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1 **On the impact of grassland management on soil carbon stocks: a worldwide meta-**
2 **analysis**

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14

15 **Abstract**

16 Grasslands occupy 70% of whole agricultural land and hold significant amounts of carbon, a
17 key element in the regulation of Earth's soils fertility, biomass production and climate.
18 Previous work has shown that carbon stocks of grassland soils have been largely depleted
19 worldwide due to missuse or mismanagement but that shifts in management could also
20 potentially increase soil carbon stocks and mitigate against the degradation of natural
21 ecosystems. However, the existing literature points to large discrepancies in the impact of
22 grassland management practices on soil carbon, which the present study investigated. Here we
23 considered 235 experimental sites in 18 countries across the world where shifts in grassland
24 management involved different grazing strategies (free, F vs controlled, C; high, H vs low, L
25 density grazers), grazers exclusion (E), mowing (M) and burning (B). The best performing
26 practice was controlled grazing with high density of grazers (CHG) with an average soil
27 organic carbon content (SOC_C) increase of 21% and with 100 % of the studies pointing to a
28 SOC_C increase. This was followed by E (14.9%; 60%) and FLG (13.3%; 80%). On average,
29 burning grasslands, decreases SOC_C by 9.3% but 31% of the studies pointed to an increase,
30 thus indicating discrepancies in the impact of grassland management. CLG and mowing did
31 not significantly impact SOC_C. These results also indicated that B decreased SOC_C the most
32 under moist to humid climates (-10.9% vs -1.7% under arid to semi-arid), while that E was
33 only beneficial in arid to semi-arid grasslands. Adoption of rotational high-intensity grazing in
34 place of free grazing grasslands, should be seriously considered by policy and decision
35 makers to mitigate against climate change while fostering economic and social development.

36

37 **Key words:** soil organic carbon, rehabilitation, grazing, multiple soil classes, meta-analysis

38

39 **Abbreviations**

| | | |
|----|----------------------|---|
| 40 | B | Burning |
| 41 | BD | Soil bulk density |
| 42 | C | Carbon |
| 43 | C/N | carbon to nitrogen ratio |
| 44 | CHG | Controlled high grazing |
| 45 | CLG | Controlled low grazing |
| 46 | CV | Coefficient of variation |
| 47 | N | Sample size |
| 48 | E | Grazer exclusion |
| 49 | FLG | Free grazing with low intensity |
| 50 | FHG | Free grazing with high intensity |
| 51 | LAT | Latitude (°) |
| 52 | LONG | Longitude (°) |
| 53 | M | Mowing with residues retained |
| 54 | MAP | Mean annual precipitation (mm year ⁻¹) |
| 55 | MAT | Mean annual air temperature (°C year ⁻¹) |
| 56 | Max | Maximum |
| 57 | Min | Minimum |
| 58 | PC | Principal Component |
| 59 | PCA | Principal Component Analyses |
| 60 | Quart | Quartile |
| 61 | SEM | Standard error of mean |
| 62 | SOC _C | Change in soil organic carbon content (%) |
| 63 | SOC _S | Soil organic carbon stocks (kg C m ²) |
| 64 | SON _C | Soil organic nitrogen content (%) |
| 65 | Z | Altitude (masl) |
| 66 | ΔSOC _C | Change in soil organic carbon content (%) |
| 67 | ΔSOC _C >0 | Number of studies with positive change in soil organic carbon content (%) |
| 68 | ΔSON _C | Change in soil organic nitrogen content (%) |
| 69 | ΔBD | Change in bulk density (%) |
| 70 | ΔC/N | Change in carbon to nitrogen ratio (%) |
| 71 | | |

72 **1 Introduction**

73 In 2015, 192 countries ratified the COP21 in Paris to keep global temperature rise this century
74 well below 2 degrees Celsius above pre-industrial levels and to pursue efforts to limit further
75 increase of the temperature. At that occasion, transfer of atmospheric carbon (C) to soils, to
76 increase the soil organic C stocks, was suggested as an effective and low cost means to meet
77 the COP21 objective (Minasny et al. 2017). Spill over benefits such as improved soil health,
78 soil fertility and reduced soil erosion will accrue to potentiate food production (Lal 2016).

79 Grasslands, which are natural ecosystems with grass as the dominant vegetation, occupy
80 about 40% of the earth's land surface area and 70% of all agricultural land (FAO, 2015).
81 Despite being mostly found under the arid to semi-arid climates of Africa, Asia and America
82 where net primary production is significantly lower than the worldwide average (Abdalla et
83 al., 2018), grassland soils store about 10% of the global soil organic C (SOC) stocks, which is
84 nearly 50% more than is stored in forests worldwide (FAO 2007). Previous work established
85 that grassland soil C stocks are sensitive to changes in land use and management. Guo and
86 Gifford (2002) reported that converting grassland to cropland is detrimental to SOC stocks
87 with a worldwide average loss of 59 %. A comprehensive meta-analysis by Dlamini et al.
88 (2016), using 628 soil profiles, revealed that grassland degradation further declines SOC
89 stocks by a worldwide average of 9 % and that this decline was associated with a loss of basal
90 grass cover. Higher average SOC stock decline (-16%) was found under dry climates (<600
91 mm of rainfall per annum) than wet climates (-8% for >1000 mm yr⁻¹ zone).

92 It was also learned from Abdalla et al. (2018) that when grazing intensity rises above carrying
93 capacity, SOC stocks decrease at an average rate of -0.9% per year, which can be attributed to
94 a net loss of grass and/or inability of frequently grazed plants to build root C reserves, thus
95 unable to sustain atmospheric C allocation to soils (Savory and Parsons, 1980). On the other
96 hand, Conant et al. (2017) reported that only 49% of worldwide studies on improved

97 grassland management practices such as low stocking rates, exclusion of grazing livestock
98 and planned rotational grazing enhanced SOC stocks, thus pointing to the absence of
99 consensus on the grassland management strategies to promote for increasing SOC stocks.
100 There is also no consensus on the impact of practices that do not involve livestock such as
101 burning, which is a common practice for increasing fodder production and quality, whilst
102 avoiding bush encroachments (Tainton, 1999). For instance, Abdalla et al. (2016) reported
103 that SOC stocks increased from 1.4 to 1.6 kg C m⁻² (i.e. by 14%) in the 0-0.2 m soil depth of a
104 long-term (70 years) annual burning trial in sub-tropical humid South Africa. However,
105 Granged et al. (2011) reported a decrease of 35% for a site in Spain, and Nardoto and
106 Bustamante (2003) reported a 13% decline for another site in Brasil. Discrepancies on the
107 impact of burning also exist within single ecosystems. For example, Snayman (2002) reported
108 a decrease of -33% while Manson et al. (2000) reported an increase of +22% in SOC stocks of
109 a grasslands subjected to burning in savannah climate of South Africa.

110 Previous studies that quantified consequences of grassland degradation (Dlamini et al., 2016),
111 conversion of forest and/or cropland to grassland, burning, fertilization and reclamation of
112 grass (Conant et al., 2001; 2018), grazing intensity (Abdalla, 2018) on grassland SOC
113 demonstrated that SOC is very sensitive to changes in management practices. The studies also
114 showed large discrepancies in the impacts of these practices on SOC changes. For instance,
115 Conant et al. (2017) reported an average SOC increase of 0.28 Mg C ha⁻¹ yr⁻¹ following
116 adoption of improved grazing but with changes in SOC stocks varying between -30% at one
117 study site and 70% at another, which points to a need to understand the reasons behind such
118 discrepancies before suggesting adoption of any given practice at large scale. Investigation of
119 inconsistencies on the effects of a given practice to enhance grassland SOC stocks thus
120 becomes an important objective. Therefore, the main purpose of the current paper was to
121 investigate discrepancies in the effects of selected grassland management practices; burning,

122 low and high intensity grazing (free or controlled), mowing and livestock exclusion, on SOC
123 stocks.

