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Declining trend of carbon in Finnish cropland soils in 1974–2009

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Abstract

Soil organic matter not only affects soil properties and productivity but also has an essential role in global carbon (C) cycle. We studied changes in the topsoil C content of Finnish croplands using a dataset produced in nationwide soil monitoring. The monitoring network consisting of fields on both mineral and organic soils was established in 1974 and resampled in 1987, 1998, and 2009. Over the monitoring period from 1974 to 2009, cultivated soils showed a continuous decline in C concentration (g kg^{-1}). In organic soils, C concentration decreased at a mean rate of 0.2–0.3% yr^{-1} relative to the existing C concentration. In mineral soils, the relative decrease was 0.4% yr^{-1} corresponding to a C stock (kg m^{-2}) loss of 220 kg ha^{-1} yr^{-1}. The change in management practices in last decades toward increasing cultivation of annual crops has contributed to soil C losses noted in this study. The results, however, suggest that the C losses result partly from other processes affecting cultivated soils such as climatic change or the continuing long-term effect of forest clearance. We estimated that Finnish cropland soils store 161 Tg carbon nationwide in the topmost 15 cm of which 117 Tg is in mineral soils. C losses from mineral soils can therefore total up to 0.5 Tg yearly.

Keywords: arable land, boreal zone, generalized linear mixed models, long-term monitoring, soil organic carbon, soil sampling

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Introduction

Most arable topsoils contain 2–10% organic matter (Bot & Benites, 2005). Although the concentration is that low it is critical for sustainable food production and has global environmental implications. Organic matter affects the chemical, physical, and biological properties of soil (Matson *et al.*, 1997; Bot & Benites, 2005). It eases the seedbed preparation and other agricultural practices and affects the yields of cultivated crops (Loveland & Webb, 2003; Brock *et al.*, 2011). Soil organic matter also has several environmental implications such as preventing risk of soil erosion (Jankauskas *et al.*, 2007) and reducing leaching of nutrients and pesticides to aquatic ecosystems (Stoate *et al.*, 2001).

Rising carbon dioxide (CO_2) concentration in the atmosphere has steered attention to the role of soil organic matter in carbon sequestration (Obersteiner *et al.*, 2010). Concerns have developed especially over the fate of terrestrial carbon under changing climate and effects of land-use change and management practices on soil carbon sequestration. Cultivated mineral soils can be either sources or sinks of carbon depending on the environmental conditions and management (Lal, 2004). Due to the high decomposition rate of drained peatland soils, it is widely acknowledged that cultivation of organic soils stimulates high emissions of CO_2 to the atmosphere (Grønlund *et al.*, 2008; Berglund & Berglund, 2010; Maljanen *et al.*, 2010). In Finland, about 11% of total arable area is organic soils, but it is estimated that most of the CO_2 emissions related to agriculture arise from the decomposition of organic soils under cultivation (Statistics Finland, 2012).

Extensive soil monitoring programs are valuable in providing information on temporal changes in soil carbon content. However, such datasets are scarce even worldwide. Therefore, most carbon change estimates in Europe have been derived either from long-term field experiments (Kirchmann *et al.*, 1994; Hopkins *et al.*, 2009) or simulation modeling (Smith *et al.*, 2005; Gervois *et al.*, 2008).

Repeated soil monitoring studies in European croplands have shown contrasting trends in carbon content. While most studies suggest that the soil carbon in mineral soils is decreasing as in England and Wales (Bellamy *et al.*, 2005), Belgium (Lettens *et al.*, 2005; Sleutel *et al.*, 2006; Meersmans *et al.*, 2009), Finland (Mäkelä-Kurtto & Sippola, 2002), Norway (Riley & Bakkegard, 2006), Austria (Dersch & Böhm, 1997), and France (Saby *et al.*, 2008), others show no unequivocal trend as in Denmark (Heidmann *et al.*, 2002) and the Netherlands (Hanegraaf *et al.*, 2009) or even increase in carbon content as in the Netherlands (Reijneveld *et al.*, 2009). The trends in carbon content can also be contrasting between regions within a country or time periods observed. Contrary to croplands, the long-term soil monitoring studies have generally shown increase in

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carbon content in permanent grasslands (e.g., Lettens *et al.*, 2005) indicating the importance of management on soil carbon sequestration.

The aim of this study was to determine the current state and temporal trend of carbon content in cropland soils in Finland based on a long-term soil monitoring study. The dependence of the trend on the management, geographical region, and soil class was also examined. Furthermore, to evaluate the atmospheric impact, the nationwide carbon stock of Finnish croplands was determined. Previously, the monitoring data has been utilized to study changes in soil chemical properties between two successive sampling times (Sippola & Tares, 1978; Erviö *et al.*, 1990; Mäkelä-Kurtto & Sippola, 2002) but this is the first analysis on the data over the whole monitoring period including the most recent survey in 2009.

Materials and methods

Description of the study area

Although Finland belongs to the boreal zone, there is high regional variation in climate, soil texture, and management of cultivated land within the country (Table 1). Therefore, agricultural production has strongly centered in the most favorable coastal region in south and west.

Finland has moist continental climate, characterized by cold winters and relatively cool summers. Temperatures in July average from 8°C to 18°C depending on the region. February is usually the coldest month, with average temperatures ranging from −14°C to −2°C. Snow covers the ground for several months in the winter time. Depth and duration of ground frost varies greatly with year and soil texture, but typically the ground frost penetrates to the topmost 10–60 cm soil layer and lasts from November until April to early June in the spring. Commercial agricultural production extends as far north as the Arctic Circle, but short and cool summers, occasional frosts during growing season and long-lasting ground frost restrict the cultivation of demanding crops in the north.

In Finland, soils were formed after the last glaciation during the last 12 000 years and geological history has strongly contributed to the formation of soils suitable for food production. Most of the agricultural lands have been established on the subaquatic clays of the coastal region formed during several evolutionary stages of the Baltic Sea. Soils in the interior of the country in the north and east are formed mainly through glacial and glaciofluvial processes and are generally coarser in texture (Table 1). Organic soils are common in the north. According to WRB classification (FAO, 1998), the Finnish cultivated soils are mostly Cambisols (34%), Podzols (27%), Regosols (19%), and Histosols (10%) (Lilja *et al.* 2006).

