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Axel Don, Jens Schumacher, Annette Freibauer

► **To cite this version:**

Axel Don, Jens Schumacher, Annette Freibauer. Impact of tropical land use change on soil organic carbon stocks - a meta-analysis. *Global Change Biology*, 2010, 17 (4), pp.1658. 10.1111/j.1365-2486.2010.02336.x . hal-00599528

**HAL Id: hal-00599528**

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Submitted on 10 Jun 2011

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



Journal:	<i>Global Change Biology</i>
Manuscript ID:	GCB-10-0535
Wiley - Manuscript type:	Review
Date Submitted by the Author:	28-Jun-2010
Complete List of Authors:	Don, Axel; Johann Heinrich von Thünen Institute, Institute of Agricultural Climate Research Schumacher, Jens; Friedrich Schiller University, Institute of Stochastic Freibauer, Annette; Johann Heinrich von Thünen Institute, Institute of Agricultural Climate Research
Keywords:	Land use change, Soil organic carbon, Deforestation, Afforestation, Subsoil, Bulk density, Tropics
Abstract:	<p>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<sub>2</sub> 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</p>

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

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3 1 **REVIEW**  
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5 2 **Impact of tropical land use change on soil organic carbon stocks – a meta-analysis**  
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8 3 Running title: Soil organic carbon and land use change  
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12 5 Axel Don<sup>1)\*</sup>, Jens Schumacher<sup>2)</sup>, Annette Freibauer<sup>1)</sup>  
13  
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16  
17 7 1) Johann Heinrich von Thünen-Institute, Institute of Agricultural Climate Research,  
18 Bundesallee 50, 38116 Braunschweig, Germany  
19 8

20 9 2) Friedrich Schiller University, Institute of Stochastic, Jena, Germany  
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27 11 \* Corresponding author:  
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29 12 Phone: ++49-531-596-2641  
30

31 13 Fax: ++49-531-596-2645  
32

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34 14 Email: axel.don@vti.bund.de  
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38 16 **Keywords:** Land use change, Soil organic carbon, Tropics, Deforestation, Afforestation,  
39 Subsoil, Bulk density  
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3 19 **Abstract**  
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5 20 Land use changes are the second largest source of human induced greenhouse gas emission,  
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7 21 mainly due to deforestation in the tropics and sub-tropics. CO<sub>2</sub> emissions result from biomass  
8  
9 22 and soil organic carbon (SOC) losses and may be offset with afforestation programs.  
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11 23 However, the effect of land use changes on SOC is poorly quantified due to insufficient data  
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13 24 quality (only SOC concentrations and no SOC stocks, shallow sampling depth) and  
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15 25 representativeness. In a global meta-analysis, 385 studies on land use change in the tropics  
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17 26 were explored to estimate the SOC stock changes for all major land use change types. The  
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19 27 highest SOC losses were caused by conversion of primary forest into cropland (-25%) and  
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21 28 perennial crops (-30%) but forest conversion into grassland also reduced SOC stocks by 12%.  
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31 33 i) the bulk density changes with land use and needs to be accounted for when calculating SOC  
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35 35 compare land use types on the same basis of soil mass. Without soil mass correction, land use  
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37 36 change effects would have been underestimated by 28%. Land use change impact on SOC  
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39 37 was not restricted to the surface soil, but relative changes were equally high in the subsoil,  
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41 38 stressing the importance of sufficiently deep sampling.  
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## 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<sub>2</sub>. 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<sup>-1</sup> 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<sup>-1</sup>. 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

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3 65 (SOC) stocks are the major uncertainty in such accounting schemes and in life cycle  
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5 66 assessments of tropical agricultural products.  
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10 68 SOC changes are controlled by i) the decomposition rate of SOC, e.g., due to changes in  
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12 69 microclimate, and ii) alterations in the quantity and quality of C cycled through the system  
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14 70 (Juo *et al.*, 1996). Land use directly affects both microclimate and quantity, quality and the  
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16 71 pathways of C input. Moreover, erosion is controlled by land use and land management and  
17  
18 72 may decrease SOC stocks in agricultural systems compared with forests. Erosion may be a  
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20 73 major pathway of SOC loss on the plot scale on insufficiently aggregated soils typical for  
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22 74 tropical regions (van Noordwijk *et al.*, 1997, Berhe *et al.*, 2007). On the other hand,  
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24 75 erodibility generally decreases with increasing topsoil SOC content, which stresses the  
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26 76 importance of SOC for soil fertility and productivity (Feller *et al.*, 1997). This is especially  
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28 77 true for tropical regions where nutrient poor, highly weathered soils are often managed with  
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30 78 few external inputs of nutrients and C. Tropical SOC stocks may be more susceptible to  
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32 79 perturbations such as land use changes with twice as high SOC turnover than in temperate  
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34 80 regions (Trumbore, 1993, Six *et al.*, 2002, Penman *et al.*, 2003). Higher temperature and soil  
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36 81 moisture regimes enhance decomposition rates and thus may speed up SOC losses. Highly  
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38 82 weathered soils, e.g., Oxisols and Ultisols, cover 60 to 70% of tropical land area. In these  
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40 83 soils low activity clays are predominant and provide less mineral surfaces for physical  
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42 84 protection and stabilisation of SOC (Feller *et al.*, 1997). However, there is an ongoing  
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44 85 discussion about whether climatic factors or the differences in soil mineralogy and land use  
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46 86 history contribute most to distinct tropical SOC dynamics (Paul *et al.*, 2008, Zinn *et al.*, 2005,  
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48 87 Zinn *et al.*, 2007, Feller *et al.*, 1997).  
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60 89 During the last years, many new research programs and projects have aimed to improve the  
90 understanding on carbon fluxes and balances in tropical soils. Hundreds of new studies were

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3 91 published but have never been analysed together. Moreover, insufficient sampling depth and  
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5 92 missing correction for differences in bulk density after land use change may have lead to  
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8 93 significant bias in previous studies (Ellert *et al.*, 1995, Baker *et al.*, 2007). Differences in  
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10 94 rooting depth and tillage on croplands directly influence the C distribution in the soil profile.  
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12 95 Shallow sampling misses C which is incorporated below the topsoil and may lead to  
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14 96 overestimations of land use change effects on soil C. Thus, the objective of this study was to  
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16 97 gather the existing high quality data sets on land use change effects and SOC for the tropics to  
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18 98 derive new estimates beyond site specific values and including also subsoil horizons. More  
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20 99 than 380 old and new data sets were compiled and quantitatively analysed. This study  
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22 100 provides the first estimate of tropical soil C stock changes after land use change for the depth  
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25 101 0-30 cm, the soil depth that has to be reported under UNFCCC, and below.  
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## 103 **2 Material and Methods**

### 104 **2.1 Data sources and compilation**

105 Data from 385 studies from 153 published and mostly peer reviewed publications on the  
106 influence of land use changes on soil organic carbon were compiled. Data were derived from  
107 39 different tropical countries covering all continents ranging from semi-arid regions such as  
108 southern Africa and northern Australia to the humid tropics along the equator. Twelve  
109 different land use change types were classified and investigated covering all land use  
110 transitions occurring in the tropical zone (Tab. 1). Most studies were conducted in paired plot  
111 design using the “space for time” approach. Since SOC may reach a new equilibrium only  
112 after several years or decades, there was almost no study with a time series going back to  
113 prior land use change conditions. Thus, for each paired site, it has to be assumed that soil  
114 conditions were similar before the land use change. Studies were rejected for the data  
115 compilation if the different land use types were i) confounded by different soil types as, e.g.,  
116 indicated by significant differences in texture, ii) sampled for different soil depth or iii)



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3 117 reporting only short term effects (< 5 years). For chronosequences only data from the longest  
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5 118 treatments of land use change were used in this study. With the exclusion of short term  
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8 119 studies, the influence of the time period since land use change on the estimated SOC changes  
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10 120 was minimized and not detectable anymore. Data on SOC stocks, bulk density and the  
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12 121 associated meta-data were compiled. Organic layers (forest floor) are rare in tropical forests  
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15 122 and the few existing data sets did not allow them to be included in this analysis.

