1. Introduction

Grasslands cover approximately 40% of the earth’s terrestrial surface area and play an important role in the global carbon (C) cycle by storing about 10% of the global soil C stocks (Suttie et al., 2005). Additionally, in grasslands soil organic C (SOC) concentration is higher at the soil surface, which may turn the C to the external factors such as climate and land management. Fire is the most common anthropogenic grassland management practice, used since the early Holocene (Behling and Pillar, 2007), because of easy application in difficult terrains and on large areas. However, other practices like mowing, grazing and fertilization are also in use (Blüthgen et al., 2012; Peng et al., 2011). All these grassland management practices have potential to influence soil C stocks and eventually CO2 emissions from the soil to the atmosphere (Peng et al., 2011; Granged et al., 2011; Jia et al., 2012). Burning is a common practice used for increasing fodder production and quality, whilst avoiding bush encroachments (Tainton, 1999). It results in increased biomass growth period and biomass production (Ojima et al., 1994), while at the same time improving grass cover and biodiversity (Boakye et al., 2013). This grassland management practice can have negative consequences on soil chemical, physical and biological properties (Andersson et al., 2004; Granged et al., 2011). Burning causes a general decline of SOC through combustion of soil organic matter (SOM) in the upper soil layer (Granged et al., 2011). For example, Granged et al. (2011) reported a 35% reduction in SOC after three years of burning with significant changes in soil physical properties, leading to increased water repellence. In addition, the high soil temperature (ST) and low soil moisture content (SWC), during fires, causes a...
sharp decrease in topsoil (0–0.05 m) biological biomass and its activity (Nardoto and Bustamante, 2003; D’Ascoli et al., 2005). Moreover, because of greater root and tuff density at the soil surface frequently-burned grasslands have the tendency to show detritus accumulation on the topsoil compared to non-burned shrubby grasslands and trees (Ansgley et al., 2002). This, together with an enhanced mineralization of SOM (Singh et al., 1991), highly affects CO2 effluxes from soil.

Several studies have reported lower CO2 emissions from soil in no burn (NB) than burned grasslands (e.g. Knapp et al., 1998; Rutigliano et al., 2007; Ward et al., 2007; Xu and Wan, 2008). For instance, Knapp et al. (1998) reported that annual burn (AB) for 17 years in eastern Kansas region, USA, resulted in 55% greater monthly CO2 emission from the soil than in NB treatment. Similarly, Xu and Wan (2008) reported 23.8% more CO2 emission in AB than NB on sandy soils of semiarid Northern China over two growing seasons, whereas Jia et al. (2012) reported in the same region 11% lower emissions with NB compared to AB but for only one growing season. Castaldi et al. (2012) also reported less CO2 emissions from unburned compared to burned plots in central Africa.

Mowing is also regarded as an improved grassland management practice (Zhou et al., 2007; Hamilton et al., 2008), which can result in a decrease of the CO2 emissions from soils (Bahn et al., 2006) by 20–50% compared to burning (Wan and Luo, 2003). However, the reasons for such a decrease are unclear. Wan and Luo (2003) explained it to be due to a decrease in photosynthetic C supply from aboveground biomass. Bahn et al. (2006) suggested that it could be a result of the depletion of easily available C substrates for the microflora. Nevertheless, others studies stated that mowing results precisely in an increase of rhizodeposition, soil microbial biomass and labile C (Zhou et al., 2007; Hamilton et al., 2008), which might suggest increased CO2 emission from the soil. Therefore, the underlying reasons for mowing effect on CO2 emissions from soils require further elucidations.

Both mowing and burning also impact the land cover evolution. Indeed, bushes often encroach into grasslands where neither burning nor mowing are applied (Trollope, 1980; Tainton, 1999; Montané et al., 2007). Montané et al. (2007) reported an increase of soil C stocks in the upper soil layers (top 15 cm depth) following shrubs encroachment into grasslands. Wang et al. (2013) also found an increase of soil C storage by shifting from grassland to woody plants.

While numerous studies exist on the impact of grasslands burning on CO2 emissions from soil and soil C stocks, the existence of discrepancies between these limits decision making on grassland management. These studies show inherent relations, related to their short duration, which long-term experiments might allow to overcome. In this study, 62-year annual burn and mow were compared against no burn treatment in an African Savannah. The no burn treatment was characterized by encroachment of large trees. Our main objective was to evaluate the impact of annual grassland burn management on SOC dynamics (C-stocks and CO2 emissions from soils) and their factors of control.

