Grazing reduces the capacity of Landscape Function Analysis to predict regional-scale nutrient availability or decomposition, but not total nutrient pools

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Running Header: Soil nutrient predictions affected by increasing grazing pressure

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Abstract

The Nutrient Cycling Index (hereafter 'Nutrient Index') derived from Landscape Function Analysis (LFA) is used extensively by land managers worldwide to obtain rapid and cost-effective information on soil condition and nutrient status in terrestrial ecosystems. Despite its utility, relatively little is known about its reliability under different management conditions (e.g. grazing) or across different climatic zones (aridity). Here we correlated the Nutrient Index, comprising measures of biocrust cover, plant basal cover, soil roughness and three attributes of surface litter cover, with empirical data on measures of soil total nutrient pools (C and N), nutrient availability (labile C, inorganic N and P), and decomposition-related enzymes at 151 locations from eastern Australia varying in grazing intensity and climatic conditions. Grazing intensity was assessed by measuring current grazing (dung production by the herbivores cattle, sheep/goats, kangaroos and rabbits), and historic grazing (the total area of livestock tracks leading from water). We used aridity (the relationship between precipitation and potential evapotranspiration) as a measure of climate. On average, the Nutrient Index was positively associated with total nutrient pools, nutrient availability and decomposition enzymes. However, further statistical modelling indicated that grazing intensity strongly reduced the link between the index and decomposition enzymes, labile C and inorganic P, but not with total nutrient pools. This grazing effect was predominantly due to cattle. Conversely, aridity had no significant effect on the predictive power of the index, suggesting that it could be used across different aridity conditions in natural ecosystems as a reliable predictor of soil health. Overall, our study reveals that the Nutrient Index is a robust predictor of total nutrient pools across different aridity and grazing conditions, but not for predicting nutrient availability or decomposition in environments heavily grazed by livestock.

Keywords: Aridity; Enzyme activities; Carbon; Nitrogen; Phosphorus; Drylands

Introduction

Rapid methods of assessing soil nutrient status have gained increasing popularity over the past few decades, particularly in arid and semi-arid environments (drylands) where monitoring extensive areas is prohibitively expensive, and where sophisticated laboratories are not always available. The use of indices or surrogates for assessing soil quality is widespread, particularly under cultivated agriculture (e.g. Granatstein and Bezdicek, 1992;

Sojka and Upchurch, 1999; Li et al., 2013; Izquierdo et al., 2005; Zornoza et al., 2015; Raiesi and Kabiri, 2016) but also in drylands (Li et al., 2013; Raiesi, 2017). The attributes used to assess quality vary substantially, from soil physical, biological, chemical and biochemical, to microbiological assays, and the advantages of different indices vary with land management type, soil type, environmental setting and available resources. Consequently, there is no universally accepted measure for assessing soil quality (Karlen, et al., 1997).

The use of simple soil indices has many advantages over traditional physical and chemical methods. First, they are relatively rapid, and more sites can be assessed without the need for expensive and detailed laboratory analyses such as soil enzymes activities (Bell et al., 2013) or mineralization rates (C or N; e.g. Picone et al., 2002). Second, data collection, and assessment and interpretation of indices or surrogates require only low levels of expertise. Third, indices are typically focussed on specific management objectives that may be closely aligned to soil policy (e.g. Griffiths et al., 2016). Notwithstanding their limitations (Blecker et al., 2012; Sojka and Upchurch, 1999), the use of indices or proxies of soil health provide valuable insights into the processes driving soil function by focussing on tangible soil and ecological attributes that are appropriate and relatively well understood by operators with only minimal training.

Landscape Function Analysis (LFA: Ludwig and Tongway, 1995) is a widely accepted technique for assessing soil nutrient status in terrestrial environments. It incorporates a quadrat-based module (Soil Surface Condition) that assesses the capacity of the soil to resist erosion, cycle nutrients and infiltrate water (Tongway, 1995). One of these indices, the LFA Nutrient Cycling Index (hereafter 'Nutrient Index'), provides information on the nutrient status (e.g. nutrient availability and mineralization) of soils (Maestre and Puche, 2009; Tongway, 1995). It is based on the close relationship among 12 readily identifiable soil surface features and underlying processes of nutrient mineralisation. These relationships have been quantified using extensive field and laboratory studies (McR Holm et al., 2002; Rezaei et al., 2006; Maestre and Puche, 2009; Zucca et al., 2013). The practicality of the Nutrient Index is based on the assumption that functional, healthy landscapes regulate critical resources such as sediment, water and organic material, which are all important components of the Nutrient Index (Sarre, 1988). Worldwide studies indicate that the values obtained from this index are highly related to laboratory and field measurements of their related processes (Maestre and Puche, 2009; McIntyre and Tongway, 2005; McR Holm et al., 2002; Rezaei et al., 2006; Tongway, 1995; Zucca, et al., 2013). Consequently, the soil Nutrient Index has been used widely, across diverse landscapes, community types, climatic zones, management scenarios and land use intensities (e.g. Eldridge et al., 2011; Eldridge et al., 2016a), and often in developing countries (Rezaei et al., 2006; Zucca et al., 2013). Given its largely global adoption, particularly in semi-arid rangelands, it is assumed that the Nutrient Index is globally relevant under a range of ecosystem conditions. Lacking, however, is an assessment of the effectiveness of the index under different land use intensity scenarios and climatic drivers, the strongest of which are grazing and increasing aridity.

