Science of the Total Environment 660 (2019) 97–104

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Short Communication

Significant build-up of soil organic carbon under climate-smart conservation farming in Sub-Saharan Acrisols

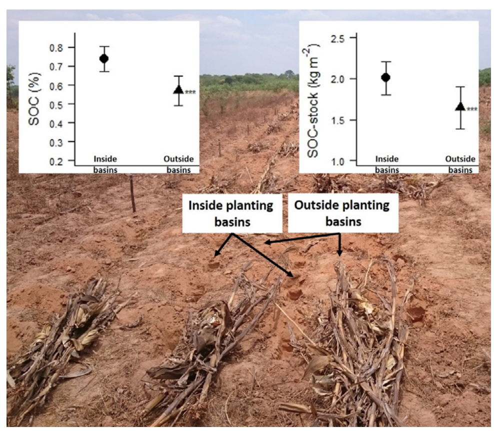
Vegard Martinsen [a,*], Jose Luis Munera-Echeverri [a], Alfred Obia [b,c], Gerard Cornelissen [a,b], Jan Mulder [a]

[a] Faculty of Environmental Sciences and Natural Resource Management, Norwegian University of Life Sciences, P.O. Box 5003, 1432 Ås, Norway
[b] Norwegian Geotechnical Institute (NGI), P.O. Box 3930, Ullevål Stadion, 0806 Oslo, Norway
[c] Department of Agronomy, Faculty of Agriculture and Environment, Gulu University, P.O. Box 166, Gulu, Uganda

HIGHLIGHTS

- Soil fertility build-up inside vs. outside planting basins under conservation farming in SSA
- Absolute increase of 0.05 t C ha^{-1} yr^{-1} inside vs. outside 20 cm deep planting basins
- Relative increase of 2.95 ‰ SOC yr^{-1} inside vs. outside 20 cm deep planting basins
- Increase in labile C, pH, CEC and potential nitrification inside planting basins

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 5 September 2018
Received in revised form 20 December 2018
Accepted 29 December 2018
Available online 04 January 2019

Editor: Jose Julio Ortega-Calvo

Keywords:
Hand-hoe tillage
Planting basin
Soil fertility
Soil sampling
Phosphorus
CEC

ABSTRACT

Conservation farming (CF) involving minimum tillage, mulching and crop rotation may offer climate change adaptation and mitigation benefits. However, reported effects of CF, as applied by smallholders, on storage of soil organic carbon (SOC) and soil fertility in Sub-Saharan Africa differ considerably between studies. This is partly due to differences in management practice, soil type and adoption level between individual farmers. Where CF involves planting basins, year-to-year changes in position of basins make SOC stock estimates more uncertain. Here we assess the difference in SOC build-up and soil quality between inside planting basins (receiving inputs of lime and fertilizer; basins opened each year) and outside planting basins (no soil disturbance or inputs other than residues) under hand-hoe tilled CF in an Acrisol at Mkushi, Zambia. Seven years of strict CF husbandry significantly improved soil quality inside planting basins as compared with outside basins. Significant effects were found for SOC concentration (0.74 ± 0.06% vs. 0.57 ± 0.08%), SOC stock (20.1 ± 2.0 vs. 16.4 ± 2.6 t ha^{-1}, 0–20 cm), soil pH (6.3 ± 0.2 vs. 4.95± 0.4) and cation exchange capacity (3.8 ± 0.7 vs. 1.6 ± 0.4 $cmol_c$ kg^{-1}). As planting basins only occupy 9.3% of the field, the absolute rate of increase in SOC, compared with outside basins, was 0.05 t C ha^{-1} yr^{-1}. This corresponds to an overall relative increase of 2.95‰ SOC yr^{-1} in the upper 20 cm of the soil. Also, hot water extractable carbon (HWEC), a proxy for labile organic matter, and

Abbreviations: CF, conservation farming; CA, conservation agriculture; SOC, soil organic carbon; CEC, cation exchange capacity; HWEC, hot water extractable carbon; P, phosphorus; K, potassium; N, total nitrogen; SOM, soil organic matter; SOC, soil organic carbon; BD, bulk density; NH_4-N, ammonium; NO_3-N, nitrate.

* Corresponding author.
E-mail address: vegard.martinsen@nmbu.no (V. Martinsen).

https://doi.org/10.1016/j.scitotenv.2018.12.452

potential nitrification rates were consistently greater inside than outside basins. The significant increase in quantity and quality of SOC may be due to increased inputs of roots, due to favorable conditions for plant growth through input of fertilizer and lime, along with increased rainwater infiltration in the basins.

1. Introduction

Soil organic matter (SOM) is important for soil structure, water holding capacity and release and retention of plant nutrients that are crucial for agricultural productivity (FAO, 2017a). Soil organic carbon (SOC) is a major constituent of SOM and SOC sequestration mitigates climate change (Lal, 2004a, 2004b), while a decline in SOC may lead to soil degradation that poses a threat to global climate and food security (Lal, 2013). Recently, the "4 per mille" initiative (4p1000.org/) was launched at the COP 21 with an aspiration to increase global SOC stocks by 4 per thousand per year (0.4%) as a compensation for the global emissions of greenhouse gases from anthropogenic sources (Minasny et al., 2017). In a survey of SOC stock and sequestration potential for 20 regions in the world, Minasny et al. (2017) found that a sequestration rate of 4‰ can be accomplished under best management practices. Also, Paustian et al. (2016) reported that improved soil management could make "climate smart soils" in terms of increased C sequestration and reduced greenhouse gas emissions. However, others such as Poulton et al. (2018) have highlighted limitations to achieving the 4‰ goal in practical agriculture due to e.g. lack of resources or because practices are uneconomic or undesirable for food production.

Adoption of sustainable production systems and practices has been suggested to increase resilience and help mitigate climate change (FAO, 2017b). Conservation agriculture (CA) comprising the principles of minimum tillage, residue retention and crop rotation (Mafongoya et al., 2016; Powlson et al., 2016) may offer climate change adaptation and mitigation benefits, due to increased SOC storage, soil fertility, water conservation and productivity (Corbeels et al., 2018; Lal, 2015; Pisante et al., 2015). However, reported effects of CA on SOC sequestration and soil quality in Sub-Saharan Africa differ considerably between studies (Cheesman et al., 2016; Corbeels et al., 2018; Corbeels et al., 2015; Sommer et al., 2018; Thierfelder and Wall, 2012) and the mitigation potential of CA systems remains unclear (Thierfelder et al., 2017). A recent review by Corbeels et al. (2018) showed that annual SOC accumulation rates in response to treatments with all three principles of CA varied enormously, from −96 to 176‰ yr^{-1}, with half of the observations reporting relative SOC build-up rates exceeding 34‰ yr^{-1}. It is believed that climatic and edaphic conditions, combined with management practices such as seeding system, degree of residue retention, fertilizer addition, weeding and crop rotation, determine whether CA has positive, negative or no effect on yields and soil fertility (Gatere et al., 2013; Giller et al., 2009; Nyamangara et al., 2014; Steward et al., 2018; Thierfelder et al., 2016).

