

Long-term monitoring and modelling of pasture regeneration and water quality from a *Bothriochloa pertusa* site in the Great Barrier Reef catchments

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ABSTRACT

There has been considerable investment in grazing land management in Australia's Great Barrier Reef catchments targeted at improving off-site runoff and water quality. This study used a systems modelling framework to analyse an 18-year monitoring data set from an Indian couch (*Bothriochloa pertusa*) dominated hillslope flume site in the Burdekin catchment, Australia. The analyses show important functional differences in *B. pertusa* pastures, relative to the native tussock pastures they replaced, with implications for how they are monitored and managed: (1) rates of landscape recovery were far slower than rates of degradation. In the practical time frames of most land managers, transition to a *B. pertusa* state is effectively irreversible, so historical management practices will have to adapt; and (2) transformations in pasture composition and structure can lead to misinterpretation of monitored trends in ground cover (a common proxy for land condition). At 50% pasture cover, *B. pertusa* pastures have only 33% of the biomass of local tussock pastures. Where *B. pertusa* invasions are occurring, inferences from positive associations between changes in cover and landscape function can break down: increases in ground cover can instead be accompanied by decreases in biomass, productivity, and runoff retention. The paper contributes an initial calibration of the GRASP pasture grazing systems model for a transformed *B. pertusa* site, capturing the observed relationships between grazing management, pasture biomass, ground cover and water quality. This serves as a starting point from which these landscapes, of growing extent and concern in Great Barrier Reef catchments, can be better represented in monitoring and modelling assessments, and for revised management options to be explored. The calibrated model was able to accurately represent long-term average runoff and sediment yield (both within 2% of observed) but was only able to adequately represent the year-to-year variation in runoff (not sediment).

Keywords: GRASP, grazing management, ground cover, hillslope soil erosion, native pasture, rangeland monitoring, runoff, state and transition, systems modelling.

Introduction

Excess sediments and associated nutrients from agricultural areas are having a detrimental impact on Australia's Great Barrier Reef (GBR) (Furnas 2003; De'ath and Fabricius 2010; Humanes *et al.* 2017; Lambert *et al.* 2021), with sediment exports estimated to be at least five times greater than before European settlement (Mariotti *et al.* 2020). Livestock grazing is the dominant land use ($\approx 70\%$) in the catchments draining to the GBR (Lewis *et al.* 2021), and improvements in the amount and quality of ground cover and pasture biomass has been demonstrated to improve hillslope scale runoff and water quality (Roth 2004; Bartley *et al.* 2006; Silburn *et al.* 2011). However, while models and remotely sensed estimates of ground cover are now widely used to infer the long-term effects of improved grazing land management on water quality in GBR catchments (Ghahramani *et al.* 2020; McCloskey *et al.* 2021), there remains a gap in published data on these links, and the effectiveness of models in representing them.

One data gap of particular concern is the implications of the widespread invasion of the exotic stoloniferous grass *Bothriochloa pertusa* ('Indian couch'), which is displacing native tussock grass pastures in many parts of the GBR catchments, fundamentally transforming their ecological function and responses to grazing (Kutt and Fisher 2011; Kutt and Kemp 2012). 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 coincides with areas prone to sediment loss (McKeon *et al.* 2004). *B. pertusa* now covers large areas of grazing land (Lebbink 2021; Lebbink *et al.* 2022; Lebbink and Fensham 2023) (Fig. 1), but the implications of this physiognomic transformation for water quality are not clear. (Botanical names throughout this paper follow the Australian Plant Names Index (APNI 2022)).

Various grazing land management practices have been used in efforts to reverse this change, including approaches that focus on reducing stocking rates to achieve appropriate levels of pasture utilisation and wet season resting (Ash *et al.* 2011; Hall *et al.* 2014; Hunt *et al.* 2014), and rotational grazing systems that often involve grazing areas for shorter durations at higher intensity (Briske *et al.* 2011; Teague *et al.* 2013; Gosnell *et al.* 2020). Regardless of the method employed, the aim is to restore previous levels of pasture cover, biomass, and land condition, with the perception that

this will also reduce runoff and improve off-site water quality. Because semi-arid systems can take 10–20 years or more to respond to improved management (O'Reagain *et al.* 2011; Bartley *et al.* 2014; Koci *et al.* 2020) and these changes are costly to measure, paddock and catchment scale hydrological models are frequently used to estimate and infer the long term effects of management changes on future offsite water quality (Carroll *et al.* 2012). Land condition is defined here as the capacity of a landscape to retain water, produce biomass, cycle nutrients and support both biodiversity and livestock production.

Few hydrological data sets are available for northern Australian grazing lands to test the links between metrics of improved land condition, runoff, and water quality. Where suitable runoff data exist, most studies were conducted under controlled conditions involving small enclosures or complete cattle removal (McIvor *et al.* 1995; Scanlan *et al.* 1996; Silburn *et al.* 2011), and very few studies have been conducted under commercial grazing conditions. Moreover, while hundreds of calibration data sets have been collected to parameterise pasture components of models such as GRASP for tussock pastures (Day *et al.* 1997; McKeon *et al.* 2010), there are fewer calibrated runoff responses, and there are no readily available models that have been parameterised for stoloniferous lawn-forming pastures like *B. pertusa*.

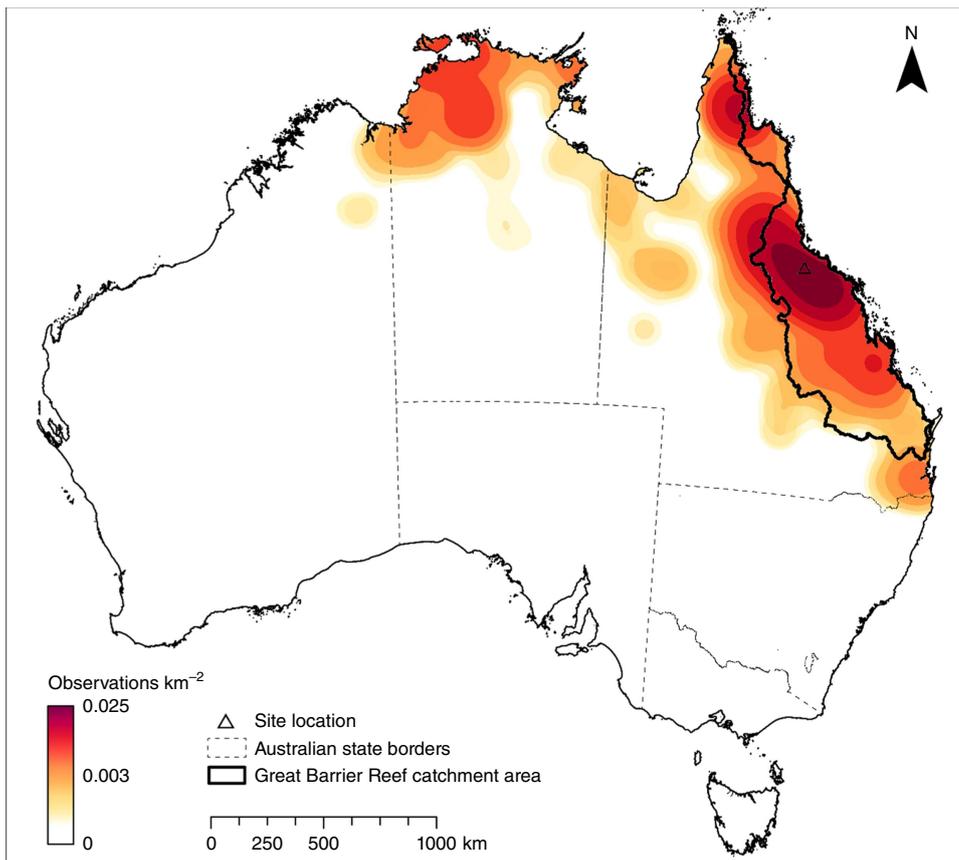


Fig. 1. Map of *Bothriochloa pertusa* distribution, in relation to Great Barrier Reef catchments, based on herbarium, observation and survey records. Source data: Karfs *et al.* 2009, ALA (2022), AVH (2022).

