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Leichhardt, land clearing and livestock: the legacy of European agriculture in the Brigalow Belt bioregion of central Queensland, Australia

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ABSTRACT

Context. The Brigalow Belt bioregion of central Queensland has been extensively developed for agriculture since exploration by Leichhardt in 1844. About 4.5 million hectares of vegetation dominated by brigalow (*Acacia harpophylla*) was cleared as part of the Land Development Fitzroy Basin Scheme, which commenced in 1962. When the *Vegetation Management Act 1999* commenced, 93% of brigalow woodland had been cleared. Grazing is the dominant land use in the Fitzroy Basin, with 2.6 million cattle over 11.1 million hectares (72% of the catchment area). This is the largest cattle herd in any natural resource management region in Australia, accounting for 25% of the state herd and 11% of the national herd. **Aims.** The Fitzroy Basin, Queensland's largest coastal catchment, drains directly to the Great Barrier Reef, and as reef health continues to decline, there has been increased focus on the impacts of land-use change and grazing management on hydrology and runoff water quality. The Brigalow Catchment Study sought to determine the impact of land clearing, land-use change and land management on hydrology, soil fertility, water quality and animal production in the Fitzroy Basin. **Methods.** The study is a paired, calibrated catchment study. Catchment hydrology, soil fertility, water quality and agricultural productivity were monitored before and after land clearing and land-use change. **Key results.** The Brigalow Catchment Study has shown that clearing brigalow for grazing in the Fitzroy Basin doubled runoff, increased peak runoff rate by 50% and increased total suspended solid loads by 80%. Soil fertility and pasture productivity also declined under grazing compared with brigalow. Overgrazing exacerbated these results, as failure to reduce stocking rate with reduced pasture productivity more than tripled runoff, peak runoff rate and total suspended solid load compared with conservatively grazed pasture. **Conclusions.** This study demonstrates the impacts of land-use change and land management on hydrology, soil fertility and water quality. The long-term data records are a model in their own right, capable of answering land-use and land-management questions beyond the initial study scope. **Implications.** Sustainable grazing management should consider the production limitations of depleted soil and pasture resources to minimise land degradation.

Keywords: agricultural systems, buffel grass, dryland farming, grazing management, pesticides, rangelands, resource management, stocking rate.

Introduction

The Brigalow Belt bioregion of Queensland and New South Wales occupies 36.7 million hectares, stretching from Dubbo in the south to Townsville in the north (Fig. 1). Since European settlement, 58% of this bioregion has been cleared. Within Queensland, rates of land clearing were among the highest in the world, with estimates of 425 000–446 000 ha cleared per year (Wilson *et al.* 2002; Lindenmayer and Burgman 2005; Reside *et al.* 2017). More than 60% of this clearing, or ~261 000 ha/year, was undertaken in the Brigalow Belt (Wilson *et al.* 2002; Cogger *et al.* 2003). It is estimated that up to 93% of brigalow scrub has

