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Soil organic carbon storage following conversion from cropland to grassland on sites differing in soil drainage and erosion history

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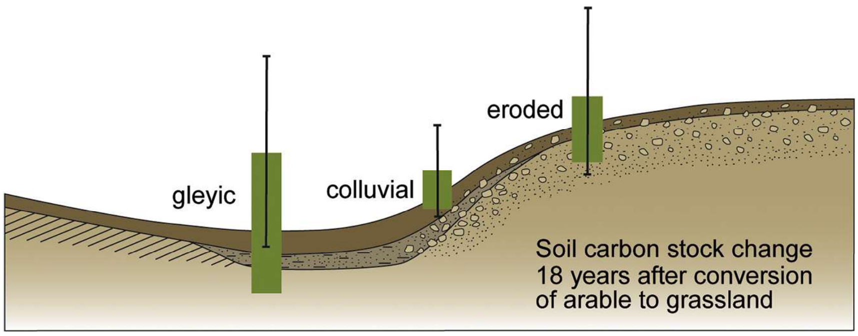
HIGHLIGHTS

- Our long-term study covered 20 yr of cropland use followed by 18 yr of grassland use.
- Only poorly drained soils sequestered significant amounts of SOC after converting cropland to grassland.
- No significant dynamic replacement of SOC was found under strongly eroded soils.
- Regulating soil drainage is most important to increase or decrease SOC stocks.

GRAPHICAL ABSTRACT



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Land use change
Cropland
Grassland
Soil drainage

ABSTRACT

Changing soil use from cropland to grassland influences organic carbon storage in a highly complex way. This includes the root/shoot allocation, the root depth distribution, the incorporation of shoot biomass and lateral organic carbon fluxes, by erosion and removal of harvested carbon, and finally the aeration by tillage. An experiment was designed allowing resampling a number of soils 18 yr after conversion to grassland (either pasture or meadow or set-aside) only 20 cm apart from the original sampling to exclude site variation. Before conversion to grassland the cropland was prone to erosion, with a mean lateral carbon flux during 20 yr prior to conversion of 13 t ha^{-1}. Harvest had removed another 29 t ha^{-1} of carbon at eroding sites. Colluvial carbon inputs had been up to 18 t ha^{-1} while harvest had removed 38 t ha^{-1} at colluvial sites. The carbon fluxes by erosion were negligible during the 18 yr period after conversion. After conversion the carbon losses by harvest also ceased at set-aside grassland and pastures while the net losses on meadows were 45 t ha^{-1}. Conversion to grassland significantly changed depth functions of carbon, stones, bulk density and porosity. Despite the large changes in carbon fluxes, carbon stocks did only change significantly within 18 yr under poorly drained, gleyic soils. Well-aerated soils did not show a significant increase in SOC stocks. This was even true for heavily eroded soils, where conversion from cropland to grassland (without erosion) should foster dynamic replacement of SOC. The widespread drainage of wet grassland soils prior to conversion to cropland thus can cause a large release of carbon, while an influence of tillage by either increasing aeration or erosion could not be detected in this study. Therefore, fostering carbon sequestration by conversion of cropland to grassland requires restoring former draining conditions.

Abbreviation: SOC, soil organic carbon.

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1. Introduction

Appropriate changes in land use and adapted soil management are widely accepted as option to sequester atmospheric carbon and thus contribute to the mitigation of fossil fuel burning on atmospheric CO_2 concentrations (IPCC, 2014). Given especially the higher topsoil SOC contents of grassland compared to cropland (Don et al., 2009; Freibauer et al., 2004; Guo and Gifford, 2002; Post and Kwon, 2000), a conversion of cropland to grassland is widely reported to have a large potential to act as sink for atmospheric CO_2 (Allard et al., 2007; A.F.G. Jacobs et al., 2007; C.M.J. Jacobs et al., 2007; Soussana et al., 2007). However, studies analysing entire soil profiles often indicate that the difference in SOC stocks between cropland and grassland are much smaller than estimated from topsoil data alone (Don et al., 2009; Guo and Gifford, 2002) and some studies even indicate that there is no significant accumulation of SOC after conversion (Gosling et al., 2017; Leifeld et al., 2011). Apart from the soil depth related differences in SOC stocks between cropland and grassland there are a number of methodological issues not fully addressed in all studies, e.g. different soil masses in case of different bulk density under different land use is ignored (see equivalent soil mass approach (e.g., Ellert and Bettany, 1995)) or the dependency of land use, soil type and SOC stocks (Wiesmeier et al., 2015) is not taken into account. Also the large-scale experience with the USA Conservation Reserve Program that lead to a widespread establishment of grassland for which often an increase in SOC was reported (for a bibliography see Allen and Vandever, 2012) may not be applicable everywhere because of the methodological issues mentioned above and because it is likely that especially cropland had been converted to grassland where prior declines in SOC were most likely due to poor cropland management (Waisanen and Bliss, 2002). Moreover, the first use of grassland or the conversion from grassland to cropland may have required other changes, especially soil drainage, which are not restored, if moving back from cropland to grassland.

The difference in SOC storage of cropland and grassland soils can either be quantified by comparison of different sites (e.g. Wiesmeier et al., 2012), or by carbon flux studies immediately after conversion (e.g. Allard et al., 2007; A.F.G. Jacobs et al., 2007; C.M.J. Jacobs et al., 2007; Leifeld et al., 2011), or by long-term studies, which compare SOC stocks before and after land-use change (e.g. Don et al., 2009; Leifeld et al., 2011). Attributing a difference in SOC to land use change in temporal comparison does not need assumptions on similarity of sites or extrapolation from short-term flux measurements and is hence a valuable complementation of other studies. One difficulty of temporal comparisons is that not only the initial and final SOC stocks have to be known but also whether other factors than conversion from cropland to grassland have changed. The most important are: (i) a change in drainage conditions, which is often associated with a first conversion from grassland to cropland, (ii) a change in lateral carbon export by harvest or import by organic manure (Johnston et al., 2009), and (iii) the redistribution and export of SOC via erosion processes (Dlugoß et al., 2012; Doetterl et al., 2016; Gregorich et al., 1998; Lal, 2003). A second difficulty of temporal comparisons, which also applies to spatial comparisons, is the fact that a difference has to be calculated. Differences are always especially prone to error because they combine the errors of the first and the second sampling. Hence, both samplings have to be done with great care and identical methodology, but also the local variability of soils has to be considered. This can be accounted for by analysing a large number of soils for both land uses like in many spatial comparisons but this is usually not possible in the case of temporal comparisons. A high precision of the location and resampling in close vicinity is then necessary to minimize the effect of short-range soil variability.

The aims of this study are (i) to determine the changes in SOC stocks 18 yr after converting arable fields to different grassland types using high accuracy resampling in relation to carbon fluxes before and after land conversion, (ii) to analyse the differences under different grassland types (meadows, pastures, set-aside) and in landscape positions differing in erosion history and drainage (eroded hillslopes, colluviated foot slopes and poorly drained valley bottoms). We hypothesize that conversion of cropland to grassland increases SOC stocks and that the increase in SOC stocks will be larger on formerly severely eroded sites, which have lost more carbon than other arable sites.

2. Material and methods

2.1. Land use

The study was conducted at the Scheyern Research Farm (Fig. 1) in southern Germany with a mean annual precipitation of 833 mm and a mean annual temperature of 7.4 °C. The sampling sites had been under cropland use for at least 20 yr prior to the first sampling. For most of the sites, cropland use is even documented since several centuries by the monastery, which owned the area. The crops during the last 20 yr before conversion were almost entirely small-grain cash crops with crop residues left in place. No organic fertilizers were applied due to lack of livestock keeping on the farm. After harvest in 1992 the land use of the farm was changed to better fulfil ecological demands such as protection from erosion and soil compaction, protection of wildlife and to improve economic returns (Auerswald et al., 2000; Auerswald and Filser, 2001). During reorganization, some former cropland sites were converted into grassland.

