Reef Rescue Research & Development

Technical Report

Runoff Nitrogen, Phosphorus and Sediment Generation Rates from Pasture Legumes: An Enhancement to Reef Catchment Modelling

RRRD009

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Department of Natural Resources and Mines, Rockhampton

Queensland Government

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Runoff Nitrogen, Phosphorus and Sediment Generation Rates from Pasture Legumes: An Enhancement to Reef Catchment Modelling (Project RRRD009)

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Cover photographs: Leucaena pasture (top left), runoff event through flume (top right), leucaena trimming (bottom left) and butterfly pea ley pasture (bottom right) are all sourced from the Brigalow Catchment Study photo archives, courtesy of the Department of Natural Resources and Mines.

This report is available from the Reef Rescue R&D website: http://www.reefrescueresearch.com.au

Executive Summary

The Fitzroy and Burdekin Basins are Queensland's largest coastal catchments, and both drain directly to the Great Barrier Reef. Greater than 80% of the catchment area in each basin is impacted by grazing, on both native and improved pastures. Despite substantial historical plantings of legume pasture species and ongoing programs of pasture improvement incorporating legumes, no evidence exists on the potential for increased exports of nitrogen in runoff from these systems to the Great Barrier Reef. This study sought to determine if broad-scale plantings of pasture legumes, particularly leucaena and butterfly pea, pose a risk to Great Barrier Reef water quality by increasing loads of nitrogen in runoff waters compared with both grass only pastures and the virgin brigalow scrub landscape.

Comparison of pasture type effects on water quality at the paddock scale was undertaken in the Fitzroy Basin using a paired, calibrated catchment study approach combined with simple regression based modelling developed from the long-term Brigalow Catchment Study. This work was complimented by plot scale rainfall simulation in the Burdekin and Burnett-Mary Basins.

At the paddock scale (12 to 24 ha) in the hydrological years 2010 and 2011, loads of total suspended solids and nitrogen (total and species) in runoff from grass only and leucaena pastures were typically lower than or equal to loads from virgin brigalow scrub. Phosphorus loads (total and species) from the pastures showed the opposite trend, being typically equal to or higher than loads from virgin brigalow scrub. In the same period, loads of all parameters from butterfly pea ley pasture were equal to or higher than loads from virgin brigalow scrub. High event mean concentrations did not necessarily equate to high loads. During 2012, no runoff occurred from the virgin brigalow scrub, so loads from all catchments were an absolute increase compared with their pre-European condition.

Plot scale rainfall simulation conducted in the Burdekin and Burnett-Mary Basins showed that runoff in the late-dry season typically had higher loads of total nutrients than runoff in the late-wet season. No significant interaction between pasture type and season for total suspended solids was observed. Rainfall simulations at the plot scale in the Burdekin and Burnett-Mary Basins also indicated that results from paddock scale catchment studies at the long-term Brigalow Catchment Study in the Fitzroy Basin are applicable to other grazing areas in the Brigalow Belt Bioregion.

Modelling flow and water quality from cropping and grazed buffel grass pasture between 1984 and 2012 showed similar trends to the rainfall simulation studies. Using virgin brigalow scrub as a reference, cropping exported more dissolved inorganic nitrogen and phosphorus, total phosphorus and total suspended solids; whilst grazing exported less total nitrogen and dissolved inorganic nitrogen, but more total and dissolved inorganic phosphorus and more total suspended solids. Cropping exports were always greater than grazing exports.

Within the grazing landscape, soil and pasture nutrient concentrations exhibited high variability and limited temporal response within the study. With no discernible period of potential high nutrient availability during the year, soil and pasture management should

focus on minimising runoff, rather than manipulation of the natural nutrient cycle to reduce risks to water quality.

Newly planted legume based ley pastures pose a risk to water quality as they contribute higher nutrient loads than grass only pasture systems, established leucaena pastures, and the virgin brigalow scrub landscape representative of the environment in its pre-European condition. However, they do reduce total suspended sediment loads compared with the cropping system that they replaced. Dissolved inorganic nitrogen loads from well-established leucaena exceeded those from grass, indicating a potential risk to water quality from the legume component of permanent pasture. This may have implications for parts of northern Australia, such as the Burdekin Basin, with large areas of naturalised S*tylosanthes spp*. pasture. These findings have been synthesised into a series of values and trends suitable for use in model development and validation to further refine estimations of the impact of changed land use, management and the adoption of leguminous pastures on water quality.

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Abbreviations

BCS	Brigalow Catchment Study
DIN	Dissolved Inorganic Nitrogen (NO_x-N + NH_4-N)
DIP	Dissolved Inorganic Phosphorus, also known as Filterable Reactive Phosphorus (FRP) and Orthophosphate (PO4-P)
EMC	Event Mean Concentration
GBR	Great Barrier Reef
IS	Insufficient Sample
NA	Not Available
NH_4-N	Ammonium-Nitrogen
NO_2-N	Nitrite-Nitrogen
NO_3-N	Nitrate-Nitrogen
NO_x-N	Oxidised Nitrogen (NO_3-N + NO_2-N)
PQL	Practical Quantitation Limit
TKN	Total Kjeldahl Nitrogen
TKP	Total Kjeldahl Phosphorus
TN	Total Nitrogen (TKN + NO_x-N)
TP	Total Phosphorus
TSS	Total Suspended Solids
WQA	Water Quality Analyser

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Chapter 1: General Introduction

Great Barrier Reef Water Quality Monitoring and Improvement

The Great Barrier Reef (GBR) covers an area of 348,000 km^2 along the north-eastern coast of Australia and is the largest reef ecosystem in the world (Great Barrier Reef Marine Park Authority 2010; United Nations Educational, Scientific and Cultural Organisation 2013). The GBR was placed on the World Heritage List in 1981, due to its outstanding beauty and high diversity of habitats and species (Great Barrier Reef Marine Park Authority 2010; United Nations Educational, Scientific and Cultural Organisation 2013). Nonetheless, terrestrial catchments adjacent to the GBR have undergone extensive anthropogenic modifications over the last 150 years, including land development for urban, mining and agricultural purposes (Department of the Premier and Cabinet 2009).

These land use changes have led to increased pollutant loads of nutrients, sediments and pesticides entering the GBR, which adversely impact the survival of this precious ecosystem (Department of the Premier and Cabinet 2009). For example, in 2007, an estimated 6.6 Mt of sediment, 16,600 t of nitrogen and 4,180 t of phosphorus reached the GBR due to loss from adjacent catchments (Department of the Premier and Cabinet 2009). Pollutant loads such as these led to a range of problems, including crown-of-thorns starfish (*Acanthaster planci*) outbreaks which lead to coral mortality and increased turbidity which adversely impacts seagrasses, a primary dietary component of the vulnerable dugong (*Dugong dugon*) (Brodie and Waterhouse 2012). A management issue that flows on from increased pollutant loads is that the residence time of nitrogen, phosphorus and sediments in the GBR has been estimated between years and decades, which will influence the recovery time of corals and seagrasses in response to remediation efforts (Brodie *et al*. 2012).

Agricultural land uses are currently the largest contributor to pollutant loads entering the GBR (Department of the Premier and Cabinet 2009), and of the 38 MHa of land within the five reef catchments adjacent to the GBR, 71% (27 MHa) is grazed and 3% (1 MHa) is cropped (Australian Bureau of Statistics 2009). To reduce the risk of declining water quality entering the GBR, the Australian and Queensland Governments enacted the Reef Water Quality Protection Plan, commonly referred to as Reef Plan, in 2009 (Department of the Premier and Cabinet 2009). A key priority of the Reef Rescue Research and Development Program, a component of Reef Plan, is to improve understanding of the link between land management practices and environmental impacts on hydrology and water quality for the grazing, sugarcane, horticulture and dairy sectors. This report specifically addresses grazing land management practices and the introduction of legume species (leucaena and butterfly pea) into the pastures of the Brigalow Belt Bioregion adjacent to the GBR.

Grazing in Queensland's Brigalow Belt Bioregion

Brigalow scrub is a general term used to refer to a vegetation community that is dominated by brigalow (*Acacia harpophylla*) trees, either in a monoculture or in association with other species, such as gidgee (*Adansonia cambagei*), belah (*Casuarina cristata*) and Dawson River blackbutt (*Eucalyptus cambageana*) (Department of Lands 1968). Brigalow scrub originally covered an area approximately 36.7 Mha from Townsville in north Queensland to Dubbo in central-western New South Wales (Cowie *et al*. 2007; Thornton *et al*. 2007). This area can be further classified into the Northern Brigalow Belt Bioregion in central-east Queensland which covers an area 59,824 km^2, and the Southern Brigalow Belt Bioregion in southern Queensland with a small patch in northern New South Wales which covers an area 56,496 km^2 (Bastin *et al*. 2008). The dominant land use in both the Northern and Southern Brigalow Belt Bioregions is grazing, which accounts for approximately 90% and 80% of the areas, respectively (Bastin *et al*. 2008). The Brigalow Belt Bioregion covers 22.1 Mha of the GBR catchment area in the Fitzroy, Burdekin and Burnett-Mary Basins (Pulsford 1993; Great Barrier Reef Marine Park Authority 2009; Australian Government 2013; Queensland Government 2013). As grazing is the predominant land use in these catchments, land development and grazing land management has a direct impact on GBR water quality.

Land development has a proud history in Queensland. The Queensland Government had an initiative in the 1930s to lease brigalow land to unemployed persons on the condition that they partially develop the land for agriculture. This initiative was motivated by drought which had destabilised the grazing industry and high unemployment due to the global economic depression (Department of Lands 1968). However, large tracts of brigalow land held as pastoral leases or grazing selections remained undeveloped in the 1950s and the Queensland Government commenced the Fitzroy Basin Land Development Scheme in 1963 to encourage increased production. This scheme resulted in the clearing of 4.5 Mha of brigalow scrub for cropping and grazing, and continued through until the 1990s (Department of Lands 1968; Partridge *et al.* 1994). Broad-scale land clearing was only halted in Queensland in 2006 (Thornton *et al.* 2012). Despite this recent history of intensive development for agriculture, the impacts of some practices, such as the introduction of legume based pastures, on water quality were poorly understood.

Current Knowledge Gap

Grazing is the dominant agricultural land use in GBR catchments of Queensland, accounting for 96% (26.2 Mha) of the area within the Fitzroy, Burdekin and Burnett-Mary Basins (Australian Bureau of Statistics 2009). The inclusion of nitrogen fixing pasture legumes into grazing systems is driven by well documented productivity gains (Lawrence and French 1992; Partridge *et al.* 1994; Humphreys and Partridge 1995). Although pasture legumes are considered good land management practice, there is no evidence from Australia to describe impacts on sediment and nitrogen concentrations and loads in runoff discharged to the GBR. At this stage, Source Catchments (www.ewater.com.au) modelling of GBR catchments under Reef Plan do not differentiate between native or improved pastures as there is no empirical evidence to support model changes.

This project sought to determine if pasture legumes, particularly of leucaena and butterfly pea, represent a risk to reef water quality through increased loads of sediment and nitrogen in runoff waters compared with grass only pastures and virgin brigalow scrub; the latter representative of the landscape in its pre-European condition.

Project Objectives

The overarching goal of this project was to quantify nitrogen generation rates in runoff water from pastures with and without legumes. This was achieved by the completion of three main objectives:

(1) Determine loads and event mean concentrations (EMCs) of nitrogen, phosphorus and sediment in runoff water from grass only pasture, butterfly pea ley pasture, and leucaena pasture at the rainfall simulator plot scale (1.7 m^2) and paddock scale (12 to 24 ha) (Chapter 3).

(2) Use historical data from the long-term Brigalow Catchment Study (BCS) site to estimate nitrogen, phosphorus and sediment loads and EMCs from virgin brigalow scrub, and use this data to provide a pre-European context for load changes due to land use conversion to agriculture (Chapter 4).

(3) Develop an understanding of the biophysical interaction of nitrogen and phosphorus in virgin brigalow scrub, grass only pasture, butterfly pea ley pasture, and leucaena pasture systems by investigating seasonal trends in soil and pasture nutrient concentrations (Chapter 5).

Chapter 2: Study Sites

One study site was selected in each of the Fitzroy, Burdekin and Burnett-Mary Basins of central Queensland (Figure 2.1). All sites were characterised by dark Vertosol soils, and in their natural state supported eucalypt and acacia woodland or bluegrass downs and brigalow-gidgee scrub vegetation communities (Reid *et al.* 1986; Shields *et al.* 1993). All three sites are in the Northern Brigalow Belt Bioregion and are in predominantly grazed catchments that flow to the GBR (Bastin *et al*. 2008). The BCS site in the Fitzroy Basin was the primary site used for data collection. Sites in the Burdekin and Burnett-Mary Basins were used to quantify runoff water quality at the small plot scale (1.7 m^2) using rainfall simulation (Chapter 3). These satellite sites were selected to broaden the applicability of land use comparison findings from the BCS in the Fitzroy Basin to other reef catchments in the Brigalow Belt Bioregion. Field inspections of the three study sites confirmed their similarities in soil and vegetation (Table 2.1). Mapping of these three catchments shows that 100% (14 Mha) of the Fitzroy Basin, 46% (6.1 Mha) of the Burdekin Basin, and 49% (2 Mha) of the Burnett-Mary Basin are within the Brigalow Belt Bioregion (Pulsford 1993; Great Barrier Reef Marine Park Authority 2009; Australian Government 2013; Queensland Government 2013).

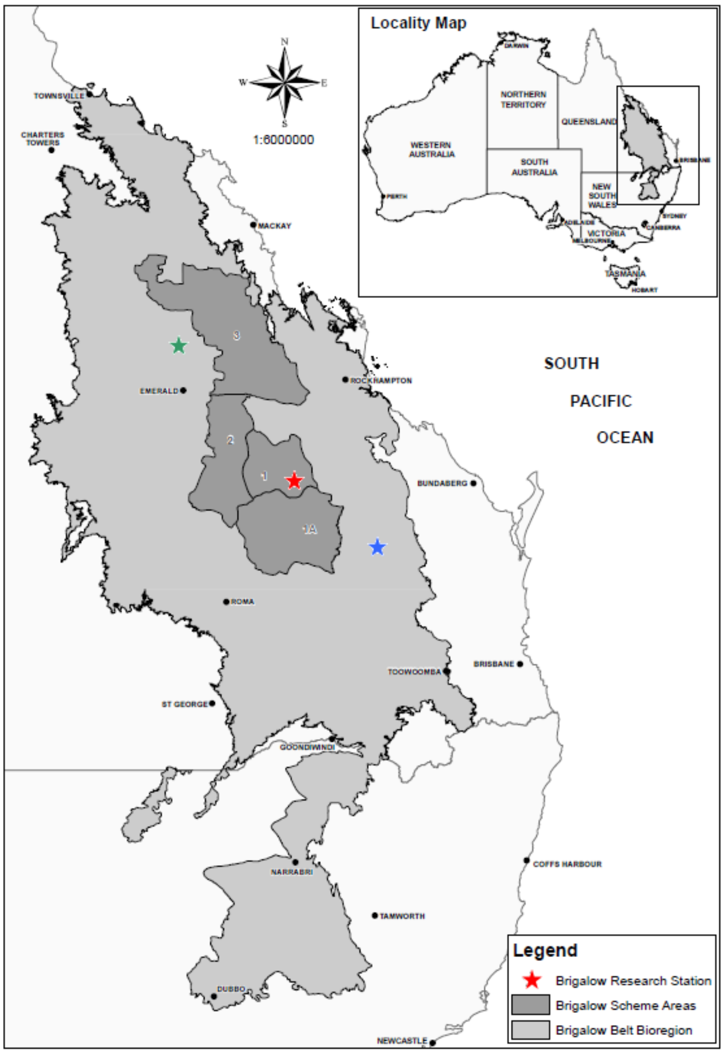

Figure 2.1: Three study sites located in the Burdekin (green star, top), Fitzroy (red star, centre), and Burnett-Mary (blue star, bottom) Basins within the Brigalow Belt Bioregion of central Queensland.

Table 2.1: Photographic comparison of study sites for the three pasture types within each of the three Great Barrier Reef catchments.

		Great Barrier Reef Catchment		
		Fitzroy Basin	**Burdekin Basin**	**Burnett-Mary Basin**
Pasture Type	**Grass**			
	Butterfly Pea			
	Leucaena			

Fitzroy Basin: Brigalow Catchment Study

The BCS (24°48'S and 149°47'E) is located near Theodore in central Queensland, and was established in 1965 to quantify the impact of land development on hydrology, productivity and resource condition. The long-term study was established to understand the effect of broad-scale clearing on the landscape associated with the Fitzroy Basin Brigalow Land Development Scheme (Cowie *et al*. 2007). The study is a paired, calibrated catchment study consisting of three catchments monitored since 1965 and an additional fourth catchment since 2010.

Soils within each catchment are predominantly grey and black Vertosols, with an average slope of 2.5%. In their virgin state, all catchments were vegetated with brigalow scrub communities (Thornton and Elledge 2012). The region has a semi-arid, subtropical climate. Summers are wet with 70% of the annual average calendar rainfall of 720 mm falling between October and March, whilst winter rainfall is low. Hydrological year (October to September) annual average rainfall is 647 mm. Rainfall is highly variable, ranging from 11 mm or less in any month to 165 mm in one day. Annual potential evaporation is 2,133 mm and average evaporation is at least twice the average rainfall in all months (Thornton *et al.* 2010).

The BCS can be separated into four experimental stages (Table 2.2; Figure 2.2):

(1) Calibration of the catchments between 1965 and 1982 (Stage I). That is, rainfall and runoff were monitored for 18 years from three contiguous catchments. Mathematical relationships were derived to predict runoff from Catchment 2 (C2) and Catchment 3 (C3) given known runoff from Catchment 1 (C1) (Thornton *et al*. 2007).

(2) Development of the catchments from 1982 to 1983 (Stage II). That is, Catchment 1 remained virgin brigalow scrub to provide a control treatment, and Catchments 2 and 3 were cleared and the fallen timber burnt *in-situ*. Catchment 2 was then developed for cropping with the construction of contour banks and grassed waterways, whilst Catchment 3 was developed for grazing by the planting of improved grass pasture.

(3) Land use comparison from 1984 until 2010 (Stage III). Catchment 2 was typically opportunity cropped with either wheat or sorghum, and zero or reduced till fallows were introduced in the 1990s. No fertiliser inputs were used (Radford *et al*. 2007). Catchment 3 was grazed at industry recommended stocking rates with utilisation to result in no less than 1000 kg/ha of pasture available at any time. During this phase, both the cropping (C2) and grazing (C3) catchments were compared with the virgin brigalow scrub catchment (C1).

(4) Pasture type comparison from 2010 until present (2013) (Stage IV). During this phase, four catchments were compared: (1) virgin brigalow scrub (C1); (2) butterfly pea (*Clitoria ternatea* cv. Milgarra) ley pasture with Rhodes grass (*Chloris gayana* cv. Callide) (C2); (3) buffel grass (*Cenchrus ciliaris* cv. Biloela) pasture (C3); and (4) leucaena (*Leucaena leucocephala* cv. Cunningham) and buffel grass pasture (C4). Ley

pasture refers to the planting of pasture species into cropping lands, typically for 3 to 5 year rotations, to improve soil fertility. It was during this stage that the fourth catchment (C4) was introduced to the study. Whilst this catchment is characterised by similar soils, slope and native vegetation as the other catchments, it had a different history of land use. This catchment had a prior history of cropping and grazing before being developed for intensive grazing with the planting of leucaena on 8 m hedgerows in 1998. Due to its later inclusion in the study, the fourth catchment did not have a runoff calibration period to allow comparison with the other three catchments.

Further details of the calibration, development, and comparison of land use stages are documented in other sources (Cowie *et al*. 2007; Radford *et al*. 2007; Thornton *et al*. 2007).

Table 2.2: Land use history of the Brigalow Catchment Study.

Catchment	Area (ha)	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 (2013)
C1	16.8	Brigalow scrub	Brigalow scrub	Brigalow scrub	Brigalow scrub
C2	11.7	Brigalow scrub	Development	Cropping	Butterfly pea
C3	12.7	Brigalow scrub	Development	Grass	Grass
C4	23.3	Brigalow scrub	NA	NA	Leucaena

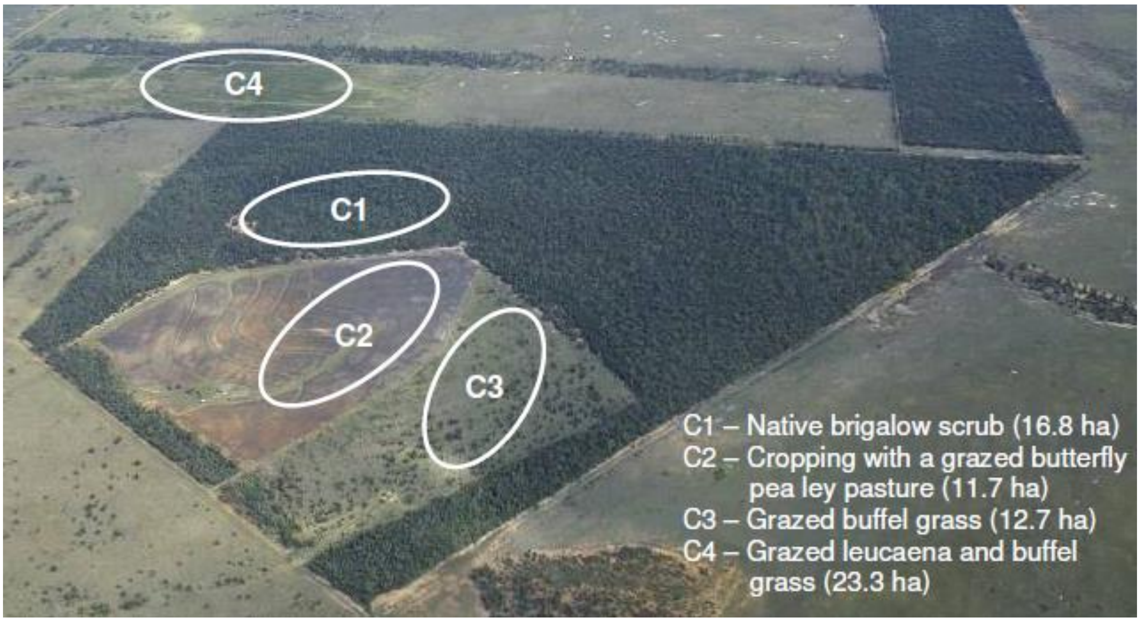

Figure 2.2: Aerial photo of the four catchments at the Brigalow Catchment Study site.

Burdekin Basin: Private Properties

Two neighbouring private properties (22°30'S and 147°30'E) were used near Clermont in central Queensland. The experimental sites were characterised by Vertosol soils and in their natural state supported bluegrass downs and brigalow-gidgee scrub vegetation communities (Shields *et al.* 1993). Three pasture types were compared: (1) grass only; (2) butterfly pea and grass; and (3) leucaena and grass.

The dominant grass species in all three pasture types were bluegrass (*Dichanthium sericeum* and *Bothriochloa spp*.) and buffel grass. Pasture in the grass only paddock was greater than 10 years old, and pasture in the butterfly pea paddock was planted in 2003 (9 years old). Leucaena was planted in 1996 (17 years old) on 10 m hedgerows.

Burnett-Mary Basin: Brian Pastures Research Station

The Brian Pastures Research Station (25°39'S and 151°45'E) is located near Gayndah in central Queensland. The experimental sites were characterised by Vertosol soils and in their natural state supported eucalypt and acacia woodland vegetation communities (Reid *et al.* 1986). Three pasture types were compared: (1) grass only; (2) butterfly pea and grass; and (3) leucaena and grass. The dominant grass species in all three pasture types were bluegrass. Pastures in the grass only and butterfly pea paddocks were greater than 10 years old, and leucaena was planted in 1995 (18 years old) on 8 m hedgerows.

Chapter 3: Impact of Pasture Type on Runoff Nitrogen, Phosphorus and Sediment Generation Rates

Summary

Trends for Model Development and Validation

- Effect of pasture type on runoff water quality at the paddock scale over three hydrological years are summarised in Table 3.1.

Table 3.1: Overall trend of pasture type effect on key nutrient and sediment parameters for runoff water quality at the paddock scale. Br = virgin brigalow scrub, BP = butterfly pea ley pasture, Gr = grass only pasture, and L = leucaena pasture.

Parameter	Load Comparison	EMC Comparison
TKN	BP > Br > Gr = L	Br > BP > Gr = L
DIN	BP > Br > L > Gr	Br > BP > L > Gr
TP	BP > Gr = L > Br	BP > L > Br > Gr
TSS	BP > Br > Gr = L	Br = BP > Gr = L

Risks

- End of dry season runoff is a greater risk to water quality than end of wet season runoff.
- Runoff in years where none would have occurred from the pre-European landscape is an absolute increase in load, irrespective of how small the load.

Key Insights

- High EMCs do not necessarily equate to high loads.
- Butterfly pea had the highest sediment load of all pastures; however, it was substantially lower than the cropping system that it replaced.
- Typically, nitrogen (total and species) and total suspended sediment loads and EMC's from grass and leucaena pastures were lower than or equal to brigalow scrub
- Nitrogen (total and species) and total suspended sediment EMC's from brigalow scrub were greater than all pasture types

Management Action

- Manage grazing pressure to ensure high cover and biomass at the end of the dry season. This protects the soil surface, increases infiltration and helps to utilise available soil moisture to minimise runoff.

Introduction

Leucaena hedgerows planted with companion pasture grasses are one of the most productive, profitable and sustainable grazing systems in tropical and subtropical Australia (Boddey *et al*. 1997; Dalzell *et al*. 2006; Shelton and Dalzell 2007). In managed agricultural scenarios, leucaena has many reported though often not measured benefits, including enhanced soil fertility, reduced soil erosion and improved runoff water quality (Dalzell *et al*. 2006; Shelton and Dalzell 2007). Despite the inclusion of pasture legumes, such as leucaena and butterfly pea, into grazing systems in central Queensland (Pengelly and Conway 2000), there are currently no quantitative studies available on the effects of legume based pastures on water quality in Australia.

Research on nutrient and sediment loads exported in runoff from leguminous pastures in central Queensland is important for understanding water quality in the GBR and implementing management practices to reduce their effects. This is due to the rapid and broad-scale inclusion of improved legume pastures in grazing lands, with an increase in the area of butterfly pea in central Queensland to more than 100,000 ha in 2004 (Collins and Grundy 2005) and an increase in the area of leucaena in northern Australia to 150,000 ha in 2006 (Dalzell *et al*. 2006). It has also been estimated that 13 Mha in northern Australia is suitable for the future establishment of leucaena, with 4 to 5 Mha of this area in the Fitzroy Basin alone (Dalzell *et al*. 2006).

This study assesses the risk of increased nutrient and sediment exports from leguminous pastures, specifically leucaena and butterfly pea, to GBR water quality. This was achieved by comparing loads and EMCs from grass only pasture, butterfly pea ley pasture and leucaena pasture to loads and EMCs from virgin brigalow scrub at the paddock scale (12 to 24 ha) under natural rainfall. In addition, loads lost from the three pasture types were compared at the plot scale (1.7 m^2) under simulated rainfall conditions.