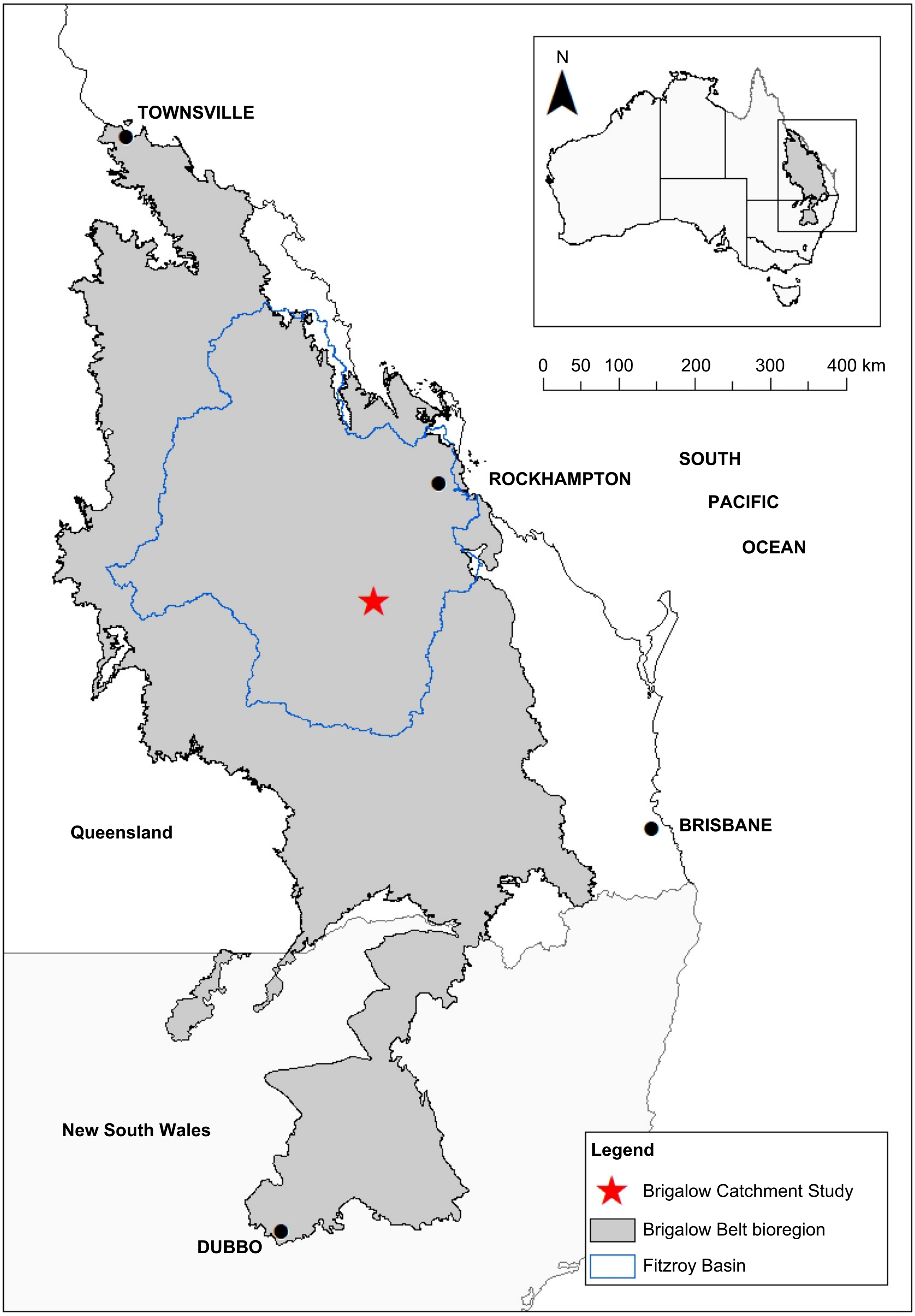

Fig. 1. Location of the Brigalow Catchment Study within the Brigalow Belt bioregion.

been cleared since European settlement (Butler and Fairfax 2003; Cogger *et al.* 2003; Tulloch *et al.* 2016). The spatial extent of this clearing in Queensland is shown in Fig. 2, which contrasts remnant (as at 2019) broad vegetation groups containing *Acacia harpophylla* with cleared land that previously supported broad vegetation groups

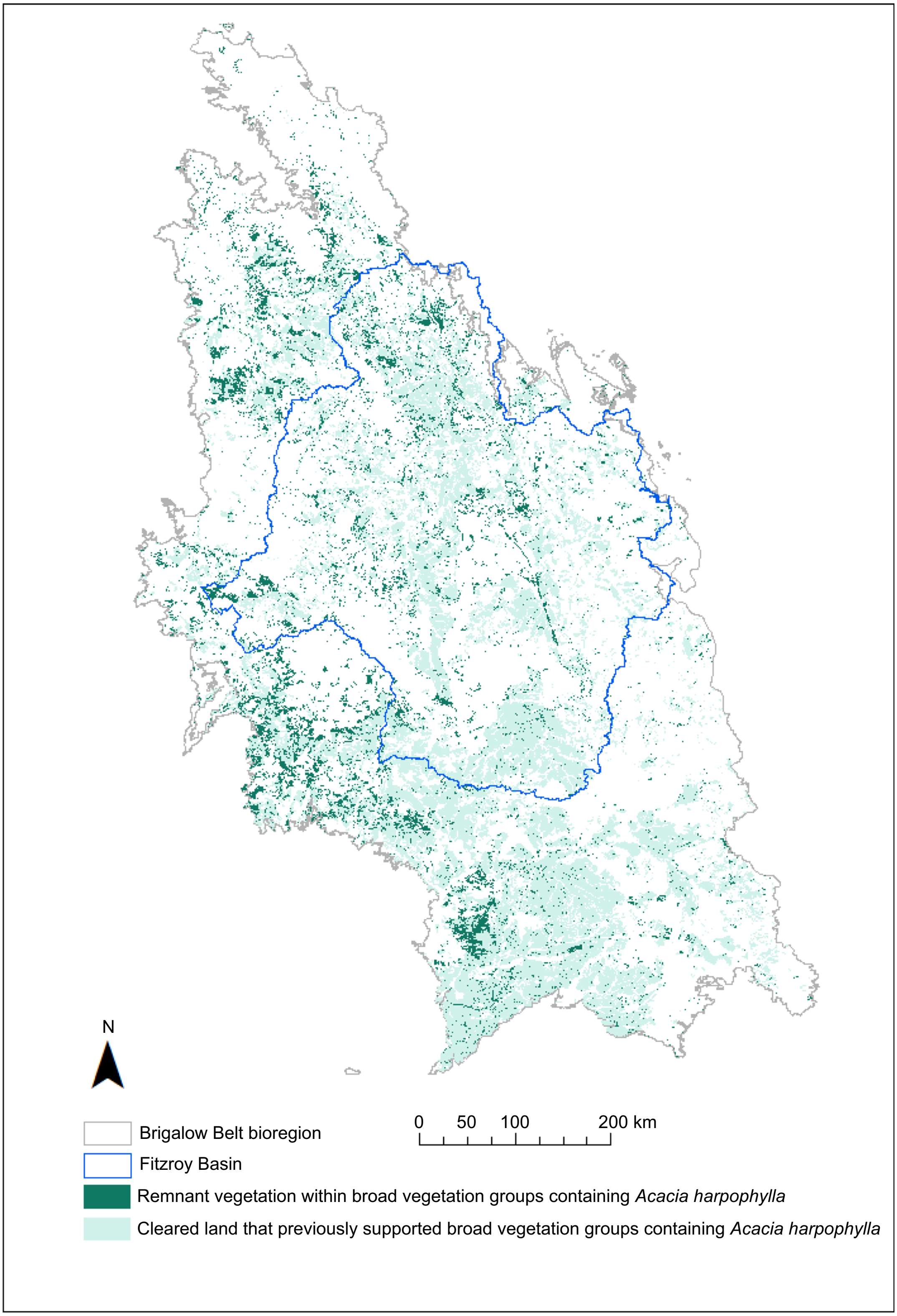

Fig. 2. Remnant (as at 2019) broad vegetation groups in the Queensland Brigalow Belt bioregion containing *Acacia harpophylla* and cleared land that previously supported broad vegetation groups containing *A. harpophylla*.

containing *A. harpophylla* (Neldner *et al.* 2019; The State of Queensland 2019, 2021*a*, 2021*b*).

