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Original research article

Global patterns of the effects of land-use changes on soil carbon stocks

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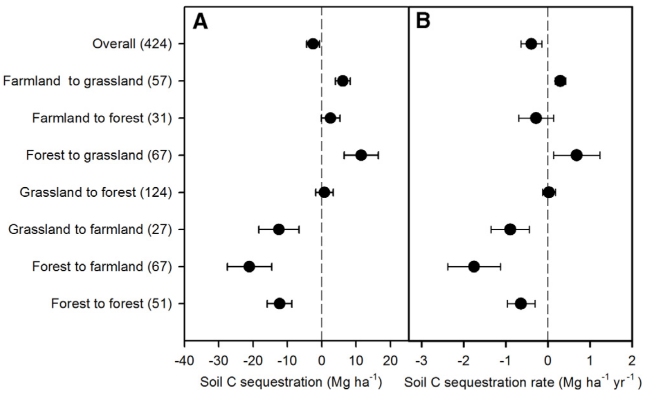
HIGHLIGHTS

- Soil C sequestrations varied in different land use change types.
- Soil C sequestration dynamics were not determined by age at the global level.
- Globally, land use conversions have significantly reduced soil C stock.

GRAPHICAL ABSTRACT



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ABSTRACT

Despite hundreds of field studies and at least a dozen literature reviews, there is still considerable disagreement about the direction and magnitude of changes in soil C stocks with land use change. This paper reviews the literature on the effects of land use conversions on soil C stocks, based on a synthesis of 103 recent publications, including 160 sites in 29 countries, with the aims of determining the factors responsible for soil C sequestration and quantifying changes in soil C stocks from seven land use conversions. The results show that as an overall average across all land use change examined, land use conversions have significantly reduced soil C stocks (0.39 Mg ha^{-1} yr^{-1}). Soil C stocks significantly increased after conversions from farmland to grassland (0.30 Mg ha^{-1} yr^{-1}) and forest to grassland (0.68 Mg ha^{-1} yr^{-1}), but significantly declined after conversion from grassland to farmland (0.89 Mg ha^{-1} yr^{-1}), forest to farmland (1.74 Mg ha^{-1} yr^{-1}), and forest to forest (0.63 Mg ha^{-1} yr^{-1}). And after conversion from farmland to forest and grassland to forest, soil C stocks did not change significantly. Globally, soil C sequestration showed a significant negative correlation with initial soil C stocks ($P<0.05$), and the

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effects of climatic factors (mean annual temperature and mean annual precipitation) on soil C sequestration varied between the land use conversion types. Also, the relationships between soil C sequestration and age since land use conversion varied in different land use change types. Generally, where the land use changes decreased soil C, the reverse process usually increased soil C stocks and vice versa. Soil C sequestration dynamics were not determined by age since land use conversion at the global level when all land use change types were combined.

1. Introduction

The terrestrial biosphere can act either as a source or as a sink for atmospheric CO_2, both the vegetation and the soil may play a part in the residual terrestrial uptake (IPCC, 2000; Guo and Gifford, 2002). Organic carbon stored in the world's soils is the largest terrestrial pool of carbon, and is at least three times larger than the pool of atmospheric carbon dioxide (Jobbágy and Jackson, 2000; Amundson, 2001). It has long been recognized that land-cover change and management can alter the amount of organic carbon stored in the soil (Laganière et al., 2010; Deng et al., 2014a), and this in turn affects both soil fertility and atmospheric carbon dioxide (CO_2) concentrations (Powers et al., 2011). Although the contributions of land-cover change to anthropogenic CO_2 atmospheric emissions have recently been revised downward, the estimated current annual contribution of 1.2 Pg, or about 12%–15% of total anthropogenic fluxes, is still significant (Van der Werf et al., 2009). Therefore, a new challenge in the context of climate change mitigation is the management of terrestrial ecosystem to conserve existing carbon stocks and to remove carbon from the atmosphere by adding to stocks.

Land use change can cause a change in land cover and an associated change in carbon stocks (Bolin and Sukumar, 2000; Deng et al., 2014a,b). The change from one ecosystem to another could occur naturally or be the result of human activity. Each soil has a carbon carrying capacity, and an equilibrium carbon content depending on the nature of vegetation, precipitation and temperature (Jobbágy and Jackson, 2000). The equilibrium between carbon inflows and outflows in soil is disturbed by land use change until a new equilibrium is eventually reached in the new ecosystem (Guo and Gifford, 2002). During this process, soil C stocks would have been changed, either as a carbon source or as a carbon sink. Despite hundreds of field studies and dozens of literature reviews, there is still considerable disagreement on the direction and magnitude of changes in soil C stocks with land-use change (Van der Werf et al., 2009). Indeed, the current knowledge remains inconclusive on both the magnitude and direction of C stock changes in mineral soils associated with land use type, management and other disturbances, and cannot support broad generalizations (IPCC, 2006).

Recently, some studies have reviewed the effects of certain land use changes on soil C stocks, for example, Post and Kwon (2000) found an average rate of soil C accumulation after afforestation of former farmlands with the value of 0.338 Mg ha^{-1} yr^{-1} although values vary greatly among studies, however, Vesterdal et al. (2002) observed that afforestation of former farmlands did not lead to increase of SOC in three decades. In addition, for a tropical regions study, the conversion of forests to farmland reduced soil C stocks by an average of 15.4% or 18.5%, respectively. Interestingly, both the conversions of forests to grassland and grassland to secondary forests increased soil C stocks, and the establishment of perennial tree plantations on lands that were previously grazed or cropped increased soil C stocks, but the conversion of unmanaged forests, or grasslands to plantations had no effect (Powers et al., 2011). Globally, although Guo and Gifford (2002) concluded that soil C stocks significantly increased after the conversion from farmland to grassland (19%), tree plantation (18%) and secondary forest (53%), they uses a relative percentage change. In order to understand how C stocks change after land use conversions, the changes need to be recalculated as absolute values to meet the challenge of managing soil C stocks world-wide and to help understand the contribution of carbon emissions due to land use change to the global climate change.

In addition, understanding the factors that govern the size of the current land carbon stock and the balance between plant carbon inputs and soil carbon losses is crucial to predicting the effects of future land use change on the net greenhouse gas balance, and to the development of policy for 'carbon conscious' management of the land surface. Although several authors (Post and Kwon, 2000; Paul et al., 2002; Laganière et al., 2010) have analyzed the factors determining soil C stocks during the establishment of perennial vegetation, a consensus on the relative significance of these factors has yet to be achieved. While Paul et al. (2002) found that climate is one of the most important factors influencing soil C change after cropland conversion, Laganière et al. (2010) concluded that climate had a smaller effect on soil C accumulation during afforestation when compared to previous land use, tree species planted, soil clay content and preplanting disturbance. However, at a global scale, we had difficulties in getting access to many factors that may influence the soil C stock after land use change from the existing field studies, except for climatic factors, so this paper mainly analyzed the effects of mean annual precipitation and temperature on soil C sequestration after land use conversions, as was done by other national or global studies (Guo and Gifford, 2002; Paul et al., 2002; Yang et al., 2011).