2. Material and methods

2.1. Study area

The experiment was conducted at Ukulinga Farm, the training and research farm of the University of KwaZulu-Natal, Pietermaritzburg, South Africa (24° 24'E, 30° 24'S) (Fig. 1). The experimental site is located on top of a small sloping plateau ranging in altitude from 847 to 838 m (Fynn et al., 2004). Soil depths vary from 0.05 m in the upslope to 0.20 m in midslope and 0.6 m at the footslope, and were classified as Plinthic Acrisols (WRB-FAO, 2006). The parent material is colluvium shale with intrusions of dolerite. The soil is acidic with a pH (KCl) of 5.5 at the top-soil and its texture is silty clay loam (37% clay, 43% silt and 20% sand).

The climate is sub-tropical humid and characterized by warm and wet summers (October–April), and cool and dry winters (May–September). Long-term (30 years) mean annual temperature and precipitation at the farm were 16 °C and 694 mm, respectively.

The native vegetation of the study area is dominated by the southern tall grassveld, which produces dense vegetation with plant heights ranging between 0.5 and 0.75 m (Fynn et al., 2004). Depending on the grassland management, some scattered trees, for instance Acacia sieberiana and some grass species such as Themeda triandra and Tristachya leucothrix are also found (Fynn et al., 2004). The native grass species (e.g. Themeda triandra and Tristachya leucothrix) all use the C4 photosynthetic path (Fynn et al., 2005).

2.2. Experimental design

The experiment involved three treatments namely; no burn (NB), annual burning (AB) and annual mowing (AM). There has been neither burning nor mowing in the NB since 1950, and these plots are now encroached by densely spread trees of Acacia sieberiana species. Long-term annual burning (AB) involves the burning of grass in the 1st week of August every year since 1950. At the time of study, the AB plots were dominated by sparse Themeda triandra grass. In the AM treatment, the grass is cut at the same time as burning and the material is removed from the treatment plots. All treatments are replicated three times by slope position (upper, mid and footslope) in a randomized block design and the plots sizes are 18.3 × 13.7 m spaced by 4 m sidewalls. The three treatments (NB, AB and AM) were represented once in three slope positions (replicate 1: upslope; replicate 2: midslope and replicate 3: footslope). There was no grazing at the experimental site since it was established in 1950. More details about the experimental site and design can be found in Tainton et al. (1978).

2.3. Soil sampling and analysis

Soil samples for evaluation of SOC content (SOCc) and soil organic nitrogen content (SONc) were collected once (at the beginning of the second year) in each plot at three randomly selected pits (0–0.2 m deep). The samples were air-dried for 48 h, then gently ground and sieved through a 2 mm sieve. Total C and N were measured in the soil samples using LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI). On the same day additional soil samples for bulk density (pb) were also collected from each plot in the middle of the 0–0.2 m layer using 7.5 cm diameter metallic cylinder core with the height of 5 cm. Soil pb was determined using the core method where the ratio of water content corrected mass to volume was computed (Grossman and Reinsch, 2002).

SOC stocks (SOCs) were calculated using the following equation (Batjes, 1996):

$$\text{SOCs} = \text{SOCc} \times \text{pb} \times \left(1 - \frac{\text{PF}}{100}\right) \frac{b}{T}$$

(1)

where SOCc is SOC stock (kg C m⁻²); SOCc is soil organic carbon content in the ≤2 mm soil material (g C kg⁻¹ soil); pb is the bulk density of the soil (kg m⁻³); T is the thickness of the soil layer (m); PF is the proportion of fragments ≥2 mm in percent; and b is a constant equal to 0.001.

The nitrogen stocks (SONs) were calculated using the same equation (Batjes, 1996), replacing SOCc by the soil nitrogen content (SONc).

