystem services (e.g. water regulation, climate regulation through carbon storage and biodiversity support) and support local livelihoods. A better understanding of soil erosion control in the humid tropics is therefore vital (Locatelli et al., 2014).

Theoretically, empirical models of erosion prediction should only be applied under conditions and for purposes similar to those of their development (e.g. predicting erosion from croplands in the United States for the RUSLE). Adapting an empirical model to out-of-range conditions would require parameter calibration, which can consume both time and resources (Nearing et al., 1994). While some studies have adapted temperate model factors to their own geographical contexts (e.g. Streck and Cogo, 2003 for surface soil consolidation and Diodato et al., 2013 for rainfall erosivity), others have directly applied models developed for a temperate context to predict soil erosion in the humid tropics (e.g. Angima et al., 2003; Hoyos, 2005).

Yet there is little consensus about the direct applicability of models such as RUSLE (and its predecessors) to a tropical context. Despite over- and under-estimation of soil loss depending on the cropping phase, Almas and Jamal (2000) found the RUSLE model to correctly predict the overall soil loss from a banana–pineapple intercropping system in Malaysia. On the other hand, Cohen et al. (2005) showed that erosion risk prediction was poorly achieved by the USLE (universal soil loss equation) in a watershed of western Kenya, and called for ground surveys to properly calibrate the USLE and similar empirical models.

In the face of this lack of agreement, studies that directly measure soil loss are of great interest as they can help shed light on the influence of vegetation and soil management on soil erosion control. Synthesizing and analyzing available data from multiple sources is necessary given the diversity of study contexts and the impossibility of drawing general conclusions from a single study.

Such syntheses are available for some regions of the world. Focussing on Europe and the Mediterranean, Maetens et al. (2012) reviewed data from 227 stations and 1056 soil erosion plots to analyze the effect of land use on erosion and runoff. They found that (semi-) natural vegetation produced lower erosion (<1 Mg/ha/yr) than vegetation directly influenced by human activities (e.g. croplands and vineyards; 6–20 Mg/ha/yr). Montgomery (2007) also compiled erosion data from globally distributed studies (some in the humid tropics) and showed that conventional agriculture, i.e. with tillage, produced 10–100 times more soil loss than conservation agriculture, i.e. with no-tillage, but conditions were highly variable. For example, plots under conventional agriculture were more erosion-prone (with maximum slope of 37° and maximum annual precipitation of 5600 mm/yr) than those of plots under conservation agriculture (17° and 2000 mm/yr).

Table 1
Land-use types and subtypes.

Land-use type	Land-use subtype	Definitions
Bare		Land has been opened and kept bare for various reasons (includes pre-sowing and post-harvesting cropland and skid trails).
	Tilled	High-disturbance soil management techniques (e.g. ploughing and raking) are used.
	Untilled	Low-disturbance soil management techniques (e.g. slash and burn and weeding with a knife) are used.
Cropland		Crops are sown and harvested within a single agricultural year, sometimes more than once (excludes perennial crops).
	Crop, non-established, without conservation practices	Crop was recently planted and crop cover is not developed; no conservation techniques are practiced.
	Crop, established, without conservation practices	Crop cover is developed; no conservation techniques are practiced.
	Crop with vegetation-related conservation practices	Crop cover may or may not be fully developed. Vegetation-related conservation techniques (e.g. hedgerows, intercropping and mulching) are practiced.
	Crop with vegetation- and soil-related conservation practices	Crop cover may or may not be fully developed. Both vegetation-related (e.g. hedgerows, intercropping and mulching) and soil-related (e.g. no-till farming and contour planting) conservation techniques are practiced.
Grassland		Vegetation is dominated by grasses (includes open grasslands and pastures).
	Pasture	Land is used for grazing and managed through agricultural practices such as seeding, irrigation and use of fertilizer.
	Open grassland	Land is unmanaged and has no trees or shrubs.
Shrubland		Vegetation is dominated by shrubs but can also include grasses, herbs and geophytes.
	Open shrubland	A transitional plant community occurs temporarily as the result of a disturbance such as logging or fire.
Tree-dominated agrosystem		Planted vegetation is dominated by trees, including perennial tree crops such as rubber, fruit and nut trees.
	Tree plantation	A group of planted trees is grown in the form of an agricultural crop, usually with the aim of harvesting wood.
	Tree crop without contact cover	A permanent crop has been planted; it has no contact cover (such as grass or cover crops) underneath.
	Tree crop with contact cover	A permanent crop has been planted and has contact cover (such as grass or cover crops) underneath.
	Simple agroforest	One woody perennial species is planted with one annual crop.
	Complex agroforest	Multiple species of woody perennials, often with natural vegetation regrowth, are planted (usually intercropped) with annual crops.
Forest		Ground is covered with natural vegetation dominated by trees (excludes tree plantations).
	Secondary forest	Forest has regenerated naturally after clear-cutting, burning or other land-clearing activities and contains vegetation in early successional stages.
	Old-growth forest	Forest is ecologically mature, containing trees of various sizes and species (the last stage in forest succession).
	Logged-over forest	Forest has been logged-over.
	Degraded forest	Forest has been degraded by human activities other than logging or by a naturally occurring event such as a fire or severe storm.

Selecting erosion measurements available for the two agriculture types under the same conditions substantially reduced the sample.

No synthesis (to our knowledge) has been done so far for the humid tropics. The purpose of this study was therefore to quantitatively analyze available data (collected via systematic review of the literature) on soil erosion in the humid tropics to study how soil and vegetation management influence soil erosion control in this region. Effects of the measurement protocol (method, duration and area) and context (rainfall, slope length, slope steepness and soil erodibility) were controlled for to keep a consistent dataset and focus on the influence of soil and vegetation management on soil erosion.

The underlying hypothesis is that land use has a limited influence on soil erosion provided vegetation cover is developed enough or good management practices are implemented. This hypothesis was previously conclusively tested in a few single studies on ecosystems such as rangelands (e.g. Snelder and Bryan, 1995; Chartier and Rostagno, 2006), but never systematically nor for the humid tropics. This study aims to contribute to the scientific understanding of the relationship between soil erosion and vegetation/soil management in the humid tropics, to help clarify the applicability of widely used models such as the RUSLE, and to provide to stakeholders involved in natural resource management and protection a synthesis on soil erosion control and its sound management.

2. Materials and methods

2.1. Materials

We searched for studies of erosion in the humid tropics, defined for the purpose of this review as the "Af" (tropical rainforest climate) and "Am" (tropical monsoon climate) Köppen climatic classes (Köppen, 1936; Peel et al., 2007). Queries were built on the conjunction of elements from three thematic clusters: "scope" and "outcome" and "measurement". The "scope" cluster corresponded to: tropic* or region (list of broadly defined relevant regions, e.g. Africa) or specific country (all countries under either Af or Am climate were considered, e.g. Brazil). The "outcome" cluster encompassed the following terms: soil erosion, water erosion, soil loss, soil depletion, land degradation, sedimentation, sediment production and siltation. The "measurement" cluster included keywords defining methodological approaches and measurement methods such as "runoff plot" and "sediment trap". In order to select studies with homogeneous land use; we excluded measures at the catchment scale. Additionally, to avoid bias in the analysis of reported measurements, indirect measures and estimates (e.g. the use of ^{137}Cs as a tracer—see Sidle et al., 2006) were not considered. As suggested by the Collaboration for Environmental Evidence (2013), a variety of peer-reviewed and grey literature sources were searched. Details about queries and sources are available in Appendix A. Queries were carried out during the second half of April 2013 in English, French and Spanish.

Searches led to 5183 references after removing duplicates. After irrelevant references were removed, based on information in article titles and abstracts about topic, geographical scope and erosion measurement method, the database shrank to 114 references. Finally, after screening the full texts of those references, we kept 55 of them (more details are available in Appendix B). For each reference, we retrieved data on soil loss (expressed as quantity of soil mass per unit of area) in one or more cases. A case was defined as one erosion measurement, characterized by an associated measurement method (profile meter, root exposition, sediment trap, unbounded plot or runoff plot, all with natural rainfall, and runoff plot with simulated rainfall), area and duration, topographical features (slope length and steepness), rainfall, and land-use type and subtype (see definitions in Table 1). For each case, building on the classification proposed by Moench (1991), vegetation cover was also described by the presence or absence of four layers: high (≥4 m), intermediate (at least 1 m but <4 m), low (at least 0.1 m but <1 m) and ground (<0.1 m).

The final data set consisted of 3649 measurements from 55 references covering 21 countries in the humid tropics (Fig. 1, Table 2). Most references originated from peer-reviewed journals ($n = 44$) and used runoff plots to quantify soil loss ($n = 48$). Publication years ranged from 1973 to 2012, with half of the references published before 1997 (Fig. 2a). The number of cases per study was highly variable, and the six references with the most cases contributed half the total number of cases in the final data set (Fig. 2b, Table 2). Study length ranged from two days (studies under simulated rainfall) to 17 years (Fig. 2c). References generally reported erosion values per rainfall event, per year or for the duration of the study (Fig. 2d). Most references assessed one to three land-use types (Fig. 2e), of which bare soils and croplands were the most studied (Fig. 2f).

Rainfall erosivity and soil erodibility were assessed for each case. An indicator of rainfall erosivity sensu Renard et al. (1997) could not be obtained or computed for most cases because monthly data were not available or because measurement duration was too short to apply an annual erosivity index. We thus used total rainfall as an indicator of rainfall erosivity based on the finding by Maetens et al. (2012) that soil loss does not correlate better with erosivity indices than with total rainfall.

For soil erodibility, we combined different indices because of the diverse ways soils were described in the studies. For each case, we calculated three soil erodibility indices from soil texture and organic matter data with an empirical table and two different equations (Stewart et al., 1975; Sharpley and Williams, 1990; Torri et al., 1997). If soil data were not available in a study, we extracted them from the ISRIC global soil dataset (resolution of 1 km) using measurement coordinates (ISRIC-World Soil Information, 2013). For each index, soils were split into low-, medium- and high-erodibility classes of equal sizes. A soil was then classified as highly erodible if it was considered highly erodible by at least two of the three indices, low if it was considered low by at least two indices and medium otherwise (more details are available in Appendix C).

