Sustainable grazing for a healthy Burdekin catchment

finalreport

Project code: B.NBP.0473

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Date published: May 2009

ISBN: 978 1 741 91364 4

PUBLISHED BY
Meat & Livestock Australia Limited
Locked Bag 991
NORTH SYDNEY NSW 2059

Can improved grazing land management reduce sediment yields delivered to the Great Barrier Reef?

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Abstract

This publication presents the results of Meat and Livestock Australia (MLA) project B.NBP.0473. This project assessed the grazing impacts on rangeland health looking specifically at the relationship between ground cover and the loss of water, sediments and nutrients from hillslopes and catchments. At the completion of this project, CSIRO was to have:

1) Evaluated the persistence and magnitude of pasture recovery and associated reductions in sediment and nutrient yields on existing field sites at Virginia Park;
2) Recommended options for recovery of scalded areas that warrant field testing;
3) Removed hydrologic equipment from Blue Range, Station Creek and Meadowvale sites for reconditioning and storage.

Task (3) has been completed, and the results from tasks (1) and (2) for sediments are outlined in this document. This one year extension project has built upon a previous 8 years of MLA funded research in the Burdekin catchment. Results from earlier components of this study can be found in Roth et al., (2003), Post et al., (2006) and Bartley et al., (2007a). Due to the timing and of the events in the 2007/08 wet season, and ability to access the field sites, very few nutrient samples were collected. The results from the nutrient analysis did not provide any further information from that published in Bartley et al., (2007a). Given the lack of additional nutrient data, and to maintain this report at a length suitable for journal publication, this document focuses on runoff and sediment loss only.

A number of findings from this study are important for graziers wanting to manage vegetation and the associated soil loss from their properties. These were outlined in the Burdekin brochure series posted to all Burdekin graziers in 2007. The results from this study are also important for the wider research and policy community both in Australia and overseas, and these findings will help guide the target setting process as part of ‘Reef Rescue’ and the ‘Caring for Country’ programs being rolled out across Queensland in 2009. To help communicate the results of this research to a broader scientific audience this final publication has been written in the format of a scientific journal paper rather than a report. The paper will be submitted to an appropriate journal once we have final confirmation from MLA.

Executive Summary

Poor land condition resulting from unsustainable grazing practices can reduce enterprise profitability and increase water, sediment and associated nutrient yields from properties and catchments. This report presents the results of an 8 year field study that evaluated the impact of grazing land best management practice (BMP) on a 13 km^2 sub-catchment of the Burdekin River in Northern Australia. Land condition recovery and changes to runoff and sediment yield were measured on hillslopes (using three flumes) and at the end of catchment (using automatic water sampling).

At the hillslope scale, average ground cover increased on all sites (from ~35% to ~75%), although biomass levels are still relatively low for this landscape type (60 to 1100 kg/ha). Further improvements in cover (to ~85%) and biomass (to ~1700 kg/ha) are recommended before this site can be considered to be in ‘good’ condition. At the catchment scale, the area of land with < 10% cover decreased from approximately 10.2% to 4.5%. Most of this recovery was on the upper and middle parts of hillslopes. The low-cover areas that did not respond to grazing management were on the lower slopes associated with the location of sodic soil and the initiation of gullies. Comparison of ground cover changes with adjacent properties suggest that grazing management, and not just improved rainfall conditions, were responsible for the improvements in cover in this study.

Hillslope runoff did decline over the study period for early wet season events, up to ~200 mm of rainfall, but after this point the amount of runoff was no longer strongly related to the amount of cover on the hillslope. Hence there was no reduction in hillslope runoff at the annual time scale with the improved cover. This is attributed to limited soil hydrological capacity, and suggests that soil condition is recovering at a slower rate than ground cover. The hillslope sediment yields declined by ~70% on two out of three hillslopes, however, where bare patches (with < 10% cover) are connected to gullies and streams, sediment yields increased. Extrapolation of the hillslope results to the catchment scale show that hillslope sediment yields did not decline between 2003 and 2007. This is due to the disproportionately high yields from scald sites particularly in high runoff years. In 2007, when there was above average rainfall, 83% of the hillslope derived fine sediment was coming from less than 5% of the catchment.

At the end of the catchment, sediment yield did not decline, and actually increased, associated with increases in rainfall and runoff during the study period. The difference in sediment yield response between hillslope and catchment scales is attributed to the contribution from gully and river bank erosion. The event mean concentration (EMC) of suspended sediment had a significant decreasing trend, however, this is appears to be a function of increasing runoff. This study has demonstrated that it is difficult to detect a change in end of catchment sediment yields in response to changed grazing intensity when the dominant sediment source is subsoil erosion. It may be possible, given enough time, that grazing land management (GLM) will produce the biomass and runoff reductions required to reduce channel erosion in this catchment. Unfortunately the time lines associated with this change are unknown, and the recovery times (assuming recovery is possible in a commercial setting) are likely to be longer than ‘target’ timelines being set by the Reef Plan. Rehabilitation of scald and gully sites are likely to be an important companion to GLM if sediment yield targets are to be met. Research into the appropriate methods and effectiveness of gully and scald rehabilitation, including the economic feasibility of such options, are needed. In summary, the ground cover improvements are likely to be advantageous for pasture growth and animal production (in the short term), however, this ground cover recovery is fragile. It is recommended that grazing BMP is maintained (and where possible increased) to facilitate improved infiltration which will help maintain pasture growth. This will also indirectly reduce the hillslope runoff that is fuelling the scald, gully and bank erosion that is impacting on downstream water quality.

Acknowledgements

The research presented in this publication was funded by Meat and Livestock Australia, CSIRO and eWATER CRC; their support is gratefully acknowledged. We also thank Rob and Sue Bennetto on 'Virginia Park' Station and the Ramsay Family on 'Meadowvale' Station for access to their properties over the last 9 years to carry out this work. Thanks also to Dr David Post and Anne-Lise Koch-Lavisse for analysis of gauge data, and to Peter Fitch, Rex Keen, Jamie Vleeshouwer, Joseph Kemei, Lindsay Whiteman of CSIRO and the late Peter Allen for the installation of field equipment and collection of samples. To David McJannet, Brendan Farthing and Freeman Cook for advice on hydrological and statistical analysis, and to Drs Christian Roth and John McIvor for comments and review on early versions of this manuscript.

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1 Introduction

Livestock grazing is Australia's largest land use occupying 58% of the continent (www.brs.gov.au/landuse). In many grazing areas poor land condition resulting from unsustainable grazing practices has reduced the productivity of land for beef production, and also increased water, sediment and nutrient yields leaving the landscape (e.g. Bartley et al., 2007b; McKeon et al., 2004). Evidence suggests that excess sediments and nutrients can also impact on the water quality and ecology of adjacent rivers and streams (e.g. McIver and McInnis, 2007; Vidon et al., 2008) and downstream ecosystems such as the Great Barrier Reef (GBR) (Fabricius, 2005; Fabricius et al., 2005; McCulloch et al., 2003).

Sediments are delivered to streams from three main processes (hillslope, gully or bank erosion). Hillslope erosion is the process that has received the most attention in the last decade in rangeland regions of northern Australia as it is the management unit of interest to most graziers (e.g. 'the paddock'). There have been a number of studies quantifying the amount of water and sediment lost from hillslopes in Australian rangelands (e.g. Bartley et al., 2006; McIvor et al., 1995; Scanlan et al., 1996) and internationally this has been a well researched field (e.g. Branson et al., 1972; Stone et al., 2008). Trimble and Mendel (1995) provide a thorough review on the range of impacts grazing and cattle can have on catchment processes including soil hydrology, hillslope runoff, bank erosion and stream channel structure. These studies have described the degradation process, however, few studies have looked at landscape recovery following cattle exclusion or reduction. For the few international studies that describe recovery, the rates of recovery vary considerably from 2.5 years for phosphorus and sediment loads (Line et al., 2000) to between 3 to 13 years for hillslope hydrology (Branson et al., 1981; Sartz and Tolsted, 1974).

In Australia, previous studies have evaluated whether changes to land management affect ground cover and land condition, particularly in a historical context (e.g. Ash et al., 2001; Bastin et al., 2001; McKeon et al., 2004). Another study found that sediment yields from hillslope plots were reduced by 50% after one year of cattle exclusion (Hawdon et al., 2008). A number of studies have attempted to link pasture condition changes to changes in water quality at the end of the catchment (e.g. O'Reagain et al., 2005), however, very few studies have had the appropriate study design or long enough data sets to provide significant results. Given that grazing lands represent ~ 76 % of the catchment area draining to the GBR (Furnas, 2003), there is a need to determine if grazing best management practice will lead to reductions in the amount of sediment not only leaving the hillslope, but reaching downstream rivers and coastal regions.

There is an increased interest in improving land management and reducing impacts to downstream ecosystems. In 2008, the Australian Government allocated $200 million, via the Reef Rescue package, to help land owners and managers implement improved land management practices to reduce the amount of nutrients, chemicals and sediments leaving their farms and impacting on Reef water quality (http://www.nrm.gov.au/funding/2008/reef-rescue.html). This investment is based on the assumption that improved land management practices will reduce sediment and nutrients delivered to downstream water bodies, yet there are very few studies that have measured this link, and the magnitude and timescales associated with the response are not well understood. There is also an increase in the number of studies that use sediment budget models (e.g. SedNet; Wilkinson et al., 2004) to run 'scenario' analysis to predict changes to downstream water quality from the implementation of best management practice (BMP), catchment changes (e.g. Bohnet et al., 2008) or investment prioritization options (e.g. Lu et al., 2004). There are, however, very little data available to determine if these models are providing sensible responses to given scenarios, and it is acknowledged that the recovery pathway is unlikely to mirror the degradation pathway. It is likely that recovery will follow a new trajectory leading to an alternative and potentially irreversible state in which ecological and hydrological processes operate on a fundamentally different scale to the original intact state (Searle et al., 2009).

The primary focus of grazing land management (GLM) in rangelands is vegetation management (Ash et al., 2001). There are four principal ways to rehabilitate or prompt recovery in rangeland vegetation: (i) reduce stock density (with or without seasonal resting), (ii) prescribed burning, (iii) sowing introduced plant species and (iv) reseeding native plant species (Noble et al., 1984). These methods are considered in the context of stock production and may not be suitable for ecological management and restoration of vegetation communities. In Northern Australia, GLM can be considered as Best Management Practice (BMP) if it is following the recommendations for commercial grazing properties in Northern Australia such as those given in Ash et al., (2001). These include (a) continuous stocking at 25% utilisation; (b) biennial wet season resting regime with an average utilisation of 35% and (c) annual early wet season resting with up to 50% utilisation. Utilisation is defined as the proportion of pasture growth consumed over a year. Wet season resting allows pasture to take advantage of summer rain without grazing.

In this report we present data that links grazing management, ground cover condition and water and sediment loss at the hillslope and catchment scale. This was achieved by establishing a monitoring program at flume sites on Virginia Park Station, in the Burdekin Catchment. In December of 2002, BMP in the form of reduced utilisation, de-stocking and rotational wet season resting grazing strategies, as recommended by the EcoGraze project (Ash et al., 2001) were implemented. For the next 6 wet seasons, changes in land condition and water and sediment runoff were measured. At the catchment scale (13 km^2), runoff and sediment yield were monitored for 8 years to determine if grazing land management changes can be detected at the end of the catchment. The catchment is the scale at which the majority of routine water quality monitoring is presently focused and this is one of the first studies looking at land condition recovery and water quality improvement on a commercial property with continuous grazing (i.e. most previous studies have only evaluated land condition recovery using complete cattle removal). It is important to note that the ground cover and pasture biomass levels at the beginning of this project were considered to be well below ‘sustainable’ conditions for this soil type (Ash et al., 2001). The term BMP is used in this report to define the strategies implemented on the property in 2002, however, ‘best management practice’ does not necessarily equate to ‘good’ or ‘sustainable’ land condition. The results are discussed in the context of water quality target setting and grazing management practises currently undertaken in Northern Australia.

2 Study Area

This study was carried out in the Weany Creek catchment (S19^{o}53’06.79’’, E146^{o}32’06.65’’), which is covered by Eucalypt savanna woodland. The catchment is contained with the Virginia Park station which a privately owned cattle grazing property. The area is representative of the highly erodible ‘gold-fields’ (granodiorite) country between Townsville and Charters Towers in North Queensland, and has been grazed for more than 100 years. Weany Creek is an ephemeral 13 km^2 sub-catchment of the larger Burdekin catchment (~130,000 km^2) in North Queensland, Australia (Figure 1). The Burdekin catchment is the second largest catchment draining into the Great Barrier Reef World Heritage Area (GBRWHA), and a number of studies have shown that sediment discharge from the Burdekin catchment is approximately 5 times greater than prior to European settlement (Furnas, 2003; McCulloch et al., 2003; McKergow et al., 2005; Neil et al., 2002). The Weany Creek catchment was chosen for this study due to its location in an area identified as having high erosion rates (Prosser et al., 2001), but also because of the willingness of the landholders to trial sustainable grazing practices.