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17 123 In the current study, primary forest is defined as natural vegetation without apparent and  
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19 124 reported human impacts. The primary forest vegetation class also comprises natural  
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21 125 vegetations with shrubland and non-managed grassland with savannah-like characters, e.g.,  
22  
23 126 the South American Cerrado. It has to be noted that there are only few remaining totally  
24  
25 127 undisturbed tropical forests leading to a rather broad definition of “primary forest” in many  
26  
27 128 studies and consequently also in our study (Lugo *et al.*, 1993). Secondary forests are managed  
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29 129 forests and forests regrown after destruction or partial exploitation of the primary forest.  
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31 130 Natural successions and fallow older than 7 years were classified as secondary forest.  
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33 131 “Grassland” comprises pastures but no natural grasslands, since natural grasslands are mostly  
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35 132 savannah type grasslands with tree and shrub vegetation. Additionally, there is no harvested  
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37 133 fraction of net primary production on natural grasslands, which directly affects the C  
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39 134 dynamics. Croplands are classified as “perennial crops,” such as sugar cane and coffee  
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41 135 plantations, and “croplands,” with annual crops such as maize and beans. Both cropland types  
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43 136 and grasslands were described as agricultural systems.  
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## 52 138 **2.2 Data treatment and missing bulk densities**

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55 139 For 81% of the reported data, SOC stocks were directly available or calculated as

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

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60 141 volume [m<sup>3</sup> ha<sup>-1</sup>] (Eq 1),

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3 142 where  $n$  was the number of soil layers, varying for each study. Bulk density data were  
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5 143 available for 52% of all reported soil horizons. Most studies reported SOC stocks (81%) but  
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7 144 19% of the studies reported only SOC concentrations. Two strategies can be used to handle  
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9 145 the problem of missing bulk density data: either all studies without bulk density  
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11 146 measurements are excluded from the meta-analysis, or estimated mean bulk densities replace  
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13 147 the unknown values to convert SOC concentrations into stocks. To quantify the difference in  
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15 148 accuracy between the two approaches we first conducted the meta-analysis (see below) based  
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17 149 only on those studies with bulk density measurements. For the second approach the data set  
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19 150 was divided in two sub datasets comprising the studies with and without bulk density  
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21 151 measurements. For those studies lacking own bulk density data, the bulk densities before and  
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23 152 after land use change were simulated as two-dimensional truncated normal random vectors  
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25 153 separately for each land use change type. For these Monte Carlo simulations the means and  
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27 154 covariances were derived from the studies with bulk density measurements. The truncation  
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29 155 was necessary to avoid unrealistic (e.g., negative) values for bulk density caused by the  
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31 156 unbounded normal distribution. The standard deviation of the estimated mean effect size from  
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33 157 10000 repeated simulations provided a direct estimate of the uncertainty introduced by using  
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35 158 an estimated mean bulk density instead of true measurements. Finally, estimates of mean  
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37 159 effect sizes and their standard errors were obtained as weighted averages of the estimates  
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39 160 from the two sub datasets. These Monte Carlo simulations revealed that for all land use  
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41 161 change types the estimated uncertainty can be reduced by including also the studies that  
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43 162 reported only SOC concentrations after converting them with weighted mean bulk densities  
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45 163 into SOC stocks. Thus, we decided to also include studies that report only SOC  
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47 164 concentrations.  
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51 166 SOC stocks were corrected to an equivalent soil mass on both land use types (Ellert *et al.*,  
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53 167 1995). Weighted mean bulk densities for each land use change type were used if bulk density  
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3 168 data were not available to perform this correction. Data reported as SOM concentrations were  
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5 169 converted to SOC by multiplying with a conversion factor of 0.58 (Mann, 1986). Studies were  
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8 170 restricted to mineral soils. Wetlands soils such as peatlands and paddy soils were not included  
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10 171 in this analysis, mainly due to an insufficient number of studies on these soil types to obtain  
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12 172 an adequate representation compared to non-wetland soils. Soil horizons down to max 100 cm  
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14  
15 173 were included in the analysis. The following relevant meta-data were also included in the  
16  
17 174 compilation: time since conversion (age), clay content (texture), soil type, mean annual  
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19 175 precipitation, mean annual temperature, soil sampling depth and other management factors  
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21 176 (tillage, species, fertilisation etc.). If some of these data were not available, data were not  
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23 177 estimated by interpolation or transfer functions and the study was excluded from the  
24  
25 178 corresponding analysis. Only for climatic data were other sources used such as long term  
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27 179 climate records of the region.  
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## 34 181 **2.2 Meta-analysis**

35  
36 182 The simplest measure of effect size  $\delta$  commonly employed in a meta-analysis is the difference  
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38 183 between control group mean  $\mu_c$  (before land use change) and treatment group mean  $\mu_e$  (after  
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40 184 land use change). We used both the absolute effect size  $\delta_{abs,i} = \mu_{e,i} - \mu_{c,i}$  and the relative effect  
41  
42 185 size  $\delta_{rel,i} = (\mu_{e,i} - \mu_{c,i}) / \mu_{c,i} * 100\%$ . To account for the different accuracy of the heterogeneous set  
43  
44 186 of studies, the mean effect size for the different land use change types was estimated as  
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46 187 weighted mean with the optimal weights being inversely proportional to the variance of the  
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48 188 single-study effect sizes. To estimate the optimal weights we had to estimate these variances.  
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51 189 Two sources of variability contribute to the uncertainty of the effect sizes, the within-study  
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53 190 variability derived from sampling and analytical errors and the between-study variability  
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55 191 derived from differences in climate, soil, plant species and land management between studies.  
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57 192 Since 73% of all studies did not report any measure of variability/accuracy for their estimated  
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59 193 means  $\mu_{c,i}$  and  $\mu_{e,i}$ , we used the available information to estimate the underlying within-study  
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3 194 variances  $\sigma_{within_{c,i}}^2$  and  $\sigma_{within_{e,i}}^2$  as a power function of SOC stocks ( $R^2=0.67$  for a linear  
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6 195 regression on a log-log scale). The standard errors of reported SOC stocks estimates for a  
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9 196 single study are then determined by the differing sample sizes  $n_i$  and can be estimated as:

