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Do regenerative grazing management practices improve vegetation and soil health in grazed rangelands? Preliminary insights from a space-for-time study in the Great Barrier Reef catchments, Australia

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ABSTRACT

Regenerative grazing, which generally involves some form of rotational grazing with strategic rest, is increasingly seen as a profitable management approach that will accelerate landscape recovery. However, there is limited quantitative evidence supporting the benefits of this approach in northern Australia. This space-for-time study collected vegetation and soil data from a range of properties in the Burdekin catchment in Queensland that have implemented regenerative grazing strategies for between 5 and 20 years. Data were also collected at adjacent control sites that did not undergo regenerative grazing, but where more traditional continuous set-stocking grazing approaches were applied. Coincident data were also collected from several sites where grazing had been excluded for ~30 years. Data suggested that improvements in vegetation, soil and land condition can be obtained from implementing regenerative grazing principles, although it is likely to take at least 3–5 years, and up to 15–20 years for statistically significant improvements to be measurable at a site, particularly for areas that are moving from a degraded baseline condition. Vegetation attributes such as plant biomass and basal area and litter incorporation all appeared to be better surrogates than percentage ground cover for representing improved landscape condition and soil health. Sites that maintained remotely sensed percentage ground cover at or above the minimally disturbed reference benchmark levels for >10 years, as well as having statistically higher biomass, basal area and litter, had significant increases in total nitrogen (TN) and soil organic carbon (SOC) relative to the local control site. Although there are indications that regenerative grazing can lead to improvements in land condition, this study does not enable us to conclude whether regenerative grazing will accelerate improvements compared with other best-practice grazing land management (GLM) approaches, and further research on the social and economic dimensions of regenerative grazing is needed.

Keywords: Burdekin, cattle grazing, grazing land management, land condition, pasture, regenerative agriculture, restoration, soil organic carbon, water quality.

Introduction

Regenerative agriculture is broadly defined as 'an alternative form of food and fibre production, concerning itself with enhancing and restoring resilient systems supported by functional ecosystem processes and healthy soils capable of producing a full suite of ecosystem services, among them soil carbon sequestration and improved soil water retention' (Gosnell *et al.* 2019). Regenerative agriculture has largely evolved out of economic and environmental challenges with industrial agriculture (De La Torre Ugarte and Hellwinckel 2010), and the need for more sustainable, closed, efficient and environmentally sustainable agricultural systems that have lower chemical use and can support climate change into the future (Sherwood and Uphoff 2000; Pearson 2007).

A subset of regenerative agriculture is the term 'regenerative grazing' (RG). Increasing numbers of the Australian Great Barrier Reef (GBR) natural resource managers and extension providers are working with 'regenerative' graziers that use a range of sophisticated grazing management, business, and production approaches to their operations. Beyond the GBR, regenerative grazing is increasingly seen as a profitable grazing management approach that will accelerate landscape recovery, improve soil condition and is considered as an important opportunity for linking sustainable grazing to environmental 'co-benefits' such as carbon (Briske *et al.* 2011; Teague *et al.* 2013; Gosnell *et al.* 2020). These regenerative grazing approaches are also capturing the attention of investors, in the banking and not-for-profit sector, looking to support sustainable agricultural practices in the GBR catchments.

Numerous terms have been used to describe potential components of regenerative grazing and landholders may use one or more approaches at different parts of their property over different time periods. Some of the terms include *spelling or strategic rest* (Ash *et al.* 2001, 2011), *rotational grazing* (Briske *et al.* 2008), *time controlled grazing* (Sanjari *et al.* 2008), *intensive rotational grazing* (Badgery 2017), *short duration rotational grazing* (Lawrence *et al.* 2019), *cell grazing* (McCosker 2000; Richards and Lawrence 2009), *Holistic Management* or *Holistic Planned grazing* (Savory 1983; Hawkins 2017; Gosnell *et al.* 2020) and *adaptive multi-paddock per herd grazing* (Teague and Kreuter 2020), to name a few. Many of the principles outlined in RG are the same as those applied in more traditional best-practice grazing land management (GLM) approaches, and many of the terms used are common to grazing systems and management generally. However, the subdivision of properties into numerous small paddocks, and alternating the use of high stocking densities (or mobbing cattle) for some part of the management cycle, with longer periods of strategic rest (sometimes termed pasture recovery), are often the key differentiators between RG and GLM. Noting that not all rotational grazing systems use high-density stocking rates.

The large number of terms used to describe RG components can be confusing, and it can be difficult to determine whether there are differences in the various approaches, or whether they are simply the same thing with a different name. Either because of the differences in epistemology and nomenclature (Gosnell *et al.* 2020), or the diverse landscapes on which these practices have been applied, there has been considerable controversy regarding the relative benefits of these methods for grazing production, soil and land condition, and ecosystem function (Abdel-Magid *et al.* 1987; Hart *et al.* 1993; Briske *et al.* 2008, 2011, 2014; Teague *et al.* 2013; Hawkins 2017; Hawkins *et al.* 2017; Teague and Barnes 2017; Teague and Kreuter 2020). It is not the intention of this paper to focus on the detailed management activities employed because there are several recent reviews of this topic and readers are encouraged to access this literature for more detail (e.g. McDonald *et al.* 2019; Gosnell *et al.* 2020; Newton *et al.* 2020). Rather, the focus of this paper is on the key biophysical outcomes following the implementation of the key principles of RG, which all involve some form of rest from grazing. In this context, we adopt the Ludwig and Bastin's (2008) definition of good range(land) condition being a system that has healthy biophysical functions that include a high capacity to retain water, capture energy, produce biomass, cycle nutrients and provide habitats for diverse populations of native animals, plants and microorganisms.

Several recent Australia-based studies have demonstrated improvements in ecological condition (namely plant richness and diversity) by applying RG practices. McDonald *et al.* (2019) found that total ground cover and animal production per hectare were significantly greater under strategic-rest grazing than under continuous grazing management, but biomass, plant richness, plant diversity and animal weight gain did not differ between the grazing treatments. Increasing the length of rest relative to graze time (so that rest:graze ratios were higher than 6:1) was associated with an increase in plant biomass, ground cover, animal weight gain and animal production per hectare when compared with continuous grazing (McDonald *et al.* 2019). Lawrence *et al.* (2019) found that under short-duration grazing, there was ~19% greater ground cover of perennial species, with higher-value forage species being more abundant, although they highlighted that there was also a large amount of unexplained variation in the plant community composition between treatments. This was similar to a study by Badgery *et al.* (2017), which demonstrated that even though productivity and cover were higher under intensive rotational grazing, grazing management had little influence on pasture composition. Translating the improvements in vegetation attributes to soil condition, Sanderman *et al.* (2015) found that 22% of the variation in soil organic carbon (SOC) stocks could be explained by management variables such as grazing rest periods; however, rainfall and temperature were the dominant predictors of SOC. Wang *et al.* (2018*a*, 2018*b*) also found that bare ground fraction, rainfall, elevation and Prescott index were the most important variables for explaining SOC stocks (Wang *et al.* 2018*a*) and that the proportion of bare ground and vegetation in different seasons derived from satellite imagery are important and practical indicators of surface SOC, particularly in the 0–5 cm layer (Wang *et al.* 2018*b*). The influence of climate variables such as temperature and vapour pressure deficit on SOC only increase in subtropical climates, but once detrended for climatic effects, SOC stocks are strongly influenced by total standing dry matter (TSDM), soil type and the dominant grass species (Allen *et al.* 2013; Bray *et al.* 2016). Allen *et al.* (2013) also found a small but significant negative relationship between stocking rate and SOC across 98 sites. Interestingly, Schatz *et al.* (2020) found lower per head and per area production from intensive rotational grazing than from continuous grazing, and topsoil organic carbon stocks did not increase with intensive grazing over a 5 year period. Hillslope scale soil condition, runoff and water quality in a subtropical area used for sheep grazing were found to improve under time-controlled rotational grazing after ~6 years (Sanjari *et al.* 2008, 2009), and several other studies have investigated the effect of different grazing management strategies for sheep-dominated enterprises, with mixed ecological and economic outcomes (see Behrendt *et al.* 2013; Scott *et al.* 2013*a*, 2013*b*; Shakhane *et al.* 2013).

Despite evidence supporting the benefits of regenerative grazing practices, many of the publications demonstrating scientifically robust changes are from temperate or sheep grazing systems (Pringle *et al.* 2014), and there is limited evidence from the more climatically variable seasonal dry tropics region of northern Australia which represents ~60% of the total Australian beef cattle herd (MLA 2020). Many of the publications assessing changes in vegetation and soil health in the subtropics have focused on land-use change between grazing and native forests (Allen *et al.* 2016; Dalal *et al.* 2021*b*), or an evaluation of different grazing management strategies on profitability (O'Reagain *et al.* 2011; Walsh and Cowley 2011, 2016; Owens *et al.* 2021). There is less published research demonstrating that regenerative grazing practices lead to improved land condition in rangeland systems above and beyond what would be expected under traditional best practice GLM (Hall *et al.* 2014; Segoli *et al.* 2015; MLA 2019), although many of these studies have been limited to study periods of ~3–6 years (Dowling *et al.* 2005; Hall *et al.* 2014).

To help fill this research gap, this paper investigates whether there are differences in the vegetation and soil properties among RG sites and locations that have not implemented any specific RG practices. Properties using RG strategies for between 5 and 20 years were identified within the subtropical Burdekin catchment, and a space-for-time field sampling design was developed that allowed vegetation and soil data to be collected. Data were also collected at adjacent control sites that had not undergone regenerative grazing, but where more traditional continuous set-stocking grazing approaches were applied. Coincident data were also collected from several sites where grazing had been excluded for ~30 years. Specific details related to actual cattle management (grazing periods versus spelling periods, level of pasture utilisation, herd composition, etc.) and the financial returns of various approaches were beyond the scope of this study due to resource limitations. The implications of these results are discussed in relation to the broader issue of improving runoff and water quality in the GBR catchments, as well as the potential benefit and timeframes associated with carbon and land restoration investments.