The soils in the catchment are generally Red Chromosols on the upper slopes and and Yellow to brown texture contrast soils with dispersive, natric B-horizons on the lower footslopes. Large bare scald patches are present on the colluvial slopes adjacent to many gully and stream networks. The canopy vegetation is composed primarily ironbark/bloodwood communities (e.g. narrow-leafed ironbark, *Eucalyptus creba* and red bloodwood, *Eucalyptus papuana*) which are located primarily on the mid and upper slopes. The lower slope sodic soil communities are dominated by more shrubby species (e.g. currant bush, *Carissa ovata* and false sandalwood, *Eremophila mitchellii)*. The ground cover is dominated by the exotic, but naturalised stoloniferous grass Indian couch (*Bothriochloa pertusa*). Native tussock grasses such as desert bluegrass (*Bothriochloa ewartiana*), Black spear grass (*Heteropogon contortus*) and Golden beard grass (*Chrysopogon fallax*) are present in small numbers within the pasture.

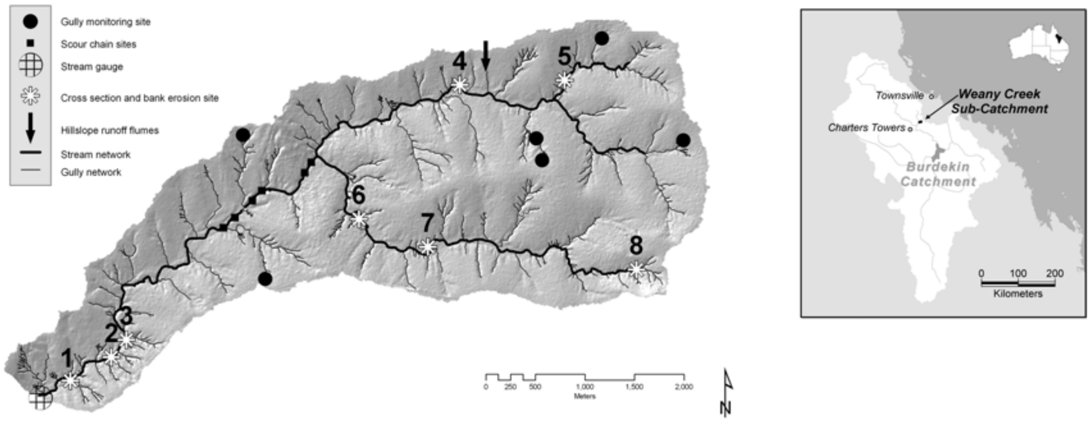


Figure 1: The Weany Creek catchment showing the stream and gully network and the location of field monitoring sites. The catchment outlet is in the southwest corner.

The three primary management practices implemented in this study included adjustment of cattle numbers to match proposed utilisation rates, an initial period of de-stocking and wet season resting in alternate years. A map of the Virginia Park property and the location of the four research demonstration paddocks that are located within the Weany Creek catchment are shown in Figure 2. It is important to point out that this grazing trial was initiated during a drought, on a property with mainly C condition land that was dominated by stoloniforous grass (> 85% Indian Couch).

Utilisation is defined as the proportion of pasture growth consumed over a year (Ash et al., 2001); however, in commercial properties such as Virginia Park this is very difficult to implement on an annual basis. The only way to assess available forage it to estimate the standing dry matter yield. Therefore in this study, utilisation rates were applied based on standing dry matter rather than the amount of pasture grown (see Post et al., 2006 for more detail).

The timing of wet season resting is given in Table 1. Between January 2003 to June 2006 Top Aires and Blackfellas paddocks were stocked to ensure a minimum residual yield of 400 kg of dry matter per hectare (DM/ha) (<35% use of standing dry matter) and 40% ground cover at the end of the dry season. Bottom Aires Paddock was stocked to ensure a minimum residual yield of 500 kg DM/ha (<35% use of standing dry matter) and 40% ground cover at the end of the dry season. Stud Paddock was set up to receive annual wet season rest, however, due to the paddock being very small and located next to the house yards it was often used as a holding paddock and temporarily used during the wet. It had <50% use of standing dry matter. Historical stocking rates for the demonstration paddocks were 1 beast to 4 hectares (ha). From 2003 to 2006 the stocking rates averaged to 1:10 ha. Further details of the stocking rates, pasture utilisation and forage budgeting methods are given in Post et al., (2006). The sustainable grazing treatments formally ended in June 2006 (with the end of project funding for cattle agistment), however, the owners of Virginia Park station have, for the most part, continued moderate stocking and wet season resting regimes until June 2008.

There was a steady increase in the annual rainfall totals at Virginia Park between 2003 and 2007. With exception of the 2006 and 2007 wet seasons all years were under the long term average (1901-2006) for nearby Fanning River rain gauge of ~584 mm (http://www.nrm.qld.gov.au/silo) (Table 1). The flume data for each year is presented as the year at the beginning of the wet season. For example, the wet season that started in 2002 and ended in 2003 is called 2002. Most of the rain falls between December and April each year but occasionally out of season events occur and therefore the hydrology data extends from July 1st to June 31st each year.

Given the experimental design used in this study, it is difficult to determine if the changes measured on the hillslopes and at the end of the catchment are due to the implementation of BMP or due to the increased rainfall over the second half of study. Due the size and logistics involved in this research no control catchments were available. We were, however, able to compare hillslope sediment yield data collected at Virginia Park with data from hillslopes that had experienced restricted grazing. These data, and observations made at other parts of the Burdekin that have not undergone BMP, suggest that the changes observed on the hillslopes in terms of cover and associated runoff and sediment yields are strongly related to the management intervention. Had 2006 and 2007 also received below average rainfall we envisage that there would have been similar results, albeit the trends may not have been as strong in some cases.

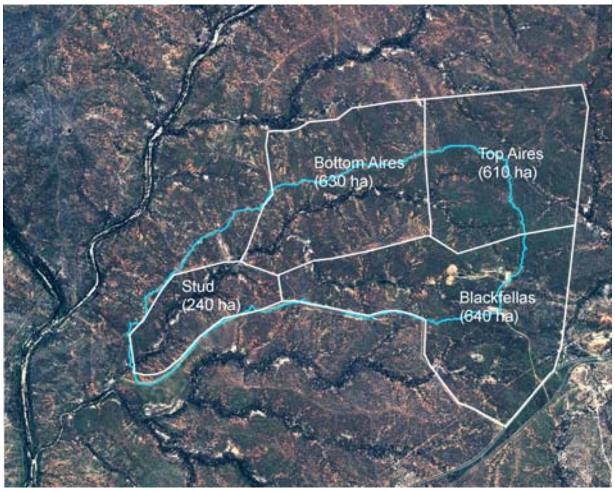


Figure 2: Study location showing the Weany Creek catchment boundary (blue line) and the paddock boundaries on Virginia Park Station (white lines). The background is a pan-sharpened real-colour image derived from the Quickbird™ satellite, taken in December 2003.

Table 1: Timing of wet season resting in each paddock during the study and annual catchment rainfall. The rainfall data for 2000-2001 was from the stream gauge, and for 2002-2007 it is an average of the flume and stream gauge rainfall data.

Paddock	2000-01	2001-02	2002-03	2003-04	2004-05	2005-06	2006-07	2007-08
Top Aires	Na	Na	Wet rest		Wet rest		Wet rest	Wet rest
Bottom Aires	Na	Na		Wet rest	Wet rest		Wet rest	Wet rest
Blackfellas	Na	Na		Wet rest		Wet rest		
Average rainfall (mm)	367	576	292	304	365	495	668	710

3 Methods

3.1 Hillslope monitoring sites

To quantify the linkage between grazing management, land condition and water and sediment loss at the hillslope scale, three hydrological flume hillslope sites were established in 2002. The flumes are located within 400 metres of each other in the Bottom Aires paddock (Figure 1 and Figure 2). There are considerable variations in ground cover pattern within and between the flume hillslopes. There are also differences in vegetation communities between the upper and lower areas of individual hillslopes. The upper and middle slopes are dominated by ironbark-bloodwood (e.g. *Eucalyptus creba)* with couch (*Bothriochloa pertusa)* as the predominant grass. The lower slopes have patches of shrubby vegetation (e.g. *Carissa)* often on or adjacent to exposed sodic soils that have little or no grass cover.

Studies have shown that degraded sites have a larger scale pattern of alternating patches of vegetation and bare ground, and intact rangelands have a finer-scale vegetation structure (Ludwig et al., 2005). Recent research by Searle et al., (2009) suggests that the patchiness of ground cover on grazed hillslopes is a relative measure of structural ecological recovery that can also be used to infer the potential functional recovery of these ecosystems following disturbance by over-grazing.

The patchiness in vegetation cover also varies with the underlying soil and vegetation type. Many riparian areas in Queensland have inherently unstable duplex soils where the clay fraction of the subsoil is high in sodium (Pressland et al., 1988). Long term overstocking on these soils can denude the pasture, remove the A horizon, and expose the dispersible subsoils. This produces areas commonly known as ‘scalds’. It was important in this study to capture some severely degraded areas of pasture as it was hypothesised, and since confirmed, that these scalds contribute a disproportionately high level of sediments to the stream network (Bartley et al., 2006).

In an attempt to capture the different spatial patterns of vegetation for this property, each hillslope has a different vegetation configuration. Flume 2 has a fine grained vegetation arrangement with no large bare patches. Flume 1 is medium grained with a number of bare patches (<6 m^2) and some areas of moderate to high cover. Flume 3 is coarser grained patch arrangement with a large scald or bare patch at the base of the hillslope (>6 m^2) and moderate to high cover at the top of the hillslope (Figure 3). Despite the differences in vegetation pattern, each of the flumes had very similar ‘average’ ground cover at the beginning of the study (see Table 3, Table 4, Table 5).

The hillslope catchment of Flume 1 is ~11,930 m^2 with a mean slope of 3.9% and slope length of 240 m. Flume 2 catchment is ~2031m^2 with a mean slope of 3.1% and slope length of 130 m, and Flume 3 catchment is ~2861 m^2 with a mean slope of 3.6 % and length of 150 m. To determine the area, slope and topography of each flume, the sites were surveyed at approximately 4 × 2 m spacings using a Wild TC 1000 total station. The data were then converted to a DEM profile using TOPOGRID within ArcInfo.

In this study we present some data from Flume 2, however, we predominantly present the results from Flumes 1 and 3 as they are located at the bottom of the hillslope and are therefore more representative of the sediment yields that will enter the stream network. Flume 2 is located at the top of the hillslope and it is not certain if and when sediment generated from this hillslope will enter the stream network. For more detail on the hillslope instrumentation see Bartley et al., (2006).

3.2 Hillslope ground cover and condition monitoring

The flume hillslope ground condition was measured using end of dry season surveys on a 4m×4m grid, with data collected from within a 1m quadrat at each grid point. The grid was later reduced to 8m×4m following initial data analysis. An adaptive sampling method was used whereby additional quadrats were also sampled at patch boundaries to help define patches. Information on vegetation/land type, landscape location, tree canopy cover was also recorded within a 10 metre radius from each sampling point.

Pasture condition metrics recorded at each grid point included the main species and/or functional group composition, biomass, percentage ground-cover, litter-cover, basal-area class, defoliation level and key soil surface condition (SSC). Erosion/deposition status and litter contribution were recorded using relevant BOTANAL (Tothill et al, 1978) and Landscape Function Analysis (LFA) methods (Tongway and Hindley, 1996). The SSC data were found to be insensitive to the changes in cover and were relatively subjective and are therefore not presented in this report. More standard measures such as soil bulk density were used (see next section) and only data relevant for linking land condition and hydrological response are presented in this document.

The condition of each hillslope was classified as A, B, C or D based on the data collected (Table 2). The ABCD landscape condition framework was initially developed by Chilcott et al., (2003) as a straight-forward method by which graziers could assess landscape condition, and estimate the long term sustainable carrying capacity (or stocking density) for their land. The framework also helped to raise awareness of environmental factors involved in animal production. Adoption of the framework has been very wide-spread, however, it is relatively subjective. It is important to note that different land types, in terms of geology and plant communities, may have different cover thresholds for ABCD land condition. To help identify the thresholds for Virginia Park station the work by McIvor et al., (1995) and Roth (2004) were also taken into consideration. In these studies 40% and 75% were considered to be the cover levels required to reduce hillslope runoff and maintain soil hydrological and biological function, respectively. The ABCD cover thresholds applied in the Weany Creek catchment are given in Table 2.

