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Effects of cattle grazing and rainfall on soil nutrients in northern Australia

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Abstract

Extensive cattle grazing is the dominant land use in northern Australia. To better understand the role of nutrient cycling in response to grazing and rainfall variability, field studies were undertaken in north-east Queensland. Organic matter in soils is critical for soil health, carbon storage, pasture productivity and nutrient cycling. Analysis of soils from the long-term grazing trial at Wambiana in north-east Queensland showed that soil organic matter was higher under heavy stocking, although grazing treatment had little effect on soil nitrogen. While this result is somewhat surprising given conventional wisdom suggests that more conservative grazing should result in higher organic matter, it is not completely inconsistent with other recent studies in northern Australia which show little effect of grazing intensity on soil organic matter. The results from this study suggest that caution is needed in recommending specific grazing management strategies as a means of storing carbon. An experimental study at the Spyglass Research Facility, where rainfall was manipulated to study the interaction between climate variability and nutrient dynamics showed that drought conditions caused an accumulation of available nitrate in the soil. While plants had higher nitrogen concentrations in drier conditions, the lower biomass resulted in less total nitrogen uptake which can explain accumulation of nitrate in the soil. The accumulation of nitrate in the soil in drought conditions can explain the high levels of animal growth following drought breaking rains. The results show that plant biomass, at least up to average rainfall, is mainly limited by water availability in this ecosystem. With higher levels of rainfall, water availability will not limit productivity and nitrogen availability will be the limiting factor.

Executive Summary

Extensive cattle grazing is the dominant land use in northern Australia. Grazing intensity and rainfall have significant effects on the dynamics of soil nutrients in these northern rangelands. However, previous studies have found variable effects of grazing pressure on soil nutrients. The equivocal nature of these results, combined with our poor understanding of the underlying processes driving nutrient dynamics in grazed tropical savannas, has made it difficult to develop recommendations for the grazing industry. Anecdotally, variations in nutrient dynamics have been implicated in variability in beef cattle performance with higher than expected animal growth in a year following a sequence of dry years, possibly due to increased pasture growth due to a build up of nitrogen in soils during the period when pasture growth was limited by low rainfall. In contrast, animal performance can be low at the end of a prolonged wet season due to nitrogen dilution in the standing pasture.

To better understand the role of nutrient cycling in response to grazing and rainfall variability, field studies were undertaken in the Charters Towers region of north-east Queensland. In the grazing management studies the focus was on moderate stocking (stocked at the long-term carrying capacity for the region) and heavy stocking (stocked at twice the long-term carrying capacity) to understand differential responses in soil nutrients to grazing, utilising the Wambiana long-term grazing study. The effects of interannual variability in rainfall on soil nitrogen and consequently plant biomass and forage quality were studied in a short-term (two year) experiment at DAFQ Spyglass Beef Research Facility.

At Wambiana, soil organic matter and mineral nitrogen in surface soils (0–10 cm depth), 11, 12 and 16 years after trial establishment on experimental plots representing moderate stocking (stocked at the long-term carrying capacity for the region) and heavy stocking (stocked at twice the long-term carrying capacity) were analysed. The effects of rainfall on soil nitrogen were tested in a rainfall manipulation experiment at Spyglass Beef Research Facility. During two years, we manipulated the amount of rainfall in relation to the natural rainfall recorded during these years.

Higher soil organic matter was found under heavy stocking, although grazing treatment had little effect on mineral and total soil nitrogen. The results found in the Brown Sodosol–Yellow Kandosol complex at the Wambiana site were not consistent with other soil organic matter studies on different soils in northern Australia There was large interannual variability in soil mineral nitrogen, but not in soil organic matter, suggesting that soil nitrogen levels observed in this soil complex may be affected by other indirect pathways, such as climate. The lack of response in organic matter to long-term grazing intensity, which is consistent with other studies that show climate and soil type are stronger determinants of soil organic carbon than grazing, suggests that caution is needed in recommending specific grazing management strategies as a means of storing carbon.

The results from the Spyglass experiment support the hypothesis that in drought conditions there is accumulation of available nitrate in the soil. Although plants had higher nitrogen concentrations in drier conditions, the lower biomass resulted in less total nitrogen uptake which can explain accumulation of nitrate in the soil in drier conditions. The results show that plant biomass, at least up to average rainfall, is mainly limited by water availability in this ecosystem. With higher levels of rainfall, water availability will not limit productivity and

nitrogen availability will be the limiting factor. Better understanding of the relationship between rainfall and nitrogen dynamics could lead to better management of stocking rates, strategies to improve animal growth and minimising land degradation from grazing practices.

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1 Background

Many of northern Australia's soils are low in nutrients. Australian landscapes are weathered relics of earlier pedological processes (Mott et al. 1985) and across northern Australia short and unreliable growing seasons and a long dry season, making it difficult for cropping to be practiced (Carberry et al. 1993). As a result, commercial cattle grazing is the dominant land use, over much of northern Australia, occupying 2.5 million km^2 of tropical semi-arid savannas and grasslands (Cook et al. 2010). Because of this large area, grazing management practices that cause changes in soil nutrient dynamics could potentially affect national- or even global-scale processes. It has been suggested that by improving land management, and hence land condition, in the livestock-grazed landscapes of northern Australia, hundreds of millions of tonnes of carbon could be sequestered in the top 10 cm of the soil (Ash et al. 1995; Garnaut 2008; Eady et al. 2009; Gifford 2010). In addition to helping reduce atmospheric CO_2 concentrations, this has the potential to improve soil and pasture productivity, improve livestock productivity and increase graziers' income through a higher value of livestock and incentives for carbon storage (Lam et al. 2013). This is especially important because grazing enterprises in north Australia are only marginally profitable (McLean et al. 2014). Improving pasture and land condition can increase rainfall infiltration and decrease erosion, which is a major threat to the water quality of the Great Barrier Reef Lagoon (O'Reagain et al. 2005).

The soils of northern Australia are heterogeneous because of factors such as underlying geology, position in the landscape, climate and land use (Mott et al. 1985). Variations in soil properties may have a strong effect on pasture growth. Mott et al. (1985) suggested that nitrogen was limiting for pasture growth on sesquioxidic soils, whereas soil water availability was limiting on the more fertile texture contrast soils. In other soils, water availability limited pasture growth in the drier years, whereas in wetter years pasture growth was limited by soil nitrogen (Mott et al. 1985). Furthermore, the effects of grazing depend on the soil type (McSherry and Ritchie 2013) and nutrient uptake by plants (Ash and McIvor 1995). In northern Australia, higher grazing intensity was found to decrease soil carbon and nitrogen on Grey Vertosol and Red Kandosols but increase soil carbon in a Brown Sodosol–Yellow Kandosol complex (Pringle et al. 2011, 2014). Such variation suggests that the effects of grazing management on soil nutrient dynamics should consider local conditions and be extrapolated with caution.

Nitrogen and carbon dynamics are inextricably connected and cannot be understood independently of each other (Wedin 1995). The net amount of nitrogen mineralisation and availability to plants may be restricted by the high C:N ratio of soil organic matter (Mott et al. 1985; Wedin 1995). It has been suggested that there is a negative relationship between average rainfall and available nitrogen (Liebig et al. 2014). Nitrogen stocks and flows not only affect primary production but they can be an important controller of secondary production e.g. domestic livestock (Myers and Robbins 1991). Therefore, it is important to manage the ecosystem for adequate available nitrogen. Even when soil nitrogen cannot be managed directly, it is important to understand its dynamics and drivers in order to better predict the amount of soil nitrogen, and consequently the amount of plant production and forage quality, under different management strategies. This information could be used by land owners to optimise their stocking rate and stocking strategies, leading to better management of the forage resource for animal productivity and maintaining land in good condition.

In northern Australia, grazing of native or introduced grasses and legumes without fertiliser application is the main management practice that affects nutrient dynamics and soil quality. It has been suggested that grazing affects soil nutrients through three main pathways in the absence of land degradation: (1) a net primary production pathway; (2) nitrogen pathways; and (3) decomposition pathways (Piñeiro et al. 2010). Because these pathways are not mutually exclusive and are affected by additional ecosystem properties, the effect of grazing on soil supply of nutrients is variable.

Available nitrogen enters the ecosystem from 1) biological N fixation; 2) mineralization; and 3) atmospheric deposition. Since the agriculture and industrial revolutions, atmospheric deposition has been the dominant source (Bobbink et al. 2010) of nitrogen addition. Although nitrogen is consumed and transformed through the system, most of it stays within the ecosystem. Between 60-90% of plant nitrogen that is consumed by livestock returns to the soil as excreta (Chen et al. 2001).

The effect of drought on the processes involved in the N cycle are largely unknown, but it has been suggested that it is of significant importance (Borken and Matzner 2009; Huntley and Walker 1982; Yahdjian et al. 2006). Most of the nitrogen in the system is unavailable to plants and needs to be transformed (e.g., mineralization and nitrification) into available nitrogen sources (e.g. nitrate, ammonium). Both mineralization and nitrification are controlled by soil moisture (Borken and Matzner 2009), so we could expect that in drier conditions there will be less available nitrogen in the soil (Tietema et al. 1992). However, in drier conditions plants have less biomass and consequently extract less nitrogen from the soil (Borken and Matzner 2009) that could result in excess soil nitrogen. This, in addition to nitrate moving more slowly through the soil and into the plants roots (Evans and Burke 2013), can result in an accumulation of available soil nitrogen in drier conditions. Furthermore, heavier rainfall increases the risk of leaching of nitrate which will leave less available soil nitrogen for the plant's roots in the soil. Since these factors are contradictory to each other, the net effect of drought on the available nitrogen in the soil depends on the ecosystem properties (Borken and Matzner 2009), although most studies found that drought increases available nitrogen (Evans and Burke 2013; Jackson et al. 1988; Kreuzwieser and Gessler 2010; Reynolds et al. 1999; Stark and Firestone 1995; Weltzin et al. 2003; Yahdjian et al. 2006). Northern Australia's dry tropic ecosystems could have different nitrogen cycle processes compared to other ecosystems around the world. This is due to a short wet, hot and humid season that enables a large pulse of biological activity in a short timeframe. During the long dry season, biological activity is reduced significantly and consequently nitrogen processes also slow down. However, the nitrogen cycle in the dry tropics has not been studied extensively (García-Méndez et al. 1991).

The effect of increased available nitrogen in the soil after droughts can increase biomass in the following year, if water availability is sufficient (Briggs and Knapp 1995; Clenton et al. 1970; Cregger et al. 2014; Knapp et al. 2008). Additionally, litter that accumulates in drought years can decompose and release more soil nitrogen in following years (Chen et al. 2001; Wedin 1995). This would suggest that in wet years when water availability does not limit productivity, the amount of rain in the previous year could control the amount of productivity through available soil nitrogen that is carried over from the previous year.

Drought is common in northern Australia and it has a large impact on livestock productivity, farm economics and land degradation (Stafford Smith et al. 2007). Most climate models predict that drought episodes will be more common and more severe with climate change (Watterson et al. 2015), further exacerbating challenges in managing climate variability. Therefore, a better understanding of the effects of dry periods and drought on soil nitrogen can have implications for better management of cattle production in these ecosystems.

2 Project objectives

We aimed to understand the effects of rainfall variability and grazing management on soil nutrient dynamics in grazing lands of north-east Queensland. In the grazing management studies we focused on moderate stocking (stocked at the long-term carrying capacity for the region) and heavy stocking (stocked at twice the long-term carrying capacity) on Brown Sodosol–Yellow Kandosol complex to understand differential responses in soil nutrients to grazing. Additionally, we aimed to understand the effects of interannual variability in rainfall on soil nitrogen and consequently plant biomass and forage quality.

3 Methodology

3.1 Effects of grazing – Wambiana grazing trial

We used soil samples that had previously been collected at a long-term cattle grazing trial conducted at Wambiana Station (20°34'S, 146°07'E), in north Queensland, Australia. The mean annual rainfall of the area is 636 mm, but highly variable (Table 1), with most of the rainfall (70%) falling between December and March (O’Reagain et al. 2009). The area has been subjected to cattle grazing at light to moderate stocking rates for at least the past 130 years before the trial’s establishment. In 1997, several ~100-ha paddocks were established to test a range of different grazing strategies (O’Reagain et al. 2009). Two contrasting strategies were selected for the present study: (1) a moderate stocking rate (MSR) treatment at ~8 ha per animal equivalent (450 kg steer), which is approximately the calculated long-term carrying capacity for the area; and (2) a heavy stocking rate (HSR) treatment at ~4 ha per animal equivalent (O’Reagain et al. 2009).

Table 1. Annual rainfall (mm year^{-1}) from July to June measured at the study area. For monthly rainfall see O'Reagain *et al.* (2009).

Year	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
Annual rainfall	380	407	388	470	469	708	898	1032	715	1240	750	606

There are three major land types within the treatment paddocks, so to minimise this source of variability all soil samples were collected from the dominant *Eucalyptus brownii* savanna woodland on the Brown Sodosol–Yellow Kandosol complex (Northcote 1960; Cannon 1997; Isbell 2002), which covers approximately 55% of the site area. Two soil samples were collected to a 30 cm depth 2 years after the commencement of the trial and general soil properties measured (Table 2). Statistical inference cannot be performed from these limited soil samples, but they suggest that the two treatments were similar at the onset of the manipulation. The area sampled in the MSR and HSR treatments was a subsample of the area sampled by Pringle et al. (2011) and Bray et al. (2014). The area sampled is a relatively homogeneous area that had no apparent differences at the onset of the manipulations, with an arbitrary fence line dividing the grazing treatments. The vegetation consists of an open woodland of *E. brownii* and an understorey of the native shrub *Carissa ovata*. The grass layer is dominated by native C4 tropical grasses, like *Bothriochloa ewartiana* and *Dichanthium fecundum*, but the exotic stoloniferous introduced grass *Bothriochloa pertusa* has spread widely on the trial area in the past 7–8 years. As a result of the grazing intensity treatment, there were major differences in both pasture total standing dry matter (TSDM) and pasture composition between the two treatments. On average, pasture TSDM was two- to three-fold greater in the MSR (range 1324–3144 kg ha^{-1}) than in the HSR treatment (range 419–1090 kg ha^{-1}). At the same time, the relative contribution to yield of the more desirable perennial grasses like *B. ewartiana* were approximately two-fold greater in the MSR than HSR treatment. In contrast, the frequency and relative contribution to yield of the exotic *B. pertusa* was far higher in the HSR treatment (P. O'Reagain, unpubl. data; O'Reagain and Bushell 2011).

Table 2. Average of two soil samples that depict general soil properties (0-30 cm) in the moderate stocking rate (MSR) and heavy stocking rate (HSR) on brown sodosol soil that were collected in 1999.

Grazing strategy	pH	% coarse sand	% fine Sand	% silt	% clay	% Organic carbon	EC (mS cm^{-1})	Phosphate (mg kg^{-1})	% Potassium
MSR	5.95	30	40	11	21	0.85	0.03	3.5	0.38
HSR	6.10	30	36	13	24	0.88	0.05	4.0	0.37

The soil samples were collected in July 2008, March–April 2009 and July 2013. Two adjacent 1-ha square-shaped areas (100 × 100 m) were selected on both sides of the arbitrary fence. In each area, 25 cores of 0–10 cm depth were sampled. However, because of a variety of issues, including missing or insufficient soil for laboratory analyses, the actual sample size was slightly lower (Table 3). Soil samples were dried at 40°C, large roots removed, soil crushed and passed through a 2 mm sieve. The dry soil was archived in sealed containers for further analysis.

Table 3. Number of soil samples analysed at 0-10 cm soil depth for the different soil measurements in the different years and in different treatments (MSR - moderate stocking rate; HSR - heavy stocking rate) on brown sodosol soil. SOM – soil organic matter.