The decision for conversion from cropland to grassland was based on the following agronomic considerations (for details see Auerswald et al., 2000): (i) All areas that were too steep to lower soil erosion below the soil loss tolerance, even if a soil conserving crop management would have been established. (ii) All areas in toe slope and valley position where problems with high soil moisture during main periods of crop management (sowing, harvest) could be expected to happen in some years were converted. (iii) In general, the layout of the arable fields at the research farm was improved by creating fields with parallel sides at multiples of the machinery width (15 m). Areas taken out of cropland use due to this optimisation were either added to adjoining grassland or were set-aside. The new grassland thus covered a wide range of geomorphological properties from flat to very steep areas. In addition, it covered a wide range of soil properties from shallow, eroded soils to deep colluvial soils and rather poorly drained, gleyic soils in the valley bottoms (Table 1).

Soils had a high fertility: Plant available water capacity always was >60 mm (mean: 133 mm). Nutrient supply and pH (Table 1) with few exceptions were within the ranges recommended as ideal by local agricultural authorities, which are 0.08–0.15 g kg^{-1} for potassium, 0.04–0.08 g kg^{-1} for phosphorous and 5.6–5.9 for pH (Wendland et al., 2018). Under these conditions, no fertilization with potassium and phosphorus is recommended for optimum growth as long as no export with harvested goods occurs (Wendland et al., 2018), as it was the case on set-aside areas and on pastures.

Land use after conversion differed between the sampling positions depending on the best agronomical use. It covered about the full range of possible grassland uses. Some sites were converted to meadow, some to pasture and some to set-aside land (Fig. 1). The new grassland did not receive any mineral fertilizer because the long-term surplus application of potassium and phosphorous fertilizers had built up sufficiently high nutrient stocks in the soils (Auerswald et al., 2001a). The pastures were rotationally stocked for beef production with animals usually shifted to neighbouring pastures three times per year (for typical stocking sequence and stocking density see Schnyder et al., 2010). The mean stocking rate was 0.65 livestock units per hectare and year (Auerswald et al., 2010). The meadows were usually cut two times per year. In some years meadows were cut a third time with clippings left in place because harvest would have been uneconomical. The set-aside areas were mowed to avoid encroachment of woody species. This was done about every eight years with the clippings left in place.

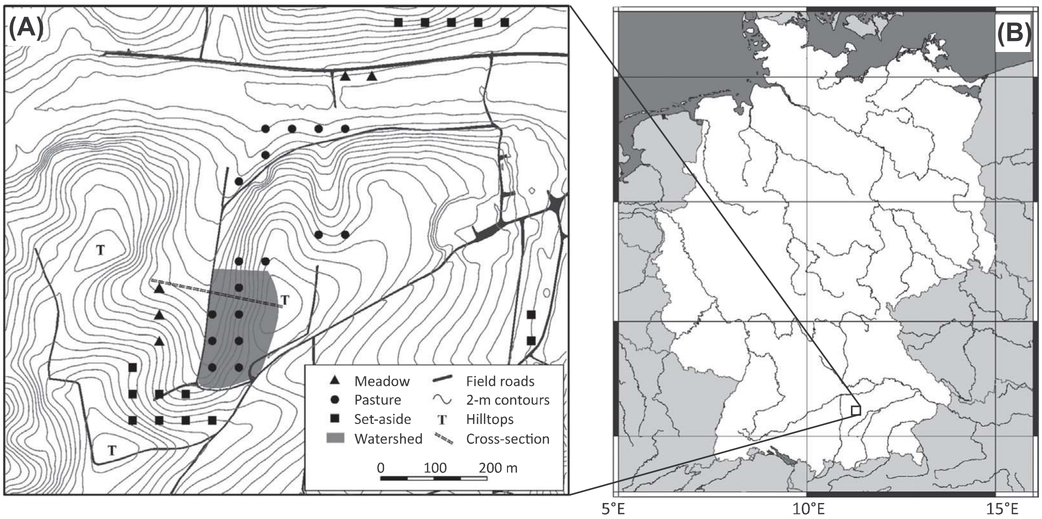

Fig. 1. (A) Position of 37 soil sampling locations, where grassland had been established after cropland use in 1992, position of the catena shown in Fig. 5 (thick dashed line) and location of the gauged watershed (grey); the quantification of erosion from $^{239+240}$Pu losses was carried out for the locations within this watershed and the two locations adjoining to the north. (B) Situation of Scheyern Research Farm within Germany (lines are major rivers).

The results of this study are hence representative for a wide range of grassland use and site conditions in Central Europe.

2.2. Sampling protocol

Before reorganization of land use, an intensive soil sampling in a 50 m × 50 m grid was carried out in 1990 and 1991 (Auerswald et al., 2001a; Sinowski et al., 1997). To level out any differences caused by the individual crops between the adjoining fields, the initial subsoil sampling was done after one year of fall wheat (*Triticum aestivum* L.) cultivation with the management operations done on all fields within one week. The sampling of the plough horizon, where a stronger crop influence could be expected, was done in the following year (1991) after another year of identical crop management (spring barley, *Hordeum vulgare* L., in this case). The soil sampling took place after the grain harvest before tillage to have a well-settled plough horizon and to facilitate depth determination. The resampling was carried out in 2010, which is 20 yr after the first sampling and 18 yr after land conversion. Both numbers will be used in the following depending on whether we refer to the sampling interval or the duration of grassland use.

Table 1
Initial soil properties (n = 37); all data apply for the original plough horizon before conversion to grassland except for the plant available water capacity (PAWcap), which was determined to the effective rooting depth (see Sinowski et al., 1997); clay is <2 μm, silt is 2–63 μm, sand is 63–2000 μm. pH was measured in 0.01 M $CaCl_2$. Calcium-acetate-lactate-extractable phosphorus P_{CAL} and potassium K_{CAL} are regarded plant available fractions (according to Schüller, 1969). SOC is soil organic carbon content and N_t total nitrogen content.

Property	Minimum	Mean	Maximum
Slope (%)	2.9	13.3	24
PAWcap (mm)	62	133	205
Plough horizon thickness (cm)	17	25.5	33
Bulk density (kg L^{-1})	1.14	1.41	1.64
Clay[a] (kg kg^{-1})	0.07	0.14	0.27
Silt[a] (kg kg^{-1})	0.19	0.34	0.58
Sand[a] (kg kg^{-1})	0.21	0.51	0.72
Stones[b] (kg kg^{-1})	0.01	0.16	0.57
pH[a]	5.1	6.0	7.3
K_{CAL}[a] (g kg^{-1})	0.10	0.20	0.32
P_{CAL}[a] (g kg^{-1})	0.02	0.07	0.14
SOC[a] (g kg^{-1})	10.4	16.9	27.4
N_t[a] (g kg^{-1})	0.95	1.71	2.96

[a] In the fine earth fraction.
[b] In the bulk soil.

The sampling positions of the initial inventory were determined within a 50 m × 50 m grid by a geodetical survey with an accuracy of <5 cm for the x, y and z coordinate (for details and accuracy see Warren et al., 2004). The resampling positions were again determined by a geodetical survey. They were shifted 20 cm to the north of the first sampling positions (independent of slope orientation).