Water stable soil aggregates were separated using wet sieving methods described by Elliott (1986). Field moist soil samples were sieved through an 8 mm sieve and air-dried. A subsample of 80 g was placed on a 2 mm sieve and submerged in water for 5 minutes followed by wet sieving for 2 mins. The wet sieving process involved moving the sieve up and down 50 times. The materials remaining on the 2 mm sieve were collected by backwashing the sieve into a pre-weighted drying pan. Eventually, four aggregate size classes were collected from each treatment (2, 0.25–2, 0.053–0.25, and >0.053 mm), by repeating the wet sieving procedure using 0.25 mm, and 0.053 mm
sized sieves. The mean weight diameter (MWD) for the water stable aggregate for each treatment was calculated using the following equation (Kemper and Rosenau, 1986):

\[
\text{MWD} = \frac{\sum_{i=1}^{n} X_i W_i}{\sum_{i=1}^{n} W_i}
\]

where \(X_i\) is the mean diameter for each fraction size, \(W_i\) is the proportional weight the fraction from the total dry weight of soil used, and \(n\) is the number of aggregate classes separated.

2.4. Measurements of CO\(_2\) emissions from soil

CO\(_2\) emissions from soil were measured once a month from March 2013 to March 2015 with ten measurements per plot using LI-COR 6400 gas exchange system (LI-COR, Lincoln, NE, U.S.A) fitted with the LI-COR 6400–09 soil respiration chamber. The closed chamber system has an internal volume of 991 cm\(^3\) and soil area of 71.6 cm\(^2\) (Healy et al., 1996). We evaluated the sample size necessary to estimate CO\(_2\) emissions measurement points within the plot with a standard error of ±10% of the mean as in Eq. (3) below:

\[
n = \frac{(CV/10)^2}{2}
\]

where \(n\) is the sample size and CV is the average coefficient of variation within the plot.

The samples size was found to be 10 points within each plot, which was randomly selected between grass and trees.

All the measurements were carried out during daylight hours, starting at 10:00 and finishing around 13:00 h. This time period was determined by a pre-experiment, which compared CO\(_2\) emissions from soil during the day and found that emission between 10:00 and 13:00 closely represented the average daily CO\(_2\) emissions from soil. CO\(_2\) fluxes from soil were expressed in two units: (1) in g CO\(_2\)-C in unit of surface per day (g CO\(_2\)-C m\(^{-2}\) day\(^{-1}\)) to evaluate the CO\(_2\) emission from soil to the atmosphere and (2) in g CO\(_2\)-C per gram of soil C per

Fig. 1. Location of the study site in South Africa and selected treatments (NB; no-burning, AB; annual burning and AM; annual mowing) and their replicate (R) in each plot (R1: upslope, R2: mid slope and R3: downslope position).
day (g CO₂-C g C⁻¹ day⁻¹) as a mean to evaluate the stability of C in the soil. The CO₂ emissions measured from soil do not include the emissions resulting from burning of plant residues.

2.5. Soil temperature and water content

Soil temperature (ST) and soil water content (SWC) were determined in conjunction with CO₂ effluxes at the 10 data points per plot. ST was evaluated by a sensor connected to the soil chamber (LI-COR 6400-09) by inserting the thermocouple close to the measurement points of CO₂ emissions at a depth of 0–0.05 m. SWC was measured for one season at the closest point to the CO₂ chamber using a Hydrosense soil moisture meter (Campbell Scientific, Inc., USA), calibrated by measurement of the meter responses at saturated soil.

2.6. Statistical analysis

Summary statistics were done for CO₂-C emissions from soil under NB, AB and AM during the whole study period, and summer and winter periods were separated (Table 2). A coefficient of variation was carried out using all the data together. The CO₂ emissions and soil properties data were analyzed as a complete randomized block design using the GENSTAT 14th Edition software. Since CO₂ emissions from soil were measured 24 times during the study period repeated analysis of variance was performed. There were systematic differences between slope positions, which however were considered in the analysis of variance as a block factor. Treatment means were compared using Tukey corrections for multiple comparisons, with significant differences defined at P < 0.05, unless specified otherwise. Finally, the study period and seasonal cumulative CO₂ emissions from soil for the treatment were compared. The CO₂ emissions from soil data were analyzed with regard to the season (winter/summer) and also to the position in the slope, the later having high contribution to the overall variability in data.

3. Results

3.1. Impact of treatments on soil properties

The mean ± SE of the soil bulk density (ρb), SOC content (SOCc), SON content (SONc), SON stocks (SONs), SOC stocks (SOCs), C:N ratio and mean weight diameter (MWD) in the top-soil (0–20 cm) under grassland and subjected to no burn (NB), annual burn (AB) and annual mowing (AM) treatments. The values are means (n = 3) ± SE.