Methods

Paddock Scale

The study was conducted on the BCS site near Theodore in central Queensland during experimental Stage IV. To examine land use effects on nutrient and sediments loads in runoff water, four catchments were monitored over the 2010 to 2013 hydrological years: (1) virgin brigalow scrub (C1); (2) cropping with a grazed butterfly pea ley pasture (C2); grazed grass pasture (C3); and grazed leucaena pasture (C4). Nonetheless, no additional runoff events were expected as the missed period is the late-dry season. Refer to 'Study Sites' in Chapter 2 for more information on each of these catchments.

Each catchment was instrumented to measure runoff using a 1.2 m steel HL flume with a 3.9 x 6.1 m concrete approach box. Water heights through the flumes were recorded using mechanical float recorders. Rainfall was recorded adjacent to each flume and at the head of the catchments (Thornton *et al.* 2007). A runoff event was defined as commencing when stage height exceeded zero and finished when it returned to zero.

Discrete water quality samples were obtained using auto-samplers and rising stage samplers. Auto-samplers were programmed to sample every 0.1 m change in absolute stage height, and rising stage samplers were installed to sample from approximately 0.05 m of stage height in 0.1 m increments. Laboratory analysis of runoff water samples were completed by Queensland Health Forensic and Scientific Services in Coopers Plains, Queensland (Table 3.2). The method numbers reported below were current as of April 2013, and although the laboratory has changed methods over the years, only comparable methods have been included.

Table 3.2: Methods used by Queensland Health Forensic and Scientific Services for nutrient and sediment analyses of water samples.

Parameter	Method
TN	Method 13802 by simultaneous persulfate digestion. For the period 2000 to 2003, method 13804 based on simultaneous Kjeldahl digestion was reported and TN was manually calculated as TKN + NO_x-N.
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.
DIN	Manually calculated as NO_x-N + NH_4-N.
TP	Method 13800 by simultaneous persulfate or Kjeldahl digestion.
DIP	Method 13799 by flow injection analysis.
TSS	Method 18211 based on gravimetric quantification of solids in water.

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, such as with rising stage samplers, 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 EMC was calculated by dividing total event load by total event flow.

Annual total load was calculated by summing all of the event based water quality loads for each catchment. Annual average EMC was calculated by averaging the event based EMCs. Annual average EMCs from the period 2000 to 2012 were used to calculate a grand average 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, primarily due to events being too small to trigger auto-samplers, estimations were obtained by multiplying the grand average annual EMC by the observed flow. The EMC for total suspended sediments from butterfly pea in 2012 was reported as ‘not available’ (NA), as there was insufficient sample for laboratory analysis; however, load was estimated using grand average EMC.

Plot Scale

Rainfall simulations in the Burdekin and Burnett-Mary Basins were used to investigate runoff water quality from three different pasture types: (1) grass only; (2) butterfly pea; and (3) leucaena. Refer to 'Study Sites' in Chapter 2 for more information. Data was collected in the late-wet and late-dry to determine seasonal effects on nutrient loads lost in runoff water; that is, the end of the summer growing season and the end of the winter dormant season. Trials for the late-wet and late-dry were conducted in May 2012 and September 2012 at the Burdekin site, and May 2012 and October 2011 at the Burnett-Mary site, respectively.

For each pasture type within each basin, four (replicate) plots 1.0 x 1.7 m were selected based on a stratified random sampling design. A three sided metal frame (0.15 m high) was placed approximately 0.05 to 0.07 m into the ground to achieve a hydraulic barrier. The short, downslope side of the plot had a separate metal plot end with a spout for collecting runoff; the plot end was pushed into the ground until the top edge was level with the soil surface. All standing dry matter was removed from the plot so that the plant biomass did not redirect rainfall outside the plot.

A standard Paddock to Reef Program storm event was applied to all plots with simulated rainfall applied at intensities between 56 and 103 mm/hr (target intensity 80 mm/hr). Rainfall events in the Burdekin Basin had an average recurrence interval between 5 and 10 years and an annual exceedance probability between 5 and 18%; whereas the Burnett-Mary Basin had an average recurrence interval between 2 and 5 years and an annual exceedance probability between 18 and 39%. Dunkerley (2008) reviewed 40 rainfall simulation studies and found that intensities typically ranged between 60 and 100 mm/hr, which more closely resemble extreme weather events rather than natural rainfall events.

The rainfall simulators used were a lightweight aluminium tube construction in a basic A-frame configuration (Figure 3.1). Three downward facing oscillating nozzles delivered a flat spray pattern of water across the plot with a fan angle of 80 degrees. A metal shroud positioned below each nozzle limited the lateral and longitudinal spray delivery, and also collected excess water which was returned to the pump delivery unit for recycling. The assembly was adjusted at each plot for adequate magnitude of nozzle sweep and symmetry of sweep relative to the simulator frame. A pressure gauge allowed the delivery pressure of water to be set at 60 kPa which was monitored throughout the simulation event. The rainfall simulators were calibrated each time they were setup by placing a plastic cover over the plot and measuring runoff volumes from timed samples. Refer to Loch *et al*. (2001) for general information on rainfall simulators.

Rainfall was applied to wet each plot until runoff flow was steady; that is, water coming out the spout did not show signs of pulsing due to the sweeping action of the nozzles delivering water to the plot. Rainfall continued on the plot for a further 30 minutes, during which time runoff samples were collected at five minute intervals (0, 5, 10, 15, 20, 25 and 30 min) in the form of a discrete 1 litre sample per interval, and one composite 1 litre sample for the event. Composite samples were collected by sampling for a set time within each five minute interval. The discrete water sample was used to calculate rainfall intensity, runoff volume and other event characteristics in Water Quality Analyser (WQA) (version 2.1.2.4;

www.ewater.com.au), and the composite water sample was used for nutrient analyses. Samples of source water were also submitted for nutrient analysis for each trial, or whenever the source changed mid-trial. Nutrient analyses were completed by the Environmental Resource Sciences Chemistry Centre in Dutton Park, Queensland (Table 3.3); and a quality assurance and control criteria was later applied to all laboratory results (Table 3.4). The plot scale data from this study was collected in collaboration with the Paddock to Reef rainfall simulation project by Thornton *et al*. (2013) (Appendix 1.5).

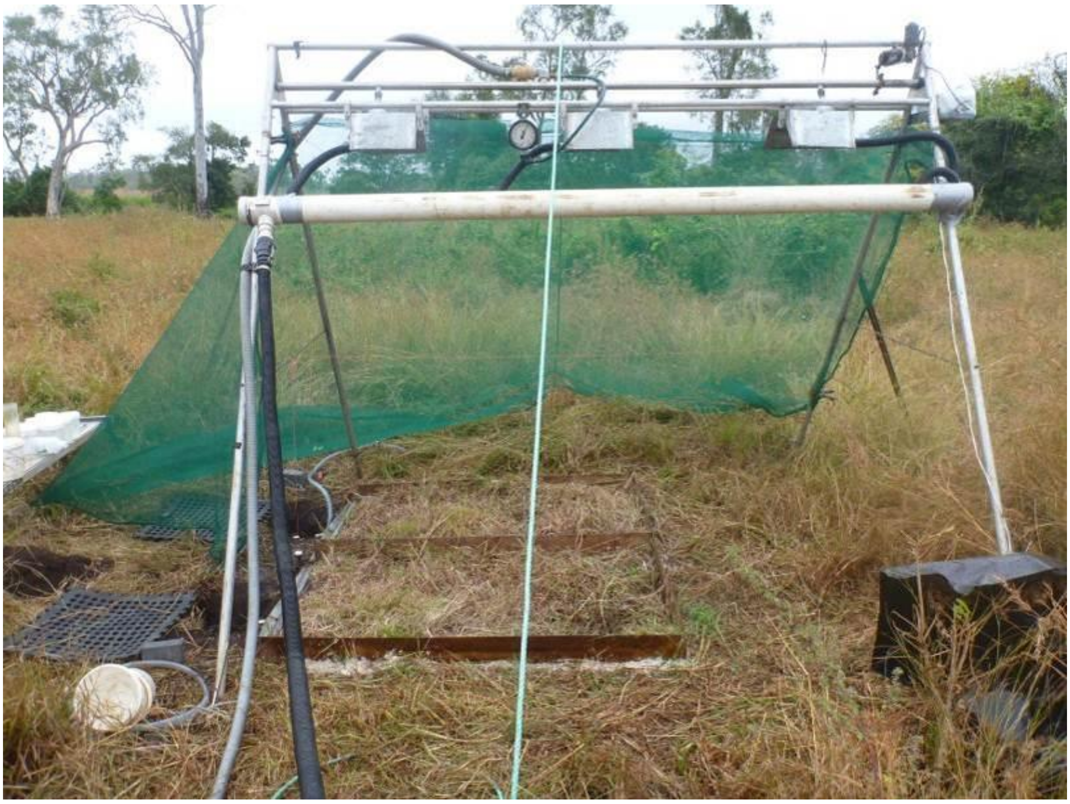

Figure 3.1: Rainfall simulation being undertaken. Water is applied via three nozzles each contained within a metal shroud (silver boxes toward the top of the picture). Two plots can be seen under the simulator. The nozzles oscillate across the plots (front to back in this picture) with runoff exiting through the plot end installed on the downslope end of the plot (left of picture).

Table 3.3: Methods used by the Environmental Resource Sciences Chemistry Centre for nutrient and sediment analyses of water samples, based on the American Public Health Association and American Water Works Association (2005).

Parameter	Method
TKN	Method 4500-Norg D using a catalysed acidic block digestion with a colorimetric segmented flow analysis.
NO_x-N	Method 4500-NO_3 using a variation of the automated cadmium reduction to determine concentrations of NO_3 and NO_2 by segmented flow analysis.
NH_4-N	Method 4500-NH_3 adapted for segmented flow analysis.
DIN	Manually calculated as NO_x-N + NH_4-N.
TKP	Method 4500-P B using a catalysed acidic block digestion with a colorimetric segmented flow analysis.
DIP	Method 4500-P G based on segmented flow analysis.
TSS	Method 2540 D based on gravimetric quantification of non-filterable solids in water.

Table 3.4: Quality assurance and control criteria applied to the laboratory results. Good = sample quality ok and data used in analyses, IS = insufficient sample, PQL = practical quantitation limit, and blank = missing data or data not used in analyses.

Runoff Water Sample (A)	Source Water Sample (B)	Corrected Runoff Result (C)
Good	Good	A - B
Good	IS	A - (0.5 * PQL)
Good	< PQL	A - (0.5 * PQL)
Good	Blank	A - (0.5 * PQL)
Good	> A	0.5 * PQL
Good	= A	0.5 * PQL
< PQL	IS	0.5 * PQL
< PQL	< PQL	0.5 * PQL
< PQL	Blank	0.5 * PQL
< PQL	> A	0.5 * PQL
< PQL	Good	0.5 * PQL
IS	IS	IS
IS	< PQL	IS
IS	Blank	IS
IS	> A	IS
IS	Good	IS
Blank	IS	Blank
Blank	< PQL	Blank
Blank	Blank	Blank
Blank	> A	Blank
Blank	Good	Blank

Nutrient loads lost in runoff, on a kg/ha basis, were then calculated by:

$$\left[\frac{\text{Corrected Runoff Result (mg/L)} * \text{WQA Runoff Volume (L)}}{\text{Plot Length (m)} * \text{Plot Width (m)}}\right] / 100$$

Comparison of nutrient load losses between seasons was made separately for each pasture type within each site using analysis of variance in GenStat (version 14.2; www.vsni.co.uk).

Results

Paddock Scale

Rainfall during 2010 was 958 mm, and at the time, was the wettest year on record at the BCS site. During 2010, butterfly pea consistently had the highest loads of all parameters, whereas grass had the smallest loads of all nitrogen parameters and total phosphorus. Loads of total, oxidised and dissolved inorganic nitrogen were higher in butterfly pea and brigalow scrub than grass and leucaena pastures (Table 3.5). The smallest loads of dissolved inorganic phosphorus were from brigalow scrub, and the smallest loads of total suspended sediments were from leucaena.

Loads from the three pasture catchments during 2010, presented as a proportion of the load lost from brigalow scrub, indicate that butterfly pea consistently had higher loads than brigalow scrub, grass only pasture, and leucaena pasture for all parameters (Figure 3.2). Furthermore, in comparison to the uncleared control brigalow scrub, grass pasture had smaller loads for all parameters except dissolved inorganic phosphorus, and leucaena had smaller loads for sediments and all nitrogen parameters except ammonium-nitrogen.

Table 3.5: Runoff event based flow and water quality data from the brigalow scrub, butterfly pea ley pasture, grass only pasture and leucaena pasture at the Brigalow Catchment Study for the 2010 hydrological year.

Parameter	Brigalow Scrub	Butterfly Pea	Grass	Leucaena
Area (ha)	16.8	11.7	12.7	23.3
Total Discharge (mm)	55	175	111	114
Events (n)	4	9	7	7
TN Load (kg/ha)	6.5	9.3	2.4	2.6
TN EMC (mg/L)	18	8	2	2
NO_x-N Load (kg/ha)	2.03	4.93	0.25	0.70
NO_x-N EMC (mg/L)	1.87	5.38	0.16	0.50
NH_4-N Load (kg/ha)	0.05	0.16	0.04	0.06
NH_4-N EMC (mg/L)	0.07	0.07	0.04	0.07
DIN Load (kg/ha)	2.09	5.09	0.29	0.76
DIN EMC (mg/L)	1.94	5.45	0.20	0.57
TP Load (kg/ha)	0.43	1.22	0.32	0.51
TP EMC (mg/L)	0.52	0.55	0.23	0.34
DIP Load (kg/ha)	0.10	0.53	0.17	0.33
DIP EMC (mg/L)	0.15	0.24	0.10	0.18
TSS Load (kg/ha)	542	1122	218	153
TSS EMC (mg/L)	587	488	189	149

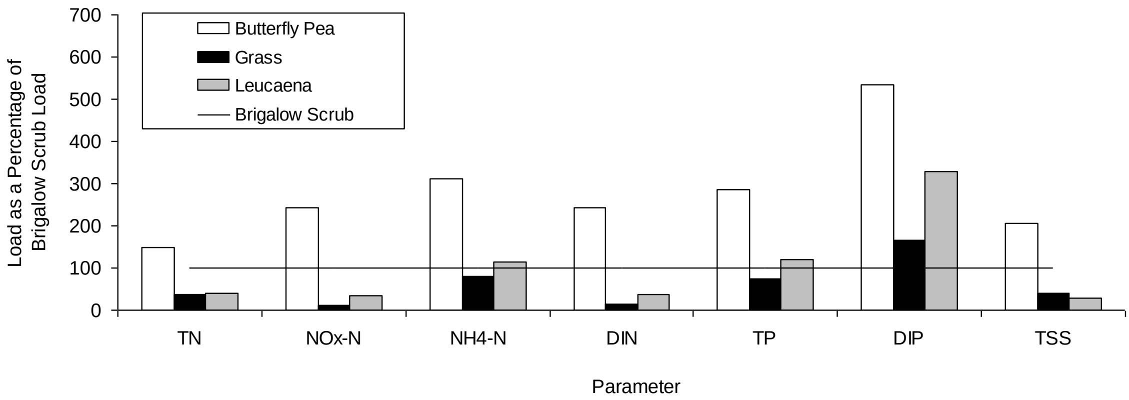

Figure 3.2: Nutrient and sediment loads (kg/ha) for the 2010 hydrological year from the butterfly pea ley pasture, grass only pasture and leucaena pasture as a percentage of the load from the virgin brigalow scrub catchment.

Rainfall during 2011 was 1,009 mm, breaking the record set the previous year. Loads in 2011 were typically higher than 2010 (Table 3.6); however, brigalow scrub exported less total suspended sediment, butterfly pea less oxidised and dissolved inorganic nitrogen, and leucaena less ammonium-nitrogen. When comparing between catchments during 2011, overall trends were similar as for 2010 (Figure 3.3). Butterfly pea had consistently higher loads than brigalow scrub, grass only pasture and leucaena pasture for all parameters except total nitrogen, which was only marginally smaller than the brigalow load. Grass pasture exported the smallest loads of oxidised and dissolved inorganic nitrogen, whereas leucaena exported the smallest loads of ammonium-nitrogen and total suspended sediments. Equally low total nitrogen loads were found in grass and leucaena. The smallest loads for total and dissolved inorganic phosphorus were from brigalow scrub.

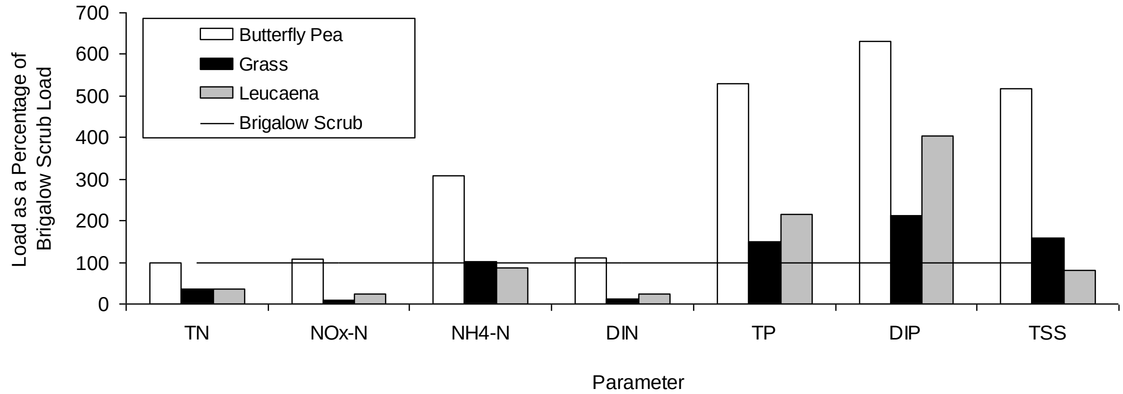

Figure 3.3: Nutrient and sediment loads (kg/ha) for the 2011 hydrological year from the butterfly pea ley pasture, grass only pasture and leucaena pasture as a percentage of the load from the virgin brigalow scrub catchment.

Rainfall during 2012 exceeded the 75th percentile of BCS records, with a total of 786 mm. However, despite above average rainfall, runoff was low for the three pasture catchments and non-existent for the brigalow scrub catchment (Table 3.7). As there was no runoff from the brigalow scrub, loads from all pasture catchments were an absolute increase compared with their pre-European condition.

Table 3.6: Runoff event based flow and water quality data from the virgin brigalow scrub, butterfly pea ley pasture, grass only pasture and leucaena pasture at the Brigalow Catchment Study for the 2011 hydrological year.

Parameter	Brigalow Scrub	Butterfly Pea	Grass	Leucaena
Area (ha)	16.8	11.7	12.7	23.3
Total Discharge (mm)	97	246	168	174
Events (n)	2	9	6	6
TN Load (kg/ha)	11.4	11.2	4.0	4.0
TN EMC (mg/L)	11.8	3.5	2.1	2.7
NO_x-N Load (kg/ha)	4.09	4.42	0.40	1.00
NO_x-N EMC (mg/L)	4.22	1.03	0.26	0.66
NH_4-N Load (kg/ha)	0.05	0.17	0.06	0.047
NH_4-N EMC (mg/L)	0.06	0.05	0.03	0.018
DIN Load (kg/ha)	4.14	4.59	0.46	1.04
DIN EMC (mg/L)	4.28	1.08	0.30	0.68
TP Load (kg/ha)	0.38	2.01	0.57	0.82
TP EMC (mg/L)	0.39	0.70	0.30	0.46
DIP Load (kg/ha)	0.13	0.81	0.27	0.52
DIP EMC (mg/L)	0.13	0.37	0.16	0.25
TSS Load (kg/ha)	297	1531	468	243
TSS EMC (mg/L)	307	343	192	167

Table 3.7: Runoff event based flow and water quality data from the virgin brigalow scrub, butterfly pea ley pasture, grass only pasture and leucaena pasture at the Brigalow Catchment Study for the 2012 hydrological year.

Parameter	Brigalow Scrub	Butterfly Pea	Grass	Leucaena
Area (ha)	16.8	11.7	12.7	23.3
Total Discharge (mm)	0	6	21	26
Events (n)	0	1	6	5
TN Load (kg/ha)	NA	0.2	0.6	0.6
TN EMC (mg/L)	NA	3.3	2.6	2.3
NO_x-N Load (kg/ha)	NA	0.05	0.13	0.23
NO_x-N EMC (mg/L)	NA	0.77	0.76	0.88
NH_4-N Load (kg/ha)	NA	0.00	0.00	0.01
NH_4-N EMC (mg/L)	NA	0.01	0.04	0.02
DIN Load (kg/ha)	NA	0.05	0.13	0.23
DIN EMC (mg/L)	NA	0.78	0.80	0.90
TP Load (kg/ha)	NA	0.06	0.07	0.19
TP EMC (mg/L)	NA	0.99	0.29	0.79
DIP Load (kg/ha)	NA	0.05	0.04	0.15
DIP EMC (mg/L)	NA	0.79	0.17	0.66
TSS Load (kg/ha)	NA	46	20	20
TSS EMC (mg/L)	NA	NA	95	49

Plot Scale

Seasonal trends in nutrient and sediment loads were observed within the three pasture types, in particular for grass and butterfly pea (Table 3.8). All statistically ($P<0.05$) different results had higher loads in the dry than wet season, except for dissolved inorganic phosphorus in the Burdekin Basin which was higher in the wet season.

Higher loads of total nitrogen were exported in the dry season from all three pasture types (Figure 3.4). Low loads were observed for nitrogen species in the dry season from grass and butterfly pea pastures; however, leucaena pastures consistently generated higher loads in the dry season (Figures 3.5 to 3.7).

Total phosphorus loads exported in runoff were similar between pasture types and basins; however, loads were considerably higher in the dry than wet season for all three pasture types (Figure 3.8). Although dissolved inorganic phosphorus loads for grass and butterfly pea did not vary much within basins for each season, loads were higher in the dry season at the Burnett-Mary site and conversely higher in the wet season at the Burdekin site (Figure 3.9). Dissolved inorganic phosphorus loads generated from leucaena pastures were higher in the

Burnett-Mary than Burdekin site, and were higher in the dry than wet season for the Burnett-Mary site.

Total suspended sediment loads were greater from the Burdekin than Burnett-Mary site, particularly from butterfly pea and leucaena pastures in the dry season (Figure 3.10). Loads exported from the three pasture types at the Burnett-Mary site were low. In contrast, total suspended sediment loads from the Burdekin site were considerably higher from butterfly pea than grass and leucaena pastures.

Table 3.8: Differences between dry (D) and wet (W) season loads for nutrient and sediment loads from the grass only, butterfly pea and leucaena pastures within the Burnett-Mary and Burdekin sites. ns = not significant (P>0.05), * = P<0.05, and ** = P<0.01.

Parameter	Burnett-Mary			Burdekin		
	Grass	Butterfly Pea	Leucaena	Grass	Butterfly Pea	Leucaena
TKN	P=0.008 (**) D>W	P=0.082 ns	P=0.033 (*) D>W	P=0.088 ns	P=0.006 (**) D>W	P=0.061 ns
NO_x-N	P=0.004 (**) D>W	P=0.022 (*) D>W	P=0.111 ns	P=0.077 ns	P=0.108 ns	P=0.071 ns
NH_4-N	P=0.013 (*) D>W	P=0.017 (*) D>W	P=0.107 ns	P=0.906 ns	P=0.887 ns	P=0.329 ns
DIN	P=0.013 (*) D>W	P=0.018 (*) D>W	P=0.108 ns	P=0.098 ns	P=0.152 ns	P=0.286 ns
TKP	P=0.008 (**) D>W	P=0.066 ns	P=0.025 (*) D>W	P=0.120 ns	P=0.002 (**) D>W	P=0.044 (*) D>W
DIP	P=0.008 (**) D>W	P=0.095 ns	P=0.005 (**) D>W	P=0.011 (*) W>D	P=0.032 (*) W>D	P=0.329 ns
TSS	P=0.084 ns	P=0.333 ns	P=0.498 ns	P=0.125 ns	P=0.016 (*) D>W	P=0.120 ns

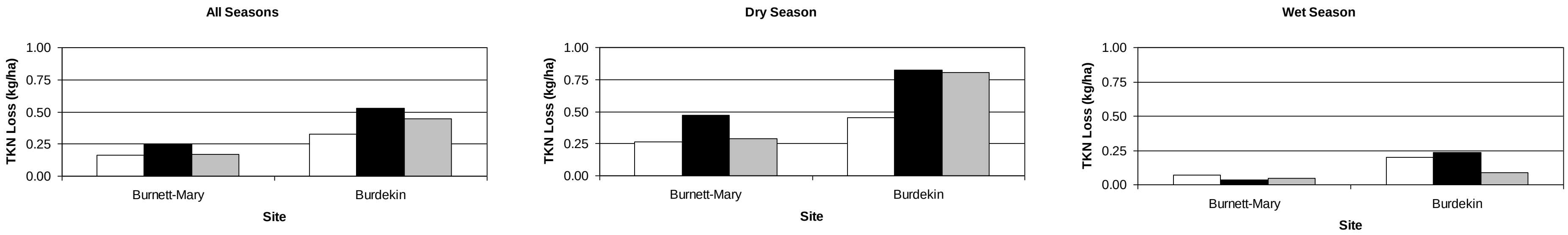

Figure 3.4: Mean total Kjeldahl nitrogen loads lost in runoff water from the grass only (white), butterfly pea (black), and leucaena pastures (grey) in the dry and wet seasons using rainfall simulation activities at the plot scale.

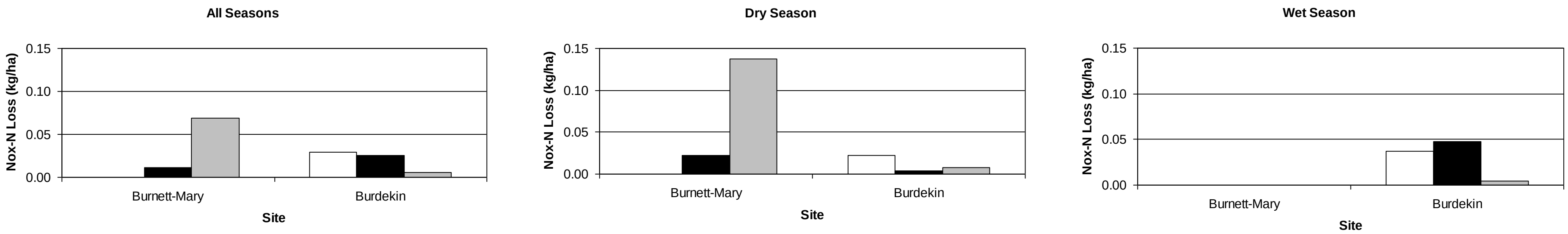

Figure 3.5: Mean oxidised nitrogen load lost in runoff water from the grass only (white), butterfly pea (black), and leucaena pastures (grey) in the dry and wet seasons using rainfall simulation activities at the plot scale.