2.2. Data analysis

All data transformation and statistical analysis were done using R (R Core Team, 2013). Due to highly skewed distributions, all

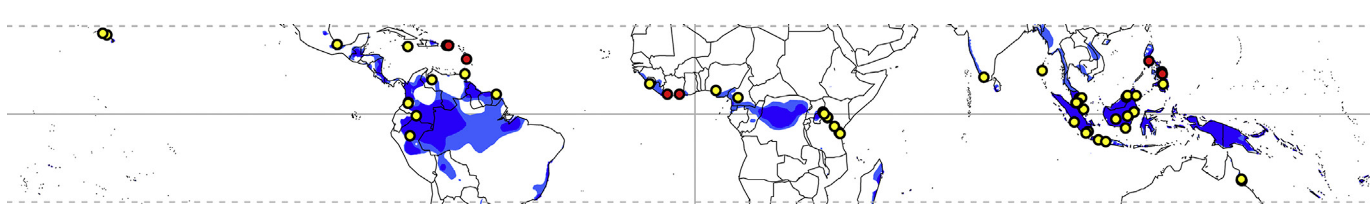

Fig. 1. Location of study sites ($n = 61$). Some dots represent several references, and some references contribute more than one dot. Red dots show locations provided by the six references with the most cases. Af (tropical rainforest) climate ranges are displayed in dark blue and Am (tropical monsoon) climate ranges in light blue. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 2
Contributing references by geographical location. References from Southeast Asia and Northeast Australia ($n=29$) made up more than half of all references ($n=55$). The 30 references with the fewest cases provided about 10% of all cases ($n=3649$). The 6 references with the most cases are printed in bold.

Reference	Country	Source type	Method	Rainfall type	Soil data[a]	Land-use type(s)[b]	Cases	Case time frame(s)	Study length
Africa ($n=11$)									
Ambassa-Kiki and Nill (1999)	Cameroon	Journal article	Runoff plot	Natural	ST + OM	3 (B, C, T)	3	Study	2 years
Boye and Albrecht (2004)	Kenya	Project report	Runoff plot	Simulation	ST + OM	1 (B)	10	Rainfall event	2 days
Collinet (1983)	Côte d'Ivoire	Project report	Runoff plot	Natural	None	2 (C, F)	24	Year, study	3 years
Collinet (1988)	**Côte d'Ivoire**	**PhD thesis**	**Runoff plot**	**Simulation**	**None**	**2 (B, C)**	**189**	**Rainfall event**	**2 months**
Defersha and Melesse (2012)	Kenya	Journal article	Runoff plot	Natural	ST + OM	3 (B, C, G)	87	Rainfall event, month	1 month
Kamara (1986)	Sierra Leone	Journal article	Runoff plot	Natural	ST + OM	2 (B, C)	14	Month	2 years
Lundgren (1980)	Tanzania	Journal article	Runoff plot	Natural	ST + OM	2 (F, T)	33	Year, study	2 years
Ngatunga et al. (1984)	Tanzania	Journal article	Runoff plot	Natural	ST + OM	3 (B, C, G)	36	Season, year	1 year
Odemerho and Avwunudiogba (1993)	Nigeria	Journal article	Runoff plot	Natural	ST	2 (C, G)	126	Rainfall event, study	5 months
Roose (1973)	**Côte d'Ivoire**	**PhD thesis**	**Runoff plot**	**Natural**	**None**	**5 (B, C, F, G, T)**	**431**	**Rainfall event, day, month, season, year**	**17 years**
Våje et al. (2005)	Tanzania	Journal article	Runoff plot	Natural	ST + OM	2 (B, C)	10	Rainfall event, season	2 years
America & North Pacific Ocean ($n=10$)									
Alegre and Cassel (1996)	Peru	Journal article	Runoff plot	Natural	OM	3 (B, C, F)	4	Study	52 months
Alegre and Rao (1996)	Peru	Journal article	Runoff plot	Natural	OM	3 (B, C, F)	50	Season, year, study	5 years
Bellanger et al. (2004)	Venezuela	Journal article	Runoff plot	Natural	ST + OM	3 (B, C, T)	41	Rainfall event, week, season	5 months
Dangler and El-Swaify (1976)	USA (Hawaii)	Journal article	Runoff plot	Simulation	None	1 (B)	16	Rainfall event	1.75 years
Francisco-Nicolas et al. (2006)	Mexico	Journal article	Runoff plot	Natural	OM	1 (C)	18	Year, study	8 years
Fritsch and Sarrailh (1986)	France (French Guiana)	Journal article	Runoff plot	Natural	None	2 (B, F)	38	Month, season, year, study	32 months
McGregor (1980)	Colombia	Journal article	Runoff plot	natural	ST	3 (C, F, G)	7	Study	8 week
Ruppenthal et al. (1997)	Colombia	Journal article	Runoff plot	Natural	None	2 (B, C)	32	Season	2 years
Sarrailh (1981)	France (French Guiana)	Project report	Runoff plot	Natural	None	2 (F, G)	50	Month, season, year, study	20 months
Wan and El-Swaify (1999)	USA (Hawaii)	Journal article	Runoff plot	Simulation	ST + OM	2 (B, C)	6	Rainfall event	2 days
SE Asia & NE Australia ($n=29$)									
Afandi et al. (2002a)	Indonesia	Journal article	Runoff plot	Natural	ST + OM	1 (T)	54	Month	3.5 years
Afandi et al. (2002b)	Indonesia	Journal article	Sediment trap	Natural	ST + OM	4 (C, F, G, T)	77	Month, study	11 months
Almas and Jamal (2000)	Malaysia	Journal article	Runoff plot	Natural	None	3 (B, C, T)	52	Season	9 months
Baharuddin et al. (1995)	Malaysia	Journal article	Runoff plot	Natural	None	3 (B, F, G)	90	Month, year	2 years
Bons (1990)	Indonesia	Conference proceedings	Runoff plot	Natural	None	2 (S, T)	2	Year, study	26 months
Chatterjea (1998)	Singapore	Journal article	Runoff plot	Natural	None	2 (B, G)	30	Rainfall event	1.3 years
Comia et al. (1994)	Philippines	Journal article	Runoff plot	Natural	ST + OM	1 (C)	16	Year, study	3 years
Daño and Siapno (1992)	Philippines	Conference proceedings	Runoff plot	Natural	None	1 (T)	22	Year, study	2 years
Hartanto et al. (2003)	Indonesia	Journal article	Runoff plot	Natural	None	2 (B, F)	135	Rainfall event, season	2.5 months
Hashim et al. (1995)	Malaysia	Journal article	Runoff plot	Natural	ST + OM	2 (B, T)	152	Rainfall event, season, study	1.5 years
Jaafar et al. (2011)	Malaysia	Journal article	Runoff plot	Natural	ST + OM	1 (F)	6	Year	1 year
Leigh (1982)	Malaysia	Journal article	Sediment trap	Natural	ST	1 (F)	11	Year	1 year
Malmer (1996)	Malaysia	Journal article	Unbounded plot	Natural	None	2 (B, F)	3	Year, study	1 year
Moehansyah et al. (2004)	Indonesia	Journal article	Runoff plot	Natural	ST	3 (C, G, T)	156	Rainfall event, season, study	8 months
Moench (1991)	India	Journal article	Runoff plot	Natural	OM	1 (T)	21	Study	9 months
Pandey and Chaudhari (2010)	India	Journal article	Runoff plot	Natural	ST	3 (C, F, T)	44	Year, study	3 years
Paningbatan et al. (1995)	**Philippines**	**Journal article**	**Runoff plot**	**Natural**	**ST + OM**	**1 (C)**	**168**	**Rainfall event, season**	**3 years**
Poudel et al. (1999)	Philippines	Journal article	Runoff plot	Natural	ST + OM	1 (C)	35	Season, study	2.5 years
Poudel et al. (2000)	Philippines	Journal article	Runoff plot	Natural	OM	1 (C)	12	Year	2.5 years
Presbitero (2003)	**Philippines**	**PhD thesis**	**Runoff plot**	**Natural**	**OM**	**2 (B, C)**	**433**	**Rainfall event**	**2.5 years**
Prove et al. (1995)	Australia	Journal article	Profile meter	Natural	None	1 (C)	14	Year	6 years
Ross and Dykes (1996)	Brunei	Book chapter	Runoff plot	Natural	ST	1 (F)	24	Month	8 months
Shimokawa (1988)	Indonesia	Book chapter	Root exposition	Natural	None	1 (F)	21	Year	1 year
Siebert and Belsky (1990)	Indonesia	Journal article	Runoff plot	Natural	ST + OM	1 (C)	3	Season	9 months
Sinun et al. (1992)	Malaysia	Journal article	Runoff plot	Natural	None	3 (B, F, G)	78	Month, year	1 year
Sudarmadji (2001)	Indonesia	Conference proceedings	Runoff plot	Natural	ST	1 (F)	3	Study	4 months
Syed Abdullah and Al-Toum (2000)	Malaysia	Journal article	Sediment trap	Natural	ST + OM	1 (F)	12	Year	1 year
van der Linden (1980)	Indonesia	Journal article	Runoff plot	Natural	ST + OM	3 (B, C, G)	88	Rainfall event, study	3 months
Verbist et al. (2010)	Indonesia	Journal article	Runoff plot	Natural	ST + OM	2 (F, T)	18	Year	4 years

Table 2 (*Continued*)

Reference	Country	Source type	Method	Rainfall type	Soil data[a]	Land-use type(s)[b]	Cases	Case time frame(s)	Study length
Caribbean islands ($n = 5$)									
Khamsouk (2001)	**France (Martinique)**	**PhD thesis**	**Runoff plot**	**Natural, simulation**	**ST + OM**	**3 (B, C, T)**	**429**	**Rainfall event**	**1.5 years**
Larsen et al. (1999)	**USA (Puerto Rico)**	**Journal article**	**Unbounded plot**	**Natural**	**ST**	**3 (B, G, S)**	**177**	**Month, season, year**	**3.75 years**
McDonald et al. (2002)	Jamaica	Journal article	Runoff plot	Natural	ST + OM	3 (B, C, F)	24	Year, study	5 years
Mohammed and Gumbs (1982)	Trinidad and Tobago	Journal article	Runoff plot	Natural	ST + OM	2 (B, C)	6	Rainfall event, season	3 months
Ramos Santana et al. (2003)	USA (Puerto Rico)	Journal article	Runoff plot	Natural	None	3 (B, G, T)	8	Month	1 month

[a] ST: soil texture; OM: organic matter.
[b] B: bare; C: cropland; G: grassland; F: forest; S: shrubland; T: tree-dominated agrosystem.

continuous variables (erosion, duration, area, rainfall, slope length and slope steepness) were $\log_{10}$-transformed to normalize their distribution. If not specified, further mention of values of these variables will refer to their $\log_{10}$-transformed values. Because null values cannot be $\log_{10}$-transformed, each null value of measured soil loss (664 values, expressed in g after transforming values reported in other units in the papers) was replaced by a random value taken from a uniform distribution in the range of 0.001–1 g, an interval arbitrarily chosen in which 1 g represents a measurement detection threshold (Chiappetta et al., 2004). After substituting the null values, measured soil loss (g) was converted into soil loss per unit of area and per year (g/m^2/yr). Replicating the substitution process 10 times, we checked that the randomness of the data replacement did not affect the subsequent results.