Year	Grazing pressure	SOM	NO3	NH4
2008	MSR	22	25	25
2008	HSR	25	25	25
2009	MSR	25	25	25
2009	HSR	23	25	25
2013	MSR	25	25	25
2013	HSR	20	20	20

Mineral N (ammonium and nitrate-N) was extracted from the soil by shaking 8 g soil in 20 mL potassium chloride (2 M) for 1 h and filtering through No. 41 filter paper. Nitrate and ammonium were determined in the KCl extracts by colourimetric methods (Best 1976; Keeney and Nelson 1982; Willis et al. 1993). Soil organic matter content was determined by the loss-on-ignition method (Segoli et al. 2012). Spain et al. (1982) demonstrated that loss-on-ignition provides a robust indicator of soil organic matter and organic carbon content in soils with low contents of carbonate and hydrous oxides, on similar soils to those examined herein.

The 2008 soils were subsampled to obtain total nitrogen and δ13C. Analysis was undertaken using an Isoprime isotope ratio mass spectrometer (IRMS), coupled to a Eurovector elemental analyser (Micromass Isoprime–Eurovector EA 3000). Approximately 20 mg fine-ground soil was weighed into an 8 × 5 mm tin (Sn) capsule and analysed against a known set of standards (ANU Sucrose for 13C).

The effects of grazing intensity were subjected to a t-test, and, when variances were significantly different between groups, a t-test with separate variance was performed (Zar 1999). The relationship between soil organic matter and δ13C was analysed using Pearson correlation (Zar 1999). All statistical analyses were conducted using STATISTICA 12.0 software (StatSoft Inc., Tulsa, OK, USA).

3.2 Effects of rainfall – Spyglass grazing trial

The study was undertaken at the Spyglass Beef Research Facility (19°29’ S, 145°43’ E) north Queensland, Australia. Mean annual rainfall (July-June) in the 40 years prior to the onset of the experiment was 651 mm (Australian Government Bureau of Meteorology). The soil is classified as Lixic Ferralsol (Soil Survey Staff 2015). The vegetation consists of an open woodland of narrow-leaved ironbark (*Eucalyptus crebra*) and long-fruited bloodwood (*Corymbia polycarpa*), with an understorey of mainly black speargrass (*Heteropogon contortus*).

This area, like many of northern Australia’s soils, is low in nutrients compared to other ecosystems in the world. This is due to Australian landscapes being weathered relics of earlier pedological processes (Mott *et al.* 1985). In addition, northern Australia has a short and unreliable growing season, and a long dry season, making it difficult for cropping to be practiced (Carberry *et al.* 1993). As a result, commercial cattle grazing is by far the dominant land use, by area, over much of northern Australia, occupying 2.5 million square km of tropical semi-arid savannas and grasslands (Cook *et al.* 2010).

In order to test the effect of the amount of rainfall on soil nitrogen dynamics, we manipulated the amount of rainfall falling on the soil over a single wet season. Since, the frequency of wetting and drying of the soils has a large effect on the nitrogen cycle (Borken and Matzner 2009), we did not change the frequency of rainfall but rather only the amount. Reduction of rainfall was achieved by rainout shelters that reduced the amount of rain arriving to the ground to half of the actual natural rainfall. The rainout shelters were constructed by building nine square metre frames in the paddock. Clear polycarbonate strips were attached to the frame to allow light penetration but captured rainfall which was then guttered out from the plot into trenches in the ground that transported the water from the plots. The trenches also prevented overland flow from entering the plots from the surrounding area. Addition of rainfall was achieved by irrigating nine square metre plots with water (mostly rainwater captured into rainwater tanks from farm buildings) within 24 hours following the rainfall event. The amount of water irrigated was based on the actual rainfall amount measured on site and it was doubled to achieve a 100% increase in rainfall arriving at the ground. Additional nine square metre plots were erected that did not manipulate the amount of rainfall arriving at the plots, but included trenches to prevent overland flow. In order to test the direct effect of available soil nitrogen, we added 50 kg/N/ha of Cal-Gran fertiliser (Incitec Pivot Fertilisers) to half of the plots a week before the first rains of the season. Within the same paddock, six blocks were erected, with six plots (three rainfall treatments and two nitrogen levels) in each block, producing six replicates for each treatment.

Above ground vegetation was mowed, before the first rain and at the end of the season, dried and weighed in the lab. Soil samples were taken by a truck-mounted soil probe up to a depth of 40 cm, at the end of the wet season at three locations within each plot. The soil core was separated to 0-10, 10-20 and 20-40 cm depths. The three samples from the 10-20 and 20-40 cm depths of each plot were mixed together to form one sample. Soils were dried at 40˚C, large roots removed, soil crushed and passed through a 2-mm sieve. The dry soil was archived in sealed containers for further analysis. Mineral N (ammonium and nitrate-N) was extracted from the soil by shaking 8 g of soil in 20 mL of potassium chloride (2 M KCl) for 1 h and filtering through No. 41 filter paper. Nitrate and ammonium were determined in the KCl extracts by colorimetric methods (Best 1976; Keeney and Nelson 1982; Willis *et al.* 1993). Total N contents was measured by inserting 50 mg of fine-ground soil into an 8-mm×5-mm tin (Sn) capsule, and analysing it by combustion using a Costech Elemental Analyser (Costech International, Milan). Allocation of biomass to above and below ground vegetation (root to shoot ratio) was taken by collecting all of the shoots, roots and rhizomes in a cylinder of 73x50 mm.

The results of rainfall and nitrogen manipulations were subjected to a two-way randomised-block ANOVA (Zar 1999). When data violated the assumption of homoscedasticity, rank transformation was applied (Conover and Iman 1981; Zar 1999). Tukey–Kramer HSD (honest significant difference) tests were used for post hoc comparisons. All statistical analyses were conducted with the STATISTICA 12.0 software (StatSoft Inc., Tulsa, OK, USA).

4 Results

4.1 Effects of grazing – Wambiana grazing trial

The soil organic matter content was higher in the heavy grazing treatment in all years (Fig 1a), although this was only significant in 2008 and 2009 (2008: *t*-test $t_{45} = -2.79$, $P < 0.01$; 2009: *t*-test $t_{46} = -2.74$, $P < 0.01$; 2013: *t*-test $t_{43} = -1.62$, $P = 0.11$). Grazing treatment had no significant effect on the $\delta^{13}C$ signature in 2008 (Fig. 2; *t*-test $t_{45} = 0.44$, $P = 0.66$). However, $\delta^{13}C$ was negatively correlated with organic matter in 2008 ($r = -0.50$, $P < 0.01$).

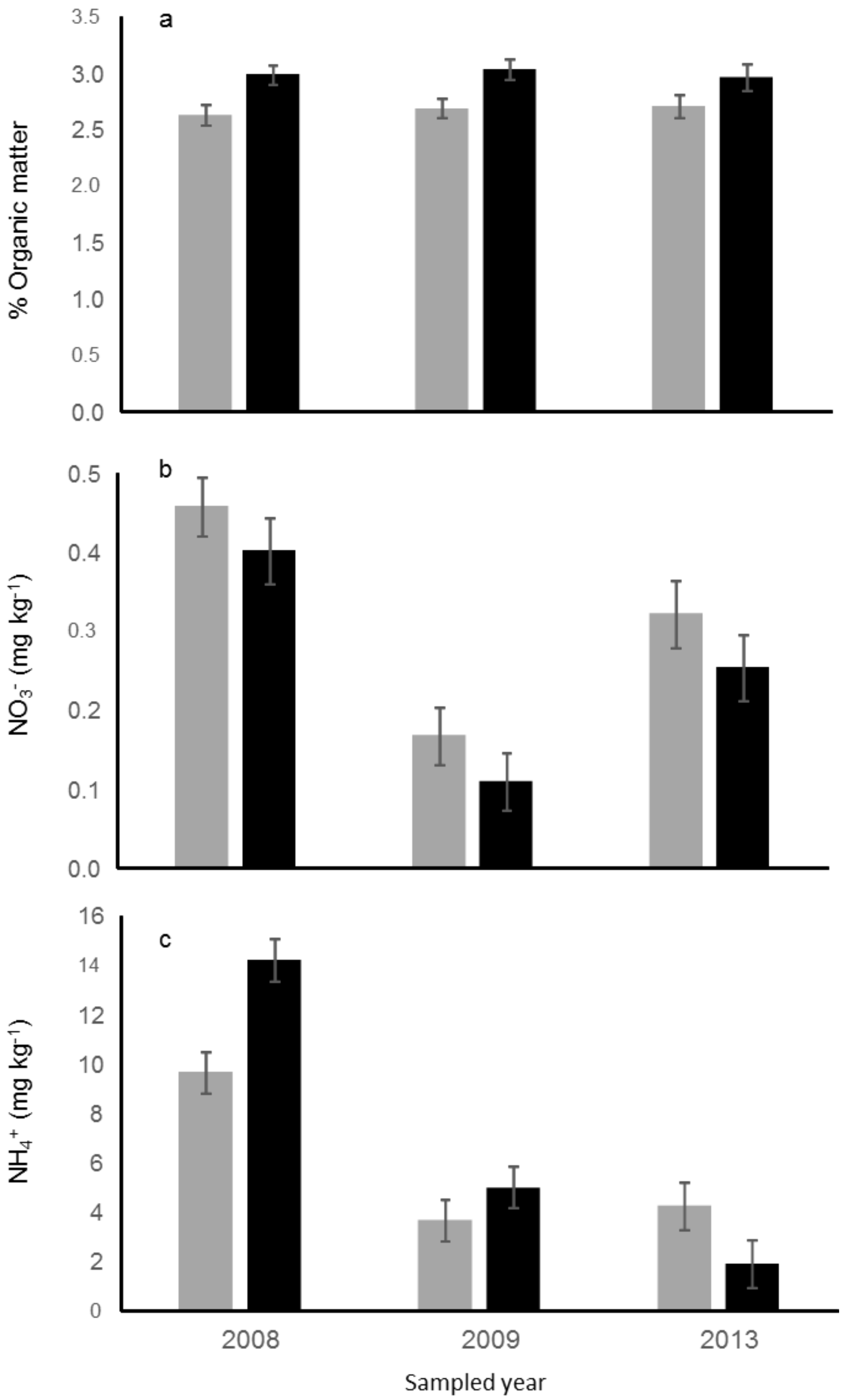

Figure 1. a) Soil organic matter, b) nitrate, and c) ammonium contents at 0-10 cm soil depth under moderate (grey) and heavy (black) grazing intensity rates on brown sodosol soil in 2008, 2009 and 2013. Data are represented as means ± SE.

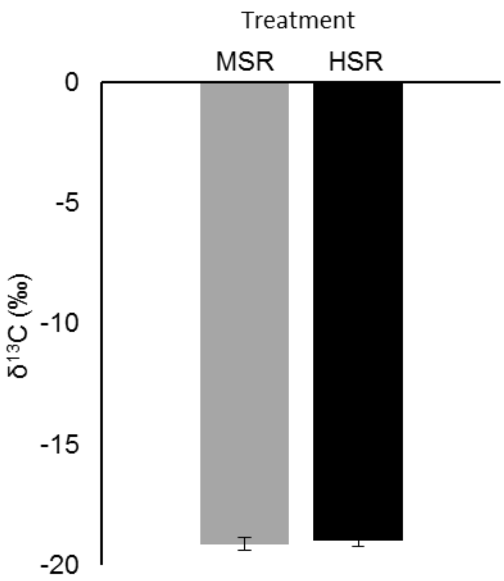

Figure 2. $\delta^{13}C$ values at 0-10 cm soil depth under moderate (grey) and heavy (black) grazing intensity rates on brown sodosol soil in 2008. Data are represented as means ± SE.

Nitrate content varied among the years (Fig. 1b) and was significantly higher in the moderate grazing treatment in 2008 and 2009 (2008: *t*-test $t_{48} = -2.04$, $P < 0.05$; 2009: *t*-test $t_{48} = -2.35$, $P = 0.02$; 2013: *t*-test $t_{43} = -0.90$, $P = 0.37$).

Ammonium content also varied among years (Fig. 1c). Ammonium content was higher in the heavy grazing treatment in 2008 and 2009, but was lower in the heavy grazing treatment in 2013 (2008: *t*-test with separate variance $t_{31,25} = -2.73$, $P = 0.01$; 2009: *t*-test with separate variance $t_{35,59} = -3.28$, $P < 0.01$; 2013: *t*-test with separate variance $t_{30,41} = 2.59$, $P = 0.01$).

Grazing treatment had no significant effect on the total nitrogen content in 2008 (Fig. 3; *t*-test $t_{45} = 1.15$, $P = 0.26$).

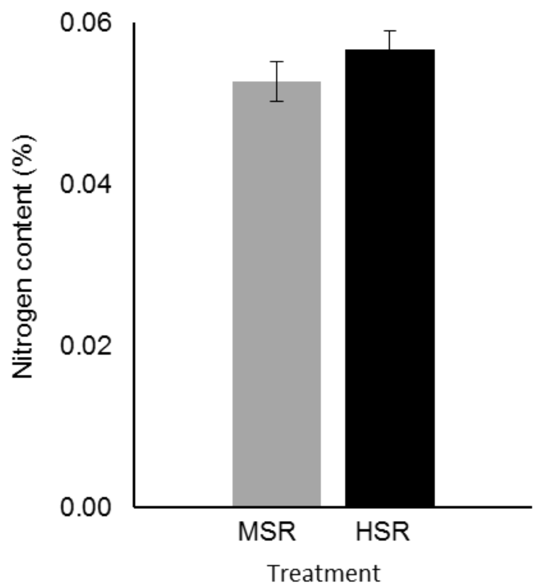

Figure 3. Total soil nitrogen content at 0-10 cm soil depth under moderate (grey) and heavy (black) grazing intensity rates on brown sodosol soil in 2008. Data are represented as means ± SE.

4.2 Effects of rainfall – Spyglass grazing trial

Between July 2012 and June 2013 there was 348 mm of rain (Fig. 4) concentrated between November and April, compared with a long term mean of 651 mm.

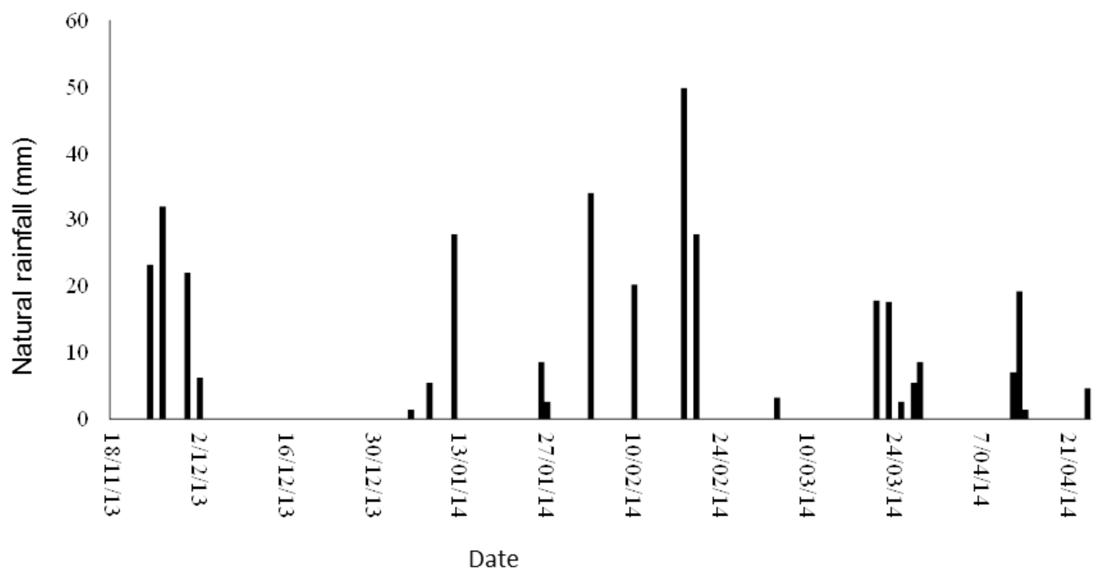

Figure 4. Rainfall events during the 2013-2014 wet season.