Grazing is a major global change driver, and overgrazing has been described as one of the most destructive landuses on the planet because of its negative effect on ecosystem processes and functions (Steinfeld et al., 2006; Eldridge et al., 2015). However, grazing provides millions of peoples and their cultures worldwide with essential goods and services. Aridity is also a significant driver and reflects potential changes that might occur under hotter and drier global climates (Maestre et al., 2015, 2016). Increasing aridity will reduce the efficiency with which plants carry out essential soil processes such as the mineralisation of organic matter (Maestre et al., 2016), and has been demonstrated to decouple nutrient cycling in global drylands (Delgado-Baquerizo et al., 2013). Both grazing aridity are expected to increase in response to a changing climate. With increases in aridity, human cultures that rely on livestock grazing for their livelihoods will be forced to exploit less suitable environments or increase their stocking rate to maintain productivity in the face of declining rainfall (Steinfeld et al., 2006; Prăvălie, 2016).

Here we evaluate the robustness of the LFA Nutrient Index in response to increasing grazing and aridity. Our intention is not to evaluate the strength of correlations between individual soil attributes and the index *per* se (Rezaei et al., 2006; Maestre and Puche, 2009) or to debate the merits or otherwise of the many indices currently used in agriculture (Zornoza et al., 2015), but rather, to examine the utility of this index in response to the two major environmental drivers. Put simply, we assess the usefulness of the index under a drier climate and a more intensively managed world. The specific components of the Nutrient Index: surface roughness, biocrust cover, plant basal cover, plant litter cover, plant litter origin, and plant litter incorporation, are expected to vary naturally across aridity gradients, providing an indication of naturally co-occurring changes in soil nutrient availability. For example, a mesic environment with a greater incorporation of litter would be expected to have a higher amount of organic C than an arid ecosystem with a much lower amount of litter incorporation. Because of this, we hypothesized that increases in aridity, naturally accompanying changes in LFA components, should have little effect on the predictability of the LFA Nutrient Index. Conversely, however, we hypothesized that grazing would strongly influence the correlation between the index and multiple empirical measures of soil function, particularly those related to nutrient availability (i.e. inorganic N and P, and labile C) and measures of organic matter decomposition (i.e. extracellular enzyme activity). However, grazing may not influence the correlation with total nutrient pools. Our reasoning is that grazing would likely influence the availability of nutrients and enzymes *via* direct additions of nutrients as urine and dung, but may not alter components of the Nutrient Index such as litter incorporation, thereby disrupting the natural capacity of the index to predict nutrient availability. Conversely, grazing would be expected to alter plant components such as vascular and non-vascular plant cover and total nutrient pools in parallel, thus maintaining the links between the index and total nutrient pools. For example, a high grazing intensity would be expected to reduce plant basal cover and soil C (Eldridge and Delgado-Baquerizo, 2016), hence plant cover would still be a good predictor of total C under high grazing scenarios.

Clarifying the extent to which grazing by different herbivores might reduce the utility of soil chemical surrogates in drylands is critically important because governments and their resource management agencies need rapid, reliable and cost-effective measures to assess changes in soil function as the planet gets warmer and drier into the next century. This is particularly important in drylands because: (1) drylands mostly occur in developing countries, which have a more limited capacity to assess nutrient availability over extensive area; (2) the effects of increases in aridity are likely to be most strongly felt, and (3) about 40% of Earth's human population currently reside in drylands (Prăvălie, 2016). The work is also important because increasing intensities of different herbivores would be expected to have different effects on surrogates of soil chemical status. For example, cattle and sheep have been shown to have strong negative effects on soil health, but kangaroos (*Macropus* spp.), which have co-evolved with soils and vegetation in Australia, have relatively benign effects (Eldridge et al., 2016b). The ability to predict soil nutrient pools, therefore, might be stronger in environments supporting low levels of livestock grazing or where kangaroos are the principal herbivores. Knowing how these different herbivores might affect the relationships between nutrient indices and different soil nutrients and enzymes is important because it provides land managers with vital information that will improve their ability to make decisions on how their management alters soil function using relatively rapid, cost effective methodologies that are readily accessible to non-professionals.

Methods

Study area

The study was undertaken in a woodland community in south-eastern Australia dominated by white cypress pine (*Callitris glaucophylla* Joy Thomps. & L.A.S. Johnson; Fig. 1). The climate is typically Mediterranean and semiarid (Aridity Index = 0.26 to 0.39; see below), with slightly greater rainfall in the east-central areas during the six warmer months, and in the south and south-west during the six cooler months. Average annual rainfall (385 to 460 mm yr^{-1}) and average temperatures (~18° C) varied little across the sites.