The sampling was done by machine augering (Ø 11 cm; inner diameter slightly smaller than the cutting ring). The first sampling was carried out to a depth of 100–120 cm while the resampling only covered 0–40 cm depth. This was based on an assessment of possible changes in soil properties, the known variability of the first sampling and the statistical confidence that could be obtained with the given number of sampling positions. The auger voids of the first sampling were refilled with a mixture of washed sand and bentonite to allow easy recognition of the weathered soil and the unweathered filling and thus to avoid resampling of the disturbed area in case of any errors in the determination of the re-sampling position. Furthermore, the bentonite content of the mixture was adjusted to yield a permeability somewhat smaller than that of the original soil. This avoided an artificial drainage in the surroundings of the sampling positions. The plough horizon was refilled with material from the surrounding plough horizon assuming that tillage had mixed the plough horizon anyhow.

The depth increments of the first sampling were: (i) the plough horizon (between 17 and 33 cm; mean 25.5 cm), (ii) the first 10 cm underneath the plough horizon, which were sampled separately because the largest changes of the subsoil were expected to happen there, (iii) all following genetic subsoil horizons. Genetic horizons larger than 40 cm were split in two, even if they appeared homogeneous. This resulted in five to six depth increments. The second sampling deviated from this scheme because the plough horizon was not existent anymore and we expected gradual changes with depth becoming smaller with increasing depth. Hence, we sampled in constant depths increments of 0–5 cm, 5–10 cm, 10–20 cm, 20–30 cm and 30–40 cm. Soils from each increment were homogenized before laboratory analysis; analytical results represent the mean of the respective increment.

2.3. Erosion rates before conversion to grassland

SOC fluxes by erosion before and after conversion to grassland were determined for those sites that had the largest losses by erosion or the largest gains by colluviation according to field evidence and model calculations (watershed in Fig. 1). For the retrospective quantification of the erosion rates usually the change in ^{137}Cs (half-live time 30.2 yr) originating from nuclear bomb testing in the 1960s is applied (Walling and He, 1999). This method was not applicable at the Scheyern site because this region had received a second and about ten times higher contamination with ^{137}Cs by the Chernobyl nuclear power plant accident, which was distributed inhomogeneously because it mainly was deposited during one thunderstorm event and because hot particles that arrived by dry deposition due to the proximity to the source cause a potentially large variation even within one site. Also the short-lived ^{134}Cs (half-live time 2.06 yr) from the Chernobyl accident did not satisfy to separate between bomb and power plant derived ^{137}Cs because the error of separation had already become too large at the time of conversion due to the decay of ^{134}Cs causing a large scatter between individual locations. Therefore we used $^{239+240}$Pu (half-live time 24,110 yr and 6564 yr) as 99% of this nuclide in Bavarian soil results from nuclear bomb testing whereas the amount of $^{239+240}$Pu deposited by the Chernobyl fallout is negligible (Schimmack et al., 2001). It is generally important to note that radionuclides originating from bomb testing are deposited more or less homogeneous during series of rainfall events over years. The $^{239+240}$Pu was measured by alpha spectrometry subsequent to an extensive radiochemical separation and purification. We will report the changes in $^{239+240}$Pu inventory together with the changes in nuclear bomb derived ^{137}Cs separated from Chernobyl derived ^{137}Cs by ^{134}Cs to show that the averages of both methods agree despite the large scatter of the $^{134/137}$Cs method. The sites of erosion measurement had been converted from grassland to cropland 20 yr before $^{239+240}$Pu was determined. The changes of $^{239+240}$Pu relative to uneroded sites could thus be converted to rates of soil loss and gain applying the methods by Walling and He (1999). The data of soil losses or gains have already been reported by Schimmack et al. (2002) and were taken from this source. More details about the measurement and comparison with ^{134}Cs are given by Schimmack et al. (2001). Schimmack et al. (2002) used a finer grid to resolve better the erosion catena but we will use only the data from the 50 m × 50 m grid points for which also all other data are available. The soil losses and gains were then converted to SOC losses and gains by multiplication of soil mass with the SOC content at the respective location. We assume that the potential error in our calculation due to preferential water erosion of SOC and radiotracers (Quinton et al., 2006; Weigand et al., 1998) is small as about half of the soil was transported via unselective tillage erosion (Schimmack et al., 2002). Moreover, enrichment by water erosion was potentially low due to the extraordinary high losses (Auerswald, 1989; Weigand et al., 1998) associated with rill erosion which is, compared to interrill erosion, hardly selective (Kuhn et al., 2010).

2.4. Erosion rates after conversion to grassland

The nuclear fallout method would potentially fail to quantify erosion under grassland due to the low erosion rates that do not exceed the error of measurement. Hence after conversion, erosion was measured directly and continuously in a small watershed (2.7 ha), which covered the locations of the $^{239+240}$Pu measurements (Fig. 1). At the outlet of this watershed, which had only pasture and some field border structures at its fringe, the runoff was concentrated, diverted (0.5%) by a Coshocton-type runoff sampler and the aliquot was collected in a large underground tank, which was emptied after each event. Before emptying, the suspension was homogenized with a submersible pump and subsamples were taken for determination of the runoff constituents. The same measuring technology was applied for 15 neighbour watersheds on the same research farm that remained under cropland use. Details on the Coshocton-type samplers, the analytical protocol and accuracy have already been given for these watersheds (e.g. Fiener and Auerswald, 2003, 2009). The measurements of erosion were stopped after eight years in 2001 because the amounts were negligible without large variation. We assume that the average rates of this 8-yr measuring period sufficiently represent the annual rates during the entire period after conversion to grassland.

2.5. Assessment of carbon fluxes due to agricultural operations before and after conversion to grassland

The management of the fall wheat and spring barley cultivation during the soil inventory (1990–1991) prior to conversion to grassland aimed to maintain a similar (moderate) intensity of agricultural management as had been applied during previous two decades. The yields determined during these two years will be assumed similar to the yields in previous years. They were determined with two approaches. At the soil sampling locations, an area of 12 m^2 was harvested with a plot harvester. For the remaining field between these locations the yield was determined with a GPS-assisted online yield determination (for details see Auerswald et al., 2001a). The yields were corrected for moisture content and converted to carbon losses with a constant factor of 0.47, which is the conversion factor recommended by the IPCC for herbaceous biomass from grassland and cropland (Verchot et al., 1996). Straw residues were not removed, as the farm had no winter shedding of animals prior to conversion. Here we will report only the data for those locations where also erosion had been quantified by $^{239+240}$Pu but the yields at other locations and between the locations of the 50 m × 50 m grid were similar. For the reported locations, the crop sequence was known for 13 years prior to conversion and we assumed that this is representative for the full 20 years since conversion from grassland, as no change in farm management had occurred during that time. For the calculation of the average carbon removal by harvest we assumed that the fall wheat yield measured in 1990 also applied in all other years with fall grain (5 yr of fall wheat and 4 yr of fall barley), while the spring barley yield measured in 1991 was assumed for the two years with spring crops (barley, oats *Avena sativa* L.) and the two years of canola (*Brassica napus* L.).

Net carbon removal with harvested goods after conversion was assumed to be zero for set-aside sites and close to zero for pastures where only meat growth during the stocking period caused a 'lateral' loss. For the meadows, the carbon removal by forage production (silage and hay) and the carbon gains by application of organic manures were determined by recording the amounts, measuring the moisture contents and converting the total masses to carbon masses with again a constant factor of 0.47.

Respiratory losses were not determined, irrespective of whether they were caused by plant respiration or respiration by soil organisms or by grazing animals.