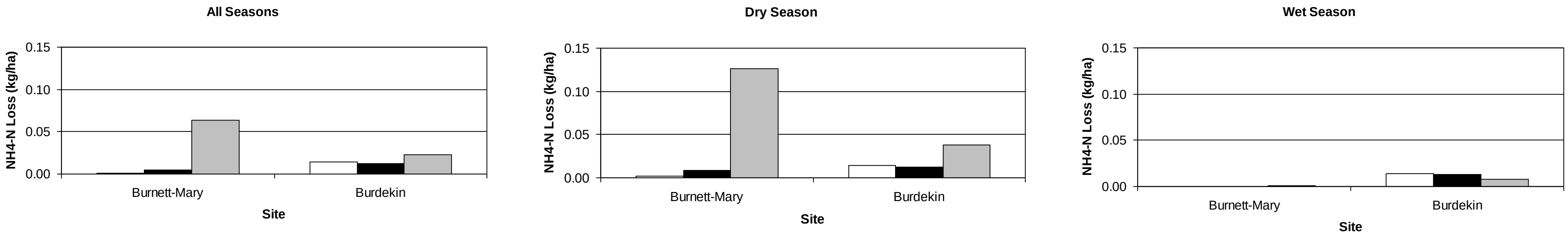

Figure 3.6: Mean ammonium-nitrogen load lost in runoff water from the grass only (white), butterfly pea (black), and leucaena pastures (grey) in the dry and wet seasons using rainfall simulation activities at the plot scale.

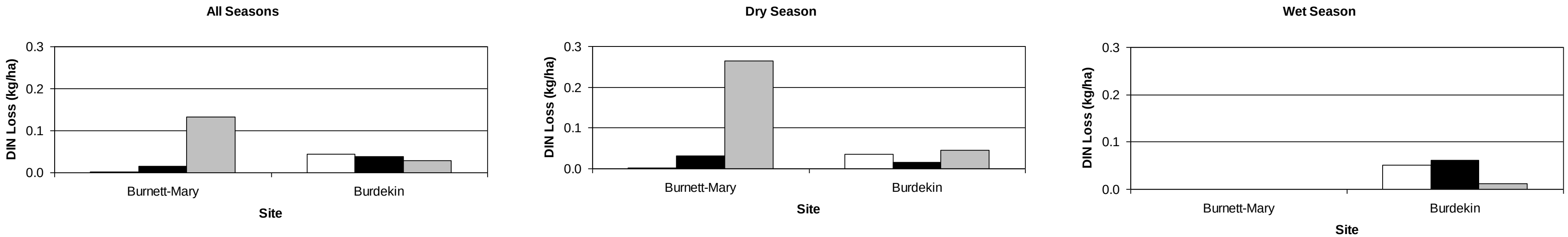

Figure 3.7: Mean dissolved inorganic nitrogen load lost in runoff water from the grass only (white), butterfly pea (black), and leucaena pastures (grey) in the dry and wet seasons using rainfall simulation activities at the plot scale.

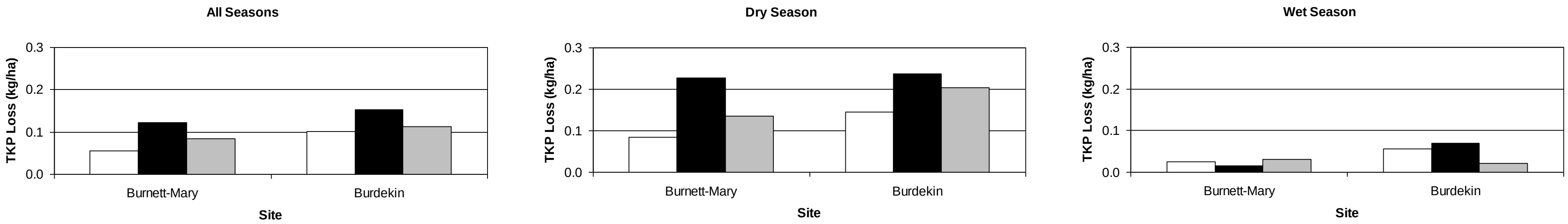

Figure 3.8: Mean total Kjeldahl phosphorus load lost in runoff water from the grass only (white), butterfly pea (black), and leucaena pastures (grey) in the dry and wet seasons using rainfall simulation activities at the plot scale.

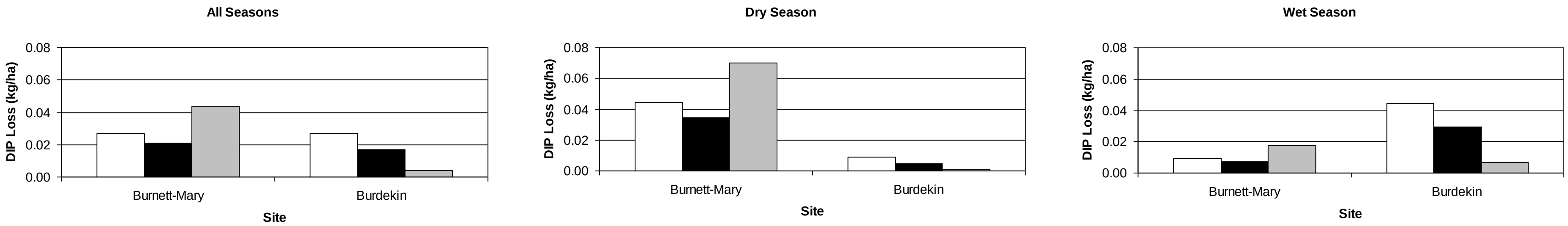

Figure 3.9: Mean dissolved inorganic phosphorus load lost in runoff water from the grass only (white), butterfly pea (black), and leucaena pastures (grey) in the dry and wet seasons using rainfall simulation activities at the plot scale.

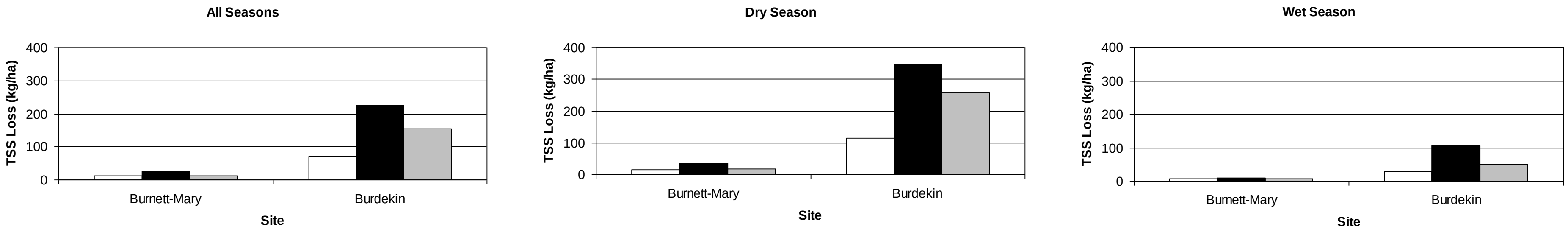

Figure 3.10: Mean total suspended solid load lost in runoff water from the grass only (white), butterfly pea (black), and leucaena pastures (grey) in the dry and wet seasons using rainfall simulation activities at the plot scale.

Discussion

Paddock Scale

Runoff water quality from pastures in the Brigalow Belt Bioregion of central Queensland varied between years and legume species. High EMCs did not necessarily equate to high loads. Typically, nitrogen (total and species) and total suspended sediment loads and EMCs from grass and leucaena pastures were lower than or equal to brigalow scrub, whilst phosphorus (total and species) loads were typically equal to or higher than brigalow scrub. Nutrient and sediment loads from young (<5 year old) butterfly pea ley pasture equaled or exceeded all other catchments, and EMCs from butterfly pea also exceeded those from grass and leucaena pasture. However, nitrogen (total and species) and total suspended sediment EMCs from brigalow scrub were greater than all pasture types.

Catchments with butterfly pea may pose a greater threat to water quality in the GBR than grass pastures with or without leucaena, due to higher nitrogen loads in runoff water. In the 2010 hydrological year, the higher nitrogen and sediment loads from the butterfly pea catchment can be partly attributed to the short time since planting; that is, the catchment had a greater risk of nutrient and sediment loss due to recent soil disturbance and reduced ground cover. However, a similar trend was also observed in 2011 with higher loads of total, oxidised and ammonium-nitrogen in runoff compared with the other catchments, despite good establishment of the butterfly pea pasture.

Whilst this data suggests that butterfly pea ley pasture poses a risk to water quality, comparison of EMC data with that generated from prior long-term cropping in the same catchment (Chapter 4) shows little change in nutrient EMCs. What is clearly evident is a major decline in total suspended sediment from an EMC of 809 mg/L from cropping to an EMC of 416 mg/L from butterfly pea. Increased cover levels and perennial vegetation when compared with a fallow period in a cropping system are the likely drivers of this reduction (Silburn *et al.* 2007). This suggests that butterfly pea ley pasture may pose no greater risk to water quality than the cropping it replaced, and may provide benefits to water quality by reducing sediment.

The balance between grass and legume in an improved pasture system is also likely to influence runoff water quality. As previously stated, high nutrient and sediment loads from newly established and young butterfly pea may be attributed to disturbance and low cover. Over time, as soil fertility increases, the balance between grass and legume in the pasture fluctuates (Jones *et al.* 1997; Orr *et al.* 2010). Where legume based ley pastures are planted as a monoculture or predominant species, it is expected that over time, grasses will increase in proportion as a result of the increased soil fertility. As this new system approaches equilibrium, it is likely that the high nutrient and sediment loads from the butterfly pea will decrease to become more like a grass monoculture pasture. Given the age of the leucaena pasture in this study, it is likely that the similarity between its nutrient and sediment loads and those of the grass only pasture can be attributed to this dynamic. Whilst the loads and EMCs of nutrients and sediment in runoff from grass and leucaena pastures are similar, it should be noted that dissolved inorganic nitrogen loads and EMCs from leucaena were consistently higher than those from grass only pasture. If this can be attributed to the legume component, then this finding has possible implications for explaining the high levels of anthropogenic dissolved inorganic nitrogen found in runoff from the Burdekin Basin (Waterhouse *et al.* 2012), given the extensive areas of pasture improvement undertaken over many decades using leguminous *Stylosanthes spp*. (McCowan *et al*. 1977; Noble *et al*. 2000; Ash *et al*. 2002).

During 2010, all catchments had an above average number of runoff events and total discharge. This trend was repeated in 2011 for all pasture land uses. Despite only having an average number of runoff events, brigalow scrub also had above average runoff. During 2012, the grass only and leucaena pastures both had an above average number of runoff events, similar to previous seasons; however, the butterfly pea ley pasture only had one runoff event whilst the brigalow scrub had no runoff. Although some land uses had an above average number of runoff events in 2012, all catchments had below average annual runoff discharge. Nil runoff from the brigalow scrub resulted in all pasture land uses having an increased load for all parameters monitored compared with their virgin condition.

The successive record wet seasons followed by an above average rainfall season, which yielded well below average runoff volume, suggests that the findings in this study represent the best case scenario for water quality. It is likely that there was a flush of plant growth in response to the high rainfall conditions, which led to increased pasture biomass and cover (Chapter 5) with a higher potential for evapotranspiration (Sims and Colloff 2012). Reduced rainfall in 2012 meant that most of the rainfall that would have become runoff in a normal run of seasons was instead captured by the increased evapotranspiration requirement of the previous year's plant growth. Although increased biomass and cover provide a benefit to water quality in their own right, pasture growth also improves soil structure resulting in greater infiltration. This together with an increased evapotranspiration requirement leads to decreased runoff (Thornton *et al.* 2007), and hence improved water quality.

A review of the literature suggests that this study is the first to provide useful insights into nutrient and sediment loss in runoff from legume based pastures in northern Australia. It is certainly the only work presenting loads and EMCs from both butterfly pea and leucaena pastures. The current target of planting leucaena over 1 Mha in Queensland within the next decade (Cooper 2013) suggests that this information is critical for managing the state's natural resources.

Plot Scale

Plot scale loads of all parameters were substantially less than those measured at the paddock scale. Comparison of total nitrogen, phosphorus and suspended sediment loads at the plot scale tended to reflect the paddock scale trend of greater loads from butterfly pea than grass or leucaena; however, the trend of grass and leucaena having similar loads is less obvious. Comparison of nutrient species between the Burdekin and Burnett-Mary Basins and pasture types was inconclusive.

The clear message from the plot scale comparison was that runoff at the end of the dry season is a higher risk to water quality than runoff at the end of the wet season. This was expected given that plant growth during the dry season is limited by both moisture and temperature, so grazing at this time will reduce cover and biomass, which can lead to increased nutrient and sediment loads. The prevalence of spring and early summer runoff from grass pasture at the paddock scale has been attributed to the same drivers (Thornton *et al*. 2007). This strongly suggests that management actions to improve water quality from grazed landscapes should focus on ensuring there is adequate cover and biomass at the end of the dry season.

Rainfall simulation plot sides were left installed in the pastures between seasonal simulations. When returning to the plots to conduct further simulations, it was noted that all legume based pasture plots suffered from dry, cracked soil around the plot edge which tended to divert substantial volumes of the applied rain into subsurface lateral flow. This behaviour was less obvious in the grass only pasture. This leakage from the plots had implications on time to runoff, proportion of rainfall as runoff, and on the movement of nutrients and sediments from the plots down the soil profile rather than in overland flow. This is in contrast to the behaviour that would be expected from the catchment under natural rainfall where similar antecedent moisture characteristics would promote a more uniform catchment runoff response.

These observations call in to question the suitability of rainfall simulation in grazed landscapes on Vertosols. Whilst plot leakage can be minimised by filling the cracks with bentonite or by wetting up the area surrounding the plot before simulating, it is likely that the best representation of catchment processes will be achieved by performing simulations immediately after significant natural rainfall has resulted in similar antecedent moisture conditions, with no cracking in the soil, at all sites.

Observations from this study on the suitability of rainfall simulations for investigating nutrient and sediment movement in grazed landscapes are in firm agreement with those of Dougherty *et al.* (2008) who wrote:

> *"Rainfall simulation on small plots provides rapid results, is an efficient way to collect data and test hypotheses, allows greater control of conditions (both treatments and rainfall/climate), and is adaptable to a range of research approaches."*
>
> *"We conclude that small-scale, high-intensity rainfall simulation provides a useful tool for studying treatment effects and processes of mobilisation in pastures, but concentration and load data should not be inferred for natural conditions at larger scales without a clear understanding of the effects of the rainfall simulation methodology on the results for the system being studied."*

In conclusion, trends observed at both the paddock and plot scales indicate that grass only and leucaena pastures pose a smaller threat to GBR water quality than ley pastures of butterfly pea, as less nitrogen and sediments are exported in runoff waters.

Chapter 4: Land Use Effect on Nitrogen, Phosphorus and Sediment Generation Rates

Summary

Trends for Model Development and Validation

- The effects of changing land use from brigalow scrub to cropping or grazing on runoff water quality at the paddock scale over 29 years are summarised in Table 4.1.

Table 4.1: Land use change effect of converting virgin brigalow scrub to cropping and grazing on key nutrient and sediment parameters for runoff water quality at the paddock scale.

	Load Change	Load Comparison	EMC Change	EMC Comparison
TKN	Cropping ↓, Grazing ↓	Cropping > Grazing	Cropping ↓, Grazing ↓	Cropping > Grazing
DIN	Cropping ↑, Grazing ↓	Cropping > Grazing	Cropping ↓, Grazing ↓	Cropping > Grazing
TP	Cropping ↑, Grazing ↑	Cropping > Grazing	Cropping ↑, Grazing ↓	Cropping > Grazing
TSS	Cropping ↑, Grazing ↑	Cropping > Grazing	Cropping ↑, Grazing ↓	Cropping > Grazing

Key Insights

- Nutrient and sediment loads from cropping are greater than loads from both grazing and the virgin brigalow landscape.

- Loads of some parameters from grazing are in fact less than those of the virgin brigalow landscape.

Management Action

- Switching to minimum and zero tillage cropping systems will have a positive impact on water quality.
- Manage grazing pressure to maintain high pasture cover and reduce sediment concentration.

Introduction

European settlement in Queensland occurred in 1824 and the landscape has since been modified for different purposes, such as agriculture and urban development. Clearing vegetation from the landscape has led to an increase in the pollutant loads that are exported in runoff water. For example, Moss *et al*. (1993) estimated that total nutrient inputs to the GBR have increased 30% since European settlement, with the rate of increase mostly since 1950. Furthermore, Kroon *et al*. (2012) estimated mean annual pollutant loads from six coastal catchments of Queensland and found that loads since European settlement in the GBR have increased 5.5 times for total suspended sediment (17,000 kt/yr), 5.7 times for total nitrogen (80,000 t/yr) and 8.9 times for total phosphorus (16,000 t/yr).

In an effort by the Queensland Government to increase production in the 1960s, broad-scale clearing occurred over most of the Fitzroy Basin and the BCS was established to investigate changes in the landscape when developed for either cropping or grazing. Past findings from the BCS have found that clearing brigalow scrub increased the frequency and magnitude of runoff, increased deep drainage, led to the leaching of soil chloride and reduced soil total nitrogen and organic carbon levels (Hunter and Cowie 1989; Cowie *et al*. 2007; Thornton *et al*. 2007; Thornton *et al*. 2010).

Under the Reef Plan, agriculture is identified as the main source of pollutants in the GBR (Department of the Premier and Cabinet 2009). Although the implementation of best management practices have improved runoff water quality at the paddock and/or property scale, total loads entering the GBR rapidly escalate when calculated at the catchment scale. For example, the Fitzroy Basin exports 4.1 Mt of total suspended sediments, 15,000 t of total nitrogen and 4,100 t of total phosphorus per year (The State of Queensland 2011). The proportion of these loads due to anthropogenic activities was 71%, 87% and 95%, respectively. These values are based on only 53% of graziers in the Fitzroy Basin maintaining land in good to very good condition and/or improving land in lesser condition (The State of Queensland 2011). Thus, there is potential to reduce total loads entering the GBR through further adoption of improved land management practices across the region.

The objective of this study was to estimate loads and EMCs of nitrogen, phosphorus and sediment in runoff from virgin brigalow scrub during the calibration phase of the BCS (1965 to 1982), and then use this information as a pre-European estimate to calculate cumulative load changes due to development of land for agriculture, specifically cropping and grazing (1984 to 2012).

Methods

Research was conducted on the BCS site near Theodore in central Queensland. This work focused on the virgin brigalow scrub (C1), cropping (C2) and grazing (C3) catchments using data from Stage I (1965 to 1982), Stage III (1984 to 2010) and Stage IV (2010 to 2012) to compare the effect of land use on nutrient and sediments loads in runoff water. Refer to 'Study Sites' in Chapter 2 for more information on each of these catchments.

Rainfall, runoff and water quality data were collected from all three catchments as described in Chapter 3. Laboratory analysis of runoff water quality samples, and nutrient and sediment load and EMC calculations were also described in Chapter 3.

Changes in nutrient and sediment loads as a result of changing land use from virgin brigalow scrub to cropping or grazing were modelled by:

Land use change effect of load = $(Q_{Obs} \times EMC_{Current}) - (Q_{Est} \times EMC_{Brigalow})$

Model parameters were defined as follows:

Q_{Obs}	=	Observed discharge from the catchment with its current land use
$EMC_{Current}$	=	Observed grand average EMC data from the catchment with its current land use
Q_{Est}	=	Estimated discharge from the catchment had it have remained virgin brigalow scrub (Thornton *et al.* 2007)
$EMC_{Brigalow}$	=	Calculated grand average EMC data from the virgin brigalow scrub catchment

Q_{Obs} was calculated from observed event based runoff from the catchments in Stages III and IV (Chapter 2). $EMC_{Current}$ was the calculated grand average EMC values from the period 2000 to 2012 (Chapter 3). Q_{Est} was calculated from the catchment calibrations developed during Stage I of the study (Chapter 2; Thornton *et al*. 2007). $EMC_{Brigalow}$ was the calculated grand average EMC values for the brigalow scrub catchment (C1) for the period 2000 to 2012 (Chapter 3). The assumption of this approach is that water quality from the three catchments in their virgin state would have been similar, and that the grand average EMC values for Catchment 1 apply to all catchments had they remained virgin brigalow scrub.

Observed loads per catchment were estimated by multiplying runoff volume by the respective catchments grand average EMC for each parameter. Predicted loads per catchment had they remained virgin brigalow scrub were modelled by multiplying predicted runoff volume for the catchment by the brigalow scrub catchment grand average EMC for each parameter. Changes in runoff water quality as a result of converting virgin brigalow scrub into agricultural land uses were then modelled by subtracting the predicted loads for the cropping and grazing catchments had they remained uncleared from their observed loads. Nutrient and sediment mean annual loads were calculated by dividing the cumulative observed load for each parameter by the number of years data; that is, 29. Mean annual land use change effects were modelled by dividing the cumulative difference in load by the number of years. Daily rainfall for the land use comparison period was collected at the BCS site and presented as a cumulative total in millimetres.

Results

Overall, mean annual nutrient and sediment loads exported in runoff water at the paddock scale were greater from cropping compared with grazing (Table 4.2). Both agricultural land uses exported less total nitrogen than if the catchments had remained brigalow scrub. The decrease in total nitrogen exported from grazing was greater than the decrease exported from cropping (Table 4.2; Figure 4.1). Land developed for grazing exported less oxidised and dissolved inorganic nitrogen than if the catchments had remained brigalow scrub; however, land developed for cropping exported more of these two parameters (Table 4.2; Figures 4.2 and 4.3).

In contrast to total nitrogen, ammonium-nitrogen loads increased as a result of land use change (Table 4.2; Figure 4.4). Total and dissolved inorganic phosphorus showed similar trends to ammonium-nitrogen with cropping exporting greater loads compared with grazing, but with both land use types exporting more than if they had remained uncleared (Table 4.2; Figures 4.5 and 4.6). Furthermore, total suspended sediment loads due to land use change were greater than if the catchments had remained uncleared; however, the loads exported from cropping were considerably higher than from grazing (Table 4.2; Figure 4.7).

Table 4.2: Mean annual loads and land use change effects for the cropping and grazing catchments; data for the period 1984 to 2012.

	Brigalow		Cropping			Grazing	
Parameter	Mean Annual EMC (mg/L)	Mean Annual Load (kg/ha)	Mean Annual EMC (mg/L)	Mean Annual Land Use Change Effect (kg/ha)	Mean Annual Load (kg/ha)	Mean Annual EMC (mg/L)	Mean Annual Land Use Change Effect (kg/ha)
TN	13.89	3.58	5.00	-0.15	1.24	2.25	-1.56
NO_x-N	4.99	1.60	2.24	0.29	0.15	0.26	-0.97
NH_4-N	0.06	0.06	0.08	0.04	0.02	0.04	0.01
DIN	5.05	1.66	2.32	0.33	0.17	0.30	-0.96
TP	0.38	0.64	0.90	0.54	0.17	0.31	0.09
DIP	0.14	0.28	0.39	0.24	0.10	0.18	0.07
TSS	362	522	730	427	103	187	22

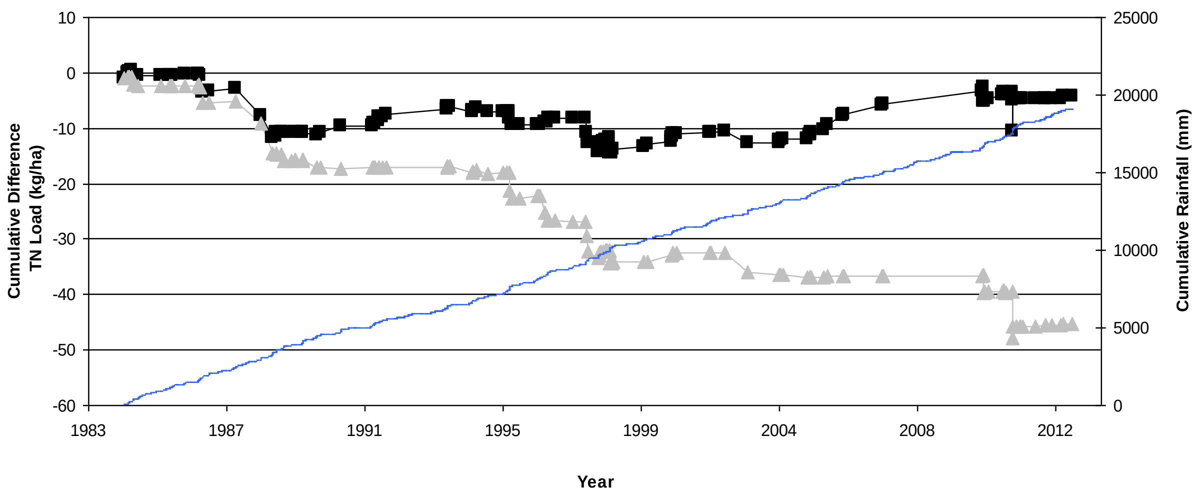

Figure 4.1: Cumulative difference between observed and predicted loads of total nitrogen from the cropping (square) and grazing catchments (triangle) as a result of changing land use from virgin brigalow scrub, and cumulative total rainfall (line) for the period 1984 to 2012.

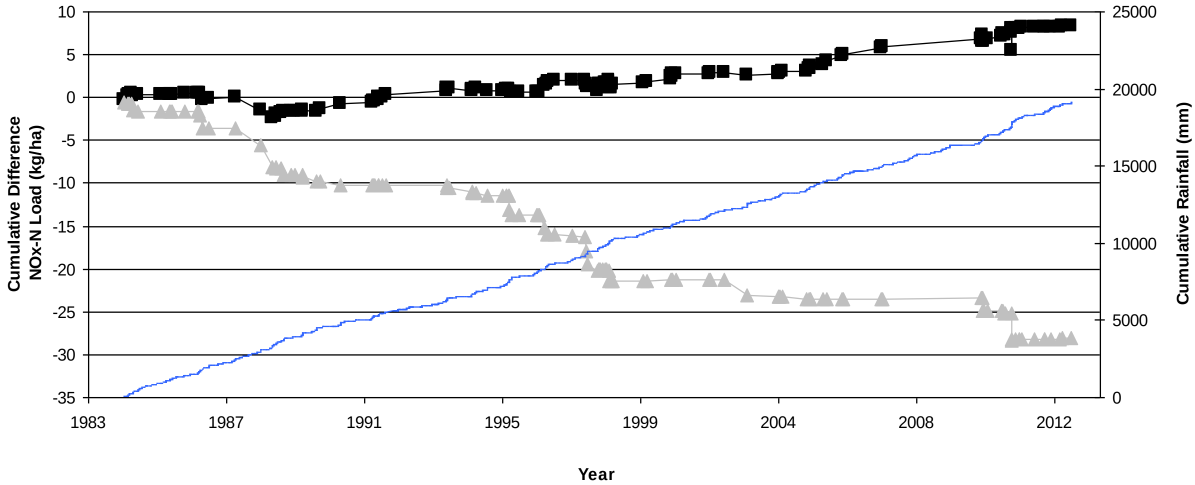

Figure 4.2: Cumulative difference between observed and predicted loads of oxidised nitrogen from the cropping (square) and grazing catchments (triangle) as a result of changing land use from virgin brigalow scrub, and cumulative total rainfall (line) for the period 1984 to 2012.