Study sites and design

All study properties were in the Burdekin catchment, which is ~130 000 km^2 and drains into the GBR Lagoon south of Townsville on the eastern coast of Australia (Fig. 1). It has the largest mean annual runoff of any of the GBR catchments at 10.29×10^6 ML, and it is also the largest contributor of anthropogenic derived fine sediment to the GBR lagoon (Mariotti *et al.* 2021; McCloskey *et al.* 2021). Due to the high sediment yields and dominance of rangeland grazing (Lewis *et al.* 2021), there is a strong focus on implementing management approaches that will potentially improve vegetation and soil condition, with the assumption that this will also lead to reduced runoff and improved water quality to the GBR.

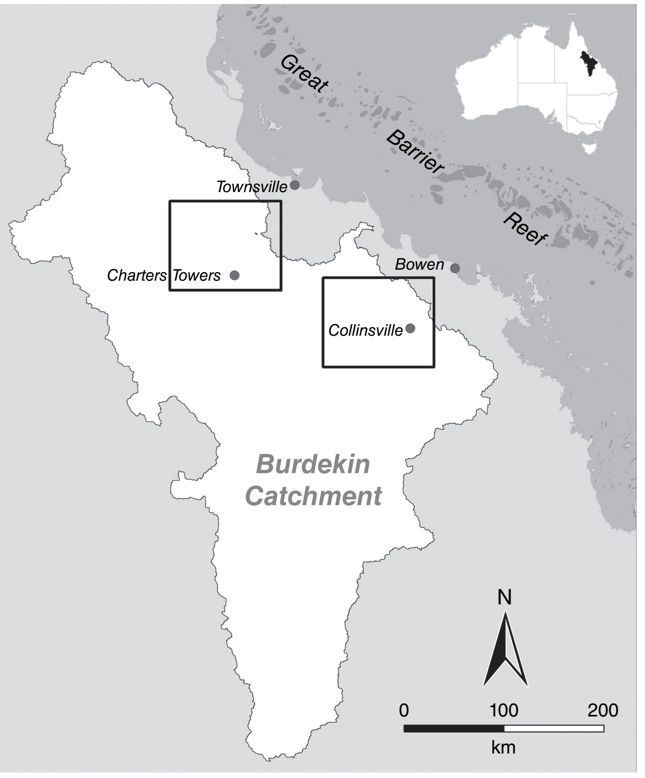

Fig. 1. General location of study properties within the Burdekin catchment, Queensland, Australia. The major towns in the region are shown for reference.

Given that changes in landscape condition can be very slow in rangeland systems (Searle *et al.* 2009), this study employed a space-for-time approach (Blois *et al.* 2013; Damgaard 2019). Approximately 10 properties were initially considered for this study, and the selection of final properties was based on (i) the history of land management at the treatment site, (ii) the ability to find a suitable paired control site that had a contrasting grazing management regime, and (iii) landholder willingness to be involved and allow access to their property for data collection. From the initial 10 properties, six properties (R1–R6) were selected for data collection. At each of the properties, the soil type, slopes, rainfall and historical vegetation changes were evaluated using a range of qualitative site assessment methods, historical records and discussions with landholders. This helped identify comparable control and treatment sites with similar landscape and rainfall attributes on (or near) each property (R1–R6; Table 1).

These properties represent a range of different management approaches from set stocking with no specific management regime, wet season resting and rotational grazing management, to complete grazing exclusion. Two of these properties, namely Virginia Park (R1) and Meadowvale (R5), were included as they had extensive previous data collected at these sites and were useful for constraining the results from the other three 'regenerative' grazing properties. Numerous studies have been published using data from Virginia Park Station, which was initially studied because of its location in an erosion-prone region (Leuning *et al.* 2005; Bartley *et al.* 2006, 2007, 2014; Wilkinson *et al.* 2018; Koci *et al.* 2020). The Meadowvale Station site was home to one of the original Queensland Department of Primary Industry (QDPI) exclosure sites, and a complete description of the site is given in Scanlan *et al.* (1996) and Hawdon *et al.* (2008). Fraser and Stone (2016) and Searle *et al.* (2009) have also published soil attribute and vegetation data from this site. Data from the nearby Department of Defence Townsville Field Training Area (TFTA), from here on represented as R6, that had very low to no grazing, but military training activities and increased fire frequency, were also included in the study (see Koci *et al.* 2020 for more details from the area).

The three 'regenerative' grazing properties, that asked not to be named, are herein called R2, R3 and R4. Each of the properties used an element of planned rotational grazing and had increased the fencing and watering-point infrastructure on their properties. The time frames associated with each of the treatments also varied among the properties (see Table 2).

For all treatment sites, except the TFTA site, a coincident 'control' site was selected either on the same property or on an adjacent neighbouring property with the same soil, vegetation, rainfall and landscape-position attributes (Table 3). The key condition for the control site was that it was to have had no specific (high-density) rotational regenerative grazing management approaches applied, and there was no planned rest as part of the management at the control sites. In most cases, the control sites were on the same property in an area that had not been included in the treatment management system due to timing or infrastructure issues. The approach involved having a 1 ha control and 1 ha treatment site as close as possible, so they were on the same land type (catena), with very similar rainfall. This provided the most suitable comparison, given the time frame for this study.

The focus of this study was primarily on capturing the key biophysical indicators, and details of the number of cattle and the period they grazed paddocks; cattle weights and herd composition were not formerly collected across all sites as coincident comparable data were not available, given the management was over a 20–30 year period for some properties. There is also the challenge of obtaining consistent data over time for the often-confused terms of stocking rate (or animals per unit area), stocking density (or animals allocated to an area for a given time period) and utilisation rate (which is the amount of pasture that is consumed), and relating these to the extended rest period commonly used in regenerative grazing (Walsh and Cowley 2011). Nonetheless, an estimate of stocking rate is provided for each control and treatment site in Table 2. To help provide comparable estimates of approximate stocking rates, two pieces of information were used. First, approximate stocking rates were calculated from collected BOTANAL data (Tothill *et al.* 1992) by using daily cattle metabolic requirements (ME) and relating those to assessed biomass removal from end of dry pasture utilisation levels collected during the study. Calculations were based on intake from an adult equivalent (AE) which can be described as a standard animal, i.e. a steer at 2.25 years old and weighing 450 kg, walking 7 km day^{-1}. The revised recommended intake for a standard animal in northern Australia is ~8 kg day^{-1} (McLennan *et al.* 2020). The second approach was to use landholder-documented or approximated stocking rates for the paddocks in which the control and treatment sites were located. Stocking rate range estimates are presented in Table 2, noting that these stocking rates are considered to be approximates only, and they should not necessarily be compared with those from more rigorous studies that have used consistent methods over time (Cowley *et al.* 2007; Hall *et al.* 2014; O'Reagain *et al.* 2018).

It is also important to note that at all properties and sites in this study, with the exception of R6, the exotic stoloniferous grass *Bothriocloa pertusa* (L.) A. Camus ('Indian couch') has largely displaced native tussock grass pastures (Kutt and Fisher 2011). The conversion from tussock to *B. pertusa*-dominated pastures typically occurs following long-term moderate to heavy grazing of fragile land types (O'Reagain *et al.* 2018) and *B. pertusa* now covers large areas of grazing land in GBR catchments (Lebbink *et al.* 2021, 2022).

Materials and methods

Rainfall during measurement period

The field data were collected across the 2020/21 wet season. Pre-wet field vegetation and soil data collection were undertaken in October and November 2020 and the post-wet field vegetation sampling was undertaken between April and June 2021. The average annual rainfall (mm) during the 2020/21 study period was between 5% and 32% lower than the long-term average across all sites (Table 4).

Remote-sensing of ground cover

In addition to the on-ground field measurements, remotely sensed ground-cover data (Department of Environment Science Queensland Government 2021) were used to

Table 1. Description of the landscape attributes associated with each of the six properties (R1–R6) documented in this study.

Item	R1	R2	R3	R4	R5	R6
Subcatchment of the Burdekin	Upper Burdekin	Lower Burdekin	Bogie	Bogie	Upper Burdekin (and Haughton)	Upper Burdekin
Size of property (ha)	~7000	~40 000	~23 000	~14 000	~3000	~11 000
Dominant soils (at the 1 ha treatment and control sites)	Red Chromosol	Sodic Vertosol	Red and Brown Chromosols	Red Chromosol	Red Chromosol	Red Chromosol
Associated soils	Sodosol	Sodosol		Sodosol	Sodosol	Rudosol
Geology						
Described for entire property – Source: Geoscience Australia, 1 M lithological mapping	Granite and Granodiorite	Alluvium overlying granite and basalt	Monzogranite, diorite and basalt	Diorite and monzogranite	Granodiorite	Granodiorite, sandstone, diorite and alluvium
Vegetation overstorey	Widely spaced Narrow-leafed Ironbark and Bloodwood	Cleared	Moderately spaced Narrow-leafed Ironbark, Ghost gum and Bloodwood	Narrow-leafed Ironbark and Bloodwood	Widely spaced Narrow-leafed Ironbark and Bloodwood	Widely spaced Narrow-leafed Ironbark and Bloodwood
Vegetation groundcover (high proportion 15–25%, moderate proportion 10–15%, low proportion <10%)	Dominated by *B. pertusa*, (Indian Couch) with other exotic grasses (*Melinis repens* (Willd.) Zizka, *Urochloa mosambicensis* (Hack.) Dandy and a high proportion of the exotic legume *Stylosanthes scabra* (*S. scabra*) (Stylo)	Dominated by *B. pertusa*, with a moderate proportion of the native annual grass *Iseilema vaginiflorum omin*	Dominated by *B. pertusa*, with a low proportion of *S. scabra* present	Dominated by *B. pertusa*, with a moderate proportion of native tussock grasses present and a high proportion of *S. scabra*	Dominated by *B. pertusa* and other exotic grasses, with *S. scabra* present in low proportions	Dominated by native tussock grasses with some *B. pertusa* incursion and a low proportion of *S. scabra*

Table 2. Description of the grazing land management (treatment) activities at each property.