Due to mulching, using crop residue, CF increases the input of organic carbon in soil, thus enhancing soil structure, water infiltration and biological activity (Lal et al., 2007; Powlson et al., 2014). In addition, no-till, due to its minimum soil disturbance, is effective in controlling soil evaporation, sequestering SOC and minimizing erosion losses (Lal et al., 2007). According to Powlson et al. (2014) potential disadvantages of no-till include relatively small additions of SOC to the whole profile (i.e. increases occur largely near the soil surface), more challenges in weed control (extra hand weeding or reliance on herbicides), increased BD and in some cases increased nitrous oxide emission. No-till in combination with residue retention and crop rotation increases crop yields under rain fed agriculture in dry climates (Pittelkow et al., 2015). However, no-till alone may reduce yields (Pittelkow et al., 2015) and inappropriate management, such as insufficient weeding and lack of early planting, constrains yields on CA farms (Gatere et al., 2013). Farmers may struggle to follow all principles of CA such as maintaining sufficient crop residues, due to e.g. burning, removal and grazing that will reduce carbon inputs to the soil (Cheesman et al., 2016; Chivenge et al., 2007; Thierfelder et al., 2013; Umar et al., 2011). In addition, CA technologies (e.g. direct seeding/dibble stick, hand hoe-basin systems and ripping) and fertilizer application rates vary among individual farmers (Johansen et al., 2012; Mafongoya et al., 2016; Thierfelder et al., 2015) resulting in variations in soil disturbance and input of organic carbon and nutrients to the soil. Together with inherent site/farm heterogenity (intrinsic soil properties, micro-climate) affecting crop production, these factors may partly explain the large variation found in the literature with respect to yield and soil quality effects of CA.

In Zambia, the CA system, with hand-hoe prepared planting basins and animal-drafted or mechanized rip lines (Johansen et al., 2012), has been promoted by the Conservation Farming Unit (CFU). CFU uses the term conservation farming (CF) for conservation tillage (i.e. minimum tillage (MT), using planting basins or ripping), retention of crop residues and the incorporation of legumes in crop rotation (CFU, 2011; Gatere et al., 2013). Recently, Sommer et al. (2018) reported reduced losses of SOC (0–15 cm) but no net carbon sequestration under CA in two long term (12 years) trials in Western Kenya. A large number of on-farm sites in Zambia (Martinsen et al., 2017) and Zimbabwe (Nyamangara et al., 2013) indicate small effects of CF on soil C stocks. Comparisons of soils under CA (up to 9 years) and adjacent conventional fields from 450 farms in 15 districts in Zimbabwe revealed generally low SOC contents (<1%) without clear difference between the two management practices (Nyamangara et al., 2013). Results from 40 on-farm sites in Zambia showed small differences in soil quality parameters between CF and conventional practices at smallholder farms after maximum 12 years since CF adoption (Martinsen et al., 2017). In both studies, there were only small differences in amount of SOC, total phosphorus and pH when comparing inside and outside CF planting basins. Martinsen et al. (2017) attributed this to a gradual year-to-year shift in position of the basins and large variability between study sites.

Here, we assess the effect of seven years of hand-hoe tilled CF on soil quality and build-up of SOC by comparing soil from inside vs. outside planting basins under controlled conditions in Acrisols, Mkushi, Zambia. Considerable attention was given to keep basins in the same position and fertilizer and lime were added to basins only. Planting basins are hypothesized to increase SOC content, due to the high root density and increased biomass production, associated with elevated soil moisture, as basins favor the accumulation of runoff from the surrounding outside basin areas.

2. Material and methods

2.1. Study design and sampling

The study was conducted on a private farm (Mt Isabel) in Mkushi (S13°45′25.7″ E29°03′55.5″), central Zambia. The average annual rainfall and temperature were 1220 mm and 20.8 °C, respectively. The soil type was sandy loam Acrisol (Obia et al., 2016). Land use prior to soil sampling in 2015 included seven years of strict conservation farming (CF) husbandry. Before application of CF, land use was conventional, including continuous maize cropping with minimal inputs of fertilizer and lime and poor weed control. Conversion to CF in 2008 included dry season (May–August) preparation of permanent planting basins using hoes (min tillage), two year crop rotation (maize-ground nuts) and residue retention (mulching without chopping, i.e. leaving plant residues on the soil surface in between planting rows). The CF practice included preparation of rows of permanent basins (Fig. 1, Fig. S1) with a spacing of 90 cm between rows and 80 cm between basins within rows (~13,890 basins ha^{-1}). Each basin has an area of ~0.07 m^2 and a volume of ~13.4 L (20 cm depth, 40 cm length, 16.7 cm width). Herbicides (Glyphosate, Atrazine/Cyanazine mix and Gramoxone) and hand weeding were used for weed control. For maize, fertilizer "Compound D" (N, P_2O_5, K_2O, 10:20:10) was applied at a rate of 200 kg ha^{-1} yr^{-1} before planting and urea (46:0:0) applied as top dressing at a rate of 100 kg ha^{-1} yr^{-1} about four to five weeks as well as eight weeks after planting (i.e. a total of 200 kg ha^{-1} yr^{-1}). Legumes used in rotation received no fertilizer; so, the site was fertilized every second year, i.e. four times in the period 2008–2015. The total amount of NPK on elemental basis corresponded to an application of 112 kg N ha^{-1} yr^{-1}, about 17.5 kg P ha^{-1} yr^{-1} and about 16.5 kg K ha^{-1} yr^{-1} during the four years when fertilizer was applied. A total amount of 2.8 t ha^{-1} (i.e. ~11 g kg $soil^{-1}$ inside CF basin) of Dolomitic lime ($CaMg(CO_3)_2$) was added to the basins in years with maize (2008, 2010, 2012 and 2014).

Soil sampling was conducted in October 2015 and in May 2016. After sampling, the soil was air dried prior to transport and analyzed. In October 2015 soil sampling was conducted at three selected plots (each ~24 m^2) in each of four blocks (each block ~250 m^2, Fig. 1a). Each plot consisted of four rows of six permanent planting basins (i.e. 24 basins per plot, Fig. 1b). At each of the 12 plots, five soil samples (0–20 cm) were collected in planting basins and three soil samples were collected between basin rows (i.e. outside basins) using a hand hoe. The five and three samples, respectively, were bulked prior to chemical analysis (n = 12 for both CF basins and outside CF basins, i.e. a total of 24 samples) to make a composite sample per plot. When sampling, we focused on the top 20 cm of the soil, which was the basin depth. In an earlier study we showed that maize roots tended not to go deeper than the basin, with 95% of maize roots occurring in the top 25 cm of the soil (Abiven et al., 2015). Undisturbed soil samples were collected at 2–7 cm soil depth using 100 cm^3 steel rings in two plots per block in basins and between rows to determine plant available water and bulk density (BD). One sample from between rows in block 1 was lost in transport, so the average of CF basins in block 1 was used to allow for a paired comparison per block (n = 7 for both CF basins and between basin rows).

In May 2016 at harvest, four soil samples at two plots per block were collected inside basins and between rows at 0–3, 3–8, 8–13 and 13–20 cm soil depth, using a cylindrical soil auger. Hot water extractable carbon (HWEC) was determined on all samples and potential N-mineralization rate was determined on samples bulked at 0–8 cm and 8–20 cm soil depth. In order to report the results for the same depth intervals, values for HWEC were bulked at the same depth intervals based on depth-weighted average for 0–3 and 3–8 cm and 8–13 and 13–20 cm, respectively. Thus, the total amount of samples at both of the depths 0–8 and 8–20 was n = 4 for both CF basins and for between rows of CF basins (i.e. a total number of 16 samples).

