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Overgrazing decreases soil organic carbon stocks the most under dry climates and low soil pH: A meta-analysis shows

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ABSTRACT

Grasslands occupy about 40% of the world's land surface and store approximately 10% of the global soil organic carbon (SOC) stock. This SOC pool, in which a larger proportion is held in the topsoil (0–0.3 m), is strongly influenced by grassland management. Despite this, it is not yet fully understood how grassland SOC stocks respond to degradation, particularly for the different environmental conditions found globally. The objective of this review was to elucidate the impact of grassland degradation on changes in SOC stocks and the main environmental controls, worldwide, as a prerequisite for rehabilitation. A comprehensive meta-analysis was conducted using 55 studies with 628 soil profiles under temperate, humid, sub-humid, tropical and semi-arid conditions, to compare SOC stocks in the topsoil of non-degraded and degraded grassland soils. Grassland degradation significantly reduced SOC stocks by 16% in dry climates (<600 mm) compared to 8% in wet climates (>1000 mm) and Asia was the most affected continent (−23.7%). Moreover, the depletion of SOC stock induced by degradation was more pronounced in sandy (<20% clay) soils with a high SOC depletion of 10% compared to 1% in clayey (≥32% clay) soils. Furthermore, grassland degradation significantly reduced SOC by 14% in acidic soils (pH ≤ 5), while SOC changes were negligible for higher pH. Assuming that 30% of grasslands worldwide are degraded, the amount of SOC likely to be lost would be 4.05 Gt C, with a 95% confidence between 1.8 and 6.3 Gt C (*i.e.* from 1.2 to 4.2% of the whole grassland soil stock). These results by pointing to greater SOC losses from grasslands under dry climates and sandy acidic soils allow identification of grassland soils for which SOC stocks are the most vulnerable, while also informing on rehabilitation measures.

1. Introduction

Soil is the third largest reservoir of carbon (C) next to the lithosphere and the oceans. Globally, soil contains about twice the amount of C in the atmosphere and more than three times in above-ground biomass (Batjes, 1996; Batjes and Sombroek, 1997; Jobbágy and Jackson, 2000). Historically, terrestrial C pools, have been largely depleted by anthropogenic activities such as deforestation, tillage and overgrazing (Lal, 2004). It has been widely argued that a shift in land use or land management in agroecosystems could potentially sequester as much as 30–40% organic C back into the soil (Lal, 2004). A meta-analysis of 74 studies by Guo and Gifford (2002) reported that conversion of croplands to grasslands could result to soil organic C (SOC) gains of 19%, while a global analysis of 115 studies by Conant et al. (2001) estimated much lower SOC gains varying from 3 to 5%. Under degraded croplands in the Highveld region of South Africa characterized by a temperate climate, with 6–8 months dry spells Preger et al. (2010) indicated that consideration of the initial level of degradation was important. In that study, they observed 30% SOC stock gains (*i.e.* 300 kg C ha^{-1} yr^{-1}) when less degraded croplands were converted to grasslands to as much as 70% (*i.e.* 500 kg C ha^{-1} yr^{-1}) for heavily degraded croplands. The variable response of SOC stocks to shifts in land use may be related to environmental factors including precipitation as shown by Guo and Gifford (2002), who reported greater SOC gains in areas with low mean annual precipitation (<500 mm) than in areas with greater mean annual precipitation (>500 mm).

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Indeed, is has been argued that because of such discrepancies in the results, the actual SOC sequestration potential of soil remains largely uncertain (Powlson et al., 2011).

Grasslands cover about 40% of the world's land surface (Suttie et al., 2005) and store approximately 10% of the global soil organic carbon (SOC) stock of 1500 Gt (West and Post, 2002). Grasslands are an essential component of the biogeochemical carbon cycle and provide key ecosystem goods and services (Suttie et al., 2005; FAO, 2010). Grasslands including both pastures and rangelands support biodiversity and are used extensively for the production of forage to sustain the world's livestock (Asner et al., 2004; Bradford and Thurow, 2006). Not only is the SOC pool in grassland soil critical for climate change, but it yields important feedbacks to soil fertility, plant productivity, soil aggregate stability, water holding capacity and overland flow regulation (Sombroek et al., 1993; Lal, 2004).

Bai et al. (2008) estimated that up to 25% of grasslands worldwide have been affected by grassland degradation. So far, the relative effect of degradation on SOC stocks has been difficult to predict because of the paucity of data (FAO, 2010), particularly in grassland soils. Because a greater proportion (ca 39–70%) of the SOC stock to 1 m depth is held in the top 0.3 m of the soil (Batjes, 1996), it has been postulated that grassland soils could be highly sensitive to, and strongly vulnerable to grassland degradation than previously thought.

However, grazing effects on SOC stocks have been found to be highly variable with some studies showing a decrease in SOC with grazing, for example, Martinsen et al. (2011) found that SOC stocks declined by 14% after 7 years of grazing in Norway, with 0.76 kg C m^{-2} in ungrazed compared to 0.64 kg C m^{-2} in heavily grazed grasslands. Steffens et al. (2008) found that 30 years of grazing in semi-arid Mongolian grasslands resulted in 45% decrease in SOC stocks, with 0.64 kg C m^{-2} in grazed compared to 1.17 kg C m^{-2} in ungrazed grasslands. Franzluebbers and Stuedmann (2009) observed that 56% of SOC stocks were lost after 12 years of grazing, with 0.051 kg C m^{-2} in heavily grazed compared with 0.117 kg C m^{-2} under ungrazed grasslands. In contrast, some studies have shown that grazing results in an increase in SOC stocks (Smoliak et al., 1972; Bauer et al., 1987; Frank et al., 1995; Derner et al., 1997), while others have reported no difference in SOC stocks after grazing (*e.g.* Johnston et al., 1971; Domaar et al., 1977). These contradictory findings demonstrate that the underlying processes affecting the response of SOC stocks to grassland degradation are not well understood. Thus, there is a need to study grassland degradation over a wide range of environmental and management conditions.

SOC losses due to degradation are dependent on soil texture. A number of studies have shown that fine-textured soils relatively have greater SOC stocks than coarse textured soils (Hassink, 1992, 1997; Bird et al., 2000; Brye and Kucharik, 2003). However, the effect of grassland degradation in soils differing in texture is largely unknown. In a previous study, Parton et al. (1987) using 560 soil profiles showed that SOC stocks and soil texture were correlated, with SOC stocks greater in fine textured soils than sandy textured soils. Across two degraded grassland soils with contrasting texture in the USA, Potter et al. (2001) found that grassland degradation reduced SOC stocks by 41% in coarse-textured than fine-textured soils. Clayey soils have a greater stabilizing influence on SOC than sandy soils, probably due to a large surface area, which form stable organo-mineral complexes that protect C from microbial decomposition (Feller and Beare, 1997; Six et al., 2000). Soil texture may interact and be confounded with other environmental factors such as climate, which may profoundly affect SOC depletion in grassland soils (Feller and Beare, 1997). Climate can impose constraints on the processes that control SOC stabilization, which may result in different changes of SOC under different environmental conditions (Virto et al., 2012). In a review of 12 studies totalling 22 data points, Conant and Paustian (2002) identified mean annual precipitation (MAP) as the main factor controlling C sequestration in degraded grasslands. Since the publication of that review, studies covering a wider range of environmental conditions have become available, thus allowing a robust evaluation of the impact of not only climate, but also, altitude, soil properties, time and grass type (C_3 *vs* C_4) and their likely interactions on SOC dynamics.