124

125 **2 Materials and methods**

126 **2.1 Database preparation**

127 A comprehensive search was performed on the Web of Science to harvest published research
128 papers on grassland management practices and their impacts on soil carbon stocks and other
129 selected soil properties such as nitrogen content, bulk density and soil water content, between
130 January and July 2017. The key words used for the search from title, abstracts and keywords
131 included “grass” AND/OR “grassland” AND/OR “veld” AND/OR “burning of grass”,
132 AND/OR “livestock enclosure” AND/OR “livestock exclusion” AND/OR “grazing, intensity”
133 AND/OR “grazing density” AND/OR “soil carbon, nitrogen stocks”. All references in the
134 papers found were also assessed in order to increase the search. The search process identified
135 342 papers. Only studies reporting on results based on grassland management trials longer
136 than three years and including soil carbon concentration and stocks in their analyses were
137 selected. Sixty-two papers met the acceptance criteria. The 62 papers reported on trials
138 performed on 235 experimental sites in 18 countries across the world (Figure 1). Table 1
139 shows a summarised version of the database. The studies had to compare management
140 practices against free grazing as reference. Different management practices were considered
141 from free grazing at high intensity (FHG) and controlled grazing at low or high intensity
142 (CLG, CHG), burning (Burn), mowing (M) and cattle exclusion (E). The intervals between
143 burning incidences varied with study site and were recorded in the same way as the duration
144 of livestock enclosure, which also varied with experimental sites.

145

146 **2.2 Data extraction**

147 *Variables of interest*

148 Data on soil organic carbon content (SOC_C), soil organic nitrogen content (SOC_N), soil
149 organic carbon stocks (SOC_S) and soil bulk density (BD) were collected from the papers.
150 Here we used data from the shallowest available soil layer where most of soil carbon lies
151 (Dlamini et al., 2015), mostly from the 0-0.05 m layer but when not available to 0.05m the
152 data came to the shallowest depth available to a maximum of 0.3m (n=15). In order to
153 compare SOC_S from different thicknesses of the top-soil layer, the data were converted to soil
154 carbon stocks per m³ of soil using the depth of soil layer considered. Moreover, the data on
155 soil organic matter content were converted to SOC_C using the 1.72 ratio (Pribyl, 2010), a
156 conversion factor that assumes organic matter contains 58% organic carbon. Data for each
157 management were subtracted from reference treatment (free grazing at low density) and
158 expressed in gram or percent difference per year to remove the differences in length of trials.
159 Finally data of carbon stocks were also reported per cubic metre of soil to improve the
160 comparisons between layers of different thicknesses. When not available in the research
161 papers, data of BD were estimated using values of SOC_C and SOC_S. The missing SOC_C or
162 SOC_S values were estimated in a similar way, and when not possible the data points had no
163 information (see table 5 for number of data points).

164 Here we considered the treatment means, the sample sizes (i.e. number of replications), and
165 the standard deviation (SD) between replications. When not directly available these
166 parameters were either estimated from the replicates or recalculated such as for SD that was in
167 some cases obtained from standard error data [SE = SD (n^{-1/2})]. In case of the results were
168 only presented in the format of figures, which occurred for two studies, the data were
169 extracted through graph digitisation (use of “GetData Graph Digitizer 2.26”)

170

171 ***Secondary variables***

172 The environmental factors were related to site location (LONG: longitude, LAT: latitude and
173 Z: altitude), climate (MAP: mean annual precipitation, and MAT: mean annual air
174 temperature), thickness of the entire soil profile and soil texture (clay, silt and sand content) of
175 the upper layer. These variables were stratified into 3 classes (low, medium and high, Table
176 3). The soil order (WRB, 2016) at each data point was considered and the climates were
177 defined following Mathew et al. (2017).

178

179 **2.3 Statistical analysis**

180 The first step of the statistical analysis was to calculate descriptive statistics for the selected
181 variables (Tables 4 and 5). Meta-analysis was then applied to the data to quantify the
182 significance of the impact of the grassland management techniques under study on SOC_s. The
183 MetaEasy software (v1.0.5, Kontopantelis and Reeves 2009) was used to compute effect sizes
184 (a quantitative measure of the magnitude of the relationship between two variables, the larger
185 the effect size the stronger the correlation between the two variables) and standard errors (SE)
186 from each technique. MetaEasy allows the standardisation of effect size according to eight
187 possible methods based on the sample size and available information, and ultimately selects
188 the method that minimises the biases (Kontopantelis and Reeves 2009). Bootstrap resampling
189 was used to assign confidence intervals (CI) at 95%.

190 The multiple correlations between grassland management techniques and soil and
191 environmental conditions were further investigated using principal component analysis (PCA)
192 (Figure 3). PCAs convert non-linearly correlated factors and variables into linear
193 combinations called principal components (Jambu, 1991) and based on the assumption of
194 normality, which was not necessarily true with the study due to the disparate nature of the
195 data.

196

197 **3 Results**

198 **3.1 Variations of soil and environmental conditions across the data set**

199 Table 4 presents the descriptive statistics depicting variability of the environmental conditions
200 across the data set. The mean annual precipitation (MAP) was 711 mm yr⁻¹ with values
201 ranging from 100 mm yr⁻¹ in China (Rong et al., 2014) to 2326 mm yr⁻¹ in Brazil (Conant et
202 al., 2017). Mean MAT was 14.0°C with data varying from -8.5°C in Canada (Donkor et al.,
203 2001) to 32°C in Ethiopia (Kassahun et al., 2012). Altitude (mean of 956 m) ranged from 4 m
204 in Brasil (Conant et al., 2017) to 3500 m in China (Wei et al., 2011). Soil texture also varied
205 greatly across sites with clay content (mean, 24.9%) varying from 3.0% in China (Pei et al.
206 2007) to 74% in Brazil (Nardoto and Bustamante, 2003).

207

208 **3.2 Variations in soil carbon and nitrogen**

209 Soil carbon and nitrogen varied greatly within the data set (Table 5). The average soil organic
210 carbon content (SOC_C) computed from all reference treatments was 2.26% with a standard
211 error mean of ±0.19% (Table 5). Values varied from 0.06% in the USA under Aridisols and
212 6% clay content (Neff et al., 2005) to 20.16% under gleyic andosols in Japan (Shimoda and
213 Takahashi, 2009). The mean soil organic nitrogen (SON_C) at the reference sites was
214 0.19±0.02 with values ranging from a minimum of 0.01% also found by Neff et al., (2005) to
215 1.17% under Mollisols with 10% clay content in Canada (Evans et al 2012). The resultant
216 average carbon to nitrogen (C/N) ratio at these sites was 11.52±1.10.

217 Soil bulk density at reference sites averaged 1.22±0.02 g cm⁻³ and soil organic carbon stocks
218 (SOC_S) was 2.02±0.08 kg C m² with values between 0.27 to 17.80 kg C m² (Table 5).

219 On average, changes in grassland management decreased BD by 0.40±0.15% yr⁻¹, enhanced
220 SOC_C by 0.71±0.52% yr⁻¹, SON_C by 1.17±0.84% yr⁻¹ and SOC_S by 0.24±0.54% yr⁻¹ (Table 5).

221 The change in SOC_C (Δ SOC_C) varied from a -25.9% yr⁻¹ under free grazing with high density
222 grazers in Argentina (Abril and Bucher 1999) to 28.9% yr⁻¹ at a livestock enclosure site in
223 China (Yang et al., 2016).

224

225 **3.2 Effects of grassland management practice on soil carbon**

226 Soil carbon stocks were significantly affected by only one out of the six grassland management
227 techniques (Fig. 2A). Controlled grazing with high density grazers (CHG) significantly
228 enhanced soil carbon stocks (SOC_S) with an effect size of 0.477 and a top-soil SOC_S increase
229 by an average 5.9% yr⁻¹ or 94.5 g m⁻³ yr⁻¹ (Figure 2A). In contrast, under the conditions of the
230 study, the effect on SOC_S was neutral for enclosure (effect size = 0.004), mowing (0.009),
231 high density free grazing (-0.013), burning (-0.002) and controlled grazing with low density
232 grazers (-0.016).