Item	R1	R2	R3	R4	R5	R6
Main treatment evaluated	Wet-season spelling and conservative stocking for ~10 years (2003–2012), then sporadic higher-density rotational grazing	Rotational grazing + high-density grazing	Rotational grazing	Planned rotational grazing	Stock exclusion (exclosure paddock)	Stock exclusion, but military training activities
Year treatment was initiated	2003	2015	2007	2000	1986	2000
Number of years treatment in place (years)	~10	~5	~13	~20	~30	~20
Treatment or grazing management system	Pasture resting in alternate summer wet seasons and forage budgeting were introduced in 2002. This approach was seen to be industry best practice at the time of implementation	For the past 5 years, this property has been managed with a time control rotational grazing system and with stock numbers based on forage availability. R2 has undergone major infrastructure improvement including extensive subdivisional fencing and the development of a fully integrated/linked stock watering system to service new paddocks	Rotational grazing since 2007. Over the past 7 years, this property has been managed through holistically planned grazing, moving mobs of 1000 head of cattle from paddock to paddock managing for pasture recovery, improved land condition and for quality of pasture	Since 2000, management practices changed to a planned rotational cell grazing system. R4 now runs two mobs for almost all cattle (except bulls). There are 36 cells for the whole property with a central water point and three or four 100 ha paddocks radiating from each trough. Depending on grass stage of growth and rainfall, the grazing time for each paddock changes throughout the year	In the exclosure there has been no cattle grazing between 1986 and 1992, 'light' grazing between 1992 and 2002 (Alewijnse 2003), and no grazing for 2002–2020 (Hawdon *et al.* 2008)	Generally no grazing, but there has been some opportunistic light grazing in some years when cattle have accessed R6. This site has greater fire frequency due to military training activities
Range of stocking densities at treatment (AE km^{-2})	Approx. 25 AE km^{-2} prior to 2001 and 5–30 AE km^{-2} (average of 13 head per km^2) thereafter, although the higher rates were part of a brief trial of high-density grazing in 2019/2020	Approx. 24–25 AE km^{-2}; however, the grazing period was usually 2 weeks at a time in the wet season and 3 weeks a time in the dry – up to ~10 weeks per year. The paddock is rested for the remainder of the year	Approx. 26–28 AE km^{-2}; however, the grazing period ranged from 8 to 12 days a few times a year. The paddock is rested for the remainder of the year	Approx. 24–25 AE km^{-2}; however, the grazing period ranged from 1 to 3 days, for a total of 8 days per year. The paddock is rested for the remainder of the year	Approx. 2 AE km^{-2} (macropods)	Approx. 2 AE km^{-2} (macropods)

(*Continued on next page*)

Table 2. (*Continued*)

Item	R1	R2	R3	R4	R5	R6
Control site nearby with no specific grazing management strategy	No specific grazing management until the past few years when the paddock has been included in a rotational system	Prior to about 2015, R2 was set stocked with occasional wet-season spelling. The control paddock was largely under set-stocking	The control paddock is used as a sacrificial paddock with horses and cows	The control site is on the adjacent landholder's property and has had long-term set-stocking in larger paddocks	The control site has had set-stocking for ~30 years. Stocking rates were approximately 40 head per km^2 prior to 2001, and 25 head per km^2 since	NA
Range of stocking densities at control (AE km^{-2})	Approx. 25 AE km^{-2} prior to 2001 and 2–20 AE km^{-2} (average of 13 head per km^2) thereafter	Approx. 38 – 42 AE km^{-2}, largely set-stocked with some wet-season spelling in large paddock (since subdivided and now part of rotational grazing system)	Approx. 17–20 AE km^{-2}, without strategic rest periods (i.e. not part of rotational system)	Approx. 22–25 AE km^{-2} without strategic rest periods (i.e. not part of rotational system)	Approx. 18 – 25 AE km^{-2} without strategic rest periods (i.e. not part of rotational system)	NA

AE, animal equivalent.

Table 3. Photos of the control and treatment sites from each property.

Property	Control	Treatment
R1		
R2		
R3		
R4		
R5		
R6		

Photos taken in ~September 2020.

Table 4. The long-term (~120-year), medium term (~30-year) and annual average rainfall during measurement period for each of the properties.

Site	Long-term average annual rainfall (mm)	~30-year annual average rainfall (mm) over the remote-sensing period, 1989–2020	Annual average rainfall (mm) during sampling period, 1 July 1 2020 – 30 June 2021	% difference between measurement period and long-term average (%)
R1	614	648	450	−27
R2	704	716	670	−5
R3	734	639	638	−13
R4	825	695	678	−18
R5	672	710	454	−32
R6	663	711	577	−13

Data derived from SILO gridded rainfall (Source: https://www.longpaddock.qld.gov.au/silo/). The rainfall season is calculated from July to June each year.

provide context for the longer-term changes in pasture at each property. The long term (~30 year) cover data were climatically corrected using the dynamic reference cover (DRCM) method outlined in Bastin *et al.* (2012). Using this approach, minimally disturbed reference areas are used to benchmark changes in cover through time. The method does not require ground-based reference sites or information about land management, and thereby provides an independent check against management data.

In summary, a minimum ground-cover image is calculated across all years to identify locations of most persistent ground cover (reference pixels) in years of lowest rainfall. A moving window approach is then applied to calculate the difference between the window's central pixel and its surrounding reference pixels (Bastin *et al.* 2012). This difference estimates ground-cover change between successive below-average rainfall years, which provides a seasonally interpreted measure of management effects.

It is acknowledged that in some rangeland areas, there are systematic biases as well as random errors in the remotely sensed cover data when compared with on-ground measurements of cover. This is particularly prevalent in the Upper Burdekin areas, or areas dominated by *B. pertusa* (Bastin *et al.* 1996; Wilkinson *et al.* 2014). Therefore, a comparison of the remotely sensed ground cover data with the field measurements of ground cover was undertaken to quantify the likely bias between cover data sets.

Vegetation measurements

Land condition indicators most correlated with soil organic carbon (SOC) stocks across previous data collected in northern Australian rangelands include measures such as tree canopy cover, grass basal area, ground cover, pasture biomass, and the density of perennial grass tussocks (Bray *et al.* 2016). Therefore, many of these attributes were the focus of data collection for this study.

For each 1 ha site, 10 (100 m) transects were evenly distributed across the site, and 10 equidistant quadrats were located along each transect. Each transect had a permanent marker at the beginning and end to facilitate repeat measures from before and after the wet season. Pasture metrics were recorded along each transect using a 1 m^2 quadrat, based on the methods of Tothill *et al.* (1992) (Table 5). Metrics included defoliation level, above-ground pasture biomass as dry matter yield (DMY), total cover, grass basal-area, litter cover, identification of all pasture species and/or functional group composition and frequency, tree canopy cover. Key soil surface condition metrics were also measured (Tongway and Hindley 1995) (Table 5). Cover and biomass estimates were calibrated against standard quadrats taken at each site by using classified quadrat photographs and cut samples. Biomass standards were oven dried to attain dry matter yield, removing vegetation water retention error among treatments.

Soil measurements

Table 6 outlines the key soil attributes that were measured at each 1 ha field site. An appropriate sampling design is important to ensure that soil characteristics are adequately captured across the study paddocks and soil profiles. To avoid potential bias, a stratified random design-based approach was employed (Papritz and Webster 1995). This approach classifies the area according to geographical coordinates, ensures a good representation of sites compared with simple random sampling and reduces sampling bias (Allen *et al.* 2010).

For each of the 1 ha sites, a *k*-means clustering approach was applied to stratify each site into five approximately equal zones. This was undertaken using the 'stratify' function from the R software package 'spcosa' (Walvoort *et al.* 2010). Two random sampling locations were then identified within each zone. This was performed by using the 'spsample' function from the R package. The randomly selected sampling locations were evaluated at each site and the function was reapplied for any locations that were near the site boundary to avoid edge effects and bias. The selected sample location coordinates were imported into a portable global positioning system (GPS) device (Trimble,

Table 5. Vegetation and land-condition attributes measured in this study; vegetation metrics were measured using the BOTANAL technique (Tothill *et al.* 1992) and soil surface condition metrics after Tongway and Hindley (1995).

Attribute	Unit	Description
Vegetation metrics		
Defoliation	%	Proportion of the potential total biomass that has been removed by grazing animals or fire
Biomass	$kg\ ha^{-1}$	Weight of standing plant material (not litter)
Cover	%	Total ground cover; anything covering the soil that is not bare ground, as seen from above.
Basal area	%	Area of soil covered by grass bases, at the soil/tussock intersection.
Litter cover	%	Proportion of total cover that is made up of dead plant material (touching the soil surface)
Tree canopy cover	%	Canopy cover directly above sample site
Species composition	%	Proportion of individual species by biomass share
Soil-surface condition metrics		
Litter incorporation	Index	Level of litter incorporation into soil surface
Soil hardness	Index	Level of soil hardness
Erosion severity	Index	Level of erosion severity

Juno SB, USA) to track each sampling locations with an average accuracy of 5 m in the field.

Soil samples were taken with 43 mm (ID) soil corer/tubes, using a pneumatic jackhammer. Samples were removed and placed on a graded half pipe to separate soil depth increments. Each incremental depth was stored in a separate labelled bag and undesirable soil increments were discarded. Samples were weighed immediately and taken to the laboratory for air-drying, grinding, sieving and analysis. Previous studies have demonstrated that the largest differences in soil attributes are generally in the 0–10 cm layer when changing land use or land management (England *et al.* 2016), and therefore only data from 0 to 10 cm layer are presented.

Statistical analysis

Due to the variation in the mean annual rainfall, soil type and geology, it is challenging to compare data among properties from different areas. This is because rainfall, climate and soil type are known to be strong drivers of vegetation and soil condition (Allen *et al.* 2013), and therefore any differences between properties are more likely to be due to environmental conditions than to management effects. Therefore, only control and treatment site data from within each property were statistically compared for changes. Analysis of basic soil chemical and physical properties were conducted to investigate consistency between controls and treatments across sites.