The objectives of this study were: (1) explore the effect of various land use changes on soil C sequestration; (2) establish the temporal pattern of soil C sequestration for different land use changes; and (3) study those factors driving the changes in soil C. To achieve these objectives we synthesized the findings of 103 recent publications from the literature in which

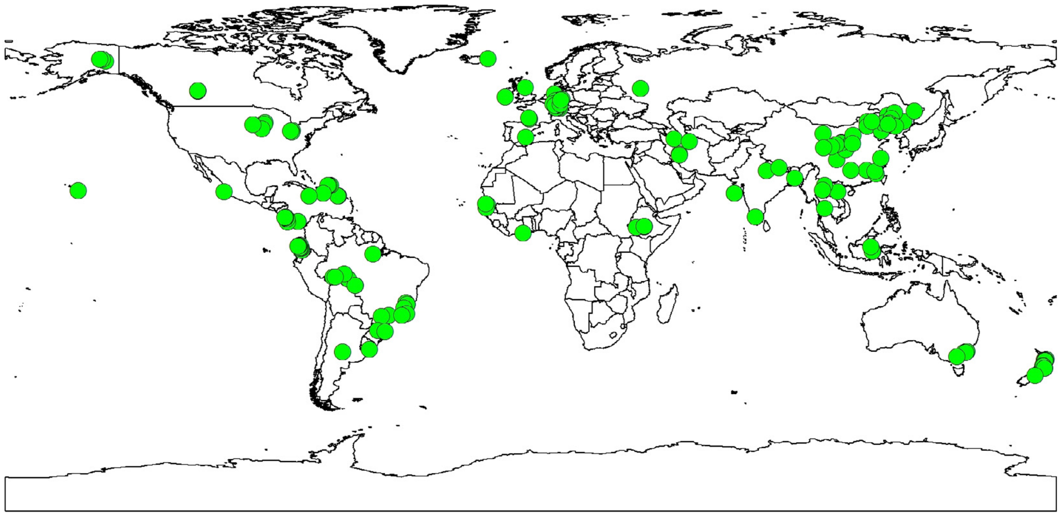

Fig. 1. Global sampling sites distribution of observed soil C change after land use conversions in the collected dataset.

seven land use conversions (from farmland to forest, from forest to farmland, from farmland to grassland, from grassland to farmland, from grassland to forest, from forest to grassland, from forest to forest) were related to changes in soil C values.

2. Materials and methods

2.1. Data preparation

We compiled data from the literature on the influence of land use change on soil C stocks and characterized each study according to the following categories of land use change: from farmland to forest, from forest to farmland, from farmland to grassland, from grassland to farmland, from grassland to forest, from forest to grassland, from forest to forest. The following criteria were used to select papers for synthesis: soil C stocks were provided or could be calculated based on SOC or SOM concentration, bulk density and soil depth; there were data on both the immediate land use (LUn) and the prior land use (LU0); the experiments used paired sites, chronosequences or retrospective designs, had similar soil conditions for both LUn and LU0; the number of years since land use conversion were either clearly given or could be directly derived. In the studies, only the first rotation of land use conversion was considered and data for 0–20 cm soil layers were extracted (Appendix Dataset S1). In addition, studies were excluded if they lacked replications or if the paired sites or sites of chronosequence were confounded by different soil types. In total, the final dataset comprised 103 studies (Appendix Table S1) most of them published between 2000 and 2014, and three studies published in 1994, 1997 and 1999, respectively, including 424 observations at 160 sites in 29 countries (Appendix Table S2) (Fig. 1). In the current study, the definition of forest is native forest before it is cleared for other land use. Pasture is land used for grazing purposes including natural grassland. Farmland includes lands cultivated for food products. The current forest is the plantation forest by human establishment. The ages of land use conversions were divided into four groups, as follows: 0–10, 11–20, 21–30, and >30 yr.

The raw data were either obtained from tables or extracted by digitizing graphs using the GetData Graph Digitizer (version 2.24, Russian Federation). For each paper, the following information was compiled: sources, location (longitude and latitude), climatic data (mean annual temperature and precipitation), land-use conversion types (including both prior and current land use types), age (years since land use conversion), soil depth from soil surface, soil bulk density, and amount of SOC or soil C stocks in each layer of 0–100 cm soil (Appendix Dataset S2).

2.2. Data calculation

If the samples reported only had SOM, their SOC were calculated by the relation between SOM and SOC. The formula for the calculation is as follows (Guo and Gifford, 2002):

$$\text{SOC} = \text{SOM} \times 0.58 \tag{1}$$

where SOC is the soil organic carbon and SOM is the soil organic matter.
The SOC stocks was calculated using the following equation (Guo and Gifford, 2002):

$$C_s = \frac{\text{SOC} \times \text{BD} \times \text{D}}{10} \tag{2}$$

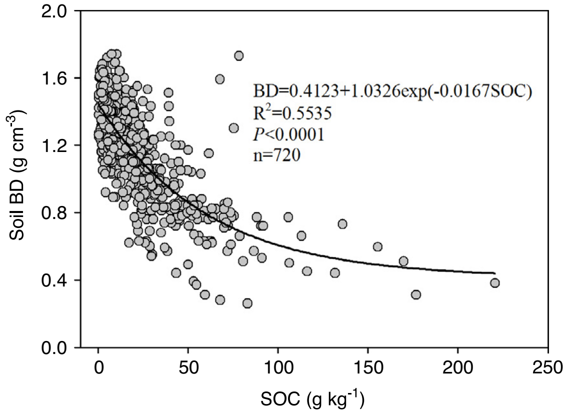

Fig. 2. Empirical functions for estimating the missing soil BD based on data from studies reporting SOC concentration and soil BD.

in which, C_s is soil organic carbon stocks (Mg ha^{-1}); SOC is soil organic carbon concentration (g kg^{-1}); BD is soil bulk density (g cm^{-3}); and D is soil thickness (cm).

Soil BD estimates are critical for calculations of C_s, but many studies did not measure this attribute. We established an empirical relationship between soil BD and SOC concentration with the reported values from the Appendix Dataset S2 (Fig. 2). Then, the missing values of soil BD were interpolated using the predicted values from the empirical functions in Fig. 2. The formula for the calculation is as follows (Eq. (3)):

$$\mathrm{BD} = 0.4123 + 1.0326e^{-0.0413\mathrm{soc}}. \tag{3}$$

Although it is desirable to compare changes in soil C stocks between land uses based on common soil mass rather than on volume because of compaction, it is impossible for us to correct data for all these studies, as not all the studies reported bulk densities, especially bulk densities for different soil depths. Thus, similar to other meta-analysis (Guo and Gifford, 2002; Laganière et al., 2010; Powers et al., 2011), we did not adjust reported data to a common mass, but we used mass-corrected soil C stocks when authors presented them. Not adjusting for an equivalent mass of soil could only result in a slight bias in the estimation of changes in soil C stocks, which is supported by our data and also by other studies (Laganière et al., 2010).