3.2. Precipitation, air and soil temperature during the study period

The precipitation, average monthly air temperature and ST for the study period (March 2013 to March 2015) are presented in Fig. 2. The highest mean air temperature of 23 °C was recorded in January and December 2014 and lowest (8 °C) in July 2013. Average ST in AB of 23.4 °C was significantly higher than AM (21.4 °C) and NB (20.7 °C) (Fig. 2B). Overall, the average summer ST of 24 °C was significantly higher than that of winter which was 19 °C. While all treatments had similar ST in winters, the summer period ST was significantly higher in AB with average of 25.5 °C compared to NB with 22.5 °C, which was not significantly different from that of AM (23.3 °C).

3.3. Seasonal variation in fluxes and cumulative CO₂-C emissions from soil

Table 2 shows summary statistics of CO₂-C emissions per m² and gram of C in the soil for the study period, broken down into summer and winter
seasons. The lowest mean CO2 emission from soil was 0.87 g CO2-C m−2 d−1 in NB during winter and the maximum was 3.34 g CO2-C m−2 d−1 in AM during summer. The average daily CO2 emissions per m2 for the study period were significantly lower in NB than AB and AM, by 30% and 34%, respectively. When expressed as mg CO2-C per m2 for the study period were significantly higher with the increase of SONs (Table 3). CO2-C emissions from soil expressed either per area basis or per gram of C in the soils are shown in Fig. 2C and D, respectively. The patterns of CO2-C fluxes from soil were similar in all treatments (NB, AB and AM) during the study period with greatest fluxes observed in summer from October to April, and lowest in winter from May to September. Regardless of treatment, on average, 65% of CO2-C emissions from soil occurred during summer, which coincided with greater precipitation (90% of total annual rainfall) and highest air temperature (average 19 °C; Fig. 2). During summer periods, CO2-C fluxes from soil were generally lower under NB than AB and AM, with greater differences at 12 out of 14 sampling events. However, there were no significant differences between AB and AM during this period. The winter periods, CO2-C emissions from soil in all treatments were statistically similar. In addition, annual burning did not show significant effect on CO2-C emissions from soil during the winter period (Fig. 3).

3.4. Controls of SOCc and CO2 emissions from soil

For all data sets, CO2 emissions from soil per unit of surface area increased significantly with the increase in SWC, SOCc, pH, ST and C:N ratio, but decreased with the increasing of SONs (Table 3). CO2-C expressed per gram of C in the soil showed the same trend, increasing with SWC followed by SOCc and SOCc, C:N, pH and MWD but decreased with increasing SONs (Table 3).

### Table 2

Summary statistics of CO2 emissions from soil per m2 and gram of carbon in the soil form no burn (NB) annual burn, (AN) and annual mow (AM) treatments during the whole study period, summer and winter. N = 24.

|                  | CO2-C (g CO2-C m−2 d−1) | mg CO2-C g C−1 d−1 |
|------------------|-------------------------|---------------------|
| **All seasons**  |                         |                     |
| Mean             | 1.80b                  | 10.5b               |
| SD               | 1.08                   | 0.3                      |
| Min              | 0.25                   | 0.15                  |
| Median           | 1.84                   | 1.07                  |
| Max              | 4.15                   | 3.04                  |
| SE               | 0.13                   | 0.11                 |
| CV               | 0.60                   | 0.68                 |
|                  |                         | 61                    |
| **Summer**       |                         |                     |
| Mean             | 2.47b                  | 14.4b               |
| SD               | 0.81                   | 0.48                  |
| Min              | 0.88                   | 0.52                  |
| Median           | 2.51                   | 1.45                  |
| Max              | 4.15                   | 2.40                  |
| SE               | 0.13                   | 0.17                 |
| CV               | 0.33                   | 0.43                 |
|                  |                         | 32                    |
| **Winter**       |                         |                     |
| Mean             | 0.87a                  | 0.63a               |
| SD               | 3.16                   | 0.15                  |
| Min              | 0.25                   | 0.15                  |
| Median           | 0.62                   | 0.36                  |
| Max              | 2.19                   | 1.27                  |
| SE               | 0.59                   | 0.59                 |
| CV               | 3.62                   | 6.18                 |

Means followed by different superscript letters (a–b) in the same row are significantly different (P < 0.05).