Barley, oat, wheat, and turnip rape are the most commonly cultivated annual crops covering 20%, 13%, 9%, and 6% of agricultural area, respectively (Tike, 2012). Annual crops are usually spring-sown varieties. Perennial crops mainly consist of silage (20%), dry hay (5%), and pasture (3%). Cultivation of annual crops is centered in the south whereas perennial crops and animal husbandry are prevailing in the north and east (Table 1). Perennial crops are usually grown for 3–4 years

Table 1 Area of cropland, basic climate, and proportional distribution of cropland area in groups of soil class and management in the four geographical regions of Finland. Regional division is shown in Fig. 1 and the definition of classification is presented in "Data classification" section. The number of sampling plots in the monitoring network in 2009 is also shown (*n*)

	North	East	West	South
Area of Cropland, 1000 ha*	73	460	724	1084
Climate 1970–2009†				
Mean temperature, °C	−0.4	2.9	2.8	4.5
Annual precipitation, mm	471	550	494	561
Growing season (T > 5°), days	127	161	160	171
Effective temperature sum, °C	730	1140	1050	1260
Soil Class, % of cropland area‡				
Clay	0 (n = 0)	15 (n = 22)	17 (n = 3)	72 (n = 81)
Fine	11 (n = 16)	20 (n = 41)	34 (n = 46)	12 (n = 67)
Coarse	57 (n = 24)	54 (n = 66)	30 (n = 49)	12 (n = 34)
Organic, 20%<OM<40%§	7 (n = 2)	8 (n = 15)	14 (n = 15)	3 (n = 7)
Organic, OM>40%§	25 (n = 9)	3 (n = 7)	5 (n = 17)	1 (n = 5)
Management 1995–2009, % of cropland area*				
Annual cropland	1 (n = 1)	13 (n = 9)	32 (n = 35)	57 (n = 94)
Perennial cropland	59 (n = 31)	22 (n = 36)	13 (n = 29)	4 (n = 16)
Crop rotation	39 (n = 19)	64 (n = 106)	55 (n = 66)	39 (n = 84)

*Land Parcel Identification System (EU, 1992).
†Finnish Meteorological Institute, 10 km × 10 km gridded dataset (Venäläinen *et al.*, 2005).
‡Finnish soil database, digital soil map at scale 1:250 000 (Lilja *et al.*, 2006).
§Database of soil fertility samples, Viljavuuspalvelu Oy.

following either cultivation of annual crops or reestablishment of perennial vegetation. Perennial crops are regularly renewed as they do not remain productive for long time periods in Finnish climatic conditions. Crop-grass rotation including the annual and perennial crops is a common management practice. In 1995–2009, 46% of cropland area included both annual and perennial crops in the cultivation scheme [Land Parcel Identification System (EU, 1992)]. In the past, the crop-grass rotation was even more common. Due to regular tillage and involvement of annual crops, practically all actively used agricultural lands in Finland fall in the "cropland" category according to the definition by the IPCC (IPCC, 2006). In Finland, there are no large areas of grazing land or permanent grasslands. Area of "grassland" as defined by IPCC is small (0.2 million ha) comprising mainly of abandoned agricultural area, but also grasslands and meadows more than 5 years old (Statistics Finland, 2012).

Soil samples and data

To study nutrient, heavy metal, and organic matter contents of Finnish cultivated soils, a nationwide survey was conducted in 1974 (Sippola & Tares, 1978) with the collection of soil samples from 2042 sampling plots growing timothy (*Phleum pratense*). Preceding the sampling, the locations of the sites were defined regionally but the final selection of the sampling plots took place in the field. Sites in close proximity to roads, electric power lines or drainage ditches were avoided.

The soil sampling was repeated in 1987 (Erviö *et al*., 1990), 1998 (Mäkelä-Kurtto & Sippola, 2002) and 2009 by taking samples at the same plots as in 1974. However, due to diminished resources, the number of sampling plots has decreased from the original 2042 plots to 1362 in 1987, 720 in 1998, and 611 in 2009. The number of plots in the network was reduced ensuring the regional coverage of the network. Figure 1 shows the network in 2009. The sampling network covers the whole country except for the most northern part of the country with a small area of cultivated soils.

In 1974, the locations of the sampling plots were shortly documented and marked on the map of scale 1:200 000. In 1987, more detailed maps were drawn and the locations of the sampling plots were tied up with landmarks to improve the precision of positioning the sampling plots in the following observation years. Therefore, the measurements in 1987, 1998, and 2009 should be more consistent in terms of accuracy in locating the plots. In 2009, coordinates of all plots were determined with GPS. In some cases, the measured C concentration shows a relatively high change between consecutive observation years (Fig. 2). Most likely this arises from inaccuracy in locating exactly the same sampling points together with the high spatial variability in soil carbon. Therefore, we prefer to talk about "sampling plots" rather than "sampling points".

The soil samples were taken from the surface layer (top 15 cm) of a square-shaped (10 m × 10 m) sampling plot. Topmost part of the sample with high density of litter and root biomass was removed from the sample. In 1974 and 1987, the samples (about 0.5 dm^3) were taken with thin bladed shovel and consisted of four subsamples collected from each corner of the sampling plot. In 1998 and 2009, the samples were pooled from about 10 subsamples taken with an auger (Ø 2 cm).

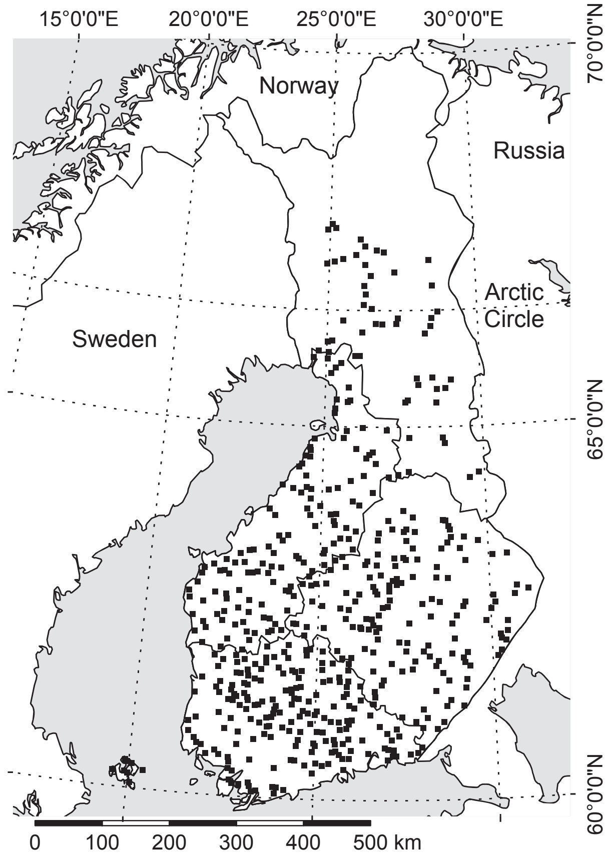

Fig. 1 Arable soil monitoring network of Finland in 2009 and the regional division (north, east, west, and south) used in this study.

The soil samples were air dried and passed through a 2 mm sieve for analysis. Roots and other fresh plant material were removed from the sample. All analyses were performed in the same laboratory (MTT Agrifood Research Finland, Jokioinen) to guarantee the consistency of methods. The C concentrations were determined by dry combustion method (Matejovic, 1993) using LECO CN-2000 analyzer (LECO, St Joseph, MI, USA) except in 1974 when wet oxidation method was used (Graham, 1948). The wet oxidation method has been reported to give on average 6–7% lower C concentrations for mineral soils and 12% lower for organic soils than dry combustion (Sippola, 1982). As Finnish agricultural soils are generally acidic, the samples were assumed to contain only organic carbon. In this study, "C concentration" means the amount of carbon relative to the soil mass (g kg^{-1}), whereas "C stock" means the total amount of carbon per unit area (kg m^{-2}). "C content" is used as a general term.