2.2. Soil analysis and statistics

A detailed description of the methods can be found in Appendix A (Supplementary data). Briefly, bulk density (BD), volume percentage water at field capacity (pF 2) and at wilting point (pF 4.2), as well as total porosity were determined using undisturbed soil cores of 100 cm^3 according to Obia et al. (2016). The amount of plant available water was calculated as the difference between volume percentage water at field capacity and at wilting point. Prior to all other analyses, soil samples were air-dried and sieved at 2 mm. Soil texture was determined using the Pipette method (Elonen, 1971). In the pipette method,

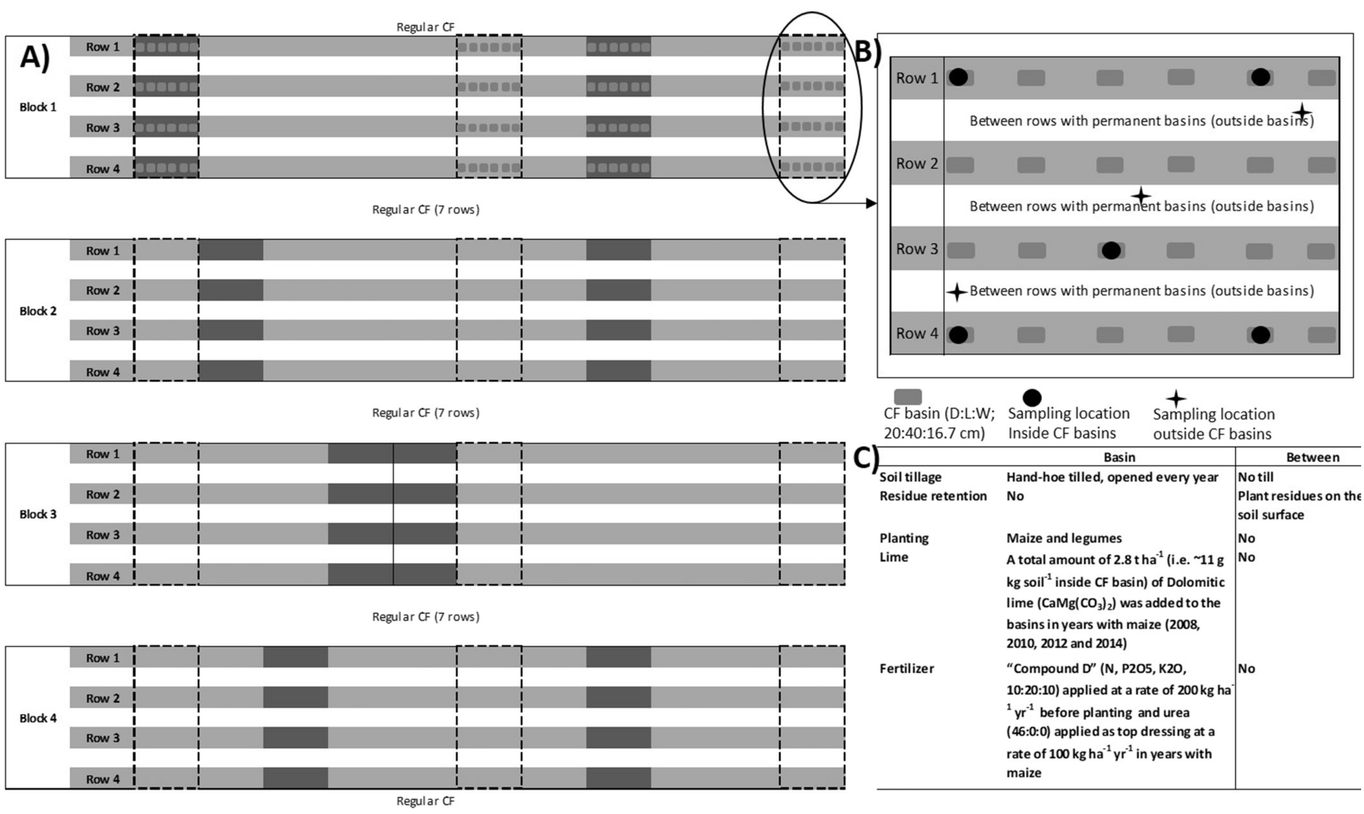


Fig. 1. A) Experimental setup, Mkushi, Zambia. The sampling was conducted at 4–5 plots (each ~24 m^2) in four blocks (each ~4 m width and 61 m length i.e. ~250 m^2). Plots used for chemical analysis and BD are highlighted with a dashed border (three plots in each block sampled October 2015) and plots used for determining hot water extractable C and potential N mineralization are highlighted dark gray (two plots per block sampled May 2016). **B**) Plot (24 m^2, 24 planting basins) used for soil sampling. **C**) Management.

soil texture classes (clay, silt, sand) are estimated in terms of size and distribution of primary particles by sieve and sedimentation analysis. Soil pH was determined in 0.01 M $CaCl_2$ using a soil to solution ratio of 1:2.5 with a digital pH meter. Subsamples of the air-dried and sieved samples were dried at 60 °C to determine dry matter content and then milled prior to determination of organic carbon (C) and total nitrogen (N). Total soil carbon was determined by dry combustion (EC12, C determinator, Leco Corporation) (Nelson and Sommers, 1982). Since soil pH was below 6.5, total C was used as a measure of soil organic carbon (SOC). Total N was determined on untreated soil samples by the Dumas method (TruSpec, CHN analyzer, Leco Corporation; Bremmer and Mulvaney, 1982). Carbon and N stocks were calculated based on volume of soil by multiplying depth of sampling, BD and elemental concentration. Mean values of BD for basins and between rows, respectively, were used per block to calculate C stocks. In addition, C stocks were calculated based on an equivalent mass of soil since equal depth sampling may overestimate C stocks in treatments with greater BD (Ellert and Bettany, 1995; Wendt and Hauser, 2013).

Exchangeable base cations (Ca^{2+}, Mg^{2+}, Na^+, K^+) and exchangeable Al^{3+} were determined in 1 M ammonium nitrate extracts (NH_4NO_3, unbuffered). Exchangeable acidity was determined by back-titration with 0.05 M sodium hydroxide to pH 7. The sum of exchangeable base cations and exchangeable acidity was assumed to equal the effective cation exchange capacity (CEC) according to Schollenberger and Simon (1945). Total, inorganic and organic P was determined according to Møberg et al. (1990). Acid oxalate extractable Fe, Al and P were determined according to van Reeuwijk (1995).

Potential nitrogen mineralization rates were determined in an incubation experiment (60 days, 20 °C) using air dried and sieved soil samples with an adjusted water content of ~31 vol% (modified after Martinsen et al. (2017); see Appendix). Rates of net ammonification and net nitrification were determined by subtracting initial (day 0) KCl-extractable soil NH_4-N and NO_3-N from final amounts (after 60 days) of extracted NH_4-N and NO_3-N, respectively. Extractable NH_4-N and NO_3-N were analyzed using a flow injection analyzer (FIA Star 5010). Hot water extraction of organic C was done to determine the labile fraction of SOM according to Ghani et al. (2003). Briefly, 30 mL of distilled water was added to polypropylene tubes with 5 g of soil. The suspensions were gently shaken in a vortex shaker prior to extraction of dissolved organic carbon in a temperature-controlled hot water bath (80 °C during 16 h). Dissolved organic carbon was measured after filtration of the extract (0.45 μm), using a total organic carbon analyzer (TOC-V CPN, Shimadzu).