Within central Queensland, European settlement of the Brigalow Belt bioregion was well established within two decades of its initial exploration by Leichhardt in 1844 (Stewart 1926). Development of the bioregion for agriculture was slow, limited by regrowth of brigalow suckers, drought and prickly pear (*Opuntia stricta*) invasion (Seabrook *et al.* 2006; Cowie *et al.* 2007). Land clearing intensified from about 1940 due to a combination of government policy seeking closer settlement, biological control of prickly pear and mechanisation, particularly post-World War II (Seabrook *et al.* 2006; Lewis *et al.* 2021). In 1962, the Brigalow Land Development Fitzroy Basin Scheme commenced, resulting in the clearing of 4.5 million hectares for cropping and grazing. This clearing represents 21% of all clearing in the Brigalow Belt bioregion and represents 32% of the Fitzroy Basin Catchment area. So as to quantify the effect of land clearing and land-use change on hydrology, soil fertility, water quality and crop and animal production, the Brigalow Catchment Study commenced in 1965 (Cowie *et al.* 2007; Thornton *et al.* 2007).

Materials and methods

Site details

The Brigalow Catchment Study (24°48′S, 149°47′E; Cowie *et al.* 2007; Figs 1, 3) is located near Theodore in the Fitzroy Basin of central Queensland. The project is a paired, calibrated catchment study consisting of three calibrated catchments monitored since 1965 (C1–C3), a fourth catchment monitored since 2010 (C4) and a fifth catchment (C5) monitored since 2014. The catchments vary in size from 12 to 23 ha. Soils within each catchment are predominantly Black and Grey Vertosols, with an average slope of 2.5%. In their virgin state, all catchments were vegetated with brigalow scrub communities. The region has a semi-arid, subtropical climate. Annual average hydrological year (October 1965–September 2018) rainfall was 648 mm.

The Brigalow Catchment Study can be separated into four experimental phases:

(1) Calibration (1965–1982)

Rainfall and runoff were monitored from three contiguous catchments for 18 years. Mathematical relationships were derived to predict runoff from Catchment 2 (C2) and Catchment 3 (C3), given known runoff from Catchment 1 (C1; Thornton *et al.* 2007).

(2) Development (1982–1983)

Catchment 1 remained virgin brigalow scrub to provide a control treatment, while Catchments 2 and 3 were cleared and the fallen timber burnt *in situ*. Catchment 2 was then developed for cropping with the construction of contour banks and grassed waterways, whereas Catchment 3 was

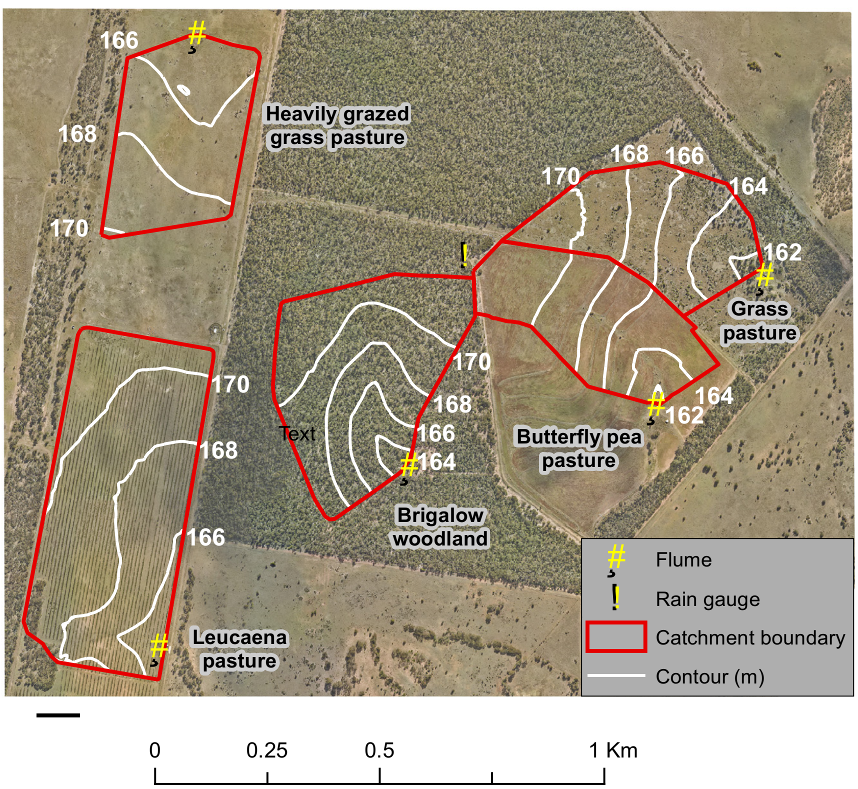

Fig. 3. An aerial view of the Brigalow Catchment Study, showing the five monitored catchments (C1–C5). C1 is a virgin brigalow scrub control, while C2–C5 all currently support grazing on improved grass or improved grass and legume pastures.

developed for grazing by the planting of improved grass pasture.

(3) Land-use comparison (1984–2010)

In C2, the first crop sown was sorghum (*Sorghum bicolor*; September 1984), followed by annual wheat (*Triticum aestivum*) for 9 years. Fallows were initially managed using mechanical tillage (disc and chisel ploughs), which resulted in a significant soil disturbance and low soil cover. In 1992, a minimum-tillage philosophy was introduced and in 1995 opportunity cropping commenced with summer (sorghum) or winter [wheat, barley (*Hordeum vulgare*) and chickpea (*Cicer arietinum*)] crops sown when soil water content was adequate. No fertiliser inputs were used (Radford *et al.* 2007). Catchment 3 was grazed at industry-recommended stocking rates, with utilisation to result in no less than 1000 kg/ha of pasture available at any time.

(4) Adaptive land management (2010–present)

Catchment 2 was planted to butterfly pea (*Clitoria ternatea* cv. Milgarra) ley pasture to restore soil fertility. Catchment 3 maintained the same treatment from the land-use comparison phase; however, management was changed from a set stocking rate on an annual basis to variable stocking rates based on measured pasture biomass with the introduction of wet-season spelling. Woody weed control was undertaken in 2011 by aerial application of 12.5 kg/ha of granular tebuthiuron, which contained 200 g/kg of active ingredient. The method and rate of application were reflective of commercial practice within the central Queensland grazing industry (Thornton and Elledge 2016).

