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	were equally high in the subsoil, stressing the importance of sufficiently deep sampling.

REVIEW

Impact of tropical land use change on soil organic carbon stocks – a meta-analysis

Running title: Soil organic carbon and land use change

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Abstract

Land use changes are the second largest source of human induced greenhouse gas emission, mainly due to deforestation in the tropics and sub-tropics. CO_2 emissions result from biomass and soil organic carbon (SOC) losses and may be offset with afforestation programs. However, the effect of land use changes on SOC is poorly quantified due to insufficient data quality (only SOC concentrations and no SOC stocks, shallow sampling depth) and representativeness. In a global meta-analysis, 385 studies on land use change in the tropics were explored to estimate the SOC stock changes for all major land use change types. The highest SOC losses were caused by conversion of primary forest into cropland (-25%) and perennial crops (-30%) but forest conversion into grassland also reduced SOC stocks by 12%. Secondary forests stored less SOC than primary forests (-9%) underlining the importance of primary forests for C stores. SOC losses are partly reversible if agricultural land is afforested (+29%) or under cropland fallow (+32%) and with cropland conversion into grassland (+26%). Data on soil bulk density are critical in order to estimate SOC stock changes because i) the bulk density changes with land use and needs to be accounted for when calculating SOC stocks and ii) soil sample mass has to be corrected for bulk density changes in order to compare land use types on the same basis of soil mass. Without soil mass correction, land use change effects would have been underestimated by 28%. Land use change impact on SOC was not restricted to the surface soil, but relative changes were equally high in the subsoil, stressing the importance of sufficiently deep sampling.

1 Introduction

Land use changes in the tropics are responsible for 12-20% of the human induced greenhouse gas emissions and are expected to remain the second largest source of greenhouse gas emission also for the future (IPCC, 2007, van der Werf *et al.*, 2009). The dominant type of land use change is the conversion of forest to agricultural systems with continuously high rates of 13 million ha being deforested per year (FAO, 2006). Governmental measures to reduce deforestation have been effective only in some countries such as Costa Rica and India during the last years. The destruction of primary forest causes a rapid biomass carbon (C) loss that is accompanied by a C loss from soils. A shift from higher to lower average total ecosystem C stocks increases in atmospheric CO_2. Soils are major carbon stores in tropical areas, with 36-60% of ecosystem C in forests being stored in soils (Dixon *et al.*, 1994, FAO, 2006, Malhi *et al.*, 1999). Tropical soils are estimated to emit 0.2 Gt C yr^{-1} due to land use changes, accounting for 10-30% of the total C emission from deforestation (Houghton, 1999, Achard *et al.*, 2004). In contrast, other land use changes may lead to increased soil organic carbon (SOC) stocks, e.g., if cropland is converted into grassland or afforested (Paul *et al.*, 2002, Guo *et al.*, 2002). However, the estimates of SOC losses and gains are subject to large errors and methodological biases (Goidts *et al.*, 2009) and the susceptibility of SOC to land use change in tropical soils is insufficiently quantified. The estimated errors of the IPCC default values (Good Practice Guidelines LULUCF) for SOC stock changes after land use change are three to four times higher for tropical than for temperate regions (Penman *et al.*, 2003). The reduction of land use changes that lead to C losses from soils and biomass could be a substantial and economically sound contribution to reduce greenhouse gas emissions (Kindermann *et al.*, 2008). Avoided deforestation activities could reduce anthropogenic C emission by 0.8 to 1.3 Gt C yr^{-1}. The most cost effective way to reduce C emissions can be achieved, if under a full carbon accounting scheme all major effects of human activities are included and reported. The land use induced changes in biomass and soil organic carbon

(SOC) stocks are the major uncertainty in such accounting schemes and in life cycle assessments of tropical agricultural products.

SOC changes are controlled by i) the decomposition rate of SOC, e.g., due to changes in microclimate, and ii) alterations in the quantity and quality of C cycled through the system (Juo *et al.*, 1996). Land use directly affects both microclimate and quantity, quality and the pathways of C input. Moreover, erosion is controlled by land use and land management and may decrease SOC stocks in agricultural systems compared with forests. Erosion may be a major pathway of SOC loss on the plot scale on insufficiently aggregated soils typical for tropical regions (van Noordwijk *et al.*, 1997, Berhe *et al.*, 2007). On the other hand, erodibility generally decreases with increasing topsoil SOC content, which stresses the importance of SOC for soil fertility and productivity (Feller *et al.*, 1997). This is especially true for tropical regions where nutrient poor, highly weathered soils are often managed with few external inputs of nutrients and C. Tropical SOC stocks may be more susceptible to perturbations such as land use changes with twice as high SOC turnover than in temperate regions (Trumbore, 1993, Six *et al.*, 2002, Penman *et al.*, 2003). Higher temperature and soil moisture regimes enhance decomposition rates and thus may speed up SOC losses. Highly weathered soils, e.g., Oxisols and Ultisols, cover 60 to 70% of tropical land area. In these soils low activity clays are predominant and provide less mineral surfaces for physical protection and stabilisation of SOC (Feller *et al.*, 1997). However, there is an ongoing discussion about whether climatic factors or the differences in soil mineralogy and land use history contribute most to distinct tropical SOC dynamics (Paul *et al.*, 2008, Zinn *et al.*, 2005, Zinn *et al.*, 2007, Feller *et al.*, 1997).

During the last years, many new research programs and projects have aimed to improve the understanding on carbon fluxes and balances in tropical soils. Hundreds of new studies were published but have never been analysed together. Moreover, insufficient sampling depth and missing correction for differences in bulk density after land use change may have lead to significant bias in previous studies (Ellert *et al.*, 1995, Baker *et al.*, 2007). Differences in rooting depth and tillage on croplands directly influence the C distribution in the soil profile. Shallow sampling misses C which is incorporated below the topsoil and may lead to overestimations of land use change effects on soil C. Thus, the objective of this study was to gather the existing high quality data sets on land use change effects and SOC for the tropics to derive new estimates beyond site specific values and including also subsoil horizons. More than 380 old and new data sets were compiled and quantitatively analysed. This study provides the first estimate of tropical soil C stock changes after land use change for the depth 0-30 cm, the soil depth that has to be reported under UNFCCC, and below.

2 Material and Methods

2.1 Data sources and compilation

Data from 385 studies from 153 published and mostly peer reviewed publications on the influence of land use changes on soil organic carbon were compiled. Data were derived from 39 different tropical countries covering all continents ranging from semi-arid regions such as southern Africa and northern Australia to the humid tropics along the equator. Twelve different land use change types were classified and investigated covering all land use transitions occurring in the tropical zone (Tab. 1). Most studies were conducted in paired plot design using the "space for time" approach. Since SOC may reach a new equilibrium only after several years or decades, there was almost no study with a time series going back to prior land use change conditions. Thus, for each paired site, it has to be assumed that soil conditions were similar before the land use change. Studies were rejected for the data compilation if the different land use types were i) confounded by different soil types as, e.g., indicated by significant differences in texture, ii) sampled for different soil depth or iii) reporting only short term effects (< 5 years). For chronosequences only data from the longest treatments of land use change were used in this study. With the exclusion of short term studies, the influence of the time period since land use change on the estimated SOC changes was minimized and not detectable anymore. Data on SOC stocks, bulk density and the associated meta-data were compiled. Organic layers (forest floor) are rare in tropical forests and the few existing data sets did not allow them to be included in this analysis.

