Examining passive riparian revegetation in degraded rivers: livestock exclusion and environmental flows

Acacia salicina seedling

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Declaration

I hereby declare that this thesis has not been previously submitted to any other institution or university for a higher degree. Except where otherwise acknowledged, this thesis is comprised entirely of my own work.

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This thesis has been formatted to ensure compliance with a page limit of 59 pages for text and figures determined by the Faculty of Science and Engineering at Macquarie University.

Abstract

Passive riparian revegetation techniques are becoming increasingly important tools in river rehabilitation. However the utility of the sediment seed bank as a passive riparian regeneration option is poorly understood. A livestock excluded site has been monitored over a three year period and compared to a continuously grazed site to test the regeneration potential of the seed bank. Livestock exclusion was successful in increasing erosion resistance (roughness) and the regeneration of native species but not successful in restoring communities or regenerating substantial numbers of woody species. The utility of the seed bank does not extend to the restoration of full communities and requires direct plantings to provide a more developed assemblage. A seedling emergence experiment has compared the effects of simulated flood durations on the seedling emergence of desirable riparian species. The inundation of bench units using environmental flows would not significantly increase or decrease recruitment of desirable species from the seed bank. Significant differences in seedling emergence and timing under different inundation periods were found to vary among species. These studies provide greater understanding of how passive revegetation utilising riparian seed banks can best achieve river rehabilitation goals and how revegetation objectives should be framed when utilising riparian seed banks.

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Chapter 1: Introduction, literature review and thesis aims

1. Introduction

The European settlement of Australia has a legacy of widespread land clearing, flow regulation, desnagging, introductions of invasive species and intensive agricultural practises which have degraded many river systems (Fryirs et al., 2008; Arthington and Pusey, 2003; Jansen and Robertson, 2001; Rutherfurd, 2000). The pressures that have previously degraded river systems and riparian zones are expected to continue (Beechie et al., 2010; Brierley et al., 2008; Wohl et al., 2005). This is despite the recognised importance of riparian zones in maintaining the chemical, ecological and geomorphic functioning of rivers (Capon and Dowe, 2007; Richardson et al., 2007; Naiman and Decamps, 1997). There has been a great increase in rehabilitation projects since the 1980's in order to counteract river degradation and more recently to mitigate the impacts of future pressures such as climate change (Bernhardt and Palmer, 2011; Ormerod, 2004). With the rise in river rehabilitation projects, ecosystem and process based approaches to river management have become best practice (Brierley and Fryirs, 2008). Passive revegetation techniques have become important tools within these contemporary approaches, involving the removal of a degrading force and allowing natural processes, such as regeneration from the seed bank, to drive recovery (Benayas et al., 2008). Passive revegetation techniques are the focus of this thesis.

The distribution and dynamics of riparian seed banks have only recently received significant attention in research and are not yet fully utilised in river management and rehabilitation (Goodson et al., 2001). As such the capacity of the seed bank to be utilised in passive revegetation is only partly understood. This lack of understanding impairs the accurate setting of rehabilitation objectives and the ability to make informed predictions as to the utility of the seed bank for revegetation, which are key considerations for rehabilitation success (Skinner et al., 2008). Further, important factors such as whether environmental cues may be used to improve recruitment from the seed bank (Williams et al., 2008), and if so what they may be, remain to be defined. This thesis will assess the efficacy of two passive revegetation techniques in order to better understand the utility of the seed bank for river rehabilitation. Firstly this thesis will quantify the success of livestock exclusion, an existing and widely used passive revegetation technique, by directly addressing its performance against two key passive revegetation objectives, which are increasing erosion resistance and restoring native species (Hunter-Central Rivers CMA, 2011). Secondly this thesis will test whether manipulating the inundation duration of bench units along a degraded river may improve the recruitment of desirable native species from the seed bank. The results will be used to determine whether environmental flows may be used as a passive revegetation technique in degraded rivers. The results of these studies will be used to better understand the utility and limitations of the seed bank for riparian revegetation and how seed banks can be best utilised in river rehabilitation.