There is therefore a need for data that can quantify the links between grazing management, pasture responses, land condition, runoff, and water quality in GBR catchments, particularly for expanding *B. pertusa* landscapes. This paper aims to address this need in two ways. First, it presents 18 years of flume and pasture measurements from monitoring a *B. pertusa* dominated hillslope located on a commercial grazing property in northeast Australia. Secondly, the GRASP grazing systems model was used to both: provide a rigorous systems framework for quantifying and interpreting trends in land condition and water quality over the course of the study; and test the suitability of the calibrated model to support future work in better representing the influence of *B. pertusa* in GBR catchments. GRASP is commonly used as a pasture growth and rangeland systems tool (Rickert and McKeon 1988; Owens et al. 2003) to simulate the effects of weather and management on pasture growth, livestock production, land condition, runoff, and erosion. Demonstrating the veracity of the GRASP model components in a site dominated by *B. pertusa*, will enhance the use of GRASP and related models in assessing water quality outcomes in GBR catchments.

Methods

Study site

This study was carried out on a ‘typical’ hillslope within Virginia Park station, a commercial cattle grazing property in the Burdekin catchment, Queensland, Australia (-19.89° , 146.54°) (Fig. 1). The main drainage system on the property is Weany Creek. The area is representative of the highly erodible granodiorite (‘gold-fields’) country between the cities of Townsville and Charters Towers. The site is a 1.2 ha hillslope within a larger 625 ha paddock, and was chosen because it has been studied and characterised in detail from research spanning more than 30 years (Gardener et al. 1990; Leuning et al. 2005; Bartley et al. 2007; Koci et al. 2020, 2021). The key landscape and hillslope attributes are summarised in Table 1.

The dominant land use has been beef cattle grazing for over 100 years. Prior to 1988, pastures at Virginia Park were dominated by tall native perennial grasses, *Heteropogon contortus*, *Dichanthium fecundum*, and native legumes, with the upper story consisting primarily of narrow-leaved ironbark (*Eucalyptus crebra*) and red bloodwood (*Corymbia erythrophloia*). However, heavy grazing by *Bos indicus* cattle between 1979 and 1988 resulted in a significant change in the pasture composition, where the dominant species shifted to *B. pertusa*, annual grasses, non-legume dicots, and legumes (Gardener et al. 1990).

This study covered the period from 2003 to 2021. At the beginning of the project, during a drought, the ground cover and pasture biomass levels were on average 63% and 350 kg of dry matter per hectare (kg DM ha^{-1}), respectively, well below that considered ‘sustainable’ in terms of long term

Table 1. Key attributes of the flume hillslope site in the Weany Creek catchment on Virginia Park Station.

Attribute	Description or value
Geology	Granodiorite
Soil type	Chromosol, sodosol
Land type, bioregion	Einaleigh Uplands
Vegetation – overstory	Dominated by Ironbark, bloodwood communities (Queensland Regional Ecosystem 9.12.1), tree basal area $5.1 \text{ m}^2 \text{ ha}^{-1}$
Vegetation – grasses	Dominated by <i>B. pertusa</i>
Long term average rainfall	634 mm
Slope gradient	3.9%
Slope length	250 m
Hillslope area	11 930 m^2

grazing management for this soil type (Ash et al. 2001). Prior to the study (1986–2002), the stocking rate was about 0.25 AE ha^{-1} , whereas in the first 10 years of the study stocking rates averaged about 0.1 AE ha^{-1} , where one animal equivalent (AE) is measured relative to a 450 kg steer at maintenance (McLean and Blakeley 2014). Previous publications documented the data outputs from this site after 6 and 10 years, respectively (Bartley et al. 2010, 2014). After the formal research project finished, stocking rates varied between years, approximated as light stocking rates from 2013 to 2016 (0.1 AE ha^{-1} , except for complete destocking in 2014) and heaving stocking rates in the five remaining years (about 0.5 AE ha^{-1} in 2012–2013 and 2017–2020).

Vegetation monitoring

Land condition on the flume hillslope was measured using a range of techniques over the 18-year measurement period (Table 2). Measurements were made at the end of the wet season and end of the dry season each year. Between 2003 and 2016, land condition was measured across the hillslope on an $8 \times 4 \text{ m}$ grid. Between 2020 and 2021, ten 100-m transects were each sampled with ten equidistant $1 \times 1 \text{ m}$ quadrats. Sampling followed the BOTANAL calibrated visual estimation method of Tohill et al. (1992). Measurements included pasture composition (by species and functional group), pasture standing biomass, percentage pasture cover (associated with standing pasture biomass), percentage ground cover (predominantly pasture cover plus a small ($\approx 5\%$) area of non-overlapping litter cover) and grass basal area. Between 2016 and 2020 field data were collected from an adjacent hillslope in the same paddock (within 500 m, sharing the same land management and comparable MODIS ground cover). Calibrated MODIS pasture and ground cover estimates (Guerschman 2020) were used when field measurements were not conducted.

Table 2. Method and timing of vegetation surveys undertaken on the flume hillslope over the 18-year period.

Method	Time frame applied
End of dry season surveys	2003–2016; 2020–2021
End of wet season surveys	2003–2007; 2020–2021
Calibrated MODIS fractional cover components	2001–2002; 2008–2016
End of dry and end of wet surveys from adjacent hillslope	2016–2019

Remote sensing of ground cover

MODIS (Moderate Resolution Imaging Spectroradiometer) satellite data were calibrated against field measurements (Table 2) to derive estimates of pasture cover and ground cover ($R^2 = 0.88$). High resolution (≤ 10 m) satellite images (Quickbird, Worldview and Sentinel-2) were analysed and calibrated using the PD54 cover method (Pickup *et al.* 1993) and used to map the distribution of ground cover for 7 years (2003–2007, 2009–2010).

Rainfall corrected trends in ground cover were derived from Landsat images using the dynamic reference cover method (DRCM) outlined in Bastin *et al.* (2012). This approach identified minimally disturbed, grazed, non-riparian reference areas in Landsat images to benchmark changes in cover through time. There may be systematic biases in the Landsat estimates of ground cover, particularly in areas dominated by *B. pertusa* (Bastin *et al.* 1996; Wilkinson *et al.* 2014a, 2014b; Bartley *et al.* 2022).