In the current study, primary forest is defined as natural vegetation without apparent and reported human impacts. The primary forest vegetation class also comprises natural vegetations with shrubland and non-managed grassland with savannah-like characters, e.g., the South American Cerrado. It has to be noted that there are only few remaining totally undisturbed tropical forests leading to a rather broad definition of "primary forest" in many studies and consequently also in our study (Lugo *et al.*, 1993). Secondary forests are managed forests and forests regrown after destruction or partial exploitation of the primary forest. Natural successions and fallow older than 7 years were classified as secondary forest. "Grassland" comprises pastures but no natural grasslands, since natural grasslands are mostly savannah type grasslands with tree and shrub vegetation. Additionally, there is no harvested fraction of net primary production on natural grasslands, which directly affects the C dynamics. Croplands are classified as "perennial crops," such as sugar cane and coffee plantations, and "croplands," with annual crops such as maize and beans. Both cropland types and grasslands were described as agricultural systems.

2.2 Data treatment and missing bulk densities

For 81% of the reported data, SOC stocks were directly available or calculated as

$$\text{SOC stock [Mg ha}^{-1}] = \sum_{i=1}^{n} \text{SOC concentration [Mg Mg}^{-1}] * \text{bulk density [Mg m}^{-3}] * \text{soil volume [m}^{3}\text{ ha}^{-1}] \quad \text{(Eq 1)},$$

where n was the number of soil layers, varying for each study. Bulk density data were available for 52% of all reported soil horizons. Most studies reported SOC stocks (81%) but 19% of the studies reported only SOC concentrations. Two strategies can be used to handle the problem of missing bulk density data: either all studies without bulk density measurements are excluded from the meta-analysis, or estimated mean bulk densities replace the unknown values to convert SOC concentrations into stocks. To quantify the difference in accuracy between the two approaches we first conducted the meta-analysis (see below) based only on those studies with bulk density measurements. For the second approach the data set was divided in two sub datasets comprising the studies with and without bulk density measurements. For those studies lacking own bulk density data, the bulk densities before and after land use change were simulated as two-dimensional truncated normal random vectors separately for each land use change type. For these Monte Carlo simulations the means and covariances were derived from the studies with bulk density measurements. The truncation was necessary to avoid unrealistic (e.g., negative) values for bulk density caused by the unbounded normal distribution. The standard deviation of the estimated mean effect size from 10000 repeated simulations provided a direct estimate of the uncertainty introduced by using an estimated mean bulk density instead of true measurements. Finally, estimates of mean effect sizes and their standard errors were obtained as weighted averages of the estimates from the two sub datasets. These Monte Carlo simulations revealed that for all land use change types the estimated uncertainty can be reduced by including also the studies that reported only SOC concentrations after converting them with weighted mean bulk densities into SOC stocks. Thus, we decided to also include studies that report only SOC concentrations.

SOC stocks were corrected to an equivalent soil mass on both land use types (Ellert *et al.*, 1995). Weighted mean bulk densities for each land use change type were used if bulk density data were not available to perform this correction. Data reported as SOM concentrations were converted to SOC by multiplying with a conversion factor of 0.58 (Mann, 1986). Studies were restricted to mineral soils. Wetlands soils such as peatlands and paddy soils were not included in this analysis, mainly due to an insufficient number of studies on these soil types to abtain an adequate representation compared to non-wetland soils. Soil horizons down to max 100 cm were included in the analysis. The following relevant meta-data were also included in the compilation: time since conversion (age), clay content (texture), soil type, mean annual precipitation, mean annual temperature, soil sampling depth and other management factors (tillage, species, fertilisation etc.). If some of these data were not available, data were not estimated by interpolation or transfer functions and the study was excluded from the corresponding analysis. Only for climatic data were other sources used such as long term climate records of the region.

2.2 Meta-analysis

The simplest measure of effect size δ commonly employed in a meta-analysis is the difference between control group mean μ_c (before land use change) and treatment group mean μ_e (after land use change). We used both the absolute effect size $\delta_{abs,i}=\mu_{e,i} - \mu_{c,i}$ and the relative effect size $\delta_{rel,i=} (\mu_{e,i} - \mu_{c,i)/} \mu_{c,i}$*100%. To account for the different accuracy of the heterogeneous set of studies, the mean effect size for the different land use change types was estimated as weighted mean with the optimal weights being inversely proportional to the variance of the single-study effect sizes. To estimate the optimal weights we had to estimate these variances. Two sources of variability contribute to the uncertainty of the effect sizes, the within-study variability derived from sampling and analytical errors and the between-study variability derived from differences in climate, soil, plant species and land management between studies. Since 73% of all studies did not report any measure of variability/accuracy for their estimated means $\mu_{c,i}$ and $\mu_{e,i,}$, we used the available information to estimate the underlying within-study variances $\sigma^2_{within_{c,i}}$ and $\sigma^2_{within_{e,i}}$ as a power function of SOC stocks (R^2=0.67 for a linear regression on a log-log scale). The standard errors of reported SOC stocks estimates for a single study are then determined by the differing sample sizes n_i and can be estimated as:

$$\hat{\sigma}_{\mu_{within,i}} = \frac{\hat{\sigma}_{within_i}}{\sqrt{n_i}} \qquad \text{Eq (2)}$$

for both means, μ_c and μ_e, respectively. The main uncertainty is derived from the field heterogeneity and therefore, the sample size n is the number of collected soil samples, even though they have been pooled to compound samples before analysis. Between-study variability was estimated by the moment estimator

$$\hat{\sigma}^2_{between} = \frac{1}{k-1}\sum_{i=1}^{k}(\mu_i - \overline{\mu})^2 - \frac{1}{k}\sum_{i=1}^{k}\frac{\hat{\sigma}^2_{within_i}}{n_i} \qquad \text{Eq (3)}$$

resulting in the following estimated variance for the absolute effect size $\delta_{abs,i}$:

$$\hat{\sigma}^2_{abs,i} = \hat{\sigma}^2_{between,c} + \hat{\sigma}^2_{\mu_{c,within,i}} + \hat{\sigma}^2_{between,e} + \hat{\sigma}^2_{\mu_{e,within,i}} - 2\,\widehat{\mathrm{cov}}(\mu_{c,i}, \mu_{e,i}) \qquad \text{Eq (4)}$$

The mean absolute effect size δ_{abs} was then estimated as the weighted average

$$\hat{\delta}_{abs} = \frac{\sum_{i=1}^{k}\left(\frac{1}{\hat{\sigma}^2_{abs,i}} \cdot \hat{\delta}_{abs,i}\right)}{\sum_{i=1}^{k}\frac{1}{\hat{\sigma}^2_{abs,i}}} \qquad \text{Eq (5)}$$

For the variance estimation of the mean relative effect sizes $\delta_{rel,i}$, we used the same weights

$$\hat{\delta}_{rel} = \frac{\sum_{i=1}^{k}\left(\frac{1}{\hat{\sigma}^2_{abs,i}} \cdot \hat{\delta}_{rel,i}\right)}{\sum_{i=1}^{k}\frac{1}{\hat{\sigma}^2_{abs,i}}} \qquad \text{Eq (6)}$$

Estimates of standard errors for the weighted means were obtained by nonparametric bootstrap based on 1000 bootstrap samples.