To examine the effect of season and management on continuous vegetation, two-way analysis of variance (ANOVA) was carried out. Similarly, a general linear model (ordinary least squares, OLS) ANOVA model was used to analyse the ordinal data by using the same two factors. Chi-squared tests for homogeneity were used to analyse the categorical species data for difference in species composition among treatments, with z-test for proportion used to test for significant difference in total species numbers among treatments. Ordinal and categorical datasets were compared among treatments only, being aggregated across seasons. All statistical analysis for the vegetation data was completed using the R package (R Core Team 2015). For the soil data, normality and equal variance tests were run to test the assumptions for Student's t-test. When the data were not normally distributed, a Mann–Whitney Rank-sum test was applied, and when the test for equal variance failed, Welch's test was applied. All the analyses were done by using SigmaPlot version 14 (Systat Software Inc., https://systatsoftware.com/sigmaplot/).

Results

Vegetation

Remote-sensing

To place the single season of on-ground measured vegetation data into a longer-term perspective, we compared the field-based percentage cover with Landsat-derived percentage cover. It has long been known that remote-sensing may over- or under-predict ground cover (Scarth *et al.* 2006), with over-prediction typical in the Burdekin basin (Wilkinson *et al.* 2014). Fig. 2 highlights that the Landsat-derived percentage cover estimated for each of the 1 ha field sites has been over-estimated for most sites in this study. The over-estimation is more pronounced for sites with 50–70% ground cover. The relationship between observed versus predicted ground cover, for the cover model used here, is generally better for sites with <20% or >80% cover (Scarth *et al.* 2006). To obtain a temporal correction for the Landsat cover data, we used a 20 year ground-based data set collected at Virginia Park (which is a similar *B. pertusa*-dominant landscape; Bartley *et al.* 2014). The

Table 6. Soil physio-chemical attributes used to verify comparability of paired sites, and the soil chemical attributes used to represent soil condition in this study.

Attribute	Unit	Primary purpose for measurement		Method
		Verification of paired site comparability	Soil health indicator likely to be affected by grazing	
Soil physico-chemical attributes				
Soil particle size distribution		√		Gee and Bauder (1986)
Soil texture		√		Determined from particle size distribution (The National Committee on Soil and Terrain 2009)
Soil acidity	pH	√		Method 4A1 (1:5 Water) (Rayment and Lyons 2011)
Cation exchange capacity (CEC)	$Cmol_c\,kg^{-1}$	√		Method 15A2 (Rayment and Lyons 2011)
Electrical conductivity (EC) 1:5 (Soil:Water)	($dS\,m^{-1}$)	√		Method 3A1 (1:5 Water) (Rayment and Lyons 2011)
Exchangeable sodium potential (ESP)		√		US Salinity Laboratory Staff (1954)
Soil chemical attributes				
Total nitrogen (TN)	$mg\,kg^{-1}$		√	Bremner (1960)
Total organic carbon (SOC)	$t\,ha^{-1}$		√	Walkley and Black (1934)
C:N ratio			√	Calculated from TOC and TN

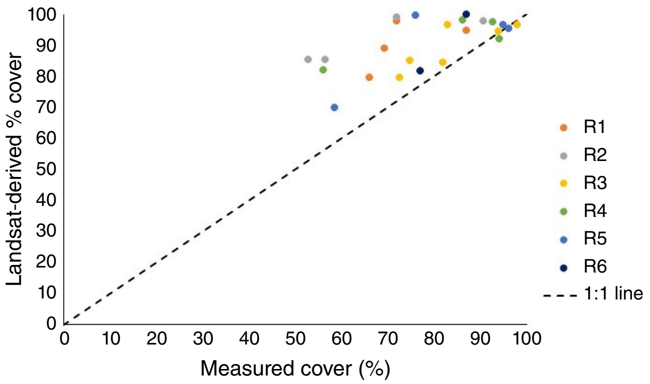

Fig. 2. Comparison of measured ground cover and Landsat-derived cover calculated for 12 1-ha sites across six properties. The comparisons were made for the end of dry 2020 (September–November) and end of wet 2021 (March–May) across control and treatment sites.

relationship between ground measured and remotely sensed data was good at $\sim r^2 = 0.8$. The resulting regression equation was applied to the Landsat cover data presented in Fig. 3. This resulted in a mean difference between the on-ground measured cover and Landsat-derived cover of -14 (± 5)% to the Landsat original values; however, this varied considerably among sites (see Supplementary Table S1).

Of the six sites, in three cases (R3, R4, R5), the treatment curves trended above the control curves following onset of treatment, although there was a ~5 year lag in the response at R4. The treatment curves at R3, R4 and R5 also tended to be above the DRCM reference cover (Fig. 3), although all treatment sites went close to, or below, the regional benchmark (dashed line) at some point following treatment, demonstrating how tenuous and fragile the cover response can be. Fig. 3 highlights the impact of frequent fire on %cover at the R6 site.

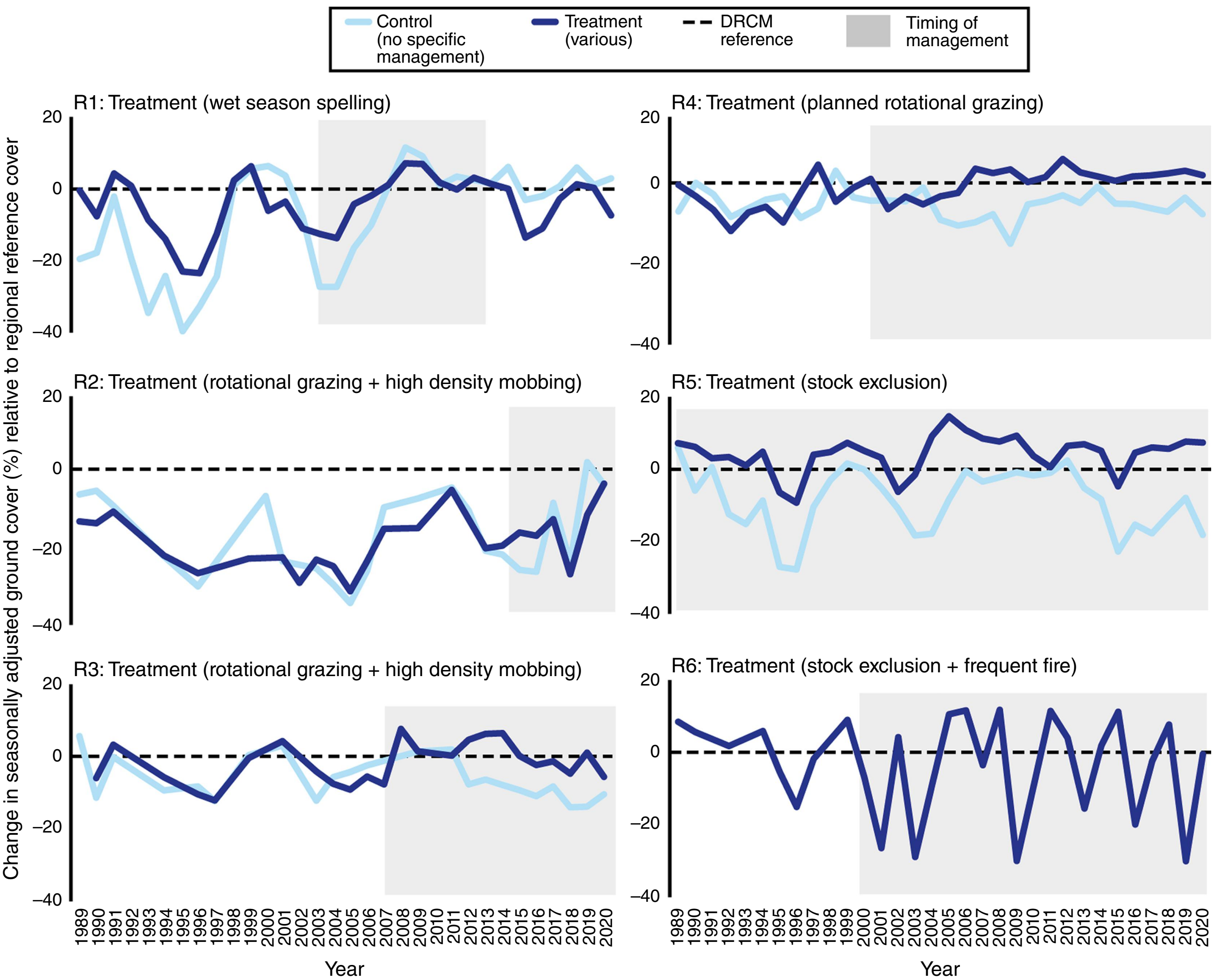

Fig. 3. Change in end of dry ground cover at each of the control (no RG) and treatment (RG at R2, R3 and R4) sites relative to the dynamic reference cover method (DRCM) pixel condition (dashed line) with grey zones representing the treatment period. These data are derived from Landsat-derived ground cover by using the method outlined in Bastin *et al.* (2012).

Vegetation data

Results of key vegetation metrics are presented in Table 7. Fig. 4 visually shows the high variability within and among sites for five key vegetation attributes (defoliation, biomass, total cover, litter cover and grass basal area). None of the sites had statistically different tree canopy between control and treatment sites, which is important because tree proximity can have a significant effect on SOC (Waters *et al.* 2015). This suggests that any differences in the vegetation attributes between control and treatment sites are likely to be due to management effects.

The percentage defoliation, or amount of pasture removed by grazing (or fire), was higher at the control sites in the pre-wet season period, than at treatment sites, for all sites except R1. The difference was not significant at Site R2 at the end of the wet season, suggesting that both the control and treatment sites at R2 had been rested. In general, the percentage defoliation was also much higher at all sites in the pre-wet period, as pasture levels are generally much lower prior to the wet season than at post-wet period.

