Global Change Biology

Global Change Biology (2014) 20, 2393–2405, doi: 10.1111/gcb.12576

REVIEW

Soil carbon stock change following afforestation in Northern Europe: a meta-analysis

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Abstract

Northern Europe supports large soil organic carbon (SOC) pools and has been subjected to high frequency of land-use changes during the past decades. However, this region has not been well represented in previous large-scale syntheses of land-use change effects on SOC, especially regarding effects of afforestation. Therefore, we conducted a meta-analysis of SOC stock change following afforestation in Northern Europe. Response ratios were calculated for forest floors and mineral soils (0–10 cm and 0–20/30 cm layers) based on paired control (former land use) and afforested plots. We analyzed the influence of forest age, former land-use, forest type, and soil textural class. Three major improvements were incorporated in the meta-analysis: analysis of major interaction groups, evaluation of the influence of nonindependence between samples according to study design, and mass correction. Former land use was a major factor contributing to changes in SOC after afforestation. In former croplands, SOC change differed between soil layers and was significantly positive (20%) in the 0–10 cm layer. Afforestation of former grasslands had a small negative (nonsignificant) effect indicating limited SOC change following this land-use change within the region. Forest floors enhanced the positive effects of afforestation on SOC, especially with conifers. Meta-estimates calculated for the periods <30 years and >30 years since afforestation revealed a shift from initial loss to later gain of SOC. The interaction group analysis indicated that meta-estimates in former land-use, forest type, and soil textural class alone were either offset or enhanced when confounding effects among variable classes were considered. Furthermore, effect sizes were slightly overestimated if sample dependence was not accounted for and if no mass correction was performed. We conclude that significant SOC sequestration in Northern Europe occurs after afforestation of croplands and not grasslands, and changes are small within a 30-year perspective.

Keywords: forest floor, land-use change, meta-analysis, meta-regression, Northern Europe, soil organic carbon, stand age

Received 25 October 2013 and accepted 28 February 2014

Introduction

Land-use change is a major driver of environmental changes at the global scale, and quantitative and qualitative knowledge about ongoing and former land use has an important value as an environmental change indicator (Gerard *et al*., 2010). One of the main interests in detecting and quantifying environmental changes originating from land-use conversion is the potential effect on the carbon (C) budget. Land-use change causes perturbation of the ecosystem and thereby influences the C fluxes and stocks (Lal, 2005). It was recently reported that the net flux of carbon derived from land-use change accounted for up to 12.5% of the anthropogenic carbon emissions from 1990 to 2010 (Houghton *et al*., 2012). Oppositely, some land-use changes have been acknowledged as tools to mitigate emissions of CO_2 by providing a sink for C; afforestation being one of them. Afforestation is defined as the planting of trees on lands which historically have not contained forest cover (IPCC, 2007). In Europe, changes in agricultural policies, timber shortage, and the recognition of afforestation measures for accounting to meet the emission reduction commitments from the Kyoto Protocol have promoted afforestation. For all these reasons, the total forest area in Europe has increased 25% since 1950, with most pronounced increments in Western and Northern Europe (Fuchs *et al*., 2013).

During the past 2 decades a number of studies have reviewed changes in soil organic carbon (SOC) stocks quantitatively, either for different land-use changes (Post & Kwon, 2000; Guo & Gifford, 2002; Don *et al*., 2011; Poeplau *et al*., 2011) or more specifically concerning changes following a certain type of conversion, such as afforestation (Paul *et al*., 2002; Berthrong *et al*.,

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2009; Laganière et al., 2010; Nave et al., 2013; Shi et al., 2013). The majority of these studies focused on large scales to investigate global patterns of SOC change with land-use change. So far, several drivers of SOC change have been identified, such as climate, former land-use, forest age, forest type, soil type (and clay content), nitrogen deposition, and management practices (Jobbágy & Jackson, 2000; Guo & Gifford, 2002; Paul et al., 2002; Callesen et al., 2003; Leifeld et al., 2005; Jandl et al., 2007; Sutton et al., 2008; Laganière et al., 2010). With regard to former land use, there is a general agreement on a net SOC gain following afforestation of croplands or other SOC-depleted systems at the global or temperate region scale (Laganière et al., 2010; Poeplau et al., 2011; Nave et al., 2013), while SOC losses are generally observed following afforestation of more SOC-rich systems such as grasslands (Guo & Gifford, 2002; Paul et al., 2002; Shi et al., 2013). A major determinant of changes in SOC is time. After land-use change, a transition phase is often reported to take place (Paul et al., 2002), and in terms of SOC this transition is characterized by a loss of SOC followed by a recovery phase, its length probably being determined by the above-mentioned drivers.

Northern Europe has been absent or sparsely represented in previous quantitative reviews on SOC and land-use change (Scott et al., 1999; Post & Kwon, 2000; Johnson & Curtis, 2001; Guo & Gifford, 2002; Paul et al., 2002 Laganière et al., 2010; Don et al., 2011; Poeplau et al., 2011; Li et al., 2012; Nave et al., 2013; Shi et al., 2013). There is a need for further knowledge in this important area, not only because Northern Europe is a hot spot of land-use change (Fuchs et al., 2013), but also due to the importance of the SOC pool in northern regions, as the ratio of SOC with respect to vegetation C increases with latitude (Lal, 2005). Moreover, an increasing number of studies reporting data on SOC changes in diverse land uses have become available for Northern Europe, thereby enabling analyses at this more restricted spatial scale. The novelty of this study also relies on the improvements incorporated to the meta-analysis methodology on SOC and land-use change by: (i) evaluating the influence of nonindependence between samples; (ii) investigating confounding effects; and (iii) estimating SOC stock changes according to equal masses.

We aimed to synthesize major changes in SOC stocks following afforestation in Northern Europe by means of a meta-analysis. We studied the relative SOC changes that took place after afforestation, with focus on former land-use, age dynamics, forest type, and soil textural class, considering possible confounding effects between these factors. Moreover, we investigated differences in the relative SOC change between soil layers (upper and deeper mineral soil) as well as the contribution of forest floor C.

Materials and methods

Data compilation

This study focuses on northern regions of Europe, covering the Nordic-Baltic countries, i.e. Denmark, Sweden, Norway, Iceland, Finland, Estonia, Latvia and Lithuania, and the British Isles (Fig. 1). We collected data and site information from various sources including gray literature and unpublished data, articles from national journals and peer reviewed literature (Table S1). Data search was done consulting the online reference database Web of Science, web sites from relevant research institutions from the countries of interest, as well as by contacting corresponding authors directly. A total of 18 studies were identified (Table S1), including 119 paired observations (k) of afforested sites with corresponding control plots (a plot in which the SOC stock from former land use prior afforestation was known and could be compared with a nearby afforested plot within the same site).