Because the response of soils to grassland degradation is expected to vary from site to site, the main objective of this review was to assess the level of SOC stock depletion in grassland soils worldwide and to identify the main environmental factors of

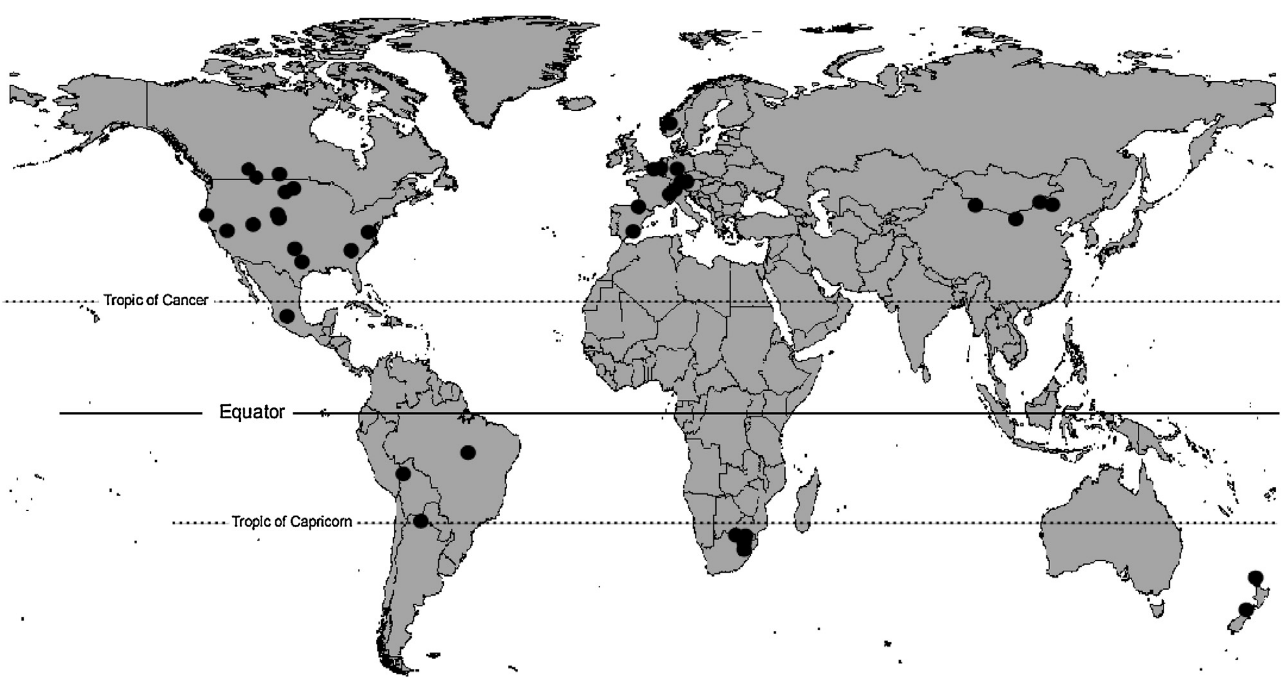

Fig. 1. Location of the study sites included in the literature review.

control. This study considered analytical data from 628 soil profiles gathered from 55 studies in temperate, humid, sub-humid, tropical and semi-arid grassland globally.

2. Materials and methods

2.1. Literature search and database construction

A literature search was conducted using online search engines (Google Scholar, ISI Web of Knowledge) and electronic bibliographic databases (Science Direct, Springerlink) and was focused on *in-situ* studies under grasslands (Fig. 1). The key words that were used to search the literature were; SOC; C sequestration; soil C storage; C depletion; grazing and grasslands. The studies (Table 1) had to meet specific criteria to be included in the data set are those that: (1) reported the concentration of SOC for a grazed grassland compared to a non-grazed grassland and the SOC was expressed as proportion of the total soil mass and the soil bulk density (ρ_b); (2) reported SOC stock; which is the quantity of carbon per unit area expressed in kg C m^{-2} or kg C ha^{-1} (Eq. (1));

Table 1
Compilation of references included in the database for analysis of the factors controlling SOC stocks in grassland soils up to 0.3 m.

Author (s).	Country	Long[a]	LAT	Z	n	MAP	MAT	SOC_C			SOC_S			SOC_D			Clay
		Degree		m.a.s.l		mm	mm	g C kg^{-1}			kg C m^{-2}			kg C m^{-3}			%
								Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	
Abril and Bucher (1999)	Argentina	−62.7	−23.3	217	3	550	22.7	33.8	22.5	45.0	6.8	5.5	8.2	34.2	27.5	41.0	17.6
Baisden and Amundson (2002)	USA	−117.9	37.9	1591	26	300	16.0	13.7	0.7	60.5	1.6	0.3	5.1	14.2	0.7	52.0	17.6
Bauer et al. (1987)	USA	−101.0	46.0	670	2	538	3.4	2.2	2.0	2.3	1.2	1.1	1.3	2.6	2.4	2.8	26.5
Bird et al. (2000)	Zimbabwe	28.3	-20.2	1400	9	630	17.7	12.9	3.3	45.4	1.1	0.3	2.0	16.4	4.9	40.0	20.0
Chuluun et al. (1999)	China	112.8	44.3	1165	8	307	2.6	61.1	7.8	82.6	3.9	0.5	5.3	65.4	8.3	88.3	10.3
Conant et al. (2003)	USA	−77.8	37.6	84	1	1075	13.5	6.5	4.8	8.8	4.4	3.2	6.0	8.8	6.4	11.9	10.3
Covaleda et al. (2011)	Mexico	−100.8	19.6	1674	14	844	16.8	48.6	48.6	48.6	2.4	2.4	2.4	24.3	24.3	24.3	34.8
Cui et al. (2005)	China	116.7	43.5	1255	21	350	0.2	12.4	7.6	16.7	3.9	3.5	4.6	14.3	8.8	19.4	21.0
Don et al. (2007)	Germany	10.4	51.0	267	48	600	8.0	11.3	0.9	44.2	1.6	0.5	2.9	15.7	1.2	57.4	26.8
Dong et al. (2012)	China	100.2	34.5	4200	6	570	−0.6	54.0	14.2	164.0	5.6	2.1	15.2	52.4	13.3	137.8	20.0
Frank et al. (1995)	USA	−99.2	46.8	573	59	404	4.4	22.0	11.4	36.1	4.7	3.3	6.1	29.7	15.4	48.7	10.0
Franzluebbers and Stuedmann (2009)	USA	−82.6	33.4	153	34	1250	16.5	13.2	4.3	24.1	3.3	1.8	4.9	18.2	6.1	32.8	10.0
Fynn et al. (2003)	South Africa	29.4	−29.6	2280	21	790	17.6	34.1	27.1	58.0	2.3	0.8	5.9	39.5	31.4	67.3	33.0
Ganjegunte et al. (2005)	USA	−104.0	41.1	1930	8	384	15.0	22.2	19.8	26.0	1.2	1.1	1.4	23.7	21.6	27.6	35.0
Garcia-Pausas et al. (2007)	Spain	−0.6	42.8	1461	14	1595	2.6	85.0	38.0	165.0	13.9	5.9	23.4	52.1	26.0	97.4	31.9
Gill (2007)	USA	−110.5	39.3	1600	5	932	1.3	33.4	16.4	53.3	5.4	3.3	10.5	35.7	21.7	69.8	24.7
Gill et al. (1999)	USA	−103.2	49.8	628	10	321	8.2	5.5	1.7	15.9	0.7	0.4	1.4	5.5	2.0	14.0	19.4
Hafner et al. (2012)	China	99.8	35.5	3440	1	582	1.7	28.0	10.1	62.1	3.1	2.1	4.2	27.2	11.3	52.2	25.0
Hassink (1997)	Netherlands	5.6	51.1	67	2	750	8.0	37.2	15.0	60.7	3.7	1.5	6.1	37.2	15.0	60.7	8.0
Hiltbrunner et al. (2012)	Switzerland	7.3	46.6	1600	63	1250	6.0	24.3	6.4	51.4	2.3	1.4	3.2	20.7	7.2	38.0	52.9
Ingram et al. (2008)	USA	−103.1	41.2	1930	2	425	15.0	14.5	7.7	26.1	1.6	1.1	2.2	17.8	11.1	27.6	10.0
Kaye et al. (2002)	USA	−103.1	40.7	1186	4	382	10.1	11.9	7.9	20.6	3.2	2.1	5.5	16.0	10.6	27.7	28.7
Leifeld and Kögel-Knabner (2005)	Germany	11.3	48.5	462	8	833	7.5	31.7	24.6	38.8	8.5	8.1	8.9	149.8	29.7	270.0	18.0
Maia et al. (2009)	Brazil	−49.7	-9.1	171	58	1950	15.1	13.0	4.1	30.0	3.9	1.4	7.5	14.9	4.7	34.2	18.0
Manley et al. (1995)	USA	-103.1	41.2	1930	3	384	13.0	14.6	14.0	15.1	5.7	5.7	5.7	18.9	18.9	19.0	10.0
Manson et al. (2009)	South Africa	29.3	-29.0	1841	9	1380	10.0	94.8	77.0	114.0	6.8	3.2	9.4	57.7	47.0	65.0	61.3
Martinsen et al. (2011)	Norway	7.9	60.8	1211	4	1000	−1.5	247.8	209.0	293.0	0.6	0.5	0.8	12.3	10.2	15.0	3.0
Masiello et al. (2004)	USA	−123.7	41.3	580	2	1000	12.0	48.8	3.4	82.1	12.5	2.3	31.8	53.1	4.7	82.1	17.6
Mchunu and Chaplot (2012)	South Africa	29.6	−26.7	1300	16	684	13.0	8.2	4.1	12.3	0.2	0.1	0.3	10.3	5.5	15.0	16.6
Medina-Roldán et al. (2012)	England	2.4	54.2	400	7	1840	2.8	193.2	163.6	222.9	6.1	5.9	6.2	30.3	29.5	31.2	10.0
Mestdagh et al. (2006)	Belgium	3.7	50.9	13	22	780	9.8	29.9	13.8	54.2	5.2	2.7	8.5	28.6	17.5	46.8	12.3
Mills and Fey (2004)	South Africa	29.5	−28.3	1718	30	1050	15.1	34.5	22.0	47.0	4.1	2.7	5.4	40.5	27.0	54.0	19.0
Mills et al. (2005)	South Africa	29.5	−28.3	1718	9	1050	15.1	34.5	22.0	47.0	4.1	2.7	5.4	40.5	27.0	54.0	19.0
Muñoz García and Faz Cano (2012)	Bolivia	−67.5	-13.3	167	12	505	4.5	52.7	30.0	91.7	1.8	0.5	3.8	32.6	5.0	76.0	16.0
Naeth et al. (1991)	Canada	−112.0	51.0	745	14	355	4.0	40.1	31.3	49.2	5.1	4.0	6.3	51.3	40.0	63.0	15.8
Neff et al. (2005)	USA	−109.9	38.3	1500	31	207	11.7	2.3	1.1	3.6	0.3	0.2	0.5	3.2	1.5	5.0	4.8
Percival et al. (2000)	New Zealand	172.5	-42.4	1365	6	1191	12.5	41.5	17.0	83.0	7.9	3.2	15.8	39.4	16.2	78.9	24.4
Piñeiro et al. (2009)	Uraguay	−56.9	-32.0	110	10	1100	17.3	23.6	15.4	30.3	9.2	6.0	11.8	30.7	20.0	39.3	25.5
Potter et al. (2001)	USA	−97.2	34.2	2438	4	842	17.0	11.2	4.7	27.1	1.9	0.9	6.2	6.3	2.9	20.5	23.3
Preger et al. (2010)	South Africa	27.2	-26.4	1456	5	641	15.5	14.5	6.0	29.7	1.6	0.9	2.2	18.3	9.0	41.6	19.3
Raiesi and Asadi (2006)	Iran	51.0	31.8	2500	4	860	6.7	21.1	19.4	23.6	7.0	6.4	8.0	23.4	21.3	26.7	50.0
Reeder and Schuman (2002)	USA	−104.0	41.1	1930	3	343	15.0	12.7	9.6	17.3	5.0	3.7	6.7	16.5	12.4	22.5	10.0
Schimel et al. (1985)	USA	−103.2	40.8	1238	4	310	8.5	5.9	0.8	20.4	1.2	0.5	2.1	8.0	1.2	26.4	23.2
Schipper et al. (2007)	New Zealand	175.3	-36.2	10	4	1266	12.6	121.2	62.6	204.2	23.0	11.9	38.8	115.1	59.5	194.0	24.2
Shi et al. (2012)	China	106.4	40.5	1038	2	353	−0.4	36.1	12.6	68.4	3.6	1.4	7.2	34.2	16.3	57.5	25.0
Sinoga et al. (2012)	Spain	−2.2	37.5	902	3	598	16.1	13.3	3.4	32.4	1.6	0.5	3.5	16.1	3.8	35.0	19.7
Smoliak et al. (1972)	Canada	−109.5	49.1	926	26	550	1.3	12.0	11.0	13.8	1.5	1.4	1.8	15.2	14.0	17.7	15.8
Steffens et al. (2008)	China	116.7	43.6	1270	2	343	0.7	24.1	17.0	31.0	1.0	0.9	1.2	25.9	21.5	28.8	15.0
Teague et al. (2011)	USA	−32.7	98.1	315	9	820	18.1	41.1	24.5	56.2	8.4	6.0	10.8	39.8	26.0	51.4	30.0
von Lützow et al. (2002)	Germany	11.3	48.5	462	8	803	7.4	31.7	28.0	38.0	9.4	8.3	11.3	47.2	41.7	56.6	14.0
Wiesmeier et al. (2012)	China	116.7	43.6	1260	1	350	0.7	17.0	13.7	21.3	1.9	1.7	2.0	18.8	17.5	20.0	20.4
Wood and Blackburn (1984)	USA	−98.6	34.0	316	14	624	17.0	33.1	23.0	45.0	1.5	1.2	1.8	51.1	41.4	61.2	30.0
Wu and Tiessen (2002)	China	102.8	37.2	2940	21	416	−0.3	67.7	37.0	85.0	7.4	4.1	9.5	49.3	27.0	63.2	27.3
Yong-Zhong et al. (2005)	China	120.7	43.0	360	48	366	6.5	2.4	2.1	2.8	0.5	0.5	0.6	3.3	2.9	3.7	2.5
Zimmermann et al. (2007)	Switzerland	8.5	46.2	1656	6	1337	4.5	35.4	27.3	43.4	10.5	8.1	12.8	52.3	40.4	64.2	26.0