Rainfall, runoff and sediment yield monitoring

Most rain falls between December and April, but occasional out of season events occur, and therefore rainfall and runoff data extend from July 1 to June 30 (used here to define the ‘water year’, each referenced by its starting year).

To measure water and sediment runoff, a large Parshall flume was installed at the bottom of the hillslope. Details of the data logger, associated instrumentation and data processing can be found in Bartley *et al.* (2006). Annual percentage runoff was calculated as the total annual runoff divided by total rainfall for that water year. Water sampling was stratified by flow depth, and collected samples were used to represent total suspended solids (TSS) concentrations between 2002 and 2013. In the rare case that samples were not collected for an event, an average TSS concentration from the wet season was applied. In 2014 a turbidity sensor was installed in the flume, and a turbidity-TSS relationship ($R^2 = 0.91$) was developed, which allowed continuous sediment loads to be derived when runoff occurred.

Collected water samples were analysed for TSS concentration (considered to represent the silt (0.002–0.063 mm) and clay (< 0.002 mm) sediment fractions). Bedload samples (generally between 0.063 and 8 mm) were collected

manually from bedload traps but these represent $< 1\%$ of total load (Bartley *et al.* 2006), and the data were not used in calculating fine sediment loads. When both concentration and discharge data were available, annual sediment loads were estimated by summing the event loads using the arithmetic mean approach (Letcher *et al.* 1999). When sediment concentration data were unavailable for an entire event, average values for that wet season were applied.

Modelling approach using GRASP

This study used GRASP (Cedar version 2.1.0), a deterministic, point-based, daily time step, native/naturalised pasture grazing systems model, developed to support research and decision making in semi-arid and tropical grasslands (Rickert and McKeon 1988; McKeon *et al.* 1990; Owens *et al.* 2003). The model and parameterisation of its key functions are described by Littleboy and McKeon (1997), with components of the model covering water balance, pasture dynamics, grazing management, runoff and soil loss. GRASP focuses on capturing and representing the gross long-term ‘average’ system behaviours that are of most relevance to property- and catchment-scale management decisions.

Data on the site soil profile (Queensland Government 2022) were used to directly estimate GRASP parameters related to rooting layer depths and water-holding characteristics of each layer (Table 3). Previous pasture measurements were used to directly calculate important parameters related to pasture growth, such as sward nitrogen, maximum pasture growth, and the relationship between pasture cover and biomass. Direct measurements for some other key parameters related to leaf senescence and defoliation were unavailable, and were instead calibrated by selecting pasture parameters that gave a good fit between modelled and observed pasture biomass. The model used daily rainfall measured at the flume site augmented with Charters Towers gap-filled historical weather station data from the SILO online repository (Jeffrey *et al.* 2001) (Table 3). Model parameters were set to represent the broad shifts in stocking rates described previously in the site description: the intention was not to mimic the full detail of month-to-month changes in stocking rate, but to capture periods of major differences in management. The model was reset to observed pasture states wherever observations were available, which had the effect of limiting the cumulative propagation of small errors that could cause progressive drift of the pasture component of the model from observations over 20 years.

GRASP contains several different sub-model options for representing runoff. Of these, the ‘curve number’ sub-model was used because it provided the best fit in a previous analysis of the first 10 years of Virginia Park data, and because it allowed explicit testing of whether ground cover affected runoff. Curve numbers range between 30 (maximum infiltration) and 100 (an impermeable surface with maximum runoff). The modified implementation of

Table 3. Data sources and attributes used in the GRASP model.

Attribute	Frequency	Units	Data source
Data inputs			
Soil layer depth	Layers to 1.1 m	cm	Local field measurements
Air dry moisture	Layers to 1.1 m	mm mm ⁻¹	Local field measurements
Wilting point	Layers to 1.1 m	mm mm ⁻¹	Local field measurements
Field capacity	Layers to 1.1 m	mm mm ⁻¹	Local field measurements
Maximum sward nitrogen	Early wet season	kg ha ⁻¹	Local field measurements
Rainfall	Daily	mm	Local weather station (or SILO)
Max temperature	Daily	°C	Local weather station (or SILO)
Min temperature	Daily	°C	Local weather station (or SILO)
Vapour pressure deficit	Daily	hPa	SILO (Charters Towers)
Pan evaporation	Daily	mm	SILO (Charters Towers)
Solar radiation	Daily	MJ m ⁻²	SILO (Charters Towers)
Time series observations (for calibrating matched data outputs)			
Pasture biomass	Twice yearly	kg ha ⁻¹	Local field measurements
Pasture cover	Twice yearly	m ² m ⁻²	Local field measurements + remote sensing
Ground cover	Twice yearly	m ² m ⁻²	Local field measurements + remote sensing
Runoff	Daily (annual totals)	mm	Flume monitoring site
Total suspended sediment	Daily (annual totals)	t ha ⁻¹	Flume monitoring site

the curve number model in GRASP (Owens *et al.* 2003) allows specifying different curve numbers for 0 and 100% ground cover (and then interpolating a curve number for intervening levels of cover). Calibration involved running factorials of these two parameters with progressively finer resolution picking the combination that gave the highest Pearson correlation between modelled and observed annual runoff until the deviation of the modelled average was within 2% of the observed value for the study period.

The sub-model for fine sediment yield was parameterised similarly to runoff, using the Rose sub-model option (Rose 1985), which has parameters for slope angle, sensitivity of soil loss to ground cover, and bare soil efficiency of entrainment.

Results

Changes in pasture composition

Although originally dominated by tussock grasses (with no *B. pertusa*), the site had been transformed by invasion of *B. pertusa* by 2002 when the flume was installed and a period of reduced grazing pressure implemented (Fig. 2). Such *B. pertusa* transformations are characteristic of heavy grazing over long periods, and are typical of State 3 and 4 grazing systems within state and transition models for this region (McIvor and Scanlan 1994; Ash *et al.* 2011). While the *B. pertusa* invasion was rapid (replacing most tussock

grasses within 9 years), the altered state has proven very resistant to improvements in grazing management, with only slow and limited recovery of native tussock grasses over the subsequent 20 years (Fig. 2).

Changes in ground cover

Ground cover at the start of the wet season is particularly important for water quality outcomes, as is the spatial arrangement of the vegetation, where cover lower on the hillslope is more important for trapping sediment (Bartley *et al.* 2010). Both the amount and spatial arrangement of end of dry season ground cover varied during the study (Fig. 3).

The weather-corrected DCRM data show a slight trend of increasing ground cover over time (not statistically significant, $P = 0.134$) (Fig. 4). Increases in ground cover occur not only from management practices, but also from changes in pasture composition (as discussed in the next section).