A number of criteria had to be fulfilled to include a study in the meta-analysis: measurements of SOC stocks and/or carbon concentration and bulk density from a single afforested plot and its corresponding control plot (which could originate from chronosequence, repeated sampling, or paired-plot studies); former land-use information on the afforested site, tree species used, soil texture information, forest age/time since conversion, study design, sampling depth, and geoclimatic information (mean annual temperature and mean annual precipitation, latitude and longitude, altitude). Whenever available, we also collected the number of replicates and standard deviations or error estimates of SOC measurements. We also contacted scientists/researchers from Estonia, Latvia, and Norway; eventually, however, the data available did not meet the criteria of the study. The number of soil samples from forest soils was missing in a single study (Cerli et al., 2006); this number was imputed by the lowest sample size available in the data set (n = 3), thereby down-weighting the contribution of the study to any meta-analysis (see Data analysis paragraph).

Data structure

The database consisted of SOC stock data from afforested and corresponding control plots. Three SOC data sets were collated: (i) 0–10 cm (k = 108); (ii) 0–20/30 cm (k = 119); and (iii) forest floors + 0–20/30 cm (k = 103). Forest floors were defined as the organic layer consisting of undecomposed or partly decomposed organic material on top of the mineral soil. The last data set was collated to determine the effect of incorporating the organic layer as part of the total forest SOC stock. As forest floor C stock data were not reported in all cases, this data set consisted of fewer observations (Table S1).

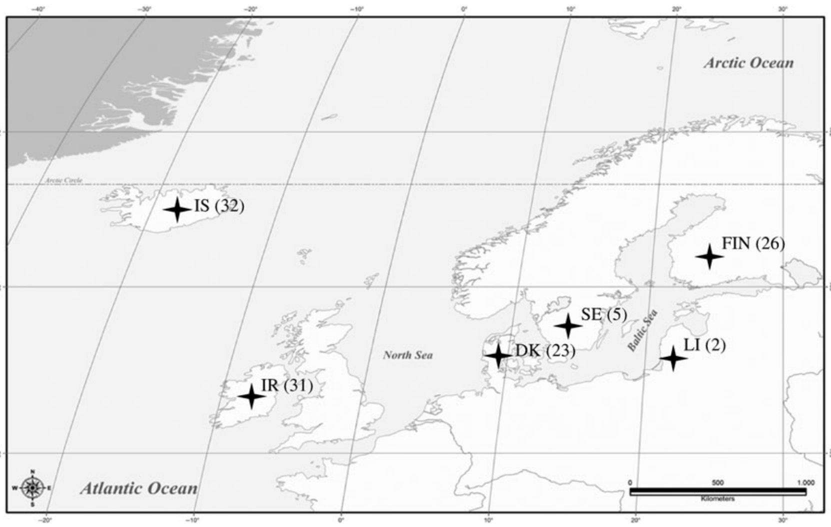

Fig. 1 Map of the Northern European countries included in this study: Denmark (DK), Finland (FIN), Ireland (IR), Iceland (IS), Lithuania, (LI) and Sweden (SE). Numbers in brackets denote the number of observations (afforested stands and their corresponding control) available from each country. © ESRI 2013.

SOC data were mostly given in stocks, but for those studies where SOC was presented in concentrations, stocks were calculated using the actual C concentration for a given layer, the bulk density (of fine soil fraction, <2 mm), and the thickness of the layer (Schrumpf *et al.*, 2011) all provided in the reference studies. Due to heterogeneity of sampling depths, it was not possible to establish a common maximum depth for the deeper soil layer data set. Therefore, we compiled a data set containing SOC stocks from 0 cm to a maximum of 20–30 cm, where the majority of data (71%) covered the 0–30 cm soil layer, while 19% of the data reported 0–25 cm and 10% reported 0–20 cm. We assumed that different maximum sampling depths did not affect the SOC trends qualitatively, as the meta-analysis was based on a relative effect measure (see below).

Differences in bulk densities and C concentrations in soils across different land uses are often rather large and therefore, comparison of SOC stocks based on fixed-depth layers (equivalent soil volume) can lead to an over- or underestimation of the SOC stocks (Ellert & Bettany, 1995; Lee *et al.*, 2009). Thus, several studies dealing with SOC stocks make use of equivalent soil masses for comparisons (Don *et al.*, 2011; Holmes *et al.*, 2011; Sauer *et al.*, 2012). Based on the study by Lee *et al.* (2009), we chose the minimum equivalent soil mass (ESM) to perform mass correction for both data sets. When C concentrations are not uniform within the soil profile, the minimum ESM correction is more accurate independently of the direction of change in the bulk density (Lee *et al.*, 2009). For the comparison of SOC stocks based on equivalent mass, the methodology described by Ellert & Bettany (1995) and Lee *et al.* (2009) was used as follows:

1. The mass per unit area of the soil according to fixed depth (M_{fd} in Mg ha^{-1}) in the afforested and control plots was calculated using the bulk density of the fine soil fraction (<2 mm) provided in the reference studies.
2. The M_{fd} in both afforested and control plots was compared and the plot with the lowest mass (lightest) was considered as the reference soil (M_{ref} in Mg ha^{-1}). To obtain equivalent mass, a certain amount of it (M_{sub} in Mg ha^{-1}) had to be subtracted from the heavy soil to match the mass of the reference soil (i.e. the lightest soil). This amount was calculated as:

$$M_{sub} = M_{fd} - M_{ref}$$

3. Finally, the C stock in equivalent soil mass was calculated as:

$$SOC_{equiv} = SOC_{fd} - SOC_{sub}$$

where SOC_{equiv} is the C stock in equivalent soil mass, SOC_{fd} is the original C stock calculated on a fixed-depth basis and SOC_{sub} is the C stock calculated for M_{sub}. All stocks are given in Mg ha^{-1}.

Data analysis

Study information was organized into categorical variables to assess the effects of former land-use, forest type, forest age, and soil textural class on SOC stocks: Former land uses were classified in four classes as either cropland, grassland, heathland, or barren land (sparsely vegetated eroded soils). Forest types were classified as coniferous, deciduous, or mixed forests based on the tree species used for afforestation. Soils were categorized in textural classes, i.e. coarse-textured or fine-textured soils, according to information derived from the reference studies. A third class was devoted to the volcanic soils from Iceland, due to their special features, which do not allow a straightforward definition of texture. Study designs were classified as chronosequences, repeated sampling studies, or paired-plot studies to facilitate further analyses of the effects of the degree of nonindependence between control and afforested soil samples on the meta-estimates.