[a] Long (longitude), Lat (latitude), Z (altitude above sea level), n (number of sites), MAP (Mean annual precipitation), MAT (mean annual temperature), SOC_C (soil organic carbon content), SOC_S (soil organic carbon stocks), SOC_D (soil organic carbon density), Clay (soil clay content).

(3) determined SOC by dry combustion using the C and N elemental analyzer.

$$SOC_S = SOC_C \times \rho_b \times T\left(1 - \frac{P}{100}\right) \quad (1)$$

where SOC_S is the soil organic carbon stock (kg C m^{-2}); SOC_C is the soil organic carbon concentration (g C kg^{-1}), ρ_b is the soil bulk density, T is the thickness of the soil layer (m); and P is the proportion of fragments of >2 mm, given as a percentage.

A quantitative database was then established in excel based on the published literature. The following characteristics were, when available, extracted from the research papers: mean altitude above sea level (Z), mean annual precipitation (MAP), mean annual temperature (MAT), and clay, silt, sand content in the topsoil (Clay, Silt, Sand). When not reported, the information on MAP and MAT was gathered from global assessments, such as the WORLDCLIM database with a spatial resolution of 30 arc seconds (approximately 1 km) (Hijmans et al., 2005). The global land cover database of the International Geosphere Biosphere Program at 1 km × 1 km spatial resolution was used for the classification of grassland areas globally (IGBP-DIS, 1998).

Environmental conditions were summarized by the number of categorical variables as described in Table 6: Soil texture was classified into three categorical textural classes: sand (<20% clay), loam (20–32% clay) and clay (>32% clay) based on the textural triangle (Shirazi and Boersma, 1984). Soil pH was classified into classes from strong acidic (≤5), weak acidic to weak alkaline (5–7) and strong alkaline (≥7). Following FAO guidelines for agro-climatic zoning (Fischer et al., 2001) MAP was divided into three precipitation classes: dry (<600 mm), intermediate (600–1000 mm) and wet (>1000 mm).

Given that various studies reported SOC for different soil surface layers (*e.g.* from 0.05 m, Dlamini et al., 2010; to 0.3 m, Maia et al., 2009), the preferred/recommended method for comparing SOC_S changes based on equivalent soil masses per area could not be used here. Hence the use of equivalent soil volumes to a fixed soil depth was selected with SOC_S data transformed into SOC densities (SOC_D, Sombroek et al., 1993; Batjes, 1996) (Eq. (2)).

The density of SOC in the topsoil was calculated as:

$$SOC_D = SOC_S \times \frac{1}{x_3} \quad (2)$$

where SOC_D is the SOC density (kg C m^{-3} up to 0.3 m depth).

The database on SOC stocks in grasslands contained 628 sites from 55 research studies across the world originating from semi-arid, temperate and tropical grassland environments. The geographical distribution of the data was as follows; North America (38%), Europe (22%), and only a few from Africa (16%), South America (7%) and New Zealand (4%). The geographic distribution of the study sites included in the review is depicted in Fig. 2. The database consisted of SOC_S, SOC_S, SOC_D, MAP, MAT, Z, LAT, LONG, ρ_b, SAND, SILT and CLAY. The grassland sites exhibited a wide range of environmental conditions (Table 2). Mean annual precipitation ranged from 240 to 2000 mm, with an average of 960 mm, while MAT ranged from −1.6 to 22.7 °C, with an average of 11.5 °C. Altitude ranged between 10 and 2721 m, with an average of 847 m. Soil texture exhibited substantial variations with CLAY

Table 2
Statistical summary of the site environmental characteristics.