Assessment of groundstorey cover and grazing intensity

We surveyed 151 woodland sites characterised by the presence of the community dominant *Callitrus glaucophylla*. At each site we positioned a 200 m long transect within which were placed five 25 m^2 (5 m x 5 m) plots (hereafter 'large quadrat') every 50 m (i.e. 0 m, 50 m, 100 m, 150 m and 200 m). A smaller (0.5 m x 0.5 m) quadrat (hereafter: 'small quadrat') was located at a consistent position within each of the larger quadrats. Within both the large and small quadrats we assessed groundstorey plant cover (defined as the foliage cover of all plants < 1 m tall).

Our sites represented different levels of current and historic grazing by different herbivores. We did this initially by using distance from permanent water, which is a useful surrogate of grazing intensity (Fensham and Fairfax, 2008). The sites spanned the full spectrum of grazing intensities, from low intensity and long ungrazed sites from conservation reserves and road verges, to intermittent grazing in forests and conservation reserves, to high levels of grazing in a range of environments. Four attributes reflected current grazing intensity, i.e. grazing within the past 2 to 5 years, and the fifth was a measure of historic grazing over the past 50-100 years. To assess current grazing, we counted dung produced by four herbivore groups: cattle (large quadrat), sheep/goats (large and small quadrats), kangaroos (large and small quadrats) and rabbits (small quadrat only). For sheep/goats and kangaroos, dung counts for the two quadrats sizes were averaged to produce a total dung/pellet density m^{-2}. For cattle, we counted dung events rather than individual pieces of dung, which are known to disintegrate. Dung and pellet counts have been used widely to estimate the abundance of large herbivores, including kangaroos (Marques et al., 2001). We then used previously developed algorithms (see Eldridge et al., 2016b) to calculate the total oven-dried mass of dung per hectare for each herbivore type based on the number of pellets recorded in the field. This total oven-dried mass of dung was used as our measure of recent grazing intensity for each herbivore (Eldridge et al., 2016b). To assess historic livestock grazing we recorded the total cross-sectional area of tracks along which livestock walk when moving to and from water (livestock tracks) along the 200 m transect at each site (cm^2/200 m).

Laboratory-based soil analyses

We collected about 500 g of soil, from the surface 5 cm, from the centre of each small quadrat, resulting in a total of 755 soil samples (151 sites each of five quadrats). The soils were air dried, passed through a 2 mm sieve to remove roots, organic debris and stones prior to chemical analyes. Sand, silt and clay contents were measured using the hydrometer method (Bouyoucos, 1962). Total C and N were assessed using high intensity combustion (LECO CNS-2000; LECO Corporation, St. Joseph, MI, USA), available (Olsen) P according to Colwell (1963). Labile carbon was assessed by measuring the change in absorbance when slightly alkaline $KMnO_4$ reacts with the most readily oxidizable (active) forms of soil C to convert Mn (VII) to Mn (II; Weil et al., 2002). Ammonium and nitrate concentrations were measured using Flow Injection Analysis (Quick-Chem8500-LACHAT) following extraction with 0.5M K_2SO_4. Four enzyme activities were measured following Bell et al., (2013). These enzymes include: β-glucosidase (starch degradation) (BG), cellobiosidase (cellulose degradation), N-acetyl-β-glucosaminidase (chitin degradation) (NAG) and phosphatase (P mineralization) (PHOS) activity (Bell et al., 2013). In brief, a mixture of 1 g of air-dried soil and 33 ml of sodium acetate buffer (pH <7.5) was shaken at 200 rpm on an orbital shaker for 30 minutes and 800 μl soil slurry was sampled and 200 μl substrate of 4-Methylumbelliferyl β-D glucopyranoside solution were added to the slurry. A solution of 1000 μl was incubated at 25 °C for 3 hours and the activity (nmol activity g^{-1} dry $soil^{-1}$ h^{-1}) was measured at the 365 nm excitation wavelength and 450 nm of emission wavelength in a microplate reader. The same procedure was used, but with different substrate solutions, for an additional three enzymes.

Assessment of the measure of soil health and soil chemistry

We used rigorous, field-based protocols to calculate the LFA Nutrient Index by assessing the status and morphology of the soil surface within the small quadrats (*sensu* Tongway, 1995). Within these quadrats, we measured 12 attributes: surface roughness, crust resistance, crust brokenness, crust stability, the percent cover of the soil affected by erosion, cover of deposited material, biocrust cover, plant basal cover, projected groundstorey plant cover, litter cover, litter origin, and the degree of litter incorporation (see Supplementary Methods and Table S1). We derived our Nutrient Index for each quadrat based on an assessment of six of the 12 attributes: surface roughness, biocrust cover, basal cover of groundstorey plants, and a combined score for litter derived from the product of its cover (% cover), origin (local or transported from elsewhere) and the degree of litter incorporation. These values were summed and divided by the maximum score of 44 to derive the index, which reflects the capacity of the soil to cycle and retain nutrients. This Nutrient Index, which is one of three indices developed as part of the Landscape Function Analysis protocol (Ludwig and Tongway, 1995), has been shown to be highly correlated with ecosystem functions related to nutrient cycling (Maestre and Puche, 2009; see Supplementary Methods S1 for specific analytical methods).