We used log-transformed response ratio (effect size) to quantify the SOC stock difference between afforested and control plots for each data set considered (Hedges *et al.*, 1999):

$$\ln(\mathrm{RR}) = \ln(\bar{X}_{\mathrm{Aff}}/\bar{X}_{\mathrm{control}})$$

where RR is the response ratio, $\bar{X}_{\mathrm{Aff}}$ is the mean SOC stock from the afforested site and $\bar{X}_{\mathrm{control}}$ is the mean SOC stock in the corresponding control site. The variance of each log response ratio was approximated as

$$\begin{aligned}\mathrm{Var}[\ln(\mathrm{RR})] &= \mathrm{Var}[\ln(\bar{X}_{\mathrm{Aff}})] + \mathrm{Var}[\ln(\bar{X}_{\mathrm{control}})] \\ &\approx \frac{(\mathrm{SD}_{\mathrm{Aff}})^2}{n_{\mathrm{Aff}}\bar{X}_{\mathrm{Aff}}} + \frac{(\mathrm{SD}_{\mathrm{Control}})^2}{n_{\mathrm{Control}}\bar{X}_{\mathrm{Control},}},\end{aligned}$$

where n_{Aff} and n_{Control} denote the numbers of replicated measurements in afforested and control plots, respectively, $\mathrm{SD}_{\mathrm{Aff}}$ and $\mathrm{SD}_{\mathrm{Control}}$ denote the standard deviations of SOC measurements in afforested and control plots, respectively, and the rest are given above.

To facilitate comparison with similar studies, ln(RR) estimates were back-transformed (as $e^{\ln(\mathrm{RR})}$) and expressed as percentage (Table S2) relative to the value of the control.

For each data set, overall meta-estimates of SOC stock change were obtained from random effects meta-analysis models in which calculated effect sizes were weighted with the inverse of their respective variances (Hedges & Olkin, 1985). Within each data set, a range of mixed effects meta-regression models were used to evaluate (by regression coefficients) the change in effect size against different study variables (see Viechtbauer, 2010 for details): (i) The array of effect sizes was regressed separately on each categorical variable (see above) coded as dummy (0/1) variables; (ii) Relative SOC stock changes over time were obtained by regressing effect sizes against information on stand age; (iii) Pair-wise comparisons of temporal SOC stock change between classes of categorical variables were achieved using multiple meta-regression models.

The level of residual variation (*heterogeneity*) from fitting each meta-regression model was used to estimate the degree to which the study variables could explain effect size differences among studies, applying restricted maximum-likelihood (REML) estimation of a Q statistic (Viechtbauer, 2007). To include a few experiments in which standard deviations were either not reported or irretrievable in the analysis, missing values were imputed by sampling with replacement before each meta-regression (Wiebe *et al.*, 2006; see Appendix S1) and averaging parameters and test probabilities across 1000 runs.

One of the central assumptions in the ordinary statistical framework of meta-analysis is that effect sizes are mutually independent. Conflicting with this assumption, common sampling designs in land-use change and afforestation studies on SOC (i.e. chronosequences, paired-plot sampling, and repeated sampling) by definition introduce correlation among the data. Unlike previous quantitative reviews of afforestation effects on SOC stocks, we used a modified statistical framework capable of accounting for these sources of nonindependence. Two types of nonindependence were identified. Firstly, some studies compared several treatments against a shared control plot (e.g. pair-wise comparison between different afforested plots and a single control or chronosequence plots sharing one control). For this reason, only 52 unique control plots were identified in comparison to the grand total of 119 effect sizes. Following Lajeunesse (2011) correlated effect size estimates were integrated by adding a covariance term to the appropriate off-diagonal entries in the covariance matrix used for meta-regression (see Appendix S2). Secondly, some studies measured the same plot before and after an afforestation period (repeated sampling), the plots acting as their own control (same-plot controls). This type of correlation, which may result in a 'downscaling' of within-study variances, was integrated by subtracting a covariance term from the appropriate diagonal entries in the covariance matrix (Lajeunesse, 2011). To evaluate the influence of these sources of nonindependence, each of the above meta-analysis models were fitted with and without accounting for study design nonindependence. The results presented in the next section are based on models accounting for nonindependent effect sizes, unless otherwise stated.

All analyses were run in the R environment (version 2.14.0; R Development Core Team 2012), using matrix notation as described in Lajeunesse (2009, 2011) and applied in the *metafor* package (Viechtbauer, 2010). R code is available upon request.

Results

We identified afforestation as a main driver of changes in SOC stocks in Northern Europe with former land-use, forest age, forest type, and soil textural class as influencing factors moderating the effect of afforestation on SOC stocks. Changes in SOC stocks with respect to mean annual temperature, mean annual precipitation, and latitude were also tested, however, no obvious patterns were identified.

Effects of former land-use, forest type, and soil textural class

Positive effects of afforestation on SOC stocks [ln (RR) > 0] were identified on former cropland, heathland, and barren land (Fig. 2a). While this trend was found for both the 0–10 cm and the 0–20/30 cm soil layers, the effects were significant only in the uppermost layer. Afforestation of grasslands had a small negative effect on SOC stocks of −8% in both soil layers that was close to significant in 0–20/30 cm (P = 0.053). The overall effect of afforestation on former heathlands was similar in both soil layers, while for former barren lands differences between layers were larger, SOC accumulation being highest in 0–10 cm. However, the uncertainty of the estimates in this former land-use class was large due to the small number of observations (k = 6, only from Iceland) and high variability in the estimates.