233 Similarly, CHG was the only technique to significantly enhance soil carbon content (SOC_C)
234 with an effect size of 0.70 and a mean percentage change of 6.6% yr⁻¹ (Fig. 2B). Finally, for
235 top-soil bulk density (BD) the effect of grassland management (Fig. 2C) was significant for
236 burning (effect size of 0.071; 1.08% yr⁻¹ increase) and controlled grazing at low density
237 (0.074; 1.07% yr⁻¹).

238

239 **3.3 Effect of soil and environmental conditions on the changes in soil carbon**

240 Figure 3 shows the variations in soil carbon stocks (Δ SOC_S) following the changes in
241 management for different soil orders. The statistics for Mollisols were performed using 64
242 data points, Alfisols (60), Oxisols (43), Aridisols (24), Vertisols (17), Entisols (13), Ultisols
243 (2) and with 14 data points with undetermined soil order. The median stock difference was
244 with -10 gC m⁻³ y⁻¹ the lowest for Oxisols, followed by Aridisols and Mollisols (2.1 and 3.2),

245 Entisols (6.0), Alfisols (8.1), Vertisols (14.0) and Ultisols (85.2), all these differences being
246 not significant at $P < 0.05$ level.

247

248 As pointed out by figure 4, management duration did not have a significant impact ($P < 0.05$)
249 on the changes of soil organic carbon stocks with average carbon gains of $30.43 \text{ gC m}^{-3} \text{ y}^{-1}$ for
250 durations lower than 10 years, $5.59 \text{ gC m}^{-3} \text{ y}^{-1}$ for 10-20 years but -15.51 for 20-30 years and
251 $-0.62 \text{ gC m}^{-3} \text{ y}^{-1}$ for more than 30 years.

252 Figure 5 shows the multiple correlations between the selected variables of interest (ΔSOC_C ,
253 ΔSOC_S , ΔSON_C , $\Delta\text{C/N}$, ΔBD) on one hand, and soil and environmental factors on the other
254 hand. On figure 3A, focusing on all management techniques, the first two axes of the PCA
255 explained 94% of the data variability, with Axis 1 explaining 66% while Axis 2 accounted for
256 28%. In axis 1, ΔSOC_C , ΔSOC_S and ΔSON_C had positive coordinates while ΔBD had negative
257 coordinates, indicating a tendency for soil C and N to increase concomitantly, and to be
258 associated to a decrease in soil compaction. Figure 3A also shows a positive correlation
259 between ΔSOC_C and ΔSON_C on one hand and soil clay content on the other, meaning that
260 changes in soil C and N were the greatest at higher soil clay content ($r=0.67$, Table 6) but also
261 warmer climatic conditions ($r=0.58$ with MAT). Changes in the C/N ratio correlated the most
262 with SOC_C , pointing to an increase in C relative to N as soil carbon content increases ($r=0.62$,
263 Table 6).

264 The PCA generated from the grazing treatments only (Fig. 4B) pointed to a correlation
265 between MAT, MAP and clay content on the one hand and SOC_C , SOC_N and SOC_S in the
266 other hand, thus showing a tendency for soil C and N to increase the most under high clay,
267 temperature and rainfall conditions. From Table 7 we also learn that controlled grazing with
268 high density grazers (CHG) had a singular behaviour with the greatest SOC_S gains under arid
269 to humid and low clay conditions. Controlled high grazing (CHG) enhanced SOC_S the most
270 under moist-humid ($233 \text{ g m}^{-3} \text{ yr}^{-1}$ or $8.9 \% \text{ yr}^{-1}$) and arid to semi-arid ($15.6 \text{ g m}^{-3} \text{ yr}^{-1}$ or 1.9
271 $\% \text{ yr}^{-1}$) and as compared to wet climates ($0.5 \% \text{ yr}^{-1}$) and the increase was greater under low
272 clay conditions ($211.7 \text{ g m}^{-3} \text{ yr}^{-1}$ or $3.15 \% \text{ yr}^{-1}$ at $<20\%$ clay) (Table 7). CHG thus appeared

273 to induce highest SOC recovery under moist-humid and sandy soil conditions than under wet
274 and clayey ones.

275 In contrast, controlled low grazing (CLG) only slightly increased SOC_s under sandy and arid
276 conditions but these differences were not significant. CLG enhanced SOC_s by 3.1% yr⁻¹ under
277 the wet climate (Table 7).

278 Finally, grass burning significantly decreased SOC_s from Arid to humid climates but not for
279 wet ones (Table 8) and lessened SOC_s for all textures. The PCA of Fig. 5D showed that the
280 impact of grazer exclusion (E) on soil C and N did not correlate with climate. However,
281 Table 7 indicated SOC_s slight gains under arid to humid conditions and significant losses
282 under wet climate (-2.1% yr⁻¹).

283

284 **4 Discussion**

285 **4.1 The impact of grassland management on soil C and other selected soil properties**

286 From our dataset, controlled grazing at high density (CHG) was the only grassland
287 management technique that significantly increased top-soil organic carbon content (SOC_C)
288 and stocks (SOC_s) as compared to controlled grazing at low density (CLG), free grazing at
289 high density (FHG), grazer exclusion (E), mowing (M), and grass burning (B). CHG yielded
290 the highest soil C increase in the study by Gao et al. (2009) in China finding a 48% SOC_C rise,
291 Chaplot et al (2016), in South Africa found a 35-45% increase, Li et al (2007) in China (20%)
292 and Manley et al 1995 in the USA found 20%. From these studies we also learn that soil C
293 recovery is likely to be due to increase in soil surface coverage by grass (Chaplot et al. 2016)
294 that potentiates C inputs to the soil by plants. Greater soil coverage by grass also decreases
295 soil C erosion during rainstorms (Mchunu and Chaplot, 2012). C increase may also be due to
296 a greater proportion of N fixing grass species that helps building soil organic matter as
297 indicated by Gao et al., (2009). CHG also resulted in higher soil compaction than all reference

298 treatments, which might be attributed to soil trampling by livestock hooves. The associated
299 decrease in soil porosity lessens oxygen and CO₂ exchanges between the atmosphere and
300 soils, thus limiting organic matter decomposition and C fluxes out of soils as pointed out by
301 previous studies such as these by Nsabimana et al., (2004), Pandey et al. (2014) and Chaplot
302 et al. (2015).

303 Using data from many sites across the world Dlamini et al (2016) showed that CHG perform
304 best in terms of SOC_c under dry and sandy soil conditions, which may be another reason for
305 discrepancies. In fact, under these conditions that C losses due to degradation were also
306 highest (-16% for dry climates vs -8% for wet; -10% for sandy vs -1% for clayey) due to a
307 combination of harsh climate and low aggregate stability, which subsequently result in poor
308 soil organic matter protection. In addition to SOC_c, CHG also enhanced SON_c and increased
309 C/N ratios.

310 Grazer exclusion (E) had an insignificant impact on soil carbon stocks under arid to semi-arid
311 and moist to humid climates (Table 7) as well under coarse textured soils, but was detrimental
312 to soil C under wet climate and clayey soils. A possible explanation for the latter is likely to
313 be the low number of data points for E under wet climates (n=4) and clayey soils (n=4).

314 Additionally, E tended to enhance soil nitrogen content but its impacts on the C/N ratio and
315 on soil bulk density were insignificant. Such results can be attributed to dominance by
316 nitrogen-fixing grass species that help building soil C or the presence of C₃ or C₄ grass
317 species of different C behavior (Ritchie and Raina, 2016; Mahaney et al., 2008).