As well as on ground field measurements of cover, high resolution Quickbird satellite images with a 2.4 m^2 resolution (Pan sharpened to 0.6 m) were analysed for each of the hillslope flume sites for the 5 years between 2003 and 2007. The imagery was classified and calibrated using the ABCD classes described in Table 2. An additional fifth land class called ‘low cover D condition’ was defined as areas having < 10 % vegetation cover. The < 10 % vegetation cover data were also available for the whole catchment for 2003, 2005 and 2007.

Table 2: ABCD cover thresholds for Virginia Park station

Class	% ground cover
A	> 70%
B	50 - 70 %
C	20 - 50 %
D	10 - 20 %
Low cover D	<10%

3.3 Hillslope runoff and sediment yield monitoring

Virginia Park station

To measure water and sediment runoff, Flume 1 used a large cut-throat flume for measuring high flows, and a combination weir for measuring low flows. Flumes 2 and 3 were 9 inch Parshall flumes. Details of the logger setup and associated instrumentation can be found in Bartley et al., (2006). The water quality samples collected from Flume 1 were stratified according to flow depth, and for Flumes 2 and 3 they were bulk samples up to 20 L, which were collected following major runoff events. All samples were analysed in the laboratory for EC, pH, turbidity, total suspended sediment (TSS) concentration and sediment particle size. TSS concentration was considered to represent the silt (0.002-0.06 mm) and clay (<0.002 mm) sediment fractions. Bedload samples (which were generally between 0.063 – 8mm) were collected manually from bedload traps in each of the 3 sites and were assessed for mass and grain size distribution. When both concentration and discharge data were available, annual sediment loads were estimated by summing the event loads using the arithmetic mean approach (Letcher et al., 1999). When sediment concentration data were unavailable for an entire event, average wet season values were applied. Maximum rainfall intensity (I_{30}) during a 30 minute period were calculated for each event and for the whole season.

Meadowvale Station

A comparison of hillslope runoff, soil infiltration and bulk density data from Virginia Park station, were compared with data collected from field sites at Meadowvale Station (S19^{o}50'30.67", E146^{o}35'19.81") which is less than 20 km from Virginia Park and has similar soils and landscape characteristics (Roth, 2004).

To determine if the sediment yield response for the Virginia Park flume sites was due to increased rainfall during the study period or a result of GLM, the flume TSS concentrations were compared with those from two non-grazed experimental hillslope runoff troughs on Meadowvale Station. These runoff troughs had no cattle grazing between 1986 and 1992, and then had 'light' grazing between 1992 and 2002 (Alewijnse, 2003). The exclosures were then re-established and not grazed for the last six years (Hawdon et al., 2008). The Meadowvale site used runoff troughs rather than flumes, and a description of the site setup and sampling regime for this site is given in Hawdon et al., (2008) and Scanlon et al., (1996). A total of 20 TSS samples were collected from the Meadowvale troughs from a range of events between 2001 and 2006. The water quality samples from Virginia Park and Meadowvale Stations were analysed at the same laboratory.

Grazing has been shown to have a significant impact on infiltration rates in rangelands (Gifford and Hawkins, 1978; Trimble and Mendel, 1995) and therefore to determine if there is a difference in the soil conditions between heavy and lightly grazed sites, soil bulk density data collected as part of the cover versus infiltration experiment in 2004 on Virginia Park Station (Bartley et al., 2006) were compared with data collected using the same methods from Meadowvale Station between 2000 and

2002. The Meadowvale station data were collected from sites that had had no cattle or kangaroo grazing for 16 years, sites that have been lightly grazed for 10 years and sites that had undergone continuous grazing (Alewijnse, 2003). Saturated hydraulic conductivity at both sites was measured using the hood infiltrometer using the methods described in Bartley et al., (2006). Virginia Park Station has been grazed for ~100 years. It is acknowledged that bulk density is a coarse surrogate for soil condition, however, other soil surface condition (SSC) data were unavailable for the Meadowvale site and bulk density is an internationally accepted metric used for soil assessments.

3.4 End-of-catchment runoff and sediment yield monitoring

The majority of water quality monitoring in the GBR catchments is at the catchment scale (> 10 km^2). To determine if water quality changes due to the reduced stocking rates and wet season resting could be identified at the end of the catchment, discharge and sediment yields were recorded at the outlet of Weany Creek using an automatic gauging station installed in 1999. Weany Creek is ephemeral and flows for ~ 5% of the year. During runoff events the gauging station recorded rainfall, stage height, stream velocity, turbidity and temperature at one-minute intervals. A 1 L water sample was collected at each 400 mm change in stage height. For this study, runoff events were defined as occurring when the water level was greater than 200 mm for at least 2 hours, with at least 12 hours since the previous event. Details of the monitoring equipment and water sampling design of the gauging site are given in Bartley et al., (2007b) and Roth et al., (2003). The discharge estimation method employed both velocity measurement and Manning's equation to derive a stage-discharge rating curve (Koch-Lavisse et al., 2009).

To estimate sediment concentration between water samples, a linear relationship between total suspended sediment (TSS) and turbidity was derived (after Gippel, 1995; Grayson et al., 1996b). The TSS-turbidity relationship was based on all of the data from 2000-2006. The TSS concentration derived was multiplied by discharge to calculate the sediment load at the catchment outlet. The event sediment loads were totalled for each year to provide an annual suspended sediment yield at the catchment outlet. It is important to point out that the bedload (coarse sediment) fraction was measured on the hillslopes but not in the stream channel.

The event mean concentration (EMC) value for each event was calculated according to Equation 1 (after Kim et al., (2004)).

$$\mathbf{EMC_T\ (mg/l)} = \frac{\text{Sediment mass}}{\text{Runoff volume}}$$

$$= \frac{\int_0^T M(t)dt}{\int_0^T Q_{TRu}(t)dt} \times 1000 \qquad \textbf{Equation 1}$$

Where M(t) is the weight of sediment (in tonnes), $Q_{TRu}(t)$ is runoff volume (in ML) during the time interval, and T is the event duration. EMC's were calculated for each of the 20 measured events, and for each flow year.

To test if there was a decline in EMC at the end of the catchment following the introduction of grazing BMP in 2002, Mann's test (Kendall, 1970), which is a non-parametric trend detection test was applied. The Mann's statistic tests the null hypothesis H_0 that the observations are randomly ordered versus the alternative of a monotonic trend over time (Chiew and McMahon, 1993; Grayson et al., 1996a). The test assumes that there is no autocorrelation in the data as this can distort the variance. The annual discharges from the gauge for the period after the grazing trials were initiated were tested for auto-correlation.

To evaluate if the change in sediment yields observed on the hillslope could be detected at the end of the catchment, a simplified sediment budget was constructed. The catchment scale cover estimates were derived from the Quickbird imagery collected at the same time each year in 2003, 2005 and 2007. To determine the amount of sediment lost from the hillslopes Flume 3 fine sediment yield data was multiplied by the proportion of the catchment with < 10 % cover. Flume 1 sediment yield data were used to represent the remainder of the catchment for that year. These data were combined to estimate the amount of fine sediment predicted to be coming from the whole hillslope area (1357 ha) for 2003, 2005 and 2007.

4 Results

4.1 Pasture and biomass change on the hillslope

The change in cover (%), pasture biomass and % of low cover D condition land for Flumes 1, 2 and 3 are given in Table 3, Table 4 and Table 5, respectively. The % cover for each hillslope at the beginning (2002) and end of the study (2007) is given in Figure 3 and demonstrates that the overall average % cover has increased on all of the hillslopes over the study period. Photographs taken at similar points on Flume 1 in 2002 and 2007 visually show the change in ground cover over time (Figure 4).

Over the study period, the change in cover varied both within, and between, hillslopes, with upper parts of the slopes recovering better than lower parts (Figure 5). Much of the cover improvements were due to increased litter, which was higher under tree canopy (e.g. Figure 7). Overall there has also been a general shift in class condition from C to B on each of the hillslopes (Figure 6) although most of the change was dominated by Indian Couch (*Bothriochloa pertusa*) recovery. By contrast, the proportion of D condition land measured on the ground remains largely the same and in some cases it increased slightly in the early years of treatment (Figure 6).

The biggest difference in cover change is noticed when comparing the % change of low cover D condition land between Flumes 1 and 3 using the Quickbird imagery (Figure 8B). Flumes 1 and 3 initially started with similar amounts of low cover D condition land in 2003. With the implementation of grazing BMP the proportion of this land class has reduced on Flume 1 but not on Flume 3. At the whole of catchment level, the Quickbird imagery showed that the proportion of low cover D condition land in 2003 was ~ 10.16% and in 2007 it was 4.51% (Figure 9A and B). It is important to note that 2003 was in the height of a drought for the area.

Table 3: Change in cover attributes for Flume 1 measured at the end of the dry season (2002-2007). Standard error (SE) in brackets

	Field data		Quickbird data
Year	Average cover (%) (SE)	Pasture biomass (kg/ha dry matter) (SE)	% of low cover D condition land < 10%
2002	61.5 (0.8)	350 (6.9)	-
2003	33.8 (0.3)	60 (4.0)	7.5
2004	44.3 (1.1)	240 (14.1)	3.2
2005	57.2 (1.1)	520 (17.9)	3.6
2006	71.7 (1.2)	915 (44.4)	1.2
2007	71.6 (1.2)	984 (39.0)	1.5

Table 4: Change in cover attributes for Flume 2 at the end of the dry season (2002-2007). Standard Error (SE) in brackets.

	Field data		Quickbird data
Year	Average cover (%) (SE)	Pasture biomass (kg/ha dry matter) (SE)	% of low cover D condition land < 10%
2002	58.0 (0.9)	393 (13.9)	-
2003	37.9 (0.5)	62 (3.2)	<1%
2004	34.1 (1.8)	153 (12.3)	<1%
2005	50.2 (1.8)	479 (22.3)	<1%
2006	74.1 (2.4)	782 (39.5)	<1%
2007	76.3 (1.5)	1123 (75.3)	<1%

Table 5: Change in cover attributes for Flume 3 at the end of the dry season (2002-2007). Standard Error (SE) in brackets.

	Field data		Quickbird data
Year	Average cover (%) (SE)	Pasture biomass (kg/ha dry matter) (SE)	% of low cover D condition land < 10%
2002	68.1 (1.3)	321 (7.5)	-
2003	45.6 (1.0)	61 (3.5)	7.7
2004	46.6 (1.4)	146 (10.5)	6.7
2005	54.4 (2.1)	510 (23.3)	6.7
2006	72.7 (2.7)	667 (38.5)	5.3
2007	74.9 (1.8)	972 (47.0)	7.0