Modelled vs observed pasture biomass and cover

GRASP modelling reflected the observed fluctuations in pasture biomass at the study site over the duration of the investigation, however, forced resets to observations were required to correct for the gradual accumulation of errors (Fig. 5). Ultimately, the pasture component of the modelling was mainly used to interpolate the vegetation dynamics between observations (Fig. 5) to provide the daily ground

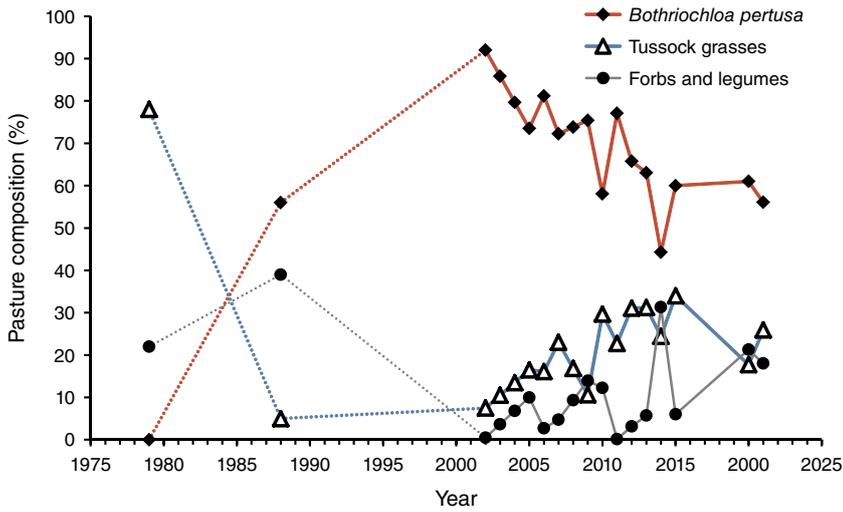


Fig. 2. Changes in relative species composition of the pasture layer at the study site measured at the end of the dry season (as percentages of total biomass). Data points for 1979 and 1988 are from previous studies at the site, predating the installation of the flume (Gardener et al. 1990).

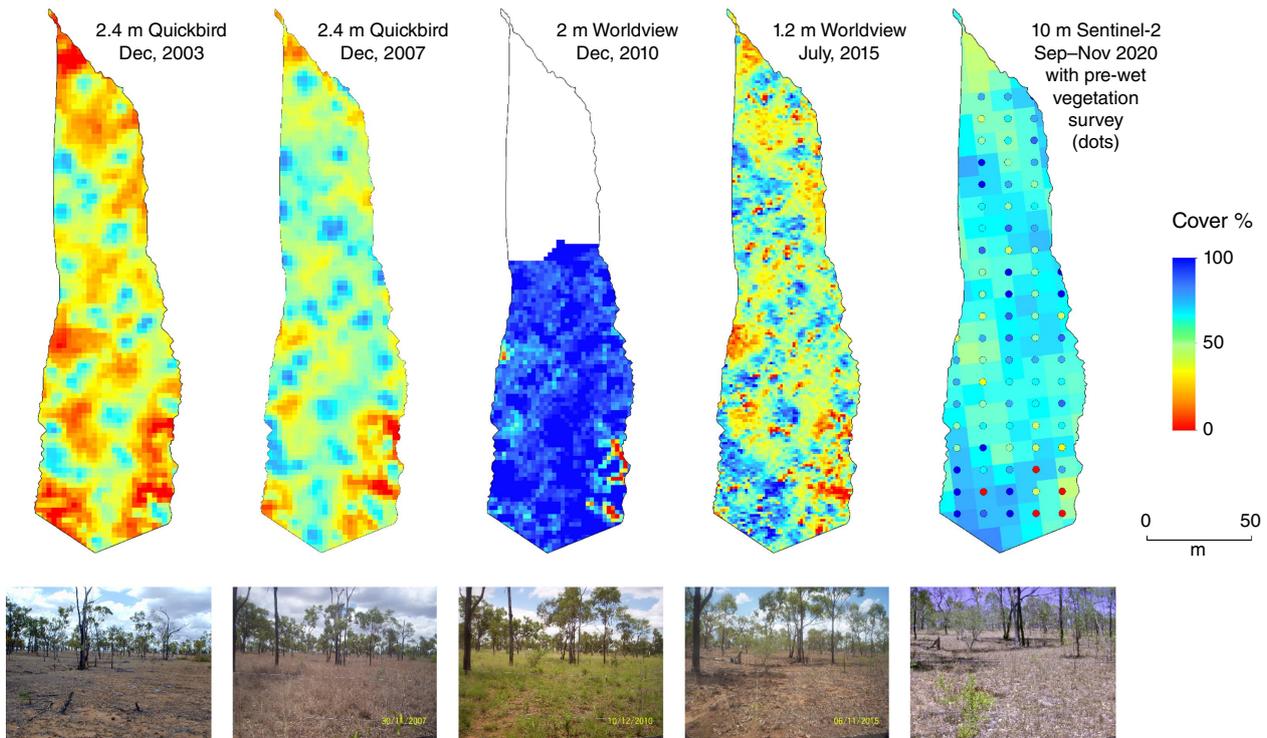


Fig. 3. Remotely sensed ground cover on the hillslope flume site in 2003, 2007, 2010, 2015 and 2020. The missing area in 2010 was due to cloud interference.

cover data required by the runoff and soil loss components of GRASP.

Pasture cover has a strongly asymptotic relationship to pasture biomass. The structure of pasture vegetation at the site was very different from typical tussock pastures in northern Australia because of the creeping, stoloniferous growth form of the dominant *B. pertusa*. *B. pertusa* has much less biomass for the same amount of pasture cover relative to a tussock grass (Fig. 6, where, at 50% pasture cover, the *B. pertusa* pasture had only a third of the biomass

expected for tussock pastures in the region). This structural change in vegetation has important implications for how satellite and field-based measures of pasture are interpreted, as demonstrated in Fig. 7, showing that pasture biomass would be greatly over-estimated if it were assumed to be a typical tussock pasture rather than calibrated for *B. pertusa*. Although pasture cover is easier to measure, biomass is a better indicator of functional land condition both in terms of water quality (the amount of pasture material that obstructs overland flow of water) (Roth 2004) and productivity (the

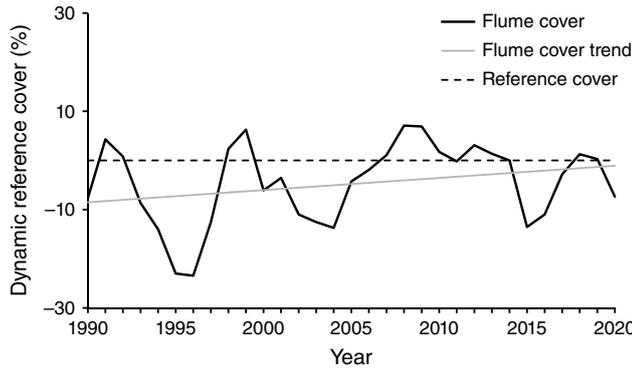


Fig. 4. Weather-corrected changes in end of dry season ground cover for the flume site (1990–2020). Data are for Landsat-derived ground cover processed using the dynamic reference cover method.

amount of forage and protein-building nitrogen available for livestock to consume). Inferences from pasture cover alone in *B. pertusa* dominated landscapes would therefore tend to give overoptimistic estimates about land condition and trend.

Hillslope rainfall, runoff, sediment concentrations and sediment yields

The long-term average measured rainfall for the site was 634 mm year⁻¹. The lowest annual rainfall over the study period was 245 mm (2003), the highest 1308 mm (2008) (shown with coincident ground cover data in Fig. 8a). The measurements of runoff (Fig. 8b) and total suspended sediment concentrations (Fig. 8c) over the same period combine to generate fine sediment loads (Fig. 8d).

