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Response of boreal clay soil properties and erosion to ten years of no-till management

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ABSTRACT

We compared soil physical, chemical and biological properties, erosion rate and carbon allocation to soil physical fractions between conventional tillage (CT) and no-till (NT) management at a clay soil site under spring cereal monoculture in southwestern Finland. Subsurface drain discharge, surface runoff and soil erosion were continuously monitored in 2008 − 2018. At the end of the 10-year monitoring period in 2018, various soil properties and earthworm total density, mass and species richness were determined. Total soil erosion was 56 % less in NT than in CT although surface water discharge was higher in NT. NT had a clear effect on the topsoil physical structure by decreasing the pore size and increasing soil aggregate size. The total soil carbon stock in the 700 kg m^{-2} mineral topsoil layer (approx. 0−60 cm layer) was slightly lower in NT (108 ± 12 Mg C ha^{-1}) than in CT (118 ± 9.0 Mg C ha^{-1}) due to lower carbon content of the 10−30 cm layer in NT. In NT the proportion of large macroaggregates was higher and more organic carbon was bound to large macroaggregates in the 0−10 cm layer which may be related to the higher abundance of earthworms in NT. The results showed that NT is an effective method to reduce erosion rates but other means to increase carbon input especially below the topsoil layer are likely required to achieve a significant increase in the carbon stock of boreal clay soils. For both tillage managements, the rate of erosion through subsurface drains depended clearly on annual precipitation and winter temperature, posing a challenge in the future climate with mild winters and more extreme discharges.

1. Introduction

Agricultural practices conserving soil and its functions are crucial to secure food production and to diminish its environmental impact (Lal, 2004). In no-till (NT) practice, sowing is performed directly to stubble after preceding harvest without ploughing or other tillage practices typical for conventional tillage (CT), thus minimizing soil disturbance. Compared to conventional tillage, NT practice reduces farm workload and fuel use and has beneficial environmental impacts like effective reduction of erosion risk (Nearing et al., 2017; Skaalsveen et al., 2019) due to improved soil structure and continuous plant cover (Seta et al., 1993; Skaalsveen et al., 2019). NT has been found to improve soil structure and soil organic carbon (SOC) content of the topsoil, water retention, plant available water capacity and soil biodiversity (Du et al., 2013; Blanco-Canqui and Ruis, 2018).

Hydrological conditions in NT deviate from those in CT with implications to observed erosion rates. On average, in most cases NT has been found to decrease surface water runoff (Skaalsveen et al., 2019) but e.g. soil clay content or field slope modify this effect (Sun et al., 2015). NT potentially decreases water discharge through the soil profile as it improves soil water retention e.g. through increased organic matter content (Rawls et al., 2003) and due to decreased pore size (Chan and Govindaraju, 2004), however, subsurface discharge is seldom monitored in field experiments.

NT also potentially enhances sequestration of SOC by reducing erosion and soil disturbances and by improving soil structure, biomass production and aggregation. While NT is known to increase SOC stock in topsoil the limited allocation of crop residue below the surface layer typically reduces the SOC stock in deeper soil layers in comparison to CT which limits the potential of NT in climate change mitigation (Luo et al., 2010; Powlson et al., 2014). In some studies, NT has induced net sequestration of carbon in the whole soil profile (Luo et al., 2010; Haddaway et al., 2017) but it is still unclear which are the drivers enabling this. In arid and semi-arid regions, increase in the available

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water capacity and biomass production compared to CT may explain the increased carbon sequestration (Bonfil et al., 1999; Huang et al., 2008). However, a recent review found that the total SOC stock of the soil profile under NT management had increased also in many humid regions (Ogle et al., 2019). As water holding capacity of soil is not often a limiting factor for plant growth in humid climate, adopting NT under those conditions can reduce productivity and C inputs to soil due to surface soil compaction, poor drainage and reduced soil aeration (Ogle et al., 2012; Pittelkow et al., 2015). Differences in the duration of the field experiments, management history of the sites or soil biological activity complicate obtaining clear conclusions even in large data compilations and further add uncertainty in the carbon sequestration potential of NT management.

Soil aggregation is considered a good indicator of sustainable soil use since the physical fractions of soil are sensitive to land management changes in the short term unlike soil chemical properties or changes in total SOC (Amézketa, 1999). NT affects the aggregate composition in two ways: it often increases the amount of carbon-rich macroaggregates but also reduces their turnover compared to CT (Six et al., 2000). The size distribution of soil aggregates and more stable aggregate structure of NT soils can be related e.g. to higher soil macrofaunal density often found in NT soils compared to CT (Briones and Schmidt, 2017). Ploughing, in turn, directly disturbs the aggregates and macrofauna, and indirectly affects the wetting, drying, thawing and freezing processes which all can break down the aggregates in cold climates (Le Guillou et al., 2012). Especially in northern conditions soil conservation is important as the growing season is short, and soils are susceptible to aggregate breakdown during the long winter season (Van Esbroeck et al., 2016; Bottinelli et al. (2017)) with consequent risk of soil erosion.

We examined the effects of NT and CT management on the rate of erosion and soil properties during and after a 10-year experiment representing typical farming systems of southwestern Finland with spring cereal production on clay soil. The aim was to quantify the benefits and trade-offs of reduced soil disturbance and to increase understanding on the related processes. We hypothesized that while NT 1) reduces water erosion it also 2) improves soil aggregate structure and 3) increases carbon stability in comparison to CT treatment.