318 In general, fire had detrimental effects on SOC, irrespective of climate and soil texture
319 (except for moist-humid climate with a 3.7% increase), but the differences with free grazing
320 were not significant. The loss of soil C can be explained by high C losses in gaseous forms
321 during combustion of organic matter (Oluwole et al., 2008) and through preferential erosion
322 of black carbon formed during burning of the grass (Rumpel et al., 2006). The loss of soil C

323 lessens the stability of soil structure (Heydari et al., 2017), which in turn increases soil C
324 losses to the atmosphere as gases from organic matter decomposition and/or via erosion by
325 water. Burning can also decrease soil basal cover and thus lessen C inputs to soils. The gains
326 could be explained by removal of low productivity grass species and/or trees of lower soil C
327 input capacity.

328

329 **4.2 Recommendations for appropriate grassland management**

330 The study pointed to controlled grazing at high intensity for short duration as the best
331 performing grassland management practice worldwide with, however, greater benefits in Arid
332 to humid climates than in the wet regions of the world. This practice was first suggested by
333 Voisin (1957) following his work under temperate climate and reworked as holistic grassland
334 management by Savory and Parsons (1980) in the dry southern Africa. Controlled high
335 density grazing (CHG) proved to be more efficient than free grazing or rotational grazing for
336 forage production (higher quality and quantity) and for rehabilitation of degraded grasslands
337 where quasi-bare soils are a common feature (Savory and Parsons, 1980). In his work Voisin
338 (1957) demonstrated that grass grazed at 3-leaf stage was not only of higher nutritional value
339 but also recovered faster as enough energy was accumulated in its stem and roots. Grazing at
340 the 3-leaf stage allows more grass to be produced as flowering is prevented, which helps
341 keeping their rooting systems and their associated water and nutrient supply functions during
342 droughts for long, thus potentially explaining the higher increase in C content and stocks in
343 high intensity short duration grazing. Voisin (1957) also demonstrated in Western Europe that
344 a grazing period of two to three days every 21 days in spring during when grass growth rate is
345 faster and every 35 days afterward was optimal. This is probably due to development of
346 higher rooting systems that allocate C to soils for a longer period as exudates. The fact that
347 more forage is harvested by cows under CHG practices and further transformed into dung

348 which has high potential to become soil organic matter is another potential process for
349 building soil C.

350 This study showed the benefits of CHG were higher for dry and sandy environments where
351 vegetative growth is low because of scarce rains and high temperatures. Improving grazing
352 management in these areas is not only important to mitigate against climate change but also to
353 ensure food security in already highly vulnerable regions. However, CHG implies a drastic
354 shift in livestock management from traditional free grazing as it requires subdivisions of
355 entire grassing areas into paddocks of similar size which need to be fenced and supplied with
356 water. Since CHG requires more labour and investment for paddock construction, accurate
357 estimation of grass stages and biomass over time for grazing planning, and moving the grazers
358 from paddock to paddock the practice has mostly been applied to highly degraded grasslands.
359 Rehabilitation of degraded grasslands using livestock requires extreme care to avoid
360 unintended consequences such as increased soil bulk density and reduced water infiltration,
361 especially nearby watering points.

362 The study results further suggests the lack of impact of Soil Order on soil C recovery, which
363 was surprising, probably owing to the that different soil orders may exhibit similar soil
364 properties. Finally, fire as a method for managing grasslands did not appear to be beneficial
365 for building soil C stocks, irrespective of the soil and environmental conditions. The fact that
366 burning is less detrimental under dry and sandy conditions than under clayey and wet ones is
367 probably due to the fact that the soils are already highly C depleted.

368

369 **Conclusions**

370 The main conclusions from the current study performed using 235 grassland experimental
371 sites from 18 countries across the world were that (i) grassland soil C and N was confirmed to
372 be significantly affected by grassland management practice but that large discrepancies in

373 such effect existed from site to site; (ii) sandy soils in grasslands of arid to semi-arid and
374 moist-humid climates have greater potential for C and N recovery following changes in
375 management practices as than clayey soils of the tropics; (iii) the best performing grassland
376 management practice was controlled rotational grazing with high density grazers and short
377 duration which gave rise to average soil C stock increase of 5.9% yr⁻¹ (iv) excluding grazers
378 or burning grasslands were not beneficial for soil C with exclosure being highly detrimental
379 under wet and clayey soils. The current widespread use of these practices needs to be
380 reassessed.

381 These results improve our knowledge on the impact of grassland management on soil C and
382 the links to climate change and soil texture. Adoption of rotational high-density grazing for
383 short duration in place of free grazing grasslands, especially on degraded grasslands, should
384 be seriously considered by policy and decision makers to mitigate against climate change
385 while fostering economic and social development. This study benefitted from the existence of
386 numerous publications that could assist decision and policy makers on determining proper
387 grassland management techniques for rehabilitating grasslands across the world. However,
388 most of the studies only reported on soil carbon content, little on N content and even less on C
389 and N stocks. In addition, only 11 studies assessed the impact of controlled high density
390 grazing thus pointing to the need to coordinate research on key soil variables and on this very
391 promising practice across different environmental conditions in order to ascertain its
392 underlying mechanisms and associated ecosystem functions of food and water security. The
393 research to come should thus consider critical variables and routinely use consistent standards
394 and inform on durations of experiments.

395 Finally, the fact that none of the study techniques was truly beneficial in terms of soil C and N
396 in the wet climate points to the need to test other alternatives.

397

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401

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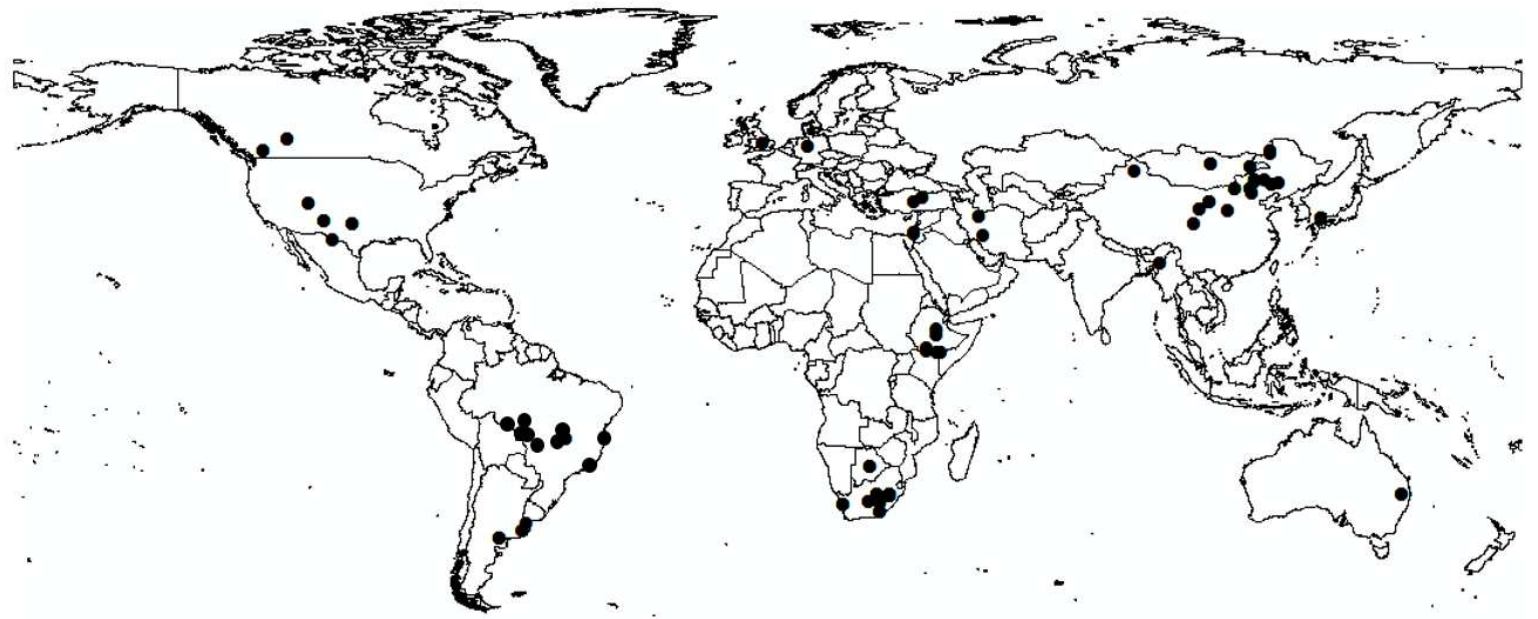