Soil bulk density and texture were determined in 2009. Samples were taken from the 0–15 cm layer using either 5 cm (n = 409, V = 294 cm^3) or 3.5 cm (n = 109, V = 144 cm^3) diameter soil core samplers. The samplers covered the whole sampling depth. Each sample was pooled from three subsamples. Samples were oven dried in 105°C overnight and weighted. Bulk densities were corrected to represent the soil fraction <2 mm in diameter by removing the weight of gravel

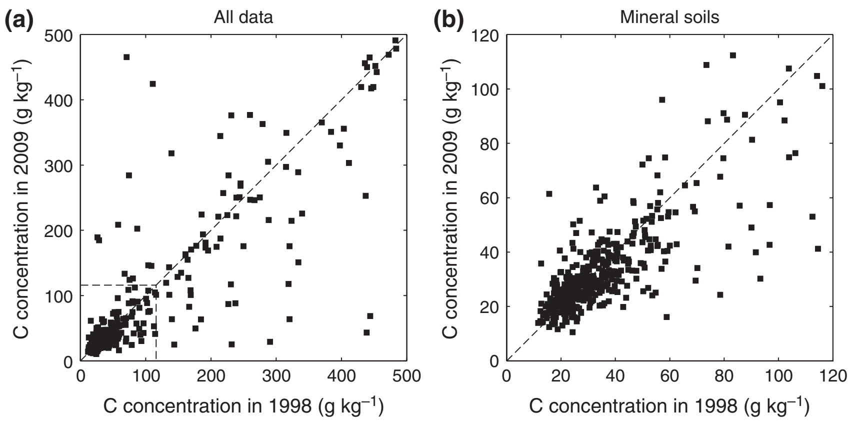

Fig. 2 Relation between measured C concentrations (g kg^{-1}) in 1998 and 2009 (a). Mineral soils are presented in separate graph (b) to clarify the range of OM < 20% (confined by the broken line in graph showing the whole dataset). The number of observation pairs is 522.

size fraction. Bulk densities were used to estimate the soil C stocks [C stock = C concentration × Bulk density × Depth (15 cm)]. The thinner soil corer (3.5 cm) tended to give systematically slightly lower bulk densities than the other sampler. As the number of samples taken with the thinner corer was low and mainly taken in one region only (east), the data obtained with thinner corer were omitted from the dataset. In addition, the bulk density measurements were missing in eight sampling plots. Soil texture was determined using the sieve-pipette method (Elonen, 1971).

Observations with missing C concentration or coordinates were removed from the dataset as well as abandoned and afforested sites. Furthermore, observations showing exceptionally high changes in C concentration between observation years (more than 50 g kg^{-1} in mineral soils and 100 g kg^{-1} in organic soils) compared with the scatter in the data (Fig. 2) were removed if the field notes indicated e.g., difficulties in locating the plots, changes in soil class or unusual management or environmental conditions (e.g., flooding or clearance of field from stones). After the removals, the number of plots was 1865 in 1974, 1315 in 1987, 641 in 1998, and 526 in 2009. The high number of rejected observations in 1974 arises mostly from inaccuracy in determining the location of the sampling plots, whereas abandonment and afforestation were the main reasons for removals in subsequent observation years.

The nationwide C stock estimate was based on combining data from the soil monitoring data and geospatial datasets. The soil monitoring data were exploited to determine the mean C stocks in the groups of soil class, management, and region (see "Data classification"). The areas of Finnish croplands in each group were extracted from geospatial datasets [Finnish soil database (Lilja *et al*., 2006), Land Parcel Identification System (EU, 1992)] using MATLAB (Version 7.9.0, The MathWorks) and ArcGIS (Version 9.3.1, ESRI Inc.) softwares.

Data classification

For statistical analyses, the data were grouped based on the region, soil class, and management. The regional division shown in Fig. 1 follows roughly the spatial distribution of soilscapes in the Finnish soil database (Lilja *et al*., 2006). In addition, the four regions (North, East, West, and South) were divided into 25, 78, 81, and 117 administrative localities, respectively (not shown). The number of sampling plots per locality ranged from 1 to 48 during the observation years.

Following the Finnish classification system, the soil class was divided into three groups of mineral soils (OM < 20%) and two groups of organic soils. Mineral soils with dominant particle size below 0.06 mm and clay content less than 30% were classified as "Fine" whereas soils with dominant particle size over 0.06 mm were classified as "Coarse". "Clay" refers to soils with more than 30% of clay regardless of particle size distribution. Organic soils were classified in soils with organic matter content (OM) between 20% and 40% (20% < OM < 40%) and highly organic soils with organic matter content more than 40% (OM > 40%). Organic matter content was derived from C concentration using the Van Bemmelen factor of 1.724.

Classification of cultivation history was based on the records of cultivated species from 1995 to 2009 [Land Parcel Identification System (EU, 1992)]. Fields growing annual or perennial crops more than 80% of the time were classified as 'Annual cropland' or 'Perennial cropland', respectively. The ones which did not fall in these classes were grouped in the class of 'Crop rotation' representing the most typical cultivation scheme in Finland.

The classification above was used to determine the soil C stocks and to examine the changes in C concentration between 1998 and 2009 in groups of management, soil class, and region. As cultivation history of the sampling sites is known from 1995 onwards and as soil class was determined by the sieve-pipette method only in the latest observation year in 2009, trends in C concentration in 1974–2009 were analyzed using simplified classification in which cultivation history was omitted and soil class was divided into mineral soils and two groups of organic soils only.

Statistical analyses

The data of the mineral and organic soils were analyzed separately. The statistical analyses of temporal changes in mean C concentration were based on the years 1987, 1998, and 2009 when C concentrations were determined with the dry combustion technique. The measurements determined by wet oxidation in 1974 were used to estimate the mean C concentrations and the 95% confidence intervals for the means of that year. The data were nonnormally distributed even after log transformation. This can be the case for response variables whose values have a lower and upper bound (Smithson & Verkuilen, 2006). The statistical analyses were based on generalized linear mixed effects models with the logit link function and the C concentration (values between 0 and 1) assumed to have a beta distribution. The use of beta distribution for modeling proportions is proposed e.g., by Gbur *et al.* (2012).