2. Materials and methods

2.1. Site and management

The study site is Kotkanoja experimental field in Jokioinen, southwestern Finland (60° 49’ N, 23° 30’ E, about 100 m a.s.l., slope 1–4%,

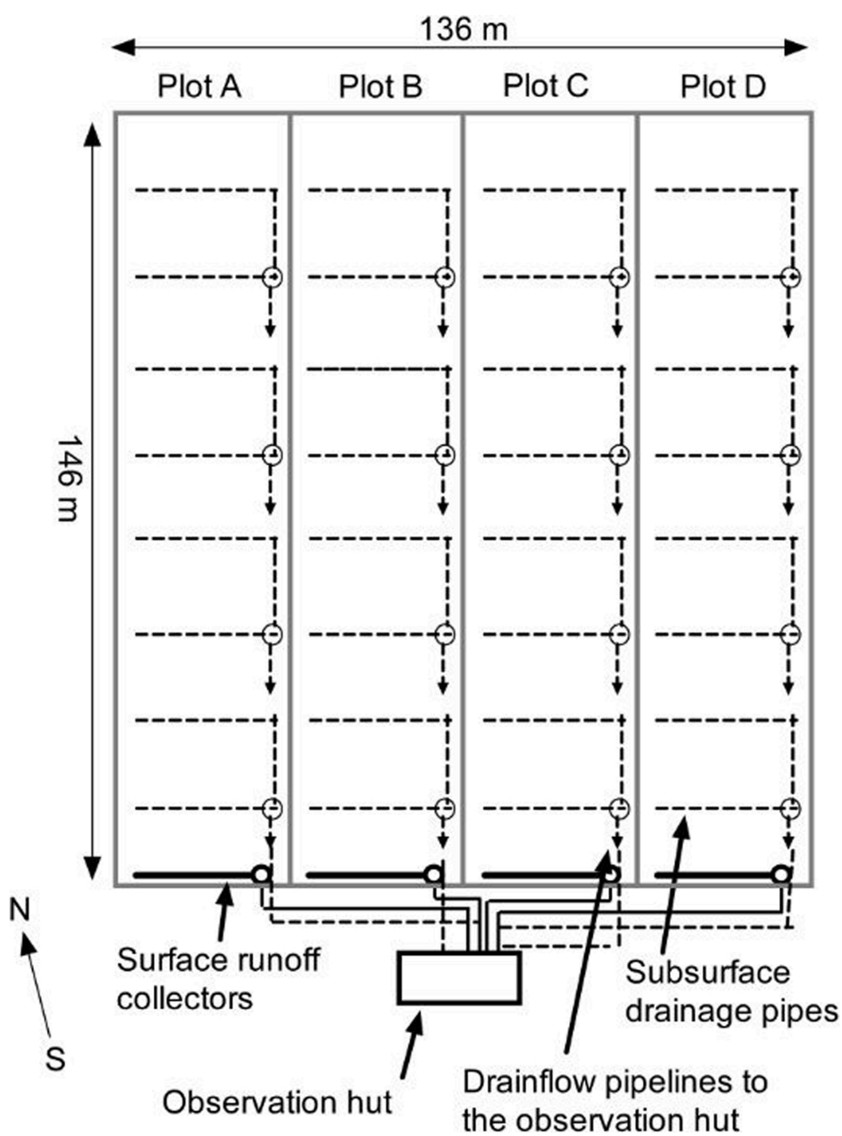

Fig. 1. The experimental field of Kotkanoja, Jokioinen, southwestern Finland.

mean 2%). The soil is a Protovertic Luvisol with average clay, silt and sand contents of 60, 16 and 24 %, respectively, in the topsoil (0−30 cm). The experimental field was established in 1976 for the purpose of studying leaching from arable land. The site has four 0.5 ha plots from which surface runoff is measured and each of the four plots is divided in four 33m × 33m subplots with identical subsurface drainage systems for drainage discharge measurement (Fig. 1). Two of the 0.5 plots (with eight subplots in total) were under CT treatment with autumn moldboard ploughing to the depth of about 20 cm and two other plots were under NT (sown with VM300SK, manufactured by Vieskan Metalli Oy, Finland) during the 10-year experimental period (2008 − 2018). Spring cereals (spring barley, spring wheat and oats) were grown on the plots using typical farm machinery and annual mineral NPK fertilizers rates (average 92−15−28 kg ha^{-1} yr^{-1}) and pesticides as needed. Dry grain yield was measured annually with combine harvester from two sites of each subplot.

2.2. Measurement of water discharge and erosion

Water discharge as surface runoff and subsurface drainage discharge was continuously monitored and sampled using a tipping bucket method as described in Turtola and Paajanen (1995) and Uusitalo et al. (2018). Subsurface drainage discharge was measured separately from each 16 subplot whereas surface runoff discharge was collected from each of the four 0.5-ha main plots. The flow-proportionally sampled water was collected on a daily to biweekly basis depending on the flow rate. Water samples were then stored at +4 °C until analyses. Quantity of eroded soil was determined by weighing the evaporation residues of the collected water samples (50 mL subsamples) after drying at 105 °C, and by multiplying with the respective amounts of discharge and summing the quantities up to get the annual erosion values. Precipitation and temperature data were obtained from an observation station of Finnish Meteorological Institute (Licence CC BY 4.0) located less than one kilometer from the site.

2.3. Soil sampling

Samples for soil organic carbon (SOC), particulate organic carbon (POC) and soil physical fractionation were collected from 0−10 cm and 10−30 cm soil layers of the experimental field in May 2018. From each 16 subplot, 2–3 replicate samples were taken, summing up to 20 samples per tillage treatment. For each sample above, several core drill samples (diameter 3 cm) were pooled to represent one square meter of the plot. In addition, one sample per each 16 subplot was taken from the depth of 30−60 cm. Additionally, for each 16 subplot, samples for bulk density, pore size distribution and saturated water conductivity were collected at the same time by using steel cylinders (5 cm in height and 200 cm^3 of volume). These undisturbed samples were taken by pressing the cylinders to the depth of 2.5–7.5, 12.5–17.5 and 22.5–27.5 cm and carefully lifting the cylinder from the soil. The samples were stored at +4 °C before analysis. The soil layers of 12.5–17.5 cm and 22.5–27.5 cm were later averaged to represent the 10−30 cm layer.