Four different sampling depths were selected in order to investigate the effects of land use change for different soil depth: Topsoil (0-10 cm), 0-30 cm depth (ploughing horizon), subsoil (>20 cm depth) and full depth (total sampling depth of each study). Studies reporting only data for some soil depth classes were however included for the analysis of the reported soil depth class which led to different numbers of studies for the four soil depth classes. There was no significant influence of the maximum sampling depth on the relative SOC changes (as fixed and variable effect in the general linear models; $F_{133.4,1}$= 0.097, p=0.76), indicating that the variability between different studies is as large as between different maximum sampling depths.

2.3 Statistical analysis

The influence of the following variables on SOC change and bulk density change was investigated on a core data set which contained all of the following variables comprising 377 land use pairs: Mean annual temperature, annual precipitation, soil mineralogy (using the three clay type classes: high activity clay, low activity clay, allophanic clay), region/continent, sampling depth and SOC stocks before land use change. Additionally, the methodological parameters maximum sampling depth and number of sample replicates were used as independent variables. We checked the effects with mixed linear models including the author of the studies as a random effect. Since no author-specific effects could be found, we used classical general linear models for the further analysis. Data are presented in the text as $F_{\text{Sum of squares, degrees of freedom}}$ and the p-value. Statistical analysis was performed using R software.

3 Results

3.1 Data quality and mass correction

The analysis of land use effects on soil carbon is hampered by the high heterogeneity of the data set including different sampling methods, sampling intervals and missing meta-data such as the land use history. Most important for improved estimates of SOC changes is the availability of bulk density data in order to account for SOC changes on an area basis and to be able to correct data for different soil mass sampled in land use types. The correction for different soil mass (bulk density) of land use types increased the land use change effect by an average of 28% (Tab. 2). Pedo-transfer functions (Post *et al.*, 2000, De Vos *et al.*, 2005, Mann, 1986) are rarely able to take these effects into account with sufficient accuracy and we were able to predict bulk density in our study only with a correlation coefficient of 0.67 (data not shown). However, the uncertainty of the estimated SOC change could be decreased by combining studies with and without density measurements. The fraction of studies reporting only SOC concentrations per land use change type was between 6 and 36%, indicating that the majority of studies reported bulk densities and stocks. The uncertainty has been reduced by on average 52% as compared to a meta-analysis only comprising studies with bulk density managements. With our approach, we take the uncertainty derived from an incomplete data set (e.g. missing bulk density data) and the uncertainty of SOC and bulk density measurements into account. The within-study uncertainty depends on the sampling and analytical errors and the soil heterogeneity in the field and could be reduced by increasing numbers of soil samples (Eq. 2).

Land use change caused changes in the bulk density of slightly lower magnitude like SOC changes and in the reverse direction (Fig. 1, Tab. 2). Since organic carbon has an inherent low bulk density, SOC concentrations directly affect soil bulk density (Lal *et al.*, 2001). However, land use changes affect bulk density beyond this effect due to compaction by animal trampling, machinery and loosening by tillage. The cultivation of forests caused bulk density to increase by 5 to 23% with the strongest increase occurring in the surface, lessening with depth with no significant effects below 20 cm depth (Fig. 1). Surprisingly, even tillage on croplands did not decrease bulk density, but cropland bulk density was always higher than grassland and forest bulk density.

Rates of land use change vary widely among tropical regions but the number of existing studies did not reflect land use change rates. Malaysia and Indonesia are among the countries with the highest emissions due to land use changes (Houghton, 2003, FAO, 2006). However, the region of South Eastern Asia was undersampled with only 11 studies reporting quantitative data on land use effects on SOC. Most regions in Africa – except for Nigeria – were also undersampled, whereas good data coverage has been reached in Central and South America, especially in Ecuador, Costa Rica and Brazil.

3.2 Primary forest to agricultural land

The conversion of native vegetation to agricultural systems caused the highest SOC losses among all land use change types (Tab. 1, Fig. 2 and 3). Native vegetation such as primary forest and native grassland stored among the highest amounts of SOC (80 ±9 Mg SOC ha^{-1}, mean sampling depth 36 cm). Conversion of primary forest to cropland (-25%) caused twice as high SOC losses than its conversion to grassland (-12%). The relative SOC loss in the subsoil was similar on grasslands but not significant for croplands due to a great variability caused by different management practices and crop types. With the cultivation of primary forests, soils were compacted and bulk density increased by 14 and 18% for grasslands and croplands, respectively. Especially if forests are converted to grasslands the correction for different soil mass exerted a strong influence on the estimated SOC changes, switching this land use change from almost C neutral to a significant C source (Tab. 2).

42% of the variability between data sets could be explained with different land use change types and the climatic factors precipitation and temperature. SOC losses increased with increasing temperature and for conversion of forest into grassland also with increasing precipitation (Tab. 3 and 4). For forest cultivation we found no uniform influence of precipitation on SOC changes and a higher uncertainty of the models as compared to forest conversion to grassland.

3.3 Secondary forest to agricultural land

Similar to primary forests also secondary forests' conversion to agriculture systems led to SOC losses. However, relative SOC loss (for all depths) was less compared to primary forests, indicating a higher vulnerability of SOC in primary forests than in secondary forests to land use changes. Surface soil SOC stocks remain unchanged when secondary forest was converted to grassland (Fig. 3). Grasslands are characterised by a steep C gradient with soil depth leading to high surface soil SOC stocks. In contrast, a smaller fraction of total SOC is stored in the surface soil of secondary forests. SOC losses after deforestation were significantly affected by climatic factors, in particular moisture conditions (mean annual precipitation) and temperature (Tab. 3, Fig. 4). Surprisingly, we found no effect of the clay content ($F_{7750,5}$; p=0.58; Fig. 5) and soil type ($F_{53213,30}$ p=0.45) on SOC losses. However, beside temperature and precipitation the differences between clay types (low activity, high activity allophonic) exerted a significant effect on SOC changes (Tab. 3). Land use change types, climate factors and clay type could explain 55% of the SOC change variance leaving almost half of the data set variability unexplained. The effect of different management practices for croplands (e.g., tillage vs. no tillage) could not be investigated due to a very small number of studies covering both land use change and different management practices.