2.6. Terminology

Following accepted recommendations (Tolhurst et al., 2005) we use 'content' to denote the ratio between the mass of a component and the total mass of the air-dry soil (SOC content, stone content; in kg kg^{-1}) while 'concentration' denotes the mass of a component per volume of soil (SOC concentration, stone concentration; in kg L^{-1}; the concentrations of solids and pores is called bulk density and porosity, respectively). The concentrations multiplied by depth are termed 'stocks' although in principle they are still concentrations (either spatial concentrations if expressed as kg m^{-2} or volume concentrations averaged over a larger depth if expressed as kg m^{-3}). For terminology related to grassland types (set-aside land, pasture, and meadow), we follow Allen et al. (2011).

2.7. Analytical protocol

For each horizon, disturbed and undisturbed samples were taken to determine stone content, SOC content and bulk density. The two uppermost horizons (plough horizon and the first 10 cm below the plough horizon in case of cropland use; first 10 cm separated in two 5 cm increments in the case of grassland use) were sampled from soil surface. Considering the large contribution of these uppermost horizons to total SOC stock and a presumably larger small-scale variability, soil cores

(100 cm^3) were taken in triplicate from these horizons and about 1 kg of disturbed soil was sampled. Given the fact that most of the stones were small (85% were between 2 and 20 mm; none were > 63 mm), the disturbed sample should also contain a representative proportion of stones. Care was taken not to select for stones/fine earth during sampling. Below the two uppermost horizons, sampling was done from the auger content. Usually, this allowed only for one core per horizon. The complete auger content except for material of the undisturbed soil core was taken as disturbed sample. The undisturbed cores were dried (105 °C, several days) and weighed to determine bulk density. The disturbed samples were broken in small pieces by hand to improve air-drying and separation of stones and fine earth. This was done with the field-moist samples and repeated during the drying process for those soils that were sticky and wet. In addition, coarse organic material (mainly large roots) was removed during this drying process. The air dry soils were then passed through a set of sieves by hand sieving to separate four stone fractions (2–6.3 mm, 6.3–20 mm, 20–63 mm, >63 mm) and the fine-earth fraction (<2 mm). A subsample of the fine-earth fraction was then ball-milled and carbon and nitrogen contents were determined by dry combustion with elemental analysers (CN-analyser 1500, Carlo-Erba, Milan, for the first sampling; NA 1110, Carlo Erba, for the second sampling). Accuracy of calibration was assured by running a solid internal lab standard (fine ground wheat flour) as a blind control after every tenth sample. The precision that was thereby achieved for soil samples that were repeatedly measured at different times was 0.04% (RMSE). All carbon was assumed to be organic as the soils contained no carbonates as verified by repeating the analysis after heating the samples (500 °C, 5 h) to differentiate carbonate carbon from SOC (Rabenhorst, 1988).

The stones (almost entirely quartz; well rounded) for half of the soil profiles of the first sampling were washed to separate adhering fine earth and the SOC content in the adhering fraction was determined as well. To estimate the amount of SOC that could not be removed by washing, the ^{137}Cs activity of the washed stones was determined by gamma spectroscopy assuming that the same proportion of ^{137}Cs and SOC would have moved into the stones. On average, the material obtained by washing of stones added 9% and material within the stones contributed another 1% of the total SOC content (Auerswald and Schimmack, 2000). This procedure was not repeated for the second sampling but we assume that these proportions remained unchanged. Further, we will not use these proportions in the calculation of stocks for an easy comparison with stocks from other sources. Nevertheless, total SOC stocks (as averaged over all soils and all horizons down to 1 m depth) would be larger by about 10% if the material adhering to and in the stones would be considered as well.

2.8. Calculations and statistical analysis

In cases where bulk density varies between treatments or dates, depth functions may just be different because of differences in pore volume. To exclude the influence of a varying pore volume and also the influence of a varying SOC content an equivalent mineral mass approach (following Ellert and Bettany (1995)) was applied where the depth coordinate is replaced by a material coordinate (MacBratney and Minasny, 2010). Accumulated mineral masses at 40 cm depth were not statistically different ($p > 0.05$) between 1990 and 2010 and hence accumulated depths become comparable as well. Also above 40 cm depth, differences between both coordinate systems were small. Thus it is justified to use the depth coordinate instead of the material coordinate. The depth coordinate is important because sampling protocols in all studies are based on depth and not on the material coordinate, which is not known at the time of sampling. We will show that in many studies sampling depth was too shallow and caused a bias between cropland and grassland even if the equivalent mineral mass approach had been used. It is hence important also to show depth.

SOC stocks of each horizon m_C, were calculated from SOC content c_C, stone volume V_S and bulk density in the fine earth fraction ρ_{FE}:

$$m_C = c_C \times \rho_{FE} \times (1 - V_S) \tag{1}$$

ρ_{FE} was calculated from the bulk density of the whole soil horizon, its stone content and a stone density of 2.6 kg L^{-1}.

The sampling depths of the first sampling differed due to the differences in plough depth and the different thicknesses of the genetic horizons, while the depths were fixed and constant for the second sampling.

For averaging the different sites, we assumed that within each sampled depth (either plough horizon, genetic horizon or fixed depth increment) the variation was low enough that the same value was valid over the entire increment. Thus, we were able to calculate averages incrementally, which is the basis for a comparison between the first and the second sampling campaign. Statistical uncertainty was quantified as 95% interval of confidence (denoted by $\pm$), which is given from the standard deviation of all values at a certain depth increment, the number of samples (n = 37 in all cases) and the respective t value ($t_{95\%,37-1} = 2.34$ in all cases). It is important to note that the comparison of means (and their interval of confidence) is only valid between the first and second sampling, while a comparison of different depths within one sampling is not possible because the same value may be used for the calculation of the mean of subsequent layers if the horizon of one soil extended over both layers.

3. Results

The sampled soils occupied a large variety of land use and site conditions (Table 2). There were soils on steep and on flat terrain, eroded sites, colluvial sites and poorly drained valley floor (gleyic) sites. Soil texture ranged from heavy loamy to sandy. The samples and thus the results covered a wide range of grassland.

All parameters of SOC stocks (Eq. (1)) changed as a result of the land use change (Fig. 2a–c). In particular bulk density and SOC (and nitrogen) content changed in opposite directions. A high SOC content was associated with a low bulk density irrespective whether the variation of both reflected the decrease of SOC content with depth within a soil profile at a certain time or whether it reflected the change of a soil horizon over time due to the land use change. Hence SOC content and bulk density followed a rather narrow universal regression, which did not differ between land uses ($c_C = 1.64–0.135\ \rho_{FE}$; $r^2 = 0.69$; n = 229). Due to the negative relation of both parameters, the change of SOC concentration was smaller than the change in SOC content. Furthermore all changes varied in sign with depth (except for nitrogen content), which again reduced the overall effect on SOC stocks for the whole profile. In particular, SOC content increased in the top 10 cm while it decreased below (Fig. 2c), bulk density decreased in the top 5 cm but increased below 10 cm (Fig. 2b) and stone content decreased in the top 5 cm but increased below (Fig. 2a). In general, the changes in the top soil were large but restricted to the top 5 to 10 cm while the changes were smaller below but extended over a wider range of depth. Except for the decrease in SOC content, the changes below 30 cm became too small to be significant.

Total nitrogen (Fig. 2d) deviated in its behaviour slightly from SOC. It also exhibited an increase in content in the top 10 cm but the decrease below was too small to be significant. In consequence, the C/N ratio decreased from around 9.5 to around 8.5 between 1990 and 2010 in all depths below 10 cm depth ($p < 0.001$). The decreasing C/N ratio thus indicated a change in the quality of the organic matter caused by a loss of carbon while nitrogen was retained.