To increase the comparability of data derived from different studies, the methodology adopted by Yang et al. (2011) was used in the present study. The original soil C data were converted to the soil C stocks in the top 20 cm using the depth functions developed by Jobbágy and Jackson (2000) according to the following equations:

$$Y = 1 - \beta^{d} \tag{4}$$

$$X_{20} = \frac{1-\beta^{20}}{1-\beta^{d0}} \times X_{d0} \tag{5}$$

where Y represents the cumulative proportion of the soil C stocks from the soil surface to depth d (cm); β is the relative rate of decrease in the soil C stocks with soil depth; X_{20} denotes the soil C stocks in the upper 20 cm; $d0$ denotes the original soil depth available in individual studies (cm); and X_{d0} is the original soil C stocks. For soil C, although Jobbágy and Jackson (2000) provided the depth distributions for 11 biome types globally, there was no significant difference ($P > 0.98$), ANOVA LSD test using PASW 18 (SPSS Inc., Chicago, IL, USA) in the depth distribution among biome types or between individual biomes and the global average. Therefore, in this study, the global average depth distributions for C were adopted to calculate the value of β (0.9786) in Equations.

It should be noted that potential uncertainties may be introduced by this dataset standardization, mainly as a result of the difference in C distribution over the soil profile between prior land use types and immediate land use sites, and among different stages of vegetation (forest, shrub and grassland) development. However, as has been stated, there was no significant difference among the 11 biome types included in Jobbágy and Jackson (2000) or between individual biomes and the global average in terms of the soil C distribution with depth. The same method (i.e., converting the original C stocks to the stocks in the top 20 cm using the depth functions in order to increase comparability) was used by Yang et al. (2011) and Li et al. (2012), and it was concluded that depth correction did not alter the overall pattern of soil C stocks dynamics during vegetation development.

C sequestration rate is estimated depending on the changes of soil C stocks in different time sequences of time. The study set the C stocks of prior land use types as the baseline for calculating the rate of C stocks change in the restoration process after land use conversions. We first calculate the carbon sequestration (Mg ha^{-1}) in every site after land use conversion (Eq. (6)),

$$\Delta C_s = C_{LUn} - C_{LU0} \tag{6}$$

in which, C_{LUn} is represent soil carbon stocks at immediate land use sites (Mg ha^{-1}), and C_{LU0} is soil carbon stocks of prior land use types before land use conversion (Mg ha^{-1}).

We used mean annual absolute rate of change in soil C stock to indicate soil C sequestration rate (R_s, Mg ha^{-1} yr^{-1}). The calculated equation is as follows:

$$\text{Soil C sequestration rate (Mg ha}^{-1}\text{ yr}^{-1}) : R_s = \frac{\Delta C_s}{\Delta Age}. \tag{7}$$

In order to reflect the dynamics of soil C stocks, soil C sequestration were summed for each category. In this case, a methodology reported previously (Luo et al., 2006, 2009) was used to calculate 95% CI of means for soil C sequestration, as shown in Eqs. (8) and (9):

$$SE_{total} = \sqrt{\frac{V_S}{N}} \tag{8}$$

$$95\%\ CI = 1.96 \times SE_{total} \tag{9}$$

where SE_{total} denotes the standard error of the relative change in soil C stock. V_S and N are the variance of relative soil C stock change and the number of observations, respectively. In this study, the 95% confidence interval (CI) was calculated for the overall data and for each category. And the observed effect sizes are considered statistically different from zero if the 95% CI does not include zero, and the grouping factors are considered significantly different from each other if their 95% CI does not overlap.

2.3. Data analysis

ANOVA results of the effects of land use conversions, years since land use conversion (Age), and their interactions on soil C sequestration in different land use conversion types with general linear model (GLM) tests. Differences were evaluated at the 0.05 significance level. Stepwise regression analysis was used to analysis the relationship between soil C sequestration (ΔC_s) and mean annual temperature (MAT), mean annual precipitation (MAP), age since land use conversions (Age), and initial soil C stocks (I) in each land use conversion types. Pearson correlation coefficients between soil C sequestration and various other factors (Age, I, MAT, and MAP) were determined following land-use conversions. All statistical analyses were performed using the software program SPSS, ver. 17.0 (SPSS Inc., Chicago, IL, USA).

3. Results

3.1. Global patterns of soil C stock changes due to land use conversions

As an overall average across all land use change examined, land use conversions had significantly reduced soil C stock (−2.52 Mg ha^{-1}, rate = 0.39 Mg ha^{-1} yr^{-1}) (Fig. 3). However, not all land use conversions have reduced soil C stock. Soil C stocks significantly increased after the conversion from farmland to grassland (+6.16 Mg ha^{-1}, rate = 0.30 Mg ha^{-1} yr^{-1}), forest to grassland (+11.53 Mg ha^{-1}, rate = 0.68 Mg ha^{-1} yr^{-1}) (Fig. 3). Soil C stocks had significantly declined after the conversion from grassland to farmland (−12.45 Mg ha^{-1}, rate = 0.89 Mg ha^{-1} yr^{-1}), forest to farmland (−21.05 Mg ha^{-1}, rate = 1.74 Mg ha^{-1} yr^{-1}) and forest to forest (−12.28 Mg ha^{-1}, rate = 0.63 Mg ha^{-1} yr^{-1}) (Fig. 3). The highest soil C stock increases occurred in land-use conversions from forest to grassland, and soil C stocks declined the most after conversion from forest to farmland. After conversions of farmland to forest and grassland to forest, soil C stocks did not change significantly (Fig. 3).

3.2. Temporal patterns of soil C stocks after land use conversions

Overall, after land use conversions, the differences in soil C sequestration were not significant among different age groups ($P > 0.05$) (Table 1), but soil C sequestration changes fluctuated over time (Fig. 4). With the land-use change from farmland to grassland, soil C sequestration change was not significant during 0–10 years, and then it was significantly increased (Fig. 5(A)). Soil C sequestration had significantly difference among different age groups ($P < 0.01$) (Table 1). With the land-use change from grassland to farmland, soil C sequestration had not significantly difference among different age groups ($P > 0.05$) (Table 1). Soil C stocks had been reducing in the first 30 years (0–30 yr), and then soil C stock began to increase (>30 yr) (Fig. 5(B)), however, soil C stock still reduced compared to the initial soil C stock. With the land-use change from farmland to forest, soil C sequestration had significant differences among different age groups ($P < 0.01$) (Table 1). Soil C sequestration was significantly reduced in the first 10 years (0–10 yr), and then it was significantly increased (>10 years) (Fig. 5(C)). With the land-use change from forest to farmland, soil C sequestration had no significant difference among different age groups ($P > 0.05$) (Table 1). Soil C stock was similar to the conversion from grassland to farmland that soil C stocks had been reducing in the first 30 years (0–30 yr), and then soil C stock began to increase (>30 yr) (Fig. 5(D)). With the land-use change from grassland to forest, soil C sequestration had significant differences among different age groups