The control sites represented areas that had not undergone any RG management in the form of high-density rotational grazing with long periods of rest and were largely representative of a set-stocking regime. The pre-wet biomass for the control sites ranged between ~100 and 1000 kg ha^{-1} (Fig. 4), and post-wet biomass ranged between ~450 and 2200 kg ha^{-1} (Fig. 4). These control-site biomass levels were generally much lower than sustainable levels measured elsewhere (Ash *et al.* 2011), although the below-average rainfall at all sites during sampling (see Table 4) may partly explain these low biomass levels. The pasture biomass levels measured at the treatment sites before the wet season (November 2020) were <1400 kg ha^{-1} (Fig. 4). The pasture biomass levels in the treatment sites at the end of the wet season (April/May 2021) were between 2 and 10 times the pre-wet biomass levels, and the biggest changes in the pre- and post-wet biomass were at the grazed RG treatment sites (R2, R3 and R4). At the ungrazed R5 treatment and R6 sites, biomass doubled between pre- and post-wet conditions (Fig. 4). Notably the proportion of the plant biomass represented by non-native or exotic grasses ranged from 47% to 98% for pre-wet and from 56% to 99% for post-wet sampling for Sites R1–R5 (Table 7). The exotic grasses were overwhelmingly dominated by *B. pertusa* (Table 7). Only site R6 had >70% native grass species representing the ground biomass.

Sites R4 and R5 treatments both showed statistically significantly improved vegetation and soil-surface condition values relative to the neighbouring control sites (Fig. 4). Site R4 treatment has undergone ~20 years of planned rotational RG and site R5 treatment was a stock exclosure that has had little or no grazing for ~30 years. Both R4 and R5 treatment sites had similar percentage cover both before (94–95%) and after (93–95%) the wet season (Fig. 4). The pasture biomass levels at the R4 treatment site were ~5400 kg ha^{-1} at the end of the wet season; however, the proportion of total biomass represented by legumes was ~66%. Site R5 had lower defoliation, which is a measure of the proportion of the potential total biomass that has been removed by grazing animals (or fire), and the proportion of total biomass represented by legumes at the R5 treatment site was ~19%. The plant basal area was statistically higher at both the R4 and R5 treatment sites than at their adjacent control sites. Litter incorporation, soil hardness, and erosion severity, which represent soil-surface condition, were significantly higher at both the R4 and R5 treatment sites, than in the control. Both the number of grass species and the proportion of native grasses has significantly improved at both R4 and R5 treatment sites, with the proportion of native grasses as a proportion of total biomass reaching ~45% at R5 and ~25% at R4. These results suggest that improved vegetation and soil-surface condition was observed at a site that has had ~20 years of rotational planned grazing, with long rest periods (R4), as well as at a site that has had ~30 years of stock exclusion (R5).

Sites R1, R2 and R3 showed greater variability for vegetation attributes between control and treatment sites. Site R1 treatment involved wet-season spelling for ~10 years, and according to the long-term remote-sensing of ground cover (Fig. 3), the treatment site increased its percentage ground cover relative to the control in the initial treatment period (between ~2003 and 2013); however, the control site percentage cover matched that in the treatment site after ~5 years and has remained higher at that site. The remote-sensing data also suggested that the spelling management practice was not necessarily maintained in more recent years, and, subsequently, percentage cover has fallen at the treatment site (see Fig. 3). The R2 treatment, which is the 'youngest' of the treatments, also did not show improvements in defoliation (a surrogate for grazing impact), indicating that the grazing management may still be fluctuating at this site or the time for improvement has been too short.

There was no difference in percentage basal area between control and treatment sites at R2 and R3. The R6 site, which has no grazing, but more regular fire, had a higher percentage grass basal area than did all other sites, which is supported by the fact that the species mix contributing to biomass at this site is ~77% native pastures, and highlighting that basal area generally increases with the proportion of tussock grasses. The regular fire at this site has resulted in an extreme fluctuation of ground cover over the past 20 years (see Fig. 3, R6); however, the pre- and post-wet season measured average ground cover was not markedly different from that at the other grazed sites. Although the litter cover was relatively low at the R6 site (compared with the R5 exclosure site), continued low rates of litter incorporation suggested that, in the absence of grazing, organic material incorporation can improve some, but not all, aspects of land condition (Fig. 4).

Table 7. Differences in vegetation attributes between control and treatment sites (significant difference at 95% confidence level is highlighted in bold).

Attribute	Period or category	R1		R2		R3		R4		R5		R6
		Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Treatment
Continuous variables												
Defoliation (%)	Pre-wet	**8 (±12)**	70 (±19)	87 (±5)	**63 (±27)**	95 (±0)	**54 (±33)**	72 (±25)	**41 (±30)**	87 (±4)	**10 (±14)**	7 (±11)
	Post-wet	**3 (±1)**	6 (±13)	3 (± 4)	4 (±12)	55 (±20)	**3 (±2)**	14 (± 24)	**2 (±0)**	14 (±22)	**0 (±0)**	4 (±5)
Biomass (kg ha^{-1})	Pre-wet	**998 (±524)**	327 (±226)	170 (±339)	**514 (±556)**	110 (±53)	**484 (±503)**	262 (±355)	**955 (±777)**	88 (±68)	**771 (±828)**	1320 (±663)
	Post-wet	**2174 (±910)**	1485 (±1107)	1632 (±905)	1834 (±732)	463 (±139)	**1953 (±1146)**	1252 (±1213)	**5400 (±3761)**	567 (334)	**1523 (±1759)**	2711 (±1580)
Species – native:exotic (% biomass)	Pre-wet	12:88	14:86	33:67	21:79	2:98	**24:76**	7:93	**34:66**	3:97	**53:47**	81:19
	Post-wet	7:93	9:91	13:87	**44:56**	1:99	**8:92**	16:84	16:84	6:94	**36:64**	72:23
Species – *B. pertusa*: exotic legume: other (% biomass)	Pre-wet	25: 27: 48	71: 16: 13	94: 0: 6	93: 0: 7	98: 1: 1	86: 8: 6	100: 0: 0	75: 14: 11	99: 1: 0	20: 17: 64	26: 3: 72
	Post-wet	53: 14: 33	61:15:24	49: 2: 49	48: 22: 30	98: 1: 1	69: 24: 7	93: 1: 6	17: 66: 17	98: 1:1	32: 19: 49	23: 13: 64
Cover (%)	Pre-wet	69 (±90)	66 (±22)	53 (±24)	56 (±27)	73 (±18)	**82 (±19)**	56 (±31)	**94 (±13)**	59 (±21)	**96 (±13)**	77 (±16)
	Post-wet	**86 (±15)**	71 (±24)	72 (±27)	**91 (±13)**	94 (±12)	98 (±6)	86 (±20)	**93 (±11)**	75 (±24)	**95 (±12)**	87 (±15)
Litter cover (%)	Pre-wet	28 (±27)	32 (±31)	**43 (±22)**	24 (±25)	16 (±19)	**63 (±27)**	24 (±23)	**58 (±20)**	7 (±17)	**87 (±17)**	20 (±23)
	Post-wet	9 (±18)	**22 (±31)**	6 (±10)	**13 (±15)**	9 (±14)	**44 (±37)**	17 (±24)	**34 (±28)**	6 (±16)	**78 (±25)**	20 (±28)
Basal area (%)	Pre-wet	**0.58 (±0.36)**	0.26 (±0.18)	0.17 (±0.32)	0.27 (±0.29)	0.30 (±0.13)	0.38 (±0.43)	0.24 (±0.13)	**0.99 (±0.90)**	0.23 (±0.2)	**0.64 (±0.7)**	1.36 (±0.89)
	Post-wet	0.53 (±0.26)	0.42 (±0.67)	0.26 (±0.34)	0.26 (±0.14)	0.58 (±0.27)	0.50 (±0.45)	0.28 (±0.15)	**0.56 (±0.64)**	0.36 (±0.15)	**0.6 (±0.70)**	0.80 (±0.55)
Tree canopy (%)	Pre and post-wet (aggregated)	20 (±36)	24 (±40)	Nil	Nil	25 (±42)	34 (±44)	28 (±43)	20 (±38)	10 (±28)	25 (±41)	20 (±35)
Categorical/ordinal variables												
Litter incorporation (count)	1 – Nil	42	31	6	3	24	**12**	50	**3**	89	**2**	53
	2 – Low	52	51	94	96	74	**70**	43	**74**	7	**25**	40
	3 – Medium	6	9	0	1	2	**8**	7	**23**	1	**42**	7
	4 – High	0	1	0	0	0	0	0	0	0	31	0

(*Continued on next page*)

Table 7. (*Continued*)

Attribute	Period or category	R1		R2		R3		R4		R5		R6
		Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Treatment
Soil hardness (count)	1 – Sand	0	0	0	0	0	0	0	**0**	0	**1**	0
	2 – Easily broken	3	8	0	0	6	1	4	1	1	73	5
	3 – Self mulching	0	1	100	100	0	0	0	0	0	0	0
	4 – Moderately hard	28	34	0	0	40	35	15	70	2	20	37
	5 – Very hard	69	58	0	0	54	64	81	29	93	6	58
Erosion severity (count)	1 – Nil	**83**	42	100	100	82	93	58	**99**	47	**90**	82
	2 – Slight	**17**	50	0	0	18	7	36	**1**	41	**10**	18
	3 – Moderate	**0**	7	0	0	0	0	5	**0**	8	**0**	0
	4 – Extensive	**0**	1	0	0	0	0	1	**0**	1	**0**	0
Species (count)	Total native:exotic	**60*** (47:13)	56 (46:10)	45 (29:16)	**69** (49:20)	41 (30:11)	**67** (49:18)	48 (37:11)	**80** (60:20)	38 (29:9)	**76** (59:17)	64 (35:29)

Species count numbers in bold indicate a significant difference in species composition and number of species unless indicated by * indicating significant difference in species composition only.