	MAP[a] mm	MAT °C	*Z* m.a.s.l	Clay %	ρ_b g cm^{-3}	pH
Minimum	207	−1.6	10	2.0	0.14	3.84
Maximum	2000	22.7	4200	70.4	1.90	8.20
Mean	898	11.4	1248	21.9	1.16	6.37
Median	790	14.3	1365	18.9	1.16	6.50
Variance	248290	39	984789	127	0	1.07
Standard deviation	498.3	6.3	992.4	11.3	0.3	1.03
Skewness	1	−0.7	1	1.2	−0.52	−0.07
Quartile1	550	6.0	171	15.0	1.00	5.50
Quartile3	1149	17.0	1841	30.0	1.34	7.20
Kurtosis	0	−1.0	1	2.3	1.58	−1.00
CV	56	55	80	51	23	16.24
SE	19.9	0.3	39.6	0.4	0.0	0.04

[a] MAP (Mean annual precipitation), MAT (mean annual temperature), *Z* (altitude above sea level), Clay (soil clay content), ρ_b (soil bulk density), pH (soil pH).

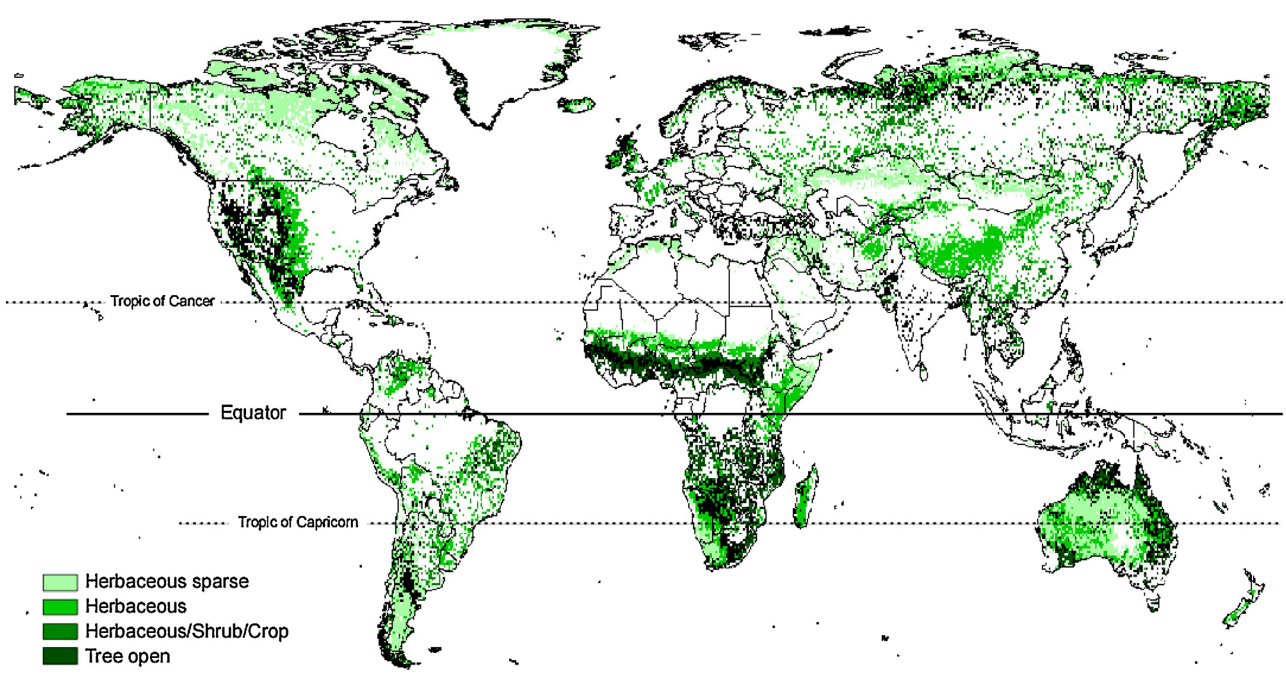

Fig. 2. The global extent of grasslands. The global land cover database of the International Geosphere Biosphere Program at 1 km × 1 km spatial resolution was used for the classification of grassland areas globally (IGBP-DIS, 1998).

ranging from 3 to 85%, while SILT and SAND ranged from 3 to 73% and from 2 to 81%, respectively.

2.2. Determination of grassland degradation impact on SOC stocks

Overgrazing is the main mechanism of grassland degradation and associated loss of SOC stocks (*e.g.* Dlamini et al., 2014) as it is detrimental to grass primary production and associated carbon inputs to soils and favor soil carbon erosion by wind and water. (*e.g.* Conant and Paustian, 2002).

Following Daily (1995), degraded grasslands exhibit a decline in grassland productivity, from moderate to substantial. Information on grass productivity and/or plant basal cover is not always available in research papers, which is not the case for grazing intensity or cattle stocking density. Here we used stocking density as indicators of grazing intensity and level of grassland degradation. Since recommendations for adequate stocking mainly depend on sites' primary production and soil characteristics (Trimble and Mendel, 1995), rates were not compared between studies. Heavy grazing was equated with low grass basal cover since heavy grazing implies grazing intensities greater than those recommended with detrimental impacts on grass coverage, (Bilotta et al., 2007). Three classes of grassland degradation were considered following the Global Assessment of Soil Degradation (GLASOD), each corresponding to a different proportion of plant basal cover, an easy to access information including using remote sensing (Seyler et al., 2001), from between 100–75% for "lightly-degraded", 75–50% ("moderately degraded") to below 25% for "heavy degraded". The losses of SOC_S (SOC_L) between the non-degraded and the degraded grassland situation was calculated as follows:

$$SOC_L = \frac{SOC_{D-ND} - SOC_{D-D}}{SOC_{D-ND}} \times 100 \quad (3)$$

where SOC_L is the difference (%) between the SOC_D in the upper soil layer to 0.3 m of the non-degraded grassland (SOC_{D-ND}) and of the most degraded grassland (SOC_{D-D}). We assumed that SOC_L can be regained with rehabilitation of degraded grassland.

Twenty eight studies from the 55 informed on the impact of grassland degradation on SOC stocks (Table 5). Since SOC_L can now be compared between different sites, a meta-analysis was performed, which included other environmental factors (mean annual precipitation, mean annual temperature, soil texture, grass type, soil pH and grazing intensity).

Table 3
Statistical summary of soil organic carbon content (SOC_C); soil organic carbon density in the non-degraded soil (SOC_{D-ND}), and degradation-induced soil organic carbon losses (SOC_L) from the global data set. Data from the upper soil layer up to 0.3 m depth.

	SOC_C g C kg^{-1}	SOC_{D-ND} kg C m^{-3}	SOC_L kg C m^{-3}
Minimum	0.7	2.8	−1.7
Maximum	293.0	137.8	124.5
Mean	34.9	34.4	17.8
Median	22.9	28.8	9.5
Variance	1602	774	697
Standard deviation	40.0	27.8	26.4
Skewness	2.9	2.1	2.9
Quartile1	11.0	17.6	3.5
Quartile3	42.6	48.7	24.8
Kurtosis	10.8	6.0	9.9
CV	115	81	148
SE	1.6	5.2	4.9

2.3. Statistical analysis

First, the basic statics was computed and this included minimum, maximum, mean, median, variance, standard deviation, skewness, 25th quartile and 75th percentiles, kurtosis, standard error (SE) and coefficient of variation (CV) (Tables 2 and 3). Second, a correlation matrix was applied to the data set to identify the Univariate relations between the SOC stocks in grassland soils and selected environmental factors (Table 4). ANOVA was then performed to test the significance of the environmental characteristics on SOC stocks. Third, a principal component analysis (PCA) was applied to the data to identify the multiple relationships between SOC stocks and the controlling environmental factors. A PCA is a statistical tool for data analysis, a dimensionality reduction technique that identifies structure in large sets of correlated multivariate data (Webster, 2001). Beyond that, multiple regression analysis was applied to the data to model and spatially display the influence of grassland degradation on SOC_D and SOC_{DC}.

2.4. Meta-analysis

The data was analyzed using MetaWin 2.1 software (Rosenburg et al., 2000). The meta-analysis was used to determine the mean effect of grassland degradation on SOC stocks in grassland soils. The natural log of the response ratio (InR) was used as an effect size for the meta-analysis. The natural log linearizes the metric by treating deviations in the numerator and denominator the same and also provides more normal sampling in small samples (Hedges et al., 1999). The response ratio was calculated as the ratio of SOC stocks between degraded and non-degraded grasslands using the following equation:

$$R = \frac{Xe}{Xc}$$

where Xe is the mean for the treatment and Xc is the mean of the control group (Rosenburg et al., 2000). A resampling based on 4999 bootstrap samples was used to generate the mean effect size of each categorical variable and 95% confidence intervals. The bootstrapping technique was used to generate confidence intervals on the mean effect size of the whole data set and for each categorical variable. The number of iterations used for bootstrapping was 4999 (Rosenburg et al., 2000). Grassland degradation effect on a response variable was considered significant if the 95% confidence interval did not overlap zero. The means of categories were considered significantly different if their 95 confidence intervals did not overlap 0 (Hedges et al., 1999). Meta-analysis was performed using a non-parametric weighting function and 95% confidence intervals (CIs) were generated using bootstrapping. Effect sizes were weighted by replication. For ease of interpretation the response ratio was transformed to percentage. These percentages represent the mean percentage change for a given site that has been degraded.