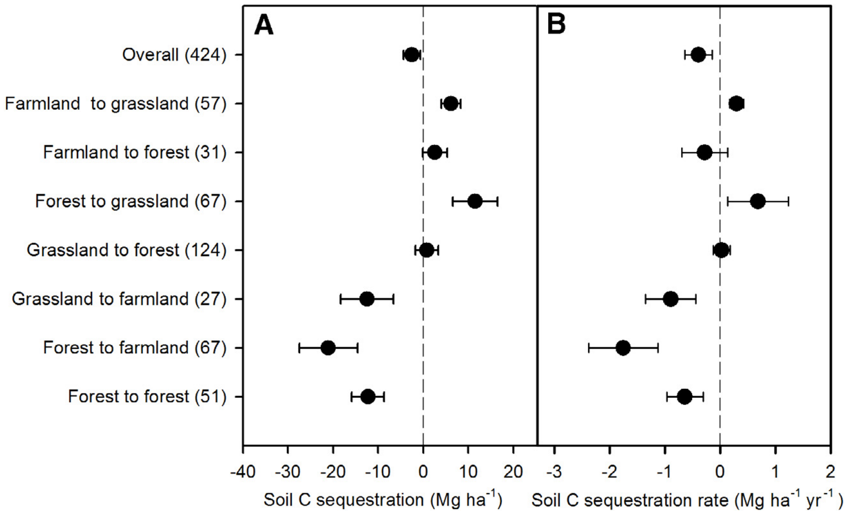

Fig. 3. The effects of the land use changes on soil C sequestration and soil C sequestration rate. Note: dots with error bars denote the overall mean values and the 95% CI, and numbers of observations are in the parenthesis. The dash line indicates $x = 0$.

Table 1
ANOVA results of between-subjects effects of years since land use conversion (Age), and their interactions on soil C sequestration in different land use conversion types (LUE) with general linear model (GLM) tests.

Land use change types	Source	df	F	Sig. (P)
Farmland to grassland	Age	3	11.793	0.000**
Grassland to farmland	Age	3	0.973	0.422
Farmland to forest	Age	3	13.535	0.000**
Forest to farmland	Age	3	1.331	0.272
Grassland to forest	Age	3	4.451	0.005**
Forest to grassland	Age	3	2.883	0.043*
Forest to forest	Age	3	3.953	0.014*
	LUE	6	27.565	0.000**
Overall	Age	3	2.307	0.076
	LUE × Age	18	2.361	0.001**

Note: Age were divided into four groups, as follows: 0–10, 11–20, 21–30, and >30 yr.
* Indicates significant at $P < 0.05$.
** Indicates significant at $P < 0.01$.

($P < 0.01$) (Table 1). Soil C stock was not significantly reduced in the first 10 years, and then it was significant increased (11–20 yr), but it also tended to significantly reduce from 21 to 30 years, and then increase after 30 years (Fig. 5(E)). With the land-use change from forest to grassland, soil C sequestration had significant differences among different age groups ($P < 0.05$) (Table 1). Soil C stock was increased after land use conversion (Fig. 5), especially after conversion for more than 30 years, soil C stock increased 24.40 Mg ha^{-1} compared with the initial soil C stock (Fig. 5(F)). With the land-use change from forest to forest, soil C stock was reduced especially after more than 30 years (−19.45 Mg ha^{-1}) (Fig. 5(G)). And soil C sequestration had significant differences among different age groups ($P < 0.05$) (Table 1).

3.3. Factor effects on soil C stocks

Pearson correlations analysis showed that in different age groups, the factors affecting soil C sequestrations were different (Table 2). However, after land use conversions, soil C sequestration had significant negative correlations with the initial soil C stocks in every land use conversion type ($P < 0.05$) (Table 2). In addition, after the conversion of farmland to grassland and forest, soil C sequestration also had significant positive correlations with the age since land use conversion, but it showed negative correlations after the conversion of forest to forest (Table 2). For the conversion of grassland to forest, soil C sequestration also had significant positive correlations with MAT (Table 2).

Stepwise regression revealed that different land use conversion types had different impact factors (Table 3). The age since land use conversion had significant effects on soil C sequestration after the conversion of farmland to grassland, farmland to forest, and forest to forest (Table 3); the initial soil C stock was the main factor affecting soil C sequestration after the conversion of grassland to farmland, forest to farmland and grassland to forest (Table 3); and climatic factors (MAT and MAP) were the main factors affecting soil C sequestration after the conversion of forest to grassland (Table 3). However, for

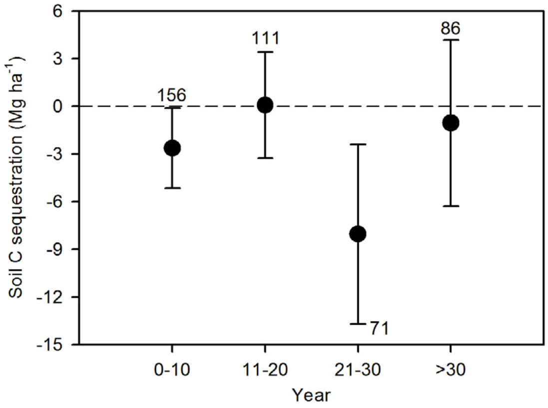

Fig. 4. The effects of the land use changes on soil C sequestration in different age groups. Note: dots with error bars denote the overall mean values and the 95% CI, and numbers of observations above the error bars.

