Agricultural land management practices and water quality in the Fitzroy Basin

Technical report for the 2015 to 2019 hydrological years



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QUEENSLAND GOVERNMENT

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Cover photographs: cattle in the heavily grazed pasture catchment (left); runoff event through a monitoring flume (centre); and a fenceline comparison of conservatively and heavily grazed pastures (right). All photographs are sourced from the Brigalow Catchment Study photo archives, courtesy of the Department of Natural Resources, Mines and Energy.

This report is available from the Brigalow Catchment Study website www.brigalowcatchmentstudy.com.

Executive Summary

Loss of sediment, particulate nitrogen and particulate phosphorus in runoff from the extensive grazing lands of the Fitzroy Basin, central Queensland, continue to contribute to the declining health of the Great Barrier Reef. Substantial investment has been made by the Australian and Queensland Governments to improve runoff water quality from grazing land; however, there is little data directly comparing the effect of grazing pressure on hydrology and water quality. This is further confounded by the difficulty of separating the impacts of climate variability from the anthropogenic impacts of changing land use from native vegetation to grazing. This study measured changes in hydrology and water quality from conservative and heavy cattle grazing pressures on rundown improved grass pastures. Conservative grazing pressure reflected the safe long-term carrying capacity for rundown buffel grass pastures, whereas heavy grazing pressure reflected stocking rates recommended for newly established pastures. This study also considered the anthropogenic effect of changing land use from brigalow scrub to an improved grass pasture with a conservative grazing pressure.

After four below-average rainfall years from 2015 to 2018 (Appendix 1.1), heavy grazing resulted in 3.6 times more total runoff and 3.3 times greater average peak runoff rate compared to conservative grazing. No runoff occurred from brigalow scrub in two of the four years, which means that no runoff would have occurred from the conservatively grazed pasture had it remained uncleared. Mean annual loads of total suspended solids, nitrogen and phosphorus (total and dissolved) in runoff were greater from the two grass pastures than from brigalow scrub, while loads from heavy grazing were greater than from conservative grazing. In contrast, event mean concentrations were lower from heavy than conservative grazing due to the dilution effect of increased runoff. In the two years with no runoff from brigalow scrub, total runoff and pollutant loads from conservatively grazed pasture were an absolute anthropogenic increase attributable to land use change.

Hydrology and water quality monitoring continued for the first six months of the 2019 hydrological year. Mean annual rainfall for this period was also below the long-term average; however, rainfall in the month of October, when runoff occurred, was the second-highest October total on record. This resulted in both the highest mean annual and event based runoff from all three catchments compared to 2015 to 2018. During 2019, loads of total suspended solids, particulate nitrogen and all phosphorus parameters remained higher from heavily than conservatively grazed pasture. However, loads of total and dissolved nitrogen were lower from heavily than conservatively grazed pasture, which is in contrast to the 2015 to 2018 period where loads were greater from heavily grazed pasture. Event mean concentrations were consistently lower from heavily grazed pasture compared to conservatively grazed pasture for both reporting periods.

Modelling of the long-term hydrology and water quality data from the Brigalow Catchment Study has shown that an unfertilised cropping system exports higher loads of total suspended solids, nitrogen and phosphorus (total and dissolved) compared to a conservatively grazed pasture (Appendix 1.2). Furthermore, grazed pasture exports higher loads of total suspended solids and phosphorus compared to brigalow scrub, but less total and dissolved inorganic nitrogen. One explanation for the variation in the magnitude and direction of pollutant differences between treatments is dilution. That is, increased runoff from either above average rainfall or a treatment effect, such as grazing pressure or a bare fallow, results in the dilution of pollutants in runoff which leads to lower event mean concentrations. This highlights the importance of reporting runoff data, as high loads are not necessarily related to high event mean concentrations.

Other research at the Brigalow Catchment Study (Appendix 1.3) investigated changes in soil fertility when changing land use from brigalow scrub to either an unfertilised cropping system or a conservatively grazed pasture. Increases in mineral nitrogen and both total and available phosphorus were found in surface soil due to ash deposition from clearing and burning native vegetation. However, total and available nitrogen and phosphorus under both agricultural systems declined over the subsequent 32 years since land use change. The effective depth of interaction for rainfall, runoff and soil is 0.1 to 4.0 cm (Sharpley 1985), so the cumulative loss of sediment and nutrients in runoff and the subsequent decline in surface soil fertility over time are interrelated. This highlights the importance of not just monitoring runoff pollutants, but also the fertility of the soil surface to improve understanding of agricultural land management impacts.

Determination of particle size distribution in both runoff and deposited material was undertaken at the Brigalow Catchment Study for the first time during the 2019 hydrological year. Land uses with high cover and high biomass had the lowest proportion of fine particles less than 16 µm in runoff. No correlation was found between loads of total suspended solids and fine particles. The proportion of ultrasonically dispersed fine particles from land uses with low cover and low biomass was 94%, which is the same as that reported at the end of catchment scale for the Fitzroy Basin. A fine particle sediment enrichment ratio of 1.6 was observed from deposited material to runoff. Despite clear trends, this data only represents a single point in time and ongoing monitoring will be essential to improve confidence in these findings.

Long-term data from the Brigalow Catchment Study has also been used to develop methods for estimation of peak runoff rate to improve erosion modelling activities in Great Barrier Reef catchments (Appendix 1.4). Four methods of estimating peak runoff rate were compared using data from three catchments, both prior to clearing brigalow scrub (1965 to 1982) and after conversion of two catchments to either cropping or grazing, while the third catchment was retained as brigalow scrub (1985 to 2004). Despite different data requirements and complexity, all four methods were easily applied with parameter values derived from widely available rainfall data, easily measured or estimated runoff volume data, and basic physical descriptors of the catchment.

In summary, the long-term Brigalow Catchment Study dataset has been fundamental for addressing numerous knowledge gaps through: (1) the provision of empirical data to support the adoption of improved agricultural land management practices; and (2) collaboration with modellers funded by the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program to further refine parameters used to report progress towards achieving the Reef 2050 Water Quality Improvement Plan 2017 to 2022 water quality targets. A conceptual model of the outputs from the Brigalow Catchment Study and how they have delivered on the objectives of the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program highlights these achievements.

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List of Units

AE/ha/yr	Adult equivalent per hectare per year
days/yr	Days per year
ha/AE	Hectare per adult equivalent
kg/ha	Kilogram per hectare
kg/ha/yr	Kilogram per hectare per year
m	Metre
mg/L	Milligram per litre
Mha	Million hectare
mm	Millimetre
mm/hr	Millimetres per hour
t/ha	Tonne per hectare
µm	Micrometre

Abbreviations

BCS	Brigalow Catchment Study
DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved Inorganic Phosphorus, also known as Filterable Reactive Phosphorus (FRP) and Orthophosphate (PO_4-P)
DON	Dissolved Organic Nitrogen
DOP	Dissolved Organic Phosphorus
EMC	Event Mean Concentration
P2R2	Phase 2 of the Paddock to Reef program
P2R3	Phase 3 of the Paddock to Reef program
PN	Particulate Nitrogen, also known as Total Suspended Nitrogen (TSN)
PP	Particulate Phosphorus, also known as Total Suspended Phosphorus (TSP)
PSD	Particle Size Distribution
QRWQP	Queensland Reef Water Quality Program
QWMN	Queensland Water Monitoring Network
RRRD	Reef Rescue Research and Development program
TDN	Total Dissolved Nitrogen
TDP	Total Dissolved Phosphorus
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids

Acknowledgments

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1 Introduction

The 2017 scientific consensus statement on Great Barrier Reef water quality identified the Fitzroy Basin as a high priority area for reducing fine sediment and particulate nutrients (Waterhouse *et al.* 2017). Grazing is the dominant land use in this region, with more than 2.6 million cattle over 11.1 Mha (Australian Bureau of Statistics 2009; Meat and Livestock Australia 2017). This is the largest cattle herd in any natural resource management region in both Queensland and Australia, accounting for 25% of the state herd and 11% of the national herd (Meat and Livestock Australia 2017). Despite the extent of the grazing industry in this region, and throughout northern Australia, there is little data directly comparing the effect of grazing pressure on hydrology and water quality. This is further confounded by the difficulty of separating the impacts of climate variability from the anthropogenic impacts of changing land use from native vegetation to grazing.

This study measured changes in hydrology, water quality, ground cover and pasture biomass from cattle grazing at conservative and heavy grazing pressures on rundown (>30 years old) improved grass pastures. Furthermore, it also considered the anthropogenic effect of changing land use from virgin brigalow scrub to an improved grass pasture with a conservative grazing pressure. Data collected from 2015 to 2018 was reported by Thornton and Elledge (2018) for the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (hereafter referred to as the Paddock to Reef program) (Appendix 1.1). Monitoring of hydrology and water quality continued for the first half of the 2019 hydrological year. The synthesis of both monitoring periods, from 2015 to 2019, is captured in the current report for the Queensland Reef Water Quality Program. In addition, the scope of the current report was broadened to include particle size distributions (PSD) of soil in runoff and in deposited material.

Documenting the link between improved land management practices and improvements in water quality underpin the adaptive management approach of the Reef 2050 Water Quality Improvement Plan 2017 to 2022 (hereafter referred to as Reef Plan), which seeks to improve the quality of water flowing from catchments adjacent to the Great Barrier Reef. Monitoring and modelling activities from the Paddock to Reef and Queensland Reef Water Quality Programs are used to evaluate progress towards the Reef 2050 Water Quality Improvement Plan targets in the Great Barrier Reef Report Cards (The State of Queensland 2018; Waterhouse *et al.* 2019). A substantial body of evidence documents the anthropogenic effects of land use change on natural resources in the Brigalow Belt bioregion, particularly in the Fitzroy Basin. A doubling of runoff (Thornton *et al.* 2007) and similar increases in peak runoff rate (Thornton and Yu 2016) have been reported as a result of land use change, which implies that there has been a subsequent anthropogenic impact on water quality. An additional aim of this study was to use 25 years of runoff data and 10 years of water quality data to determine loads of total suspended solids, nitrogen and phosphorus in runoff from cropping and grazing compared to virgin brigalow scrub. The anthropogenic effect of changing land use from native vegetation to agriculture was published by Elledge and Thornton (2017) in the journal of Agriculture, Ecosystems and Environment (Appendix 1.2).

During an independent review of the Paddock to Reef program in 2015, the methods used by paddock monitoring, paddock modelling and catchment modelling to calculate an event mean concentration (EMC) were rigorously debated. Similar comments were also reiterated to authors during the journal review process for Elledge and Thornton (2017). As the Great Barrier Reef Report Card is underpinned by these monitoring and modelling activities, it was necessary to validate the method used to derive EMCs. To address this knowledge gap, four methods were compared using 16 years (2000 to 2015) of water quality data from five catchments of the long-term Brigalow Catchment Study (BCS). These results are reported in Appendix 1 of Thornton and Elledge (2018).

Validation of the EMC method was undertaken with data from catchments that had undergone land use change from virgin brigalow scrub to agriculture 18 years prior to the start of the dataset. However, soil fertility in these catchments has been shown to limit plant growth within 12 years of land use change (Radford *et al.* 2007), which had also occurred prior to the start of the dataset. Given water quality loads are a result of the interaction between runoff and surface soil (Lin *et al.* 2006; Sharpley 1985), it is possible that changes in soil fertility as a result of land use change would also result in changes to water quality over time. Thus, surface soil fertility (0 to 10 cm) was investigated from 1981 (pre-clearing) to 2014 by Thornton and Shrestha (Unpublished). This is a draft manuscript that has received approval by the Queensland Government for external release to the journal Soil Research (Appendix 1.3). These results facilitate modelling by numerically describing the starting condition of the landscape and mathematically defining fertility trends over time. Discussion on the mechanisms of change further informs process based models, assisting in moving forward from traditional empirical black box (conceptual) models.

In the future, long-term soil fertility and water quality data from the BCS can be integrated to investigate the hypothesis that changes in soil fertility, as a result of land use change, would also result in changes to water quality over time. This is relevant to the Paddock to Reef program as temporal changes in water quality as a result of fertility decline from a consistently managed, single land use catchment, cannot be resolved by implementing APSIM, HowLeaky or eWater Source models in their current frameworks. Testing of this hypothesis is required to determine if there is a need to change the model frameworks.

Other identified research priorities for the modelling components of the Paddock to Reef program included developing spatially derived peak runoff rates to allow the modified universal soil loss equation (M-USLE) for erosion modelling to be implemented within eWater Source (Carroll and Yu 2018). Hydrological characteristics of the BCS are already well documented (Thornton *et al.* 2007; Thornton and Yu 2016), so runoff data from the same catchments were used to identify a suitable method to derive peak runoff rate. Four methods that used either site specific regression, rainfall rate, curve number or infiltration rate as the primary runoff driver were evaluated against observed peak runoff rate by Thornton and Yu (Unpublished). This is a draft manuscript that has received approval by the Queensland Government for external release to the journal of Soil Research (Appendix 1.4).

The current report demonstrates the integration and synthesis of knowledge obtained from strategic investments from various Reef Plan programs along with long-term foundational datasets from the BCS. A conceptual model of the outputs generated from the BCS during the period of Reef Plan funding (2010 to 2019) is provided in Figure 1. The technical reports and journal papers listed provide a body of evidence that: (1) documents anthropogenic impacts on soil and water resources; (2) demonstrates management practices that can improve water quality outcomes from grazing land; and (3) provides data to further refine modelling components of the Paddock to Reef program.

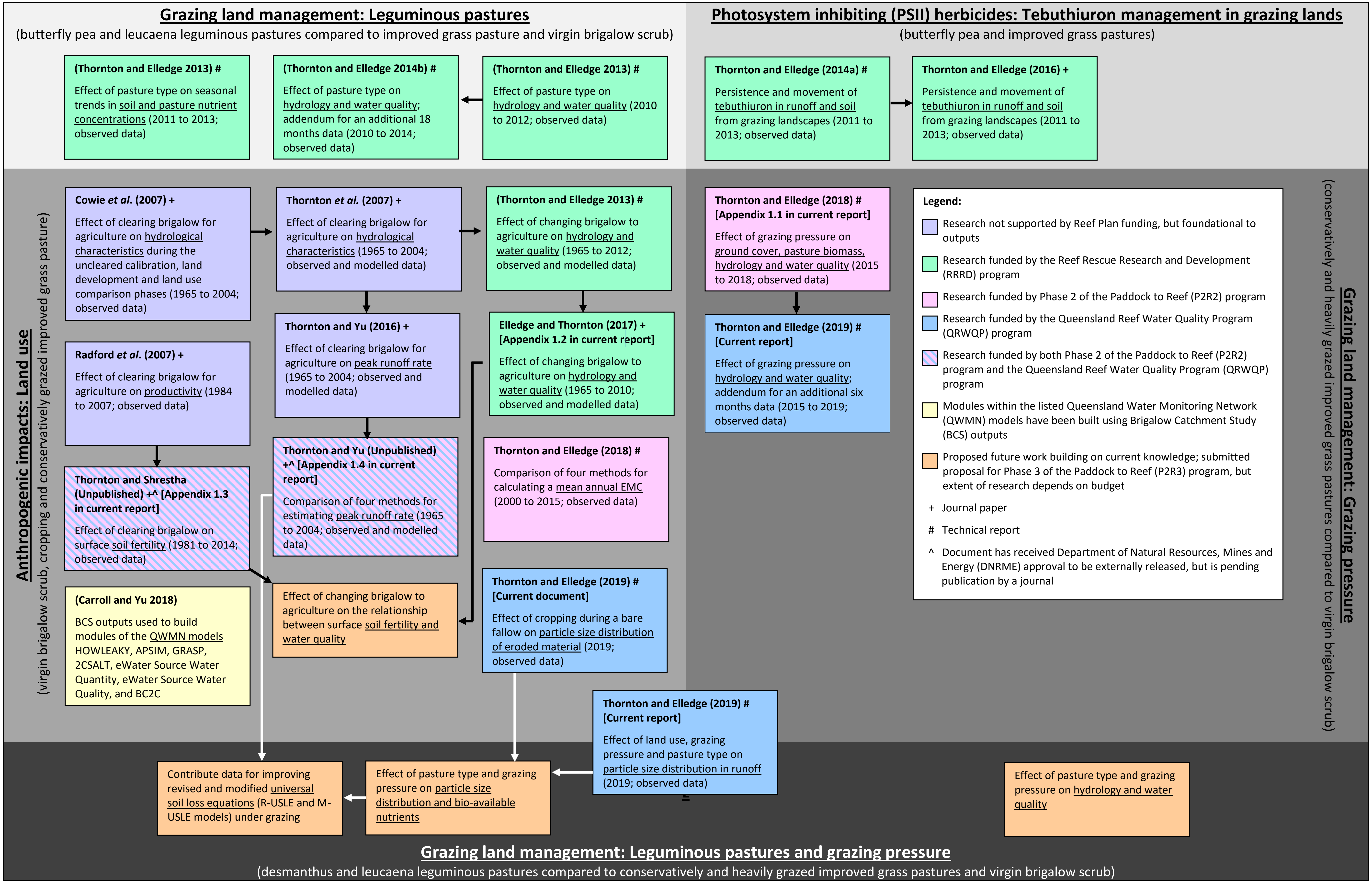

Figure 1: Conceptual model of technical reports and journal papers that have been delivered by the Brigalow Catchment Study project from 2010 to 2019 while supported by various Reef Plan programs.

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2 Methods

This report is an addendum to the paddock scale water quality monitoring that occurred from 2015 to 2018 at the BCS, near Theodore in central Queensland (Thornton and Elledge 2018). This study includes an additional six months data collected during the wet season of the 2019 hydrological year (October to March). While not a complete hydrological year, no additional runoff events were expected as April to October encompasses the dry season. This report only updates the hydrology and water quality results, not the ground cover and pasture biomass results. Monitoring for the 2019 hydrological year was also expanded to include measurements of PSD of soil in both runoff and deposited material, as described in Section 2.3.

2.1 Site Description and Treatments

A comprehensive description of the study site, experimental design, analytical methods and data analyses are provided in Thornton and Elledge (2018). Grazing management during the 2019 wet season in relation to the 2015 to 2018 grazing pressures are summarised in Table 1 and Table 2.

Table 1: Annual stocking rates in adult equivalents (AE) per hectare per year and also in hectare per AE for the two pastures.

Year	Stocking rate (AE/ha/yr)		Stocking rate (ha/AE)	
	Conservative grazing	**Heavy grazing**	**Conservative grazing**	**Heavy grazing**
2013	Destocked	0.09	Destocked	1.90
2014	0.19	Destocked	0.67	Destocked
2015	0.20	0.83	3.86	0.81
2016	0.13	0.20	1.47	1.32
2017	0.19	0.26	4.42	1.11
2018	Destocked	0.85	Destocked	0.52
2019	0.06	0.13	4.35	0.96

Table 2: Annual number of non-grazed days (spelling) for the two pastures.

Year	**Pasture spelled (days/yr)**	
	Conservative grazing	**Heavy grazing**
2013	365	303
2014	320	365
2015	80	33
2016	297	286
2017	76	180
2018	365	146
2019	264	319

Measurement of PSD in runoff was undertaken during the 2019 hydrological year from all five catchments of the BCS. This includes Catchments 2 and 4 which were not incorporated in the Paddock to Reef program report (Thornton and Elledge 2018), but have been previously monitored and reported as part of the Reef Rescue Research and Development program (Thornton and Elledge 2013; Thornton and Elledge 2014b). During the 2018 and 2019 hydrological years, Catchment 2 was a cropping treatment in fallow with minimal cover levels after the butterfly pea ley pasture was terminated by disc ploughing and other operations to prepare the paddock for replanting (Table 3). Catchment 4 was a grazed leucaena and grass treatment with an average stocking rate of 0.18 AE/ha/yr, or alternatively 1.45 ha/AE, and 271 non-grazed days in the year. Pasture biomass from Catchment 4 was 0.2 t/ha in the 2018 late dry season. PSD of deposited material was only monitored from Catchment 2.

Table 3: Fallow management operations performed in Catchment 2 over the 2018 to 2019 hydrological years.

Date	**Operation**	**Description**
12/10/2017	Tillage	Termination of previous pasture by tillage with an offset disc plough resulting in full profile inversion
12/11/2017	Herbicide	Application of non-selective herbicides aiming for 100% plant mortality
06/03/2018	Tillage	Tillage with an offset disc plough resulting in full profile inversion
13/09/2018	Herbicide	Application of non-selective herbicides aiming for 100% plant mortality
01/11/2018	Herbicide	Application of non-selective herbicides aiming for 100% plant mortality
17/12/2018	Tillage	Tillage with a scarifier resulting in disturbance of the soil surface, but not inversion of the profile

2.2 Hydrology and Water Quality

Monitoring of hydrology and water quality from the brigalow scrub and two grass pastures, with either conservative or heavy grazing pressure, for the 2015 to 2018 hydrological years are outlined in Thornton and Elledge (2018). Monitoring continued for the first six months of the 2019 hydrological year (October 2018 to March 2019) using the same methods, except that laboratory analyses were undertaken by the Department of Environment and Science Chemistry Centre.

2.3 Particle Size Distribution

2.3.1 Runoff

Monitoring of PSD from all five catchments of the BCS commenced in 2019. Analysis of PSD was performed by laser diffraction of runoff samples. This was undertaken by the Department of Environment and Science Chemistry Centre using a Malvern Mastersizer 3000E (lens range 0.02 to 2,000 µm) according to methods developed in accordance with the instrument operating procedures, Australian standard AS 4863.1-2000 and method 2560 D (Standards Australia International 2000). Samples were analysed both as-received and after drying and grinding; the latter where samples were air dried at 40˚C and ground to pass a 2 mm sieve. No other pre-treatments, such as chemical digestion, were performed. The Mastersizer was operated at a pump speed of 2,800 revolutions per minute, with sonication by probe performed for 150 seconds with 20 µm of tip displacement. PSDs using similar methods have been reported by a number of Great Barrier Reef sediment and erosion studies (Bainbridge *et al.* 2016; Eyles *et al.* 2018; Garzon-Garcia *et al.* 2018).

The Udden–Wentworth size classification, rounded to zero decimal places, was adopted for this study (Leeder 1982). Particles less than 4 µm are classified as clay, particles 4 µm to less than 16 µm are very fine and fine silt, particles 16 µm to less than 20 µm are medium silt, particles 20 µm to less than 63 µm are medium and coarse silt, and particles 63 µm to 2,000 µm are sand. Fine particles less than 16 µm are considered to be the greatest risk to Great Barrier Reef water quality; however, modelling components of the Paddock to Reef program focus on fine particles less than 20 µm (Bartley *et al.* 2017). References to fine particles in this report refer to particles less than 16 µm. Data on particles less than 20 µm are provided to assist with modelling.

Three dispersion methods were also compared for the determination of PSD: (1) non-dispersed which represents runoff with particles that have recently detached from the soil surface; (2) mechanically dispersed which represents how particles might present in a river system under flow conditions; and (3) ultrasonically dispersed which represents disaggregation to primary particles and information on the shearing resistance of clay particles. Sample numbers that are missing from events in the results section are due to the automatic samplers not detecting liquid when triggered.

2.3.2 Deposited Material

Within the cropping bare fallow (Catchment 2) of the BCS, eroded material from the hillslope that was deposited at the end of three waterways (one grassed and two immediately adjacent to the bare fallow) were sampled for determination of PSD. A composite sample of four to six cores was collected from each site where the depth of deposited material was greater than 0.10 m. Cores were collected by manually pushing a 0.042 m diameter coring tube into the deposit. Laser diffraction of deposited material was undertaken by the Department of Natural Resources, Mines and Energy using a Malvern Mastersizer 2000E (lens range 0.02 to 2,000 µm) according to the method of Eyles *et al.* (2018). In addition to this method, which required samples to be dried and ground, PSD of deposited material was also determined for the unprocessed, as-received sample.

3 Results

3.1 Hydrology

Total annual rainfall at the study site was below the long-term mean annual rainfall of 643 mm (October 1965 to March 2019) in all five hydrological years (Figure 2). Rainfall was in the 32nd percentile in 2015 (563 mm), the 30th percentile in 2016 (562 mm), the lowest on record in 2017 (272 mm), the 42nd percentile in 2018 (584 mm), and the 4th percentile for the first six months of the 2019 hydrological year (363 mm). However, the 2019 runoff event occurred in the second wettest October on record (1965 to 2019).

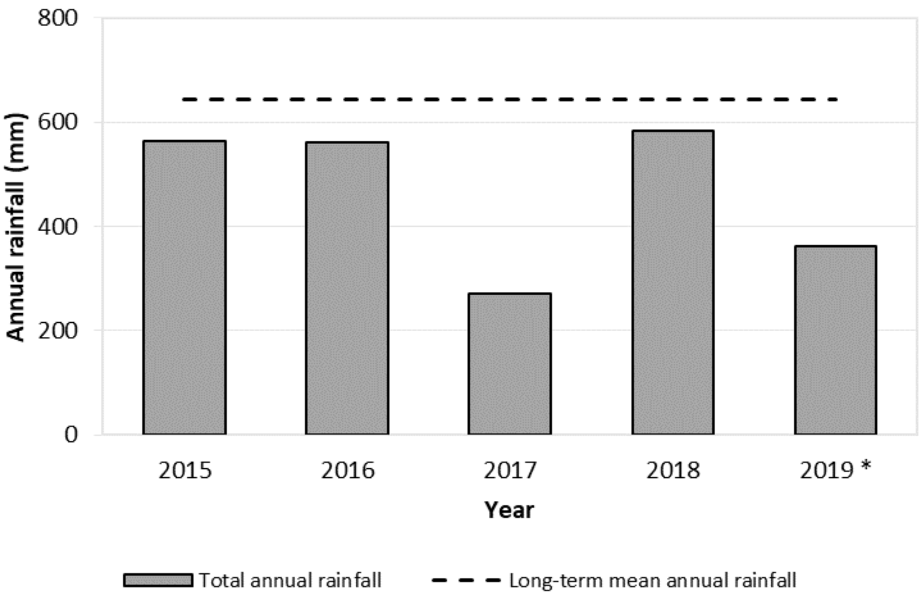

*Figure 2: Total annual hydrological year rainfall for 2015 to 2019 relative to the long-term mean annual rainfall for the Brigalow Catchment Study. * Rainfall in 2019 is an incomplete hydrological year from October 2018 to March 2019 only.*

Similar to rainfall, runoff for the five hydrological years was below the long-term mean annual runoff (1985 to 2019) for the brigalow scrub and conservatively grazed catchments (Figure 3). The heavily grazed catchment was instrumented in 2014, at the commencement of this study, and mean annual runoff was based on five years (2015 to 2019) of data. Runoff from brigalow scrub was in the 31st percentile in 2015, no runoff occurred in 2016 or 2017, the 28th percentile in 2018, and the 56th percentile for the first six months of the 2019 hydrological year. Runoff from the conservatively grazed catchment was in the 34th percentile in 2015, the 29th percentile in 2016, no runoff occurred in 2017, the 15th percentile in 2018, and the 42nd percentile for the first six months of 2019. The heavily grazed catchment had the same amount of runoff (28 mm) in both 2015 and 2016, no runoff occurred in 2017, and runoff in 2018 and 2019 was 68% and 172% of the 2015 to 2016 average, respectively. Although 2019 is an incomplete hydrological year and mean annual rainfall is currently below the long term average, runoff contributed to 98% of the total runoff over the total five years presented in this report for brigalow scrub, and contributed to 44% and 39% of runoff from conservatively and heavily grazed pastures, respectively.

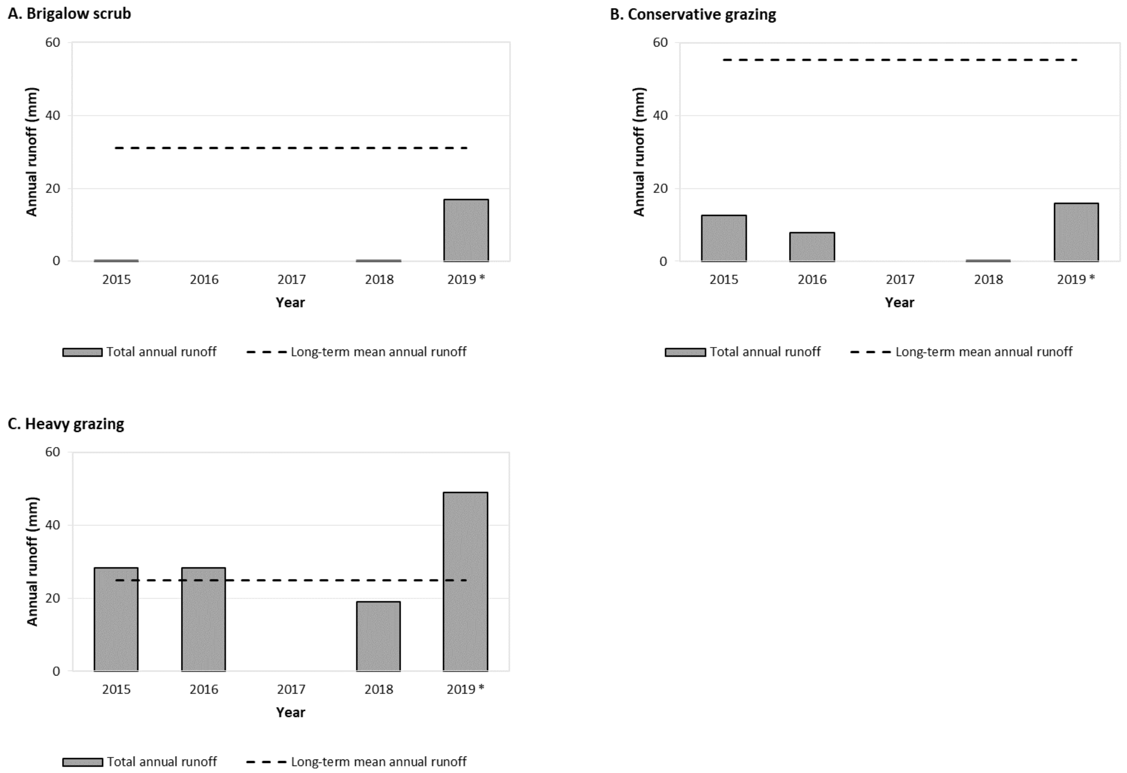

*Figure 3: Total annual hydrological year runoff for 2015 to 2019 relative to the long-term mean annual runoff for the three catchments. Long-term means were based on 35 years (1985 to 2019) data for the brigalow scrub and conservatively grazed catchments, and five years data (2015 to 2019) for the heavily grazed catchment. * Runoff in 2019 is an incomplete hydrological year from October 2018 to March 2019 only.*

Hydrological data and water quality sampling effort for 2015 to 2019 are summarised in Table 4. Over the five hydrological years, there was a total of three events from the brigalow scrub catchment, five events from the conservatively grazed catchment, and six events from the heavily grazed catchment. Although the number of events and total runoff was low in these below-average rainfall years, when runoff did occur, the heavily grazed catchment had consistently greater runoff than the conservatively grazed catchment. A similar trend was also observed for peak runoff rates with both average and maximum values greatest from the heavily grazed pasture.

Using the hydrological calibration developed during Stage I (1965 to 1982) (Thornton and Elledge 2018), runoff characteristics for the conservatively grazed pasture (Catchment 3) can be estimated had it remained brigalow scrub (Table 5). In 2015, conservatively grazed pasture generated 65 times more total runoff and 13 times greater peak runoff than uncleared estimates for this catchment. As no runoff occurred from the brigalow scrub catchment (Catchment 1) in 2016 and 2017, there would have been no runoff from Catchment 3 in an uncleared state. Total runoff and peak runoff from the brigalow scrub and conservatively grazed pasture catchments were similar in both 2018 and 2019 (Table 4), which means that there was negligible difference between observed and estimated uncleared runoff from the conservatively grazed catchment in that year (Table 5).

*Table 4: Observed annual hydrological year summaries of runoff and sampling effort for three catchments. * Runoff in 2019 is an incomplete hydrological year from October 2018 to March 2019 only.*

Parameter	Year	Brigalow scrub	Conservative grazing	Heavy grazing
Number of events	2015	1	2	2
	2016	0	1	1
	2017	0	0	0
	2018	1	1	2
	2019 *	1	1	1
Number of samples	2015	0	3	21
	2016	0	2	6
	2017	0	0	0
	2018	0	0	4
	2019 *	7	7	12
Total runoff (mm)	2015	0.2	13	28
	2016	0	8	28
	2017	0	0	0
	2018	0.1	0.1	19
	2019 *	17	16	49
Average peak runoff rate (mm/hr)	2015	0.1	2.6	6.4
	2016	0	1.0	2.6
	2017	0	0	0
	2018	0.1	0.1	2.6
	2019 *	7	6	20
Maximum peak runoff rate (mm/hr)	2015	0.1	3.1	6.5
	2016	0	1.0	2.6
	2017	0	0	0
	2018	0.1	0.1	4.7
	2019 *	7	6	20

*Table 5: Predicted annual hydrological year summaries of runoff from the conservatively grazed pasture catchment had it remained uncleared brigalow scrub. * Runoff in 2019 is an incomplete hydrological year from October 2018 to March 2019 only.*

Parameter	Year	Conservative grazing
Estimated uncleared runoff (mm)	2015	0.2
	2016	0
	2017	0
	2018	0.1
	2019 *	12
Increase in runoff under pasture (mm)	2015	12
	2016	8
	2017	0
	2018	0
	2019 *	3
Estimated uncleared average peak runoff rate (mm/hr)	2015	0.2
	2016	0
	2017	0
	2018	0.4
	2019 *	6
Increase in average peak runoff rate under pasture (mm/hr)	2015	2.4
	2016	1.0
	2017	0
	2018	0
	2019 *	0

3.2 Water Quality

Loads and EMCs of total suspended solids, nitrogen and phosphorus for the first six months of the 2019 hydrological year are presented in Table 6. Results for the 2015 to 2018 hydrological years have previously been presented in Appendix 2 of Thornton and Elledge (2018). There was no runoff, and hence no water quality from any catchment in 2017.

Loads of total suspended solids and all nitrogen and phosphorus parameters from heavily grazed pasture were between 1.4 and 3.7 times greater than from conservatively grazed pasture from 2015 to 2018. During 2019, loads of total suspended solids, particulate nitrogen and all phosphorus parameters from heavily grazed pasture were 1.0 to 2.3 times greater than conservatively grazed pasture. In contrast, loads of total and dissolved nitrogen were lower from heavily grazed pasture. EMCs were consistently lower from heavily grazed pasture, being only 30% to 90% of that from conservatively grazed pasture from 2015 to 2018 and only 13% to 68% for 2019.

In the four hydrological years (2015 to 2018) previously reported, loads of all water quality parameters from brigalow scrub were negligible due to no runoff in two of the four years, and less than 0.2 mm of runoff in the other two years. Consequently, no water quality samples were collected from this catchment and all data presented were estimations based on observed runoff and long-term EMCs. Using the hydrological calibration developed during Stage I (1965 to 1982) (Thornton and Elledge 2018), there would have been virtually no runoff from the conservatively grazed catchment in all four years had it remained brigalow scrub. Hence all loads of total suspended solids, nitrogen and phosphorus in runoff from the conservatively grazed catchment are an absolute anthropogenic increase attributable to changing land use from brigalow scrub to grazed pasture.

Table 6: 2019 hydrological year loads and event mean concentrations (EMCs) for total suspended solids, nitrogen and phosphorus in runoff.

	Parameter	Brigalow scrub	Conservative grazing	Heavy grazing
TSS	Total load (kg/ha/yr)	171	203	229
	Mean EMC (mg/L)	989	1,379	470
TN	Total load (kg/ha/yr)	5.36	1.46	1.37
	Mean EMC (mg/L)	31.07	9.91	2.81
PN	Total load (kg/ha/yr)	3.59	1.01	1.07
	Mean EMC (mg/L)	20.79	6.89	2.20
TDN	Total load (kg/ha/yr)	1.77	0.45	0.20
	Mean EMC (mg/L)	10.27	3.03	0.42
DON	Total load (kg/ha/yr)	0.36	0.18	0.08
	Mean EMC (mg/L)	2.10	1.24	0.17
DIN	Total load (kg/ha/yr)	1.41	0.26	0.12
	Mean EMC (mg/L)	8.17	1.79	0.25
TP	Total load (kg/ha/yr)	0.41	0.23	0.32
	Mean EMC (mg/L)	2.37	1.60	0.65
PP	Total load (kg/ha/yr)	0.38	0.20	0.21
	Mean EMC (mg/L)	2.22	1.39	0.43
TDP	Total load (kg/ha/yr)	0.03	0.03	0.06
	Mean EMC (mg/L)	0.16	0.21	0.13
DOP	Total load (kg/ha/yr)	0.00	0.00	0.01
	Mean EMC (mg/L)	0.03	0.03	0.02
DIP	Total load (kg/ha/yr)	0.02	0.03	0.06
	Mean EMC (mg/L)	0.13	0.18	0.12

3.2.1 Total Suspended Solids

Mean annual load of total suspended solids from the heavily grazed pasture was 3.2 times greater than from the conservatively grazed pasture for 2015 to 2018, but was only 1.1 times greater in 2019 (Figure 4). Overall, 2019 loads were 401 times greater from brigalow scrub, 14 times greater from conservatively grazed pasture, and 5 times greater from heavily grazed pasture compared to the four years previously reported. Mean annual EMC for total suspended solids was considerably greater in 2019 compared to the previous four years for both the conservatively grazed (278 mg/L and 1,379 mg/L, respectively) and heavily grazed pastures (235 mg/L and 470 mg/L, respectively).

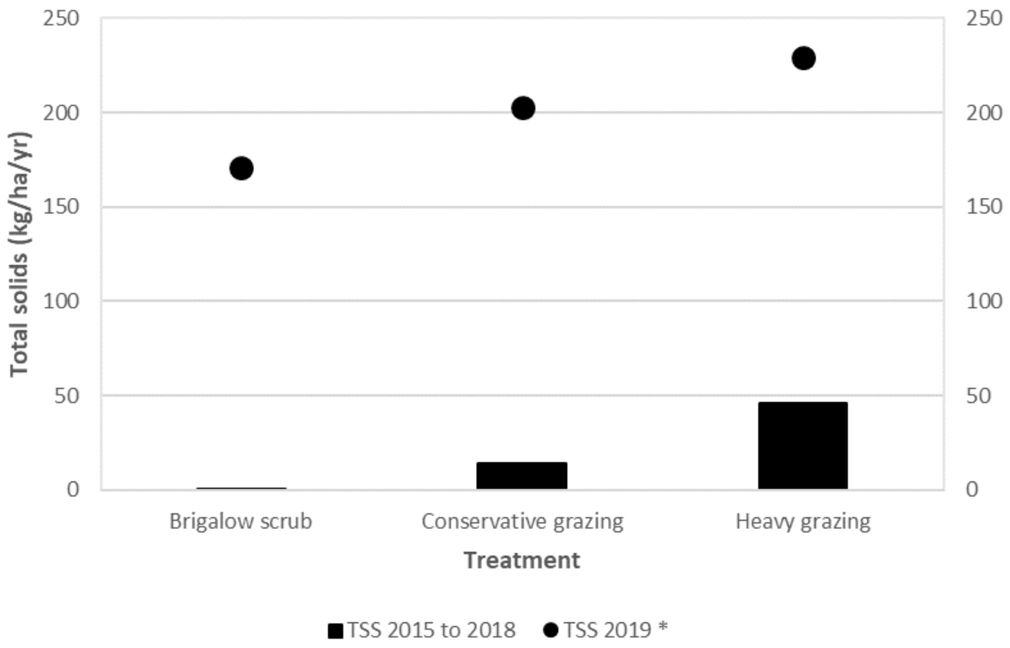

Figure 4: Mean annual load of total suspended solids (TSS) in runoff from 2015 to 2018 compared to the first six months of 2019 (incomplete hydrological year).*

3.2.2 Nitrogen

Mean annual load of total nitrogen from the heavily grazed pasture was 1.6 times greater than from the conservatively grazed pasture for 2015 to 2018, but was only 90% of the load in 2019 (Figure 5). Overall, 2019 loads were 496 times greater from brigalow scrub, 5 times greater from conservatively grazed pasture, and 3 times greater from heavily grazed pasture compared to the four years previously reported. For 2015 to 2018, the dominant pathway of nitrogen loss was in a dissolved form from brigalow scrub (based on estimates of limited data) with no clear trend for the two pasture catchments (Table 7). In contrast, particulate nitrogen was the dominant pathway of loss in 2019 from all three catchments. Mean annual EMCs of total nitrogen from the heavily grazed pasture were similar for the 2015 to 2018 (2.4 mg/L) and the 2019 (2.8 mg/L) reporting periods. In contrast, the conservatively grazed pasture had a higher EMC in 2019 (9.9 mg/L) than for 2015 to 2018 (6.5 mg/L). Sufficient runoff to calculate an EMC from brigalow scrub only occurred in 2019 (31.1 mg/L), which was at least 3.1 times greater than both pasture grazing pressures.

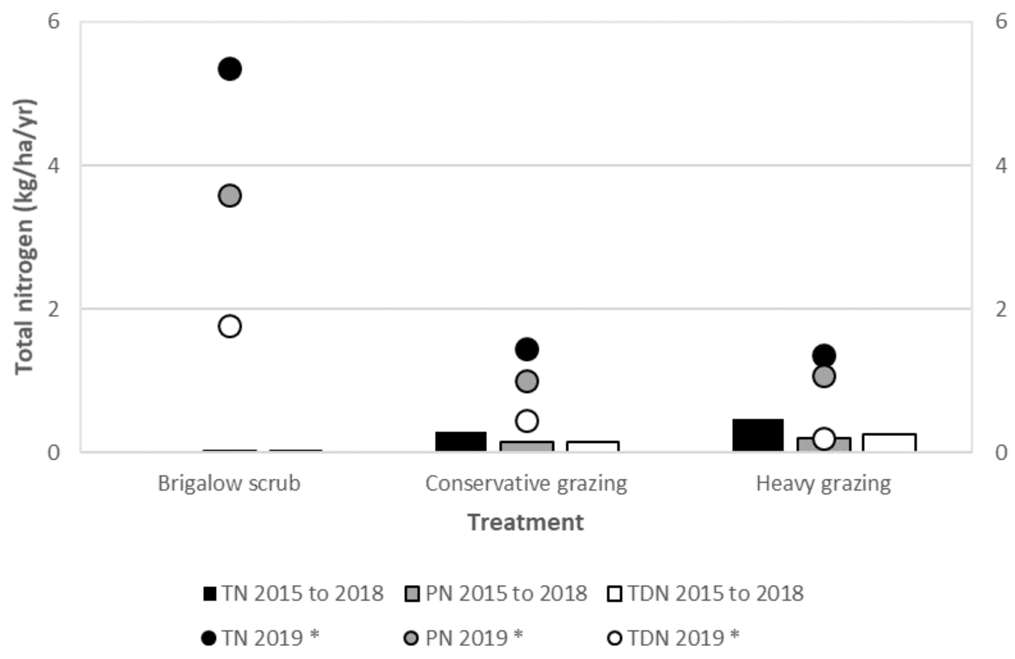

Figure 5: Mean annual loads of total nitrogen (TN), particulate nitrogen (PN) and total dissolved nitrogen (TDN) in runoff from 2015 to 2018 compared to the first six months of 2019 (incomplete hydrological year).*

Table 7: Dominant pathway of nitrogen loss in runoff from 2015 to 2019.

Year	Brigalow scrub	Conservative grazing	Heavy grazing
2015	Dissolved	No dominant	No dominant
2016	No runoff	No dominant	Dissolved
2017	No runoff	No runoff	No runoff
2018	Dissolved	Dissolved	Particulate
2019	Particulate	Particulate	Particulate

Mean annual load of total dissolved nitrogen from the heavily grazed pasture was 1.7 times greater than from the conservatively grazed pasture for 2015 to 2018, but was only 50% of the load in 2019 (Figure 6). Overall, 2019 loads were 257 times greater from brigalow scrub and 3 times greater from conservatively grazed pasture compared to the four years previously reported, while heavily grazed pasture had only 80% of the previously reported mean annual load. Organic and inorganic fractions generally contributed similar amounts towards total dissolved nitrogen from the two pasture catchments. Although there was limited data from brigalow scrub, estimations indicate a greater contribution of dissolved inorganic nitrogen towards total dissolved nitrogen. Mean annual EMCs of total dissolved nitrogen from the conservatively grazed pasture were similar for the 2015 to 2018 (3.1 mg/L) and the 2019 (3.0 mg/L) reporting periods. In contrast, the heavily grazed pasture had a lower EMC in 2019 (0.4 mg/L) than for 2015 to 2018 (1.3 mg/L). Sufficient runoff to calculate an EMC from brigalow scrub only occurred in 2019 (10.3 mg/L), which was at least 3.4 times greater than both pastures.