A fourth catchment, C4, was added to the study at this time. The land use of C4 was grazing on improved leucaena (*Leucaena leucocephala* cv. Cunningham) and buffel grass pasture. This catchment had a prior history of cropping and grazing before the planting of leucaena on 8 m hedgerows in 1998.

A fifth catchment, C5, was added to the study in 2014 (Thornton and Elledge 2021). Catchment 5 was also a grazed catchment with improved pasture of buffel grass and purple pigeon grass (*Seteria incrassate*); however, stocking rates in this catchment were as recommended for newly established buffel grass pasture, rather than stocking rates suitable for rundown buffel grass pasture. This catchment also had a prior history of cropping and grazing before its inclusion in the study.

The two catchments added to the long-term study during the adaptive land-management phase were characterised by soils, slope and native vegetation similar to the three original catchments. They also shared a common land-use history prior to their inclusion in this study (Thornton and Elledge 2021). A calibration period in an uncleared state before their inclusion in the study was impossible due to their prior history of agricultural land use. Thus, although the two new catchments have their own unique hydrological characteristics, their relationship to the original catchments in an uncleared state is unknown. Despite this lack of calibration, hydrological responses in these catchments have been clearly attributed to treatment effects rather than a legacy response to historical land use (Thornton and Elledge 2021).

Rainfall and runoff measurements

Each catchment was instrumented to measure runoff by installation of a 1.2-m steel HL flume with a 3.9 m × 6.1 m approach box. Water height through the flumes was recorded using mechanical float recorders. Rainfall was recorded adjacent to each flume and at the head of the catchments using tipping bucket rain gauges with a 0.5 mm bucket (Thornton *et al.* 2007).

Drainage measurements

Deep drainage under native vegetation was determined using steady-state chloride mass balance (Silburn *et al.* 2009). Transient chloride mass balance was used to calculate deep drainage for various periods since clearing. These approaches rely on the water-soluble nature of chloride and assume complete mixing of the soil and water and one-dimensional downward piston flow below the root zone. Both methods require an estimate of chloride input in infiltration and consideration of other potential sources and outputs. Chloride input was determined via soil sampling similar to soil-fertility parameters (below); however, samples were taken down the profile rather than confined to the surface 0.1 m. The soil profile samples used for this deep drainage and chloride mass analysis were taken in 1981 (pre-development), 1983, 1985, 1987, 1990, 1997 and 2000.

Soil fertility measurements

Within each catchment, three permanent monitoring sites were established to monitor soil fertility. Establishment of the 20 m × 20 m sites was undertaken by using double stratification. Initial stratification was based on soil type and slope position, with a monitoring site in an upper and lower-slope position on Vertosols, and the third on a Sodosol. Secondary stratification was by way of 10 subunits, each 4 m × 10 m, within each site. Soil samples were collected from the surface 0.1 m of the soil profile at each monitoring site, by using manual coring tubes of 0.05-m diameter. Samples were a composite of a minimum of eight (20 pre-clearing in 1981, and in 2008 and 2014) 0.05-m-diameter cores, with two cores (five pre-clearing in 1981, and in 2008 and 2014) being taken from around four fixed points within each subunit. Soil samples were collected annually from pre-clearing in 1981 to 1987 and then in 1990, 1994, 1997, 2000, 2003, 2008 and 2014, with samples being retained after analysis in a long-term storage archive. Sample preparation and analytical methods are given in Thornton and Shrestha (2021).

Water quality measurements

Discrete water quality samples were obtained using autosamplers (Thornton and Elledge 2016, 2021; Elledge and Thornton 2017). Auto-samplers were programmed to sample every 0.1 m change in absolute stage height. Event-based water quality loads were calculated by dividing the hydrograph into sampling intervals, multiplying the discharge in each interval by the sample concentration, and summing the loads over all the intervals. The intervals were defined as the start of flow to the mid-point of sample one and sample two, the mid-point of sample one and sample two to the mid-point of sample two and sample three, and so on. Event-based event mean concentrations (EMCs) were calculated by dividing the total event load by the total event flow. Mean annual EMC was calculated by averaging the event-based EMCs. These values were then averaged to determine the long-term EMC for each catchment. To calculate cumulative long-term water quality loads, observed event flow from 1984 to 2010 was multiplied by the long-term EMC (2000–2010) for the respective catchment.

Ground cover and pasture biomass measurements

Ground cover was assessed quarterly using VegMachine® (Fitzroy Basin Association 2018) and compared to similar land types throughout the Fitzroy basin by using the method of Thornton and Elledge (2021). The BOTANAL method of Tothill *et al.* (1978) was used to estimate pasture biomass one to two times per year over the total grazed area of the pasture catchments, excluding the shade lines. Pasture assessments occurred in the late wet and/or the late dry season.

Animal productivity measurements

Catchment 3 has typically been stocked with *Bos indicus* cattle breeds. At a minimum, cattle are weighed on entry to and exit from the catchment. Where possible, animals are fasted for 24 h with water but without feed prior to weighing. When fasting is not possible, fasted weights are estimated as 0.94% of non-fasted weights. This correlation was determined from observed data collected from 1983 to 2005. The study has continuously held an approval to use animals for a scientific purpose, currently granted by the Queensland Department of Agriculture and Fisheries Animal Ethics committee.

Results and discussion

What are the impacts of land-use change on hydrology?

In their virgin state, the catchments behaved similarly, with the average annual runoff being 5% of the annual rainfall. Once cleared, total runoff from the cropping catchment increased to 11% of the annual rainfall and total runoff from the grazing catchment increased to 9% of the annual rainfall; however, timing of the individual runoff events varied between land uses. This increase in runoff reflects water-use patterns that are much more seasonal than those of native vegetation. Both annual cropping and introduced pasture have significant periods of the year without transpiring plants to extract water from depth. It has been suggested that this change in the water-use pattern is the dominant mechanism responsible for hydrologic change, with soil cover, structural decline, and surface roughness being secondary factors (Thornton *et al.* 2007).