Figure 1 Global distribution of the sites where data used in the analysis

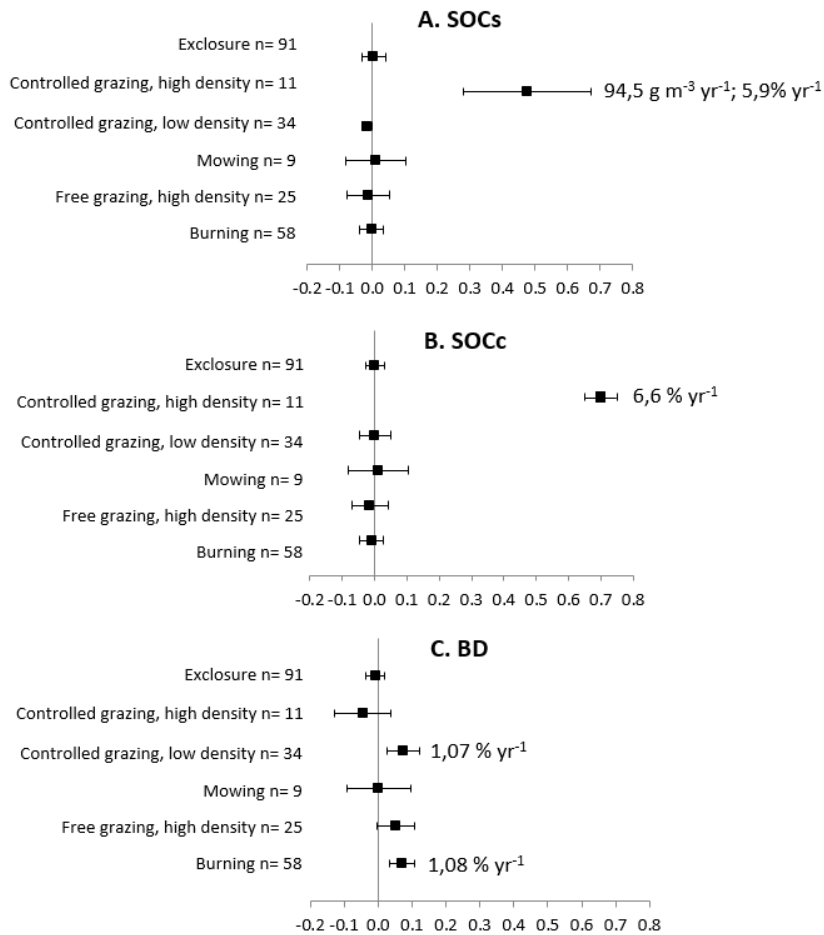


Figure 2. Soil response to selected grassland management practices as compared to free grazing at low density as control. A: Soil organic carbon stocks; B: Soil organic carbon content; C: soil bulk density. Error bars are effect size means \pm 95% bootstrap CIs. Where the CIs do not overlap the vertical $x=0$ line, the effect size for a practice is significant. n is the number of studies and the percentage corresponding to the bar indicates the mean percentage changes per year when changes are significant. Data for the surface soil layer are considered here.

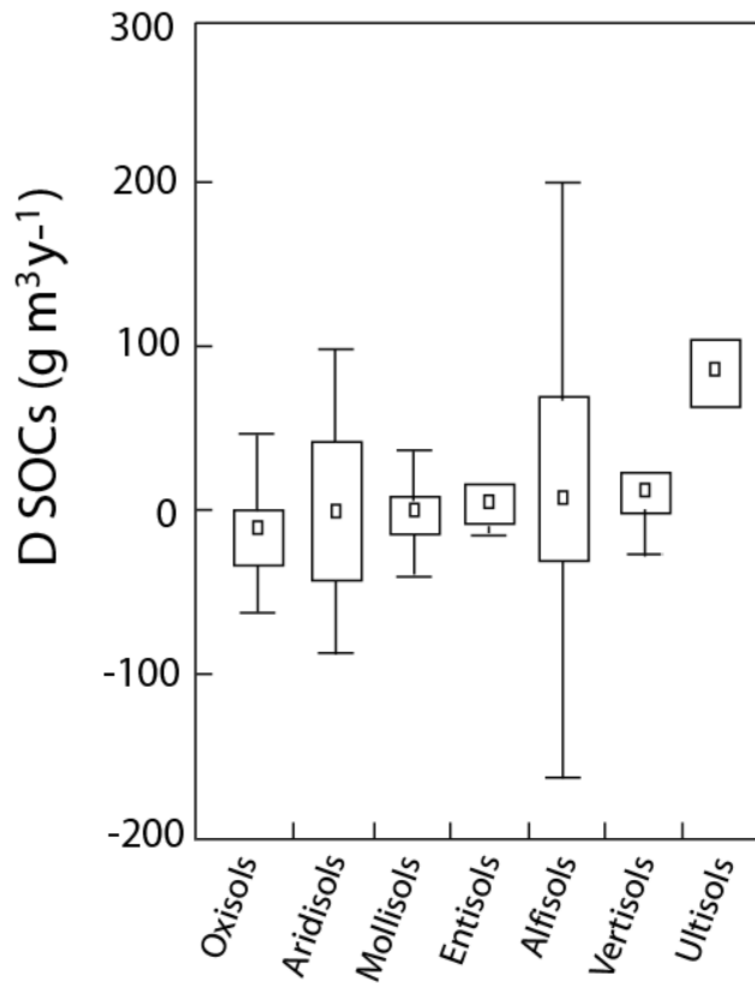
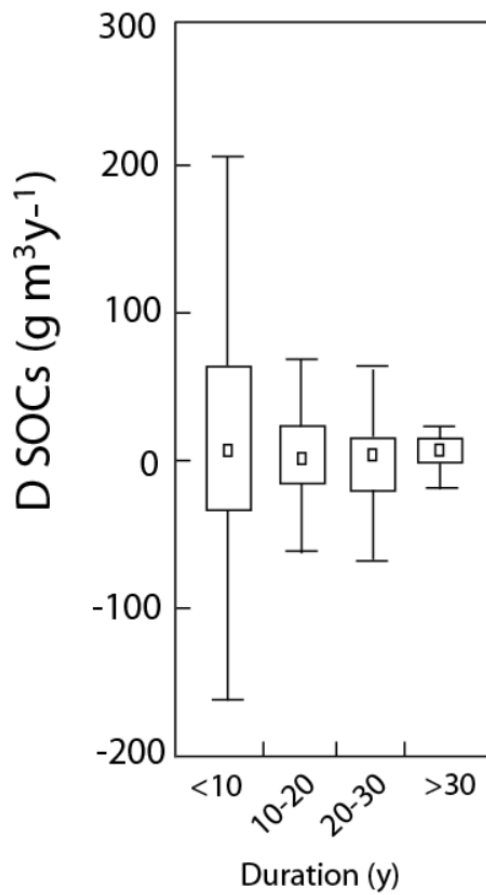


Figure 3. Impact of soil orders (WRB, 2015) on the changes of soil organic carbon stocks following rehabilitation. The general statistics are median, 1st and 3rd quartile, non-outlier minimum and maximum.



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- Figure 4. Impact of management duration (years) on the changes of soil organic carbon stocks following rehabilitation. The general statistics are median, 1st and 3rd quartile, non-outlier minimum and maximum.