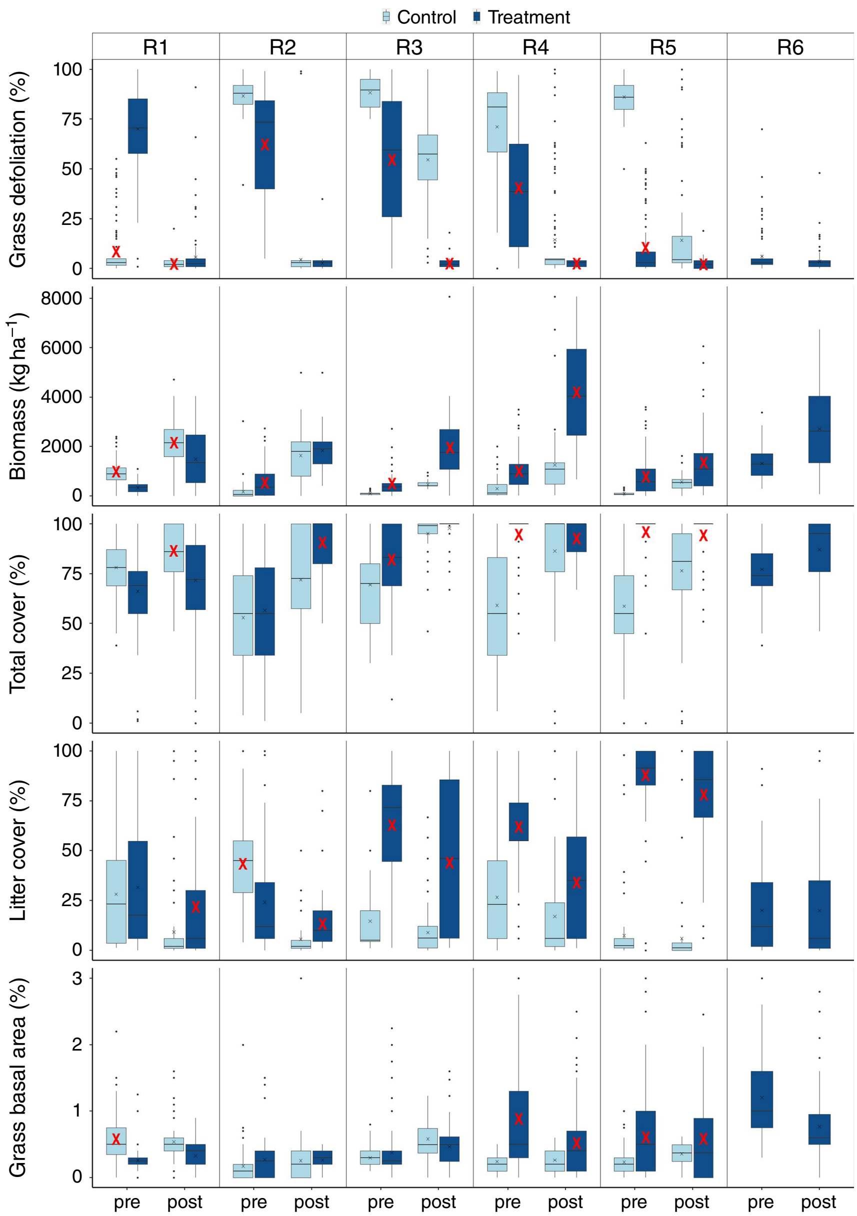

Fig. 4. Key vegetation metrics measured at the control and treatment sites in November 2020 (pre-wet) and April 2021 (post-wet). X indicates a significant difference at 95% confidence level between control and treatment sites.

Soil data

For the six soil attributes measured to test the physio-chemical comparability across the five paired treatment and control sites, there was only one variable that was identified as being statistically different, namely, pH at Site R1, with that in the treatment being significantly higher than in the control (Table 8). Given that there was no significant difference in any of the other physio-chemical attributes between control and treatment sites, we assume that management activities are a likely cause of any difference in soil condition as observed in the soil health indicator data shown in Supplementary Table S1.

All the chemical soil-health indicators, total nitrogen (TN), soil organic carbon (SOC) and C:N ratio showed significant differences at some sites (Fig. 5). Both TN and SOC were higher in the treatment sites for Sites R4 and R5. The SOC was also higher at R1 treatment, which was unexpected, because there was no change in any of the vegetation attributes to explain the increase in SOC (see Fig. 4). We assume this result to be likely due to a difference in soil type (this site did have higher a pH than the control) or some other non-management-related factor. Site R2 treatment also had a small, marginally significant difference in SOC, despite having inconsistent changes in the vegetation metrics. This may have been related to weed infestation at this site in recent years. Comparing across all sites, R3 treatment had the highest overall TN and SOC values of any site; however, given TN and SOC was high at both the treatment and control site, there was no significant increase attributed to RG at R3.

Table 8. Differences in mean value of soil attributes used as soil-health indicators between control and treatment sites for the 0–10 cm layer (significant difference at 95% confidence level is highlighted in bold).

Attribute	Unit	R1		R2		R3		R4		R5		R6
		Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Treatment
Soil acidity/alkalinity	pH	7.0	**7.8**	8.3	8.2	6.7	6.7	6.8	6.6	6.8	6.9	7.0
Electronic conductivity (EC)	$dS\ m^{-1}$	0.04	0.06	0.23	0.17	0.11	0.05	0.08	0.07	0.04	0.05	0.02
Cation exchange capacity (CEC)	$Cmol_c\ kg^{-1}$	7.88	10.26	49.56	53.45	25.14	16.06	10.64	13.45	7.34	14.53	
Exchangeable sodium potential (ESP)	%	1.83	2.25	4.3	2.61	0.57	1.5	1.19	0.96	2.25	1.68	
Percentage sand	%	62.2	71.2	17.8	24.6	52.3	53.7	66.9	56.3	68.8	57.5	75.4
Percentage silt	%	7.9	8.3	19.6	19.2	17.2	17.4	9.3	13.3	7.1	8.7	6.8
Percentage clay	%	29.2	20.4	63.3	56.2	30.5	28.9	23.7	30.4	24.2	33.8	20.4
Soil texture		Sandy clay loam	Sandy clay loam	Clay	Clay	Sandy clay loam	Sandy clay loam	Sandy clay loam	Sandy clay loam	Sandy clay loam	Sandy clay loam	Sandy clay loam
Total nitrogen (TN)	%	0.04	0.05	0.05	0.06	**0.12**	0.10	0.06	**0.11**	0.05	**0.07**	0.05
Total organic carbon (SOC)	%	0.71	**0.91**	0.97	**1.08**	2.28	2.12	1.28	**2.15**	0.93	**1.28**	1.13
C:N ratio (derived from individual samples)		19.03	19.84	19.93	20.45	19.77	**21.54**	22.43	21.32	21.35	20.05	22.23

All data were collected in October 2020.

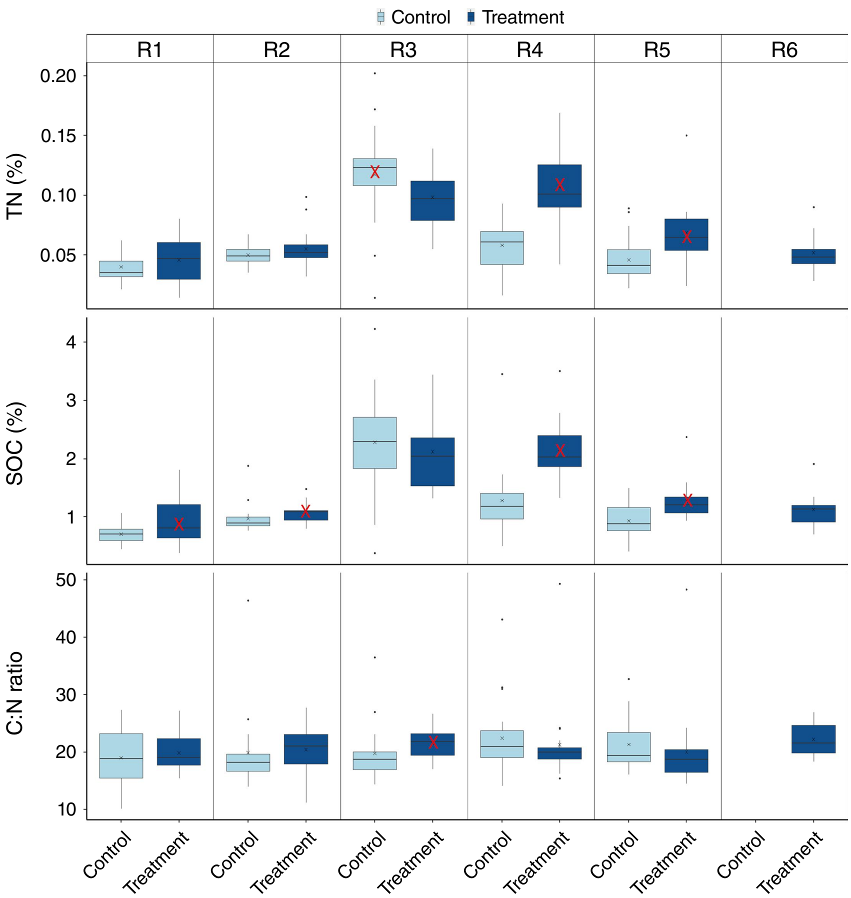

Fig. 5. Total nitrogen (TN)%, and soil organic carbon (SOC)%, and C:N ratio for the 0–10 cm layer at each site. X indicates significant difference at 95% confidence level between control and treatment sites.

Conversely, at Site R4 treatment, which has undergone regenerative grazing and included extended rest periods for ~20 years, there has been a statistically significant improvement in all the vegetation attributes (relative to the control site) and a subsequent doubling or 100% increase in TN and a 74% increase in SOC relative to the control site. At Site R5, where the grazing exclosure has not had any grazing for ~30 years, there was also a ~40% increase in TN and a ~35% increase in SOC relative to the control site. Interestingly, both R4 and R5 treatments had considerably higher TN and SOC than did the R6 site, which has had low or no grazing, but increased fire, over the past ~20 years (Fig. 5). The only site that had a statistically significant difference in the C:N ratio was at R3, with the value for the treatment site being higher than that for the control site.