Rainfall had a significant effect on the amount of soil nitrate left in the soil at the end of the season at all three depths (0-10 cm – rank transformed data; two-way ANOVA; $F_{(2,63)}$ = 10.53, $P < 0.01$; 10-20 cm – rank transformed data; two-way ANOVA; $F_{(2,14)}$ = 9.78, $P < 0.01$;

20-40 cm – rank transformed data; two-way ANOVA; $F_{(2,20)} = 11.01$, $P < 0.01$), with increasing soil nitrate at lower rainfalls (Fig. 5). Addition of fertiliser had no significant effect at any depth.

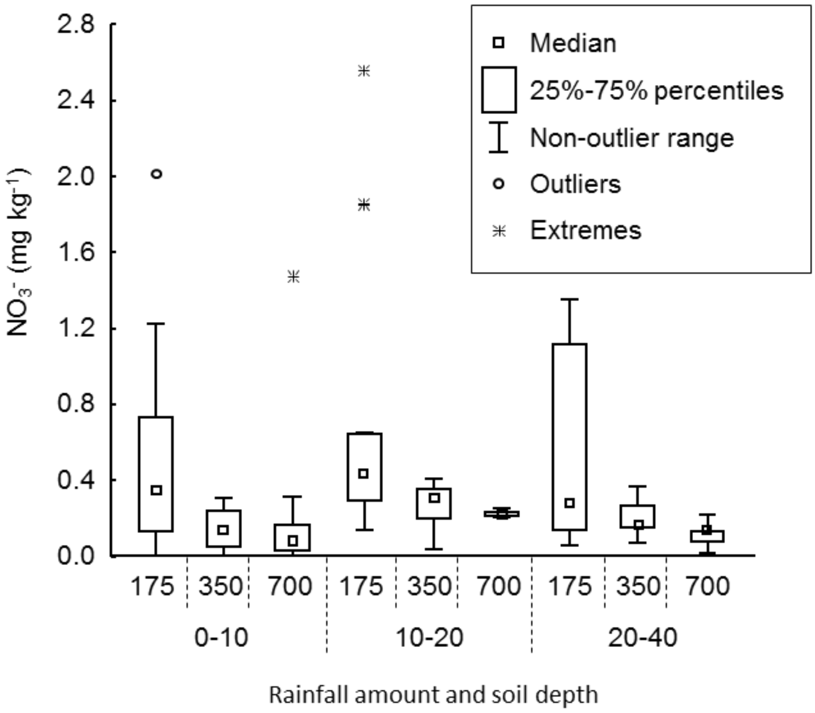

Figure 5. Nitrate content at 0-10, 10-20 and 20-40 cm depth, under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Neither rainfall nor nitrogen addition had an effect on ammonium (Fig. 6) or total nitrogen (Fig. 7) at any depth.

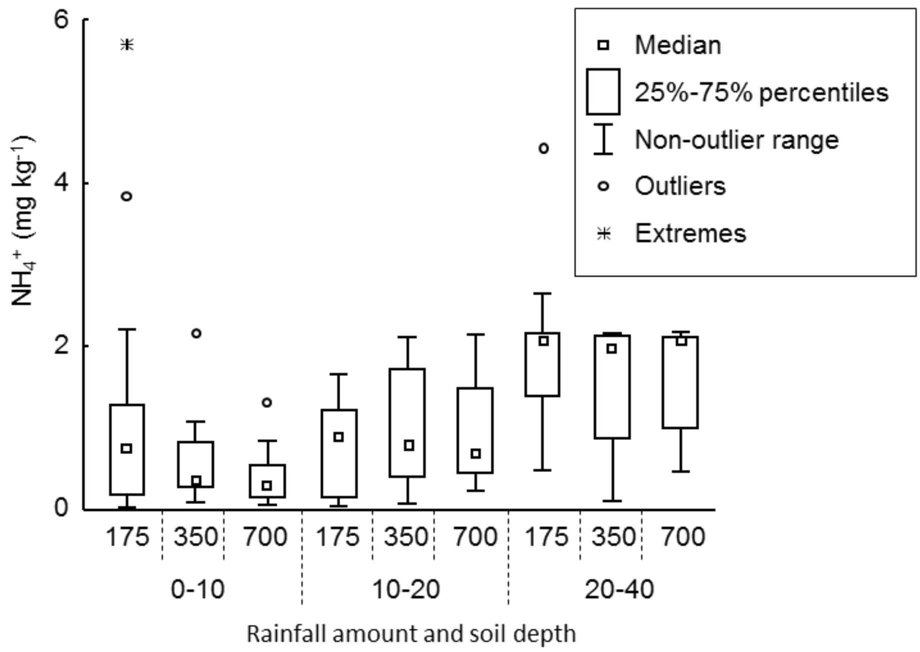

Figure 6. Ammonium content at 0-10, 10-20 and 20-40 cm depth, under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

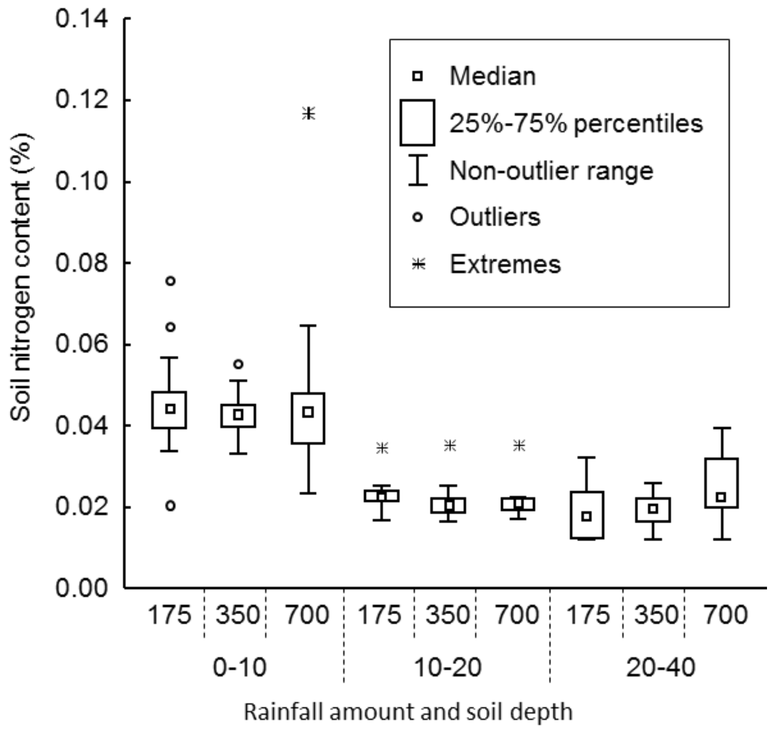

Figure 7. Total nitrogen abundance at 0-10, 10-20 and 20-40 cm depth, under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Rainfall had a significant effect on the nitrogen concentration in the shoot of black speargrass (rank transformed data; two-way ANOVA; $F_{(2,60)}$ = 18.28, P < 0.01), with higher values at lower rainfalls (Fig. 8). Nitrogen addition had no effect on shoot nitrogen.

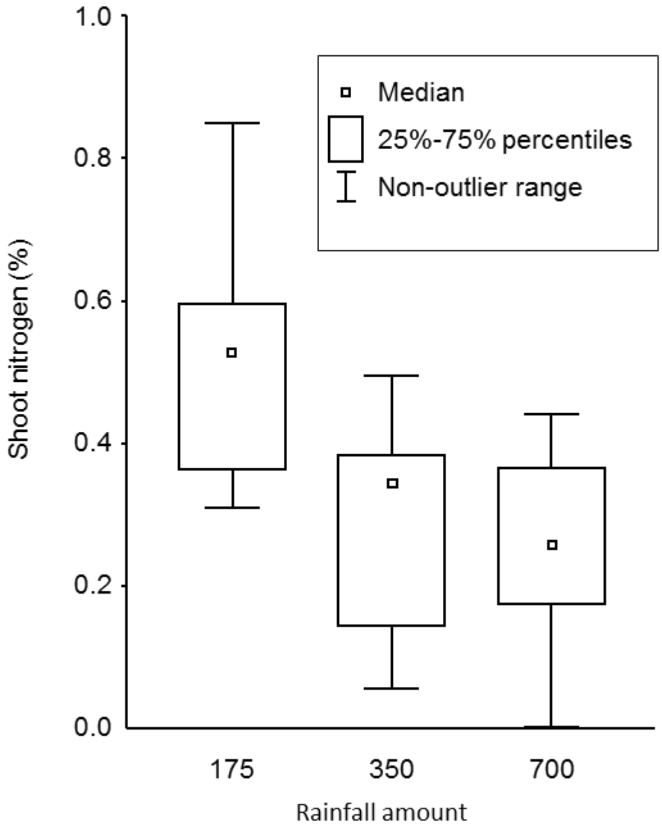

Figure 8. Shoot nitrogen concentration under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Rainfall had a significant effect on the root to shoot ratio (rank transformed data; two-way ANOVA; $F_{(2,30)} = 7.58$, $P < 0.01$), with increased allocation to below ground biomass at lower rainfalls (Fig. 9). Nitrogen addition had no effect on the root to shoot ratio.

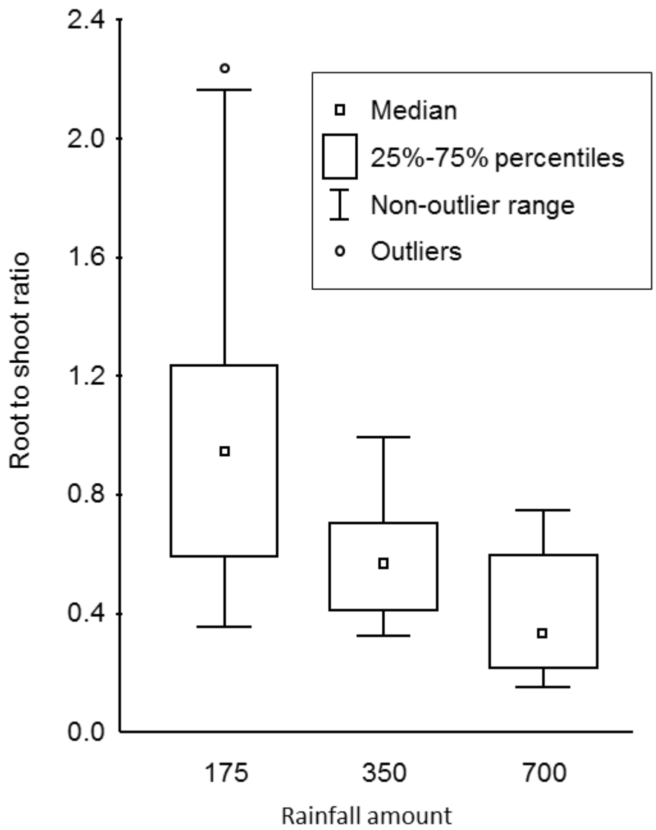

Figure 9. Root to shoot ratio under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Rainfall had a significant effect on the total biomass (rank transformed data; two-way ANOVA; $F_{(2,25)} = 18.75$, $P < 0.01$), with more biomass with increasing rainfall (Fig. 10). Nitrogen addition did not have a significant effect on total plant biomass.

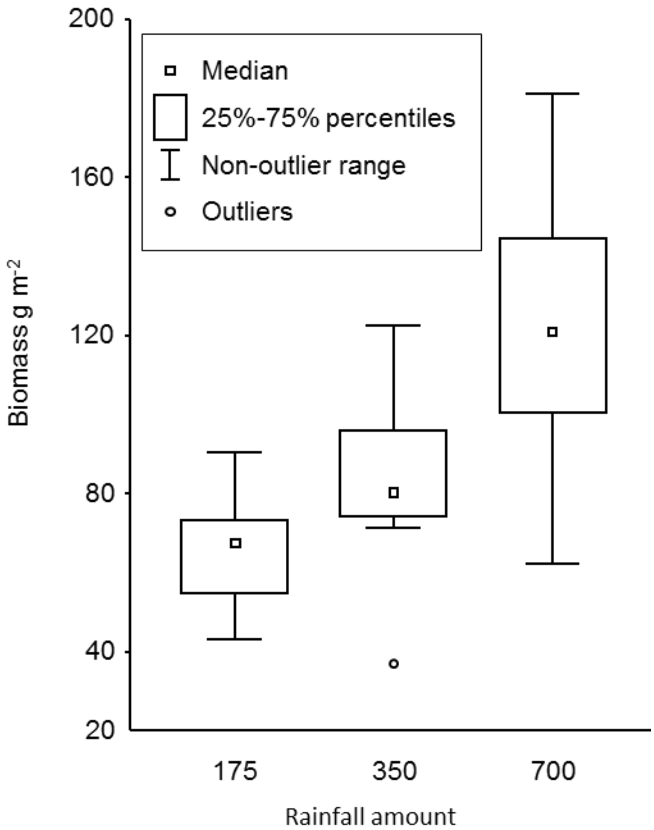

Figure 10. Total biomass under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

5 Discussion

Commercial grazing is the dominant commercial land use in the tropical semi-arid ecosystems of northern Australia (Cook et al. 2010). It is expected that grazing management will affect nutrient cycling in soils. We used a long-term grazing trial to test the effects of cattle stocking rates on interannual differences in soil organic matter, soil total nitrogen and soil mineral nitrogen. The results suggest that on Brown Sodosol–Yellow Kandosol complex, the HSR treatment had higher soil organic matter compared with the MSR treatment, consistent with Pringle et al. (2011), and with little interannual variability (Fig. 1a). The relatively small increase in soil organic matter under the HSR suggests a negligible ecological effect at the local scale. However, extrapolation of this small change over the large area occupied by commercial cattle grazing in northern Australia could affect national- or even global-scale processes (Ash et al. 1995). In contrast with soil organic matter, interannual variability was greater for soil mineral nitrogen, with no consistent effect of grazing treatment (Fig. 1b, c). This suggests that soil mineral nitrogen is relatively resistant to grazing, even at different rainfall intensities.

Pringle et al. (2011) explained the higher SOC in the HSR on this soil complex by a lower δ13C signature, suggesting a higher contribution to SOC from C3 shrubs and trees relative to the C4 perennial grasses (Krull and Bray 2005). After a fire in the treatments in 1999, the cover of the sprawling shrub *C. ovata* increased faster in the HSR than MSR treatment (Bray et al. 2014), but there was no significant difference in the total woody vegetation cover between the treatments (Bray et al. 2014; P. O'Reagain, unpubl. data). Preliminary soil analyses suggest that soil under *C. ovata* shrubs contains more carbon than surrounding soil (S. Bray, unpubl. data). The present data support the conclusions of Pringle et al. (2011) because we found a negative correlation between δ13C and soil organic matter, although no significant difference in δ13C was found between the grazing treatments. It is possible that our finding reflects the effects of depth and spatial scale on the strength of the correlation, or that the present experiment was too short to detect the effects of grazing on δ13C in total organic carbon (Pringle et al. 2011). Perhaps examining the δ13C by aggregate fractions could enhance the sensitivity and enable detection of the effects of grazing (Krull and Bray 2005).

The results found in the Brown Sodosol–Yellow Kandosol complex at the Wambiana site are not consistent with other soil organic matter studies on different soils in northern Australia (Ash et al. 1995; Holt 1997; Badgery et al. 2013; Cowie et al. 2013; Pringle et al. 2014). Pringle et al. (2011) found that SOC stocks (soil organic C concentration × bulk density × soil depth) at 0–10 cm depth were lower under high than low stocking rates on different soil types, namely a Grey Vertosol and a Red Kandosol, at the same trial site. In a recent survey of 98 sites, Allen et al. (2013) found a small but significant negative relationship between stocking rates and SOC. However, they concluded that the main drivers for SOC were climate and soil type. It was suggested that stocking rates affect SOC through changes in TSDM, grass species composition and root mass (O'Reagain and Bushell 2011; Badgery et al. 2013). In another long-term 26-year study of sheep grazing intensity on a Grey Vertosol, Pringle et al. (2014) found no significant difference in SOC stocks between six pasture utilisation rates ranging from 0% (exclosure) to 80% utilisation. However, there was a significant loss of total nitrogen from the topsoil under intensive pasture utilisation. Furthermore, a slight visual trend of declining SOC was found at higher utilisation rates.