2. Literature review

"Restoration programs are doomed unless we all know what we are aiming for and what we are up against."
(Robertson, 1997, p. 216)

Contemporary Australian river rehabilitation

The restoration of fluvial systems to pre-disturbance states is largely considered impossible since the degree of changes to boundary conditions (water, sediment and vegetation interactions) within which rivers operate, have been fundamentally altered (Wohl and Merritts, 2007; Newson and Large, 2006; Hughes et al., 2005; Shields et al., 1999). River rehabilitation seeks to return rivers to a more natural state, rather than a pre-disturbance state (Dollar, 2004; Rutherfurd and Gippel, 2001). In recent times there has been a move away from traditional hard engineering works in river management towards process and ecosystem based rehabilitation techniques (Fryirs et al., 2008). Traditional river management techniques attempted to enforce structural changes upon rivers to modify their behaviour, creating system instability, changes in fluvial function and severe ecological damage (Fryirs et al., 2008; Hey, 1994). Ultimately traditional engineering techniques have been implemented at great environmental cost and in many parts of the world have become socially unacceptable (Brierley and Fryirs, 2008; Hey, 1997; Holling and Meffe, 1996). Ecosystem based rehabilitation focuses on understanding the biophysical interactions at the ecosystem scale rather than focussing on a single species or reach (Brierley and Fryirs, 2008; Browman et al., 2004). Process based approaches seek to establish acceptable rates of natural processes, such as erosion and sedimentation, which create and maintain ecological and physical systems (Beechie et al., 2010). The focus on restoring process rather than just structure allows for natural variability and works with natural processes (Wohl et al., 2005). It is now accepted that river management should be interdisciplinary and address multiple causes for multiple outcomes (Dollar et al., 2007). These contemporary approaches focus on building the capacity of systems to recover from rather than resist disturbance, while implicitly placing value on sustainability and biodiversity (Brierley and Fryirs, 2008).

Recognition of the importance of riparian vegetation in maintaining river function and the rise in rehabilitation projects has led to increased interest in revegetation from both agencies and communities, but there remains a lack of understanding of processes in the regeneration of riparian vegetation (Pettit and Froend, 2001b; Lovett and Price, 1999; Tabacchi et al., 1998; Naiman and Decamps, 1997). Riparian revegetation has become a major tool in Australian river rehabilitation projects. However this is often undertaken using direct riparian planting which is an expensive exercise, costing millions of dollars in Australia annually (Shelly et al., 2009; Lovett and Price, 1999). Passive regeneration is a less costly and less intrusive alternative to active management (Schneider, 2007). Passive riparian revegetation works by protecting riparian areas from disturbance and allowing natural processes, such as regeneration from the seed bank, to be the main drivers of recovery (Gould and Spink, 2012; Benayas et al., 2008; Kauffman et al., 1997). The advantages of this approach is great flexibility in the scale and placement of exclusion zones, a reduction in cost and prevention of further direct human disturbance to the site (Corr, 2003; Price and Lovett, 1999). Passive regeneration often serves as the first practical on site step in revegetation projects (Kauffman et al., 1997). The seed bank is increasingly being recognised as an important component in river rehabilitation, but the utility of riparian seed banks for use in passive revegetation, particularly in Australia has not been fully explored (Jensen et al., 2008; Middleton, 2003; Goodson et al., 2001). Therefore questions remain as to the exact capacity of regeneration from the seed bank to fulfil revegetation objectives and how managers should frame their objectives around this uncertainty. The setting of clear objectives and evaluation of outcomes are key steps in determining project success, yet these steps are often not adequately implemented (Skinner et al., 2008, Rutherfurd et al., 2004; Ladson et al., 1999). Further, the possibility of using environmental cues to promote the regeneration of desirable species from the seed bank (Williams et al., 2008; Britton and Brock, 1994), may be a potential tool in revegetation but has yet to be fully explored.