Comparison of mean soil parameters for basins and between rows was done using two-sided paired *t*-tests. Difference between means and 95% confidence intervals of the estimated differences are shown in Figs. 2 and 3 and in Table 1. To assess propagation of error (Ku, 1966),

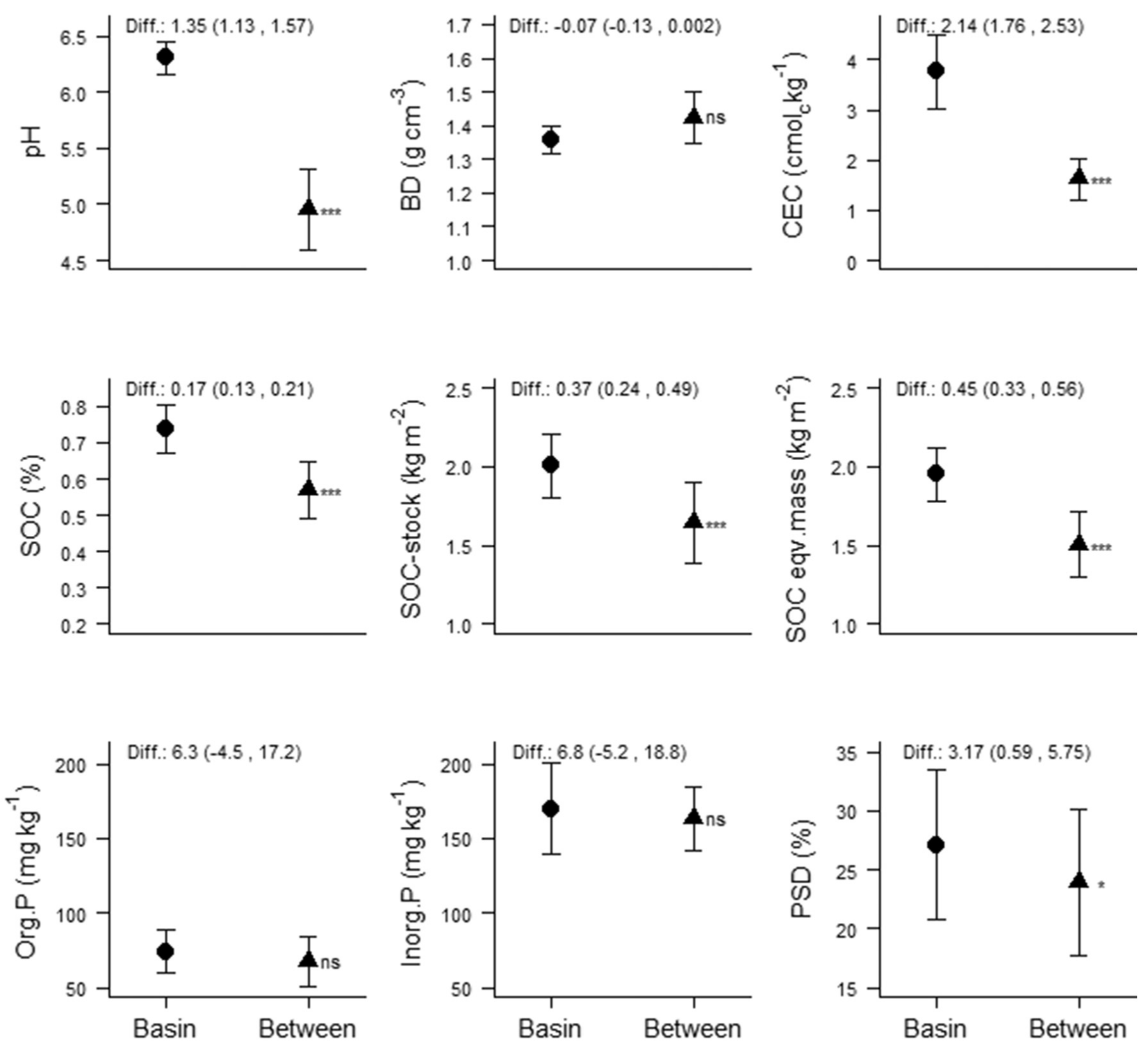


Fig. 2. Mean (± sd) soil properties (0–20 cm) of CF basins (inside) and in between rows of CF basins (outside), Mkushi, Zambia. Difference between means and 95% confidence intervals of the estimated differences are shown. N = 12 for both CF basins and between rows of CF basins except for bulk density (BD, n = 7). "ns"; p > 0.05, "*";p < 0.05, "**";p < 0.01 and "***";p < 0.001 based on two-sided paired *t*-tests. BD is bulk density, CEC is cation exchange capacity, SOC is soil organic carbon, Org. P and Inorg. P are organic and inorganic P, respectively. PSD is P saturation degree (see Appendix for calculation). The equivalent mass of soil is 265 kg soil (BD 1.33 g cm^{-3}, depth 20 cm at CF basin in block four).