Table 4
Correlation matrix of soil organic carbon content (SOC_C), soil organic carbon stocks (SOC_S), soil organic carbon density (SOC_D) and selected environmental factors.

	Z	MAP	MAT	Clay	ρ_b	pH
SOC_C	0.33*	0.26*	−0.37*	−0.38*	−0.87*	−0.29
SOC_S	−0.36*	0.58*	0.36*	0.17*	0.17*	0.07
SOC_D	−0.17*	0.36*	0.17*	0.06	0.08	0.27*
SOC_L	−0.09	0.18*	0.29*	0.10	0.04	0.02

* Significant correlation at $P < 0.05$ level.

Table 5
Environmental characteristics and soil organic carbon in the top-soil up to 0.3 m of degraded and non-degraded grasslands.

Reference	Country	Location	n	T	MAP mm	MAT °C	Z m	Clay %	SOC_{D-ND} kg C m^{-3}	SOC_{D-D}
Abril and Bucher (1999)	Argentina	Salta	3	0.20	550	22.7	217	17.6	41.0	27.5
Bauer et al. (1987)	USA	North Dakota	2	0.46	538	3.4	670	26.5	2.8	2.4
Chuluun et al. (1999)	China	Mongolia	8	0.06	307	2.6	1165	10.3	88.3	8.3
Cui et al. (2005)	China	Inner Mongolia	21	0.30	350	0.2	1255	21.0	17.6	11.0
Dong et al. (2012)	China	Qinghai-Tibetan	6	0.13	570	−0.6	4200	20.0	137.8	13.3
Frank et al. (1995)	USA	Mandan, N.D	59	0.19	404	4.4	573	10.0	48.7	15.4
Franzluebbers and Stuedmann (2009)	USA	Georgia	34	0.15	1250	16.5	153	10.0	32.0	6.1
Ganjegunte et al. (2005)	USA	Cheyenne	8	0.05	384	15.0	1930	35.0	21.6	21.8
Gill (2007)	USA	Utah	5	0.15	932	1.3	1600	24.5	50.3	21.7
Hafner et al. (2012)	China	Qinghai-Tibetan	1	0.17	582	1.7	3440	25.0	41.0	11.3
Hiltbrunner et al. (2012)	Switzerland	Fribourg	63	0.15	1250	6.0	1600	52.9	32.0	7.2
Ingram et al. (2008)	USA	Cheyenne	2	0.10	425	15.0	1930	10.0	21.6	11.1
Manley et al. (1995)	USA	Cheyenne	3	0.30	384	13.0	1930	10.0	18.9	19.0
Martinsen et al. (2011)	Norway	Burskerud County	4	0.05	1000	−1.5	1211	3.0	13.8	10.2
Mchunu and Chaplot (2012)	South Africa	Bergville	3	0.02	684	13.0	1300	16.6	15.0	5.5
Medina-Roldán et al. (2012)	England	Yorkshire Dales	7	0.20	1840	2.8	400	10.0	29.5	31.2
Naeth et al. (1991)	Canada	Alberta	14	0.10	355	4.0	745	15.8	55.0	40.0
Neff et al. (2005)	USA	Utah	31	0.10	207	11.7	1500	4.8	5.0	1.5
Piñeiro et al. (2009)	Uraguay	Rio de la Plata	10	0.30	1100	17.3	110	25.5	36.3	23.0
Potter et al. (2001)	USA	Oklahoma	4	0.33	842	17.0	2438	23.3	13.8	3.7
Raiesi and Asadi (2006)	Iran	Shahrekord	4	0.30	860	6.7	2500	50.0	26.7	22.2
Reeder and Schuman (2002)	USA	Cheyenne	3	0.30	343	15.0	1930	10.0	19.4	12.5
Smoliak et al. (1972)	Canada	Alberta	26	0.10	550	1.3	926	15.8	14.0	14.4
Teague et al. (2011)	Texas	USA	9	0.23	820	18.1	315	30.0	50.6	26.0
Wiesmeier et al. (2012)	China	Inner Mongolia	1	0.10	350	0.7	1260	20.4	20.0	18.1
Wood and Blackburn (1984)	USA	Texas	14	0.03	624	17.0	316	30.0	55.9	46.8
Wu and Tiessen (2002)	China	Tianzhu	21	0.15	416	−0.3	2940	27.3	57.8	27.0
Yong-Zhong et al. (2005)	China	Naiman County	48	0.15	366	6.5	360	2.5	3.7	2.9

n (number of sites), T (thickness of the considered top-soil layer, m), MAP (Mean annual precipitation), MAT (mean annual temperature), Z (altitude above sea level), Clay (soil clay content), SOC_{D-ND} (soil organic carbon density in the non degraded top-soil), SOC_{D-D} (soil organic carbon density in the degraded top-soil).

Table 6
List of the categorical variables describing the environmental conditions.

Category	Class	Definition
Mean annual precipitation (MAP)	Low	<600 mm
	Medium	600–1000 mm
	High	>1000 mm
Mean annual temperature (MAT)	Low	<0 °C
	Medium	0–10 °C
	High	>10 °C
Soil texture	Low	<20% clay
	Medium	20–32% clay
	High	>32% clay
Soil pH	Strong acidic	>5
	Weak acidic to weak alkaline	5–7
	Strong alkaline	>7

3. Results and discussion

3.1. Global distribution of SOC stocks in grassland soils

SOC concentration (SOC_C) ranged between 0.2 and 293 g C kg^{-1}, with a coefficient of variation (CV) of 115% (Table 3). The average SOC_C in the topsoil of both degraded and non-degraded grasslands worldwide computed from 628 observations was 34.9 g C kg^{-1} with a standard error (±) of 1.6 g C kg^{-1} (Table 3). SOC stocks (SOC_S) ranged between 0.1 and 38.8 kg C m^{-2}, with an average of 5.0 ± 0.2 kg C m^{-2}. SOC density (SOC_D) ranged from a minimum of 0.7 kg C m^{-3} to a maximum of 194.0 kg C m^{-3}. These values are higher to those reported by Baisden and Amundson (2002) with SOC_D of 14.2 kg C m^{-3} for sandy loamy grassland soils under a Mediterranean climate (hot, dry summers and cool, wet winters) in USA and lower to SOC_D of 194.0 kg C m^{-3} reported by Schipper et al. (2007) under a latitudinal gradient of 36–46°S in New Zealand grassland soils. The average SOC_D was 32.2 ± 1.3 kg C m^{-3}. A CV of 88% suggested the high spatial variations of SOC stocks in grassland soils (0–0.3 m) worldwide.

The greatest SOC_S were found in temperate regions (Fig. 3). This trend can be explained by lower average temperatures, which slow the rate of decomposition, hence accumulating SOC as pointed out by Davidson and Janssens (2006). The lowest SOC stocks were observed in arid to semi-arid grassland soils. This trend of lower SOC stocks in semi-arid to arid grassland environments may be explained by the low rainfall amounts, which result in low biomass production and organic matter decomposition, thus reducing C inputs into the soil (Amundson et al., 1989; De Deyn et al., 2008).