Statistical procedures

We used a two-stage process to examine the extent to which increases in grazing and aridity altered the strength of relationships among the Nutrient Index and various measures of soil chemistry and enzyme activity. We first calculated the correlations (Spearman's ρ) among the Nutrient Index, and the four enzymes, total and labile C, total N, dissolved inorganic nitrogen (DIN: sum of NH_4^+ and NO_3^-) and available P. Spearman's ρ was used as a measure of our effect-size, as it is robust to deviations from normality and is largely used in the ecological literature (see Nakagawa and Cuthill, 2007 for a review). We then tested the skewness of the Spearman's ρ values.

In a second stage we used the principles of structural equation modelling (SEM) to explore relationships among the nine Spearman's ρ values and grazing intensity, aridity and ground cover. Structural equation modelling tests the plausibility of a causal model, based on *a priori* information, in explaining the relationships among different variables. We formulated an *a priori* model whereupon we predicted that both grazing intensity and aridity would have direct effects on the Spearman's ρ values, but also indirect effects *via* changes in groundstorey plant cover. Structural equation modelling allowed us to partition direct and indirect effects of one variable upon another and to estimate the strengths of these multiple effects. This is particularly important in grazing studies where grazing has both direct effects on soils, for example, by removing surface crusts or compacting the soil surface, and indirect effects, *via* removal of plant material (herbivory) and therefore decomposition processes (Eldridge et al., 2016b).

We combined the effects of recent and historic grazing into a single composite variable ('grazing'). Increases in this composite variable corresponded to increasing total grazing pressure. The use of composite variables collapses the effects of multiple, conceptually-related variables into a single combined effect, aiding the interpretation of model results (Grace, 2006). We included aridity in the models because it has been shown to be a useful tool to account for spatial variability in sites (Delgado-Baquerizo et al., 2013) and potentially provides insights into the effects of rainfall and evapotranspiration on the hydrology. Aridity was calculated as 1- AI, where aridity is precipitation/potential evapotranspiration, obtained from Worldclim interpolations (Hijmans et al., 2005). Aridity Index was subtracted from 1 so that increasing aridity corresponded with increased dryness.

We used goodness of fit probability tests to determine the absolute fit of the best models. This goodness of fit test estimates the probability that our observed data fit the *a priori* model described above. Thus high probability values indicate that these models are highly plausible causal structures underlying the observed correlations. Models with the strongest measures of fit (e.g., low χ^2, high GFI, and high NFI) were interpreted as showing the best fit to our data. All SEM analysis was conducted using AMOS Software Version 22. The stability of these models was evaluated as described in Reisner et al. (2013).

Results

Correlations (Spearman ρ) among the Nutrient Index and nutrient concentrations and enzyme activity were all positive (0.22 ± 0.04; mean ± SE; Fig. 2) and strongly left skewed (Fig. 3; Supplementary Table S2), indicating the generally high positive correlations between the Nutrient Index and total nutrient pools, nutrient availability and the activity of enzymes related to organic matter decomposition.

Our structural equation models indicated that increased grazing intensity reduced the correlation between the Nutrient Index and nutrient availability (inorganic P and labile C) and enzyme activities related to organic matter decomposition (Fig. 4). However, we did not find any effect of grazing intensity on total nutrient pools (i.e. total C and N) or inorganic N (Fig 5). Unlike increases in grazing, however, increases in aridity had no effects on the correlations among the Nutrient Index and any nutrients or enzymes. Our results also indicate that all of the effects were direct, i.e. there were no indirect effects of either aridity or grazing mediated by changes in ground cover. We also found strong positive effects of both increases in grazing intensity and aridity on plant cover.

The standardised total effects (the sum of direct and indirect effects) of aridity or different measures of grazing, on nutrients and enzymes showed that the suppressive effect of grazing on the correlations between the Nutrient Index and the four enzymes was due almost entirely to increases in the intensity of cattle grazing (Table 1). Apart from the suppressive effect of cattle grazing and the stimulatory effect of historic grazing on available P, there were no clear grazing intensity trends for the other nutrient relationships. The total standardised effects of aridity on nutrients and enzymes were extremely small (Table 1).

Discussion

Our study provides solid evidence that the LFA Nutrient Index is a robust predictor of total nutrient pools irrespective of grazing intensity, but not of nutrient availability or decomposition under high levels of grazing. Thus, while the index is an extremely useful and cost-effective proxy of processes driving specific soil functions, increases in grazing intensity will strongly reduce its predictive power; thus its utility in a more intensively managed world. Our results indicate the weakness of using this index without first considering grazing intensity, particularly if sites are heavily grazed. Increases in grazing intensity will make the adoption of this index more problematic for land managers, increasing their reliance on more traditional, costly laboratory methods for assessing nutrient status. Interestingly, our results further suggest that the Nutrient Index is still useful for total nutrient pools, nutrient availability and decomposition across different aridity regimes. Thus, our study suggest that the index is a good predictor for nutrient assessments in drylands under low grazing intensity, an important contextual message that need to be considered by land managers and policy makers using these indices.