Due to the change of all parameters entering Eq. (1), the change of SOC stocks can only be assessed by considering them all. Furthermore, due to the opposite behaviour of top soil and subsoil the overall effect can only be assessed by integrating over accumulated mineral mass or depth. Accumulated mineral mass down to 40 cm depth was 570 kg m^{-2} (SD 40 kg m^{-2}) in 1990 and 588 kg m^{-2} (SD 41 kg m^{-2})

Table 2
Relative distribution of sampling sites according to land use and site properties (n = 37) and organic carbon stock changes 18 yr after conversion to grassland for an identical mineral mass of 588 kg m^{-2} corresponding to a depth of ca. 40 cm; site properties including the classification as more or less eroded[a], colluvial or gleyic soils were determined during an intensive soil inventory of the entire research farm (Auerswald et al., 2001b; Sinowski and Auerswald, 1999); '±' denotes the 95% interval of confidence in organic carbon stock; significant increases are presented in bold.

Land use		All	Meadows	Pastures	Set-aside	Carbon stock change (t ha^{-1})	Stock change rate (t ha^{-1} yr^{-1})
		1.00	0.14	0.46	0.40	+**5.8** ± **4.3**	+**0.3** ± **0.2**
Gradient	<12°	0.46	0.60	0.53	0.33	+**8.7** ± **6.6**	+**0.4** ± **0.3**
	>12°	0.54	0.40	0.47	0.67	+3.5 ± 6.0	+0.2 ± 0.3
Landscape position	±Eroded[a]	0.50	0.67	0.31	0.67	+5.3 ± 7.1	+0.3 ± 0.4
	Colluvial	0.29	0.33	0.25	0.33	+2.4 ± 4.3	+0.1 ± 0.2
	Gleyic	0.21	0	0.44	0	+**11.4** ± **9.0**	+**0.6** ± **0.4**
Soil texture	Sandy	0.60	1.00	0.82	0.33	+5.5 ± 7.1	+0.3 ± 0.4
	Loamy	0.36	0	0.18	0.60	−0.2 ± 6.5	+0.0 ± 0.3
	Heavy loamy	0.04	0	0	0.07	+3.9	+0.2
Carbon stock change (t ha^{-1})		+**5.8** ± **4.3**	+18.2 ± 22.7	+6.9 ± 7.8	+1.7 ± 4.3		
Stock change rate (t ha^{-1} yr^{-1})		+**0.3** ± **0.2**	+0.9 ± 1.1	+0.3 ± 0.4	+0.1 ± 0.2		

[a] Cropland soils on sloping land were classified as more or less eroded if no colluvium was present because the degree of truncation could not be determined unequivocally for the mostly deep cambic horizons.

in 2010. Accumulated masses are identical when 40 cm in 2010 is compared to 41.5 cm depth in 1990 (Fig. 3a, b). In the topsoil, SOC stocks increased very highly significantly ($p < 0.001$). The largest effect was found when integrating over the upper 10 cm of the soil. With increasing integration depth, the effect decreased again and the scatter increased. Nevertheless, the SOC gain was still significant in 40 cm depth (even at $p = 0.01$) although the lower confidence limit indicated only a minimum increase in SOC stocks of 1.7 t ha^{-1} (0.09 t ha^{-1} yr^{-1}). Considering equivalent mineral masses, the minimum increase in SOC stocks was only 1.5 t ha^{-1} (~1% increase in 20 yr). The relative increase was considerably stronger for the nitrogen, where the increase in the topsoil was not partly balanced by depletion below. Thus the lower confidence limit indicated a minimum increase by 0.5 t ha^{-1} nitrogen (0.03 t ha^{-1} yr^{-1}) down to 40 cm depth.

There was a small SOC gain for all groups of land use, slope gradient, erosion/deposition state, drainage and soil texture (Table 2) but, except

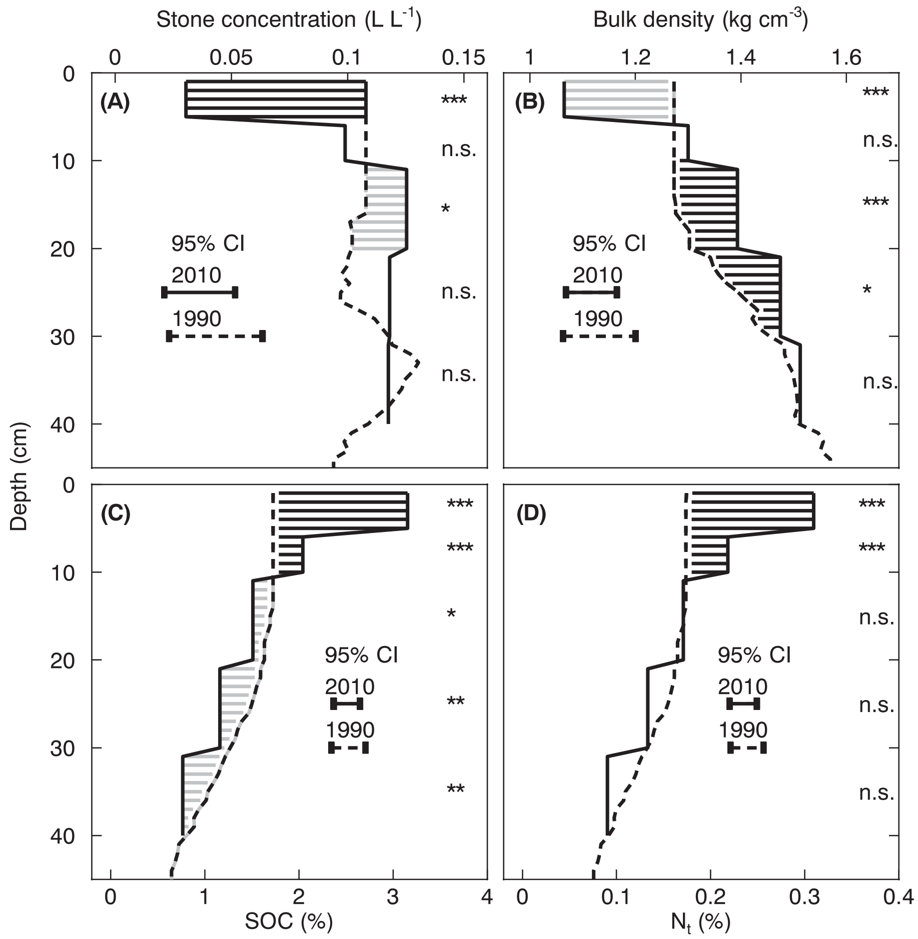

Fig. 2. Depth functions of stone concentration (panel A), bulk density of the fine earth fraction (panel B), organic carbon content (panel C), and total nitrogen content (panel D) from the 1990 sampling (dashed line) and the 2010 sampling (solid line); confidence intervals (CI) given in the legends are the average for all depths; ***, **, * and n.s. denote significant differences between both dates at $p < 0.001$, $p < 0.01$, $p < 0.05$ and $p \geq 0.05$. Dark horizontal lines between both dates indicate significant changes that cause an increase in organic carbon stocks, while grey horizontal lines indicate significant changes that cause a decrease in organic carbon stocks.