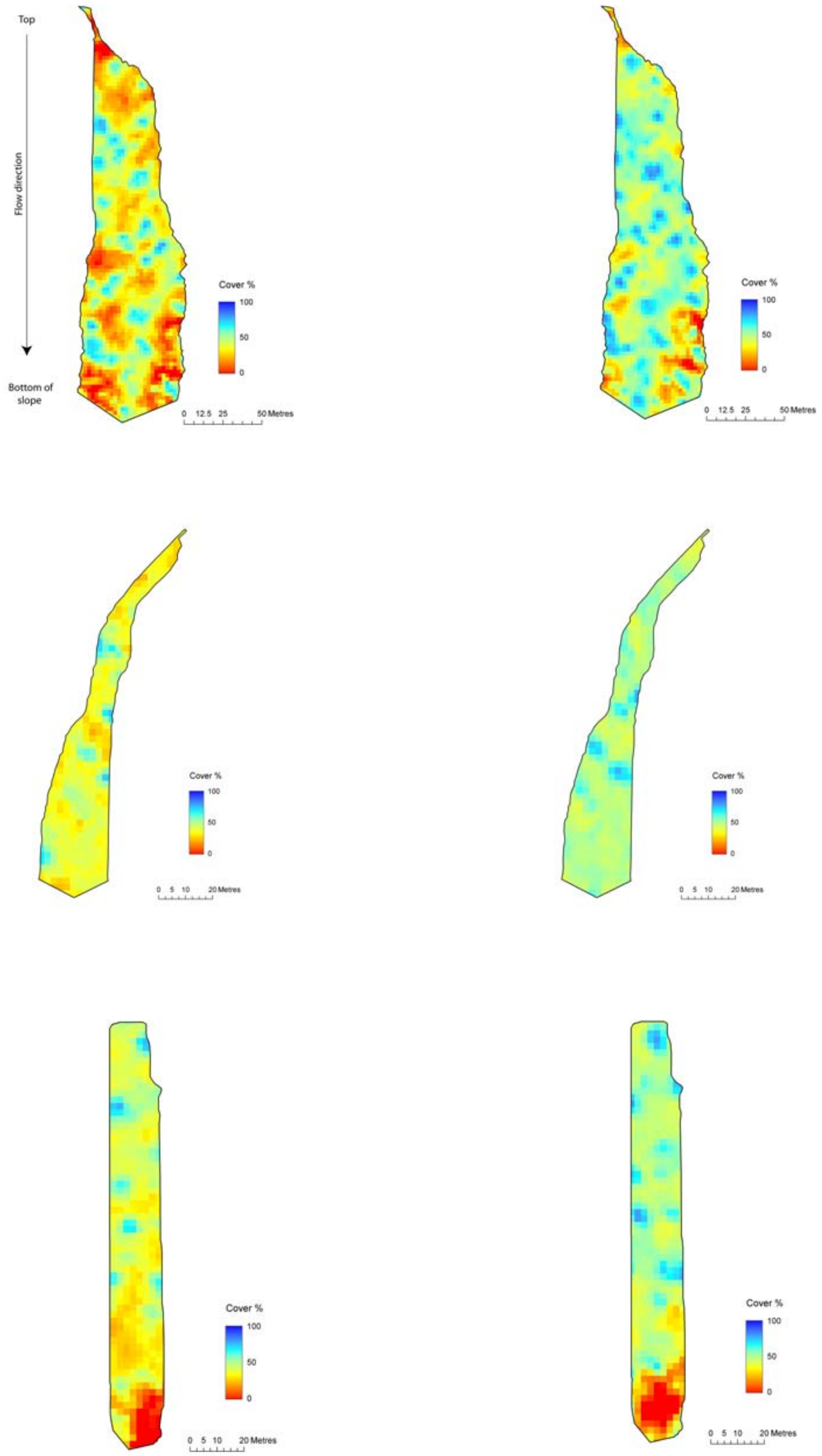
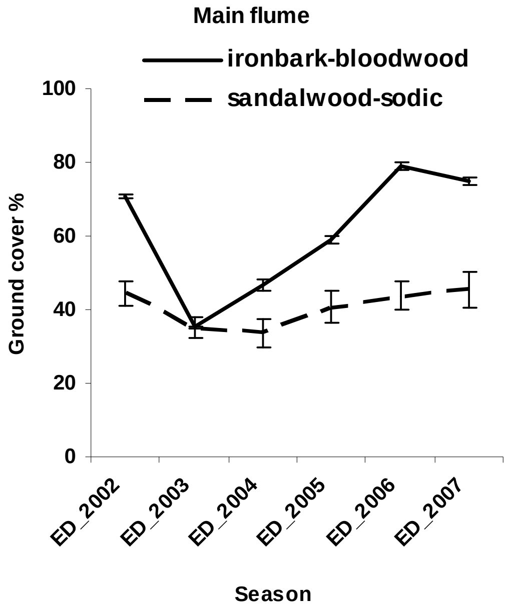
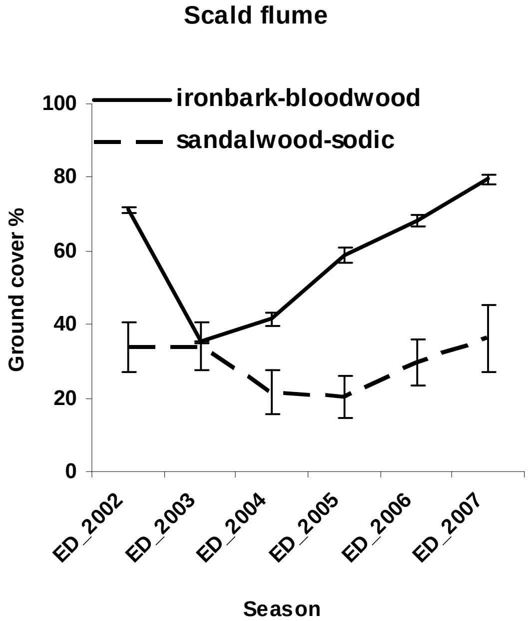


Figure 3: Quickbird derived cover (%) on each of the three hillslope flume sites in 2003 (left) and in 2007 (right). (A) Flume 1, (B) Flume 2 and (C) Flume 3. All slopes are aligned with the same flow direction. Note scale differences between Flume 1, 2 and 3. The contour interval is 0.5 metres. Quickbird imagery was not available for the beginning of the study in 2002.

Figure 4: Visual evidence of the changes in cover between 2002 (left) and 2007 (right) on Flume 1. Note: the left photo is beginning of wet season and right photo is end of wet season. Photos taken at equivalent time periods are not available.

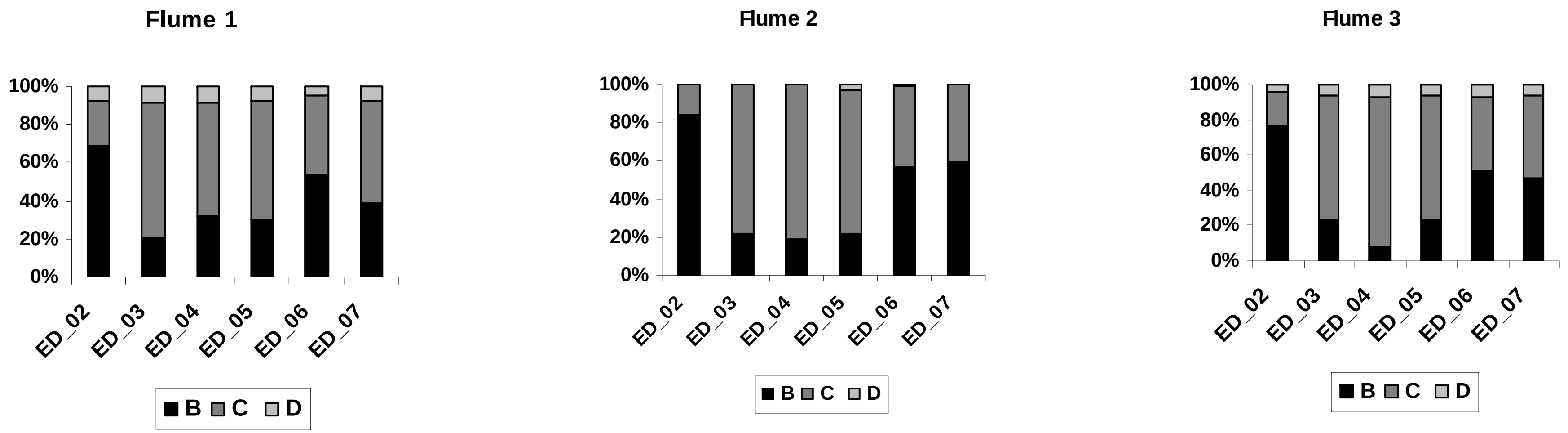


(A)



(B)

Figure 5: End of dry (ED) season ground cover levels for the upper slope (ironbark-bloodwood) areas and lower slope (sandalwood-sodic) areas for (A) Flume 1 and (B) Flume 3.



(a) (b) (c)

Figure 6: Trends in ABCD land condition 2002-2007 for (a) Flume 1, (b) Flume 2 and (c) Flume 3 catchments at Virginia Park station. ED = end of dry season

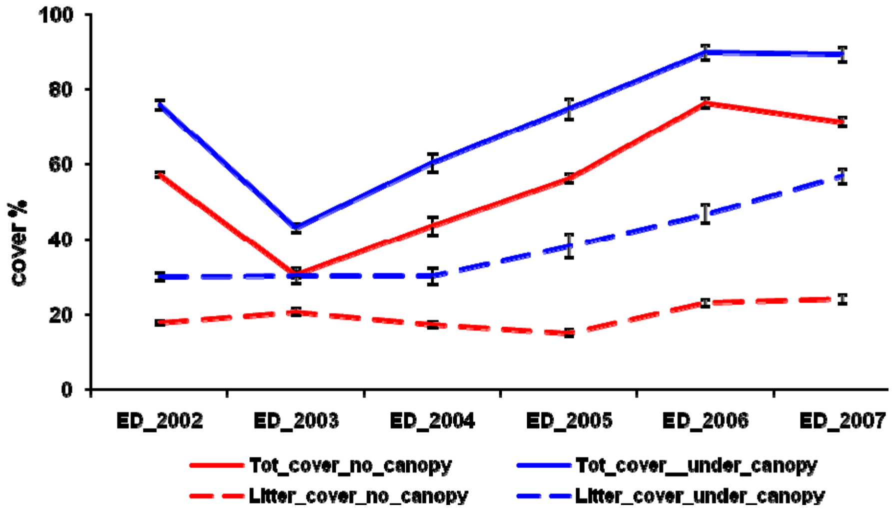
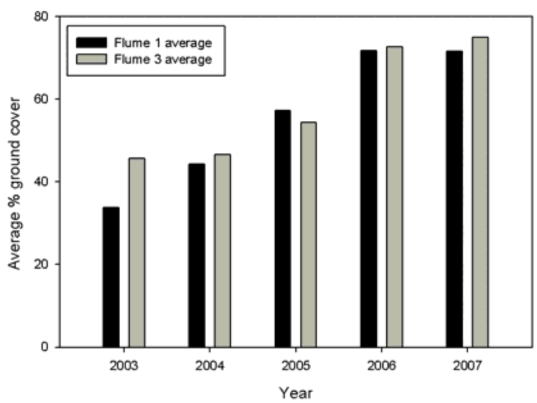
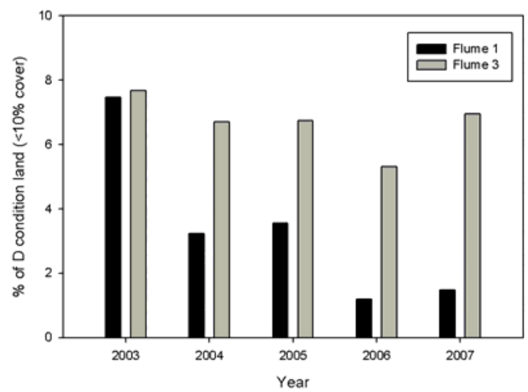


Figure 7: Variation in cover with and without canopy on Flume 1. ED = end of dry

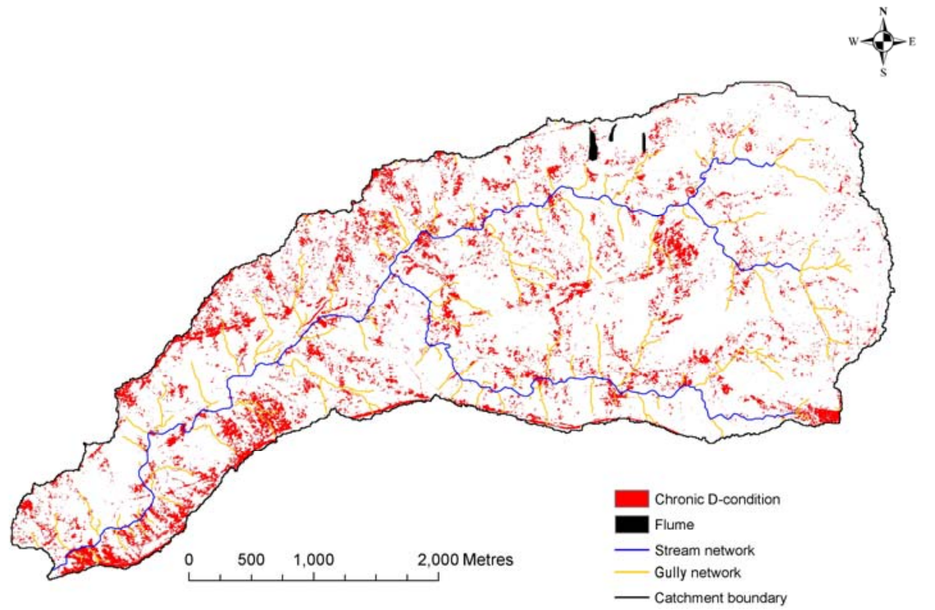


(A)

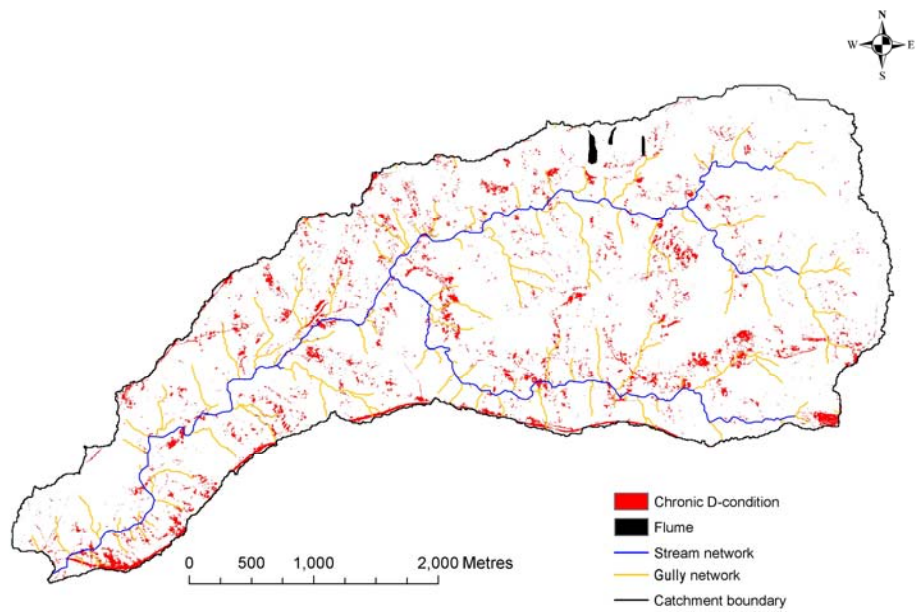


(B)

Figure 8: (A) Changes in % average cover on Flume 1 and 3 between 2003 and 2007 measured on the ground; and (B) Changes in % of low cover D condition land (less than 10%) between Flume 1 and 3 between 2003 and 2007 measured using landsat Quickbird imagery.