Prior to land development, average peak runoff rates from the three brigalow scrub catchments were 3.2, 5 and 2 mm/h for Catchments 1–3 respectively. Peak runoff rates increased significantly from both the cropping and grazing catchments after adjusting for the underlying variation in peak runoff rate due to climatic variation between the pre- and post-development periods. The average peak runoff rate increased by 5.4 mm/h (96%) for the cropping catchment and by 2.6 mm/h (47%) for the grazing catchment. Increases in peak runoff rate were most prevalent in smaller events, with an average recurrence interval of less than 2 years under cropping and 4 years under grazing. Curve fitting to observed annual maximum series data suggests maximum peak runoff rates of 44 and 53 mm/h under cropping for a 1 in 10 and 1 in 20 years average recurrence interval respectively. Under grazing, maximum peak runoff rates of 16 and 17 mm/h were estimated for a 1 in 10 and 1 in 20 years average recurrence interval respectively. This is likely an underestimate due to missing data during large events in this catchment. Soil moisture is a key driver of both runoff and peak runoff rate in this landscape (Thornton *et al.* 2007; Thornton and Yu 2016) .

Steady-state chloride mass balance indicated deep drainage of 0.13–0.34 mm/year across all catchments prior to land development. Large losses of soil chloride occurred under cropping and smaller losses occurred under grazing. Transient chloride mass balance gave average deep drainage of 59 and 32 mm/year for cropping and grazing catchments respectively, during the development phase (1981–1983) when the land was bare following clearing of native vegetation and prior to establishment of crops or pastures (Silburn *et al.* 2009). In the 16.7 years following the establishment of agricultural land uses (1983–2000), transient chloride mass balance gave average deep drainage of 19.8 (range 3.3–50) and 0.16 (−2.2 to 1.4) mm/year respectively, in the cropping and grazing catchments. The drainage rate under grazing was similar to that under brigalow scrub (Silburn *et al.* 2009).

What are the impacts of land-use change on soil fertility?

Initial clearing and burning of brigalow scrub resulted in a temporary increase of mineral nitrogen, total and available phosphorus, total potassium and total sulfur in the surface soil (0–0.1 m), as a result of soil heating and the ash bed effect. Over the subsequent 32 years, fertility declined significantly. Under cropping, organic carbon declined by 46%, total nitrogen by 55%, total phosphorus by 29%, bicarbonate-extractable phosphorus by 54%, acid-extractable phosphorus by 59%, total sulfur by 49%, total potassium by 9% and exchangeable potassium by 63% from post-burn, pre-cropping levels (Thornton and Shrestha 2021). While declines in mineral-associated and particulate organic carbon as a result of long-term cropping were expected, the significant decline in resistant organic carbon was an unanticipated result (Dalal *et al.* 2021).

Fertility also declined under grazing but in a pattern different from that observed under cropping. Organic carbon showed clear fluctuation, but it was not until the natural variation in soil fertility over time was separated from the anthropogenic effects of land-use change that a significant decline was observed (Thornton and Shrestha 2021). However, only 58% of the organic carbon under pasture is derived from the original brigalow vegetation, with the remainder being derived from buffel grass (Dalal *et al.* 2011). This suggests that a poorly managed overgrazed pasture would also lead to a decline in organic carbon. No significant change in mineral-associated, particulate or resistant organic carbon was found (Dalal *et al.* 2021). Total nitrogen declined by 22%. Total phosphorus declined by 14%, equating to only half of the decline under cropping. Bicarbonate-extractable phosphorus declined by 64% and acid-extractable phosphorus by 66%, both being greater than the decline observed under cropping. Total sulfur declined by 23%, less than half of the decline under cropping. A similar decline in total potassium was observed under both land uses, with a 10% decline under grazing. Exchangeable potassium declined by 59% (Thornton and Shrestha 2021).

The primary mechanism of nutrient loss depended on the specific land use and nutrient in question but included removal in grain and beef, mineralisation and oxidation, redistribution and stratification within the soil profile and nutrient pools due to plant growth and litter recycling, uptake and storage in above ground biomass, and loss in runoff and leaching (Thornton and Shrestha 2021).

What are the impacts of land-use change on water quality?

Long-term water quality modelling indicated that changing land use from virgin brigalow scrub to cropping or grazing increased loads of total suspended solids, total and dissolved inorganic phosphorus, and ammonium nitrogen. The well managed (unfertilised) pasture system had less nitrogen in runoff than in runoff from virgin brigalow scrub (Elledge and Thornton 2017). In years when runoff occurred from the agricultural catchments, but no runoff occurred from the virgin brigalow scrub, water quality loads were entirely anthropogenic and totally attributable to land-use change (Thornton and Elledge 2021). Certain agricultural management activities also increase risk to water quality. In the cropping catchment, the largest event-based load of total suspended solids followed a chickpea crop with mechanical tillage for weed control in the fallow prior to and following the crop. Chickpeas leave little stubble cover to protect the soil surface from raindrop impact, so preserving stubble cover with zero-till fallow management both before and after the crop would likely have resulted in better water quality. Indeed, the best stubble and fallow management combination to reduce erosion in dryland grain farming systems in central Queensland is zero-till fallow management following a wheat crop (Carroll *et al.* 1997; Thomas *et al.* 2007). The establishment stage of pasture is, not unexpectedly, the period of greatest risk to water quality in that management system. The risk then declines over time with water quality trending towards that of long-term grazed landscapes (Thornton and Elledge 2014).

What are the impacts of grazing management on ground cover, biomass and water quality?