3.4 Primary to secondary forest

Management of primary forest with wood extraction and planting of productive tree species caused a mean SOC loss of 7% or 9 Mg SOC ha^{-1} (Tab. 1). One major difference between primary and secondary forest is the SOC distribution in the soil profile (Fig. 2 and 3), with a higher surface SOC fraction in primary forest compared to secondary forests. 7 Mg SOC ha^{-1} (-15%) were lost in the upper 10 cm only after conversion of primary forest to secondary forest. However, there was no significant SOC change below 20 cm depth. SOC losses were accompanied by a 6% increase in the soil bulk density leading to 28% higher SOC changes after mass correction as compared with no mass corrected data (Tab. 2).

3.5 Afforestation and fallow

SOC losses due to deforestations were partly reversible by afforestations of croplands or grasslands. Estimated mean SOC stock gains for afforestations were even higher than SOC losses from deforestations, with highest SOC gains in afforestations on former croplands (+ 50%) compared to afforestations on former grassland (18%, Tab. 1). Afforested grassland stored particular low amounts of SOC (60 ±9 Mg ha^{-1}) since afforestations were mainly conducted on degraded grasslands or on marginal land with intrinsic low SOC storage capacity (Tab. 1). Similarly, the termination of cropping leading to natural succession (fallow) on croplands took place mainly on degraded land as indicated by low SOC stocks (Tab. 1). Fallow increased SOC stocks by 32% indicating a rapid recovery of SOC stocks.

3.6 Conversion of grassland to cropland and vice versa

Typical land use change cascades in the tropics are first the conversion of forest into grassland for cattle grazing and at a later stage the conversion of grassland into cropland. Cropland establishment on grasslands reduced SOC stocks by 6 Mg C ha^{-1} (-10%) but this effect was restricted to the uppermost soil horizon. Subsoil below 20 cm depth was not significantly affected by these land use changes due to high C input with tillage. Several studies reported lower subsoil SOC stock in grassland compared to croplands (Fujisaka 1998, Huges, 2000, Freitas 2000). Cropland conversion or re-conversion to grassland increased SOC stocks by 8 Mg C ha^{-1} (+26%), which is more than the SOC loss after cropland establishment on grassland. Similar to the afforestation of croplands, this indicates that croplands management causes SOC losses leading to lower initial SOC stocks of croplands before conversion into grasslands (Tab. 1).

3.7 Perennial crops and plantations

The conversion of primary forests to perennial crops caused an even higher C loss of than the conversion to cropland (-30%, Tab. 1). In contrast, the conversion of secondary forests to perennial crops seems to hardly affect SOC stocks. This may be partly explained by higher mean SOC stock before land use change in primary forests than in secondary forests of studies reporting conversion into perennial crops. For all land use change types SOC losses and gains were weakly positively correlated with SOC stock before land use change. Most data on soil carbon on perennial crops were reported from sugar cane plantations (28% of studies including perennial crops), fruit tree plantations (including banana) (12%) and cacao plantations (9%). These findings are based on 35 studies and indicate that a permanent vegetation cover does not always prevent SOC losses under intensive management when SOC rich forests are converted to perennial crop plantations. SOC changes may be different in low input agro-forestry type perennial croplands which are not well covered in this meta-analysis.

4 Discussion

4.1 Deforestation and afforestation

A large number of studies on land use change effects were conducted during the last years, 25% of the studies in this meta-analysis were published during the last five years and 67% during the last 10 years. Moreover, it was only recently that more studies also included deeper soil horizons down to 100 cm depth. Former reviews calculated higher global and tropical SOC stock changes after cultivation of forests compared to our study (Davidson *et al.*, 1993, Guo *et al.*, 2002, Detwiler, 1986, Paustian *et al.*, 1997, Amundson, 2001). This can partly be explained with an improved data quality and quantity, e.g., with deeper sampling and more data on bulk density changes. Detwiler (1986) found twice as high SOC losses after deforestation than reported in our study (-20% for forest to grassland and -40% for forest to cropland). A range of SOC losses from -24 to -43% was reported for cultivated tropical soils (Davidson *et al.*, 1993). The IPCC guidelines set a default value of -31 and -42% SOC for dry and wet tropical regions after forest cultivation, respectively, which is higher than calculated in our study (Tab. 5). Our study confirmed the impact of soil moisture and precipitation on SOC dynamics with higher SOC changes in regions with higher precipitation for most land use change types (Tab. 3, Fig. 4). Soils in humid regions maybe more vulnerable to land use changes than in dryer regions (Brown *et al.*, 1990, Amundson, 2001). The impact of precipitation seems to be stronger when forests are converted into grasslands than for forest conversion into cropland (Tab. 5). In a global analysis Guo and Gifford (2002) found conversion of forest to grassland to increase SOC stocks by 9% (2002), no SOC change has to be assumed as default value under IPCC and is reported in other reviews (Lugo *et al.*, 1993, Cerri *et al.*, 2004, Penman *et al.*, 2003). In contrast, we found tropical forests lost 12% SOC after grassland establishment (Tab. 1). These differences can be partly attributed to improved data quality and the application of a soil mass correction which accounts for changes in different bulk densities in different land use types (Ellert *et al.*, 1995, Gifford *et al.*, 2003, de Moraes *et al.*, 1996). Detwiler (1986) tried already to overcome the problem of different soil mass but had to rely on calculated and not measured bulk densities. Soils were compacted by 10 and 16% due to forest conversion into grassland and cropland, respectively. Most tropical grasslands are under higher grazing pressure, a higher biomass fraction is exported (harvest) and fertilizer input is low compared to temperate grasslands. Improved grassland management with the application of fertilizers would help to increase productivity and SOC stocks compared to extensive pastures (Soussana *et al.*, 2007, Ammann *et al.*, 2007). Roots are a more effective pathway to build up SOC stocks than foliar litter, which explains high grassland SOC (Lugo *et al.*, 1993, Rasse *et al.*, 2005) and relatively small SOC losses if forests are converted to grasslands as compared to croplands.

A major proportion of total SOC change occurred during the first few years after cultivation of forest, indicating that these soils contain large amounts of labile SOC, potentially vulnerable to degradation upon human-induced land-use changes (Solomon *et al.*, 2007). A new equilibrium of SOC has been reached most often within 3 to 10 years (Houghton, 1999, Feller *et al.*, 1997, Detwiler, 1986, Davidson *et al.*, 1993). Other studies found 20 to 40 years (Solomon *et al.*, 2007, Sa *et al.*, 2001, Riezebos *et al.*, 1998, Cerri *et al.*, 2007). In our study, the average time period since deforestation was 22 years, and 33 years since afforestation, indicating that major parts of SOC changes are captured within this time period.