On average, the LFA Nutrient Index was a relatively good proxy of both total and available nutrient pools, as indicated by the distribution of left-skewed correlations (Figs. 2 & 3) and consistent with results of previous global studies, particularly from drylands. For example, Maestre and Puche (2009) showed that the nutrient index was strongly correlated with soil variables highly indicative of microbial activity such as pH, total soil N and P, soil respiration, and the activity of phosphatase and β-glucosidase at 29 arid grassland sites in Spain. The index has also been shown to be highly correlated with soil organic C and total N in studies in Australia (McR Holm et al., 2002, Tongway and Hindley, 2003), Iran (Ata Rezaei et al., 2006) and Spain (Maestre and Cortina, 2004). Similarly, Munro et al. (2012) demonstrated that values of the Nutrient Index increased with increasing age of tree plantings and found that the index was most strongly influenced by vegetation cover rather than more subtle soil surface features. Paz-Jimenez et al. (2002) demonstrated strong links between the activity of some extracellular soil enzymes such as phosphomonoesterase and β-glucosidase, and agricultural practices, but did not report any effects of grazing. However, Seaborn (2005) showed that the index was a good predictor of soil health measures (soil respiration, mineralisable N) at one of four mining sites in tropical and sub-tropical Australia.

Despite the generally positive correlations, there was a wide range of positive and negative correlations for all variables, indicating that potential site- or soil-specific conditions might reduce the universality of the index. For example, correlations for available P were highly variable and about half that of other nutrients (Fig. 2), possibly due to differences in the type of parent material type or depth to bedrock, which are difficult to identify using quadrat-based LFA methods. Interestingly, we found that correlations for available P on sites with sandy surface textures (sand hills with substantial European rabbit *Oryctolagus cuniculus* activity) were almost twice those on plains with loamy to clay-loam surface textures (ρ = 0.19 *cf.* 0.10 for sand hill and plains, respectively). Intense rabbit activity on sandy soils leads to considerable soil destabilisation (Eldridge et al., 2016b), potentially exposing P-rich subsoil

(Vandandorj et al., 2017). Rabbits have also been shown to enhance litter cover and thus affect the Nutrient Index by favouring large exotic Mediterranean forbs with substantial litter at the expense of smaller native forbs (Leigh et al., 1987; Vandandorj et al., 2017). Relatively high levels of available P at sites with high index values (resulting from herbivory-induced competitive exclusion) coupled with high levels of available P at low index values (*via* rabbit engineering effects of exposing soil P, but covering surface litter, biocrusts and plants) would result in generally equivocal values of P across the range of the index.

Our SEM models showed that increased grazing intensity reduced the strength of correlations between the Nutrient Index, and nutrient availability (inorganic P and labile C) and decomposition enzymes. The only exception to this was the availability of inorganic N. Conversely, correlations for total C and N remaining unaffected by increasing grazing intensity. Thus land use intensification associated with grazing disrupts the capacity of the index to predict soil functions (fast variables) that occur over short time scales. This indicates to us two things. First, the index is relatively robust to changes in grazing intensity for slow nutrient pools (total C and N), which are more strongly related to long-term changes in nutrient availability and reflect differences in persistent soil characteristics that have developed over long time periods such as soil texture. This is consistent with the observation that total C pools are relatively insensitive to changes in management, such as conservation tillage, compared with more labile forms such as labile C (Weil et al., 2002; Rabbi et al., 2015). Second, the path coefficients between grazing and the four measures of enzyme activity related to C, N and P mineralisation were strongly negative, indicating that increased grazing intensity will decouple the link between the index and the more labile soil enzymes and nutrient forms (Vandandorj et al., 2017). Furthermore, this decoupling was largely due to cattle grazing, consistent with the largely negative effects of cattle on soil surface morphology (Eldridge et al., 2016b). Although grazing has been shown to have negative effects on the Nutrient Index (e.g. Eldridge et al., 2013), in the present study, the index was a good proxy of slow variables such as total C and total N, irrespective of grazing intensity. Heavy grazing would likely reduce organic matter inputs into the soil, reducing substrates for microbial growth (Northup et al., 1999).