In order to analyze the effect of soil or vegetation management on soil erosion, we controlled first for the effect of the measurement protocol (method, duration and area) (Hair et al., 2006). Annual soil loss values obtained from extrapolation of measures taken over a single rain event are likely to be larger than values from measures over one year, and soil loss values per unit of area are probably higher in small plots than in larger areas because of sediment deposition (Boix-Fayos et al., 2006). We used only the two quantitative descriptors of measurement protocol (area and duration), as they were good proxies for method (60% correct determination, jackknifed classification following discriminant function analysis). We transformed the $\log_{10}$ values of soil loss and context variables (rainfall, soil erodibility, slope length and slope steepness) into the residuals resulting from a linear regression against duration, area and the interaction between the two variables (all three significant at $p < 0.001$; Table D1). Residuals were further adjusted to correspond to a reference protocol of measurements over one year and 100 m^2 (this value corresponding to the order of magnitude of the median area).

We then controlled for the effect of context on soil loss by calculating the residuals of a general linear model relating soil loss to context (values of rainfall, slope length and slope steepness, after factoring out the effects of protocol, as well as soil erodibility classes). All the context variables had a significant effect on soil loss ($p < 0.05$; Table D2). The residuals were adjusted to a "reference scenario" with the median values for annual rainfall (exclusively from cases where rainfall was measured for one year or more), slope length, slope steepness (back-transformed values being 2444 mm, 16.4 m and 16.5%, respectively), and a soil erodibility of class "medium".

All subsequent statistical analyses (ANOVA and Tukey's HSD) used these $\log_{10}$-transformed soil loss values, corrected for the effect of the measurement protocol and context and scaled to correspond to a reference scenario. We tested for differences (at $p < 0.001$) in soil loss depending on (1) land-use type, (2) land-use subtype and (3) the number and (4) nature of layers constituting the vegetation cover. As six references provided half the total number of cases, we tested whether they had a dominant effect on the overall results. To do so, we reanalyzed the data after removing these references one by one, but no significant changes in the results and no changes in the findings were observed.

3. Results

Soil loss was maximum on bare soils and strikingly exceeded that of all other land-use types (Fig. 3). Minimum soil loss was found in forests. Croplands had the second highest soil loss value among land-use types. Mean soil loss values for grasslands and shrublands were about half that of croplands. The ratio (of geometric means in the natural scale) shrank to 1:3 for mean soil loss between tree-dominated agrosystems and croplands. The erosion rate in forests was ca. one-tenth and one-150th than that of croplands and bare soils, respectively. The ratio of soil loss values between two consecutive land uses (sorted by decreasing mean soil loss) was much higher between bare soils and croplands (ca. 20:1) than between other land-use types (ratios below 3:1).

Soil loss differed significantly between subtypes of land uses within the same type. Soil loss was minimum for tree crops with contact cover (e.g. grass or cover crop) and maximum on tilled bare soils, with a ratio of 1:1,200 between the two values (Fig. 4). Among bare soils, soil loss was 40% higher with tillage than without (the latter still had a high absolute value of soil loss). Among croplands, recently planted crops without vegetation-related conservation practices (e.g. hedgerows, mulching or intercropping) had erosion rates similar to those of bare soils (either tilled or not), whereas well-established crops on similar lands reduced soil loss by 89% on average. Vegetation-related conservation practices reduced soil loss by 93% in recently planted cropland but did not reduce soil loss significantly in land with established crops. Simultaneous soil- and vegetation-related conservation practices (e.g. no-till farming and hedgerows) decreased soil loss in croplands (up to 99% compared to no conservation practices in land with recently planted crops).

Among tree-dominated agrosystems, tree crops with contact cover faced 99% less soil loss on average than tree crops without contact cover. Simple agroforests had greater soil loss than complex ones (3:1 ratio); however, the difference was not significant. Among the five least erosion-prone land-use subtypes, three were of forest type (old-growth, secondary, and logged-over forests).

The number of layers constituting the vegetation cover had a significant impact on soil loss. Soil loss was maximal without any layer and minimal with four layers. Soil loss was one-tenth as much with one layer as without, and one-70th as much with two layers as without (Fig. 5). The 90% reduction in soil loss between one and two layers was also significant. Conversely, no significant difference in mean soil loss was found between two and four layers.

The type of layers constituting the vegetation cover had a significant impact on soil loss. The presence of high, intermediary,

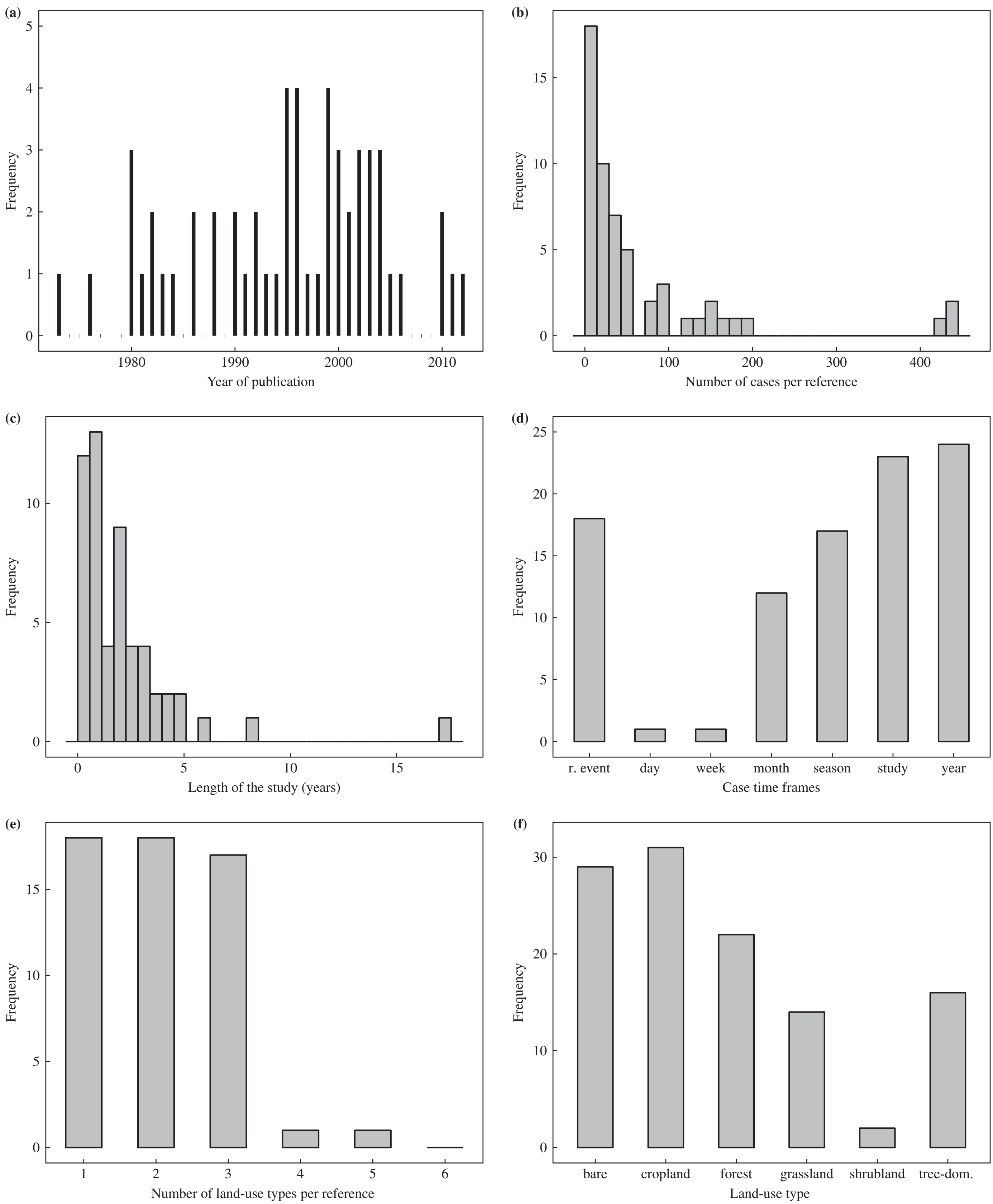

Fig. 2. Frequency distribution of (a) year of publication of the contributing references ($n = 55$), (b) number of cases per reference (total cases = 3649), (c) length of the study, (d) case time frames, (e) number of land-use types investigated per reference, (f) land-use types investigated. Total for (d) >55 because some references provide data on more than one time frame; total for (f) >55 because most references reported on more than one land use. R. event: rainfall event; tree-dom.: tree-dominated agrosystem.