The model for the C concentrations of the mineral soils in 1974 had region as a fixed effect and locality within region as a random effect which was assumed to be normally distributed with zero mean and unequal variance for each region. The initial model for the longitudinal data in the years 1987–2009 had region, year and their interaction as fixed effects. The random locality effect was assumed to be normally distributed with zero mean and constant variance. The C concentrations measured in the same sampling plots in two or three observation years tended to be intercorrelated which was taken into account with the covariance structure of the repeated measurements. The initial model included the most complex covariance structure in which both variances and pairwise covariances of observations at various measuring times were unequal. This unstructured covariance model was estimated for each region separately and appeared most appropriate when it was compared with simpler structures by likelihood ratio tests (SAS Institute Inc. 2009). The data did not support equality of the covariance structures across regions or homogeneity of the variances or equality of the pairwise covariances within the covariance structures of the regions. Furthermore, no clear autoregressive pattern (Schabenberger & Pierce, 2002) was shown in the covariance estimates. After finding the "best" covariance model, the mean C concentrations for the 12 region-by-year combinations and the 95% confidence intervals for the means were estimated on the basis of the model. Furthermore, the mean part (the fixed effects part) of the model was refined by treating year as a continuous variable and modeling trends over years using first-degree polynomials (i.e., straight lines). The mean part was thus of the following form: $\text{logit}(\mu_i) = a_i + b_i\,\text{YEAR}_j$, where μ_i is the mean C concentration, a_i is the intercept, b_i is the slope for region i, YEAR_j is the jth year during the period 1987–2009 and $\text{logit}(\mu_i) = \ln[\mu_i/(1-\mu_i)]$. The mean C concentration for each region as a function of year can be solved from the above equation as follows: $\mu_i = [1 + \exp(-a_i - b_i\,\text{YEAR}_j)]^{-1}$. The dependency of the intercept and slope on region was tested through Wald F-test.

The C concentrations of the organic soils were analyzed similarly with a few exceptions. The model for the year 1974 had soil class and soil class-by-region interaction as additional fixed effects besides region. The variation in C concentrations between localities within the regions was allowed to be unequal for the two organic soil classes (20% < OM < 40%, OM > 40%). The initial model for the data of the years 1987–2009 had soil class as an additional fixed effect and also the interactions of soil class with region, year, and region-by-year. The unstructured covariance model appeared most appropriate when estimated for the two soil classes separately.

The topsoil C stock was analyzed as a log-normal distributed variable according to mixed effects models. In estimating the nationwide C stock in 2009 for organic soils, the fixed effects were soil class, region, and soil class-by-region interaction and locality was the random effect. For mineral soils, the effects of management and region-soil class grouping and their interaction were the fixed effects. The temporal change in mean C stock in the years 1987–2009 was estimated for mineral soils using a model which had the same fixed and random effects and the same covariance structure than the final model for the C concentrations. Because bulk densities were measured only in 2009 they were estimated for all years by a pedotransfer function (PTF) presented in the results section.

The models were fitted using maximum likelihood estimation based on adaptive Gaussian quadrature (Gbur *et al.*, 2012) excluding the models for the longitudinal data in the years 1987–2009 which were fitted using the pseudo likelihood estimation method (Wolfinger & O'Connell, 1993). The significances of the model terms were determined using Wald F-tests with the denominator degrees of freedom calculated using the containment method (SAS Institute Inc. 2009). The statistical analyses were performed by the GLIMMIX procedure in version 9.2 of the SAS/STAT software (SAS Institute Inc. 2009). The adequacy of the models was checked graphically. To identify atypical and influential observations, Pearson-type residuals (SAS Institute Inc. 2009) were plotted against the indices of the observations. The removal of large-scale and small-scale spatial dependencies among the observations by the effects of region and locality within region was checked through empirical semivariograms of the residuals (Schabenberger & Pierce, 2002). According to the diagnostic plots, the models fitted the data adequately.

Results

Change in C concentration in 1974–2009

In mineral soils, there was evidence of a decreasing trend in the mean C concentration over the years 1987, 1998, and 2009 when C concentration was determined by dry-combustion technique (Fig. 3). The trends for the regions were approximately linear on the logit scale and could be modeled by regression lines. The slopes of the regression lines tended to depend on region ($F_{3,1782} = 2.75$, $P = 0.04$ for region-by-year interaction) because the decreasing trend was less obvious in the north. However, the small number of observations in the north in the last two observation years causes uncertainty to the results as shown by the wide confidence intervals for the means. Therefore, regression lines with common slope for the regions were considered adequate models for the trends on the logit scale (slope = –0.0044, standard error = SE = 0.0008,

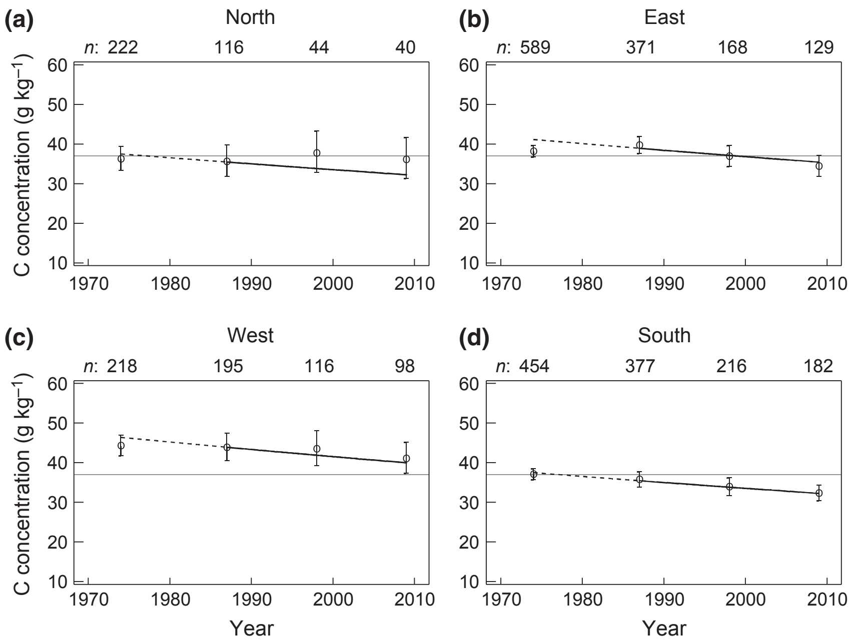

Fig. 3 Development of C concentration in mineral soils in the north (a), east (b), west (c) and south (d). Estimated mean C concentrations with 95% confidence intervals (CI) for the means and modeled trends over years. Wider CI indicates lesser precision (reliability), and narrower CI indicates greater precision in the estimated means. The trend models were based on the data of the years 1987, 1998, and 2009 (dry combustion method, solid line) and were used to "predict" the means of the year 1974 (wet oxidation method, broken line). The number of sampling plots in each region-by-year combination is denoted by n. The horizontal reference lines denote the overall mean 37 g kg^{-1} of the years 1987–2009.

$P < 0.0001$ for the hypothesis H_0: slope = 0). The trends are slightly nonlinear on the original scale (g kg^{-1}) indicating that the C concentration decreased at a mean rate of 0.4% yr^{-1} relative to the existing C concentration in all regions (Fig. 3).

In the south and north, the estimated intercepts of the parallel regression lines were approximately equal, indicating that the same regression line was adequate to describe the trends in both regions. However, the pairwise comparisons of the remaining three intercepts (west, east, and common for the south and north) were all statistically significant ($P < 0.01$), i.e., the vertical distances of the regression lines for the three region groups differed. The mean C concentration was highest in west and lowest in the south and north (Fig. 3).