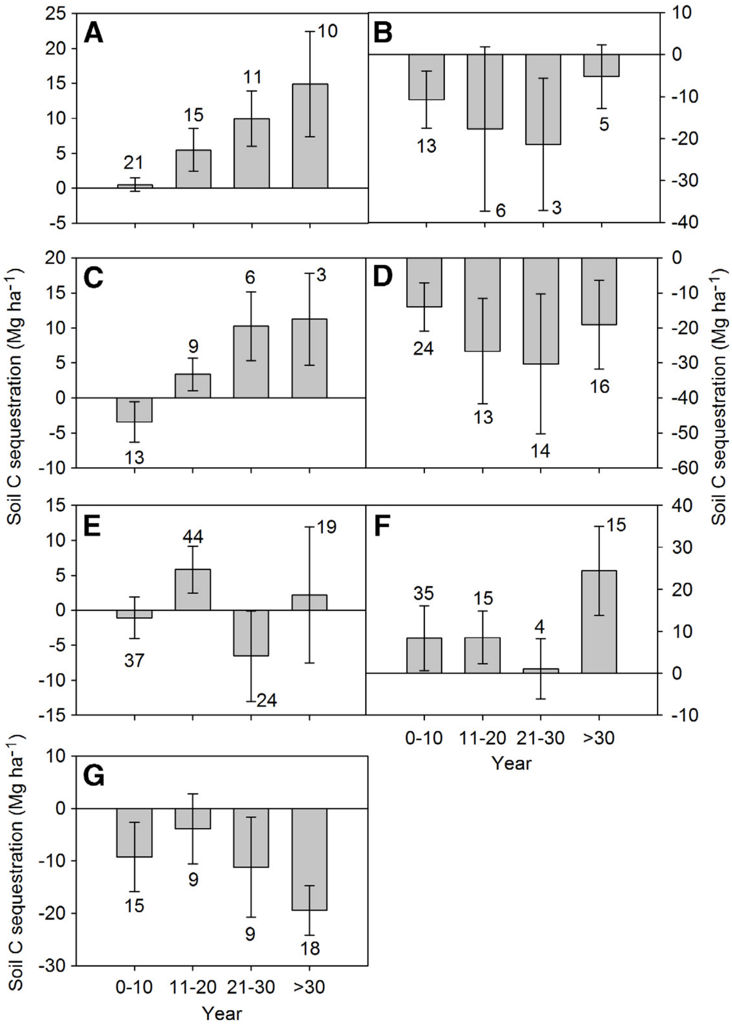

Fig. 5. The effects of the land use changes on soil C stocks in different age groups after the conversion. Note: A, farmland to grassland, B, grassland to farmland, C, farmland to forest, D, forest to farmland, E, grassland to forest, F, forest to grassland, G, forest to forest. Bar charts with error bars denote the overall mean values and the 95% CI, and numbers of observations above the error bars.

all land use conversions, the initial soil C stocks and climatic factors had significant effects on soil C sequestration after land use conversions, particularly the initial soil C stocks (Table 3).

Table 2
Pearson correlation coefficients between soil C sequestration and factors: age after land use conversion (Age), initial soil C stocks (I), mean annual temperature (MAT), mean and annual precipitation (MAP) following land-use conversions.

Land use change types	Age groups	Age (yr)	I (Mg ha^{-1})	MAT (°C)	MAP (mm)
Farmland to grassland	0–10	−0.260 (n = 21)	0.252 (n = 21)	−0.083 (n = 21)	−0.014 (n = 21)
	11–20	0.539* (n = 15)	0.658** (n = 15)	0.167 (n = 15)	−0.137 (n = 15)
	21–30	0.407 (n = 11)	−0.334 (n = 11)	−0.7** (n = 11)	−0.003 (n = 11)
	>30	0.714* (n = 10)	0.898** (n = 10)	−0.393 (n = 10)	0.047 (n = 10)
	All	0.739** (n = 57)	0.639** (n = 57)	−0.061 (n = 57)	0.228 (n = 57)
Grassland to farmland	0–10	−0.499 (n = 13)	−0.601* (n = 13)	0.375 (n = 13)	0.135 (n = 13)
	11–20	−0.504 (n = 6)	−0.913* (n = 6)	0.469 (n = 6)	0.424 (n = 6)
	21–30	−0.665 (n = 3)	−0.663 (n = 3)	0.181 (n = 3)	−0.996 (n = 3)
	>30	0.630 (n = 5)	−0.980** (n = 5)	0.481 (n = 5)	0.526 (n = 5)
	All	0.017 (n = 27)	−0.741** (n = 27)	0.331 (n = 27)	0.147 (n = 27)
Farmland to forest	0–10	0.357 (n = 13)	−0.165 (n = 13)	−0.640* (n = 13)	−0.600* (n = 13)
	11–20	0.181 (n = 9)	0.598 (n = 9)	−0.067 (n = 9)	0.493 (n = 9)
	21–30	−0.253 (n = 6)	−0.218 (n = 6)	0.432 (n = 6)	0.570 (n = 6)
	>30	0.844 (n = 3)	0.727 (n = 3)	−0.727 (n = 3)	0.727 (n = 3)
	All	0.767** (n = 31)	−0.415* (n = 31)	−0.01 (n = 31)	−0.345 (n = 31)
Forest to farmland	0–10	−0.246 (n = 24)	−0.637** (n = 24)	0.137 (n = 24)	0.013 (n = 24)
	11–20	0.252 (n = 13)	−0.699** (n = 13)	0.194 (n = 13)	−0.031 (n = 13)
	21–30	−0.121 (n = 14)	−0.874** (n = 14)	−0.326 (n = 14)	−0.358 (n = 14)
	>30	−0.170 (n = 16)	−0.701** (n = 16)	−0.333 (n = 16)	0.108 (n = 16)
	All	−0.096 (n = 67)	−0.730** (n = 67)	−0.098 (n = 67)	−0.079 (n = 67)
Grassland to forest	0–10	0.096 (n = 37)	−0.465** (n = 37)	0.221 (n = 37)	0.015 (n = 37)
	11–20	0.435** (n = 44)	−0.178 (n = 44)	0.156 (n = 44)	0.093 (n = 44)
	21–30	−0.251 (n = 24)	−0.609** (n = 24)	0.300 (n = 24)	0.128 (n = 24)
	>30	0.015 (n = 19)	−0.723** (n = 19)	0.221 (n = 19)	0.082 (n = 19)
	All	0.01 (n = 124)	−0.408** (n = 124)	0.241** (n = 124)	0.09 (n = 124)
Forest to grassland	0–10	−0.212 (n = 33)	−0.374* (n = 33)	0.201 (n = 33)	0.370* (n = 33)
	11–20	−0.152 (n = 15)	−0.174 (n = 15)	−0.551* (n = 15)	−0.384 (n = 15)
	21–30	−0.938 (n = 4)	−0.925 (n = 4)	0.461 (n = 4)	−0.926 (n = 4)
	>30	−0.203 (n = 15)	0.075 (n = 15)	−0.488 (n = 15)	−0.452 (n = 15)
	All	0.225 (n = 67)	−0.416** (n = 67)	−0.107 (n = 67)	−0.021 (n = 67)
Forest to forest	0–10	−0.029 (n = 15)	−0.441 (n = 15)	0.328 (n = 15)	0.406 (n = 15)
	11–20	−0.071 (n = 9)	0.599 (n = 9)	−0.715* (n = 9)	−0.286 (n = 9)
	21–30	−0.191 (n = 9)	−0.689* (n = 9)	−0.316 (n = 9)	−0.414 (n = 9)
	>30	−0.095 (n = 18)	−0.171 (n = 18)	−0.153 (n = 18)	0.059 (n = 18)
	All	−0.356* (n = 51)	−0.425** (n = 51)	−0.094 (n = 51)	−0.026 (n = 51)
Overall	0–10	−0.107 (n = 156)	−0.410** (n = 156)	0.115 (n = 156)	0.106 (n = 156)
	11–20	0.123 (n = 111)	−0.454** (n = 111)	0.217* (n = 111)	0.172 (n = 111)
	21–30	−0.180 (n = 71)	−0.753** (n = 71)	−0.148 (n = 71)	−0.096 (n = 71)
	>30	−0.035 (n = 86)	−0.468** (n = 86)	−0.260* (n = 86)	0.036 (n = 86)
	All	**−0.010 (n = 424)**	**−0.504** (n = 424)**	**−0.03 (n = 424)**	**0.07 (n = 424)**

Note: (value) indicates the number of observations.
* Correlation is significant at the 0.05 level (2-tailed) ($P < 0.05$).
** Correlation is significant at the 0.01 level (2-tailed) ($P < 0.01$).