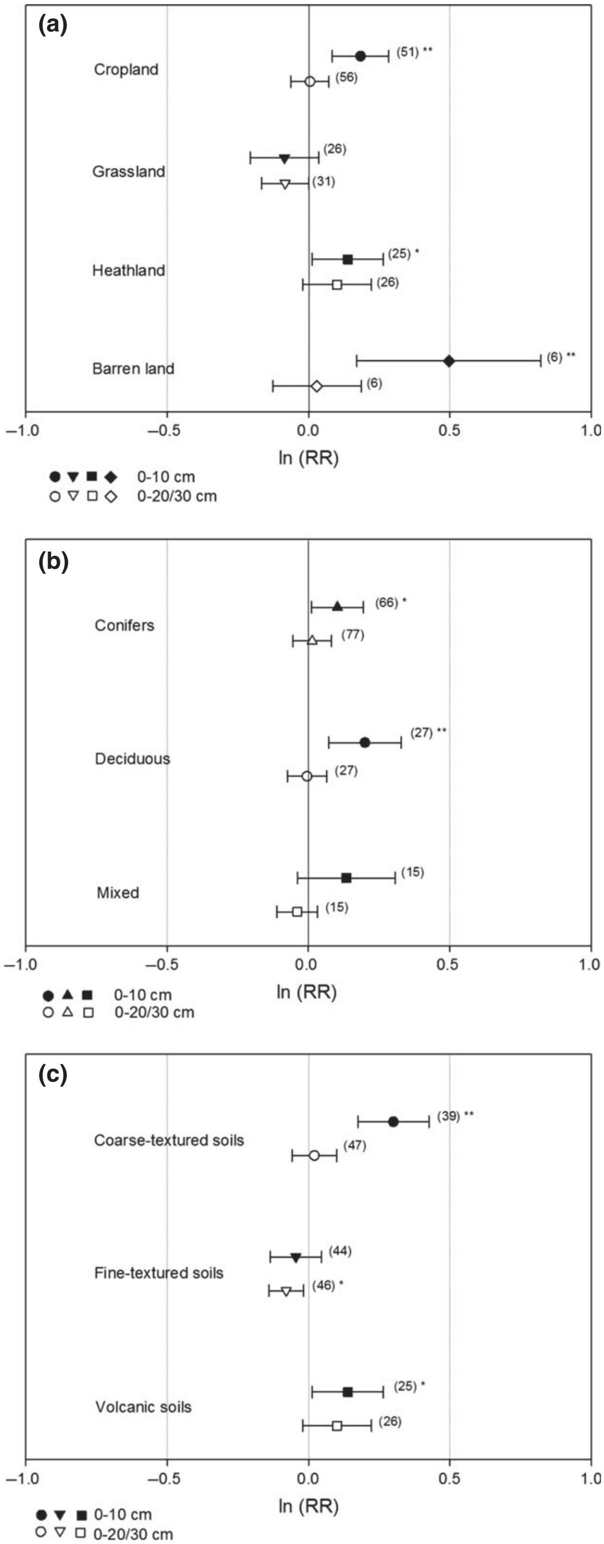

Fig. 2 Meta-estimates of effect size (log response ratios) based on the former land-use (a), forest type (b), and soil textural class (c). Numbers in brackets denote the number of observations and error bars are 95% confidence intervals. Significant results refer to $P < 0.05$ (*), $P < 0.01$ (**), and $P < 0.001$ (***).

There was no significant impact of afforestation on SOC stocks in 0–20/30 cm of the mineral soil in the three forest types (Fig. 2b). Nevertheless, significant positive effects of afforestation with conifers (11%, $P = 0.03$) and deciduous species (22%, $P < 0.01$) were found in the 0–10 cm soil layer, while there was no significant effect of afforestation with mixed species.

With regard to soil textural class, the largest positive effect of afforestation on SOC stocks was found in coarse-textured soils (35%, $P < 0.05$) and afforestation on volcanic soils (15%, $P < 0.05$) in 0–10 cm (Fig. 2c). Afforestation on fine-textured soils resulted in a significant negative effect on SOC stocks −8% in 0–20/30 cm (Fig. 2c). Strong confounding effects (overlap) were found between soil textural classes and former land-use classes. The results from these confounding effects are presented later.

The contribution of forest floors to SOC stock change

There was no significant effect of afforestation on SOC stock in the mineral soil (0–20/30 cm) in all three forest types (Fig. 3). However, including forest floor C in the analysis induced a positive effect on C stocks in all forest types (Fig. 3), in particular for conifers where the estimate increased from 1% (not significant) to 13% ($P < 0.05$). This corresponded well with the significantly (ANCOVA, $P < 0.01$) larger accumulation rate of forest floor C stocks over time in conifers

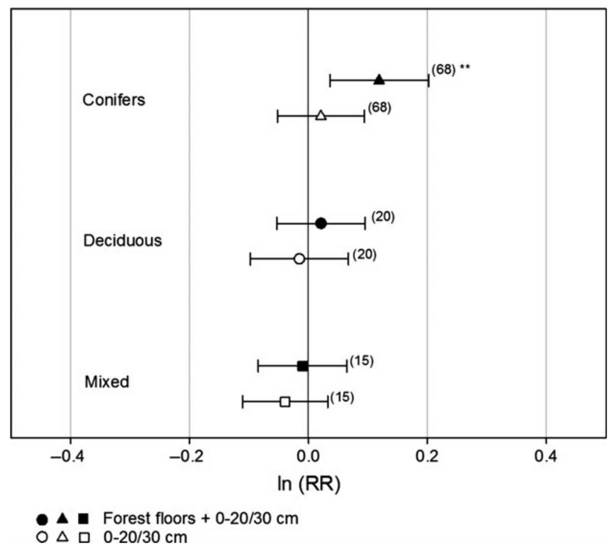

Fig. 3 Meta-estimates of effect size (log response ratios) based on the three forest type classes including forest floor C. Numbers in brackets denote the number of observations and error bars are 95% confidence intervals. Significant results refer to $P < 0.05$ (*), $P < 0.01$ (**), and $P < 0.001$ (***).

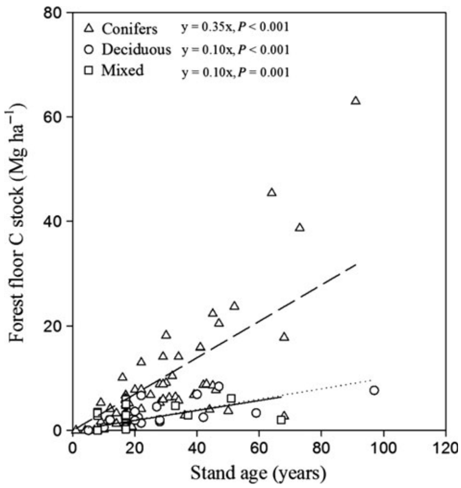

Fig. 4 Forest floor C stocks in the three different forest types as affected by stand age.

(0.35 Mg ha^{-1} yr^{-1}) compared to the deciduous and mixed forests (0.10 Mg ha^{-1} yr^{-1}) (Fig. 4).

Age dynamics

Meta-regression. Meta-regressions were carried out to investigate the temporal development of relative SOC stock changes (expressed by log response ratios) following afforestation. Meta-regression analyses including all observations from the 0–10 cm and 0–20/30 cm data sets (i.e. without considering variable classes) showed that overall SOC stocks increased with forest age, although only significantly in the 0–20/30 cm soil layer (0.4% yr^{-1}). Meta-regression on age by different former land-use classes indicated that afforestation on former croplands significantly ($P \leq 0.01$) increased SOC stocks both in the 0–10 cm soil layer (by 1.1% yr^{-1}, Fig. 5a) and in the 0–20/30 cm soil layer (by 0.8% yr^{-1}, Fig. 5b).

Afforestation on barren land, heathland, and grassland had no significant effect on SOC stocks in any of the soil layers (Fig. 5a and b), similar to the mean meta-estimate results (Fig. 2a).

When forest floors were included in the meta-regression analysis of the 0–20/30 cm soil layer by different former land uses, the positive rate of change in SOC following afforestation in former croplands increased from 0.8% yr^{-1}(Fig. 5b) to 1.3% yr^{-1} ($P < 0.01$, Figure S1). Forest floors also positively influenced the remaining former land uses, e.g. it compensated for the (small) negative effect of grassland afforestation (Fig. 5b, Figure S1), yet none of these trends were significant.