(A)



(B)

Figure 9: Classified Quickbird image of Weany Creek in (A) 2003 and (B) 2007, showing the areas in low cover D condition (<10% ground cover). The stream and gully network is shown in blue and the location of the flumes are shown in black with Flume 1 on the left, Flume 2 in centre and Flume 3 on right.

4.2 Hillslope runoff and sediment yields

The results show that Flume 3 has consistently higher % runoff and sediment yield for the length of the study (Figure 10). Over the six year study Flume 3 had 5.8 times more runoff and 88 times higher sediment yield than Flume 2, and 2.9 times more runoff and 27 times higher sediment yield than Flume 1.

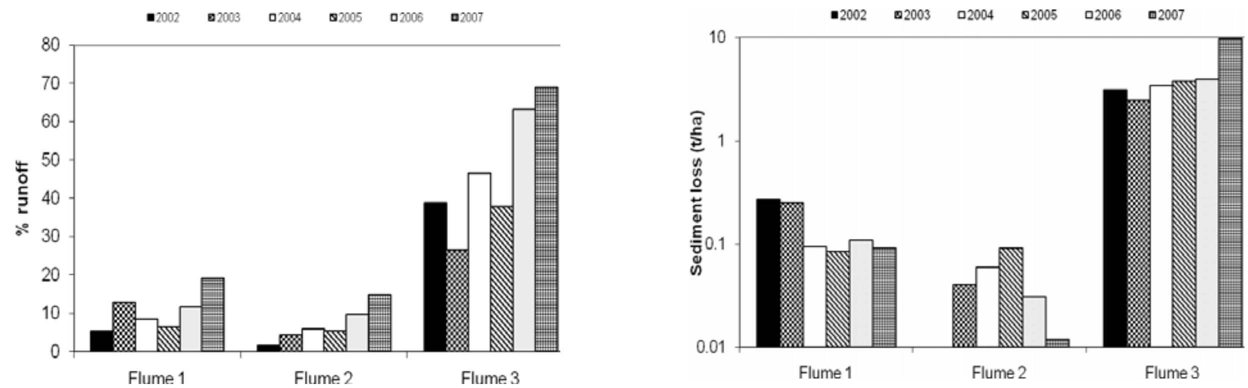
Flumes 1 and 3 had different responses in terms of the annual % runoff (which is the amount of rainfall that turns into runoff) over the study period. Figure 11A shows that for Flume 1, there has been a nine fold increase in runoff for Flume 1 (between 2002 and 2007) and for Flume 3 there has been five fold increase for Flume 3 over the same period (Figure 12A).

The runoff data show a different pattern when evaluated for lower rainfall amounts. Figure 13 shows that there is strong relationship between the amount of cover and hillslope runoff up to 200 mm of rain for Flume 1. [This rainfall amount was chosen as it was the only time in the 6 year data set for Flume 1 where all years had more than 0.1 mm of runoff] (see Figure 14). The rainfall-runoff relationship is strong (r^2 =0.92) for rainfall up to 200 mm despite the variation in rainfall intensities between years (Table 6). This pattern was not as strong on Flume 3, although it does appear that runoff has also reduced slightly on Flume 3 at least for very low rainfall events (< 50-100 mm) (

Figure 15). This suggests that cover is important for the first storm events particularly on hillslopes without large bare patches in the main flow path. For the lower rainfall amounts (<200 mm) it does appear that there has been a shift in the amount of infiltration over the 6 year study for Flume 1. Figure 14 shows that in 2003 (when there is < 35% cover), 200 mm of rain yielded ~32 mm of runoff. In 2007 (when cover was ~ 72%), there was < 3 mm of runoff for the same amount of rainfall.

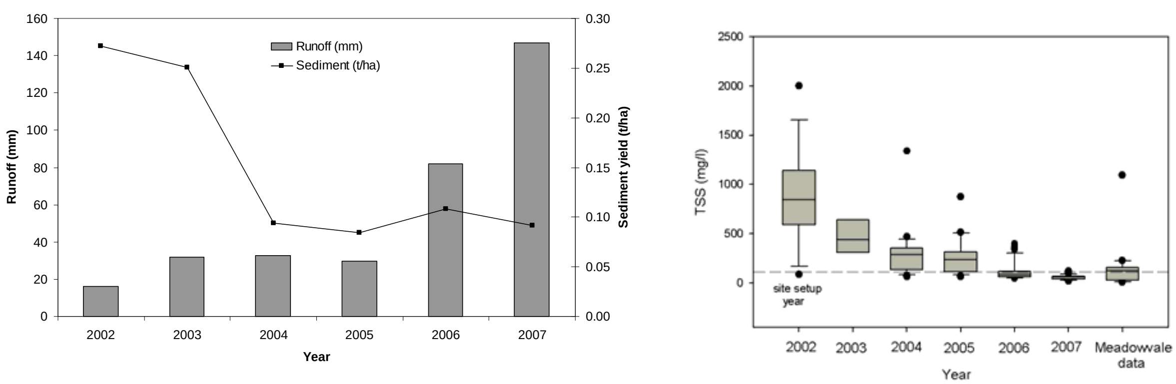
Figure 16 presents the bulk density data from the Virginia Park and Meadowvale sites. These data show that the bulk density of the soil declines with reduced grazing impact and there appears to be a threshold bulk density value at approximately 1.5 g/cm^3 below which infiltration starts to increase considerably. Similar results have been shown in other studies (e.g. Sartz and Tolsted, 1974). At Virginia Park station 95% of the bulk density values measured are greater than 1.5 g/cm^3. It is likely that the bulk density values are a surrogate or representative of a range of variables including surface cover, location of the patch on the hillslope, amount and location of litter sources, history of grazing and soil ecology.

Despite little change in hillslope hydrology at the annual time step over the 6 year flume trial, there has been a decline in total sediment yields on two of the three flumes (Figure 11 and Figure 12). Figure 10B shows that there has been a 66% and 70% decline in sediment yield for Flumes 1 and 2 between 2002-2007, respectively. At Flume 3 there was a 210% increase over the study period. As well a decline in total sediment yield, Flume 1 also shows a decline in the TSS concentration over the 6 year study (Figure 11B). The mean TSS concentrations collected at Flume 1 for 2005 were not significantly greater (p<0.05) than the mean TSS concentration at the Meadowvale cattle exclosure sites. Hence, the water quality coming off Flume 1 is now equivalent to a site that has not been grazed for 5 years. Unlike Flume 1 there was not a decrease in total sediment yield or TSS concentration at Flume 3 and the TSS concentrations are still well above the stock free values measured at Meadowvale (Figure 12B). There was insufficient TSS data for Flume 2 to present as a single site. On all of the flumes the TSS concentration declined as cover increased yet the rate and amount of reduction was very different for Flumes 1 and 2 compared to Flume 3. Figure 17A shows that as average cover increases beyond 60-70% on Flumes 1 and 2, TSS values are equivalent to the ungrazed TSS data. On Flume 3 average ground cover does reach more than 70% yet TSS values are still very high. Figure 17B shows that the amount of D condition land on a hillslope greatly influences the sediment concentrations in the water leaving the hillslopes.



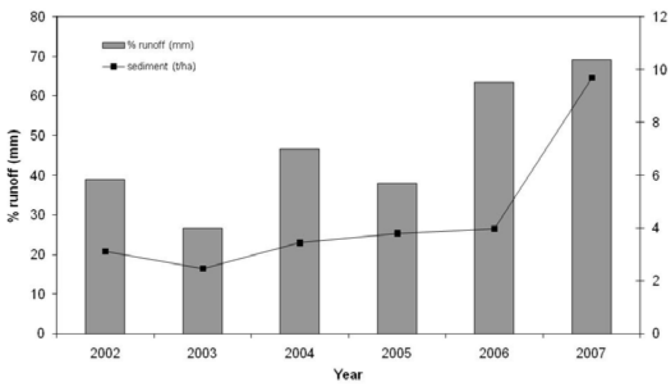
(A) (B)

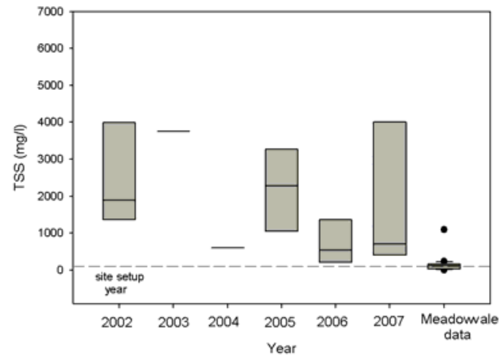
Figure 10: (A) Comparison of runoff as a percentage of rainfall for each of the three flumes over the 6 years of measurement; (B) Comparison of the total (suspended + bedload) sediment loss for each of the three flumes over the 6 years (note the log scale)



(A) (B)

Figure 11: (A) Changes in runoff (mm) and sediment yield (t/ha) over the 6 year study period at Flume 1; (B) Total suspended sediment (TSS) values from Flume 1 compared with the Meadowvale data from the grazing exclosures described in Hawdon et al., (2008).

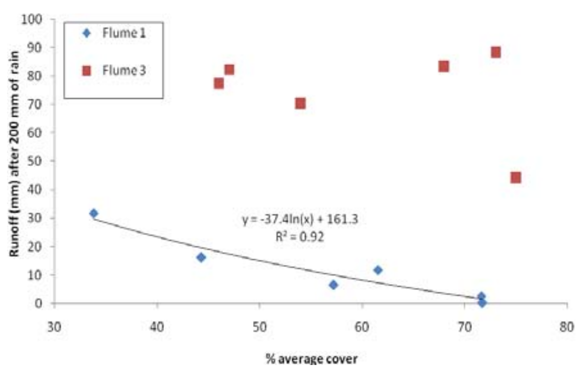


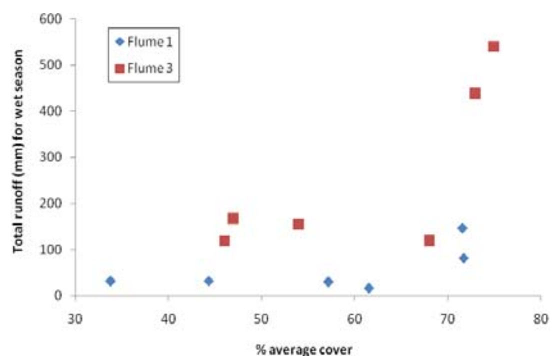


(A)

(B)

Figure 12 (A): Changes in runoff and total sediment yield (t/ha) over the 6 year study period at Flume 3 (scald flume); (B) TSS values from Flume 3 compared with the Meadowvale data from the grazing exclosures described in Hawdon et al., (2008).





(A)

(B)

Figure 13: The variation in runoff (mm) with % cover for (A) the first 200 mm of rainfall each year for Flumes 1 and 3 and (B) for the total runoff for Flumes 1 and 3.