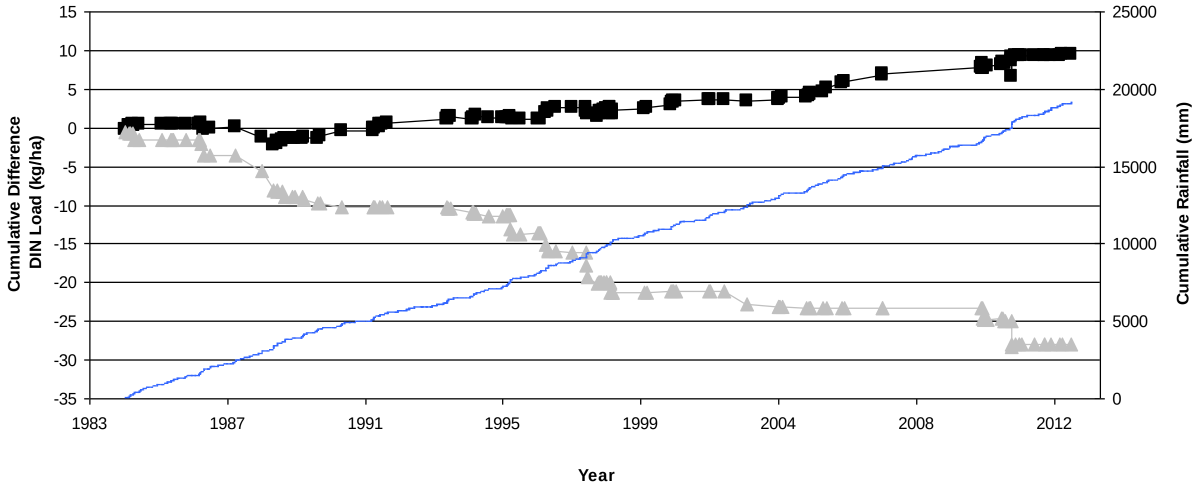

Figure 4.3: Cumulative difference between observed and predicted loads of dissolved inorganic nitrogen from the cropping (square) and grazing catchments (triangle) as a result of changing land use from virgin brigalow scrub, and cumulative total rainfall (line) for the period 1984 to 2012.

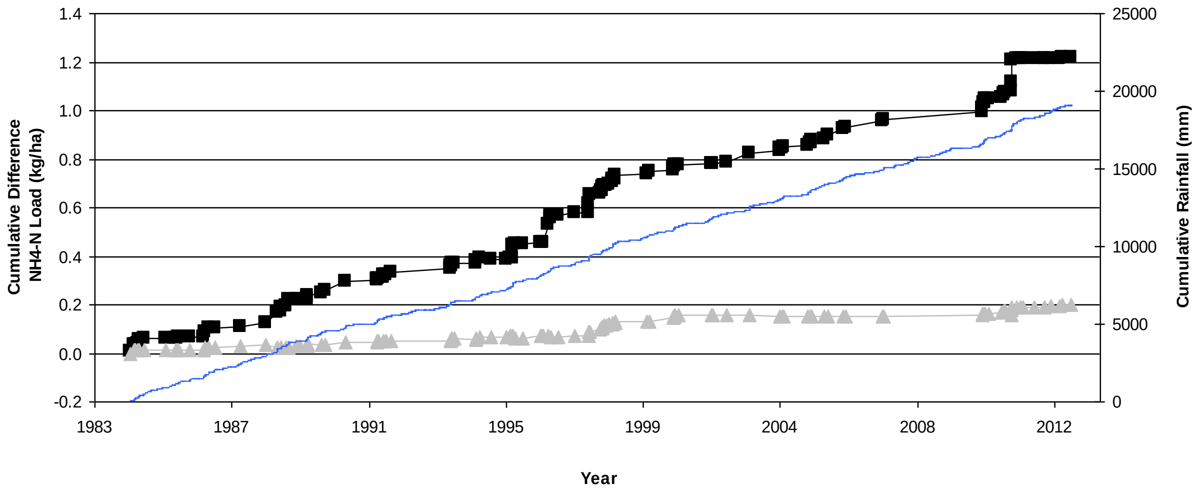

Figure 4.4: Cumulative difference between observed and predicted loads of ammonium-nitrogen from the cropping (square) and grazing catchments (triangle) as a result of changing land use from virgin brigalow scrub, and cumulative total rainfall (line) for the period 1984 to 2012.

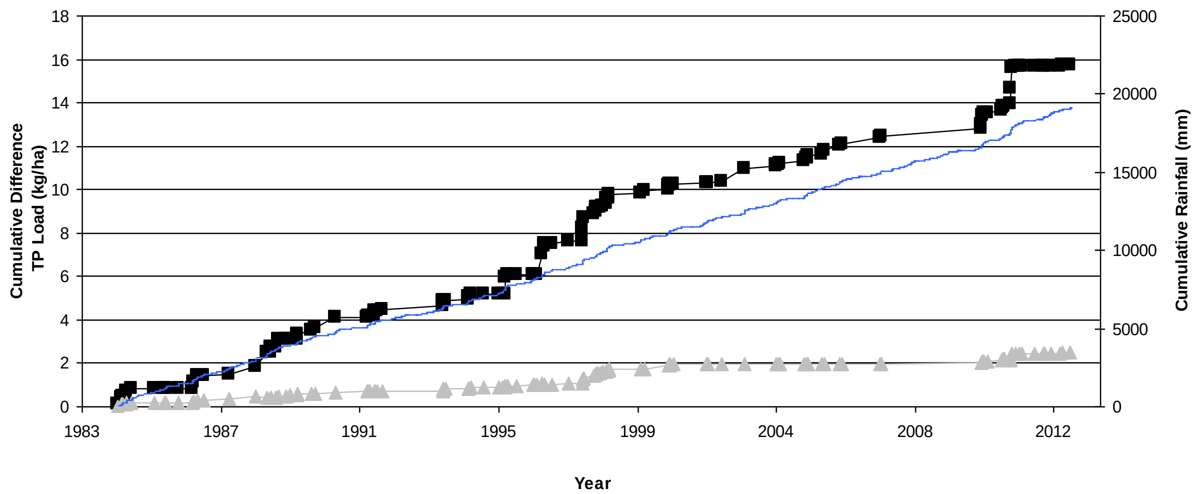

Figure 4.5: Cumulative difference between observed and predicted loads of total phosphorus from the cropping (square) and grazing catchments (triangle) as a result of changing land use from virgin brigalow scrub, and cumulative total rainfall (line) for the period 1984 to 2012.

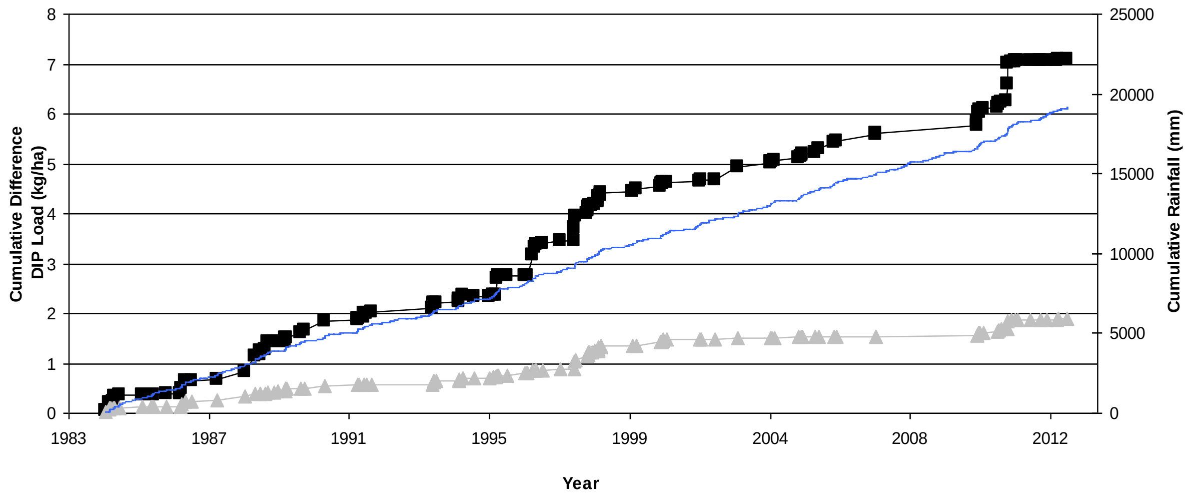

Figure 4.6: Cumulative difference between observed and predicted loads of dissolved inorganic phosphorus from the cropping (square) and grazing catchments (triangle) as a result of changing land use from virgin brigalow scrub, and cumulative total rainfall (line) for the period 1984 to 2012.

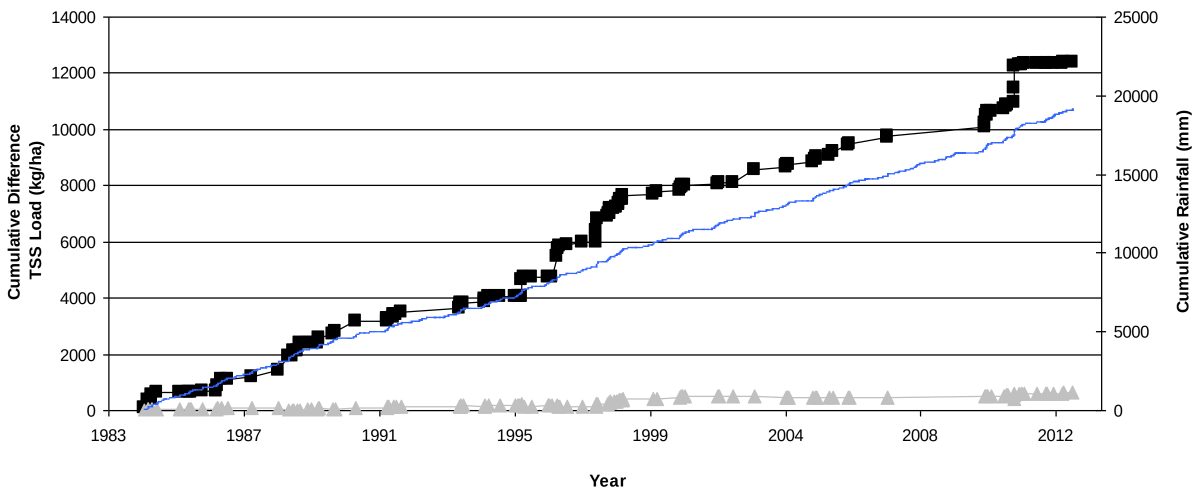

Figure 4.7: Cumulative difference between observed and predicted loads of total suspended solids from the cropping (square) and grazing catchments (triangle) as a result of changing land use from virgin brigalow scrub, and cumulative total rainfall (line) for the period 1984 to 2012.

Discussion

Modelling of land use impacts on water quality has shown higher loads of nitrogen (total and species), phosphorus (total and species) and total suspended sediments in runoff from cropping compared with grazing. Changing land use from brigalow scrub to cropping decreased total nitrogen load, but increased loads of nitrogen species, phosphorus (total and species) and total suspended sediments. Changing land use from virgin brigalow scrub to grazing not only decreased total nitrogen load, but also oxidised and dissolved inorganic nitrogen load. However, loads of ammonium-nitrogen, phosphorus (total and species) and total suspended sediment increased. No changes in water quality from the cropping catchment could be detected as a result of changed management practices during the 1990s.

An annual load of 522 kg/ha of sediment in runoff from cropping in this study is approximately half of the 1,200 kg/ha annual load reported in a similar study of zero-till dryland cropping on Vertosols in the Fitzroy Basin (Murphy *et al.* 2011). Both of these loads can be considered low when contrasted to the 4,010 kg/ha load from a similar cropping study on Vertosols in the Fitzroy Basin managed with conventional cultivation (Carroll *et al.* 1997). Furthermore, all of these loads are less than the 8,480 kg/ha reported from cropping on shallow, more erodible sodic duplex soils on greater slopes in a nearby area of the Fitzroy Basin (Stevens *et al.* 2006). A summary of sediment loads from dryland cropping in 21 Australian locations gives a mean concentration of 2,501 mg/L with 10th and 90th percentiles of 162 mg/L and 5,339 mg/L, respectively (Bartley *et al.* 2012). Given the effect of management practices noted in the Fitzroy Basin, the loads estimated in this study correspond well to other local data. Furthermore, this data suggests that transitioning to minimum and zero tillage cropping systems will have a positive impact on water quality.

Literature on nutrient loads and concentrations from cropping systems is sparse compared with sediment loads. Of the 21 dryland cropping sites considered in Bartley *et al.* (2012), none were cited as measuring nutrient concentrations and being representative of more than 90% of the catchment land use. Comparisons with other studies are also confounded by the application of fertiliser which is prone to movement in runoff. For example, total nitrogen load and EMC from cropping in this study is less than that measured from other cropping studies in the Fitzroy Basin which had applied fertiliser (Stevens *et al.* 2006; Murphy *et al.* 2011); but is slightly higher than the mean of 17 sites Australia wide (Bartley *et al.* 2012), some of which would be located on less fertile soils from which you would expect lower nutrient loads. Total phosphorus load and EMC from cropping in this study is also less than that measured for another study in the Fitzroy Basin (Stevens *et al.* 2006), but is equal to the mean of the 17 sites Australia wide (Bartley *et al.* 2012).

A review of grazing studies throughout northern Australia found that sediment concentrations in runoff were typically in the range of 100 to 300 mg/L, but could be substantially lower for sites with high cover and low slope (Silburn *et al.* in press). Although the total suspended sediment EMC of 187 mg/L from grazing in this study is contained within this range, a higher EMC of 500 mg/L from a similar grazed landscape in the Fitzroy Basin has been reported (Murphy *et al.* 2011). The difference between studies is probably associated with pasture cover levels which were lower (>50%) in Murphy *et al.* (2011) compared with consistently higher levels in this study (>80%).

As for cropping systems, data for nutrient runoff from grazing is sparse, particularly for nutrient species as opposed to total nitrogen and phosphorus (Silburn *et al.* in press). Within Queensland, total nitrogen and phosphorus loads appear to be reasonably consistent around 2.0 mg/L (ranging from 0.4 to 9.6 mg/L) and 0.5 mg/L (0.4 to 1.5 mg/L), respectively. However, lower total phosphorus concentrations (0.04 mg/L) were reported in areas of less sediment generation, such as low slopes. Data from streams in the Northern Territory have similar loads for total nitrogen, but lower loads for total phosphorus. Mean annual loads were generally 1 to 2 kg/ha/yr for total nitrogen and 0.2 to 0.3 kg/ha/yr for total phosphorus, except where erosion rates were considerably lower or higher due to specific landscape processes, such as high cover or bare scalds (Silburn *et al.* in press). Total nitrogen load and EMC for this study are comparable to these values. Load of total phosphorus is also comparable; however, EMC in this study is 40% lower, which is similar to that of other well managed grazed landscapes in the Fitzroy Basin (Murphy *et al.* 2011; Silburn *et al.* in press).

The development of brigalow lands for cropping or grazing in central Queensland has had significant impacts on the landscape. Some effects can be attributed primarily to land use change, such as the doubling of runoff when virgin brigalow scrub is developed for either cropping or grazing (Thornton *et al*. 2007). Other effects, such as increased peak runoff rate, are more pronounced under cropping than grazing (Thornton 2012); whilst other impacts, such as increased deep drainage, are confined to cropping rather than grazing, which continues to mimic the virgin brigalow landscape in its pre-European condition (Silburn *et al*. 2009). This study suggests that nutrient and sediment loads from cropping are greater than loads from both grazing and the virgin brigalow landscape. In contrast, loads of some parameters from grazing are in fact less than those of the virgin brigalow landscape, and where loads are greater, the increase is less than 30% of the increase under cropping.

Chapter 5: Seasonal Nitrogen and Phosphorus Cycling in Soil and Pasture

Summary

Trends for Model Development and Validation

- Effect of pasture type on soil and plant nutrient concentrations over two years are summarised in Table 5.1.

Table 5.1: Comparison of pasture type effects on key soil and plant nutrient parameters. Br = virgin brigalow scrub, BP = ley pasture of butterfly pea, G = grass only pasture, L = leucaena pasture, LL = leucaena leaves.

Parameter	Pasture Type Comparison
Soil TKN	Br > Gr > L > BP
Soil NO_3-N	Br > BP = Gr = L
Soil Colwell P	BP > Br > Gr > L
Plant TKN	LL > BP > Gr = L
Plant TKP	LL = BP > Gr > L

Risks

- Risk is consistent throughout the year due to high variability in concentrations and the slow response of landscapes to changed nutrient cycling.

Key Insights

- Soil total and nitrate-nitrogen were higher in virgin brigalow scrub than the three pasture types.
- Soil available phosphorus was higher in butterfly pea than in grass only and leucaena pastures.
- Plant total nitrogen and phosphorus were higher in leucaena leaves and butterfly pea than grass only and the grass component of leucaena pastures.

Management Action

- The transfer of nitrogen from legume to grass appears to be inefficient at the BCS site; however, the application of fertilisers to increase plant biomass and nitrogen uptake may also increase nutrient loads in runoff if surplus to system requirements.
- Soil and pasture management should focus on minimising runoff, rather than manipulation of the natural nutrient cycle to reduce risks to water quality.

Introduction

Declining soil fertility is a major threat to future productivity from cereal cropping and monoculture grass pastures in central Queensland. This is due to the continuous cropping or grazing of land over many years with little or no input of nutrients via fertiliser applications or legume crop rotations (Armstrong *et al*. 1997; Collins and Grundy 2005).

Leguminous pasture species are able to restore soil fertility in degraded lands by fixing atmospheric nitrogen, which then becomes available in the soil for other plants to use as the legume decomposes. This process can be achieved using either short-term ley pastures, such as butterfly pea, for 3 to 5 years, or long-term permanent pastures, such as leucaena, which can be used for 25 or more years (Collins and Grundy 2005). For example, a soil restoration trial using butterfly pea in Mexico found increased levels of both total nitrogen (0.30 to 0.48%) and phosphorus (2.0 to 10.2 mg/kg) over 180 days during the summer growing season (Alderete-Chavez *et al*. 2011).

Beef production on leguminous pastures in central Queensland is becoming more common, as there is an increase in the availability of data which demonstrates greater nutritive value of forage and consequently greater live weight gains of cattle from leucaena versus grass only pastures (Collins and Grundy 2005; Dalzell *et al*. 2006; Thornton and Buck 2011; Elledge and Thornton 2012). However, although it is known that leucaena and butterfly pea are summer growing, there is limited data available which demonstrates nutrient concentrations of pastures with and without legumes in addition to their link with soil fertility throughout the year.

The objective of this study was to monitor soil and pasture nutrients on a quarterly basis over two years to develop an understanding of the seasonal changes in nitrogen and phosphorus concentrations in three pasture types.

Methods

Research was conducted at the BCS site near Theodore in central Queensland using Stage IV data from all four catchments: (1) virgin brigalow scrub; (2) grass only pasture; (3) butterfly pea ley pasture; and (4) leucaena pasture. Refer to 'Study Sites' in Chapter 2 for more information. The BCS is an ideal study site for assessing soil restoration via the use of leguminous pasture species on degraded cropping land and monoculture grass pastures, as there have not been any fertiliser inputs or legume species used since its commencement in 1965.

Three permanent monitoring sites 20 x 20 m are located within each of the four catchments, and each monitoring site is bounded by a series of 1 x 1 m cells. For each sampling period, six cells were randomly selected from the bounded area adjacent to each monitoring site for soil and pasture sampling. Six soil samples consisting of two bulked cores 5 cm in diameter and 0 to 10 cm depth from the soil surface were collected from these cells; that is, 72 samples per survey period. All nutrient analyses were completed by the Environmental Resource Sciences Chemistry Centre in Dutton Park, Queensland. Soil samples were analysed for total Kjeldahl nitrogen (method 7A2), nitrate-nitrogen (method 7C2), ammonium-nitrogen (method 7C2) and Colwell phosphorus (method 9B2) based on methods described in Rayment and Higginson (1992).

Six pasture samples consisting of all standing biomass were collected from the same 1 x 1 m cells used for soil sampling, but only within the three pasture land use catchments; that is, 54 samples per survey period. Pasture samples were used to calculate biomass (kg/ha) on a dry weight basis and were primarily comprised of butterfly pea and grass for Catchment 2, buffel grass for Catchment 3, and young leucaena plants between hedgerows and grass for Catchment 4. In addition, the leaves from two leucaena stems were collected at each monitoring site within the leucaena catchment (3 samples per survey period) for all sampling periods except early September and late November 2011 (data not collected). Nutrient analyses for total Kjeldahl nitrogen and phosphorus in pasture were also conducted by Environmental Resource Sciences Chemistry Centre.

Data on soil and pasture condition were sampled at approximately three month intervals from May 2011 to March 2013 to capture the break in summer, autumn, winter and spring months; summer months in Australia being December to February.

Values in laboratory reports that were at the practical quantitation limit (PQL) were changed to 50% of the reported value to obtain a numeric value that could be used for analyses. That is, for example, <2 was changed to 1. Data were analysed using GenStat (version 14.2; www.vsni.co.uk) to assess catchment and season main and interaction effects. Inspection of the residual plots for soil data indicated that transformations were necessary to stabilise the variance prior to analyses; total nitrogen was arcsine transformed, and nitrate-nitrogen, ammonium-nitrogen and phosphorus were log transformed. Soil data were then analysed using an analysis of variance. Inspection of the residual plots for pasture data indicated that transformations were also required; total nitrogen and phosphorus were arcsine transformed and biomass was square root transformed. Restricted maximum likelihood analyses were used for pasture data to account for the imbalance that resulted from quadrats with no yield. Missing values in total nitrogen and phosphorus data where there was no plant yield were not estimated in analyses. If the interaction between pasture type and catchment was not significant (P<0.05) in the restricted maximum likelihood analyses, it was removed from the model so the main effects could be tested. A combined analysis of catchments for both soil and pasture data was also performed as a restricted maximum likelihood meta-analysis to accommodate for the different catchment variances The comparison of levels within the main effects was done using a 5% least significant difference (P<0.05).

Results

Soil

Soil total nitrogen concentrations were highest in brigalow scrub, and then declined in order from grass only pasture, leucaena pasture and then butterfly pea ley pasture (Table 5.2; Figure 5.1). Despite this consistent trend across all sampling periods, no statistical difference between catchments was detected (P>0.10). However, concentrations in spring were statistically lower than in the other seasons (P<0.05).

Soil nitrate-nitrogen concentrations were considerably higher in the brigalow scrub catchment (Table 5.2); however, the grand mean value may have been skewed by the observed peaks for the November 2011 and March 2012 sampling periods (Figure 5.2). Due to the low practical quantification limit reported for 49% of the data, catchment and seasonal differences were not calculated.

Overall, concentrations of soil ammonium-nitrogen were similar among the four catchments with no statistical differences detected (P>0.10) (Table 5.2). However, high variability within the data was observed (Figure 5.3). Seasonal differences were detected with concentrations in summer statistically greater than in autumn and spring (P<0.001).

Soil Colwell phosphorus concentrations were highest in the butterfly pea ley pasture, and then declined in order from brigalow scrub, grass only pasture, and then leucaena pasture (Table 5.2). Again, high variability within the data was observed (Figure 5.4) and no difference could be detected between catchments (P<0.10) or seasons (P<0.10).

Table 5.2: Soil nutrient grand means (± standard errors) for the virgin brigalow scrub, butterfly pea ley pasture, grass only pasture, and leucaena pasture; data for the period 2011 to 2013.

Catchment	TKN (%)	NO_3-N (mg/kg)	NH_4-N (mg/kg)	Colwell P (mg/kg)
Brigalow Scrub	0.190 (0.006)	5.9 (0.9)	3.3 (0.2)	13.2 (0.7)
Butterfly Pea	0.092 (0.001)	2.3 (0.2)	3.3 (0.4)	16.5 (0.6)
Grass	0.144 (0.003)	2.1 (0.1)	3.7 (0.2)	8.9 (0.3)
Leucaena	0.114 (0.002)	1.9 (0.1)	3.2 (0.2)	6.2 (0.3)

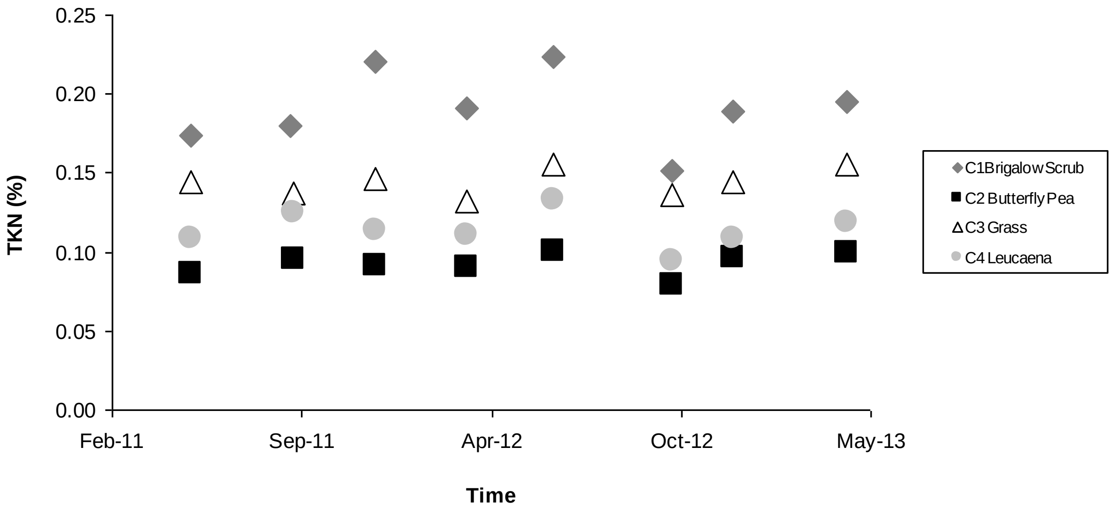

Figure 5.1: Mean concentration of total Kjeldahl nitrogen in soil 0 to 10 cm from the ground surface in three pasture land use types compared with virgin brigalow scrub.