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$$11 \hat{\sigma}_{\mu_{within,i}}^2 = \frac{\hat{\sigma}_{within,i}^2}{\sqrt{n_i}} \quad \text{Eq (2)}$$

$$12$$

$$13$$

$$14$$

15 198 for both means,  $\mu_c$  and  $\mu_e$ , respectively. The main uncertainty is derived from the field  
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18 199 heterogeneity and therefore, the sample size  $n$  is the number of collected soil samples, even  
19  
20 200 though they have been pooled to compound samples before analysis. Between-study  
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22 201 variability was estimated by the moment estimator

$$23$$

$$24$$

$$25 202 \hat{\sigma}_{between}^2 = \frac{1}{k-1} \sum_{i=1}^k (\mu_i - \bar{\mu})^2 - \frac{1}{k} \sum_{i=1}^k \frac{\hat{\sigma}_{within,i}^2}{n_i} \quad \text{Eq (3)}$$

$$26$$

$$27$$

$$28$$

29 203 resulting in the following estimated variance for the absolute effect size  $\delta_{abs,i}$ :

$$30$$

$$31$$

$$32 204 \hat{\sigma}_{abs,i}^2 = \hat{\sigma}_{between,c}^2 + \hat{\sigma}_{\mu_c,within,i}^2 + \hat{\sigma}_{between,e}^2 + \hat{\sigma}_{\mu_e,within,i}^2 - 2 \hat{cov}(\mu_{c,i}, \mu_{e,i}) \quad \text{Eq (4)}$$

$$33$$

$$34$$

35 205 The mean absolute effect size  $\delta_{abs}$  was then estimated as the weighted average

$$36$$

$$37$$

$$38 206 \hat{\delta}_{abs} = \frac{\sum_{i=1}^k \left( \frac{1}{\hat{\sigma}_{abs,i}^2} \cdot \hat{\delta}_{abs,i} \right)}{\sum_{i=1}^k \frac{1}{\hat{\sigma}_{abs,i}^2}} \quad \text{Eq (5)}$$

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45 207 For the variance estimation of the mean relative effect sizes  $\delta_{rel,i}$ , we used the same weights

$$46$$

$$47$$

$$48 208 \hat{\delta}_{rel} = \frac{\sum_{i=1}^k \left( \frac{1}{\hat{\sigma}_{abs,i}^2} \cdot \hat{\delta}_{rel,i} \right)}{\sum_{i=1}^k \frac{1}{\hat{\sigma}_{abs,i}^2}} \quad \text{Eq (6)}$$

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55 209 Estimates of standard errors for the weighted means were obtained by nonparametric  
56  
57 210 bootstrap based on 1000 bootstrap samples.

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59 211 Four different sampling depths were selected in order to investigate the effects of land use  
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212 change for different soil depth: Topsoil (0-10 cm), 0-30 cm depth (ploughing horizon),

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3 213 subsoil (>20 cm depth) and full depth (total sampling depth of each study). Studies reporting  
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5 214 only data for some soil depth classes were however included for the analysis of the reported  
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7  
8 215 soil depth class which led to different numbers of studies for the four soil depth classes. There  
9  
10 216 was no significant influence of the maximum sampling depth on the relative SOC changes (as  
11  
12 217 fixed and variable effect in the general linear models;  $F_{133,4,1} = 0.097$ ,  $p=0.76$ ), indicating that  
13  
14 218 the variability between different studies is as large as between different maximum sampling  
15  
16 219 depths.  
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20 220

### 21 221 **2.3 Statistical analysis**

22 222 The influence of the following variables on SOC change and bulk density change was  
23  
24 223 investigated on a core data set which contained all of the following variables comprising 377  
25  
26 224 land use pairs: Mean annual temperature, annual precipitation, soil mineralogy (using the  
27  
28 225 three clay type classes: high activity clay, low activity clay, allophanic clay), region/continent,  
29  
30 226 sampling depth and SOC stocks before land use change. Additionally, the methodological  
31  
32 227 parameters maximum sampling depth and number of sample replicates were used as  
33  
34 228 independent variables. We checked the effects with mixed linear models including the author  
35  
36 229 of the studies as a random effect. Since no author-specific effects could be found, we used  
37  
38 230 classical general linear models for the further analysis. Data are presented in the text as  $F_{\text{Sum of}}$   
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40 231 squares, degrees of freedom and the p-value. Statistical analysis was performed using R software.  
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## 50 233 **3 Results**

### 51 234 **3.1 Data quality and mass correction**

52 235 The analysis of land use effects on soil carbon is hampered by the high heterogeneity of the  
53  
54 236 data set including different sampling methods, sampling intervals and missing meta-data such  
55  
56 237 as the land use history. Most important for improved estimates of SOC changes is the  
57  
58 238 availability of bulk density data in order to account for SOC changes on an area basis and to  
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3 239 be able to correct data for different soil mass sampled in land use types. The correction for  
4  
5 240 different soil mass (bulk density) of land use types increased the land use change effect by an  
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7  
8 241 average of 28% (Tab. 2). Pedo-transfer functions (Post *et al.*, 2000, De Vos *et al.*, 2005,  
9  
10 242 Mann, 1986) are rarely able to take these effects into account with sufficient accuracy and we  
11  
12 243 were able to predict bulk density in our study only with a correlation coefficient of 0.67 (data  
13  
14  
15 244 not shown). However, the uncertainty of the estimated SOC change could be decreased by  
16  
17 245 combining studies with and without density measurements. The fraction of studies reporting  
18  
19 246 only SOC concentrations per land use change type was between 6 and 36%, indicating that  
20  
21 247 the majority of studies reported bulk densities and stocks. The uncertainty has been reduced  
22  
23  
24 248 by on average 52% as compared to a meta-analysis only comprising studies with bulk density  
25  
26  
27 249 managements. With our approach, we take the uncertainty derived from an incomplete data  
28  
29 250 set (e.g. missing bulk density data) and the uncertainty of SOC and bulk density  
30  
31 251 measurements into account. The within-study uncertainty depends on the sampling and  
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34 252 analytical errors and the soil heterogeneity in the field and could be reduced by increasing  
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36 253 numbers of soil samples (Eq. 2).  
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41 255 Land use change caused changes in the bulk density of slightly lower magnitude like SOC  
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43 256 changes and in the reverse direction (Fig. 1, Tab. 2). Since organic carbon has an inherent low  
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45 257 bulk density, SOC concentrations directly affect soil bulk density (Lal *et al.*, 2001). However,  
46  
47  
48 258 land use changes affect bulk density beyond this effect due to compaction by animal  
49  
50 259 trampling, machinery and loosening by tillage. The cultivation of forests caused bulk density  
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52  
53 260 to increase by 5 to 23% with the strongest increase occurring in the surface, lessening with  
54  
55 261 depth with no significant effects below 20 cm depth (Fig. 1). Surprisingly, even tillage on  
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58 262 croplands did not decrease bulk density, but cropland bulk density was always higher than  
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60 263 grassland and forest bulk density.

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3 265 Rates of land use change vary widely among tropical regions but the number of existing  
4  
5 266 studies did not reflect land use change rates. Malaysia and Indonesia are among the countries  
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7  
8 267 with the highest emissions due to land use changes (Houghton, 2003, FAO, 2006). However,  
9  
10 268 the region of South Eastern Asia was undersampled with only 11 studies reporting  
11  
12 269 quantitative data on land use effects on SOC. Most regions in Africa – except for Nigeria –  
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14  
15 270 were also undersampled, whereas good data coverage has been reached in Central and South  
16  
17 271 America, especially in Ecuador, Costa Rica and Brazil.  
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20 272

### 21 22 273 **3.2 Primary forest to agricultural land**

23  
24 274 The conversion of native vegetation to agricultural systems caused the highest SOC losses  
25  
26 275 among all land use change types (Tab. 1, Fig. 2 and 3). Native vegetation such as primary  
27  
28 276 forest and native grassland stored among the highest amounts of SOC ( $80 \pm 9$  Mg SOC ha<sup>-1</sup>,  
29  
30 277 mean sampling depth 36 cm). Conversion of primary forest to cropland (-25%) caused twice  
31  
32 278 as high SOC losses than its conversion to grassland (-12%). The relative SOC loss in the  
33  
34 279 subsoil was similar on grasslands but not significant for croplands due to a great variability  
35  
36 280 caused by different management practices and crop types. With the cultivation of primary  
37  
38 281 forests, soils were compacted and bulk density increased by 14 and 18% for grasslands and  
39  
40 282 croplands, respectively. Especially if forests are converted to grasslands the correction for  
41  
42 283 different soil mass exerted a strong influence on the estimated SOC changes, switching this  
43  
44 284 land use change from almost C neutral to a significant C source (Tab. 2).  
45  
46 285 42% of the variability between data sets could be explained with different land use change  
47  
48 286 types and the climatic factors precipitation and temperature. SOC losses increased with  
49  
50 287 increasing temperature and for conversion of forest into grassland also with increasing  
51  
52 288 precipitation (Tab. 3 and 4). For forest cultivation we found no uniform influence of  
53  
54 289 precipitation on SOC changes and a higher uncertainty of the models as compared to forest  
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60 290 conversion to grassland.