Table 3
Stepwise regression to detect factors (Age, I, MAT, and MAP) determining soil C sequestration following land-use conversions.

Land use change types	Equations	R^2	Sig. (P)	n
Farmland to grassland	$\Delta C_s = 0.31\text{Age} + 0.20\text{I} - 4.30$	0.630	0.001**	57
Grassland to farmland	$\Delta C_s = -0.48\text{I} + 10.01$	0.537	0.000**	27
Farmland to forest	$\Delta C_s = 0.52\text{Age} - 5.76$	0.589	0.000**	31
Forest to farmland	$\Delta C_s = -0.53\text{I} - 0.25\text{Age} + 19.22$	0.546	0.000**	67
Grassland to forest	$\Delta C_s = -0.35\text{I} + 0.01\text{MAP} + 9.10$	0.402	0.000**	124
Forest to grassland	$\Delta C_s = -1.33\text{MAT} + 0.01\text{MAP} - 0.36\text{I} + 0.17\text{Age} + 38.44$	0.189	0.010**	67
Forest to forest	$\Delta C_s = -0.18\text{Age} - 7.18$	0.127	0.010*	51
Overall	$\Delta C_s = -0.385\text{I} + 0.007\text{MAP} - 0.385\text{MAT} + 11.87$	0.332	0.000**	424

Note: ΔC_s is soil C sequestration following land-use conversions; Age (yr) is the restoration age; I (Mg ha^{-1}) is the initial soil C stocks; MAT (°C) is the average annual temperature; MAP (mm) is the average annual precipitation.
* Indicates significant at $P < 0.05$.
** Indicates significant at $P < 0.01$.

4. Discussion

4.1. Effects of land use changes on soil C stocks

Globally, land-use change can cause a change in soil C stock (Fig. 3). As an overall average across all land use change examined, land use conversions had significantly reduced soil C stock (-2.52 Mg ha^{-1}) (Fig. 3(A)). Similarly, Guo and Gifford (2002) reported land use changes reduced soil C stocks by 9%, but their study only showed the relative percentage change, not the absolute change in soil C sequestration after land use conversions, so our study fills this gap. Guo and Gifford (2002) also reported that soil C stocks decline after land use changes from grassland to plantation (−10%), native forest to plantation (−13%), native forest to farmland (−42%) and grassland to farmland (−59%), and soil C stocks increase after land use changes from native forest to grassland (+8%), farmland to grassland (+19%), farmland to plantation (+18%), and farmland to secondary forest (+53%). In our study, soil C stocks significantly increased after the conversion from farmland to grassland (+6.16 Mg ha^{-1}) and forest to grassland (+11.53 Mg ha^{-1}) (Fig. 3(A)), and soil C stocks significantly declined after the conversion from grassland to farmland (−12.45 Mg ha^{-1}), forest to farmland (−21.05 Mg ha^{-1}) and forest to forest (−12.28 Mg ha^{-1}) (Fig. 3(A)). In addition, after conversion of farmland to forest and grassland to forest, soil C sequestration had no significant change, (Fig. 3(A)), which is similar with the study of Powers et al. (2011), which reported that the conversion of unmanaged forests, grasslands, or savannas to plantations had no effect in the tropical regions. And Powers et al. (2011) also reported that the conversion of forests to shifting cultivation or permanent crops reduced soil C stocks by an average of 15.4% or 18.5%, respectively, and interestingly, both the conversions of forests to pastures and pastures to secondary forests increased soil C stocks, and the establishment of perennial tree plantations on lands that were previously grazed or cropped increased soil C stocks. Generally, where the land use changes decreased soil C, the reverse process usually increased soil C stocks and vice versa.

In addition, despite the considerable soil C sequestration potential that afforestation offers, many studies have reported contradictory findings. Afforestation resulted in either a decrease (Farley et al., 2004) or an increase in soil C stocks (Lemma et al., 2006), or had a negligible effect (Smal and Olszewska, 2008). Also, at the global scale, we found similar patterns (Fig. 3): decline (forest to forest); increase (farmland to forest), and unchanged (grassland to forest). At the national scale, Deng et al. (2014a) calculated the average soil C sequestration rate in the top 20 cm soil to be 0.33 Mg ha^{-1} yr^{-1} following perennial vegetation establishment (mainly forest and shrub) from cropland in China. In addition, the global average soil C sequestration rates following the conversion of cultivated land to forest, shrub and grassland are 0.45, 0.47 and 1.1 Mg ha^{-1} yr^{-1}, respectively (IPCC, 2000; Vleeshouwers and Verhagen, 2002; Shi et al., 2013). In contrast, after the conversion from forest to farmland, a significant decline in soil C stocks was found in many studies (Guo and Gifford, 2002; Murty et al., 2002; Wei et al., 2014). For example, Wei et al. (2014) reported that the largest decrease in soil C stocks was observed in temperate regions (−52%) followed by tropical regions (−41%) and boreal regions (−31%), and for the global scale, soil C stocks declined about 43.1% compared to prior forest land; and the 30% decrease observed by Murty et al. (2002). Also, with the conversion from grassland to farmland, similar pattern was observed, where soil C stocks declined after land use conversion (Fig. 3).