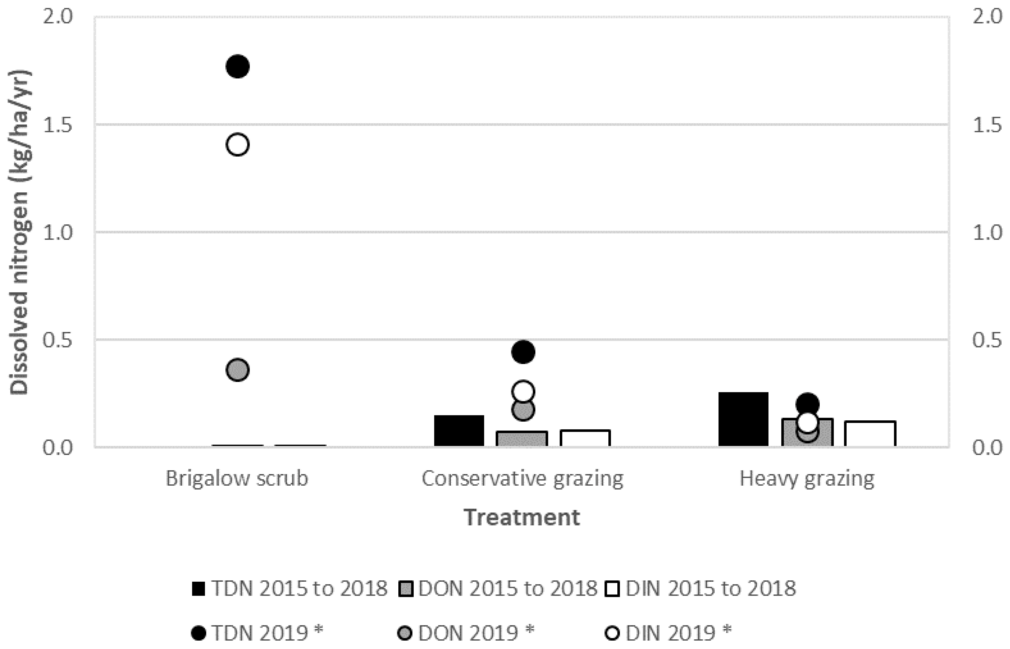

Figure 6: Mean annual loads of total dissolved nitrogen (TDN), dissolved organic nitrogen (DON) and dissolved inorganic nitrogen (DIN) in runoff from 2015 to 2018 compared to the first six months of 2019 (incomplete hydrological year).*

3.2.3 Phosphorus

Mean annual load of total phosphorus from the heavily grazed pasture was 2.6 times greater than from the conservatively grazed pasture for 2015 to 2018, but was only 1.3 times greater in 2019 (Figure 7). Overall, 2019 loads were 755 times greater from brigalow scrub, 6 times greater from conservatively grazed pasture, and 3 times greater from heavily grazed pasture compared to the four years previously reported. For 2015 to 2018, the dominant pathway of phosphorus loss was in a particulate form from brigalow scrub (based on estimates of limited data) with no clear trend for the two pasture catchments (Table 8). In contrast, particulate phosphorus was the dominant pathway of loss from all three catchments in 2019. Mean annual EMCs of total phosphorus from the heavily grazed pasture were similar for the 2015 to 2018 (0.5 mg/L) and the 2019 (0.6 mg/L) reporting periods. In contrast, the conservatively grazed pasture had a higher EMC in 2019 (1.6 mg/L) than for 2015 to 2018 (0.8 mg/L). Sufficient runoff to calculate an EMC from brigalow scrub only occurred in 2019 (2.4 mg/L), which was at least 1.5 times greater than both pastures.

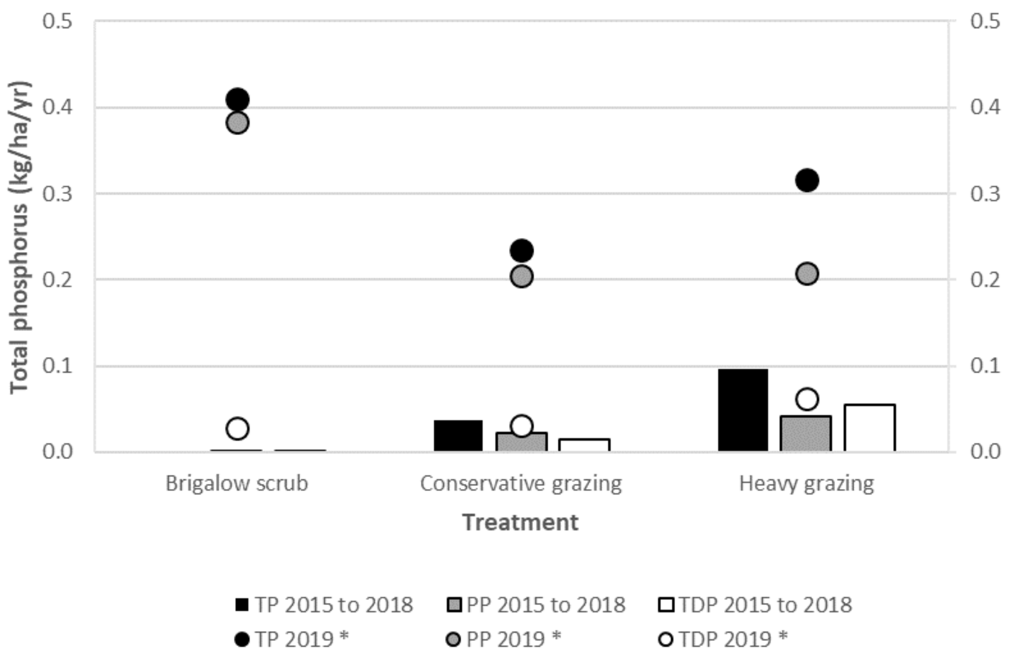

Figure 7: Mean annual loads of total phosphorus (TP), particulate phosphorus (PP) and total dissolved phosphorus (TDP) in runoff from 2015 to 2018 compared to the first six months of 2019 (incomplete hydrological year).*

Table 8: Dominant pathway of phosphorus loss in runoff from 2015 to 2019.

Year	Brigalow scrub	Conservative grazing	Heavy grazing
2015	Particulate	Particulate	No dominant
2016	No runoff	No dominant	Dissolved
2017	No runoff	No runoff	No runoff
2018	Particulate	No dominant	Particulate
2019	Particulate	Particulate	Particulate

Mean annual load of total dissolved phosphorus from the heavily grazed pasture was 3.6 times greater than from the conservatively grazed pasture for 2015 to 2018, and was 2.0 times greater in 2019 (Figure 8). Overall, 2019 loads were 2.0 times greater from conservatively grazed pasture and 1.1 times greater from heavily grazed pasture compared to the four years previously reported. Dissolved inorganic phosphorus was the greatest fraction of total dissolved phosphorus from all three catchments over all five years; on average accounting for 80% from brigalow scrub, 83% from conservatively grazed pasture and 88% from heavily grazed pasture. Mean annual EMCs of total dissolved phosphorus also showed limited variation from all catchments in all years (<0.4 mg/L).

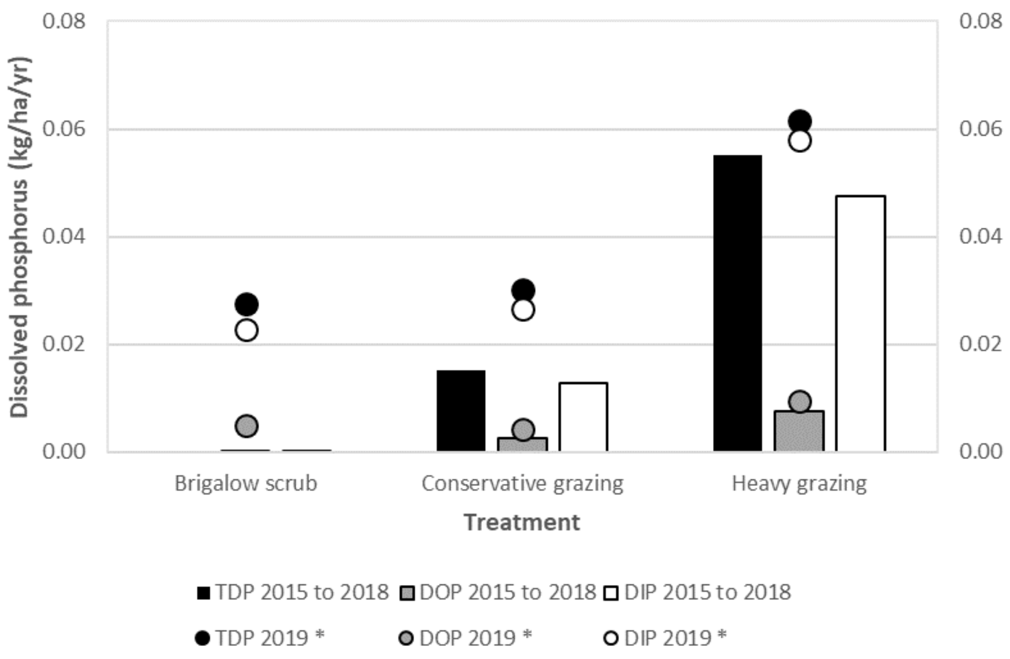

Figure 8: Mean annual loads of total dissolved phosphorus (TDP), dissolved organic phosphorus (DOP) and dissolved inorganic phosphorus (DIP) in runoff from 2015 to 2018 compared to the first six months of 2019 (incomplete hydrological year).*

3.3 Particle Size Distribution

3.3.1 Runoff

Water samples were collected throughout a runoff event from all five catchments in October 2018. Particle size distribution was similar between non-dispersed and mechanically dispersed methods, so only mechanical and ultrasonic results are presented in Figure 9. Similar trends were observed between these two methods, but ultrasonic dispersion typically resulted with a greater proportion of clay particles (<4 µm) than mechanical dispersion. Samples from brigalow scrub and conservatively grazed pasture typically had particles from all size classes. There was a general trend for the proportion of silt particles (4 to <63 µm) to increase and the proportion of clay particles (<4 µm) to decrease over time from these two catchments, but the trend was more evident from the conservatively grazed pasture. Runoff PSDs from the remaining three catchments which had minimal or no ground cover, due to either heavy grazing pressure or a bare fallow, were dominated by fine particles (<16 µm) with an increase in the proportion of clay particles over time. There was no linear or exponential correlation (*P*= 0.6 and *P*= 0.87, respectively) between loads of total suspended solids in runoff and the proportion of fine particles.

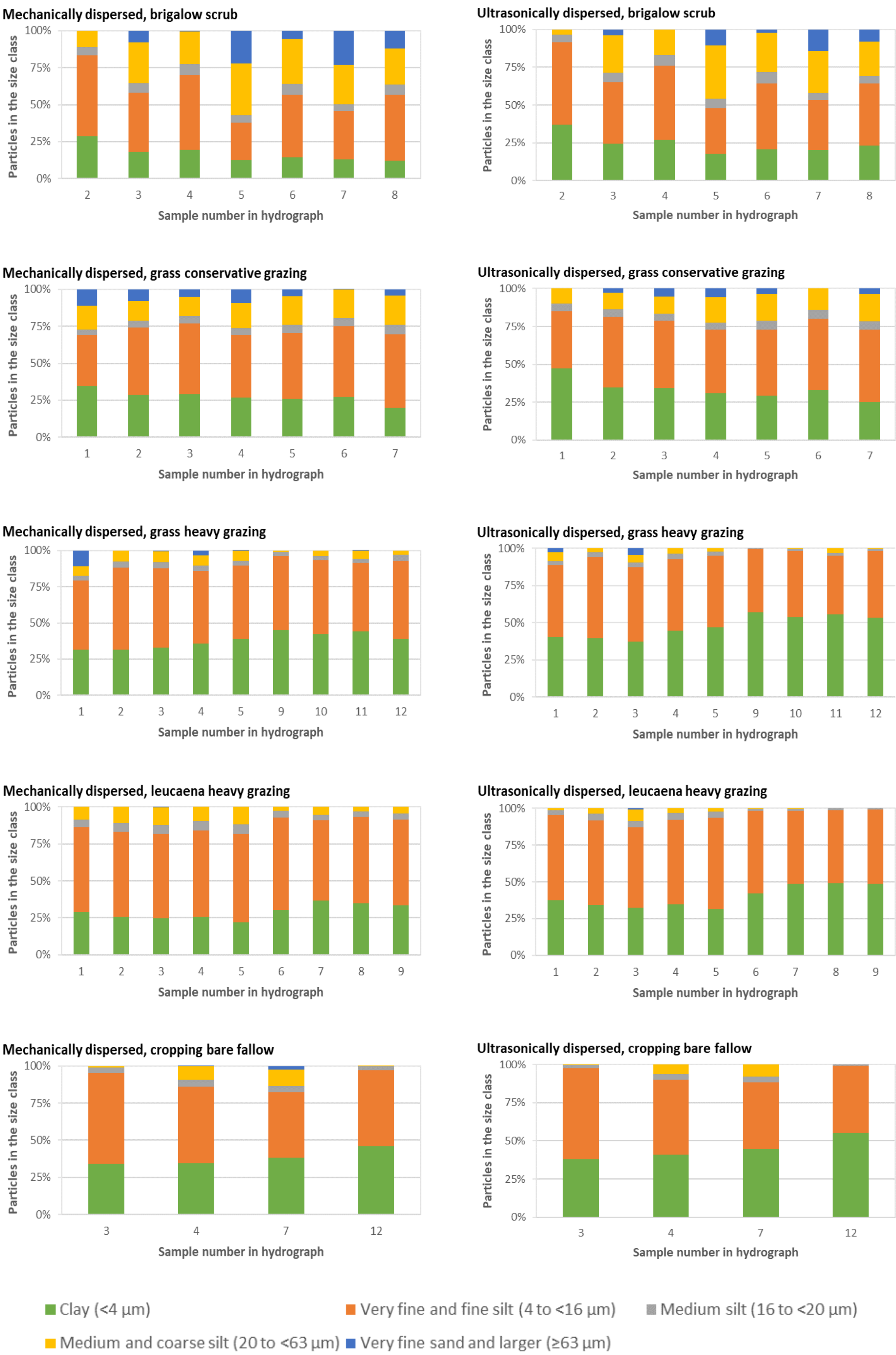

Figure 9: Particle size distributions of (as-received) runoff that were analysed by mechanical and ultrasonic dispersion methods after collection throughout events from five catchments.

3.3.2 Deposited Material

Overall, non-dispersed and mechanically dispersed samples that were dried and ground had a greater proportion of clay (<4 µm) and very fine and fine silt (4 to <16 µm) particles compared to samples analysed as-received (Figure 10). The proportion of these finer particles (<16 µm) was also greater from a deposit at the end of a grassed waterway compared to deposits immediately adjacent to the cropping bare fallow. Ultrasonically dispersed samples had less sand particles (≥63 µm) and more fine particles (<16 µm) compared to non-dispersed and mechanically dispersed methods, which was also a trend observed for samples analysed both as-received and after being dried and ground.

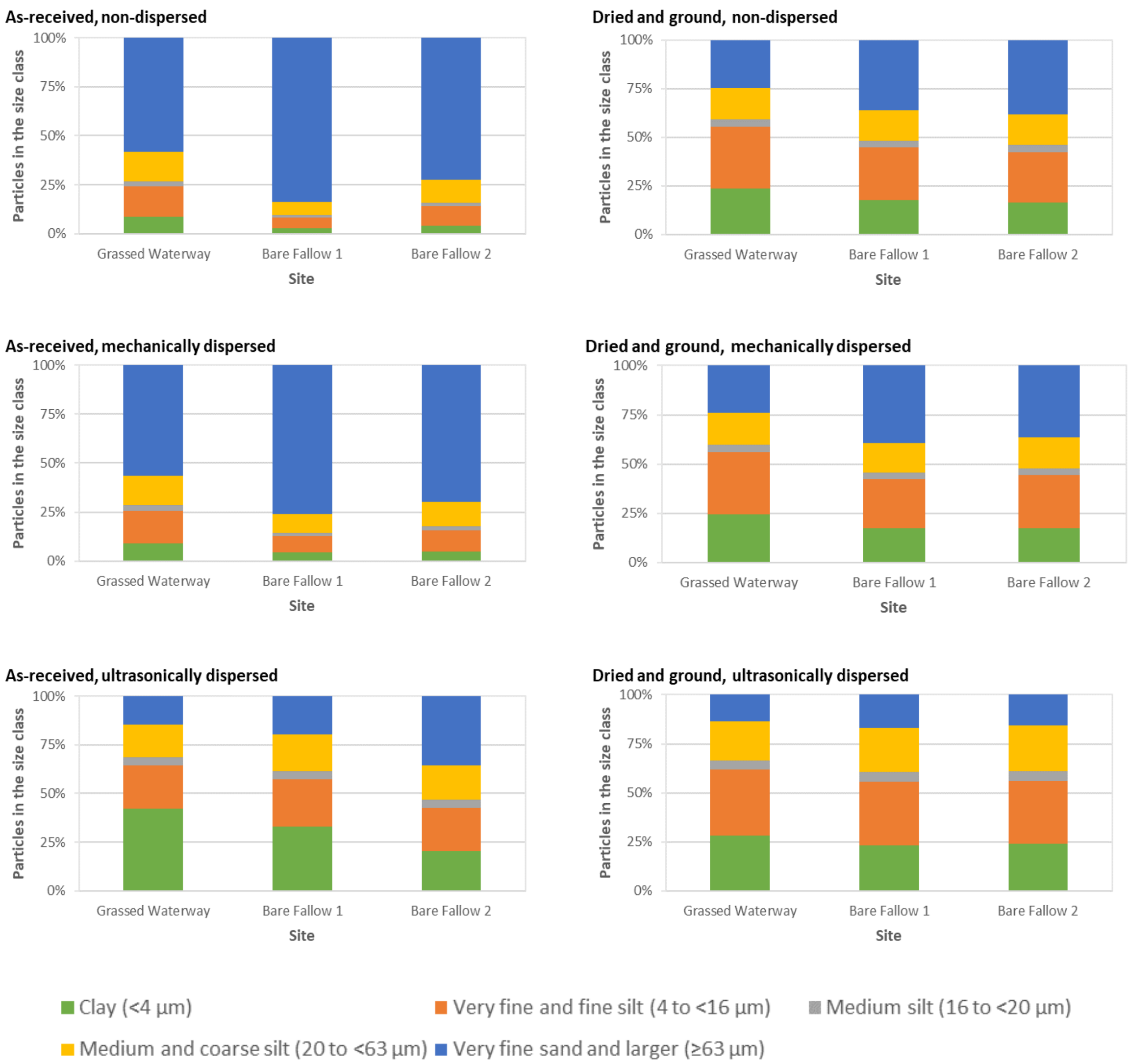

Figure 10: Particle size distributions of deposited material collected from three sites within the cropping bare fallow that were analysed both as-received and after drying and grinding for three dispersion methods.

When data from the three sites within the cropping bare fallow were averaged, the PSD of samples analysed both as-received and after drying and grinding were similar between non-dispersed and mechanically dispersed methods (Figure 11). However, dried and ground samples had at least 2.6 times more fine particles (<16 µm) than as-received samples for both dispersion methods. Similar to an earlier observation, ultrasonically dispersed samples had less sand particles (≥63 µm) and more fine particles (<16 µm) compared to non-dispersed and mechanically dispersed methods for samples analysed both as-received and after drying and grinding. Ultrasonic dispersion of as-received samples resulted in 3.0 times more clay (<4 µm) and very fine and fine silt (4 to <16 µm) particles compared to mechanical dispersion. This same trend was observed for dried and ground samples, although the magnitude of difference was lower with only 1.2 times more fine particles (<16 µm) by ultrasonic dispersion.

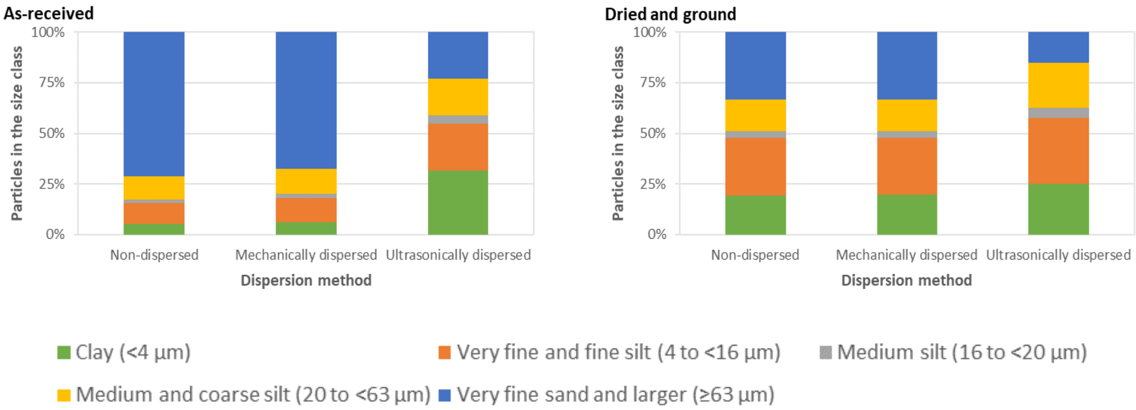

Figure 11: Particle size distributions of deposited material from the cropping bare fallow that were analysed both as-received and after drying and grinding for three dispersion methods.

4 Discussion

4.1 Hydrology

This study adds to the body of evidence from the BCS that shows clearing brigalow scrub for cropping or grazing increases total runoff and peak runoff rate (Thornton *et al.* 2007; Thornton and Yu 2016). In agreement with findings reported from 2015 to 2018, monitoring during 2019 continued to demonstrate that heavy grazing pressure further increases runoff and peak runoff rate compared to conservative grazing pressure (Thornton and Elledge 2019). Monitoring in 2019 also continued to illustrate the variability of rainfall, and subsequently runoff, that is characteristic of the semi-arid subtropical Brigalow Belt bioregion. Whilst total rainfall in the first six months of 2019 was low (4th percentile), the study experienced the second wettest October on record, accounting for 46% of total rainfall in the additional six months monitored. This rainfall generated a single runoff event from all five catchments, which yielded the highest mean annual and event based runoff from 2015 to 2019.

4.2 Water Quality

During the 2019 hydrological year, heavily grazed pasture had higher loads of total suspended solids and both total and dissolved phosphorus compared to conservatively grazed pasture. This is in agreement with previous observations of grazing land management and water quality reported for 2015 to 2018 (Thornton and Elledge 2019). In contrast, loads of total nitrogen and most dissolved nitrogen parameters were higher from conservatively than heavily grazed pasture. Both conservatively and heavily grazed pasture had higher loads of total suspended solids than brigalow scrub; however, brigalow scrub had the highest loads of total and dissolved nitrogen. This reflects the long-term water quality comparison between conservatively grazed pasture and brigalow scrub (Elledge and Thornton 2017).

Loads of total suspended solids, total nitrogen, dissolved inorganic nitrogen, total phosphorus and dissolved inorganic phosphorus from brigalow scrub, conservatively grazed pasture and heavily grazed pasture were all within the ranges previously reported for the individual catchments of the BCS (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014b; Thornton and Elledge 2019).

The observation that EMCs of all monitored parameters were lower from heavily than conservatively grazed pasture was also repeated in 2019. Lower EMCs are a result of increased runoff under higher grazing pressure which dilutes pollutants in runoff (Thornton and Elledge 2019). The complex interplay between runoff, load and EMC is highlighted by total suspended solids data from the three catchments. That is, runoff from brigalow scrub was similar to that of conservatively grazed pasture, while runoff was three times greater from heavily grazed pasture. Conversely, EMCs of total suspended solids from brigalow scrub and conservatively grazed pasture were two and three times greater than heavily grazed pasture, respectively. Nonetheless, loads of total suspended solids from all three catchments averaged 200 kg/ha (range 171 to 229 kg/ha). Clearly the observation that high EMCs do not necessarily equate to high loads continues to apply in below-average rainfall years, as it did in above average rainfall years (Thornton and Elledge 2013). This is reflected internationally, with loads typically correlated with flow rather than EMC (Water Environment Federation and the American Society of Civil Engineers 1998).

From 2015 to 2018, nitrogen lost in runoff from brigalow scrub was predominately in the dissolved form while phosphorus lost in runoff was predominately in the particulate form. In contrast, nitrogen and phosphorus was lost from both grazed pastures in particulate and dissolved forms. During 2019, particulate nitrogen and phosphorus were the dominant forms lost in runoff from all catchments. Although dissolved nitrogen was only a minor contribution to total nitrogen lost in runoff during 2019, dissolved nitrogen lost from brigalow scrub was predominantly dissolved inorganic nitrogen. In contrast, dissolved nitrogen lost from the two pasture catchments contained substantial proportions of both organic and inorganic nitrogen. This was in agreement with their behaviour from 2015 to 2018.

Storm flow is largely responsible for erosion and delivery of sediment to the end of catchments during large flood events (Waterhouse *et al.* 2017). Loads of total suspended solids from all catchments in the single 2019 runoff event exceeded the total cumulative load from 2015 to 2018. Loads of total nitrogen and phosphorus were equal to at least 74% of the cumulative load from 2015 to 2018. These high loads from an individual event compared to short-term mean annual loads clearly demonstrate that storm flow events can also dominate the loss of pollutants at the paddock scale. While acute loads of pollutants are noted at the end of catchment during large and infrequent events, chronic lower loads of anthropogenically-derived sediment and nutrients are lost at the paddock scale in drier years. These chronic loads are demonstrated by the occurrence of runoff from the two grazed pastures when the pre-European ecosystem would have yielded no runoff. The data shows poor grazing management leads to greater runoff and pollutant loads than well managed conservatively grazed pasture, which further exacerbates this trend. Furthermore, the dominant pathway of pollutant loss changed from both particulate and dissolved nutrients during small events for the below-average rainfall years of 2015 to 2018, to particulate dominated losses during a single large event in 2019. This highlights the need to understand not only the processes involved, but how the processes and pathways may vary as a result of climatic sequences. This interaction is a priority knowledge gap identified in the 2017 Scientific Consensus Statement (Waterhouse *et al.* 2017) and is clearly addressed by coupling the short-term data of this study with long-term data from the BCS.

4.3 Particle Size Distribution

4.3.1 Particle Size Distribution in Runoff

Determination of PSD in natural runoff from the five catchments in 2019 was a first for the BCS. It also appears to be the first published data from Australia that tracks PSD throughout the hydrograph at the paddock scale, and the first data providing a PSD comparison between native vegetation, cropping and grazing land uses, and grazing land management practices, as a result of a single storm event. As the catchments are contiguous, comprised of the same soils and subject to the same environmental conditions and the same rainfall, the PSDs can be wholly attributed to the treatment effects.

In this study, the land use and management interactions that resulted in high cover and biomass, that is brigalow scrub and conservatively grazed pasture, had the lowest proportion of fine particles (<16 µm) in runoff regardless of the dispersion method used. There was also a trend for the proportion of fine particles to decrease over time through the event. Decreasing erosion of fine particles with time is indicative of supply exhaustion (Durnford and King 1993), attributed to the ability of high cover and biomass to minimise aggregate disruption and generation of fine particles by rainfall and overland flow. Conversely, greater than 90% of the particles in runoff from bare fallow and heavily grazed pasture, with no and low cover and biomass respectively, were fine particles. These two treatments had a trend for fine particles less than 4 µm to increase over time through the event, attributed to the lack of cover resulting in aggregate disruption, generating a continuous source of fine particles (Loch and Donnollan 1983). The lack of correlation between loads of total suspended solids and fine particles in runoff from this study reflects findings from the Burdekin Basin. That is, the highest loads of fine particles, which are the most important from a land management and water quality perspective, are not necessarily derived from areas yielding the highest load of total suspended solids (Bainbridge *et al.* 2014).

The proportion of ultrasonically dispersed fine particles (<16 µm) from bare fallow and heavily grazed pasture in this study was 94%. This is the same as that reported at the end of catchment scale for the Fitzroy Basin (Garzon-Garcia *et al.* 2018), and similar to the 90% reported for the Upper Burdekin (Bainbridge *et al.* 2014). In contrast, the proportion of ultrasonically dispersed fine particles from brigalow scrub and conservatively grazed pasture was 72%. The direct extrapolation of this data suggests that the end of catchment PSD for the Fitzroy Basin is indicative of a low cover and biomass landscape. As the Fitzroy Basin is clearly not a low cover and biomass landscape (The State of Queensland 2017), the end of catchment PSD indicates the preferential instream transport and/or enrichment of the fine particle fraction.

Although there was a clear land use and management practice effect on PSD in runoff, more data is required to improve confidence in these findings. Process understanding of why the lower cover and biomass treatments yielded more fine particles is essential to inform options for improving grazing land management. If measured declines in soil organic carbon with land use change and increased grazing pressure (Dalal *et al.* 2011; Thornton and Shrestha Unpublished, Appendix 1.3) are resulting in decreased aggregate stability, and hence more fine particles, then basic grazing management principles, such as stocking to safe long-term carrying capacity as described in Thornton and Elledge (2018, Appendix 1.1), are likely to improve water quality. However, if higher proportions of fine particles in runoff are a legacy of decades of soil chemical and structural change, such as the doubling of fine sediment in runoff from simulated rainfall at the long-term cropping catchment of the BCS compared to the long-term conservatively grazed pasture (Eyles *et al.* 2018), then more complex management intervention strategies are likely necessary.

4.3.2 Particle Size Distribution in Deposited Material

Determination of PSD of deposited material eroded from the long-term cropping catchment was also a first for the BCS. Runoff occurred when the catchment was in bare fallow which lead to substantial erosion. The 988 kg/ha of total suspended solids lost in runoff was 1.9 times the long-term annual average of 525 kg/ha (Elledge and Thornton 2017), so the resultant PSD should represent that for high rates of hillslope erosion in this landscape. The PSD of deposited material was dependent on sample preparation and dispersion method. Preparation by drying and grinding increased the proportion of all size classes smaller than sand (< 63 µm) compared to as-received samples. Samples subjected to the same preparation had similar non-dispersed and mechanically dispersed PSDs. Ultrasonic dispersion of both as-received and dried and ground deposited material increased the proportion of all size classes smaller than sand compared to the other dispersion methods.

Sediment enrichment ratios, being the ratio of fine particles in runoff compared to deposited material, were greatest from mechanically dispersed as-received samples, with a ratio of 5. This is similar to the enrichment ratio of 4.8 for the less than 20 µm fraction of runoff from a bare Vertosol used for cropping in the nearby Nogoa subcatchment of the Fitzroy Basin (Silburn and Glanville 2002). Enrichment ratios for ultrasonically dispersed as-received samples and both mechanically and ultrasonically dispersed dried and ground samples were similar (range 1.6 to 1.9). These enrichment ratios were all greater than the ratio of 1.1 previously reported for the long-term cropping catchment when comparing proportions of the ultrasonically dispersed less than 20 µm fraction contained in runoff generated from simulated rainfall, with the same size fraction of the surface soil (Eyles *et al.* 2018).

While three dispersion methods were utilised to determine PSD of soil in runoff and deposited material, it is acknowledged that none of them are likely to accurately reflect in-situ field conditions (Garzon-Garcia *et al.* 2018). Non-dispersed and mechanically dispersed PSDs may give some indication of the likely PSD of naturally aggregated particles in runoff. Ultrasonically dispersed PSDs approximate the true distribution of the absolute particle size and is the only method available that can be assumed to give comparable results across samples and studies (Garzon-Garcia *et al.* 2018). Despite similarities between the PSDs of runoff in this study and that reported for the Fitzroy Basin by Garzon-Garcia *et al.* (2018), and also similarities between fine particle enrichment ratios in this study compared to those reported by Eyles *et al.* (2018) using simulated rainfall at this site, the data only represents a single point in time. Ongoing monitoring will be essential to improve confidence in these findings.

4.4 Improving Grazing Management to Benefit Water Quality

Monitoring of hydrology, water quality, ground cover and pasture biomass from 2015 to 2018 by Thornton and Elledge (2018) concluded that 3.4 ha/AE is a safe long-term carrying capacity for rundown (30 to 40 years old) buffel grass pasture established on predominantly clay soils previously dominated by brigalow scrub. Failure to reduce stocking rates on rundown pastures to match the safe long-term carrying capacity increased runoff, and subsequently increased loads of total suspended solids, nitrogen and phosphorus in runoff. Although there was limited water quality data collected during these four below-average rainfall years, both total nitrogen and phosphorus loads had substantial contributions of particulate and dissolved fractions in both the conservatively and heavily grazed pastures.

An additional six months monitoring of hydrology and water quality was undertaken in 2019. In agreement with earlier findings, heavily grazed pasture had the highest runoff and highest loads of total suspended solids, particulate nitrogen and all phosphorus parameters compared to conservatively grazed pasture. Event mean concentrations continued to be lower from heavily grazed pasture compared to conservatively grazed pasture. In contrast with earlier findings, extreme rainfall resulted in particulate nitrogen and phosphorus being the dominant pathway of loss from brigalow scrub, conservatively grazed pasture and heavily grazed pasture. Particle size distribution in runoff was measured for the first time at the Brigalow Catchment Study in this period. Conservatively grazed pasture had the lowest proportion of fine particles less than 16 µm in runoff and exhibited supply exhaustion. Conversely, greater than 90% of the particles in runoff from heavily grazed pasture were fine particles less than 16 µm with no evidence of supply exhaustion. These findings support the earlier conclusion that conservative grazing pressure is a realistic option for landholders to improve runoff water quality.

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Appendix 1: Publications

Technical Reports

The current report is an addendum to a Paddock to Reef program technical report published in 2018:

(1) Thornton C. M. and Elledge A. E. (2018). Paddock scale water quality monitoring of grazing management practices in the Fitzroy Basin: Technical report on the effect of grazing pressure on water quality for the 2015 to 2018 hydrological years. Report to the Paddock to Reef Program. Department of Natural Resources, Mines and Energy, Rockhampton. [**Appendix 1.1**]

Journal Papers

Three journal papers that used BCS data were published during the funded period:

(1) Elledge A. and Thornton C. (2017). Effect of changing land use from virgin brigalow (*Acacia harpophylla*) woodland to a crop or pasture system on sediment, nitrogen and phosphorus in runoff over 25 years in subtropical Australia. *Agriculture, Ecosystems and Environment* **239**, pp. 119-131. [**Appendix 1.2**]
(2) Thornton C. and Elledge A. (2016). Tebuthiuron movement via leaching and runoff from grazed Vertisol and Alfisol soils in the Brigalow Belt bioregion of central Queensland, Australia. *Journal of Agricultural and Food Chemistry* **64** (20), pp. 3949-3959.
(3) Thornton C. M. and Yu B. (2016). The Brigalow Catchment Study: IV. Clearing brigalow (*Acacia harpophylla*) for cropping or grazing increases peak runoff rate. *Soil Research* **54** (6), pp. 749-759.

Two additional journal papers that used BCS data were prepared during the funded period. They have both received approval from the Department of Natural Resources, Mines and Energy to release externally, but are pending submission to the journal of Soil Research:

(1) Thornton C. and Shrestha K. (Unpublished). The Brigalow Catchment Study: VI. Clearing and burning brigalow (*Acacia harpophylla*) in Queensland, Australia, temporarily increases surface soil fertility prior to nutrient decline under cropping or grazing. [**Appendix 1.3**]

(2) Thornton C. and Yu B. (Unpublished). The Brigalow Catchment Study: V. A comparison of four methods to estimate peak runoff rate for small catchments before and after land use change in the Brigalow Belt bioregion of central Queensland, Australia. [**Appendix 1.4**]

Conference Papers and Presentations

Eight seminars that used BCS data were presented at conferences and workshops during the funded period:

(1) Elledge A. and Thornton C. (2018). The Brigalow Catchment Study: The legacy of land clearing and European agriculture in the Brigalow Belt. *Central region vegetation management team meeting*, Department of Natural Resources, Mines and Energy, Rockhampton.

(2) Elledge A. E. and Thornton C. M. (2018). The Brigalow Catchment Study: The impacts of developing *Acacia harpophylla* woodland for cropping or grazing on hydrology, soil fertility and water quality in the Brigalow Belt bioregion of Australia. *Natural resource science in action: Connecting people, science and purpose*, Toowoomba.

(3) Thornton C. and Elledge A. (2018). The Brigalow Catchment Study: The legacy of land clearing and European agriculture in the Brigalow Belt. *Fitzroy Basin Association annual general meeting*, Rockhampton.

(4) Thornton C. and Elledge A. (2018). The Brigalow Catchment Study: The legacy of land clearing and European agriculture in the Brigalow Belt. *Department of Environment and Science central region compliance team meeting*, Rockhampton.

(5) Thornton C. and Elledge A. (2019). The Brigalow Catchment Study: The legacy of land clearing and European agriculture in the Brigalow Belt. *Fitzroy Basin Association regional science forum on Paddock to Reef*, Rockhampton.

(6) Thornton C. and Elledge A. (2019). The Brigalow Catchment Study: The legacy of land clearing and European agriculture in the Brigalow Belt. *Department of Agriculture and Fisheries economist team meeting*, Rockhampton.

(7) Thornton C. M. and Elledge A. E. (2018). The Brigalow Catchment Study: The impacts of developing *Acacia harpophylla* woodland for cropping or grazing on hydrology, soil fertility and water quality in the Brigalow Belt bioregion of Australia. Occasional Report No. 31. *Farm environmental planning – Science, policy and practice*, Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand. pp. 1-8.

(8) Thornton C., Elledge A., Shrestha K., Wallace S., Bosomworth B. and Yu B. (2017). The Brigalow Catchment Study: The impacts of developing *Acacia harpophylla* woodland for cropping or grazing on hydrology, soil fertility and water quality in the Brigalow Belt bioregion of Australia. *International interdisciplinary conference on land use and water quality: Effect of agriculture on the environment*, The Hague, Netherlands.

Brigalow Catchment Study Field Tours

Five field tours of the BCS were conducted during the funded period:

(1) Afshin Ghahramani (Agricultural Systems Modeller) from the University of Southern Queensland, Bofu Yu (Head of Environmental Engineering) from Griffith University, Marianna Joo (Water Planning Ecologist) from the Department of Environment and Science, and students from both universities visited to discuss the long-term study and establish a collaboration to improve the soil erosion component of the HOWLEAKY model (May 2018).
(2) Dipaka Sena (Environmental Engineer) from the Indian Institute of Soil and Water Conservation had a research fellowship with the University of Southern Queensland to improve erosion modelling used by the Paddock to Reef program (September 2018).
(3) Mandy Downs (Executive Director of Operations Support for Natural Resources and Science Champion) and Darren Moor (Executive Director of Central Region and Water Champion) from the Department of Natural Resources, Mines and Energy to provide awareness of the project to higher level managers.
(4) Review panel for the paddock and catchment modelling components of the Paddock to Reef program. Attendees included Daren Harmel (Director for Agricultural Resources Research) and Tim Green (Agricultural Engineer) from the United States Department of Agriculture, Paul Lawrence (Executive Director for Science Delivery and Knowledge and Chair of the Queensland Water Modelling Network) from the Department of Environment and Science, and the Paddock to Reef program modellers David Waters, Mark Silburn, Shawn Darr and Cameron Dougall (April 2019).
(5) Jon Duncan (Hydrologist) from Pennsylvania State University and Anna Lintern (Civil Engineer) from Monash University s visited to discuss the management of diffuse pollution from the long-term BCS in Australia compared to the long-term Chesapeake Bay Program in the United States of America, which was used to design the Paddock to Reef program (June 2019).

Website

A portal for the BCS (www.brigalowcatchmentstudy.com) was developed during the funded period which provides access to rainfall and runoff data from all five monitored catchments, in addition to information on publications that have resulted from the long-term BCS.

Appendix 1.1: Thornton and Elledge (2018)

Department of Natural Resources, Mines and Energy

Paddock scale water quality monitoring of grazing management practices in the Fitzroy Basin

Technical report on the effect of grazing pressure on water quality for the 2015 to 2018 hydrological years



Craig M Thornton and Amanda E Elledge

QUEENSLAND GOVERNMENT

Australian Government
Department of the Environment and Energy

Supported by the Australian and Queensland Government's Paddock to Reef Program

This publication has been compiled by Land and Water Science, Department of Natural Resources, Mines and Energy, Rockhampton.

This report should be cited as:

Thornton C. M. and Elledge A. E. (2018). Paddock scale water quality monitoring of grazing management practices in the Fitzroy Basin: Technical report on the effect of grazing pressure on water quality for the 2015 to 2018 hydrological years. Report to the Paddock to Reef Program. Department of Natural Resources, Mines and Energy, Rockhampton.

Cover photographs: cattle in the heavily grazed pasture catchment (left); runoff event through a monitoring flume (centre); and a fenceline comparison of conservatively and heavily grazed pastures (right). All photographs are sourced from the Brigalow Catchment Study photo archives, courtesy of the Department of Natural Resources, Mines and Energy.

This report is available from the Brigalow Catchment Study website www.brigalowcatchmentstudy.com.

Executive Summary

Loss of sediment, particulate nitrogen and particulate phosphorus in runoff from the extensive grazing lands of the Fitzroy Basin, central Queensland, continue to contribute to the declining health of the Great Barrier Reef. Substantial investment has been made by the Australian and Queensland Governments to improve runoff water quality from grazing lands; however, there is little data directly comparing the effect of grazing pressure on hydrology and water quality. This is further confounded by the difficulty of separating the impacts of climate variability from the anthropogenic impacts of changing land use from native vegetation to grazing. This study measured changes in hydrology, water quality, ground cover and pasture biomass from conservative and heavy cattle grazing pressures on rundown (>30 years old) improved grass pastures. It also considered the anthropogenic effect of changing land use from brigalow scrub to an improved grass pasture with a conservative grazing pressure. The paddock-scale (12.0 to 16.8 ha) study was conducted at the long-term Brigalow Catchment Study, located in the Fitzroy Basin of central Queensland, Australia.

Conservative grazing pressure averaged 5.9 ha/AE, which was a lighter stocking rate than the calculated safe long-term carrying capacity of 3.4 ha/AE for the rundown pasture. This was due to below average rainfall which limited pasture growth over the four hydrological years of this study (October 2014 to September 2018). Mean annual rainfall at the study site ranged from 272 mm in 2017 to 584 mm in 2018, which was well below the long-term average of 648 mm. Heavy grazing pressure averaged 1.9 ha/AE, which reflected stocking rates recommended for newly established buffel grass pasture rather than for rundown pasture.

Heavy grazing resulted in 3.6 times more total runoff compared to conservative grazing (18.8 mm/yr cf. 5.2 mm/yr) and 3.3 times greater average peak runoff rate (2.9 mm/hr cf. 0.9 mm/hr). No runoff occurred from brigalow scrub in two of the four years, which means that no runoff would have occurred from the conservatively grazed pasture had it remained uncleared. Runoff from the conservatively grazed pasture in these two years was an absolute anthropogenic increase attributable to land use change.

Runoff loads of total suspended solids and total, particulate and dissolved nitrogen and phosphorus were greater from the two grass pastures than from brigalow scrub, while loads from heavy grazing were greater than from conservative grazing. Heavy grazing resulted in 3.2 times greater load of total suspended solids than from conservative grazing (46 kg/ha/yr cf. 14 kg/ha/yr), 1.6 times greater load of total nitrogen (0.46 kg/ha/yr cf. 0.29 kg/ha/yr) and 2.6 times greater load of total phosphorus (0.10 kg/ha/yr cf. 0.04 kg/ha/yr). Total nitrogen and phosphorus loads from grass pastures had substantial contributions of both particulate and dissolved fractions regardless of grazing pressure, and the dominant fraction varied between years. Particulate and dissolved loads of nitrogen and phosphorus from heavily grazed pasture were between 1.4 and 3.7 times greater than from conservatively grazed pasture. In the two years with no runoff from brigalow scrub, water quality loads from the conservatively grazed pasture were also an absolute anthropogenic increase. In contrast to loads, event mean concentrations for all water quality parameters were lower from heavy than conservative grazing due to the dilution effect of increased runoff.

At the commencement of this study, the conservatively and heavily grazed pastures started in a similar condition with a comparable proportion of bare ground (12.3% cf. 13.4%) and pasture biomass (6.9 t/ha cf. 6.2 t/ha). After four below average rainfall years, heavy grazing of rundown pasture resulted in 2.5 times more bare ground than the conservatively grazed pasture (14.9% cf. 5.9%) and only 8% of the pasture biomass (0.4 t/ha cf. 5.3 t/ha).

Thornton and Elledge 2018

A safe long-term carrying capacity for rundown buffel grass pasture established on predominantly clay soils, previously dominated by brigalow scrub, was 3.4 ha/AE. Exceeding the safe long-term carrying capacity during this four year study increased runoff and subsequently increased loads of total suspended solids in runoff. Loads of total, particulate and dissolved nitrogen and phosphorus in runoff also increased under heavy grazing pressure. Ground cover and pasture biomass are both indicators of land condition and decreased under heavy grazing pressure. This study compliments other research that has reported improved land condition and reduced economic risk after transitioning from heavy to conservative grazing pressure. Thus, conservative grazing pressure is a realistic option for landholders to improve land condition, business profitability and runoff water quality.

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List of Units

AE/ha/yr	Adult equivalent per hectare per year
cf.	Confer or compare with
days/yr	Days per year
ha	Hectare
ha/AE	Hectare per adult equivalent
ha/AE/yr	Hectare per adult equivalent per year
ha/head	Hectare per head
kg	Kilogram
kg/ha	Kilogram per hectare
kg/ha/yr	Kilogram per hectare per year
kg/head	Kilogram per head
m	Metre
m^2	Square metre
mg/L	Milligram per litre
Mha	Million hectare
mm	Millimetre
mm/hr	Millimetres per hour
mm/yr	Millimetres per year
t/ha	Tonne per hectare

Abbreviations

AMC	Annual Mean Concentration
BCS	Brigalow Catchment Study
C1	Catchment 1; virgin brigalow scrub which is an ungrazed control
C3	Catchment 3; grass pasture with conservative grazing pressure
C5	Catchment 5; grass pasture with heavy grazing pressure
DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved Inorganic Phosphorus, also known as Filterable Reactive Phosphorus (FRP) and Orthophosphate (PO_4-P)
DON	Dissolved Organic Nitrogen
DOP	Dissolved Organic Phosphorus
EMC	Event Mean Concentration
NH_4-N	Ammonium-Nitrogen
NO_x-N	Oxidised Nitrogen
PN	Particulate Nitrogen, also known as Total Suspended Nitrogen (TSN)
PP	Particulate Phosphorus, also known as Total Suspended Phosphorus (TSP)
TDN	Total Dissolved Nitrogen
TDP	Total Dissolved Phosphorus
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids

Acknowledgments

This study was funded by the Australian and Queensland Government Paddock to Reef Program, and the Department of Natural Resources, Mines and Energy. The authors thank past and present staff from the Department of Natural Resources, Mines and Energy and the Queensland Department of Agriculture and Fisheries that contributed to the long-term Brigalow Catchment Study datasets that have been used in this report. Finally, we thank our industry collaborator Elrose Brahman Stud for their input into the study, and in particular Walter and Leicha Gleeson from Brigalow Station for their assistance with on-ground cattle operations.