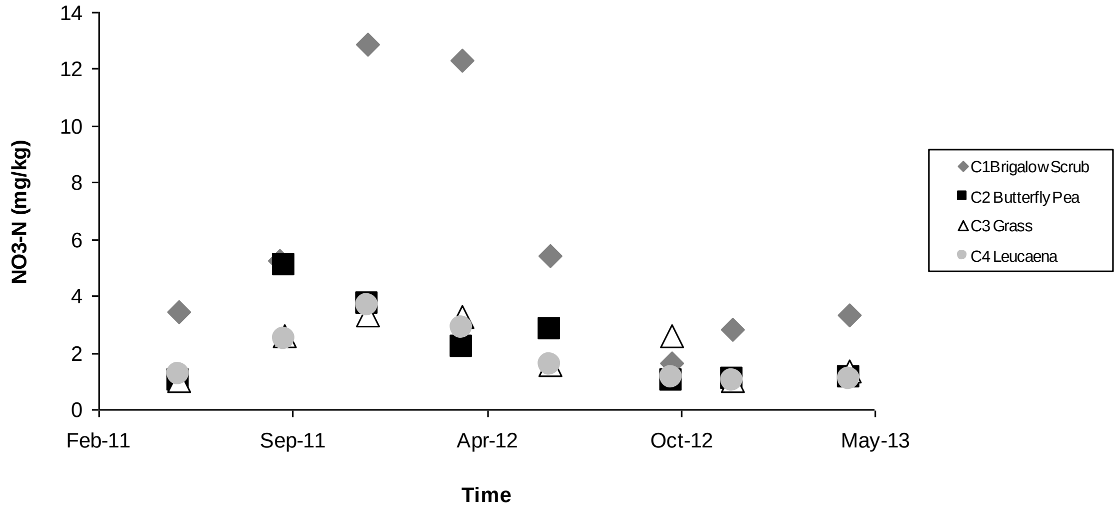

Figure 5.2: Mean concentration of nitrate-nitrogen in soil 0 to 10 cm from the ground surface in three pasture land use types compared with virgin brigalow scrub.

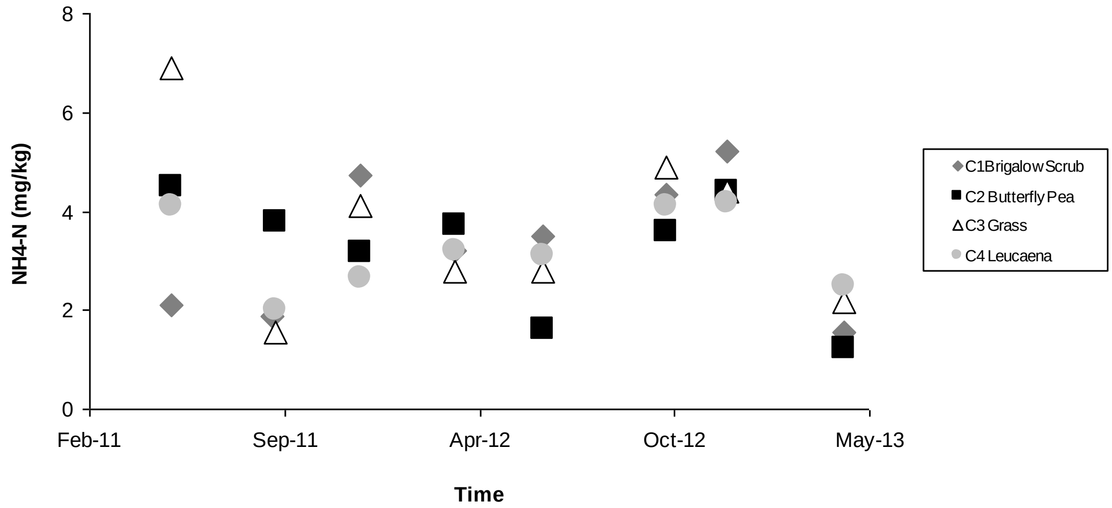

Figure 5.3: Mean concentration of ammonium-nitrogen in soil 0 to 10 cm from the ground surface in three pasture land use types compared with virgin brigalow scrub.

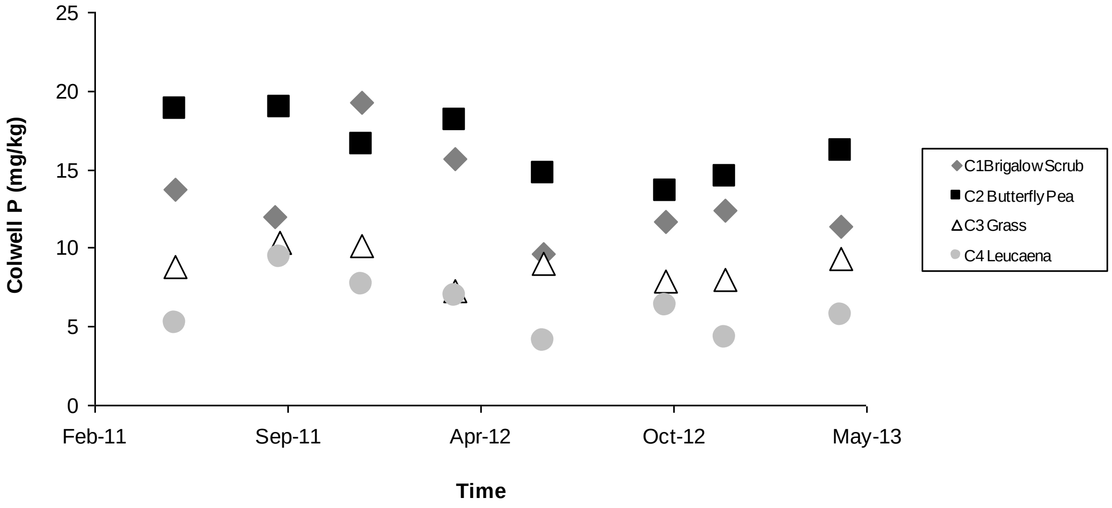

Figure 5.4: Mean concentration of Colwell phosphorus in soil 0 to 10 cm from the ground surface in three pasture land use types compared with virgin brigalow scrub.

Pasture

Plant total nitrogen concentrations were consistently higher in leucaena leaves than in the pasture of all catchments for all the seasons sampled (Table 5.3; Figure 5.5). There was a catchment and season interaction effect detected ($P<0.05$). Differences between the three pasture types were also observed with concentrations in the butterfly pea ley pasture more variable and overall higher than the grass dominated samples from both the grass only and leucaena catchments ($P<0.01$). There was also evidence of a seasonal effect with concentrations highest in autumn and lowest in spring ($P<0.001$).

Plant total phosphorus concentrations were highest in leucaena leaves and lowest in the predominantly grass component of the leucaena pasture (Table 5.3; Figure 5.6), but results varied within each pasture type. There was a catchment and season interaction effect detected ($P<0.01$). Concentrations were statistically higher in autumn from leucaena pasture ($P<0.01$), and in both winter and autumn from butterfly pea pasture ($P<0.01$). However, no evidence of a seasonal effect was detected in grass only pasture ($P>0.10$).

The grass only pasture had higher biomass than the butterfly pea and leucaena pastures ($P<0.01$) (Table 5.3; Figure 5.7). Seasonal trends were also evident with greater biomass in autumn and winter compared with spring and summer ($P<0.001$). Although there was high variability within all three pasture types, temporal variation may be partly biased by grazing pressure, as biomass tended to increase during periods of cattle removal.

Table 5.3: Pasture nutrient and biomass grand means (± standard errors) for the butterfly pea ley pasture, grass only pasture, leucaena pasture, and leaves from leucaena stems; data for the period 2011 to 2013.

Catchment	TKN (%)	TKP (%)	Biomass (kg/ha)
Butterfly Pea	1.36 (0.06)	0.157 (0.006)	1205 (105)
Grass	0.60 (0.02)	0.127 (0.003)	4601 (183)
Leucaena	0.60 (0.03)	0.092 (0.004)	1407 (95)
Leucaena Leaves	3.32 (0.13)	0.161 (0.012)	1021 (191)

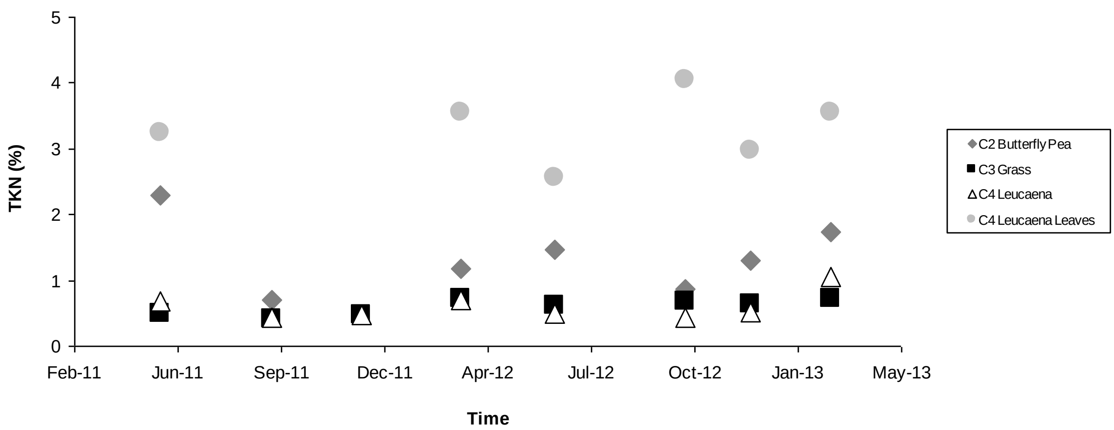

Figure 5.5: Mean concentration of total Kjeldahl nitrogen in plants from the three pasture land use types and leaves from leucaena stems.

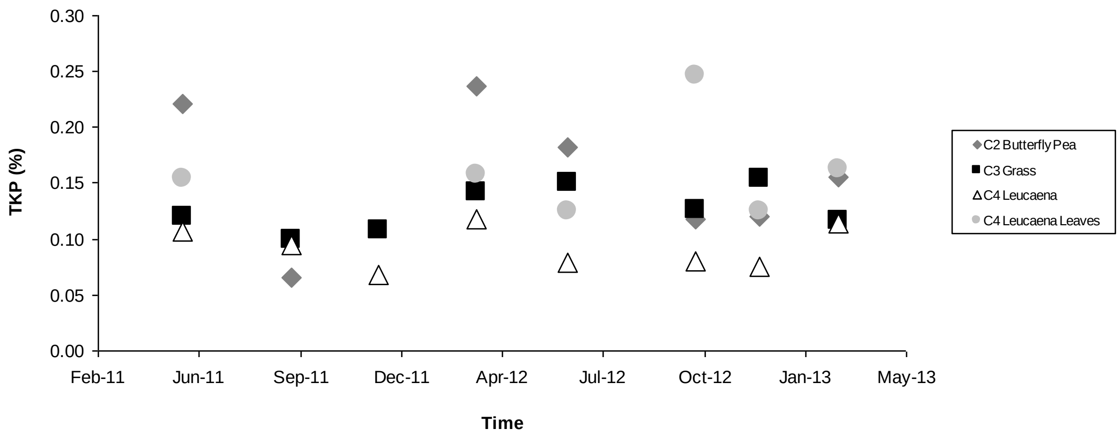

Figure 5.6: Mean concentration of total Kjeldahl phosphorus in plants from the three pasture land use types and leaves from leucaena stems.

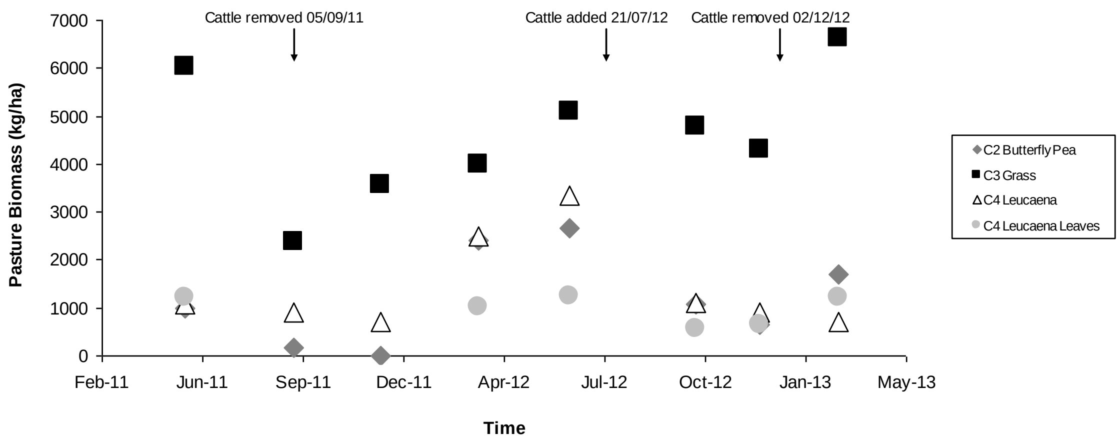

Figure 5.7: Mean total plant biomass from the three pasture land use types and leaves from leucaena stems. Cattle removed from all catchments in September 2011, but only added and removed from the leucaena pasture during 2012.

Discussion

Leguminous pasture species are able to greatly contribute to productive and sustainable agricultural systems in the tropics of Australia (Boddey *et al*. 1997; Dalzell *et al*. 2006; Shelton and Dalzell 2007). This is because legumes can provide companion pasture grasses with nitrogen either directly via the transfer of fixed nitrogen or indirectly by not competing for soil nitrogen (Vandermeer 1989). The potential for greater beef production from legume pastures in central Queensland has resulted in a number of studies investigating plant nutritive content and cattle live weight gains, in particular for leucaena (Clem and Hall 1994; Thornton and Buck 2011; Jones *et al*. 2000; Elledge and Thornton 2012). For example, Elledge and Thornton (2012) reported greater live weight gains per hectare from leucaena pasture versus grass only pasture, which was attributed to the higher crude protein content of leucaena leaves. Another reason for the increased uptake of leguminous pastures in central Queensland is that they are able to remain productive during the dry season. This is due to their deeper root system which enables them to utilise subsoil moisture and nutrients beyond the reach of grass roots (Shelton and Dalzell 2007; Radrizzani *et al*. 2010).

Garcia *et al*. (1996) reviewed 65 publications on the nutritive value and productivity of leucaena and reported a median forage value of 3.52 g/100 g dry matter (range 2.24 to 4.80 g/100 g dry matter) for nitrogen and 0.26 g/100 g dry matter (range 0.14 to 0.38 g/100 g dry matter) for phosphorus. A contrast of these values to plant total nitrogen (3.33%) and phosphorus (0.16%) from this study shows that although phosphorus levels in this study were slightly lower, both nitrogen and phosphorus are within the range of values reported by Garcia *et al*. (1996). Furthermore, plant nitrogen and phosphorus concentrations from this study are consistent with another Queensland study on cracking clay soils which reported respective concentrations of 2.87% and 0.13% in leucaena leaf tips (Clem and Hall 1994).

Similarly, Jones *et al*. (2000) compared the nutritive value of butterfly pea across three ley legume sites in Queensland that were all on heavy clay soils with a mean annual rainfall of 700 mm. The authors found that butterfly pea leaves comprised 66.6% of the samples and that total nitrogen in the leaves and stems was 4.39% and 2.48%, respectively; whereas total phosphorus was 0.44% and 0.36%, respectively. Clem and Hall (1994) reported lower concentrations of total nitrogen (3.07%) and phosphorus (0.27%) in butterfly pea leaf tips at the end of the growing season. However, a comparison of both these studies to the mean total nitrogen (1.36%; range 0.71 to 2.29%) and phosphorus (0.16%; range 0.07 to 0.24%) concentrations from butterfly pea leaves and stems in this study indicate that concentrations are considerably lower at the BCS. Although Jones *et al*. (2000) sampled plants after a period of active growth when it would be expected that concentrations are higher, concentrations in this study demonstrated limited seasonal fluctuations over the two years. Furthermore, the work of Jones *et al.* (2000) focused solely on butterfly pea plants, whilst the values in this study were derived from samples of total standing dry matter, including grasses, which would result in a lower estimate of nutrient content compared with that of the legume alone.

Another explanation for the lower concentrations of plant total nitrogen and phosphorus in leguminous pastures at the BCS is that fertilisers have never been applied and the butterfly pea ley pasture is relatively young (<5 years old). Furthermore, the catchment with a ley pasture of butterfly pea was originally setup to investigate fertility decline with intensive cropping which occurred for 27 years before the butterfly pea was planted (Chapter 4). This trend of lower nutrients at BCS is supported by literature which demonstrated increased biomass and nitrogen uptake by maize ally crops where nitrogen fertiliser had been applied in addition to the prunings of leucaena hedgerows (Xu *et al*. 1993). A survey of beef producers in Queensland reported a 58% (n=18) decline in leucaena productivity from pastures more than 10 years old, despite no fertiliser application to 86.2% (n=25) of pastures (Radrizzani *et al*. 2010). Although fertilisers may be applied to leguminous pastures to increase the uptake of nitrogen by companion grasses, this may also increase nutrient loads in runoff if surplus to system requirements.

Radrizzani *et al*. (2011) reported higher total nitrogen in soil from leucaena pasture versus grass only pasture, in addition to concentration increases with age (20, 31 and 38 years old) for both pasture types. More specifically, the authors found total nitrogen concentrations of 0.17% from 31 year old grass pasture and 0.14% from 20 year old leucaena pasture. These values are comparable to results from this study, which were 0.14% from 31 year old grass pasture and 0.11% from 15 year old leucaena pasture. This comparison of studies demonstrated similar concentrations of soil total nitrogen from grass pastures with and without leucaena at two sites from central Queensland.

In contrast to other studies which reported more nitrogen in grass (Rao and Giller 1993; Jayasundara *et al*. 1997) and soil (Radrizzani *et al*. 2011) from leucaena pasture versus grass monocultures, this study found that concentrations in grass from both pasture types exhibited little variation. Furthermore, this study found that soil total nitrogen was higher in grass only versus leucaena pasture, despite both pasture types responding similarly to each season over the two years. These results indicate that the transfer of nitrogen from leucaena to grass is inefficient or not occurring at the BCS site. Nonetheless, the presence of leucaena in the pasture system would still have positive production benefits by reducing its competition with grass for available soil nitrogen (Vandermeer 1989), and also by providing greater cattle live weight gains per hectare due to its greater nutritive value (Elledge and Thornton 2012). The principle of limited nitrogen transfer from legume to grass is supported by Xu *et al*. (1993), who investigated the efficiency of nitrogen transfer from leucaena to maize (*Zea mays*) ally crops. These authors found that 52 days after leucaena prunings were applied, 45.1% was still in leucaena residues, 25.2% was lost from the system, 24.9% was found in the soil and only 4.8% was taken up by the maize. Furthermore, Rao and Giller (1993) conducted a glasshouse experiment to quantify the transfer of nitrogen from leucaena to grass. These authors found that 42 to 54% of nitrogen in leucaena was derived by fixation, and that only 3.07 to 3.87% of nitrogen in grass came from leucaena. It is expected that butterfly pea would behave in a similar manner, as it has been found that plant total nitrogen derived from fixation was over 45% (Armstrong *et al*. 1997). The higher concentrations of soil total and nitrate-nitrogen from virgin brigalow scrub versus the three pasture types is probably due to brigalow (*A. harpophylla*) also being a legume.

There is currently a paucity of information on soil nutrient concentrations from butterfly pea ley pastures from other Australian sites to compare with results from this study; however, the usefulness of this legume for restoring soil fertility can be demonstrated using historical data from the BCS. That is, Thornton *et al*. (2010) reported a soil total nitrogen concentration of 0.074% from the cropping catchment (Chapter 2). This survey was conducted in 2008 and another crop rotation was implemented after this time, which may have further reduced the soil nitrogen concentration. However, a ley pasture of butterfly pea was planted in this catchment in January 2010 to restore soil fertility. At the commencement of this study in May 2011, more than 15 months after the butterfly pea was planted, total nitrogen in the soil was 0.086%, which then increased to 0.099% in April 2013. Thus, total nitrogen concentrations in the soil increased 0.025% in three years.

Although there is some literature to compare soil and plant nutrient status between pastures with and without legumes, there is currently a paucity of information on seasonal trends on this topic. This is important as legume growth is dominated in the summer months (Collins and Grundy 2005; Dalzell *et al*. 2006), and it is at this time of year that nutrient concentrations in soil and plants are expected to be highest. However, the end of the dry season is when runoff poses the greatest risk to GBR water quality (Chapter 3) and there is currently no seasonal data from other Australian sites which can demonstrate a link between soil and plant nutrient status from pasture legumes and water quality. An earlier trial at the BCS compared soil and pasture nutrient concentrations from the grass only and leucaena pastures collected between December 2008 and March 2010 (Elledge and Thornton 2012). A comparison of results between the earlier and current studies shows limited temporal variability in the concentration of soil nitrate-nitrogen from both pasture types. Total nitrogen showed the same trend of limited temporal variability in both studies (unpublished data). In contrast, ammonium-nitrogen in soil exhibited considerable temporal variation in both studies. Although concentrations of ammonium-nitrogen were higher over winter in the earlier study, the current study found statistically higher concentrations in summer compared with autumn and spring. No temporal trend was detected for soil available phosphorus in the current study despite higher concentrations over the summer months in the earlier study. A possible explanation for some of the soil parameters exhibiting more temporal variation in the current study is the above average rainfall that occurred during this period (Chapter 3) which is likely to have resulted in increased litter cycling with the potential for increased nutrient stratification in the organic and uppermost soil layer.

Further comparison of the earlier (Elledge and Thornton 2012) and current studies from the BCS site show that plant total nitrogen did not exhibit much variability for either the grass only or leucaena pasture types, but concentrations were statistically higher in autumn in the current study. Plant total phosphorus in both studies showed more temporal variability; however, seasonal trends differed. The earlier study showed higher concentrations in the summer months, whereas the current study found statistically higher concentrations in autumn than in summer or spring. Pasture biomass did not fluctuate much in the earlier study as catchments were stocked continuously according to feed availability. However, biomass in the current study showed considerable variation due to the removal of cattle from all pasture catchments in September 2011, and the then addition of cattle to the leucaena pasture for about five months the following year. Thus, the increase in pasture biomass observed during the destocked periods may have no relationship to season.

This work highlights the high variability of soil and plant nutrients within the landscape and that longer timeframes, in excess of two years, are required to detect change. With no discernible period of potential high nutrient availability during the year, soil and pasture management should focus on minimising runoff, rather than manipulation of the natural nutrient cycles to reduce risk to water quality.

Chapter 6: Conclusions

This project explored whether broad-scale pasture legume plantings, particularly of leucaena and butterfly pea, represent a risk to reef water quality by increasing loads of nitrogen in runoff compared with grass only pastures or virgin brigalow scrub. Virgin Brigalow Scrub is representative of the landscape in its pre-European condition and should be used as the benchmark for hydrology and water quality assessment. It has been clearly illustrated at the paddock scale that butterfly pea ley pasture which is less than three years old lost more nutrient and sediment in runoff than grass and leucaena pastures and virgin brigalow scrub. However, total suspended sediment loads from the butterfly pea ley pasture were substantially smaller than loads from the cropping system that it replaced. Runoff water quality was similar from grass and long-term leucaena. However, dissolved inorganic nitrogen loads and EMCs from leucaena were consistently higher than those from grass. Plot scale comparison of pasture types conducted in both the Burdekin and Burnett-Mary Basins highlights the end of the dry season as the greatest risk for high nutrient and sediment loads in runoff.

Comparison of pasture type effect on water quality in the Brigalow Belt Bioregion was preceded by extensive land use change. Over a 29 year period, the conversion of brigalow lands to cropping typically increased nutrient and sediment loads in runoff compared with virgin brigalow scrub. In contrast, conversion of brigalow lands to grazing reduced loads of some nutrients. Nutrient and sediment loads from cropping exceeded those from grazing. No change in water quality from cropping could be detected over time despite change from conventional tillage and annual wheat cropping to minimum and zero tillage and opportunity cropping.

The quantification of both the land use change effect and the pasture type effect on nutrients and sediment in runoff provides empirical evidence to support model development and testing. Comparison of load and EMC data for the various land uses and pasture types at the paddock scale provides important insight that high loads are not necessarily related to high EMCs. This is critical when extrapolating work from catchments where load or EMC is reported in the absence of flow.

Data presented on land use effects on nutrient and sediment loads is considered robust given that it is based on 18 years of baseline flow data, a 29 year comparison of land use and flow, and a 13 year comparison of land use and water quality. By comparison, data presented on pasture type effects on nutrient and sediment loads reflects only three years of record; a typical project cycle. The climatic sequence during this three year study was back to back record wet seasons followed by a season exceeding the 75th percentile for rainfall.

Increases in runoff associated with land development and changes in water quality associated with both land development and pasture type were still detected in this extremely wet period. However, they may not be representative of system dynamics in the longer term. High cover, high biomass and the exhaustion of easily mobilised nutrients and sediments within the catchment as a result of an extended wet period are likely to result in lower loads than may be expected during a drier period. This was certainly observed within the plot scale work (Chapter 3) and also the whole of catchment scale in the Fitzroy Basin (Packett, R. *pers. comm*. 2013 Department of Natural Resources and Mines).

A comparison of soil and plant nutrients in three different pasture types with and without legumes detected no seasonal differences in concentrations over a two year period. This indicates that the risk to runoff water quality is consistent throughout the year for grass only, butterfly pea and leucaena pastures. Overall, the grass only pasture had similar plant total nitrogen and slightly higher soil total and species nitrogen than the leucaena pasture, indicating that the transfer of nitrogen from leucaena to grass is inefficient at the BCS site. Nitrogen concentrations in soil and plants from butterfly pea were similar to the other two pasture types; however, butterfly pea had considerably higher available phosphorus in the soil.

Future Directions

Future research into grazing land management and its impacts on GBR water quality should heed key issues identified from the 2009 to 2013 Paddock to Reef Program (Silburn *et al*. in press). A synthesis of this program found that there were no sites with a direct comparison of grazing management practices, nor did any historical sites focus on the effects of grazing management on nutrient water quality. In addition, data for nutrient runoff was sparse, particularly for nutrient species as opposed to parameters such as total nitrogen and phosphorus which were more commonly reported. Furthermore, data for the effects of pasture management on nitrogen and phosphorus in runoff from northern Australia were virtually non-existent.

These issues could be addressed in future projects by the inclusion of the Department of Natural Resources and Mines owned BCS site. The BCS is underpinned by 49 years of continuous data on rainfall and runoff, in addition to 13 years of runoff water quality data. This state owned asset provides scope for the creation of a paired, replicated and/or nested catchment grazing land management study, and is able to be tailored to specific research questions rather than adapting research to unsuitable commercial operations which may have insecure tenure.

The BCS is an ideal site for future research, as it is located within the Fitzroy Basin which is Queensland's largest reef catchment (Australian Bureau of Statistics 2009). The Fitzroy Basin, closely followed by the Burdekin Basin, has the highest risk of adversely impacting GBR water quality due to the effects of grazing (Waterhouse *et al.* 2012). The risk to water quality in the Fitzroy Basin is high as it has more than 71% of its 15.6 Mha impacted by grazing (Australian Bureau of Statistics 2009). Furthermore, it has 2.8 million head of cattle (Johnston *et al.* 2008) which is the greatest number within a reef catchment, and 77% more cattle than the Burdekin Basin (Heather and Clouston 2006). The potential for well-established legume based pastures to contribute to loads of dissolved inorganic nitrogen, as demonstrated in Chapter 3, is also of particular relevance to the Burdekin Basin where large areas of naturalised S*tylosanthes spp*. may be contributing to anthropogenic loads at the catchment scale. Additionally, grazing in the Fitzroy Basin is undertaken on soils inherently more fertile than in the Burdekin Basin (Shelton and Dalzell 2007), leading to a greater risk of erosion based nutrient enrichment in runoff.