The variability and inconsistency of results within and between studies across northern Australia indicate that caution is required when making claims on the benefits of improved grazing management on soil C stocks and nutrients in northern Australia. Although heavy grazing increased SOC on one soil in the present study, most other studies have found that heavy grazing can have severe negative effects, such as increased soil erosion (Eldridge 1998; O'Reagain et al. 2005), woody encroachment (Daryanto et al. 2013), changed species composition (Orr et al. 2010; Ash et al. 2011; Orr and Phelps 2013), reduced ground cover and soil surface condition (Ash et al. 2011). In addition, it has been shown that, in the longer term, heavy grazing reduces animal production and profitability (O'Reagain et al. 2009; O'Reagain and Bushell 2011).

Combining results from different soils can increase the predictive value of models, which are important for underpinning the calculation of Australia's national greenhouse accounts. National predictions of the effects of land use change could to include soil type in their assessments to reduce uncertainties and errors (Segoli et al. 2013). Furthermore, it is important for the landowner to know which soils are likely to sequester carbon with changes in land use. More research is needed on different soil types and variability within paddocks in order to build a practical managerial plan and make more accurate predictions on the long-term consequences of different management systems on soil carbon, carbon sequestration and greenhouse gas accounting. Special attention should be given to the effect of woody vegetation on soil carbon.

The interannual variability in soil nitrate content was greater than the effect of grazing intensity, probably because of the natural variability of nitrate content at seasonal and annual time scales (Liebig et al. 2014). For example, the lower nitrate content in 2009 than the other two years may reflect sampling season, because 2009 sampling was performed during the growing season when nitrate uptake by plants would be high as opposed to the other years, when the sampling was performed in the dry season (Chen et al. 2001). The higher amount of nitrate and ammonium in 2008 could be explained by the below-average rainfall observed during previous years (Table 1) that caused an accumulation of mineral nitrogen (Liebig et al. 2014). An absence of reliable measured data before the commencement of the experimental grazing trial restricts our capacity to attribute observed changes to long-term management. However, the fact that no biologically meaningful differences in mineral nitrogen were apparent after 20 years suggests that climate, rather than grazing intensity, is the main driver of mineral nitrogen dynamics. This is probable because atmospheric deposition is the dominant source of reactive nitrogen (Bobbink et al. 2010) and most of the plant nutrients consumed by livestock return to the soil (Chen et al. 2001).

Our results from the Spyglass experiment support our hypothesis that in drought conditions there is accumulation of available nitrate in the soil. This is consistent with other studies in other ecosystems (Evans and Burke 2013; Stark and Firestone 1995; Yahdjian et al. 2006). The accumulation could be due to a number of mechanisms that are not mutually exclusive. 1) Plants take up less nitrate in drier conditions due to plants growing less and therefore demanding less nitrogen (Borken and Matzner 2009). Although plants had higher nitrogen concentrations in drier conditions (Fig. 5), their lower biomass (Fig. 7) resulted in less total nitrogen uptake which can explain accumulation of nitrate in the soil in drier conditions. 2) Lower soil moisture limits diffusion of nitrate and ammonium to the plant roots and microbes (Evans and Burke 2013). As expected, plants allocated more biomass to roots in drier conditions, although nitrogen addition did not affect this relationship. 3) Reduced leaching of nitrate into the deeper soils when there is insufficient rainfall for deep drainage (Cregger et al. 2014). Since there was no increase in nitrogen in deeper soils, we do not expect leaching to be an important factor at the simulated rainfall conditions (Fig. 4). This accumulation of available nitrogen in drought conditions causes an asynchrony between the available nitrogen and plant/microbial demand (Augustine and McNaughton 2004), where nitrate levels are higher when soil moisture is lower and plants/microbes cannot utilize the nitrogen .

Our results did not show any effect of rainfall on soil ammonium or total nitrogen in the soil. Soil ammonium did show a non-significant trend of higher values in drier conditions at shallow depths, but we would have expected a stronger effect in the manipulated dry treatment. This is surprising since nitrification is considered to be mainly mediated by soil moisture (REF) and therefore ammonium should accumulate in low soil moisture conditions. A possible explanation for us not finding any effect could be that our sample size was too small to detect an effect considering the high heterogeneity in soil ammonium.

The addition of nitrogen did not have a significant effect on plant biomass and this may be explained by our experimental design. Our rainfall addition manipulation aimed at obtaining an above average wet year, assuming that rainfall received during the experiment was around average. However, as we experienced a dry year, the rainfall addition manipulation simulated an average year. Therefore, it is possible that there wasn't sufficient water available to remove the primary limitation of water (Bennett and Adams 2001). Alternatively, in the two weeks following the addition of nitrogen it rained 85 mm, and then it didn't rain for over a month (Fig. 1). This pulse of nitrogen followed by drying of the soil could have caused denitrification of the added nitrogen. Of course, the straightforward explanation that nitrogen does not limit productivity in this system is valid. Our data tends to support that the rainfall pattern denitrified the added nitrogen, since we found no effect of added nitrogen on soil nitrogen.

Our results show that plant biomass, at least up to average rainfall, is mainly limited by water availability in this ecosystem. We hypothesised that with higher levels of rainfall, water availability will not limit productivity and nitrogen availability will be the limiting factor. Therefore, the amount of soil nitrogen will determine the amount of plant biomass. We predict that in these scenarios, the previous year's amount of rainfall will determine the current year's plant biomass. Understanding this relationship can help us understand the dynamics of both soil nitrogen and productivity in these ecosystems. Additionally, quantification of the effects of rainfall on soil nitrogen and consequently productivity should improve our modelling and predictions of ecosystem productivity. This could lead to better management of stocking rates that increase profitability and reduce land degradation. Furthermore, this could also have implication for the best years of rehabilitation of degraded habitats and control actions of invasive species.

6 Conclusions/recommendations

This post-doctoral study aimed to better understand the interactions between grazing and rainfall and soil nutrients in the semi-arid grazing lands of northern Australia. Pasture and animal production in these semi-arid tropical regions are primarily limited by water but nutrient availability can have a significant effect on pasture and animal production. However, our understanding of the interactions between grazing, rainfall and nutrient dynamics is poorly understood, as are the implications for red meat production.

The soil analysis of long-term grazing treatments in the Wambiana grazing study showed soil organic matter to be slightly higher in the high stocking rate paddocks compared with the moderate stocking rate paddocks. Other studies have tended to show a slight decrease in soil organic matter in response to high grazing pressure but as in this study the differences are small and the biggest driver of soil organic carbon is climate and the inputs from pasture growth rather than grazing. This has important implications for producers wanting to implement grazing strategies to sequester carbon because the results from this and other studies suggest that the changes in soil carbon stocks maybe small depending on soil type, climate and vegetation. More research is needed on different soil types and variability within paddocks in order to build practical management strategies and make more accurate predictions on the long-term consequences of different management systems on soil carbon, carbon sequestration and greenhouse gas accounting.

Nitrogen, a key driver of protein in the diet of cattle, did not show a strong response to grazing treatments at the Wambiana site. Interannual variability in response to climate had a bigger influence on soil nitrogen. This suggests there is little option for influencing soil nitrogen through grazing management.

The rainfall manipulation experiment at Spyglass supported the hypothesis that in drought conditions there is accumulation of available nitrate in the soil. Although plants had higher nitrogen concentrations in drier conditions, their lower biomass resulted in less total nitrogen uptake which can explain accumulation of nitrate in the soil in drier conditions. This has implications for animal production in the year following a sequence of dry years as animal performance is likely to be much higher than average, not simply because of a return to higher pasture growth but that the protein content of this pasture is likely to be higher.

The results from the Spyglass study also showed that plant biomass, at least up to average rainfall, is mainly limited by water availability in this ecosystem. With higher levels of rainfall, water availability will not limit productivity and nitrogen availability will be the limiting factor. It was hoped that the experiment would be able to simulate much higher than average rainfall to observe the effect on nutrient dynamics under conditions where nutrients are likely to become limiting. However, the very dry years experienced during the experiment made this approach impractical. The implications of very wet years on nutrient dilution and reduced protein content in pasture for animal growth therefore remain unresolved. It is believed that better understanding of the relationship between rainfall and nitrogen dynamics could lead to better management of stocking rates, strategies to improve animal growth, and minimise land degradation.

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8 Outcome

The results of this project were published as a manuscript titled "Managing cattle grazing intensity: effects on soil organic matter and soil nitrogen" in Soil Research. An additional manuscript titled "Enhancement of soil nitrate by drought in the dry tropics" has been submitted to Soil Research. This project was also included in the conference proceedings of Soil Change Matters.

Details of these publications and communications are shown in section 10.

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10 Publications and communications from this project

Angela Anderson (2013) A range of research projects in the pipeline - Spyglass Beef Research Facility. Effects of nitrogen dynamics on pasture availability and quality. FUTURE BEEF NORTHERN MUSTER 12 December 2013. http: //www.futurebeef.com.au

Moran Segoli, Steven Bray, Diane Allen, Ram Dalal, Ian Watson, Andrew Ash, Peter O'Reagain (2014) Managing cattle for sustainable soil properties: interactions between stocking rates and rainfall. Conference Proceedings SOIL CHANGE MATTERS-PAPERS AND ABSTRACTS, Bendigo Victoria, Vic Dept. Environ. & Primary Ind. pp 139-143.

Moran Segoli, Steven Bray, Diane Allen, Ram Dalal, Ian Watson, Andrew Ash, and Peter O'Reagain (2015). Managing cattle grazing intensity: effects on soil organic matter and soil nitrogen. *Soil Research*, **53**, 677–682.

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11 Appendix 1

Managing cattle grazing intensity: effects on soil organic matter and soil nitrogen

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Managing cattle grazing intensity: effects on soil organic matter and soil nitrogen

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Abstract. Extensive cattle grazing is the dominant land use in northern Australia. It has been suggested that grazing intensity and rainfall have profound effects on the dynamics of soil nutrients in northern Australia's semi-arid rangelands. Previous studies have found positive, neutral and negative effects of grazing pressure on soil nutrients. These inconsistencies could be due to short-term experiments that do not capture the slow dynamics of some soil nutrients and the effects of interannual variability in rainfall. In a long-term cattle grazing trial in northern Australia on Brown Sodosol–Yellow Kandosol complex, we analysed soil organic matter and mineral nitrogen in surface soils (0–10 cm depth) 11, 12 and 16 years after trial establishment on experimental plots representing moderate stocking (stocked at the long-term carrying capacity for the region) and heavy stocking (stocked at twice the long-term carrying capacity). Higher soil organic matter was found under heavy stocking, although grazing treatment had little effect on mineral and total soil nitrogen. Interannual variability had a large effect on soil mineral nitrogen, but not on soil organic matter, suggesting that soil nitrogen levels observed in this soil complex may be affected by other indirect pathways, such as climate. The effect of interannual variability in rainfall and the effects of other soil types need to be explored further.

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Introduction

Many of northern Australia's soils are low in nutrients. This is because Australian landscapes are weathered relics of earlier pedological processes (Mott *et al*. 1985). In addition, northern Australia has a short and unreliable growing season and a long dry season, making it difficult for cropping to be practiced (Carberry *et al*. 1993). As a result, commercial cattle grazing is by far the dominant land use, by area, over much of northern Australia, occupying 2.5 million km^2 of tropical semi-arid savannas and grasslands (Cook *et al*. 2010). Because of this large area, grazing management practices that cause changes in soil nutrient dynamics could potentially affect national- or even global-scale processes. For example, it has been suggested that by improving land management, and hence land condition, in the livestock-grazed landscapes of northern Australia, hundreds of millions of tonnes of carbon could be sequestered in the top 10 cm of the soil (Ash *et al*. 1995; Garnaut 2008; Eady *et al*. 2009; Gifford 2010). In addition to helping reduce atmospheric CO_2 concentrations, this has the potential to improve soil and pasture productivity, improve livestock productivity and increase graziers' income through a higher value of livestock sales and incentives for carbon storage. This is especially important because grazing enterprises in north Australia are only marginally profitable (McLean *et al*. 2014). Finally, improving pasture and land condition can increase rainfall infiltration and decrease erosion, which is a major threat to the water quality of the Great Barrier Reef Lagoon (O'Reagain *et al*. 2005).

The soils of northern Australia are highly heterogeneous because of factors such as underlying geology, position in the landscape, climate and land use (Mott *et al*. 1985). Variations in soil properties may have a strong effect on pasture growth. For example, Mott *et al*. (1985) suggested that nitrogen was limiting for pasture growth on sesquioxidic soils, whereas soil water availability was limiting on the more fertile texture contrast soils. In other soils, water availability limited pasture growth in the drier years, whereas in wetter years pasture growth was limited by soil nitrogen (Mott *et al*. 1985). Furthermore, the effects of grazing depend on the soil type (McSherry and Ritchie 2013) and nutrient uptake by plants (Ash and McIvor 1995). For example, in northern Australia, higher grazing intensity was found to decrease soil carbon and nitrogen on Grey Vertosol and

Red Kandosol but to increase soil carbon on a Brown Sodosol–Yellow Kandosol complex (Pringle *et al*. 2011, 2014). Such variation suggests that the effects of grazing management on soil nutrient dynamics should consider local conditions and be extrapolated with caution.

Nitrogen and carbon dynamics are inextricably connected and cannot be understood independently of each other (Wedin 1995). The net amount of nitrogen mineralisation and availability to plants may be restricted by the high C : N ratio of soil organic matter (Mott *et al*. 1985; Wedin 1995). In addition, it has been suggested that there is a negative relationship between average rainfall and available nitrogen (Liebig *et al*. 2014). Therefore, it is important to manage the ecosystem for adequate available nitrogen. Even when soil nitrogen cannot be managed directly, it is important to understand its dynamics and drivers in order to better predict the amount of soil nitrogen, and consequently the amount of plant production and forage quality, under different management strategies. This information could be used by land owners to optimise their stocking rate and stocking strategies, leading to better management of their forage resource for animal productivity and maintaining land in good condition.

In northern Australia, grazing of native or introduced grasses and legumes without fertiliser application is the main management practice that affects nutrient dynamics and soil quality. It has been suggested that grazing affects soil nutrients through three main pathways in the absence of land degradation: (1) a net primary production pathway; (2) nitrogen pathways; and (3) decomposition pathways (Piñeiro *et al*. 2010). Because these pathways are not mutually exclusive and are affected by additional ecosystem properties, the effect of grazing on soil supply of nutrients is variable.

Manipulated field trials addressing the effects of grazing on soil supply of plant nutrients in the dry tropics are limited and often short term (Dalal and Carter 2000; Nelson and Roth 2004; for long-term studies, see Holt 1997; Pringle *et al*. 2011, 2014; Allen *et al*. 2013). However, it can take many years for some soil nutrients to respond to changes in land use or grazing intensity (Daryanto and Eldridge 2010; Segoli *et al*. 2012). Therefore, we took the opportunity of using soil samples collected in three different years from the dominant soil complex in an existing long-term grazing trial established in 1997 (O'Reagain *et al*. 2009) in order to study the effect of grazing intensity (i.e. stocking rates) and interannual rainfall variability on total soil nitrogen, mineral nitrogen and carbon.