### Table 3

Coefficients of determination (r) between CO2 emissions from soil per m2 and per gram of carbon in the soil and multiple soil factors: top-soil organic carbon and nitrogen (SOCc; SONc), SOCc stocks (SOCc); SON stocks (SONs), carbon to nitrogen ratio (C:N), soil bulk density (pB); Mean weight diameter (MWD); soil temperature (ST); and soil water content (SWC).

| SOCc | SONc | SOCc | SONs | C:N | pB | MWD | ST | SWC |
|------|------|------|------|-----|----|-----|----|-----|
| g CO2-C m−2 | mg CO2-C g C−1 | g CO2-C g C−1 | mg CO2-C g C−1 | g CO2-C g C−1 | mg CO2-C g C−1 | g CO2-C g C−1 | mg CO2-C g C−1 | g CO2-C g C−1 |
| 0.50 | 0.15 | 0.84 | 0.73 | 0.59 | 0.70 | 0.49 | 0.38 | 0.67 |

*Statistically significant determinants at P < 0.05 level.

### 4. Discussion

#### 4.1. Long-term burning and mowing impacts on CO2 emissions from soil

The overall 30% greater CO2 emissions from soil in AB than NB (Table 2) implies significant stimulation of CO2 emissions when grassland management changes from no burn to annual burn systems. This finding is consistent with several studies worldwide which reported greater CO2-C emissions from soil per m2 in “burn” compared to “no burn” treatment (e.g. Knapp et al., 1998; Rutigliano et al., 2007; Ward et al., 2007; Jia et al., 2012). The greater CO2 efflux from soil has been attributed to an increase in nitrogen availability from burning which enhances microbial respiration. Another explanation could be a change in organic matter quality consequent to burning. Assuming that there is a greater proportion of charcoal with low aggregation potential, the charcoal lowers the soil aggregate stability and consequently the protection of C from the decomposers. There was, therefore, likely lower C protection under long-term annual burning at the study site which resulted in NB having 26% lower CO2-C emissions per gram of C from soil than in AB (Table 2). In support of this, some literature showed that the SOC stability is strongly related to the stabilization of soil aggregates (e.g. Singh et al., 2009; Carrizo et al., 2015). However, there are studies who observed reductions (e.g. Ma et al., 2004) or no changes of CO2-C emissions from soil in response to fire (Castaldi et al., 2012), for example, laboratory experiments by Guerrero et al. (2001) reported an increase in aggregate stability after fire despite a decrease in soil organic matter.

Consistent with AB, AM stimulated greater CO2 emissions from soil than NB during the study period, which agreed with the results by Antonsen and Olsson (2005) and Li and Sun (2011). Mowing increases ST due to the removal of vegetation and exposure of the top-soil to direct sunlight, which enhances microbial activity and plant-roots growth. Another explanation for stimulation of CO2 emissions from soil by mowing was given by Antonsen and Olsson (2005) who indicated the stimulation of arbuscular mycorrhizal fungi growth after mowing. However, contrary to our results, some studies (e.g. Bremer et al., 1998; Han et al., 2012; Jia et al., 2012), found mowing to decrease CO2 emissions from soil in comparison to NB and they attributed it to the reduction in canopy photosynthesis.

In the present study, lower CO2 emissions from the soil under NB with tree encroachment than AB and AM were explained by difference of microclimatic conditions due to difference of soil cover. Tree coverage can significantly influence CO2 emissions from soils due to its effect on soil microclimate (Smith and Johnson, 2004; McCulley et al., 2007). For example, Smith and Johnson (2004) reported that soil respiration rate in woodlands was 38% less compared to grasslands, which they explained by the change in the soil microclimate (moisture and temperature). Carbone et al. (2008) also reported 86% lower CO2 emissions from NB encroached by shrubs than pure grass over five months (910 vs. 126 g C m−2), which was attributed to differences in soil water availability and below ground C allocation and plant productivity. However, Smith and Johnson (2004) found no difference in fine root biomass between grasslands and woodlands.
4.2. Seasonal change effects on CO2 emissions from soil

In the present study, significant differences in CO2-C per m2 and per g of C in the soil between treatments were only observed in summer (rainy) seasons (Table 2 and Fig. 2) coinciding with high precipitation, air and ST, which was consistent with what Chen et al. (2002) found in relatively similar conditions in the tropical savannah of northern Australia. They found that 70% of CO2-C emissions from soil were emitted during rainy seasons, where 95% of the precipitation occurred during this period. The greater CO2-C emissions from soil during the summer seasons were attributed to higher temperature and precipitation. In this study, CO2-C emissions from soil were positively correlated to ST \( (r = 0.47 \) and \( 0.34 \) for CO2-C per m2 and per g of C in the soil, respectively) and SWC \( (r = 0.67 \) and \( 0.72 \) for CO2-C per m\(^{-2} \) and per g of C in the soil, respectively). It is well known that soil temperature and water content are the most essential microclimatic factors controlling CO2-C emissions from soils. Thus, the absence of significant differences in CO2 emissions from soil among the treatments over the dry period can be explained by lower soil moisture in the root zone, which reduces fine root growth and soil microbial activity (Chen et al., 2002).