1 Introduction

The 2017 scientific consensus statement on Great Barrier Reef water quality identified the Fitzroy Basin as a high priority area for reducing fine sediment and particulate nutrients (Waterhouse *et al.* 2017). Grazing is the dominant land use in this region, with more than 2.6 million cattle over 11.1 Mha (Australian Bureau of Statistics 2009; Meat and Livestock Australia 2017). This is the largest cattle herd in any natural resource management region in both Queensland and Australia, accounting for 25% of the state herd and 11% of the national herd (Meat and Livestock Australia 2017). The 2016 Great Barrier Reef report card noted that only 29% of grazing in the Fitzroy Basin was under best management practices compared to the 90% target (The State of Queensland 2017a). Progress to reduce anthropogenic end-of-catchment loads for this region was classed as very poor due to reductions of only 9.6% for sediment, 4.7% for particulate nitrogen and 8.5% for particulate phosphorus compared to the 20% targets. This is despite greater reductions in sediment and particulate nutrients compared to the prior year, which was mainly achieved by excluding cattle from streambanks in high risk areas (The State of Queensland 2017b).

In contrast, the Burdekin Basin had sediment reductions of 17.7% which was attributed to management practices such as pasture budgeting to determine carrying capacity and the adoption of wet season spelling (The State of Queensland 2017b). These practices are commonly recommended to maintain or improve ground cover (Jones *et al.* 2016; Moravek *et al.* 2017; O'Reagain *et al.* 2011), as high cover is known to reduce runoff, and hence also sediment and nutrients exported in runoff (Murphy *et al.* 2008; Nelson *et al.* 1996; Schwarte *et al.* 2011; Silburn *et al.* 2011). For example, in the Burdekin Basin, O'Reagain *et al.* (2008) compared a light stocking rate which had 20 to 25% pasture utilisation to a heavy stocking rate which had 40 to 50% pasture utilisation. In below average rainfall years, the heavy stocking rate had less ground cover, a greater frequency and intensity of runoff, and higher sediment concentrations in runoff. However, there was little difference between the two stocking rates in high rainfall years due to high ground cover (O'Reagain *et al.* 2008).

Moravek *et al.* (2017) reviewed economic literature on grazing management practices and found that there is not always a win-win situation between business profitability and environmental outcomes, such as reduced sediment in runoff. This is possibly the reason that so few landholders use the recommended practices of reduced stocking rates and wet season spelling. For example, of the total area mainly used for grazing in Queensland, only 6% (7.4 Mha) is under tactical grazing which involves a range of management practices to meet various animal and pasture objectives (Australian Bureau of Statistics 2017). Furthermore, 25% of Queensland agricultural businesses that mainly used land for grazing did not spell pasture between grazing periods (Australian Bureau of Statistics 2017). Although spelling pasture has been shown to increase biomass, seasonal conditions can actually have a stronger effect on ground cover and pasture biomass (Jones *et al.* 2016). This further highlights the importance of managing grazing pressure to maintain landscape resilience, particularly during periods of below average rainfall (Edwards 2018).

This study provides more evidence for adopting the recommended management practices of a safe long-term carrying capacity and wet season spelling for improved water quality outcomes by:

(1) Quantifying the impact of conservative and heavy grazing pressure on ground cover, pasture biomass, hydrology, and both loads and event mean concentrations (EMCs) of total suspended solids, nitrogen and phosphorus in runoff over four hydrological years (2015 to 2018); and

(2) Determining the anthropogenic impact of grazing by comparing hydrology and both loads and event mean concentrations (EMCs) of total suspended solids, nitrogen and phosphorus in runoff from a conservatively grazed pasture to virgin brigalow scrub, which is representative of the pre-European landscape.

2 Methods

2.1 Site Description

The Brigalow Catchment Study (BCS) is a paired, calibrated catchment study located (24°48'S and 149°47'E) near Theodore in central Queensland, Australia. It was established in 1965 to quantify the impact of land development for agriculture on hydrology, productivity and resource condition (Cowie *et al.* 2007). The study site was selected to represent the Brigalow Belt bioregion which covers an area of approximately 36.7 Mha from Townsville in north Queensland to Dubbo in central-western New South Wales (Thornton *et al.* 2007) (Figure 1). In its native state, the site was dominated by brigalow (*Acacia harpophylla*), either in a monoculture or in association with other species, such as belah (*Casuarina cristata*) and Dawson River blackbutt (*Eucalyptus cambageana*) (Johnson 2004). The extant uncleared vegetation of the BCS is classified as regional ecosystems 11.4.8, woodland to open forest dominated by *Eucalyptus cambageana* and *Acacia harpophylla*, and 11.4.9, open forest and occasionally woodland dominated by *Acacia harpophylla* (Queensland Government 2014). Slope of the land averages 2.5% (1.8% to 3.5%) and soils are an association of Vertosols, Dermosols, Sodosols and Chromosols. These soil types are representative of 75% of the Fitzroy Basin under grazing: 28% Vertosols; 28% Sodosols; 11% Dermosols; and 8% Chromosols (Roots 2016). The region has a semi-arid, subtropical climate and mean annual hydrological year (October 1965 to September 2018) rainfall at the site was 648 mm.

2.2 Long-Term Brigalow Catchment Study

The BCS can be separated into four experimental phases: (1) calibration of three catchments in an uncleared state from 1965 to 1982; (2) development of two catchments for agriculture from 1982 to 1983; (3) comparison of cropping and grazing land use to virgin brigalow scrub from 1984 to 2010; and (4) a comparison of leguminous and non-leguminous pastures to virgin brigalow scrub during the adaptive land management phase from 2010 to 2014 (Table 1). The adaptive land management phase involved the transition of the cropping catchment into a grazed ley pasture to improve soil fertility, and the addition of two new catchments; a grazed leucaena-grass pasture and a heavily grazed grass pasture. This phase continued from 2015 to 2018, but with a focus on comparing improved grass pasture with conservative and heavy grazing pressures to virgin brigalow scrub.

The 18 year calibration period for the three long-term catchments in Stage I means that runoff characteristics from the original cropping and grazing catchments can be estimated had they remained brigalow scrub. A calibration period for the two new catchments was not possible as they had been developed for agriculture sometime between 1965 and 1969, which was 40 to 50 years prior to their inclusion in the study. Thus, although the two new catchments have their own unique hydrological characteristics, their relationship to the three long-term catchments in an uncleared state is unknown. Further details on these experimental phases are documented in other sources (Cowie *et al.* 2007; Radford *et al.* 2007; Thornton *et al.* 2007; Thornton and Elledge 2013).

Thornton and Elledge 2018

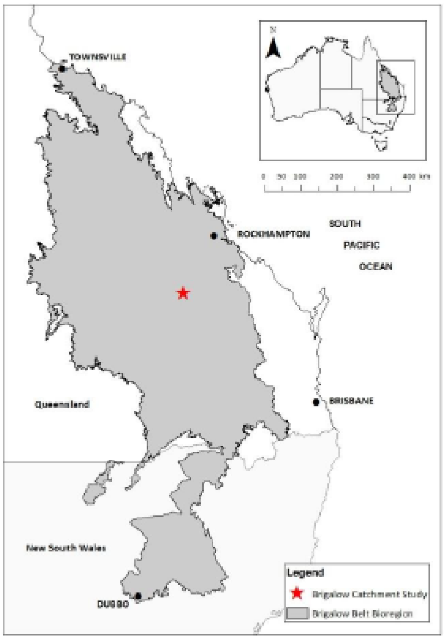

Figure 1: Location of the Brigalow Catchment Study within the Brigalow Belt bioregion of central Queensland.

Table 1: Land use history of the Brigalow Catchment Study.

Catchment	Land use by experimental stage			
	Stage I **Jan 1965 to Mar 1982**	**Stage II** **Mar 1982 to Sep 1983**	**Stage III** **Sep 1984 to Jan 2010**	**Stage IV** **Jan 2010 to Present (2018)**
C1	Brigalow scrub	Brigalow scrub	Brigalow scrub	Brigalow scrub
C2	Brigalow scrub	Development	Cropping	Ley pasture
C3	Brigalow scrub	Development	Grass pasture	Grass pasture
C4	NA	NA	NA	Leucaena pasture[1]
C5	NA	NA	NA	Grass pasture[2]

[1] *Monitoring in the C4 leucaena pasture commenced in 2009.*

[2] *Monitoring in the C5 grass pasture commenced in 2014.*

3

2.3 Treatments

Although all five catchments described above were continually monitored as part of the long-term BCS, this report only considers the conservatively grazed pasture (Catchment 3), the heavily grazed pasture (Catchment 5) and the brigalow scrub (Catchment 1) land uses (Figure 2; Table 2). The period of reporting is from the adaptive land management phase for the 2015 to 2018 hydrological years (October 2014 to September 2018). All references to years are based on hydrological years.

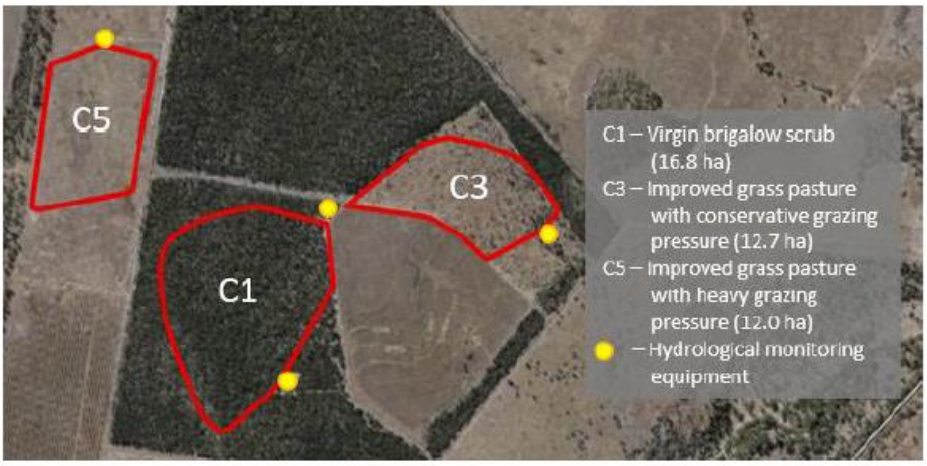

Figure 2: Aerial photo of the Brigalow Catchment Study showing the hydrological (runoff) boundaries and location of monitoring equipment within the three catchments.

The brigalow scrub catchment was retained in its virgin uncleared condition. This was an ungrazed control treatment representative of the Brigalow Belt bioregion in its pre-European condition. This catchment has Vertosols and Dermosols (clay soils) covering approximately 70% of the hydrological area and Sodosols over the remaining 30% (Cowie *et al.* 2007). The conservatively grazed catchment was a buffel grass (*Cenchrus ciliaris* cv. Biloela) pasture. This catchment has Vertosols and Dermosols (clay soils) covering approximately 58% of the hydrological area and Sodosols over the remaining 42% (Cowie *et al.* 2007). The heavily grazed catchment was a purple pigeon grass (*Setaria incrassata*) and buffel grass (*Chenchrus ciliaris*) pasture. This catchment has Vertosols covering approximately 90% of the hydrological area and Chromosols over the remaining 10% (unpublished BCS data).

The two pastures were spelled prior to the commencement of this study in October 2014. The conservatively grazed pasture was spelled between September 2011 and October 2014, with the exception of grazing between December 2013 and February 2014. The heavily grazed pasture was grazed from July 2012 to December 2012 and then spelled until October 2014. Stocking rates were set based on pasture biomass and have been converted to adult equivalents per hectare per year (AE/ha/yr) to account for differences in the size of cattle, and also the length of time the pastures were grazed (Table 3). Stocking rates in hectares per an adult equivalent (ha/AE) are also provided; however, this gives no indication of the time that the pasture was stocked. An adult equivalent is equal to a 450 kg non-lactating animal. Recommended stocking rates are about 2 ha/head for newly established buffel grass pasture and about 3 ha/head for rundown buffel grass pasture, which can occur in as little as five to ten years after establishment (Noble *et al.* 2000; Peck *et al.* 2011). Spelling was defined as the number of days annually that pasture wasn't grazed (Table 4). Overall, the conservatively grazed pasture had lower stocking rates and greater periods of spelling.

Thornton and Elledge 2018

Table 2: Description of the three Brigalow Catchment Study treatments reported for the 2015 to 2018 hydrological years.

Parameter	Brigalow scrub	Conservative grazing	Heavy grazing
Alternative catchment name	Catchment 1 or C1	Catchment 3 or C3	Catchment 5 or C5
Hydrological area (ha)	16.8	12.7	12.0
Total grazed area (ha)	0.0	17.0	25.0
Land use	Virgin brigalow scrub	Improved grass pasture	Improved grass pasture
Cattle stocking philosophy	Ungrazed control	Conservative stocking rate	High stocking rate
Pasture spelling philosophy	Ungrazed control	Wet season spell	Limited spelling
Pasture biomass philosophy	Not applicable	Minimum 1,000 kg/ha	No minimum limit
Photo			

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Table 3: Annual stocking rates in adult equivalents (AE) per hectare per year and also in hectare per AE for the two pastures.

Year	Stocking rate (AE/ha/yr)		Stocking rate (ha/AE)	
	Conservative grazing	Heavy grazing	Conservative grazing	Heavy grazing
2013	Destocked	0.09	Destocked	1.89
2014	0.19	Destocked	0.67	Destocked
2015	0.20	0.83	3.86	0.81
2016	0.13	0.20	1.47	1.32
2017	0.19	0.26	4.42	1.11
2018	Destocked	0.86	Destocked	0.51

Table 4: Annual number of non-grazed days (spelling) for the two pastures.

Year	Pasture spelled (days/yr)	
	Conservative grazing	Heavy grazing
2013	365	303
2014	320	365
2015	80	33
2016	297	286
2017	76	180
2018	365	146

2.4 Hydrology

Rainfall and runoff were monitored over four hydrological years from October 2014 to September 2018. Rainfall was measured using a 0.5 mm tipping bucket rain gauge located at the head point of the three long-term catchments (Thornton *et al.* 2007). Each catchment was instrumented to measure runoff using a 1.2 m steel HL flume with a 3.9 x 6.1 m approach box. Water heights through the flume were recorded using a pressure transducer with a mechanical float recorder backup. Stage heights were converted to discharge using a rating table (Brakenseik *et al.* 1979), while peak runoff rate was calculated on an event basis from instantaneous peak height. A runoff event commenced when stage height exceeded zero and finished when it returned to zero. Further details on calculating total runoff and peak runoff rates are documented in other sources (Thornton *et al.* 2007; Thornton and Yu 2016).

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2.5 Water Quality

Discrete water quality samples were obtained over four hydrological years (October 2014 to September 2018) using an auto-sampler located at the flume of each catchment. Auto-samplers were programmed to sample every 0.1 m change in stage height. Laboratory analyses of runoff samples were undertaken by Queensland Health Forensic and Scientific Services (Table 5), with some parameters calculated by difference (Table 6).

Table 5: Methods used by Queensland Health Forensic and Scientific Services for total suspended solids and nutrient analyses of runoff water samples.

Parameter	Method
TSS	Method 18211 based on gravimetric quantification of solids in water
TN / TDN	Method 13802 by simultaneous persulfate digestion
NO_x-N	Method 13798 based on flow injection analysis of nitrogen as oxides
NH_4-N	Method 13796 based on flow injection analysis of nitrogen as ammonia
TP / TDP	Method 13800 by simultaneous persulfate or Kjeldahl digestion
DIP	Method 13799 by flow injection analysis

Table 6: Equations used to estimate nutrient parameters that were not directly measured.

Parameter	Equation
PN	TN - TDN
DON	TDN - DIN
DIN	NO_x-N + NH_4-N
PP	TP - TDP
DOP	TDP - DIP

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 resulting loads from all intervals. The intervals were defined as the start of flow to the midpoint of sample one and sample two, the midpoint of sample one and sample two to the midpoint of sample two and sample three, and so on. Total annual load was calculated by summing all of the event based water quality loads, and load in kg/ha was calculated by accounting for hydrological catchment area.

Event based EMCs were calculated by dividing total event load by total event flow, and mean annual EMCs were calculated by averaging the event based EMCs within each year. Mean annual EMCs from 2000 to 2018 were used to calculate a long-term EMC for each catchment. The method used for calculating a mean annual EMC is described in Appendix 1. Where water quality data was not captured due to flows being too small to trigger auto-samplers, load estimations were obtained by multiplying the long-term EMC by observed flow. Only observed (measured) event based EMCs were included in the calculation of mean annual EMCs.

Dominant pathways of nitrogen and phosphorus loss in runoff were determined by the proportion of particulate and total dissolved fractions. That is, if total dissolved nitrogen was greater than 60% of total nitrogen it was considered to be transported primarily in a dissolved phase, and if less than 40% it was transported primarily in a particulate phase. If the value was between 40% and 60%, it was considered to have no dominant pathway of loss. The same method was applied to total phosphorus and total dissolved phosphorus.

2.6 Ground Cover

Ground cover from the total grazed area of the two pasture catchments, excluding the shade lines, was compared from October 2012 to April 2018 using VegMachine® (Fitzroy Basin Association 2018). This is an online tool that uses satellite imagery to summarise spatial and temporal changes in cover; that is, cover at or near ground level which excludes higher cover such as tree and shrub canopies. Seasonal deciles were also reported for total (green and non-green) cover, where total cover and bare ground equal 100%. Quarterly data from Autumn (March to May) 1988 to Summer (December to February) 2012/2013 are used as a baseline, and then every season is ranked (expressed as a decile) against all corresponding values for that season in the baseline period (Trevithick 2017). For example, total cover from spring (September to November) 2013 is ranked against total cover in all the spring images from the baseline period.

2.7 Pasture Biomass

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 two pasture catchments, excluding the shade lines. Pasture assessments occurred in the late wet and/or the late dry season. The late wet season is typically the end of the pasture growing season, and the late dry season provides an indicator of the remaining pasture available for cattle grazing until suitable conditions for growth occur. Pasture biomass was visually estimated for up to 300 0.16 m^2 quadrats in each catchment at each sampling period. Visual estimates were calibrated against a set of 10 quadrats which were cut, dried and weighed.

2.8 Qualitative Pasture Assessments

A photographic comparison of the conservatively and heavily grazed pastures during the late wet and late dry seasons over the 2015 to 2018 hydrological years is also provided. This is to help the reader visualise how ground cover and pasture biomass measurements appear in the field. BOTANAL measurements of pasture biomass and photographs may have occurred at different times within the season.

During July 2018, a visual comparison of pasture condition was also made between the conservatively and heavily grazed pastures of the BCS with five other heavily grazed properties under different ownership elsewhere in the Fitzroy Basin.

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3 Results

3.1 Hydrology

Total annual rainfall at the study site was below the long-term mean annual rainfall of 648 mm (October 1965 to September 2018) in all four hydrological years (Figure 3). Rainfall was in the 31st percentile in 2015 (563 mm), the 29th percentile in 2016 (562 mm), the lowest on record in 2017 (272 mm) and in the 40th percentile in 2018 (584 mm).

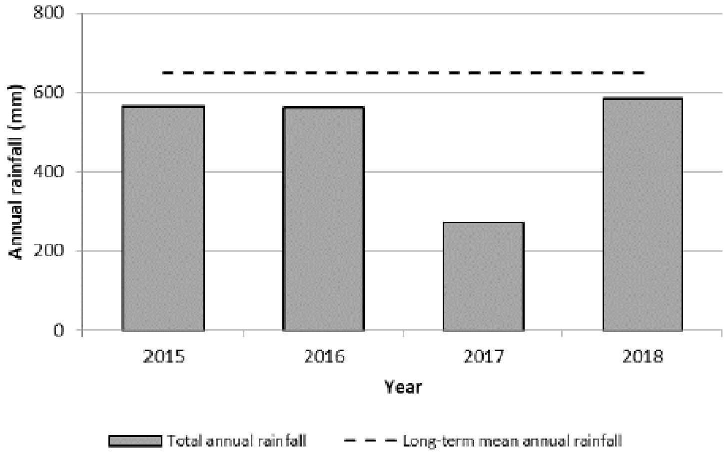

Figure 3: Total annual hydrological year rainfall for 2015 to 2018 relative to the long-term mean annual rainfall for the Brigalow Catchment Study.

Similar to rainfall, runoff for the four hydrological years was below the long-term mean annual runoff (1985 to 2018) for the brigalow scrub and conservatively grazed catchments (Figure 4). The heavily grazed catchment was only instrumented in 2014, at the commencement of this study, and mean annual runoff was based on four years (2015 to 2018) data. Runoff from brigalow scrub was in the 32nd percentile in 2015, no runoff occurred in 2016 and 2017, and in 2018 was in the 29th percentile. Runoff from the conservatively grazed catchment was in the 35th percentile in 2015, the 30th percentile in 2016, no runoff occurred in 2017, and in 2018 was in the 15th percentile. The heavily grazed catchment had the same amount of runoff (28 mm) in both 2015 and 2016, no runoff occurred in 2017, and in 2018 runoff was 68% of the 2015 to 2016 average.

Hydrological data and water quality sampling effort for 2015 to 2018 are summarised in Table 7. Over the four hydrological years, there was a total of two events from the brigalow scrub catchment, four events from the conservatively grazed catchment, and five events from the heavily grazed catchment. Although the number of events and total runoff was low in these below average rainfall years, when runoff did occur, the heavily grazed catchment had consistently greater runoff than the conservatively grazed catchment. A similar trend was also observed for peak runoff rates with both average and maximum values greatest from the heavily grazed pasture.

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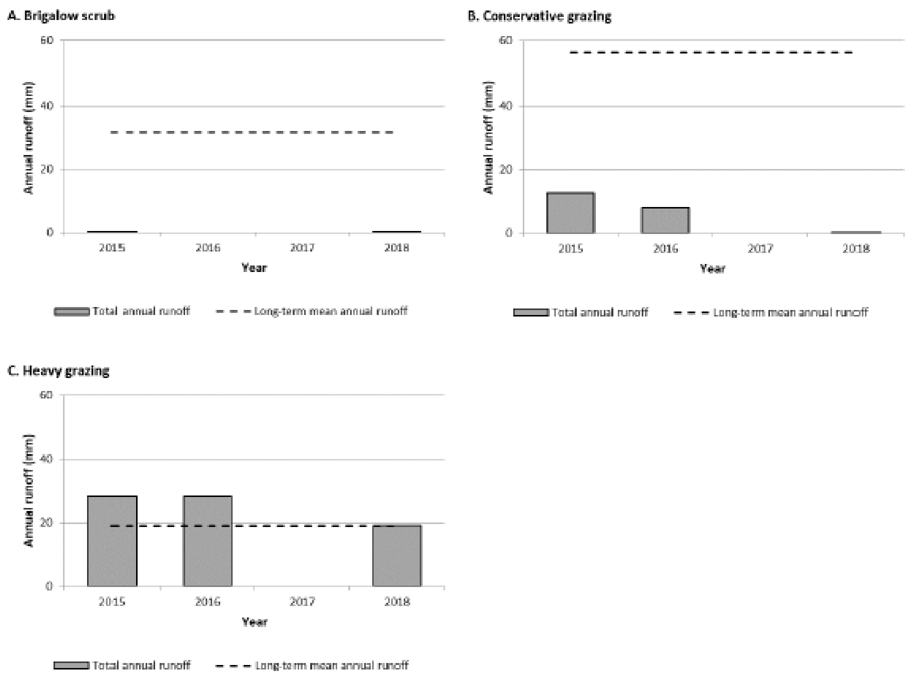

Figure 4: Total annual hydrological year runoff for 2015 to 2018 relative to the long-term mean annual runoff for the three catchments. Long-term means were based on 34 years (1985 to 2018) data for the brigalow scrub and conservatively grazed catchments, and four years data (2015 to 2018) for the heavily grazed catchment.

Using the hydrological calibration developed during Stage I (1965 to 1982), runoff characteristics for the conservatively grazed pasture (Catchment 3) can be estimated had it remained brigalow scrub (Table 8). In 2015, conservatively grazed pasture generated 65 times more total runoff and 13 times greater peak runoff than uncleared estimates for this catchment. As no runoff occurred from the brigalow scrub catchment (Catchment 1) in 2016 and 2017, there would have been no runoff from Catchment 3 in an uncleared state. Total runoff and peak runoff from the brigalow scrub and conservatively grazed pasture catchments were the same in 2018 (Table 7), which means that there were negligible difference between observed and estimated uncleared runoff from the conservatively grazed catchment in that year.

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Table 7: Observed annual hydrological year summaries of runoff and sampling effort for three catchments.

Parameter	Year	Brigalow scrub	Conservative grazing	Heavy grazing
Number of events	2015	1	2	2
	2016	0	1	1
	2017	0	0	0
	2018	1	1	2
Number of samples	2015	0	3	21
	2016	0	2	6
	2017	0	0	0
	2018	0	0	4
Total runoff (mm)	2015	0.2	13	28
	2016	0	8	28
	2017	0	0	0
	2018	0.1	0.1	19
Average peak runoff rate (mm/hr)	2015	0.1	2.6	6.4
	2016	0	1.0	2.6
	2017	0	0	0
	2018	0.1	0.1	2.6
Maximum peak runoff rate (mm/hr)	2015	0.1	3.1	6.5
	2016	0	1.0	2.6
	2017	0	0	0
	2018	0.1	0.1	4.7

Table 8: Predicted annual hydrological year summaries of runoff from the conservatively grazed pasture catchment had it remained uncleared brigalow scrub.

Parameter	Year	Catchment 3
Estimated uncleared runoff (mm)	2015	0.2
	2016	0
	2017	0
	2018	0.1
Increase in runoff under pasture (mm)	2015	12
	2016	8
	2017	0
	2018	0
Estimated uncleared average peak runoff rate (mm/hr)	2015	0.2
	2016	0
	2017	0
	2018	0.4
Increase in average peak runoff rate under pasture (mm/hr)	2015	2.4
	2016	1.0
	2017	0
	2018	0

3.2 Water Quality

Loads and EMCs of total suspended solids, nitrogen and phosphorus are presented in Appendix 1. Results for 2015 are presented in Table A1, 2016 in Table A2 and 2018 in Table A3. There was no runoff, and hence no water quality from any catchment in 2017.

Loads of total suspended solids and all nitrogen and phosphorus parameters from heavily grazed pasture were between 1.4 and 3.7 times greater than from conservatively grazed pasture. In contrast, EMCs were consistently lower from heavily grazed pasture, being only 30% to 90% of that from conservatively grazed pasture. Loads of all water quality parameters from brigalow scrub were almost negligible due to no runoff in two of the four hydrological years, and less than 0.2 mm of runoff in the other two years. Consequently, no water quality samples were collected from this catchment and all data presented are estimations based on observed runoff and long-term EMCs. Using the hydrological calibration developed during Stage I (1965 to 1982), there would have been virtually no runoff from the conservatively grazed catchment in all four years had it remained brigalow scrub. Hence all loads of total suspended solids, nitrogen and phosphorus in runoff from the conservatively grazed catchment are an absolute anthropogenic increase attributable to changing land use from brigalow scrub to grazed pasture.

3.2.1 Total Suspended Solids

Mean annual load of total suspended solids from the heavily grazed pasture was 3.2 times greater than from the conservatively grazed pasture (Figure 5). The mean annual EMC for total suspended solids was 277.7 mg/L from conservatively grazed pasture and 234.7 mg/L from heavily grazed pasture.

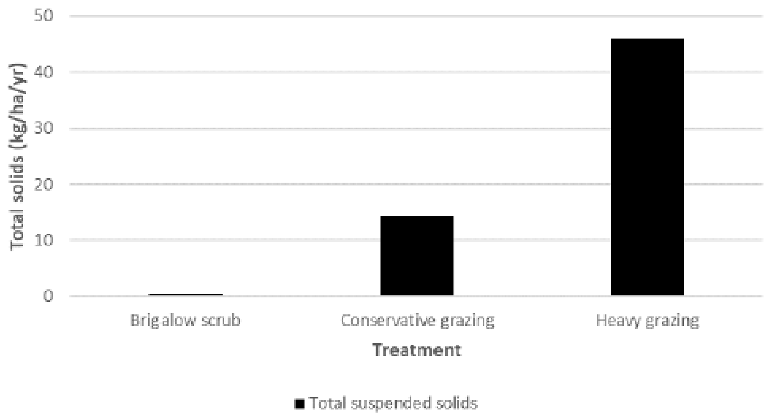

Figure 5: Mean annual load of total suspended solids in runoff from 2015 to 2018.

3.2.2 Nitrogen

Mean annual load of total nitrogen from the heavily grazed pasture was 1.6 times greater than from the conservatively grazed pasture (Figure 6). Total nitrogen was composed of similar amounts of particulate and total dissolved nitrogen irrespective of grazing pressure; 49% and 51% for conservatively grazed pasture and 45% and 55% for heavily grazed pasture, respectively. Although there was limited data from brigalow scrub, estimations indicate a greater contribution of total

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dissolved nitrogen (64%) than particulate nitrogen (36%) towards total nitrogen. The dominant pathway of nitrogen loss was in a dissolved form from brigalow scrub, but was unclear for the two pasture catchments (Table 9). The mean annual EMC for total nitrogen was 6.5 mg/L from conservatively grazed pasture and 2.4 mg/L from heavily grazed pasture; particulate nitrogen was 3.4 mg/L and 1.1 mg/L; and total dissolved nitrogen was 3.1 mg/L and 1.2 mg/L, respectively.

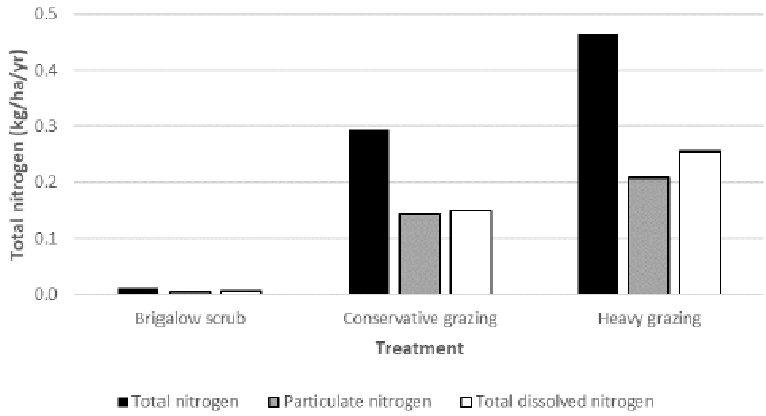

Figure 6: Mean annual load of total, particulate and dissolved nitrogen in runoff from 2015 to 2018.

Table 9: Dominant pathway of nitrogen loss in runoff from 2015 to 2018.

Year	Brigalow scrub	Conservative grazing	Heavy grazing
2015	Dissolved	No dominant	No dominant
2016	No runoff	No dominant	Dissolved
2017	No runoff	No runoff	No runoff
2018	Dissolved	Dissolved	Particulate

Mean annual load of total dissolved nitrogen from the heavily grazed pasture was 1.7 times greater than from conservatively grazed pasture (Figure 7). Dissolved organic and inorganic fractions contributed similar amounts towards total dissolved nitrogen from the two pasture catchments; 47% and 53% for conservatively grazed pasture and 53% and 47% for heavily grazed pasture, respectively. Although there was limited data from brigalow scrub, estimations indicate a greater contribution of dissolved inorganic nitrogen (66%) than dissolved organic nitrogen (34%) towards total dissolved nitrogen. Oxidised nitrogen was the greatest fraction of dissolved inorganic nitrogen from all catchments; 99% for brigalow scrub, 94% for conservatively grazed pasture and 88% for heavily grazed pasture. The mean annual EMC for dissolved organic nitrogen was 1.3 mg/L from conservatively grazed pasture and 0.7 mg/L from heavily grazed pasture; and dissolved inorganic nitrogen was 1.8 mg/L and 0.6 mg/L, respectively.

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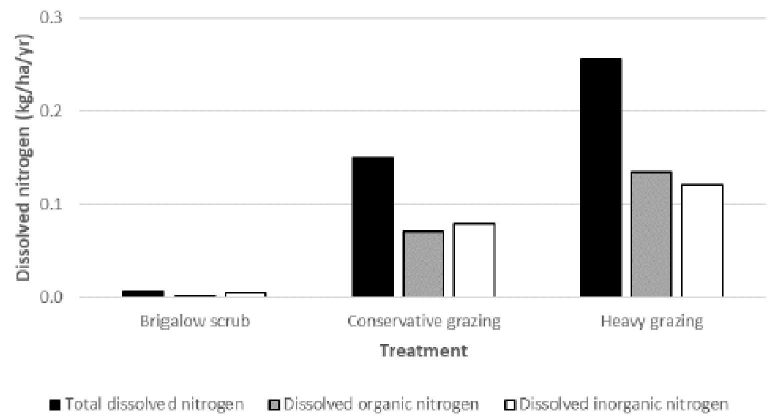

Figure 7: Mean annual load of dissolved nitrogen fractions in runoff from 2015 to 2018.

3.2.3 Phosphorus

Mean annual load of total phosphorus from the heavily grazed pasture was 2.6 times greater than from conservatively grazed pasture (Figure 8). Total phosphorus was composed of similar amounts of particulate and total dissolved phosphorus irrespective of grazing pressure; 59% and 41% for conservatively grazed pasture and 43% and 57% for heavily grazed pasture, respectively. Although there was limited data from brigalow scrub, estimations indicate a greater contribution of particulate phosphorus (72%) than total dissolved phosphorus (28%) towards total phosphorus. The dominant pathway of phosphorus loss was in a particulate form from brigalow scrub, but was unclear for the two pastures (Table 10). The mean annual EMC for total phosphorus was 0.81 mg/L from conservatively grazed pasture and 0.49 mg/L from heavily grazed pasture; particulate phosphorus was 0.50 mg/L and 0.22 mg/L; and total dissolved phosphorus was 0.31 mg/L and 0.27 mg/L, respectively.

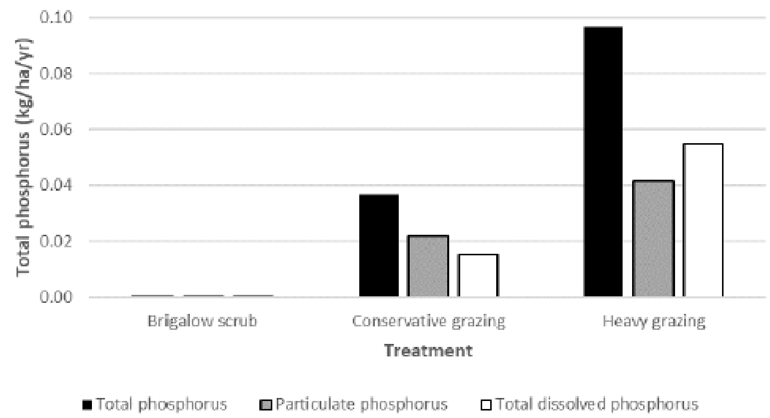

Figure 8: Mean annual load of total, particulate and dissolved phosphorus in runoff from 2015 to 2018.

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Table 10: Dominant pathway of phosphorus loss in runoff from 2015 to 2018.

Year	Brigalow scrub	Conservative grazing	Heavy grazing
2015	Particulate	Particulate	No dominant
2016	No runoff	No dominant	Dissolved
2017	No runoff	No runoff	No runoff
2018	Particulate	No dominant	Particulate

Mean annual load of total dissolved phosphorus from the heavily grazed pasture was 3.6 times greater than from conservatively grazed pasture (Figure 9). Dissolved inorganic phosphorus was the greatest fraction of total dissolved phosphorus from all catchments; 78% from brigalow scrub, 84% from conservatively grazed pasture and 86% from heavily grazed pasture. The mean annual EMC for dissolved inorganic phosphorus was 0.26 mg/L from conservatively grazed pasture and 0.23 mg/L from heavily grazed pasture; and dissolved organic phosphorus was 0.05 mg/L and 0.04 mg/L, respectively.

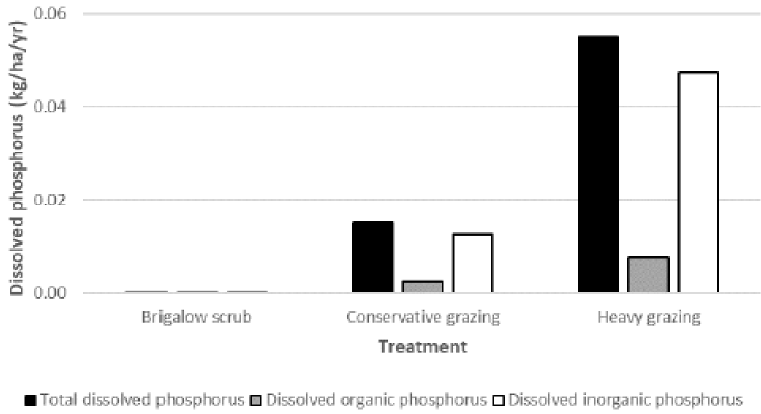

Figure 9: Mean annual load of dissolved phosphorus fractions in runoff from 2015 to 2018.

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3.3 Ground Cover

In the two years prior to the commencement of this study, the two pastures were extensively spelled with less than nine weeks of grazing at conservative stocking rates. During this time, the effect of season on cover can be observed with both pastures having higher proportions of bare ground in the late dry season (Figure 10). At the commencement of this study in October 2014, the proportion of bare ground was similar in the conservatively (12.3%) and heavily grazed pastures (13.4%). At this time, 95% of the conservatively grazed pasture had cover levels of 78% or higher and 95% of the heavily grazed pasture had similar cover levels of 73% or higher. In April 2018, the amount of bare ground in the heavily grazed pasture (14.9%) was 2.5 times greater than in the conservatively grazed pasture (5.9%). Ground cover in the conservatively grazed pasture remained relatively constant during the study with 95% of the pasture having cover levels of 84% or higher in April 2018. However, cover levels across 95% of the heavily grazed pasture decreased to 57% or higher by January 2018 before increasing to 76% or higher in April 2018, similar to the distribution of cover at the commencement of the study. This analysis showed that the conservatively and heavily grazed pastures started in a similar condition, but an increase in bare ground and a corresponding decrease in ground cover were observed over time in the heavily grazed pasture.

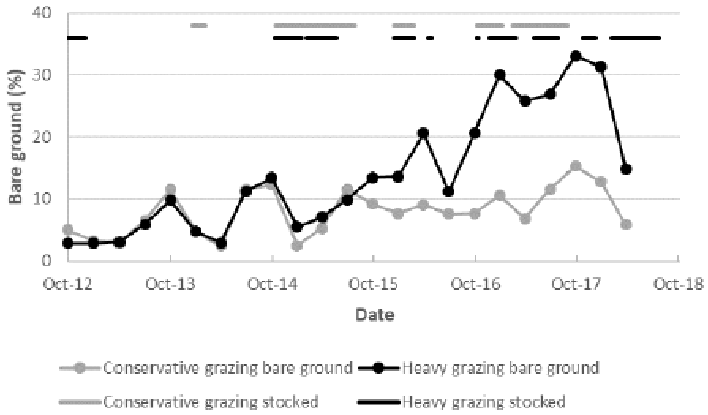

Figure 10: Measurements of bare ground in the two pastures related to cattle stocking.

3.4 Pasture Biomass

Overall, the heavily grazed catchment had lower pasture biomass than the conservatively grazed catchment (Figure 11). In the 2014 late wet season, prior to the commencement of the study, there was similar biomass in both the conservatively (6.9 t/ha) and heavily grazed pastures (6.2 t/ha). Biomass in the 2015 late wet season had increased 2.7 t/ha in the conservatively grazed pasture (9.6 t/ha) with little change in the heavily grazed pasture (6.5 t/ha). Biomass in the heavily grazed pasture went from 90% of the biomass in the conservatively grazed pasture in 2014 to 68% in 2015.

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In the 2016 late wet season, biomass had reduced 53% under conservative grazing (4.5 t/ha) and 57% under heavy grazing (2.8 t/ha) compared to the previous year (Figure 11). The difference in biomass between the two pastures was 63%, similar to the previous year. Biomass continued to decline in both pastures over the next six months, with a 43% reduction in the conservatively grazed pasture to 2.5 t/ha and a much greater 83% reduction in the heavily grazed pasture to 0.5 t/ha. Biomass in the heavily grazed pasture during the 2016 late dry season was reduced to 19% of that from the conservatively grazed pasture.

In the 2017 late dry season, biomass had increased to 5.0 t/ha under conservative grazing and 3.1 t/ha under heavy grazing (Figure 11). Pasture biomass in the heavily grazed catchment increased to 62% of that from the conservatively grazed catchment, similar to the 2015 and 2016 late wet seasons. In the 2018 late dry season, biomass had increased 5% under conservative grazing (5.3 t/ha) whereas biomass under heavy grazing (0.4 t/ha) had declined 86% compared to the previous year. Biomass in the heavily grazed pasture during the 2018 late dry season was reduced to 8% of that from the conservatively grazed pasture.

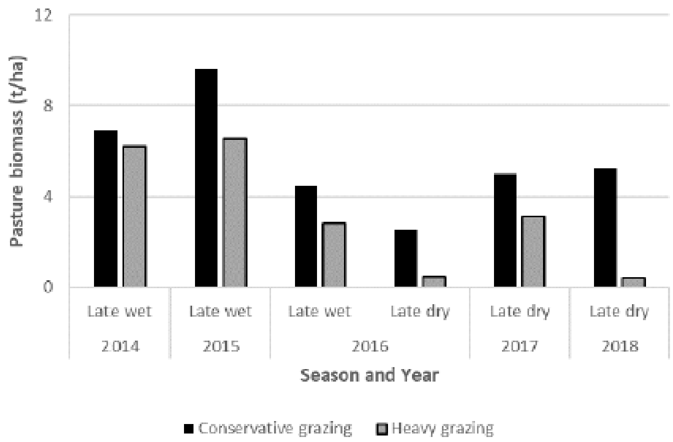

Figure 11: Pasture biomass in the two pastures from 2015 to 2018.

3.5 Qualitative Pasture Assessment

Table 11 provides a visual comparison of the conservatively and heavily grazed pastures during the late wet and late dry seasons over the 2015 to 2018 hydrological years. These photographs show lower ground cover and pasture biomass from the heavily grazed pasture compared to the conservatively grazed pasture. Table 12 provides a visual comparison of the two BCS pastures with five other grazed properties under different ownership in the Fitzroy Basin. The five properties appear to have lower ground cover and pasture biomass than the heavily grazed pasture.

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Table 11: Photographic comparison of ground cover and pasture biomass from the two pastures in the late wet and late dry seasons from 2015 to 2018.

Year	Late wet season		Late dry season	
	Conservative grazing	**Heavy grazing**	**Conservative grazing**	**Heavy grazing**
2015			No photo	No photo
2016				
2017				
2018				

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Table 12: Photographic comparison of the two Brigalow Catchment Study pastures compared to five other heavily grazed properties within the Fitzroy Basin during the 2018 late dry season.

Site and grazing pressure	Landscape	Ground Cover
Brigalow Catchment Study Conservative grazing		
Brigalow Catchment Study Heavy grazing		
Property 1 Fitzroy Basin Heavy grazing		
Property 2 Fitzroy Basin Heavy grazing		
Property 3 Fitzroy Basin Heavy grazing		
Property 4 Fitzroy Basin Heavy grazing		
Property 5 Fitzroy Basin Heavy grazing		

4 Discussion

4.1 Effect of Grazing Pressure on Hydrology

Changing land use from virgin brigalow scrub to conservatively grazed pasture at the long-term BCS has doubled total runoff (Thornton *et al.* 2007) and increased average and maximum peak runoff rates by 1.5 times and 3 times, respectively, when runoff occurred from both catchments (Thornton and Yu 2016). Over the four below average rainfall years of this study, heavy grazing of rundown pasture at stocking rates recommended for newly established pasture resulted in 3.6 times more total runoff and 3.3 times greater average peak runoff rate than the conservatively grazed pasture. At the end of the four year study, the heavily grazed pasture had 2.5 times more bare ground and only 8% of the pasture biomass compared to the conservatively grazed pasture. In years when no runoff occurred from brigalow scrub, total runoff from the conservatively grazed pasture was an absolute anthropogenic increase attributable to land use change. Runoff is known to increase with a decline in ground cover and/or biomass (Bartley *et al.* 2010; McIvor *et al.* 1995; Silburn *et al.* 2011), so an increase in runoff from the heavily grazed catchment was expected. This reflects numerous other studies that have reported greater runoff from grazed than ungrazed areas and/or pastures with higher stocking rates (Duniway *et al.* 2018; Filet and Osten 1996; Mapfumo *et al.* 2002; O'Reagain 2011; Silcock *et al.* 2005; van Oudenhoven *et al.* 2015).

Ground cover is an easily measured and visually evident indicator of land condition. While increases in runoff are commonly attributed to or observed in partnership with declining ground cover, the landscape response is more complex. For example, Thornton *et al.* (2007) showed that changed water use patterns was the primary driver of increased runoff when native vegetation was replaced with improved grass pasture, and that increased compaction and reduced ground cover, soil structure and infiltration rate were secondary drivers. Increased runoff, and subsequently increased loads of nutrients in runoff, are effectively a reduction in plant available water capacity and fertility of soils which leads to reduced pasture growth.