In a previous study at the same location using data from a single sampling date, Pringle *et al*. (2011) showed that soil texture and stocking rate interact to affect soil organic carbon (SOC) stock. They found that the heavier grazing intensity treatment had a higher SOC stock in the dominant Brown Sodosol–Yellow Kandosol complex compared with the moderate grazing intensity treatment. Based on these results, we predicted that the heavy grazing treatment would have higher soil organic matter and lower soil nitrogen compared with the moderate grazing intensity treatment. However, interannual variation in rainfall may have a stronger effect on the soil's supply of plant nutrients, hence masking the long-term effects of grazing management in short-term studies.

Materials and methods

We used soil samples that were collected at a long-term cattle grazing trial conducted at Wambiana Station (20°34′S, 146°07′E), in north Queensland, Australia. The mean annual rainfall of the area is 636 mm, but is highly variable (Table 1), with most of the rainfall (70%) falling between December and March (O'Reagain *et al*. 2009). The area has been subjected to cattle grazing at light to moderate stocking rates for at least the past 130 years before the trial's establishment. In 1997, several ~100-ha paddocks were established to test a range of different grazing strategies (for more detail, see O'Reagain *et al*. 2009). Two contrasting strategies were selected for the present study: (1) a moderate stocking rate (MSR) treatment at ~8 ha per animal equivalent (450 kg steer), which is approximately the calculated long-term carrying capacity for the area; and (2) a heavy stocking rate (HSR) treatment at ~4 ha per animal equivalent (O'Reagain *et al*. 2009).

There are three major land types within the treatment paddocks, so to minimise this source of variability all soil samples were collected from the dominant *Eucalyptus brownii* savanna woodland on the Brown Sodosol–Yellow Kandosol complex (Northcote 1960; Cannon 1997; Isbell 2002), which covers approximately 55% of the site area. Two soil samples were collected to a 30 cm depth 2 years after the commencement of the trial and general soil properties measured (Table 2). Statistical inference cannot be performed from these limited soil samples, but they suggest that the two treatments were similar at the onset of the manipulation. The area sampled in the MSR and HSR treatments was a subsample of the area sampled by Pringle *et al*. (2011) and Bray *et al*. (2014). The area sampled is a relatively homogeneous area that had no apparent differences at the onset of the manipulations, with an arbitrary fence line dividing the grazing treatments. The vegetation consists of an open woodland of *E. brownii* and an understorey of the native shrub *Carissa ovata*. The grass layer is dominated by native C_4 tropical grasses, like *Bothriochloa ewartiana* and *Dichanthium fecundum*, but the exotic stoloniferous introduced grass *Bothriochloa pertusa* has spread widely on the trial area in the past 7–8 years. As a result of the grazing intensity treatment, there were major differences in both pasture total standing dry matter (TSDM) and pasture composition between the two treatments. On average, pasture TSDM was two- to threefold greater in the MSR (range 1324–3144 kg ha^{-1}) than in the HSR treatment (range 419–1090 kg ha^{-1}). At the same time, the relative contribution to yield of the more desirable perennial grasses like *B. ewartiana*

Table 1. Annual rainfall from July to June measured at the study area
For monthly rainfall see O'Reagain *et al*. (2009)

Year	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
Annual rainfall (mm year^{-1})	380	407	388	470	469	708	898	1032	715	1240	750	606

Table 2. Average of two soil samples that depict general soil properties (0–30 cm) in the moderate stocking rate (MSR) and heavy stocking rate (HSR) treatments on Brown Sodosol–Yellow Kandosol complex collected in 1999

EC, Electrical conductivity

Grazing strategy	pH	Coarse sand (%)	Fine sand (%)	Silt (%)	Clay (%)	Organic C (%)	EC (mS cm^{-1})	Phosphate (mg kg^{-1})	Potassium (%)
MSR	5.95	30	40	11	21	0.85	0.03	3.5	0.38
HSR	6.10	30	36	13	24	0.88	0.05	4.0	0.37

were approximately twofold greater in the MSR than HSR treatment. In contrast, the frequency and relative contribution to yield of the exotic *B. pertusa* was far higher in the HSR treatment (P. O'Reagain, unpubl. data; O'Reagain and Bushell 2011).

The soil samples were collected in July 2008, March–April 2009 and July 2013. Two adjacent 1-ha square-shaped areas (100 × 100 m) were selected on both sides of the arbitrary fence. In each area, 25 cores of 0–10 cm depth were sampled. However, because of a variety of issues, including missing or insufficient soil for laboratory analyses, the actual sample size was slightly lower (Table 3). Soil samples were dried at 40°C, large roots removed, soil crushed and passed through a 2-mm sieve. The dry soil was archived in sealed containers for further analysis.

Mineral N (ammonium and nitrate-N) was extracted from the soil by shaking 8 g soil in 20 mL potassium chloride (2 M) for 1 h and filtering through No. 41 filter paper. Nitrate and ammonium were determined in the KCl extracts by colourimetric methods (Best 1976; Keeney and Nelson 1982; Willis *et al*. 1993). Soil organic matter content was determined by the loss-on-ignition method (Segoli *et al*. 2012). Spain *et al*. (1982) demonstrated that loss-on-ignition provides a robust indicator of soil organic matter and organic carbon content in soils with low contents of carbonate and hydrous oxides, on similar soils to those examined herein.

The 2008 soils were subsampled to obtain total nitrogen and δ13C. Analysis was undertaken using an Isoprime isotope ratio mass spectrometer (IRMS) coupled to a Eurovector elemental analyser (Isoprime-EuroEA 3000, Isoprime Ltd, Stockport, UK) with HCl pre-treatment (concentration 10%) to remove carbonates (see Pringle *et al*. (2011), for further detail). Approximately 20 mg fine-ground soil was weighed into an 8 × 5 mm tin (Sn) capsule and analysed against a known set of standards (ANU Sucrose for ^{13}C).

The effects of grazing intensity were subjected to a *t*-test, and, when variances were significantly different between groups, a *t*-test with separate variance was performed (Zar 1999). The relationship between soil organic matter and $\delta^{13}C$ was analysed using Pearson correlation (Zar 1999). All statistical analyses were conducted using STATISTICA 12.0 software (StatSoft Inc., Tulsa, OK, USA).

Results

The soil organic matter content was higher in the heavy grazing treatment in all years (Fig. 1*a*), although this was only significant in 2008 and 2009 (2008: *t*-test $t_{45}=-2.79$, $P<0.01$; 2009: *t*-test $t_{46}=-2.74$, $P<0.01$; 2013: *t*-test $t_{43}=-1.62$, $P=0.11$). Grazing treatment had no significant effect on the $\delta^{13}C$ signature in 2008 (Fig. 2; *t*-test $t_{45}=0.44$, $P=0.66$). However, $\delta^{13}C$ was negatively correlated with organic matter in 2008 ($r=-0.50$, $P<0.01$).

Table 3. Number of soil samples analysed at 0–10 cm depth for the different soil measurements in the different years and in different treatments on Brown Sodosol–Yellow Kandosol complex

SOM, Soil organic matter; MSR, moderate stocking rate; HSR heavy stocking rate

Year	Grazing pressure	SOM	NO_3	NH_4
2008	MSR	22	25	25
2008	HSR	25	25	25
2009	MSR	25	25	25
2009	HSR	23	25	25
2013	MSR	25	25	25
2013	HSR	20	20	20

Nitrate content varied among the years (Fig. 1*b*) and was significantly higher in the moderate grazing treatment in 2008 and 2009 (2008: *t*-test $t_{48}=-2.04$, $P<0.05$; 2009: *t*-test $t_{48}=-2.35$, $P=0.02$; 2013: *t*-test $t_{43}=-0.90$, $P=0.37$).

Ammonium content also varied among years (Fig. 1*c*). Ammonium content was higher in the heavy grazing treatment in 2008 and 2009, but was lower in the heavy grazing treatment in 2013 (2008: *t*-test with separate variance $t_{31,25}=-2.73$, $P=0.01$; 2009: *t*-test with separate variance $t_{35,59}=-3.28$, $P<0.01$; 2013: *t*-test with separate variance $t_{30,41}=2.59$, $P=0.01$).

Grazing treatment had no significant effect on the total nitrogen content in 2008 (Fig. 3; *t*-test $t_{45}=1.15$, $P=0.26$).

Discussion

Commercial grazing is the dominant land use in the tropical semi-arid ecosystems of northern Australia (Cook *et al*. 2010). It is expected that grazing management will affect nutrient cycling in soils. We used a long-term grazing trial to test the effects of cattle stocking rates on interannual differences in soil organic matter, soil total nitrogen and soil mineral nitrogen. The results suggest that on Brown Sodosol–Yellow Kandosol complex, the HSR treatment had higher soil organic matter compared with the MSR treatment, consistent with Pringle *et al*. (2011), and with little interannual variability (Fig. 1*a*). The relatively small increase in soil organic matter under the HSR suggests a negligible ecological effect at the local scale. However, extrapolation of this small change over the large area occupied by commercial cattle grazing in northern Australia could affect national- or even global-scale processes (Ash *et al*. 1995). In contrast with soil organic matter, interannual variability was greater for soil mineral nitrogen, with no consistent effect of grazing treatment (Fig. 1*b*, *c*). This suggests that soil mineral nitrogen is relatively resistant to grazing, even at different rainfall intensities.

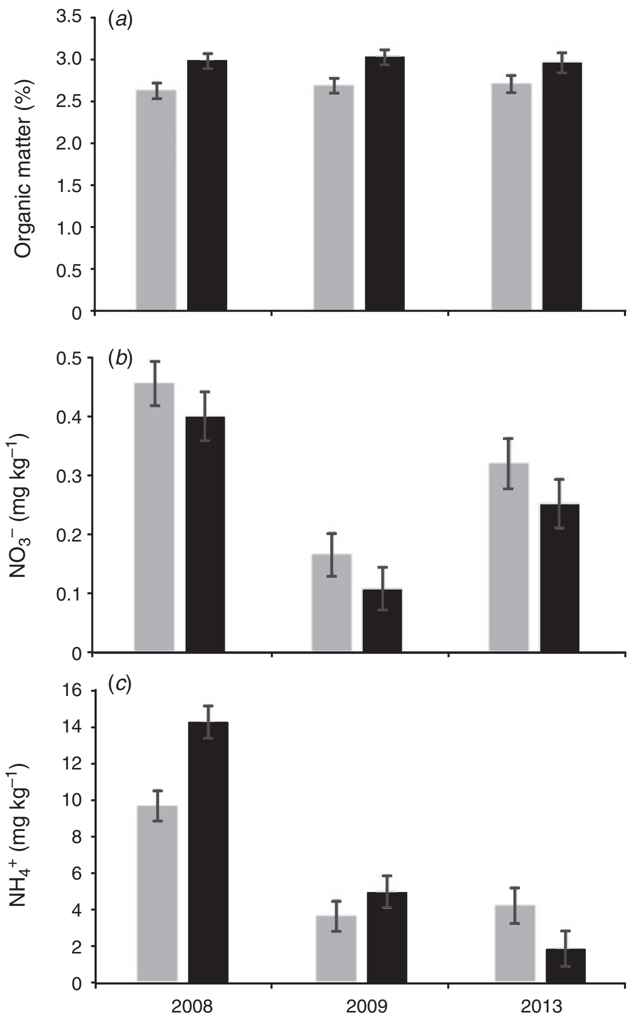

Fig. 1. (*a*) Soil organic matter, (*b*) nitrate, and (*c*) ammonium contents at 0–10 cm soil depth under moderate (grey) and heavy (black) grazing intensity rates on Brown Sodosol–Yellow Kandosol complex in 2008, 2009 and 2013. Data are the mean ± s.e.

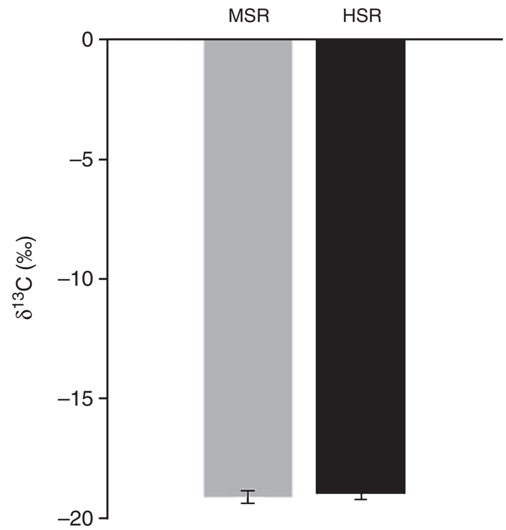

Fig. 2. $\delta^{13}C$ values at 0–10 cm soil depth under moderate and heavy stocking rates (MSR and HSR, respectively) on Brown Sodosol–Yellow Kandosol complex in 2008. Data are the mean ± s.e.

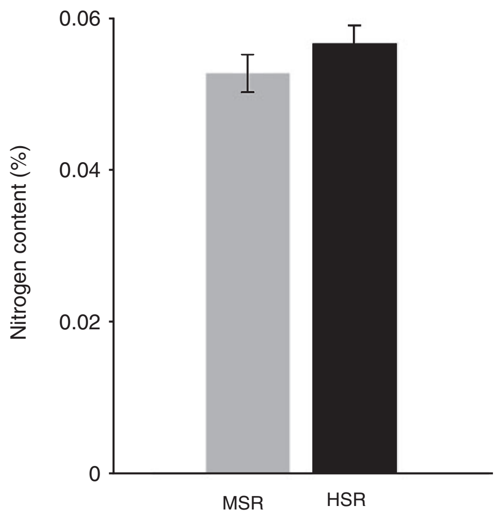

Fig. 3. Total soil nitrogen content at 0–10 cm soil depth under moderate and heavy stocking rates (MSR and HSR, respectively) on Brown Sodosol–Yellow Kandosol complex in 2008. Data are the mean ± s.e.

Pringle *et al*. (2011) explained the higher SOC in the HSR on this soil complex by a lower $\delta^{13}C$ signature, suggesting a higher contribution to SOC from C_3 shrubs and trees relative to the C_4 perennial grasses (Krull and Bray 2005). After a fire in the treatments in 1999, the cover of the sprawling shrub *C. ovata* increased faster in the HSR than MSR treatment (Bray *et al*. 2014), but there was no significant difference in the total woody vegetation cover between the treatments (Bray *et al*. 2014; P. O'Reagain, unpubl. data). Preliminary soil analyses suggest that soil under *C. ovata* shrubs contains more carbon than surrounding soil (S. Bray, unpubl. data). The present data support the conclusions of Pringle *et al*. (2011) because we found a negative correlation between $\delta^{13}C$ and soil organic matter, although no significant difference in $\delta^{13}C$ was found between the grazing treatments. It is possible that our finding reflects the effects of depth and spatial scale on the strength of the correlation, or that the present experiment was too short to detect the effects of grazing on $\delta^{13}C$ in total organic carbon (Pringle *et al*. 2011). Perhaps examining the $\delta^{13}C$ by aggregate fractions could enhance the sensitivity and enable detection of the effects of grazing (Krull and Bray 2005).

The results found in the Brown Sodosol–Yellow Kandosol complex at the Wambiana site are not consistent with other soil organic matter studies on different soils in northern Australia (Ash *et al*. 1995; Holt 1997; Badgery *et al*. 2013; Cowie *et al*. 2013; Pringle *et al*. 2014). For example, Pringle *et al*. (2011) found that SOC stocks (soil organic C concentration × bulk density × soil depth) at 0–10 cm depth were lower under high than low stocking rates on different soil types, namely a Grey Vertosol and a Red Kandosol, at the same trial site. In a recent survey of 98 sites, Allen *et al*. (2013) found a small but significant negative relationship between stocking rates and SOC. However, they concluded that the main drivers for SOC were climate and soil type. It was suggested that stocking rates affect SOC through changes in TSDM, grass species composition and root mass (O'Reagain and Bushell 2011; Badgery *et al*. 2013). In another long-term 26-year study of sheep grazing intensity on a Grey Vertosol, Pringle *et al*. (2014) found no significant difference in SOC stocks between six pasture utilisation rates ranging from 0% (exclosure) to 80% utilisation. However, there was a significant loss of total nitrogen from the topsoil under intensive pasture utilisation. Furthermore, a slight visual trend of declining SOC was found at higher utilisation rates.