3.2. Grassland degradation impact on SOC stocks

Overall, grassland degradation induced an average 9% decline in SOC_S (Table 5), with a 95% confidence interval of −14% to −4% (Fig. 5). The meta-analysis revealed that changes in SOC_S were affected by the intensity of degradation, with SOC_S being

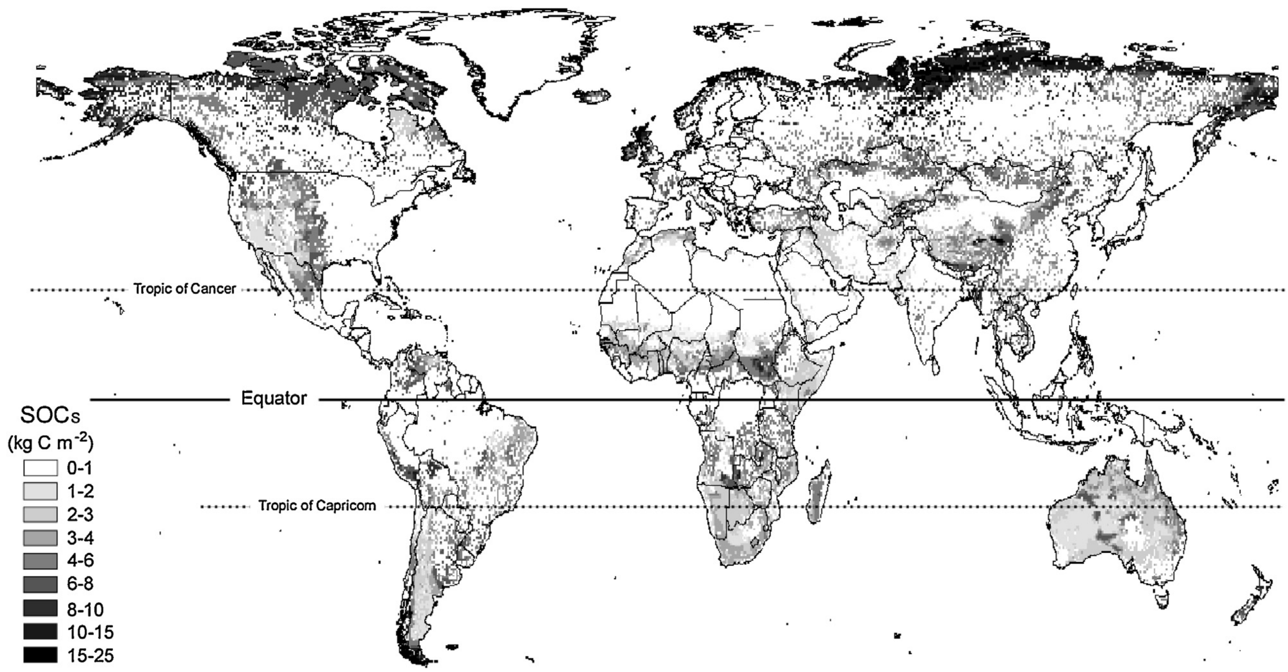

Fig. 3. Map showing the spatial variation of SOC stocks to 0.3 m throughout grassland soils worldwide. The information was extracted from the FAO-UNESCO SOC map of the world.

significantly reduced by 13% when the grassland was heavily degraded and 7% when moderately degraded (Fig. 5). This resulted in a trend of declining differences in SOC stocks between degraded and non-degraded grasslands, with no differences between degraded and non-degraded grasslands when the grasslands were lightly degraded. Hence, the lower degradation intensity decreases the SOC_S by 6%. This initial level of degradation has been shown to affect the SOC sequestration potential of grassland soils. In a temperate soil, characterized by different intensities of degradation, Preger et al. (2010) found SOC_S losses of as much as 30% in lightly degraded grasslands and up to 70% under the most degraded ones.

3.3. Relationship between SOC stocks and selected environmental factors

3.3.1. Mean annual precipitation and temperature

The climatic variables, MAP and MAT explained much of the variability of SOC concentration, stocks and densities in grassland soils worldwide. Correlation matrix (Table 4) showed that MAP was correlated positively with SOC_C ($r = 0.26$; $P < 0.05$), SOC_S ($r = 0.58$; $P < 0.05$) and SOC_D ($r = 0.36$; $P < 0.05$). MAT correlated negatively with SOC_C ($r = -0.37$), but positively with SOC_S and SOC_D.

Such an impact of climatic factors (MAP and MAT) on the spatial variations of SOC in grassland soils was consistent with classical studies based on soil-forming factors (Jenny, 1941). Based on a comprehensive analysis of 2700 soil profiles, Post et al. (1982) and Jobbágy and Jackson (2000) found that SOC_D increased with increasing MAP.

Greater SOC stocks under wet climates can be attributable to the high productivity of grasslands in wet environments, which allocate a high proportion of C below ground (Guo and Gifford 2002; Jobbágy et al., 2000). On the other hand, grassland soils in dry climates do not receive adequate C inputs owing to the low precipitation causing low biomass production to replenish the SOC lost through grassland degradation.

3.3.2. Altitude

SOC in grassland soils was found to be related to Z ($P < 0.05$) with SOCc increasing as Z increases, while SOC_S and SOC_D decreased with increasing Z. Lower SOC_S and SOC_D at higher altitudes was corroborated by results of Garcia-Pausas et al. (2007) in the Pyrenees mountain grasslands of Spain. These authors explained this result by the lower MAT conditions of higher altitudes, ranging between $-0.7\,°C$ and $5\,°C$, which limit net primary productivity. Moreover, the decrease of organic matter and root turnover with soil cooling is likely to explain the higher topsoil SOC_C, as pointed out by Hitz et al. (2001) along an altitude gradient varying between 1665 and 2525 m.a.s.l across a Swiss alpine grassland. Other studies have shown that a combination of cold temperatures, water logging conditions and substrate quality favors soil organic matter stabilization (Hobbie et al., 2000; Grosse et al., 2011; Baumann et al., 2009). Hobbie et al. (2000) suggested the production at high altitudes of substantial fraction of substrate quality that decomposes slowly.

3.3.3. Soil texture

Overall, SOC_C in grassland soils negatively correlated with Clay ($r = -0.38$, $P < 0.05$). This was a surprising trend as greater SOC_C in fine textured soils is attributed to the stabilization of SOC by clay and silt particles, which protect organic matter from decomposers (Parton et al., 1987; Six et al., 2002). Additionally, fine textured soils tend to have higher nutrient and water holding capacities, impacting on biomass production and carbon inputs to soils (Skjemstad et al., 1996). Among the few available studies in grassland soils, Garcia-Pausas et al. (2007) in the Pyrenees mountain grasslands of Spain found SOC_C to be 36% higher in silty loamy soils ($93.9\,g\,kg^{-1}$) compared to sandy soils ($60\,g\,kg^{-1}$). SOC_C was 40% higher in fine compared to coarse textures soils in several grassland soils in the Netherlands (110 *vs* $60\,g\,kg^{-1}$, Hassink, 1992). An average 190% difference was observed by Brye and Kucharik (2003) across two topochronosequences in the USA (8.9 *vs* $26.4\,g\,kg^{-1}$). This rate was as high as 270% (19.7 *vs* $7.3\,g\,kg^{-1}$) in a tropical grassland of Zimbabwe (Bird et al., 2000). In this context, what may explain the negative correlation between clay content and SOC_C as observed in the present study? The likely explanation is the presence in the data base of sites from the high latitudes under coarse soils, but high SOC_C (*e.g.* 247.3 g kg^{-1}: Martinsen et al., 2011) because of low MAT and associated low activity of decomposers.

SOC_S positively correlated with Clay (r = 0.17), but surprisingly, soil texture had no impact on SOC_D (Table 4).

3.3.4. Multiple correlations between SOC and the environment

Further insights on the relationship between SOC and the environment were depicted in Fig. 4. The first two axes of the PCA explained 76% of the total data variation within the data set. The first PCA axis (Axis 1), which accounted for 43% of the variance was positively correlated with Sand and Z and negatively correlated with Clay and MAT and to a lesser extent to MAP. Axis 1 could thus be interpreted as an axis of "tropicality", opposing clayey soils submitted to high MAT and precipitation to sandy soils of low MAT and MAP. The second PCA axis (Axis 2), which accounted for 33% of the data variance opposed pH with positive coordinates to MAP and was thus interpreted as an axis of leaching. The soils submitted to the higher precipitation having lost their cations and are thus characterized by lower pH.

From Fig. 4 we learned that SOD_D and SOC_S have the tendency to increase with tropicality, and that SOC_C decreased with tropicality and increased with soil leaching.

3.4. Impact of controlling factors on the depletion of SOC stocks

Grassland degradation reduced SOC_S in grassland soils, although the effect size varied with degradation intensity, classes for climatic factors (MAP and MAT), soil texture, soil pH, and grass type (Table 6).

3.4.1. Mean annual precipitations

Changes in SOC_S induced by grassland degradation were found to be significantly correlated with MAP (Fig. 6), with areas

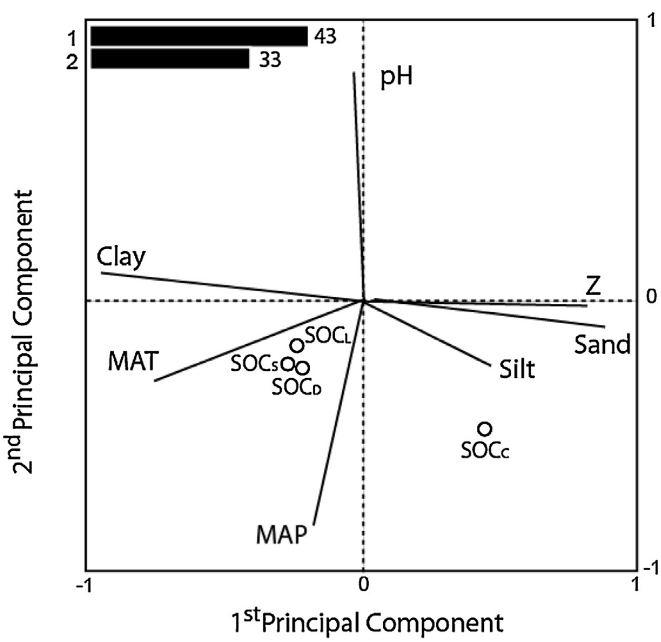

Fig. 4. Principal component analysis (PCA) scatter diagrams for mean annual precipitation (MAP), mean annual temperature (MAT) and soil Clay, Silt and Sand content as variables for Axis generation, and soil organic carbon content (SOC_C), stocks (SOC_C), and density (SOC_D) at the non-degraded treatments and soil organic carbon losses due to degradation (SOC_L), as secondary variables.