4.3. Relevance of grassland burning for C emissions in Africa: end note

Since burning will also produce CO2 and other greenhouse gases such as CO, NOx and non-methane hydrocarbons (Jain et al., 2006) and the fact that burned area represents up to 80% of the total grassland area in some region (Csiszar et al., 2005), the implications on GHG to the atmosphere through AB at global scale are huge. Annual burning and respiration have already been reported to induce significant C losses into the atmosphere (van der Werf et al., 2006), it suggests \(-95\% \) of the 58 Pg C year\(^{-1} \) fixed by plant through net primary production would eventually be emitted into the atmosphere. Van der Werf et al. (2006) estimated the greatest emissions as a result of biomass burning to come from Africa (49%), South America (13%), equatorial Asia (11%), boreal regions (9%), and Australia (6%).

5. Conclusions

In this study performed on long-term (62-years) grassland management trial in South Africa, CO2-C emissions from soil under no burned grasslands were compared to annual burning and mowing treatments. Annual burning and mowing resulted in 30% and 34%, respectively, greater CO2-C emissions per m2 than no-burning. These differences could be explained by lower stability of SOC in annual burning treatment. Since in this study the higher the aggregation (mean weight diameter), the lower the CO2 effluxes from soil which propose a decrease in soil organic matter physical protection occurs upon annual burning and mowing and that is likely to be one of the causes of the increased CO2 efflux from soil compared to no-burned grasslands.

They are several implications of these results. The first one is that burning, which is a common practice in grasslands of the developing world, should be avoided because of a significant increase in CO2 emissions from soil. Greenhouse gases other than CO2 are also emitted during burning and these need to be further investigated. This result directly implies that alternative grassland management practices have to be found. While burning abandonment appeared to lessen CO2 emissions, it poses a major threat to forage production as grass species get replaced by woody ones. The third implication is that grass mowing is beneficial for avoiding the release of fire-derived GHGs emissions and maintaining the grass sward in good conditions. Mowing, however, slightly decrease grass palatability (Fynn et al., 2004) and this might constitute a major limitation for its broad adoption.

There is a need to find grassland management emitting low amount of CO2 while sustaining high grass diversity. Following our results, a combination of mowing and burning and/or controlled grazing (such as the high density short duration one: Chaplot et al., 2016) should be tested. Further research needs also to be performed on the underlying
reasons of the variation in CO₂ emissions from soil between different treatments.

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References

Andersson, M., Michelsen, A., Jensen, M., Köllner, A., Gashaw, M., 2004. Carbon stock, soil respiration and microbial biomass in fireprone tropical grassland, woodland and forest ecosystems. Soil Biol. Biochem. 36, 1707–1717.

Anson, K.J., Bugas, W.A., Heuer, M.L., Kramp, B.A., 2002. Bowen ratio/energy balance and scaled leaf measurements of CO₂ flux over burned Prosopis savannah. Ecol. Appl. 12, 948–961.

Anthonio, T., Olsson, P.A., 2005. Relative importance of burning, mowing and species translocation in the restoration of a former boreal hayfield: results of plant diversity and the microbial community. J. Appl. Ecol. 42, 337–347.

Bath, M., Knapp, M., Garajova, Z., Pfahring, N., 2006. Root respiration in temperate mountain grasslands differing in land use. Glob. Chang. Biol. 12, 995–1006.

Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163.

Beihling, H., Pillar, V.D., 2007. Late quaternary vegetation, biodiversity and fire dynamics on the southern Brazilian highland and their implication for conservation and management of modern Araucaria forest and grassland ecosystems. Philos. Trans. R. Soc. Lond. 362, 243–251.