4.2. Temporal changes in soil C stocks after land use conversions

Globally, as an overall average across all land use changes examined, the changes in soil C sequestration with age were not significant ($P > 0.05$). For a specific land use conversion type, in contrast, age since land use conversion can effect soil C sequestration. In our study, soil C sequestration was significant different among different age groups in 5 of 7 land use conversion types (Table 1). Deng et al. (2014a) reported after the conversion from farmland to forest, soil C stocks showed an initial loss in soil during the first few years (∼5 yr), followed by a gradual return of C stocks to levels comparable to those in the control agricultural soil, and then increasing to generate net C gains in some cases; similar temporal patterns of soil C stock changes following farmland conversion have been reported in a number of studies (Turner et al., 2005; Ritter, 2007; Karhu et al., 2011). The duration of the initial decrease in C was reported to last for 3–35 years following agricultural abandonment and there was a significant net accumulation of soil C by 30 years after afforestation (Paul et al., 2002). Nevertheless, Wei et al. (2014) observed that the conversion of forests to agricultural land caused a rapid initial decrease in SOC stocks, followed by a slow decline. In addition, with afforestation on grassland, soil C stocks showed the trend of declined–increased–declined–increased following time (Fig. 5(E)). The possible mechanism was that due to afforestation activity, soils of the previous grassland suffered human disturbance, leading to soil C loss due to strong soil respiration, and following forest development, soil C input into the soils was increased, especially understory vegetation biomass input into the soils, then the forest coverage increased, understory vegetation biomass decreased, so leading to soil C loss again, but then forest litter and dead roots input into the soil were increased, so soil C stocks increased again. In contrast, after the forest was converted to grassland, soil C stocks have been increased on the whole (Fig. 5(F)). However, after afforestation on native forest, soil C stocks were always lower than the values in a comparable natural forest, which agrees with the finding of Smal and Olszewska (2008). For the conversion from farmland to grassland, soil C stocks have been increased after farmland abandonment, which was supported by some studies (Mensah et al., 2003; Nelson et al., 2008; Deng et al., 2013; De Baets et al., 2013; Deng et al., 2014b,c). For example, De Baets et al. (2013) found a logarithmic increase since farmland abandonment. On the contrary, after the conversion from grassland to farmland, soil C stocks had been reducing in the first 30 years (Fig. 5(B)). The temporal patterns were similar to the conversion from forest to farmland (Fig. 5(D)). Houghton (2003) indicated that clearing forests for new agricultural land caused a release of carbon to the atmosphere. The carbon initially held in trees and other vegetation is released through decomposition of above- and below-ground plant material left in the soil at the time of clearing. Even if the productivity of the new farmland is as high as it was in the forest or grassland, less of the crop production accumulates as litter, most of it is harvested and subsequently consumed or respired. Hence, the process of the conversion from forest or grassland to farmland and management afterward reduces carbon input from litter and enhances the carbon output via breaking the protection of soil organic matter (Guo and Gifford, 2002). For example, Aslam et al. (1999) found that adoption of no-tillage could protect soils from biological degradation and maintain soil quality as compared with plow tillage management after land use change from grassland (pasture) to farmland. However, through long term tillage management, such as increased organic fertilizer input to the topsoil, soil C stock did not always decline. It could reach a new equilibrium point and gradually accumulate at a slower rate (Guo and Gifford, 2002). As in our study, after forest or grassland converted to farmland, soil C stock began to increase after 30 years of cultivation (Fig. 5(B), (D)).

4.3. Factors affecting soil C stocks

Soil C sequestration due to land use change is likely to be affected by multiple factors such as climatic condition, soil texture, site preparation and management, vegetation type, land use history, etc. (Post and Kwon, 2000; Paul et al., 2002; Vesterdal et al., 2002; Guo and Gifford, 2002; Laganière et al., 2010; Li et al., 2012; Shi et al., 2013; Deng et al., 2014). Many studies have reported that age since land use changes plays a consistent and key role in estimated soil C stocks (Guo and Gifford, 2002; Laganière et al., 2010; Li et al., 2012; Shi et al., 2013; Deng et al., 2014). For example, Shi et al. (2013) found that stand age can play an important role in carbon sequestration after afforestation of agricultural sites in the globe. Deng et al. (2014a) have reported that restoration age was the main factor affecting soil C stock change after cropland conversion in China. In our study, our results showed that after the conversion of farmland to grassland and forest, the age since land use conversion also had significant positive correlations with soil C sequestration, but also negative correlations with soil C sequestration after the conversion of forest to forest (Table 2), and for grassland to farmland, forest to farmland, grassland to forest, and forest to grassland, soil C sequestration had no significant correlations with the age since land use conversion (Table 2). Wei et al. (2014) have reported that soil C sequestration has significantly positive correlations with the age. Therefore, these results showed that the relationships between soil C sequestration and age since land use conversion varied between study scales and land use change types.

Climate may affect soil C accumulation through biotic processes associated with the productivity of vegetation and decomposition of organic matter (Li et al., 2012). At the global scale, the restoration of soil C stocks after afforestation was found to vary with climate (Laganière et al., 2010). Post and Kwon (2000) found a tendency for rates of soil C accumulation to increase from temperate regions to subtropical regions, and they inferred that the amounts of organic matter inputs which increased with temperature and moisture were the major factors determining the rate of accumulation. Our results show that soil C sequestration at a global scale had no significant correlation with mean annual temperature and precipitation ($P > 0.05$), but soil C sequestration is positively correlated with MAP, demonstrating that precipitation increases the magnitude of soil C sequestration after land use change, and negatively correlated with MAT, which may be because higher temperature leads to higher losses of soil carbon through decomposition of soil organic matter (Deng et al., 2014b). In addition, our results revealed that the effects of climatic factors (MAT and MAP) on soil C sequestration were closely related to the land use conversion type (Table 3). For example, soil C sequestration had significant positive correlations with MAT for the conversion of grassland to forest (Table 2), and MAT and MAP were the main factors affecting soil C sequestration after the conversion of forest to grassland (Table 3).

Moreover, for all land use conversions in the globe, soil C sequestration showed a significant negative correlation with initial soil C stocks ($P < 0.05$) (Table 2), which was the same with the studies on soil C sequestrations since farmland conversion (Zhang et al., 2010; Deng et al., 2014a). However, in a smaller study scale, i.e. the Loess Plateau, soil C sequestration showed a significant positive correlation with initial soil C stocks ($P < 0.05$) (Deng et al., 2014b). This may be related to the different decomposition rates to be found under the varying nutrient conditions associated with different climate regimes (Zhang et al., 2010; Deng et al., 2014b). The synthesis revealed that initial soil C stocks had strong relations with MAT and MAP (Zhang et al., 2010; Deng et al., 2014b). Also, in our study, we found that the initial soil C stocks and climatic factors (MAT and MAP) had significant effects on soil C sequestration after land use conversions, and were mainly affected by the initial soil C stocks (Table 3), a finding from which Vesterdal et al. (2002) had earlier inferred that slower rates of decomposition might make soil carbon storage increase faster in more nutrient-poor soils following afforestation, and vice versa.

4.4. Uncertainty analysis

This synthesis offers the most accurate estimate on soil carbon sequestration following land use change across the entire global. Strict accuracy is limited due to the uneven distribution of data collected in each category (Age group and land use change types). Additionally, many of the studies have no long term observations and consequently, these measurements may add to the uncertainty. In addition, in our study, we ignored the effect of bulk density substituting Eq. (3) (Fig. 2), i.e., the will be equal as long as SOC is the same. In reality, however, bulk density would not only be significantly different among different sites but also experience significant change after land use conversion within a site. Soil types also a limiting factor effecting on the relationship between SOC and soil bulk density. In spite of that, the concern of only the upper soil layer and soil carbon sequestration in the deeper soils also changed due to land use changes, so the problem of the only upper layer consideration have a high influence on the quantity of the result. In future, we should focus on the effects of land use change on deeper soil carbon sequestration to build a functional relationship between topsoil carbon sequestration and deeper soil carbon sequestration.