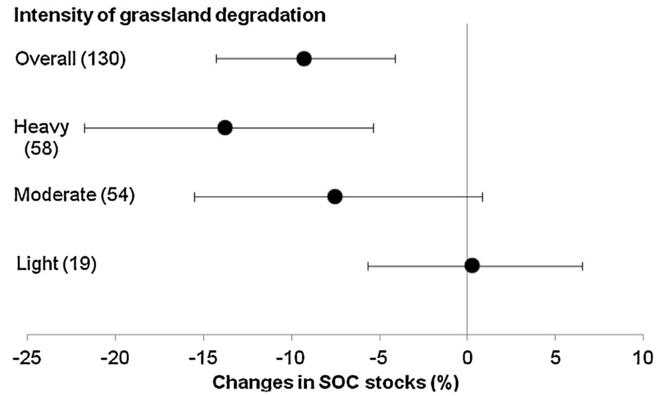

Fig. 5. Influence of degradation intensity on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap zero. The number of observations in each class is shown in parenthesis.

receiving 600 mm or less of rain per year showing a greater decline in SOC_S (−16%) than areas receiving 600–1000 mm (−1%). The much higher impact of grassland degradation on SOC_S under the drier climates could be explained by a low vegetation density, which induces upon disappearance, a sharp decrease in soil moisture, thus hampering carbon inputs to the soil.

The results of this meta-analysis are also in agreement to an early review of 22 studies examining the SOC sequestration potential of degraded grasslands worldwide by Conant and Paustian (2002), who observed that MAP was the main factor controlling SOC sequestration in degraded grasslands. Similarly to the results obtained in this meta-analysis, Conant and Paustian (2002) found that grassland degradation led to a greater depletion of SOC_S under dry climates (<333 mm). In contrast, they reported a greater SOC sequestration potential of up to 93% for grassland sites in wet climates (≤1800 mm yr^{-1}).

3.4.2. Temperature

Interestingly, this meta-analysis showed that grassland degradation significantly reduced SOC_S by 35% in sites with temperatures ≤0 °C, while there was no significant effect on SOC_S for the 0–10 °C and ≥10 °C MAT categories (Fig. 7). A closer look at the studies where SOC_S declined as a consequence of grassland degradation revealed that these were mainly sites with coarse

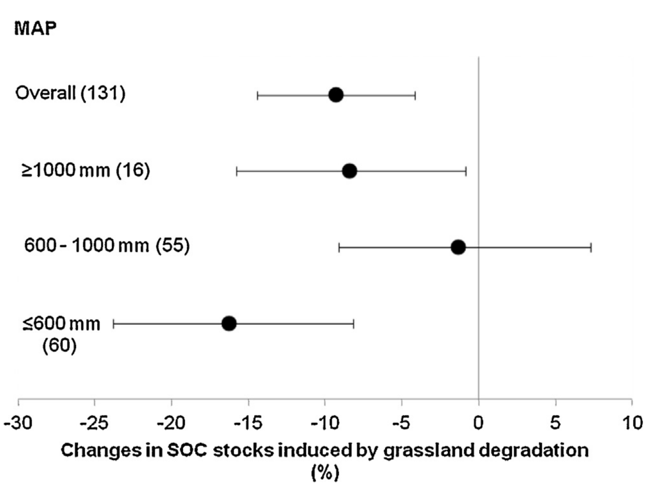

Fig. 6. Influence of mean annual precipitation (MAP) on changes in SOC stocks as affected by grassland degradation. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap zero. The number of observations in each class is shown in parenthesis.

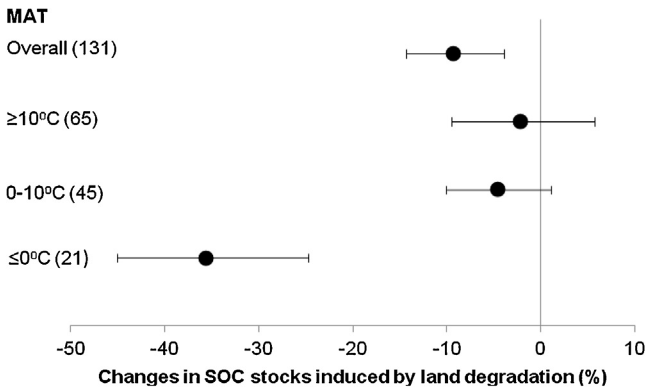

Fig. 7. Influence of mean annual temperature (MAT) on changes in SOC stocks as affected by grassland degradation. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap zero. The number of observations in each class is shown in parenthesis.

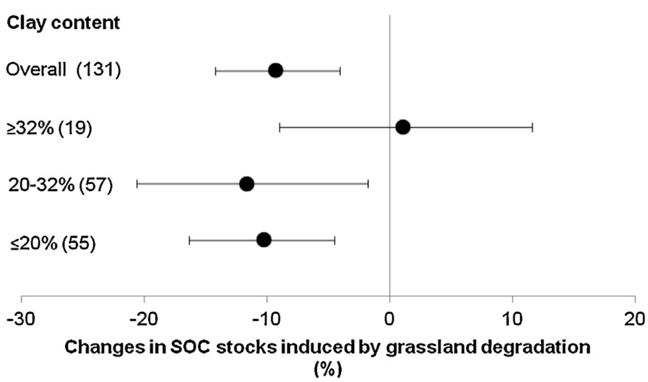

Fig. 8. Influence of clay content on changes in SOC stocks consecutive to grassland degradation.Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap zero. The number of observations in each class is shown in parenthesis.

textured soils (*e.g.* Chuluun et al., 1999; Steffens et al., 2008; Martinsen et al., 2011; Medina-Roldán et al., 2012).

3.4.3. Soil texture

Meta-analysis revealed that grassland degradation had a significant negative effect on coarser-textured than clayey-textured grassland soils (Fig. 8). On average, grassland degradation resulted in a 12% decline in SOC_S in loamy soils (20–32% clay), 10% in sandy soils (<20% clay) and there was a negligible effect (1%) in soils with a clay content of more than 32%. A comparable response of loamy and sandy soils was observed, however, the difference was not significant. The correlation between SOC_S and soil texture corroborates most previous studies on grassland soils, which have indicated that grassland degradation depletes SOC more in coarse textured soils. For example, Potter et al. (2001) in the USA who examined the impact of grassland degradation on SOC stocks across two degraded grassland soils with contrasted soil textures found that degradation significantly reduced SOCs by 41% (56.7 t ha^{-1} in coarse textured *vs* 95.7 t ha^{-1} in fine-textured soils).

Not only can intensification of degradation lead to significant depletion of SOC_S, but it can induce shifts in the distribution of soil texture itself. A recent study by Dong et al. (2012) in the Qinghai–Tibetan Plateau in China found that grassland degradation led to a shift in soil texture from loamy toward sandy-loamy soils, with the likely cause being soil erosion. They postulated that this shift toward more sandy induced changes in the carbon storage capacity of the soil.

The limited effect of grassland degradation on the depletion of SOC stocks in clayey soils can be explained by several reasons. Clay particles associate with organic compounds, thereby contribute to the formation of stable organo-mineral complexes (Six et al., 2002). These stable complexes lead to the stabilization of SOC through physical protection against decomposition (von Lützow et al., 2006), especially within micro-aggregates (Feller and Beare, 1997; Six et al., 2000). Furthermore, soils with high clay content have been shown to have better water holding capacity and infiltration rates (when well aggregates as in the tropics), which might stimulate biomass production and consequently increase carbon inputs into the soil (Burke et al., 1989; Schimel et al., 1994).