Thirty-year threshold and interaction groups. The temporal patterns observed indicated that the direction of change in SOC stocks is highly variable during the initial phase following afforestation (0–30 years) and detectable gains in SOC stocks appear in later stages (Fig. 5). Overall meta-estimates based on complementary subsets of studies (age<30 years or age>30 years) indicated a negative effect of afforestation on SOC stocks in the former land-use classes within the first 30 years after afforestation, however only significant for former croplands (Fig. 6). After 30 years, the effect of afforestation on SOC change was significantly positive on former croplands and heathlands (Fig. 6), especially for afforestation of croplands (38% gain in SOC) despite the small amount of observations ($k = 9$) available. On the contrary, there was still no significant change in SOC stocks of former grasslands ($P = 0.1$) 30 years after afforestation (Fig. 6).

We found that 85% of the afforested heathlands were coniferous forests, while in former croplands conifers represented 66% of the observations. In grasslands, forest types were slightly more balanced, although conifers were still predominant (58%). Strong confounding was also observed between former land-use classes and soil textural classes. For example, in former grasslands more than 80% of the afforested-control pairs were on fine-textured soils, while all heathland observations were on volcanic soils, meaning that the effect of former land-use and soil textural class could not be separated. Forest type and soil textural class were also confounded in some cases as the majority of coniferous forests were planted on coarse-textured soils (40%) and more than half of the observations from deciduous forests were on fine-textured soils. To avoid the limitations resulting from these confounding effects (i.e. overlap between variable classes), we evaluated the effects on SOC within major interaction groups (such as cropland × conifer, grassland × fine-textured soils, etc.) across the three main categorical variables: former land-use, forest type, and soil textural class (Table 1). All interaction groups consisted of five or more observations and originated from at least two different studies. Meta-estimates for the interaction groups that fulfilled these criteria were then calculated with respect to the 30-year threshold (Table 1).

During the first 30 years after afforestation, only the interaction groups cropland × conifers, and conifers × coarse-textured soils showed a significant negative response of SOC after afforestation (Table 1), and the negative effect on SOC in these interaction groups was larger than those observed for croplands or conifers alone. More than 30 years after afforestation, the effects on SOC stocks became mainly significantly positive (except for former grasslands and fine-textured soils and their interaction groups), and meta-estimates for some interaction groups were more comparable to those for the categorical variables alone (i.e.

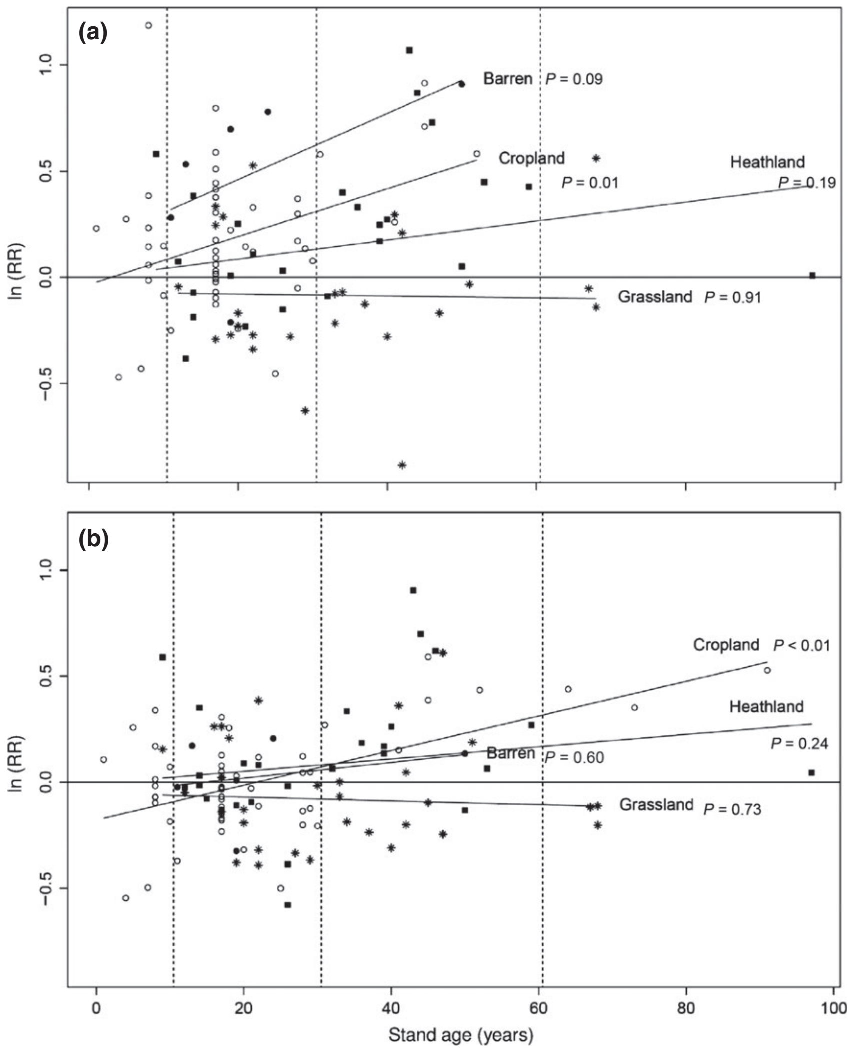

Fig. 5 Meta-regression of the former land-use classes: barren (closed circles), cropland (open circles), heathland (squares), and grassland (stars) against age, for the (a) 0–10 cm (R^2 = 0.26) and (b) 0–20/30 cm (R^2 = 0.17) data sets.

croplands × conifers or croplands × coarse-textured soils had similar meta-estimates as the cropland or conifer class alone). Also, in the interaction group conifers × fine-textured soils, the significantly positive effect of afforestation with conifers (Figs 2b and 3) was masked by the effect of fine-textured soils in the interaction group, resulting in a significant SOC loss after afforestation similar to the negative effect on SOC stocks of fine-textured soils alone.

Effects of mass correction and accounting for sample dependence

Meta-regressions based on mass-corrected and fixed-depth SOC stock data in 0–10 cm (Table S3) indicated that without mass correction, the SOC stock effects were generally overestimated. This was especially the case for former croplands, which showed an accumulation of 1.1% yr^{-1} in the mass-corrected data as compared to the 1.5% yr^{-1} in the fixed-depth data. On the other hand, the effect of mass correction was negligible in the analysis of the 0–20/30 cm data set (Table S3).

With regard to the effect of accounting for sample dependence, in most of the cases, the meta-regression analysis indicated some overestimation when this issue was ignored, e.g. the meta-estimates for the conifer class decreased from 1.10 to 1.03 gain in SOC in the 0–10 cm data set and from 1.9 to 1.4 in the 0–20/30 cm data set when sample dependence was accounted for (Table S4).