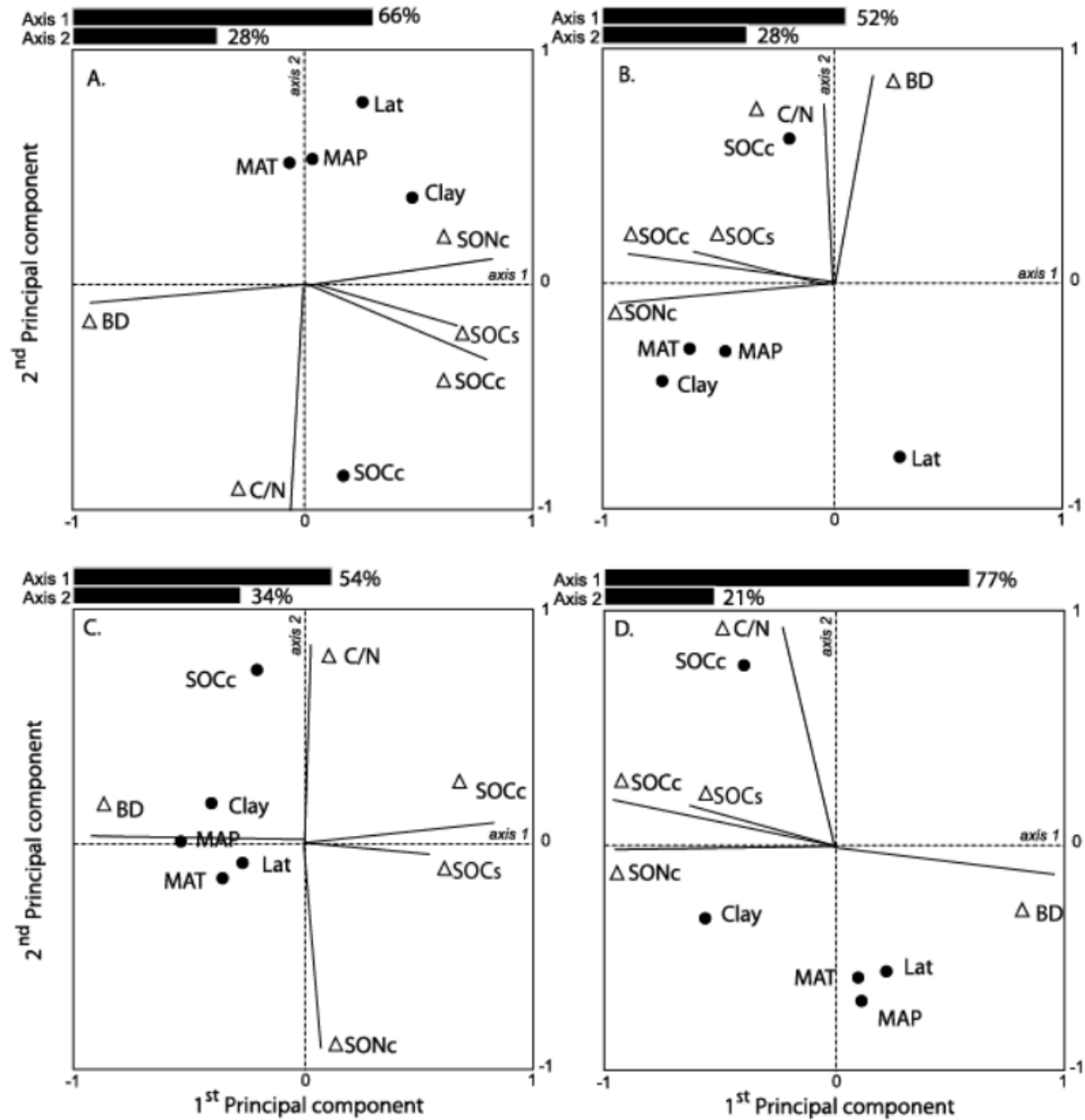


Figure 3 Principal component analysis for selected grassland management practices (all treatments, A; grazing treatments, B; Fire treatments; C Exclosure treatments, D) and showing the correlation between the changes in selected soil properties and soil and environmental variables.

Table 1 Compilation of references (authors and years), country of reference, investigated practices (B, burn; G, Grazing; E, enclosure), mean treatment duration and selected environmental factors (MAP, mean annual precipitation; MAT, mean annual temperature), soil type as provided by the authors and equivalent WRB (2016) soil order.