The trends in mean C concentration over the years 1987–2009 for both groups of organic soils (20% < OM < 40% and OM > 40%) could be described by declining regression lines on the logit scale. Furthermore, the slopes of the lines did not depend on soil class ($F_{1,287} = 2.04$, $P = 0.15$ for soil class-by-year interaction) or on region ($F_{3,284} = 1.05$, $P = 0.37$ for region-by-year interaction) or on both factors ($F_{3,278} = 1.86$, $P = 0.14$ for the three factor interaction). Consequently, a common slope model was sufficient for the trends on the logit scale (slope = –0.0032, SE = 0.0013, $P = 0.01$ for the hypothesis H_0: slope = 0). On the original scale, the relative decrease in C concentration is 0.3% yr^{-1} and 0.2% yr^{-1} in the two groups of organic soils 20% < OM < 40% and OM > 40%, respectively (Fig. 4). The modeled trends are presented separately for all eight groups despite the high uncertainty in the groups where the number of observations is small and the confidence intervals for the estimated means are wide. According to the estimated means, it seems that especially in the soil groups of 20% < OM < 40% in the north and OM > 40% in the south there is no evidence of declining trend (Fig. 4). However, more data are needed to draw reliable conclusions whether the lack of trend is real or just due to random variation.

The differences in the intercepts of the parallel regression lines between the four regions did not depend significantly on soil class ($F_{3,281} = 1.14$, $P = 0.33$ for soil class-by-region interaction). For both groups of organic soils, the mean C concentration was highest in the north, second highest in the west and lowest in the south and east. When all pairwise differences in the distances of the parallel regression lines were examined between the regions, the differences between north and other regions and between west and south were statistically significant ($P < 0.04$, Fig. 4).

In 1974, the C concentration was determined by the wet oxidation method which has been shown to

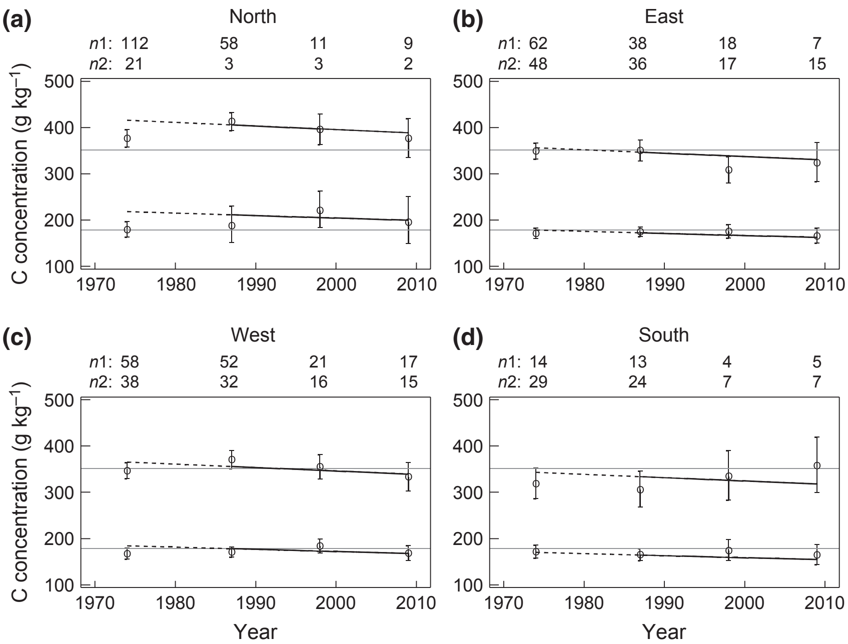

Fig. 4 Development of C concentration in organic soils in the north (a), east (b), west (c) and south (d). Estimated mean C concentrations with 95% confidence intervals for the means and modeled trends over years. Upper lines represent soils with OM > 40% ($n1$ = the number of sampling plots) and lower lines soils with 20% < OM < 40% ($n2$ = the number of sampling plots). The trend models were based on the data of the years 1987, 1998, and 2009 (dry combustion method, solid line) and were used to "predict" the means of the year 1974 (wet oxidation method, broken line). The horizontal reference lines denote the overall means 178 and 351 g kg^{-1} of the two soil groups in the years 1987–2009.

produce lower C concentrations than dry combustion (see "Materials and methods"). Taking the different methods and the number of observations into account, the means estimated on the basis of measurements in 1974 were consistent with the higher concentrations "predicted" by the trend models for the year 1974 (Figs 3 and 4). This suggests that the decline in C concentration has been close to linear throughout the monitoring period in 1974–2009.

Topsoil C stock and its change

The nationwide C stock in the topsoil (0–15 cm) was estimated to be 161 Tg, of which 117 Tg is in mineral soils. The estimates were determined by dividing the monitoring data in groups of soil class, management, and region and by combining the estimated mean C stocks of the groups with the area of Finnish croplands in each group (Figs 5 and 6, areas summarized in Table 1). In mineral soils, the mean C stocks varied between 41 and 67 Mg ha^{-1} depending on the management, soil class, and region (Fig. 5). In the groups of organic soils, the range of mean C stocks was between 128 and 202 Mg ha^{-1} (Fig. 6).

In mineral soils, the change in C stock can be estimated on the basis of relative change in C concentration and nationwide C stock. By assuming that the relative decrease in C stock corresponds to the decrease in topsoil C concentration (0.4% yr^{-1}), the annual C losses from the cropland mineral soils (0–15 cm) are on average 220 kg ha^{-1} yr^{-1}.

The soil C loss in mineral soils was also estimated by determining the topsoil C stocks (0–15 cm) of the sampling plots and modeling the C stocks in 1987–2009 similar to C concentrations. The C stocks of the sampling plots in all observation years were calculated as C concentration × bulk density × depth (15 cm), where bulk density = 1.52−0.280 × ln [C concentration (%)]. The PTF for bulk density was determined on the basis of the bulk densities of mineral soils (n = 345) measured in 2009. In mineral soils, the estimate for a common slope of the regression line describing the temporal trend in mean C stock over the years 1987–2009 in the four regions was −0.0031 (SE = 0.0005) on the log scale. This indicates that the C stock decreased at a mean rate of 0.3% yr^{-1} relative to the existing C stock. The rate of decrease corresponds to soil C loss of 170–200 kg ha^{-1} yr^{-1} depending on the region and year.