Discussion

Influence of former land use

Land that has been cultivated, degraded (deforestation, overgrazing, etc.), or eroded is characterized by low soil

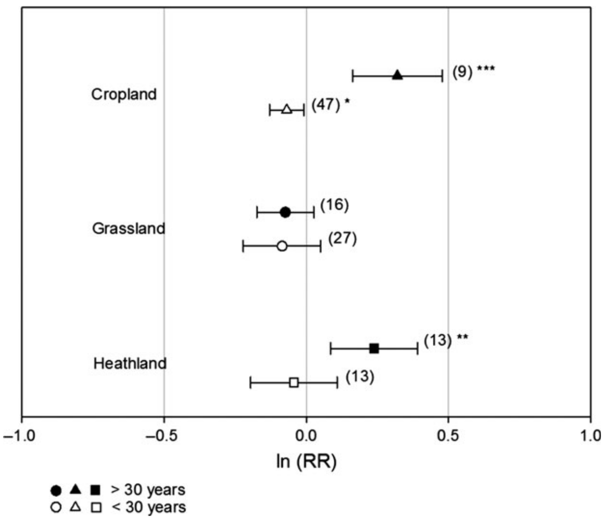

Fig. 6 Meta-estimates of effect size (log response ratios) based on the former land-use classes for the 0–20/30 cm data set less than 30 years (open symbols) and more than 30 years (closed symbols) since afforestation. The barren land class is not included due to insufficient available data. Numbers in brackets denote the number of observations and error bars are 95% confidence intervals. Significant results refer to $P < 0.05$ (*), $P < 0.01$ (**), and $P < 0.001$ (***).

organic matter content and is therefore expected to be more prone to gain C following woody vegetation encroachment with afforestation (Nave *et al.*, 2013). Our results demonstrate that in Northern Europe, afforestation on soils that have been depleted in SOC (such as croplands, heathlands, and barren lands) due to low organic matter inputs, disturbances (e.g. tillage and harvest in croplands), high decomposition rates, and/or erosion, may become net C sinks 3 decades after afforestation. More SOC-rich soils like those found in grasslands do not become SOC sinks following afforestation within the same time span and instead maintain the same SOC stock or experience small SOC losses. Grasslands maintain a permanent vegetation cover and litter inputs (and thereby SOC) often occur belowground with enhanced root turnover supporting high microbial activity (Don *et al.*, 2011; Li *et al.*, 2012). With time since afforestation, the above- to belowground biomass ratio increases compared to former grassland, where belowground C inputs are more important (Peichl *et al.*, 2012). Also, it has been hypothesized that microbial products of decomposition from labile plant material (e.g. fine roots) may be the main precursor of stable soil organic matter due to organo-mineral interactions (Rasse *et al.*, 2005; Cotrufo *et al.*, 2013).

While the positive effect of afforestation on SOC was significant in the 0–10 cm data set following afforestation on former croplands, on heathlands and barren

Table 1 Meta-estimates of SOC stock effect for variable classes and major interaction groups across former land-use (FLU), forest type (FT), and soil textural class (ST) in the 0-20/30 cm data set. The number of observations is denoted by '*k*'. Not determined (nd) indicates that the interaction group did not fulfill the criteria and was therefore not analyzed. Significant results refer to $P < 0.05$ (*), $P < 0.01$ (**), and $P < 0.001$ (***), 'ns' refers to nonsignificant

	≤30 years			>30 years		
	k	ln(RR)		*k*	ln(RR)	
Variable class						
FLU						
Cropland	47	−0.07	*	9	0.32	***
Grassland	15	−0.09	ns	16	−0.07	ns
Heathland	13	−0.04	ns	13	0.24	**
FT						
Conifers	48	−0.10	**	29	0.20	***
Deciduous	21	−0.01	ns	6	−0.01	ns
ST						
Coarse-textured soils	35	−0.07	ns	12	0.28	***
Fine-textured soils	32	−0.06	ns	14	−0.12	**
Volcanic soils	13	−0.04	ns	13	0.24	**
Interaction group						
FLU : FT						
Cropland × Conifers	28	−0.11	**	9	0.32	***
Cropland × Deciduous	9	0.02	ns	nd	nd	nd
Grassland × Conifers	9	−0.07	ns	9	−0.04	ns
Heathland × Conifers	11	−0.05	ns	11	0.26	**
FLU : ST						
Cropland × Fine-textured soils	20	−0.06	ns	nd	nd	nd
Cropland × Coarse-textured soils	27	−0.08	ns	8	0.38	***
Grassland × Fine-textured soils	12	−0.06	ns	13	−0.11	*
Heathland × Volcanic soils	13	−0.04	ns	13	0.24	**
FT : ST						
Conifers × Fine-textured soils	17	−0.08	ns	7	−0.14	**
Deciduous × Fine-textured soils	12	−0.01	ns	nd	nd	nd
Conifer × Coarse-textured soils	20	−0.13	*	11	0.29	***
Conifers × Volcanic soils	11	−0.05	ns	11	0.26	**
Deciduous × Coarse-textured soils	7	0.00	ns	nd	nd	nd

lands, no significant effects were found in the 0–20/30 cm soil layer (Fig. 2a). These results suggest that it requires a longer period of time to observe similar effects in the deeper soil layer (0–20/30 cm). The uppermost layers of the mineral soil are more rapidly affected by land-use changes as they are directly in contact with the aboveground inputs of C (via foliar materials, twigs, branches, etc.), but also in the case of former cropland soils, the cessation of tillage can be the cause for greater SOC stocks after afforestation near the top of the soil profile. Carbon inputs from upper compartments may be incorporated in deeper soil layers, through e.g. soil fauna (Don *et al.*, 2008) or leaching of dissolved organic C from forest floors (Hansson *et al.*, 2010). These mechanisms in addition to C inputs from roots would all favor SOC accumulation in deep soil layers in the long term.

In a recent global meta-analysis by Shi *et al.* (2013), SOC change in the upper 10 cm after afforestation of former cropland resulted in a positive mean response ratio of 1.7, compared to the 1.2 found in this study (Table S2), and for afforestation of grasslands, we found a very similar estimate of 0.9 (C loss). Other global meta-analyses showed that SOC stocks after afforestation in croplands increased by 26% (Laganière *et al.*, 2010) and 18% (Guo & Gifford, 2002). Similarly, we found an increase in 20% for the 0–10 cm layer in Northern Europe; however, this effect was absent in the 0–20/30 cm layer (1%) (Fig. 2a; Table S2), suggesting that the process of SOC accumulation in deeper soil layers is slower than in surface soil layers. These results altogether highlight that despite the difference in geographical scales (global vs. regional) and thus in climatic conditions, the influence of previous land use is prevailing. However, future studies should focus on the mechanisms in control of SOC dynamics by investigating how the amount and distribution of C between genetic soil horizons changes with land use.