Persistent heavy grazing also changes the species composition of pasture over time leading to a decline in desirable (perennial, palatable and productive) species and an increase in less desirable (annual, unpalatable and less productive) species. For example, studies in the Burdekin Basin have attributed the transition of productive native grass species, such as black speargrass (*Heteropogon contortus*) and desert bluegrass (*Bothriochloa ewartiana*), to the less productive and less drought tolerant Indian couch (*Bothriochloa pertusa*) to a combination of drought and heavy grazing (Bartley *et al.* 2014; Spiegel 2016). Therefore, runoff, plant available water capacity, pasture growth and changes in pasture species composition are all intrinsically linked by the management of grazing pressure.

Intervention to break the cycle of declining land condition can be achieved with the adoption of improved management practices; however, the time required to restore healthy eco-hydrological function may vary from years to decades (Bartley *et al.* 2014; Hawdon *et al.* 2008; Roth 2004; Silcock *et al.* 2005). For example, a landholder in the Burdekin Basin reported improved land condition with the adoption of a safe long-term carrying capacity and pasture spelling (Landsberg *et al.* 1998). The property had reduced income during the three year transition phase; however, it became profitable with less cattle once the perennial grasses recovered. Other research in the Burdekin Basin clearly indicates that sustainable grazing management is profitable over extended time periods and varying climatic cycles (O'Reagain *et al.* 2011). Nonetheless, from both an environmental and economic perspective, it is better to improve grazing management before a dramatic decline in land condition occurs.

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4.2 *Effect of Grazing Pressure on Water Quality*

Heavily grazed pasture had higher loads and lower EMCs for all water quality parameters compared to conservatively grazed pasture. In years when no runoff occurred from brigalow scrub, total runoff and subsequent loads of total suspended solids and nutrients from the conservatively grazed pasture were an absolute anthropogenic increase attributable to land use change. Over four below average rainfall years, this study typically had lower loads and higher EMCs than previously reported for the BCS during wetter periods and over longer timeframes (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014). These trends indicate that increased flow, whether from above average rainfall or a treatment (grazing pressure) effect, results in dilution of total suspended solids and nutrients leading to lower EMCs. However, the dilution effect was not strong enough to result in reduced loads. Dilution effects have been reported for sediment and nutrient concentrations within events (Schepers and Francis 1982), within seasons (Hay *et al.* 2006; Schepers *et al.* 1982), in the transition from dry to wet seasons (Vink *et al.* 2007), and also on an annual basis over multiple years (Bartley *et al.* 2014; Miller *et al.* 2017). This study reflects other publications that have reported increased loads with increased flow (Hay *et al.* 2006; Schepers *et al.* 1982).

4.2.1 Total Suspended Solids

Runoff from heavily grazed pasture had 3.2 times more total suspended solids load than the conservatively grazed pasture. An increase in suspended solids with a decrease in ground cover is the same as the trend observed between runoff and cover in this study, which is a relationship often cited in the literature (Bartley *et al.* 2010; McIvor *et al.* 1995; Silburn *et al.* 2011). VegMachine® analysis showed decreased ground cover with increased grazing pressure. Despite similar cover levels in the two pastures initially, there was 2.5 times more bare ground in the heavily grazed pasture after four years compared to the conservatively grazed pasture. Mean annual loads for both the conservatively (14 kg/ha/yr) and heavily grazed pastures (46 kg/ha/yr) during the four below average rainfall years of this study were considerably lower than observed from the conservatively grazed pasture during an extremely wet period from 2010 to 2012, a return to average conditions from 2013 to 2014, and also modelled loads for the period 1984 to 2010 (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014). Mean annual load from these three periods was 258 kg/ha/yr (range 20 to 468 kg/ha/yr). Loads from this study were also lower than more erosive landscapes with shallower soils elsewhere in the Fitzroy Basin (Silburn *et al.* 2011) and in the nearby Burdekin Basin (Bartley *et al.* 2014; Hawdon *et al.* 2008).

Mean annual EMCs of total suspended solids from both the conservatively (278 mg/L) and heavily grazed pastures (235 mg/L) were similar to those previously reported for the conservatively grazed pasture during wetter periods and over longer timeframes (301 mg/L; range 95 to 916 mg/L) (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014). These values also fit within the ranges reported for grazing on both improved and native pastures dominated (>90%) by a single land use (Bartley *et al.* 2012). Bartley *et al*. (2012) reviewed water quality data from across Australia and found that EMCs of total suspended solids were lower from forests than improved pasture, and both these land uses were lower than from native pastures. In contrast, EMCs from brigalow scrub of the BCS were generally higher than from conservatively grazed pasture when runoff occurred from both catchments (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014). This highlights the importance that hydrological characteristics, vegetation type and landscape condition (i.e. ground cover) have on the resulting total suspended solids loads and concentrations. Data from the BCS is able to fill the knowledge gap of water quality from brigalow lands in the Fitzroy Basin, which can further refine estimations of the impact of grazing land management on Great Barrier Reef water quality.

4.2.2 Nitrogen

Similar total suspended solids, loads of all nitrogen parameters during the four below average rainfall years were greater from heavily than conservatively grazed pasture while EMCs were lower from heavily grazed pasture. This reflects other studies that have reported greater nitrogen loads from grazed than ungrazed areas and also from heavier than lighter grazing pressures (Daniel *et al.* 2006; Park *et al.* 2017). Mean annual loads of total nitrogen (0.29 kg/ha/yr) and dissolved inorganic nitrogen (0.08 kg/ha/yr) from the conservatively grazed pasture in this study were lower than previously reported during wetter periods and over longer timeframes; 2.6 kg/ha/yr (range 0.6 to 5.1 kg/ha/yr) and 0.37 kg/ha/yr (range 0.06 to 0.81 kg/ha/yr), respectively (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014).

In contrast, EMCs of total nitrogen (6.49 mg/L) and dissolved inorganic nitrogen (1.81 mg/L) from the conservatively grazed pasture in this study were higher than previously reported; 2.4 mg/L (range 2.0 to 3.2 mg/L) and 0.41 mg/L (range 0.11 to 0.80 mg/L), respectively (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014). EMCs for these two nitrogen parameters were within the range for improved pastures in Australia, but exceeded the range for native pastures when the majority of the upstream area was under a single land use (Bartley *et al.* 2012). However, under the more rigorous criteria of upstream area dominated (>90%) by a single land use, the total nitrogen EMC in this study exceeded the ranges for both improved and native pastures. Comparable data was not available for dissolved inorganic nitrogen.

These high EMCs are likely a reflection of the high soil fertility of brigalow lands compared to the rangeland, savannah and woodland landscapes from which comparable data was available. This is supported by long-term total nitrogen (14.4 mg/L; range 9.9 to 20.2 mg/L) and dissolved inorganic nitrogen (4.82 mg/L; range 1.94 to 7.01 mg/L) EMCs from brigalow scrub (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014) which greatly exceed the ranges given for forest in Bartley *et al.* (2012). Furthermore, modelling of long-term water quality indicates that brigalow scrub has higher loads and concentrations of nitrogen (total and dissolved) compared to conservatively grazed pasture (Elledge and Thornton 2017). This is in contrast to a number of Australian and international studies that have noted higher loads of nitrogen from pasture than forest (Quinn and Stroud 2002; Udawatta *et al.* 2011; Vink *et al.* 2007). This highlights the uniqueness of brigalow lands where nitrogen fixation by brigalow (*Acacia harpophylla*) leads to high soil fertility, and hence higher losses of nitrogen in runoff, compared to other landscapes (Thornton and Elledge 2018; Webb *et al.* 1982; Yule 1989).

The limited data collected during this study showed that nitrogen lost in runoff from brigalow scrub was predominately in the dissolved phase. This phase was dominated by dissolved inorganic nitrogen which in turn was dominated by oxidised nitrogen. In contrast, nitrogen from the two pastures was lost in both particulate and dissolved phases. Both dissolved organic and inorganic nitrogen made substantial contributions to the dissolved phase. Oxidised nitrogen dominated the dissolved inorganic nitrogen fraction. This reflects numerous authors that have highlighted the importance of dissolved organic nitrogen when considering nitrogen losses (Alfaro *et al.* 2008; Robertson and Nash 2008; Van Kessel *et al.* 2009). This is certainly the case for grazed landscapes, as dissolved organic nitrogen is known to increase with the application of cattle urine and dung (Van Kessel *et al.* 2009; Wachendorf *et al.* 2005), and concentrations have also been shown to increase with increased grazing pressure (Owens *et al.* 1989).

4.2.3 Phosphorus

Similar to total suspended solids and nitrogen, loads of all phosphorus parameters during the four below average rainfall years were greater from heavily than conservatively grazed pastures while EMCs were lower from heavily grazed pastures. This reflects other studies that have reported greater phosphorus loads from grazed than ungrazed areas and also from heavier than lighter grazing pressures (Butler *et al.* 2008; Daniel *et al.* 2006; Park *et al.* 2017; Vink *et al.* 2007). Mean annual loads of total phosphorus (0.04 kg/ha/yr) and dissolved inorganic phosphorus (0.01 kg/ha/yr) from the conservatively grazed pasture in this study were lower than previously reported during wetter periods and over longer timeframes; 0.38 kg/ha/yr (range 0.07 to 0.76 kg/ha/yr) and 0.20 kg/ha/yr (range 0.04 to 0.42 kg/ha/yr), respectively (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014).

In contrast, EMCs of total phosphorus (0.81 mg/L) and dissolved inorganic phosphorus (0.26 mg/L) from the conservatively grazed pasture were higher than previously reported; 0.32 mg/L (range 0.23 to 0.41 mg/L) and 0.17 mg/L (range 0.10 to 0.22 mg/L), respectively (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014). The total phosphorus EMC fits within the range for both improved and native pastures within Australia (Bartley *et al.* 2012). Although the EMC for dissolved inorganic phosphorus was just above the range for improved pastures, it greatly exceeded the range for native pastures (Bartley *et al.* 2012). Similarly, the EMC for dissolved organic phosphorus in this study (0.05 mg/L) greatly exceeded both the improved and native pasture ranges of Bartley *et al.* (2012).

Similar to the response for nitrogen, these high EMCs are likely a reflection of the high soil fertility of brigalow lands. This is supported by long-term EMCs of total phosphorus (0.79 mg/L; range 0.32 to 2.19 mg/L) and dissolved inorganic phosphorus (0.16 mg/L; range 0.10 to 0.29 mg/L) from brigalow scrub (Elledge and Thornton 2017; Thornton and Elledge 2013; Thornton and Elledge 2014) which greatly exceed the ranges given for forest in Bartley *et al.* (2012). Furthermore, soil phosphorus levels prior to land development at the BCS were considered moderate (13.7 mg/kg; range 13.3 to 14.0 mg/kg) based on the classification of Ahern *et al.* (1994). Levels increased rapidly becoming high to very high (34.7 mg/kg; range 24 to 44 mg/kg) following clearing and burning due to the resulting ash bed. However, soil phosphorus levels under grazing then declined back to a moderate level (12.6 mg/kg; range 11.0 to 14.6 mg/kg) over the next 32 years (unpublished BCS data). This is in stark contrast to the low, deficient (very low) and acute (extremely low) status of soil phosphorus given to 72% of the central and north-east Queensland grazing lands (Ahern *et al.* 1994) and the deficient and acute status given to 68% of northern Australian soils (McCosker and Winks 1994).

Phosphorus loss from uncultivated fields and grazed pasture is typically in the dissolved phase, which is dominated by dissolved inorganic phosphorus (Alfaro *et al.* 2008; Gillingham and Gray 2006; Potter *et al.* 2006; Robertson and Nash 2008). The limited data collected during this study showed that phosphorus loss from brigalow scrub may be dominated by particulate phosphorus while the grass pastures lost phosphorus in both particulate and dissolved phases. Higher EMCs of dissolved inorganic phosphorus from conservatively grazed pasture compared to brigalow scrub has previously been attributed to the presence of grazing animals and their dung (Elledge and Thornton 2017), which is in agreement with the literature (Schepers *et al.* 1982; Vadas *et al.* 2011).

4.3 *Stocking Rates and Safe Long-Term Carrying Capacity*

Published stocking rates for buffel grass pastures on brigalow lands vary from 2 ha/head to 10 ha/head (Graham *et al.* 1991; Lawrence and French 1992; Noble *et al.* 2000; Partridge *et al.* 1994; Paton *et al.* 2011; Peck *et al.* 2011). Some authors acknowledge that stocking rates should be adjusted for landscape and seasonal variability (Graham *et al.* 1991; Lawrence and French 1992; Paton *et al.* 2011), while others note that stocking rates should be reduced over time as pasture productivity declines (Noble *et al.* 2000; Partridge *et al.* 1994; Peck *et al.* 2011). For example, Noble *et al.* (2000) recommends 2 ha/head on newly established buffel grass pastures and 3 ha/head on rundown buffel grass pastures. Daily live weight gains of 0.5 kg/head are considered possible from newly established pastures (Lawrence and French 1992; Radford *et al.* 2007); however, stocking rates should be adjusted to achieve daily weight gains of 0.4 kg/head on rundown pastures (Partridge *et al.* 1994).

In line with these recommendations and to maintain industry relevance, the average stocking rate of the conservatively grazed pasture during this study was 0.17 AE/ha/yr, which equates to 5.9 ha/AE. Historically, stocking rates for this pasture were 2.2 ha/AE on newly established buffel grass pasture when the study commenced, and decreased to 3.8 ha/AE over the next 21 years (Radford *et al.* 2007). The average long-term (1984 to 2017) stocking rate was 3.3 ha/AE (unpublished BCS data). Daily weight gains in the order of 0.5 kg/head were achieved initially and have been obtained periodically since (Radford *et al.* 2007; Thornton and Buck 2011); however, maintaining the 2.2 ha/AE stocking rate during the first 11 years following pasture establishment saw daily weight gains decline to about 0.3 kg/head (Radford *et al.* 2007).

The average stocking rate in the heavily grazed pasture was 0.54 ha/AE/yr, which equates to 1.9 ha/AE. Despite the age of the pasture (40 to 50 years old), this stocking rate was similar to recommended stocking rates for newly established buffel grass pastures. Given the difficulties encountered in changing the traditional paradigm of "more cattle means more money" towards lighter stocking rates despite equal or greater economic return (Moravek *et al.* 2017; O'Reagain *et al.* 2011; Stockwell *et al.* 1991), it is likely that high stocking rates are still used within the industry. This is supported by the qualitative pasture assessment in this study which shows better management of the heavily grazed pasture of the BCS compared to five properties in the Fitzroy Basin. Thus, ground cover, pasture biomass, hydrology and water quality data for the heavily grazed pasture in this report may still be an underestimate for some properties.

The concept of safe long-term carrying capacity for sustainable grazing management benefits productivity, land condition and runoff water quality by balancing pasture utilisation with pasture growth (O'Reagain *et al.* 2014). A utilisation rate between 15 and 30% of pasture growth has been considered a safe long-term carrying capacity (O'Reagain *et al.* 2011; Peck *et al.* 2011). Safe long-term carrying capacity can be calculated using pasture biomass, dietary intake requirements of cattle and pasture utilisation rates. For the conservatively grazed pasture, a safe long-term carrying capacity was 3.4 ha/AE based on long-term pasture biomass of 3,500 kg/ha (Radford *et al.* 2007), an estimated dietary intake of 2.2% bodyweight per day (Minson and McDonald 1987) and a high but still economically viable utilisation rate of 30% (Bowen and Chudleigh 2017). Although a safe long-term carrying capacity can be calculated for a specific location, stocking rates should be adjusted annually at the end of the summer growing period to account for pasture biomass (Lawrence and French 1992).

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4.4 Implications for the Grazing Industry

Long-term data from the BCS suggests that a stocking rate of 3.4 ha/AE is a safe long-term carrying capacity for rundown (30 to 40 years old) buffel grass pasture established on predominantly clay soils previously dominated by brigalow scrub. This recommendation is based on long-term pasture biomass and cattle live weight gains from the study site, and stocking rates may need to be reduced at other locations unable to produce similar amounts of pasture biomass (average 3,500 kg/ha). Failure to reduce stocking rates on rundown pastures to match safe long-term carrying capacity led to increased runoff, and subsequently increased loads of total suspended solids, nitrogen and phosphorus in runoff. While limited water quality data was collected during the four below average rainfall years of this study, total nitrogen and phosphorus loads both had substantial contributions of particulate and dissolved fractions. Although heavily grazed pasture had the highest runoff and greatest loads of all total suspended solids and nutrient parameters, it had the lowest EMCs. This indicates that total runoff and peak runoff rate were key drivers of runoff loads. Heavy grazing pressure reduced ground cover which demonstrates the value of ground cover as an indicator of degraded land condition. This study compliments other research that has reported improved land condition and reduced economic risk by transitioning from heavy to conservative grazing pressures. This demonstrates that reducing grazing pressure is a realistic option for landholders that will also have benefits for runoff water quality.

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Appendix 1: EMC Method Comparison

Introduction

The traditional method for calculating an event mean concentration (EMC) is total load for all years divided by total flow for all years. However, this method can be confounded by both the time step of the input data (i.e. daily, monthly, yearly or event based) and the need to develop a mean or representative EMC from multiple time steps, events and/or sites. To overcome these issues, the Brigalow Catchment Study (BCS) has historically calculated a mean EMC as the arithmetic mean of all annual EMCs, where each annual EMC was calculated as the arithmetic mean of all event based EMCs in a year. Comments received during the Paddock to Reef independent review in October 2015 indicated that this method may be mathematically invalid, and similar comments were reiterated to authors during the review process for Elledge and Thornton (2017). A validation of the applicability of this method was required, as EMC data from the BCS has been used in APSIM, HowLeaky? and Source Catchments modelling which all underpin the Paddock to Reef Program.

Method

A comparison of methods for calculating a mean EMC was undertaken using 16 years of water quality data from the five catchments of the BCS (Figure A1). This data was collected from 2000 to 2015 during parts of the land use comparison (Stage III) and adaptive land management (Stage IV) phases. Table 1 in Section 2.2 shows the land use in these catchments. Note data in this appendix uses different catchments and time periods compared to the rest of the report. Further details on these catchments are provided in other documents (Cowie *et al.* 2007; Elledge and Thornton 2017; Radford *et al.* 2007; Thornton *et al.* 2007; Thornton and Elledge 2013).

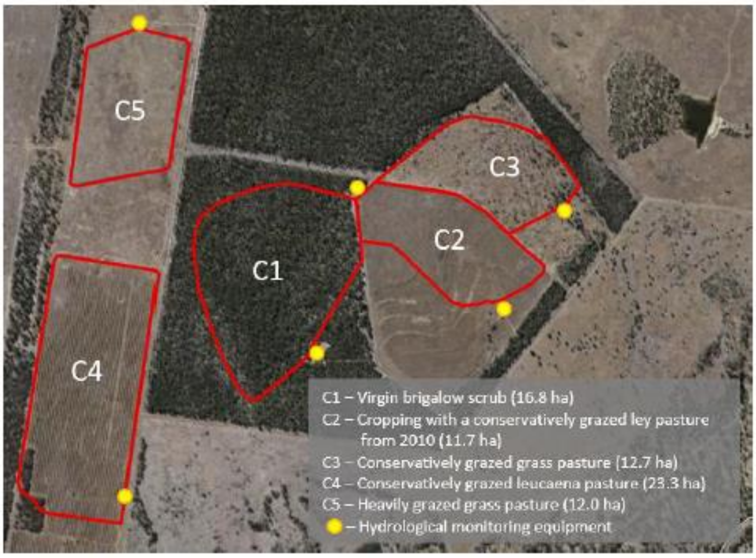

Figure A1: Aerial photo of the Brigalow Catchment Study showing the hydrological (runoff) boundaries and location of monitoring equipment within the five catchments.

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All available water quality data from the five catchment was used, including total and dissolved fractions of solids, nitrogen, phosphorus and carbon. Four methods were used to calculate a mean EMC:

(1) Total load for all years divided by total flow for all years (traditional method);
(2) Arithmetic mean of all event based EMCs, where each EMC was calculated as total load for an event divided by total flow for an event;
(3) Arithmetic mean of all annual EMCs, where each annual EMC was calculated as the arithmetic mean of all event based EMCs in a year (historically used for BCS data including the water quality results in this report);
(4) Arithmetic mean of all annual mean concentrations (AMCs), where each AMC was calculated as total load in a year divided by total flow in a year.

The EMCs for Methods 2 to 4 were plotted against the EMC for Method 1, and a regression analysis was performed to determine their correlation.

Results and Discussion

Three alternative methods for calculating a mean EMC were compared to the traditional method (Figure A2). Regression analyses showed that between 95% and 97% of the variability can be explained by the linear models, indicating that all four methods are equally valid. The BCS will continue to use the arithmetic mean of all annual EMCs to calculate a long-term EMC.

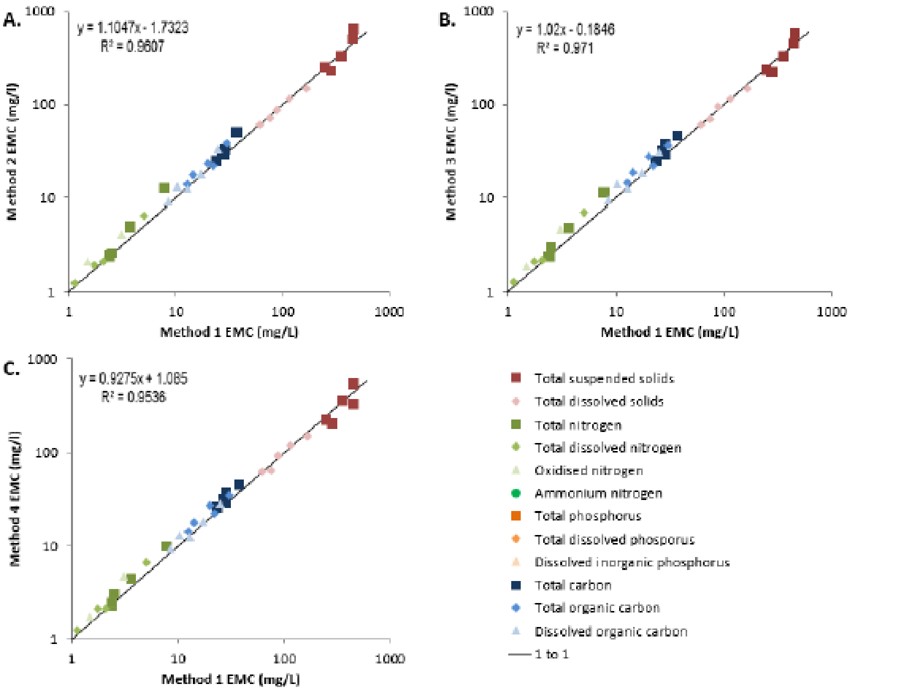

Figure A2: Three alternative methods for calculating an event mean concentration (EMC) compared to the traditional method (Method 1) using 16 years of water quality data from the Brigalow Catchment Study. Note that not all parameters are visible due to overlaying data points and very low values.

Thornton and Elledge 2018

Appendix 2: Tabulated Annual Loads and EMCs

Table A1: 2015 hydrological year loads and event mean concentrations (EMCs) for total suspended solids, nitrogen and phosphorus in runoff.

	Parameter	Brigalow scrub	Conservative grazing	Heavy grazing
TSS	Total load (kg/ha/yr)	1	20	101
	Mean EMC (mg/L)	No data	99	321
TN	Total load (kg/ha/yr)	0.03	0.70	0.69
	Mean EMC (mg/L)	No data	7.06	2.37
PN	Total load (kg/ha/yr)	0.01	0.36	0.40
	Mean EMC (mg/L)	No data	4.03	1.39
TDN	Total load (kg/ha/yr)	0.02	0.35	0.28
	Mean EMC (mg/L)	No data	3.03	0.98
DON	Total load (kg/ha/yr)	0.01	0.18	0.20
	Mean EMC (mg/L)	No data	1.25	0.69
DIN	Total load (kg/ha/yr)	0.01	0.17	0.08
	Mean EMC (mg/L)	No data	1.78	0.29
TP	Total load (kg/ha/yr)	<0.01	0.10	0.17
	Mean EMC (mg/L)	No data	1.00	0.58
PP	Total load (kg/ha/yr)	<0.01	0.06	0.09
	Mean EMC (mg/L)	No data	0.68	0.31
TDP	Total load (kg/ha/yr)	<0.01	0.04	0.08
	Mean EMC (mg/L)	No data	0.32	0.28
DOP	Total load (kg/ha/yr)	<0.01	0.01	0.01
	Mean EMC (mg/L)	No data	0.05	0.04
DIP	Total load (kg/ha/yr)	<0.01	0.03	0.07
	Mean EMC (mg/L)	No data	0.27	0.23

Table A2: 2016 hydrological year loads and event mean concentrations (EMCs) for total suspended solids, nitrogen and phosphorus in runoff.

	Parameter	Brigalow scrub	Conservative grazing	Heavy grazing
TSS	Total load (kg/ha/yr)	No runoff	36	36
	Mean EMC (mg/L)	No runoff	456	125
TN	Total load (kg/ha/yr)	No runoff	0.47	0.79
	Mean EMC (mg/L)	No runoff	5.92	2.80
PN	Total load (kg/ha/yr)	No runoff	0.22	0.17
	Mean EMC (mg/L)	No runoff	2.78	0.61
TDN	Total load (kg/ha/yr)	No runoff	0.25	0.62
	Mean EMC (mg/L)	No runoff	3.13	2.18
DON	Total load (kg/ha/yr)	No runoff	0.10	0.26
	Mean EMC (mg/L)	No runoff	1.30	0.92
DIN	Total load (kg/ha/yr)	No runoff	0.15	0.36
	Mean EMC (mg/L)	No runoff	1.83	1.26
TP	Total load (kg/ha/yr)	No runoff	0.05	0.14
	Mean EMC (mg/L)	No runoff	0.61	0.49
PP	Total load (kg/ha/yr)	No runoff	0.02	0.03
	Mean EMC (mg/L)	No runoff	0.32	0.11
TDP	Total load (kg/ha/yr)	No runoff	0.02	0.11
	Mean EMC (mg/L)	No runoff	0.30	0.38
DOP	Total load (kg/ha/yr)	No runoff	<0.01	0.01
	Mean EMC (mg/L)	No runoff	0.04	0.05
DIP	Total load (kg/ha/yr)	No runoff	0.02	0.10
	Mean EMC (mg/L)	No runoff	0.25	0.34

Thornton and Elledge 2018

Table A3: 2018 hydrological year loads and event mean concentrations (EMCs) for total suspended solids, nitrogen and phosphorus in runoff.

Parameter		Brigalow scrub	Conservative grazing	Heavy grazing
TSS	Total load (kg/ha/yr)	0.7	0.3	47
	Mean EMC (mg/L)	No data	No data	257
TN	Total load (kg/ha/yr)	0.02	<0.01	0.38
	Mean EMC (mg/L)	No data	No data	1.99
PN	Total load (kg/ha/yr)	0.01	<0.01	0.26
	Mean EMC (mg/L)	No data	No data	1.41
TDN	Total load (kg/ha/yr)	0.01	<0.01	0.12
	Mean EMC (mg/L)	No data	No data	0.58
DON	Total load (kg/ha/yr)	<0.01	<0.01	0.07
	Mean EMC (mg/L)	No data	No data	0.37
DIN	Total load (kg/ha/yr)	0.01	<0.01	0.05
	Mean EMC (mg/L)	No data	No data	0.21
TP	Total load (kg/ha/yr)	<0.01	<0.01	0.08
	Mean EMC (mg/L)	No data	No data	0.40
PP	Total load (kg/ha/yr)	<0.01	<0.01	0.05
	Mean EMC (mg/L)	No data	No data	0.26
TDP	Total load (kg/ha/yr)	<0.01	<0.01	0.03
	Mean EMC (mg/L)	No data	No data	0.15
DOP	Total load (kg/ha/yr)	<0.01	<0.01	<0.01
	Mean EMC (mg/L)	No data	No data	0.02
DIP	Total load (kg/ha/yr)	<0.01	<0.01	0.03
	Mean EMC (mg/L)	No data	No data	0.13

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Appendix 3: Publications

Journal Papers

Three journal papers that used BCS data were published during the funded period:

(1) Elledge A. and Thornton C. (2017). Effect of changing land use from virgin brigalow (*Acacia harpophylla*) woodland to a crop or pasture system on sediment, nitrogen and phosphorus in runoff over 25 years in subtropical Australia. *Agriculture, Ecosystems and Environment* **239**, pp. 119-131.
(2) Thornton C. and Elledge A. (2016). Tebuthiuron movement via leaching and runoff from grazed Vertisol and Alfisol soils in the Brigalow Belt bioregion of central Queensland, Australia. *Journal of Agricultural and Food Chemistry* **64** (20), pp. 3949-3959.
(3) Thornton C. M. and Yu B. (2016). The Brigalow Catchment Study: IV. Clearing brigalow (*Acacia harpophylla*) for cropping or grazing increases peak runoff rate. *Soil Research* **54** (6), pp. 749-759.

Conference Papers and Presentations

Three seminars that used BCS data were presented at conferences during the funded period:

(1) Elledge A. E. and Thornton C. M. (2018). The Brigalow Catchment Study: The impacts of developing Acacia harpophylla woodland for cropping or grazing on hydrology, soil fertility and water quality in the Brigalow Belt bioregion of Australia. Natural resource science in action: Connecting people, science and purpose, Toowoomba.
(2) Thornton C., Elledge A., Shrestha K., Wallace S., Bosomworth B. and Yu B. (2017). The Brigalow Catchment Study: The impacts of developing Acacia harpophylla woodland for cropping or grazing on hydrology, soil fertility and water quality in the Brigalow Belt bioregion of Australia. International interdisciplinary conference on land use and water quality: Effect of agriculture on the environment, The Hague, Netherlands.
(3) Thornton C. M. and Elledge A. E. (2018). The Brigalow Catchment Study: The impacts of developing Acacia harpophylla woodland for cropping or grazing on hydrology, soil fertility and water quality in the Brigalow Belt bioregion of Australia. Occasional Report No. 31. Farm environmental planning – Science, policy and practice, Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand. pp. 1-8.

Website

A portal for the BCS (www.brigalowcatchmentstudy.com) was developed during the funded period which provides access to rainfall and runoff data from all five monitored catchments, in addition to information on publications that have resulted from the long-term BCS.

Appendix 1.2: Elledge and Thornton (2017)

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journal homepage: www.elsevier.com/locate/agee

Effect of changing land use from virgin brigalow (*Acacia harpophylla*) woodland to a crop or pasture system on sediment, nitrogen and phosphorus in runoff over 25 years in subtropical Australia

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ABSTRACT

Native vegetation has been extensively cleared for agricultural systems worldwide, resulting in increased pollutant loads that often have adverse impacts downstream. This study uses 25 years of flow data and 10 years of sediment, nitrogen and phosphorus (total and dissolved) event mean concentrations from paired catchments to quantify the effect of changing land use from virgin brigalow (*Acacia harpophylla*) woodland in a semi-arid subtropical region of Australia into an unfertilised crop or conservatively grazed pasture system. Both the cropped and grazed catchments exported higher loads of sediment and phosphorus than the virgin brigalow catchment; however, the grazed catchment exported less total, oxidised and dissolved nitrogen than the virgin brigalow catchment. The cropped catchment exported higher loads of all water quality parameters compared to the grazed catchment. The simple hydrology and water quality model presented was effective for measuring the effect of land use change on runoff water quality. Variations in water quality between the three catchments are likely due to the presence of native legumes, ground cover, tillage practices and pasture rundown.

1. Introduction

Worldwide, the total area of forests in 2010 was estimated to be four billion hectares, or 31% of the total land area (Food and Agriculture Organization of the United Nations, 2010). Deforestation is typically associated with natural causes, such as fire and drought, and change of land use to agriculture. However, rates of net gain and loss vary between country and agro-ecological zones (Food and Agriculture Organization of the United Nations, 2010). For example, in Australia the Fitzroy Basin Land Development Scheme commenced in 1963 resulting in 4.5 Mha of virgin brigalow woodland being cleared for agriculture. This scheme continued through to the 1990s (Department of Lands, 1968; Partridge et al., 1994), with broad-scale clearing in Queensland only ceasing in 2006 (Thornton et al., 2012). In 2009, 74.8% (11.7 Mha) of the Fitzroy Basin was being used for agricultural purposes, with 71.5% grazed and 3.2% cropped (Australian Bureau of Statistics, 2009).

Pollutant loads exported in runoff have increased from natural rates as a consequence of broad-scale clearing of native vegetation and subsequent change of land use to agriculture. For example, Kroon et al. (2012) estimated that since European settlement mean annual loads exported from six catchments along the coast of Queensland, Australia, into the Great Barrier Reef have increased 5.5 times for total suspended sediment (17,000 kt yr^{-1}), 5.7 times for total nitrogen (80,000 t yr^{-1}) and 8.9 times for total phosphorus (16,000 t yr^{-1}). Transport of sediment and nutrients from the landscape into the Great Barrier Reef causes increased eutrophication and turbidity (Brodie et al., 2011; Hansen et al., 2002), which can lead to crown-of-thorns starfish (*Acanthaster planci*) outbreaks and coral mortality (Brodie and Waterhouse, 2012; De'ath et al., 2012). The impact of different agricultural activities on downstream water quality is an issue in common with other parts of Australia and the world (Barlow et al., 2007; Bossa et al., 2012; Brion et al., 2011; Dilshad et al., 1996; Jarvie et al., 2010; Lal, 1996; Singh and Mishra, 2014; Vink et al., 2007).

It is well documented that runoff volume and/or sediment load increase when native forest is cleared for agriculture (Cowie et al., 2007; Hunter and Walton, 2008; Siriwardena et al., 2006; Thornton et al., 2007). Numerous studies have also demonstrated higher runoff volume and/or sediment loads from cropped than grazed areas (Freebairn et al., 2009; Murphy et al., 2013; Stevens et al.,

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2006; Wilson et al., 2014). However, studies that have reported nutrient loads from agricultural systems tend to focus on total loads rather than dissolved loads (O'Reagain et al., 2005; Povilaitis et al., 2014; Stevens et al., 2006; Wilson et al., 2014). Dissolved nutrients pose a great risk to aquatic systems, as they are less likely to settle than nutrients bound to sediment (Silburn et al., 2007). For example, Devlin and Brodie (2005) mapped flood plumes from rivers exporting into the Great Barrier Reef over nine years and found that most suspended solids and associated particulate nutrients were deposited within 10 km of the river mouth while dissolved nutrients were transported with the plume 50–200 km from the river mouth.

Studies that have reported both total and dissolved nutrients are typically at the catchment scale (Joo et al., 2012; Li et al., 2014; Packett et al., 2009), but catchments often have multiple land uses within the monitored area so it is difficult to separate the impacts of each land use on water quality (Bartley et al., 2012; Li et al., 2014; Povilaitis et al., 2014). Bartley et al. (2012) reviewed 755 sediment, nitrogen and phosphorus data points from studies across Australia for use in catchment scale water quality models. They found that a catchment with less than 90% of a specific land use could have its water quality signature influenced by the other land uses, whereas a catchment dominated by a single land use (>90%) was a more appropriate representation of that specific land use. However, using data from sites with more than 90% of the area dominated by a single land use dramatically reduced the number of data points and also biased data towards smaller plot sizes for intensive land uses, such as sugar cane, which rarely cover large areas of a catchment (Bartley et al., 2012). Thus, there is currently a paucity of total and dissolved water quality data from areas greater than plot scale that are dominated by a single land use.

This study investigates the impact of changing land use from a virgin brigalow woodland into a crop or pasture system on runoff water quality. It models data based on a 17 year calibration period of three catchments in their virgin condition before changing the land use of two catchments to agriculture, and subsequent monitoring of all three catchments to collect 25 years flow and 10 years water quality data. The model presented uses long-term event mean concentrations (EMCs) with a regression based flow model described by Thornton et al. (2007). This research is unique as it: 1) reports on total and dissolved nitrogen and phosphorus in addition to sediment; and 2) compares both cropped and grazed catchments with a virgin woodland control catchment. This study improves understanding on the impact of agriculture on runoff water quality relative to the pre-European landscape and provides a comparison of water quality from crop and pasture systems.

2. Methods

2.1. Site description

The Brigalow Catchment Study (24°48′S and 149°47′E) is a paired, calibrated catchment study located near Theodore in central Queensland, Australia (Fig. 1). It was established in 1965 to quantify the impact of land development for agriculture on hydrology, productivity and resource condition (Cowie et al., 2007). The study site was selected to represent the Brigalow Belt Bioregion which covers an area approximately 36.7 Mha from Townsville in north Queensland to Dubbo in central-western New South Wales (Thornton et al., 2007). The site in its native state was dominated by brigalow (*Acacia harpophylla*) trees, either in a monoculture or in association with other species, such as belah (*Casuarina cristata*) and Dawson River blackbutt (*Eucalyptus cambageana*) (Johnson, 2004). The extant uncleared vegetation at the Brigalow Catchment Study is classified as regional ecosystems 11.4.8, woodland to open forest dominated by *Eucalyptus cambageana* and *Acacia harpophylla*, and 11.4.9, open forest and occasionally woodland dominated by *Acacia harpophylla* (Queensland Government, 2014). Slope of the land averages 2.5% (range from 1.8 to 3.5%) and soils are an association of black and grey Vertosols, black and grey Dermosols, and black and brown Sodosols. Vertosols and Dermosols (clay soils) cover approximately 70% of Catchments 1 and 2, and 58% of Catchment 3; Sodosols cover the remaining area (Cowie et al., 2007). These soil types are representative of 67% of the Fitzroy Basin under grazing: 28% Vertosols, 28% Sodosols and 11.3% Dermosols (Roots, 2016). The region has a semi-arid, subtropical climate and mean annual hydrological year (October 1965 to September 2014) rainfall at the site was 661 mm.

2.2. Calibration and development of catchments

Three contiguous catchments were monitored for rainfall and runoff from 1965 to 1982 (17 years). Each catchment was instrumented to measure runoff using a 1.2 m steel HL flume with a 3.9 × 6.1 m concrete approach box. Water heights through the flumes were recorded using mechanical float recorders and converted to discharge using a rating table. Rainfall was recorded adjacent to each flume and at the top of the catchments using a tipping bucket rain gauge (Thornton et al., 2007). A runoff event was defined as commencing when stage height exceeded zero and finished when it returned to zero. These data were used to derive mathematical relationships to predict runoff from Catchment 2 (C2) and Catchment 3 (C3) given known runoff from Catchment 1 (C1) (Thornton et al., 2007). During this period, it was found that C2 and C3 in their uncleared state had 95% and 72% of the runoff from C1, respectively. Each catchment had its own intrinsic hydrological signature; for example, C3 had more runoff events but less total runoff volume on an annual basis compared to C1 and C2. Nonetheless, approximately 5% of the mean annual rainfall become runoff in all three catchments (Thornton et al., 2007).

Land development occurred between 1982 and 1983; that is, C1 remained virgin brigalow woodland to provide an uncleared control treatment, while C2 and C3 were cleared using a chain dragged between two dozers and the fallen timber burnt in-situ (Cowie et al., 2007). C2 was then developed for cropping with the construction of contour banks and grassed waterways, while C3 was developed for grazing by the planting of improved buffel grass pasture (Fig. 2).

2.3. Land use comparisons

Rainfall and runoff were monitored from the virgin brigalow woodland (C1), cropped (C2) and grazed (C3) catchments from 1984 until 2010 (Thornton and Elledge, 2013). This equates to 25 full hydrological years (October to September) monitoring and two incomplete hydrological years; July 1984 to September 1984, and October 2009 to January 2010. Over the 25 years, C2 had one sorghum crop followed by nine monoculture wheat crops, and then was opportunity cropped with sorghum (*Sorghum bicolor*), wheat (*Triticum* spp.), barley (*Hordeum vulgare*) or chick peas (*Cicer arietinum*). Zero or reduced till fallows were introduced in 1990. There were no fertiliser inputs in the cropped catchment (Radford et al., 2007). C3 was grazed at industry recommended stocking rates with utilisation to result in no less than 1000 kg ha^{-1} of pasture available at any time. Conservative management of this catchment has resulted in groundcover averaging 91% since 2000 (earlier data not available), which is greater than paddocks of the same land type within a 50 km radius which averaged only 74% (Fitzroy Basin Association, 2016). The foliage projective cover of tree regrowth in C3 has remained below 15% (Department of Science, Information Technology and Innovation, 2016). There was

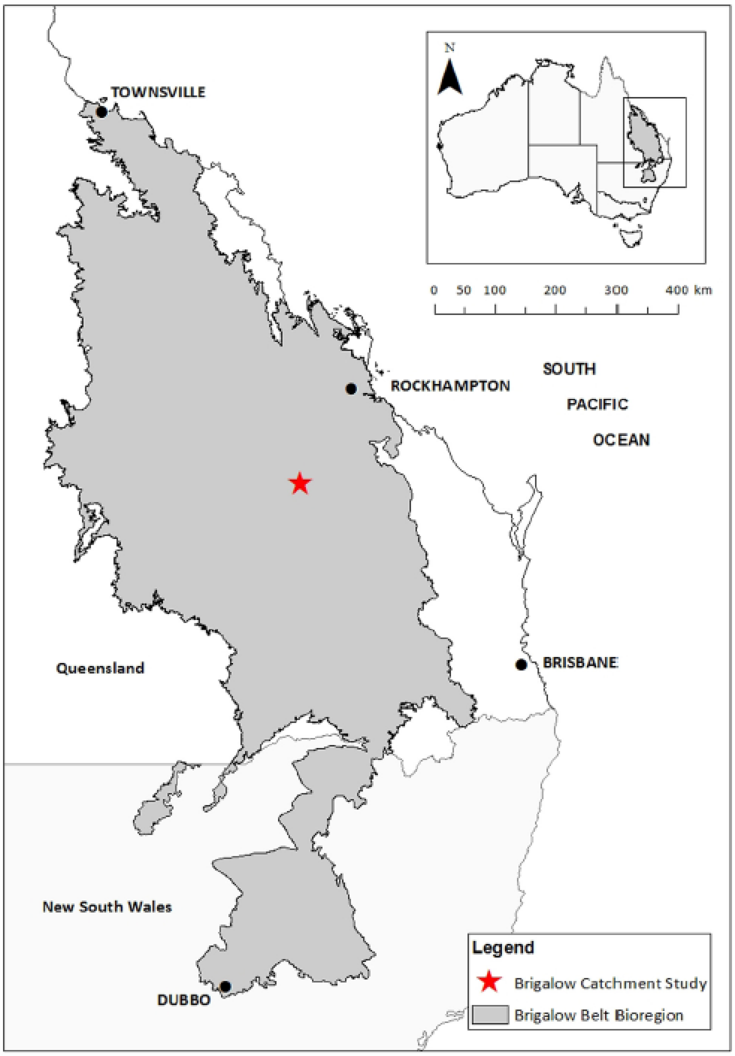

Fig. 1. Location of the Brigalow Catchment Study within the Brigalow Belt Bioregion of central Queensland, Australia.

no fertiliser inputs or supplement feeding in the pasture catchment (Radford et al., 2007).

Discrete water quality samples were obtained using auto-samplers from 2000 to 2010. Auto-samplers were programmed to sample every 0.1 m change in absolute stage height. Runoff samples were analysed for seven parameters by Queensland Health Forensic and Scientific Services (https://www.health.qld.gov.au/qhcss/qhss/) (Table 1).

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 midpoint of sample one and sample two, the midpoint of sample one and sample two to the midpoint of sample two and sample three, and so on. Where samples were only collected on the rising limb of the hydrograph, the event peak was considered to be the end of the sampling interval for the last discrete sample, and the mean concentration of the discrete samples was applied to flow from the event peak to the event end. Event based EMCs were calculated by dividing total event load by 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. Where missing flow data occurred, estimations were obtained using the "HowLeaky?" modelling approach of Thornton et al. (2007). Where missing water quality data occurred, estimations were obtained by multiplying the long-

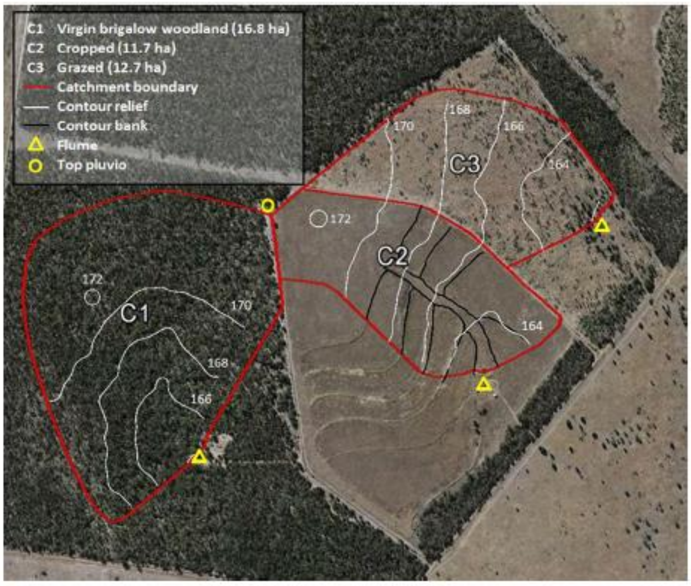

Fig. 2. Aerial photo of the three catchments monitored at the Brigalow Catchment Study following land use change of two catchments from virgin brigalow woodland to crop and pasture systems.

term EMC by the observed flow. Mean annual load was calculated by dividing the cumulative observed load for each catchment by the number of full hydrological years monitoring data (n = 25). The effect of changing land use from virgin brigalow woodland to crop or pasture systems on sediment, nitrogen and phosphorus loads (kg ha^{-1}) on an event basis were calculated by: (Table 2)

$$\left(\frac{(Q_{Obs} \times EMC_{Current}) - (Q_{Est} \times EMC_{Brigalow})}{1,000,000}\right) \div Area$$

Observed load was calculated by multiplying the observed event flow from 1984 to 2010 by the long-term EMC (2000 to 2010) for the respective catchment. Predicted load was calculated by multiplying the estimated flow of C2 and C3 had they remained virgin brigalow woodland (using the relationship of flow between the catchments during the calibration phase from 1965 to 1982; for example, C2 in an uncleared state had 95% of the runoff from C1) by the EMC for the virgin brigalow catchment. Mean annual land use change effect was calculated by dividing the cumulative difference in observed and predicted loads by the number of full hydrological years of monitoring data (n = 25). The assumptions of this approach are that water quality from the three catchments in their virgin state would have been similar, and that the long-term EMC values for C1 apply to all catchments had they remained virgin brigalow woodland.