2.4. Soil organic carbon

The 20 replicate samples per tillage treatment for SOC analysis were air dried and sieved (2 mm). Carbon concentration was analyzed using dry combustion (LECO TruMac CN, LECO corporation, MI). The same analysis had been done for samples collected in Oct 2008. SOC stocks for 700 kg m^{-2} mineral soil layer of all subplots were calculated using the recommended equivalent soil mass method (Ellert and Bettany, 1995; Wendt and Hauser, 2013; Haddaway et al., 2017) as described in detail by Heikkinen et al. (2020).

Particulate organic matter in the samples was fractionated by wet sieving a 20 g soil sample through a 0.053 mm sieve to separate the sample to particulate organic matter (> 0.053 mm) and to silt and clay (< 0.053 mm) (Martínez-Mena et al., 2012). A few grams of each fraction were collected for carbon analysis.

2.5. Aggregate size

For physical soil fractionation the soil samples were sieved through an 8 mm sieve and air dried. The wet sieving method of Elliott (1986) was used to fractionate one or two 50 g soil samples from each subplot to large macroaggregates (> 2 mm), small macroaggregates (250–2000 μm) and microaggregates (0.053 – 0.25 mm). The mass of silt and clay (< 0.053 mm) that had passed through the last sieve was derived by subtracting the weight of the aggregates from the original 50 g sample. A few grams of each fraction were collected for total carbon analysis. For silt and clay, the carbon content was estimated by using the values of the original sample and the other fractions. Mean weight diameter of the aggregates was calculated as described by van Bavel (1950).

2.6. Pore size distribution

Water retention curves, measured from twelve undisturbed 200 cm^3 cylinder samples per treatment, were used to determine soil pore size distribution. First, the samples were moistened and kept at saturation point (-0.15 kPa matric potential) for two weeks. Thereafter they were placed on pressure plates and equilibrated at matric potentials of -0.5 kPa and −10 kPa using hanging water columns to drain large and medium sized pores (>30 μm and 0.2−30 μm), respectively. The same samples were used to determine bulk density as in Blake and Hartge (1986). The wilting point and the volume of small pores (<0.2 μm) was determined using 20 replicates per treatment, by an osmotic method (Aura, 1975), where a few grams of soil from the layers of 0−10 cm and 10−30 cm were dried to wilting point (respective to −1500 kPa) with polyethylene glycol to leave only small pores filled with water. Thereafter the samples were dried in an oven (105 °C) and weighted for the remaining water.

2.7. Hydraulic conductivity

Soil hydraulic conductivity was measured from 9 replicates of 200 cm^3 cylindrical undisturbed samples (7 cm in diameter) from both tillage treatment from layers 0−10 cm and 10−30 cm by the constant pressure method (Youngs, 1991). Samples were first wetted to the saturation point for two weeks. During the 3-hour measurement, water level of the sample was kept constant and all water draining through the sample was measured at several time points. The results were calculated by fitting the following equation to the data points;

$$I_C = K_{Sat} * t + S * t^{0.5},$$

where I_c is cumulative infiltration, K_{Sat} is the conductivity of the saturation point, t is time and S is the sorption of water (Philip, 1969).

2.8. Earthworm sampling and analysis

Earthworms were sampled in 2018 between 25 Sept and 4 Oct. Based on daily measurements at sampling sites, soil temperature at the 0−7 cm depth was 7−10 °C and soil moisture content 23–37 % (TDR reading at the depth of 0−15 cm). This indicates that conditions were favorable for the earthworm activity in the topsoil and for an efficient earthworm sampling. While one of the no-till plots (D) had been moldboard ploughed in July 18, 2018 it was nevertheless included in the earthworm part of the study as an earlier NT treatment because it was considered that a single ploughing instance would not essentially change the earthworm community. This was regarded plausible also because the ploughing was done during exceptionally dry soil conditions when a large proportion of adult endogeic and anecic earthworms reside below the ploughing depth.

Samples were taken with combined soil hand-sorting and AITC (mustard oil) extraction following ISO 23611-1, 2018 with some modifications (Nuutinen, 2019). From each of the 16 subplots, three samples were taken along a transect through the subplot with two samples at 5 m distance from the opposite subplot margins and one sample at the subplot center (appr. 16 m from the margins). At the sampling point, a soil sample with an area of 25cm × 25 cm and depth of 20 cm was taken with a spade, placed on a white sheet, and earthworms were hand-sorted from the sample. Simultaneously with the hand-sorting, AITC solution was poured at the bottom of the sampling pit to collect deep burrowing earthworms. The chemical extraction lasted for 30 min and the solution (max. 5 L) was added according to the infiltration rate. During the sampling, earthworms were picked in tap water and then stored in 4% formalin in the field.

After a few weeks storage in the laboratory, the samples were transferred to 70 % ethanol. The masses of the specimens as well as their species were determined when possible following Sims and Gerard (1999). To estimate the total abundance per square meter, the hand-sorting and chemical extraction samples were combined, and the numbers and masses multiplied by 16. For biomass, all specimens of the sample were used whereas for density estimation broken specimens lacking anterior end were excluded. Means of the subplot samples for earthworm density, mass and species number were used in the statistical analyses.