Grazing land management had significant impacts on ground cover, biomass and water quality. The use of tebuthiuron for woody weed control resulted in detection of tebuthiuron in runoff more than 1200 days after application, with a loss of about 1% of applied herbicide. Tebuthiuron concentrations in runoff declined exponentially with time, rainfall and runoff since application, while half-lives of tebuthiuron in soil averaged 100 days. Loss of tebuthiuron in runoff was in the dissolved phase, with no correlation to total suspended solids (Thornton and Elledge 2016).

Failure to reduce stocking rates on old, rundown pastures to match safe long-term carrying capacity led to 2.5 times more bare ground and only 8% of the pasture biomass compared with conservative grazing at a safe long-term carrying capacity. Increased bare ground and decreased biomass lead to increased runoff, and subsequently increased loads of total suspended solids, nitrogen and phosphorus in runoff. Specifically, heavy grazing resulted in 3.6 times more total runoff, 3.3 times the peak runoff rate and 3.2 times more total suspended solids lost in runoff than with conservative grazing. Loads of total suspended solids, nitrogen and phosphorus in runoff were also greater from heavy than from conservative grazing (Thornton and Elledge 2021).

What are the changes in animal productivity over 38 years of grazing?

Animal productivity at the study site reflected broader changes in the resource base over time. Beef production of about 100 kg/ha.year was possible immediately post-clearing and development (Anderson 1989). Beef production declined slowly during the first 8 years after grazing commenced, with limited variation in stocking rate (Radford *et al.* 2007; Fig. 4). This continued for a further 2 years despite substantial reductions in stocking rate in accordance with reductions in pasture biomass. After a decade of grazing, beef production declined to about 70 kg/ha.year. Stocking rates have typically declined over the subsequent three decades, although substantial year-to-year variation was evident. Beef production continued at a reduced rate with increasing periods of destocking. During the most recent decade of monitoring (March 2010–August 2021), beef production averaged 55 kg/ha per grazing period. The length of each continuous grazing period averaged 6 months at a stocking rate of 0.18 adult equivalent animals/ha.year. The pasture development and grazing regime were estimated to generate a private rate of return on capital invested of between 2% and almost 5% (Stephens *et al.* 2008).

How representative is the Brigalow Catchment Study of the Fitzroy Basin?

The applicability of these findings from the Brigalow Catchment Study to the Fitzroy Basin, and by extension the wider Brigalow Belt bioregion, can be determined by a comparison of the regional ecosystems that dominate the study site with those found within the basin. The extant uncleared vegetation of the Brigalow Catchment Study is classified as regional ecosystems 11.4.8, *Eucalyptus cambageana* woodland to open forest with *A. harpophylla* or *Acacia argyrodendron* on Cainozoic clay plains, and 11.4.9, *A. harpophylla* shrubby woodland with *Terminalia oblongata* on Cainozoic clay plains (The State of Queensland 2020; Thornton and Elledge 2021). The location of these regional ecosystems, and others containing *A. harpophylla*, that support grazing within the Fitzroy Basin is shown in Fig. 5 (after Thornton and Elledge 2021). These regional ecosystems account for 32% of the Fitzroy Basin (Thornton and Elledge 2021).

Regional ecosystems are defined by three major attributes, the first of which is bioregions (Neldner *et al.* 2019). Bioregions are defined as landscapes clustered according to similar attributes including climate, lithology, geology, landforms and vegetation (Thackway and Cresswell 1995). As bioregions are embedded within regional ecosystems, it is likely that, as for the Fitzroy Basin, about 32% of the Brigalow Belt bioregion is directly represented by the Brigalow Catchment Study. While the trends and process understanding obtained from the Brigalow Catchment

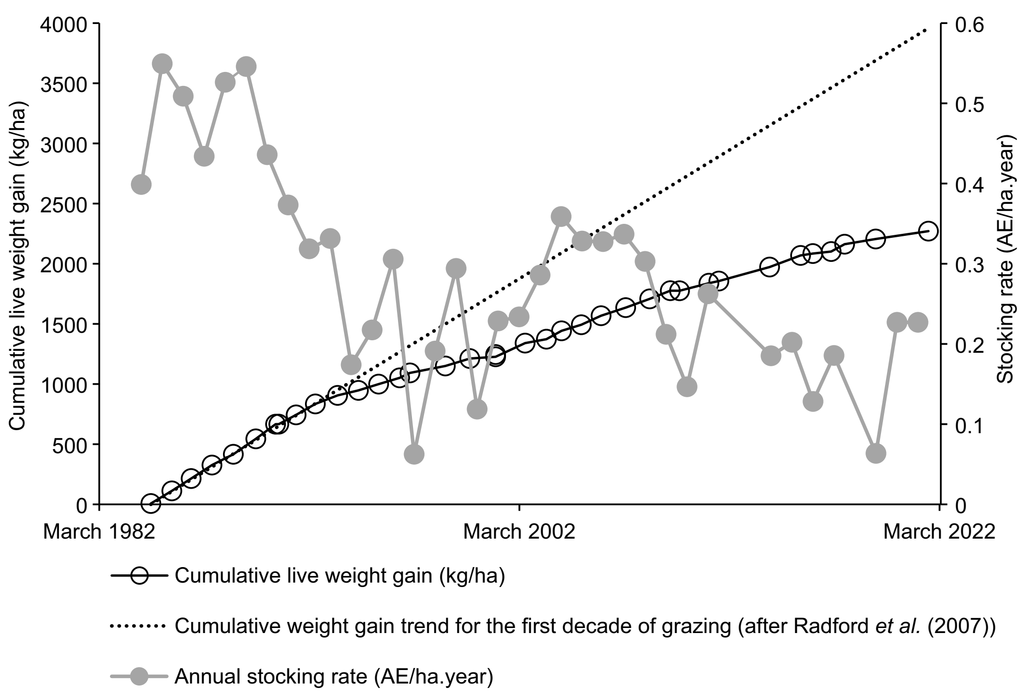

Fig. 4. Cumulative beef production (kg/ha.year) and annual hydrological-year stocking rates [animal equivalents (AE)/ha.year] since the commencement of grazing in the long-term conservative grazing catchment of the Brigalow Catchment study.