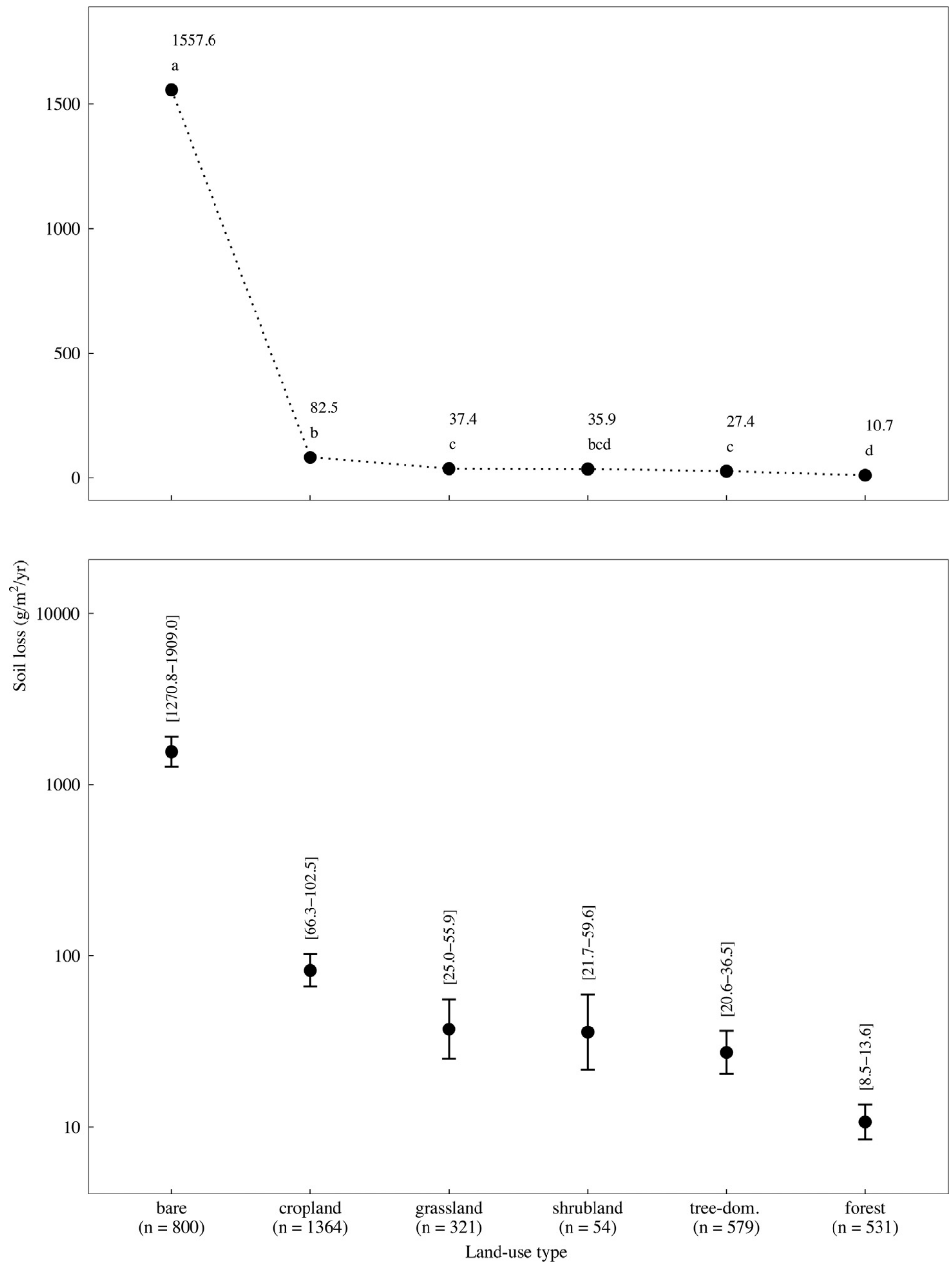

Fig. 3. Impact of land-use type on soil loss under reference scenario (significant difference at $p < 0.001$). Geometric means along with 95% confidence intervals on the natural scale are plotted on a $\log_{10}$ scale for the sake of readability (bottom panel). Log_{10}-transformed mean soil loss values with the same letter are not significantly different (Tukey's HSD, $p < 0.01$). Geometric means are also plotted on the natural scale to highlight the loss of soil erosion control from cropland to bare land (top panel). Tree-dom.: tree-dominated agrosystem.

low and ground layers influenced soil loss significantly and differently (Table 3): soil loss under a unique layer of high vegetation (≥4 m) was twice that occurring on bare soils, whereas other layers decreased soil loss compared to bare soils by a factor of 5, 8 and 5 for intermediary, low and ground layers respectively, and a factor of 200 for a combination of the three layers.

4. Discussion

4.1. Soil erosion is concentrated in space and time

Soil erosion control can abruptly be lost when vegetation cover is not developed enough and/or when poor soil and vegetation management practices are implemented (Figs. 3–5). While we found the ratio of soil loss values between bare soils and croplands to be ca. 20:1 in the humid tropics, the ratio ranged from 2:1 to 10:1 in Europe and the Mediterranean (Cerdan et al., 2010; Maetens et al., 2012). This suggests that soil erosion control is still provided in the humid tropics to a certain extent for crop- and grass-dominated land uses but is alarmingly depleted in bare soils, with dramatic consequences on soil loss. The 2-order-of-magnitude difference in soil loss between one and zero vegetation layer also suggests that some vegetation cover is necessary for soil erosion control to be provided. Consequently, bare soils should be avoided at all times.

The abrupt loss of soil erosion control depicted in Figs. 3–5 suggests that, in most land uses, erosion is concentrated spatially (over bare soil, e.g. logging roads or non-protected crop fields between rotations) and temporally (e.g. before vegetation is fully established). Soil loss was lowest in plots under tree crops with

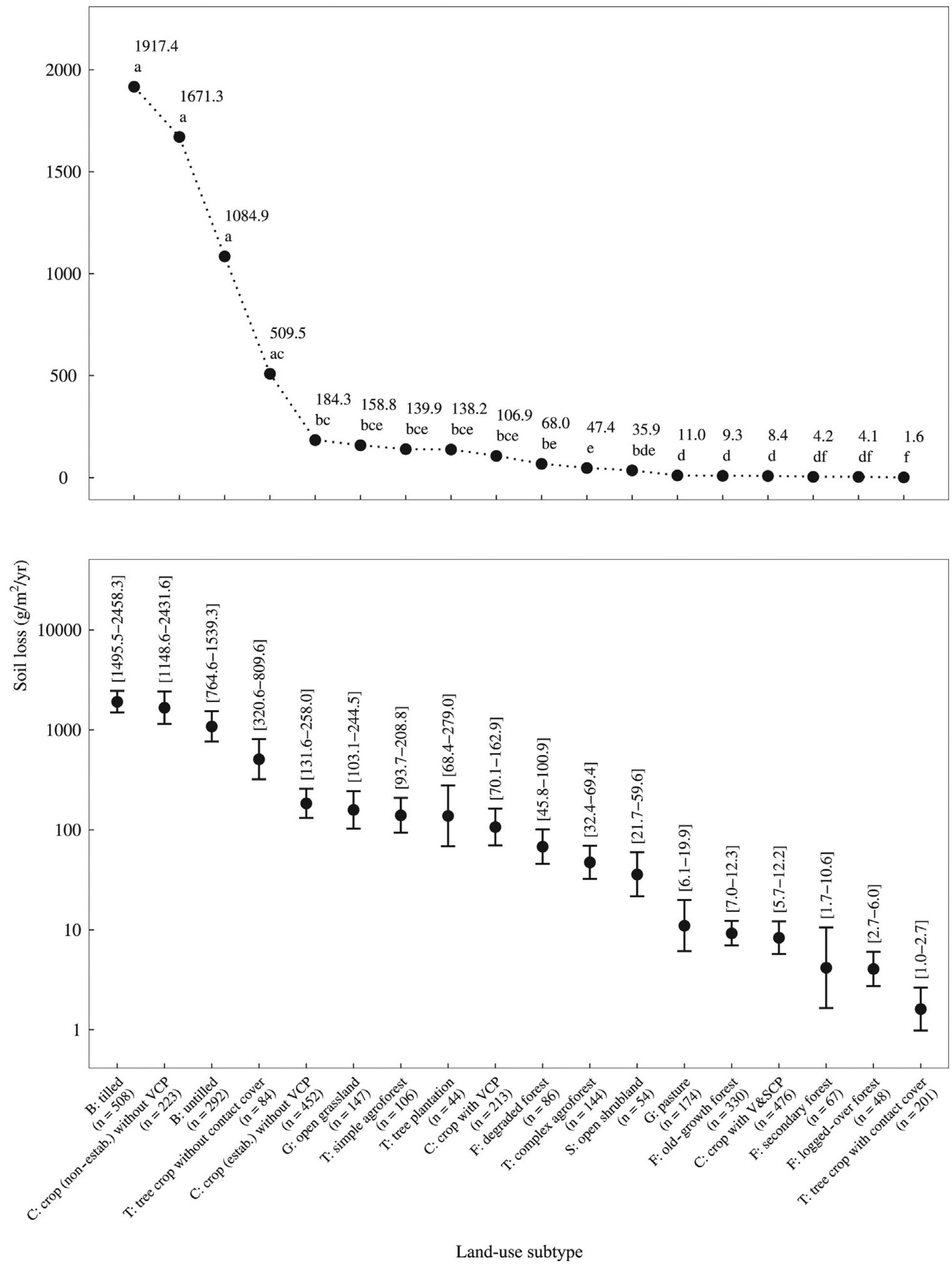

Fig. 4. Impact of land-use subtype on soil loss under reference scenario (significant difference at $p < 0.001$). Geometric means along with 95% confidence intervals on the natural scale are plotted on a $\log_{10}$ scale for the sake of readability (bottom panel). Log_{10}-transformed mean soil loss values with the same letter are not significantly different (Tukey's HSD, $p < 0.01$). Geometric means are also plotted on the natural scale to highlight the loss of soil erosion control from tree crops with contact cover to tilled bare soils (top panel). B: bare; C: cropland; G: grassland; F: forest; S: shrubland; T: tree-dominated agrosystem; estab.: established; VCP: vegetation-related conservation practice(s); V&SCP: vegetation- and soil-related conservation practice(s).

contact cover, but such crops might not be totally erosion-neutral. Similarly, the fact that soil loss in logged-over forests is not different from that in old-growth forests should not lead to the delusive conclusion that logging does not increase soil erosion. Bare soil elements exclusively related to logging and farming (e.g. roads and trails) contribute to disproportionately increase the overall erosion rate of such activities (e.g. Rijsdijk, 2005; Gómez-Delgado, 2010). Much attention should therefore be given to managing these elements (e.g. through water diversion, use of vegetative buffer strips and trail consolidation) so as to reduce the overall impact of such activities.