2.9. Statistical analysis

Soil properties after 10 years under the two tillage managements (CT and NT) and accounting for the two layers (0−10 cm and 10−30 cm) when appropriate were compared by generalized linear mixed models (GLMM) having management, layer and their interaction as fixed effects. Due to skewed distribution of some variables the assumptions of gamma (with a log link) and lognormal (with an identity link) distributions were used for SOC and large macroaggregate mass-%, and for saturated hydraulic conductivity, respectively. Each horizontal row in the field was used as a block, and thus, both managements had two plots per row. Block and its interaction with management were used as random effects. Correlated measurements from two layers within a plot were taken into account using homogeneous or heterogeneous compound symmetry (CS or CSH) covariance structure. The latter allows different variances for both layers. In post hoc tests, managements were compared only in both layers separately.

The monitoring results in the time series from 2008 to 2017 were analyzed using GLMM with the assumptions of gamma distribution (with a log link) for erosion quantities in subsurface drainage and surface runoff and the assumptions of Gaussian distribution (with an identity link) for subsurface and surface runoff discharges having management (CT and NT), dry matter cereal yield, mean winter temperature and annual precipitation as fixed effects. Some interactions could not be studied due to data limitations, but all relevant interactions were tested and found to be non-significant. For subsurface discharge, block and the interaction of year and management were denoted as random effects. The main random effect of year was omitted based on non-significance and due to possible correlation with annual precipitation and winter temperature. However, both models led to the same interpretations of fixed effects. For surface runoff, in turn, the effect of year and the interaction of year and management were denoted as random effects.

Residuals of all models were checked graphically and found to be adequate. The residual pseudo-likelihood (REPL) estimation method was used for models with the assumptions of gamma distribution and the residual maximum likelihood (REML) for others. Degrees of freedom were calculated using the Kenward–Roger method. Pairwise comparisons of the means of the managements were analyzed using the method of Westfall, with a significance level of $\alpha = 0.05$. The analyses were performed using the GLIMMIX procedure of the SAS Enterprise Guide 7.15 (SAS Institute Inc., Cary, NC, USA).

3. Results

3.1. Soil carbon

There was no significant difference in the SOC content of the 0−10 cm soil layer between NT and CT but the SOC content of the 10−30 cm layer was lower in NT (Table 1). The SOC stock calculated for equivalent soil masses was higher in NT compared to CT only in the layer

Table 1
Soil properties (mean with lower and upper bounds of the 95 % confidence interval) in 0-10 and 10-30 cm soil depth. Exponentiated distribution is marked with *-sign and p-values indicating statistically significant differences between the treatments at the level <0.05 are in bold.

	CT	NT	P
		0−10cm	
SOC %	2.75 (2.46;3.07)	2.95 (2.64;3.29)	0.344
SOC kg m^{-2}	3.16 (2.88;3.44)	3.56 (3.28;3.84)	0.051
POC kg m^{-2}	1.01 (0.88;1.15)	1.30 (1.17;1.43)	**0.011**
Bulk density g cm^{-3}	1.15 (1.06;1.24)	1.21 (1.12;1.30)	0.297
Saturated hydraulic conductivity cm h^{-1} *	7.07 (1.80;27.8)	4.92 (1.14;21.14)	0.818
Large pores Vol %	14.3 (11.3;17.3)	10.5 (7.5;13.6)	0.143
Medium pores Vol %	15.9 (12.9;19.0)	18.3 (15.3;21.3)	**0.018**
Small pores Vol %	25.2 (22.6;27.8)	27.0 (24.4;29.6)	0.143
Large macroaggregates mass-%	8.02 (6.20;10.4)	30.3 (23.4;39.3)	**< 0.001**
Small macroaggregates mass-%	52.2 (47.2;57.3)	43.8 (38.7;48.8)	**< 0.001**
Microaggregates mass-%	31.0 (24.4;37.6)	20.4 (13.8;27.0)	**0.002**
Mean weight diameter mm	0.735 (0.62;0.85)	1.31 (1.20;1.43)	**< 0.001**
		10−30cm	
SOC %	2.43 (2.18;2.71)	1.87 (1.67;2.08)	**0.004**
SOC kg m^{-2}	6.28 (5.74;6.82)	4.91 (4.37;5.45)	**0.004**
POC kg m^{-2}	1.77 (1.48;2.06)	1.40 (1.11;1.69)	0.074
Bulk density g cm^{-3}	1.29 (1.22;1.37)	1.33 (1.25;1.40)	0.297
Saturated hydraulic conductivity cm h^{-1} *	0.39 (0.10;1.53)	0.59 (0.15;2.33)	0.818
Large pores Vol %	7.52 (6.15;8.89)	7.04 (5.67;8.41)	0.454
Medium pores Vol %	18.4 (15.4;21.5)	15.5 (12.4;18.5)	**0.013**
Small pores Vol %	29.3 (26.5;32.1)	29.8 (26.9;32.7)	0.726
Large macroaggregates mass-%	8.96 (6.21;12.9)	9.57 (6.63;13.8)	0.774
Small macroaggregates mass-%	57.6 (51.6;63.6)	54.9 (48.9;60.9)	0.424
Microaggregates mass-%	24.3 (17.7;31.0)	27.4 (20.8;34.0)	0.071
Mean weight diameter mm	0.80 (0.69;0.92)	0.80 (0.69;0.91)	0.929