Blüthgen, N., Dornheim, C.F., Prati, D., Klaus, V.H., Kleinebecker, T., Hözel, N., Alt, F., Böch, S., Gockel, S., Hemp, A., 2012. A quantitative index of land-use intensity in grasslands: integrating mowing, grazing and fertilization. Basic Appl. Ecol. 13, 207–220.

Boyle, M.K., Little, L.T., Panagoul, M.D., Jansen, E.R., 2013. Effects of burning and grazing on plant species percentage cover and habitat condition in the highland grassland of Mpuamulanga Province, South Africa. J. Anim. Plant Sci. 23, 603–610.

Bremer, D.J., Ham, J.M., Owenby, C.E., Knapp, A.K., 1998. Responses of soil respiration to clipping and grazing in a tallgrass prairie. J. Environ. Qual. 27, 1530–1548.

Carbone, M.S., Winston, G.C., Trumbore, S.E., 2008. Soil respiration in perennial grass and shrub ecosystems: linking environmental controls with plant and microbial sources on seasonal and diel timescales. J. Geophys. Res. 113, G02022. http://dx.doi.org/10.1029/2007JG000611.

Carrizo, M.E., Alesso, CA, Cosentino, D., Imhoff, S., 2015. Aggregation agents and structural stability in soils with different texture and organic carbon contents. Sci. Agric. 72, 75–82.

Castaldi, S., de Grandcourt, A., Rasile, A., Skiba, U., Valentini, R., 2012. CO₂, CH₄ and N₂O fluxes from soil of a burned grassland in Central Africa. Bieogeosciences 7, 3459–3471.

Chau, P., Dillman, P., Chiev, P., 2016. Vegetation-mediated changes in microclimate reduce soil respiration as woodylands expand into grasslands. Ecology 85, 3348–3361.

Singh, R.S., Rabgabanshi, A.S., Singh, J.S., 2009. Soil physicochemical properties in a grassland and agroecosystem receiving varying organic inputs. Soil Sci. Soc. Am. J. 73, 1530–1538.

Singh, R.S., Rabgabanshi, A.S., Singh, J.S., 1991. Nitrogen mineralization in dry tropical savanna: effects of burning and grazing. Soil Biol. Biochem. 23, 269–273.

Smith, D.L., Johnson, L., 2004. Vegetation-mediated changes in microclimate reduce soil respiration as woodylands expand into grasslands. Ecology 85, 3348–3361.

Sutcliffe, J.M., Reynolds, S.G., Batello, C., 2005. Grasslands of the World. Food and Agricultural Organization of the United Nations, Rome, Italy.

Tainton, N.M., 1999. Veld management in South Africa. University of Natal Press, Pietermaritzburg, p. 364.

Tainton, N.M., Boosen, P.D., Brosney, D.J., Nash, R.C., 1978. Long term effect of burning and mowing on tall grassveld in Natal: dry matter production. Proceedings of the Annual Congresses of the Grassland Society of Southern Africa, p. 173–177.

Van der Werf, G.R., Randerson, J.T., Giglio, L., Collatz, G.J., Kasibhatla, P.S., Arantes Jr., E., 2006. Inter-annual variability in global biomass burning emissions from 1997 to 2004. Atmos. Chem. Phys. 6, 3423–3441.

Wan, S., Luo, Y., 2003. Substrate regulation of soil respiration in a tallgrass prairie: results of a clipping and shading experiment. Glob. Biogeochem. Cycles 17, 1–12.

Wang, W., Zeng, W., Chen, W., Zeng, H., Fang, J., 2013. Soil respiration and organic carbon dynamics with grassland conversions to woodlands in temperate China. Plos One 8, e71986. http://dx.doi.org/10.1371/journal.pone.0071986.

Ward, S.E., Bardgett, R.D., McNamara, N.P., Adamson, J.K., Ostle, N.J., 2007. Long-term consequences of grazing and burning on northern peatland carbon dynamics. Ecosystems 10, 1060–1083.

WRB-FAO, 2006. World reference base for soil resources. World Soil Resources Report No. 103. Rome, Italy.

Xu, W., Wan, S., 2008. Water- and plant-mediated responses of soil respiration to topography, fire, and nitrogen fertilization in a semiarid grassland in northern China. Soil Biol. Biochem. 40, 679–687.

Zhou, Z., Sun, O.J., Huang, J., Li, L., Liu, P., Han, X., 2007. Soil carbon and nitrogen stores and storage potential as affected by land-use in an agro-pastoral ecton of northern China. Biogeochemistry 82, 127–138.