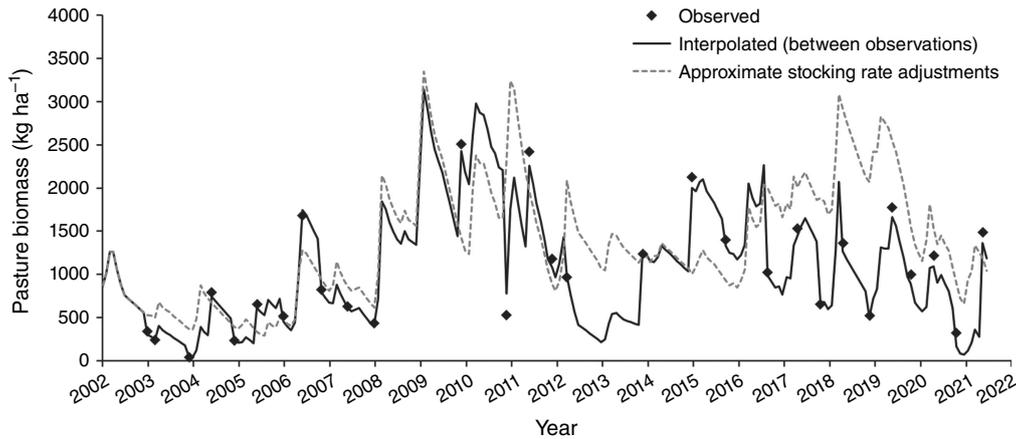


Fig. 5. Fluctuations in pasture biomass over the duration of the investigation at the Virginia Park flume site. The dashed line is the unconstrained estimate from the GRASP model with approximated adjustments to stocking rates to represent periods of substantial differences in management. Field observations are marked by diamonds, and the solid line is where the model was used to interpolate between observations.

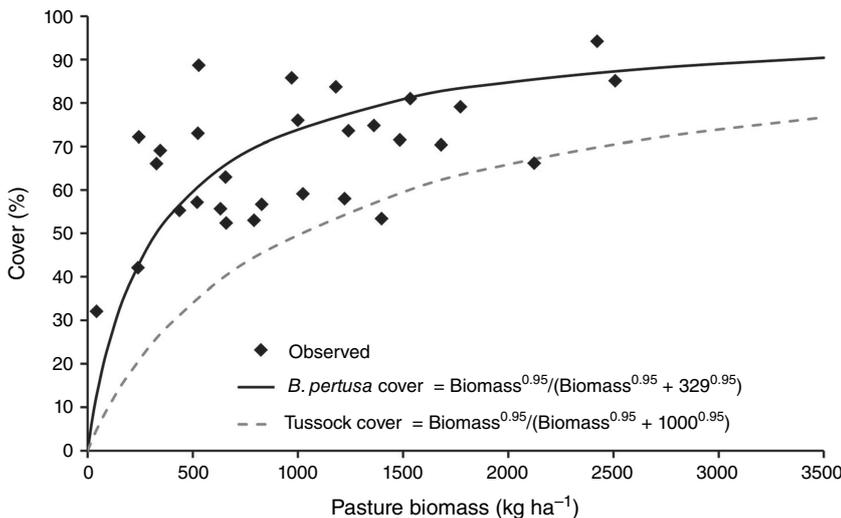


Fig. 6. Contrasting relationships between pasture cover and biomass for pastures dominated by *Bothriochloa pertusa* (diamonds: end of dry season observations at Virginia Park; solid line: calibrated curve in the form of the equation used in GRASP) and typical tussock grasses in the region (dashed line: parameterised curve for tussock grasses in the region).

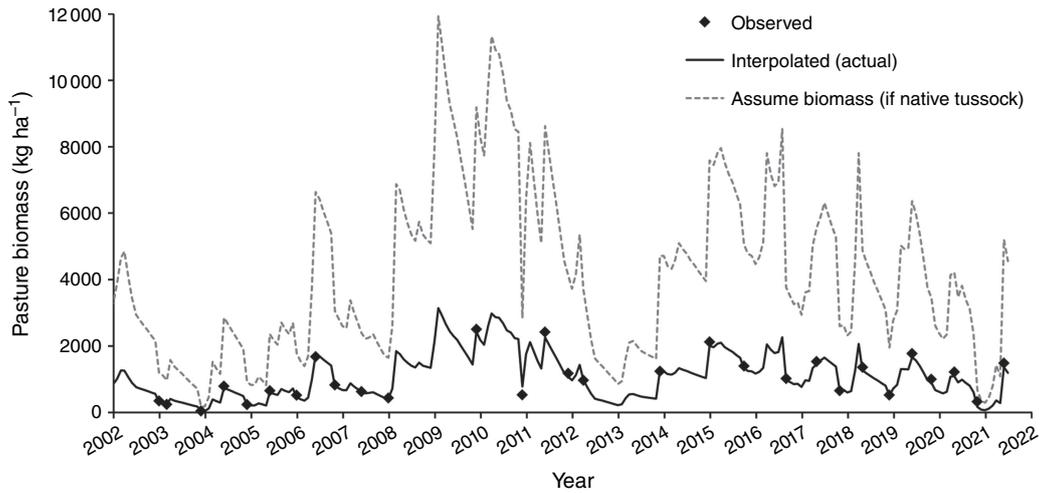


Fig. 7. Fluctuations in biomass for *Bothriochloa pertusa* pastures at the study site for observed data (diamonds) with modelled interpolations (black line) compared to the biomass that would be implied from observed pasture cover alone if pastures were assumed to have the typical structure of tussock grasses (dashed line: calculated using the inverse relationship from Fig. 6 for pasture cover-vs-biomass in tussock pastures).