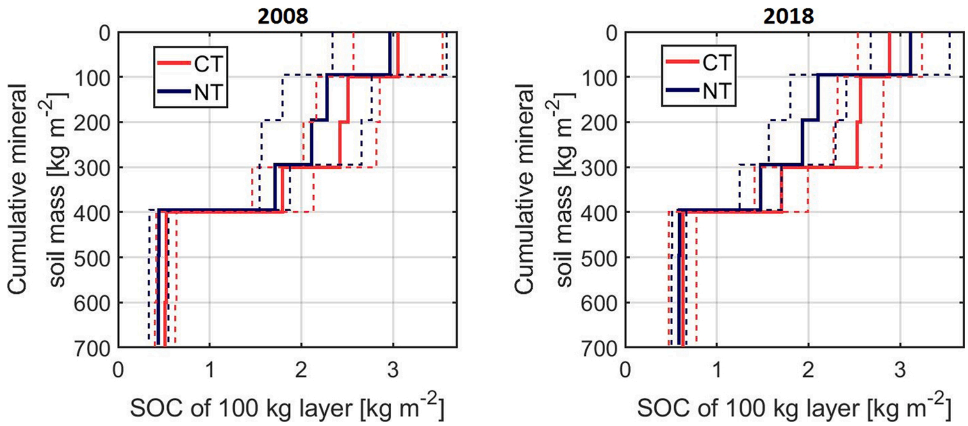

Fig. 2. SOC stock (solid line) and its standard deviation (dashed line) for CT and NT in each 100 kg mineral soil layer in 2008 and 2018. Values for the layers are interpolated from those measured for the 0-10, 10-30 and 30-60 cm layers.

corresponding to the top 100 kg of soil per square meter (Fig. 2). Below that, NT had a lower or equal SOC stock compared to CT. In the 700 kg m^{-2} mineral soil layer (representing approx. the 0−60 cm layer), there was a significant ($P = 0.025$) difference in total SOC stocks between the treatments: the total SOC stock of NT was 104 ± 10 Mg C ha^{-1} which was slightly lower than that in CT (116 ± 9 Mg C ha^{-1}). However, this cannot be interpreted as a decrease caused by NT since the SOC stocks measured at the beginning of the experiment in 2008 were already lower in NT (106 ± 16 Mg C ha^{-1}) compared to CT (114 ± 10 Mg C ha^{-1}) but with no significant ($P = 0.275$) difference (Fig. 2).

The stock of POC was higher in NT compared to CT in the 0−10 cm soil layer (Table 1). NT management also affected the distribution of SOC in the other soil physical fractions remarkably, especially in the topsoil (Fig. 3). In the 0−10 cm layer of NT soil, the amount of large macroaggregate bound SOC was 4.2 times that of CT ($P < 0.001$). Stock of SOC allocated in microaggregates of the same layer, in turn, was significantly lower in NT compared to CT ($P < 0.001$). The proportion of silt and clay bound SOC was significantly lower in the topsoil of NT ($P = 0.033$) and there was an indication for lower amount ($P = 0.076$) also in the 10−30 cm layer of NT compared to in CT. Statistically significant differences between CT and NT were not observed for small macroaggregates in either layer.

3.2. Soil structure

Bulk density did not differ between the tillage treatments although the average values tended to be higher in NT compared to CT (Table 1). Saturated hydraulic conductivity had high variability with no statistically significant differences between treatments. The proportion of medium-sized soil pores was significantly higher in NT than in CT in the 0−10 cm layer whereas in the deeper layer it was lower in NT. No significant differences in the proportion of large and small pores were observed. The mean weight diameter of soil aggregates was almost twice in the 0−10 cm layer of NT soil compared to that in CT. The proportion of large macroaggregates by mass-percent was in NT over three times that of CT while the contents of small macroaggregates and microaggregates were significantly lower in NT compared to CT in the 0−10 cm layer. The aggregate structure did not differ between the treatments in the 10−30 cm layer.

3.3. Earthworm abundance and diversity

Earthworm mean total density and mass were higher under NT compared to CT, however, the variation between subplots was large and the differences were only almost statistically significant (Table 2). Although the mean number of species was low in both treatments, it was significantly higher in NT. Five earthworm species were found in NT (the proportion of identified individuals in brackets): topsoil dwelling (endogeic) *Aporrectodea caliginosa* Sav. (58 %), deep burrowing (anecic) *Lumbricus terrestris* L. (29 %) and litter dwelling (epigeic) *L. castaneus* Sav.(5.5 %), *Dendrobaena octaedra* Sav.,(5.5 %) and *L. rubellus* Hoffm. (2%). CT had only two species: dominating *A. caliginosa* (84 %) and *L. terrestris* (16 %) and no litter dwelling species present.

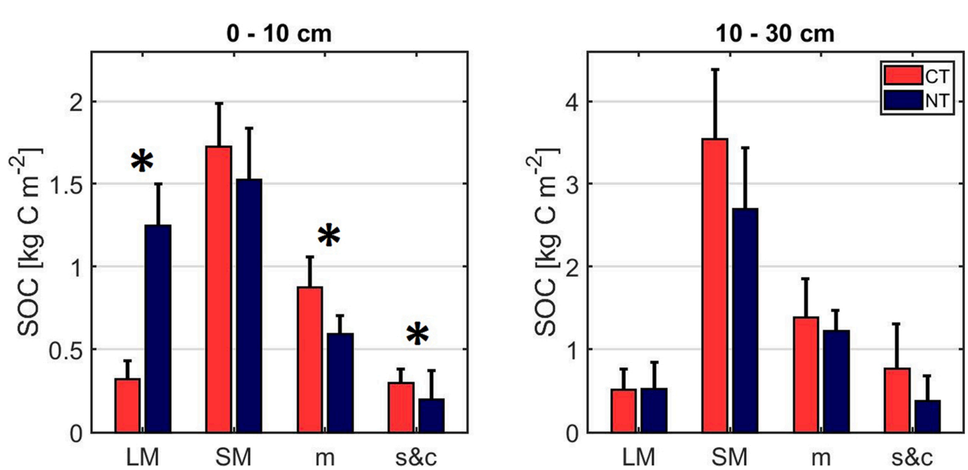

Fig. 3. Distribution of SOC in soil physical fractions in the 0-10 and 10-30 cm layers of conventionally tilled (CT) and no-till (NT) soil. LM = large macroaggregates, SM = small macroaggregates, m = microaggregates and s&c = silt and clay. Statistically significant differences ($p < 0.05$) are marked with the asterisk and the error bars denote standard deviation.