Discussion

Space-for-time versus longitudinal studies – the trade off

The results presented here generally validated the utility of the space-for-time approach, but, as discussed later, in future studies we would include additional metrics to make the approach even more robust. It is acknowledged that long-term studies of landscape change in response to variable grazing management (e.g. Bartley *et al.* 2014; O'Reagain *et al.* 2018; Dalal *et al.* 2021*a*) provide more robust data than space-for-time studies that may lead to erroneous conclusions (Damgaard 2019). However, given the current pressures on identifying approaches for improving landscape condition, there is a need for long-term studies to be supplemented with space-for-time approaches such as those presented here. This is particularly the case where we are trying to identify processes of landscape repair or recovery, as opposed to land degradation. For example, Pringle *et al.* (2011) suggested that ~12 years might be too short to see coarse-scale effects of grazing management on variables such as soil organic carbon (SOC). Hence, in a practical sense, we do not have time to wait >12+ years to identify approaches that may be useful at regenerating grazed landscapes. However, this need for urgency does come with a caution. Although studies such as that by Blois *et al.* (2013) have suggested that the judicious use of space-for-time approaches is appropriate in many cases, we need to be mindful that these results may not capture the detailed and specific management activities and processes such as climate extremes (e.g. droughts) that can have important implications on study results. For these reasons, the results from this study are considered as a preliminary insight into the likely changes that could occur for sites under similar climate, soil, vegetation, and stock management conditions.

Vegetation and soil metrics for measuring improvements in land condition

There were insufficient data in this study to conduct robust statistical analysis of the relationship between the key vegetation and soil attributes. Nonetheless, these data suggested that there are some attributes that are better at estimating improved land condition than are others. We propose that land condition represents a combination of vegetation and soil metrics, with the assumption that improvement in both vegetation and soil attributes will lead to improved land condition, and more specifically, improved hydrological function (see Fig. 6).

In this study, grass biomass was found to be a better discriminator of soil condition than was percent cover. Biomass, of pastures and trees, is not only indicative of potential productivity above ground, but also below ground as root mass, improving biological activity, carbon storage,

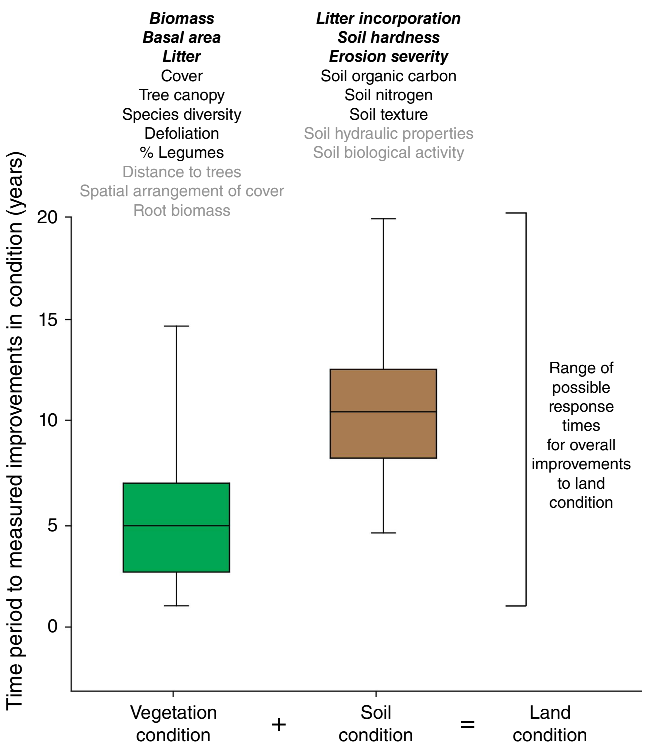

Fig. 6. Conceptual diagram showing the range in the likely response-time periods for improved vegetation and soil condition, which together lead to improved land condition. The key metrics for measuring vegetation and soil condition are listed, with those in black **bold** measured in this study, and the most influential metrics in *italics*. Metrics listed in grey are considered important, but were not measured in this study.

and water infiltration (Northup *et al.* 1999; Jackson and Ash 2001). Also, sites dominated by *B. pertusa*, which includes all sites in this study (with exception of Site R6), can often have relatively high percentage cover, because *B. pertusa* is a stoloniferous (runner) grass, and the higher cover does not necessarily represent higher biomass or basal area. Due to the utility of remote-sensing data products, percentage cover is often used as an indicator of land condition (Karfs *et al.* 2009; Beutel *et al.* 2021). However, percentage cover of pasture often reflects shorter-term changes in land condition and is strongly related to local rainfall and recent grazing pressure and is not a strong predictor of soil hydrological function (Roth 2004). The spatial arrangement of cover, rather than average cover, has also been found to be a better indicator of runoff and erosion in rangelands (Bartley *et al.* 2006).

Plant basal area is considered a useful metric for explaining long-term land-condition changes that happen slowly (Northup *et al.* 2005; Searle *et al.* 2009), and can be a good indicator of historical longer-term impacts and landscape condition. There is evidence for higher basal area leading to better soil moisture (Northup *et al.* 2005; Searle *et al.* 2009), and others have found that there is a strong link between higher basal area and biomass, and biomass and infiltration (Fraser and Stone 2016). The R2 and R3 treatment sites are showing statistically significant improvements in many of the vegetation attributes, including biomass, cover, litter cover and species diversity, yet percentage basal area has not changed at these sites (Fig. 4). Basal area generally increases with the proportion of tussock grasses, suggesting that plant percentage basal area is potentially slower to change following improved grazing management than are other vegetation attributes. Good basal area values in the Burdekin region are generally in the range of ~2–3% (Rogers *et al.* 1999), and the basal area even at the 'improved' treatment sites is still relatively low, with R4 at ~0.56–0.99%, and R5 at 0.6–0.64%. This is likely to reflect the widespread dominance of *B. pertusa* at these sites. This may also reflect the new 'normal' for improved land condition in these *B. pertusa*-dominated landscapes.

In this study, a statistically significant improvement in litter was linked to improvements in TN and SOC at some, but not all, sites. Litter can be a source of, and trap for, organic matter, which can in turn improve SOC in the landscape (Hodgkinson and Freudenberger 1997; Orgill *et al.* 2017). Unfortunately, litter, biomass and basal area are not necessarily related in linear and uniform ways (Lodge and Murphy 2002) and are not necessarily well correlated to percentage ground cover. Litter is also strongly linked to tree cover in rangelands (Jackson and Ash 1998).

The proportion of litter incorporated into soil, soil hardness and erosion severity are also useful indictors for distinguishing improved land condition at the site scale, and they seem to correlate with improvements in the other key vegetation metrics (Fig. 4). Roth (2004), using similar soil surface-condition attributes, established a strong relationship between soil surface condition, infiltration and sediment, nitrogen, and phosphorus concentrations in runoff. Fraser and Stone (2016) also found that soil texture was a useful predictor of soil infiltration rates, as well as influencing the amount of soil carbon at a site.

Total nitrogen (TN), SOC and C:N ratio are useful, but these metrics are also strongly related to *in situ* soil and climate conditions (Bray *et al.* 2016), and therefore using these data on their own to infer changes in land condition needs caution. For example, the R3 treatment site had a significantly higher C:N ratio than did the corresponding control site, but without a coincident increase in TN% or SOC%. This is an unusual result and does not conform with those of other studies that have generally shown that higher concentrations of TN and SOC and C:N ratio were each associated with higher ground cover of perennial plants (Waters *et al.* 2015). TN, SOC and C:N metrics are best used in conjunction with vegetation and surface-condition metrics and/or in longitudinal studies looking at changes in these attributes over time. Although not explicitly assessed in this study, other studies have also shown that distance to trees is an important component influencing SOC (Bray *et al.* 2016; Orgill *et al.* 2017), and this should also be addressed in any future work linking metrics to land condition.

Our results are similar to those of Bray *et al.* (2016) who found that following the removal of climate effects, land condition indicators most correlated with SOC stocks were tree basal area, tree canopy cover, ground cover, pasture biomass, and the density of perennial grass tussocks. Allen *et al.* (2013) also found that significant reductions in SOC stocks were associated with decreasing pasture total standing dry matter and the dominant grass category, with Exotic Other (>25% *B. pertusa* and *Urochloa mosambicensis* (Hack.) Dandy) being associated with significantly lower SOC than the dominant grass categories Exotic Buffel (>25% *Cenchrus ciliaris* L.) and Native Grass (25% native pasture species). In contrast, Orgill *et al.* (2018) found that removing grazing pressure has been shown to lower soil carbon stocks in south-eastern Australia, and Pringle *et al.* (2011) and Segoli *et al.* (2015) both showed that high stocking rates or heavier grazing intensity can produce higher SOC and higher soil organic matter. The proportion of legumes represented at the treatment sites did not seem to influence soil conditions adversely; however, it is something to consider in the long-term, given that legume dominance has been associated with soil acidification, nutrient depletion and increased soil erosion (Noble *et al.* 2000).

In this study we propose that multiple vegetation and soil attributes will be needed to measure improved land condition. No single attribute will adequately represent the likely recovery pathway from a degraded condition, with high runoff and poor water quality, to a healthy and improved condition. In this context, Fig. 6 becomes a working hypothesis or conceptual framework requiring additional testing. Noting that this framework should be applied to grazed hillslopes and it is not necessarily transferrable to rangelands that are highly dissected with severe and active gully erosion. It is likely that similar attributes will be important for gullied landscapes; however, the recovery time lags will be longer (Bartley *et al.* 2020).

Improving hillslope land condition – how long does it take?

Results from this study suggest that improvements in vegetation and soil and, thus, land condition can be obtained from implementing a range of RG principles in semiarid rangeland areas of northern Australia. The question is, what is the timeframe for recovery, and can RG accelerate improved land condition compared with cattle exclusion? Depending on whether the improvements were initiated during dry years or wet years, and the extent to which biomass was utilised by cattle, improvements in land condition using RG might be quite rapid. Although our study showed that, after 20 years, we were able to distinguish significant differences in vegetation, TN and SOC parameters at R4 and R5 treatment sites, this may have occurred a lot sooner. In fact, Fig. 3 suggests that the RG treatments in R3 and R4 started affecting ground cover, as measured by remote-sensing, a lot sooner, implying that the observed level of soil improvement may also have occurred sooner than after 20 years at R4, and changes were imminent at R3. Unfortunately, it is not currently possible to use remote-sensing to estimate the other key variables, such as basal area and litter. Although we can observe the onset of change using remote-sensing of percentage cover, the study design employed here does not allow for the specific point in time to be determined whereby statistical improvements become measurable at a site, particularly for sites that are moving from a very degraded baseline condition.