Reforestation and afforestation were found to successfully recover SOC stocks (Silver *et al.*, 2000, Post *et al.*, 2000, Bashkin *et al.*, 1998). Cropland afforestation increased SOC stocks by 33 Mg ha^{-1} which is slightly lower than the mean SOC accumulation of 41 Mg ha^{-1} after 80 years reported from Silver *et al.* (2000). SOC gains with afforestation were higher than SOC losses after deforestation (Tab. 1). Forest establishment has mainly been performed with highly productive tree species like eucalyptus with a low litter quality and high recalcitrance. High SOC accumulation in secondary forest may be also fostered by a low initial SOC content in the afforested degraded agricultural land (Lugo *et al.*, 1993). Agricultural management on highly weathered soils often lead to a rapid decline in soil fertility, leaving degraded land for forest regrowth, as it is part of the traditional shifting cultivation system.

4.2 Agricultural systems

Low SOC stocks in croplands have important implications for crop production since organic matter supplies most of the nitrogen and parts of the phosphorous taken up by unfertilized crops (Sanchez, 1976). SOC is essential for the retention of nutrients and water in highly weathered soils with low cation exchange capacity. Tropical regions cover very different stages of agricultural mechanisation and development with various management options on croplands, including organic amendments and different tillage practice, and a high number of different crops. Thus, the estimated effect of cultivation can only set a mean value for regions but cannot be applied to specific field sites. Climatic and soil parameters could only explain 55% of the data variability. For agricultural systems, the biomass fraction left for SOC build-up (crop residuals) is strongly controlled by management practices including the selection of crop species. Improved cropland management may partly offset SOC losses due to deforestation (Lugo *et al.*, 1993); 13% of croplands included in this meta-analysis reported similar or higher SOC stocks than in forests. Additional effort with field data collection is necessary to quantify the effect of different management options on a global scale. Moreover, insufficient sampling depths were found to obscure conclusions on management and land use effects on the SOC balance (Baker *et al.*, 2007).

Regular soil disturbance during tillage or harvest is one of the main reasons for low cropland SOC stocks (Lal, 1998). Grasslands, pastures and perennial crops, unlike croplands, maintain a permanent vegetation cover and a high root turnover leading to high SOC input. We found surprisingly high SOC losses after primary forest was converted to perennial cropland or grassland. The amount of crop residuals returned to the soil directly affect SOC, and most perennial crops such as sugar cane plantations, are managed with high intensity and high biomass export (Graham *et al.*, 2002). Similar to other cropland types, different management options and land use history determined the amount of SOC loss after cultivation of primary forests and, on the other hand, the sequestration potential if perennial cropland and grassland is afforested.

4.3 Soil characteristics and erosion

Differences in SOC change are expected to be attributable to soil parameters such as soil type, texture and clay mineral type (de Moraes *et al.*, 1996, Feller *et al.*, 1997, Hartemink, 1997, Davidson *et al.*, 1993). Clay type was found to explain only additional 13% of the SOC variability, beside land use change type, temperature and precipitation (Tab. 3). No influence on SOC change could be attributed to soil type and clay content (Fig. 5), which is in line with findings from Davidson *et al.* (1993). Soil parameters' influence maybe obscured by dominant other factors such as climate and management. Additionally, the data availability was low with only 22% of all studies reporting clay content. Highly weathered soils, such as Ferralsols were found to loose more SOC after cultivation than other soil types (Hartemink, 1997), but this could not be confirmed in our study.

Erosion is a major factor affecting SOC stocks that is directly related to land use and forest clearing (Nye *et al.*, 1964, van Noordwijk *et al.*, 1997, Wairiu *et al.*, 2003). Soils under low vegetation cover (agricultural systems, conventional tillage), on steep slopes and under high precipitation intensity are most prone to erosion. However, adequate data was not available in this meta- analysis to assess the proportion of erosion-triggered SOC loss. Some areas may even gain SOC with deposition of eroded material, leaving the question open of whether erosion decreases or increases the terrestrial C sink (Berhe *et al.*, 2007, Lal, 2003, Van Oost *et al.*, 2007).

4.4 SOC changes in the surface soil and the subsoil

The SOC in topsoil is supposed to be more prone to land use change and other perturbations than subsoil (Siband, 1974, Veldkamp *et al.*, 2003, Veldkamp, 1994). We found equally high relative subsoil SOC stock changes compared with surface soil horizons after conversion of native forests to agriculture systems (Fig. 2 and 4). Native forests stored higher amounts of subsoil C which are lost upon cultivation compared with secondary forests. The mean soil sampling depth did not contribute to an explanatory model indicating that the relative SOC change is only weakly related to the soil depth. In contrast, absolute SOC changes significantly decrease with soil depth owing to decreasing absolute SOC stocks in deeper horizons. In fact, tillage may even increase subsoil SOC stocks in croplands due to C rich topsoil being mixed with deeper horizons (Hughes *et al.*, 2000, Fujisaka *et al.*, 1998). A sampling depth as deep as the tillage depth is the minimum to quantify land use change effects. Our results indicate that at least the conversion of native forests also affects subsoil SOC below 20 cm depth and a comprehensive assessment should also include subsoil horizons, if possible down to 100 cm depth. In order to estimate subsoil SOC changes with land use change it is even more important to ensure comparable soil intrinsic conditions on paired or chronosequence sites since SOC stabilisation in the subsoil is highly dependent on soil mineralogy, texture and other soil parameters. The high variability of subsoil SOC change may be a result of the variability of these soil intrinsic parameters.

4.5 Bulk density change and its impact of SOC stock estimates

The relative changes in bulk density were almost as high as the relative SOC changes, e.g., cultivation of forest increased bulk density by 16 % (Fig. 1, Tab. 2). Bulk density changes are important to account for SOC stocks changes, since SOC stocks linearly depend on both SOC concentration and bulk density. Moreover, bulk density change causes a sampling bias if sampling of each land use type is performed at the same sampling depth (Ellert *et al.*, 1995, Gifford *et al.*, 2003, Davidson *et al.*, 1993). If bulk density increased with land use change, the soil is compacted and sampling down to the same sample sampling depth would lead to higher sampled soil mass than in the corresponding land use type. Since soil mass and soil carbon are ultimately linked, sampling of more soil mass results in higher SOC stocks (Davidson *et al.*, 1993). Thus, the effect of land use change is underestimated, in our study by an average 28% (Tab. 2). This can only be completely corrected if bulk density data prior to land use change are available and it can partly be corrected if bulk density data were recorded after land use change for both land use types (Lee *et al.*, 2009). We found mass correction strongly influencing the effect size (SOC change) with up to three times higher mean SOC changes than estimated without mass correction. However, Monte Carlo simulations revealed that the exclusion of studies that report only SOC concentration would increase the uncertainty of the estimated SOC change by 52%. The high diversity of soil types, climate conditions, vegetation and management types call for as many studies as possible to be included in such meta-analysis, even though not all studies provide the full parameter set. This confirms earlier findings that fewer bulk density than SOC concentration measurements are necessary to estimate SOC stocks (Don *et al.*, 2007). Coefficient of variation (CV) of all studies was 2.7 times lower for bulk density than for SOC concentration (29 and 81% for bulk density and SOC concentration, respectively) indicating that even with few bulk density measurements, uncertainties on land use change effect can be reduced considerably.