Table 2
Annual means (lower and upper bounds of the 95 % confidence interval) of subsurface drainage and surface runoff discharge, erosion in subsurface and in surface discharge, cereal dry matter yield data and earthworm data in conventionally tilled (CT) and no-till (NT) plots. Discharge, erosion rates and spring cereal yields are based on 10-year time series. Earthworm abundance and species number estimates are based on a single sampling in autumn 2018. Results of erosion and discharge rates are based on the model as stated in chapter 2.9 Statistical analysis. Exponentiated distribution is marked with *-sign and p-values indicating significant differences between the treatments at the level <0.05 are in bold.

	CT	NT	P
Subsurface discharge mm yr^{-1}	201 (164;238)	183 (146;220)	0.406
Surface discharge mm yr^{-1}	63.4 (34.4;92.3)	97.0 (68.0;126)	**0.003**
Subsurface erosion Mg ha^{-1} yr^{-1}	1.27 (0.89;1.81)	0.505 (0.35;0.72)	< **0.001**
Surface erosion Mg ha^{-1} yr^{-1}	0.25 (0.17;0.38)	0.16 (0.11;0.25)	**0.040**
Yield kg ha^{-1} yr^{-1}	3670 (2940;4400)	3070 (2340;3800)	< **0.001**
Earthworm density ind. m^{-2}	36.0 (8.3;63.7)	62.1 (46.7;77.4)	0.065
Earthworm mass g m^{-2}	8.40 (0;24.3)	24.9 (9.0;40.7)	0.057
Number of species *	1.28 (1.01;1.61)	2.21 (1.77;2.77)	< **0.001**

3.4. *Erosion rates and crop yield*

During the 10-year experiment, annual erosion rates in surface runoff and subsurface drainage discharge were significantly lower in NT compared to CT (Table 2). Because we found a persistent difference between the experimental plots not caused by the current treatments in the case of SOC we decided to check for such differences also in erosion rates. During the period of extensive grassland management preceding the tillage experiment, erosion rates were almost similar in the different plots (results not shown), and thus the differences observed in 2008–2018 can be considered due to the tillage treatments. Erosion in subsurface drainage from CT increased immediately after the end of grass phase and the beginning of the NT and CT treatments in 2008. The 10-year sum of erosion from both sources (surface and subsurface discharge) was 56 % lower in NT, the yearly differences between treatments varying between 16 and 71 % during the experiment. Water volumes as subsurface discharge did not differ between the treatments but the surface discharge was significantly higher from NT, 1.6 times of that measured from CT.

Both water discharge and erosion rates followed the pattern in annual precipitation with the exception of CT in 2015 when ploughing was followed by repeated rainfall and warm temperature in the late autumn (Fig. 4). Precipitation was found to be the only significant factor to increase subsurface water discharges in the 10-year experiment ($P < 0.001$). Higher winter-time temperature, annual precipitation and more intensive tillage management can be used to explain increased erosion rate in subsurface discharge (Table 3). For erosion in surface runoff, only tillage management explained the annual rate significantly. Mean monthly erosion rates varied highly during the experiment and especially for CT erosion in subsurface discharge showed high variation and high rates in early winter months of November and December (Fig. 5).

The mean crop yield during the 10-year experimental period was significantly lower in NT compared to CT with a mean difference of 17 % (650 kg ha^{-2} yr^{-1}) between the treatments (Table 2).

4. Discussion

As hypothesized, we observed that erosion was significantly lower in NT compared to autumn moldboard ploughing during the 10-year experiment which is in line with the results of several other studies (Langdale et al., 1979; Schuller et al., 2007; DeLaune and Sij, 2012). The observed differences in erosion were obviously not due to differences in discharge volumes since e.g. erosion through subsurface drains was significantly lower in NT despite the similar subsurface discharges in both treatments. Moreover, the erosion in surface runoff was also clearly less in NT although the surface runoff volume was higher.

Our results deviated from the mainstream as it is more common to find less runoff in NT compared to CT (Sun et al., 2015). However, the site characteristics with a gentle slope and high soil clay content likely explain this discrepancy as was also suggested by the meta-analysis by Sun et al. (2015) that found NT causing less decrease in runoff at sites with gentle slope and no decrease in runoff at sites with high clay content. The findings on the lower proportion of large pores, higher bulk