Forest type and the contribution of forest floors

We found significantly positive effects on SOC of conversion into coniferous and deciduous forests (Figs 2b and 3). In Northern Europe, conifers have dominated in afforestation due to their high production value and lower cultivation costs (Stanturf & Madsen, 2006). In our data set, 66% of the observations reported C stocks from coniferous forests, and therefore the low amount of data from the other two forest types (deciduous and mixed forests) limits the possibility of drawing further conclusions. Unbalanced sources of data, i.e. when some categories in certain variables are clearly dominating, are a common limitation in synthesis studies. Nevertheless, studies consisting of a more balanced data set (Paul *et al.*, 2002; Berthrong *et al.*, 2009; Laganière *et al.*, 2010) reported tree species effects on the recovery of SOC stocks after afforestation, and SOC accumulation was highest in deciduous forests. We also found a larger significant effect of deciduous afforestation on SOC stock change for the 0–10 cm layer (Fig. 2b), while the change was smaller but still significant for coniferous afforestations. Different species-related trends in SOC change were thus only observed for forest floor C stocks (Fig. 4); the significantly higher rates of C sequestration found in conifer forest floors compared to deciduous and mixed forests, agree with forest floor data from afforestation chronosequence studies in Denmark (Vesterdal *et al.*, 2002; Bárcena TG, Gundersen P and Vesterdal L, in preparation). In contrast, Poeplau *et al.* (2011) found no significant differences in forest floor C accumulation between conifer and deciduous forests in the temperate zone. Accumulation of forest floor with stand age requires that plant litter production exceeds decomposition (Zak *et al.*, 1990; Thuille *et al.*, 2000). Tree species differences in forest floor C have been attributed to differences in litter quality that in turn affect decomposition rates (Vesterdal & Raulund-Rasmussen, 1998; Vesterdal *et al.*, 2012). Temperature, precipitation, soil properties, etc. varying across climatic zones may also lead to differences in forest floor C accumulation within the same forest type. In this study, some sites belong to the subboreal/boreal zone with slow C turnover and are in many cases afforested with conifer species, such as Norway spruce, which are more prone to accumulate C in organic layers due to the low quality of the litter as compared to broadleaf litter (Zhang *et al.*, 2008; Hansson *et al.*, 2011; Vesterdal *et al.*, 2012). Conifer litter is known to acidify soils, especially in combination with low buffer capacity that is typically found in nutrient poor coarse-textured soils (Baritz *et al.*, 2010). In Europe, coarse-textured soils are commonly managed with conifers (40% of the data in our database belong to this interaction group) resulting in high forest floors C stock (Baritz *et al.*, 2010), as we observed in this study (Fig. 4).

The importance of considering the forest floors as an integral part of the forest soil has been emphasized in previous studies (Scott *et al.*, 1999; Laganière *et al.*, 2010; de Schrijver *et al.*, 2012). Forest floor C has been shown to offset SOC losses in the mineral soil after afforestation on former cropland (Vesterdal *et al.*, 2002) and grasslands (Poeplau *et al.*, 2011), which indicates that omission of C stocks in forest floors results in an incomplete estimation of SOC. In our study, forest floors had a positive influence in the effect sizes observed for the three forest types (Fig. 3), especially for conifers, where the positive effect of afforestation on SOC became significant when forest floors were included.

Effects of soil textural class

In Northern Europe, coarse-textured soils and volcanic soils were the most prone to gain SOC following afforestation, while more fine-textured soils affected SOC sequestration negatively (Fig. 2c). This finding contradicts the general pattern of higher SOC accumulation in fine textured in soils that has been observed previously (Paul *et al*., 2002; Laganière *et al*., 2010). However, there is still an ongoing discussion on the effects of soil type and texture on SOC stocks with land-use change, as other studies have not detected any effect from texture (Karhu *et al*., 2011; Vindušková & Frouz, 2013). Also, it has been pointed out that the clay type (low vs. high activity clays) rather than the clay content affects SOC sequestration (Don *et al*., 2011). In this study, results from soil textural classes were strongly confounded with the other major categorical variables former land use and forest type. Therefore, we extend the discussion of meta-estimates derived from the three soil textural classes to the interaction group paragraph.

Age dynamics

Meta-regression. Previous studies have suggested a faster response in SOC change in superficial mineral soil layers compared to deeper soil layers after land-use change (Paul *et al*., 2002; Shi *et al*., 2013). However, in our study, the lack of data from old afforested stands (>60 years) in the 0–10 cm data set resulted in a lower overall (i.e. without considering variable classes) C accumulation rate for this layer (0.2% yr^{-1}) compared to 0–20/30 cm (0.4% yr^{-1}) when analyzed by meta-regression. Carbon accumulation rates after afforestation on former cropland (0.8% yr^{-1} in 0–20/30 cm, Fig. 5b) are within the range found in similar studies, e.g. the 1.51% C gain yr^{-1} observed in a global quantitative review (Paul *et al*., 2002) or 0.83% yr^{-1} for the temperate region (Poeplau *et al*., 2011) and are somewhat higher than the 0.15% yr^{-1} found in the United States (Nave *et al*., 2013).

Temporal patterns of SOC change based on meta-regression indicated that positive (and significant) changes following afforestation require long periods of time, as it was also suggested by other recent meta-regressions (Nave *et al*., 2013).

Thirty-year threshold and interaction groups. SOC dynamics in afforested stands were highly dependent on forest age (Fig. 5). At initial stages, trees have little impact on the soil and the legacy of the previous land use in terms of soil microbial activity, physical soil properties, availability of soil nutrients, etc. probably play a major role. As forests develop, input of C from litterfall increases and stabilizes approximately 20–30 years after afforestation with canopy closure (Vesterdal *et al*., 2007). Previous studies have suggested that afforestation on agricultural land requires more than 30 years before significant increases in SOC can be detected (Paul *et al*., 2002; Laganière *et al*., 2010; Nave *et al*., 2013). In afforested croplands, Poeplau *et al*. (2011) found that the (positive) rate of change in C was five times larger when calculated at 100 years as compared to 20 years. Similarly, a continuous increase in SOC sequestration rates was observed 15 years after afforestation in the United States (Nave *et al*., 2013). Therefore, according to the patterns that have been observed in the above-mentioned studies, we consider a reasonable approach to compare changes in SOC stocks in early and later stages following afforestation by establishing a threshold at 30 years' time. Our data confirmed the validity of the threshold, with clear differences between estimates calculated for both periods, especially with regard to the direction of change, shifting from negative to positive effects in SOC before and after 30 years since afforestation, respectively (Fig. 6; Table 1).