Attention must also be given to temporal transitions between land uses, for example when establishing crops or plantations. Although this finding has been reported before (Sarrailh, 1981; Baharuddin et al., 1995; Anderson and Macdonald, 1998; Bruijnzeel et al., 1998; Rijsdijk, 2005; Defersha and Melesse, 2012), our study brings a strong quantitative endorsement to it because of the number of studies and cases taken into consideration.

Studies investigating the consequences of land-use changes for soil erosion often used a synchronic approach (comparing different land uses in different plots to infer the consequences of a conversion, in a single plot, from one land use to the other). Unlike a diachronic approach measuring soil loss before, during and after land use change (e.g. Fritsch and Sarrailh, 1986; Malmer, 1996), a synchronic approach does not record the transition (e.g. through clear-cutting or tillage) from one land use to the other. This transition appears to be

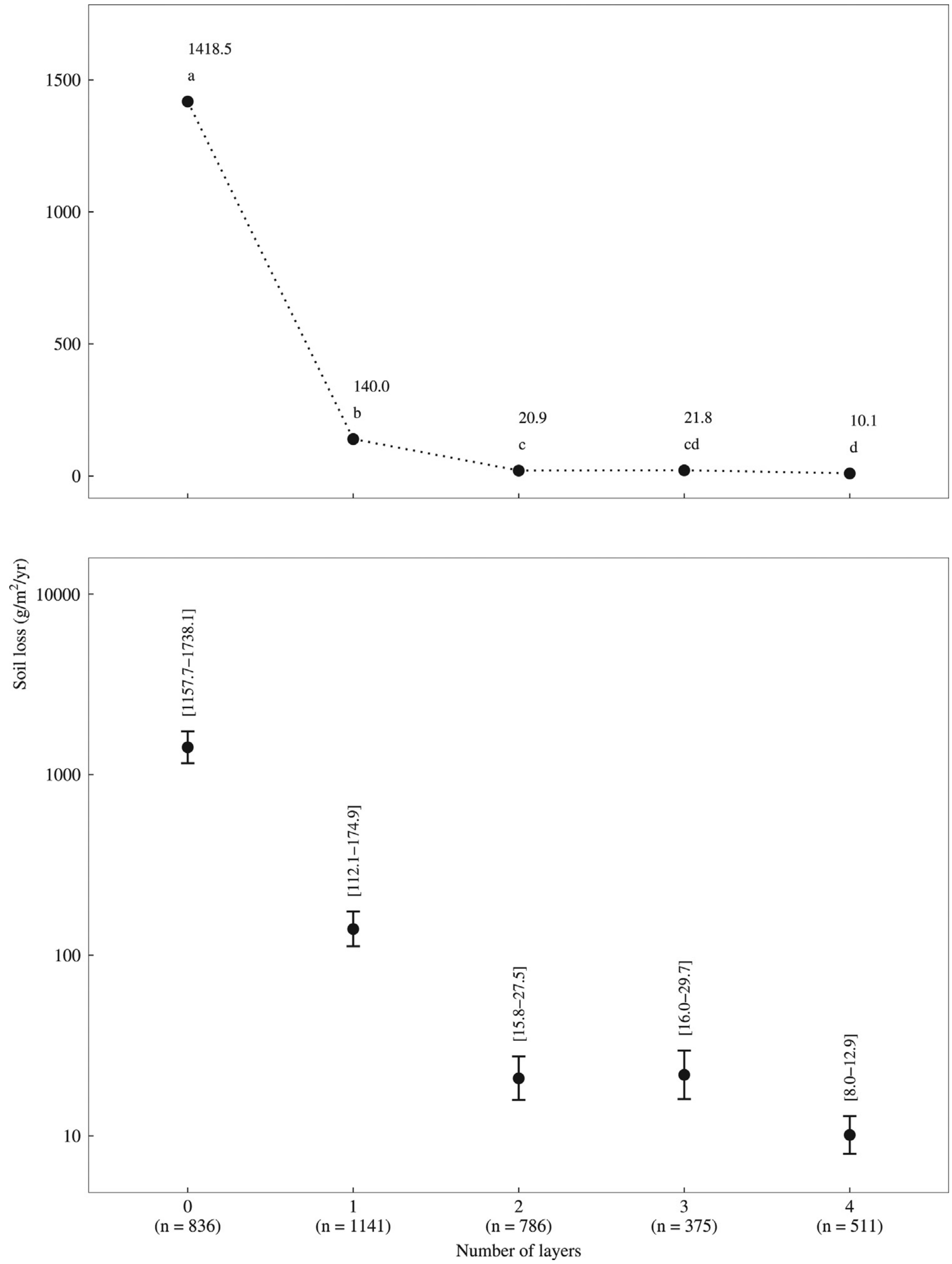

Fig. 5. Impact of the number of vegetation layers on soil loss under reference scenario (significant difference at $p < 0.001$). Geometric means along with 95% confidence intervals on the natural scale are plotted on a $\log_{10}$ scale for the sake of readability (bottom panel). Log_{10}-transformed mean soil loss values with the same letter are not significantly different (Tukey's HSD, $p < 0.01$). Geometric means are also plotted on the natural scale to highlight the loss of soil erosion control from one layer of vegetation to none (top panel).

critical for understanding the consequences of land-use changes for soil loss in the humid tropics, where vegetation regrowth is rapid but most of the annual soil loss is potentially caused by a limited number of extreme rainfall events (e.g. Poudel et al., 1999; Defersha and Melesse, 2012). Comparing synchronic and diachronic approaches for soil carbon sequestration assessment, Costa Junior et al. (2013) found that results depended on the selected approach, and recommended use of the diachronic approach whenever possible. Because of intrinsic variations in soil characteristics (e.g. texture) between sites under the same land use or management practice, a diachronic approach should always be preferred. On the other hand, a synchronic approach using multiple replicates makes it possible to highlight trends in the consequences of land use change or management.

In this respect, the sequence of land uses—bare untilled, cropland, open grassland, open shrubland, secondary forest and old-growth forest—can be interpreted as snapshots of different successional stages following shifting cultivation (after clearing, cultivation, and subsequent natural regeneration). This review showed that soil erosion decreased along the sequence, attesting to the recovery of soil erosion control. Martin et al. (2013) highlighted a similar increasing trend for carbon storage and plant diversity during post-disturbance forest recovery. This suggests a synergy (or a joint increase in multiple ecosystem services following implementation of a practice—forest regeneration in this case) between soil erosion control, carbon storage and plant diversity. But the evaluation of a wider range of ecosystem services (including e.g. water regulation) is advised so as to avoid

Table 3
Coefficients of the generalized linear model regression of annual soil loss ($\log_{10}$-transformed values) against presence/absence of high ($\geq$4 m), intermediate (1 m $\leq$ height < 4 m), low (0.1 m $\leq$ height < 1 m) and ground (<0.1 m) vegetation layers.

	Estimate	Standard error	*p*
Intercept (bare)	2.97	0.044	***
High	0.22	0.071	**
Intermediary	−0.66	0.054	***
Low	−0.91	0.058	***
Ground	−0.71	0.068	***
Adjusted R^2: 0.204			
Number of observations: 3649			

** $p < 0.01$.
*** $p < 0.001$.

promoting measures (e.g. afforestation) that would be detrimental for the delivery of other services.

4.2. What matters in soil erosion control by vegetation?

The change of slope in Fig. 4 highlights four land uses in which soil erosion control is depleted. In addition to two situations of bare soils, recently planted croplands without vegetation-related conservation practices also provide a low level of soil erosion control. This highlights the importance of good management of croplands: vegetation-related conservation practices (such as hedgerows) can ensure that, even during inter- or early-rotation periods when crop cover is not yet developed, erosion can be prevented or minimized.

Tree crops without contact cover also provide critically low levels of soil erosion control, which is confirmed by the analysis of the effect of vegetation layers: the presence of a sole high layer increases erosion compared to bare soil. This is consistent with other studies that pointed out the role of tree canopy in modifying rainfall kinetic energy (e.g. Wiersum, 1985; Brandt, 1988; Calder, 2001). Leaves of the canopy layer help break the kinetic energy of raindrops, but secondary drops falling from the canopy (particularly from large leaves) are often larger than the raindrops and reach the ground with a higher kinetic energy than in areas without a canopy layer (Wiersum, 1985; Brandt, 1988). This results in increased soil erosion, particularly when the canopy is high and there is no understorey vegetation. Teak (*Tectonia grandis* L.f.) plantations, for example, have often been associated with high erosion rates because of lack of understorey and large tree leaves (Calder, 2001). But a recent study showed that poor vegetation and soil management rather than intrinsic teak leaf morphology was responsible for those high erosion rates (Fernández-Moya et al., 2014).

Litter and understorey both help break the kinetic energy of raindrops and therefore decrease splash erosion (Brandt, 1988). Multiple layers of vegetation are necessary in plantations to minimize soil erosion, and non-compliance with sound management rules (e.g. the repeated use of fire to clear ground cover and understorey) directly and dramatically increases soil loss (Wiersum, 1984). Overall, whatever the land use, we found low and ground layers of vegetation to be essential in decreasing soil loss (Table 3). This is consistent with plot-derived results from northern Vietnam, which identified a critical value of understorey biomass (130 g/m^2) above which soil loss was negligible (Anh et al., 2014). Therefore, low and ground covers should be restored and/or maintained whatever the land use.

4.3. Soil erosion under human-impacted or managed vs. natural vegetation

This study also showed that the difference between "human-impacted or managed" and "natural" vegetation does not explain soil loss in the humid tropics (although intuitively one would expect lower soil erosion under natural vegetation). For example, we found that soil loss in old-growth forest is higher than in tree crops with contact cover. Soil erosion is a natural phenomenon that also occurs in old-growth forest despite its complex vegetation structure and high ground cover (mostly leaf litter or wood debris). In Tanzania, Lundgren (1980) suggested that good land management practices (e.g. mulching and no burning) accounted for lower erosion rates in agrosystems than in natural forest, even though this observation was made during normal rainfall conditions and it was impossible to predict how the human-managed system would have reacted to extreme rainfall events. In South Andaman island, Pandey and Chaudhari (2010) showed that coconut plantations with a contact cover of *Pueraria phaseoloides* had similar soil loss as nearby native evergreen forest and therefore recommended the use of contact cover in plantations for soil erosion control on the island.

Our quantitative analysis strongly supports the idea that no land use (except bare soils) is erosion-prone per se and that sound management of soil and vegetation can reduce soil erosion in managed areas to levels even lower than in areas under natural vegetation.