Three of the key management priorities for reducing the relative risk of pollutants to the GBR, as identified in the Relative Risk Assessment (Waterhouse *et al.* 2012), can be cost effectively investigated at the BCS (Table 6.1). Four benefits of the inclusion of the BCS site in future projects are: (1) avoiding expenditure on the development of new sites; (2) getting research activities on-ground in a short time period as management practice treatments are already imposed and monitoring equipment is installed; (3) 49 years of historical data collection; and (4) an experimental design that can be tailored to meet a specific research need and can be adaptively managed should the need arise throughout the course of a program.

Table 6.1: Management priorities for reducing pollutant risks to the Great Barrier Reef that can be investigated in future research projects using the Brigalow Catchment Study (modified from Waterhouse *et al.* 2012).

Relative Priority	Management Priorities		
	Region	Pollutant Management	Key Land Use
2	Burdekin	Erosion management	Grazing
2	Fitzroy	Erosion management	Grazing
4	Fitzroy	Pesticide reduction	Grazing
2	Fitzroy	DIN reduction	Cropping

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Appendix 1: Publications

There are five publications that have resulted from this project:

(1) ELLEDGE, A. and THORNTON, C., 2012. The Brigalow Catchment Study: Comparison of soil fertility, forage quality and beef production from buffel grass vs. leucaena-buffel grass pastures, L.L. BURKITT and L.A. SPARROW, eds. In: *Proceedings of the 5th Joint Australian and New Zealand Soil Science Conference: Soil solutions for diverse landscapes*, 2-7 December 2012, Australian Society of Soil Science Inc, pp. 181-184.

(2) ELLEDGE, A. and THORNTON, C., 2012. The Brigalow Catchment Study: Nitrogen runoff generation rates from pasture legumes and changes since land development, S. WESTA, ed. In: *Proceedings of the 34th Hydrology and Water Resources Symposium*, 19-22 November 2012, pp. 1000-1007.

(3) THORNTON, C. and ELLEDGE, A., 2012. The Brigalow Catchment Study: Increases in runoff associated with land development can still be detected in flood events at a small catchment scale, S. WESTRA, ed. In: *Proceedings of the 34th Hydrology and Water Resources Symposium*, 19-22 November 2012, pp. 1566-1570.

(4) THORNTON, C. and YU, B., 2012. The Brigalow Catchment Study: Effects of land development on peak runoff rate and its prediction in central Queensland, Australia, S. WESTRA, ed. In: *Proceedings of the 34th Hydrology and Water Resources Symposium*, 19-22 November 2012, pp. 362-369.

(5) THORNTON, C., COWIE, B., DAVISON, L., TANG, W. and ELLEDGE, A., 2013. Appendix 14: Nitrogen generation rates from grass pastures over-sown with butterfly pea or leucaena in the Burnett-Mary and lower Burdekin catchment. In*: Final report: Paddock to Reef: Paddock Scale Monitoring and Modelling - Rainfall Simulation Program 2010 – 2013.* pp. 59-61.

Appendix 1.1: Publication 1

The Brigalow Catchment Study: Comparison of soil fertility, forage quality and beef production from buffel grass vs. leucaena-buffel grass pastures

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Abstract

This paper assesses the fertility status of soils and forages in two grazed pasture systems in central Queensland: 1) buffel grass only, and 2) leucaena-buffel grass. Trends observed in our preliminary results indicate that the paddock with leucaena had similar or lower soil fertility (0 to 10 cm) than the grass only paddock based on concentrations of available phosphorus, organic carbon, nitrate and ammonia; however, the leucaena paddock had similar or better pasture quality based on crude protein and total phosphorus. This can be partly explained by the deeper root system of leucaena which can utilize nutrients further down the soil profile than grass pastures. Furthermore, greater live weight gains of cattle in the leucaena-grass paddock can be attributed to higher crude protein concentrations found in leucaena vs. grass leaves.

Introduction

Leucaena hedgerows planted with companion pasture grasses are one of the most productive, profitable and sustainable grazing systems in tropical and subtropical Australia (Dalzell *et al.* 2006, Shelton and Dalzell 2007). In managed agricultural scenarios, leucaena has many reported benefits, including reduced soil erosion, improved runoff water quality, and enhanced soil fertility (Dalzell *et al.* 2006, Shelton and Dalzell 2007). Beef production on leguminous pastures in central Queensland is becoming more common, as there is an increase in the availability of data that demonstrates greater biomass and nutritive value of forage from leucaena-grass paddocks vs. grass only paddocks. This paper links data on soil and pasture fertility status with beef production per hectare between pastures with and without legumes.

Methods

Study site

This research was conducted on the Brigalow Catchment Study (BCS) site near Theodore in central Queensland. The BCS commenced in 1965 and is an ongoing long-term study on the impact of land development on hydrology, productivity and resource condition. This project compares two paddocks from the BCS site: 1) buffel grass (*Cenchrus ciliaris* cv. Biloela) pasture that was originally cleared in 1982 and planted in 1983, and 2) leucaena (*Leucaena leucocephala* cv. Cunningham) and buffel grass pasture that was originally cleared ~1968 and cropped for 10 to 15 years before converting into a grazed paddock. The leucaena was planted in 1998 on 8 m hedgerows. Both paddocks are predominantly grey and black Vertosols with an average slope of 2.5%, and neither paddock has a history of fertiliser application.

Soil fertility and forage quality

Soil was sampled at approximately three month intervals from December 2008 until October 2009. Six samples 0 to 10 cm from the soil surface were collected from three permanent monitoring sites within both the grass only (N=18) and leucaena-grass paddock (N=18) for each sampling period. The samples were analysed for Colwell available phosphorus (P), Walkley and Black organic carbon (OC), nitrate (NO_3-N), and ammonia (NH_4-N). Pasture was sampled at approximately three month intervals from December 2008 until March 2010. Six samples 1x1 m were collected from the same three permanent monitoring sites within both the grass only (N=18) and leucaena-grass paddock (N=18) for each sampling period. The samples were analysed for total Kjeldahl nitrogen (TKN) and phosphorus (TKP). Crude protein (CP) was later calculated as 6.25 x TKN.

Beef production

Two drafts of weaner cattle were grazed on the grass only and leucaena-grass pastures; the first draft from May 2008 to May 2009 and the second draft from June 2009 to March 2011. Similar stocking rates were used for the first grazing period with 2.1 ha per head for the grass only paddock and 2.2 ha per head for the leucaena-grass paddock. In the second grazing period, the stocking rate was decreased in the grass only paddock to 3.4 ha per head and increased in the leucaena-grass paddock to 1.5 ha per head to match feed availability. Production was measured as cumulative weight gain of the cattle per ha. Values reported are based on results from Thornton and Buck (2011).

Statistical analyses

Means and standard errors were calculated in GenStat (v.14) for soil and pasture fertility. Furthermore, one-way ANOVA's with protected Fisher's LSD for pasture type were performed (P<0.05).

Results

Soil fertility and forage quality

Overall, there was a trend of lower organic carbon and available phosphorus in soils of the leucaena-grass pastures (OC mean 1.22% ± S.E. 0.03%; P mean 9.25 mg/kg ± S.E. 0.29 mg/kg) than in the grass only pastures (OC mean 1.77% ± S.E. 0.04%; P mean 11.89 mg/kg ± S.E. 0.35 mg/kg) (Fig.1). Both of these nutrients exhibited little variation, with an overall detectable difference between paddocks (OC $F_{1, 178}$=85.96, P<0.001; P $F_{1, 178}$=29.13, P<0.001).

Nitrate concentrations were higher in the grass only (mean 3.64 mg/kg ± S.E. 0.25 mg/kg) than leucaena-grass paddock (mean 2.99 mg/kg ± S.E. 0.26 mg/kg) (Fig.1). Nitrate in the grass only paddock was lower in the wet season from November to March than in the dry season; however, the late commencement of sampling from the leucaena-grass paddock does not currently allow seasonal variations to be determined. Overall, large variability was observed in the results for the last three sampling periods and no difference could be detected between the two pasture systems ($F_{1, 178}$=3.01, P=0.084).

Ammonia was similar between the two pasture systems in the last three sampling periods; however, concentrations were overall higher in the leucaena-grass (mean 5.50 mg/kg ± S.E. 0.27 mg/kg) than grass only paddock (mean 4.81 mg/kg ± S.E. 0.22 mg/kg) (Fig.1). Little variation was observed in the data, but an overall difference between the paddocks was detected ($F_{1, 178}$=3.97, P=0.048).

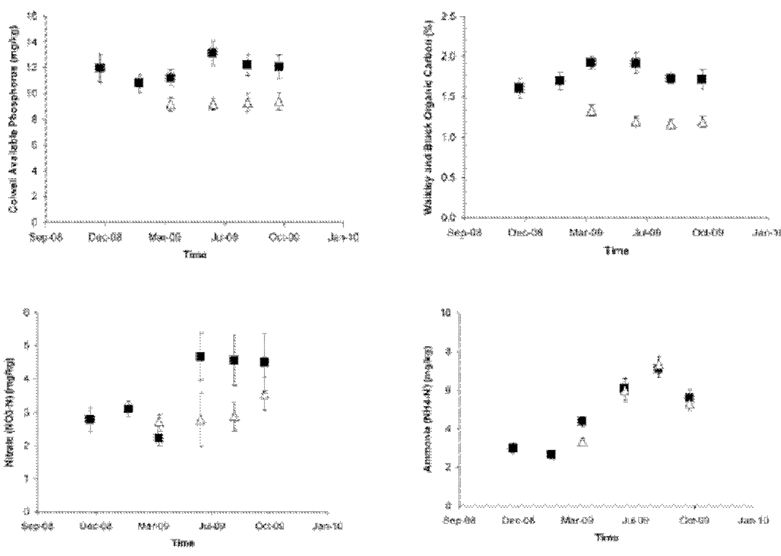

Figure 1. Fertility of soil (0 to 10 cm) based on available phosphorus, organic carbon, nitrate and ammonia concentrations in the grass only (black square) and leucaena-grass (white triangle) pastures; mean ± standard error bars.

Forage yield was greater in the grass only (mean 1617.0 kg/ha ± S.E. 134.5 kg/ha) than in the leucaena-grass paddock (mean 906.5 kg/ha ± S.E. 56.3 kg/ha), whilst yield from leucaena leaves was much lower (mean 385.9 kg/ha ± S.E. 51.3 kg/ha) (Fig.2). Although variability within sampling periods was larger in the grass only paddock, overall, differences were detected between all three pasture types ($F_{2,336}$=30.59, P<0.001).

Crude protein of pasture grasses from the grass only (mean 4.44% ± S.E. 0.15%) and the leucaena-grass paddock (mean 5.88% ± S.E. 0.27%) were similar over time (Fig.3). In contrast, crude protein of leucaena leaves (mean 16.70% ± S.E. 0.59%) from the latter paddock was much higher than from the companion pasture grasses in the same paddock. Variability within each treatments sampling period was small, and overall, a difference between all three pasture types was detected ($F_{2,318}$=322.54, P<0.001).

Total phosphorus concentrations were similar between pasture grasses from the grass only (mean 0.10% ± S.E. 0.005%) and the leucaena-grass paddock (mean 0.12% ± S.E. 0.004%), and also leucaena leaves (mean 0.12% ± S.E. 0.004%) from the latter paddock (Fig.3). Although data from all three pasture types exhibited variability in concentrations overtime, overall the grass only paddock had a detectably lower concentration that the other two pasture types ($F_{2,318}$=10.20, P<0.001).

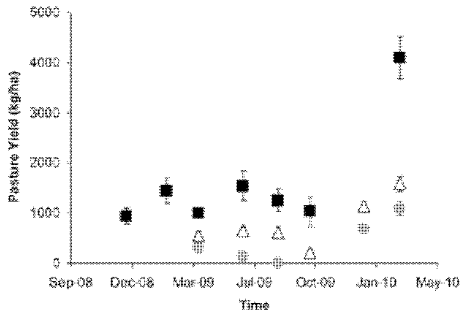

Figure 2. Total yield of pasture from the grass only (black square) and leucaena-grass (leucaena = grey circle, and grass = white triangle) paddocks; mean ± standard error bars.

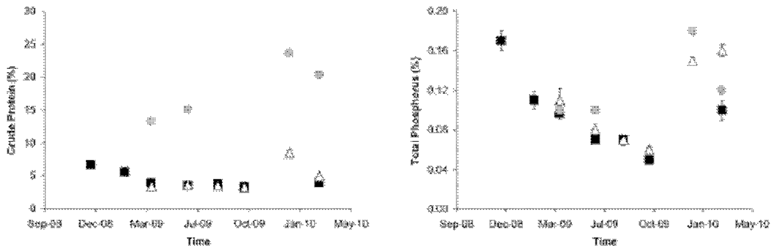

Figure 3. Nutritive value of pastures based on crude protein and total phosphorus in the grass only (black square) and leucaena-grass (leucaena = grey circle, and grass = white triangle) paddocks; mean ± standard error bars.

Beef production

During the first grazing period when the two pasture systems had similar stocking rates, the cumulative live weight gain of cattle per ha was comparable, though slightly higher in the leucaena-grass (94 kg/ha) than in the grass only paddock (81 kg/ha) (Fig.4). During the second grazing period when stocking rates were adjusted to match feed availability, beef production per hectare was much higher in the leucaena-grass paddock (117 kg/ha) whereas it remained consistent in the grass only paddock (57 kg/ha) (Fig.4).

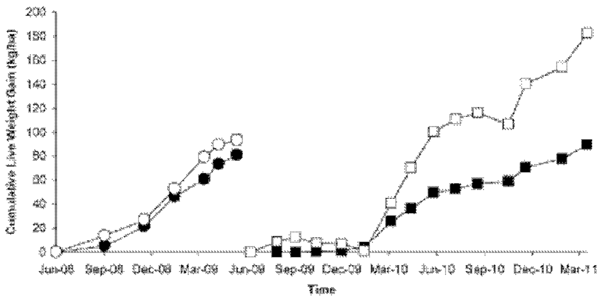

Figure 4. Cumulative live weight gain of cattle grazed on grass only (black shapes) and leucaena-grass (white shapes) paddocks at similar (circle) and feed on offer (square) stocking rates. The two stocking rate treatments used a different draft of cattle. Source: Thornton et al. (2010), Thornton and Buck (2011).

Discussion

In this study, soil fertility was similar or lower in the leucaena-grass paddock than the grass only paddock. However, the nutritive value of grass forages was similar or better for the leucaena-grass paddock. The contrasting results can be partly explained by soil sampling depth (0 to 10 cm) and the physiology of the studied plants. That is, leucaena has a deep root system that can access subsoil moisture and nutrients that are typically beyond the reach of grass roots. The deep root system of leucaena enables the plant to remain productive during the dry season; thus, enabling continued cattle production (Radrizzani *et al.* 2010). However, the results in this paper are in contrast to other literature which discuss the importance of grass in leguminous pastures to utilise mineral nitrogen (Fillery 2001) and improvements to soil fertility through nitrogen fixation (Shelton and Dalzell 2007). Thus, it is surprising that the nutritive value of grass from both the grass only and leucaena-grass paddock are similar over time, as it indicates that the transfer of nitrogen from leucaena to the grass is inefficient or not occurring at all. Based on feed availability, the leucaena-grass paddock was able to stock more cattle resulting in greater beef production per hectare than the grass only paddock. Due to the lower concentrations of organic carbon, phosphorus and nitrate in soils from the leucaena-grass paddock, the greater cattle stocking rates and live weight gains observed can be attributed to the higher concentrations of crude protein found in the leucaena leaves (Shelton and Dalzell 2007).

Acknowledgments

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Appendix 1.2: Publication 2

The Brigalow Catchment Study: Nitrogen runoff generation rates from pasture legumes and changes since land development

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Abstract

Grazing is the dominant agricultural land use in reef catchments of central Queensland, and despite the rapid and broad-scale inclusion of pasture legumes into grazing systems, there is currently no information available on the loads and impacts of nitrogen on the Great Barrier Reef (GBR) lagoon from legume based pastures.

This paper assesses the risk of nitrogen runoff from legume pastures to water quality in the GBR by comparing runoff water quality from grass pastures with and without nitrogen fixing legumes, namely leucaena and butterfly pea. Furthermore, historical data from the long-term internationally recognised Brigalow Catchment Study (BCS), near Theodore in central Queensland, is used to provide estimations of pre-European nitrogen loads in runoff waters to determine how nitrogen dynamics have changed with land development.

Trends observed in these results indicate that pastures with leucaena and buffel grass pose a smaller threat to water quality in the GBR than both buffel grass only pastures and ley pastures of butterfly pea, as less nitrogen and sediments are exported in runoff waters. Furthermore, based on 20 years of historical data from the BCS, the cumulative effect of grazing on total and oxidised nitrogen was smaller than the effect from cropping and the predicted effect had the native brigalow scrub not been cleared.

1. INTRODUCTION

Grazing is the dominant agricultural land use in reef catchments, accounting for 96% (26.2 million ha) of land use within the Burdekin, Fitzroy and Burnett-Mary catchments (Australian Bureau of Statistics, 2009). In this region of central Queensland, 9.1 million hectares are sown to improved pastures (Australian Bureau of Statistics, 2009). Nitrogen fixing legumes, such as butterfly pea and leucaena, are often introduced into pastures to improve the fertility of the system and increase ground cover. However, runoff is one of many processes that can reduce nitrogen availability in the system, resulting in lower productivity.

Leucaena hedgerows planted with companion pasture grasses are one of the most productive, profitable and sustainable grazing systems in tropical and subtropical Australia (Dalzell *et al*, 2006; Shelton & Dalzell, 2007). In managed agricultural scenarios, leucaena has many reported benefits, including reduced soil erosion, improved runoff water quality, and enhanced soil fertility (Dalzell *et al*, 2006; Shelton & Dalzell, 2007). Beef production on leguminous pastures in central Queensland is becoming more common, as there is an increase in the availability of data that demonstrates greater nutritive value of forage and consequently greater live weight gains of cattle from leucaena-grass paddocks versus grass only paddocks (Elledge & Thornton, 2012; Thornton & Buck, 2011). Although there is a paucity of information on beef production benefits from butterfly pea, interest in this legume as a ley pasture system is becoming more popular in cropping areas as declines in soil fertility limit production (Pengelly & Conway, 2000). Furthermore, there is currently limited research available on water quality from leucaena and butterfly pea pasture systems in Australia.

Despite the rapid and broad-scale inclusion of pasture legumes into grazing systems in central Queensland (Pengelly & Conway, 2000), there is currently no information on the loads and impacts of nitrogen on the GBR from these catchments. This knowledge gap is a weakness in current catchment models, as they do not attribute different nitrogen generation rates to pastures which do or do not have legumes. Thus, the two objectives of this paper are:

1. Assess the quality of water discharging into the GBR by calculating nitrogen loads in runoff waters from three pasture land uses: i) cropping catchment with a grazed butterfly pea ley pasture, ii) grazed buffel grass catchment, and iii) grazed leucaena and buffel grass catchment.
2. Use historical data from the BCS to estimate pre-European nitrogen loads from the cropping and pasture catchments (i and ii above) in order to determine nitrogen dynamic changes with land development.

2. METHODS

2.1. Study Site

This research was conducted on the BCS site near Theodore in central Queensland. The BCS commenced in 1965 and is an ongoing long-term study on the impact of land development on hydrology, resource condition and productivity. This paper compares data from four catchments (Figure 1): 1) native brigalow scrub (*Acacia harpophylla*) which is an uncleared control; 2) butterfly pea (*Clitoria ternatea* cv. Milgarra) paddock that was originally cleared in 1982 and cropped for 27 years before conversion into a grazed ley pasture in 2010; 3) grazed buffel grass (*Cenchrus ciliaris* cv. Biloela) paddock that was originally cleared in 1982 and planted in 1983; and 4) leucaena (*Leucaena leucocephala* cv. Cunningham) and buffel grass paddock that was originally cleared ~1968 and cropped for 10 to 15 years before conversion into a grazed pasture. Leucaena was planted in 1998 on 8 m hedgerows. Soils in all catchments are predominantly grey and black Vertosols with an average slope of 2.5%, and there is no history of fertiliser application.

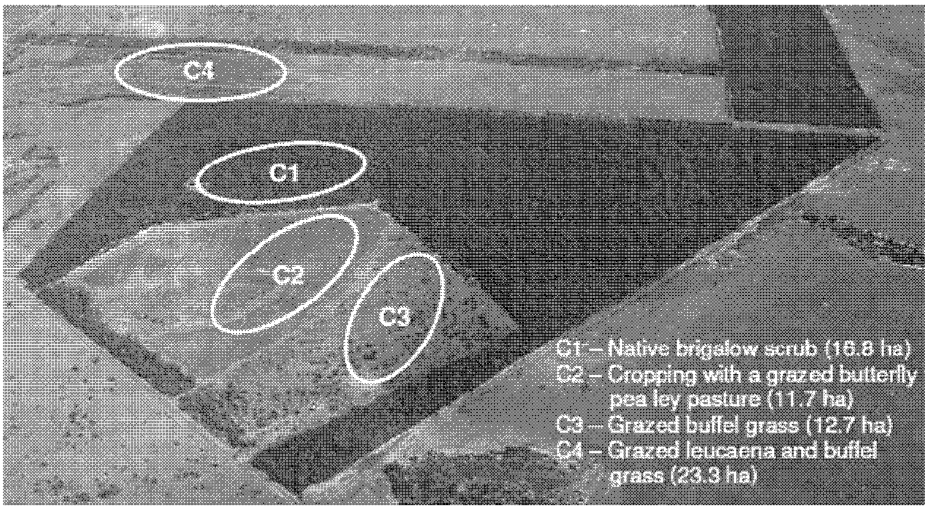

Figure 1. Ariel photo of the Brigalow Catchment Study showing the four catchments compared in this paper.

2.2. Leguminous Pasture Effect on Water Quality

Rainfall and runoff were monitored over a 17 year calibration period (1965 to 1982) from the three catchments referred to elsewhere in this paper as native brigalow scrub (C1), cropping with a butterfly pea ley pasture (C2), and grazed buffel grass only pasture (C3). In their virgin condition, runoff was approximately 5% of annual rainfall for all three catchments. Furthermore, the amount of runoff from C2 and C3 prior to their development was 95% and 72% of runoff from C1, respectively (Thornton *et al*, 2007). Calibration of the leucaena-grass catchment did not occur at this time due to its latter inclusion in the BCS.

Measurement of nitrogen in runoff waters was undertaken for all four land use types for two hydrological years: 2010 and 2011. All catchments were instrumented to measure rainfall and runoff, and to collect event based water quality samples. Total loads and event mean concentrations were calculated for total nitrogen, oxidized nitrogen (nitrate NO_3–N + nitrite NO_2–N), and ammonia (NH_4-N). No samples were collected from the leucaena-grass catchment for water quality in the 2010 wet season. Total rainfalls for the 2010 and 2011 wet seasons were 958 mm and 1009 mm, respectively.

2.3. Land Development Effect on Water Quality

Total nitrogen, oxidised nitrogen and ammonia loads were determined for runoff data for the period 1984 to 2004 in the native brigalow (C1), cropped (C2) and grazed catchments (C3). This was achieved by converting runoff depth (mm) to volume discharge (L), and then calculating load (kg/ha) using event mean concentrations (EMC) for the period 2000 to 2010. EMC data for total nitrogen only covers the period 2004 to 2010 due to a change in laboratories, and thus analytical methods. Loads were calculated for observed data in all three catchments and predicted estimates for the cropped and grazed catchments if they had not been cleared. Predicted estimates were calculated using EMC data from C1 applied to the predicted runoff from C2 and C3 had they not been cleared, as described in Thornton *et al* (2007). The effect of land use on runoff water quality for the cropped and grazed catchments was determined by subtracting their predicted uncleared loads from observed loads.

3. RESULTS

3.1. Leguminous Pasture Effect on Water Quality

For the 2010 wet season, nitrogen and sediment loads from the newly established butterfly pea in the cropping catchment were much higher than results from the pasture catchment with only grass (Figure 2). Loads in the grass only catchment were also lower than results from native brigalow scrub.

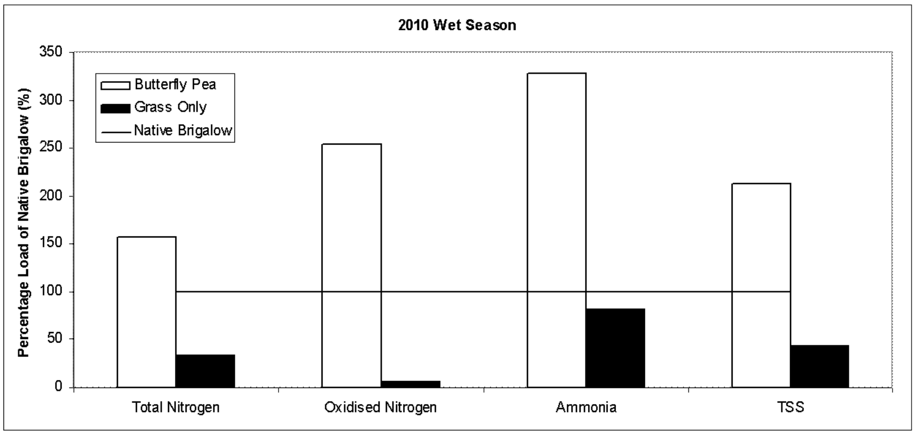

Figure 2. Nitrogen and total suspended sediment (TSS) loads in runoff waters from the butterfly pea and grass only catchments. Values are presented as a percentage of the native brigalow scrub loads.

For the 2011 wet season, nitrogen and sediment loads in all three developed catchments were greater than in the native brigalow scrub (Figure 3). Nitrogen loads were consistently higher in the butterfly pea catchment; and oxidized nitrogen in the grass only catchment was the closest value to the native brigalow scrub. Sediment loads were considerably higher in the butterfly pea and grass only catchments compared to the native brigalow scrub.

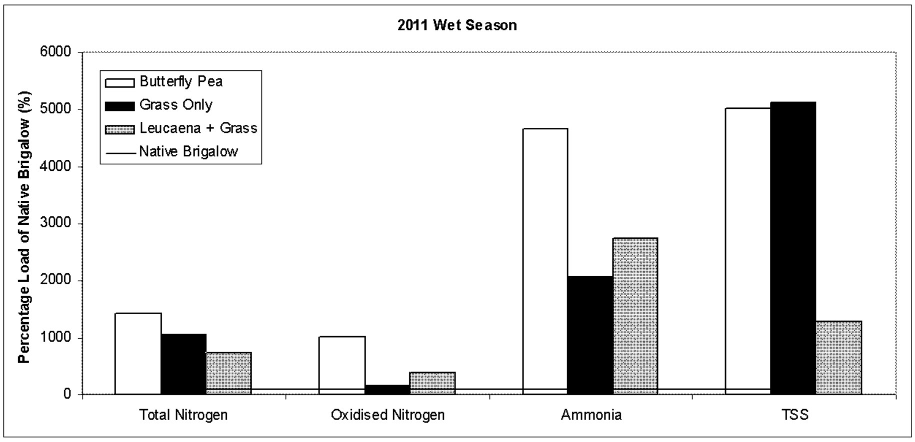

Figure 3. Nitrogen and total suspended sediment (TSS) loads in runoff waters from the butterfly pea, grass only, and leucaena-grass catchments. Values are presented as a percentage of the native brigalow scrub loads.