In meta-analyses and other quantitative reviews, confounding effects are a common obstacle that complicates the interpretation of results and have often not been addressed statistically. Therefore, there is still a need to deal with the limitations arising from confounding effects (Paul *et al*., 2002). To our knowledge, no previous synthesis studies in the context of land-use change and SOC have quantified confounding effects between variables. In some studies, conflicting data were excluded from the analysis if confounded effects were identified (Scott *et al*., 1999; Don *et al*., 2011; Li *et al*., 2012). In other cases, confounding effects were detected and discussed as limitations of the data, but no analyses were performed (Johnson & Curtis, 2001; Nave *et al*., 2009, 2013; Jerabkova *et al*., 2011). We evaluated confounding effects by analyzing major interaction groups in separate analyses in an attempt to optimize the interpretation of the data. We observed that the interaction groups between former land-use classes and forest types and soil textural classes, respectively, were often dominated by the direction of changes in the former land-use class alone, and that the meta-estimates observed for the variable classes alone were either compensated or enhanced when incorporated in an interaction group.

In former grasslands, the negative effect of afforestation on SOC could hardly be detached from the effect of soil textural class, as grassland afforestation was mainly on fine-textured soils. The effect of afforestation on SOC was negative in both the grassland class alone and the interaction group grassland × fine-textured soil (Table 1). On the one hand, these results are in line with the SOC loss reported for afforestation on former grasslands (Guo & Gifford, 2002; Paul *et al.*, 2002; Shi *et al.*, 2013), but on the other hand they contrast the hypothesis of a higher SOC accumulation potential in fine-textured soils due to formation of organo-mineral complexes (Paul *et al.*, 2002; Callesen *et al.*, 2003; Laganière *et al.*, 2010), which provide physical protection against decomposition. This discrepancy regarding our findings of negative effects on SOC after afforestation on fine-textured soils (Fig. 2c) could be attributed to the influence of other mechanisms, such as the initial SOC status in grassland that was the dominating former land use within fine-textured soils. Continued belowground litter inputs through the dense fine root system of grasslands decrease after tree planting/afforestation. The initial SOC from grassland is most likely quickly mineralized by the microbial community while aboveground litter inputs from the young forest cannot provide the same quantity and quality of litter directly to the soil resulting in a SOC loss (or no change) following this land-use change.

A large proportion of deciduous forests in our database were planted on fine-textured soils. Compared to poor coarse-textured soils (mainly planted with conifers in our database), fine-textured soils are more nutrient rich and therefore C mineralization may be faster than accumulation also because deciduous litter is often more easily decomposed than coniferous litter (Vesterdal & Raulund-Rasmussen, 1998; Berg *et al.*, 2000; Thuille & Schulze, 2006). Thus, decomposition may then override C accretion in the deciduous × fine-textured soils interaction group, also because many aboveground inputs may be already mineralized in the litter layer (Schmidt *et al.*, 2011), resulting in negative effects of afforestation on SOC. Also, other recent studies have reported higher SOC stocks in Norway spruce compared to broadleaved species as well (Gurmesa *et al.*, 2013; Vesterdal *et al.*, 2013).

We conclude that the confounding effect between variable classes has an impact on the estimate of the effect of afforestation on SOC, which can be examined by comparing the interaction group estimates with the variable class estimates alone. In particular, our results suggest that former land use in Northern Europe is a major contributing factor in control of the changes in SOC stocks after afforestation.

Effects of mass correction and accounting for sample dependence

Results based on mass-corrected and fixed-depth SOC stock data indicated that without mass correction, SOC stock change was generally overestimated in 0–10 cm (Table S3), while only negligible differences were found for the 0–20/30 cm layer. These observations agree with the idea of larger effects of mass correction in surface soil layers, which are more prone to be affected by bulk density and C concentration changes and comprise higher amounts of roots and residues (Lee *et al.*, 2009). We did not find a strong influence of mass correction in the effect sizes as others have seen (Don *et al.*, 2011), however, we hypothesize that this could be due to the young age of most of the afforestation plots, in which differences in bulk density with respect to the control plots are not as large as in later stages. Nevertheless, it is important to consider the impact of bulk density in SOC stocks when comparing different land uses, to provide more accurate estimates.

The violation of the assumption of independent effect sizes is a common problem in ecological meta-analyses (Hedges *et al.*, 2010), however, the consequences of this issue have only been addressed recently. Curtis & Queenborough (2012) found that the response ratio obtained for a nitrogen addition meta-analysis study (Nave *et al.*, 2009) decreased from 1.21 to 1.17 if they accounted for sample nonindependence, i.e. accounting for the use of the same control plot for multiple afforested plots. Similarly, we observed some differences between estimates when not accounting for nonindependence of control plots (Table S4). Differences among estimates were not dramatic and the identified trends persisted. However, it remains unknown whether this assumption which was ignored in previous meta-analyses of SOC and LUC could have had more important consequences due to stronger effects of sample nonindependence. Therefore, future studies should use the statistical tools for more rigorous analysis that are now available and thereby provide more precise estimates in which the nature of the data is taken into account.

Conclusion

Afforestation in Northern Europe had a positive effect on SOC stocks approximately 3 decades after land-use change, with the exception of afforestation on grasslands. We found the largest SOC sequestration impact when former croplands were afforested, and this effect was enhanced when conifer species were planted, especially when forest floors were accounted for.

Changes in SOC stocks following afforestation were slower in Northern Europe as compared to estimates from tropical, temperate, or global data sets, most likely due to the influence of less favorable climatic conditions that induce slow tree growth and thereby low C input rates. The good correspondence between previous estimates of SOC change on different former land uses and the results presented here clearly suggests that previous land use plays a major role on SOC change after afforestation.

The improvement of the analytical methods by considering the limitations of confounding effects at global or regional scales, the influence of nonindependence of samples in the meta-estimates, and the use of mass correction are all state-of-the-art tools that should be incorporated in synthesis studies of SOC and land-use change, and further efforts are needed to address their influence and facilitate a more accurate interpretation of large data sets.

Acknowledgements

We thank Michael Wellock, Antti Wall, Luisella Celli, Veiko Uri, and Karna Hansson for providing valuable additional information from published studies and Gudmundur Halldórsson for allowing the use of unpublished data. This study was launched by the Forest Soil Carbon-sink Nordic Network (FSC-Sink) funded by the Nordic Council of Ministers and contributes to the Nordic Centre of Advance Research on Environmental Services (CAR-ES) funded by SNS (SamNordisk Skogsforskning).

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Imputation.
Appendix S2. Adjustment for correlated effect sizes.
Figure S1. Meta-regression of the former land-use classes: cropland (open circles), heathland (squares), and grassland (stars) against age, for the 0–20/30 cm data set including forest floors.
Table S1. Reference studies used in the meta-analysis.
Table S2. Back-transformed response ratios (RR) also expressed as percentage of change.
Table S3. Meta-regression estimates for the mass-corrected and nonmass-corrected SOC stocks in the 0–10 cm and 0–20/30 cm data sets.
Table S4. Meta-regression estimates from analyses with and without accounting for sampling depence in the 0–10 cm and 0–20/30 cm data sets.