3. Results

3.1. Hydrology

Total annual rainfall exceeded the long-term mean annual rainfall of 661 mm for the Brigalow Catchment Study in 10 out of the 25 full hydrological years monitored (Fig. 3). Observed mean annual runoff from the cropped and grazed catchments were 2.48 times (65.8 mm) and 1.97 times (52.2 mm) greater than observed runoff from the virgin brigalow woodland (26.5 mm), respectively. Similarly, observed runoff from the cropped catchment was 2.60 times greater than predicted runoff from this catchment had it remained uncleared (25.3 mm), and observed runoff from the

Table 1
Methods used by Queensland Health Forensic and Scientific Services for sediment, nitrogen and phosphorus analyses of water samples.

Parameter	Method
Total Suspended Solids	Method 18211 based on gravimetric quantification of solids in water.
Total Nitrogen	Method 13802 by simultaneous persulfate digestion. For the period 2000 to 2003, method 13804 based on simultaneous Kjeldahl digestion was reported and total nitrogen was manually calculated as total Kjeldahl nitrogen + oxidised nitrogen.
Oxidised Nitrogen	Method 13798 based on flow injection analysis of nitrogen as oxides.
Ammonium Nitrogen	Method 13796 based on flow injection analysis of nitrogen as ammonia.
Dissolved Inorganic Nitrogen	Manually calculated as oxidised nitrogen + ammonium nitrogen.
Total Phosphorus	Method 13800 by simultaneous persulfate or Kjeldahl digestion.
Dissolved Inorganic Phosphorus	Method 13799 by flow injection analysis; also known as orthophosphate.

Table 2
Model parameters were defined as follows.

Parameter		Description
Q_{Obs}	=	Observed discharge from the catchment under current land use (L event^{-1})
$EMC_{Current}$	=	Observed long-term event mean concentration from the catchment under current land use (mg L^{-1})
Q_{Est}	=	Estimated discharge from the catchment had it remained virgin brigalow woodland (L event^{-1}) (Thornton et al., 2007)
$EMC_{Brigalow}$	=	Observed long-term event mean concentration from the virgin brigalow catchment (mg L^{-1})
Area	=	Catchment area (ha)

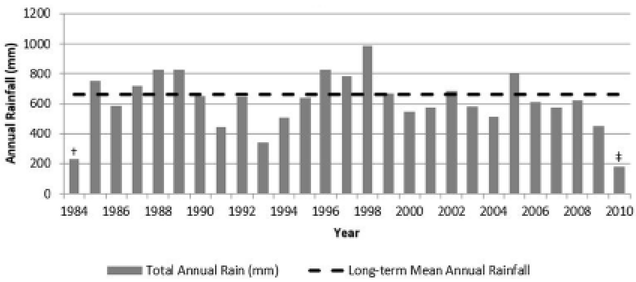

Fig. 3. Total annual hydrological year rainfall (mm) for 1984 to 2010 relative to the long-term mean annual rainfall for the Brigalow Catchment Study. † Total rainfall only from 25/07/1984, as this relates to the first runoff event recorded at the Brigalow Catchment Study following land development. ‡ Total rainfall only to 19/01/2015, as event data after this date was excluded from the presented model due to a change in management practices.

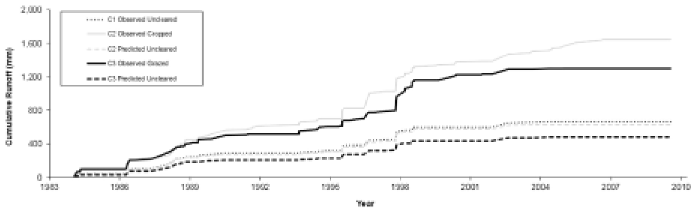

Fig. 4. Cumulative runoff (mm) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted runoff for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

grazed catchment was 2.74 times greater than predicted runoff from this catchment had it remained uncleared (19.0 mm). The rate of increase in cumulative runoff was greater in years with above average rainfall, particularly from 1987 to 1989 and 1996 to 1999 (Fig. 4). Over the 25 year period, the virgin brigalow catchment discharged a total of 663 mm runoff over 45 days, the cropped catchment discharged a total of 1647 mm runoff over 99 days, and the grazed catchment discharged a total of 1304 mm runoff over 80 days.

Table 3
Long-term event mean concentrations (mg L^{-1}) of sediment, nitrogen and phosphorus for the virgin brigalow woodland, cropped and grazed pasture catchments over 10 years (2000–2010).

Parameter	Event Mean Concentration (mg L^{-1})		
	Woodland (C1)	Crop (C2)	Pasture (C3)
Total Suspended Solids	307	798	229
Total Nitrogen	9.85	5.37	2.17
Oxidised Nitrogen	6.27	2.17	0.07
Ammonium Nitrogen	0.06	0.11	0.04
Dissolved Inorganic Nitrogen	6.32	2.27	0.11
Total Phosphorus	0.32	0.93	0.41
Dissolved Inorganic Phosphorus	0.12	0.35	0.22

3.2. Event mean concentrations

Long-term EMCs for the three monitored catchments from 2000 to 2010 are presented in Table 3. Concentrations of total, oxidised and dissolved inorganic nitrogen from virgin brigalow woodland were 1.83, 2.89 and 2.78 times greater than concentrations from the cropped catchment and 4.53, 95.10 and 59.89 times greater than concentrations from the grazed catchment, respectively. In contrast, concentrations of total suspended solids, total and dissolved inorganic phosphorus, and ammonium nitrogen from the cropped catchment were 2.60, 2.90, 3.00 and 1.73 times greater than concentrations from the virgin brigalow catchment and 3.49, 2.26, 1.57 and 2.67 times greater than concentrations from the grazed catchment, respectively.

Overall, the proportion of dissolved inorganic phosphorus that comprised total phosphorus was 37% from the virgin brigalow catchment, 38% from the cropped catchment and 55% from the grazed catchment. The proportion of ammonium nitrogen that comprised dissolved inorganic nitrogen was 1% from the virgin brigalow catchment, 5% from the cropped catchment and 38% from the grazed catchment.

3.3. Sediment, nitrogen and phosphorus loads

Cumulative loads of sediment, nitrogen and phosphorus are presented in Figs. 5–11. Similar to runoff, the rate of increase in cumulative loads was greater in years with above average rainfall, particularly from 1987 to 1989 and 1996 to 1999.

The cropped catchment exported more sediment and nutrients (total and dissolved) than the grazed catchment over the 25 year period (Table 4). Changing land use from virgin brigalow woodland to either agricultural system increased loads of total suspended solids, total and dissolved inorganic phosphorus, and ammonium nitrogen; the cropped catchment exported 6.45, 7.21, 7.45 and 4.29 times greater loads and the pasture catchment exported 1.46, 2.53, 3.75 and 1.27 times greater loads, respectively. In contrast, loads of oxidised and dissolved inorganic nitrogen were higher from virgin brigalow woodland than both agricultural systems; 1.16 and 1.12 times greater than loads from the cropped catchment and 48.34 and 30.44 times greater than loads from the grazed catchment, respectively. The virgin brigalow and cropped catchments exported 2.30 and 3.12 times greater total nitrogen than the pasture catchment, respectively.

Observed mean annual loads of total suspended solids, total phosphorus and dissolved inorganic phosphorus from the cropped catchment were 6.88, 7.70 and 7.95 times greater, respectively, than predictions from this catchment had it remained uncleared (Table 4). Total and ammonium nitrogen were also 1.42 and 4.57 times greater than uncleared predictions, whereas uncleared predictions of oxidised and dissolved inorganic nitrogen were conversely 1.09 and 1.05 times greater than the observed means from this catchment under cropping, respectively. Observed mean annual loads of total suspended solids, total and dissolved inorganic phosphorus, and ammonium nitrogen from the grazed catchment were 1.80, 3.11, 4.61 and 1.56 times greater, respectively, than predictions from this catchment had it remained uncleared (Table 4). In contrast, uncleared predictions of total, oxidised and dissolved inorganic nitrogen were 1.65, 39.36 and 24.79 times greater than the observed means from this catchment under grazing, respectively.

3.4. Effect of land use change on water quality

Over the 25 year period, the mean annual effect of changing land use from virgin brigalow woodland to crop or pasture resulted in 449 kg ha^{-1} yr^{-1} and 53 kg ha^{-1} yr^{-1} more total suspended solids in runoff, respectively (Table 5). Similarly, more total phosphorus, dissolved inorganic phosphorus and ammonium nitrogen were exported from crop and pasture systems than virgin brigalow woodland. Crops exported total nitrogen at an average rate of 1.04 kg ha^{-1} yr^{-1} more than if the catchment had remained uncleared, whereas pasture exported 0.74 kg ha^{-1} yr^{-1} less than if the catchment had remained uncleared. Although the cropped catchment exported more total nitrogen than its uncleared predictions, less oxidised and dissolved inorganic nitrogen were exported.

4. Discussion

4.1. Event mean concentrations

The simple hydrology and water quality model presented was effective at quantifying the effect of changing land use from virgin brigalow woodland to crop and pasture systems; however, it is likely that the results are an underestimate of the true change. Although 25 years (1984–2010) of flow data was available for these three catchments, comprehensive water quality data had only been collected for the last 10 years of this period (2000–2010). If sediment and nutrient EMCs from the agricultural catchments immediately after land use change were higher to that observed later in the study, the model would underestimate change. For example, wildfires in natural areas have been reported to increase sediment, nitrogen and phosphorus losses in runoff which typically remain high for about a year or until the soil stabilises and vegetation establishes (Ice et al., 2004; Smith et al., 2011; Stein et al., 2012). Historical cover data for the two agricultural catchments in this study indicate that plant cover was established by December 1983; C2 had 53% cover in June and 95% cover before the first crop was harvested in December, whereas C3 had 6.5% pasture cover in June and 96% cover by December. However, it is possible that the earlier runoff events may have had elevated sediment and nutrients in runoff as a residual impact of clearing and burning the catchments despite established cover.

Nonetheless, this study provides a rigorous estimate of sediment, nitrogen and phosphorus (total and dissolved) loads exported in runoff from these three catchments over 25 years. Bartley et al. (2012) reviewed sediment and nutrient concentration data from Australia suitable for catchment water quality models. Where upstream land use was dominated by more than 90% modified grazed pasture, they reported concentrations of 322 mg L^{-1} (10th and 90th percentiles 39 and 390 mg L^{-1}; n = 9 sites) for total suspended solids, 3.04 mg L^{-1} (10th and 90th 1.65 and 4.92 mg L^{-1}; n = 9 sites) for total nitrogen, and 0.73 mg L^{-1} (10th and 90th percentiles 0.17 and 2.17 mg L^{-1}; n = 17 sites) for total phosphorus. EMCs from the grazed catchment in this study for total suspended solids (229 mg L^{-1}), total nitrogen (2.17 mg L^{-1}) and total phosphorus (0.41 mg L^{-1}) are within the range of values reported by Bartley et al. (2012).

There were no dryland crop sites in the review by Bartley et al. (2012) that were dominated (> 90%) by a single upstream land use. However, plot and catchment scale data for sites with dryland crops as the main land use reported concentrations of 2501 mg L^{-1} (10th and 90th percentiles 162 and 5339 mg L^{-1}; n = 21 sites) for total suspended solids, 1.99 mg L^{-1} (10th and 90th 0.71 and 3.38 mg L^{-1}; n = 17 sites) for total nitrogen, and 0.85 mg L^{-1} (10th and 90th 0.096 and 1.65 mg L^{-1}; n = 17 sites) for total phosphorus (Bartley et al., 2012). As for the grazed catchment, EMCs from the cropped catchment in this study were within the range of values reported by Bartley et al. (2012) for total suspended solids (798 mg L^{-1}) and total phosphorus (0.93 mg L^{-1}), but total nitrogen (5.37 mg L^{-1}) values from the cropped catchment in this study were higher despite no fertiliser applications. The lower total nitrogen values reported by Bartley et al. (2012) may be partly explained by: (1) diversity of study locations, including variations in the physical and chemical structure of soil; (2) data collection from different spatial scales (plot versus small, medium and large catchments); and (3) less than 90% of the upstream catchment areas were dominated by dryland crops, which due to a potential matrix of soil type, land use and ground cover provide a less accurate comparison than if data was collected from a single land use.

Soil characteristics and land use history are of particular interest when comparing runoff water quality studies, as physically more sediment and particulate nutrients are expected from sodic soils which readily erode (Gray and Murphy, 2002) and chemically soil fertility declines over time. For example, total soil nitrogen (0–10 cm) has been shown to decline with an increase in cropping history ranging from 0 to 70 years (Dalal and Mayer, 1986a, 1986b). Following colonisation of Australia in 1788, clearing land for agriculture started in the southern states and slowly headed north to Queensland (Australian Government, 2015). For example, 85% (407,840 ha) of cropping in Australia was conducted in the southern states of Victoria, South Australia and New South Wales in 1860 with only 0.3% (1357 ha) occurring in Queensland

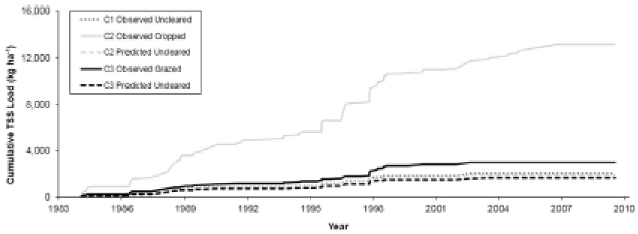

Fig. 5. Cumulative load (kg ha^{-1}) of total suspended sediments (TSS) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted loads for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

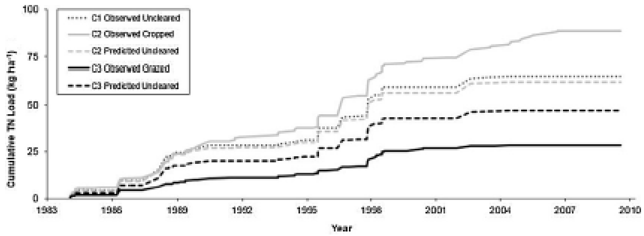

Fig. 6. Cumulative load (kg ha^{-1}) of total nitrogen (TN) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted loads for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

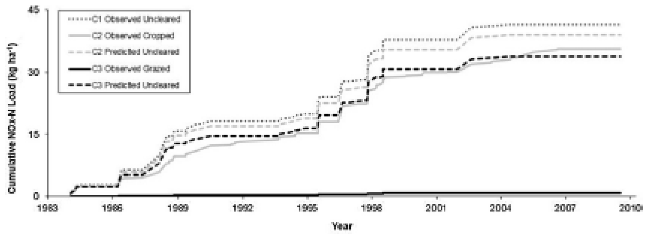

Fig. 7. Cumulative load (kg ha^{-1}) of oxidised nitrogen (NOx-N) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted loads for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

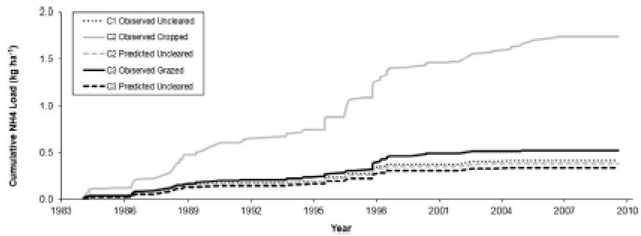

Fig. 8. Cumulative load (kg ha^{-1}) of ammonium nitrogen (NH_4-N) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted loads for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

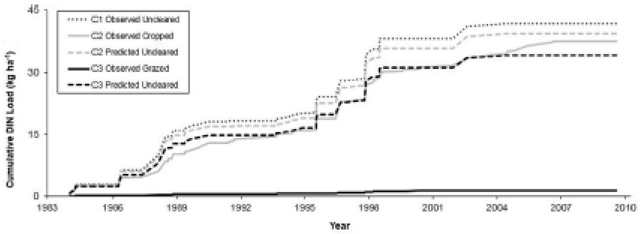

Fig. 9. Cumulative load (kg ha^{-1}) of dissolved inorganic nitrogen (DIN) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted loads for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

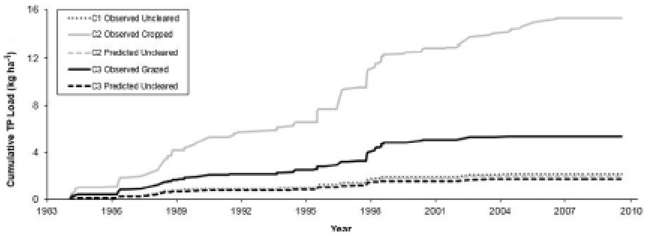

Fig. 10. Cumulative load (kg ha^{-1}) of total phosphorus (TP) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted loads for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

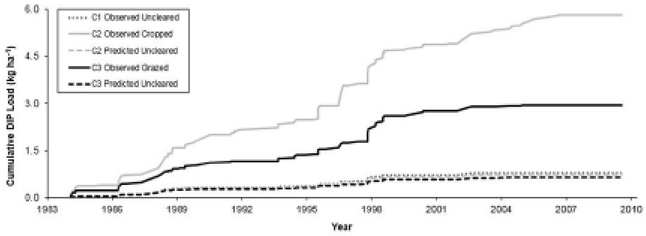

Fig. 11. Cumulative load (kg ha^{-1}) of dissolved inorganic phosphorus (DIP) from the virgin brigalow woodland (C1), crop (C2) and pasture (C3) catchments, and predicted loads for the cropped and grazed catchments had they remained virgin brigalow woodland. Data for the period July 1984 to January 2010; however, no events occurred between March 2007 and January 2010.

Table 4
Observed mean annual sediment, nitrogen and phosphorus loads (kg ha^{-1} yr^{-1}) from the virgin brigalow woodland, cropped and grazed pasture catchments over 25 years (1984–2010); and predicted mean annual loads from the cropped and grazed catchments had they remained virgin brigalow woodland.

Parameter	Load (kg ha^{-1} yr^{-1})				
	Woodland (C1)	Crop (C2)	Pasture (C3)	C2 Predicted Uncleared	C3 Predicted Uncleared
Total Suspended Solids	81	525	119	76	66
Total Nitrogen	2.61	3.53	1.13	2.49	1.87
Oxidised Nitrogen	1.66	1.43	0.03	1.56	1.35
Ammonium Nitrogen	0.02	0.07	0.02	0.02	0.01
Dissolved Inorganic Nitrogen	1.68	1.50	0.06	1.57	1.37
Total Phosphorus	0.08	0.61	0.21	0.08	0.07
Dissolved Inorganic Phosphorus	0.03	0.23	0.12	0.03	0.03

Table 5
Mean annual effect of changing land use from virgin brigalow woodland to crop and pasture systems on sediment, nitrogen and phosphorus loads (kg $ha^{-1} yr^{-1}$) over 25 hydrological years (1984–2010).

Parameter	Mean Annual Land Use Change Effect (kg $ha^{-1} yr^{-1}$)	
	Crop (C2)	Pasture (C3)
Total Suspended Solids	449	53
Total Nitrogen	1.04	−0.74
Oxidised Nitrogen	−0.13	−1.32
Ammonium Nitrogen	0.05	0.01
Dissolved Inorganic Nitrogen	−0.07	−1.31
Total Phosphorus	0.53	0.15
Dissolved Inorganic Phosphorus	0.20	0.09

(Australian Bureau of Statistics, 2007). As a result, soils in the southern states where cropping has occurred for over 150 years are likely to be less fertile than in the Fitzroy Basin of Queensland where land development for cropping only commenced about 50 years ago. The shorter history of cropping at this study site in the Fitzroy Basin would also explain, at least in part, the higher total nitrogen in runoff compared to other areas of Australia which were included in the Bartley et al. (2012) review.

In contrast to total nutrients, there are limited data available on dissolved concentrations of nitrogen and phosphorus in runoff from cropped and/or grazed areas. Murphy et al. (2013) reported 5.9 mg L^{-1} oxidised nitrogen and 0.017 mg L^{-1} dissolved phosphorus from a cropped area over one wet season; whereas concentrations over 10 years used in this study were 2.17 mg L^{-1} and 0.14 mg L^{-1}, respectively. The paucity of studies that have reported on dissolved nutrients from comparative single land use systems over enough wet seasons to account for annual variability makes interpretation difficult at present. However, the EMCs used in the model presented in this study are within the range of sediment and total nitrogen and phosphorus values reported from other cropped and grazed sites. This suggests that the results are comparable to other areas dominated by similar agricultural systems.

4.2. Effect of land use change on water quality

Differences in runoff volume between the catchments can be attributed to variable water use patterns of the different vegetation types with ground cover, structural decline and surface roughness being secondary factors (Thornton et al., 2007). Clearing virgin brigalow woodland for agriculture is known to increase runoff volume (Siriwardena et al., 2006; Thornton et al., 2007), and it is well established that runoff volume and sediment loads are higher from cropped than grazed areas (Freebairn et al., 2009; Murphy et al., 2013; Sharpley and Smith, 1994; Silburn et al., 2007; Stevens et al., 2006). Both these trends were observed in this study. However, Australian literature currently provides an incomplete story on the impacts of changing land use for these two agricultural systems on nutrients in runoff. For example, Stevens et al. (2006) reported higher loads of total nitrogen and phosphorus from cropped than grazed areas but nothing on dissolved species, while Murphy et al. (2013) reported total and dissolved concentrations of nitrogen and phosphorus from cropped areas but nothing from grazed areas. This gap is also found in international studies; for example, in the southwestern United States of America, Sharpley and Smith (1994) reported higher loads of nitrogen and phosphorus (total and dissolved) following change of native grasslands to conventional tilled (fertilised) wheat but nothing from grazed areas. This highlights the uniqueness of this study's design which has collected long-term data on total and dissolved nutrients in runoff from both cropping and grazed areas concurrently with an uncleared control. In this study, more sediment and phosphorus (total and dissolved) were exported in runoff from both agricultural systems than virgin brigalow woodland. Changing land use to a pasture system also had less impact on runoff water quality than changing land use to a crop system for all sediment, nitrogen and phosphorus parameters reported.

The findings in this study also support other research which have reported a correlation between sediment and total phosphorus loss, and runoff and dissolved inorganic phosphorus loss (Gillingham and Thorrold, 2000; Hansen et al., 2002; Sharpley and Smith, 1990; Yuan et al., 2013). The considerably higher sediment loss from the cropped catchment is most likely the result of bare and/or low cover fallow management and tillage practices which are associated with erosion (Freebairn et al., 1993). Exports of sediment and total phosphorus increased at relatively proportional rates; however, the overall contribution of dissolved inorganic phosphorus to total phosphorus remained similar between the cropped catchment (38%) and its uncleared prediction (37%). This indicates that phosphorus from the cropped catchment was mainly exported in a particulate phase. Although this does not take into account the contribution of dissolved organic phosphorus which was not measured in this study, data from a nearby study has shown that dissolved organic phosphorus contributes only 3 to 5% of the total phosphorus load (Rogusz et al., 2013). This supports phosphorus from the cropped catchment being mainly exported in a particulate phase.

In contrast, the lower loss of sediment from the virgin brigalow and grazed catchments can be attributed to the higher proportion of litter and pasture cover, respectively, which protects the soil surface from raindrop impact. High ground cover also helps maintain high infiltration rates, which reduces runoff and subsequently erosion (Freebairn and Wockner, 1986; Silburn et al., 2011). Although conservative grazing of the unfertilised pasture resulted in only a 1.80 times increase in sediment compared to uncleared predictions for this catchment, total phosphorus increased 3.11 times and dissolved inorganic phosphorus 4.61 times. Furthermore, the overall contribution of dissolved inorganic phosphorus to total phosphorus increased from 37% for the uncleared prediction to 55% under grazing. The inclusion of dissolved organic phosphorus would increase the proportion of total phosphorus transported in the dissolved phase. The transport of phosphorus in mainly a dissolved phase is not surprising given the negative inverse relationship reported between dissolved phosphorus and sediment by Sharpley et al. (1981), and the management of this catchment to maintain high pasture cover and minimise erosion, which subsequently reduces particulate phosphorus loss (Sharpley et al., 1994).

The enrichment of dissolved phosphorus in runoff from the grazed catchment may also be explained by the presence of cattle, as grazing animals can return 60 to 99% of the nutrients they ingest back into the pasture system via dung and urine (Haynes and Williams, 1993). Dung is the main form that phosphorus is excreted from animals, and it often has a higher inorganic content than the pasture ingested (Haynes and Williams, 1993). For example, sheep dung has been shown to contain 80% inorganic phosphorus compared to only 64% from the pasture ingested

(Haynes and Williams, 1993). Australian data indicates that a 400 kg beef cattle steer maintaining body weight will excrete 2.8 kg of faecal dry matter per day (Department of Agriculture and Fisheries, 2011) which contains 2.1 g of phosphorus per kg of faecal dry matter (Jackson et al., 2012). Given the grazed catchment in this study is typically stocked at one 300 kg animal per 2.2 ha, approximately 0.71 kg $ha^{-1} yr^{-1}$ of phosphorus is returned to the soil surface via dung.

Virgin brigalow woodland at this site is representative of the broader brigalow landscape in its pre-European condition, and the high nitrogen concentrations in runoff relative to the agricultural systems are likely due to the leguminous brigalow (*Acacia harpophylla*) tree which dominates the vegetation community. Although the concentration of total nitrogen in runoff from the virgin brigalow catchment was higher than the unfertilised cropped catchment, the mean annual load exported was less. This is a function of greater runoff volume and the number of days on which runoff occurred from the cropped catchment; that is, a total of 1647 mm was discharged over 99 days from the cropped catchment compared to only 663 mm discharged over 45 days from the virgin brigalow catchment over the 25 year period. This trend is similarly reported by Thornton et al. (2007) who found that in the same catchment 5.7% of rainfall become runoff in an uncleared state which increased to 11.0% under cropping.

In contrast to total nitrogen, oxidised and dissolved inorganic nitrogen both had higher concentrations and loads from virgin brigalow woodland compared to cropping. The contribution of particulate nitrogen to the total cumulative load of total nitrogen was 36% for virgin brigalow woodland and 58% for cropping; where particulate nitrogen was calculated as total nitrogen minus dissolved inorganic nitrogen. This indicates that total nitrogen load was dominated by a dissolved phase in the virgin brigalow woodland but a particulate phase in cropping. However, this does not take into account the contribution of dissolved organic nitrogen which was not measured in this study. The literature shows that dissolved organic nitrogen load in runoff can equal dissolved inorganic nitrogen loads (Heathwaite and Johnes, 1996; Martinelli et al., 2010; Rogusz et al., 2013), providing further evidence that the total nitrogen load from virgin brigalow woodland was dominated by a dissolved phase. It also suggests that total nitrogen load in cropping was likely to be equally comprised of both dissolved and particulate nitrogen if not dominated by a dissolved phase.

Although mean annual loads presented in this study are based on calculations using the 10 years of available EMC data, it is expected that total nitrogen concentrations in runoff would decline from the cropped catchment over the 25 years as nitrogen was removed from the system. Measurements of total soil nitrogen from 1981 to 2008 (the last sampling period before conversion of the cropped catchment into a ley pasture in 2010) shows that nutrient rundown occurred in the absence of fertiliser inputs and the export of nitrogen in grain and runoff. That is, concentrations of total soil nitrogen in the virgin brigalow catchment remained relatively stable (mean 0.176%) whereas concentrations in the cropped catchment declined from 0.197% in 1981 to 0.076% in 2008 (unpublished data). This result is expected given the previously reported declines in grain yield and grain nitrogen from the cropped catchment over 23 years (Radford et al., 2007).

Both agricultural systems had more ammonium nitrogen in runoff than the virgin brigalow woodland; 2% contribution to the total cumulative load of total nitrogen compared to less than 1%, respectively. However, the overall small contribution of ammonium to total nitrogen is most likely due to soil bacteria which rapidly convert ammonium into nitrate given ideal moisture and temperature conditions (Price, 2006). Cumulative losses of ammonium in runoff from this study were more similar to sediment, and hence phosphorus, than other nitrogen parameters. This trend has been reported in other studies and is attributed to the adsorption of ammonium onto sediment particles (Heathwaite and Johnes, 1996; Johnes and Burt, 1991). That is, ammonium (NH_4^+) is a positively charged cation which is attracted to the negatively charged surface of organic matter and clay particles, whereas nitrate (NO_3^-) is a negatively charged anion repelled by the soil and subsequently more readily lost via leaching and runoff.

The grazed catchment exported considerably less nitrogen than the virgin brigalow and cropped catchments. Loads of oxidised and dissolved inorganic nitrogen from pasture were almost negligible, and the higher total nitrogen loads can be attributed to particulate nitrogen. That is, dissolved inorganic nitrogen contributed only 5% to the total cumulative load of total nitrogen which indicates that particulate nitrogen strongly dominates transport from the grazed catchment. A dissolved organic nitrogen load equal to the dissolved inorganic nitrogen load would still result in particulate nitrogen strongly dominating transport from the grazed catchment. However, Radford et al. (2007) reported only 1.6 kg $ha^{-1} yr^{-1}$ of nitrogen removal in cattle from the grazed catchment over 23 years compared to 36.1 kg $ha^{-1} yr^{-1}$ nitrogen removal in grain from the cropped catchment. These observations imply that sown pasture in the grazed catchment is a sink for nitrogen. This is known as pasture rundown which occurs when mineral nitrogen becomes immobilised in soil organic matter and established grass plants, rather than a net loss from the system (Lawrence et al., 2014; Robertson et al., 1997). This is reflected in the soil mineral nitrogen data for the grazed catchment which remained relatively stable from 1983 to 2008 following an initial peak in 1982 due to clearing and burning of the catchment (unpublished data). Although rundown can lead to a decline in pasture productivity (Lawrence et al., 2014; Myers and Robbins, 1991), the grazed catchment in this study has always been conservatively managed to maintain high pasture cover.

Lower nitrogen from the grazed catchment compared to the cropped catchment can also be attributed to the higher percentage of nitrogen removed by pasture (3.5%) compared to cereal grains (1.8%) (The State of Victoria, 2015). As pasture tends to uptake more nitrate, a component of both oxidised and dissolved inorganic nitrogen, there is less available in the effective depth of interaction (0.1–4 cm) (Sharpley, 1985) to be dissolved and transported with water in runoff or as leachate. Loads of nitrogen in runoff have also been shown to be lower from pasture cut to 155 mm above the ground than 47 mm (Mundy et al., 2003). This supports the use of management practices that promote higher pasture biomass to reduce runoff volume and hence improve water quality, such as wet season spelling and conservative stocking rates based on feed availability.

4.3. Effect of management practices

Management practices, such as fertiliser application and tillage method, are two factors that affect the quality of surface runoff from crop systems. Sharpley and Smith (1994) found that fertiliser applications on conventionally tilled wheat resulted in a 17-fold nitrogen and 30-fold phosphorus increase in runoff. Physical and chemical degradation of soils from cropped land is a slow process (Silburn et al., 2007), and based on more than 25 years of fertility rundown at this site, soil fertility and consequently nutrient loads in runoff are expected to be lower than fertilised crops. For example, Murphy et al. (2013) reported 7–8 kg ha^{-1} oxidised nitrogen (approximately 20% of the total nitrogen applied at planting) in runoff from a fertilised crop, whereas this study in a similar area of central Queensland, Australia, reported 1.43 kg ha^{-1} oxidised nitrogen from an unfertilised crop.

Conventional tillage practices are reported to have higher runoff volume and/or erosion loss than no-till crop systems (Carroll et al., 1997; DeLaune and Sij, 2012; Ehigiator and Anyata, 2011). No-till practices have higher stubble cover which reduces overland flow velocity and the ability of water to detach and transport sediment (Rose and Freebairn, 1985). Cover levels above 30% have been suggested as critical for erosion control in crop systems (Carroll et al., 1997). Thus, management practices that retain cover and reduce runoff are also useful for reducing loads of sediment and some nutrients (Bartley et al., 2014a; Hansen et al., 2002; McIvor et al., 1995). For example, Sharpley and Smith (1994) found that changing a crop system from conventional to no-till reduced soil loss 18-fold, nitrogen loss four-fold and phosphorus loss three-fold, but an increase in bioavailable phosphorus was observed. Similarly, DeLaune and Sij (2012) reported a five-fold reduction in soil loss from no-till compared to conventional tilled systems. These authors also observed a trend of lower total phosphorus, dissolved phosphorus and ammonia nitrogen but higher nitrate nitrogen from no-till systems, although differences were not statistically different ($P < 0.05$) (DeLaune and Sij, 2012). Minimum tillage was introduced to the cropped catchment at the Brigalow Catchment Study in 1992 with intermittent use of conventional tillage practices in 1994, 1997 and 2007. Small increases in cumulative runoff and loads of sediment and nutrients can be seen around the periods when conventional tillage had been reintroduced; however, hydrology appears to be a stronger influence with the two main periods of an increased rate in cumulative loads (1991 to 1994 and 1996 to 1999) coinciding with periods of above average rainfall over multiple years. This supports the recommendation that management practices that reduce runoff also reduce sediment and nutrient loads.

Runoff water quality from pasture systems is similarly affected by cover. Silburn et al. (2011) suggested that more than 50% ground cover should be maintained in grazed areas to reduce excessive runoff and soil loss. This recommendation was based on a seven year study in a semi-arid area of Queensland which exported 30 to 50% of rainfall as runoff when cover was less than 20%, but averaged only 5.9% when cover was greater than 50%. The trend of reduced runoff, and hence reduced sediment and nutrients exported in runoff, from grazed land with higher ground cover is supported by numerous authors (Murphy et al., 2008; Nelson et al., 1996; Schwarte et al., 2011). Bare areas (scalds) have a low tolerance to soil erosion due to low total water-holding capacity which results in lower infiltration and hence increased runoff compared to areas with greater cover (Silburn et al., 2011). However, management practices such as reduced stocking rates and rotational wet season resting have been shown to increase ground cover (Bartley et al., 2010, 2014b). The pasture system at the Brigalow Catchment Study is conservatively grazed and aims to maintain at least 80% cover and less than 30% pasture utilisation, which is considered a well-managed system. Hence, the loads of sediment, nitrogen and phosphorus exported from this site may be lower than other areas which have higher stocking rates and greater pasture utilisation.

5. Conclusions

The simple hydrology and water quality model presented was based on a 17 year calibration period of the Brigalow Catchment Study in its native condition, and 25 years flow and 10 years water quality monitoring following land use change to agriculture. The model indicated that changing land use from virgin brigalow woodland to a well-managed (unfertilised) pasture system decreased nitrogen in runoff compared to runoff from virgin brigalow woodland which dominated the landscape during pre-European times; however, both crop and pasture systems increased loads of sediment and phosphorus. Overall, crops posed a greater risk to downstream water quality, and subsequently the end of catchment marine system, than pasture.

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Appendix 1.3: Thornton and Shrestha (Unpublished)

The Brigalow Catchment Study: VI[*]. Clearing and burning brigalow (*Acacia harpophylla*) in Queensland, Australia, temporarily increases surface soil fertility prior to nutrient decline under cropping or grazing

Running head

Clearing brigalow decreases soil fertility

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Abstract

In the Brigalow Belt bioregion of Australia, clearing of brigalow (*Acacia harpophylla*) scrub vegetation for agriculture has altered nutrient cycling over millions of hectares. In order to quantify the effect of this vegetation clearing and land use change on soil fertility, the Brigalow Catchment Study commenced in 1965. 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 to 0.1 m) as a result of soil heating and the ash bed effect. Fertility declined significantly over the subsequent 32 years. 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% and total potassium by 9% from post-burn, pre-cropping levels. Fertility also declined under grazing but in a different pattern to that observed under cropping. Organic carbon showed clear fluctuation but it was not until the natural variation in soil

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fertility over time was separated from the anthropogenic effects of land use change that a significant decline was observed. 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 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. The primary mechanism of nutrient loss depended on the specific land use and nutrient in question.

Additional keywords: land use change; land development; Fitzroy Basin; cultivation; cattle; grain; beef.

Introduction

Soil fertility decline, soil structural decline and erosion are all considered to be consequences of changing land use from virgin forest to cropping and grazing. Nutrient cycling in undisturbed virgin ecological systems is essentially a steady state closed system, where soil nutrients are consumed by the growing plants and then released back to the soil via leaf litter, wood debris and roots (Moody 1998). In contrast, cropping and grazing systems disturb this cycle by removing nutrients in harvested products and animals (Radford *et al.* 2007); via increased surface runoff (Elledge and Thornton 2017; Thornton *et al.* 2007); via increased leaching (Silburn *et al.* 2009); and via increased gaseous losses from soil and animals (Dalal *et al.* 2013; Huth *et al.* 2010). Disturbance of nutrient cycles and increased losses of soil nutrients affect the viability and sustainability of farming systems. Increased nutrient loads lost to the environment impacts ecosystem health, resulting in substantial investment in harm minimisation and remediation programs worldwide (Carroll *et al.* 2012).

In the Brigalow Belt bioregion of Australia, clearing of brigalow (*Acacia harpophylla*) scrub and land use change has substantially altered nutrient cycling over a large area. The bioregion occupies 36.7

million hectares of Queensland and New South Wales, stretching from Dubbo in the south to Townsville in the north of Australia. Since European settlement, 58% of this bioregion has been cleared. The bioregion contains Queensland's largest catchment, the Fitzroy Basin, which drains directly into the Great Barrier Reef lagoon. In 1962, the Brigalow Land Development Fitzroy Basin Scheme commenced, resulting in the Government-sponsored clearing of 4.5 million hectares for cropping and grazing. This clearing represents 21% of all clearing in the bioregion and 32% of the Fitzroy Basin area (Thornton *et al.* 2007). Broad scale land clearing continued in the basin until 2006 (McGrath 2007). In the preceding decade, rates of land clearing in Queensland were among the highest in the world with estimates of between 425,000 ha and 446,000 ha cleared per year (Lindenmayer and Burgman 2005; Reside *et al.* 2017; Wilson *et al.* 2002). More than 60% of this clearing, or about 261,000 ha/yr was undertaken in the Brigalow Belt (Cogger *et al.* 2003; Wilson *et al.* 2002). It is estimated that 85% to 90% of brigalow scrub has been cleared since European settlement (Cogger *et al.* 2003; Tulloch *et al.* 2016).

In order to quantify the effect of this scale of vegetation clearing and land use change on soil fertility, the Brigalow Catchment Study (BCS) commenced in 1965. The objective of this study was to evaluate whether clearing of brigalow scrub for cropping or grazing would alter the dynamics of soil organic carbon, nitrogen, phosphorus, sulfur and potassium over time. It was hypothesised that land development for cropping would lead to a significant decline in soil fertility while less or no change was expected with land development for grazing. It was also expected that the trends noted by Radford *et al.* (2007), i.e. unchanged concentrations of soil organic carbon and total nitrogen under brigalow scrub and grazing land uses but significant decline under cropping, would continue; however, the planting of legume ley pasture may enhance nutrient status in soil under the cropping land use.

As resourcing pressures limit the commencement and continuation of long-term studies there is an increasing trend towards modelling. This study facilitates modelling by numerically describing the starting condition of the landscape and mathematically defining fertility trends over time. Discussion on the mechanisms of change further informs process based models, assisting in moving forward from traditional empirical black box models. The BCS continues today having adapted to answer new research questions, and having answered questions unanticipated at its inception more than five decades ago.

Materials and Methods

The BCS is described in detail by Cowie *et al.* (2007); changes in runoff volume and peak runoff rate are given in Thornton *et al.* (2007), Thornton and Yu (2016), and Thornton and Yu (2017); agronomic and soil fertility results are given in Radford *et al.* (2007); the deep drainage component of the water balance is given in Silburn *et al.* (2009); and changes in water quality are given in Thornton and Elledge (2016) and Elledge and Thornton (2017).

Site location and climate

The study site is located at 24.81°S, 149.80°E at an altitude of 151 m above sea level, located within the Dawson sub-catchment of the Fitzroy basin, central Queensland, Australia. The region has a semi-arid, subtropical climate. Summers are wet, with 70% of the annual average (1964 to 2014) hydrological year (October to September) rainfall of 661 mm falling between October and March, while winter rainfall is low (Fig. 1). Average monthly temperature ranges from a minimum of 6.3°C in July to a maximum of 33.8°C in January (Fig. 1).

Fig. 1.

Experimental design

The BCS is a paired, calibrated catchment study consisting of three small catchments, C1, C2 and C3, ranging from 11.7 to 16.8 ha in size. Within each catchment, three permanent monitoring sites were established to monitor soil fertility. Establishment of the 20 m by 20 m sites was done using double stratification. Initial stratification was based on soil type and slope position with a monitoring site allocated to both an upper and lower-slope position on Vertosols, and the third on a Sodosol. Secondary stratification was by way of 10 sub-units, each 4 m by 10 m, within each monitoring site.

Soil types and vegetation

Soil types were typically characterised by fine-textured dark cracking clays (Black and Grey Vertosols), non-cracking clays (Black and Grey Dermosols) and thin layered dark and brown sodic soils (Black and Brown Sodosols) (Isbell 1996, R. J. Tucker, pers. comm.). Approximately 70% of C1 and C2 and 58% of C3 were comprised of Vertosols and Dermosols (clay soils); the remaining area in each catchment was occupied by Sodosols. The plant-available water holding capacity of these soils ranged from 130 to 200 mm in the surface 1.4 m of the soil profile. Average slope of the catchments is 2.5%. The catchments consisted of good quality agricultural land, all equally suitable for cropping or grazing.

Vegetation was typical of the Brigalow Belt bioregion, dominated by brigalow (*Acacia harpophylla)*, as described in detail by Cowie *et al.* (2007). In their native "brigalow" state, the catchments were composed of three major vegetation communities, identified by their most common canopy species; brigalow (*Acacia harpophylla*), brigalow–belah (*Casuarina cristata*) and brigalow–Dawson Gum (*Eucalyptus cambageana*). Understoreys of all major communities were characterised by *Geijera* sp. either exclusively, or in association with *Eremophila* sp. or *Myoporum* sp.

Site history and management

The study has had four experimental stages (Table 1). Stage I, the calibration phase, monitored rainfall and runoff from the catchments, allowing an empirical hydrological calibration between catchments to be developed. The permanent monitoring sites were established in each catchment during this stage. Baseline measurements of soil fertility were taken in 1981.

Table 1.

Stage II, the land development phase, commenced in March 1982 when vegetation in C2 and C3 were developed by clearing with traditional bulldozer and chain methods. Catchment 1 was retained as an uncleared, undisturbed control. In C2 and C3, the fallen timber was burnt *in situ* in October 1982. Following burning, residual unburnt timber in C2 was raked to the contour for secondary burning. Narrow-based contour banks were then constructed at 1.5 m vertical spacing. A grassed waterway was established to carry runoff water from the contour channels to the catchment outlet. In C3, residual unburnt timber was left in place, and in November 1982 the catchment was sown to buffel grass (*Cenchrus ciliaris* cv. Biloela). The second soil fertility assessment was undertaken in December 1982, soon after burning.

Stage III, the land use comparison phase, commenced in 1984. In C2, the first crop sown was sorghum (*Sorghum bicolor*) (September 1984), followed by annual wheat (*Triticum aestivum*) for nine years. Fallows were initially managed using mechanical tillage (disc and chisel ploughs), which resulted in 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 nutrient inputs were used. In C3, the buffel grass pasture established well with >5 plants/m^2 and 96% groundcover achieved before cattle grazing commenced in December 1983. Stocking rate was 0.3 to 0.7 head/ha (each stock typically 0.8 adult equivalent), adjusted to maintain pasture dry matter levels >1000 kg/ha without nutrient inputs, feed or nutrient supplementation.

Stage IV, the adaptive land management phase, commenced in 2010. To sustain productive agricultural systems representative of commercial enterprises in the Brigalow Belt bioregion, management strategies to maintain or enhance soil fertility were implemented. In C2, the legume butterfly pea (*Clitoria ternatea*) was planted as a ley pasture in January 2010. The butterfly pea was left ungrazed to establish and set seed until March 2011 when grazing commenced. In September 2011, cattle were removed from both C2 and C3 to allow spelling of the pastures over the 2011/12 and 2012/13 wet seasons. Grazing recommenced in December 2013 when the catchments were "crash grazed" with high stocking rates of 0.5 adult equivalents/ha in C2 and 1.4 adult equivalents/ha in C3 for 45 days to reduce rank pasture growth. Subsequently, grazing continued at conservative stocking rates of about 0.3 adult equivalents/ha with regular periods of pasture spelling.

Soil sampling

Soil water in the surface 0.1 m of the soil profile was determined gravimetrically according to the methods of Cowie *et al.* (2007).

Soil samples for fertility analysis were collected from the surface 0.1 m of the soil profile at each monitoring site using manual coring tubes of 0.05 m diameter. Samples were typically a composite of eight 0.05 m-diameter cores. The eight cores were comprised of two cores sampled adjacent to each of four fixed locations within each sub-unit. More intensive sampling was undertaken pre-clearing in 1981, and in 2008 and 2014. In these years samples were a composite of 20 cores, with five cores sampled adjacent to each of the four fixed locations. 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 retained after analysis in a long-term storage archive.