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6 292 **3.3 Secondary forest to agricultural land**  
7  
8 293 Similar to primary forests also secondary forests' conversion to agriculture systems led to  
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10 294 SOC losses. However, relative SOC loss (for all depths) was less compared to primary  
11  
12 295 forests, indicating a higher vulnerability of SOC in primary forests than in secondary forests  
13  
14  
15 296 to land use changes. Surface soil SOC stocks remain unchanged when secondary forest was  
16  
17 297 converted to grassland (Fig. 3). Grasslands are characterised by a steep C gradient with soil  
18  
19 298 depth leading to high surface soil SOC stocks. In contrast, a smaller fraction of total SOC is  
20  
21 299 stored in the surface soil of secondary forests. SOC losses after deforestation were  
22  
23 300 significantly affected by climatic factors, in particular moisture conditions (mean annual  
24  
25 301 precipitation) and temperature (Tab. 3, Fig. 4). Surprisingly, we found no effect of the clay  
26  
27 302 content ( $F_{7750,5}$ ;  $p=0.58$ ; Fig. 5) and soil type ( $F_{53213,30}$   $p=0.45$ ) on SOC losses. However,  
28  
29 303 beside temperature and precipitation the differences between clay types (low activity, high  
30  
31 304 activity allophonic) exerted a significant effect on SOC changes (Tab. 3). Land use change  
32  
33 305 types, climate factors and clay type could explain 55% of the SOC change variance leaving  
34  
35 306 almost half of the data set variability unexplained. The effect of different management  
36  
37 307 practices for croplands (e.g., tillage vs. no tillage) could not be investigated due to a very  
38  
39 308 small number of studies covering both land use change and different management practices.  
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### 48 310 **3.4 Primary to secondary forest**

49  
50 311 Management of primary forest with wood extraction and planting of productive tree species  
51  
52 312 caused a mean SOC loss of 7% or 9 Mg SOC ha<sup>-1</sup> (Tab. 1). One major difference between  
53  
54 313 primary and secondary forest is the SOC distribution in the soil profile (Fig. 2 and 3), with a  
55  
56 314 higher surface SOC fraction in primary forest compared to secondary forests. 7 Mg SOC ha<sup>-1</sup>  
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58 315 (-15%) were lost in the upper 10 cm only after conversion of primary forest to secondary  
59  
60 316 forest. However, there was no significant SOC change below 20 cm depth. SOC losses were



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3 317 accompanied by a 6% increase in the soil bulk density leading to 28% higher SOC changes  
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5 318 after mass correction as compared with no mass corrected data (Tab. 2).  
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### 10 320 **3.5 Afforestation and fallow**

11  
12 321 SOC losses due to deforestations were partly reversible by afforestations of croplands or  
13  
14 322 grasslands. Estimated mean SOC stock gains for afforestations were even higher than SOC  
15  
16 323 losses from deforestations, with highest SOC gains in afforestations on former croplands (+  
17  
18 324 50%) compared to afforestations on former grassland (18%, Tab. 1). Afforested grassland  
19  
20 325 stored particular low amounts of SOC ( $60 \pm 9 \text{ Mg ha}^{-1}$ ) since afforestations were mainly  
21  
22 326 conducted on degraded grasslands or on marginal land with intrinsic low SOC storage  
23  
24 327 capacity (Tab. 1). Similarly, the termination of cropping leading to natural succession (fallow)  
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26 328 on croplands took place mainly on degraded land as indicated by low SOC stocks (Tab. 1).  
27  
28 329 Fallow increased SOC stocks by 32% indicating a rapid recovery of SOC stocks.  
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### 36 331 **3.6 Conversion of grassland to cropland and vice versa**

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38 332 Typical land use change cascades in the tropics are first the conversion of forest into  
39  
40 333 grassland for cattle grazing and at a later stage the conversion of grassland into cropland.  
41  
42 334 Cropland establishment on grasslands reduced SOC stocks by  $6 \text{ Mg C ha}^{-1}$  (-10%) but this  
43  
44 335 effect was restricted to the uppermost soil horizon. Subsoil below 20 cm depth was not  
45  
46 336 significantly affected by these land use changes due to high C input with tillage. Several  
47  
48 337 studies reported lower subsoil SOC stock in grassland compared to croplands (Fujisaka 1998,  
49  
50 338 Huges, 2000, Freitas 2000). Cropland conversion or re-conversion to grassland increased  
51  
52 339 SOC stocks by  $8 \text{ Mg C ha}^{-1}$  (+26%), which is more than the SOC loss after cropland  
53  
54 340 establishment on grassland. Similar to the afforestation of croplands, this indicates that  
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56 341 croplands management causes SOC losses leading to lower initial SOC stocks of croplands  
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58 342 before conversion into grasslands (Tab. 1).  
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5 344 **3.7 Perennial crops and plantations**

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8 345 The conversion of primary forests to perennial crops caused an even higher C loss of than the  
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10 346 conversion to cropland (-30%, Tab. 1). In contrast, the conversion of secondary forests to  
11  
12 347 perennial crops seems to hardly affect SOC stocks. This may be partly explained by higher  
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14  
15 348 mean SOC stock before land use change in primary forests than in secondary forests of  
16  
17 349 studies reporting conversion into perennial crops. For all land use change types SOC losses  
18  
19 350 and gains were weakly positively correlated with SOC stock before land use change. Most  
20  
21 351 data on soil carbon on perennial crops were reported from sugar cane plantations (28% of  
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23 352 studies including perennial crops), fruit tree plantations (including banana) (12%) and cacao  
24  
25 353 plantations (9%). These findings are based on 35 studies and indicate that a permanent  
26  
27 354 vegetation cover does not always prevent SOC losses under intensive management when SOC  
28  
29 355 rich forests are converted to perennial crop plantations. SOC changes may be different in low  
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31 356 input agro-forestry type perennial croplands which are not well covered in this meta-analysis.  
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38 358 **4 Discussion**39  
40 359 **4.1 Deforestation and afforestation**