Most studies have correlated the Nutrient Index with total nutrient pools such as total C and N, simply because these variables are routinely assessed in many soil studies (e.g. McR. Holm et al., 2002; Tongway and Hindley, 2003; Ata Rezaei et al., 2006). In Spain, the index has been shown to be highly correlated with soil respiration and phosphatase and β-glucosidase activities across two widely different soils (Mayor, 2008; Maestre and Puche, 2009). While the Nutrient Index was successful in predicting total pools (total C and N), this correlation was independent on grazing intensity making it particularly useful for assessing slow nutrient pools that may take millennia to change. Short-term cycling of carbon compounds (labile C), which are known to change across seasons and days (Weil et al., 2002), was susceptible to grazing intensity and thus we recommend caution when using the index to assess it without considering grazing history. However, fast variables such as microbial biomass, labile forms of carbon and nitrogen and biochemical attributes such as soil enzymes are more responsive to management practices and changes in land use practices than slow variables such as total C (Weil et al., 2002; Gil-Sotres et al., 2005; Bastida et al., 2006) and are therefore most likely to be affected by grazing. Subtle changes in land use intensity that increase litter cover and incorporation such as conservative (low risk) stocking are likely to be reflected in changes in fast variables such as enzyme activity rather than slow variables such as total concentrations of N and C, which operate at longer time scales.

The ability to predict labile or total nutrient pools or enzyme activity with the Nutrient Index was unrelated to changes in aridity, possibly due to the small extent of our aridity gradient, but also because of changes in the components of the index are expected to co-occur with changes in total and available nutrient pools, i.e., not influencing the capacity of the LFA index to predict nutrient availability. Our results suggest that the index is a robust predictor for multiple indices of nutrient availability across different aridity regimes. This information supports its use in natural drylands. However, it is expected that increasing aridity associated with climate change will likely reduce the area of land suitable for grazing (Steinfeld et al., 2006), placing increasing pressure on land managers, likely forcing them to increase stocking rates in order to maintain production under a drier climate (McKeon et al., 2009). In the long term this will likely reduce the effectiveness of the Nutrient Index for monitoring changes in ecosystem functions associated with nutrients and enzyme activities.

Conclusions

Soil health indices such as the LFA Nutrient Index can provide land managers with critical knowledge that allows them to assess and monitor trends in soil function as we move towards a drier climatically uncertain future. Compared with other soil quality systems such as the

Soil Quality Index, the LFA Nutrient Index is relatively simple and intuitive, requiring few attributes that can be assessed by relatively unskilled technicians after minimal training. Our results provide a context for using the index across different aridity and grazing intensity conditions. Thus, our results suggest that the nutrient index is a robust index for predicting total nutrient pools across different aridity and grazing conditions but not for nutrient availability or decomposition under elevated grazing conditions. Therefore, we recommend the use of this index in natural ecosystem with low grazing intensity, and advice that should be taken in consideration by land use managers and policy makes using this index.

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Table 1**.** Summary of standardised total effects (the sum of direct and indirect effects) of aridity and the five measures of 'Grazing' on the correlations among the LFA nutrient index and soil enzymes and nutrients.

Enzymes and nutrients	Aridity	Grazing				
		Cattle	Sheep	Rabbit	Kangaroo	Tracks
Phosphatase	0.09	-0.28	0.05	-0.06	-0.04	0.09
Available P	0.02	-0.11	0.03	0.01	0.04	0.14
NAG	0.08	-0.30	0.10	-0.11	-0.03	0.10
Dissolved inorganic N	-0.06	-0.06	0.07	0.07	0.07	0.05
Total N	0.08	-0.06	0.01	-0.02	-0.04	0.02
Cellobiosidase	0.10	-0.32	0.07	-0.07	-0.02	0.07
β-glucosidase	0.09	-0.35	0.07	-0.09	0	0.10
Labile C	-0.02	-0.06	0.11	-0.07	0.01	0.03
Total C	0.04	-0.05	0.05	-0.02	-0.04	0.04

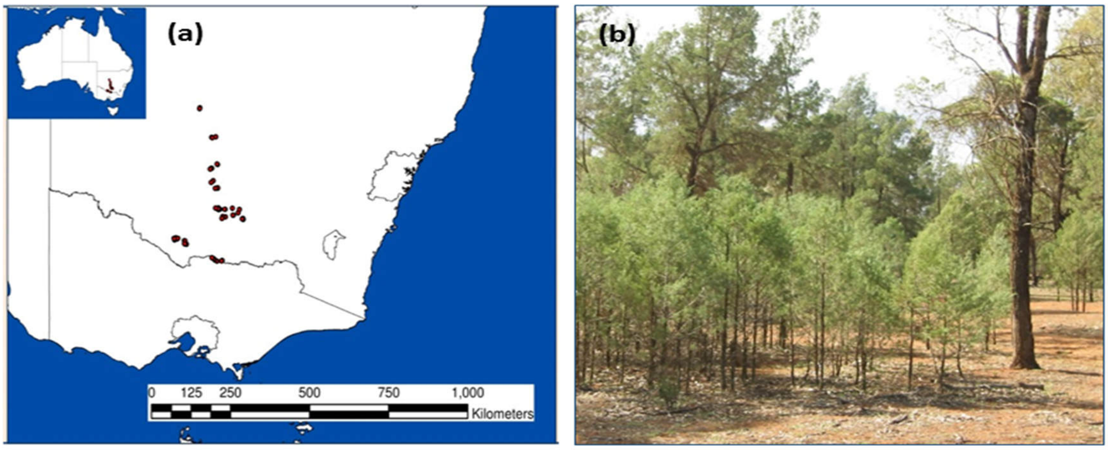

Figure 1. (a) Location of the study area in eastern Australia and (b) a view of the *Callitrus glaucophylla* community.