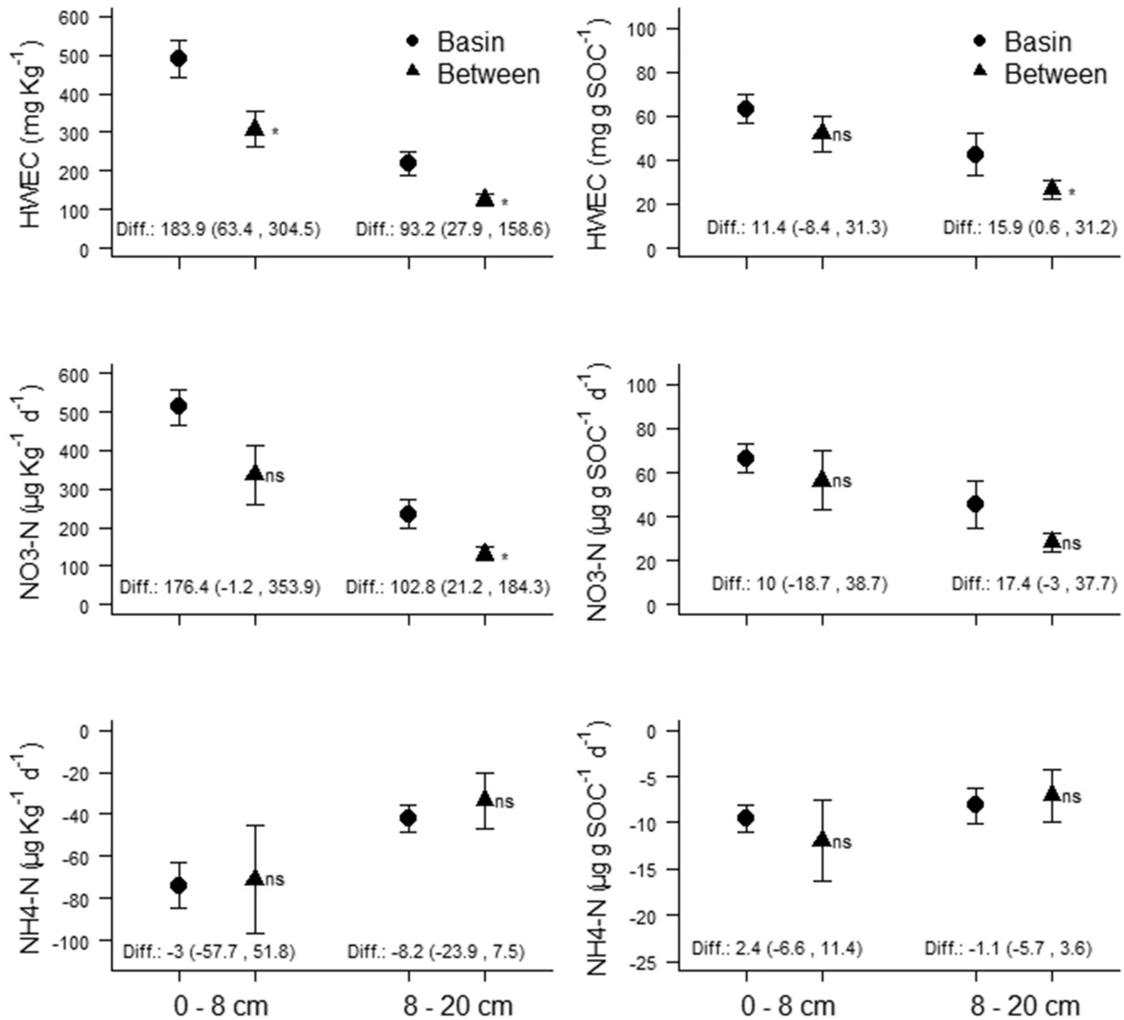


Fig. 3. Mean (± sd) hot water extractable carbon (HWEC, mg kg soil^{-1} and mg g SOC^{-1}) and net rates of NO_3-N and NH_4-N production (µg kg soil^{-1} day^{-1} and µg g SOC^{-1} day^{-1}) during 60 days of incubation of soils (0–8 cm and 8–20 cm) from inside CF basins and in rows between CF basins (outside), Mkushi, Zambia. Difference between means and 95% confidence intervals of the estimated differences are shown. N = 4 for both CF basins and between rows of CF basins at each of the depths (total n = 16). Positive and negative rates of NO_3-N and NH_4-N indicate a net mobilization and net immobilization of N, respectively. "ns"; $p > 0.05$, "*"; $p < 0.05$, "**"; $p < 0.01$ and "***"; $p < 0.001$ based on two-sided paired *t*-tests between CF basins and between rows of CF basins for each soil depth separately.

for normalized values (i.e. CEC per g of SOC, HWEC in mg g SOC^{-1} and net rates of NO_3-N and NH_4-N in µg g SOC^{-1} day^{-1}), standard deviation (sd) was calculated as $sd = |R| * \sqrt{(sdX/X)^2 + (sdY/Y)^2}$. Here R is the normalized mean, sdX is standard deviation of SOC, X is mean of SOC, and sdY and Y and is standard deviation and mean of HWEC or net rates of NO_3-N and NH_4-N, respectively. Linear regression was used to assess the relationship between CEC, SOC and pH, and between net nitrification rate and HWEC. The statistical software package "R" version 3.4.4 (R-Core-Team, 2018) was used for all statistical analysis.

3. Results and discussion

3.1. Soil organic carbon

Seven years of strict CF husbandry including all three elements of CF and basins in fixed positions resulted in significant differences in properties of soil inside and outside planting basins (Table 1, Fig. 2). Concentrations of SOC were significantly greater inside than outside basins (0.74 ± 0.06% vs. 0.57 ± 0.08%, respectively). Although bulk density did not differ significantly (1.36 ± 0.04 g cm^{-3} vs. 1.42 ± 0.08 g cm^{-3} inside and outside basins, respectively), SOC stocks were significantly greater inside basins (2.01 ± 0.20 vs. 1.64 ± 0.26 kg m^{-2}). Also, if based on the equivalent mass of soil, the SOC stock was significantly greater inside than outside basin (Table 1).

After seven years of CF the SOC stock was 0.365 kg m^{-2} greater inside than outside basins (Fig. 2). This corresponds to an increase of 0.522 t C ha^{-1} yr^{-1}, assuming unchanged SOC stocks outside basins since adoption of CF. Based on the pool of SOC outside basins (16.4 t C ha^{-1}, Fig. 2) the relative increase in SOC in basins (viz. the upper 20 cm of the soil) was 31.8‰ per year. Since the fraction of the field occupied by basins per ha at the experimental farm was ~9.3%, the corrected absolute change in organic C-stock of the field was 3.65 ∗ 0.093 = 0.34 t C ha^{-1} after seven years. This corresponds to an absolute annual increase of 0.05 t C ha^{-1} yr^{-1} or a relative increase of 2.95‰ per year, which is smaller than aimed for in the four per mille initiative 4p1000.org/;(Minasny et al., 2017). Note that this increase is limited to the upper 20 cm of the soil and that the total increase in SOC stocks may have been a bit higher. Previously, smaller differences in SOC between inside and outside planting basins were reported (Martinsen et al., 2017; Nyamangara et al., 2013). As discussed by Martinsen et al. (2017) the small differences between inside and outside basins may in part be due to large variability between study sites and year to year movement of basins. Yet, in the present study, with fixed basins and controlled conditions, the differences were significant. Clearly, SOC inside planting basins is affected by maize plants, in particular maize roots,

Table 1
Mean (± sd) soil properties (0–20 cm) of CF basins (inside) and in between rows of CF basins (outside basins), Mkushi, Zambia after 7 years of CF. Difference between means and 95% confidence intervals of the estimated differences are shown.

	Basin	Between	Diff.	95% CI
	Mean sd	Mean sd		
Clay/%	9.23 (±0.85)	9.48 (±1.04) ns	−0.25	(−1.04, 0.54)
Silt/%	21.45 (±3.11)	22.00 (±3.39) ns	−0.55	(−3.33, 2.23)
Sand/%	69.34 (±2.73)	68.53 (±2.93) ns	0.81	(−1.40, 3.02)
Total porosity[a]/%	51.42 (±1.32)	51.30 (±0.75) ns	0.11	(−0.98, 1.21)
FC (pF 2)[a]/Vol%	21.91 (±1.53)	22.18 (±1.68) ns	−0.27	(−1.81, 1.27)
WP (pF 4.2)[a]/Vol%	5.40 (±0.54)	5.10 (±0.58) ns	0.30	(−0.16, 0.76)
PAW[a]/Vol%	16.51 (±1.36)	17.08 (±1.55) ns	−0.57	(−1.93, 0.79)
Tot.N/%	0.03 (±0.01)	0.02 (±0.01) ns	0.01	(−0.002, 0.02)
Tot.N-stock (0–20 cm)/kg m^{-2}	0.08 (±0.04)	0.05 (±0.03) ns	0.02	(−0.01, 0.06)
Ca/$cmol_c$ kg^{-1}	2.49 (±0.56)	0.86 (±0.35) ***	1.63	(1.26, 2.01)
Mg/$cmol_c$ kg^{-1}	1.03 (±0.18)	0.34 (±0.12) ***	0.69	(0.54, 0.83)
K/$cmol_c$ kg^{-1}	0.19 (±0.03)	0.19 (±0.05) ns	−0.01	(−0.04, 0.02)
Na/$cmol_c$ kg^{-1}	0.05 (±0.07)	0.07 (±0.09) ns	−0.02	(−0.06, 0.03)
H/$cmol_c$ kg^{-1}	0.00 (±0.00)	0.16 (±0.34) ns	−0.16	(−0.37, 0.06)
Base saturation/%	100.00 (±0.00)	90.75 (±18.14) ns	9.25	(−2.28, 20.78)
Total P/mg kg^{-1}	243.98 (±31.67)	230.86 (±29.25) ns	13.13	(−4.12, 30.37)
Al (Ox)/mmol kg^{-1}	15.72 (±3.51)	17.95 (±4.83) ***	−2.23	(−3.31, −1.14)
Fe (Ox)/mmol kg^{-1}	6.94 (±0.96)	7.19 (±1.17) ns	−0.25	(−0.58, 0.07)
P (Ox)/mmol kg^{-1}	3.06 (±0.81)	2.94 (±0.71) ns	0.11	(−0.16, 0.39)
PSC/mmol kg^{-1}	11.33 (±2.14)	12.57 (±2.95) **	−1.24	(−1.92, −0.56)

[a] N = 12 for both CF basins and between CF basins except for total porosity, volume % water at field capacity (FC) and wilting point (WP) and amount of plant available water (PAW) where n = 7 for each. Fe (Ox), Al (Ox) and P (Ox) is oxalate extractable Fe, Al and P, respectively. PSC is phosphorus sorption capacity calculated according to Breeuwsma and Silva (1992), see Appendix. CEC is cation exchange capacity (unbuffered, sum of base cations and acidity). CN ratio was not calculated because Tot. N < 0.05%. "ns"; p > 0.05, "*"; p < 0.05, "**"; p < 0.01 and "***"; p < 0.001 based on two-sided paired *t*-tests.

and the micro-topography that favors the accumulation of runoff water (Obia et al., unpublished). The significant net increase of SOC inside basins compared with outside basins thus indicates that these factors are more important than the input of crop residues, which were primarily added in between rows.