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42 360 A large number of studies on land use change effects were conducted during the last years,  
43  
44 361 25% of the studies in this meta-analysis were published during the last five years and 67%  
45  
46 362 during the last 10 years. Moreover, it was only recently that more studies also included deeper  
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48 363 soil horizons down to 100 cm depth. Former reviews calculated higher global and tropical  
49  
50 364 SOC stock changes after cultivation of forests compared to our study (Davidson *et al.*, 1993,  
51  
52 365 Guo *et al.*, 2002, Detwiler, 1986, Paustian *et al.*, 1997, Amundson, 2001). This can partly be  
53  
54 366 explained with an improved data quality and quantity, e.g., with deeper sampling and more  
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56 367 data on bulk density changes. Detwiler (1986) found twice as high SOC losses after  
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58 368 deforestation than reported in our study (-20% for forest to grassland and -40% for forest to  
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3 369 cropland). A range of SOC losses from -24 to -43% was reported for cultivated tropical soils  
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5  
6 370 (Davidson *et al.*, 1993). The IPCC guidelines set a default value of -31 and -42% SOC for dry  
7  
8 371 and wet tropical regions after forest cultivation, respectively, which is higher than calculated  
9  
10 372 in our study (Tab. 5). Our study confirmed the impact of soil moisture and precipitation on  
11  
12 373 SOC dynamics with higher SOC changes in regions with higher precipitation for most land  
13  
14 374 use change types (Tab. 3, Fig. 4). Soils in humid regions maybe more vulnerable to land use  
15  
16 375 changes than in dryer regions (Brown *et al.*, 1990, Amundson, 2001). The impact of  
17  
18 376 precipitation seems to be stronger when forests are converted into grasslands than for forest  
19  
20 377 conversion into cropland (Tab. 5). In a global analysis Guo and Gifford (2002) found  
21  
22 378 conversion of forest to grassland to increase SOC stocks by 9% (2002), no SOC change has to  
23  
24 379 be assumed as default value under IPCC and is reported in other reviews (Lugo *et al.*, 1993,  
25  
26 380 Cerri *et al.*, 2004, Penman *et al.*, 2003). In contrast, we found tropical forests lost 12% SOC  
27  
28 381 after grassland establishment (Tab. 1). These differences can be partly attributed to improved  
29  
30 382 data quality and the application of a soil mass correction which accounts for changes in  
31  
32 383 different bulk densities in different land use types (Ellert *et al.*, 1995, Gifford *et al.*, 2003, de  
33  
34 384 Moraes *et al.*, 1996). Detwiler (1986) tried already to overcome the problem of different soil  
35  
36 385 mass but had to rely on calculated and not measured bulk densities. Soils were compacted by  
37  
38 386 10 and 16% due to forest conversion into grassland and cropland, respectively. Most tropical  
39  
40 387 grasslands are under higher grazing pressure, a higher biomass fraction is exported (harvest)  
41  
42 388 and fertilizer input is low compared to temperate grasslands. Improved grassland management  
43  
44 389 with the application of fertilizers would help to increase productivity and SOC stocks  
45  
46 390 compared to extensive pastures (Soussana *et al.*, 2007, Ammann *et al.*, 2007). Roots are a  
47  
48 391 more effective pathway to build up SOC stocks than foliar litter, which explains high  
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50 392 grassland SOC (Lugo *et al.*, 1993, Rasse *et al.*, 2005) and relatively small SOC losses if  
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52 393 forests are converted to grasslands as compared to croplands.  
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3 395 A major proportion of total SOC change occurred during the first few years after cultivation  
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5 396 of forest, indicating that these soils contain large amounts of labile SOC, potentially  
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7 397 vulnerable to degradation upon human-induced land-use changes (Solomon *et al.*, 2007). A  
8  
9 398 new equilibrium of SOC has been reached most often within 3 to 10 years (Houghton, 1999,  
10  
11 399 Feller *et al.*, 1997, Detwiler, 1986, Davidson *et al.*, 1993). Other studies found 20 to 40 years  
12  
13 400 (Solomon *et al.*, 2007, Sa *et al.*, 2001, Riezebos *et al.*, 1998, Cerri *et al.*, 2007). In our study,  
14  
15 401 the average time period since deforestation was 22 years, and 33 years since afforestation,  
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17 402 indicating that major parts of SOC changes are captured within this time period.  
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22 403  
23  
24 404 Reforestation and afforestation were found to successfully recover SOC stocks (Silver *et al.*,  
25  
26 405 2000, Post *et al.*, 2000, Bashkin *et al.*, 1998). Cropland afforestation increased SOC stocks by  
27  
28 406 33 Mg ha<sup>-1</sup> which is slightly lower than the mean SOC accumulation of 41 Mg ha<sup>-1</sup> after 80  
29  
30 407 years reported from Silver *et al.* (2000). SOC gains with afforestation were higher than SOC  
31  
32 408 losses after deforestation (Tab. 1). Forest establishment has mainly been performed with  
33  
34 409 highly productive tree species like eucalyptus with a low litter quality and high recalcitrance.  
35  
36 410 High SOC accumulation in secondary forest may be also fostered by a low initial SOC  
37  
38 411 content in the afforested degraded agricultural land (Lugo *et al.*, 1993). Agricultural  
39  
40 412 management on highly weathered soils often lead to a rapid decline in soil fertility, leaving  
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42 413 degraded land for forest regrowth, as it is part of the traditional shifting cultivation system.  
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## 51 415 **4.2 Agricultural systems**

52 416 Low SOC stocks in croplands have important implications for crop production since organic  
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54 417 matter supplies most of the nitrogen and parts of the phosphorous taken up by unfertilized  
55  
56 418 crops (Sanchez, 1976). SOC is essential for the retention of nutrients and water in highly  
57  
58 419 weathered soils with low cation exchange capacity. Tropical regions cover very different  
59  
60 420 stages of agricultural mechanisation and development with various management options on

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2  
3 421 croplands, including organic amendments and different tillage practice, and a high number of  
4  
5 422 different crops. Thus, the estimated effect of cultivation can only set a mean value for regions  
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7  
8 423 but cannot be applied to specific field sites. Climatic and soil parameters could only explain  
9  
10 424 55% of the data variability. For agricultural systems, the biomass fraction left for SOC build-  
11  
12 425 up (crop residuals) is strongly controlled by management practices including the selection of  
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14  
15 426 crop species. Improved cropland management may partly offset SOC losses due to  
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17 427 deforestation (Lugo *et al.*, 1993); 13% of croplands included in this meta-analysis reported  
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19 428 similar or higher SOC stocks than in forests. Additional effort with field data collection is  
20  
21 429 necessary to quantify the effect of different management options on a global scale. Moreover,  
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23 430 insufficient sampling depths were found to obscure conclusions on management and land use  
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25  
26 431 effects on the SOC balance (Baker *et al.*, 2007).

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31 433 Regular soil disturbance during tillage or harvest is one of the main reasons for low cropland  
32  
33 434 SOC stocks (Lal, 1998). Grasslands, pastures and perennial crops, unlike croplands, maintain  
34  
35 435 a permanent vegetation cover and a high root turnover leading to high SOC input. We found  
36  
37 436 surprisingly high SOC losses after primary forest was converted to perennial cropland or  
38  
39 437 grassland. The amount of crop residuals returned to the soil directly affect SOC, and most  
40  
41 438 perennial crops such as sugar cane plantations, are managed with high intensity and high  
42  
43 439 biomass export (Graham *et al.*, 2002). Similar to other cropland types, different management  
44  
45 440 options and land use history determined the amount of SOC loss after cultivation of primary  
46  
47 441 forests and, on the other hand, the sequestration potential if perennial cropland and grassland  
48  
49 442 is afforested.