4.4. Differences in soil erosion control between tropical vs. temperate regions

Comparing the effect of land use on soil erosion in the humid tropics (this review) and in temperate regions (Renard et al., 1997; Burke and Sugg, 2006), we found that changes in soil erosion control along a gradient of land uses had similar shape in both temperate and tropical areas (Fig. 6). A difference between these climatic zones is observed in grasslands and croplands, where soil erosion control is higher in the humid tropics than suggested by the RUSLE. Our analysis shows a much more pronounced threshold effect in the relation between vegetation and soil erosion control than given by the RUSLE, which suggests that soil erosion is more concentrated in space and time in the humid tropics than elsewhere. The difference can be explained by the more rapid development of dense vegetation protecting soil in croplands and grasslands of the humid tropics. Because of the "universal" nature of the mechanism of soil erosion, the RUSLE, an empirically-based model that integrates all the factors known to influence soil erosion (e.g. soil erodibility, rainfall erosivity), could potentially be

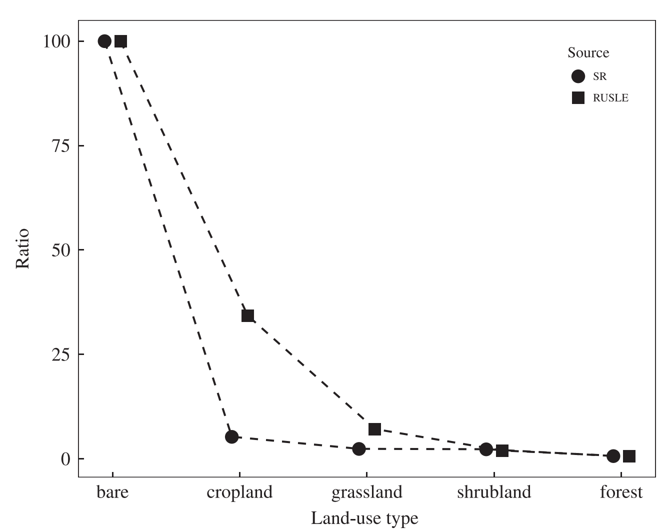

Fig. 6. Ratio of cover-management factors for the RUSLE for 5 different land uses (reference being erosion on bare soils), and ratio of soil loss per land use to soil loss on bare soils from our systematic review (SR).

used to predict soil erosion for any geographical context. But factors' parameters were computed from data collected exclusively in temperate regions and the direct application of the RUSLE to a tropical context would lead to soil loss misestimation especially for croplands and grasslands. Properly calibrating all RUSLE factors' parameters (especially those related to soil and vegetation management) using data acquired in a tropical context is therefore critical to achieve accurate prediction of soil erosion in the humid tropics.

4.5. Limitations of the study

This analysis faced challenges related to data availability. As soils were sometimes poorly described, we had to use a global database to estimate texture and carbon content, which probably influenced the accuracy of our soil erodibility indices. The structure of the vegetation cover (e.g. number and height of layers, planting density and presence or absence of ground cover) was not always well described. For example, Sinun et al. (1992) studied an abandoned logging track where a sharp decrease in soil loss was recorded over time; but while soil loss was measured on a monthly basis over one year, vegetation was not described over time. Two noticeable exceptions were Khamsouk (2001) and Presbitero (2003), in which vegetation cover was regularly and systematically estimated, but with different approaches (e.g. crown cover and contact cover).

The aim of this study was to quantitatively analyze soil erosion control in the whole humid tropics, but references only covered 21 countries and some sub-regions were critically under-represented, e.g. the Brazilian part of the Amazon and the Congo basin (Fig. 1, Table 2). Yet Köppen climatic classes "Af" and "Am" are homogeneous in term of temperature, rainfall pattern and vegetation type (Köppen, 1936), which supports the applicability of this study's findings to under-represented sub-regions. Research should nevertheless be carried out in the Amazon and the Congo basin to document the effect of local human activities (e.g. small- and large-scale agriculture, fuelwood collection and industrial logging) on soil erosion.

Because six references (from four countries) represented half the total number of cases, we tested for their dominant effect on the overall results, but no such effect was found; this further supports the relevance of this study to the whole humid tropics. Mean annual soil loss values in this study appeared to be in the line of benchmarks provided by other studies. For example, annual erosion rates ranged from 0.1 to 90 and 3 to 750 Mg/ha in humid West Africa for croplands and bare soils, respectively (Morgan, 2005), compared to 1 and 16 Mg/ha on average in our analysis. Other benchmarks are 0.03 to 6.2, 0.1 to 5.6, and 1.2 to 183 Mg/ha for old-growth forests and tree crops with and without contact cover, respectively (Wiersum, 1984), compared to 0.1, 2 and 5 Mg/ha in our analysis.

Since we used $\log_{10}$-transformed data to carry out statistical analyses, back-transforming means led to geometric means in the natural scale that are intrinsically less sensitive to extreme values (Bland and Altman, 1996). This explains the fact that our values lie in the lower part of the range.

5. Conclusion

Soil erosion in the humid tropics is dramatically concentrated both spatially (over bare soil) and temporally (before vegetation cover establishes), and low and ground layers of vegetation are essential in mitigating soil erosion. Because soil erosion appears more concentrated in space and time in the humid tropics than elsewhere, models developed in temperate regions should not be directly applied in the humid tropics, and thorough research should be conducted to calibrate model parameters. As a preliminary step to answer the UN call for action to reverse land degradation (UN, 2012), we stress the need to establish standard measurement procedures for soil erosion and influencing factors, to mirror what was achieved for terrestrial carbon measurement (Walker et al., 2012). For improving soil and vegetation management, uncovered or unprotected soils should be avoided at all times, and low and ground layers of vegetation should be restored and/or maintained whatever the land use.

No land use (except bare soils) is erosion-prone per se and natural resource managers and policy makers need to promote sound management of soil and vegetation (e.g. contour planting, no-till farming, intercropping and use of cover crops) to reduce soil loss from erosion-prone landscape elements. Because of the relative affordability and simplicity of such management practices, substantial decrease in soil loss can be attained at the catchment or regional scale with limited financial and technical means. Since soil erosion appears to decrease during the different phases of forest regeneration, soil ecosystem services (e.g. nutrient cycling, flood regulation, water purification), the delivery of which is greater in healthier soils, might be good candidates for ecosystem services bundling with biodiversity protection and carbon storage.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2015.01.027.

References

Afandi, Manik, T.K., Rosadi, B., Utomo, M., Senge, M., Adachi, T., Oki, Y., 2002a. Soil erosion under coffee trees with different weed management in humid tropical hilly area of Lampung, South Sumatra, Indonesia. J. Jpn. Soc. Soil Phys. 91, 13–14.

Afandi, Rosadi, B., Maryanto, Nurarifani, Utomo, M., Senge, M., Adachi, T., 2002b. Sediment yield from various land use practices in a hilly tropical area of the Lampung region, South Sumatra, Indonesia. J. Jpn. Soc. Soil Phys. 91, 25–38.

Alegre, J.C., Cassel, D.K., 1996. Dynamics of soil physical properties under alternative systems to slash-and-burn. Agric. Ecosyst. Environ. 58, 39–48.

Alegre, J.C., Rao, M.R., 1996. Soil and water conservation by contour hedging in the humid tropics of Peru. Agric. Ecosyst. Environ. 57, 17–25.

Almas, M., Jamal, T., 2000. Use of RUSLE for soil loss prediction during different growth periods. Pak. J. Biol. Sci. 3, 118–121.

Ambassa-Kiki, R., Nill, D., 1999. Effects of different land management techniques on selected topsoil properties of a forest Ferralsol. Soil Tillage Res. 52, 259–264.

Anderson, D.M., Macdonald, L.H., 1998. Modelling road surface sediment production using a vector geographic information system. Earth Surf. Processes Landforms 23, 95–107.

Angima, S.D., Stott, D.E., O'Neill, M.K., Ong, C.K., Weesies, G.A., 2003. Soil erosion prediction using RUSLE for central Kenyan highland conditions. Agric. Ecosyst. Environ. 97, 295–308.

Anh, P.T.Q., Gomi, T., MacDonald, L.H., Mizugaki, S., Khoa, P.V., Furuichi, T., 2014. Linkages among land use, macronutrient levels, and soil erosion in northern Vietnam: a plot-scale study. Geoderma 232, 352–362.

Baharuddin, K., Mokhtaruddin, A.M., Nik Muhamad, M., 1995. Surface runoff and soil loss from a skid trail and a logging road in a tropical forest. J. Trop. For. Sci. 7, 558–569.