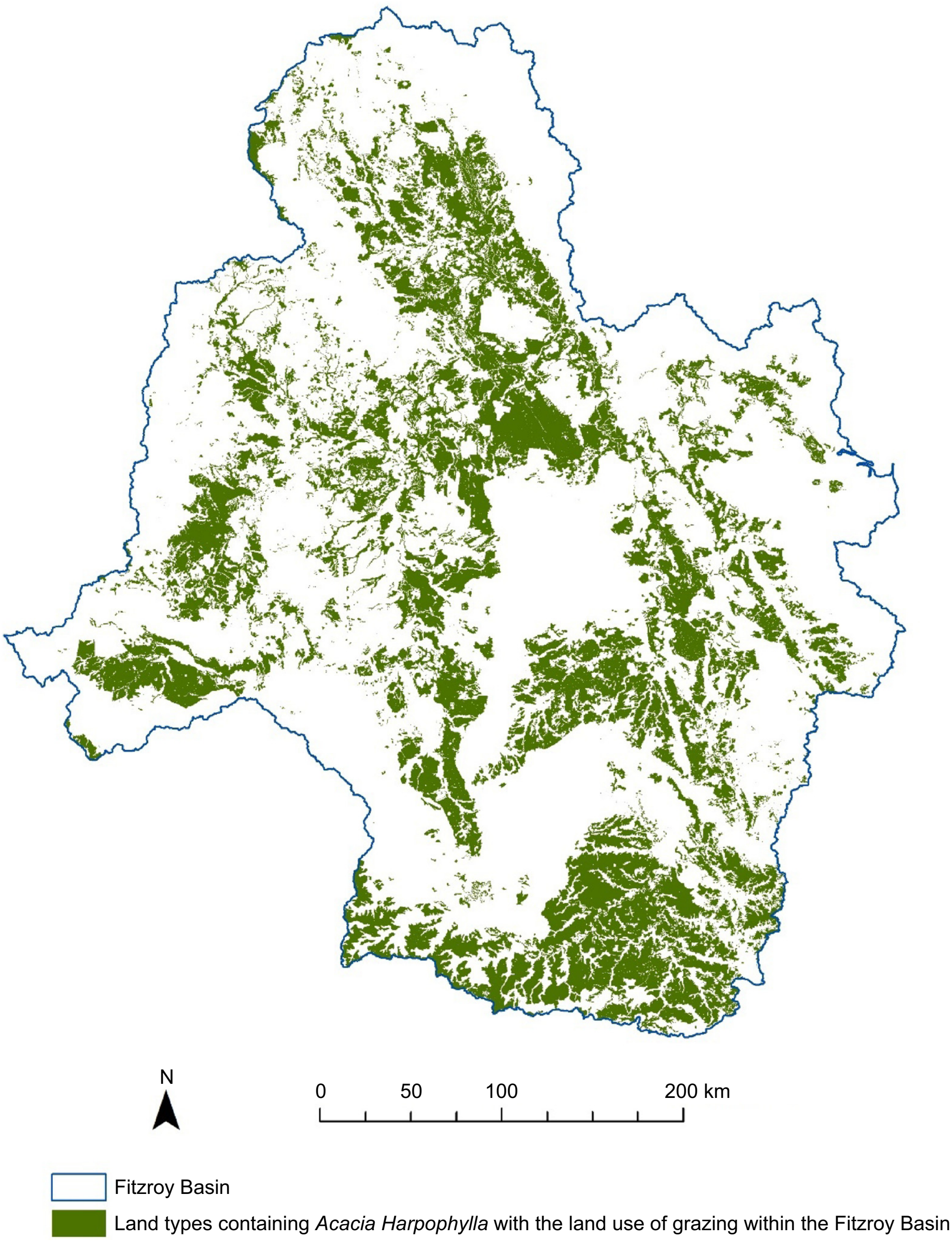

Fig. 5. Land types containing brigalow (*Acacia harpophylla*) with the land use of grazing (as at 2017) (green) within the Fitzroy Basin (blue outline; Thornton and Elledge 2021).

Study are no doubt applicable to the wider bioregion, they also broadly reflect our understanding of catchment processes throughout semiarid, subtropical Queensland. Animal performance and water quality data from the grazed catchments has relevance to the 5.8 million hectares of sown pastures dominated by buffel grass on landscapes cleared of brigalow and gidgee woodlands; the management principles will apply to the further 25.9 million hectares of tropical grass pastures in which buffel grass is common (Peck *et al.* 2011).

How does land management affect the wider catchment?

These findings from the Brigalow Catchment Study give an indication of the effects of land clearing and land-use change on natural resources across the Fitzroy Basin and the Brigalow Belt bioregion. It also demonstrates how the effects of management action, such as, for example, differences in stocking rate, can exceed that of land-use change. Understanding these effects is essential for sustainable management of the bioregion, given the intensive cropping and grazing enterprises it supports. For example, the Brigalow Belt bioregion contains the northern grain zone, the most productive broadacre farming land in Australia and includes the Peak Downs region of central Queensland, the Darling Downs region of southern Queensland, and the Liverpool Plains of northern New South Wales. Similarly, the Fitzroy Basin is home to 2.6 million cattle. This is the largest cattle herd in any natural resource management region in Australia, accounting for 25% of the state herd and 11% of the national herd (Thornton and Elledge 2021).

The impacts of cropping and grazing in the Fitzroy Basin are environmentally important as the catchment discharges directly to the Great Barrier Reef lagoon and, according to the 2017 Scientific Consensus Statement, key Great Barrier Reef ecosystems continue to be in poor condition (Waterhouse *et al.* 2017). This is largely due to the collective impact of terrestrial runoff associated with past and ongoing catchment development, coastal development activities, extreme weather events and climate-change impacts (Waterhouse *et al.* 2017). While uptake of improved grazing management to reduce sediment loss in the Fitzroy Basis has improved between Great Barrier Reef Report Card 2016 and Report Card 2019, it is still considered poor, with a D-score for adoption of best management practices (The State of Queensland 2017, 2021*c*). This highlights the need for further understanding of erosion mechanisms and grazing management practices to improve water quality outcomes in the Fitzroy Basin.