5 Conclusions

The conversion of forest, especially primary forests into agricultural systems always lead to SOC losses, but losses are reversible to a high degree if, e.g., agricultural land is afforested or properly managed. For the SOC balance of a land use system, the harvested fraction of net primary production seems to be more important than its disturbance frequency, e.g., with tillage or climate or soil characteristics.

Mass correction of SOC stock estimates is crucial in order to estimate land use change effects since land use change is always accompanied by bulk density changes. The comparison of SOC stocks based on different soil mass deeply confound estimates of SOC changes. This meta-analysis provides soil mass-corrected estimates to improve the current UNFCCC default values. Mean SOC changes were smaller than reported in previous reviews even though soil mass correction increased land use change effects on SOC by 28% on average.

The global data coverage does not mirror the current hot spots of land use changes. New effort are needed to quantify the effect of land use changes in South East Asia and Africa, also taking to account carbon rich wetland forests and degradation cascades within land use classes.

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Table 1: Mean absolute and relative SOC stocks changes and reported minimum and maximum relative SOC stock changes for different land use change types. Additionally, SOC stocks before land use change, mean full available sampling depth (on average 32 cm), mean time interval between the two land use systems and the number of studies included in this meta analysis is displayed with standard error of the mean in brackets.

Land use change (LUC) type	Absolute SOC change	Relative SOC change			SOC prior LUC	Sampling depth	Time after LUC	Number of studies
	Mg ha-1	%	Min %	Max %	Mg ha-1	cm	Years	
Primary forest to grassland	-12.6 (±3.0)	-12.1 (±2.3)	-73	51	73 (±7)	36 (±3)	25 (±3)	93
Primary forest to cropland	-20.1 (±5.2)	-25.2 (±3.3)	-80	58	83 (±9)	36 (±4)	28 (±4)	56
Primary forest to perennial crops	-32.0 (±3.5)	-30.3 (±2.7)	-62	6	105 (±20)	48 (±8)	49 (±12)	20
Primary forest to secondary forest	-12.6 (±2.4)	-8.6 (±2.0)	-64	72	91 (±9)	39 (±4)	28 (±3)	71
Secondary forest to grassland	-11.0 (±3.4)	-6.4 (±2.5)	-71	72	85 (±6)	43 (±3)	27 (±2)	66
Secondary forest to cropland	-25.8 (±6.9)	-21.3 (±4.1)	-74	53	88 (±12)	39 (±5)	36 (±7)	26
Second. forest to perennial crops	-5.6 (±3.0)	-2.4 (±4.2)	-46	243	90 (±17)	51 (±9)	23 (±4)	15
Grassland to secondary forest	12.4 (±6.1)	17.5 (±8.0)	-35	282	60 (±9)	35 (±6)	28 (±4)	32
Cropland to secondary forest	33.2 (±10.5)	50.3 (±11.9)	-63	67	70 (±9)	44 (±6)	32 (±7)	25
Grassland to cropland	-6.0 (±5.7)	-10.4 (±6.1)	-41	167	64 (±15)	38 (±11)	22 (±5)	15
Cropland to grassland	7.6 (±5.8)	25.7 (±11.1)	-32	362	61 (±17)	40 (±10)	21 (±6)	16
Cropland to fallow	8.9 (±2.9)	32.2 (±16.1)	-73	51	43 (±7)	20 (±2)	≤ 7	21

689 Table 2: Effect of mass correction on SOC stock change estimates and relative changes in
690 bulk density after land use change per land use change type for studies reporting bulk density
691 data (± SE in brackets).

Land use change type	**SOC stock changes [Mg ha^{-1}]**		**Relative bulk density changes**
	with mass correction	no mass correction	[%]
Primary forest to grassland	-12.1 (±2.3)	-4.9 (±2.5)	14.0 (±2.2)
Primary forest to cropland	-25.2 (±3.3)	-22.3 (±3.1)	17.8 (±3.5)
Primary forest to perennial crops	-30.3 (±2.7)	-23.2 (±2.7)	22.8 (±6.2)
Primary forest to secondary forest	-8.6 (±2)	-6.7 (±2.1)	5.7 (±2.7)
Secondary forest to grassland	-6.4 (±2.5)	-4.1 (±2.6)	5.4 (±2.3)
Secondary forest to cropland	-21.3 (±4.1)	-19.2 (±4.1)	11.6 (±4.4)
Grassland to secondary forest	17.5 (±8.0)	13.1 (±8.3)	-6.4 (±3.8)
Cropland to secondary forest	50.3 (±11.9)	42.9 (±11.5)	5.0 (±5.4)
Grassland to cropland	-10.4 (±6.1)	-8.8 (±6.2)	-2.7 (±5.8)
Cropland to grassland	25.7 (±11.1)	25.5 (±10.4)	1.9 (±5.3)
Cropland to fallow	32.2 (±16.1)	27.3 (±15.8)	14.0 (±2.2)

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Table 3: General linear model with degrees of freedom (Df), sum of squares (sum of Sq), F-value, P-value. LUC= land use change type, MAT= Mean annual temperature, MAP= Mean annual precipitation, max depth= maximum sampling depth [cm], significance codes: '***' <0.001, '**' 0.001, '*' 0.05, 'n.s.' not significant.

Models	**Df**	**Sum of Sq**	**F**	**P-value**		**Explained variance [%]**
rel. SOC change ~ LUC	5	113095	38.39	<0.000	***	23
rel. SOC change ~ LUC+MAT	1	992	1.68	0.196	n.s.	
rel. SOC change ~ LUC * MAT	6	15093	4.27	<0.000	***	33
rel. SOC change ~ LUC * MAT * MAP	12	16108	2.27	0.009	**	42
rel. SOC change ~ LUC * MAT * MAP * Clay mineral type	34	44853	2.24	<0.000	***	55
rel. SOC change ~ LUC	5	117882	29.23	<0.000	***	
rel. SOC change ~ LUC + max depth	1	213	0.26	0.608	n.s.	
rel. SOC change ~ LUC * max depth	5	14199	1.95	1.955	n.s.	
rel. SOC stock change ~ LUC	4	141916	20.35	<0.000	***	
rel. SOC stock change ~ LUC + Soil type	9	10463	0.67	0.739	n.s.	
rel. SOC stock change ~ LUC * Soil type	45	42750	0.28	0.280	n.s.	

Table 4: Relative SOC stocks changes (± standard error) for different climate and soil conditions derived from a general linear model (Tab. 3).

Land use change	MAT [°C]	MAP [mm]	rel. SOC change
Forest to grassland	20	1000	-1.2 (±1.3)
		2000	-2.7 (±0.2)
		4000	-5.6 (≤0.1)
	23	1000	-4.2 (±1.5)
		2000	-6.4 (±3.6)
		4000	-10.8 (±5.8)
	26	1000	-7.2 (±3.1)
		2000	-10.2 (±6.2)
		4000	-16.0 (±9.3)
Forest to cropland	20	1000	-29.5 (±19.3)
		2000	-25.8 (±17.0)
		4000	-18.2 (±6.5)
	23	1000	-29.9 (±20.4)
		2000	-28.4 (±19.6)
		4000	-25.2 (±14.7)
	26	1000	-30.3 (±19.9)
		2000	-30.9 (±21.3)
		4000	-32.2 (±20.3)

MAT: Mean annual temperature

MAP: Mean annual precipitation

Table 5: Fraction of original soil carbon stock for 0-30 cm depth remaining after land use change. Revised default values for tropical regions from the IPCC Good Practice Guidelines (GPG) LULUCF (Penman *et al.*, 2003) and from this meta-analysis.