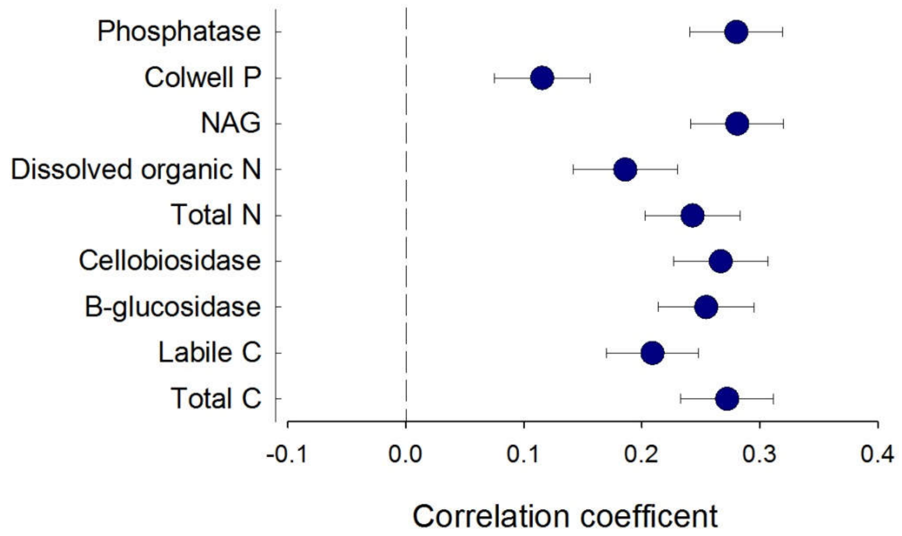

Figure 2. Mean (± 95% CI) correlation between the LFA nutrient index and soil nutrient concentrations and enzyme activities.

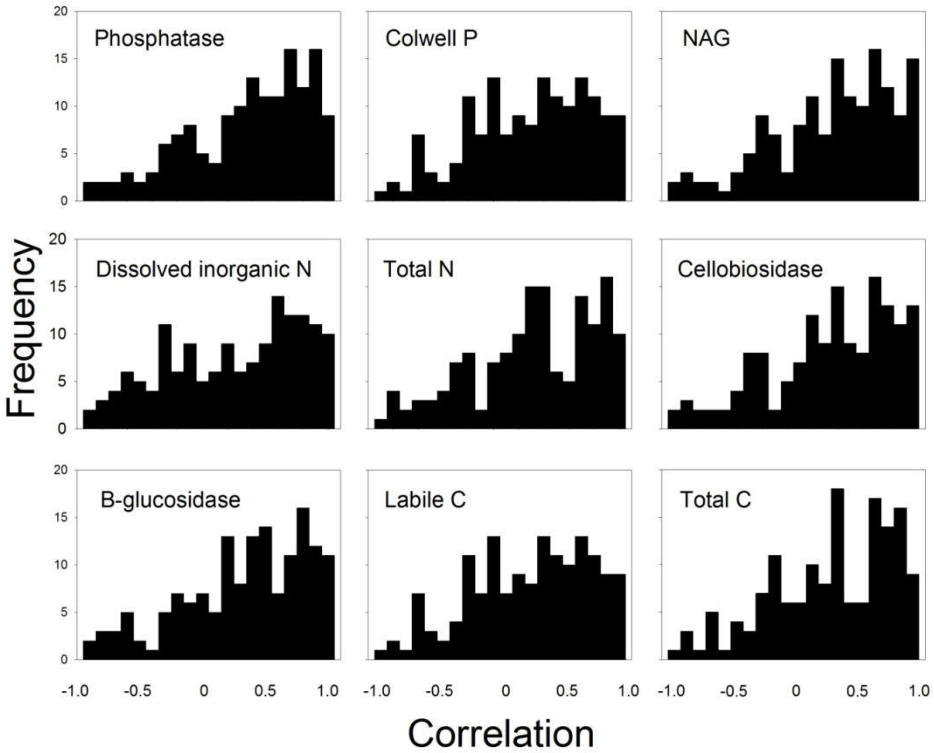

Figure 3. Frequency distribution of correlations between the LFA nutrient index and soil nutrient concentrations and enzyme activities.