3.4.4. Soil pH

Across all comparisons, changes in SOC_S were the greatest in acidic soils (pH < 5), with an average loss of 14% (Fig. 9). There are several explanations for such a trend. Various studies have shown that soil pH influences the decomposition of soil organic matter through the hydrolysis and protonation processes (Motavalli et al., 1995; Andersson and Ingvar Nilsson, 2001; Aciego Pietri and Brookes, 2008) that regulates organic matter solubilization and complexation, and its sorption and desorption on mineral surfaces (van Bergen et al., 1997). According to Motavalli et al. (1995) and Janssens et al. (2010) soil acidification reduces the enzymatic activity thus lessening decomposition of plant material. Another possible explanation for the significant impact of soil pH on the degradation-induced changes in SOC stocks could be that at low soil pH, the soil structure becomes fragile because of weak bounds between base cations such as Ca, K and Mg and soil particles (Jobbágy and Jackson, 2003; Berthrong et al., 2009). Under heavily degraded grasslands in China, Wu and Tiessen (2002) found that grassland degradation reduced the cation exchange capacity (CEC) by 18% which triggered irreversible nutrient losses and biomass production (Jobbágy and Jackson, 2003). Lower SOC_S in degraded grasslands might come as a result of low nutrient availability and associated decline in biomass production. Under very acidic conditions (pH = 3.8) in a South African degraded grassland, Mchunu and Chaplot (2012) found grassland degradation to reduce SOC_S by as much as 63%.

Interestingly, grassland degradation had no impact on SOC_S for soils with a pH ranging between 5 and 7, which had been previously noticed by Dong et al. (2012) in a Chinese grassland.

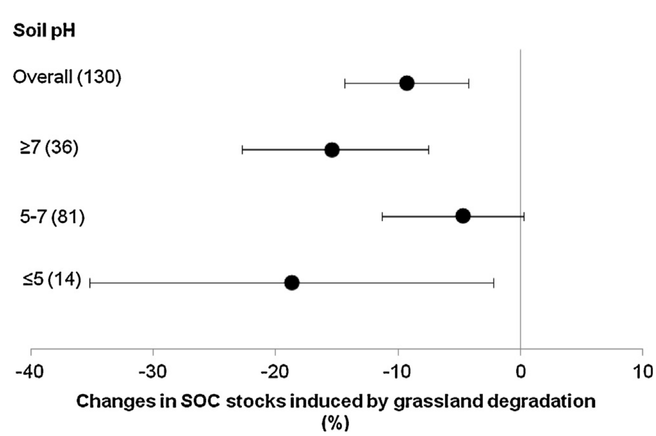

Fig. 9. Influence of soil pH on changes in SOC stocks due to grassland degradation. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap zero. The number of observations in each class is shown in parenthesis.

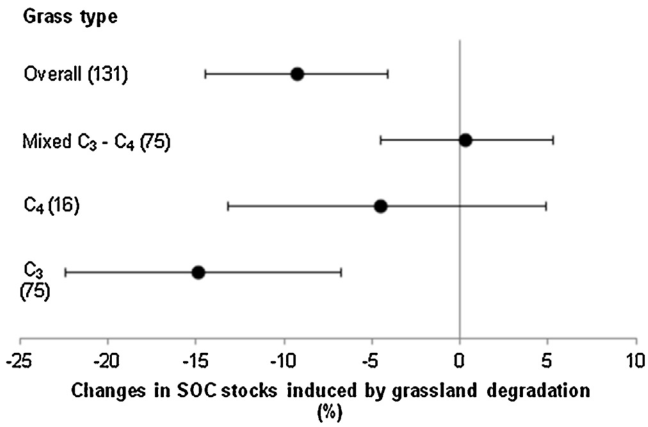

Fig. 10. Influence of grass type on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap zero. The number of observations in each class is shown in parenthesis.

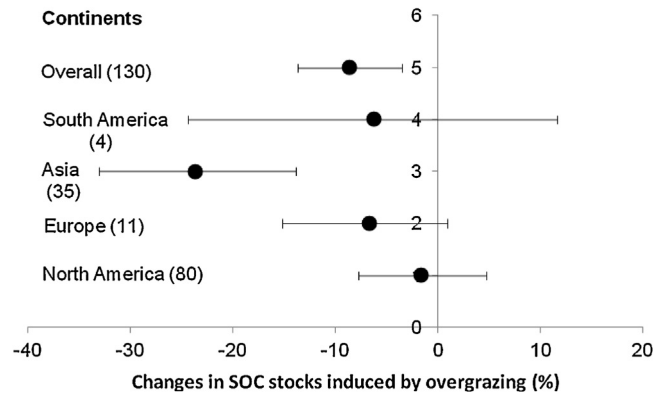

Fig. 11. Changes in SOC stocks for the different continents. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap zero. The number of observations in each class is shown in parenthesis.

3.4.5. Grass type

The depletion of SOC_S as a result of grassland degradation was found to be related to plant photosynthetic pathway (C_3 for grasses adapted to cool-season conditions *vs* C_4 for grasses adapted to warm-season conditions). Grassland degradation impact on SOC_S was greater for C_3 grasses (−14%) compared to C_4 grasses (−5%; Fig. 10). This is consistent with a recent review by McSherry and Ritchie (2013) which found that grazing reduced SOC_S the most in C_3 grasses and explained these differences by plant response to grazing. Grazing through defoliation may indeed modify the grass composition by altering the relative abundance of C_3 and C_4 grass species (Chapin et al., 1997; Bardgett and Wardle, 2003), which in turns affects the amount and dynamics of SOC stocks because of changes in the quality and quantity of carbon inputs (Derner et al., 2006; De Deyn et al., 2008; Klumpp et al., 2009). In loamy soils of the Qinghai–Tibetan Plateau in China, Dong et al. (2012) found that grassland degradation resulted in an increase in C_3 grasses (*Ligularia Virgaurea*) and a decrease in C_4 grasses (*Kobresia capillifolia*). This change in grass species composition reduced C accumulation in the soil since C_3 and C_4 grasses have different C allocation strategies, therefore, an increase in C_3 grasses and a decrease in C_4 grasses will result to lower SOC_S because C_4 grasses have a higher root-to-shoot ratios and greater transfer of photosynthate belowground (Frank et al., 1995; Reeder et al., 2004). Derner et al. (2006) found for instance that the above-ground biomass of C_4 grasses was 44% lower in degraded than in non-degraded grasslands, while above-ground biomass of C_3 grasses was 76% lower degraded than in non-degraded grasslands. It is also because the characteristics of grass species also control soil aggregation (Conant et al., 2001) and carbon release through soil respiration (De Deyn et al., 2008) that changes in species diversity have the potential to regulate SOC_S.

3.5. Multiple correlations between the losses in grassland SOC stocks and the environment

As for SOC_S and SOC_D, the losses of SOC stocks (SOC_L) following grassland degradation increased with increasing tropicality and soil leaching (Fig. 4).

3.6. Average depletion of SOC stocks per continent

Asia was the continent where grassland degradation had the greatest impact on SOC_S depletion (Fig. 11) with an average SOC_L of −23.7%, far greater than Europe (−6.7%), South America (−6.3%) and North America (−1.6%). Surprisingly, several studies in South America and Asia reported SOC_S gains following grassland degradation which are likely to be explained by changes in grass species.

4. Conclusion

In this study of 628 soil profiles from 55 temperate, sub-tropical and semi-arid sites our main objective was to quantify the impact of grassland degradation on SOC stocks and to identify the main environmental factors of controls, worldwide. Two main conclusions can be drawn. The first one is that the worldwide average grassland SOC stock depletion was 9% with values ranging between 13% for heavily degraded to 7% in lightly degraded soils. The second conclusion is that there was a general trend for grassland degradation to have a more pronounced impact on the reduction of SOC stocks under wet climates with acidic soils compared to basic soils from dry climates. Proper management of soil pH in grasslands should thus constitute an important way to mitigate against degradation-induced losses of SOC, if not against grassland degradation itself.

Assuming that 30% (an increase from 25% in 2008, Bai et al., 2008) of world grasslands are nowadays affected by degradation, the amount of SOC likely be lost is estimated as 4.05 Gt C with a 95% confidence between 1.8 and 6.3 Gt C (*i.e.* from 1.2 to 4.2% of the whole grassland soil stock). This implies that a similar amount of atmospheric C could be potentially sequestrated in soils through grassland rehabilitation. In a context where the international community recently recognised soil degradation to constitute a major threat and committed to rehabilitate degraded lands as during the COP20 in Lima, these results should guide adapted protection measures and strategies to mitigate grassland degradation. Global carbon models could also benefit from this newly acquired knowledge. There were however several limitations to this study. Following the initial work by Conant and Paustian (2002) on the same issue, several new data were available, especially in South America, Africa and Asia and have thus been used in the present analysis. However, these available studies do consider the same soil layers which significantly limits inter-site comparisons, thus calling for greater coordination and standardization in soil sampling procedures.

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