The variability and inconsistency of results within and between studies across northern Australia indicate that caution is required when making claims on the benefits of improved grazing management on soil C stocks and nutrients in northern Australia. Although heavy grazing increased SOC on one soil in the present study, most other studies have found that heavy grazing can have severe negative effects, such as increased soil erosion (Eldridge 1998; O'Reagain *et al*. 2005), woody encroachment (Daryanto *et al*. 2013), changed species composition (Orr *et al*. 2010; Ash *et al*. 2011; Orr and Phelps 2013), reduced ground cover and soil surface condition (Ash *et al*. 2011). In addition, it has been shown that, in the longer term, heavy grazing reduces animal production and profitability (O'Reagain *et al*. 2009; O'Reagain and Bushell 2011).

In addition, combining these results with other results from different soils can increase the predictive value of models, which are important for underpinning the calculation of Australia's national greenhouse accounts. National predictions of the effects of land use change need to include soil type in their assessments to reduce uncertainties and errors (Segoli *et al*. 2013). Furthermore, it is important for the landowner to know which soils are likely to sequester carbon with changes in land use. More research is needed on different soil types and variability within paddocks in order to build a practical managerial plan and make more accurate predictions on the long-term consequences of different management systems on soil carbon, carbon sequestration and greenhouse gas accounting. Special attention should be given to the effect of woody vegetation on soil carbon.

The interannual variability in soil nitrate content was greater than the effect of grazing intensity, probably because of the natural variability of nitrate content at seasonal and annual time scales (Liebig *et al*. 2014). For example, the lower nitrate content in 2009 than the other two years may reflect sampling season, because 2009 sampling was performed during the growing season when nitrate uptake by plants would be high as opposed to the other years, when the sampling was performed in the dry season (Chen *et al*. 2001). The higher amount of nitrate and ammonium in 2008 could be explained by the below-average rainfall observed during previous years (Table 1) that caused an accumulation of mineral nitrogen (Liebig *et al*. 2014). An absence of reliable measured data before the commencement of the experimental grazing trial restricts our capacity to attribute observed changes to long-term management. However, the fact that no biologically meaningful differences in mineral nitrogen were apparent after 20 years suggests that climate, rather than grazing intensity, is the main driver of mineral nitrogen dynamics. This is probable because atmospheric deposition is the dominant source of reactive nitrogen (Bobbink *et al*. 2010) and most of the plant nutrients consumed by livestock return to the soil (Chen *et al*. 2001). This also emphasises the importance of studying the effect of rainfall on soil nitrogen and the carry-over effect from previous years (Liebig *et al*. 2014). Better understanding of the dynamics of soil nitrogen and its drivers should improve our ability to predict levels of available soil nitrogen. This will likely lead to better management of stocking rates and grazing strategies that increase profitability and reduce land degradation.

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12 Appendix 2

Enhancement of soil nitrate by drought in the dry tropics

Enhancement of soil nitrate by drought in the dry tropics

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Introduction

Productivity in most arid and semi-arid ecosystems is primarily limited by water availability, and in years that there is sufficient water, nitrogen availability usually becomes a secondary, but important, limiting factor (Harpole *et al.* 2007; Lauenroth *et al.* 1978; Wedin 1995). Nitrogen stocks and flows not only affect primary production but they can be an important controller of secondary production e.g. domestic livestock (Myers and Robbins 1991). Available nitrogen enters the ecosystem from 1) biological N fixation; 2) mineralization; and 3) atmospheric deposition. Since the agriculture and industrial revolutions, atmospheric deposition can be the dominant source (Bobbink *et al.* 2010) of nitrogen addition. Although nitrogen is consumed and transformed through the system, most of it stays within the ecosystem. For example, 60-90% of plant nitrogen that is consumed by livestock returns to the soil as excreta (Chen *et al.* 2001).

The effect of drought on the processes involved in the N cycle are largely unknown, but it has been suggested that it is of significant importance (Borken and Matzner 2009; Huntley and Walker 1982; Yahdjian *et al.* 2006). Most of the nitrogen in the system is unavailable to plants and needs to be transformed (e.g., mineralization and nitrification) into available nitrogen sources (e.g., nitrate, ammonium). Both mineralization and nitrification are controlled by soil moisture (Borken and Matzner 2009), so we could expect that in drier conditions there will be less available nitrogen in the soil (Tietema *et al.* 1992). However, in drier conditions plants have less biomass and consequently extract less nitrogen from the soil (Borken and Matzner 2009) that could result in excess soil nitrogen. This, in addition to nitrate moving more slowly through the soil and into the plants roots (Evans and Burke

2013), can result in an accumulation of available soil nitrogen in drier conditions. Furthermore, heavier rainfall increases the risk of leaching of nitrate which will leave less available soil nitrogen for the plant's roots in the soil. Since these factors are contradictory to each other, the net effect of drought on the available nitrogen in the soil depends on the ecosystem properties (Borken and Matzner 2009), although most studies found that drought increases available nitrogen (Evans and Burke 2013; Jackson *et al.* 1988; Kreuzwieser and Gessler 2010; Reynolds *et al.* 1999; Stark and Firestone 1995; Weltzin *et al.* 2003; Yahdjian *et al.* 2006). Northern Australia's dry tropic ecosystems could have different nitrogen cycle processes compared to other ecosystems around the world. This is due to a short wet, hot and humid season that enables a large pulse of biological activity in a short timeframe. During the long dry season, biological activity should be reduced significantly and consequently nitrogen processes should also slow down. However, the nitrogen cycle in the dry tropics has not been studied extensively (García-Méndez *et al.* 1991).

The effect of increased available nitrogen in the soil after droughts can increase biomass in the following year, if water availability is sufficient (Briggs and Knapp 1995; Clenton *et al.* 1970; Cregger *et al.* 2014; Knapp *et al.* 2008). Additionally, litter that accumulates in drought years can decompose and release more soil nitrogen in following years (Chen *et al.* 2001; Wedin 1995). This would suggest that in wet years when water availability does not limit productivity, the amount of rain in the previous year could control the amount of productivity through available soil nitrogen that is carried over from the previous year.

Drought is common in northern Australia and it has a large impact on livestock productivity, farm economics and land degradation (Stafford Smith *et al.* 2007). Most climate models predict that drought episodes will be more common and more severe with climate change (Watterson *et al.* 2015), further exacerbating challenges in managing climate variability. Therefore, a better understanding of the effects of drought on soil nitrogen can have implications for better management of cattle production in these ecosystems.

We aimed to understand the effects of interannual variability in rainfall on soil nitrogen and consequently plant biomass and forage quality. Specifically, we hypothesised that dry years will cause an accumulation of soil nitrogen, while wet years will leave less nitrogen in the soil. This will have consequences on the productivity of vegetation in the following year.

Methods

The study was undertaken at the Spyglass Beef Research Facility (19°29′ S, 145°43′ E) north Queensland, Australia. Mean annual rainfall (July-June) of the area in the 40 years prior to the onset of the experiment was 651 mm (Australian Government Bureau of Meteorology). The soil is classified as Lixic Ferralsol (Soil Survey Staff 2015) (Table 1). The vegetation consists of an open woodland of narrow-leaved ironbark (*Eucalyptus crebra*) and long-fruited bloodwood (*Corymbia polycarpa*), with an understorey of mainly black speargrass (*Heteropogon contortus*).

This area, like many of northern Australia's soils, is low in nutrients compared to other ecosystems in the world. This is due to Australian landscapes being weathered relics of earlier pedological processes (Mott *et al.* 1985). In addition, northern Australia has a short and unreliable growing season, and a long dry season, making it difficult for cropping to be practiced (Carberry *et al.* 1993). As a result, commercial cattle grazing is by far the dominant land use, by area, over much of northern Australia, occupying 2.5 million square km of tropical semi-arid savannas and grasslands (Cook *et al.* 2010).

In order to test the effect of the amount of rainfall on soil nitrogen dynamics, we manipulated the amount of rainfall falling on the soil over a single wet season. Since, the frequency of wetting and drying of the soils has a large effect on the nitrogen cycle (Borken and Matzner 2009), we did not change the frequency of rainfall but rather only the amount. Reduction of rainfall was achieved by rainout shelters that reduced the amount of rain arriving to the ground to half of the actual natural rainfall. The rainout shelters were constructed by building nine square meter frames in the paddock. Clear polycarbonate strips were attached to the frame to allow light penetration but captured rainfall which was then guttered out from the plot into trenches in the ground that transported the water from the plots. The trenches also prevented overland flow from entering the plots from the surrounding area. Addition of rainfall was achieved by irrigating nine square meter plots with water (mostly rainwater captured into rainwater tanks from farm buildings supplemented with water extracted from a farm dam) within 24 hours following the rainfall event. The amount of water irrigated was based on the actual rainfall amount measured on site and it was doubled to achieve a 100% increase in rainfall arriving at the ground. Additional nine square meter plots were erected that did not manipulate the amount of rainfall arriving at the plots, but included trenches to prevent overland flow. In order to test the direct effect of available soil nitrogen, we added 50 kg/N/ha of Cal-Gran fertilizer

(Incitec Pivot Fertilisers) to half of the plots a week before the first rains of the season, scattered on the soil surface. Within the same paddock, six blocks were erected, with six plots (three rainfall treatments and two nitrogen levels) in each block, producing six replicates for each treatment.

Above ground vegetation was mowed, before the first rain and at the end of the season, dried and weighed in the lab. Soil samples were taken by a truck-mounted soil probe up to a depth of 40 cm, at the end of the wet season at three locations within each plot. The soil core was separated to 0-10, 10-20 and 20-40 cm depths. The three samples from the 10-20 and 20-40 cm depths of each plot were mixed together to form one sample. Soils were dried at 40˚C, large roots removed, soil crushed and passed through a 2-mm sieve. The dry soil was archived in sealed containers for further analysis. Mineral N (ammonium and nitrate-N) was extracted from the soil by shaking 8 g of soil in 20 mL of potassium chloride (2 M KCl) for 1 hr and filtering through No. 41 filter paper. Nitrate and ammonium were determined in the KCl extracts by colorimetric methods (Best 1976; Keeney and Nelson 1982; Willis *et al.* 1993). Total N contents was measured by inserting 50 mg of fine-ground soil into an 8-mm×5-mm tin (Sn) capsule, and analysing it by combustion using a Costech Elemental Analyser (Costech International, Milan). Allocation of biomass to above and below ground vegetation (root to shoot ratio) was taken by collecting all of the shoots, roots and rhizomes in a cylinder of 73x50 mm. The shoots (leaves and stems) were analysed for nitrogen concentration.

The results of rainfall and nitrogen manipulations were subjected to a two-way randomised-block ANOVA (Zar 1999). When data violated the assumption of homoscedasticity, rank transformation was applied to the data (Conover and Iman 1981; Zar 1999). Tukey–Kramer hsd tests were used for post hoc comparisons. All statistical analyses were conducted with the STATISTICA 12.0 software (StatSoft Inc., Tulsa, OK, USA).

Results

Between July 2012 and June 2013 there was 348 mm precipitation (Fig. 1) concentrated between November and April, compared with a long term mean of 651 mm.

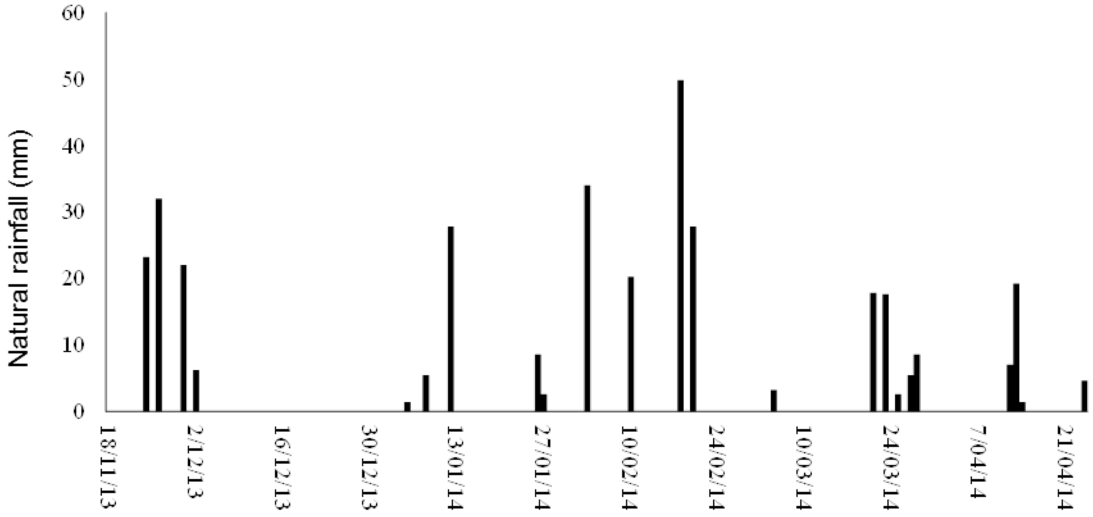

Figure 1. Rainfall events during the 2013-2014 wet season.

Rainfall had a significant effect on the amount of soil nitrate left in the soil at the end of the season at all three depths (0-10 cm – rank transformed data; two-way ANOVA; $F_{(2,63)}$ = 10.53, P < 0.01; 10-20 cm – rank transformed data; two-way ANOVA; $F_{(2,14)}$ = 9.78, P < 0.01; 20-40 cm – rank transformed data; two-way ANOVA; $F_{(2,20)}$ = 11.01, P < 0.01), with increasing soil nitrate at lower rainfalls (Fig. 2). Addition of fertilizer had no significant effect at any depth.

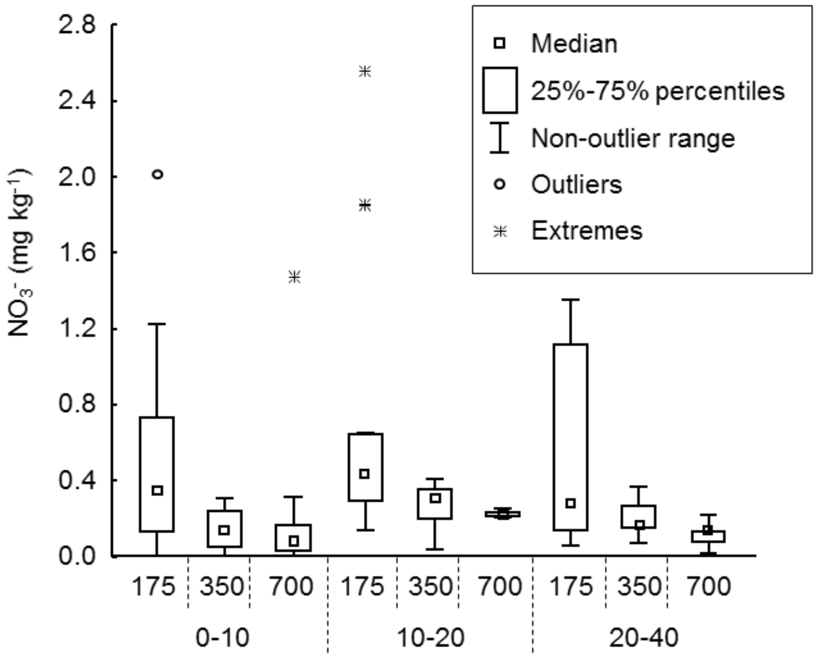

Figure 2. Nitrate content at 0-10, 10-20 and 20-40 cm depth, under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Neither rainfall nor nitrogen addition had an effect on ammonium (Fig. 3) or total nitrogen (Fig. 4) at any depth.