Seed bank dynamics and utilisation in passive riparian revegetation

There are two types of seed bank; aerial and sediment or soil seed banks (Leck, 2012). Aerial seed banks are collections of viable seeds or propagules held in the canopies of some species of tree, shrub and sedge (Hamilton-Brown et al., 2009; Lamont & Enright, 2000). Soil or sediment (hereafter referred to as sediment) seed banks are the collection of viable seeds and propagules held within the soil or sediment profile (Poiani et al., 1989; Thompson, 1987). This thesis is concerned with sediment seed banks in riparian zones. Sediment seed banks are important stores of reproductive material especially for understory species, which may rely on seed banks to buffer their populations against extended periods of poor conditions or disturbances (Leck, 2012; Fenner, 2000; Thompson, 2000). Abernethy and Willby (1999) link the importance of the seed bank for community regeneration to the scale of disturbances, periods of good or poor conditions and the condition of standing vegetation. If poor conditions are prolonged the importance of the seed bank for regeneration increases. Despite their importance the role of seed banks in riparian vegetation dynamics is poorly understood (Goodson et al., 2001; Pettit and Froend, 2001a). The composition of riparian seed banks are governed by numerous factors including disturbance frequencies, standing vegetation, flow regulation, hydrological adaptations of seeds, seed persistence, seed dormancy and the requirement of environmental cues or conditions to stimulate germination (Greet et al., 2013b; Williams et al., 2008; Corenblit et al., 2007; Gurnell et al., 2006; Goodson et al., 2001). Therefore the composition of the seed bank will be specific at the geomorphic unit (landform) scale as each unit is formed and maintained by varying hydrological and geomorphological processes, contributing to the spatial heterogeneity of seed banks (O'Donnell et al., 2014a).

Dissimilarity between the seed bank and standing vegetation has been widely observed (Tererai et al., 2015; O'Donnell et al., 2014b; Cui et al., 2013; Lu et al., 2010; Bossuyt and Honnay, 2008; Jensen et al., 2008; Williams et al., 2008; Hopfensperger, 2007). Many woody perennials have developed alternate strategies for regeneration to utilisation of the sediment seed bank such as vegetative regeneration from rhizomes and lignotubers or via serotiny, where seed is stored in plant canopies (Jensen et al., 2008; Abernethy and Willby, 1999). Williams et al. (2008) note that the canopy structure of standing vegetation may affect propagule inputs to the seed bank by regulating seed transport by wind and germination by regulating microclimate. Disturbance history also plays a role in structuring the standing and seed bank composition, with high disturbance sites likely to have standing vegetation and seed banks more similar to one another (Bossuyt and Honnay, 2008). The difference in composition between seed bank and standing vegetation is further explained through the biophysical interactions occurring at a site. The biological controls include the reproductive strategies of species i.e. amount of seed released and persistence of the seed, requirement of environmental cues for germination and adaptation to dispersal mechanisms (Gurnell et al., 2006; Goodson et al., 2001). The physical controls include hydrological and geomorphological controls. For example flood frequency, duration and the fluvial re-working of geomorphic units (O'Donnell et al., 2015; Goodson et al., 2002). An example of the interaction between the physical and biological is the dominance of sedges and rushes within the seed bank where they occur (O'Donnell et al., 2014b; Williams et al., 2008). This is due to the small size and prolific amounts of seed produced by these species, which are readily dispersed by hydrochory and incorporated into the seed bank (Williams et al., 2008). Riparian seed banks have been found to be more abundant throughout the sediment profile than terrestial seed banks. Abernethy and Willby (1999) and O'Donnell et al. (2014b) found only a relatively small vertical decrease in stratigraphic seed bank density in comparison to terrestrial seed banks, attributed to geomorphic reworking of sediment post deposition by flood events.

The potential benefits of utilising riparian seed banks in passive revegetation as opposed to direct planting or other passive techniques include reducing cost in comparison to direct plantings, low maintenance requirements, limiting further human disturbance to the site and using indigenous species (Gould and Spink, 2012; Schneider, 2007; Gurnell et al., 2001). The dominance of early successional species in riparian seed banks however limits the potential usefulness of seed banks in riparian revegetation (O'Donnell et al., 2014a; Greet et al., 2013b; Bossuyt and Honnay, 2008). Late successional and woody species are desirable for revegetation since they provide valuable habitat, important erosion resistance during high flows and eventually contribute large woody debris to the river (Hubble et al., 2010; Bragg and Kershner, 1999; Tabacchi et al., 1998). These species are generally not well represented in riparian seed banks and those that are present may require specific cues to germinate (Williams et al. 2008). The presence of weeds in riparian seed banks is another limitation, particularly in agricultural areas (O'Donnell et al., 2014a; Corr, 2003; Jansen and Robertson, 2001). The effect of these biophysical limitations on revegetation success may be increased by the lack of understanding surrounding the utility of seed banks in passive revegetation. The implications are that mismanagement may lead to in-effective outcomes. For instance applying passive revegetation to a river bank which requires physical re-enforcement by tree species, since trees are unlikely to regenerate substantially under passive revegetation (Hopfensperger, 2007).