Bellanger, B., Huon, S., Velasquez, F., Vallès, V., Girardin, C., Mariotti, A., 2004. Monitoring soil organic carbon erosion with δ13C and δ15N on experimental field plots in the Venezuelan Andes. Catena 58, 125–150.
Bland, J.M., Altman, D.G., 1996. Transformations, means, and confidence intervals. Br. Med. J. 312, 1079.
Boix-Fayos, C., Martínez-Mena, M., Arnau-Rosalén, E., Calvo-Cases, A., Castillo, V., Albaladejo, J., 2006. Measuring soil erosion by field plots: understanding the sources of variation. Earth Sci. Rev. 78, 267–285.
Bons, C.A., 1990. Accelerated erosion due to clearcutting of plantation forest and subsequent Taungya cultivation in upland West Java, Indonesia. In: Ziemer, R.R., O'Loughlin, C.L., Hamilton, L.S., International Union of Forestry Research Organizations (Eds.), Research Needs and Applications to Reduce Erosion and Sedimentation In Tropical Steeplands. IAHS Publication No. 192. International Association of Hydrological Sciences, Wallingford, Oxfordshire, UK, pp. 279–288.
Boye, A., Albrecht, A., 2004. Soil erodibility control and soil carbon losses under short term tree fallows in western Kenya. Bull. Réseau Eros. 23, 123–143.
Brandt, J., 1988. The transformation of rainfall energy by a tropical rain-forest canopy in relation to soil-erosion. J. Biogeogr. 15, 41–48.
Bruijnzeel, L.A., Van Eijk, B., Purwanto, E., 1998. Runoff and soil loss from bench terraces in upland West Java, Indonesia and implications for process modelling. In: Summer, W., Klaghofer, E., Zhang, W. (Eds.), Modelling Soil Erosion, Sediment Transport and Closely Related Hydrological Processes. IAHS Publication No. 249. International Association of Hydrological Sciences, Wallingford, Oxfordshire, UK, pp. 211–220.
Burke, L., Sugg, Z., 2006. Hydrologic modeling of watersheds discharging adjacent to the Mesoamerican Reef. World Resources Institute, Washington, DC.
Calder, I.R., 2001. Canopy processes: implications for transpiration, interception and splash induced erosion, ultimately for forest management and water resources. Plant Ecol. 153, 203–214.
Cerdan, O., Govers, G., Le Bissonnais, Y., Van Oost, K., Poesen, J., Saby, N., Gobin, A., Vacca, A., Quinton, J., Auerswald, K., Klik, A., Kwaad, F.J.P.M., Raclot, D., Ionita, I., Rejman, J., Rousseva, S., Muxart, T., Roxo, M.J., Dostal, T., 2010. Rates and spatial variations of soil erosion in Europe: a study based on erosion plot data. Geomorphology 122, 167–177.
Chartier, M.P., Rostagno, C.M., 2006. Soil erosion thresholds and alternative states in northeastern patagonian rangelands. Rangeland Ecol. Manage. 59, 616–624.
Chatterjea, K., 1998. The impact of tropical rainstorms on sediment and runoff generation from bare and grass-covered surfaces: a plot study from Singapore. Land Degrad. Dev. 9, 143–157.
Chiappetta, P., Roubaud, M.C., Torrésani, B., 2004. Blind source separation and the analysis of microarray data. J. Comput. Biol. 11, 1090–1109.
Chomitz, K.M., Kumari, K., 1998. The domestic benefits of tropical forests: a critical review. World Bank Res. Obser. 13, 13–35.
Cohen, M.J., Shepherd, K.D., Walsh, M.G., 2005. Empirical reformulation of the universal soil loss equation for erosion risk assessment in a tropical watershed. Geoderma 124, 235–252.
Collaboration for Environmental Evidence, 2013. Guidelines for Systematic Review and Evidence Synthesis in Environmental Management. Version 4.2. Environmental Evidence: www.environmentalevidence.org/Authors.htm.
Collinet, J., 1983. Hydrodynamique superficielle et érosion comparées des sols représentatifs des sites forestiers et cultivés de la station écologique de Taï (Sud-Ouest ivoirien): premier bilan sur parcelles expérimentales recevant des pluies naturelles (campagnes 1978-1979-1980) et simulées (campagne de novembre 1978 et mars 1979). Cahiers ORSTOM.
Collinet, J., 1988. Comportements hydrodynamiques et érosifs de sols de l'Afrique de l'Ouest: évolution des matériaux et des organisations sous simulations de pluies. Strasbourg, France.
Comia, R.A., Paningbatan, E.P., Håkansson, I., 1994. Erosion and crop yield response to soil conditions under alley cropping systems in the Philippines. Soil Tillage Res. 31, 249–261.
Costa Junior, C., Corbeels, M., Bernoux, M., Piccolo, M.C., Neto, M.S., Feigl, B.J., Cerri, C.E.P., Cerri, C.C., Scopel, E., Lal, R., 2013. Assessing soil carbon storage rates under no-tillage: comparing the synchronic and diachronic approaches. Soil Tillage Res. 134, 207–212.
Dangler, E.W., El-Swaify, S.A., 1976. Erosion of selected Hawaii soils by simulated rainfall. Soil Sci. Soc. Am. J. 40, 769–773.
Daño, A.M., Siapno, F.E., 1992. The effectiveness of soil conservation structures in steep cultivated mountain regions of the Philippines. In: Walling, D.E., Davies, T. R.H., Hasholt, B. (Eds.), Erosion, Debris flows and Environment in Mountain Regions. IAHS Publication No. 209. International Association of Hydrological Sciences, Wallingford, Oxfordshire, UK, pp. 399–405.
Defersha, M.B., Melesse, A.M., 2012. Field-scale investigation of the effect of land use on sediment yield and runoff using runoff plot data and models in the Mara River basin, Kenya. Catena 89, 54–64.
Diodato, N., Knight, J., Bellocchi, G., 2013. Reduced complexity model for assessing patterns of rainfall erosivity in Africa. Global Planet. Change 100, 183–193.
El-Swaify, S.A., Dangler, E.W., Armstrong, C.L., 1982. Soil erosion by water in the tropics. College of Tropical Agriculture and Human Resources. University of Hawaii, Honolulu.
Fernández-Moya, J., Alvarado, A., Forsythe, W., Ramírez, L., Algeet-Abarquero, N., Marchamalo-Sacristán, M., 2014. Soil erosion under teak (*Tectona grandis* L.f.) plantations: general patterns, assumptions and controversies. Catena 123, 236–242.
Francisco-Nicolas, N., Turrent-Fernandez, A., Oropeza-Mota, J.L., Martinez-Menes, M.R., Cortes-Flores, J.I., 2006. Soil loss and erosion-productivity relationships in four soil management systems. Terra Latinoamericana 24, 253–260.
Fritsch, J.M., Sarrailh, J.M., 1986. Les transports solides dans l'écosystème forestier tropical humide guyanais: effets du défrichement et de l'aménagement de pâturages. Cah. ORSTOM, Sér Pédol. 22, 209–222.
Gómez-Delgado, F., 2010. Hydrological, Ecophysiological and Sediment Processes in a Coffee Agroforestry Basin: Combining Experimental and Modelling Methods to Assess Hydrological Environmental Services. Montpellier, France.
Goujon, P., 1968. Conservation des sols en Afrique et à Madagascar: 1ère partie: les facteurs de l'érosion et l'équation universelle de Wischmeier. Bois For. Trop. 118, 3–17.
Hair, J.F., Tatham, R.L., Anderson, R.E., Black, W., 2006. Multivariate Data Analysis, 6th Ed Pearson Prentice Hall, Upper Saddle River, NJ.
Hartanto, H., Prabhu, R., Widayat, A.S.E., Asdak, C., 2003. Factors affecting runoff and soil erosion: plot-level soil loss monitoring for assessing sustainability of forest management. For. Ecol. Manage. 180, 361–374.
Hashim, G.M., Ciesiolka, C.A.A., Yusoff, W.A., Nafis, A.W., Mispan, M.R., Rose, C.W., Coughlan, K.J., 1995. Soil erosion processes in sloping land in the east coast of Peninsular Malaysia. Soil Technol. 8, 215–233.
Hoyos, N., 2005. Spatial modeling of soil erosion potential in a tropical watershed of the Colombian Andes. Catena 63, 85–108.
ISRIC - World Soil Information, 2013. SoilGrids: an automated system for global soil mapping. Available for download at http://soilgrids1km.isric.org.
Jaafar, O., Syed Abdullah, S.M., Al-Toum, S., 2011. The Tekala Forest Reserve: a study on surface wash and runoff using close system erosion plots. Geogr. Malay. J. Soc. Space 7, 1–13.
Kamara, C.S., 1986. Mulch-tillage effects on soil loss and soil properties on an ultisol in the humid tropics. Soil Tillage Res. 8, 131–144.
Khamsouk, B., 2001. Impact de la culture bananière sur l'environnement: influence des systèmes de cultures bananières sur l'érosion, le bilan hydrique et les pertes en nutriments sur un sol volcanique en Martinique (cas du sol brun rouillé à halloysite). Montpellier, France.
Köppen, W., 1936. Das geographisca System der Klimate. In: Köppen, W., Geiger, G. (Eds.), Handbuch der klimatologie. Gebrüder Borntraeger, Berlin, Germany, pp. 1–44.
Lal, R., 1990. Soil erosion in the tropics: principles and management. McGraw-Hill, New York.
Lal, R., 2003. Soil erosion and the global carbon budget. Environ. Int. 29, 437–450.
Larsen, M.C., Torres-Sánchez, A.J., Concepción, I.M., 1999. Slopewash, surface runoff and fine-litter transport in forest and landslide scars in humid-tropical steeplands luquillo experimental forest, Puerto Rico. Earth Surf. Processes Landforms 24, 481–502.
Leigh, C., 1982. Sediment transport by surface wash and throughflow at the Pasoh Forest Reserve Negri Sembilan, Peninsular Malaysia. Geogr. Ann. Ser. A. Phys. Geogr. 64, 171–180.
Locatelli, B., Imbach, P., Vignola, R., Metzger, M.J., Hidalgo, E.J.L., 2011. Ecosystem services and hydroelectricity in Central America: modelling service flows with fuzzy logic and expert knowledge. Reg. Environ. Change 11, 393–404.
Locatelli, B., Imbach, P., Wunder, S., 2014. Synergies and trade-offs between ecosystem services in Costa Rica. Environ. Conserv. 41, 27–36.
Lundgren, L., 1980. Comparison of surface runoff and soil loss from runoff plots in forest and small-scale agriculture in the Usambara Mts. Tanzania. Geogr. Ann. Ser. A. Phys. Geogr. 62, 113–148.
Maetens, W., Vanmaercke, M., Poesen, J., Jankauskas, B., Jankauskien, G., Ionita, I., 2012. Effects of land use on annual runoff and soil loss in Europe and the Mediterranean: a meta-analysis of plot data. Prog. Phys. Geogr. 36, 597–651.
Malmer, A., 1996. Hydrological effects and nutrient losses of forest plantation establishment on tropical rainforest land in Sabah, Malaysia. J. Hydrol. 174, 129–148.
Martin, P.A., Newton, A.C., Bullock, J.M., 2013. Carbon pools recover more quickly than plant biodiversity in tropical secondary forests. Proc. R. Soc. Lond., Ser. B: Biol. Sci. 280, 20132236.
McDonald, M.A., Healey, J.R., Stevens, P.A., 2002. The effects of secondary forest clearance and subsequent land-use on erosion losses and soil properties in the Blue Mountains of Jamaica. Agric. Ecosyst. Environ. 92, 1–19.
McGregor, D.F.M., 1980. An investigation of soil erosion in the Colombian rainforest zone. Catena 7, 265–273.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Synthesis. Island Press, Washington, DC.
Moehansyah, H., Maheshwari, B.L., Armstrong, J., 2004. Field Evaluation of Selected Soil Erosion Models for Catchment Management in Indonesia. Biosys. Eng. 88, 491–506.
Moench, M., 1991. Soil erosion under a successional agroforestry sequence: a case study from Idukki District, Kerala, India. Agrofor. Syst. 15, 31–50.
Mohammed, A., Gumbs, F.A., 1982. The effect of plant spacing on water runoff: soil erosion and yield of maize (*Zea mays* L.) on a steep slope of an ultisol in Trinidad. J. Agric. Eng. Res. 27, 481–488.
Montgomery, D.R., 2007. Soil erosion and agricultural sustainability. Proc. Natl. Acad. Sci. USA 104, 13268–13272.
Morgan, R.P.C., 2005. Soil Erosion and Conservation. Wiley-Blackwell, Oxford, UK.
Nearing, M.A., Lane, L.J., Lopes, V.L., 1994. Modeling soil erosion. In: Lal, R. (Ed.), Soil Erosion: Research Methods. Soil and Water Conservation Society and St. Lucie Press, Ankeny, Iowa, pp. 127–156.
Ngatunga, E.L.N., Lal, R., Uriyo, A.P., 1984. Effects of surface management on runoff and soil erosion from some plots at Mlingano, Tanzania. Geoderma 33, 1–12.
Odemerho, F.O., Avwunudiogba, A., 1993. The effects of changing cassava management practices on soil loss: a Nigerian example. Geogr. J. 159, 63–69.
Pandey, C.B., Chaudhari, S.K., 2010. Soil and nutrient losses from different land uses and vegetative methods for their control on hilly terrain of South Andaman. Indian J. Agric. Sci. 80, 399–404.