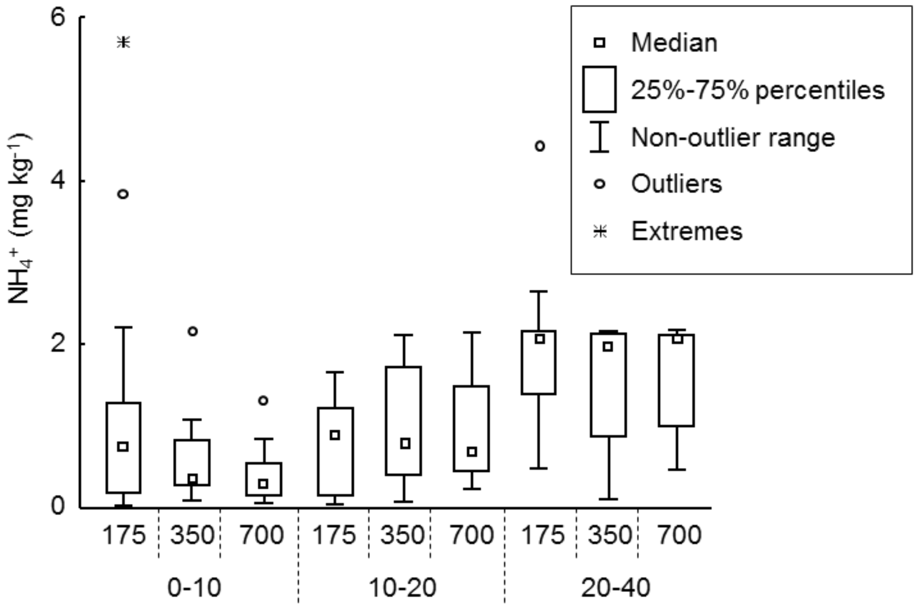

Figure 3. Ammonium content at 0-10, 10-20 and 20-40 cm depth, under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

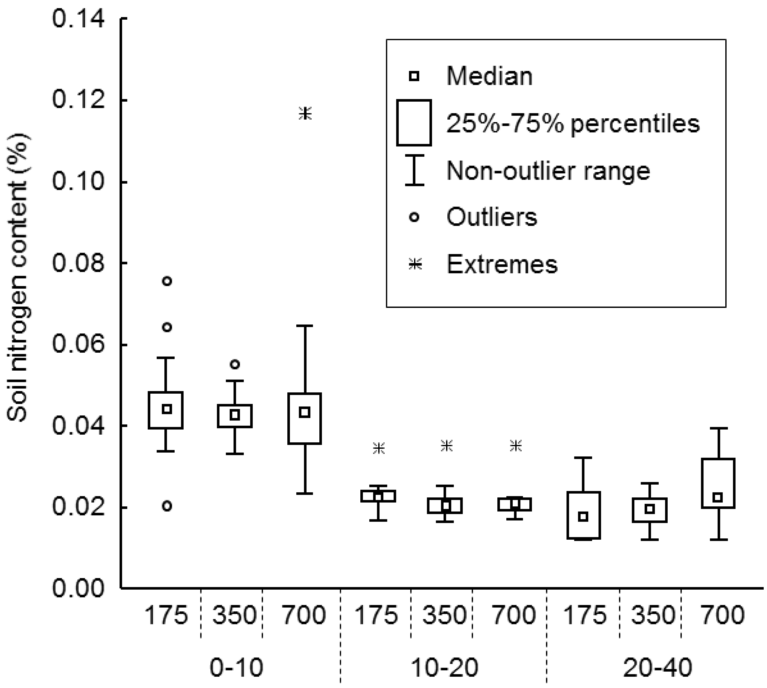

Figure 4. Total nitrogen abundance at 0-10, 10-20 and 20-40 cm depth, under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Rainfall had a significant effect on the nitrogen concentration in the shoot of black speargrass (rank transformed data; two-way ANOVA; $F_{(2,60)}$ = 18.28, P < 0.01), with higher values at lower rainfalls (Fig. 5). Nitrogen addition had no effect on shoot nitrogen.

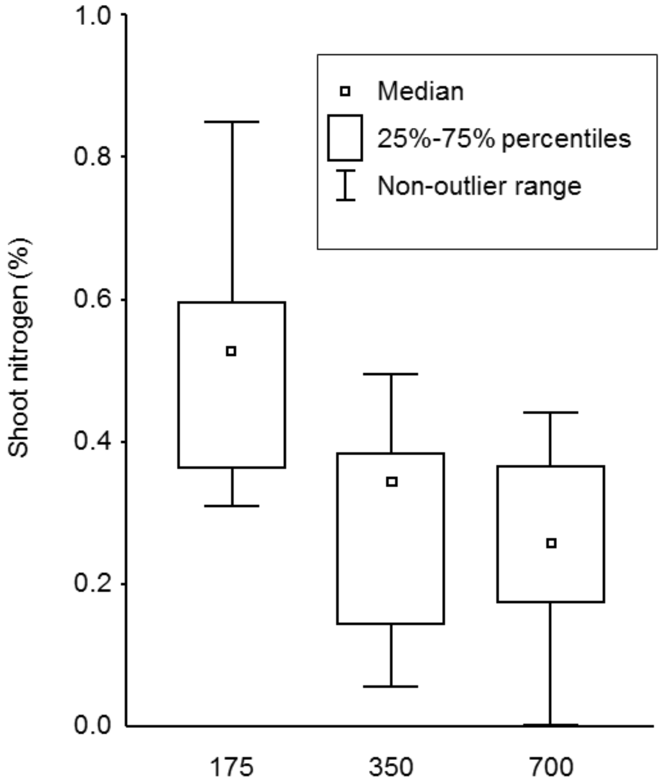

Figure 5. Shoot nitrogen concentration under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Rainfall had a significant effect on the root to shoot ratio (rank transformed data; two-way ANOVA; $F_{(2,30)}$ = 7.58, P < 0.01), with increased allocation to below ground biomass at lower rainfalls (Fig. 6). Nitrogen addition had no effect on the root to shoot ratio.

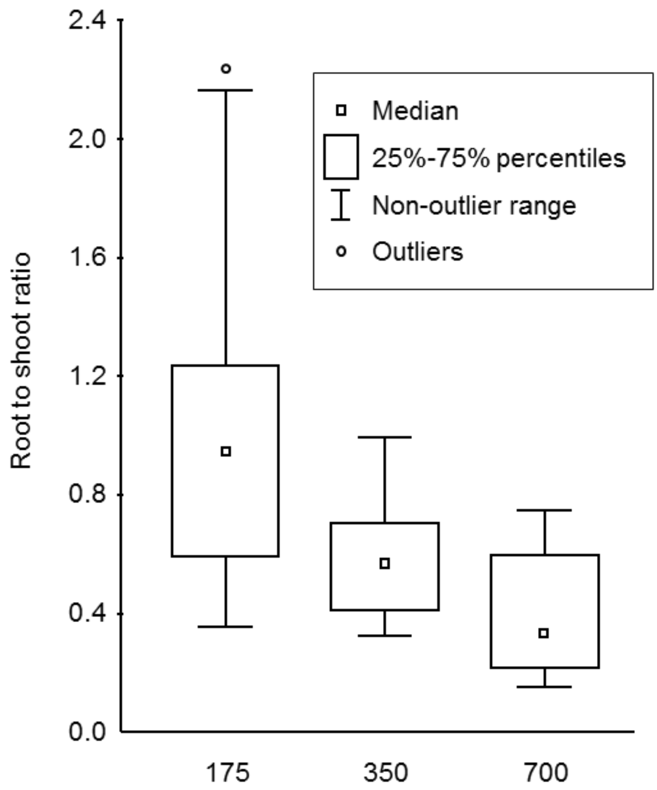

Figure 6. Root to shoot ratio under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Rainfall had a significant effect on the total biomass (rank transformed data; two-way ANOVA; $F_{(2,25)} = 18.75$, $P < 0.01$), with more biomass with increasing rainfall (Fig. 7). Nitrogen addition did not have a significant effect on total plant biomass.

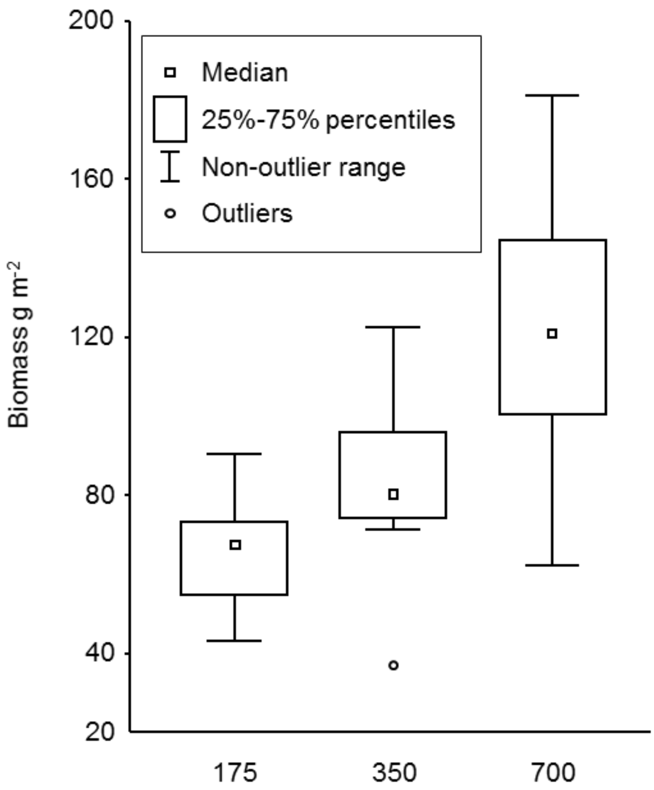

Figure 7. Total biomass under three rainfall amounts (*i.e.,* 175, 350 and 700 mm per year).

Discussion

In most semi-arid ecosystems water availability is the primary limiting factor for productivity, with nitrogen availability acting as a secondary limiting factor in wetter years (Harpole *et al.* 2007). The amount of available soil nitrogen can be affected by the previous year's rainfall (Borken and Matzner 2009). We used rainfall manipulations and addition of nitrogen to test the effect of drought on soil nitrogen dynamics and productivity. To the best of our knowledge, this is the first published study in the semi-arid grasslands that examined the simultaneous effect of nitrogen enrichment with increased and decreased rainfall manipulations (Bobbink *et al.* 2010; Lee *et al.* 2014). Overall, our results support our hypothesis that in drought conditions there is accumulation of available nitrate in the soil. This is consistent with other studies in other ecosystems (Evans and Burke 2013; Stark and Firestone 1995; Yahdjian *et al.* 2006). The accumulation could be due to a number of mechanisms that are not mutually exclusive. 1) Plants take up less nitrate in drier conditions due to plants growing less and therefore demanding less nitrogen (Borken and Matzner 2009). Although plants had higher nitrogen concentrations in drier conditions (Fig. 5), their lower biomass (Fig. 7) resulted in less total nitrogen uptake which can explain accumulation of nitrate in the soil in drier conditions. 2) Lower soil moisture limits diffusion of nitrate and ammonium to the plant roots and microbes (Evans and Burke 2013). As expected, plants allocated more biomass to roots in drier conditions, although nitrogen addition did not affect this relationship. 3) Reduced leaching of nitrate into the deeper soils when there is insufficient rainfall for deep drainage (Cregger *et al.* 2014). Since there was no increase in nitrogen in deeper soils, we do not expect leaching to be an important factor at the simulated rainfall conditions (Fig. 4). This accumulation of available nitrogen in drought conditions causes an asynchrony between the available nitrogen and plant/microbial demand (Augustine and McNaughton 2004), where nitrate levels are higher when soil moisture is lower and plants/microbes cannot utilize the nitrogen.

Our results did not show any effect of rainfall on soil ammonium or total nitrogen in the soil. Soil ammonium did show a non-significant trend of higher values in drier conditions at shallow depths, but we would have expected a stronger effect in the manipulated dry treatment. This is surprising since nitrification is considered to be mainly mediated by soil moisture and therefore ammonium should accumulate in low soil moisture conditions. A possible explanation for us not finding any effect could be that our sample size was too small to detect an effect considering the high heterogeneity in soil ammonium.

The addition of nitrogen did not have a significant effect on plant biomass and this may be explained by our experimental design. Our rainfall addition manipulation aimed at obtaining an above average wet year, assuming that rainfall received during the experiment was around average. However, as we experienced a dry year, the rainfall addition manipulation simulated an average year. Therefore, it is possible that there wasn't sufficient water available to remove the primary limitation of water (Bennett and Adams 2001). Alternatively, in the two weeks following the addition of nitrogen it rained 85 mm, and then it didn't rain for over a month (Fig. 1). This pulse of nitrogen followed by drying of the soil could have caused denitrification of the added nitrogen. Of course, the straightforward explanation that nitrogen does not limit productivity in this system is valid. Our data tends to support that the rainfall pattern denitrified the added nitrogen, since we found no effect of added nitrogen on soil nitrogen.

Our results show that plant biomass, at least up to average rainfall, is mainly limited by water availability in this ecosystem. We hypothesised that with higher levels of rainfall, water availability will not limit productivity and nitrogen availability will be the limiting factor. Therefore, the amount of soil nitrogen in the soil will determine the amount of plant biomass. We predict that in these scenarios, the previous year's amount of rainfall will determine the current year's plant biomass. Understanding this relationship can help us understand the dynamics of both soil nitrogen and productivity in these ecosystems. Additionally, quantification of the effects of rainfall on soil nitrogen and consequently productivity should improve our modelling and predictions of ecosystem productivity. This could lead to better management of stocking rates that increase profitability and reduce land degradation. Furthermore, this could also have implication for the best years of rehabilitation of degraded habitats and control actions of invasive species.

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13 Appendix 3

A range of research projects in the pipeline

A range of research projects in the pipeline

Spyglass Beef Research Facility

SEVERAL research projects are under way on Spyglass Beef Research Facility, including tracking Chital deer in an effort to assist in management plans, establishing monitoring sites in the process to improve long-term carrying capacity information, developing better pasture species, and calf hydration trials to understand reasons for calf mortality.

CHITAL DEER PROJECT

BIOSECURITY Queensland (BQ), with the assistance of the University of Queensland, is leading a pilot study to measure movement patterns and habitat use of Chital deer. To date, minimal research has been conducted on these species in North Queensland. Chital deer are a declared Class Two pest as they have adverse economic impacts on graziers through the eating of substantial amounts of pasture.

In August 2013, five adult Chital deer on Spyglass were fitted with GPS collars (two female, three male). The collars will be on the animals for approximately six months. A key outcome so far is that successful dosage rates and methods for darting of Chital have been established. The information collected on ranging behaviour (particularly habitat use) drinking frequency and ranging area, will help in developing management plans, including control measures, for the species.

A collared Chital stag on Spyglass Research Station is being used to measure movement patterns and habitat use of Chital deer.

Outcomes of this work will determine if funding should be sought to further the study. If so, this could involve requesting collaring deer on properties in the vicinity of Spyglass.

MAPPING AND MODELLING OF LAND TYPES

A NEW project is underway on Spyglass, lead by Queensland Department of Agriculture, Fisheries and Forestry (QDAFF) scientist Giselle Whish, which aims to provide improved property mapping and long-term carrying capacity information to assist in grazing land management decision-making.

Five monitoring sites have been established on the more common land types that occur on Spyglass. The sites cover a range of productivity from the most productive Loamy alluvials, Box and Narrow-leaved Ironbark through to low productivity Box and Narrow-leaved Ironbark.

Over the next two years, tree cover, rainfall, pasture and soil measurements collected from the fenced sites will be used to calibrate the GRASP pasture and animal growth model. With improved estimates of pasture productivity for each paddock on Spyglass, stocking rates and long term carrying capacities can be determined, a range of grazing management options can be explored and the financial implications of these options can be determined.

QDAFF scientist, Lester Pahl, ensuring no animals will graze this monitoring site established on Spyglass.

QDAFF scientists, Chris Holloway and Lester Pahl, establishing a monitoring site on Spyglass where over the next two years scientists will measure rainfall, collect soil samples, and cut pasture from different quadrats within the wooden stake layout.