3.2. Land Development Effect on Water Quality

After 20 years of land development, total and oxidized nitrogen from the cropped catchment were not dissimilar to predicted uncleared native brigalow scrub with only 7 and 8 kg/ha lost in runoff waters, respectively (Figures 6 and 7). In contrast, the grazing catchment exported 55 kg/ha less total nitrogen and 27 kg/ha less oxidized nitrogen than predicted estimates of uncleared land. Both the cropped (1.08 kg/ha) and grazed catchments (0.17 kg/ha) lost ammonia in runoff waters, but loads were considerably higher in the cropped catchment (Figure 8). Loads of total suspended sediments were also considerably higher in the cropped (9603 kg/ha) than grazed catchment (793 k/ha) over the 20 year period.

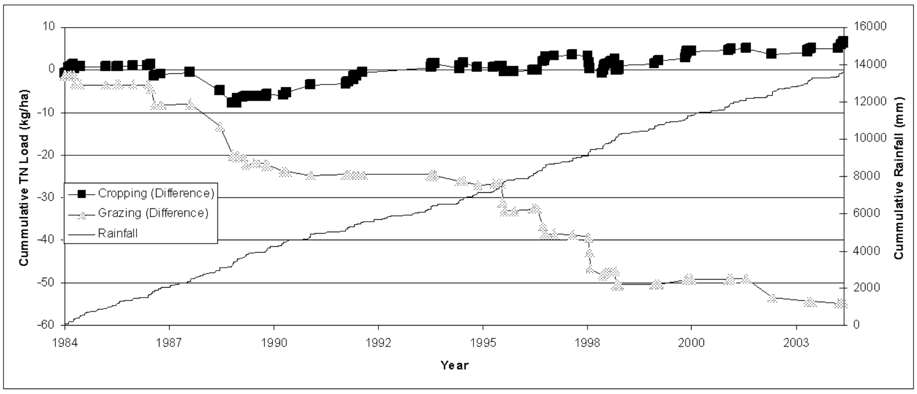

Figure 6. Cumulative total nitrogen (TN) loads from cropping (square) and grazing catchments (triangle), and cumulative total rainfall (line) for the period 1984 to 2004.

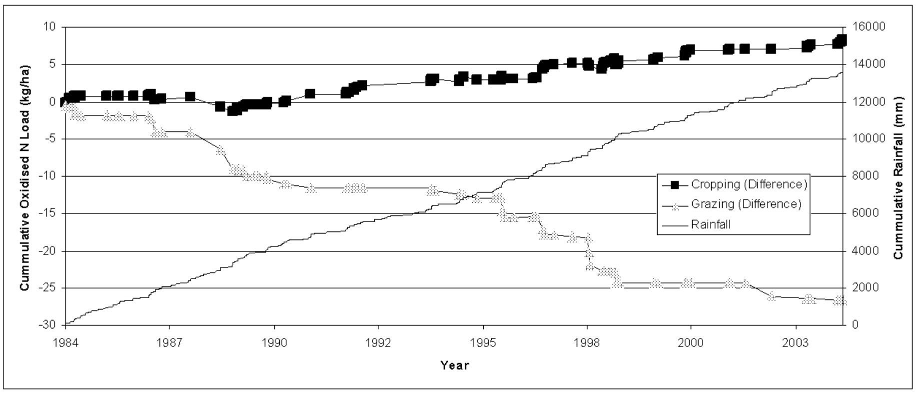

Figure 7. Cumulative oxidised nitrogen (N) loads from cropping (square) and grazing catchments (triangle), and cumulative total rainfall (line) for the period 1984 to 2004.

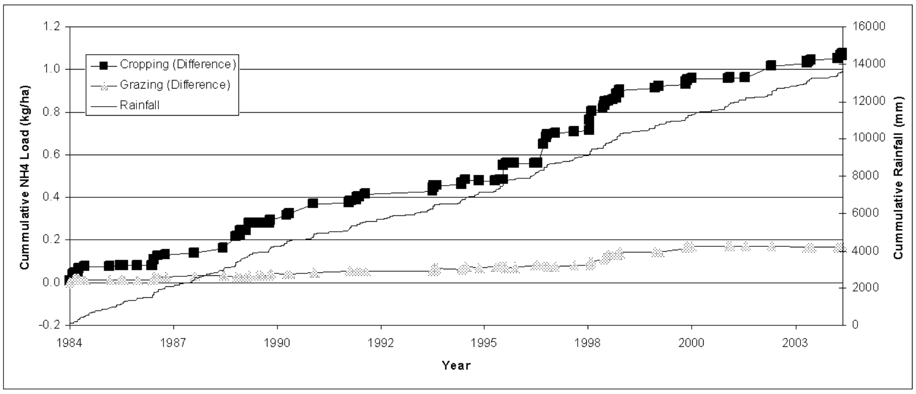

Figure 8. Cumulative ammonia (NH_4) loads from cropping (square) and grazing catchments (triangle), and cumulative total rainfall (line) for the period 1984 to 2004.

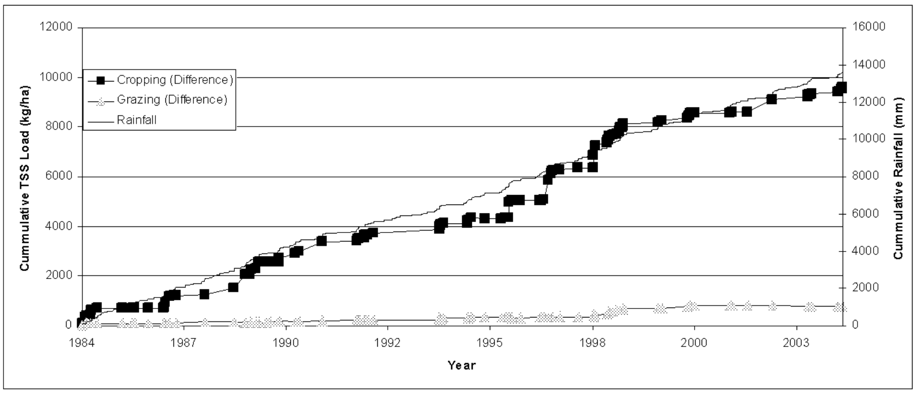

Figure 9. Cumulative total suspended sediment (TSS) loads from cropping (square) and grazing catchments (triangle), and cumulative total rainfall (line) for the period 1984 to 2004.

4. DISCUSSION

Runoff water quality from pastures in the brigalow bioregion of central Queensland varied between years and also the type of legume species present. Catchments with butterfly pea may pose a greater threat to water quality in the GBR than grass pastures with and without leucaena based on higher nitrogen loads in runoff water. In the 2010 hydrological year, the higher nitrogen and sediment loads in the butterfly pea catchment can be partly attributed to the short time since planting, that is, the catchment had a greater risk of nutrient and sediment loss due to recent soil disturbance and reduced ground cover. However, a similar trend was also observed in the second year of the study with higher concentrations of ammonia, total nitrogen and oxidised nitrogen in the butterfly pea catchment.

Although there is only one year of data for the leucaena-grass catchment, to date, our results indicate that this legume exports less nitrogen and sediment in runoff waters than grass pastures with and without butterfly pea. In regards to beef production, the leucaena-grass catchment is able to carry more cattle per hectare than grass only paddocks when stocked according to feed availability (Thornton & Buck, 2011; Thornton *et al*, 2010). The resulting greater live weight gains per hectare in the leucaena-grass paddock can be associated with higher concentrations of crude protein found in the leucaena leaves versus grass leaves (Elledge & Thornton, 2012). Thus, leucaena-grass catchments are able to produce more beef per hectare whilst exporting less nitrogen and sediments in runoff waters than grass only pastures.

Using historical data from the BCS, we found that the cumulative effect of cropping over 20 years on cleared brigalow lands was negligible for total and oxidized nitrogen in runoff waters. However, considerably smaller loads were exported in runoff waters from the grazed catchment compared to its uncleared predictions. Furthermore, grazing resulted in negligible losses of ammonia and sediments compared to if the land had remained uncleared, whereas cropping had considerably higher loads in runoff waters. This is consistent with other literature which demonstrates greater sediment losses from cropping than grazing (Freebairn *et al*, 2009; Murphy *et al*, 2012). Although minimum tillage was initiated at the BCS in 1992, this management effect can not be clearly observed due to the absence of runoff events immediately following the implementation of this practice. These results also support Silburn *et al* (2007), who reported that conservative farming practices generally reduce sediment losses, but concentrations of dissolved nutrients transported in runoff, such as nitrate and ammonia, can be higher in conservative compared versus intensely tilled systems.

Nitrogen and sediment loads were lower from grass pastures in 2010 compared to butterfly pea pastures. In addition, the grazed catchment exported less total and oxidized nitrogen compared to cropped and native brigalow scrub in the 20 year period since land development. There is currently no data for the effect of land development on the leucaena-grass catchment due to the later inclusion of this treatment into the BCS, but it is hypothesized that a similar trend of less nutrient and sediment exporting in runoff waters would be observed to the grass only catchment. Thus, grazing in reef catchments exports less nitrogen and sediments than cropping, and in the case of total and oxidized nitrogen, observed loads from the grazed catchment were lower than predicted loads had the catchment remained uncleared.

5. ACKNOWLEDGEMENTS

This work was jointly funded by the Department of Natural Resources and Mines and the Reef Rescue Research and Development Program (Project RRRD009) of the Australia Government's Caring for our Country initiative. The authors thank past and present staff from the Department of Natural Resources and Mines and the Department of Agriculture, Fisheries and Forestry that worked on the Brigalow Catchment Study.

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Appendix 1.3: Publication 3

The Brigalow Catchment Study: Increases in runoff associated with land development can still be detected in flood events at a small catchment scale

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Abstract

The Brigalow Catchment Study has unequivocally shown that developing brigalow lands for cropping or for pasture doubles runoff volume and more than doubles peak runoff rate at the small catchment scale (<20 ha). However, the persistence of these land use effects on catchment response during high flow conditions are hotly debated within the international literature. Typical hypotheses are that these changes are reflected in smaller events and that the effects of forests, land cover and land use tend to converge during high flows, which have been defined as events with a return period of as short as two and ten years.

During 2010 and 2011, much of Queensland, Australia was subjected to consecutive record wet seasons. Data from the Brigalow Catchment Study, near Theodore in central Queensland, showed 2010 to be the third wettest year since records commenced in 1965; this was eclipsed by 2011 rainfall totals. These extreme seasons provided a unique opportunity to investigate the impacts of land use change on runoff and peak runoff rate in high flow conditions using a paired, calibrated catchment approach.

During the height of flooding in 2011, the brigalow scrub yielded 183 mm of runoff, cropping yielded 224 mm and pasture yielded 197 mm. Gross increases in runoff were 28% from the cropping and 50% from the pasture compared to that expected had they remained brigalow scrub. Increases in runoff as a result of land development were found in the three wettest years on record using a number of analytical approaches.

These findings lend strong support to the hypothesis that increases in runoff associated with land development can still be detected in flood events at a small catchment scale.

1. INTRODUCTION

Review of the literature shows that the impacts of forests on flood events are unclear (Alila *et al* 2009; van Dijk *et al* 2009). Forests are generally thought to have a flood mitigation effect (Bradshaw *et al* 2007), however this is likely restricted to more frequent, less extreme rainfall and subsequent flooding (Bathurst *et al*, 2011; van Dijk *et al*, 2009). Some studies fail to differentiate between the effects of land use change and forest presence or absence on runoff, and often use the terms interchangeably. Paired catchment studies are often cited as key references for determining the effects of forest cover on flood events, however debate continues on the suitability of methods for looking at changes in both magnitude and frequency (Alila *et al*, 2009; Alila *et al*, 2010; Lewis *et al*, 2010).

The 1998, 2010 and 2011 hydrological years at the Brigalow Catchment Study site in central Queensland, Australia, all had rainfall totals greater than the 15 year recurrence period. The 2011 season was the wettest in the study history, resulting in the most runoff from each catchment in both a season and in a single event. These extreme wet seasons were chosen to determine if increases in runoff due to land use change are still prevalent in high flow conditions on both an annual and individual event basis using both regression techniques typically associated with paired catchment studies and flow duration curves.

2. METHODS

2.1. Experimental site and study history

The BCS is a paired, calibrated catchment study consisting of three catchments. The study was established in 1965 to determine the impact on hydrology, productivity and resource condition when brigalow land is cleared for cropping and grazing. The study has been thoroughly documented elsewhere in this series (Elledge & Thornton 2012; Thornton & Yu 2012).

The study consists of three adjoining catchments, each of approximately 15 ha. Soils within each catchment are predominantly grey and black Vertosols, with an average slope of 2.5%. In their native state, all catchments were vegetated with brigalow scrub communities. 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. Water height through the flumes was recorded using mechanical float recorders. Rainfall was recorded at the head of the three catchments.

During Stage I of the study, from 1965 to 1982, runoff was measured from the catchments in their native state. Stage II of the study commenced in 1982, when two of the three catchments were cleared with bulldozer and chain, and the fallen timber burnt in-situ. Catchment 2 (C2) was developed for cropping, with the construction of contour banks and grassed waterways, while catchment 3 (C3) was developed for grazing by the planting of improved pasture. Catchment 1 (C1) was left in its native state as a control. Stage III of the study was land use comparison, commencing in 1984.

2.2. Methods of analysis

Increase in runoff due to land use change was determined using two methods. Firstly, the calibrated catchment regression analysis approach of Thornton *et al* (2007) was used to compare Stage I runoff from C2 and C3 against C1. These relationships were then applied to C1 runoff from Stage III to estimate runoff from C2 and C3 had they not been cleared and developed for agriculture. The difference between observed runoff during Stage III and these estimates of runoff from the catchments is attributed to land use change. The three wettest hydrological years on record, 1998, 2010 and 2011 were chosen for this analysis. This approach was applied on an event basis, on an annual basis, and using the annual maxima sequence.

Secondly, catchment by stage flow duration curves were constructed using the program HYFLOW (Kisters, 2010) to show the percentage of time that a specific 9 am total daily discharge was equaled or exceeded. The order and magnitude of the curves between catchments was compared between Stage I and Stage III. Changes in the flow duration curve for C1 reflect changes in climatic sequence, while changes in the relationships between C2 and C3 to C1 reflect changes in runoff due to land development.

3. RESULTS

Average hydrological year rainfall at the site in the period 1965 to 2011 was 647 mm. Total rainfall for 2011, the wettest year on record was 1009 mm. Total rainfall for 1998, the second wettest year was 987 mm, while 2010, the third wettest year was 958 mm.

Equations 1 and 2 describe the event based runoff relationships of the catchments in their native state and allow the prediction of runoff from C2 and C3 given the known runoff volume from C1. The equations are robust, including an event of over 100 mm from C1 (Figure 1).

$$\text{C2 runoff (mm)} = \text{C1 runoff (mm)} \times 0.9539 \quad (R^2 = 0.95,\ n = 37) \qquad (1)$$

$$\text{C3 runoff (mm)} = \text{C1 runoff (mm)} \times 0.7176 \quad (R^2 = 0.887,\ n = 40) \qquad (2)$$

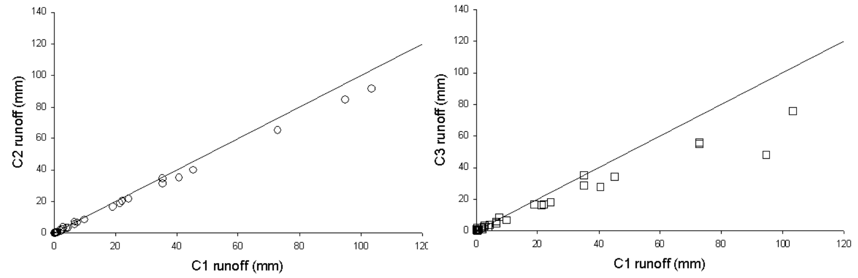

Figure 1 Event based runoff data collected during Stage I of the study shows a robust linear relationship between the catchments.

On an annual hydrological year basis, developing brigalow land for either cropping or grazing consistently shows an increase in runoff during extremely wet years (Table 1). This finding continues to hold on an individual event basis when using annual maximum series for the three years (Table 2).

Table 1. Observed annual runoff for the three wettest hydrological years of the study, compared to estimated runoff from C2 and C3 had they not been developed.

	C1	C2 - cropping				C3 – grazed pasture			
Year	Total runoff (mm)	Total runoff (mm)	Total estimated runoff* (mm)	Increase (mm)	Increase as % of estimated runoff*	Total runoff (mm)	Total estimated runoff* (mm)	Increase (mm)	Increase as % of estimated runoff*
1998	121	231	115	116	100	287	87	200	231
2010	47	175	45	130	293	111	34	46	137
2011	184	246	175	71	41	220	132	88	67

* Estimate of runoff from the catchment had it remained undeveloped, using either Equation 1 or 2.

Table 2. Annual maximum series runoff for the three wettest hydrological years of the study, compared to estimated runoff from C2 and C3 had they not been developed.

	C1	C2 - cropping				C3 – grazed pasture			
Event	Total runoff (mm)	Total runoff (mm)	Total estimated runoff* (mm)	Increase (mm)	Increase as % of estimated runoff* (mm)	Total runoff (mm)	Total estimated runoff* (mm)	Increase (mm)	Increase as % of estimated runoff* (mm)
Apr 1998	50	84	47	37	77	115	36	79	222
Feb 2010	44	63	42	21	51	50	31	18	59
Dec 2010	183	224	175	49	28	197	131	65	50

* Estimate of runoff from the catchment had it remained undeveloped, using either Equation 1 or 2.

Flow duration curves for Stage I show that total daily flows from C1 exceed those of C2 and C3 across the full range of probabilities (Figure 2). During Stage III, low flows (flows exceeded for >70% of the time) show similar probabilities of occurrence. However, flows exceeded for <25% of the time in C2 and C3 had greater daily totals than C1 (Figure 2).

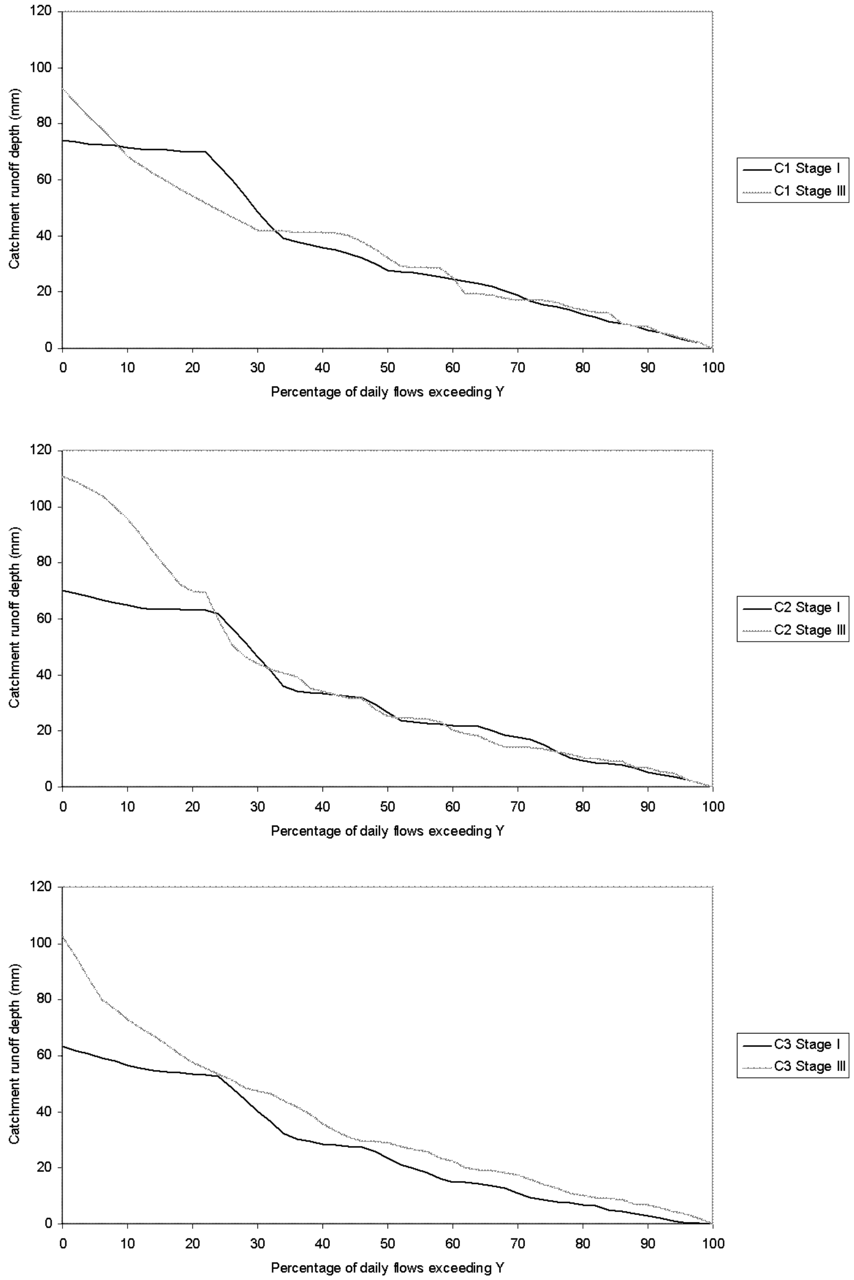

Figure 2 Flow duration curves for the three catchments in both Stage I and Stage III. The curves are derived from daily 9 am total catchment runoff depth data.

4. DISCUSSION

The paired, calibrated catchment study approach shows that increases in runoff from catchments developed for cropping or for grazing on improved pasture can be seen for extreme wet years and for individual high flow events. During 2011, the wettest year on record, with the highest ever total annual runoff from C1, total annual runoff increase from C2 was 169% of the long-term mean annual increase (71 mm cf. 42mm), while total annual increase from C3 was 231% of the long-term mean annual increase (88 mm cf. 38 mm) (Thornton *et al*, 2007). On an individual event basis, the increase in runoff from C2 as a result of land development was 117% of the long-term mean annual increase, while the increase in runoff from C3 as a result of land development was 171% of the long-term mean annual increase.

This pairing of events as a consequence of identical climatic sequence, irrespective of land use or location is robust for a number of reasons. Firstly, all flows were considered in developing the regression equations 1 and 2 to describe runoff from the catchments in their native condition. Secondly, when using these equations to estimate flows from C2 and C3 had they not been developed, the corresponding runoff from C1 was within the range of observations used to generate equations 1 and 2 for all events except for the largest on record. Thirdly, when considering the three years of data presented in the annual approach, all events were considered.

Implicit in its approach, annual maxima analysis does not consider all events. It may also suffer from the maxima event in one catchment in a given year being generated from a different climatic sequence to the maxima in an adjacent catchment. This was not the case in this analysis, with the annual maxima in all catchments generated from the same climatic sequence, making this analysis simply a subset of all three years of paired, calibrated catchment data. While it may be argued that inferring from a small sample of events that increases in runoff as a result of land development are still present in flood events, this pattern reflects that found in the long-term data from this site, and continues to hold for the largest event on record.

In addition to examining increases in flood magnitude as a result of land development, flow duration curves for C2 and C3 indicate higher daily flow totals for all probabilities of occurrence, as a result of land development. The greatest increase in magnitude is shown in flows occurring less than 25% of the time.

5. ACKNOWLEDGEMENTS

This work was jointly funded by the Department of Natural Resources & Mines and the Reef Rescue Research and Development Program (Project RRRD009) of the Australia Government's Caring for our Country initiative. The authors thank past and present staff from the Department of Natural Resources and Mines and the Department of Agriculture, Fisheries and Forestry that worked on the Brigalow Catchment Study.

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Appendix 1.4: Publication 4

The Brigalow Catchment Study: Effects of land development on peak runoff rate and its prediction in central Queensland, Australia

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Abstract

Commencing in 1965 and continuing today, the Brigalow Catchment Study in central Queensland has measured both runoff volume and peak runoff rate from three catchments (11.7 to 16.8 ha) which were initially covered with native brigalow scrub. Thirty-eight years of data were used to assess the accuracy of three different methods for estimating peak runoff rate, and then, to quantify the changes in peak runoff rate as a result of clearing two of the three catchments for either cropping or pasture.

Three methods were used to estimate the peak runoff rate for the 3 catchments: 1) multi-variable regression; 2) Soil Conservation Service Curve Number (SCS-CN) method; and 3) Spatially Variable Infiltration model (SVIM). Regression analysis showed that peak runoff rate was strongly correlated with total runoff volume, in addition to 3 rainfall related variables. Regression method yielded the best results of the three methods tested. Estimates using the SCS-CN method were strongly correlated with observed peak runoff rate for all catchments. However, the method typically underestimated for small events and overestimated for large events. Estimates using SVIM with linear kinematic wave routing were strongly correlated with observed peak runoff rate for all catchments.

Prior to land use change, peak runoff rate from the three brigalow scrub catchments averaged 3.4 mm/hr. A significant increase in peak runoff rate from both the cropping and pasture catchments was clearly evident and attributed to land use change, with the average peak runoff rate increased by 9.1 mm/hr for the cropping catchment and 3.4 mm/hr for the pasture catchment. This equates to an increase of 263% and 164%, respectively, to estimates of their peak runoff rate had they not been cleared. Increases in the peak runoff rate were largest for smaller storm events with an average recurrence interval of less than two years under cropping and less than four years under pasture.

1. INTRODUCTION

Measurement and estimation of runoff volume (Q_{tot}) and peak runoff rate (Q_p) have been the focus of substantial hydrological research worldwide. In Queensland, large tracts of native vegetation have been cleared for agriculture, resulting in substantial hydrologic change in the landscape. Australia's longest-running paired catchment study, the Brigalow Catchment Study (BCS), was established in 1965 to monitor hydrologic change associated with land development, particularly that of the 1960s Land Development Fitzroy Basin Scheme. The Scheme was the largest Government sponsored land development initiative in Australia, resulting in the clearing of 4.5 Mha of brigalow lands for cropping or grazing (Cowie *et al*, 2007). The BCS has unequivocally shown that developing brigalow for cropping or for grazing doubles runoff volume (Thornton *et al*, 2007). However, to date, little research has been done to examine suitable techniques for estimating Q_p, or to quantify the changes in Q_p when brigalow is cleared for cropping or for grazing.

Available methods for estimation of Q_{tot} and Q_p vary both in complexity and in the applicability of their results spatially. Locally accurate site-specific models have been derived empirically from research plots (Fentie *et al*, 2002). However, location-specific and empirical models may not be suitable for use in other catchments due to input data requirements or because the scale at which the relationships were derived is not representative of runoff processes in the broader landscape. Simple models

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requiring fewer, easily obtainable parameters may be more readily applicable to other catchments (Boughton, 1995), but the accuracy of Q_{tot} and Q_p estimations from this method may be poor or the resolution too coarse for certain applications. Complex physical process-based models may provide accurate estimations of Q_{tot} and Q_p at suitable resolution, however the application of these models to a new location is potentially hampered by the availability of data for parameterisation (Post & Jakeman, 1999).