Measurements of agricultural productivity and nutrient removal

In the cropped catchment, grain yield, nitrogen and phosphorus content were measured according to the methods of Radford *et al.* (2007). Grain sulfur content was estimated as grain nitrogen multiplied by 10% (Byers *et al.* 1987; Győri 2005). Grain potassium content was estimated as 0.46% of grain yield (Mengel and Kirby 1982).

In the grazed catchment, cattle live weight gain was measured according to the method of Radford *et al.* (2007). Nutrient export in of beef was estimated as live weight gain multiplied by 2.4% for nitrogen (Radford *et al.* 2007), 0.71% for phosphorus (Gibson *et al.* 2002), 0.16% for sulfur (Ad Hoc Committee on Air Emissions from Animal Feeding Operations 2003) and 0.2% for potassium (Whitehead 2000). Nitrogen volatilisation losses from cattle urine and faeces was estimated as nitrogen intake multiplied by 19.77% (Laubach *et al.* 2013). Nitrogen intake was estimated as dietary biomass intake multiplied by dietary nitrogen content. Daily dietary biomass intake was estimated as fasted animal live weight at entry to the catchment multiplied by 2% per day of grazing (Minson and McDonald 1987). Dietary nitrogen content was determined using the FNIRS technique of Dixon and Coates (2010).

Soil physical and chemical analyses

Soil bulk density was measured pre-clearing in 1981, then post-clearing in 1984, 1987, 1994, 1997, 2000 and 2014. Sample cores not contaminated by rocks or organic matter >2mm were dried at 40°C then weighed. The tip diameter of the coring tubes was measured in field with the external wall of the tube marked at 0.1 m to indicate the depth of sampling. Bulk density was calculated as the mass of 105°C oven-dry soil per volume of core sampled.

Chemical analyses were performed by the Queensland Government soil laboratory network, formerly at Biloela and Indooroopilly; now centralised at the Chemistry Centre, EcoSciences Precinct, Dutton Park, in the Department of Science, Information Technology and Innovation. Prior to analyses, soil samples were dried at 40°C and ground to pass through a 2 mm sieve. Samples were then analysed for soil organic carbon, total nitrogen, mineral nitrogen (ammonium-nitrogen (NH_4-N) and nitrate-nitrogen (NO_3-N)), total phosphorus, available phosphorus (bicarbonate-extractable phosphorus and acid-extractable phosphorus), total sulfur and total potassium. Organic carbon (OC) was determined by the dichromate oxidation method of Walkley and Black (1934) followed by titration, or after 1997, using a colorimetric procedure with sucrose standards (Sims and Haby 1971) as described in method 6A1 in Rayment and Higginson (1992); these methods are well correlated (R^2 = 0.96) (Cowie *et al.* 2002). Total nitrogen (TN) was determined by macro-Kjeldahl digestion (Bremner 1965). Mineral nitrogen was determined by the potassium chloride extraction method described in method 7C2 in Rayment and Higginson (1992). Total phosphorus (TP) was determined using the X-ray fluorescence (XRF) method described in method 9A1 in Rayment and Higginson (1992). Bicarbonate-extractable phosphorus (P(B)) was determined using a modification of the Colwell (1963) method described in method 9B2 in Rayment and Higginson (1992) while acid-extractable phosphorus (P(A)) was determined using a modification of the Kerr and von Stieglitz (1938) method described in method 9G2 in Rayment and Higginson (1992). Total sulfur (TS) and total potassium (TK) were determined using the X-ray fluorescence (XRF) method described in methods 10 A1 and 17A1 respectively, in Rayment and Higginson (1992).

The number of samples analysed varied between soil samplings (Table 2). At a minimum, a composite sample comprised of a subsample of each of the 10 sub-units in a monitoring site was generated for analysis. This composite sample was representative of at least 80 soil cores from within a monitoring site. Alternatively, a sample from each of the sub-units in a monitoring site was generated for analysis. This resulted in 10 samples, with each being representative of at least eight soil cores.

Table 2.

Approaches for assessing fertility decline

Comparison of observed soil fertility data

The observed soil fertility of a catchment was calculated as the average of the analytical results for all composite samples from the three monitoring sites within the catchment at the time of sampling. Changes in soil fertility over time since burning were assessed using both linear and exponential regression analysis tools in the statistical software package Genstat (VSN International 2016).

Calibrating to account for natural fertility change

The paired catchment design of the experiment allowed for the natural variation in soil fertility over time to be separated from the anthropogenic effects of land use change. This was done by dividing the observed soil fertility of C2 and C3 by the observed soil fertility of the control catchment C1. Analysis of these ratios accounts for likely change in the soil fertility of C2 and C3 had they remained uncleared and provides a more accurate estimation of change rather than simply comparing the observed fertility over time to pre-clearing levels. As for the observed data, changes in soil fertility over time since burning were assessed using regression analysis.

Results

Grain and beef production and associated nutrient removal

Grain production in C2 yielded 49,460 kg/ha of grain over 30 years (Fig. 2). This removed 958 kg/ha of nitrogen, 130 kg/ha of phosphorus, 96 kg/ha of sulfur and 228 kg/ha of potassium from the catchment. Removal of grain (P <0.001, R^2 = 99%) (Equation 1), nitrogen (P <0.001, R^2 = 99%)

(Equation 2) and phosphorus (P <0.001, R^2 = 99%) (Equation 3) over time since the first crop was planted all showed exponential trends.

$$C2\ grain\ removal\ (kg/ha) = 223{,}373 - 220{,}280 \times (0.999^x) \quad (1)$$

$$C2\ nitrogen\ removal\ (kg/ha) = 1{,}521 - 1{,}460 \times (0.999^x) \quad (2)$$

$$C2\ phosphorus\ removal\ (kg/ha) = 1{,}044 - 1{,}035 \times (0.999^x) \quad (3)$$

Where ^{x} is years since the first crop was planted.

Beef production in C3 yielded 1,910 kg/ha of beef over 31 years (Fig. 2). This removed 46 kg/ha of nitrogen, 14 kg/ha of phosphorus, 3 kg/ha of sulfur and 4 kg/ha of potassium from the catchment. A further 71 kg/ha of nitrogen was removed via volatilisation from urine and faeces. Removal of beef over time since grazing commenced showed an exponential trend (P <0.001, R^2 = 99%) (Equation 4) (Fig. 2). As the nitrogen and phosphorus content of beef were estimated based on a percentage of live weight gain, the response curve for their removal from the catchment over time mirrored that of total beef removal.

Fig. 2.

$$C3\ beef\ removal\ (kg/ha) = 2{,}765 - 2{,}786 \times (0.999^x) \quad (4)$$

Where ^{x} is years since grazing commenced.

Trends in bulk density

Pre-clearing oven-dry bulk density for the three catchments in 1981 averaged 1.15 g/cm^3 (range 1.1 g/cm^3 to 1.22 g/cm^3). Over the following 32 years there was no significant linear or exponential change in bulk density in C1 (P = 0.498 and P = 0.773 respectively). Clearing and burning followed by 30 years of cropping resulted in a significant linear increase in bulk density (P = 0.062, R^2 = 44%). Fitting an exponential curve maintained the significance of the regression but improved the coefficient of determination (P = 0.06, R^2 = 63%). Ratios of C2/C1 bulk density showed no significant linear or exponential change (P = 0.136 and P = 0.292 respectively). Clearing and burning followed by 31 years of grazing resulted in a linear increase in bulk density (P = 0.097, R^2 = 35%). No significant exponential change was detected (P = 0.14). Ratios of C3/C1 bulk density mirrored both the linear and exponential results of the observed data (P = 0.053, R^2 = 47% and P = 0.132 respectively).

Observed bulk density in C2 and C3 post-clearing and burning was consistently higher than it was pre-clearing. Average bulk density post-clearing and burning was 116% of pre-clearing bulk density in C2 and 118% in C3%. In the same period, bulk density in C1 declined to 98% of 1981 levels. Ratios of C2/C1 and C3/C1 bulk density were also higher post-clearing and burning, increasing to 119% and 120% of their respective pre-clearing ratios. As the average increase in bulk density in C2 and C3 equated to an additional 192 tonnes of soil in the surface 0.1 m of the soil profile, soil nutrient loss in kg/ha post-clearing and burning was calculated using the average bulk density of a catchment in that period, being 1.30 g/cm^3 in C2 and 1.34 g/cm^3 in C3.

In 1984 and 1987, soil water content was measured within two weeks of soil sampling occurring. In 1984, available soil water and bulk density at time of sampling was 5 mm and 1.18 g/cm^3 respectively in C1; 14 mm and 1.26 g/cm^3 in C2; and 17 mm and 1.30 g/cm^3 in C3. In 1987, available soil water and bulk density at time of sampling was 4 mm and 1.21 g/cm^3 respectively in C1; 23 mm and 1.21 g/cm^3 in C2; and 12 mm and 1.33 g/cm^3 in C3.

Trends in observed soil fertility data

Organic carbon

Pre-clearing, OC levels in the three catchments averaged 2.08% (range 1.93% to 2.25%). From 1981 to 2014, OC in C1 averaged 2.15% with no significant linear or exponential trend (P = 0.061 and P = 0.066 respectively) (Fig. 3). Unlike C1, OC in C2 showed a significant exponential decline of 46% from

2.25% in 1981 to 1.21% in 2014 (P <0.001, R^2 = 88%) (Equation 1 in Table 3) (Fig. 3). In C3, OC showed no significant linear or exponential trends from 1981 to 2014 (P = 0.293 and P = 0.343 respectively) (Fig. 3). However, this analysis masks a significant exponential decline of 28% from 1.93% in 1981 to 1.39% in 2000 (P <0.001, R^2 = 79%) (Equation 2 in Table 3) (Fig. 3) followed by an increase from 2000 to 2014.

Table 2 and Fig. 3.

Total nitrogen

Pre-clearing, TN levels in the three catchments averaged 0.18% (range 0.163% to 0.197%). From 1981 to 2014, TN in C1 averaged 1.75% with no significant linear or exponential trend (P = 0.191 and P = 0.161 respectively) (Fig. 4). Unlike C1, TN in C2 showed a significant exponential decline of 55%, or 1,050 kg/ha, from 0.197% in 1981 to 0.088% in 2014 (P <0.001, R^2 = 91%) (Equation 3 in Table 3) (Fig. 4). Similar to C2, C3 showed a significant exponential decline of 22%, or 143 kg/ha, from 0.163% in 1981 to 0.128% in 2014 (P = 0.01, R^2 = 49%) (Equation 4 in Table 3) (Fig. 4).

These declines were exceeded when considering only the period from 1981 to 2008, prior to the commencement of the adaptive land management phase to enhance soil fertility. In this period, TN in C2 showed a significant exponential decline of 61%, or 1,201 kg/ha while TN in C3 showed a significant exponential decline of 24%, or 192 kg/ha. From 2010 to 2014, during the adaptive land management phase, TN in C1 and C3 had similar increases of 2.4% and 2.9% respectively; however, TN in C2 increased by 15.3%, or 151 kg/ha.

Fig. 4.

Mineral nitrogen

Pre-clearing, ammonium-nitrogen levels in the three catchments averaged 5.19 mg/kg (range 4.87 mg/kg to 5.5 mg/kg) and nitrate-nitrogen averaged 2.46 mg/kg (range 1.74 mg/kg to 3.4 mg/kg). Average mineral nitrogen, being the sum of ammonium- and nitrate-nitrogen, was 7.65 mg/kg (range 6.61 mg/kg to 8.58 mg/kg) (Fig. 5 to Fig. 7). In the first sampling post-burning, ammonium-nitrogen in C2 and C3 spiked to an average of 8.9 times their pre-clearing levels when adjusted for the natural increase in ammonium-nitrogen observed in C1 (Fig. 5). This spike was short lived and by the following sampling, less than one year post-burning, ammonium-nitrogen levels in C2 and C3 declined back to that of C1. Ammonium-nitrogen levels fluctuated at all subsequent samplings with C1 typically having highest levels and C2 and C3 having similar, lower levels.

Fig. 5.

Nitrate-nitrogen in C2 and C3 had a similar spike post-clearing, increasing to an average of 7.5 times their pre-clearing levels when adjusted for the natural decline in nitrate-nitrogen observed in C1 (Fig. 6). The spike was observed after the ammonium-nitrogen spike had declined back to pre-clearing levels. Elevated nitrate-nitrogen levels were observed in C2 for at least eight years post-burning after which levels and fluctuations were similar to those observed in C1. Elevated nitrate-nitrogen levels in C3 declined within two years of burning and typically remained less than those observed in C1 with substantially less fluctuation.

Fig. 6.

Total mineral nitrogen showed a post-burning spike in C2 and C3 of 5.1 times their pre-clearing mineral nitrogen when adjusted for the natural increase in mineral nitrogen observed in C1 (Fig. 7). These increases declined substantially within one year post-burning and fluctuated similarly to mineral nitrogen levels in C1 up to five years post-burning. From this point mineral nitrogen in C1 and C2 had similar levels and fluctuations however levels in C3 were typically lower with less fluctuation.

Fig. 7.

Total phosphorus

Pre-clearing, TP levels in the three catchments averaged 0.031% (range 0.029% to 0.035%). In C1, TP showed a significant linear and exponential (Equation 5 in Table 3) increase of 14% from 0.029% in 1981 to 0.033% in 2014 (P <0.001, R^2 = 76% and P <0.001, R^2 = 77% respectively) (Fig. 8). This increase was not constant over time with no significant linear or exponential trend occurring prior to 2003 (P = 0.082 and P = 0.15 respectively).

Clearing and burning C2 and C3 increased TP by an average of 4%. Post-burning, TP in C2 showed a significant exponential decline of 29%, or 131 kg/ha, from 0.036% in 1982 to 0.027% in 2014 (P <0.001, R^2 = 91%) (Equation 6 in Table 3) (Fig. 8). Similarly, TP in C3 showed a significant exponential decline of 14%, or 59 kg/ha, from 0.032% in 1982 to 0.027% in 2014 (P = 0.009, R^2 = 53%) (Equation 7 in Table 3) (Fig. 8). Visually, the decline in C3 was most prevalent from 1982 to 1997 followed by an increase from 2000 to 2014. This is supported by linear regression showing increasing P-values and decreasing R^2 with each successive sampling from 1997 onwards. Fitting an exponential curve showed similar results with R^2 declining from 81% at 2003 to 52% at 2008.

Fig. 8.

Bicarbonate-extractable phosphorus

Pre-clearing, P(B) levels in the three catchments averaged 13.67 mg/kg (range 13.3 mg/kg to 14 mg/kg). From 1981 to 2014, P(B) in C1 averaged 14.31 mg/kg and showed no significant linear or exponential trend (P = 0.063 and P = 0.18 respectively) (Fig. 9). Clearing and burning C2 and C3 increased P(B) by an average of 2.5 times pre-clearing levels. After this initial increase a significant exponential decline occurred between 1982 and 2014 in both C2 (P <0.001, R^2 = 88%) (Equation 8 in Table 3) and C3 (P <0.001, R^2 = 92%) (Equation 9 in Table 3) (Fig. 9). Thirty two years after the increase in P(B) levels as a result of burning, P(B) levels in C2 had declined to 15.9 mg/kg, equal to 114% of its pre-clearing level; P(B) levels in C3 had declined to 12.63 mg/kg, equal to 95% of its pre-clearing level. On a kg/ha basis, this was a decline of 18 kg/ha in C2 and 23 kg/ha in C3.

Fig. 9.

Acid-extractable phosphorus

The behaviour of P(A) in all three catchments mirrored that of P(B). Pre-clearing, P(A) levels in the three catchments averaged 26 mg/kg (range 25 mg/kg to 26.3 mg/kg). From 1981 to 2014, C1 P(A) averaged 23.48 mg/kg and showed no significant linear or exponential trend (P = 0.063 and P = 0.18 respectively) (Fig. 10). Clearing and burning C2 and C3 increased P(A) by an average of 2.2 times pre-clearing levels. After this initial increase a significant exponential decline occurred between 1982 and 2014 in both in C2 (P <0.001, R^2 = 91%) (Equation 10 in Table 3) and C3 (P <0.001, R^2 = 97%) (Equation 11 in Table 3) (Fig. 10). At 32 years post-burning, P(A) levels in C2 had declined to 24.63 mg/kg, equal to 94% of its pre-clearing level; P(A) levels in C3 had declined to 19.57 mg/kg, equal to 73% of its pre-clearing level. On a kg/ha basis, this was a decline of 36 kg/ha in C2 and 39 kg/ha in C3.

Fig. 10.

Total sulfur

Pre-clearing, TS levels in the three catchments averaged 0.021% (range 0.02% to 0.023%). In C1, TS showed a significant linear and exponential (Equation 12 in Table 3) increase of 9% from 0.021% in 1981 to 0.022% in 2014 (P = 0.002, R^2 = 55% and P = 0.008, R^2 = 51% respectively) (Fig. 11). As for TP, this increase was not constant over time with no significant linear trend occurring prior to 2000 (P = 0.058) or exponential trend prior to 2003 (P = 0.145).

Clearing and burning C2 and C3 increased TS by an average of 6%. Post-burning, TS in C2 showed a significant exponential decline of 49%, or 153 kg/ha, from 0.024% in 1982 to 0.012% in 2014 (P <0.001, R^2 = 90%) (Equation 13 in Table 3) (Fig. 11). Data from C3 did not meet the assumptions for valid statistical testing so no statement of significance can be made about trends over the entire 32 year post-burning period. However, the calculated loss of TS was 23%, or 67 kg/ha, from 0.022% in

1982 to 0.017% in 2014. Visually, the increase in TS associated with clearing and burning declined rapidly from 1982 to 1984 followed by a gradual increase with a substantial spike in 2008 (Fig. 11). The initial decline from 1982 to 1987 was exponential (P = 0.009, R^2 = 93%). An exponential curve could be fitted to the data up to 2003 (P = 0.001, R^2 = 80%); however, inclusion of the 2008 data resulted in a non-significant regression (P = 0.286). No significant linear trend occurred from 1984 to 2000 (P = 0.211); however, incremental inclusion of data from 2003 to 2014 showed significant increases in TS (P = 0.005 to 0.037, R^2 = 35% to 60%).

Fig. 11.

Total potassium

Pre-clearing, TK levels in the three catchments averaged 0.483% (range 0.248% to 0.716%). In C1, TK averaged 0.716% and showed no significant linear or exponential trend from 1981 to 2014 (P = 0.084 and P = 0.119 respectively) (Fig. 12).

Clearing and burning C2 and C3 increased TK by an average of 5%. Post-burning, TK in C2 showed a significant exponential decline of 9%, or 579 kg/ha, from 0.506% in 1982 to 0.461% in 2014 (P = 0.004, R^2 = 61%) (Equation 14 in Table 3). Post-burning, TK in C3 showed a significant exponential decline of 10%, or 364 kg/ha, from 0.264% in 1982 to 0.237% in 2014 (P <0.001, R^2 = 94%) (Equation 15 in Table 3) (Fig. 12). At 32 years post-burning, TK levels in C2 had declined to 0.461%, equal to 95% of its pre-clearing level; TK levels in C3 had declined to 0.237%, equal to 96% of its pre-clearing level.

Fig. 12.

Trends after accounting for natural fertility change

Organic carbon

Similar to the observed C2 OC data, the C2/C1 OC ratio also showed a significant exponential decline from 1981 to 2014 (P <0.001, R^2 = 91%) (Equation 1 in Table 4). However the 54% decline in the ratio was greater than the 46% decline in the observed C2 OC data. In contrast to the observed C3 OC data, the C3/C1 OC ratio showed a significant exponential decline of 21% (P = 0.05, R^2 = 32%) from 1981 to 2014 (Equation 2 in Table 4). The exponential decline of 24% (P = 0.002, R^2 = 74%) in the C3/C1 OC ratio between 1981 and 2000 was similar to the observed data.

Total nitrogen

The C2/C1 TN ratio behaved similarly to the observed C2 TN data. The ratio showed a significant exponential decline of 53% from 1981 to 2014 (P <0.001, R^2 = 92.8%) (Equation 3 in Table 4). Prior to the commencement of the adaptive land management phase the ratio showed a significant exponential decline of 58 % from 1981 to 2014 (P <0.001, R^2 = 92%). From 2010 to 2014, during the adaptive land management phase, the ratio increased by 13%. The C3/C1 TN data also behaved similarly to the observed C3 TN data. The ratio showed a significant exponential decline of 18% from 1981 to 2014 (P = 0.004, R^2 = 57%) (Equation 4 in Table 4). From 2010 to 2014, during the adaptive land management phase, the ratio increased by 1%.

Total phosphorus

Compared to the observed TP data, both C2/C1 and C3/C1 TP ratios showed smaller increases with clearing and burning, averaging 2%, but greater declines over time. In C2, the C2/C1 TP ratio showed a significant exponential decline of 36% from 1982 to 2014 (P <0.001, R^2 = 95%) (Equation 5 in Table 4). In C3, the C3/C1 TP ratio showed a significant exponential decline of 23% from 1982 to 2014 (P <0.001, R^2 = 75%) (Equation 6 in Table 4).

Bicarbonate-extractable phosphorus

Compared to the observed P(B) data, both C2/C1 and C3/C1 P(B) ratios showed greater increases with clearing and burning, averaging 2.7 times the pre-clearing ratio, but similar declines over time from 1984 to 2014. The significant exponential decline in the C2/C1 ratio (P <0.001, R^2 = 86%)

(Equation 7 in Table 4) to 114% of its pre-clearing ratio over 32 years post-burning, equalled the change in the observed data. The significant exponential decline in the C3/C1 ratio (P <0.001, R^2 = 91%) (Equation 8 in Table 4) to 95% of its pre-clearing ratio also equalled the change in the observed data.

Acid-extractable phosphorus

As for the P(B) ratios, both C2/C1 and C3/C1 P(A) ratios showed greater increases with clearing and burning compared to the observed P(A) data, averaging 2.4 times the pre-clearing ratio. However, over the 32 years post-burning, the P(A) ratios showed a smaller decline than the observed data. From 1982 to 2014, the C2/C1 P(A) ratio had a significant exponential decline (P <0.001, R^2 = 97%) (Equation 9 in Table 4) to 102% of its pre-clearing ratio while the C3/C1 P(A) ratio had a significant exponential decline (P <0.001, R^2 = 97%) (Equation 10 in Table 4) to 80% of its pre-clearing ratio.

Total sulfur

Clearing and burning C2 and C3 increased ratios of C2/C1 and C3/C1 TS by an average of 6%, equalling the average increase in the observed data. Post-burning, the C2/C1 TS ratio showed a significant exponential decline of 53% from 1982 to 2014 (P <0.001, R^2 = 87%) (Equation 11 in Table 4), similar to the observed data. In contrast to the observed C3 TS data, which did not meet the assumptions for valid statistical testing, the C3/C1 TS ratio could be fitted with a significant exponential decline curve (P = 0.009, R^2 = 53%) for the whole post-burning period (Equation 12 in Table 4). The decline in the C3/C1 TS ratio from 1982 to 2014 was 29%.

Total potassium

Clearing and burning C2 and C3 increased ratios of C2/C1 and C3/C1 TK by an average of 4%, similar to the observed data. Post-burning, the ratios for both catchments showed significant exponential declines, similar to the observed data. From 1982 to 2014 the C2/C1 TK ratio declined by 10% (P =

0.001, R^2 = 68%) (Equation 13 in Table 4) and the C3/C1 TK ratio declined by 12% (P <0.001, R^2 = 85%) (Equation 14 in Table 4).

Comparison of approaches for assessing fertility decline

All of the significant declines in observed soil fertility post-burning in both C2 and C3 (Table 3) were confirmed by the ratio analysis (Table 4). When the observed soil fertility data from C2 was adjusted for the natural variation in soil fertility in the control catchment, the R^2 of the exponential decline curves increased by an average of 3% with a maximum change of 12%. When this adjustment was made for C3, the R^2 of the exponential decline curves increased by an average of 9%; however, the maximum change was 42%. While observed C3 OC and TS data showed no significant change in the 32 years post-burning, adjusting for the natural variation in soil fertility in the control catchment revealed a significant decline, similar to C2.

Correlations between soil nitrogen and phosphorus decline and removal in produce

The sum of total nitrogen removed from C2 in grain between soil samplings showed an exponential correlation with soil TN (P = 0.061, R^2 = 54%) (Equation 5). The sum of total phosphorus removed showed an exponential correlation with TP (P = 0.014, R^2 = 75%) (Equation 6), P(A) (P = 0.01, R^2 = 78%) (Equation 7), and P(B) (P = 0.061, R^2 = 54%) (Equation 8).

$$C2\,TN\,(\%) = 0.0811 + 0.0993 \times (0.997^{\,total\ nitrogen\ removed\ in\ grain\ (kg/ha)}) \qquad (5)$$

$$C2\,TP\,(\%) = 0.02739 + 0.0085 \times (0.970^{\,total\ phosphorus\ removed\ in\ grain\ (kg/ha)}) \qquad (6)$$

$$C2\,P(A)\,(mg/kg) = 34.26 + 37.1 \times (0.945^{\,total\ phosphorus\ removed\ in\ grain\ (kg/ha}) \qquad (7)$$

$$C2\,P(B)\,(mg/kg) = 18.55 + 13.59 \times (0.971^{\,total\ phosphorus\ removed\ in\ grain\ (kg/ha)}) \qquad (8)$$

The sum of total nitrogen and total phosphorus removed from C3 in beef showed no significant correlation with soil TN (P = 0.907) and soil TP (P = 0.702) respectively. The sum of total phosphorus removed showed an exponential correlation with P(A) (P <0.001, R^2 = 97%) (Equation 28), and P(B) (P = 0.002, R^2 = 75%) (Equation 9).

$$C3\ P(A)\ (mg/kg) = 19.83 + 27.63 \times (0.781^{\ total\ phosphorus\ removed\ in\ beef\ (kg/ha}) \quad (9)$$

$$C3\ P(B)\ (mg/kg) = 12.26 + 12.63 \times (0.709^{\ total\ phosphorus\ removed\ in\ beef\ (kg/ha)}) \quad (10)$$

Discussion

Nutrient cycling in natural ecosystems can be considered a steady-state, closed system, with nutrients being taken up from the soil by plant roots and being recycled back to the soil through leaf and litter fall and root decay (Murty *et al.* 2002; Radford *et al.* 2007). Under this hypothesis it is expected that no change in soil fertility carbon would occur under brigalow scrub. This was generally supported by the study data with no significant change in organic carbon, total nitrogen, bicarbonate- and acid-extractable phosphorus and total potassium. Radford *et al.*'s (2007) study of organic carbon and total nitrogen at this site from 1981 to 2003 also supports the hypothesis. However, as rainfall patterns fluctuate over time, extended wet periods are likely to result in increased nutrient uptake from deeper down the soil profile by the extending root systems of actively growing plants, followed by increased leaf and litter fall and root decay. This may lead to measurable nutrient redistribution at particular timescales within an otherwise steady-state ecosystem. This redistribution may account for the increases noted in total phosphorus and total sulfur.

Irrespective of the analysis methodology, two distinct trends in soil fertility were observed as a result of land development and land use change. The first trend was for clearing and burning to release a flush of nutrients which subsequently declined over time to near, or below, pre-clearing levels. The clearest display of this trend was in mineral nitrogen and available phosphorus with smaller increases in total phosphorus, total sulfur and total potassium. The second trend was an ongoing decline in fertility commencing at clearing. This was observed in organic carbon and total nitrogen. Both of these trends reflect predictions that clearing brigalow followed by subsequent exploitative land use would result in declining nutrient availability and landscape productivity (Dowling *et al.* 1986).

The effect of land clearing and burning on soil bulk density

Worldwide, an increase in bulk density as a result of land development and long-term cropping or grazing is commonplace (Dalal *et al.* 2005; Dalal and Mayer 1986b; Murty *et al.* 2002). The primary mechanism for increase is physical compaction by machinery and animal hoof traffic, and the degradation of soil structure and loss of organic matter in tilled soil. Conceptually, land use change followed by more than 30 years of either cropping or grazing should have increased bulk density in both the cropped and grazed catchments of this study. Although the significance of trends identified via regression analysis varied, all comparisons of pre-clearing bulk density with long-term averages under cropping and grazing showed an increase with land development. In the same period, bulk density under brigalow remained constant. Changes in the ratios of bulk density between the developed catchments and the control catchment also suggested an increase with land development.

Determining change in bulk density was confounded due to it only being measured in seven of the fourteen sampling events. In addition to limited data, other confounding issues include differing soil water content between samplings and the corresponding shrinking and swelling characteristics of Vertosols; and the ability of the chosen core diameter to obtain representative samples, particularly in heavily cracked dry soils, in wet soils prone to compaction or distortion and in soils prone to shattering (Al-Shammary *et al.* 2018; Berndt and Coughlan 1977; Coughlan *et al.* 1987).

Coughlan *et al.* (1987) stress the influence of soil water content on bulk density and note that the swelling of Vertosols with increasing soil water and the resultant reduction in bulk density complicates the comparison of measurements over time. On two occasions soil water was measured within two weeks of a soil sampling event that had measured bulk density. In both instances, soil water under cropping and grazing was substantially greater than under brigalow. However, bulk densities of the agricultural catchments continued to be similar or higher than that of the brigalow catchment despite likely reductions in observed bulk density due to increased soil water storage. This provides additional evidence that an increase in bulk density has occurred with land development and long-term cropping or grazing. Other than variations in soil water content, the primary limitation to measuring bulk density in this study is likely to be sampling error associated with loss of sample and inaccurate core trimming in friable soils or due to shattering of dry soil during coring.

The effect of land clearing and burning on soil fertility

Increases in mineral nitrogen, total phosphorus, available phosphorus, total sulfur and total potassium as a result of burning vegetation on the soil surface, as observed in this study, are well documented in both the Australian and international literature (Brennan *et al.* 2004; Butler *et al.* 2017; Carreira and Niell 1995; Castelli and Lazzari 2002; Ellis and Graley 1983; Fraser and Scott 2011; Kyuma *et al.* 1985; MacDermott *et al.* 2017). The increase has been attributed to nutrient release from plant material and deposition in ash, and is often referred to as the ash bed effect (Castelli and Lazzari 2002; Cowie *et al.* 2007; Herpin *et al.* 2002; Kyuma *et al.* 1985; May and Attiwill 2003; Raison 1979; Roder *et al.* 1993). These increases are typically restricted to the surface few centimetres of the soil profile (Castelli and Lazzari 2002; Ellis and Graley 1983; Kyuma *et al.* 1985).

Decreases in soil organic carbon and total nitrogen as a result of burning are also well documented in Australian and international literature (May and Attiwill 2003; Oyedeji *et al.* 2016). However, some studies, including a meta-analysis, have shown no change in total nitrogen as a result of burning (Guinto *et al.* 2001; Wan *et al.* 2001). Initial soil nitrogen level, soil clay content and fire intensity can account for these contrasting observations. Firstly, low fertility soils may have already lost their most fire-susceptible nitrogen fractions. Secondly, clay particles within soil assist in physically protecting organic matter from the effects of fire, therefore soils with varying clay content are likely to display different responses to burning (Guinto *et al.* 2001). Finally, low intensity fires have been shown to increase total nitrogen whereas high intensity fires decrease total nitrogen (Raison 1979). The fire intensity resulting from the burning of pulled brigalow scrub would be similar to that of slash fires and wildfires, providing intense heat for long periods, hence the observation of a loss of total nitrogen with burning in this study (Hobley *et al.* 2017; Johnson 1964; Raison 1979).

The effect of land use change on soil carbon

The decline in organic carbon when brigalow scrub was developed for cropping supports the earlier findings of Radford *et al.* (2007) at this site, and mirrors the response of other pre-clearing Australian and international landscapes developed for, and managed as, long-term cropping (Collard and Zammit 2006; Murty *et al.* 2002). The decline is typically restricted to the surface soil layers no deeper than 1 m (Dalal *et al.* 2005). The mechanism is usually attributed to the removal of nutrients in harvested grain, reduced carbon inputs, and the impacts of tillage on soil structure, chemical and biological processes including shattering, redistribution, oxidation and decomposition (Murty *et al.* 2002).

The finding of no significant change in observed organic carbon when brigalow scrub was developed for grazing is in agreement with the findings of other studies conducted at this site (Dalal *et al.* 2011; Dalal *et al.* 2013; Radford *et al.* 2007). The international review of Murty *et al.* (2002) concluded that on average, the conversion of forest to uncultivated grazing does not lead to a loss of organic carbon; however, this does not hold for all specific sites. Within Australia, Harms *et al.* (2005)

reported organic carbon losses from coarse textured soils such as Kandosols as a result of changing land use from native vegetation to grazing, but found no change in Sodosols and Vertosols, which reflect the soil types of this study. However, while no decline in organic carbon was observed after clearing brigalow followed by grazing for 31 years, a significant decline was evident during the first 17 years of grazing. When the observed organic carbon data was adjusted for the natural variation in soil fertility in the control catchment, a statistically significant decline in the organic carbon ratios between the catchments was found for the entire study period. These alternative approaches suggest that a decline in organic carbon has occurred.

Further evidence of organic carbon decline under grazing at this site is evident in the observation that organic carbon derived from the original brigalow vegetation comprised only 58% of measured organic carbon while buffel grass derived organic carbon contributed the remaining 42% (Dalal *et al.* 2011). Without this replacement of carbon by buffel grass, a greater decline in total organic carbon would have occurred. As growth of buffel grass is highly responsive to seasonal rainfall trends, variation in the observed organic carbon data could indicate changes in carbon inputs and nutrient redistribution within a steady state ecosystem, as hypothesised could occur under brigalow scrub. The literature also shows that there is potential for increased organic carbon sequestration with low precipitation and decreased sequestration with high precipitation (McSherry and Ritchie 2013). This suggests that carbon sequestration at the study site is likely to vary temporally due to the variable semi-arid climate, further explaining fluctuations in observed organic carbon.

The effect of land use change on soil total nitrogen

As for organic carbon, the decline in total nitrogen when brigalow scrub was developed for cropping supports the earlier findings of Radford *et al.* (2007) at this site. Significant loss of total nitrogen following the conversion of forest to cropping or multiple years of cultivated cropping alone was also found in other long-term studies (Anaya and Huber-Sannwald 2015; Dalal *et al.* 2005; Dalal and

Mayer 1986b) and international reviews (Murty *et al.* 2002). Removal of nitrogen in grain has been identified as the primary mechanism of total nitrogen loss (Dalal *et al.* 2005; Dalal and Mayer 1986a) and was shown by Radford *et al.* (2007) to account for 39% of the total nitrogen lost from the surface 0.3 m of the soil profile at this site between 1981 and 2003. In agreement with these finding, regression analysis showed nitrogen removed from the cropped catchment as grain accounted for 54% of the variation in total nitrogen from 1981 to 2014. On a kg/ha basis, nitrogen removed from catchment in grain accounted for 80% of the total nitrogen lost from the surface 0.1 m of the soil profile prior to the planting of legume ley pasture. In contrast, the equivalent of 8% of soil total nitrogen decline was lost in runoff (Elledge and Thornton 2017).

The increase in total nitrogen from 2008 to 2014 may be attributed to nitrogen fixation by the butterfly pea legume ley pasture planted in 2010. The ley pasture was planted in order to arrest declining total nitrogen that was limiting the productivity of dryland farming in the catchment (Huth *et al.* 2010; Radford *et al.* 2007). The ability of butterfly pea to increase total nitrogen is well documented in central Queensland (Collins and Grundy 2005).

With no pasture legumes to maintain fertility, clearing brigalow scrub for grazing resulted in ongoing total nitrogen decline from 1981 to 2014. This supports the findings of Dalal *et al.* (2013) who found significant decline in total nitrogen at this site 23 years after clearing brigalow scrub for grazing. However, both of these studies contrast with the findings of Radford *et al.* (2007). This is likely due to differences in sampling strategies, analytical methods, and the specific comparisons being made. This current study reports the longest period of record, used the most intensive sampling strategy, consistent analytical methodology and compared each catchment to its starting soil fertility, so should be considered the most robust. Globally, the conversion of forest to uncultivated grazing generally does not lead to a loss of total nitrogen, however this does not hold for all specific sites (Murty *et al.* 2002). This is reflected in the contrasting conclusions of Australian studies. For example, Harms *et al. (2005)* found no significant loss of total nitrogen across multiple paired sites encompassing the same soil and vegetation as Dalal *et al. (2005)*. In contrast, a single paired site study by Dalal *et al. (2005)* found a decrease in total nitrogen when mulga forest were developed for grazed pasture with the majority of loss occurring from the surface 0.1 m of the soil profile. Removal of total nitrogen in beef accounted for less than half of this loss with additional potential losses via deep drainage.

The decline in total nitrogen in this study showed no correlation with nitrogen removal in beef and on a kg/ha basis, removal in beef accounted for 32% of the total nitrogen lost from the surface 0.1 m of the soil profile. This is comparable to the equivalent of 25% of soil total nitrogen decline lost in runoff (Elledge and Thornton 2017). Losses of nitrogen through volatilisation from urine and faeces was estimated to remove 71 kg/ha of nitrogen, equivalent to 49% of total nitrogen loss. Annual buffel grass yields have been shown to be in the order of 3, 000 kg/ha (Myers and Robbins 1991). Previous work at this site has shown the standing above ground biomass of buffel grass was 4, 601 kg/ha and contained the equivalent of 27.6 kg/ha of nitrogen, equivalent to 19% of total nitrogen loss (Thornton and Elledge 2013). Annual root growth biomass estimations at this site are similar to above ground biomass (Dalal *et al.* 2013) and are likely to have similar nitrogen contents (Robertson *et al.* 1993), potentially accounting for a similar proportion of total nitrogen loss. The work of Graham *et al.* (1985), on similar vegetation and soil associations elsewhere within the Fitzroy basin, suggests that this is likely an underestimation having measured 207 kg/ha of nitrogen in buffel grass roots to 0.3 m. The combination of annual above and below ground plant growth and litter deposition over 32 years likely accounts for the majority of total nitrogen decline and immobilisation in plant biomass under grazing although significant losses occur via removal in beef, volatilisation and runoff.

The effect of land use change on soil mineral nitrogen

The immediate, short term increase in ammonium-nitrogen post-burning in C2 and C3, followed by a delayed, longer-lived increase in nitrate-nitrogen clearly demonstrates the generalised pattern of available nitrogen response to fire, as documented in the meta-analyses of Boerner *et al.* (2009) and Wan *et al.* (2001). The mechanism of increase is attributed to ammonium-nitrogen liberation from organic matter followed by its nitrification to nitrate-nitrogen. This is supported by previous work at this site attributing many of the changes in soil chemistry after burning to the effects of soil heating (Hunter and Cowie 1989). Subsequent declines over time were attributed to runoff losses, plant uptake and microbial immobilisation (Hunter and Cowie 1989).

The extended period of elevated nitrate-nitrogen under cropping is likely to reflect the stimulating influence of fallow tillage on nitrogen mineralisation as described by Dalal and Mayer (1986b). This is supported by the observed decline in mineral nitrogen around 15 years post-burning that corresponds to a change in cropping management practices to minimum tillage and opportunity cropping. These practices reduce tillage and shorten fallows, leading to reduced mineralisation combined with increased nitrogen uptake due to increased cropping frequency. Declining total nitrogen is also likely to result in declining mineral nitrogen under continuous cropping. This is demonstrated elsewhere within the Dawson sub-catchment of the Fitzroy basin where mineral nitrogen levels of Vertosols after more than 30 years of cropping were 82% lower than adjacent Vertosols still supporting native brigalow scrub (Shrestha *et al.* 2015).

The rapid decline of nitrate-nitrogen in C3 is likely due to uptake by the newly planted buffel grass pasture. Similar pastures in central Queensland have been shown to be highly productive in the first two years after planting due to high levels of available nitrogen, with productivity declining over time as available nitrogen declines and nitrogen immobilisation occurs (Myers and Robbins 1991). Decline and immobilisation in the grazed catchment at this site is demonstrated after the first two to three years in the ongoing low levels and minimal fluctuation of total and mineral nitrogen compared to that under cropping and brigalow. It is further demonstrated by the decline in pasture productivity and cattle live weight gain over time at this site as described by Radford *et al.* (2007).

The effect of land use change on soil total phosphorus

While the enrichment of surface soil with phosphorus as a result of burning was clear, in the absence of fertilisation, phosphorus depletion commenced immediately. Within four years, total phosphorus was depleted to near or below pre-clearing levels. Removal of phosphorus in grain was equivalent to 95% of total phosphorus lost under cropping; however, removal of phosphorus in beef was only equivalent to 22% of the loss of total phosphorus under grazing. Removal of total phosphorus in runoff was equivalent to 12% of the total decline under cropping and 11% of the total decline under grazing (Elledge and Thornton 2017). Extraction of phosphorus from the soil profile below 0.1 m is clearly occurring under cropping given that total phosphorus removal in grain and runoff exceeded the measured total phosphorus decline in the top 0.1 m of the soil profile.

Other Queensland and international studies have also reported declines in total phosphorus under cropping (Bowman *et al.* 1990; Song *et al.* 2011; Standley *et al.* 1990; Wang *et al.* 2012; Zhang *et al.* 2006). Typically, the decline could be almost entirely accounted for in crop removal (Dalal 1997). However, changes in total phosphorus under grazing are typically less pronounced and the mechanism for change less obvious. Erosion and leaching losses are acknowledged to play some role in total phosphorus decline under grazing however they are unlikely to be a key decline mechanism, particularly in flat landscapes with high clay content soils such as Vertosols (Townsend *et al.* 2002). Internationally, the removal of phosphorus in beef was poorly correlated with total phosphorus decline and hence was unlikely to be a key decline mechanism (McGrath *et al.* 2001; Townsend *et al.* 2002). These observations lead Townsend et al. (2002) to conclude that the bulk of total phosphorus decline must be occurring by other mechanisms.

Previous work has shown the above ground biomass of buffel grass in the grazed catchment contained the equivalent of 5.8 kg/ha of phosphorus (Thornton and Elledge 2013). Assuming the soil contribution to phosphorus in above ground biomass is equal to one third of the phosphorus content of the biomass grown each season, this transfer over 32 years is equivalent to the amount of total phosphorus removed from the soil. The cycling of phosphorus from soil to plant to animal waste is also likely to account for some of the phosphorus lost given that phosphorus in dung can exceed that contained within both the above-ground plant and litter biomass (Dubeux Jr *et al.* 2007), and its deposition on the soil surface increases its susceptibility to loss in runoff (McGrath *et al.* 2001). The key mechanisms of decline in total phosphorus under grazing in this study is likely to be redistribution into plant biomass and litter with additional smaller losses through runoff and removal in beef.

The effect of land use change on soil available phosphorus

Similar to total phosphorus, the enrichment of surface soil with available phosphorus as a result of burning was clear and in the absence of fertilisation, depletion commenced immediately. Under cropping, bicarbonate-extractable phosphorus was still above pre-clearing levels 32 years post-burning while acid-extractable phosphorus had declined below pre-clearing levels. Under grazing, both acid and bicarbonate-extractable phosphorus declined below pre-clearing levels within 14 years post-burning.

Other long-term Queensland studies conducted at Chinchilla and Mt. Murchison on Vertosols that originally supported brigalow vegetation associations, also found declines in available phosphorus as a result of cropping (Dalal 1997; Thomas *et al.* 1990). The declines were attributed to removal of phosphorus in grain, transformation within soil, and runoff and erosion processes. However, at Mt. Murchison, it was noted that phosphorus removal by the crop and stubble could not be accounted for simply in terms of acid- and bicarbonate-extractable phosphorus (Thomas *et al.* 1990). Greater retention of bicarbonate-extractable phosphorus in treatments with higher soil biomass and the replacement of depleted bicarbonate-extractable phosphorus with phosphorus from other pools (Standley *et al.* 1990) further indicates that land use change alters the speciation and cycling of phosphorus in soil. Similar declines in available phosphorus are noted internationally (Nancy Mungai *et al.* 2011; Song *et al.* 2011). They are also attributed to cultivation and erosion-induced declines in soil structure leading to reductions in soil organic matter, promoting microbial cycling of available phosphorus (Zhang *et al.* 2006). Harvest losses were also noted as a decline mechanism. In this study, phosphorus removal in grain was better correlated with total phosphorus than with either measure of available phosphorus. As total phosphorus accounts for losses from the organic pool, this suggests that both the inorganic and organic phosphorus pools are depleted by grain removal. The key mechanism of decline in available phosphorus under cropping in this study is likely to be removal in grain combined with cycling into other phosphorus pools.

The levels of phosphorus enrichment and decline following land use change for grazing in this study exceed that reported by Sangha *et al.* (2005) for grazing systems developed on similar vegetation and soil associations elsewhere within the Fitzroy basin. Their study found no difference in bicarbonate-extractable phosphorus under uncleared brigalow compared to sites cleared for grazing five, twelve and thirty three years previously. This lack of difference in bicarbonate-extractable phosphorus is likely due to the space-for-time paired site approach, which fails to guarantee the same starting condition for each pair; the impacts of grazing on the uncleared control plots; and the bicarbonate-extractable phosphorus levels of their uncleared control being only 56% of the BCS brigalow scrub control in this study.

Internationally, changing land use from virgin forest to grazing has also resulted in an initial flush of available phosphorus followed by a decline (McGrath *et al.* 2001; Townsend *et al.* 2002). Pasture growth and above-ground biomass accounted for some of the decline; however, beef production was poorly correlated. While removal of phosphorus in beef showed no correlation with total phosphorus in this study, it explained 97% of the decline in acid-extractable phosphorus and 75% of the decline in bicarbonate-extractable phosphorus. This suggests that any loss of phosphorus from the organic pool is likely being replaced from the inorganic pool (Fonte *et al.* 2014; Garcia-Montiel *et al.* 2000; McGrath *et al.* 2001; Townsend *et al.* 2002). The key mechanism of decline in available phosphorus under grazing in this study is likely to be removal in beef combined with cycling into other phosphorus pools. Additional losses are likely through the cycling of phosphorus from soil to plant to animal waste with smaller losses in runoff.