SOC stock change	Climate regime	This meta analysis	Uncertainty [%]	Revised GPG default	Error [%]
LUC native vegetation to cropland	Dry	0.76	2	0.69	38
LUC native vegetation to cropland	Wet	0.68	7	0.58	42
LUC native vegetation to grassland		0.91	3	1	

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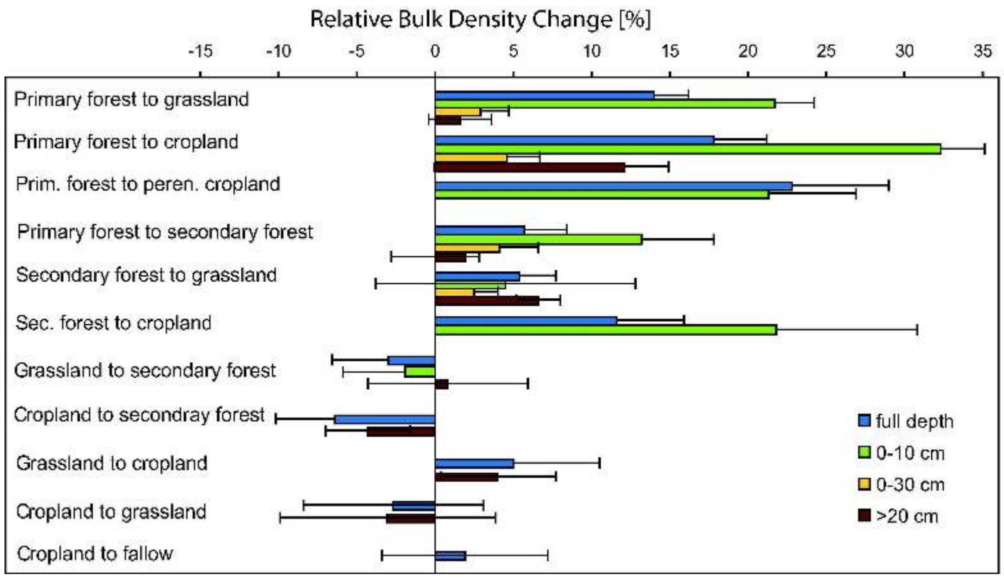
Figure 1: Weighted average relative bulk density change [%] for different soil depth for different land use change types derived from all studies reporting bulk density measurements.

Figure 2: Absolute SOC stock changes [Mg C ha^{-1}] for different soil depth for different land use change types. The different depth increments are covered by different numbers of studies.

Figure 3: Relative SOC stock changes [%] for different soil depth for different land use change types. The different depth increments are covered by different numbers of studies.

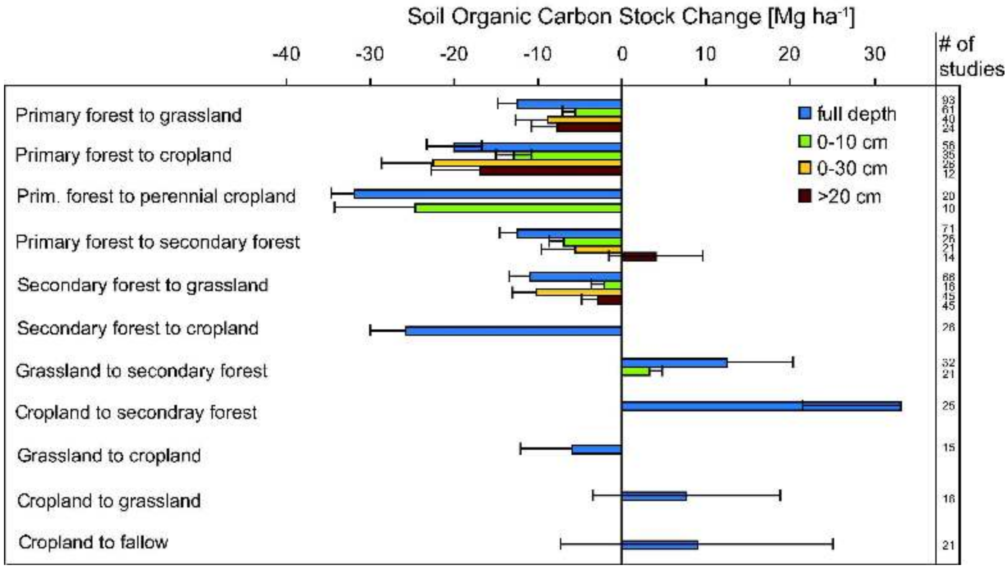
Figure 4: SOC stock change [Mg ha^{-1}] after conversion to grassland (open symbols) and cropland (filled symbols) vs. Mean Annual Precipitation [mm].

Figure 5: SOC stock change [Mg ha^{-1}] after conversion to grassland (filled symbols) and cropland (open symbols) vs. mean content clay of the soil [%].

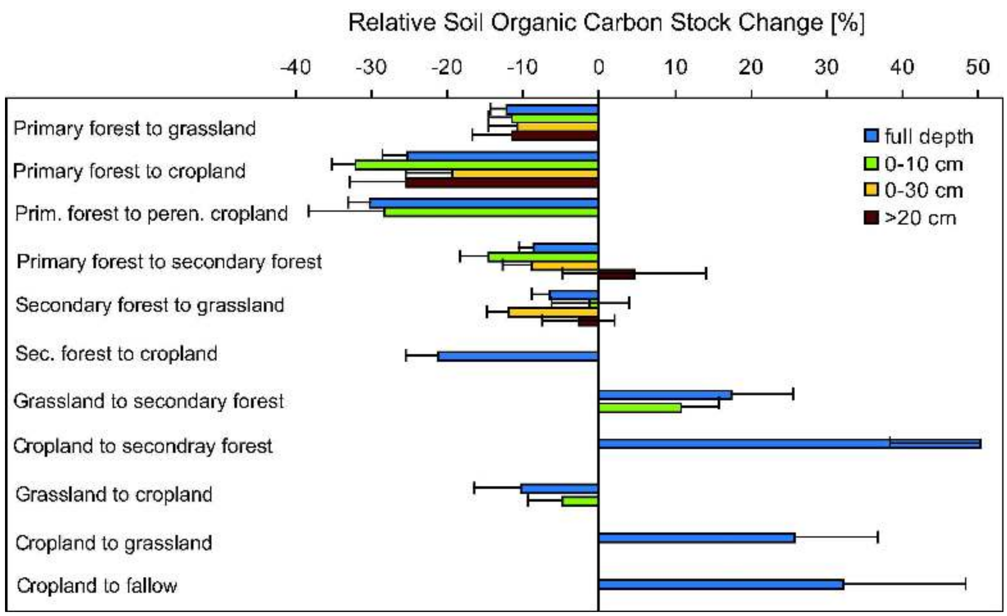


Weighted average relative bulk density change [%] for different soil depth for different land use change types derived from all studies reporting bulk density measurements.