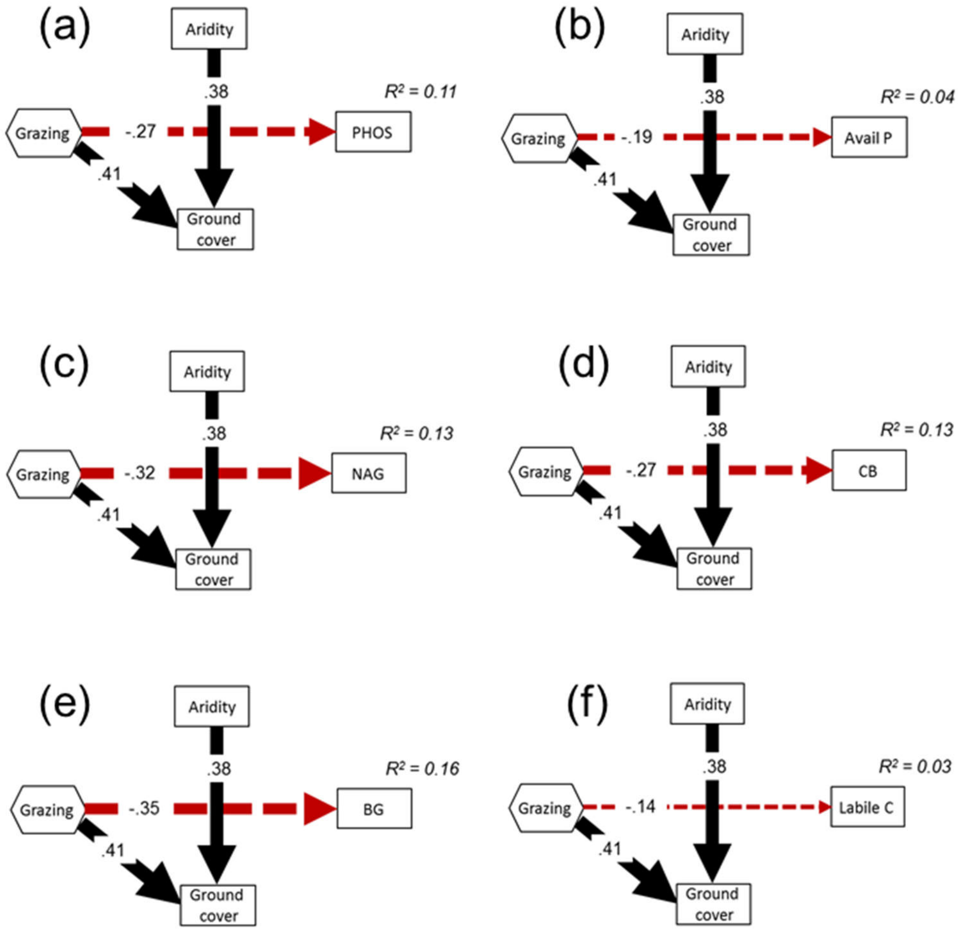

Figure 4. Structural equation models for measures of phosphorus (a-b), nitrogen (c-e) and carbon (f) functions in relation to the composite variable 'Grazing', and aridity and groundstorey plant cover. Grazing is a composite variable comprising recent grazing by all herbivores, and historic grazing by livestock. Standardized path coefficients, embedded within the arrows, are analogous to partial correlation coefficients, and indicate the effect size of the relationship. Continuous and dashed arrows indicate positive and negative relationships, respectively. The width of arrows is proportional to the strength of path coefficients. The proportion of variance explained (R^2) appears is shown in each figure. Only significant pathways are shown in the models. Model fit: χ^2 = 2.40, df = 5, P = 0.79. NFI = 0.97.

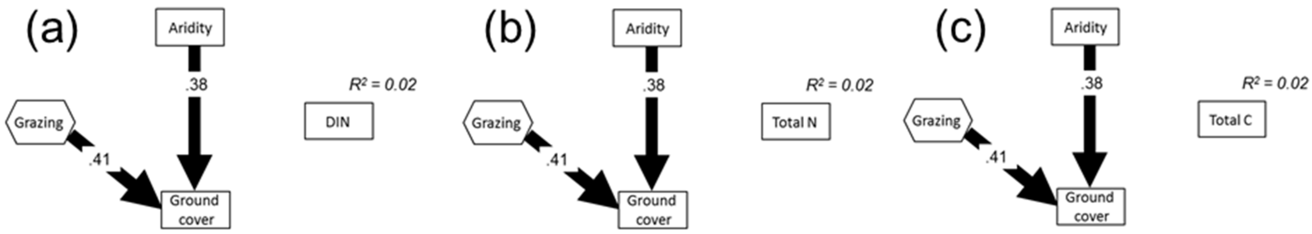

Figure 5. Structural equation models for measures of nitrogen (a-b) and carbon (c) functions in relation to the composite variable 'Grazing', and aridity and groundstorey plant cover. Grazing is a composite variable comprising recent grazing by all herbivores, and historic grazing by livestock. Standardized path coefficients, embedded within the arrows, are analogous to partial correlation coefficients, and indicate the effect size of the relationship. Continuous and dashed arrows indicate positive and negative relationships, respectively. The width of arrows is proportional to the strength of path coefficients. The proportion of variance explained (R^2) appears is shown in each figure. Only significant pathways are shown in the models. Model fit: χ^2 = 2.40, df = 5, P = 0.79. NFI = 0.97.