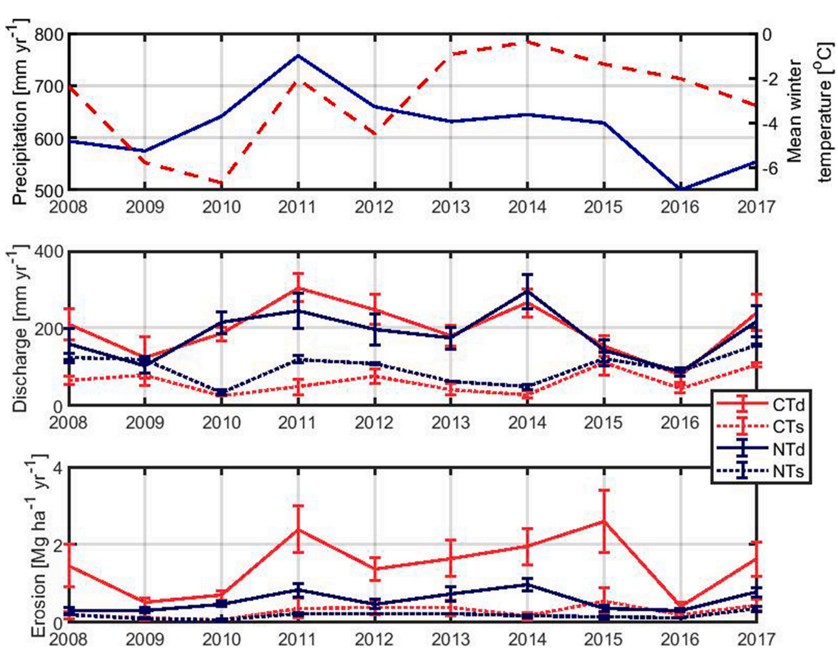

Fig. 4. Annual precipitation (solid line) and mean winter temperature of Nov-Mar (dashed line), annual amount of water and erosion matter (± standard deviation) in the subsurface drainage discharge and surface discharge of conventionally tilled and no-till plots (CTd, CTs, NTd and NTs, respectively). The annual values represent the period from September to August of next year.

Table 3
Results of statistical modelling to explain erosion rates in subsurface drainage and surface runoff discharge from conventionally tilled (CT) and no-till (NT) plots. Erosion (log(g ha^{-1} yr^{-1})), dry grain yield (kg ha^{-1}), mean winter temperature of Nov - Mar (°C), precipitation (mm). P-values indicating significant differences at the level <0.05 are in bold.

	Effect	Estimate	SE	DF	t value	P
Subsurface erosion	Intercept	10.928	0.905	16.6	12.1	**<0.001**
	Management CT	0.921	0.185	16.5	4.99	**<0.001**
	Management NT	0	.	.	.	.
	Yield	−0.00006	0.00005	144	−1.34	0.182
	Mean winter temperature	0.121	0.0463	16.1	2.62	**0.018**
	Precipitation	0.00445	0.00140	15.9	3.19	**0.006**
Surface erosion	Intercept	12.431	0.268	9.12	46.4	**<0.001**
	Management CT	0.398	0.137	9	2.9	**0.018**
	Management NT	0	.	.	.	.
	Mean winter temperature	0.138	0.0731	8	1.89	0.095

density by average and tendency of lower hydraulic conductivity in NT compared to CT are in line with the observed increase in surface discharge. Also, the largest difference between surface discharge was in the spring during snowmelt, when soil in NT remained longer in frost, and thus infiltration was lower compared to CT (Uusitalo et al., 2018).

These results corroborate those of Lipiec et al. (2006) but were contrary to findings of Fuentes et al. (2004) which showed significantly higher porosity and saturated hydraulic conductivity in NT topsoil compared to CT. Average macroporosity of the 0−30 cm layer of the NT plots was actually below the 10 % threshold found to represent soil compaction that reduces crop production (Aura, 1983). Enhanced earthworm abundance and diversity which is typical for no-till soils (Kladivko et al., 1997; Johnson-Maynard et al., 2007; Briones and Schmidt, 2017) can maintain high macroporosity in no-till soils (VandenBygaart et al., 1999). In the present case earthworm densities, however, remained relatively low also in NT. Most likely this relates to the high soil clay content which is known to limit population densities of earthworms in boreal arable soils (Nieminen et al., 2011). Especially the deep burrowing earthworms such as *L. terrestris,* can counteract the effects of lower porosity in NT by creating preferential water flow paths through the soil profile (Shipitalo and Le Bayon, 2004). Because of the direct contacts of burrows with the subdrains (Nuutinen and Butt, 2003) they could even increase subsurface flow and erosion. That seemed not to be the case at our site as NT management with the highest occurrence of anecic species reduced erosion by both discharge pathways.

The highest monthly erosion rates occurred in the winter months,

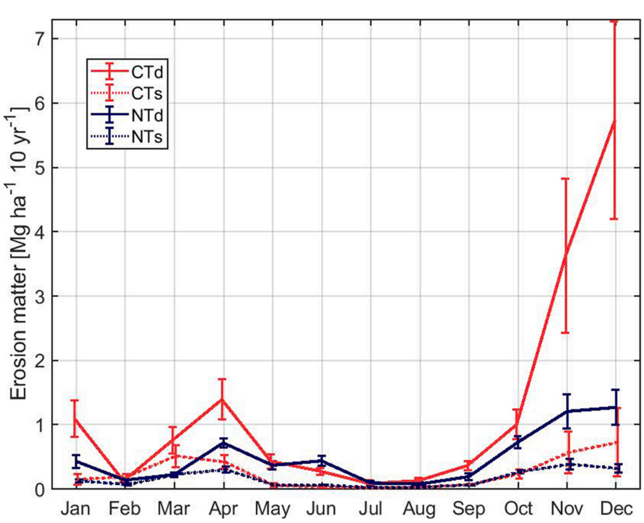

Fig. 5. Ten-year sum of monthly erosion matter in the subsurface drainage discharge (N = 8 per treatment) and surface discharge (N = 2 per treatment) of conventionally tilled and no-till plots (CTd, CTs, NTd and NTs, respectively) during Sep 2008 - Aug 2018. The error bars denote standard deviation. Tillage has been implemented around the end of October.

especially in CT with the bare soil surface, and the differences between the tillage managements are partly explained by the straw cover protecting the soil e.g. from raindrops in NT (Lu et al., 2016). In boreal climatic conditions, the non-vegetated period for conventional tillage practice of autumn ploughing is long, from September/October to May and thus the frequency and intensity of rainfall have a high potential impact on erosion losses. In this study, an increase in mean temperature during winter (Nov - Mar) had a significant effect on erosion rates due to higher occurrence of rain as water, bare non-frozen soil and freeze-thaw cycles. Earlier Puustinen et al. (2007) also reported increased erosion rates in milder winter conditions in Finland. This clearly suggests that future climate warming poses a threat of increased erosion rates in northern European conditions and calls for improved soil management. Postponing soil tillage to spring and keeping soil surface covered by straw overwinter have been found to decrease erosion rates considerably (Lundekvam, 2007; Turtola et al., 2007; Bechmann, 2012; Skøien et al., 2012; Ulén et al., 2012; Starkloff et al., 2017).