Given that vegetation and soil condition are best represented by a range of attributes, and these attributes will respond at different rates, the time scales for overall land-condition improvement will vary widely (Fig. 6). Based on this study, vegetation condition is likely to take between ~3 and 15 years to shift, and soil condition ~5–20 years. Overall land condition in highly degraded grazed rangelands is likely to take between ~5 and 20 years to demonstrate measurable improvements, and the final condition may be very different from the original (less disturbed) condition (Westoby *et al.* 1989). These timeframes compare reasonably well with earlier work, as Roth (2004) presented results on infiltration indicating that cattle exclusion can lead to recovery of soil hydrological function in timeframes <15 years. However, Hawdon *et al.* (2008) and Bartley *et al.* (2014) found that although sediment yields may decline with improved cover, changes to hillslope and catchment percentage runoff can take >10 years. This again suggests that land condition represents more than just percentage ground cover, and these other attributes need to be included to fully capture land-condition change. Generally, although there are indications that RG can lead to faster recovery than does cattle exclusion (inferred from the comparison of R4 Treatment with R5 Treatment), this study does not enable us to conclude whether RG can accelerate recovery compared with other best-practice GLM approaches, as claimed by some practitioners of RG.

Linking land condition and grazing management

This study has demonstrated that well managed rotational planned grazing will likely yield better vegetation and soil condition outcomes than at sites that do not use periods of strategic rest as part of their grazing management. This finding is not that surprising, and the results would have been more powerful and meaningful if these sites had been compared to well managed continuous grazing systems that matched stocking rates to the carrying capacity of the land and regularly rested the country (cf. Hall *et al.* 2014). Because of time and budget constraints, this comparison was not possible; however, previous studies suggest that direct comparison among different grazing systems is all but impossible because of confounding variables (amount and seasonality of rainfall, soil variation, prior land use, livestock breeds etc.; Briske *et al.* 2008). Managerial variability is seldom recognised and documented, which makes it difficult to disentangle the influence of graziers' management skills from the ecological processes (Briske *et al.* 2008). Given time constraints, this study made a conscious decision not to focus on the specific management practices for each property, but instead to focus on the biophysical outcomes of collective change over time. This is because each landholder makes unique and often opportunistic decisions according to the unfolding wet season, and/or commercial imperatives and private aspirations that are not replicable over the 5–30 year time frame represented in this study. Although this study has shown that regenerative grazing leads to land-condition benefits, owing to the study constraints and a lack of comparable grazing management approaches, it is possible that well managed continuous grazing (which is not the same as set-stocking) could have yielded the same results. There is also the challenge of distinguishing between the infrastructure associated with a grazing management approach (e.g. fencing and watering points) (Hart *et al.* 1993) and the grazing management regime (e.g. stocking rate, rest etc.). Thus, we propose that future studies should undertake a more systemic approach and seek to attain specific socio-economic and grazing enterprise management data in conjunction with landscape attributes, to enable a more complete understanding of the interactions among the grazing management system, managerial skills, and whole of enterprise management.

The negative consequences of high stocking rates should not be interpreted as a condemnation of continuous grazing at appropriate stocking rates (Briske *et al.* 2008). However, given that animals do not graze uniformly over the landscape, but repeatedly consume preferred plants and patches of vegetation, multi-paddock grazing, that includes periods of strategic rest, can prevent or reverse rangeland degradation caused by area- and patch-selective overgrazing that develops within single paddocks that are stocked continuously (Teague *et al.* 2013). This is provided that the smaller paddocks or cells are not (re-)exposed to excessive grazing pressure that would negate the benefit of better spatial utilisation of forage. It is well accepted that longer-term rest and reduced stocking, especially during favourable conditions for plant growth, contribute to the sustainability and recovery of grazed systems (Briske *et al.* 2008). Although this study suggests that planned rotational grazing is one mechanism to achieving this, the literature suggests that there are various ways a landholder can reach this outcome (Abdel-Magid *et al.* 1987; Hart *et al.* 1993; Briske *et al.* 2008; Teague *et al.* 2013; Gosnell *et al.* 2020). The main point being that well planned grazing, regardless of the grazing system adopted, is the key to improved land condition.

The role of fire as an alternative management tool to stimulate and rejuvenate pastures was not explicitly tested in this study. The R6 treatment site had a very high fire frequency owing to military training activities and is not considered representative for the region. The R5 treatment exclosure site has not been burnt in the past ~20 years, and although the conditions at this site are better than at the R5 control site, the pasture at this site has largely gone moribund, which may be preventing any further improvements in pasture condition. Hunt *et al.* (2014) proposed that judicious use of fire is a key component of sound grazing management in northern Australia, and future comparisons of land condition should be made with sites that have had suitable fire frequency (Walsh *et al.* 2014).

Translating improvements in vegetation and soil to runoff and water quality for the GBR

There is evidence to suggest that coral reproduction in the GBR is vulnerable to both declining water quality and warming temperatures, with each stressor compounding the other (Humanes *et al.* 2017). Therefore, any efforts to improve land condition and offsite runoff and water quality from land adjacent to the GBR will have an added benefit of increasing the resilience of vulnerable marine systems to climate change and other disturbances (Wenger *et al.* 2016).

Despite field-derived percentage ground cover not being a strong differentiator for sites that demonstrated improved soil condition in this study (e.g. TN and SOC), the long-term trend in percentage cover as measured using remote-sensing did suggest a link between improved soil condition, and sites that maintained ground cover at or above the minimally disturbed DRCM reference benchmark levels for >10 years (e.g. treatment sites R4 and R5; see Fig. 3).

This study did not directly measure the off-site effects of improved land management on runoff and water quality, and explicit information on how improved vegetation and soil condition relate to improvements in soil health and soil hydrological function for large grazing enterprises is scarce. It is even rarer to find studies that translate the patch-scale data to catchment-scale responses, because these studies can take several decades (Ludwig *et al.* 2007; Koci *et al.* 2020). Continuous heavy stocking rates, without long periods of rest, are known to triple total runoff, peak runoff and suspended sediment loads in other catchments draining to the GBR (Thornton and Elledge 2021), although recovery timeframes once stocking rates and management have been improved are less certain. Data from a grazed hillslope adjacent to the R5 site showed that it eroded approximately three times more sediment (~2.30 t ha^{-1} year^{-1}) over a 5-year period than did a near-by hillslope that was not grazed (0.69 t ha^{-1} year^{-1}) (Hawdon *et al.* 2008). There was little difference between sediment yields prior to the removal of cattle, with erosion rates being 0.33 and 0.44 t ha^{-1} year^{-1} for the grazed and ungrazed sites respectively (Hawdon *et al.* 2008). Whereas Koci *et al.* (2020) provided evidence that at the catchment scale, cattle exclusion at the R6 site has led to improvements in hydrological function by reducing runoff and sediment discharge after about 15 years (noting that the first 7 years were all well below-average rainfall). Other studies have suggested that maintaining ground cover at or above ~70% is important for reducing runoff and excess erosion (Sanjari *et al.* 2009; Silburn *et al.* 2011). Maintenance of ~70% average cover is best partnered with little or no bare ground, and increased biomass, basal area and litter, which can be challenging to achieve in *B. pertusa*-dominated landscapes (Bartley *et al.* 2014).

It can take between 4 and 20 years to confidently detect the effects of improved agricultural management on water quality (Melland *et al.* 2018), and in semiarid rangeland environments, time periods of up to ~60 years may be needed to see changes in catchment runoff following a dramatic change in vegetation or severe soil degradation (Wilcox *et al.* 2008). It is likely that additional space-for-time studies combined with modelling approaches using local field data will be needed to test these assumptions and to supplement the much-needed long-term longitudinal studies of landscape repair.

Conclusions and areas of further research

We have demonstrated that using a time for space approach can help overcome the need to monitor landscape change over decadal timeframes. This study focused on the biophysical outcomes that could be achieved over time with elements of regenerative grazing, and demonstrated that some attributes respond quicker to improved land management (e.g. percentage ground cover); however, converting these changes into sustained and measurable improvements in vegetation, soil and land condition is challenging. It is likely going to take between 3 and 15 years for the key vegetation metrics to respond to changed grazing management, and in the order of 5–20 years to be able to confidently detect changes in soil condition. Improvements to overall land condition as it affects water quality leaving grazed catchments, may take between 5 and 20 years to measure with statistical confidence against the backdrop of a variable and changing climate.

This is the first paper, to our knowledge, that has proposed a framework and associated attributes to represent improved land condition for the purpose of improving off-site runoff and water quality. To improve our ability to detect changes in land condition at larger scales, more work is needed to enable the detection of changes in vegetation attributes such as biomass, by using remotely sensed products (e.g. Chen *et al.* 2021). Then, more innovative methods for benchmarking changes owing to management can be applied at property and regional scale (e.g. Donohue *et al.* 2022). It is noted that commercial methods to estimate pasture biomass have recently become available (e.g. https://www.cibolabs.com.au/), and are starting to be used in areas that have suitable calibration data.

This study focused on the biophysical outcomes from RG, yet for these approaches to be applied at scale, more information on the social and economic benefits of these approaches is needed. For example, de Villiers *et al.* (2014) found that farmers practicing Holistic Management™ had a higher social capital in that they participated more in groups, which is likely to lead to increased learning and adaptive behaviour. Thus, the farm-level benefits besides production, i.e. socio-ecological aspects, should be included in future research on production rangelands (Hawkins 2017). Landholders involved in this study also suggested that there are long-term financial gains to be made from these approaches, and a targeted economic analysis to quantify the gross marginal gains would be highly beneficial. It is likely that there are also long-term ecological benefits of these management systems at whole of enterprise levels, and additional data are needed to support this hypothesis. Each of these gains should also be put into context against the increased knowledge, skill, time and financial investments required to implement these regenerative grazing approaches across large rangeland enterprises (e.g. varying from ~5000 to 50 000 ha).

Supplementary material

Supplementary material is available online.

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Data availability. Data will be made available on the CSIRO Data Access Portal (https://data.csiro.au/).

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