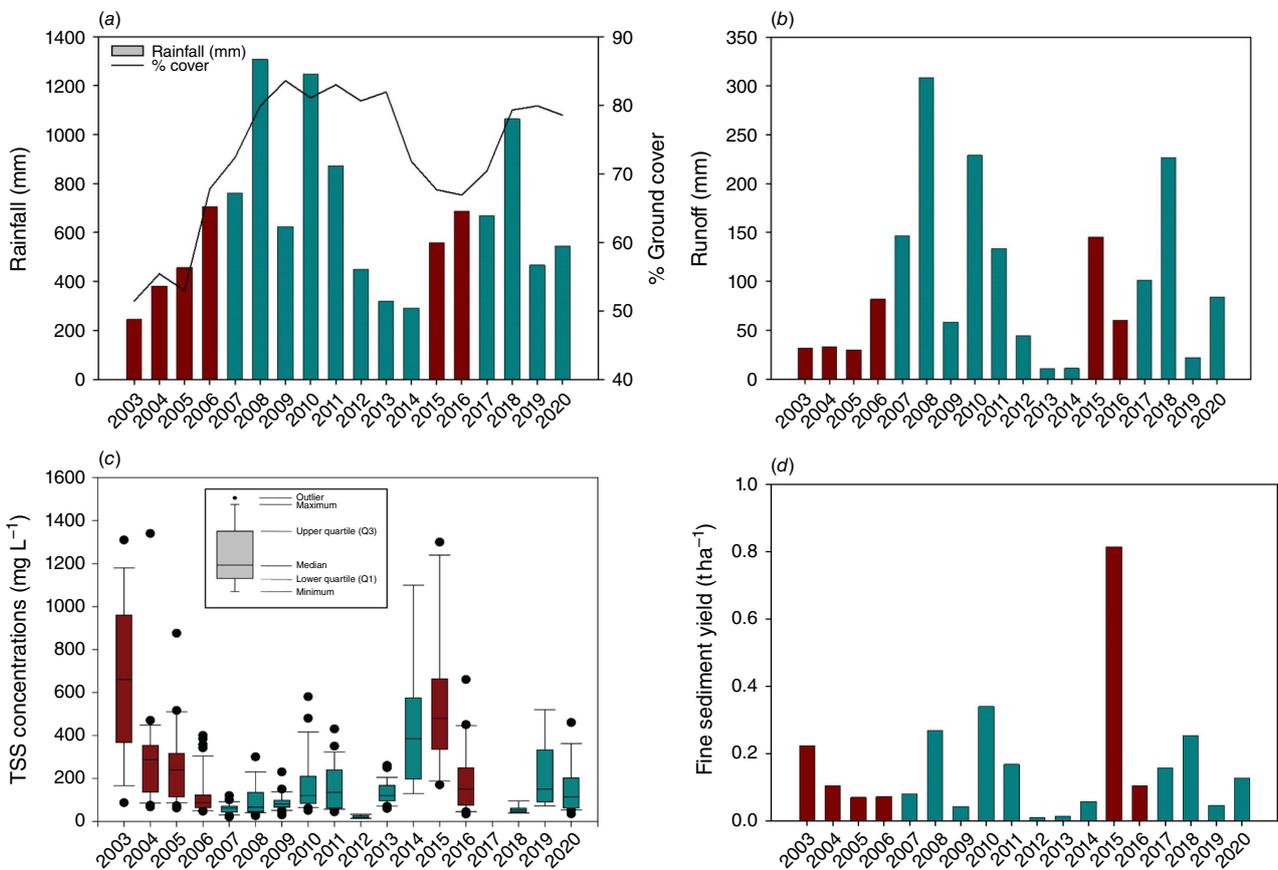


Fig. 8. (a) Annual rainfall and end of dry season ground cover (%) (calibrated MODIS estimate); (b) annual runoff (mm); (c) TSS concentrations (mg L^{-1}); and (d) fine sediment yield (t ha^{-1}) for the flume site from 2003 to 2020. Red bars/plots represent years when cover $<70\%$ and blue bars/plots when cover $>70\%$. All years are numbered as the starting year of that ‘water year’ (July–June). Box plots for (c) show upper quartile, median and lower quartile; whisker lines show maximum and minimum; and points show outliers.

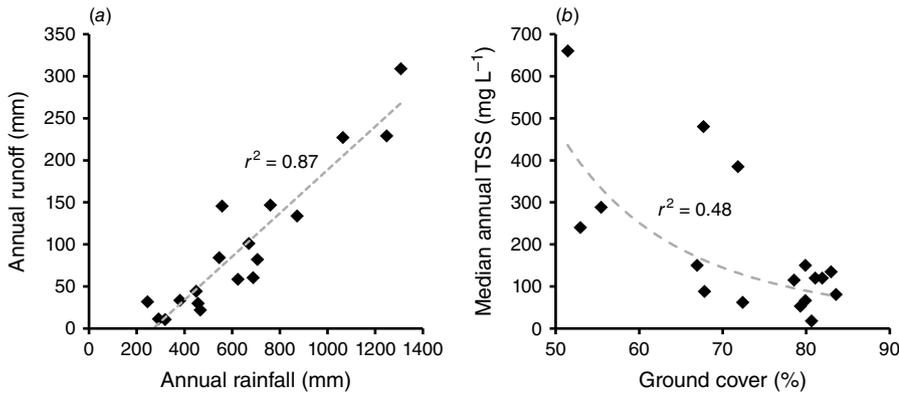


Fig. 9. (a) Linear relationship between annual rainfall (mm) and runoff (mm) for the 18-year record at the flume site; and (b) power curve relationship between end of dry season ground cover (calibrated MODIS estimate) and the median annual measured TSS concentration. Outliers in (b) are associated with large rainfall (high intensity) events.

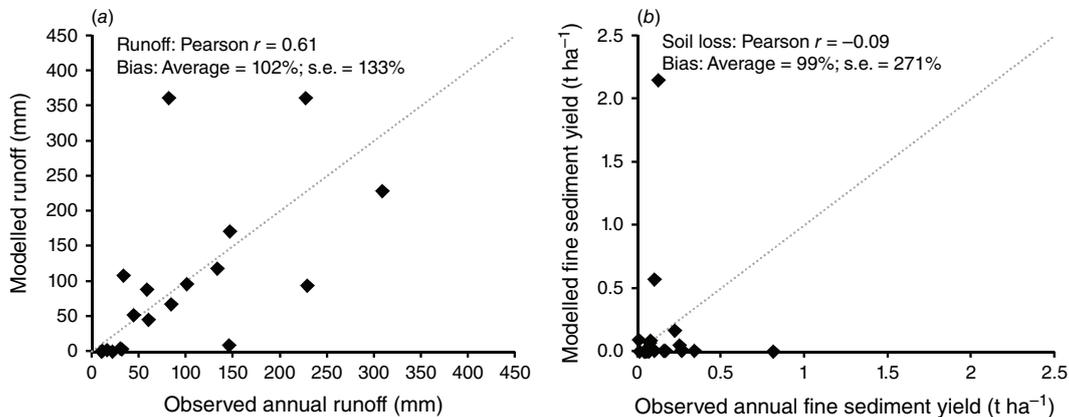


Fig. 10. Calibrations of the (a) runoff and (b) soil-loss sub-models in GRASP for annual observations vs modelled estimates. These calibrations are for year-to-year variations at a single site. The dotted line is the 1:1 relationship. Averages for annual runoff were 93.5 ± 20.1 (observed) and 95.1 ± 26.7 mm (modelled). Average annual sediment yields were 0.169 ± 0.043 (observed) and 0.168 ± 0.117 t ha⁻¹ (modelled). Annotated biases express the modelled averages; and standard errors (s.e.) as percentages of the observed values.

The data provide support for a previously documented functional water quality target for end of dry season ground cover of $\approx 70\%$ (Sanjari et al. 2009; Bartley et al. 2014): summarising the data in Fig. 8 as the averages for years when ground cover was below the 70% threshold vs averages when the 70% threshold was exceeded showed that despite rainfall (506 mm vs 718 mm) and runoff (64 mm vs 115 mm) being substantially higher in years with $>70\%$ cover, total sediment yield (0.23 t ha⁻¹ vs 0.13 t ha⁻¹) and TSS concentration (318 mg L⁻¹ vs 119 mg L⁻¹) were reduced. However, in the 2015/2016 wet season, a single event produced 90% of the annual hillslope sediment loss when 101 mm of rain fell in an hour on the 1st of March 2016. The 70% ground cover target for this site is higher than the 40% target recommended in the region for sites with a larger proportion of tussock grasses (Ash et al. 2001).

There was a strong linear correlation between the inter-annual variations in runoff and rainfall ($R^2 = 0.87$) (Fig. 9a), and a moderate relationship between the inter-annual variations in percent runoff and fine sediment yield ($R^2 = 0.47$). Using multiple linear regression, including

rainfall and ground cover, the addition of ground cover improved the prediction of annual hillslope runoff marginally ($R^2 = 0.90$).