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### 53 444 **4.3 Soil characteristics and erosion**

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56 445 Differences in SOC change are expected to be attributable to soil parameters such as soil type,  
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58 446 texture and clay mineral type (de Moraes *et al.*, 1996, Feller *et al.*, 1997, Hartemink, 1997,

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3 447 Davidson *et al.*, 1993). Clay type was found to explain only additional 13% of the SOC  
4  
5 448 variability, beside land use change type, temperature and precipitation (Tab. 3). No influence  
6  
7 449 on SOC change could be attributed to soil type and clay content (Fig. 5), which is in line with  
8  
9 450 findings from Davidson *et al.* (1993). Soil parameters' influence maybe obscured by  
10  
11 451 dominant other factors such as climate and management. Additionally, the data availability  
12  
13 452 was low with only 22% of all studies reporting clay content. Highly weathered soils, such as  
14  
15 453 Ferralsols were found to loose more SOC after cultivation than other soil types (Hartemink,  
16  
17 454 1997), but this could not be confirmed in our study.  
18  
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23  
24 456 Erosion is a major factor affecting SOC stocks that is directly related to land use and forest  
25  
26 457 clearing (Nye *et al.*, 1964, van Noordwijk *et al.*, 1997, Wairiu *et al.*, 2003). Soils under low  
27  
28 458 vegetation cover (agricultural systems, conventional tillage), on steep slopes and under high  
29  
30 459 precipitation intensity are most prone to erosion. However, adequate data was not available in  
31  
32 460 this meta- analysis to assess the proportion of erosion-triggered SOC loss. Some areas may  
33  
34 461 even gain SOC with deposition of eroded material, leaving the question open of whether  
35  
36 462 erosion decreases or increases the terrestrial C sink (Berhe *et al.*, 2007, Lal, 2003, Van Oost *et*  
37  
38 463 *al.*, 2007).  
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#### 465 **4.4 SOC changes in the surface soil and the subsoil**

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47 466 The SOC in topsoil is supposed to be more prone to land use change and other perturbations  
48  
49 467 than subsoil (Siband, 1974, Veldkamp *et al.*, 2003, Veldkamp, 1994). We found equally high  
50  
51 468 relative subsoil SOC stock changes compared with surface soil horizons after conversion of  
52  
53 469 native forests to agriculture systems (Fig. 2 and 4). Native forests stored higher amounts of  
54  
55 470 subsoil C which are lost upon cultivation compared with secondary forests. The mean soil  
56  
57 471 sampling depth did not contribute to an explanatory model indicating that the relative SOC  
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59 472 change is only weakly related to the soil depth. In contrast, absolute SOC changes  
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3 473 significantly decrease with soil depth owing to decreasing absolute SOC stocks in deeper  
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6 474 horizons. In fact, tillage may even increase subsoil SOC stocks in croplands due to C rich  
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8 475 topsoil being mixed with deeper horizons (Hughes *et al.*, 2000, Fujisaka *et al.*, 1998). A  
9  
10 476 sampling depth as deep as the tillage depth is the minimum to quantify land use change  
11  
12 477 effects. Our results indicate that at least the conversion of native forests also affects subsoil  
13  
14 478 SOC below 20 cm depth and a comprehensive assessment should also include subsoil  
15  
16 479 horizons, if possible down to 100 cm depth. In order to estimate subsoil SOC changes with  
17  
18 480 land use change it is even more important to ensure comparable soil intrinsic conditions on  
19  
20 481 paired or chronosequence sites since SOC stabilisation in the subsoil is highly dependent on  
21  
22 482 soil mineralogy, texture and other soil parameters. The high variability of subsoil SOC change  
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24 483 may be a result of the variability of these soil intrinsic parameters.  
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#### 31 485 **4.5 Bulk density change and its impact of SOC stock estimates**

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34 486 The relative changes in bulk density were almost as high as the relative SOC changes, e.g.,  
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36 487 cultivation of forest increased bulk density by 16 % (Fig. 1, Tab. 2). Bulk density changes are  
37  
38 488 important to account for SOC stocks changes, since SOC stocks linearly depend on both SOC  
39  
40 489 concentration and bulk density. Moreover, bulk density change causes a sampling bias if  
41  
42 490 sampling of each land use type is performed at the same sampling depth (Ellert *et al.*, 1995,  
43  
44 491 Gifford *et al.*, 2003, Davidson *et al.*, 1993). If bulk density increased with land use change,  
45  
46 492 the soil is compacted and sampling down to the same sample sampling depth would lead to  
47  
48 493 higher sampled soil mass than in the corresponding land use type. Since soil mass and soil  
49  
50 494 carbon are ultimately linked, sampling of more soil mass results in higher SOC stocks  
51  
52 495 (Davidson *et al.*, 1993). Thus, the effect of land use change is underestimated, in our study by  
53  
54 496 an average 28% (Tab. 2). This can only be completely corrected if bulk density data prior to  
55  
56 497 land use change are available and it can partly be corrected if bulk density data were recorded  
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58 498 after land use change for both land use types (Lee *et al.*, 2009). We found mass correction  
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3 499 strongly influencing the effect size (SOC change) with up to three times higher mean SOC  
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5 500 changes than estimated without mass correction. However, Monte Carlo simulations revealed  
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7 501 that the exclusion of studies that report only SOC concentration would increase the  
8  
9 502 uncertainty of the estimated SOC change by 52%. The high diversity of soil types, climate  
10  
11 503 conditions, vegetation and management types call for as many studies as possible to be  
12  
13 504 included in such meta-analysis, even though not all studies provide the full parameter set.  
14  
15 505 This confirms earlier findings that fewer bulk density than SOC concentration measurements  
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17 506 are necessary to estimate SOC stocks (Don *et al.*, 2007). Coefficient of variation (CV) of all  
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19 507 studies was 2.7 times lower for bulk density than for SOC concentration (29 and 81% for bulk  
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21 508 density and SOC concentration, respectively) indicating that even with few bulk density  
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23 509 measurements, uncertainties on land use change effect can be reduced considerably.  
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## 32 511 **5 Conclusions**

33  
34 512 The conversion of forest, especially primary forests into agricultural systems always lead to  
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36 513 SOC losses, but losses are reversible to a high degree if, e.g., agricultural land is afforested or  
37  
38 514 properly managed. For the SOC balance of a land use system, the harvested fraction of net  
39  
40 515 primary production seems to be more important than its disturbance frequency, e.g., with  
41  
42 516 tillage or climate or soil characteristics.  
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45 517 Mass correction of SOC stock estimates is crucial in order to estimate land use change effects  
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47 518 since land use change is always accompanied by bulk density changes. The comparison of  
48  
49 519 SOC stocks based on different soil mass deeply confound estimates of SOC changes. This  
50  
51 520 meta-analysis provides soil mass-corrected estimates to improve the current UNFCCC default  
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53 521 values. Mean SOC changes were smaller than reported in previous reviews even though soil  
54  
55 522 mass correction increased land use change effects on SOC by 28% on average.  
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58 523 The global data coverage does not mirror the current hot spots of land use changes. New  
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60 524 effort are needed to quantify the effect of land use changes in South East Asia and Africa, also



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3 525 taking to account carbon rich wetland forests and degradation cascades within land use  
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5 526 classes.