Relation of management, soil class, and region to change in C concentration in 1998–2009 in mineral soils

The starting C concentration in 1998 differed among the groups of management, soil class, and region and

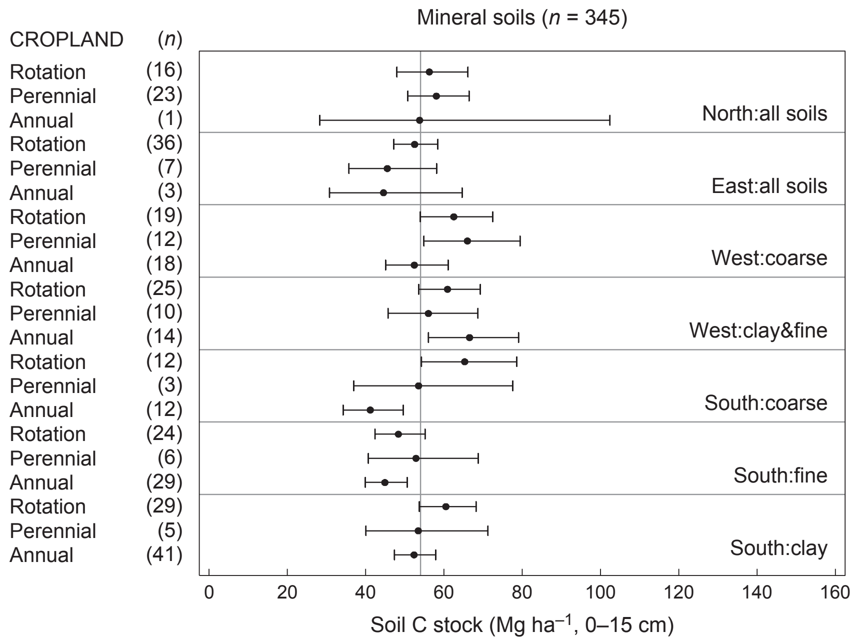

Fig. 5 Mineral soils in 2009. Mean soil C stocks with 95% confidence intervals for the means in the groups of soil class, region, and management. The vertical reference line denotes the overall mean 54 Mg ha^{-1}. Soil classes in north, east, and west were combined to get sufficient number of observations in each group of region and management. C stock has been calculated as C concentration × bulk density × depth (15 cm). The number of sampling plots is denoted by *n*.

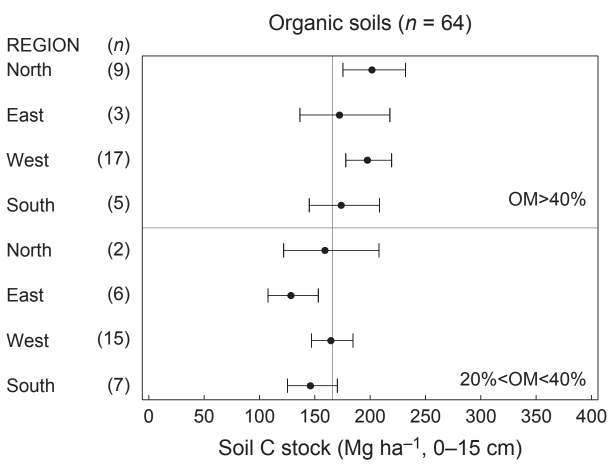

Fig. 6 Organic soils in 2009. Mean soil C stocks with 95% confidence intervals for the means in the groups of soil class and region. The vertical reference line denotes the overall mean 166 Mg ha^{-1}. C stock has been calculated as C concentration × bulk density × depth (15 cm). The number of sampling plots is denoted by *n*.

the managements before the year 1995 were unknown. Therefore, the log-transformed C concentration of the year 1998 and its interactions with management and region-soil class grouping were included as covariates in a generalized linear mixed model which had the C concentration of the year 2009 as a beta-distributed response variable, the logit link function and locality within region as a random effect. This enabled to examine how the groups compare in terms of mean C concentration in 2009, after controlling for the starting C concentration in 1998.

The covariate-adjusted means in Fig. 7 show the differences in mean C concentration in 2009 among the management practices, the soil classes, and the regions, had the starting C concentration in 1998 been equal in the groups (i.e., 38 g kg^{-1}, the mean C concentration of the year 1998). Among the management practices, the means of the fields growing predominantly annual crops were lowest in 2009 in all groups of region and soil class. Furthermore, when a decrease from the level of 1998 occurred, it was highest among the annual crops (Fig. 7). The marginal mean of annual cropland, i.e., the average of the means of the seven region-soil class groups, differed statistically significantly ($P < 0.03$) from the marginal means of perennial cropland and crop rotation, whereas the difference between the latter two means was minor. However, the mean C concentration decreased from the level of 1998 also for perennial cropland and crop rotation excluding clay soils in the south and all soils in the west. Within the soil classes in the south, the means of clay soils were highest in 2009 for all management practices. Furthermore, there was no decrease in mean C concentration from the level of 1998 in clay soils. The marginal mean

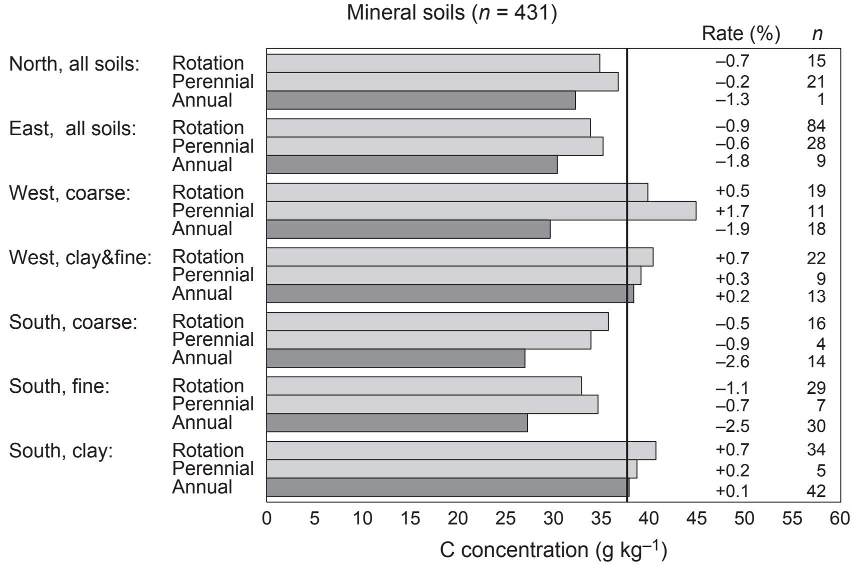

Fig. 7 Mean C concentrations in 2009 for the groups of management, soil class, and region after adjusting for the starting C concentration in 1998. The covariate-adjusted means were calculated at the mean C concentration of the year 1998 (C_{1998} = 38 g kg^{-1}, vertical reference line). The relative mean rates of change (% yr^{-1}) were calculated as (100 × (adjusted mean-38)/38)/11, and n is the number of sampling plot pairs in 1998 and 2009 which had mineral soil as soil class in both years. The number of clay soils was 0 in the north, 3 in the west, and 19 in the east. Of the latter clay soils 15 were in the class of crop rotation.

of clay soils in the south (averaged across the management groups), differed statistically significant ($P < 0.03$) from the marginal means of fine and coarse soils, but the difference between the means of fine and coarse soils was not discernible.

Discussion

According to the results, the C concentration has decreased in both mineral- and organic soils across the country. The relative decrease rate (0.2–0.4% yr^{-1}) is comparable with some earlier studies in the northern part of Europe. In Norway, the mean relative decline rate was around 1% yr^{-1} in cultivated soils (Riley & Bakkegard, 2006) whereas in England and Wales the decline was about 0.6% yr^{-1} on average in all land-use types (Bellamy *et al*., 2005). Slightly decreasing, although statistically nondiscernible trend in topsoil C concentration was also found in Danish agricultural soils (Heidmann *et al*., 2002). As soil organic matter is directly proportional to C content, topsoil C concentration is an important indicator of soil quality. However, from the perspective of atmospheric CO_2 and the carbon cycle, it is the change in C stock (the amount of carbon per unit area) which matters. We discuss first the representativity of our findings i.e., whether the decreasing trend can be generalized across all cultivated land in Finland, then the significance of the results for C stock estimation and finally the possible factors explaining the observed trends.