| n | Authors | Soil Type | Soil Order | Countries | Practices | Duration y ⁻¹ | MAP mm y ⁻¹ | MAT °C y ⁻¹ |
|----|-----------------------------|------------------------|------------|--------------|-----------|-----------------------------|---------------------------|---------------------------|
| 1 | Abril and Bucher 1999 | Aridic Haplustoll | Aridisols | Argentina | G | NA | 550 | 23 |
| 2 | Harris et al 2007 | Calcistolls | Mollisols | Argentina | E, B | 25,2 | 400 | 16 |
| 3 | Lavado et al 1996 | Typic Natraquoll | Mollisols | Argentina | G | 38,8 | 950 | 15 |
| 4 | Picone et al 2003 | Typic Natraquoll, | Mollisols | Argentina | B | 4,0 | 1029 | 15 |
| 5 | Sanjari et al 2008 | NA (clay to clay loam) | Alfisols | Australia | G, E | NA | 645 | 15 |
| 6 | Kgosikoma et al 2015 | NA loam to sand | Entisols | Botswana | G, B | 25,0 | 426 | 22 |
| 7 | Conant et al 2017 | NA | Oxisols | Brazil | G, E, B | 16,0 | 1616 | 23 |
| 8 | Nardoto and Bustamante 2003 | Oxisols | Oxisols | Brazil | B | NA | 1350 | 21 |
| 9 | Donkor et al 2001 | Luvicols | Alfisols | Canada | E | 1,0 | 334 | -8.5 |
| 10 | Evans et al 2012 | Chernozems | Mollisols | Canada | E | 0,5 | 270 | 15 |
| 11 | Krzic et al 2013 | Chernozems | Mollisols | Canada | G | 6,5 | 409 | 7.2 |
| 12 | Cao et al 2013 | NA | Mollisols | China | G, E | NA | 280 | 3.4 |
| 13 | Chen and Tang 2016 | Calcic Chernozem, | Mollisols | China | G, E | 1,3 | 279 | 2.4 |
| 14 | Gao et al 2009 | Mat Cry-gelic Cambisol | Mollisols | China | G | NA | 750 | 1.1 |
| 15 | Jiao et al 2016 | "diluvial desert soil" | Aridisols | China | G | 25,8 | 185 | 8.3 |
| 16 | Liu et al 2012 | NA | Mollisols | China | G, E | 13,0 | 308 | 2.1 |
| 17 | Pei et al 2007 | Typic Calciorthid | Mollisols | China | E | 13,5 | 134 | 9 |
| 18 | Rong et al 2014 | Calciorthid | Mollisols | China | G, E | 25,0 | 100 | 3.5 |
| 19 | Steffens et al 2008 | Calcic Chernozems | Mollisols | China | G, E | 17,0 | 343 | 0.7 |
| 20 | Su et al. 2003 | Sandy chestnut | Mollisols | China | G | 5,0 | 366 | 6.5 |
| 21 | Su et al 2004 | Sandy chestnut | Mollisols | China | E | 25,0 | 366 | 6.5 |
| 22 | Su et al 2005 | Sandy chestnut | Mollisols | China | E | 10,0 | 366 | 6.5 |
| 23 | Wang et al 2016 | Orthic Aridisol | Aridisols | China | G, E | NA | 400 | 1.5 |
| 24 | Wei et al 2011 | sub-alpine meadow | Entisols | China | G | 2,0 | 620 | 1.2 |
| 25 | Wu et al 2014 | Chernozem/chestnut | Mollisols | China | G | 25,0 | 339 | -2.2 |
| 26 | Xu et al 2011 | Calcic-orthic | Aridisols | China | G, E | 25,0 | 400 | 1.2 |
| 27 | Xu et al 2014 | Calcic-orthic | Aridisols | China | G, E | 25,0 | 400 | 1.5 |
| 28 | Yang et al 2016 | Chernozem | Mollisols | China | G | 12,0 | 378 | 7.3 |
| 29 | Zhao et al 2007 | Chernozium | Mollisols | China | G, E | 21,3 | 343 | 0.7 |
| 30 | Zhou et al 2010 | NA ("loess soils") | Mollisols | China | G, E | 7,0 | 505 | 8.8 |
| 31 | Kassahun et al 2012 | Vertisols | Vertisols | Ethiopia | G | 7,0 | 250 | 32 |
| 32 | Yusuf et al 2015 | Chromic Cambisol | Mollisols | Ethiopia | G, E | 3,0 | 550 | 20 |
| 33 | Nusse et al 2017 | Vertisol | Vertisols | Germany | G | 25,0 | 879 | 8.2 |
| 34 | Devi et al 2014 | NA (clayey loam) | Alfisols | India | G | 30,6 | 1522 | 19 |
| 35 | Zarekia et al 2012 | NA (sandy loam) | Entisols | Iran | G, E | 25,0 | 200 | 19 |
| 36 | Koyama et al 2015 | NA | Mollisols | Japan | B | 30,0 | 232 | 0.2 |
| 37 | Shimoda and Takahashi 2009 | Gleyic | Ultisols | Japan | G | 26,0 | 1724 | 16 |
| 38 | Barger et al 2004 | NA (coarse textured) | Mollisols | Mongolia | G | NA | 399 | 0 |
| 39 | Han et al 2008 | Chernozem | Mollisols | Mongolia | G | NA | 400 | -0.2 |
| 40 | Niu et al 2011 | calcisols | Mollisols | Mongolia | G, E | 21,8 | 210 | 8 |
| 41 | Al-Seekh et al 2009 | Rendzinas | Entisols | Palestine | E | 6,5 | 325 | 32 |
| 42 | Abdalla et al 2016 | Westleigh | Alfisols | South Africa | B | 42,5 | 694 | 16 |
| 43 | Chaplot et al 2016 | Acrisols | Alfisols | South Africa | G, E | 4,0 | 684 | 13 |
| 44 | Manson et al 2007 | Hutton | Alfisols | South Africa | B | 25,0 | 1380 | 25 |
| 45 | Materechera et al 1998 | Glenrosa | Alfisols | South Africa | B | 13,5 | 500 | 21 |
| 46 | Oluwole et al 2008 | Glenrosa | Alfisols | South Africa | G, B | 8,0 | 500 | 21 |
| 47 | Snayman 2002 | Bloemdal | Alfisols | South Africa | B | 1,0 | 560 | 25 |
| 48 | Oğuz, et al 2015 | NA | Aridisols | Turkey | G, E | 4,0 | 306 | 9.4 |
| 49 | Yalçın et al 2016 | Vertisol | Vertisols | Turkey | E | 10,0 | 797 | 14 |
| 50 | Ward et al 2016 | NA | Alfisols | UK | G | 25,0 | 885 | 9.7 |
| 51 | Bauer et al 1987 | Haploborolls | Mollisols | USA | E | 9,0 | 412 | 6.3 |
| 52 | Frank et al 1995 | Haploborolls | Mollisols | USA | G, E | 25,0 | 414 | 5.5 |
| 53 | Gardner 1950 | Whitehouse- | Entisols | USA | E | NA | 431 | 11.6 |
| 54 | Gill 2007 | Cryoborolls | Entisols | USA | E | NA | 932 | 1.1 |
| 55 | Manley et al 1995 | Ascalon/Altvan | Aridisols | USA | G | NA | 405 | 7.6 |
| 56 | Neff et al 2005 | Haplocambids | Aridisols | USA | G, E | 25 | 207 | 12 |
| 57 | Seithheko et al 1993 | Aridic | Aridisols | USA | G | 3 | 400 | 15 |
| 58 | Schuman et al 1999 | Argiustolls | Mollisols | USA | G | 8 | 387 | 8 |
| 59 | Teague et al 2010 | Tilvern and Wichita | Alfisols | USA | G, B | 7,5 | 648 | 17 |
| 60 | Weitz and Wood, 1986 | Leeray clay | Vertisols | USA | G | 25 | 361 | 9.9 |
| 61 | Wood and Blackburn 1984 | Leeray clay | Vertisols | USA | G | 10 | 624 | 17.2 |
| 62 | Nyamadzawo et al 2014 | Haplic Lixisols | Alfisols | Zimbabwe | G, B | 25 | 750 | 19 |

Table 2 Definitions of the environmental factors and soil properties used in the data-analysis

| Environmental factors, soil properties and techniques | Symbols | Units | Definitions |
|--|----------------|-----------------------|---|
| Mean annual precipitation | MAP | mm year ⁻¹ | Long-term (at least 30 year) mean annual precipitation for the study location from the papers |
| Mean annual temperature | MAT | °C | Long-term (at least 30 year) mean annual air temperature for the study location from the papers |
| Longitude | LONG | ° | Longitude of the midpoint of study site as given in paper |
| Latitude | LAT | ° | Latitude of the midpoint of study site as give in paper |
| Altitude | Z | m | Average elevation above sea level of the study site given in the papers |
| Clay content | Clay | % | Average clay content |
| Sand | Sand | % | Average sand content |

Table 3 List of controlling factor classes describing the environmental conditions used in the analyses.

| Environmental factors | Remarks | Class range | Name |
|--|----------------------|-------------------------------|-------------|
| Climate (MAP, mm year ⁻¹ ; MAT, °C year ⁻¹) | Warm and dry | MAT: 3.5-31.9 MAP 100-300 | Arid |
| | Cool and dry | MAT: -2.2-25.0 MAP:300-550 | Semi-arid |
| | Cool and moist | MAT: 1.2-18.6 MAP: 550-700 | Moist |
| | Warm and wet | MAT: 8.2-25.0 MAP:700-1350 | Humid |
| | Warm and wet | MAT:15.6-18.9 MAP: >1350 | Wet |
| Clay (%) | Average clay | 0-20 | Low |
| | | 20-40 | Medium |
| | | >40 | High |
| Sand (%) | Average sand content | 0-20 | Low |
| | | 20-40 | Medium |
| | | >40 | High |
| Grazing intensity: number of animal units stocked per unit area per year (AU ha ⁻¹ yr ⁻¹) | Light grazing | 0.4-2.5 | LG |
| | High density | >2.5 | HG |
| Livestock enclosure: period of time a plot has been excluded from grazing and/or burning (years) | | ≤10 | E≤10 |
| | | 10-20 | E10-20 |
| | | 20-30 | E20-30 |
| | | ≥30 | E≥30 |
| Burning frequencies: number of years from one incidence of burning grass to the other (years) | | 1-2 | B1-2 |
| | | 3-5 | B3-5 |
| | | >5 | B>5 |

Climate classes were adapted from Mathew et al. (2017); other factor classes were adapted from Mutema et al (2015).

Table 4 Overall descriptive statistics of the controlling factors selected for this study. Data from 235 experimental sites worldwide.

| | Mean | Media | Min | Max | Q1 | Q3 | CV | SEM | Skew | Kurt |
|-------------|-------------|--------------|------------|------------|-----------|-----------|-----------|------------|-------------|-------------|
| MAP | 711 | 494 | 100 | 2326 | 343 | 879 | 77 | 36,1 | 1.4 | 1.4 |
| MAT | 14.0 | 15.0 | -8.5 | 32.0 | 6.7 | 20.8 | 69.6 | 0.6 | 0.0 | 0.0 |
| Z | 956 | 1012 | 4 | 3500 | 418 | 1375 | 67 | 44.8 | 0.7 | 0.6 |
| Clay | 24.9 | 23.0 | 3.0 | 74.0 | 11.0 | 33.0 | 68.7 | 1.5 | 1.0 | 1.0 |

Mean; Median, Min=minimum value; Max=maximum value; Q1 and Q3=25th and 75th percentiles; CV=coefficient of variation; SEM=standard error of mean; Skew=skewness; Kurt=kurtosis; MAP=mean annual precipitation (mm yr⁻¹); MAT=mean annual air temperature (°C); Z=altitude (m); Clay= clay content (%).