Despite a moderate relationship between median TSS concentration and ground cover ($R^2 = 0.48$) (Fig. 9b), the average annual ground cover alone was not particularly useful at predicting year-to-year variation in sediment yield (t ha⁻¹) ($R^2 < 0.2$) due to the dominant influence of weather and interactions between the variables within and between years.

Modelled runoff and sediment

The calibrated GRASP curve number sub-model for runoff matched both the long-term observed average annual runoff and provided a reasonable approximation of year-to-year variations (Pearson $r = 0.61$) (Fig. 10a). The curve numbers that gave the best fit were 39.8 for 100% cover and 82.2 for 0% cover. This demonstrates that, despite its relatively low biomass, there was still a strong relationship between temporal changes in ground cover and runoff for *B. pertusa*

pastures. The calibrated GRASP soil-loss sub-model matched the long-term average annual fine sediment yield but did not capture the fine-scale detailed processes that give rise to annual variations in soil loss ($r = -0.09$) (Fig. 10b).

Discussion

Land condition and water quality trends

Rainfall, and associated runoff, was the dominant, natural perturbation driving year-to-year variation in sediment loads (Fig. 9). A combination of rainfall amount and the number of large ($> 50 \text{ mm h}^{-1}$) rainfall events accounts for most of the short-term fluctuations in erosion in these systems (Dunkerley 2021), while the modifying influence of grazing management on runoff is more difficult to detect. However, the systems modelling analysis, by accounting for the complex landscape interactions, detected the influence of ground cover on runoff, as evidenced by the large differences in curve number between 0 and 100% cover. While ground cover has a relatively minor influence on the proportion of rainfall that becomes runoff at the annual time scale, it also affects suspended sediment (TSS) concentration, and thus has been shown to be an important parameter for predicting sediment loads at both hillslope (Tiwari *et al.* 2020) and larger catchment scales (Kuhnert *et al.* 2012). In fact, Kuhnert *et al.* (2012) found that TSS concentrations at the Burdekin catchment scale (end of system, Inkerman Bridge) decreased 2.1% for each 1% increase in ground cover, e.g. if ground cover improves by $\approx 10\%$, then TSS concentration reduces by $\approx 20\%$. Data presented here provide further evidence that ground cover does influence sediment concentrations, particularly when cover falls below 70%. In very wet years, high intensity rainfall events become more influential as the wet season progresses and can generate large amounts of runoff, regardless of the amount of cover, as the soil profile is saturated (Bartley *et al.* 2006; Jarihani *et al.* 2017). Cover is likely to exert a stronger influence on sediment loads in lower rainfall years and at the start of the wet season, or when comparing nearby properties that experience similar weather but have contrasting management.

Other plant attributes such as pasture biomass and plant basal area are likely better predictors of infiltration, and thus runoff, particularly in *B. pertusa* dominated rangelands (Roth 2004). There has been some recent progress in estimating biomass using remote sensing (Chen *et al.* 2021), however, the methods for hindcasting ground cover data using these methods and linking to historical runoff data are limited, and current approaches cannot adequately differentiate between pasture species (e.g. tussock vs stoloniferous grasses). In this regard, the data show important differences in the pasture cover–biomass relationships for pastures dominated by *B. pertusa* vs tussock grasses (Fig. 6),

where *B. pertusa* pastures have far less biomass at the same cover than the pastures they replace. This would lead to overoptimistic assessments of land condition and trend (including the inferred sediment yields for a given observed ground cover) if changes in pasture composition and physiognomy are not considered. As *B. pertusa* expands in many grazed parts of the GBR catchments, traditional cover-based approaches to monitoring the influence of grazing land management on land condition and water quality could be compromised.

Long-term monitoring revealed several other insights into the altered functioning of transformed *B. pertusa* landscapes, with implications for management. Data show strong asymmetry in the rates of landscape degradation and recovery, where tussock grass regeneration rates have been far slower than their displacement rate by *B. pertusa* (Fig. 2). At best, there are long time lags between improved land management practices and water quality outcomes for the GBR; at worst, *B. pertusa* lawns are very resistant to change, and represent a transition to a relatively stable new pasture state (McIvor and Scanlan 1994; Ash *et al.* 2011) that needs to be managed differently to the native tussock pastures it replaced. The asymmetry in recovery vs degradation also has important implications for managing risks of adjusting stocking rates relative to (uncertain) future pasture growth, since any degradation from setting a stocking rate adjustment too high would require multiple years of intervention (with reduced stocking) for recovery. Conservative stocking rate strategies are therefore particularly favoured in land types with such asymmetric risks associated with irreversible transitions.

Comparison with other rangeland areas in the GBR catchments

The 18-year annual average fine sediment yield was $\approx 0.16 \text{ t ha}^{-1} \text{ year}^{-1}$, which is summarised alongside other hillslope fine sediment yields measured at grazing sites in the GBR catchments in Table 4. Given the variability in slope, rainfall, vegetation cover, grazing intensities and size of the hillslopes, it is not necessarily appropriate to compare these data, as factors such as plot area are known to affect hillslope sediment yield calculations (Ludwig *et al.* 2007). However, this summary does highlight how variable measured hillslope sediment yields can be between sites and years.

Erosion rates derived using cosmogenic beryllium (^{10}Be isotope) suggest that the pre-grazing baseline representing the long-term average erosion rate (≈ 100 to $> 10\,000$ years) for this site is likely to be $\approx 0.27 \text{ t ha}^{-1} \text{ year}^{-1}$ (Bartley *et al.* 2015). Therefore, the contemporary ≈ 18 -year average erosion rate from this flume hillslope site is within the range of tolerable or acceptable erosion rates (Montgomery 2007; Bui *et al.* 2011). However, data presented in Table 4 demonstrate that many hillslopes in the GBR catchments, including adjacent hillslopes on this property, have average hillslope soil erosion rates several times higher than this baseline value,

Table 4. Summary of measured hillslope sediment yields using natural rainfall from grazed sites in Queensland, Australia.