Representativity of the results

Since the monitoring network has gradually thinned out from the original 2042 sampling plots in 1974 to 611 in 2009, there is the possibility that the selection procedure of the sampling network could have systematically favoured either higher or lower C concentrations. However, as the selection procedure did not involve the C concentration as a criterion, and as regional representativity of the dataset was ensured over the monitoring, we found it appropriate to analyze the data using all retained observations from a particular sampling year to estimate the yearly means as accurately as possible. The analyses were also done using only those plots which were included in the network in 2009 and the slopes of the regression lines describing the temporal trends (on the logit scale) did not differ essentially from those obtained using the whole dataset (the slopes were −0.0044 (SE = 0.0008) and −0.0025 (SE = 0.0013) for the mineral and organic soils, respectively).

It is generally thought that the soils growing perennial grasses can store more carbon than those with annual crops due to the lesser soil disturbance. As the network was originally established on fields growing timothy, the trends of C concentration might be affected compared with a sample network which would have been originally formed randomly in relation to arable use. However, although presently perennial croplands are slightly overrepresented in the dataset, proportionally the number of sampling plots in different management classes corresponds closely to current use of Finnish croplands (Table 1). This indicates that the monitoring network was initially established in typical cultivated land and therefore the sampling plots of our study can be considered to represent the Finnish croplands fairly well. In the past, the farming in Finland was notably different from today. Until the 1960s–1970s, farms were smaller in size, cultivation included diverse cropping system and field crop cultivation was usually accompanied by animal husbandry (Markkola, 2004). Ley rotation, including timothy as the most common grass species, was an integral part of the cropping system. Specialization of farms and monoculture are the phenomena of recent decades.

The sampling network deviates more from all cultivated land with respect to proportional distribution of soil classes than with respect to management (Table 1). The inconsistency arises most probably from the inaccuracy of the geospatial dataset. There are no exact field-level datasets available on soil classes in Finland and Table 1 is based on the soil map at scale 1:250 000. Therefore, the areas of large landforms are likely overestimated whereas the areas of small-scale landforms are underestimated. Determining the areas of organic soils is especially problematic as pointed out by Turunen (2008).

Trends in C concentration and indicative changes in C stocks

Limitations of soil inventories to reliably assess changes in regional scale, C stocks are widely recognized (Smith *et al.*, 2007; Goidts *et al.*, 2009; Schrumpf *et al.*, 2011). Usually, like in our case, long-term inventory networks were originally established to monitor soil productivity rather than to assess soil carbon. Therefore, large uncertainties are often introduced by the sampling procedures used (Ellert *et al.*, 2007; Schrumpf *et al.*, 2011). Inventories are commonly based on fixed-depth sampling rather than the method of equivalent soil mass as recommended by e.g., Lee *et al.* (2009). In addition, high spatial variability in soil carbon (Röver & Kaiser, 1999; VandenBygaart & Kay, 2004; Zhang *et al.*, 2011) demands using composite samples, marking the sampling sites and high sampling density (Goidts *et al.*, 2009). As annual changes in soil C stock are small relative to the large amount of carbon present in the soil, uncertainties in the sampling and analysis procedures can jeopardize the attempts to detect small changes in soil C stock or result in erroneous interpretations. The observed decrease in C concentration does not necessarily mean a change in C stock and soil C loss of similar magnitude because of possible deepening of the plough layer and compaction of soil. This is especially true for mineral soils where topsoil C stock represents a considerable part of the total C stock in the soil.

In Finnish agriculture, tractors and ploughs got more massive especially in the period from 1960s to 1980s (Markkola, 2004) which might have caused increased mixing of the subsoil with surface soil. Erviö (1995) estimated that the depth of the plough layer increased on average 1–2 cm in Finnish experimental sites during the period of 1960–1991. As the C concentration is much lower in the subsoil than in the surface, even a slight increase in the depth of the plough layer notably dilutes the C concentration of the topsoil. Furthermore, the assumption behind using repeated measurements of topsoil C concentration to examine the temporal trend in soil C stock is that the surface soil of arable land is vertically homogeneous due to the repeated mixing of soil. However, the C content tends to decrease with depth even within the topsoil layer especially in no-tillage cropping system (Angers *et al.*, 1997; Blanco-Canqui & Lal, 2008; Karhu *et al.*, 2011). Therefore, possible soil compaction due to heavier machinery results in the sampling of relatively deeper soil layers and as a consequence in lower C concentrations. The inconsistency between C concentration and C stock thus arises from the fixed-depth sampling scheme, and could be partly avoided using the method of equivalent soil mass in temporal comparisons (Ellert & Bettany, 1995) and by sampling well below the depth of plough layer.

Despite the uncertainties, our results strengthen the evidence of C losses in cropland soils noted in several other studies in Europe (Janssens *et al.*, 2003; Bellamy *et al.*, 2005). Major changes in agricultural practices and machinery took place from the 1960s to 1980s resulting in intensification of farming (Markkola, 2004) while more recently the management has developed toward conservation farming such as reduced or no-tillage systems. If the observed decline in C concentration arose from nothing but the deepening of plough layer and soil compaction, we would expect the rate of decrease to level off toward the present time. This is not the case as shown in Fig. 3; the decrease in C concentration has instead been continuous (except in the north where the number of sampling plots was lowest) and close to linear from the beginning of the monitoring to present. This indicates that the decrease in mean C concentration is mainly associated with true loss of carbon from the soil (i.e., C stock change).

In our study, the change in C stock based on the PTF is about 25% lower than when estimated using C concentrations. In the approach based on PTF, the bulk density is calculated as a function of C concentration: the lower the C concentration, the higher the bulk density tends to be. Therefore, the resulting bulk densities in different years do not necessarily represent the same soil layer expressed as soil mass. With the observed decrease in C concentration in 1974–2009, the soil bulk density and thus the mass of the 15 cm soil layer gradually increases. As a consequence, the soil C loss is underestimated. With respect to climatic impact, it is critical that the temporal trends in C stock are based on the equivalent soil masses (Ellert & Bettany, 1995; see also Supporting Information). In fixed-depth sampling scheme such as ours Lee *et al.* (2009) recommended the use of C concentrations to describe trends in C stock.

Nationwide C stock of 117 Tg (0–15 cm) in mineral soils corresponds to an average C stock of 53 Mg ha^{-1} which is comparable with Swedish agricultural soils (90 Mg ha^{-1}, 0–25 cm) as presented by Andrén *et al.* (2008). The C stock below plough layer is poorly known due to lack of extensive datasets, but based on the soil profile studies by Yli-Halla *et al.* (2000) total C stock in mineral soils is at least two to three times the C stock in the topsoil. For comparison, total C stock in Finnish forested mineral soils and peatland are estimated to be 921–1315 Tg (Kauppi *et al.*, 1997; Liski & Westman, 1997; Ilvesniemi *et al.*, 2002) and 5304 Tg (Turunen, 2008), respectively. As the area of forested mineral soils in Finland is about seven times greater than the area of cultivated land, the croplands store on average considerably more carbon per surface area than forested soils. High soil C stock arises from the fact that cultivated lands have been established on the most fertile sites. Average soil C loss (220 kg ha^{-1} yr^{-1}) in mineral soils corresponds to nationwide C losses of 0.5 Tg which is about 2.5% of the total greenhouse gas emissions of Finland.