The effect of land use change on soil sulfur

As for phosphorus, surface soil was enriched with sulfur as a result of burning and in the absence of fertilisation, depletion commenced immediately. Other studies, both in the Brigalow Belt bioregion and internationally, attribute sulfur decline under cropping to mineralisation associated with cultivation (Dalal and Mayer 1986b; Kopittke *et al.* 2016; Wang *et al.* 2006). Decline under grazing has also been attributed to accelerated mineralisation with additional declines as a result of reduced inputs of plant residues, particularly in arid, low-fertility landscapes, and losses in runoff and leaching (Steffens *et al.* 2008; Wiesmeier *et al.* 2009).

Sulfur is a constituent of organic matter and has similar responses under agriculture as nitrogen (Kopittke *et al.* 2016; Williams 1962). The rapid decline in sulfur within two years of burning mirrors that of total and mineral nitrogen, suggesting its removal from soil by actively growing crops and pasture in response to the ash bed effect. Leaching losses are also likely during this time given deep drainage through the soil profile increased from <1 mm/yr pre-clearing to 59 mm/yr under development for cropping and 32 mm/yr under development for grazing (Silburn *et al.* 2009). Some ongoing loss of easily leached sulfur fractions may have occurred under cropping where deep drainage averaged 19.8 mm/yr; however, leaching losses under grazing are unlikely with deep drainage returning to near pre-clearing levels of <1 mm/yr.

While some of the continued sulfur decline under cropping can certainly be attributed to mineralisation associated with tillage, estimates of grain sulfur content combined with measured yield data indicate that 63% of the lost sulfur can be accounted for in crop removal. In contrast, estimates of the sulfur content of beef combined with measured live weight gain data indicate that only 5% of the lost sulfur can be accounted for in beef removal. This is supported by the observed sulfur data showing continued decline under cropping but little change under grazing after the initial decline in the ash bed effect. Thus removal of sulfur in agricultural products is a major pathway under cropping but is negligible under grazing.

The effect of land use change on soil potassium

As for phosphorus and sulfur, surface soil was enriched with potassium as a result of burning, and in the absence of fertilisation, depletion commenced immediately. Both cropping and grazing land uses lost similar amounts of potassium over the 32 years post-burning. Potassium decline has been noted in cropping systems worldwide, particularly where crop residue removal was practiced in addition to grain removal (Chen *et al.* 2006; Karlen *et al.* 2013; Rezapour *et al.* 2013). Decline has also been noted under grazing systems, typically with erosion as the primary loss mechanism, while reafforestation of grazing lands has been shown to increase surface soil potassium (Cheng *et al.* 2016; Huth *et al.* 2012; Liu *et al.* 2010; Sangha *et al.* 2005).

While some potassium is removed in grain, potassium in crop residues greatly exceeds that removed in grain (Chen *et al.* 2006). This implies that removal of potassium in beef is greatly exceeded by the potassium retained in pasture and litter. Despite similar percentage declines in potassium under both cropping and grazing, potassium removal in grain accounted for 39% of the total decline under cropping while removal in beef accounted for only 1% of the decline under grazing. This suggests that removal of potassium in agricultural produce is not the primary loss mechanism.

Potassium is relatively immobile in soil and prone to surface stratification, but can be leached slowly and lost in runoff (Bertol *et al.* 2007; Drew and Saker 1980). The return of crop residues and buffel grass litter to the soil surface promotes stratification in both the cropping and grazing systems of this study, leaving nutrients vulnerable to loss in runoff. Given that changing land use from brigalow scrub to cropping or grazing doubled runoff (Thornton *et al.* 2007), and similar potassium losses were found under both cropping and grazing, it is likely that loss in runoff is the primary loss mechanism at this site. Drainage is unlikely to be a primary loss mechanism given drainage under the two systems is two orders of magnitude apart and does not reflect the similar potassium losses from the surface soil of each system.

Conclusion

Development of brigalow scrub for cropping or grazing significantly altered soil nutrient balances. Initial clearing and burning resulted in a temporary increase, or flush, of mineral nitrogen, total and available phosphorus, total potassium and total sulfur in the surface soil (0 to 0.1 m) as a result of soil heating and the ash bed effect. Over the 32 years since changing land use from brigalow scrub to cropping, surface soil fertility has declined significantly. Specifically, organic carbon has 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% and total potassium by 9% from post-burn, pre-cropping levels. This decline in fertility has limited crop yields and would have had an economic impact on a commercial cropping enterprise (Radford *et al.* 2007). However, the planting and maintenance of a butterfly pea legume ley pasture increased total nitrogen by 15% within five years. The limited grazing of the ley pasture that was undertaken would have provided some economic benefit to offset the foregone cropping opportunities.

Surface soil fertility has also declined under grazing over the same period but in a different pattern to 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. Total nitrogen declined by 22% and in the absence of a legume in the pasture, no fertility restoration occurred. Total phosphorus declined by 14%, equating to only half of the decline under cropping. Bicarbonate-extractable phosphorus declined by 64% and acid-extractable phosphorus declined by 66%; both greater than the decline observed under cropping, possibly due to immobilisation as organic phosphorus. 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. As for cropping, this fertility decline has limited pasture production and hence beef production. Despite these production limitations, the grazing system is representative of much of the extensive grazing undertaken in northern Australia.

The primary mechanism of nutrient loss depended on the 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. The addition of legumes into both the cropping and grazing systems would assist in fertility restoration however, particularly in the case of cropping, may not enable continued production without fertility decline (Huth *et al.* 2010). In contrast to the fertility decline of the agricultural land uses, surface soil fertility of the brigalow scrub remained in relative equilibrium.

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Table 1.

Catchment	Area (ha)	Land use by experimental stage			
		Stage I (Jan 1965 to Mar 1982)	Stage II (Mar 1982 to Sep 1984)	Stage III (Sep 1984 to Jan 2010)	Stage IV (Jan 2010 to Jun 2014)
C1	16.8	Brigalow scrub	Brigalow scrub	Brigalow scrub	Brigalow scrub
C2	11.7	Brigalow scrub	Development	Cropping	Ley pasture
C3	12.7	Brigalow scrub	Development	Grazing	Grazing

Table 2.

Year	Number of samples analysed per catchment								
	OC	TN	NH_4-N	NO_3-N	TP	P(B)	P(A)	TS	TK
1981	3	3	30	30	3	3	3	3	3
1982	3	3	30	30	3	3	3	3	3
1983	3	3	30	30	3	3	3	3	3
1984	30	30*	3	3	3	3	6	3	3
1985	3**	3	30	30	3	3	3	3	3
1986	30	3	3	3	3	3	6	3	3
1987	30	30	30***	30***	3	3	6	3	3
1990	30	3	30	30	3	3	6	3	3
1994	3	3	3	3	3	3	6	3	3
1997	30	3	30****	30****	3	3	6	3	3
2000	3	3	3	3	3	3	6	3	3
2003	3	3	3	3	3	3	6	3	3
2008	30	30	30	30	30	30	30	30	30
2014	30	3	30	30	30	30	30	30	30

*C1 = 27, **C1 = 30, ***C1 = 25, C3=17, ****C2 = 29

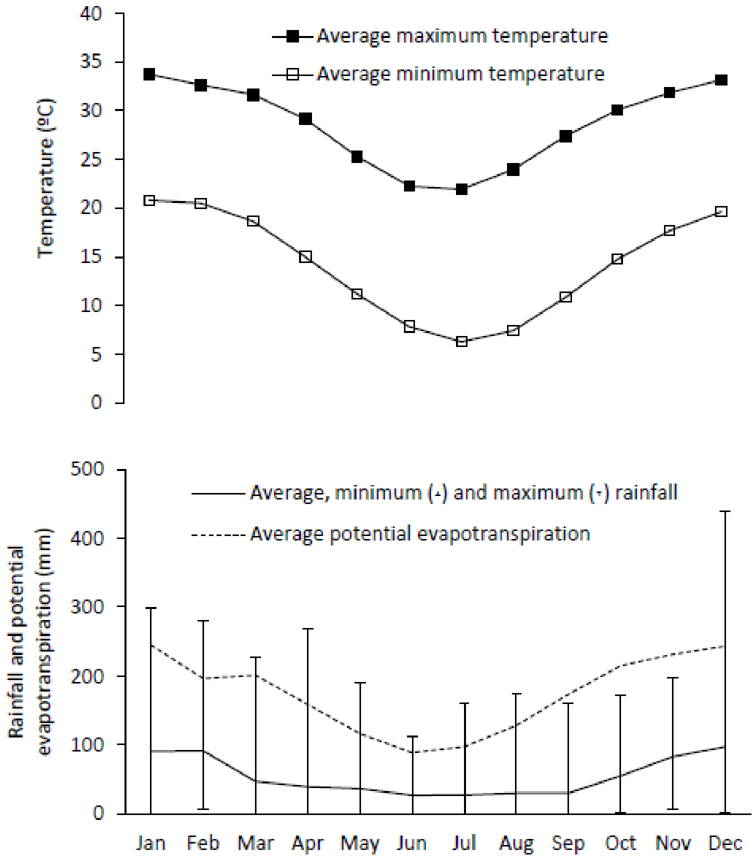
Table 3. Exponential equations describing trends in soil fertility over time where *x* is years since burning.

Parameter	Catchment	Period	Exponential trend equation	*P*	R^2	Equation number
Organic carbon	C1	1981 to 2014		NS		
	C2	1981 to 2014	$C2\ OC\ (\%) = 1.203 + 0.842 \times (0.823^x)$	<0.001	0.88	1
	C3	1981 to 2000	$C3\ OC\ (\%) = 1.53 + 0.207 \times (0.49^x)$	<0.001	0.79	2
Total nitrogen	C1	1981 to 2014		NS	0.39	
	C2	1981 to 2014	$C2\ TN\ (\%) = 0.0866 + 0.104 \times (0.84^x)$	<0.001	0.91	3
	C3	1981 to 2014	$C3\ TN\ (\%) = 0.12 + 0.028 \times (0.611^x)$	0.049	0.54	4
Total phosphorus	C1	1981 to 2014	$C1\ TP\ (\%) = 0.0265 + 0.0023 \times (1.035^x)$	<0.001	0.77	5
	C2	1982 to 2014	$C2\ TP\ (\%) = 0.0268 + 0.0097 \times (0.875^x)$	<0.001	0.91	6
	C3	1982 to 2014	$C3\ TP\ (\%) = 0.027 + 0.0053 \times (0.478^x)$	0.009	0.53	7
Bicarbonate-extractable phosphorus	C1	1981 to 2014		NS		
	C2	1982 to 2014	$C2\ P(B)\ (mg/kg) = 18.14 + 15.7 \times (0.852^x)$	<0.001	0.88	8
	C3	1982 to 2014	$C3\ P(B)\ (mg/kg) = 12.62 + 22.86 \times (0.744^x)$	<0.001	0.92	9
Acid-extractable phosphorus	C1	1981 to 2014		NS		
	C2	1982 to 2014	$C2\ P(A)\ (mg/kg) = 31.02 + 29.75 \times (0.849^x)$	<0.001	0.91	10
	C3	1982 to 2014	$C3\ P(A)\ (mg/kg) = 21.3 + 39.42 \times (0.818^x)$	<0.001	0.97	11
Total sulfur	C1	1981 to 2014	$C1\ TS\ (\%) = 0.0249 - 0.0043 \times (0.984^x)$	0.008	0.51	12
	C2	1982 to 2014	$C2\ TS\ (\%) = 0.0135 + 0.0095 \times (0.715^x)$	<0.001	0.9	13
	C3	1982 to 2014		NA		
Total potassium	C1	1981 to 2014		NS		
	C2	1982 to 2014	$C2\ TK\ (\%) = 0.457 + 0.039 \times (0.893^x)$	0.004	0.61	14
	C3	1982 to 2014	$C3\ TK\ (\%) = 0.239 + 0.027 \times (0.806^x)$	<0.001	0.94	15

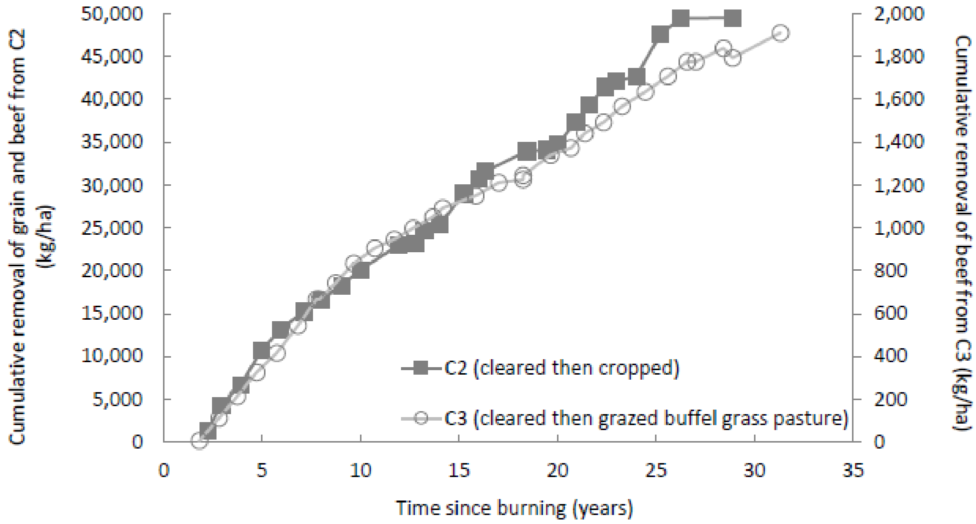
Table 4. Exponential equations describing the ratios of soil fertility over time in catchments two and three to the soil fertility in catchment 1, where *x* is years since burning.

Parameter	Ratio	Period	Exponential trend equation	*P*	R^2	Equation number
Organic carbon	C2/C1	1981 to 2014	$C2/C1\ OC = 0.5251 + 0.4332 \times (0.8649^x)$	<0.001	0.91	1
	C3/C1	1981 to 2012	$C3/C1\ OC = 0.7613 + 0.02445 \times (0.1095^x)$	0.05	0.32	2
Total nitrogen	C2/C1	1981 to 2014	$C2/C1\ TN = 0.5059 + 0.5222 \times (0.8496^x)$	<0.001	0.93	3
	C3/C1	1981 to 2014	$C3/C1\ TN = 0.7071 + 0.0681 \times (0.290^x)$	0.043	0.33	4
Total phosphorus	C2/C1	1982 to 2014	$C2/C1\ TP = 0.7334 + 0.5014 \times (0.9406^x)$	<0.001	0.95	5
	C3/C1	1982 to 2014	$C3/C1\ TP = 0.8621 + 0.2019 \times (0.8091^x)$	<0.001	0.75	6
Bicarbonate-extractable phosphorus	C2/C1	1982 to 2014	$C2/C1\ P(B) = 0.913 + 1.556 \times (0.9216^x)$	<0.001	0.86	7
	C3/C1	1982 to 2014	$C3/C1\ P(B) = 0.768 + 1.746 \times (0.8197^x)$	<0.001	0.91	8
Acid-extractable phosphorus	C2/C1	1982 to 2014	$C2/C1\ P(A) = 1.0516 + 1.6841 \times (0.9018^x)$	<0.001	0.97	9
	C3/C1	1982 to 2014	$C3/C1\ P(A) = 0.7736 + 1.952 \times (0.8565^x)$	<0.001	0.97	10
Total sulfur	C2/C1	1982 to 2014	$C2/C1\ TS = 0.6177 + 0.4874 \times (0.756^x)$	<0.001	0.87	11
	C3/C1	1982 to 2014	$C3/C1\ TS = 0.7736 + 0.337 \times (0.245^x)$	0.009	0.53	12
Total potassium	C2/C1	1982 to 2014	$C2/C1\ TK = 0.58 + 0.112 \times (0.971^x)$	0.001	0.68	13
	C3/C1	1982 to 2014	$C3/C1\ TK = 0.32862 + 0.04139 \times (0.8752^x)$	<0.001	0.85	14

956 **Figures**



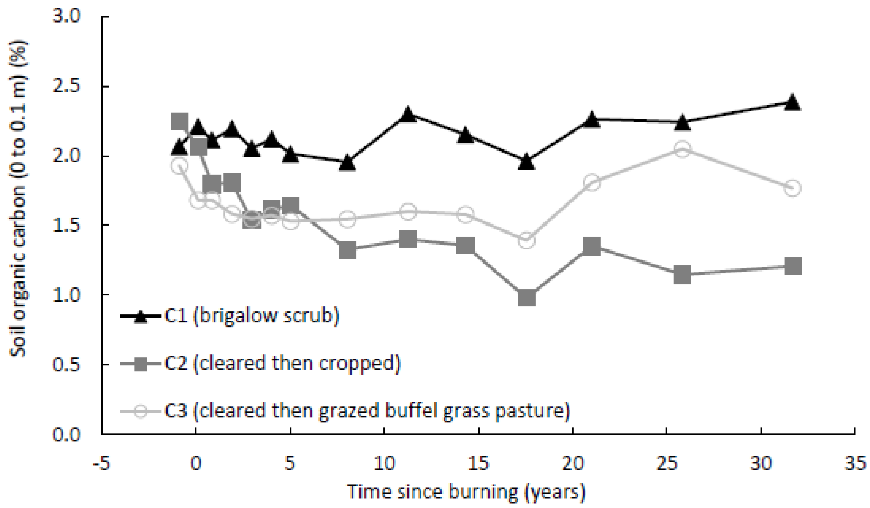
957 **Fig. 1.**



Cumulative removal of grain and beef from C2 (kg/ha)
50,000
45,000
40,000
35,000
30,000
25,000
20,000
15,000
10,000
5,000
0
Cumulative removal of beef from C3 (kg/ha)
2,000
1,800
1,600
1,400
1,200
1,000
800
600
400
200
0
0
5
10
15
20
25
30
35
Time since burning (years)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)

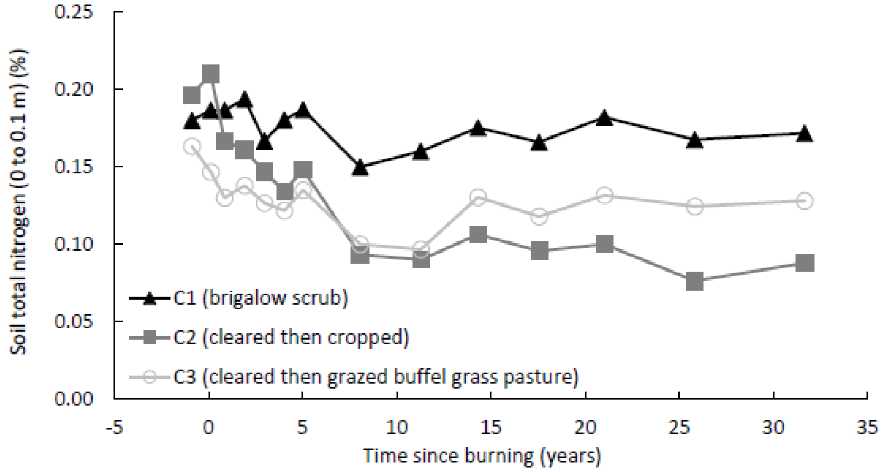
958

959 **Fig. 2.**


3.0
2.5
2.0
1.5
1.0
0.5
0.0
Soil organic carbon (0 to 0.1 m) (%)
C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)
-5
0
5
10
15
20
25
30
35
Time since burning (years)

960

961 **Fig. 3.**


0.25
0.20
0.15
0.10
0.05
0.00
Soil total nitrogen (0 to 0.1 m) (%)
-5
0
5
10
15
20
25
30
35
Time since burning (years)
C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)

962

963 **Fig. 4.**

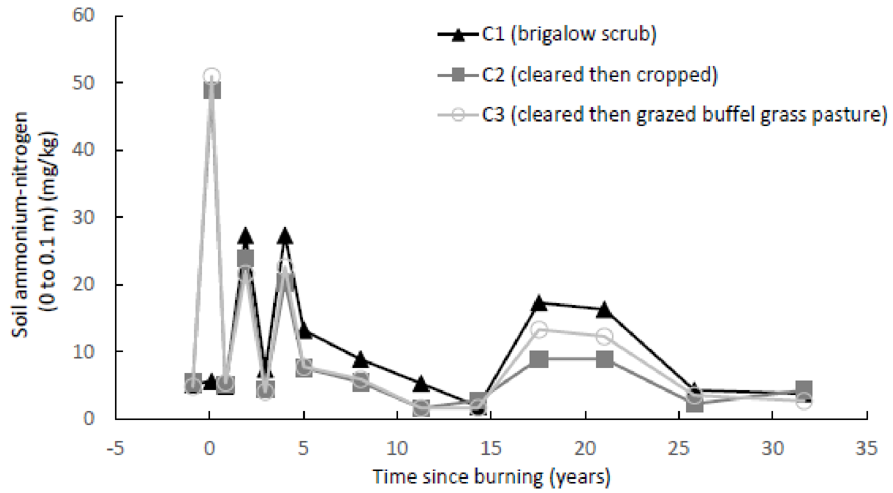
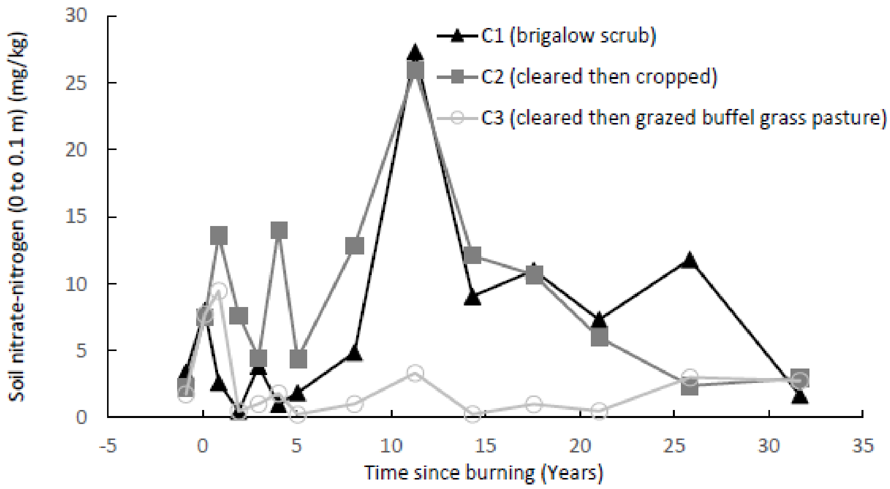

C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)
Soil ammonium-nitrogen (0 to 0.1 m) (mg/kg)
Time since burning (years)
60
50
40
30
20
10
0
-5
0
5
10
15
20
25
30
35

Fig. 5.


Soil nitrate-nitrogen (0 to 0.1 m) (mg/kg)
30
25
20
15
10
5
0
-5
0
5
10
15
20
25
30
35
Time since burning (Years)
C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)

966

967 **Fig. 6.**

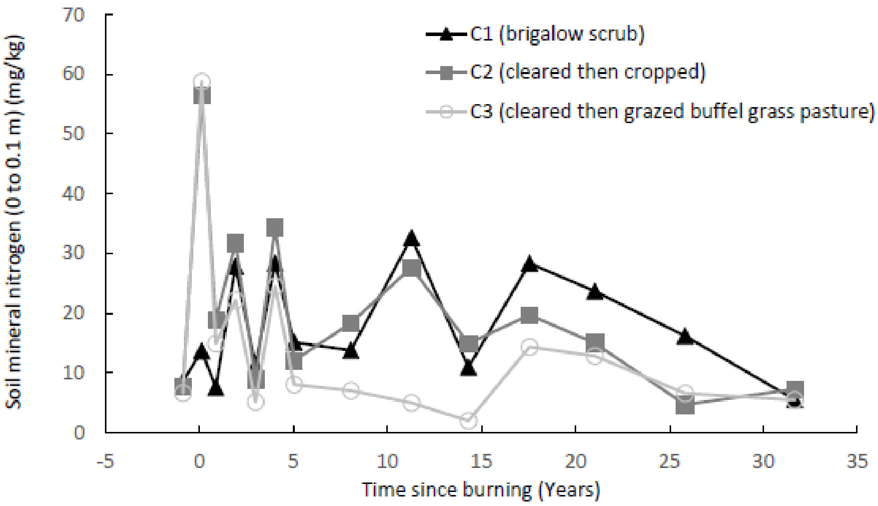
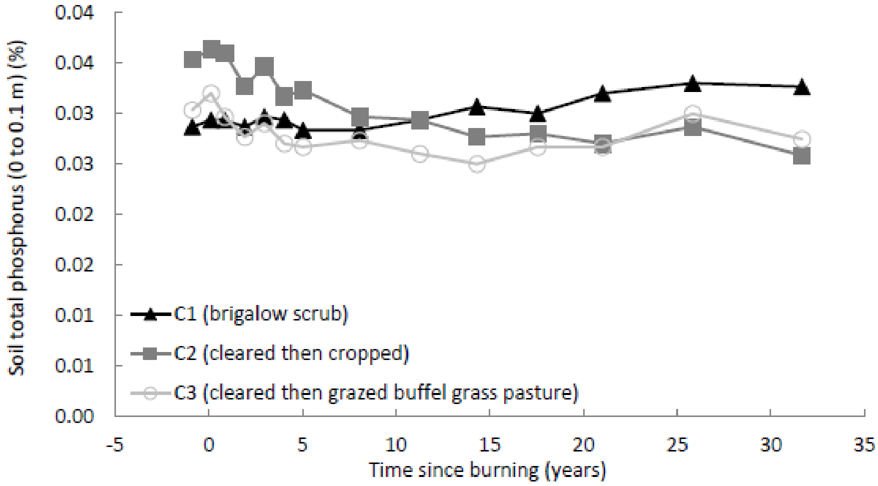

C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)
Soil mineral nitrogen (0 to 0.1 m) (mg/kg)
70
60
50
40
30
20
10
0
-5
0
5
10
15
20
25
30
35
Time since burning (Years)

Fig. 7.


0.04
0.04
0.03
0.03
0.02
0.02
0.01
0.01
0.00
Soil total phosphorus (0 to 0.1 m) (%)
C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)
-5
0
5
10
15
20
25
30
35
Time since burning (years)

970

971 **Fig. 8.**

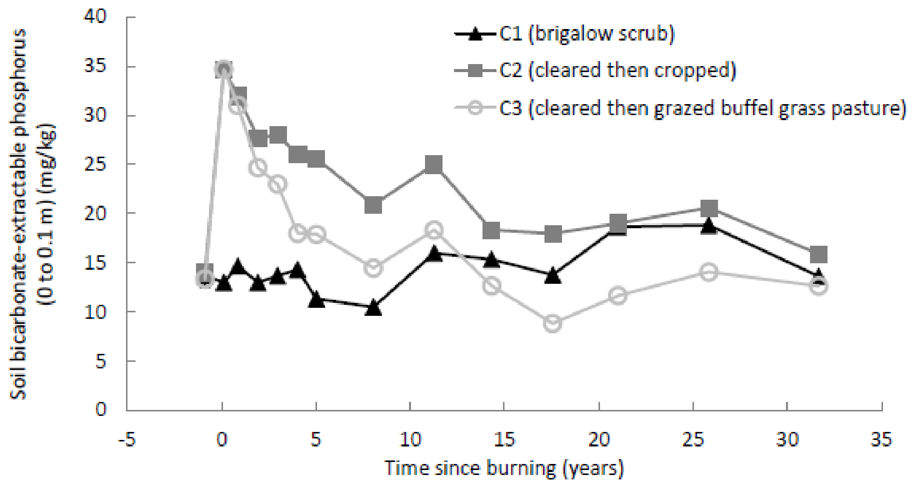

C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)
Soil bicarbonate-extractable phosphorus
(0 to 0.1 m) (mg/kg)
Time since burning (years)
40
35
30
25
20
15
10
5
0
-5
0
5
10
15
20
25
30
35

Fig. 9.

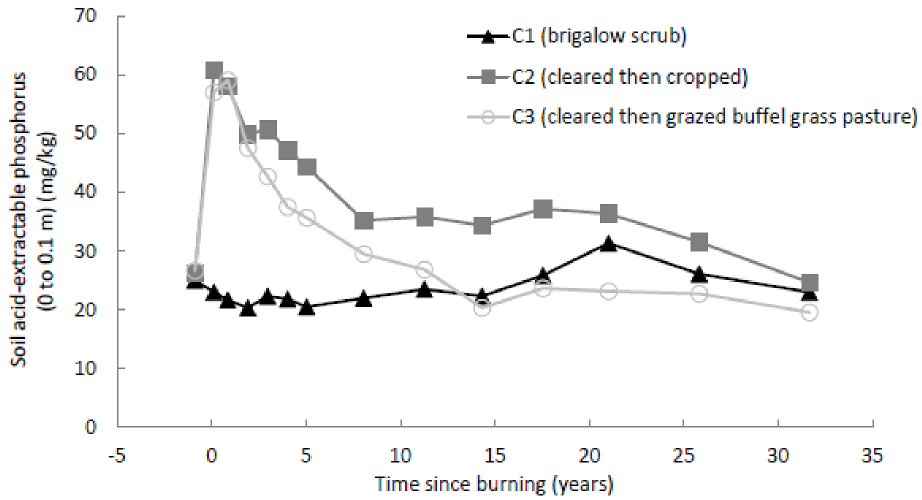
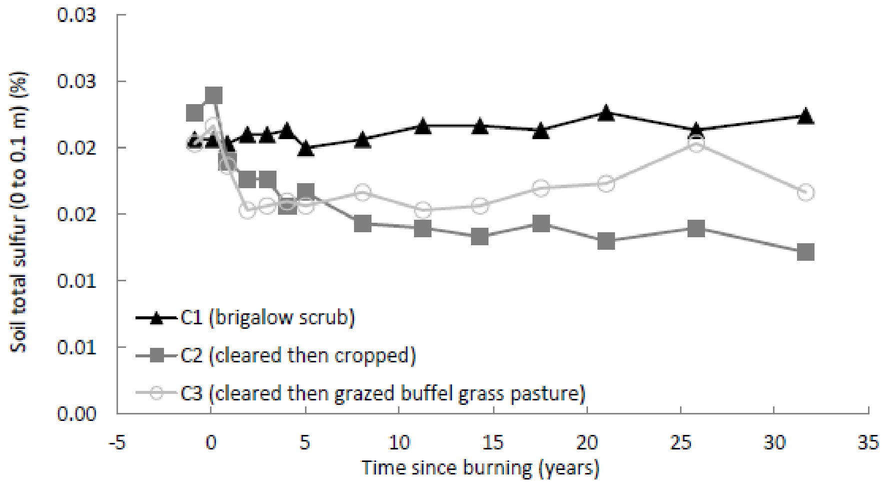

Soil acid-extractable phosphorus
(0 to 0.1 m) (mg/kg)
0
10
20
30
40
50
60
70
-5
0
5
10
15
20
25
30
35
Time since burning (years)
C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)

Fig. 10.



977

978 **Fig. 11.**

53

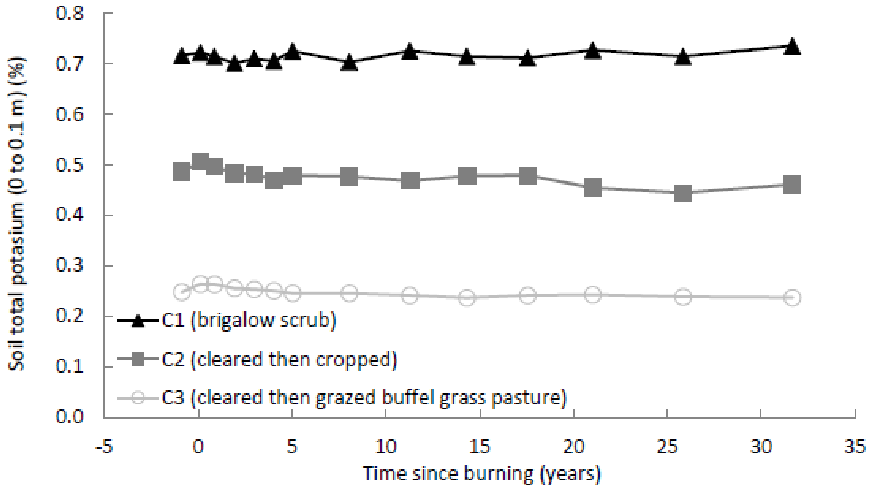

Soil total potassium (0 to 0.1 m) (%)
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
C1 (brigalow scrub)
C2 (cleared then cropped)
C3 (cleared then grazed buffel grass pasture)
-5
0
5
10
15
20
25
30
35
Time since burning (years)

Fig. 12.

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Appendix 1.4: Thornton and Yu (Unpublished)

The Brigalow Catchment Study: V*. A comparison of four methods to estimate peak runoff rate for small catchments before and after land use change in the Brigalow Belt bioregion of central Queensland, Australia

Running head

Estimating peak runoff rate in the Brigalow Belt

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Abstract

Estimation of peak runoff rate has been the focus of worldwide hydrological and soil erosion research. The results of these studies are intrinsically linked to the environment in which they were conducted, often limiting their applicability at alternative, potentially ungauged locations. This study evaluated the suitability of four simple methods to estimate peak runoff rate in the Brigalow Belt bioregion; (1) multiple regression models, (2) the scaling technique, (3) the Natural Resources Conservation Service curve number and graphical peak discharge method, and (4) the variable infiltration rate method. The performance of each method was assessed against data from the long-term Brigalow Catchment Study over a control period (1965–1982) monitoring virgin brigalow scrub prior to land use change, and a comparison period (1985–2004) when two of three catchment were converted for cropping and grazing respectively.

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The best estimates of peak runoff rate were obtained using multiple regression models (R^2 = 0.90; E = 0.63) or the scaling technique (R^2 = 0.90; E = 0.73). Good results were obtained using the variable infiltration rate method (R^2 = 0.88; E = 0.71). Estimations using the Natural Resources Conservation Service method gave an R^2 value of 0.85, however the Nash-Sutcliffe coefficient of efficiency was typically negative (E = -4.2) because the method systematically underestimated the peak runoff rate. Despite different data requirements and complexity, all four methods are easily applied with parameter values derived from widely available rainfall data, easily measured or estimated runoff volume data and basic physical descriptors of the catchment. The ability to simply estimate peak runoff rate addresses a current research priority for Queensland Government erosion modelling activities in Great Barrier Reef catchments.

Additional keywords: peak discharge; modelling; clearing; hydrology; *Acacia harpophylla*

Introduction

Estimation of runoff volume (Q_{tot}) and peak runoff rate (Q_p) for ungauged catchments has been the focus of substantial hydrological research worldwide (Dilshad and Peel 1994; Hawkins 1993; Post and Jakeman 1999). For example, the International Association of Hydrological Sciences initiative Predictions in Ungauged Basins devoted a decade (2003–2012) towards achieving major advances in the capacity to make hydrological predictions in ungauged basins (Hrachowitz et al. 2013; Sivapalan et al. 2003). Despite the investment of nearly 70 years of research in this field since the endeavours of Mockus (1949), limited availability of peak runoff rate data or, in the absence of data, models to estimate peak runoff rate, are still being identified as an impediment to soil erosion research (Silburn 2011).

Long-term monitoring at the Brigalow Catchment Study (BCS) in the Brigalow Belt bioregion of Queensland, Australia, has clearly demonstrated the increases in Q_{tot} and Q_p when virgin brigalow scrub is cleared for cropping or grazing (Thornton *et al.* 2007; Thornton and Yu 2016). As with all long-term data collection, equipment failure and subsequent periods of missing data were unavoidable and estimation techniques had to be employed. The objective of this study is to examine the suitability of four simple methods for the estimation of Q_p in three small (12–17 ha) catchments with land uses of virgin brigalow scrub, cropping, and grazing. The methods were (1) multiple regression analysis (Thornton and Yu 2016), (2) the scaling technique (Yu and Rose 1999), (3) the Natural Resources Conservation Service curve number and graphical peak discharge method (U.S. Department of Agriculture 1986; U.S. Department of Agriculture 2001), and (4) the variable infiltration rate method (Yu *et al.* 1997; Yu *et al.* 2001; Yu and Rosewell 1998).

Regression analysis is a basic statistical technique replete throughout hydrological literature (Beven 2000). Historically, it has been applied to data from similar paired catchment studies also located in semi-arid sub-tropical Queensland (Fentie *et al.* 2002; Freebairn *et al.* 2009). The scaling technique describes the relationship between runoff rate, total rainfall, and total runoff, similar to regression analysis. This was pertinent given that previous work at this site showed total rainfall to be the best predictor of Q_{tot}, while Q_{tot} was the best predictor of Q_p (Thornton 2012; Thornton and Yu 2016). The Natural Resources Conservation Service method is ubiquitous and a most enduring method for estimating the volumes and peak rate of surface runoff from ungauged catchments (Boughton 1989; Lyon *et al.* 2004). Infiltration modelling is an accepted alternative to the Natural Resources Conservation Service method for estimation of Q_p (Connolly 1998) and the variable infiltration rate method has proved to be the most suitable of eight methods for estimating runoff rates from grazing catchments in the nearby Nogoa subcatchment of the Fitzroy basin (Fentie *et al.* 2002). The performance of each method was assessed against observed peak runoff rates from the BCS using both graphical comparison and commonly used model performance indicators.

Evaluating the suitability of simple models for the estimation of Q_p at the small catchment scale, and by extension, in the wider 36.7 Mha of brigalow belt bioregion in Queensland and northern New South Wales, will be of direct benefit to hydrological modelling, providing a necessary hydrologic parameter for runoff driven soil erosion modelling in this landscape. The importance of this is highlighted by ongoing investments in modelling of erosion and water quality, particularly across the 42.4 Mha Great Barrier Reef Catchments, despite ongoing resourcing pressures limiting the commencement and continuation of long-term studies which underpin the models themselves (Great Barrier Reef Marine Park Authority 2014). Indeed the spatial derivation of Q_p has been identified as a research priority for the improvement of the eWater Source Catchment modelling framework (Carroll and Yu 2018), which is critical to the reporting framework of the Australian and Queensland governments' Reef 2050 Water Quality Improvement Plan 2017-2022 (The State of Queensland 2018).

Materials and methods

Site description

The BCS was established in 1965 to determine the impact on hydrology, productivity and resource condition when brigalow land is cleared for cropping or grazing. It is a paired, calibrated catchment study consisting of three contiguous catchments, identified by topographic survey. The areas of the catchments are 16.8 ha (catchment 1 – C1), 11.7 ha (catchment 2 – C2) and 12.7 ha (catchment 3 – C3). The catchments comprised good quality agricultural land, all equally suitable for cropping or grazing (Webb 1971). The BCS is located in central Queensland, Australia at 24.81°S, 149.80 °E using the Geocentric Datum of 1994 (Australian Government - Geoscience Australia 2006).

The BCS rationale, aims and history along with physical characteristics including location, experimental design, climate, vegetation and soils have been documented extensively (Cowie *et al.* 2007; Lawrence and Sinclair 1989; Radford *et al.* 2007; Silburn *et al.* 2009; Thornton *et al.* 2007; Thornton and Elledge 2016; Thornton and Shrestha 2017; Thornton and Yu 2016). Climate, land use and hydrological data used for this study have been collected as a part of the long-term BCS. A brief description of the site and experimental treatments follows.

Climate

The climate is semi-arid to subtropical with wet summers and low winter rainfall. Average maximum monthly temperature (1890 to 2004) for summer was 33.1 °C, while minimum temperature in winter averaged 6.5°C. Annual hydrological year rainfall during the study period (October 1965 to September 2004) ranged from 342 to 785 mm with an average of 646 mm. December, January and February had the highest average monthly rainfall (97 mm, 91 mm and 87 mm, respectively). Spring and summer rainfall (September to February) is characterised by high intensity, short duration storms with high temporal and spatial variability. Average annual potential evaporation at the nearby Bureau of Meteorology station 035149 was in excess of 2100 mm/yr during the study period. Average monthly evaporation exceeds average monthly rainfall in all months of the year (Thornton *et al.* 2007).

Soil types

Soil types in the catchments comprise associations of Black and Grey Vertosols, some with gilgais, Black and Grey Dermosols, and sub-dominant Black and Brown Sodosols (R.J. Tucker, pers. comm.) (Isbell 1996). Clay soils (Vertosols and Dermosols) occupy approximately 70% of C1 and C2, and 58% of C3. Sodosols occupy the remaining area in these catchments. Soils have a plant available water capacity ranging from 160 to 200 mm in the surface 1.4 m. Mean slope of the catchments is 2.5% (Cowie *et al.* 2007).

Vegetation

Before clearing, the study site was composed of three major vegetation communities, identified by their most common canopy species: brigalow (*Acacia harpophylla*), brigalow – belah (*Casuarina* *cristata*) and brigalow – Dawson Gum (*Eucalyptus cambageana*). Understories of all major communities are characterized by *Geijera* sp. either exclusively, or in association with *Eremophila* sp. or *Myoporum* sp. (Johnson 2004). Projected canopy cover ranges from zero in non-vegetated areas to 100% in treed areas. Litter levels (both leaf and wood) range from 1.9 t/ha in non-vegetated areas to 29 t/ha in treed areas .(Dowling *et al.* 1986)

Site history and management

The study has been divided into three distinct experimental stages (Table 1) (Thornton *et al.* 2010). Stage I commenced in 1965 with the three catchments retained in their virgin state for calibration purposes. Rainfall and runoff data were collected to describe differences in catchment hydrological responses to a range of weather sequences.

Stage II commenced in March 1982 with C2 and C3 cleared by bulldozer and chain. The fallen timber was burnt *in-situ* in October 1982. Residual unburnt timber in C2 was raked to the contour line and burnt. Narrow based contour banks at 1.5 m vertical spacing were constructed and a grassed waterway later established. In C3, unburnt timber was left in place, and in November 1982 the catchment was sown by throwing buffel grass seed (*Cenchrus ciliaris* cv. Biloela) on the soil surface.

During Stage III, comparison of the effect of land use change commenced with cropping in C2 and grazing in C3. Sorghum was planted in C2 in September 1984 followed by nine annual wheat crops commencing in 1985. Fallow management in this period was entirely mechanical tillage. A minimum tillage and opportunity cropping philosophy was adopted in the early 1990s and has continued with either a summer (sorghum) or winter (wheat) crop sown whenever soil moisture was adequate. Grazing in C3 commenced in December 1983. Stocking rate varied between 0.29 and 0.71 head/ha (each beast typically 0.8 adult equivalents), adjusted to maintain pasture dry matter levels greater than 1000 kg/ha. There was no feed supplementation.

Rainfall and runoff data

Rainfall and runoff data were analysed on an event basis. A rainfall event was defined as one or more wet days separated from other events by at least one dry day. Only rainfall events that produced runoff were considered in this study. Rainfall and runoff observations for the BCS are presented in Thornton *et al.* (2007) while peak runoff rate observations are presented in Thornton and Yu (2016).

Rainfall data used in this study were collected from a 0.5 mm tipping bucket recorder located at the head point of the catchments (Figure 1). Raw data were stored and manipulated using the Hydstra database (Kisters 2014). Where data were aggregated, 15 minute totals commenced from midnight while daily totals were the previous 24 hours to 9 am. Rainfall intensity (I_x) was calculated as the peak intensity over x minutes within the event. Antecedent rainfall (A_x) was calculated as the sum of daily rainfall totals over x number of days until 9 am on the day the event commenced.

Storm energy (E) was not measured at this site. The technique of Rosewell (1986) was used to estimate the total storm energy from observed tipping bucket rainfall intensity data. Storm erosivity (EI_{30}) was calculated as the product of storm energy and peak 30 minute rainfall intensity (Yu and Rosewell 1998).

Each catchment was instrumented to measure runoff using a 1.2 m steel HL flume with a 3.9 m by 6.1 m concrete approach box (Brakenseik *et al.* 1979) located at the outlet point of each catchment (Figure 1). Water height through the flumes was recorded with mechanical float recorders. As for rainfall data, raw runoff data were stored and manipulated using the Hydstra database (Kisters 2014). Observed stage height data (m) were converted to runoff depth (mm) and flow rate (mm/hr), eliminating the effect of catchment size. Peak runoff rate was calculated on an event basis from the observed instantaneous peak height.

Methods to estimate peak runoff rate

1) Multiple regression models

Thornton and Yu (2016) developed linear multiple regression models to estimate Q_p for each catchment and stage using local climate and catchment condition data. All regression models considered the parameters total runoff (Q_{tot}), total rainfall (P), storm energy (E), storm erosivity (EI_{30}), peak rainfall intensity (I), antecedent rainfall (A) and total soil water (TSW). Each parameter was tested individually for a significant correlation (P <0.05) with dependent parameter Q_p. Significant parameters were then combined and an all-subsets regression performed using the statistical software program GenStat v14.1 (VSN International 2011) . The final models only included significant constants and coefficients. To allow numerical evaluation of Q_p regression models, a split sample approach was used. The models were developed on data collected in odd years and then validated on data collected in even years. The models for each of the catchments in Stage I and III of the study are given in Table 2.

2) The scaling technique for estimating peak runoff rate

The scaling technique relates peak runoff rate to rainfall, runoff volume and peak rainfall intensity as follows:

$$Q_p = \alpha_p \times \frac{Q_{tot}}{P_{tot}} \times I_x \qquad (1)$$

where Q_p is the peak runoff rate (mm/hr), Q_{tot} is total runoff volume (mm), P is total rainfall (mm), I_x is rainfall intensity for a given time interval x and α_p is a dimensionless scaling parameter (Yu and Rose 1999). As rainfall intensity data for the site is available on a number of time intervals, simple calibration of α_p, the scaling parameter, was undertaken to determine the best estimate of α_p given peak rainfall intensity during 6, 10, 15, 20, and 30 minute and 1, 2, 3, 4, 6, 12, 18, 24 hr intervals.