169x96mm (600 x 600 DPI)



Absolute SOC stock changes [Mg C ha-1] for different soil depth for different land use change types.
The different depth increments are covered by different numbers of studies.
186x105mm (600 x 600 DPI)



Relative SOC stock changes [%] for different soil depth for different land use change types. The different depth increments are covered by different numbers of studies.
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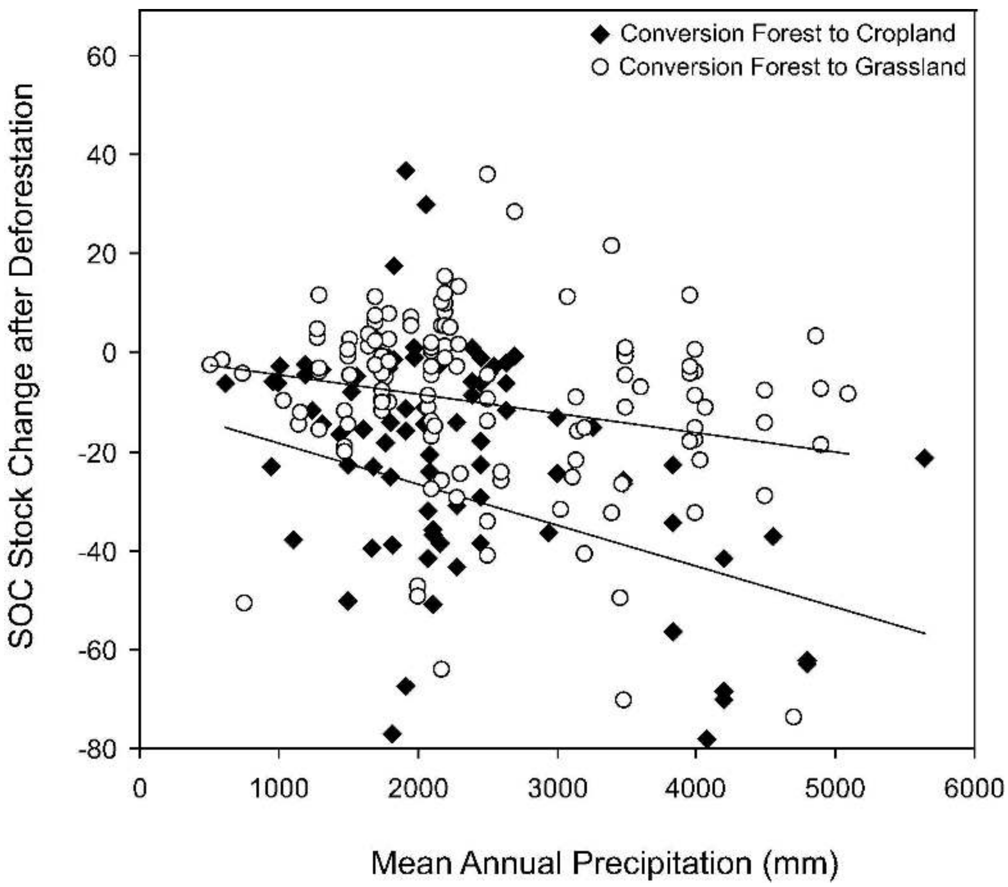
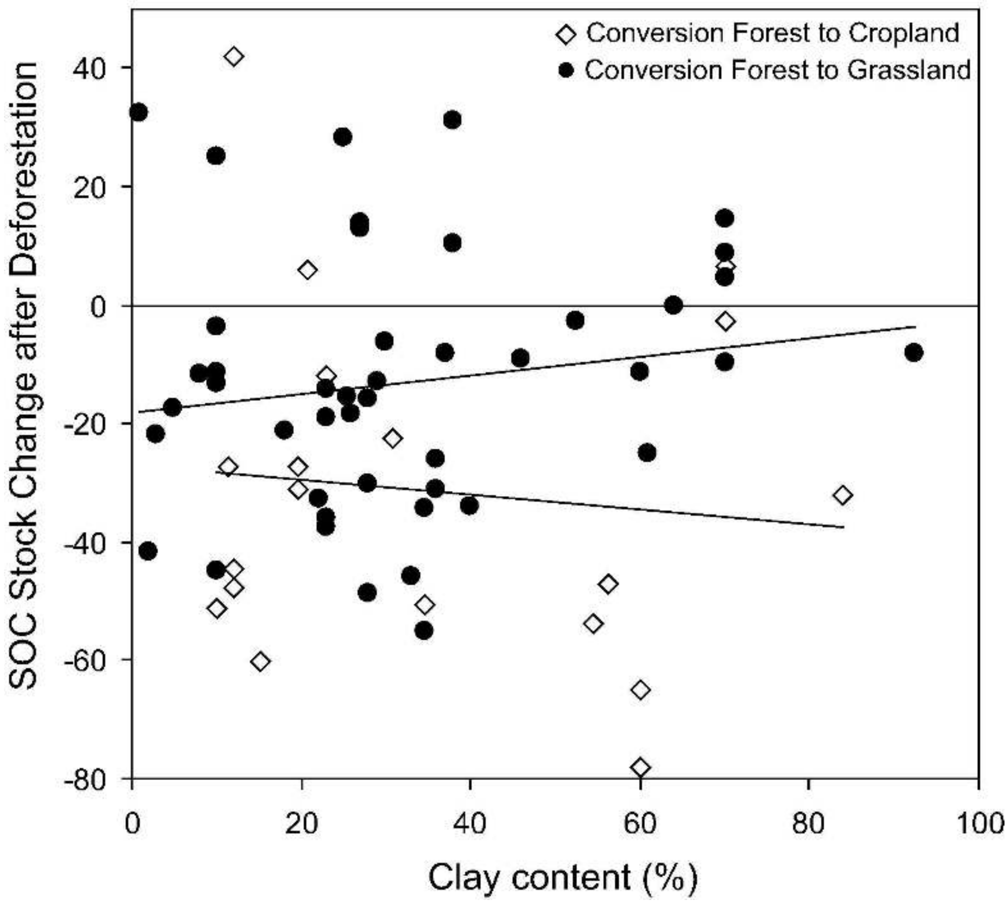


SOC stock change [Mg ha-1] after conversion to grassland (open symbols) and cropland (filled symbols) vs. Mean Annual Precipitation [mm].
171x150mm (600 x 600 DPI)



SOC stock change [Mg ha-1] after conversion to grassland (filled symbols) and cropland (open symbols) vs. mean content clay of the soil [%].
154x139mm (600 x 600 DPI)