Conflict of interest

The authors declare they have no actual or potential competing financial interests.

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Appendix A. Supplementary data

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References

Amundson, R., 2001. The carbon budget of soils. Annu. Rev. Earth Planet Sci. 29, 535–562.
Aslam, T., Choudhary, M.A., Saggar, S., 1999. Tillage impacts on soil microbial biomass C, N, and P, earthworms and agronomy after two years of cropping following permanent pasture in New Zealand. Soil Tillage Res. 51, 103–111.
Bolin, B., Sukumar, R., 2000. Global perspective. In: Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J., Dokken, D.J. (Eds.), Land Use, Land Use Change, and Forestry. Cambridge University Press, Cambridge, UK.
De Baets, S., Meersmans, J., Vanacker, V., Quine, T.A., Van Oost, K., 2013. Spatial variability and change in soil organic carbon stocks in response to recovery following land abandonment and erosion in mountainous drylands. Soil Use Manag. 29, 65–76.
Deng, L., Liu, G.B., Shangguan, Z.P., 2014a. Land use conversion and changing soil carbon stocks in China's 'Grain-for-Green' Program: a synthesis. Global Change Biol. 20, 3544–3556.
Deng, L., Shangguan, Z.P., Sweeney, S., 2013. Changes in soil carbon and nitrogen following land abandonment of farmland on the Loess Plateau, China. PLoS One 8, e71923.
Deng, L., Shangguan, Z.P., Sweeney, S., 2014b. "Grain for Green" driven land use change and carbon sequestration on the Loess Plateau, China. Sci. Rep. 4, 7039.
Deng, L., Sweeney, S., Shangguan, Z.P., 2014c. Long-term effects of natural enclosure: carbon stocks, sequestration rates and potential for grassland ecosystems in the Loess Plateau. Clean Soil Air Water 42, 617–625.
Farley, K.A., Kelly, E.F., Hofstede, R.G.M., 2004. Soil organic carbon and water retention following conversion of grasslands to pine plantations in the Ecuadoran Andes. Ecosystems 7, 729–739.
Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biol. 8, 345–360.
Houghton, R.A., 2003. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850–2000. Tellus B 55, 378–390.
IPCC, 2000. Land Use, Land-use Change, and Forestry. Cambridge University Press, Cambridge, UK.
IPCC, 2006. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), Forest Land. IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, Vol. 4. IGES, Hayama, Japan, p. 83. (Chapter 4).
Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecol. Appl. 10, 423–436.
Karhu, K., Wall, A., Vanhala, P., Liski, J., Esala, M., Regina, K., 2011. Effects of afforestation and deforestation on boreal soil carbon stocks: comparison of measured C stocks with Yasso07 model results. Geoderma 164, 33–45.
Laganière, J., Angers, D.A., Paré, D., 2010. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. Global Change Biol. 16, 439–453.
Lemma, B., Kleja, D.B., Nilsson, I., Olsson, M., 2006. Soil carbon sequestration under different exotic tree species in the southwestern highlands of Ethiopia. Geoderma 136, 886–898.
Li, D.J., Niu, S.L., Luo, Y.Q., 2012. Global patterns of the dynamics of soil carbon and nitrogen stocks following afforestation: a meta analysis. New Phytol. 195, 172–181.
Luo, Y.Q., Hui, D., Zhang, D., 2006. Elevated CO_2 stimulates net accumulations of carbon and nitrogen in land ecosystems: a meta-analysis. Ecology 87, 53–63.
Luo, Y.Q., Sherry, R., Zhou, X., Wan, S.Q., 2009. Terrestrial carbon-cycle feedback to climate warming: experimental evidence on plant regulation and impacts of biofuel feedstock harvest. Global Change Biol.-Bioenergy 1, 62–74.
Mensah, F., Schoenau, J.J., Malhi, S.S., 2003. Soil carbon changes in cultivated and excavated land converted to grasses in east-central Saskatchewan. Biogeochemistry 63, 85–92.
Murty, D., Kirschbaum, M.F., McMurtrie, R.E., McGilvray, H., 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? A review of the literature. Global Change Biol. 8, 105–123.
Nelson, J.D.J., Schoenau, J.J., Malhi, S.S., 2008. Soil organic carbon changes and distribution in cultivated and restored grassland soils in Saskatchewan. Nutr. Cycl. Agroecosyst. 82, 137–148.
Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. For. Ecol. Manag. 168, 241–257.
Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: Processes and potential. Global Change Biol. 6, 317–327.
Powers, J.S., Corre, M.D., Twine, T.E., Veldkamp, E., 2011. Geographic bias of field observations of soil carbon stocks with tropical land-use changes precludes spatial extrapolation. Proc. Natl. Acad. Sci. USA 108, 6318–6322.
Ritter, E., 2007. Carbon, nitrogen and phosphorus in volcanic soils following afforestation with native birch (Betula pubescens) and introduced larch (Larix sibirica) in Iceland. Plant Soil 295, 239–251.
Shi, S.W., Zhang, W., Zhang, P., Yu, Y.Q., Ding, F., 2013. A synthesis of change in deep soil organic carbon stores with afforestation of agricultural soils. Forest Ecol. Manag. 296, 53–63.

Smal, H., Olszewska, M., 2008. The effect of afforestation with Scots pine (Pinus silvestris L.) of sandy post-arable soils on their selected properties. II. Reaction, carbon, nitrogen and phosphorus. Plant Soil 305, 171–187.
Turner, J., Lambert, M.J., Johnson, D.W., 2005. Experience with patterns of change in soil carbon resulting from forest plantation establishment in eastern Australia. Forest Ecol. Manag. 220, 259–269.
Van der Werf, G.R., Morton, D.C., DeFries, R.S., Olivier, J.G., Kasibhatla, P.S., Jackson, R.B., Collatz, G.J., Randerson, J.T., 2009. CO_2 emissions from forest loss. Nature Geosci. 2, 737–738.
Vesterdal, L., Ritter, E., Gundersen, P., 2002. Change in soil organic carbon following afforestation of former arable land. Forest Ecol. Manag. 169, 137–147.
Vleeshouwers, L.M., Verhagen, A., 2002. Carbon emission and sequestration by agricultural land use: a model study for Europe. Globe Change Biol. 8, 519–530.
Wei, X.R., Shao, M.A., Gale, W., Li, L.H., 2014. Global pattern of soil carbon losses due to the conversion of forests to agricultural. Sci. Rep. 4, 1062.
Yang, Y.H., Luo, Y.Q., Finzi, A.C., 2011. Carbon and nitrogen dynamics during forest stand development: a global synthesis. New Phytol. 190, 977–989.
Zhang, K., Dang, H., Tan, S., Cheng, X., Zhang, Q., 2010. Change in soil organic carbon following the 'Grain-for-Green' programme in China. Land Degrad. Dev. 21, 16–28.