3.2. Hot water extractable carbon and potential N mineralization

The amount of hot water extractable carbon (HWEC), a measure of labile SOC that correlates with microbial biomass in soils with low content of SOC (Sparling et al., 1998; Wang and Wang, 2011), was consistently greater inside than outside basins (Fig. 3). The use of HWEC is a sensitive method to determine effects of changes in soil management on soil carbon (Ghani et al., 2003). Greater amounts of HWEC inside than outside basins indicate a greater pool of labile SOC inside basins that may be lost easily, due to decomposition and reduced C inputs (Chivenge et al., 2007; Six et al., 2002). The fraction of HWEC of SOC was 6.4 ± 0.7% and 5.2 ± 0.82% inside and outside basins, respectively, at 0–8 cm soil depth and 4.2 ± 0.9% and 2.7 ± 0.4% inside and outside basins, respectively, at 8–20 cm soil depth (Fig. 3). The differences of HWEC as a fraction of SOC between inside and outside basins were only statistically significant at 8–20 cm soil depth ($p = 0.045$). The HWEC fraction of SOC in the upper 8 cm of the Mkushi soil was at the high end of values found in a review by von Lützow et al. (2007), who reported the fraction of HWEC to vary between 1% and 5% of total SOC.

Rates of nitrification followed the same pattern as that of HWEC, with greater nitrification rates inside than outside basins (Fig. 3). However, the difference was only significant at 8–20 cm soil depth (235 ± 38 vs 133 ± 18 μg NO_3-N kg $soil^{-1}$ day^{-1}). Ammonium was immobilized throughout the incubation period at both soil depths (Fig. 3). Net potential nitrification rates were significantly correlated with HWEC (1.04 μg increase in NO_3-N d^{-1} per mg of HWEC, $R^2 = 96$, $p < 0.001$). Previously, Ghani et al. (2003) found a significant correlation between mineralizable N and HWEC in allophanic soils from New Zealand. Curtin et al. (2017) also reported a significant correlation between HWEC and mineralizable N in soil samples from 130 sites representing major agricultural regions of New Zealand.

3.3. Soil acidity

Soil pH was significantly higher inside basins than outside basins (6.30 ± 0.15 vs. 4.95± 0.37; $p < 0.05$). Marginally higher soil pH inside planting basins compared with outside basins was reported previously by Nyamangara et al. (2013) for smallholder farms in Zimbabwe. In the present study a total amount of 2.8 t ha^{-1} of dolomitic lime was added to the basins in years with maize, corresponding to ~11 g kg $soil^{-1}$ (i.e. an alkalinity of ~240 $mmol_c$ kg $soil^{-1}$). This is well in excess of the amount of exchangeable acidity outside the basins (1.6 $mmol_c$ kg^{-1}, Table 1) and shows that the rate of liming was enough to eliminate soil acidity inside the planting basins in addition to neutralizing the annual acid input, due to carbonic acid and cation uptake by plants (van Breemen et al., 1984).

3.4. Cation exchange capacity

The cation exchange capacity (CEC) was significantly greater inside (3.76 ± 0.73 $cmol_c$ kg^{-1}) than outside (1.62 ± 0.41 $cmol_c$ kg^{-1}) basins. Normalizing CEC per g of SOC also revealed significantly greater CEC inside (0.51 ± 0.11 $cmol_c$ g SOC^{-1}) than outside (0.29 ± 0.08 $cmol_c$ g SOC^{-1}) basins. The larger CEC inside basins is due to greater amounts of SOC, in addition to a higher pH (6.30 ± 0.15 vs. 4.95 ± 0.37), which causes an increase in the number of binding sites per g SOC (Gruba and Mulder, 2015). There was a significant linear relationship between CEC and amount of SOC (0.87 ± 0.20 and 0.26 ± 0.17 $cmol_c$ increase in CEC per g of SOC for inside and outside basins, respectively, $R^2 = 0.89$, $p < 0.001$) with a significantly ($p = 0.03$) greater slope (i.e. stronger increase in CEC per unit increase in SOC) inside than outside basins. The importance of soil organic matter controlling CEC in this sandy loam is similar to previously reported values (Martinsen et al. (2017) from different sites in the eastern and central provinces of Zambia (0.54–0.81 $cmol_c$ increase in CEC per g of SOC).

3.5. Phosphorus, potassium and nitrogen

The amount of total (Table 1), organic and inorganic P (Fig. 2) did not differ significantly between inside and outside basins. However, the P saturation degree (%PSD, Fig. 2) was significantly greater inside than outside basins mainly because of a greater P sorption capacity (due to more Al oxides and lower pH) outside basins (Table 1). This indicates a slightly higher availability of P inside as compared to outside basins. The amount of total N was low and often below detection limit (0.05% N). Low N status of agricultural soils is common in Sub-Saharan Africa (Martinsen et al., 2017; Mloza-Banda et al., 2016). Exchangeable potassium (K^+) was ~0.19 $cmol_c\ kg^{-1}$ both inside and outside the basins (Table 1). The non-significant differences in N, P and K between outside and inside the planting basins suggest that the amount of fertilizer added in years with maize (i.e. 8.1 g $basin^{-1}$, 1.3 g $basin^{-1}$ and 1.2 g $basin^{-1}$ of N, P and K, respectively) is about the same as the sum of the amount lost, including leaching (mostly N and K) and removal in biomass at harvest. Yields under CF for the season 2015–2016 (sampled May 2016) were 5.2 ($\pm$0.84 SD) t ha^{-1} and 4.6 ($\pm$0.44 SD) t ha^{-1} for maize grain and maize stover (stems and leaves), respectively. Assuming average grain yields of 5.2 t ha^{-1} and assuming that all stover (4.6 t ha^{-1}) is used as residue (i.e. returned to the soil between basins), the amount of NPK removed from basins in years with maize corresponded to ~5.7 g $basin^{-1}$, 1.1 g $basin^{-1}$ and 1.2 g $basin^{-1}$ of N, P and K, respectively (Table S1.). These numbers are close to those for annual inputs, when also taking into account some loss of N through leaching and gaseous emissions (McNeill and Unkovich, 2007).

3.6. Soil physical properties

The fraction of clay, silt and sand were similar inside and outside planting basins (Table 1). In addition, soil physical properties including texture, BD, soil porosity, percentage water at field capacity and wilting point as well as the amount of plant available water were similar to values reported in previous studies from the same area (Obia et al., 2017; Obia et al., 2016) and did not differ between inside and outside basins (Table 1, Fig. 2). The lack of significant differences in BD may have been due to i) soil heterogeneity such that the difference was rendered insignificant, ii) smaller differences in BD than expected, because increased termite activity under mulch in between the rows (Mutsamba et al., 2016) or iii) limited compaction outside basins in the oxide-rich Acrisols. The observed lack of effect on water retention characteristics is contrary to what would be expected of soil with increased soil organic matter (Obia et al., 2016). Despite being significant, the increase in SOC (from 0.57 to 0.74%) apparently has been too small to cause an increase in porosity, water content at field capacity and wilting point.