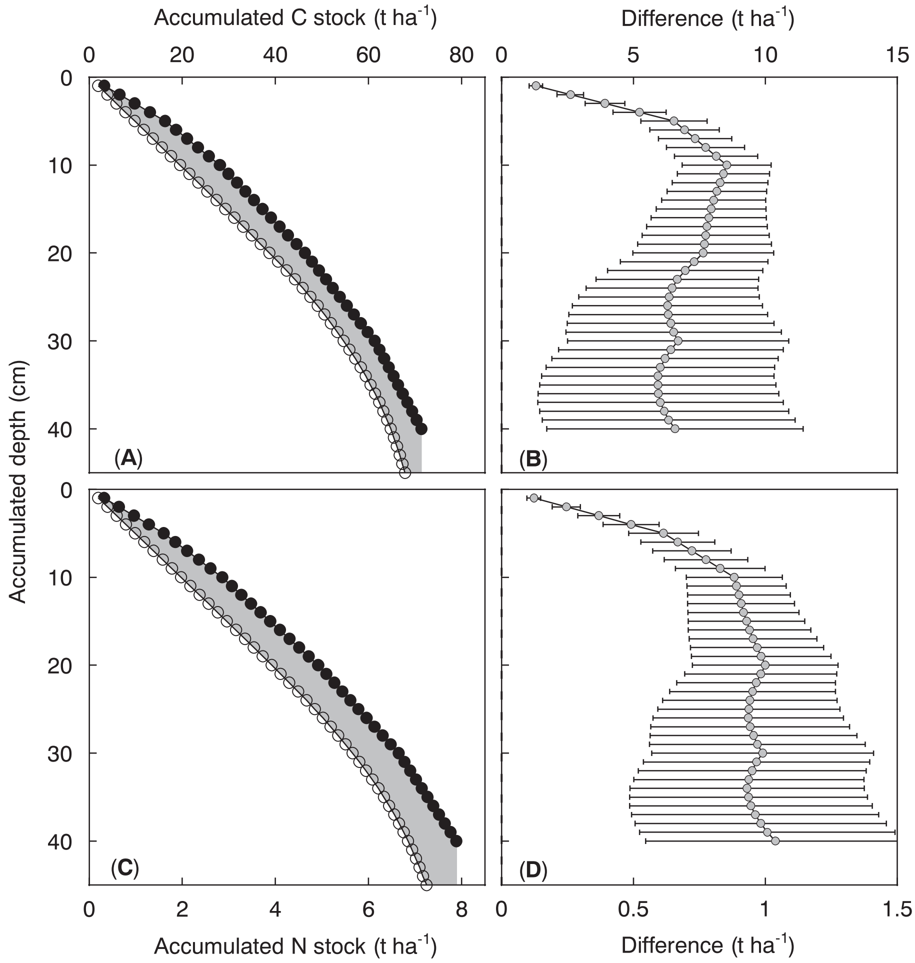

Fig. 3. Accumulated organic carbon (panel A) and nitrogen (panel C) stocks for the 1990 (○) and the 2010 sampling (●); the difference between both (shaded in grey in panels A and C) is shown in (B) and (D) together with the 95% percent intervals of confidence. Mineral masses are equivalent when 40 cm in 2010 is compared with 41.5 cm depth in 1990.

for gleyic soils, this gain was not statistically different from zero with 95% intervals of confidence being typically larger than the means. The gleyic soils and those soil groups, which included the gleyic soils, namely the slope gradient class <12% and the whole data set, exhibited significant changes in SOC stocks. The weakly drained, gleyic soils seem to have additionally sequestered carbon after grassland establishment and this then caused the gain being significant for the other mentioned groups. An influence of texture on carbon gain of the gleyic soils could be excluded because the gleyic soils ranged between the other soils (e.g.: clay content for gleyic soils 12% to 31% and 6% to 40% for all soils, silt content for gleyic soils 21% to 53% and 11% to 55% for all soils, with averages of clay or silt contents being identical for both groups) and texture in general had no influence on carbon stock change (Table 2). The interaction between landscape position and land use (Fig. 4) indicated that no grassland use exhibited advantages despite the large changes in agriculturally influenced carbon fluxes.

Harvest during cropland use had removed on average 2.8 t ha^{-1} yr^{-1} carbon ($\pm$0.1 t ha^{-1} yr^{-1}). This removal remained similar on meadows (gross removal 2.9 $\pm$ 0.6 t ha^{-1} yr^{-1}, returns 0.4 $\pm$ 0.4 t ha^{-1} yr^{-1}, net removal 2.5 $\pm$ 0.7 t ha^{-1} yr^{-1}), whereas carbon removal from pasture (0.02 t ha^{-1} yr^{-1}) and set-aside sites practically was nil. This was not reflected in the change of SOC stocks. The larger average of pasture as indicated in Table 2 is mostly the result of the poorly drained gleyic soils being restricted to this use.

Also the change in erosion was large. The most eroding site had lost 13 t ha^{-1} (0.7 t ha^{-1} yr^{-1}) of SOC during the 20 yr of cropland use prior to conversion back to grassland (Fig. 5), whereas the colluvial sites had gained 18 t ha^{-1} SOC during that time. Taking the differences in carbon harvest at erosional and colluvial sites into account, the total C removal from erosional sites was 39 t ha^{-1} (2.2 t ha^{-1} yr^{-1}) while the removal by harvest still exceeded the gain by colluviation on the colluvial sites yielding a net C removal of 16 t ha^{-1} (0.9 t ha^{-1} yr^{-1}). Erosional and colluvial sites thus differed in the net C removal by 23 t ha^{-1} (1.3 t ha^{-1} yr^{-1}), which is about the difference in SOC stocks found prior to conversion to grassland. With conversion to grassland (pasture

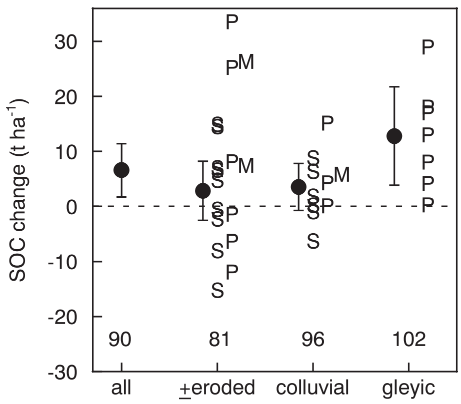

Fig. 4. Change in organic carbon stocks for a soil mass of 588 kg m^{-2} (soil depth of ca. 40 cm) 18 yr after conversion to grassland depending on landscape position and grassland use; S, M and P denote set-aside, meadow and pasture use. Error bars around means (●) give the 95% interval of confidence. Numbers inside the plotting area give the mean organic carbon stocks in 0–100 cm depth before conversion.

	Accumulation	E/A	Erosion
Sampling points	3	2	4
Pu redistribution (% of reference)	79	7	-44
Cs redistribution (% of reference)	75	-6	-34
Soil loss/gain (t ha^{-1} yr^{-1})	56	3	-37
Wheat yield in 1991 (t ha^{-1} yr^{-1})	5.1	4.1	3.9
Erosion C gain/loss (t ha^{-1})	18	-2	-13
Harvested C (t ha^{-1})	-38	-30	-29
C stock 1991 (t ha^{-1})	108	118	89
C stock change (t ha^{-1})	7	-1	7