Table 6: Annual rainfall and rainfall intensities for the Flume 1 site. I_{30} is the maximum rainfall intensity in a 30 minute period

Year	Annual Rainfall (mm)	I_{30} for events up to 200 mm rainfall (mm/hr)	I_{30} for whole wet season (mm/hr)
2002	304	66.4	66.4
2003	245	52.0	52.0
2004	382	44.0	31.6
2005	457	40.8	40.8
2006	706	35.6	26.4
2007	760	45.2	24.8

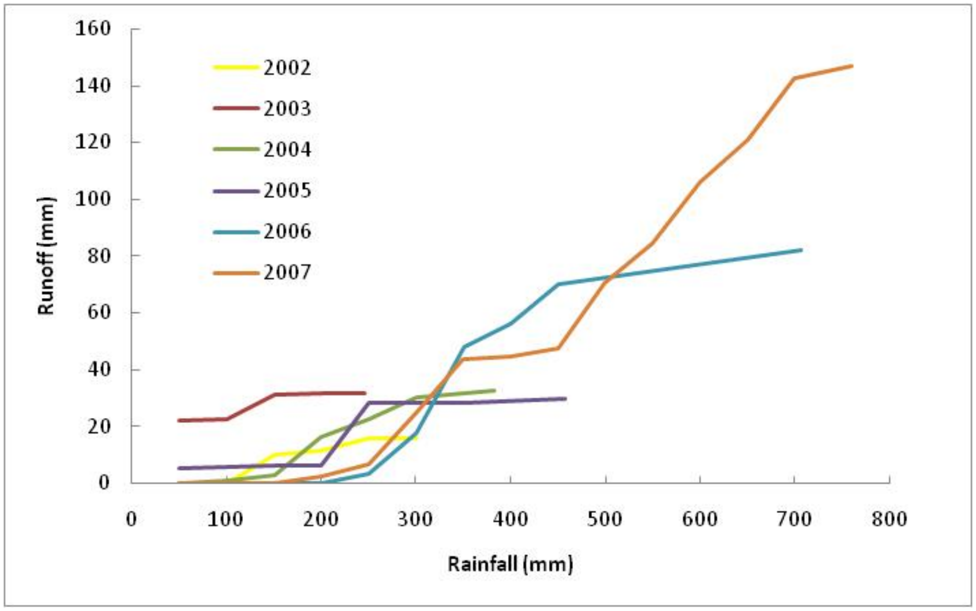


Figure 14: Amount of runoff (mm) for increasing rainfall (mm) for Flume 1 over the 6 years

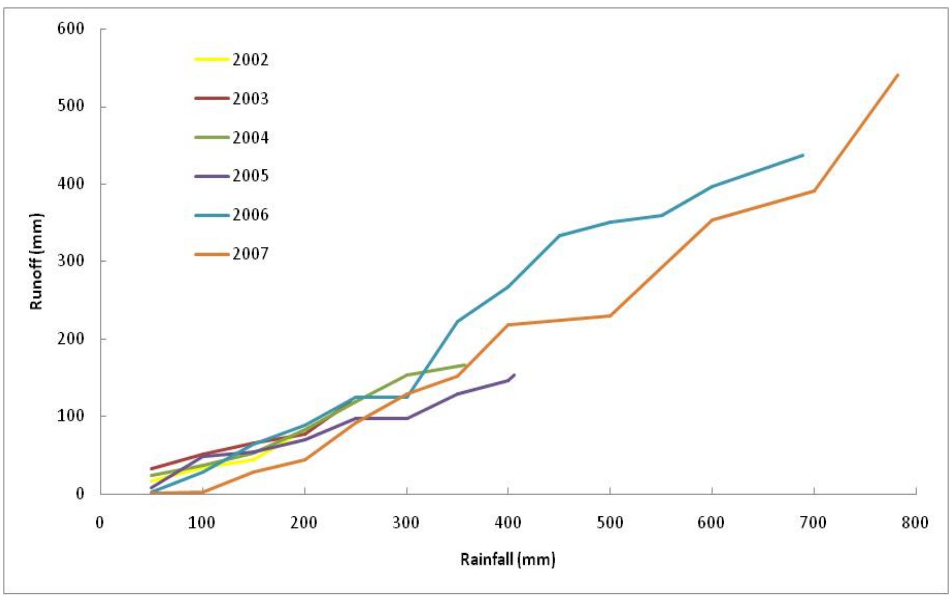


Figure 15: Amount of runoff (mm) for increasing rainfall (mm) for Flume 3 over the 6 years

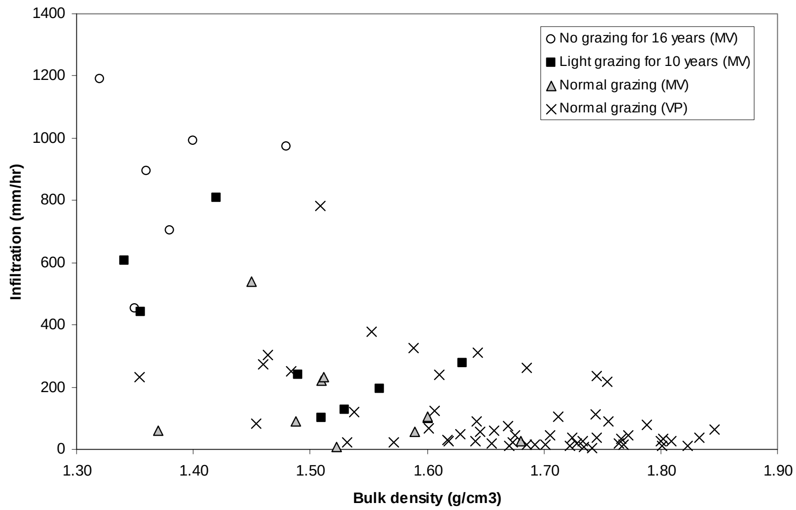
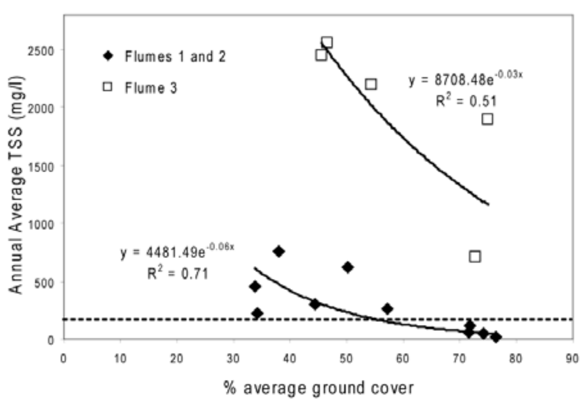
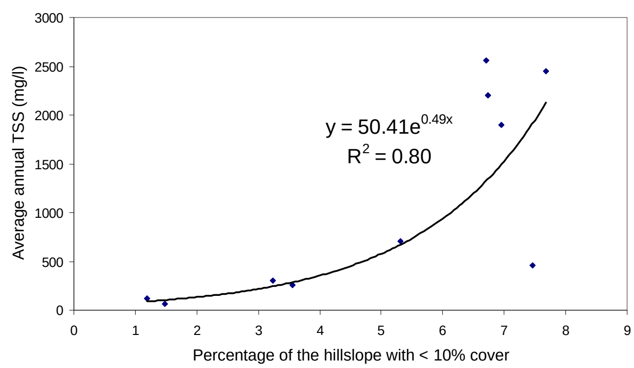


Figure 16: Bulk density values and corresponding infiltration rates measured (using hood permeameter) on a range of sites that have undergone different levels of grazing impact on Meadowvale Station (MV) and Virginia Park Station (VP) between 2000 and 2004.





(B)

(A)

Figure 17(A): Relationship between average annual ground cover (%) and average annual suspended sediment concentration for the three hillslope flumes. The dashed line represents the median TSS concentration (122 mg/l) from the nongrazed plots at Meadowvale; and (B) Relationship between the average annual TSS concentration from Flumes 1 and 3 and the proportion of low cover D condition land on the hillslope.

4.3 End of catchment runoff and sediment yields

The stream gauge recorded 20 events over the 8 year measurement period. There were four events not recorded due to damage to the turbidity sensor during the 2006 wet season. Consequently, loads were only calculated for the first, and largest event in 2006, which comprised 70% of total annual discharge. Annual sediment load calculations for 2006 are thus considered as an underestimate.

End of catchment sediment loads for Weany Creek for the eight year monitoring period are given in Table 7. The turbidity sensor was mounted ~ 300 mm off the bed of the river, no turbidity data were collected below this point. In low flow years (e.g. 2003) the depth is < 300 mm for <5 % of the time and in high flow years (e.g. 2006) <10% of the time. Therefore, the total load each year may underestimate the true value by 5 – 10%.

There was a steady increase in rainfall over the 8 year study and considerable variation in the runoff pattern particularly between years with similar rainfall (e.g. 2003 and 2004). There can be more than 100 mm of difference between total rain between the flume and stream gauge site in any one year which can be attributed to the highly isolated storms during the wet season.

Despite the variation, the annual sediment loads largely follow annual total rainfall conditions (Figure 18) and there is a strong linear relationship between annual runoff and sediment yield (Figure 19). The strength of this relationship ($r^2 = 0.85$) suggests that for the 8 years of this study there has been sufficient sediment available for erosion and transport (i.e. the catchment is supply unlimited). A similar although slightly weaker ($r^2 = 0.84$) relationship is obtained for the event data. It is hypothesised that if the sediment supply is reduced or becomes depleted by land management improvements then this relationship will weaken as there will be less sediment for the water to transport.

The change in sediment yield and EMC of suspended sediment following the introduction of best management practice is presented in Figure 18. Statistical analysis of the EMC for the 16 runoff events recorded after the implementation of BMP (data not shown) determined that although the sediment yield has increased with time (primarily due to increases in rainfall and thus runoff) there has been a statistically significant decreasing trend in suspended sediment EMC ($p=0.0051$). There was no autocorrelation in the annual at the $p<0.05$ level although this result is considered to be relatively weak due to the small size of the data set and this statistic may change with more data. Despite the strength in the trend relationship, it may be that this result is due to natural climate variability and/or other cycles of sediment delivery, rather than the implementation of BMP's in the catchment. The average EMC value of the four events measured prior to the implementation of BMP was lower (2082 mg/l) than the average of the 16 events after the implementation of BMP's (2749 mg/l) (data not shown).

Table 7: Rainfall, runoff and sediment loads measured at the stream gauge

Year (wet season begins)	Rainfall (mm) at gauge	Runoff (mm)	% Runoff	Sediment yield (t)	Number of TSS samples collected	No. of events
2000	367	18.13	4.94	404	19	2
2001	576	28.67	4.98	1010	45	2
2002	280	11.97	4.27	433	20	3
2003	362	8.90	2.46	507	10	3
2004	364	25.22	6.93	727	12	1
2005	559	9.62	1.72	385	19	4
2006*	638	111.25	17.44	2642	29	1
2007	649	141.51	21.80	1994	37	4

* the gauge was damaged this wet season and therefore the sediment load is considered to be a slight underestimate

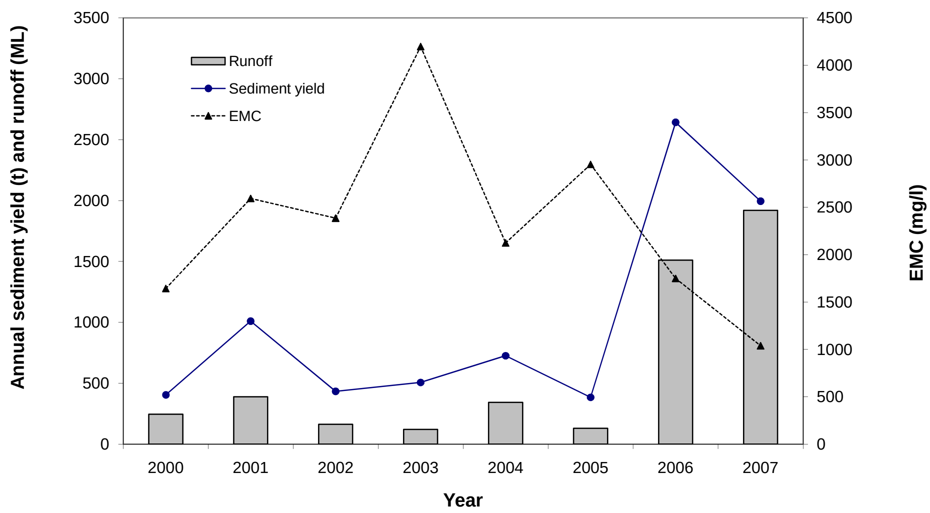


Figure 18: Runoff, sediment yield and annual event mean concentration (EMC) over the 8 year study period.

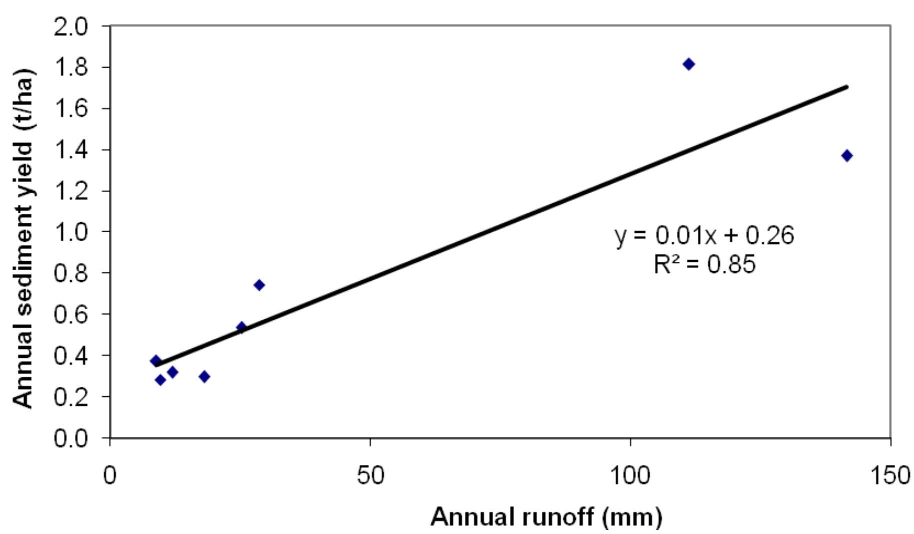


Figure 19: Relationship between annual runoff (mm) and annual sediment yield (t/ha) in Weany Creek.