4. Conclusion

Seven years of CF, following recommended guidelines, using the same basin location each year, significantly increased storage of SOC inside planting basins. The increase in SOC was most likely caused by increased inputs of roots, due to favorable conditions for plant growth through increased water availability and input of fertilizer and lime. In addition, biogeochemical properties such as pH, CEC, HWEC and potential nitrification rates were significantly enhanced inside planting basins after seven years of strict CF husbandry. Our study highlights the important role of basins in build-up of SOC and underscores the importance of appropriate soil sampling schemes to account for the spatial variability between inside and outside basins when studying effects of CF vs. other management practices.

Acknowledgements

The study was funded by the Faculty of Environmental Sciences and Natural Resource Management at the Norwegian University of Life Sciences and by the R&D project "Soil Benefits Parameters Research, Conventional and Conservation Agriculture" funded by the Conservation Farming Unit (CFU) in Zambia with support from DFID. Thanks to Jeremy Selby for allowing access to his fields, for careful management of the research plots and for his hospitality. CENA farms and YARA are acknowledged for donation of fertilizers. Victor Shitumbanuma at the Department of Soil Science, School of Agriculture, University of Zambia is gratefully acknowledged for constructive inputs to the project. Magdalena Rygalska, Irene E. Dahl, Oddny Gimmingsrud, Valentina Zivanovic, Kelvin and Edward Bwalya are acknowledged for their technical assistance in the lab and in the field.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.12.452.

References

Abiven, S., Hund, A., Martinsen, V., Cornelissen, G., 2015. Biochar amendment increases maize root surface areas and branching: a shovelomics study in Zambia. Plant Soil 1–11.
Breeuwsma, A., Silva, S., 1992. Phosphorus Fertilization and Environmental Effects in the Netherlands and the Po Region (Italy). Rapport-DLO Staring Centre, Wageningen, p. 39 (57).
Bremmer, J.M., Mulvaney, C.S., 1982. Nitrogen-total. In: Page, A.L., Miller, R.H., Keeney, D.R. (Eds.), Methods of Soil Analysis Part 2 Agronomy 9. vol. 2. Am. Soc. of Agr., Madison, Wisconsin, USA, pp. 595–624.
CFU, 2011. In: Aagaard, P.J. (Ed.), The Practice of Conventional and Conservation Agriculture in East and Southern Africa. Conservation Farming Unit (CFU), pp. 1–80 conservationagriculture.org.
Cheesman, S., Thierfelder, C., Eash, N.S., Kassie, G.T., Frossard, E., 2016. Soil carbon stocks in conservation agriculture systems of Southern Africa. Soil Tillage Res. 156, 99–109.
Chivenge, P.P., Murwira, H.K., Giller, K.E., Mapfumo, P., Six, J., 2007. Long-term impact of reduced tillage and residue management on soil carbon stabilization: implications for conservation agriculture on contrasting soils. Soil Tillage Res. 94, 328–337.
Corbeels, M., Thierfelder, C., Rusinamhodzi, L., 2015. Conservation Agriculture in Sub-Saharan Africa. In: Farooq, M., Siddique, M.K.H. (Eds.), Conservation Agriculture. Springer International Publishing, Cham, pp. 443–476.
Corbeels, M., Cardinael, R., Naudin, K., Guibert, H., Torquebiau, E., 2018. The 4 per 1000 goal and soil carbon storage under agroforestry and conservation agriculture systems in sub-Saharan Africa. Soil Tillage Res. https://doi.org/10.1016/j.still.2018.02.015 (in press).
Curtin, D., Beare, M.H., Lehto, K., Tregurtha, C., Qiu, W., Tregurtha, R., et al., 2017. Rapid assays to predict Nitrogen mineralization capacity of agricultural soils. Soil Sci. Soc. Am. J. 81, 979–991.
Ellert, B.H., Bettany, J.R., 1995. Calculation of organic matter and nutrients stored in soils under contrasting management regimes. Can. J. Soil Sci. 75, 529–538.
Elonen, P., 1971. Particle-size analysis of soil. Acta Agralia Fennica 122, 1–122.
FAO, 2017a. Soil Organic Carbon: the hidden potential. Food and Agriculture Organization of the United Nations, Rome, Italy, p. 90.
FAO, 2017b. The future of food and agriculture – Trends and challenges. Rome.
Gatere, L., Lehmann, J., DeGloria, S., Hobbs, P., Delve, R., Travis, A., 2013. One size does not fit all: conservation farming success in Africa more dependent on management than on location. Agric. Ecosyst. Environ. 179, 200–207.
Ghani, A., Dexter, M., Perrott, K.W., 2003. Hot-water extractable carbon in soils: a sensitive measurement for determining impacts of fertilisation, grazing and cultivation. Soil Biol. Biochem. 35, 1231–1243.
Giller, K.E., Witter, E., Corbeels, M., Tittonell, P., 2009. Conservation agriculture and smallholder farming in Africa: the heretics' view. Field Crop Res. 114, 23–34.
Gruba, P., Mulder, J., 2015. Tree species affect cation exchange capacity (CEC) and cation binding properties of organic matter in acid forest soils. Sci. Total Environ. 511, 655–662.
Johansen, C., Haque, M.E., Bell, R.W., Thierfelder, C., Esdaile, R.J., 2012. Conservation agriculture for small holder rainfed farming: opportunities and constraints of new mechanized seeding systems. Field Crop Res. 132, 18–32.
Ku, H.H., 1966. Notes on the use of propagation of error formulas. J. Res. Natl. Bur. Stand. C. Eng. Instrument. 70C, 263–273.
Lal, R., 2004a. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627.
Lal, R., 2004b. Soil carbon sequestration to mitigate climate change. Geoderma 123, 1–22.
Lal, R., 2013. Food security in a changing climate. Ecohydrol. Hydrobiol. 13, 8–21.
Lal, R., 2015. Sequestering carbon and increasing productivity by conservation agriculture. J. Soil Water Conserv. 70, 55A–62A.
Lal, R., Reicosky, D.C., Hanson, J.D., 2007. Evolution of the plow over 10,000 years and the rationale for no-till farming. Soil Tillage Res. 93, 1–12.
Mafongoya, P., Rusinamhodzi, L., Siziba, S., Thierfelder, C., Mvumi, B.M., Nhau, B., et al., 2016. Maize productivity and profitability in conservation Agriculture systems across agro-ecological regions in Zimbabwe: a review of knowledge and practice. Agric. Ecosyst. Environ. 220, 211–225.
Martinsen, V., Shitumbanuma, V., Mulder, J., Ritz, C., Cornelissen, G., 2017. Effects of hand-hoe tilled conservation farming on soil quality and carbon stocks under on-farm conditions in Zambia. Agric. Ecosyst. Environ. 241, 168–178.
McNeill, A., Unkovich, M., 2007. The nitrogen cycle in terrestrial ecosystems. In: Marschner, P., Rengel, Z. (Eds.), Nutrient Cycling in Terrestrial Ecosystems. 10. Springer Berlin Heidelberg, pp. 37–64.