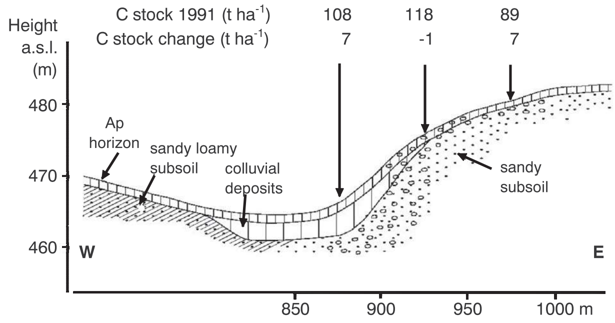

Fig. 5. Cross-section as determined by a soil survey for the most eroded slope where erosion had been quantified with the $^{239+240}$Pu method. Depth of the soil substrates is exaggerated by a factor of five compared to the elevation scale (left axis). The data on carbon fluxes by erosion and harvest and the data on carbon stocks were averaged over 2 to 4 locations spaced 50 m perpendicular to the slope. The top positions experienced large erosion losses while accumulation occurred at the bottom positions. Erosion and accumulation were balanced (E/A) for the middle positions. Harvested carbon and carbon erosion/deposition are total masses for the 20 yr period of cropland use before re-conversion to grassland. These lateral carbon fluxes practically became zero after conversion to grassland. Carbon stock changes (between 1991 and 2010) were not significant.

in the case of the slope shown in Fig. 5), the lateral organic carbon fluxes practically ceased. Although there were still 62 runoff events during the 8 yr measuring period with a mean annual surface runoff of 6.0 mm yr^{-1}, the losses of solids were negligible due to the very low sediment concentration (0.95 ± 0.35 g L^{-1}). Thus, the runoff after conversion caused a total sediment loss (organic + mineral) of 0.06 t ha^{-1} yr^{-1} and a total carbon loss of <0.05 t ha^{-1} during the 18 yr period. Despite this large decrease in lateral SOC losses on erosional and gains on colluvial sites, the total increase in stocks was small and insignificant on this specific slope (Fig. 5) as on all erosional and colluvial sites (Fig. 4).

4. Discussion

The main findings were that the changes in SOC stocks were small and significant changes were restricted to gleyic soils despite the large changes in agricultural carbon fluxes (erosion + harvest) caused by the conversion to grassland. However, this conversion had changed all depth functions of soil properties, which are needed for the calculation of SOC stocks. We will discuss both findings in the following.

The increased carbon sequestration of the gleyic soils after grassland establishment can be expected from the well-known finding that grassland responds stronger to water availability than other land uses. In his seminal book on the influence of soil forming factors (Jenny, 1941) writes "Under conditions of reasonably constant soil forming factors, the nitrogen and organic-matter content of surface soils becomes higher as the moisture increases. The relationship is especially pronounced for grassland soils." The reason why the reaction of grassland towards moisture is stronger than that of cropland or forest land is also well understood. It rests in the phytomer concept allowing the clonal growth of grasses and the short phyllochron (Ryser and Urbas, 2000; Schleip et al., 2013) needed to compensate the frequent defoliation that grasses experience in contrast to both other land uses. This defoliation requires leaf expansion during the growing period to restore the photosynthetic apparatus. There is no other physiological process in plants than leaf expansion, which depends more on water availability (Kramer and Boyer, 1995; Tomer et al., 2005).

Our findings that changes in SOC contents after conversion from cropland to grassland are very much dependent on soil type, and especially gleyic or more general soils with water stagnation develop higher SOC stocks under grassland, are also supported by a soil type specific SOC stock analysis comparing croplands and grassland in Southern Germany (Wiesmeier et al., 2015). In their analysis Wiesmeier et al. (2015) found most pronounced differences between cropland and grassland SOC stocks under soils with water stagnation.

Except of gleyic soils this study finds little evidence for additional C sequestration in grasslands 18 yr after conversion from long-term cropland use. This result differs from findings of others (e.g. Poeplau et al., 2011; Wiesmeier et al., 2012). This is surprising, as two conditions of the experiment should especially favour high C sequestration under grassland. (i) The prior cropland produced cash crops that were sold and led to a carbon removal from the site of production while the grassland use except for the meadows had only small lateral carbon fluxes. (ii) About 50% of the sampling positions had experienced severe erosion during the past (Table 2). Long-term average annual soil losses due to water and tillage erosion had caused soil losses of up to 52 t ha^{-1} yr^{-1}, which is about ten times larger than the average erosion rates by sheet and rill erosion in Germany (Auerswald et al., 2009) and Europe (Cerdan et al., 2010). Translocation by tillage was responsible for about half of the total loss, which explains the large colluviation found at the bottom slopes (Schimmack et al., 2002). Erosional carbon losses had become practically nil after conversion to grassland as already suggested by extensive modelling studies (FAPRI, 2007). Thus, especially SOC stocks at erosional sites (Table 2; Fig. 5) should recover after minimizing erosion, due to processes of dynamic replacement (Doetterl et al., 2016; Harden et al., 1999).

The question then arises, why an effect was found in other studies. The results of this study point to three reasons:

Several studies restrict the analysis of differences in cropland vs. grassland SOC stocks to the upper 10 to 20 cm (Baer et al., 2002; Burke et al., 1995; McLauchlan et al., 2006; Yu et al., 2017) or even to only the top 5 cm (Robles and Burke, 1997) which leads to a bias in land use specific SOC contents towards grassland. This problem was already described in earlier studies which indicate that differences between cropland and grassland SOC stocks decrease with increasing soil depth taken into account (Don et al., 2009; Guo and Gifford, 2002).

There are several methodological issues not fully addressed in all studies evaluating the effect of cropland to grassland conversion. (i) If conversion leads to a change in soil mass in different soil increments to be compared, this needs to be taken into account when calculating SOC stocks (Ellert and Bettany, 1995). In this study the difference in soil mass under cropland and grassland was small (3.8% in upper 40 cm), with a not significant higher soil mass under grassland. (ii) It is important to consider the temporal changes in all soil properties affecting SOC stock. In this study conversion to grassland not only changed the SOC content but practically all soil properties which are needed to calculate the SOC stocks, namely the depth functions of SOC content, bulk density and stone content. In temporal comparisons with a land-use change where not all depth functions are determined with sufficient accuracy but similarity between land uses is assumed, a large bias can occur. This is especially true for bulk density, which is more laborious to determine and which is not available in many data sets. Also stone contents are partly not reported. The close and inverse correlation between SOC content and bulk density compensates some of the change in SOC content. Thus, carbon sequestration under grassland will become falsely superior without proper consideration of bulk density (see also Jiang et al., 2011; Manrique and Jones, 1991).

Assuming identical soil and site conditions in lateral comparisons is unlikely to be true in general because farmers usually consider the production value of their soils in their decisions. Where grassland occurs the probability is thus high that the site conditions do not allow for the economically more favourable cropland use. Even selection of sampling pairs in rather close vicinity does not solve the problem. This is exemplarily shown in our 100 ha study area, where 90% of the total variation in subsoil texture was only found within 25 m distance (Sinowski and Auerswald, 1999). Hence, two sampling points at a distance of only 25 m would not have the same subsoil texture. Very likely grassland will be found in considerably wetter positions than those of cropland sites (Wiesmeier et al., 2012). Higher C stocks under grassland found in lateral comparisons thus likely reflects the influence of drainage but not that of land use. This has major implications as re-establishing grassland would not produce the observed effect as long as this conversion is not accompanied by a simultaneous restoration of drainage. Similar reasoning can be put forward for regional or continental sampling schemes, where climatic conditions likely differ between cropland and grassland sites.

Wiesmeier et al. (2012) already pointed to the fact that the larger SOC storage of their grassland soils resulted from the large proportion of gleyic soils under grassland. Their study covered the same region in which also our research site was situated. Restricting their database (Schmidt et al., 1992) to gleyic valley floor soils, on which both land uses can be found, yields similar SOC stocks for both land uses arranging along the 1:1 line result (Fig. 6). Differences in SOC stocks between typical, dry cropland sites and typical, wet grassland sites are hence caused by different moisture conditions but not by different land use (see also Meersmans et al., 2008 for Belgium). The widespread amelioration of drainage conditions (Hirt et al., 2005) extended cropland at the expense of grassland (Van der Ploeg et al., 1999) and downsized the carbon sequestration potential in soils. Re-establishing grassland to increase carbon sequestration calls for re-establishing formerly wetter moisture conditions.