Table 5 Overall descriptive statistics of the selected variables. Soil organic carbon content (SOC_C, %); soil organic nitrogen content (SON_C, %), Soil organic carbon stocks (SOC_s, kg C m³); soil bulk density (BD in g cm⁻³) ; variation in SOC_C, SON_C, SOC_s and BD (Δ SOC_C, Δ SON_C, Δ SOC_s and Δ BD in percentage difference per year from the reference (Ref) and the considered treatment (Treat), C/N ratio.

| | n | Mean | median | min | max | Q1 | Q3 | CV | SEM | Skew | Kurt |
|---|----------|-------------|---------------|------------|------------|-----------|-----------|-----------|------------|-------------|-------------|
| SOC_C Ref | 446 | 2.26 | 1.44 | 0.06 | 20.16 | 0.96 | 2.83 | 116 | 0.19 | 3.87 | 3.84 |
| SOC_C Treat | 446 | 2.34 | 1.42 | 0.12 | 26.04 | 0.95 | 2.53 | 135 | 0.22 | 4.75 | 4.72 |
| ΔSOC_C | 446 | 0.71 | 0.34 | -38.7 | 42.9 | -0.85 | 1.69 | 933 | 0.52 | 0.37 | 0.38 |
| SON_C Ref | 301 | 0.19 | 0.15 | 0.01 | 1.17 | 0.11 | 0.21 | 104 | 0.02 | 3.29 | 3.24 |
| SON_C Treat | 301 | 0.19 | 0.15 | 0.01 | 1.16 | 0.09 | 0.21 | 92 | 0.02 | 2,98 | 2.93 |
| ΔSON_C | 301 | 1.17 | 0.02 | -25.9 | 28.9 | -1.58 | 2.08 | 715 | 0.84 | 0.76 | 0.75 |
| SOC_s Ref | 367 | 3,02 | 1,93 | 0,08 | 12,70 | 1,34 | 4,27 | 86,96 | 0,17 | 1,72 | 1,71 |
| SOC_s Treat | 367 | 3,04 | 1,95 | 0,15 | 14,50 | 1,33 | 4,44 | 91,74 | 0,18 | 1,93 | 1,91 |
| ΔSOC_s | 367 | 0.24 | 0.089 | -40.6 | 41.3 | -1.05 | 0.98 | 748 | 0.54 | 0.08 | 0.08 |
| C/N Ref | 301 | 11.52 | 10.42 | 0.00 | 102 | 8.81 | 13.59 | 91 | 1.10 | 7.34 | 7.22 |
| C/N Treat | 301 | 11.72 | 10.10 | 0.00 | 103 | 8.26 | 14.37 | 92 | 1.14 | 6.78 | 6.67 |
| ΔC/N | 301 | -0.43 | -0.07 | -56 | 87 | -8.87 | 7.36 | 433 | 1.98 | 0.96 | 0.94 |
| BD Ref | 386 | 1.22 | 1.17 | 0.63 | 1.77 | 1.03 | 1.43 | 20 | 0.02 | 0.09 | 0.09 |
| BD Treat | 386 | 1.25 | 1.22 | 0.53 | 2.11 | 1.07 | 1.42 | 20 | 0.02 | -0.04 | -0.03 |
| ΔBD | 386 | -0.40 | -0,25 | -10,71 | 13,91 | -0,76 | 0,01 | 559 | 0,15 | 0,24 | 0,23 |

Table 6 Spearman rank correlation coefficients between environmental factors and the soil properties under different types of change in grassland management: grazing intensity, grazer exclusion (E), grass burning (B).

| | -----Grazing----- | | | | -----B----- | | | | -----E----- | | | |
|----------------------------|----------------------|----------------------|--------------------|-------------------|----------------------|----------------------|--------------------|-------------------|----------------------|----------------------|--------------------|-------------------|
| | ΔSOC_c | ΔSON_c | $\Delta\text{C/N}$ | ΔBD | ΔSOC_c | ΔSON_c | $\Delta\text{C/N}$ | ΔBD | ΔSOC_c | ΔSON_c | $\Delta\text{C/N}$ | ΔBD |
| MAP | 0.27* | -0.39* | 0.12 | -0.49* | 0.15 | -0.14 | 0.53* | -0.33* | 0.14 | -0.07 | 0.23* | 0.13 |
| MAT | 0.58* | 0.54* | 0.37* | -0.41* | 0.02 | -0.33* | 0.02 | -1.00* | -0.18 | 0.03 | 0.04 | 0.08 |
| LAT | -0.20 | 0.31* | -0.04 | 0.27* | -0.17 | -0.36* | 0.21* | -0.33* | 0.26* | 0.18 | 0.04 | -0.38* |
| Z | -0.06 | -0.19 | 0.35* | 0.04 | 0.02 | -0.02 | 0.29* | -0.33* | -0.15 | -0.12 | 0.03 | 0.31* |
| CLAY | 0.67* | 0.33* | 0.47* | 0.37* | -0.74* | -0.62* | 0.24* | nd | 0.10 | 0.43* | -0.45* | -0.32* |
| SOC_c ref | -0.05 | -0.11 | 0.62* | 0.23* | 0.29* | -0.30* | 0.41* | -1.00* | -0.26* | -0.18 | 0.08 | -0.18 |
| BD | 0.52* | 0.29* | -0.72* | -0.39* | 0.74* | 0.32* | 0.32* | 0.33* | -0.05 | 0.04 | -0.05 | 0.20 |

* Significant correlations at $p < 0.05$

nd not determined due lack of adequate data

MAP=mean annual precipitation (mm yr^{-1}); MAT=mean annual air temperature ($^{\circ}\text{C}$); LAT=latitude ($^{\circ}$); Z=altitude above sea level (m); Clay=clay content (%); reference organic carbon content (SOC_c ref); soil bulk density (BD, g cm^{-3}).

Table 7 Climate and soil clay content impact on the change in soil organic carbon stocks (Δ SOCs).

| <i>Climate</i> | CLG | | | CH | | | E | | Burn | | | n |
|---------------------|-------------------------|----------|----------|-------------------------|----------|----------|-------------------------|--------------------------|-------------|-------------------------|--------------------------|----------|
| | g m⁻¹ | % | n | g m⁻¹ | % | n | g m⁻³ | % yr⁻¹ | n | g m⁻³ | % yr⁻¹ | |
| Arid/semi-arid | 25.4 | 1.25 | 23 | 15.6* | 1.9* | 6 | 1.86 | 0.98 | 46 | -819.2* | -15.3* | 14 |
| Moist/humid | 65.3 | 1.98 | 26 | 233* | 8.9* | 3 | 38.7 | 3.8 | 20 | -143.8* | -9.1* | 13 |
| Wet | 47.9 | 3.12 | 245 | 6.2 | 0.5 | 1 | -34.4* | -2.1* | 4 | 63.73 | 0.24 | 31 |
| <i>Clay content</i> | | | | | | | | | | | | |
| 0-20 | 43.1 | 2.77 | 148 | 211.7* | 3.15* | 6 | 53.35 | 4.81 | 13 | -6.07 | -1.51 | 13 |
| 20-40 | 166.3* | 4.79* | 61 | nd | nd | 4 | -10.6 | -2.44 | 12 | -93.22 | -6.35 | 17 |
| >40 | 7.0 | 1.18 | 69 | 43.5 | 0.65 | 2 | -51.71 | -1.87 | 4 | -17.71 | -0.20 | 9 |

* Significant correlations at $p < 0.05$