4.3.1 Linking hillslope erosion to end of catchment sediment yields

Table 8 presents the amount of sediment coming from hillslope erosion for 2003, 2005 and 2007 at the catchment scale. The results show that although the amount of sediment has generally declined for hillslopes with > 10% cover (which represent 90-95% of the catchment), the total amount of sediment lost from hillslopes is highly biased by the small area of land with <10% cover, and overall the total fine sediment yield from hillslopes has not declined at the catchment scale. In 2007, the amount of sediment coming from the 5% of the catchment with <10% cover is now yielding more sediment (567 t or ~83%) than the 95% of the catchment that has >10% cover (119 t or 17%).

This analysis also suggests that there is deposition of fine sediment from hillslope erosion on the lower foot-slopes and/or the bed of gullies and the stream channel, as the amount of fine sediment being eroded from hillslopes is higher than the amount of sediment leaving the catchment in 2003 and 2005 (Table 7). This highlights the importance of deposition in these catchments and the non-linearity of sediment yields at different spatial scales. The proportion of fine sediment coming from hillslope erosion at the end of the catchment has declined between 2003 and 2007 suggesting that gully and bank erosion are larger contributors to end of catchment sediment yields in higher rainfall years.

Table 8: Contribution of fine sediment from hillslope erosion at the end of the catchment

Year	% of catchment with <10% cover	Fine sediment from <10% cover hillslopes (t)*	% of catchment with >10% cover	Fine sediment from >10% cover hillslopes (t)**	Estimated total fine sediment yield from hillslopes (t)	% of fine sediment from <10% cover areas
2003	10.16	227	89.84	305	532	43
2005	9.39	438	90.61	103	542	81
2007	4.51	567	95.49	119	686	83

*Flume 3 data used to estimate <10% over sediment yields

** Flume 1 data used to estimate >10% cover sediment yields

5 Discussion

5.1 The impact of grazing best management practice on vegetation

This study demonstrated that GLM (in the form of reduced utilisation, destocking and wet season resting) can improve the average vegetation cover on hillslopes within 5 years of implementation, however, pasture biomass levels are still relatively low on this property (~1000 kg/ha) compared to other management trials that have undergone grazing improvements on similar landscapes (e.g. Ash et al., (2001) recommends ~1700 kg/ha biomass for Goldfields country). GLM can also reduce the size and abundance of low cover patches in the upper slope areas. Large (> 6 m^2) D condition patches (with less than 10% cover) that are located at the base of a hillslope will either need more time for biomass levels to increase and for GLM to be effective on reducing runoff, or mechanical measures will be required to increase cover on these sites.

The exponential relationship between % ground cover and biomass for this catchment suggest that continued ground cover improvements will yield proportionally more biomass (Figure 15). The relationship suggests that this property needs ~83 % cover to reach a biomass level of 1700 kg/ha which is considered as ‘good’ land condition by Ash et al., (2001) for this soil type. Increasing the cover from ~ 75% to ~85% may be challenging from a grazing enterprise point of view, however, this extra 10% cover may be the threshold amount needed to help increase biomass, increase root density, increase infiltration and reduce runoff, particularly for the larger rainfall events.

Flume 1 has higher tree canopy cover than Flume 3 and analysis of end of dry season data indicates that areas immediately under or adjacent to live tree canopy have up to 20% more ground cover and over 100% more litter cover than areas away from tree canopy (P<0.005). Areas under tree canopy also have up to 45% more pasture biomass than equivalent areas away from canopy, while frequency of native perennial (or 3P) grasses is 27% higher under tree canopy (p<0.005). These 3P grasses play a crucial role in providing the architecture to trap litter and sediment on hillslopes. Their root structure also provides deeper infiltration and nutrient storage than other exotic species. Rainfall simulation experiments conducted on a range of tussock species (e.g. *Heteropogon contortous*) found infiltration down to 1m below soil depth (Roth, 2004).

As well as tree canopy cover, it seems that the size, number, location and interconnectedness or leakiness (Ludwig et al., 2006) of patches, particularly the lower cover patches, is important. Flume 1 has a much more 'patchy' cover arrangement with high and low cover patches mixed together (see Figure 3A). The increased canopy cover on Flume 1 has also provided more litter which can form litter bridges and reduce the connectivity between patches. Litter, and the amount of canopy cover as a primary source of litter, appears to be important for increasing infiltration and other studies have shown litter to be important for nutrient availability and pasture quality (Jackson and Ash, 1998). Flume 3 has a coarser grained ground cover pattern with the upper part of the slope having B and C condition patches and the lower slope having one large low cover D condition patch (Figure 3C). The size of the low cover D condition patch and location of the site at the bottom of the slope, as well as the reduced canopy cover, means that there is less time for litter to sit on the D condition patch as it is in the main flow path. The main sodic soil or low cover D condition patches on Flume 1 were not in the main flow path of the hillslope.

The recovery of the different vegetation patches is also related to grazing selectivity with C condition patches up to twice as likely to be repeatedly heavily grazed compared to A and B condition patches. D condition patches, often concentrated in lower slope sodic soil communities, were up to four times more likely to be heavily grazed (see Corfield and Abbott, 2008; Post et al., 2006). The lack of recovery in the D condition sites may also be related to the different recovery capabilities of the dominant vegetation groups that occupy the upper and mid slopes (e.g. *Eucalyptus creba)* with those of the lower slope sodic soil communities (e.g. *Carissa, Eremophila* and other shrubby species). This study demonstrates how changing and opening up (or coarsening) the spatial structure of vegetation in these landscapes may result in detrimental changes in pasture composition and cover (e.g. invasion of Indian couch) or have irreversible consequences such that the recolonisation of the bare patches may be severely impaired or prevented altogether (Ludwig and Tongway, 1996).

Despite the improvements in ground cover observed following the grazing management changes, it is important to highlight how fragile this recovery is, particularly when levels of biomass productivity remained very low in 5 out of 7 years. This landscape is dominated by the stoloniferous exotic grass Indian Couch and although the proportion of native perennial (3P) grasses on the hillslopes has increased, a return to increased stock numbers and no wet season resting could easily return these hillslopes to pre-trial conditions and jeopardise the full recovery of these sites. In fact, it may be necessary to further reduce current levels of grazing pressure to increase the rate of recovery.

The use of the ABCD condition classes in this study has highlighted that land condition recovery can be a slow and staggered process and that different parts of the landscape can recover at different rates. It also highlights that surface cover alone can be deceiving, and that the condition of the soil and the amount of litter associated with a given amount of cover can be an important influence on the potential infiltration capacity of the hillslope. The classification of ground condition and cover into classes makes it possible to isolate the areas that are having the biggest impact on hillslope hydrology.

This property level analysis shows that it may be possible to rehabilitate some areas of low cover D condition land with improved GLM alone, however, ~4.51% did not change after 5 years. These sites may recover with more time, however, it is likely that mechanical intervention and/or complete exclosure from grazing and diversion of overland flow may be required. It is also important to point out that the techniques used for restoration of the low cover bare patches are likely to differ between the upper non-sodic and lower sodic soil areas (see Section 5.4 for more detail).

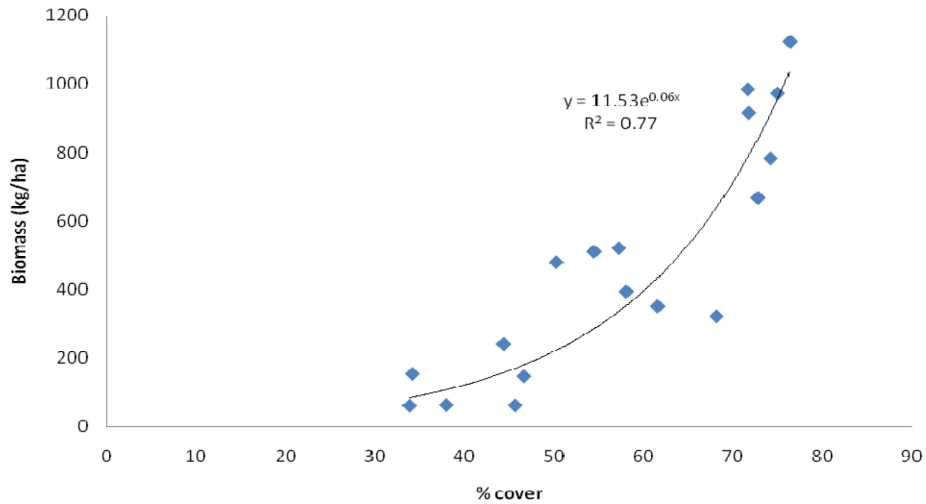


Figure 20: Relationship between % cover and pasture biomass for all three hillslopes over the 6 year monitoring period.

5.2 The impact of grazing best management practice on hillslope hydrology and sediment yields

Cover was found to have an important influence on runoff for lower rainfall amounts (up to ~200 mm). After ~ 200 mm of rain, the amount of runoff is no longer strongly related to the amount of cover on the hillslope. Due to the variation in the intensity, distribution and total rainfall amount between years, it is difficult to determine the exact saturation point of the hillslope soil profile. The soil depth on these hillslopes is ~70cm. It is likely that the saturation point is much less than 200 mm and will vary considerably with rainfall intensity and the amount of cover, soil depth and location on the hillslope. McIvor et al., (1995) and Scanlan et al., (1996) also noted that cover had little influence on runoff for high intensity events and for large events (> 100 mm).

There are a number of reasons as to why the runoff has not changed in accordance to the surface cover changes at the annual time scale. Firstly, it is likely that the excessive stocking over the last few decades, and the associated removal and compaction of the A-horizon, has increased the bulk density of the soil. The second possible reason is related to the recovery of soil ecology (e.g. earth worm and termite activity). Holt et al., (1996) showed that both infiltration rates and Acari and termite populations were lower in heavily grazed sites compared with lightly grazed areas. Similar findings for earthworms are described in the review by Drewry (2006). Therefore, it is possible that there is a lag in the recovery of the soil condition at Virginia Park, and it may take several years (or decades) before the ecology of the soil returns to its previous structure and infiltration capacity. The third possible reason is that after ~200 mm of rainfall, overland flow on these hillslopes occurs mainly as a result of saturated or excess flow. It appears that the soil profile becomes saturated, or infiltration is limited due to poor soil structure, and any additional rainfall becomes runoff. This response may be due to natural soil conditions, but it is more likely due to the grazing history at the site.

It is likely that a continued increase in biomass levels will see an improvement in the hydrological recovery of these hillslopes as the increased above ground biomass will also improve below ground root densities which will enhance infiltration. A number of other studies in the Burdekin region have shown that although sediment yields can differ greatly between different grazing treatments, mean annual runoff does not (Hawdon et al., 2008; McIvor et al., 1995; Scanlan et al., 1996). Importantly, the maximum length of any of these studies was 6 years, and it is likely that soil hydrological recovery may take > 10 years (e.g. Branson et al., 1981; Drewry, 2006). The recovery rate will also depend on the level of future grazing pressure. Good rainfall and low stocking densities may see further recovery and improvements within 10 years. If drought conditions return and/or stocking densities are increased, the recovery may take a lot longer (e.g. 20 years) or even be jeopardised all together.

Despite the lack of hydrological recovery at the annual time scale, hillslope sediment yields declined where scalds were not connected to gullies and streams. This is demonstrated by the decline in sediment yields from Flume 1 and 2 over the 6 year study period and the increase in yields from Flume 3. On the Flume 3 hillslope there is greater D patch connectivity or leakiness (Ludwig et al., 2002) as most of the low cover D condition land is near to (within ~40 m) a gully or stream network. There are some low cover D condition patches and rill features on the scalded sections of Flume 1, however, these sites are not in the main flow path and are more than 40 m from a gully or stream network. The proportion of shrubs such as current bush (*Carissa ovata* and Sandalwood *Eremophila mitchellii)* is also higher on and adjacent to the Flume 1 scalds, and this vegetation type (often associated with lower slope sodic soils) has been shown to have higher infiltration rates than surrounding areas as stock do not eat or trample on this vegetation (Roth, 2004).