Paningbatan, E.P., Ciesiolka, C.A.A., Coughlan, K.J., Rose, C.W., 1995. Alley cropping for managing soil erosion of hilly lands in the Philippines. Soil Technol. 8, 193–204.
Peel, M.C., Finlayson, B.L., McMahon, T.A., 2007. Updated world map of the Köppen-Geiger climate classification. Hydrol. Earth Syst. Sci. 11, 1633–1644.
Poudel, D.D., Midmore, D.J., West, L.T., 1999. Erosion and productivity of vegetable systems on sloping volcanic ash-derived Philippine soils. Soil Sci. Soc. Am. J. 63, 1366–1376.
Poudel, D.D., Midmore, D.J., West, L.T., 2000. Farmer participatory research to minimize soil erosion on steepland vegetable systems in the Philippines. Agric. Ecosyst. Environ. 79, 113–127.
Presbitero, A.L., 2003. Soil Erosion Studies on Steep Slopes of Humid-tropic Philippines. Griffith, Australia.
Prove, B.G., Doogan, V.J., Truong, P.N.V., 1995. Nature and magnitude of soil erosion in sugarcane land on the wet tropical coast of north-eastern Queensland. Aust. J. Exp. Agric. 35, 641–649.
R Core Team, 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org/.
Ramos Santana, R., Martínez, G., Macchiavelli, R., Rodríguez, J.E., Guzmán, J.L., 2003. Potential of trees grasses, and turf legumes for restoring eroded soils. Commun. Soil Sci. Plant Anal. 34, 2149–2162.
Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D., Yoder, D., 1997. Predicting soil erosion by water: a guide to conservation planning with the revised universal soil loss equation (RUSLE). Agriculture Handbook No. 703. U.S. Department of Agriculture, Washington DC.
Rijsdijk, A., 2005. Evaluating sediment sources and delivery in a tropical volcanic watershed. In: Horowitz, A.J., Walling, D.E. (Eds.), Sediment Budgets 1. IAHS Publication No. 291. International Association of Hydrological Sciences, Wallingford, Oxfordshire, UK, pp. 1–9.
Roose, E., 1973. Dix-sept années de mesures expérimentales de l'érosion et du ruissellement sur un sol ferrallitique sableux de basse Côte d'Ivoire: contribution à l'étude de l'érosion hydrique en milieu intertropical. Abidjan, Côte d'Ivoire.
Ross, S.M., Dykes, A., 1996. Soil conditions: erosion and nutrient loss on steep slopes under mixed dipterocarp forest in Brunei Darussalam. Monogr. Biol. 74, 259–270.
Ruppenthal, M., Leihner, D.E., Steinmüller, N., El-Sharkawy, M.A., 1997. Losses of organic matter and nutrients by water erosion in cassava-based cropping systems. Exp. Agric. 33, 487–498.
Saatchi, S.S., Harris, N.L., Brown, S., Lefsky, M., Mitchard, E.T.A., Salas, W., Zutta, B.R., Buermann, W., Lewis, S.L., Hagen, S., Petrova, S., White, L., Silman, M., Morel, A., 2011. Benchmark map of forest carbon stocks in tropical regions across three continents. Proc. Natl. Acad. Sci. USA 108, 9899–9904.
Sarrailh, J.M., 1981. Parcelles élémentaires d'étude du ruissellement et de l'érosion: analyse des résultats obtenus durant les deux premières campagnes de mesure. Ecosyst. For. Guyan. Bull. Liaison Groupe Trav. ECEREX 4, 45–51.
Sharpley, A.N., Williams, J.R., 1990. EPIC-erosion/productivity impact calculator: 1. Model documentation. USDA Technical Bulletin No. 1768. U.S. Department of Agriculture, Washington, DC.
Shimokawa, E., 1988. Effect of a fire of tropical rainforest on soil erosion. In: Tagawa, H., Nirawan, N. (Eds.), A Research on the Process of Earlier Recovery of Tropical Rainforest after a Large Scale Fire in Kalimantan Timur, Indonesia. Occasional paper N°14, Kagoshima University, Research Center for the South Pacific, Japan, pp. 2–11.
Sidle, R.C., Ziegler, A.D., Negishi, J.N., Nik, A.R., Siew, R., Turkelboom, F., 2006. Erosion processes in steep terrain – truths myths, and uncertainties related to forest management in Southeast Asia. For. Ecol. Manage. 224, 199–225.
Siebert, S.F., Belsky, J.M., 1990. Bench terracing in the Kerinci uplands of Sumatra, Indonesia. J. Soil Water Conserv. 45, 559–562.
Sinun, W., Meng, W.W., Douglas, I., Spencer, T., 1992. Throughfall, stemflow, overland flow and throughflow in the Ulu Segama Rain Forest, Sabah, Malaysia. Philos. Trans. R. Soc. Lond., Ser. B: Biol. Sci. 335, 389–395.
Snelder, D.J., Bryan, R.B., 1995. The use of rainfall simulation tests to assess the influence of vegetation density on soil loss on degraded rangelands in the Baringo District, Kenya. Catena 25, 105–116.
Stewart, B.A., Woolhiser, D.A., Wischmeier, W.H., Caro, J.H., Frere, M.H., 1975. Control of Water Pollution from Cropland. U.S. Environmental Protection Agency, Washington, DC.
Strassburg, B.B.N., Kelly, A., Balmford, A., Davies, R.G., Gibbs, H.K., Lovett, A., Miles, L., Orme, C.D.L., Price, J., Turner, R.K., Rodrigues, A.S.L., 2010. Global congruence of carbon storage and biodiversity in terrestrial ecosystems. Conserv. Lett. 3, 98–105.
Streck, E.V., Cogo, N.P., 2003. Reconsolidation of the soil surface after tillage discontinuity, with and without cultivation, related to erosion and its prediction with RUSLE. Rev. Bras. Cienc. Solo 27, 141–151.
Sudarmadji, T., 2001. Impact of logging and forest fires on soil erosion in tropical humid forest in Kalimantan. Rehabilitation of Degraded Tropical Forest Ecosystems: Workshop Proceedings, 2–4 November 1999, Center for International Forestry Research, Bogor, Indonesia, pp. 35–44.
Syed Abdullah, S.M., Al-Toum, S., 2000. A study on surface wash and runoff using open system erosion plots. Pertanika J. Trop. Agric. Sci. 23, 43–53.
Torri, D., Poesen, J., Borselli, L., 1997. Predictability and uncertainty of the soil erodibility factor using a global dataset. Catena 31, 1–22.
Tropek, R., Sedlacek, O., Beck, J., Keil, P., Musilova, Z., Simova, I., Storch, D., 2014. Comment on High-resolution global maps of 21st-century forest cover change. Science 344, 981.
UN, 2012. Conference on sustainable development, Outcome of the conference, A/CONF.216/L.1. Rio de Janeiro, Brazil.
Våje, P.I., Singh, B.R., Lal, R., 2005. Soil erosion and nutrient losses from a volcanic ash soil in Kilimanjaro Region, Tanzania. J. Sustainable Agric. 26, 95–117.
van der Linden, P., 1980. The application of a parametric grey box type approach to investigate surface runoff and erosion during rainstorms. An erosion plot case study in central Java, Indonesia. Indones. J. Geogr. 10, 23–42.
Verbist, B., Poesen, J., van Noordwijk, M., Widianto Suprayogo, D., Agus, F., Deckers, J., 2010. Factors affecting soil loss at plot scale and sediment yield at catchment scale in a tropical volcanic agroforestry landscape. Catena 80, 34–46.
Walker, S.M., Pearson, T.R.H., Casarim, F.M., Harris, N., Petrova, S., Grais, A., Swails, E., Netzer, M., Goslee, K.M., Brown, S., 2012. Standard Operating Procedures for Terrestrial Carbon Measurement: Version 2012. Winrock International, Arlington, Virginia.
Wan, Y., El-Swaify, S.A., 1999. Runoff and soil erosion as affected by plastic mulch in a Hawaiian pineapple field. Soil Tillage Res. 52, 29–35.
Wiersum, K.F., 1984. Surface erosion under various tropical agroforestry systems. In: O'Loughlin, C.L., Pearce, A.J. (Eds.), Effects of Forest Land Use on Erosion and Slope Stability. International Union of Forestry Research Organizations, Austria, pp. 231–239.
Wiersum, K.F., 1985. Effects of various vegetation layers in an Acacia auriculiformis forest plantation on surface erosion in Java, Indonesia. In: El-Swaify, S.A., Moldenhauer, W.C., Lo, A. (Eds.), Soil Erosion and Conservation. Soil Conservation Society of America, Ankeny, Iowa, pp. 79–89.
Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses: a guide to conservation planning. Agriculture Handbook No. 537. U.S. Department of Agriculture, Washington, DC.