Minasny, B., Malone, B.P., McBratney, A.B., Angers, D.A., Arrouays, D., Chambers, A., et al., 2017. Soil carbon 4 per mille. Geoderma 292, 59–86.

Mloza-Banda, H.R., Makwiza, C.N., Mloza-Banda, M.L., 2016. Soil properties after conversion to conservation agriculture from ridge tillage in Southern Malawi. J. Arid Environ. 127, 7–16.

Møberg, J.P., Petersen, L., Borggaard, O.K., 1990. Geologi og jordbundslære - pedologiske analyser (English: Geology and Pedology - Pedological Analysis). Den kongelige veterinær- og landbohøjskole, København.

Mutsamba, E.F., Nyagumbo, I., Mafongoya, P., 2016. Termite prevalence and crop lodging under conservation agriculture in sub-humid Zimbabwe. Crop Prot. 82, 60–64.

Nelson, D.W., Sommers, L.E., 1982. Total carbon, organic carbon and organic matter. In: Page, A.L., Miller, R.H., Keeney, D.R. (Eds.), Methods of Soil Analysis Part 2 Agronomy 9. vol. 2. American Society of Agronomy, Madison, Wisconsin, USA, pp. 539–579.

Nyamangara, J., Masvaya, E.N., Tirivavi, R., Nyengerai, K., 2013. Effect of hand-hoe based conservation agriculture on soil fertility and maize yield in selected smallholder areas in Zimbabwe. Soil Tillage Res. 126, 19–25.

Nyamangara, J., Marondedze, A., Masvaya, E.N., Mawodza, T., Nyawasha, R., Nyengerai, K., et al., 2014. Influence of basin-based conservation agriculture on selected soil quality parameters under smallholder farming in Zimbabwe. Soil Use Manag. 30, 550–559.

Obia, A., Mulder, J., Martinsen, V., Cornelissen, G., Børresen, T., 2016. In situ effects of biochar on aggregation, water retention and porosity in light-textured tropical soils. Soil Tillage Res. 155, 35–44.

Obia, A., Børresen, T., Martinsen, V., Cornelissen, G., Mulder, J., 2017. Effect of biochar on crust formation, penetration resistance and hydraulic properties of two coarse-textured tropical soils. Soil Tillage Res. 170, 114–121.

Obia, A., Cornelissen, G., Martinsen, V., Mulder, J., 2019. Conservation tillage and biochar improve soil water content and moderate soil temperature in a tropical Acrisol. Soil Tillage Res. (unpublished).

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G.P., Smith, P., 2016. Climate-smart soils. Nature 532, 49–57.

Pisante, M., Stagnari, F., Acutis, M., Bindi, M., Brilli, L., Di Stefano, V., et al., 2015. Conservation Agriculture and climate change. In: Farooq, M., MKH, Siddique (Eds.), Conservation Agriculture. Springer International Publishing, Cham, pp. 579–620.

Pittelkow, C.M., Liang, X., Linquist, B.A., van Groenigen, K.J., Lee, J., Lundy, M.E., et al., 2015. Productivity limits and potentials of the principles of conservation agriculture. Nature 517, 365–368.

Poulton, P., Johnston, J., Macdonald, A., White, R., Powlson, D., 2018. Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom. Glob. Chang. Biol. 24, 2563–2584.

Powlson, D.S., Stirling, C.M., Jat, M.L., Gerard, B.G., Palm, C.A., Sanchez, P.A., et al., 2014. Limited potential of no-till agriculture for climate change mitigation. Nat. Clim. Chang. 4, 678–683.

Powlson, D.S., Stirling, C.M., Thierfelder, C., White, R.P., Jat, M.L., 2016. Does conservation agriculture deliver climate change mitigation through soil carbon sequestration in tropical agro-ecosystems? Agric. Ecosyst. Environ. 220, 164–174.

R-Core-Team, 2018. R: A Language and Environment for Statistical Computing Version 3.4.4. R Foundation for Statistical Computing, Vienna, Austria.

Schollenberger, C.J., Simon, R.H., 1945. Determination of exchange capacity and exchangeable bases in soil - ammonium acetate method. Soil Sci. 59, 13–24.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. Plant Soil 241, 155–176.

Sommer, R., Paul, B.K., Mukalama, J., Kihara, J., 2018. Reducing losses but failing to sequester carbon in soils – the case of conservation agriculture and integrated soil fertility management in the humid tropical agro-ecosystem of Western Kenya. Agric. Ecosyst. Environ. 254, 82–91.

Sparling, G., Vojvodić-Vuković, M., Schipper, L.A., 1998. Hot-water-soluble C as a simple measure of labile soil organic matter: the relationship with microbial biomass C. Soil Biol. Biochem. 30, 1469–1472.

Steward, P.R., Dougill, A.J., Thierfelder, C., Pittelkow, C.M., Stringer, L.C., Kudzala, M., et al., 2018. The adaptive capacity of maize-based conservation agriculture systems to climate stress in tropical and subtropical environments: a meta-regression of yields. Agric. Ecosyst. Environ. 251, 194–202.

Thierfelder, C., Wall, P.C., 2012. Effects of conservation agriculture on soil quality and productivity in contrasting agro-ecological environments of Zimbabwe. Soil Use Manag. 28, 209–220.

Thierfelder, C., Mwila, M., Rusinamhodzi, L., 2013. Conservation agriculture in eastern and southern provinces of Zambia: long-term effects on soil quality and maize productivity. Soil Tillage Res. 126, 246–258.

Thierfelder, C., Rusinamhodzi, L., Ngwira, A.R., Mupangwa, W., Nyagumbo, I., Kassie, G.T., et al., 2015. Conservation agriculture in southern Africa: advances in knowledge. Renewable Agric. Food Syst. 30, 328–348.

Thierfelder, C., Matemba-Mutasa, R., Bunderson, W.T., Mutenje, M., Nyagumbo, I., Mupangwa, W., 2016. Evaluating manual conservation agriculture systems in southern Africa. Agric. Ecosyst. Environ. 222, 112–124.

Thierfelder, C., Chivenge, P., Mupangwa, W., Rosenstock, T.S., Lamanna, C., Eyre, J.X., 2017. How climate-smart is conservation agriculture (CA)? – its potential to deliver on adaptation, mitigation and productivity on smallholder farms in southern Africa. Food Sec. 1–24.

Umar, B.B., Aune, J.B., Johnsen, F.H., Lungu, O.I., 2011. Options for improving smallholder conservation agriculture in Zambia. J. Agric. Sci. 3, 50–62.

van Breemen, N., Driscoll, C.T., Mulder, J., 1984. Acidic deposition and internal proton sources in acidification of soils and waters. Nature 307, 599–604.

van Reeuwijk, L.P., 1995. Acid oxalate extractable Fe, Al, Si. In: van Reeuwijk, L.P. (Ed.), Procedures for Soil Analysis, fifth edition International soil reference and information centre. FAO, Wageningen, pp. 12-5–12-6.

von Lützow, M., Kögel-Knabner, I., Ekschmitt, K., Flessa, H., Guggenberger, G., Matzner, E., et al., 2007. SOM fractionation methods: Relevance to functional pools and to stabilization mechanisms. Soil Biol. Biochem. 39, 2183–2207.

Wang, Q.K., Wang, S.L., 2011. Response of labile soil organic matter to changes in forest vegetation in subtropical regions. Appl. Soil Ecol. 47, 210–216.

Wendt, J.W., Hauser, S., 2013. An equivalent soil mass procedure for monitoring soil organic carbon in multiple soil layers. Eur. J. Soil Sci. 64, 58–65.