In terms of hillslope recovery, this study has shown that changes in runoff due to land use change are much less sensitive than using sediment yield or concentration data. This can largely be explained by the fact that soil erosion rates decrease exponentially with increasing vegetation cover (Gyssels et al., 2005). The lack of hydrological recovery in this study appears to be linked to the structure of the soil and further research on the role of soil macrofauna in recovering landscapes is warranted. The use of periodic bulk density measurements may also provide an insight into the improvement (or lack of) infiltration capacity of the soil over time.

5.3 The impact of grazing best management practice on catchment hydrology and sediment yields

At the catchment scale, the amount of sediment coming from hillslope erosion has not declined over the study period. This is because the hillslope sediment yields are highly skewed by the small area of <10% cover land which is producing disproportionally high sediment yields.

This study has shown that there was a decline in the suspended sediment EMC values at the end of the catchment after 6 years, although this is likely to be related to the variation in rainfall rather than the implementation of BMP. The amount of data required to identify a statistically significant change in the mean is directly proportional to the inter-annual variability of the record, and less dependent on the length of data available (Chiew and McMahon, 1993). This means that for areas in the dry tropics, with high inter-annual variability in rainfall and runoff, long data records will be required to identify if changes in water quality are due to land management change or natural variation.

There has not been a reduction in end of catchment sediment loads, and in fact yields have increased over the study. This additional sediment has come from gully (and to a lesser extent bank) erosion in this catchment (Bartley et al., 2007b). This is supported by the fact that at the annual time scale hillslope runoff has not reduced, and therefore the excess runoff is available to erode gullies and banks. Other studies have also shown that sediment yield at the catchment scale (>10 km^2) is controlled by the amount, intensity and spatial pattern of rainfall as well as the channel properties/dimensions and sediment transport capacity of the stream (Lane et al., 1997). Processes at the hillslope scale, such as ground cover, remain important, but are subordinate to the catchment scale (Lane et al., 1997).

5.4 Priorities for on-ground restoration in grazing lands

For there to be a reduction in sediment yields at the catchment outlet, the priority rehabilitation sites for grazing lands in the Burdekin catchment appear to be gullies and low cover D condition land; although these are priorities under the assumption that GLM on hillslopes is also maintained. In terms of prioritising effort, stream bank rehabilitation, and associated riparian vegetation is important for preventing further erosion and providing habitat. In general, however, bank erosion rates measured over a 40 year period along the main channel of the Burdekin catchment have been shown to be very low by world standards (Bainbridge, 2004). Gullies on the other hand, have recently been shown to be the major process contributing sediment to the adjacent Fitzroy catchment (Hughes et al., 2009) and other parts of Northern Australia (Brooks et al., 2007; Wasson et al., 2002). It is difficult to establish how much higher the current gully and bank erosion rates are compared to pre-European times. There is evidence that suggests that sediment yields did increase in the late 1800's (McCulloch et al., 2003).

The Regional Natural Resource Management (NRM) Board in the Burdekin catchment that is responsible for on-ground investment or restoration has specified that there needs to be a 50% reduction in the amount of D-condition land in critically located areas by 2024 (Abbott et al., 2008; Dight, 2009). This is important for two reasons as (1) they have been shown to be contributing most of the sediment from hillslopes in rangelands and are unlikely to recover with grazing management alone; and (2) they appear to be the first stage of gully development in these landscapes. If gullies are allowed to develop, they can cause a peak in catchment sediment yields 2–3 orders of magnitude larger than yields prior to gully development, and this peak can last for several decades (Wasson *et al.*, 1998).

The results in this study indicate that it is possible to rehabilitate and reduce the size and abundance of D condition patches on the upper and mid slope areas with grazing management. However, grazing management is unlikely to be enough to reduce the impact of low cover D condition land on the lower slope sodic soil areas adjacent to gullies. Evidence for rangeland rehabilitation and restoration, and in particular scald reclamation, are highly variable (e.g. Green, 1989; McKeon et al., 2004; Pressland and Graham, 1989). Some studies have shown that scald recovery can occur under normal stocking rates, particularly in wet years in low-lying areas where scalds are under water for several weeks (Condon, 1986; Condon, 2002), and a few studies have shown that hillslope runoff and soil loss can be dramatically reduced when scalds are rehabilitated (Alchin, 1983). Others have found that perennial grasses have been unable to recolonise scalded claypans even after twenty years of exclosure (Silcock and Beale, 1986). Various reclamation methods for scalded areas are discussed in the literature as early as the 1950's including the use of gypsum on seedling establishment (Jones, 1962), waterponding on scalds (Cunningham, 1970; Cunningham, 1978; Jones, 1967), use of ripping and contour furrows (Cunningham, 1976). Reclaimed scalds have been shown to have improved moisture, reduced surface salt levels and greater soil cracking (for water infiltration) than scalded soils (Jones, 1966).

5.5 Areas of further research

There are five specific areas of further research that are needed that will help determine the most effective method of restoration for these sites, and if indeed hydrological recovery can be achieved.

1. There have been few documented studies in North Queensland regions to look at the specific methods and vegetation species that could/should be used to aid scald rehabilitation. Anecdotal evidence from the Burdekin suggests that there has been success rehabilitating scalded land using a range of restoration techniques (Bob Shepherd, Queensland DPI, pers. comm), although none of these experiments have been formally evaluated. Therefore it is important that these techniques are tested before broad scale recommendations for scald rehabilitation are made as it is possible that disturbing the fragile sodic soils associated with scalds could cause further erosion. The rehabilitation approach used will need to be made using knowledge of the soils, vegetation, slope and rainfall characteristics of a site. As each of these will vary subtly across the landscape, at this stage it is not possible to make generic recommendations. It is important to highlight that any mechanical rehabilitation is done in conjunction with continued (and potentially improved) GLM on the hillslopes.

2. Recent research suggests that gully bed vegetation can induce sediment deposition and gully stabilisation and reduce sediment yields from gully systems in the long term (Molina et al., 2009). Therefore targeted management of near-riparian scalds and gullies including stock exclusion or management, as well as gully revegetation are likely to be an important addition to the current GLM approach and warrant further testing.
3. Gullies are known to mature naturally with time (Graf, 1977; Rutherfurd et al., 1997), and therefore prior to any large scale gully rehabilitation project it would be pertinent to determine where gullies are in their erosion lifecycle, as there is little point investing money and rehabilitating mature gully systems. A rigorous geomorphic study using dating techniques such as OSL (optical stimulated luminescence) (e.g. Rustomji and Pietsch, 2007) could help determine whether the amount of sediment from gullies is increasing, has stabilised or is decreasing.
4. The Meadowvale data indicate that, in the longer-term, GLM has potential to reduce hillslope runoff. Therefore long (> 10 year) data sets on the hydrological recovery of sites such as Virginia Park are required. Reduction in hillslope runoff will be important to ensure the success of grazing BMP and gully and scald revegetation, and a return to overstocking may quickly return this system to its former highly degraded state, reducing the potential for decreasing hillslope, gully and bank erosion. Further work should also determine biomass as well as cover thresholds for the different pasture types.
5. There is a need to determine if the hillslope hydrology in these landscapes is being driven by saturated overland flow (due to naturally shallow soils) or hortonian overland flow (related to degraded soils with poor infiltration). This would help determine if the soil has it been moved to a new degraded state and is unlikely to ever increase infiltration, or if the infiltration rate is related to the damaged soil structure which means there is an opportunity for hydrologic recovery in the future.

Future studies that are planning to assess recovery of degraded landscapes should also endeavour, where possible, to have a pre-calibration period (e.g. Thornton et al., 2007) so that the natural hydrologic variability can be established prior to treatment. This will provide a much more robust data set for evaluating if recovery has occurred. It will also enable the impact of GLM to be separated from the effects of changes in rainfall.

5.6 Implications for management

This study has shown that it is important to identify the dominant source of sediment when attempting to reduce sediment yields at the catchment scale. In this study sub-soil erosion is dominant (Bartley et al., 2007b), and thus subsoil (e.g. gully) erosion will need to managed along side hillslope erosion before there will be a decline in sediment yields in this catchment. This is in contrast to studies where hillslope surfaces are the dominant source (e.g. Foster and Walling, 1994). It is possible, given enough time, that GLM will produce the required biomass and runoff reductions required to reduce channel erosion in this catchment. Unfortunately the time lines associated with this change are unknown, and the recovery times (assuming recovery is possible) are likely to be longer than ‘target’ timelines being set by the Reef Plan. Thus, in the short term (5-10 years), GLM will not provide substantial reductions in catchment sediment yield on its own on a working grazing property. It may be possible to reduce sediment yields within 10 years if grazing is halted altogether, although this is not a viable option in the majority of cases. International studies are increasingly showing that sediment yields in many catchments are insensitive to land use change mitigation programs at least in the short term (Walling and Collins, 2008). The lack of response of catchment yield in this study imply that it is unlikely that voluntary GLM will be effective at reducing sediment yields to the GBR. It appears that more intensive, and probably externally funded, restoration measures will be required.

Lessons also need to be learnt from the Ord River Catchment Regeneration Project (ORCRP) in Western Australia where extensive stock reduction and remedial works were undertaken in rangelands where serious erosion was identified (Fitzgerald, 1976). The approach included fencing, grazing control, and large pasture re-establishment. This program was considered to have achieved 'spectacular' results in as little as 8 years, however, more recent research has shown that after almost 30 years, the ORCRP has had no measurable effect on the sedimentation rate in Lake Argyle (Wasson et al., 2002). This is because the scheme invested a lot of money (the exact amount cannot be found) into hillslope rehabilitation, yet gully erosion was the main source of sediment and therefore sediment yields did not decline (Wasson et al., 2002). In this particular study, overland flow appears not to have been significantly impeded by either the revegetation or absorption banks.

In 2003, the Reef Water Quality Protection Plan was released with the aim of halting and reversing the decline in water quality entering the Reef within 10 years (Commonwealth and Queensland Governments, 2003). Despite the positive direction of the results obtained in this study, linking the changes observed from a headwater catchment, with the quality of water entering the Great Barrier Reef (GBR) is much more difficult. At present, there are no known examples anywhere in the world where the net flux of sediment reaching tidal waters has been shown to be reduced through a stream or catchment restoration project (Palmer, In Press). This suggests that using such short time lines (e.g. 10 years) are inappropriate and may only give false expectations regarding what can practically be achieved in terms of improved water quality entering the GBR.

The results of this study also demonstrate the importance of monitoring data to build process understanding rather than relying only on models for simulating the effects of changes to land management practice. Erroneous conclusions regarding the effectiveness of GLM at catchment scale could have been arrived at by scaling up the hillslope flume data, or by using models of hillslope erosion only (e.g. Dalzell et al., 2001).

Similarly, this study has highlighted the need to monitor a range of variables at different spatial scales. Had this study only monitored sediment yields at the end of the catchment, the impact GLM is having on reducing hillslope sediment yields would not have been identified. In addition, different data sets (on-ground field measurements and Quickbird satellite imagery) are both important for assessing the change in ground cover and land condition through time due to changes in grazing land management. The satellite imagery provides important insights into the cover story, however, it will not negate the need for detailed on-ground measurements. The on-ground field data is vital not only for calibrating the structural cover information and checking the accuracy of the satellite imagery. It also provides information on the function of the different components of hillslope ecology, hydrology and soil condition that cannot yet be captured by satellite imagery.

6 CONCLUSIONS

This paper presented the results of an 8 year field study that evaluated the impact of improved grazing management on land condition recovery and the consequent loss of water and sediments from hillslopes and catchments.

This study has shown that following the implementation of grazing BMP pasture biomass and ground cover increased, and hillslope sediment yields declined where scalds were not connected to gullies and streams. Where large low cover patches are located at the base of a hillslope, adjacent to gully or riparian areas, sediment yield did not decline. More intensive rehabilitation will be needed to improve remaining low cover D condition sites associated with lower slope locations closer to gully and stream networks. Although the term BMP is used throughout this document, it may have been more appropriate to use the term 'improved' grazing management. Further improvements are considered necessary on this property, including an increase in cover to ~83% to match recommended pasture biomass yields for this soil type, before 'best' management practice is obtained.

This study also demonstrated that there has been a reduction in runoff with increased cover for low rainfall amounts (up to ~200 mm). For higher rainfall amounts runoff is not strongly related to the amount of cover on the hillslope. This study also demonstrated that runoff is less sensitive to land use change (in the short term) compared to sediment yield. Longer data sets are required to determine if and when hydrological recovery will occur in these landscapes.

Improved GLM did not result in significant reductions in sediment yield at catchment scale, due to inter-annual variability in rainfall, and dominance of subsoil erosion from gullies and stream banks. Further research is required into the effectiveness of scald and gully erosion treatments which were not considered in the study.

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