Similar to mineral soils, C concentration declined in organic soils but at the lower relative rate of 0.2% and 0.3% yr^{-1} for the soil groups OM > 40% and 20% < OM < 40%. Consequently, the relative decrease rate in C concentration was inversely proportional to the original concentration contradictory to findings by Bellamy *et al.* (2005). This is reasonable when considering the properties of the highly organic peat soils (OM > 40%) with thick layers of practically pure organic matter. In these conditions, even a drastic loss in carbon is not necessarily reflected to topsoil C concentration. As carbon in topsoil 15 cm represents a minor part of total C stock in organic soils and C loss is likely to result in subsidence of organic layer, any attempts to convert topsoil C concentrations into stock changes are highly uncertain. Although we were not able to define the exact range for C loss from organic soils, our study is in agreement with gas flux measurements reviewed by Maljanen *et al.* (2010) showing that cultivated organic soils are a significant source of CO_2. Compared with mineral soils (220 kg C ha^{-1} yr^{-1}), the emissions from organic soils are high being 3200–5900 kg C ha^{-1} yr^{-1} on average (Maljanen *et al.*, 2007).

Reasons for the decline in C stock

The observed soil C loss in mineral soils could be linked with (i) relatively late establishment of Finnish cultivated land, (ii) change in agricultural practices and (iii) climate change.

Forest soils store more carbon than cultivated soils and clearing the forest for agriculture leads to soil C losses (Guo & Gifford, 2002; Karhu *et al.*, 2011). Although the greatest change in soil carbon occurs over the first 20 years after deforestation (Poeplau *et al.*, 2011) soils also contain large amount of stabilized or highly decomposed organic matter with a turnover rate ranging from decades to millennia (Jenkinson & Rayner, 1977; Trumbore, 1997). As a considerable proportion of the cultivated area in Finland has been cleared as late as in the 20th century, cultivated soils might not have reached a new carbon content equilibrium after deforestation. The slow decline in soil carbon observed in this study could therefore be the result of the decomposition of the soil carbon pools with long residence time.

Intensification of agricultural practices has contributed to widespread soil degradation in Europe in the recent decades (Stoate *et al.*, 2001). A shift from perennial grasslands and crop rotation to cultivation of annual crops has led to increased soil exposure to wind and water erosion and as a consequence loss of the carbon rich topsoil. Increased tillage and adoption of cropping system using little or no organic manure are claimed to deteriorate the soil structure and the processes forming carbon stabilizing soil aggregates (West & Post, 2002; Mikha & Rice, 2004). More efficient removal of plant residues from fields due to improved machinery and due to introduction of varieties with higher harvest index (the ratio of harvestable product to the total aboveground weight of the crop) might have decreased the carbon return to the soil (Smith *et al.*, 2007). Changes in management practices in Finland have generally been similar to those elsewhere in Europe. In Finland, about 0.5 million ha of perennial croplands were abandoned or converted to annual crops over the time period from 1970 to 2010 (Tike, 2012). At the same time, the number of cattle decreased from 1.8 million to 0.9 million (Tike, 2010) accompanied by the change in manure management toward the increased use of liquid slurry at the expense of solid manure. Furthermore, the area of subsurface drainage increased by about twofold being close to 60% of agricultural land area in 2010 (Salaojayhdistys, 2011). Although it is not clear that all above-mentioned aspects, such as change in manure management or increased subsurface drainage, lead to soil C losses, they demonstrate the huge change in agricultural practices in the last few decades.

The annual mean temperature has increased by about 0.7 °C in Finland in 1901–2000 (Jylhä *et al.*, 2004). No regional scale trend in precipitation has been observed in Finland over the same time period although increased precipitation has been reported elsewhere in northern latitudes (Räisänen & Alexandersson, 2003). In agricultural land, the soil C stock reflects the balance between inputs from plant residues and organic amendments and losses due to decomposition and erosion which all are to some extent climate-dependent processes. While climate warming most probably increases the decomposition of soil organic matter in humid boreal zone, it also enhances the growth of the biomass and as a consequence carbon return to the soil. The overall effect depends on whether the enhanced growth is large enough to counteract the increased decomposition. In agricultural land, the carbon return is also greatly attributed to the plant breeding and the selection of plant species and cultivars (Olesen *et al.*, 2011).

According to the covariance analysis of the years 1998–2009 the shift toward cultivation of annual crops in last decades can be considered to contribute to C losses noted in this study (Fig. 7). The finding is in agreement for instance with studies by Van Wesemael *et al.* (2010) and Lal (2004). Cultivation of annual crops seems to decrease the soil carbon more on texturally coarser soils than on fine-grained soils (Fig. 7) reflecting the influence of silt-and clay-protected carbon and aggregation on the stabilization of soil organic matter (Six *et al.*, 2002). Changes in management practices cannot, however, explain the nationwide decline alone. Interestingly, the relative soil C loss rate is similar in all regions (although less obvious in the north) (Fig. 3) despite the fact that regions differ considerably with relation to management practices (Table 1). This indicates that at least to some extent soil C loss also results from the processes affecting cultivated soils nationwide such as climatic change or the continuing long-term effect of deforestation. Bellamy *et al.* (2005) postulated the climate change to be the driving force behind the soil C losses in England and Wales. As soils consist of carbon pools of which decomposition rates range of several orders of magnitude and as carbon is continuously entering and leaving the soil, the contribution of different factors to soil C stock loss could be revealed best using process-based models (e.g., Yasso07 (Tuomi *et al.*, 2009) or RothC [Coleman & Jenkinson, 1999)] as was done in the studies by Smith *et al.* (2007), Van Wesemael *et al.* (2010) or Gervois *et al.* (2008).

On the basis of the existing data, we were able to affirm the declining trend of topsoil carbon in Finnish arable lands. However, question remains where the lost carbon resides, i.e., to what extent it has accumulated in terrestrial and aquatic habitats through erosion–deposition processes (e.g., Mattsson *et al.*, 2005; Van Oost *et al.*, 2007) or emitted to the atmosphere as CO_2 as a result of decomposition. The observed trend in C content poses a challenge to the development of cultivation methods that maintain the soil quality and ensure proper conditions for food production in the future. The results stress the importance of soil monitoring networks as a tool to keep track on changes in soil properties in agricultural land. However, by improving the precision of locating the sampling sites, by increasing the area of the sampling plots and the plot-level sampling density, and by using equivalent soil mass method, the national soil monitoring could be further developed to better serve the needs of carbon inventories.

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Supporting Information

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

Fixed-depth sampling: Whether to use soil c concentration or pedotransfer function to describe trends in c stocks?