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10 528 **Acknowledgements**

11  
12 529 We thank the library of the Max Planck Institute for Biogeochemistry (Jena) and of the  
13  
14 530 Johann Heinrich von Thünen Institute (Braunschweig) for their support. We thank Jana  
15  
16 531 Niebuhr for her help with extracting the uncertainties from the studies. This study was partly  
17  
18 532 financed by the Federal Ministry of Education and Research (BMBF) (FKZ 01LG0801A).  
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For Review Only

682 **List of Tables**

683

684 Table 1: Mean absolute and relative SOC stocks changes and reported minimum and  
 685 maximum relative SOC stock changes for different land use change types. Additionally, SOC  
 686 stocks before land use change, mean full available sampling depth (on average 32 cm), mean  
 687 time interval between the two land use systems and the number of studies included in this  
 688 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	
		Min	Max					
	Mg ha <sup>-1</sup>	%	%	%	Mg ha <sup>-1</sup>	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 ( $\pm$  SE in brackets).

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

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693 Table 3: General linear model with degrees of freedom (Df), sum of squares (sum of Sq), F-  
 694 value, P-value. LUC= land use change type, MAT= Mean annual temperature, MAP= Mean  
 695 annual precipitation, max depth= maximum sampling depth [cm], significance codes: '\*\*\*'  
 696 <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 *						55
Clay mineral type	34	44853	2.24	<0.000	***	
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.	

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

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

699 MAT: Mean annual temperature

700 MAP: Mean annual precipitation

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2  
3 701 Table 5: Fraction of original soil carbon stock for 0-30 cm depth remaining after land use  
4  
5 702 change. Revised default values for tropical regions from the IPCC Good Practice Guidelines  
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7  
8 703 (GPG) LULUCF (Penman *et al.*, 2003) and from this meta-analysis.  
9

<b>SOC stock change</b>	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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3 705 **List of Figures**  
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8 707 Figure 1: Weighted average relative bulk density change [%] for different soil depth for  
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10 708 different land use change types derived from all studies reporting bulk density measurements.  
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15 710 Figure 2: Absolute SOC stock changes [ $\text{Mg C ha}^{-1}$ ] for different soil depth for different land  
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17 711 use change types. The different depth increments are covered by different numbers of studies.  
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22 713 Figure 3: Relative SOC stock changes [%] for different soil depth for different land use  
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24 714 change types. The different depth increments are covered by different numbers of studies.  
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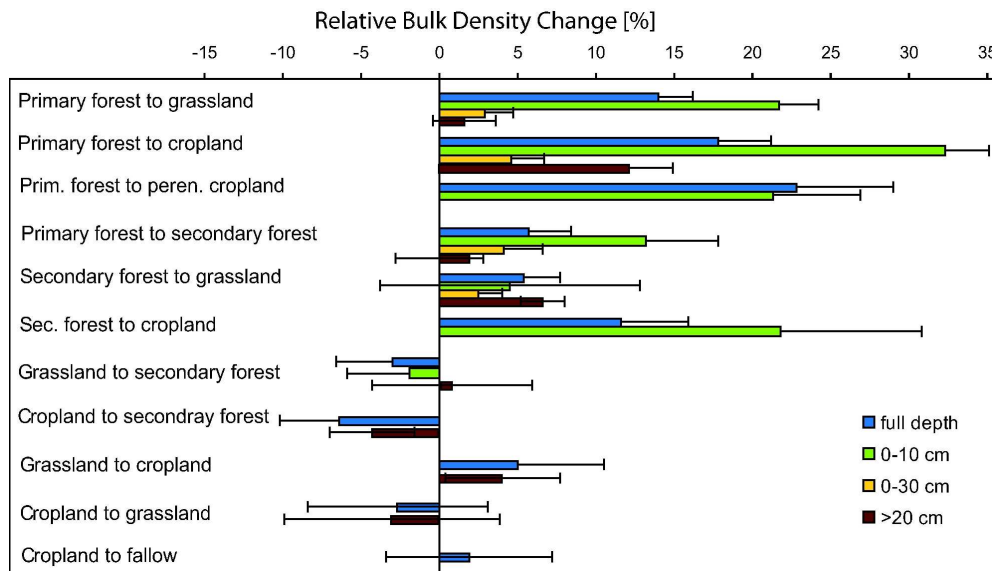
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29 716 Figure 4: SOC stock change [ $\text{Mg ha}^{-1}$ ] after conversion to grassland (open symbols) and  
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31 717 cropland (filled symbols) vs. Mean Annual Precipitation [mm].  
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36 719 Figure 5: SOC stock change [ $\text{Mg ha}^{-1}$ ] after conversion to grassland (filled symbols) and  
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38 720 cropland (open symbols) vs. mean content clay of the soil [%].  
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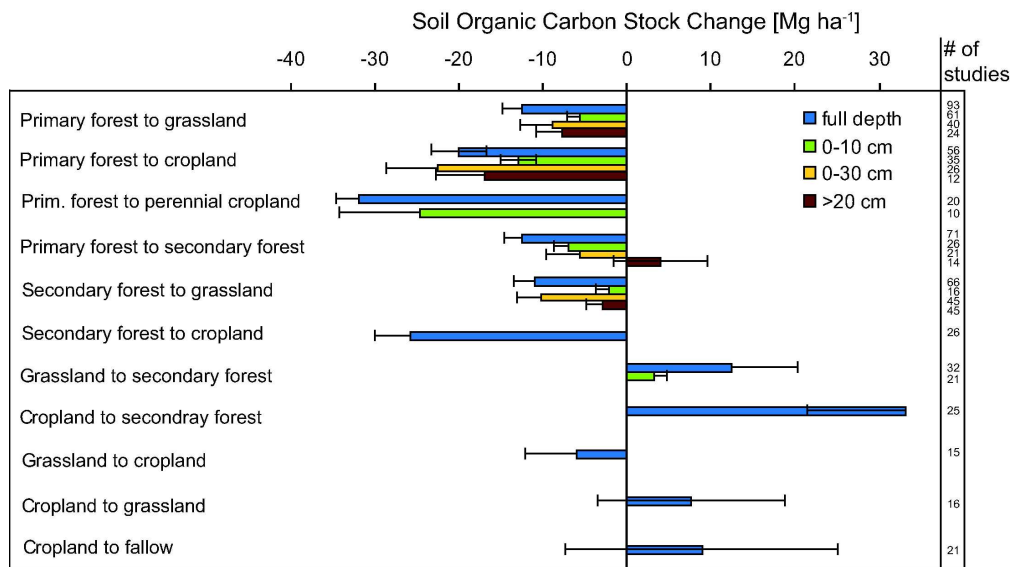


Weighted average relative bulk density change [%] for different soil depth for different land use change types derived from all studies reporting bulk density measurements.