Under the Reef 2050 Water Quality Improvement Plan (https://www.reefplan.qld.gov.au/), using policy driven by best available science, work to decrease land-based runoff in Great Barrier Reef catchments is now well advanced. Significant efforts have been made to implement improved land management practices throughout reef catchments so as to decrease the flow of nitrogen, pesticides and sediments to the reef. The success of reef plan is measured by the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (Carroll *et al.* 2012). The program uses monitoring and modelling tools at the paddock, catchment and marine scale to enable reporting in the short-to-medium term (Waterhouse 2018). The findings from studies such as the Brigalow Catchment Study are extrapolated across subcatchments by using models such as HowLeaky and APSIM (Keating *et al.* 2003; Ghahramani *et al.* 2020). The outputs are then aggregated and routed to the basin outlet by using the Great Barrier Reef Dynamic SedNet model (McCloskey *et al.* 2021*b*). Data from sites such as the Brigalow Catchment Study are critical to ensure the accuracy and credibility of models within the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program modelling framework (Jakeman *et al.* 2019). For example, the Brigalow Catchment Study had a long association with the design, calibration and validation of the HowLeaky model. The HowLeaky model was based on the PERFECT water balance model (Littleboy 1989), which was extensively validated at the study site (Lawrence 1990; Lawrence *et al.* 1991, 1993; Littleboy *et al.* 1992). Study data were also used to calibrate and validate the HowLeaky water balance submodel and to design, calibrate and validate the phosphorus and pesticide submodels (Thornton *et al.* 2007; Robinson *et al.* 2011; Shaw *et al.* 2011). Within the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program, data from the existing project have been used to model runoff, erosion, dissolved inorganic nitrogen and atrazine loss from cropping (Ghahramani *et al.* 2020), to estimate erosion from grazed hillslopes and to validate remotely sensed ground-cover inputs to the model (Dougall and McCloskey 2017), and to calibrate and validate estimates of both erosion and nutrient loss estimations (McCloskey *et al.* 2021*a*, 2021*b*).

Using revised land management data, water quality improvements from continually improving land management practices can be estimated, allowing the Reef 2050 Water Quality Improvement Plan program to evaluate, prioritise and continuously improve the efficiency and effectiveness of its on-ground actions. Results show progress in some areas; however, faster uptake of improved land management practices is required to meet the water quality targets. The Reef Water Quality Report Card 2019 shows that across all reef catchments, the modelling suggests a 14.6% reduction in sediment loads to the Great Barrier Reef (The State of Queensland 2021*c*). Within the Fitzroy Basin, the modelling suggests a sediment-load reduction of 10%.

This is a common story worldwide. The 2017 International Land Use and Water Quality conference (http://www.luwq2017.nl/) demonstrated that many countries have water quality targets but are struggling to meet them. Improved land management practices are being adopted but often will not deliver the magnitude of change that is needed to meet targets and ensure the health of waterways into the future. This is more than a green environmental issue. These are the same waterways and aquifers that provide our drinking water.

This highlights the need to develop, test and understand new land management practices to improve water quality, and will result in the next generation of new research questions for the Brigalow Catchment Study.

Recommendations for further research

The limitations of this study are common to paired catchment studies, which were traditionally limited by the period of record, the climatic sequence experienced during that period, the continuity of monitoring equipment and analytical techniques, and the lack of replication inherent in the experimental design. Calibration of the catchments in their virgin state and decades of monitoring, including extremes of both above- and below-average rainfall, suggest that surface water hydrology is well understood at this site. Similarly, the statistical rigour of the long-term soil fertility monitoring suggests good understanding of surface soil nutrient dynamics. However, ongoing leaching of chloride from the soil profile is now limiting chloride mass balance drainage modelling, and hence the understanding of drainage at the site. This can be rectified by the addition of chemical tracers such as bromide as a replacement for naturally occurring chloride.

While surface and profile hydrology are well understood, the effects of managing grazing pressure by varying stocking rate on runoff water quality has only been investigated during the longest continuous period of below-average rainfall since data collection at the study site began in 1965. Monitoring during average and above-average years is essential to determine whether water quality responses to grazing management in wet years follow the trends seen to date. Additionally, the water quality data investigations presented in this paper commenced nearly two decades after land clearing and land-use change. As soil nutrient fluxes associated with these perturbations had stabilised prior to the commencement of water quality monitoring, it is likely that measured sediment and nutrient loads in runoff would have been exceeded by losses that occurred soon after land clearing. Inclusion of unpublished historical water quality data captured closer to the time of land clearing and land-use change in future analysis will assist in validating the assumption of greater sediment and nutrient loads in runoff than in the monitoring period reported.

Conclusions

This 57-year longitudinal study clearly shows the impacts of land-use change and land management on hydrology, soil fertility and water quality. The long-term data records can be considered a model in their own right and are capable of answering questions well beyond the initial scope of the study. Given the level of foresight and investment that is required to implement and maintain these experiments, it is unlikely that new studies of this nature will be commissioned. Revisiting these historical datasets and adapting the design of the ongoing experiment will allow researchers to answer new questions not thought of, or not of concern when this study commenced nearly six decades ago.

Accessing the Brigalow Catchment Study

The Brigalow Catchment Study data portal provides easy access to additional information about the study and its publications. The portal also provides real time viewing of rainfall and runoff data from the study catchments. Please connect with the Brigalow Catchment Study at www.brigalowcatchmentstudy.com.

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Data availability. The data that support this study will be shared upon reasonable request to the corresponding author.

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