Objectives of the research were 1) to evaluate various methods for peak runoff estimation; 2) to determine the effect of land use change on peak runoff rate for these small rural catchments. For this paper, Q_p was estimated using three methods; (1) multi-variable regression analysis; (2) Soil Conservation Service Curve Number (SCS-CN) and graphical peak discharge methodologies; and (3) a spatially variable infiltration model (SVIM). The best of the three methods was used to estimate Q_p when data were missing. Changes in peak runoff rate due to land development were evaluated by comparing various aspects of the peak runoff rate between catchments.

2. METHODS

2.1. Experimental site

The BCS is a paired, calibrated catchment study consisting of three catchments, and was established in 1965 to determine the impact on hydrology, productivity and resource condition when brigalow land is cleared for cropping and grazing. The study rationale, aims and history along with physical characteristics including location, experimental design, climate, vegetation and soils have been thoroughly documented elsewhere (Lawrence & Sinclair, 1989; Cowie *et al,* 2007; Radford *et al,* 2007; Thornton *et al,* 2007). A brief description of the site and experimental treatments follows.

The BCS (24.81°S, 149.80 °E) lies in the south eastern section of the northern brigalow bioregion and is contained within the Dawson subcatchment of the Fitzroy Basin, central Queensland, Australia. The climate is a semi-arid to sub-tropical and has mean rainfall of 697 mm during the study period. Mean annual evaporation is 2100 mm/year, and exceeds mean monthly rainfall in all months.

Soil types in the catchments comprise associations of Black and Grey Vertosols, some with gilgais, Black and Grey Dermosols, and Black and Brown Sodosols (Isbell, 1996). Clay soils (Vertosols and Dermosols) occupy approximately 70% of catchments 1 and 2, and 58% of catchment 3, while Sodosols occupy the remaining area. Mean slope of the catchments is 2.5%. Before clearing, the catchment 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 spp.* either exclusively, or in association with *Eremophila spp.* or *Myoporum spp.*

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. Water height through the flumes was recorded using mechanical float recorders. Rainfall was recorded adjacent to each flume and at the head of the catchments.

The study has been divided into three distinct experimental stages (Table 1). During Stage I the three catchments were retained in their virgin state. Rainfall and runoff data were collected to describe differences in catchment hydrological responses to a range of weather sequences.

Stage II commenced in March1982 when catchment 2 (C2) and catchment 3 (C3) were cleared with bulldozer and chain. The fallen timber was burnt in-situ in October of the same year. Residual unburnt timber in C2 was raked to the contour line and burnt. Narrow based contour banks at 1.5m 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. Catchment 1 (C1) was left untouched as a control.

In C2, cropping commenced in September 1984 with the planting of sorghum 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 or winter crop (sorghum and wheat or barley) 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 equivalent), adjusted to maintain pasture dry matter levels greater than 1000 kg/ha. There was no feed supplementation.

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

		Land use by experimental stage		
		Stage I	Stage II	Stage III
Catchment	Area (ha)	(Jan 1965 to Mar 1982)	(Mar 1982 to Sep 1984)	(Sep 1984 to Dec 2004)
1	16.8	Virgin brigalow scrub	Virgin brigalow scrub	Virgin brigalow scrub
2	11.7	Virgin brigalow scrub	Development	Cropping
3	12.7	Virgin brigalow scrub	Development	Improved pasture

2.2. Methods of analysis

Estimation of Q_p was undertaken using three methods: (1) multi-variable regression analysis; (2) Soil Conservation Service Curve Number and graphical peak discharge methodologies; and (3) a spatially variable infiltration rate model.

Multiple regression analysis was used to explore relationships between Q_p and independent variables describing climate and catchment condition. All models considered the parameters Q_{tot}, total rainfall (P), storm energy (E), storm erosivity (EI30), rainfall intensity (I) (peak intensity at 6, 10, 15, 20 and 30 min and 1, 2, 3, 4, 6, 12, 18, 24 hr intervals), antecedent rainfall total (A) (2, 3, 5, 10, 20 and 30 day), and total soil water (TSW). Each parameter was tested individually for a significant correlation (*p*-value <0.05) with Q_p. Significant parameters were then combined and an all-subsets regression performed.

Curve Number (CN) and graphical peak discharge (GPD) method for estimation was undertaken using locally calculated CN values, time of concentration calculated using the SCS lag method, a type II rainfall distribution, and equation based calculation of unit peak discharge (USDA NRCS, 1986). Spatially Variable Infiltration model (VIR) estimations were undertaken on a 15 minute time step, also using time the of concentration calculated by the SCS lag method, and with a linear solution to the kinematic wave approximation used to route rainfall excess to the catchment outlet (Yu 1997; Yu *et al* 1997).

Evaluation of changes in peak runoff rate due to land development using a simple comparison of observed data and calibrated catchment analysis followed the approach of Thornton *et al* (2007).

3. RESULTS

Equations 1 to 3 describe the multiple regression models developed for each catchment during Stage III. Events with Q_p > 1 mm/hr were better estimated than events with Q_p < 1 mm/hr (Figure 1). Irrespective of catchment or stage, Q_{tot} gave the best correlation of an individual variable with Q_p. Models with the single variable Q_{tot} resulted in a significant regressions for all catchments (p-value <0.001) with minor or no reduction in R^2 (0.93, 0.80 and 0.83 for C1, C2 and C3 respectively) compared to multi-variable models.

$$\text{C1 Stage 3} \log Q_p = 0.7095 \times \log Q_{tot} + 0.02266 \times I_{2hr} - 0.491 \quad (R^2 = 0.93) \quad (1)$$

$$\text{C2 Stage 3} \log Q_p = 0.7966 \times \log Q_{tot} - 0.02568 \times P + 0.1192 \times E \quad (R^2 = 0.89) \quad (2)$$

$$\text{C3 Stage 3} \log Q_p = 0.5692 \times \log Q_{tot} + 0.01832 \times I_{1hr} - 0.3335 \quad (R^2 = 0.87) \quad (3)$$

The GPD method gave good estimations of Q_p across all catchments and stages (Figure 2). Linear regression analysis gave R^2 >0.7 in all instances. These high R^2 values disguise the fact that the method typically under-estimates Q_p in small events and over-estimates Q_p in large events. On average, in events where $Q_{p\text{-observed}}$ >5 mm/hr, 71% of estimations were greater than the observed data.

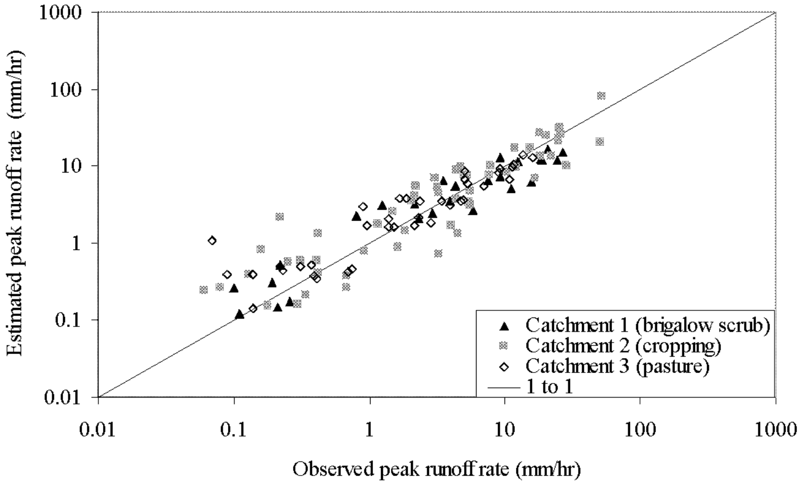


Figure 1. Observed peak runoff rate compared with estimated peak runoff rate using multiple regression models (Equations 1 to 3) for the three catchments during Stage III.

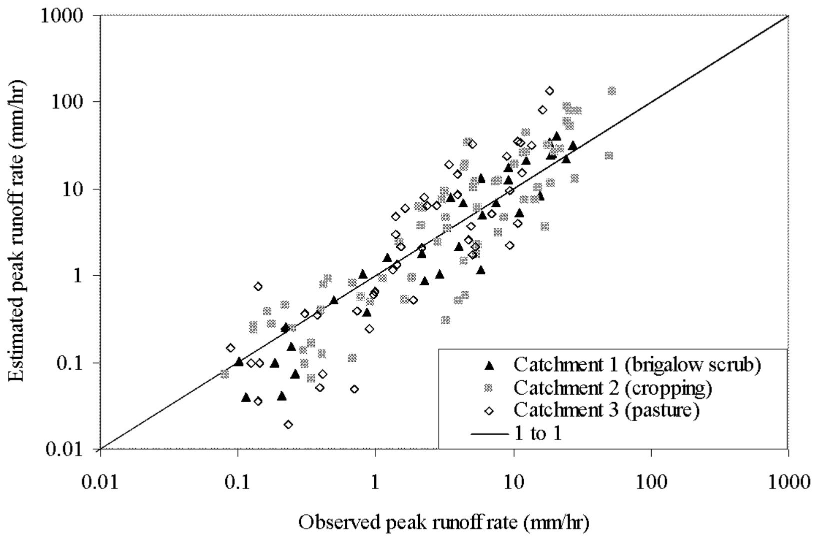


Figure 2. Observed peak runoff rate compared to GPD method estimated peak runoff rate for the three catchments during Stage III.

The SVIM method gave better estimates of Q_p compared to the GPD method, with linear regression analysis giving R^2 >0.8 for all catchments in Stage III (Figure 3). However, the lag in the routing component was too short, with 94% of all routed peaks occurring prior to the observed peak.

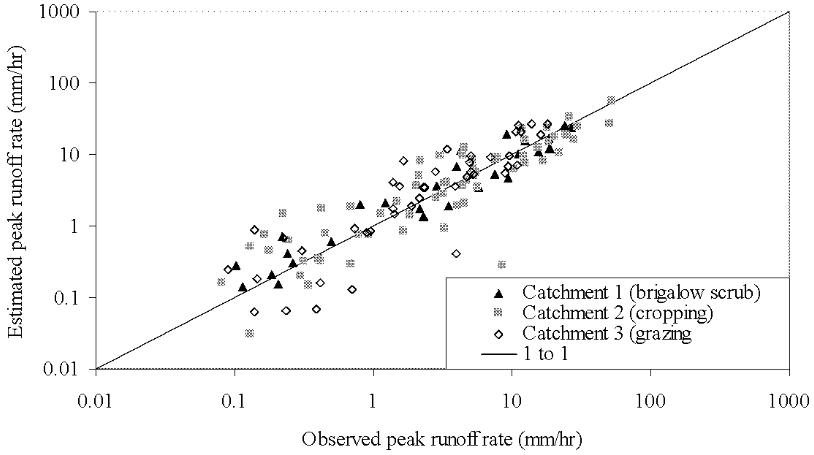


Figure 3. Observed peak runoff rate compared to the VIR method estimated peak runoff rate for the three catchments during Stage III.

Average observed Q_p for the three catchments in both Stage I and III are shown in Table 2 using the complete data set. Analysis of variance of observed Q_p showed a significant increase in average Q_p between Stage I and III for all catchments (p-value <0.05). Regression analysis showed strong correlation of Q_p between the catchments in Stage I (Figure 4) (Equations 4 and 5).

Table 2. Summary of observed peak runoff rate.

Catchment	Stage	Total number of events	Average Q_p (mm/hr)	Maximum Q_p (mm/hr)
1	I	36	3.0	31.7
	III	37	6.6	27.0
2	I	34	4.8	33.5
	III	72	14.7	52.7
3	I	73	1.9	28.7
	III	60	8.8	50.2

$$\log Q_p \text{ C2 (mm/hr)} = \log Q_p \text{C1(mm/hr)} \times 0.9431 \qquad (R^2 = 0.99, n = 25) \qquad (4)$$

$$\log Q_p \text{ C3 (mm/hr)} = \log Q_p \text{C1(mm/hr)} \times 0.8176 + 0.2303 \qquad (R^2 = 0.92, n = 24) \qquad (5)$$

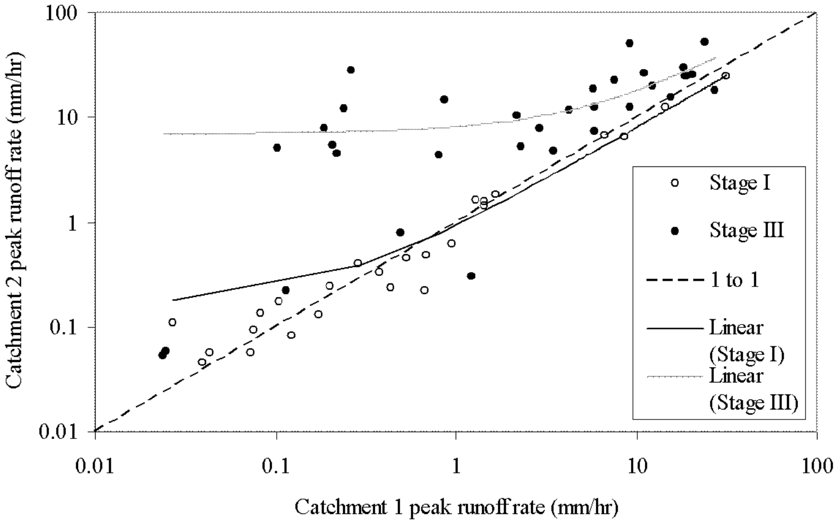
However, the correlation was much weaker In Stage III (Figure 4) (Equations 6 and 7).

$$\log Q_p \text{ C2 (mm/hr)} = \log Q_p \text{C1(mm/hr)} \times 0.686 + 1.289 \qquad (R^2 = 0.50, n = 32) \qquad (6)$$

$$\log Q_p \text{ C3 (mm/hr)} = \log Q_p \text{C1(mm/hr)} \times 0.499 + 1.185 \qquad (R^2 = 0.36, n = 19) \qquad (7)$$

Equations 4 and 5 were used to estimate Q_p from C2 and C3 in Stage III, had they not been cleared. Both catchments showed a trend of larger observed Q_p than that estimated by their pre-clearing behaviour (Figure 4). In C2, 94% of events had a higher Q_p while in C3, 80% of events had a higher Q_p. Observed average Q_p from C2 was 14.7 mm/hr, an increase of 9.1 mm/hr from its estimated Q_p of 5.6 mm/hr had it not been cleared, while the observed average Q_p from C3 was 8.8 mm/hr, an increase of 3.4 mm/hr from its estimated Q_p of 5.4 mm/hr had it not been cleared. The maximum increase in Q_p was 43 mm/hr in C2 and 34 mm/hr in C3.

In addition to direct comparison of the Q_p for the same set of storm events, partial series of peak runoff rates were prepared for each catchment in each of the two stages. For given average recurrence interval, the ratios of C2 over C1 and C3 over C1 show the change in peak runoff rate with the same frequency of occurrence (Figure 5). It is clear that the change in Q_p with land development is most prominent in events with a short average recurrence interval (Figure 5). Under cropping, events with an average recurrence interval greater than two years had similar ratios between the two stages, indicating less change in Q_p with land development during large storm events. Grazed pasture exhibited a similar trend, with little change in the ratios of Q_p for events with an average recurrence interval greater than four years.



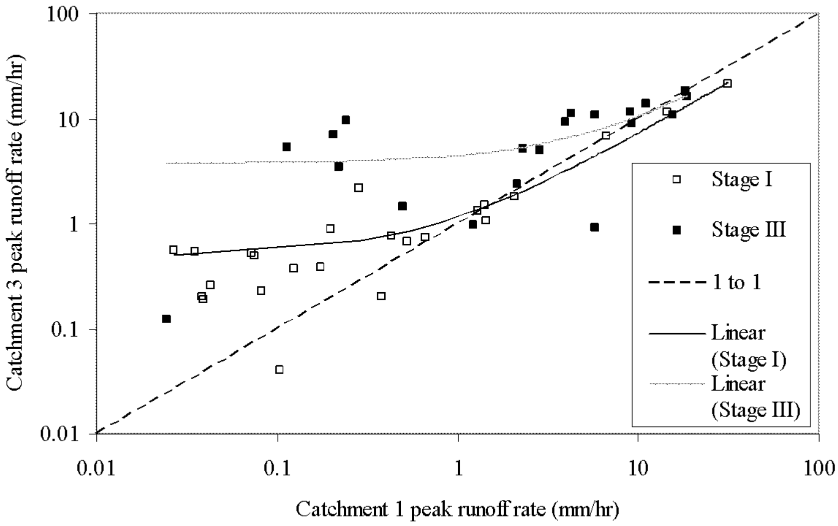


Figure 4. Peak runoff rate for C2 and C3 compared to C both pre- and post-clearing.

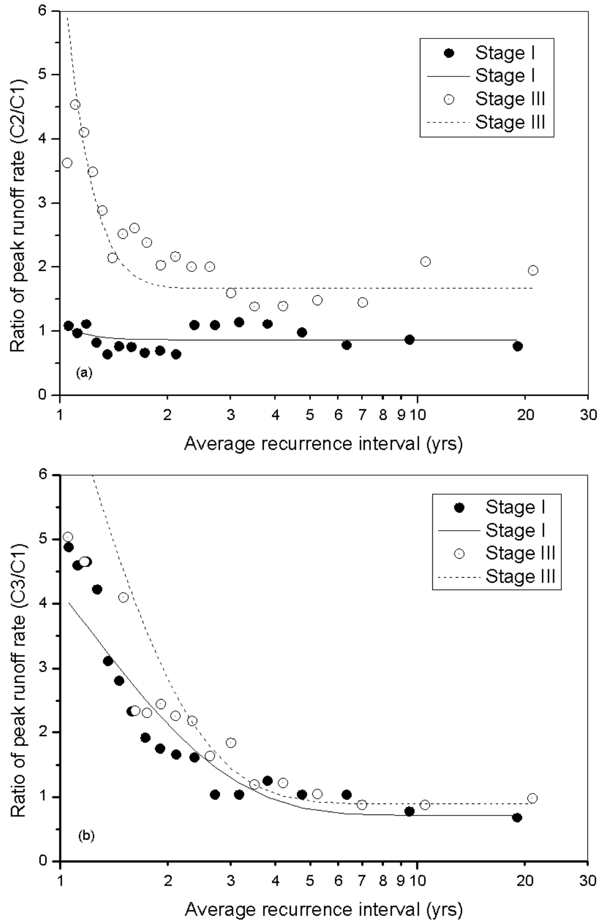


Figure 5. Ratios of peak runoff rate for C2:C1 and C3:C1 pre- and post-clearing.

4. DISCUSSION

Numerical assessment of model performance using R^2 indicates that the site-specific multiple regression models gave the best estimation of Q_p, followed by the SVIM and the SCS-CN methods. This assessment clearly indicates that multiple regression models have given the best estimations of Q_p, when data on observed peak runoff rate are available for model calibration, and when a method is needed to fill in the missing data. For ungauged sites, SVIM and the SCS-CN methods can be used to estimate peak runoff rate when data on rainfall intensity and catchment characteristics are available,

The literature shows that changes in Q_{tot} generally change Q_p (Leitch & Flinn, 1986; Bari & Smettem, 2006), and that the direction of change in Q_{tot} is generally mirrored by the change in Q_p (Rallison, 1982). As development of brigalow scrub to either cropping or pasture has been shown to have increased Q_{tot}, it would be expected that Q_p would also increase due to land use change. A simple comparison of observed Q_p data in this paper did show an increase in Q_p. Calibrated catchment methodology supported this, showing land use change increased average Q_p by 9.1 mm/hr to 14.7 mm/hr for the cropping catchment (C2) and by 3.4 mm/hr to 8.9 mm/hr for the pasture catchment (C3).

This supports the earlier conclusions of Lawrence & Sinclair (1989) who, when analysing study data from 1984 to 1987, found average increases in Q_p of 9.5 mm/hr in C2 and 4.3 mm/hr in C3. This trend is reflected internationally, where typically, higher Q_p are observed from agricultural watersheds compared to forested watersheds (Cox *et al*, 2006). Events with an average recurrence interval of less than two years showed the greatest increase in Q_p when brigalow land was developed for cropping, while events with an average recurrence interval of less than four years showed the greatest increase when brigalow land was developed for grazing.

5. ACKNOWLEDGEMENTS

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Appendix 1.5: Publication 5

Appendix 14

Nitrogen generation rates from grass pastures over-sown with butterfly pea or leucaena in the Burnett-Mary and lower Burdekin catchment.

Craig Thornton, Bruce Cowie, Luke Davison, Wendy Tang, Amanda Elledge

SUMMARY TABLE:

Catchment: Burnett Mary (Brian Pastures Research Station), Burdekin (Clermont)
ASC: NA **SALI:** BPRS – BPS29, Clermont – KCM883

Site Locality	**Season**	**Pasture cover**	**N Loss (kg/ha)**	
			TN	**DIN**
Brian Pastures Research Station (BPRS)	Late Dry	Grass	0.26	<LOR
		Butterfly Pea	0.50	0.03
		Leucaena	0.27	0.05
	Late Wet	Grass	0.05	<LOR
		Butterfly Pea	0.07	<LOR
		Leucaena	0.03	<LOR
Clermont	Late Dry	Grass	0.05	<LOR
		Butterfly Pea	0.45	0.04
		Leucaena	0.70	0.03
	Late Wet	Grass	0.47	0.04
		Butterfly Pea	0.83	0.02
		Leucaena	0.81	0.05

Background

The trend for grazing legume introduction to grass only pastures has well documented productivity benefits but the effects on runoff nitrogen loads are unknown. Two pasture legumes with a rapidly expanding footprint in the Burnett Mary and Burdekin catchment are butterfly pea and leucaena. The expansion of leucaena and butterfly pea has been driven by the broader grazing industry.

This project aims to determine whether i) a grass pastures over-sewn with butterfly pea or leucaena generates higher N runoff concentrations or loads than a grass only pasture, and ii) whether there is a seasonal effect on runoff N generation for these three pasture types.

These findings will support data collected from the Brigalow Catchment Study where runoff nitrogen generation from native Brigalow scrub, grass leucaena pasture and grass only pasture are compared.

Methods

At two locations (Brian Pastures Research Station and Clermont) runoff nitrogen generation rates were measured on three pasture types (grass only, grass and butterfly pea and grass and leucaena) in the late wet and late dry season.

Paired rainfall simulation plots (1.7 metres long by 1.0 metre wide) with two replicates received simulated rainfall (average 80 millimetres per hour) until runoff commenced and then for a further 30 minutes. Samples were collected during the runoff event to determine runoff rate and nitrogen concentration in runoff (refer to Appendix 15 pictures 10 – 12 and 16)

Results

The soil moisture is higher at BPRS than Clermont, but within each site the soil moisture is similar regardless of pasture cover. **Figure 1** shows similar or higher initial infiltration in the late wet season at BPRS and Clermont.

Low ground cover in the late dry season, runoff total nitrogen (TN) loss is expected to be higher. This trend is seen at the Brian Pastures Research Station (BPRS) and Clermont sites (see **Figure 2**). Trials conducted during the late dry season, grass over-sewn with butterfly pea had the highest runoff TN loss at both BPRS and Clermont. Grass over-sewn with leucaena had similar runoff TN loss to grass at BPRS.

Cracking clay soils in the late wet season increases infiltration such that runoff TN is expected to be lower.

The general runoff TN loss was higher in the late dry season at BPRS and Clermont, regardless of pasture cover (**Figure 2**). Within the late wet season at BPRS and Clermont, all three pasture covers had similar runoff TN losses.

The late dry season would generate higher runoff TN losses, where grass over-sewn with butterfly pea would generate the highest loss. There was no notable difference in runoff TN losses in the late wet season.

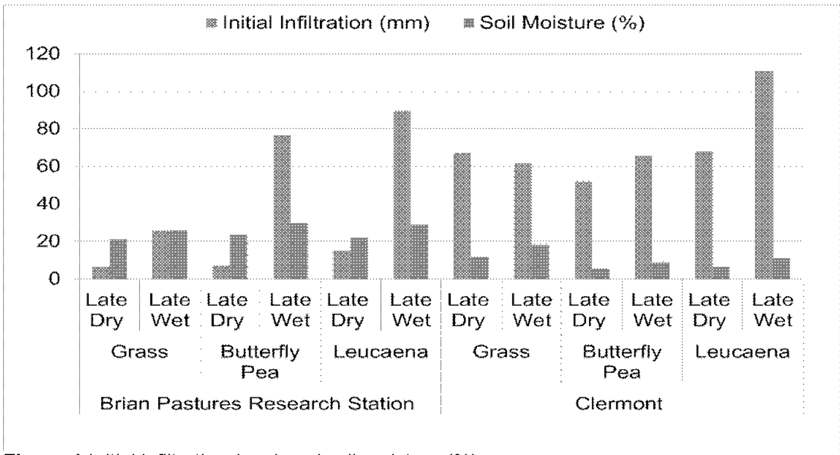

Figure 1 Initial infiltration (mm) and soil moisture (%).

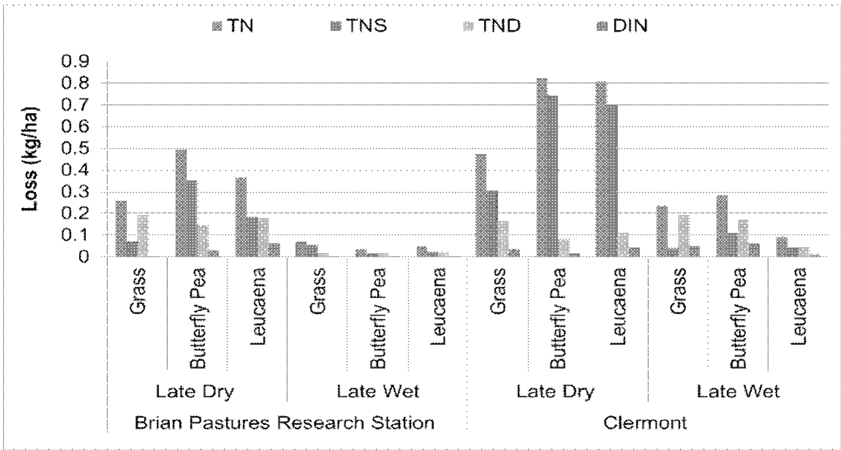

Figure 2 Nitrogen loss (kg/ha) at Brian Pastures and Clermont.

Key Findings

- There was a notable effect of season on the nitrogen loss at the sites planted with butterfly pea and leucaena with higher TN losses observed in the late dry season.
- There was no obvious late wet season effect on the runoff N losses at BPRS and Clermont.

Conclusions

The presence of leguminous pastures will yield higher nitrogen loss loads in the late dry season. Furthermore, the presence/absence of leguminous pastures and the effect on nitrogen loss loads is less evident in the late wet season.

Partners

- Department of Natural Resources & Mines
- Land owners in the Clermont district who allowed access to experiment sites.
- Staff at the Brian Pastures Research Station