Catchment	Location	Length of data set (years)	Approximate grazing intensity	Fine sediment yield ($\text{t ha}^{-1} \text{ year}^{-1}$)	Reference
Burdekin	Upper Burdekin	≈18	Range ≈0.08 beasts ha^{-1} to ≈0.25 beasts ha^{-1}	0.16 $\text{t ha}^{-1} \text{ year}^{-1}$ Range 0.009–0.8 $\text{t ha}^{-1} \text{ year}^{-1}$	This study
Burdekin	Upper Burdekin	≈10	Range ≈0.08 beasts ha^{-1} to ≈0.25 beasts ha^{-1}	Range 0.004–≈4.0 $\text{t ha}^{-1} \text{ year}^{-1}$ (hillslopes with scald/ bare areas)	Bartley et al. (2014)
Burdekin	Upper Burdekin	≈7	Range ≈0.25 beasts ha^{-1} to ≈0.4 beasts ha^{-1}	1.32 $\text{t ha}^{-1} \text{ year}^{-1}$ Range 0.33–2.30 $\text{t ha}^{-1} \text{ year}^{-1}$	Hawdon et al. (2008)
Burdekin	Upper Burdekin	≈3	Low 0.1 beasts ha^{-1} to High 0.5 beasts ha^{-1}	1.08 $\text{t ha}^{-1} \text{ year}^{-1}$ Range 0.12–1.96 $\text{t ha}^{-1} \text{ year}^{-1}$	McIvor et al. (1995)
Burdekin	Upper Burdekin	≈6	No grazing + grazing, no stocking rates provided	Range ≈0.00–≈1.00 $\text{t ha}^{-1} \text{ year}^{-1}$ (total load, not fine sediment load)	Scanlan et al. (1996)
Fitzroy	Dawson	≈25	Managed at industry recommended stocking rate with utilisation of no less than 1000 kg ha^{-1} of pasture available at any time	0.12 $\text{t ha}^{-1} \text{ year}^{-1}$	Elledge and Thornton (2017)
Fitzroy	Nogoa	≈7	No grazing + grazing, no stocking rates provided	Range ≈0.00–≈18.00 $\text{t ha}^{-1} \text{ year}^{-1}$ (yield varied with cover and soil type, with highest yields for sandstone and mudstone with 0% cover). Noting slopes were steeper at ≈15%	Silburn (2011a)
Murray-Darling Basin	Treverton Creek	6	Comparison of time-controlled (TC) and continuous (Cont) grazing. Stocking rate (SR) for TC = ≈12.6 DSE ha^{-1} for 12 days then 100 days rest. Cont SR = 1.6 DSE ha^{-1}	Range 0.44–2.06 $\text{t ha}^{-1} \text{ year}^{-1}$	Sanjari et al. (2009)

DSE, Dry sheep equivalent.

particularly where there are large bare or scalded areas void of vegetation (Bartley *et al.* 2014).

Silburn *et al.* (2011) captured many of the key factors driving hillslope erosion rates in the rangelands draining to the GBR, including the significance of the 50–70% ground cover threshold required to reduce hillslope soil erosion and runoff, the difference between erosion rates predicted from stoloniferous and tussock grasses, the dominance of low-cover bare patches and large rainfall events on erosion rates, and the time lags associated with improved ground cover and reduced erosion rates.

Model calibration and evaluation

Given the large natural variability represented in hillslope erosion measurements in grazed rangelands (Table 4), and the complex interactions and feedbacks among variables, it is difficult to attribute measured changes in hillslope sediment yields to changed grazing management in real time. Robust models are therefore very useful for providing insights and scenario analysis into the potential effects of rangeland management interventions.

The main method for assessing investments in the GBR catchments is the SourceCatchments model (Wilkinson *et al.* 2014b; McCloskey *et al.* 2021). The key sub-modelling tool used to estimate sediment loss from grazed hillslopes within the SourceCatchments model is the Revised Universal Soil Loss Equation (RUSLE) (Renard *et al.* 1994; Rosewell and Loch 2002; Silburn 2011b). Numerous publications outline how the RUSLE is not necessarily suitable for rangeland environments (Weltz *et al.* 1998; Spaeth *et al.* 2003); primarily due to the higher levels of heterogeneity in soils and plant properties at the plot scale, relative to cropping landscapes (Nearing *et al.* 2011). Another challenge is that RUSLE does not explicitly consider runoff. GBR specific studies have also identified that the RUSLE generally overpredicts hillslope erosion (Hughes and Croke 2011; Brooks *et al.* 2014). Bartley *et al.* (2006), and other studies, have determined that biomass and the arrangement of ground cover are likely better predictors of runoff than average hillslope cover, although it is unlikely that these attributes will replace average ground cover in predictive models any time soon. Despite these findings, RUSLE continues to be used in the GBR due to its utility and ability to parameterise models that can analyse large spatial areas using sparse data sets. The RUSLE is considered broadly suitable as a prioritisation or ranking tool when used in a relative sense for comparing hillslope erosion rates between regions, acknowledging that there may be high prediction error at any one location (Alewell *et al.* 2019). However, this means that it may be problematic to apply the SourceCatchments model and associated tools in accounting and financial payment frameworks (e.g. Reef Credits). This is because these models have relatively high uncertainty, compared to on-ground measured data, when estimating runoff or sediment loss at small spatial

scales. There is therefore a need to consider incorporating a broader suite of modelling tools for GBR rangelands, such as GRASP (or its component sub-models).

This study is the first to apply the GRASP model to a *B. pertusa* dominated commercial grazing property. When calibrated using local data on soils and pastures, GRASP performed well at representing long-term average runoff and sediment yield (both within 2% of observed; Fig. 10), but was only able to adequately represent the year-to-year variation in runoff (Pearson $r = 0.61$), and not fine scale variations in sediment. Erosion models (and complex full-systems models like GRASP) are more suited to representing long-term, broadscale average behaviours. The GRASP calibration in this paper represents a single data point (capturing long-term variation at a single spatial location). A larger number of water quality data sets, from sites with contrasting management and ground cover, would be required to properly evaluate how well GRASP and RUSLE (or other alternatives) can capture the effects of different grazing land management across the GBR catchments (at paddock to catchment scales).

The calibrated model provides valuable insights into what further large-scale encroachment of *B. pertusa* into existing tussock pastures would mean for GBR catchments. Given the strongly altered physiognomy when tussock pastures are replaced by *B. pertusa* (as characterised by the contrasting pasture cover–biomass curve relationships), a scaling factor may need to be considered when modelling water quality from un-calibrated *B. pertusa* dominated landscapes. These changes will have important implications for sites currently being modelled under the Paddock to Reef program (Carroll *et al.* 2012) given the increasing and pervasive presence of *B. pertusa* across large parts of the grazing areas draining to the GBR lagoon. The findings presented here will also improve the ability to model how landscape regeneration from changed grazing land management affects runoff and water quality, including being able to evaluate revised guidelines for safe stocking rates (that incorporate the consequences of the asymmetric rates of land degradation and recovery).

Conclusions

This study of a long-term monitoring site has shown major functional differences in *B. pertusa* pastures, relative to the native tussock pastures they replace, with important implications (both for beef production and GBR water quality) for how they are managed and modelled, and how monitoring data are interpreted:

- (1) Rates of regeneration in land condition are much slower than the rates at which *B. pertusa* initially replaced tussock grasses. In the practical time frames of most land managers (10–20 years), degradation to a *B. pertusa*

state is effectively irreversible. The risks of 'irreversible' pasture transitions from management decisions when stocking rates are adjusted too high, suggests that conservative stocking rate strategies would be particularly favoured in these land types. Where transitions do occur, *B. pertusa* pastures need to be managed as a new land type, accounting for the functional changes that have occurred.

- (2) Changes in species composition and vegetation structure can lead to misinterpretation of monitored trends in ground cover (which is often used as an economical proxy for land condition). Where *B. pertusa* is invading, inferences from positive associations between changes in cover and landscape function can break down: increases in ground cover can instead be accompanied by decreases in biomass, productivity, and sediment retention.
- (3) The paper contributes an initial GRASP model calibration for a transformed *B. pertusa* site, capturing the observed relationships between grazing management, pasture biomass, ground cover and water quality. This serves as a starting point from which these landscapes, of growing extent and concern in GBR catchments, can be better represented in monitoring and modelling assessments, and for revised management options to be explored.

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

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