We may ask why the differences in land use do not produce an effect in stocks although they pronouncedly modify the depth functions. Factors influencing carbon sequestration are (i) (net) carbon fixation, (ii) above-below ground allocation and (iii) soil aeration:

(i) Grassland provides a continuous plant cover, but the difference to cropland is usually restricted to months of low growth potential. In contrast, defoliation during the main growing season is typical for used grassland but not for crops. In consequence, differences in adsorbed solar irradiance and gross primary production are small (Black et al., 2006).

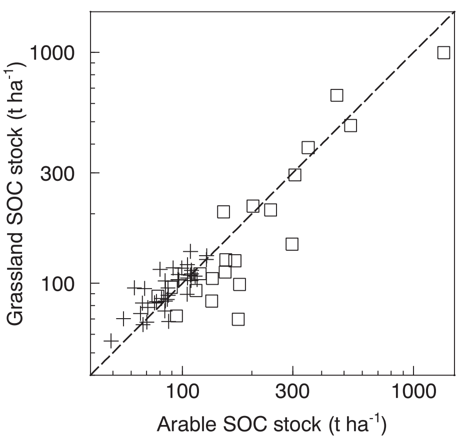

Fig. 6. Lateral (□) and temporal (+) comparisons of organic carbon stocks under cropland and grassland use; data for lateral comparisons are Bavarian valley floor soils taken from Schmidt et al. (1992) (each data point is the average of several soil profiles sampled during regional soil surveys); data for temporal comparisons are from this study. Stocks for the Bavarian soils were determined to 1 m depth. Data of this study were estimated for 1 m depth under cropland use; under grassland use, we assumed that carbon below 40 cm depth had not changed. Both axes are log-scaled.

(ii) The frequent removal of leaves from grassland restricts the amount of carbon available for root growth and initiates new leaf and root formation (Garay et al., 2000; Yang et al., 1998). Hence many roots can be found in the topsoil (70% in the upper 7 cm according to Matthew et al., 2001) while their depth penetration and carbon allocation is limited.

(iii) The main argument, why cropland use could favour carbon respiration, is the soil aeration induced by tillage. Usually the soil resettles quickly and this is additionally favoured by agricultural practices like rolling. Hence, most of the effect disappears within one month as shown by Franzluebbers et al. (1995) with monthly measurements of bulk density during a 3-yr crop rotation (Fig. 7a). Converting bulk density data by Franzluebbers et al. (1995) to porosities shows that their soil returned quickly to a porosity of 50 to 55% before the new tillage started. During cropland use, porosity on our sites (again calculated from bulk density) was 52% (SD 5%) at the respective time, which is practically identical to the data by Franzluebbers et al. (1995). On the other hand, grassland topsoils are characterized by a lower bulk density than cropland soils (Fig. 2b), which corresponds to a larger porosity. The data of Franzluebbers et al. (1995) and the data for the grassland of this study yielded identical ranges (Fig. 7). The only difference was that porosity in the plough layer changed over time under cropland use while it changed over depth under grassland. In consequence, large porosities, which are found under cropland use only during short periods after tillage, are present under grassland all the year in the topmost soil. Grassland soils are thus better aerated than cropland in those parts of the soil where most of the carbon is allocated. The dominant influence of biopores compared to tillage-induced pores is also corroborated by the study of Auerswald et al. (1996) who increased tillage drastically by maintaining seedbed conditions throughout five years on 32 arable soils that had been selected to be representative for all arable soils in southern Germany and thus also included the range of soils of this study. Despite this intensive tillage without any input of organic carbon, the SOC pool had decreased after five years by on average only 7% of the initial values. The decrease of the individual soils correlated with earthworm biomass, which in turn was correlated to biopore area that controlled gas diffusion from below the soil surface.

Vitousek and Howarth (1991) and Luo et al. (2004) proposed that carbon sequestration in soil may be limited by the co-sequestered nitrogen. Hence, carbon sequestration after conversion of cropland to grassland might be limited due to reduced N supply. This was already indicated in earlier studies finding no significant increase of SOC stocks after conversion from cropland to grassland (Gosling et al., 2017). However, at our test site N limitations are unlikely as from wet and gaseous deposition and from biological N fixation we can safely expect that 50 to 100 kg ha^{-1} yr^{-1} N were added to the sites (see results of Schwertl et al. (2005) on a neighbouring farm where also cropland sites had been converted to grassland). For a C/N ratio of 10 and an experimental period of 18 years this would allow for an accumulation of 9 to 18 t ha^{-1} of carbon. Nitrogen limitation can also not explain why C/N ratios became lower but would predict the opposite. Finally, losses of N by nitrate reduction increasing N limitation can especially be expected for the gleyic soils but it were the gleyic soils where we found net carbon sequestration.

5. Conclusion

Overall, we can conclude that conversion of cropland to grassland causes large changes in agricultural carbon fluxes (erosion + harvest) but changes in SOC stocks are small. Within the time frame of the study, eroded sites did not exhibit a significantly larger carbon

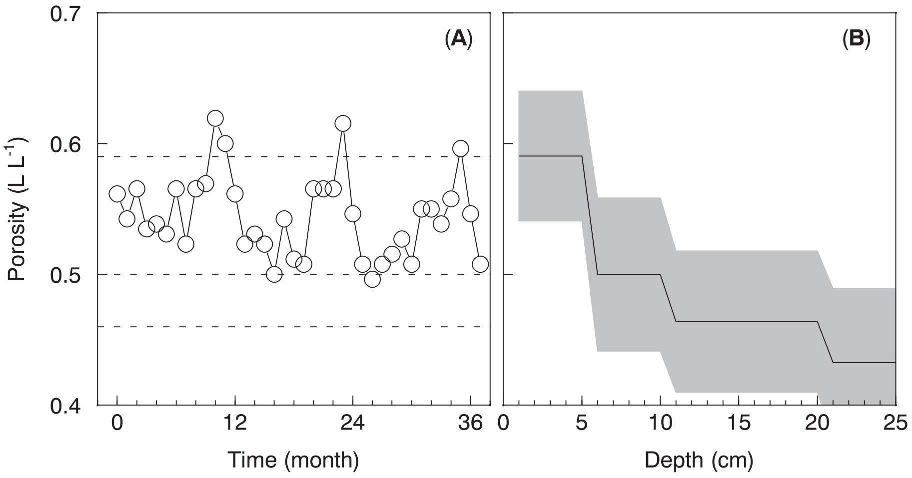

Fig. 7. Comparison of topsoil porosities under cropland (a) and grassland use (b). Data in (a) were calculated from Franzluebbers et al. (1995) and are displayed over a 3-yr crop rotation. Dashed horizontal lines indicate the averages from the soil layers under grassland in (b). Data in (b) are from this study and are displayed over depth. Grey shading is ±1 standard deviation.

sequestration that would compensate the preceding large carbon loss by erosion. Such a conversion, however, changes all depth functions of soil properties, which are needed for the calculation of SOC stocks. Measurement of these properties before and after land use changes is indispensable for quantification of SOC stock changes. In our study significant increases in SOC stocks by conversion were restricted to gleyic soils. Hence, manipulation of soil drainage seems to be an efficient way to either increase or decrease SOC stocks.

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