169x96mm (600 x 600 DPI)

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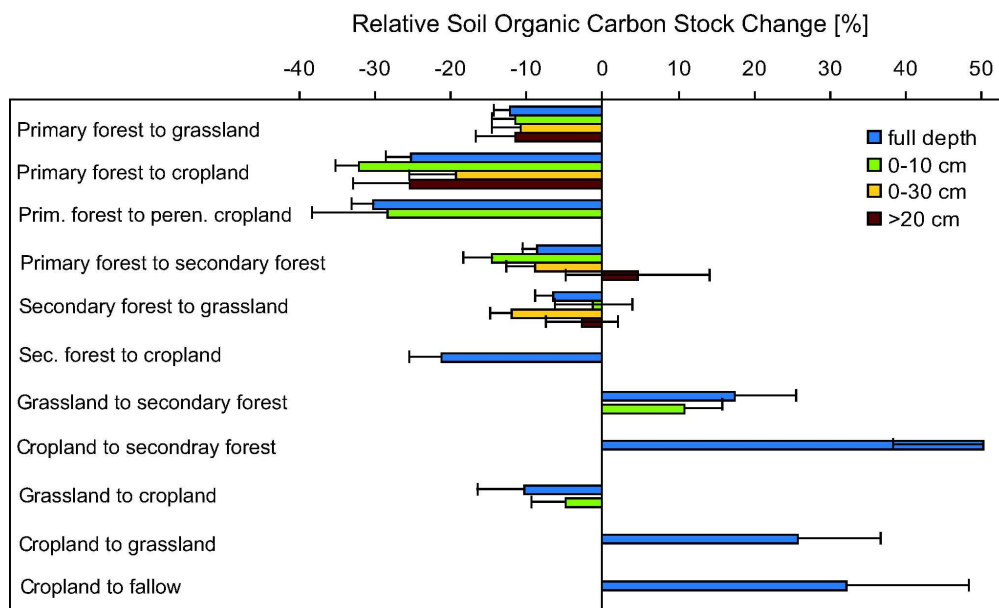
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Absolute SOC stock changes [Mg C ha<sup>-1</sup>] for different soil depth for different land use change types.  
 The different depth increments are covered by different numbers of studies.  
 186x105mm (600 x 600 DPI)

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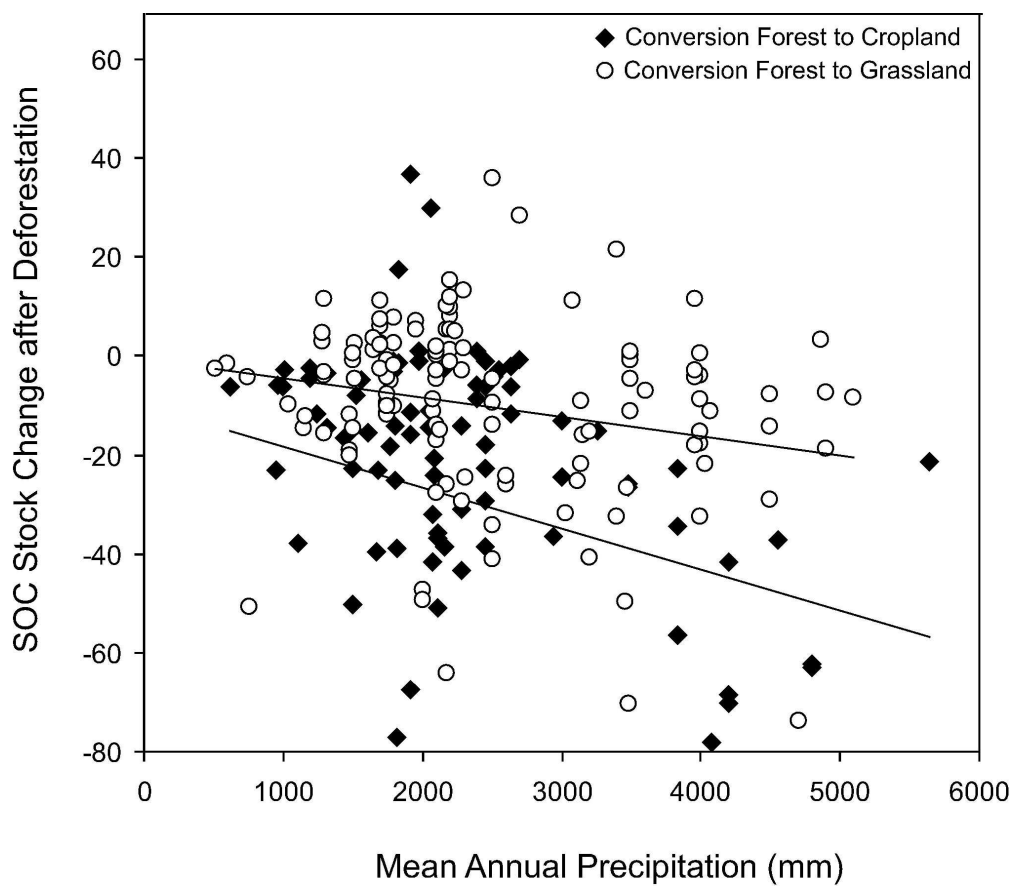
Relative SOC stock changes [%] for different soil depth for different land use change types. The different depth increments are covered by different numbers of studies.  
173x104mm (600 x 600 DPI)

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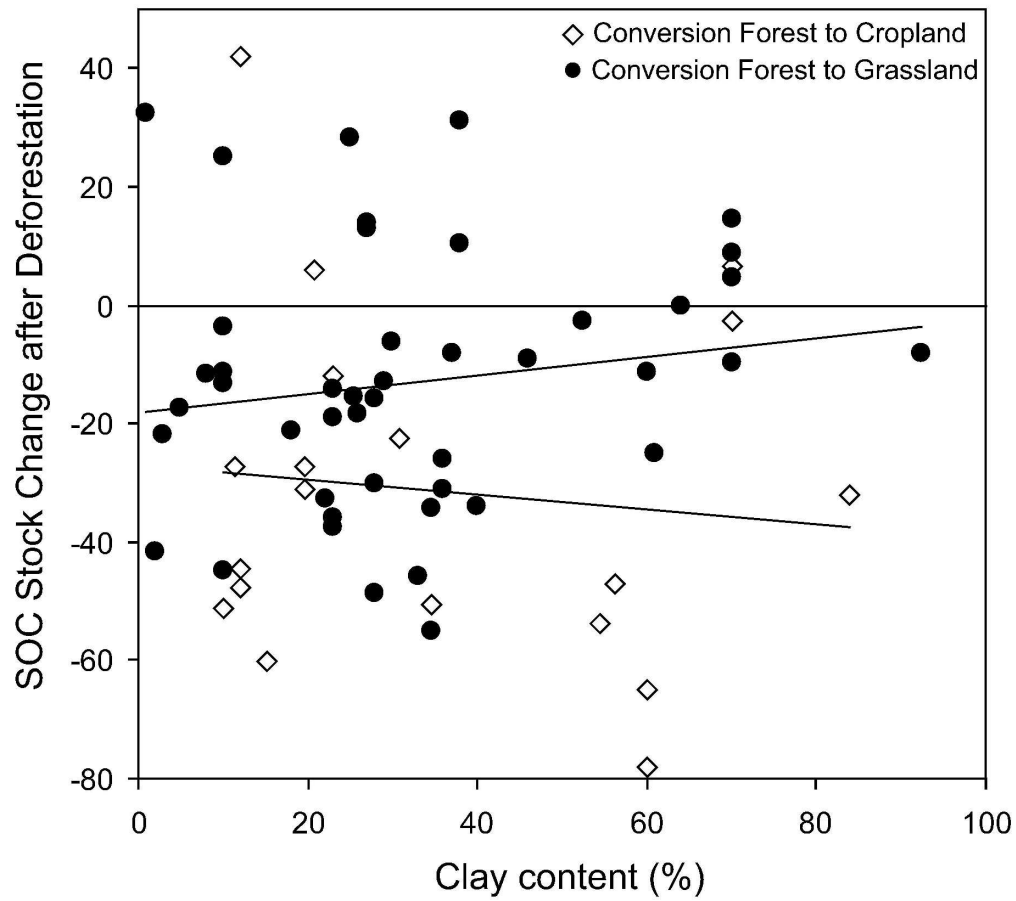


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SOC stock change [Mg ha<sup>-1</sup>] after conversion to grassland (open symbols) and cropland (filled symbols) vs. Mean Annual Precipitation [mm].  
171x150mm (600 x 600 DPI)





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