There is a slight indication that NT is accruing SOC to topsoil while CT is decreasing it (Fig. 2). This may further improve the erosion resistance of the soil surface in NT as it has been found that the ratio clay %/SOC% could be a simple indicator of the erosion risk of clay soils (Soinne et al., 2016). At our site, this ratio was 20, which is at the higher end of the range reported by Soinne et al. (2016) suggesting that e.g. increased carbon input would be a measure to improve the resilience of such fields in future climatic conditions.

In line with our second hypothesis, low disturbance frequency in NT plots for 10 years (and reduced tillage or grass cover since 1991) led to enhanced soil aggregate formation compared to CT plots as also found e.g. by Blanco-Canqui et al. (2009). Despite their relatively low numbers, earthworms may have contributed to the observed differences in soil structure as the earthworm density in the NT soil tended to be higher compared to CT. Earthworms affect soil aggregation by producing casts which can develop in stable macroaggregates as they age (Shipitalo and Le Bayon, 2004). The soil stabilizing effect is partly related to the ability of earthworm mucus to promote organo-mineral complexes in soil (Guhra et al., 2020). Previous work has documented the increase of *L. terrestris* population density at the NT plots of our experiment (Nuutinen et al., 2011), a finding corroborated by the present findings. Due to its semi-sedentary lifestyle, midden construction and casting on soil surface, this species can have a strong effect on topsoil aggregate structure at its living sites. In the arable clays of the region, the density of *L. terrestris* was positively associated with the percentage of large macroaggregates (Singh et al., 2015) and a carbon stabilization effect of the species was noticed by Sheehy et al. (2019). Thus, we can anticipate that the higher occurrence of earthworms, and *L. terrestris* in particular, had partly mediated the observed changes in soil aggregation and carbon dynamics during the NT management of our site.

In 2018, NT had lower SOC stock in the 700 kg m^{-2} mineral soil layer than CT but based on the historical data there has been a similar difference at least since 1990 (results not shown) and thus we cannot confirm the effect of NT on the total SOC stock based on the latest 10-year dataset. The trend in SOC stock, however, suggests that the stock has increased in CT while it has decreased in NT in 2008–2018. This may be related to the lower crop productivity in NT which might reduce SOC stocks in NT as reported by Ogle et al. (2012). The yield level was relatively low in both treatments but noticeable lower in NT than in CT which is in line with other studies reporting such differences and stating that the difference in yield can be related to crop species, aridity, residue management, duration of the experiment and fertilization (So et al., 2009; Pittelkow et al., 2015). In boreal conditions, the main reasons may be later sowing time and slower plant growth in spring in NT due to soil compaction, excessive moisture and low temperature in the topsoil.

The observation that NT had higher SOC stock than CT in topsoil but lower below that layer is in line with the latest data compilations (Ogle et al., 2019; Powlson et al., 2014; Haddaway et al., 2017). The result suggests that there is unused carbon sequestration potential in the deeper soil layers at least in the NT treatment. Adopting more versatile management options such as conservation agriculture with e.g. deep-rooted species in diversified crop rotations connected to NT could lead to a better carbon balance also deeper in the soil (Palm et al., 2014).

Despite the tendency of lower total carbon stock of the soil profile in NT compared to CT, the amount of aggregate-protected carbon was higher in NT corroborating our third hypothesis. The same has been observed also in other tillage experiments on clay soil sites of the region (Sheehy et al., 2015). It seems that all SOC accrued in the 0−10 cm layer of NT was allocated to the large macroaggregate fraction while the proportion of the smaller fractions and SOC in them declined. This was likely related to gradual growth of the small fractions and their development to larger aggregates under continued long-term NT management (Tisdall and Oades, 1982). NT also significantly increased the content of particulate organic matter carbon in topsoil which is known to improve erosion resistance (Pikul et al., 2007). The opposite was true in the 10−30 cm layer but the topsoil conditions are more crucial for erosion abatement and for soil biota. Particulate organic matter, as easily decomposable material, is essential for soil microbial activity and increased particulate organic matter is often connected to increased microbial abundance (Helgason et al., 2010). Through higher concentration of particulate organic matter, NT could eventually increase the soil carbon stocks if the carbon from the microbial biomass turnover increases more than soil organic matter decomposition (Prommer et al., 2020).

5. Conclusions

Based on these results, the most significant environmental impact of NT management was the approximately one ton reduction in annual erosion rate per hectare compared to CT. Quitting tillage improved soil and carbon stability by increasing the average size and carbon content of soil aggregates probably due to both less soil disturbance and higher soil stabilizing activity by soil biota. The reduced erosion rate in NT was related to the lower erodibility of the soil rather than differences in water discharge as the loss of erosion matter in subsurface discharge was more significantly reduced than the flow rate. On the other hand, the increase in surface runoff did not enhance erosion in runoff. These results suggest that NT management can be recommended for clay soils in northern Europe especially in regions where annual cropping dominates, and fields are increasingly imposed to winter rains with shorter duration of the snow cover in future climatic conditions. A noticeable downside was the poorer crop production in NT compared to CT which also decreases carbon input to soil. This diminishes carbon sequestration especially in the soil layers below 10 cm where there seems to prevail unutilized potential for carbon sequestration. Thus, further improvements in the management of similar sites should be designed to increase the carbon input to the deeper soil layers e.g. by deep-rooted plants, a practice also potentially decreasing the risk of soil compaction in NT.

Declaration of Competing Interest

The authors report no declarations of interest.

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