3) The Natural Resources Conservation Service methodology

The Natural Resources Conservation Service (NRCS) (formerly the Soil Conservation Service) curve number (*CN*) method (U.S. Department of Agriculture 2001) provides an estimate of Q_{tot} which is then used with the Graphical Peak Discharge (GPD) method (U.S. Department of Agriculture 1986) to estimate Q_p. As measured Q_{tot} was available, it was not necessary to use the *CN* method to estimate it. However, as local determination of *CN* values was always the intention of the method (Van Mullem *et al.* 2002), *CN* values for each catchment by experimental stage were calculated from pairs of P:Q_{tot} observations for a single storm. These *CN* values allow this method o be applied where measured Q_{tot} is not available.

The following equation describes the rainfall-runoff relationship used in the *CN* method (U.S. Department of Agriculture 2001):

$$Q_{tot} = \frac{(P - I_a)^2}{(P - I_a) + S} \text{ if } P > I_a \text{ and } Q_{tot} = 0 \text{ if } P < I_a \qquad (2)$$

where Q_{tot} is runoff, P is rainfall, I_a is an initial abstraction or retention parameter (rainfall that does not run off) and S is a site index defined as the maximum detention, or the maximum possible difference between P and Q_{tot} as P approaches infinity. P, Q_{tot}, I_a, S are measured in inches.

Historical field data gave the empirical relationship:

$$I_a = 0.2S \qquad (3)$$

Substituting (3) into (2) gives what is commonly termed the familiar equation:

$$Q_{tot} = \frac{(P-0.2S)^2}{P+0.8S} \qquad (4)$$

The retention parameter *S* is related to a curve number (*CN*) as follows:

$$S = \frac{1000}{CN} - 10 \qquad (5)$$

where S is measured in inches. *CN* equals 100 when S = 0, and *CN* approaches to 0, as S goes to infinity. A *CN* is calculated by solving equation 4 for *S* (equation 6 below) and equation 5 for *CN* (Boughton 1989; Hawkins 1973; Hawkins 1993):

$$S = 5[P + 2Q_{tot} - (4Q_{tot}^{2} + 5PQ_{tot})^{1/2}] \qquad (6)$$

The observation that rainfall events of similar magnitude generate varying amounts of runoff demonstrates that *CN* varies from event to event (U.S. Department of Agriculture 2001). The original *CN* method stated that antecedent moisture condition (AMC) was the most significant variable explaining this variation (Van Mullem *et al.* 2002). The NRCS classification of AMC is given in Table 3 (Boughton 1989; Chow *et al.* 1988; Dilshad and Peel 1994). As the BCS is dominated by perennial vegetation and opportunity cropping, the AMC grouping for growing season was appropriate. To make the *CN* values calculated using equations 5 and 6 widely applicable, some method of optimisation to account for AMC must be undertaken and an average set of *CN* values produced (Boughton 1989).

CN values for AMC I (*CN*(I)) and AMC III (*CN*(III)) conditions can be calculated from a *CN* value for AMC II (*CN*(II)) using equations 7 and 8 (Chow *et al.* 1988):

$$CN(I) = \frac{4.2 \times CN(II)}{10 - 0.058 \times CN(II)} \tag{7}$$

$$CN(III) = \frac{23 \times CN(II)}{10 + 0.13 \times CN(II)} \tag{8}$$

If *CN* values are calculated for enough events using pairs of P:Q_{tot} observations, statistically the mean of the calculated *CN* values should be a reasonable estimation of the true mean *CN* value for the catchment. This mean *CN* value can then be considered as *CN*(II), and *CN*(I) and (III) can be calculated using equations 7 and 8. If the number of events is small, an alternative approach is to assume that each calculated *CN* value is *CN*(II), and calculate *CN*(I) and (III) for each event. Each group of *CN* values can then be averaged to obtain *CN* values for AMC (I), (II) and (III). A simpler approach is to group the calculated event *CN* values into AMC groups depending on the observed antecedent rainfall. The groups of *CN* values can then be averaged to obtain AMC (I), (II) and (III). The performance of *CN* values optimised by each method was assessed by using the NRCS *CN* method to estimate runoff volume for each event and comparing the estimate to the observed value.

As this study had measured values of Q_{tot}, the *CN* method was not required and only the GPD method to estimate Q_p was used. The GPD method was developed from hydrograph analyses with *TR-20 Computer Program for Project Formulation – Hydrology* (U.S. Department of Agriculture 1983; U.S. Department of Agriculture 1986; Ward 1995). The equation for calculating peak discharge is:

$$Q_p = q_u A Q_{tot} F \tag{9}$$

where Q_p is peak discharge (cubic feet per second, cfs), q_u is unit peak discharge (cfs per square mile per inch of runoff, csm/in) (see equations 10 to 12), A is drainage area (mi^2), Q_{tot} is total runoff volume (inches) and F is an adjustment factor for ponds and swamps.

Unit peak discharge (q_u) for use in equation 9 requires an estimation of the time of concentration (t_c) for the catchment. Time of concentration can be estimated by a number of methods including the NRCS lag method. As this method has been shown to have one of the lowest biases (Ward 1995), it was used exclusively for estimation of t_c. The NRCS lag equation is:

$$t_l = \frac{L^{0.8}(S+1)^{0.7}}{1900\,Y^{0.5}} \qquad (10)$$

where t_l is lag time (hr), L is the hydraulic length of the catchment (ft), S is a function of the NRCS *CN* method (equations 2 to 5) and Y is the average land slope (%) (Ward 1995). Lag time is related to t_c as follows (Ward 1995):

$$t_l = 0.6t_c \qquad (11)$$

Having estimated t_c using equations 10 and 11, estimation of q_u was undertaken using the United States Department of Agriculture Natural Resource Conservation Service (1986) equation-based method for a Type II rainfall distribution. This distribution represents regions in which high rates of runoff from small areas are usually generated from summer thunderstorms (U.S. Department of Agriculture 1973), which was applicable to the study site.

The equation for estimating q_u is:

$$\log(q_u) = C_0 + C_1 \log(t_c) + C_2[\log(t_c)]^2 \quad (12)$$

where q_u is unit peak discharge (csm/in), t_c is time of concentration (equations 10 and 11) and C_0, C_1 and C_2 are coefficients chosen from lookup tables depending on the rainfall distribution and ratio of I_a / P (from equations 2 to 5) (U.S. Department of Agriculture 1986). The coefficients are given in Table 4.

4) The variable infiltration rate method for estimating peak runoff rate

From first principles the variable infiltration rate (VIR) method assumes runoff is equal to rainfall minus abstraction (which can be considered to include infiltration, surface storage, interception and evapotranspiration) (Connolly *et al.* 1997; Thornton *et al.* 2007). If it is assumed that at the commencement of runoff, surface storage, interception losses and evapotranspiration are negligible, runoff rate (Q_i) (mm/hr) can be estimated as rainfall rate (P_i) (mm/hr) less infiltration rate (f_i) (mm/hr) for a given time interval (Yu *et al.* 1998). This can be written as:

$$Q_i = P_i - f_i \quad (13)$$

The unknown infiltration rate f_i is constrained by two limitations as follows (Yu *et al.* 1998):

$$\sum_{i=1}^{n}(P_i - f_i)\Delta t = Q_{tot} \quad (14)$$

and

$$f_i \leq P_i \quad (15)$$

where Q_{tot} is the total runoff volume (mm) for the event, Δt is the time interval at which rainfall rate is measured and $n\Delta t$ is the duration of the runoff event.

Maximum infiltration rate has been shown to vary spatially across the landscape (Yu 1997; Yu *et al.* 1998; Yu *et al.* 1997). Yu *et al.* (1997) and Yu *et al.* (1998) accounted for this variability, describing the spatial variation in maximum infiltration rate with an exponential distribution, with the actual rate of infiltration given by:

$$f_i = I(1 - e^{P_i/I}) \qquad (16)$$

where I is interpreted as a spatially-average maximum infiltration rate. To determine I, equation 16 can be substituted into equation 14 as follows:

$$\sum_{i=1}^{n}[P_i - I(1 - e^{P_i/I})]\Delta t - Q_{tot} = 0 \qquad (17)$$

and equation 17 solved numerically when both rainfall rate (P_i) and total runoff volume (Q_{tot}) are known (Yu *et al.* 1998). Equation 17 presents a root-finding problem which can be solved by numerical methods, of which the most suitable for this purpose is Brent's method (Press *et al.* 1989). Brent's method combines root bracketing, bisection and inverse quadratic interpolation (Brent 1973; Press *et al.* 1989), guaranteeing a unique solution for I, the spatially–averaged maximum infiltration rate from which Q_p is calculated (Yu 1997).

Once I is known, peak rate of rainfall excess, R_p, can be evaluated as follows:

$$R_p = P_p - I(1 - e^{-\frac{P_p}{I}}) \qquad (18)$$

where *Pp* is the peak rainfall intensity. R_p is an approximation of Q_p for small areas where time lag can be ignored. For large areas, the literature shows that VIR estimations of runoff rate can be routed to a catchment outlet using a linear approximation to a kinematic wave, assuming a constant lag time between rainfall excess and runoff (Yu 1999; Yu *et al.* 1997; Yu *et al.* 2000b). The routing equation is written:

$$Q_i = \alpha Q_{i-1} + (1-\alpha)R_i \qquad (19)$$

where Q_i is the estimated runoff rate at the catchment outlet and R_i is the rainfall excess rate. The parameter α is related to the lag time of runoff within the catchment (t_l , equation 11) and the time interval of measurement (Δt), and is given as (Yu *et al.* 1997):

$$\alpha = \frac{t_l}{t_l + \Delta t} \qquad (20)$$

This study will use the software program Generation Of Synthetic Hydrograph (GOSH) (Yu 1997) to solve equation 17 and hence Q_p. GOSH uses Brent's method to solve equation 17 given known rainfall rates and Q_{tot}. GOSH outputs include both *I* and Q_p.

Assessment of method performance

Method performance was assessed against observed runoff data using several criteria, similar to the approaches of Refsgaard and Knudsen (1996), Lørup *et al.* (1998) and Legates and McCabe Jr (1999). Graphical comparison comprised overlay plots of simulated and observed Q_p data. Numerical evaluation compared R^2 and E (Nash and Sutcliffe 1970) between observed and estimated Q_p data. All R^2 presented are adjusted R^2. Adjusted R^2 has the advantage over statistic r^2 in that it takes account of the number of parameters that have been fitted in the model (VSN International 2011). As Q_p was not normally distributed, log transformation $\log(Q_p + 1)$ was performed to observed and estimated data to allow for valid statistical testing.

The coefficient of efficiency (E) expresses the proportion of variance of the observed data which can be accounted for directly by the estimated data as follows (Nash and Sutcliffe 1970):

$$\mathrm{E} = 1 - \frac{\Sigma(Q_{Obs} - Q_{Est})^2}{\Sigma(Q_{Obs} - \bar{Q}_{Obs})^2} \quad (21)$$

where Q_{Obs} is the observed peak runoff rate, Q_{Est} is the estimated peak runoff rate and $\bar{Q}_{Obs}$ is the average observed peak runoff rate. This is a better indicator of model performance than statistic R^2, which has been shown to be insensitive to additive and proportional differences between observed and estimated values (Legates and McCabe Jr 1999). Values of E range from $-\infty$ to 1. An E value of 1 means perfect agreement between the observed and estimated data; an E value of 0 means that the modelled estimate is no better predictor than a value equal to the observed mean; and a negative E value means that the modelled estimate is a worse predictor than an estimation made using the mean of the observed data (Chiew and McMahon 1993; Legates and McCabe Jr 1999; Yu *et al.* 2000a; Yu *et al.* 2000b).

Results

Estimations of peak runoff rate using multiple regression models

Regression models of Q_p during Stage I (Table 2) provide good estimations of both the development and validation data (Figure 2). Little bias is evident despite the wide range of observed Q_p data. However, Catchment 2 regressions yielded poor results for very small observed Q_p values. Where observed Q_p values less than 0.1 mm/hr were used as input data to develop the regression models and validated against observed Q_p values less than 0.4 mm/hr, the regressions gave negative results (data not shown). Regression models of Stage III data (Table 2) also provide good estimations of both the development and validation data however events with Q_p greater than 1 mm/hr were better estimated than events with Q_p less than 1 mm/hr (Figure 3).

Simple optimisation of the scaling technique parameters

During Stage I the best estimates of α_p for all catchments (highest E values) were obtained using peak one hour rainfall intensity measurements. During Stage III, the best estimates of α_p for C1 and C3 were obtained using peak six hour and two hour rainfall intensity measurements respectively. The best estimates of α_p based on pairs of P:Q_{tot} observations are given in Table 5.

Estimations of peak runoff rate using the scaling technique

During Stage I the scaling technique gave good estimations of Q_p from C1 and C2 however the method typically underestimated Q_p from C3 where observed Q_p data was less than 1 mm/hr (Figure 4). During Stage III the scaling technique gave good estimations of Q_p from C1. Catchment 2 showed wide scatter in estimations across the range of observed Q_p data. Estimates from C3 continued to be poor where observed Q_p data was less than 1 mm/hr (Figure 5).

Calculation of Curve Numbers to estimate runoff volume prior to the estimation of peak runoff rate

The average *CN* calculated from pairs of observed Stage I P:Q_{tot} data was *CN* 58 for all catchments. Average *CN* decreased to *CN* 53 for C1 in Stage III, however *CN* increased for both C2 and C3 to *CN* 67 and *CN* 64, respectively (Table 6). Observed peak runoff rates showed that during Stage III, C3 had proportionally more small events than the other catchments. If this bias is eliminated by removing all events where Q_{tot} <1 mm, the average calculated *CN* for both C2 and C3 in Stage III is *CN* 67.

CN values were optimised using both the equation based method (equations 7 and 8) and by averaging the calculated *CN* values for individual events grouped according to AMC condition. With the equation based methods there was little difference in *CN*(I) and (III) values obtained whether the equations were applied to the average of the calculated *CN* values for individual events, or applied to each calculated *CN* value and then averaged. The difference in CN values between the methods was a maximum of three for *CN*(I) values and one for *CN*(III) values. Optimising *CN* values using observed AMC resulted in *CN*(I) and (II) values higher than, and *CN*(III) values typically lower than, those given by the equation based methods. In all instances *CN* values optimised using the observed AMC condition provided the best estimate of Q_{tot} (Table 6).

Estimations of peak runoff rate using the graphical peak discharge method

The GPD method gave good estimations of Q_p across all catchments in Stage I and III however more scatter is evident in Stage III estimations (Figures 6 and 7).The method typically under-estimates Q_p in small events and over-estimates Q_p in large events. For Stage I events where observed Q_p data was greater than 5 mm/hr, 83% of estimated Q_p values were greater than the observations. This decreased for Stage III events where observed Q_p data was greater than 5 mm/hr, when only 56% of estimations were greater than the observations.

Estimations of peak runoff rate using the variable infiltration rate method

On average, the VIR method with no routing component over-estimated Q_p for 88% of events, with the time of peak occurring prior to the observed peak in 92% of events. In all cases, routing of VIR estimated runoff resulted in Q_p equal to, or smaller than, the non-routed estimations. During Stage I the routed VIR method gave good estimations of Q_p from C1 and C2 however the method typically underestimates Q_p from C3 where observed Q_p data was less than 1 mm/hr (Figure 8). During Stage III the method gave good estimations of Q_p from all catchments; however, for C2 and C3, events with Q_p greater than 1 mm/hr were better estimated than events with Q_p less than 1 mm/hr (Figure 9). Routing typically delayed the estimated peak, with an average of 97% of Stage I peaks and 100% of Stage III peaks occurring after the estimated non-routed peak. However, the delay was not long enough and on average 91% of routed peaks occurred prior to the observed peak.

Quantitative assessment of method performance

Numerical evaluation criteria R^2 and E calculated using observed and estimated Q_p data for all methods is shown in Table 7. Values of R^2 were greater than 0.9 for all methods in Stage I and greater than 0.8 for all methods in Stage III. When averaged across all catchments, the scaling technique had the highest R^2 and E for Stage I, whilst the regression models had the highest R^2 and E for Stage III. When averaged across all catchments and stages, regression models and the scaling technique had the equal highest R^2 whilst the scaling technique had the highest E.

Using a split sample approach, regression models of Q_p developed on data collected in odd years were validated against Q_p data collected in even years. During Stage I, regression models gave an R^2 of 0.89 or greater for all catchments. Catchment 1 had the lowest E of 0.35 while C2 and C3 had substantially higher E of 0.64 and 0.59 respectively. There was little change in R^2 in Stage III, with R^2 of 0.87 or greater for all catchments; however, E values improved to 0.67 or greater for all catchments.

Regression analysis of GPD estimated Q_p against observed Q_p gave R^2 greater than 0.73 in all instances. These high R^2 values disguise the tendency of the method to under-estimate Q_p in small events and over-estimate Q_p in large events. This is evident in the negative E values for all catchments in Stage I, and in C2 and C3 in Stage III. The GPD method consistently gave the lowest R^2 and E of all four methods.

Regression analysis of non-routed VIR estimated Q_p against observed Q_p showed strong correlations with R^2 greater than 0.7 in all instances; however, the tendency of the method to over-estimate Q_p resulted in low and negative E values. With the addition of routing, improved R^2 were obtained for all catchments, with R^2 greater than 0.9 in Stage I and greater than 0.8 in Stage III. As the routed method did not suffer the gross over-estimation of Q_p that the non-routed method exhibited, all values of E were greatly improved. Despite typical R^2 and E values greater than 0.8, the method gave poor estimations of C3 in Stage III, with an E value of 0.11

When averaged across all catchments and stages, the scaling technique was the best performing method when evaluated using both R^2 and E. While R^2 and E decreased slightly between Stage I and Stage III for C1 and C3, and E of 0.25 for C2 in Stage III was a marked decrease.

Different input variables are required for the different methods of Q_p estimation (Table 8). If local calibration is not required or if insufficient data is available to do so, the method with the lowest data requirement are the multiple regression models, which can be applied in alternative locations as a single parameter model, only requiring an estimate of Q_{tot}. Ranking of the methods from the lowest data requirement to the highest data requirement gives multiple regression models < scaling technique < VIR method < NRCS methodology.

Discussion

Comments on the NRCS-CN method for estimating runoff volume

The best agreement between observed and estimated runoff volume using the NRCS *CN* method was obtained using *CN* values that were the average of *CN* values calculated from pairs of $P{:}Q_{tot}$ data grouped according to AMC. As daily rainfall data for Australia is widely available via tools such as SILO (Queensland Government 2015), assigning an AMC condition to a calculated *CN* value based on the NRCS classification of AMC (Table 3) is straightforward. Substantial improvement in runoff volume estimations were obtained by this method compared to using average *CN* values optimised for AMC by the use of formula.

Average overall and AMC II optimised *CN* values (57 and 63 respectively) calculated for brigalow forest agree with those initially reported by Boughton (1989), who analysed the first three years of this dataset. Boughton (1989) also cites unpublished data for a further 10 years of record and reports optimised *CN* values of 73, 71 and 70 for C1, C2 and C3, respectively. The AMC II optimised *CN* value of 81 calculated for cropping is within the range reported by Freebairn and Boughton (1981) for cracking clays in southern Queensland. The AMC II optimised *CN* value of 67 calculated for grazing is greater than the range reported by Cao et al. (2011) for pasture and grazing treatments on predominantly medium and heavy clay soils throughout New South Wales; however, AMC III optimised *CN* value of 77 calculated for grazing was within the reported range.

Average overall and AMC II calculated *CN* values for agricultural land uses are lower than those suggested by the NRCS *CN* tables. Assuming a hydrological soil group of B or C (moderate and low infiltration rates respectively when thoroughly wetted; moderately fine to moderately course textures and moderately fine to fine textures respectively; moderate and low rates of water transmission respectively), NRCS suggested *CN* values for cropping are 83 and 88 for fallows with residual stubble, 75 and 82 for straight rowed crops with residual stubble and 74 and 81 for contoured crops with residual stubble. Suggested *CN* values for continuously grazed pasture with >75 % cover are *CN* value 61 and 74 for hydrological soil groups B and C respectively, which are closer to those calculated in this study than the suggested *CN* values for cropping. The calculated *CN* value for brigalow scrub is similar to the suggested *CN* value of 55 for woodland on hydrological soil group B, and less than the suggested *CN* value of 70 for woodland on hydrological soil group C.

Comparing the performance of the four estimation methods

This study has shown that regression models, the VIR method and the scaling technique all produce acceptable estimations of Q_p when compared using both graphical and numerical assessments of method performance. Numerical assessment of method performance across all catchments and stages using R^2 indicated that the site-specific multiple regression models and the scaling technique gave the best estimation of Q_p, followed by the VIR and the NRCS method. Assessment of method performance using E indicated that the scaling technique continued to give the best estimation of Q_p, followed by the VIR method, multiple regression models and the NRCS method. This assessment clearly indicates that the multiple regression models and scaling technique give the best estimations of Q_p, however the choice of which method is best employed can also be influenced by external factors such as data requirements.

Typically all methods gave better estimations during Stage I of the study. This is likely due to the smaller variability in catchment hydrology when all catchments contained virgin brigalow scrub compared to their changed dynamics when converted to land uses of cropping or grazing (Thornton *et al.* 2007; Thornton and Yu 2016). With the exception of the regression models, events where observed Q_p data was less than 1 mm/hr were most difficult to estimate, with C3 in Stage I consistently underestimated. This is not necessarily reflected in the E values, particularly for the VIR method and scaling technique, however this is likely explained by the fact that as a numerical indicator, E tends to overemphasize the matching of high flow values at the expense of low flow values (Krause *et al.* 2005; Patil and Stieglitz 2014; Patil *et al.* 2014).

Each of the methods has different data and computational requirements. Common to each method is the requirement for an estimation of Q_{tot}. If Q_{tot} is unknown, runoff volume will have to be estimated separately. The *CN* values calculated in this study provide a basis for doing so in other semi-arid subtropical catchments. Regression models could also be developed, however regression models of Q_{tot} for this site, obtained with the same methodology used to develop the regression models in this study, gave poorer results than regression models of Q_p (Thornton and Yu 2016). Daily time step hydrological modelling at this site has yielded better estimates of Q_{tot} than either regression modelling or the *CN* method (Thornton *et al.* 2007).

It is not surprising that multiple regression models, the VIR method and the scaling technique all generate good estimates of Q_p given that they all capture relationships between observed rainfall and runoff data. Given that rainfall is the primary driving mechanism controlling watershed runoff (Fernandez and Garbrecht 1994) and that total rainfall was the best single-estimator of Q_{tot} in regression models at this site (Thornton and Yu 2016), the regression models of Qp inherently capture the dynamic between rainfall, runoff and peak discharge. This dynamic is directly captured in the variables of the VIR method and scaling technique, whereas the *CN* method relies on a general empirical relationship.

Unlike regression models and the scaling technique, both NRCS and VIR methods require some physical knowledge of the catchment to estimate lags and time of concentration. Information such as slope, hydraulic length and ponded area are all simple parameters likely to be easily determined and should not preclude the use of either method. Examination of contour mapping should provide the basic physical catchment characteristics required.

All methods require rainfall data. Easily obtained rainfall total data is necessary for both the NRCS method, the VIR method and the scaling technique and improves estimations from some regression models. Rainfall data at a sub-daily timescale is not required for the NRCS method, but adds value to some regression models, allowing for calculation of parameters such as E and EI_{30}. It is essential for the VIR method and scaling technique. As for daily rainfall data it is relatively simple to obtain sub-daily data in formats such as the six-minute rainfall data, which is available on request from the Australian Bureau of Meteorology (Bureau of Meteorology 2016).

All of these methods have simple computational requirements. With an estimate of Q_{tot}, it is possible to estimate Q_p by hand using regression models. If no local calibration is undertaken the scaling technique is also able to be undertaken by hand. The NRCS method is only marginally more complicated and with the assistance of tables of coefficients, may also be performed by hand. By modern computing standards the computational requirements of the VIR method program are very basic. Compilation of the input files for the program is easily performed by simple spreadsheet packages, which are also of assistance in performing routing calculations.

Whilst it is clear that the NRCS method is the least suitable for the estimation of Q_p none of the other three methods should be excluded on the basis of performance. The correct method is likely to be the one that the user is able to meet the data requirements for and has the skills to implement. If a user was implementing the regression models in a dissimilar geographical region a simple check on the validity of the output may be obtained by use of one of the other methods in parallel, particularly in an ungauged catchment with no data available to undertake model validation.

Conclusions

The aim of this study was to evaluate the suitability of four simple methods to estimate peak runoff rate in small (12–17 ha) catchments with land uses of virgin brigalow scrub, cropping or grazing in the semi-arid subtropical brigalow (*Acacia harpopylla*) region of central Queensland, Australia. The four methods were (1) multiple regression models, (2) the scaling technique, (3) the Natural Resources Conservation Service curve number and graphical peak discharge method, and (4) the variable infiltration rate method. Of the four methods evaluated, the best estimations of peak runoff rate were obtained using either multiple regression models or the scaling technique. Good results were also obtained using the VIR method of estimating peak runoff rate however the computational requirements of this method were greater than that needed to use multiple regression models or the scaling technique. Estimations of peak runoff rate using the Natural Resources Conservation Service method gave good R^2 however Nash-Sutcliffe method efficiencies were typically negative, rendering the method unsuitable for use at this scale in this region. None of the four methods should be excluded on the basis of data requirements. Parameterisation is a simple task for all methods, utilising widely available rainfall data, easily measured or estimated runoff volume data and basic physical descriptors of the catchment.

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Tables

Table 1. The land use history of the three catchments of the Brigalow Catchment Study.

Catchment	Area (ha)	Land use by experimental stage		
		Stage I (Jan 1965–Mar 1982)	Stage II (Mar 1982–Sep 1984)	Stage III (Sep 1984–Dec 2004)
C1	16.8	Virgin brigalow scrub	Virgin brigalow scrub	Virgin brigalow scrub
C2	11.7	Virgin brigalow scrub	Development	Cropping
C3	12.7	Virgin brigalow scrub	Development	Improved pasture

Table 2. Multiple regression models for the estimation of peak runoff rate from the three catchments of the Brigalow Catchment Study. Log Q_p is log transformed (log (x+1) peak runoff rate, log Q_{tot} is log transformed (log (x+1) total runoff, P is total rainfall, E is storm energy, $A_{2\ day}$ is antecedent rainfall in the two days prior to the event and EI_{30} is storm erosivity (Thornton and Yu, 2016).

Stage	Catchment	Land use	Regression model of peak runoff rate ($logQ_p$)	R^2
Stage I	C1	Brigalow scrub	$0.524 \times logQ_{tot}$	0.82
	C2	Brigalow scrub	$0.8483 \times logQ_{tot} - 0.0188 \times P + 0.0787 \times E$	0.96
	C3	Brigalow scrub	$0.5767 \times logQ_{tot} + 0.0122 \times E + 0.0073 \times A_{2\ day}$	0.94
Stage III	C1	Brigalow scrub	$0.6767 \times logQ_{tot}$	0.82
	C2	Cropping	$0.815 \times logQ_{tot} - 0.0238 \times P + 0.1096 \times E$	0.75
	C3	Improved pasture	$0.466 \times logQ_{tot} + 0.0006 \times EI_{30}$	0.92

Table 3. Antecedent moisture condition classification based on 5-day antecedent rainfall.

AMC condition	5–day antecedent rainfall (mm)	
	Dormant season	Growing Season
I	<13	<36
II	13–28	36–53
III	>28	>53

Table 4. Values of the coefficients required to estimate unit peak discharge (q_u) using equation 12. Coefficients are chosen depending on the ratio I_a / *P* and rainfall distribution type. If I_a / *P* is outside of the given range, then the boundary value should be used. Linear interpolation is used between the given values (U.S. Department of Agriculture, 1986).

Rainfall distribution type	I_a/P	C_o	C_1	C_2
I	0.10	2.30550	-0.51429	-0.11750
	0.20	2.23537	-0.50387	-0.08929
	0.25	2.18219	-0.48488	-0.06589
	0.30	2.10624	-0.45695	-0.02835
	0.35	2.00303	-0.40769	0.01983
	0.40	1.87733	-0.32274	0.05754
	0.45	1.76312	-0.15644	0.00453
	0.50	1.67889	-0.06930	0.0
Ia	0.10	2.03250	-0.31583	-0.13748
	0.20	1.91978	-0.28215	-0.07020
	0.25	1.83842	-0.25543	-0.02597
	0.30	1.72657	-0.19826	0.02633
	0.50	1.63417	-0.09100	0.0
II	0.10	2.55323	-0.61512	-0.16403
	0.30	2.46532	-0.62257	-0.11657
	0.35	2.41896	-0.61594	-0.08820
	0.40	2.36409	-0.59857	-0.05621
	0.45	2.29238	-0.57005	-0.02281
	0.50	2.20282	-0.51599	-0.01259
III	0.10	2.47317	-0.51848	-0.17083
	0.30	2.39628	-0.51202	-0.13245
	0.35	2.35477	-0.49735	-0.11985
	0.40	2.30726	-0.46541	-0.11094
	0.45	2.24876	-0.41314	-0.11508
	0.50	2.17772	-0.36803	-0.09525

Table 5. The optimised intensity intervals and α_p values determined from observed rainfall total, rainfall intensity and runoff data.

Catchment	Stage	Intensity interval	α_p
1	I	1 hr	1.123
	III	6 hr	4.466
2	I	1 hr	1.024
	III	1 hr	1.383
3	I	1 hr	1.104
	III	2 hr	1.271

Table 6. CN values calculated from pairs of P:Q_{tot} observations (presented both as an overall average and as an average of CN values grouped according to AMC condition) and evaluation of their suitability for estimating total runoff.

						R^2 log Q(obs) v log Q(est)	
Catchment	Stage	Average CN[1]	CN(II)[2]	CN(I)[2]	CN(III)[2]	Using average CN[1]	Using AMC Grouped CN[2]
1	I	58	61	58	69	0.53	0.53
	III	53	68	53	55	0.54	0.55
2	I	58	59	55	78	0.6	0.65
	III	67	81	65	71	0.51	0.54
3	I	58	62	56	71	0.58	0.64
	III	64	67	61	77	0.2	0.23

[1]Calculated on an event basis using the method of Hawkins (1993) and averaged across all events

[2]Calculated on an event basis using the method of Hawkins (1993) and averaged across all events grouped according to AMC condition (Table 3)

Table 7. Comparison of method performance based on the numerical indicators R^2 and *E*.

		Regression models		Scaling technique		NRCS method		VIR method	
		E	R^2	E	R^2	E	R^2	E	R^2
Catchment	Stage	(R^2 based on log-transformed data; E based on normal data)							
1	I	0.35	0.90	0.97	0.95	-0.75	0.92	0.90	0.91
	III	0.67	0.93	0.77	0.92	0.47	0.89	0.81	0.89
2	I	0.64	0.94	0.77	0.96	-3.29	0.92	0.81	0.94
	III	0.68	0.89	0.25	0.78	-1.50	0.78	0.80	0.81
3	I	0.59	0.89	0.82	0.93	-0.44	0.87	0.82	0.92
	III	0.86	0.87	0.79	0.85	-19.69	0.73	0.11	0.82
Stage I average		0.53	0.91	0.85	0.95	-1.49	0.90	0.84	0.92
Stage III average		0.74	0.90	0.60	0.85	-6.91	0.80	0.57	0.84
Overall average		0.63	0.90	0.73	0.90	-4.20	0.85	0.71	0.88

Table 8. Minimum variable and parameter sets required to utilise each of the methods evaluated.

Method	Variable and parameter requirements
Multiple Regression Modelling of Q_p	Q_{tot} (as a minimum)
Scaling Technique	α_p, Q_{tot}, P, I
NRCS Curve Number	P, CN
NRCS Graphical Peak Discharge	A, Q_{tot}, F, t_c, L, S, Y, CN, P
Variable Infiltration Rate	P_i, Q_{tot}, t_i, α

Figures

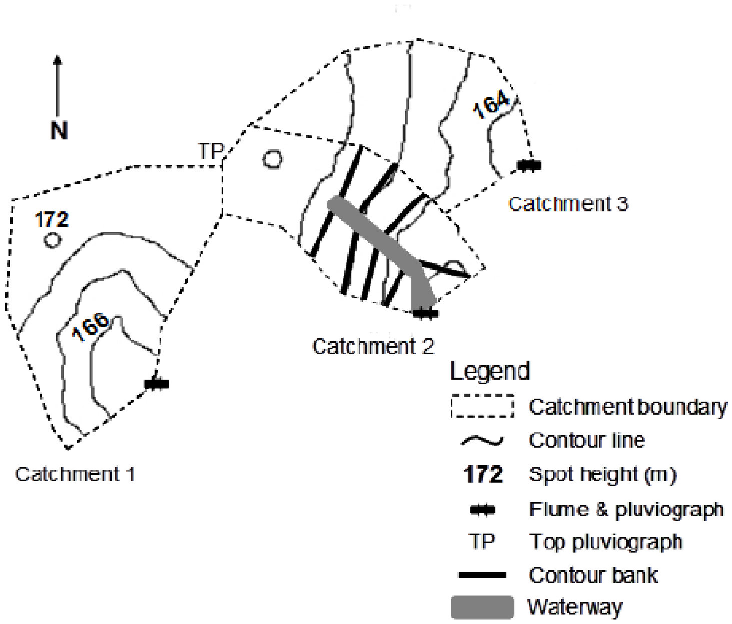

Figure 1. Schematic diagram of the Brigalow Catchment Study showing catchment boundaries, contour banks, waterways and the location of rainfall and runoff recording stations.

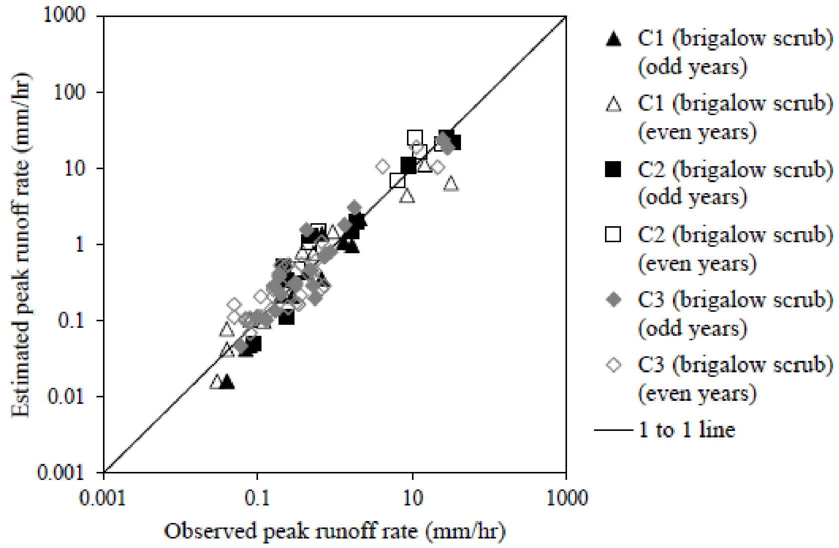

Figure 2. Observed peak runoff rate data compared with estimated peak runoff rate data using multiple regression model equations (Table 2) for the three catchments during Stage I.

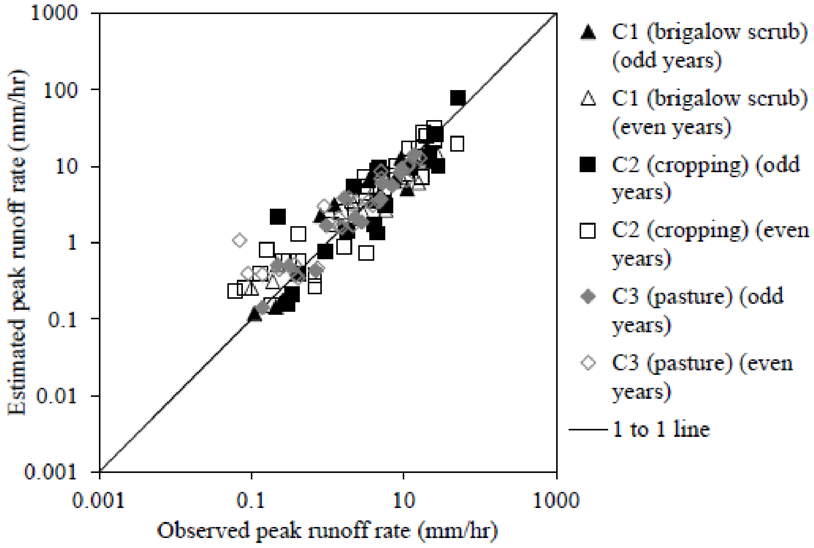

Figure 3. Observed peak runoff rate data compared with estimated peak runoff rate data using multiple regression model equations (Table 2) for the three catchments during Stage III.

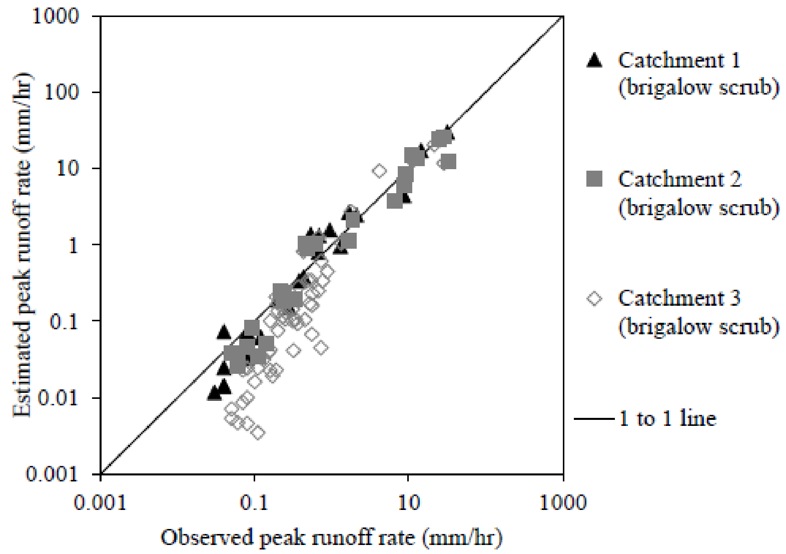

Figure 4. Observed peak runoff rate data compared to the scaling technique estimated peak runoff rate data for the three catchments during Stage I.

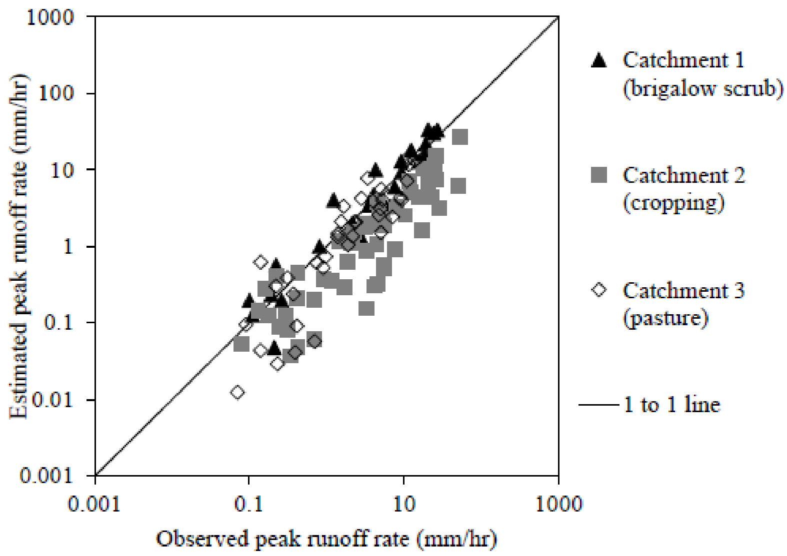

Figure 5. Observed peak runoff rate data compared to the scaling technique estimated peak runoff rate data for the three catchments during Stage III.

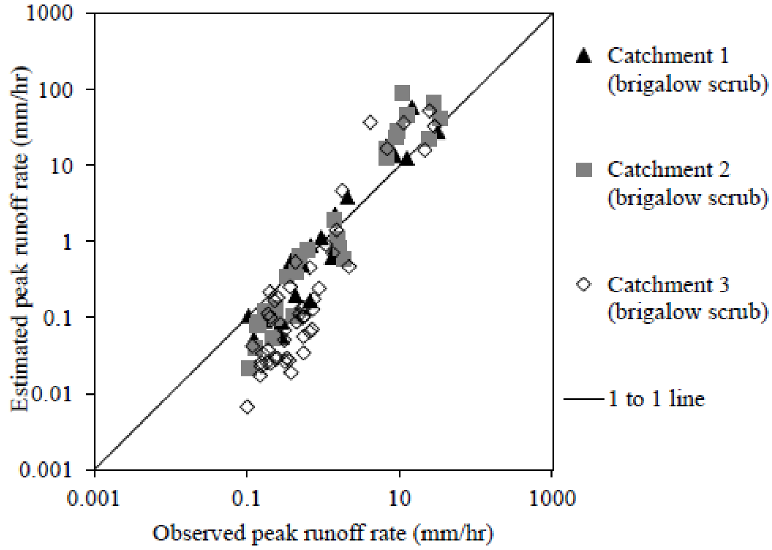

Figure 6. Observed peak runoff rate data compared to NRCS method estimated peak runoff rate data for the three catchments during Stage I.

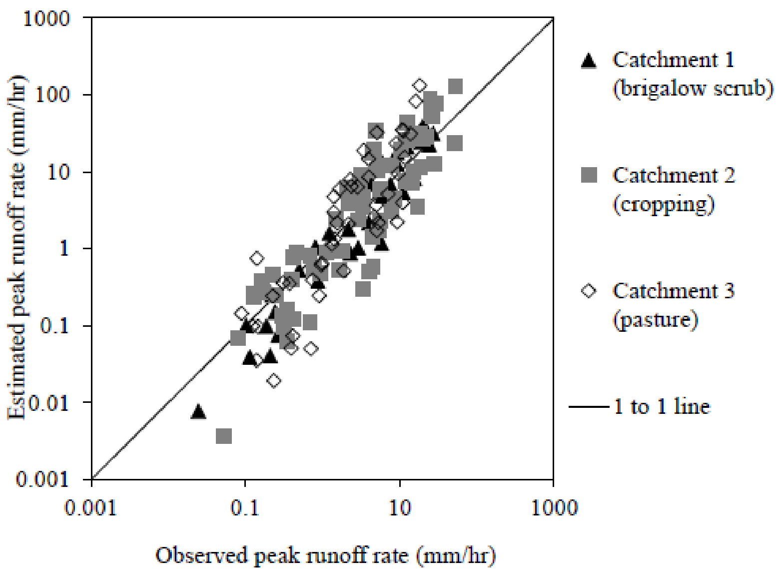

Figure 7. Observed peak runoff rate data compared to NRCS method estimated peak runoff rate data for the three catchments during Stage III.

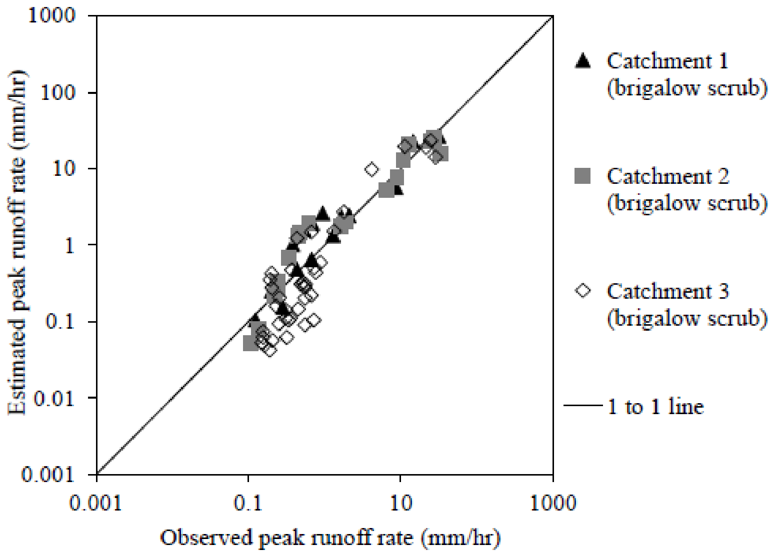

Figure 8. Observed peak runoff rate data compared to the routed VIR method estimated peak runoff rate data for the three catchments during Stage I.

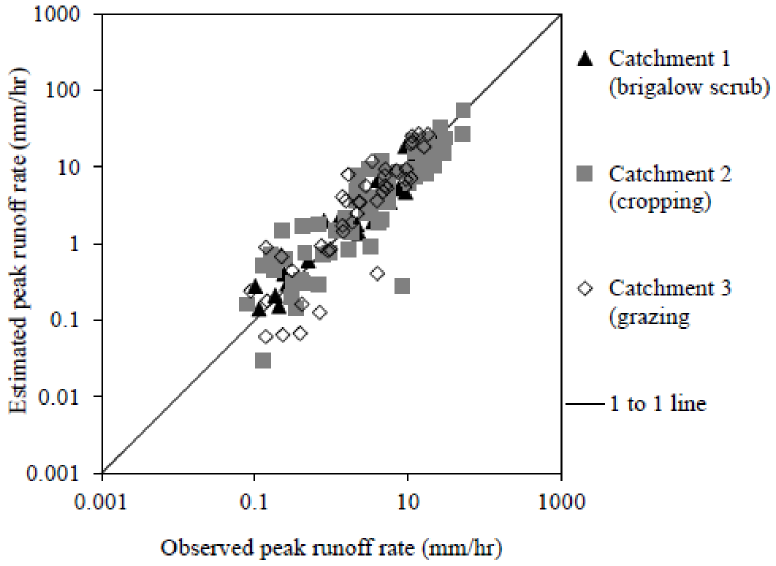

Figure 9. Observed peak runoff rate data compared to the VIR method estimated peak runoff rate data for the three catchments during Stage III.