DEVELOPMENT OF NEW PASTURES

THE use of sown tropical grasses and legumes can significantly increase the productivity of beef growing and breeding enterprises in north Queensland. However, many areas have few, or no, well-adapted grasses or legumes and recently developed cultivars have not been comparatively assessed across a range of land types.

Over the next five years, QDAFF FutureBeef pasture research and beef extension staff, partnering with MLA, seek to compare the persistence and productivity of a range of new pasture plants with older cultivars.

Comparison is to be conducted on a range of soil types and rainfall environments in northern and central Queensland, targeting improved nutrition of younger livestock. 'Spyglass' is being used to represent red earths in the Burdekin catchment. Key genera include: (legumes) Centrosema, Chamaecrista, Clitoria, Desmanthus, Leucaena, Macroptilium, Stylosanthes; (grasses) Bothriochloa, Brachiaria, Chloris, Dichanthium, Digitaria, Heteropogon, Panicum, Urochloa.

The project is in its infancy. To date, the project team, generously supported by beef producers, have identified nine properties in the Northern and Southern Gulf and Burdekin grazing districts, defined sites considered representative of the land type, installed stock/kangaroo fences at five (including Spyglass) and made arrangements for the completion of the others. Seed for sowing has been sourced and testing begun to determine sowing rates.

The project is a collaborative effort between QDAFF research scientists Kendrick Cox, Mark Keating and Steven Dayes and FutureBeef extension staff Joe Rolfe, Bernie English and Emma Hegarty.

CALF DEHYDRATION

THE Cash Cow project, and others, has shown calf loss between confirmed pregnancy and weaning to be as high as 40 percent, and consistently between 10pc and 20pc in some areas of north Australia. Obvious effects on profitability occur. Previous research shows the greatest loss occurring within a week of birth.

The Cash Cow project showed most losses are associated with nutritional and environmental factors, with occasional significant loss due to diseases like Pestivirus. It is plausible that a high proportion of elevated losses in north Australia is associated with poor calf hydration and/or vigour, and that this may be as much a problem with the cow as the calf.

Two experiments have been undertaken at Spyglass to investigate calf hydration.

These studies will provide the basis for conducting further research that will hopefully lead to practical solutions in the future.

In a preliminary study led by Geoffry Fordyce of QAAFI, newborn Brahman calves were dehydrated over three days. This study provided a way to objectively measure the degree of dehydration.

These techniques require further development for use in systematic research under extensive grazing conditions. The experiment showed that newborn calves experiencing milk depravation lose an average of 7pc of live weight daily under comfortable conditions (20 degrees Celsius), but twice this when maximum temperatures are in the vicinity of 40 degrees Celsius.

It also showed that when calves lose 15pc of their live weight, which is equivalent to 20pc dehydration, in as little as one to three days under tropical conditions, some calves are unable to recover without intervention.

A recent study by QDAFF scientist, Jarud Muller, focused more on milk supply in the first week of life.

The aim of the study was to measure normal variation in milk production and delivery in newly-calved Brahman cows, and whether this is possible to measure using a range of indirect measures.

This is complex research as direct measurement of milk supply is not possible at present under range conditions. Indirect measures included a range of calf measures including weight, udder and teat measures, and behavioural observations (such as evidence of sucking). Analyses of the data have not been completed so no results are currently available.

EFFECTS OF NITROGEN DYNAMICS ON PASTURE AVAILABILITY AND QUALITY

NITROGEN and water are the key limiting factors for pasture and animal production in northern Australia. Nitrogen dynamics are strongly affected by rainfall.

Low nitrogen availability (and low animal production) often follows big wet seasons which cause nitrogen dilution. The opposite can occur following years of drought, which leads to high nitrogen availability (and high animal production) in response to a build up in available soil nitrogen.

In order to test the effects of the amount of rainfall in one year on the nitrogen availability in the second year, the amount of rainfall was manipulated in small plots at Spyglass.

A rainout shelter is being used to simulate the effect of drought by reducing the amount of rain falling to the ground by 25 percent.

'Drought' was achieved by building rainout shelters that reduce the amount of rain falling on the ground and 'wet' season was achieved using special irrigation. Soil and plants are being collected.

The results of this study will help incorporate the nitrogen dynamics and its effect on pasture availability and quality into existing animal production models. This will enable to increase animal production while maintaining land conditions. This project is being led by Moran Segoli of CSIRO: 0498 538 788, moran.segoli@csiro.au.

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14 Appendix 4

From rags to riches: the story of carbon, nutrients and pasture with dairy compost application

From rags to riches: the story of carbon, nutrients and pasture with dairy compost application

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Around the world, dairy farmers are transforming dairy waste to compost for land application. In southeastern Australia, farmers are using composted dairy waste to increase production and reduce costs. In addition, the farmers are considering the benefits of compost for increasing sequestration of soil carbon, and on-farm nutrient retention. The "Carbon Farming Initative" in Australia is exploring the option to allow farmers to trade Carbon Credits for carbon stored in the soil. Compost also retains vital nutrients, such as N, on farm rather than importing N in the form of mineral fertilisers. Composting also reduces greenhouse gas emissions, such as CH4, compared to when stored in effluent ponds. This project will investigate if dairy compost applied to pasture improves carbon sequestration, nutrient retention and pasture production. In this project dairy compost, made from dairy effluent, feedpad waste, spoilt sillage and wood mulch, was applied onto a 1Ha field and companion plots at a rate of 0, 3, 6 and 12 t/ha. The field plot is open to grazing and normal farm management practices. The companion plots are being subjected to simulated grazing (mowing). The trials, currently underway will run for 18 months. Along with preliminary soil carbon results, this work will also include preliminary data for total and plant available nutrients, and farm biomass production. The outcomes of this research, and benefits it finds for "Carbon Farming" and nutrient retention has practical, policy and economic applications for world wide markets.

Managing cattle for sustainable soil properties: interactions between stocking rates and rainfall

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Introduction

Understanding the drivers of the dynamics of plant available nutrients in soil is important for the productivity and sustainability of rangelands grazed for livestock production, as well as identifying potential for increasing carbon in soils. In different rangeland ecosystems, grazing can increase, decrease or not affect the plant available nutrients in soil (Pineiro, Paruelo *et al.* 2010). Although many studies have examined the effects of grazing on plant available nutrients worldwide, there is a lack of manipulated field trials addressing the effect of grazing in the dry tropics (but see (Allen, Pringle *et al.* 2013;

Holt 1997; Pringle, Allen *et al.* 2011). One of the problems with studying the effect of grazing on plant available nutrients in soil is the slow response of the nutrients in soil to manipulations (Holt 1997). Therefore we used a 17 year grazing manipulation to examine the effect of grazing intensity on soil organic matter and soil mineral nitrogen.

Methods

We used soil samples that were collected at a long-term cattle-grazing trial conducted at Wambiana station (20°34' S, 146°07' E), north Queensland, Australia. Mean annual rainfall of the area is 636 mm, with most of the rainfall (70%) falling between December and March (O'Reagain, Bushell *et al.* 2009). In 1997 ~100ha paddocks were established with different grazing pressure: 1) Moderate stocking rate (MSR) at ~8 ha per animal equivalent (450 kg steer) and 2) Heavy stocking rate (HSR) at ~4 ha per animal equivalent (O'Reagain, Bushell *et al.* 2009).

There was considerable soil and vegetation heterogeneity within treatment paddocks, so to minimise this source of variability all soil samples were collected in a *Eucalyptus brownii* community on black sodosols and Yellow Kandosol in the MSR and HSR treatments sharing a common boundary (Pringle, Allen *et al.* 2011). The soil samples were collected in July 2008, March-April 2009 and July 2013. 25 cores of 0-10 cm depth were sampled from 1-ha square area. Soil samples were dried at 40˚C, large roots removed, soil crushed and passed through a 2 mm sieve. The dry soil was archived in sealed containers for further analysis.

Available N (ammonium and nitrate-N) was extracted from the soil by shaking 8 grams of soil in 20 ml of potassium chloride (2 M KCl), shaken for 1 hr and filtered through No. 41 filter paper. Nitrate and Ammonium were determined in the KCl extracts by colorimetric methods (Best 1976; Keeney and Nelson 1982; Willis, Schwab *et al.* 1993). Organic matter (OM) content was determined by loss-on-ignition method (Segoli, Ungar *et al.* 2012).

The effects of grazing intensity were subjected to a one-way ANOVA. When data violated ANOVA's homogeneity of variances assumptions, reciprocal transformation was applied to the data (Zar 1999). When the distributions of the residuals were non-normal, a Mann-Whitney U-test was applied (Mann and Whitney 1947; Zar 1999). All statistical analyses were conducted with the STATISTICA 12.0 software (StatSoft Inc., Tulsa, OK, USA).

Results and discussion

Soil organic matter was higher in the heavy grazing treatment in all years (Fig. 1), although this was only significant in 2009 (2008: one-way ANOVA, $F(1,45) = 0.886$, $P = 0.07$; 2009: one-way ANOVA, $F(1,44) = 6.874$, $P = 0.01$; 2013: one-way ANOVA, $F(1,43) = 2.630$, $P = 0.11$). The apparent positive effect of grazing on soil organic matter could be due to a preference of cattle to consume grasses and switch to shrubs and trees when grasses are limited. Therefore, in the heavy grazing treatment there will be a dominance of C3 woody trees and shrubs relative to C4 tropical grasses. This will cause higher sequestration of carbon into the soil (Pringle, Allen *et al.* 2011).

Nitrate showed variation among the years (Fig. 2) and was significantly higher in the moderate grazing treatment of 2009 (2008: one-way ANOVA, $F(1,41) = 0.603$, $P = 0.44$; 2009: Mann-Whitney U-test, adjusted $Z = -2.594$, $P < 0.01$; 2013: Mann-Whitney U-test, adjusted $Z = 0.054$, $P = 0.96$). The variability among years in soil nitrate emphasises the relatively high intra- and inter- annual variability compared to differences in grazing management. For example, lower nitrate in 2009 compared to the other two years may reflect sampling season, since 2009 sampling was performed during the growing season when nitrate uptake by plants would be high as opposed to the other years where the sampling was done in the dry season.

Ammonium also showed variation among the years (Fig. 3), but without any significant difference between the grazing treatments (2008: reciprocal-transformed data, one-way ANOVA, $F(1,41) = 0.002$, $P = 0.96$; 2009: reciprocal-transformed data, one-way ANOVA, $F(1,48) = 1.500$, $P = 0.23$; 2013: reciprocal-transformed data, one-way ANOVA, $F(1,31) = 1.494$, $P = 0.23$). The variability among years in soil ammonium also emphasises the relatively high intra- and inter- annual variability compared to differences in grazing management.

Our results suggest that heavy grazing does not have negative effects on the availability of soil nutrient in this system, and may even sequester higher amount of carbon in the soil. However, this should not be taken as a managerial recommendation since ecosystem function, productivity and sustainability are further controlled by additional factors (e.g. plant cover, leakage, plant community, erosion) all of which may be affected by heavy grazing. Furthermore, evidence exists that results are likely to vary according to soil type (Pringle, Allen *et al.* 2011), topographic unit (O'Reagain, Brodie *et al.* 2005), vegetation type (Richards, Brackin *et al.* 2012) and rainfall dynamics (O'Reagain, Bushell *et al.* 2009).

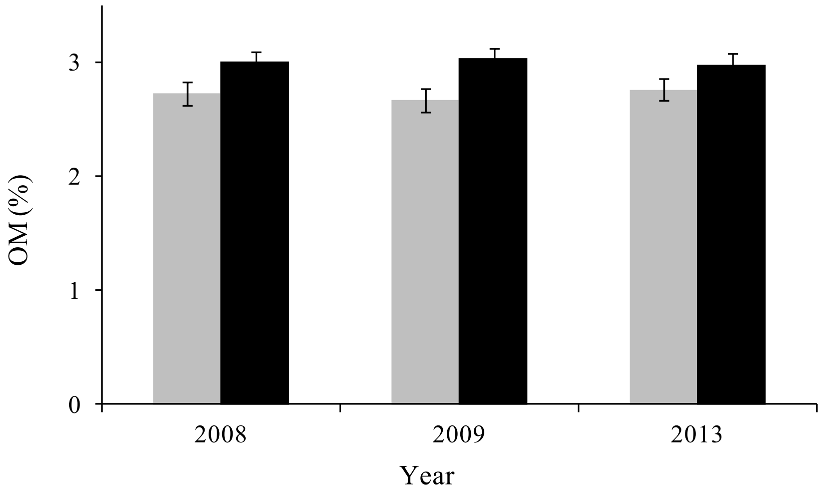

Figure 1. Soil organic matter under moderate (grey) and heavy (black) cattle stocking rates. Data are represented as means ± SE.

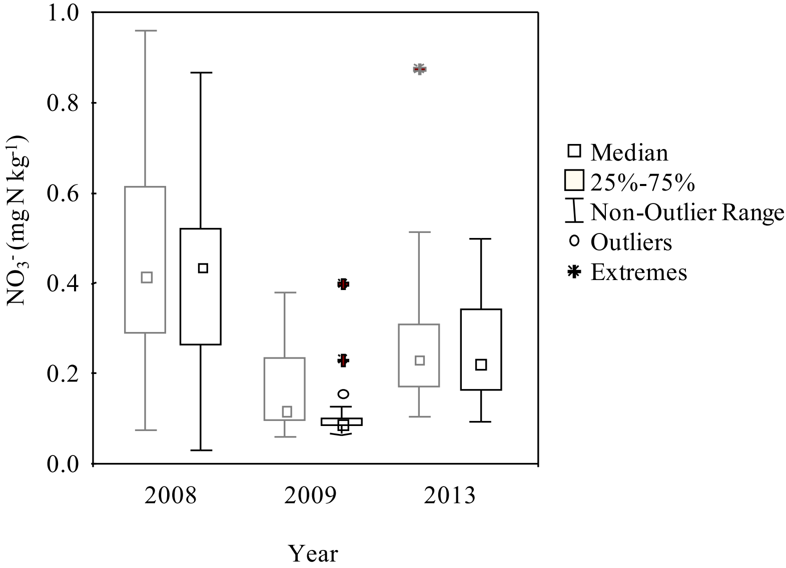

Figure 2. Box plot of soil nitrate under moderate (grey) and heavy (black) cattle stocking rates.

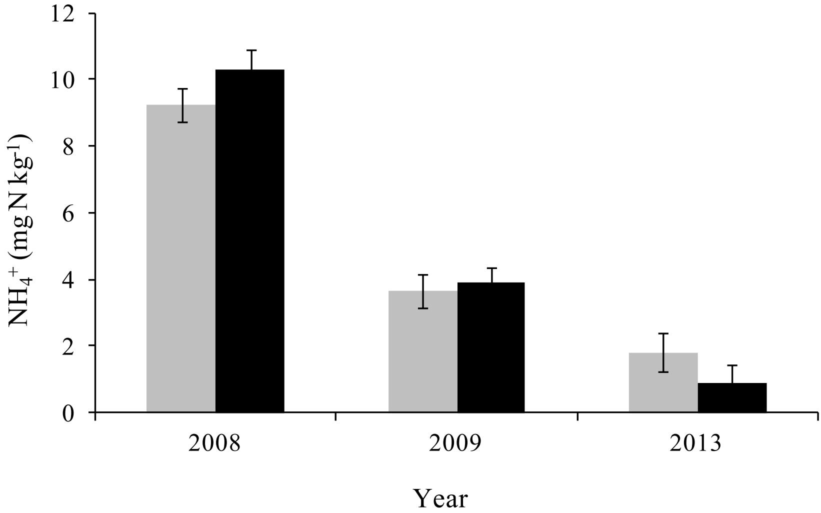

Figure 3. Soil ammonium under moderate (grey) and heavy (black) cattle stocking rates. Data are represented as means ± SE.

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