Bench units have been identified as having the greatest potential for passive revegetation utilising the seed bank in south-eastern Australia, due to their high degree of seed bank diversity and abundance in comparison to instream bar or floodplain units (O'Donnell et al., 2014a; O'Donnell et al., 2014b). Benches are geomorphic units of major sediment storage, which form in widened channels and are commonly comprised of coarser fraction sediments deposited by within-channel processes (Brierley and Fryirs, 2005; Erskine and Livingstone, 1999). Bench units are key targets for revegetation as they store significant amount of sediment, contribute to channel contraction processes and river recovery, they are important sites of vegetation regeneration and organic matter accumulation (Brierley & Fryirs, 2005; Vietz et al., 2004; Changxing et al., 1999; Junk et al., 1989). Scouring of bench surfaces by flooding provide suitable substrate for germination (Goodson et al. 2001). Bench units have been used as targets for environmental flows for geomorphic maintenance and recruitment of organic matter (Farquharson et al., 2011; Vietz et al., 2005). As such bench units will be the geomorphic unit focussed on in this thesis.

<u>Examining livestock exclusion and environmental flows as passive riparian revegetation techniques</u>

Livestock exclusion

Livestock grazing is a common land use across much of Australia and causes considerable detrimental effects to both the riparian condition and geomorphic integrity of rivers (Robertson, 1997; Trimble and Mendel, 1995; Fleischner, 1994; Wilson, 1990). Livestock exclusion is a common and widespread riparian revegetation technique in river rehabilitation globally and within Australia (Rutherfurd and Gippel, 2001; Zöckler 2000; FISRWG 1998). Livestock exclusion involves excluding an area from grazing and allowing it to passively regenerate (Schneider, 2007). This passive revegetation technique has been found to be successful in increasing vegetation cover, vegetation height, root growth, leaf litter, biodiversity, water infiltration rate, reducing exposed surfaces and streambank erosion, improving water quality and riparian condition (Batchelor et al., 2015; Herbst et al., 2012; Burger et al., 2010; Kauffman et al., 2004; Line, 2003; Jansen and Robertson, 2001; Green and Kauffman, 1995; Kauffman et al., 1983). Different types of river are affected to different degrees and in different manners by grazing (Myers and Swanson, 1992). Additionally vegetation responses to livestock exclusion are not uniform across sites (Sarr, 2002; Kauffman et al., 1984; Platts, 1979). Therefore understanding the success of livestock exclusion in a specific setting remains a useful exercise. This study will directly assess the success of livestock exclusion based on two key passive revegetation objectives of increasing erosion resistance and restoring nearby extant or pre-existing native vegetation (Hunter-Central Rivers CMA, 2011). Direct assessment against objectives is considered key in understanding not only how the biophysical aspects of a site may constrain the success of passive revegetation, but also how management expectations and objectives may affect success (Skinner et al., 2008; Ladson et al., 1999). In this way the responses and utility of the seed bank for passive revegetation under livestock exclusion will be explored. Understanding the response of the environment to the removal of the degrading pressure is a key part of improving techniques and directing further management steps (Pettit and Froend, 2000).

Numerous studies have reported significant increases in vegetation cover and reductions in exposed sediment under livestock exclusion as indicators of erosion resistance (Burger et al., 2010; Carline and Walsh, 2007; Hoover et al., 2001; Robertson and Rowling, 2000; Green and Kauffman, 1995; Schulz and Leininger, 1990). This thesis will also use these criteria to make quantitative comparisons between a livestock excluded and continuously grazed site. Several studies have identified decreases in sediment transport and erosion of stream banks, due to the removal of grazing pressure and trampling. For example (Kauffman et al., 1983) found streambank erosion to be significantly reduced under livestock exclusion, regardless of vegetation type after two years. Batchelor et al. (2015) reported a decrease in both channel widths and bank erosion after 24 years of livestock exclusion. Reductions in suspended sediment, attributed to increased bank stability and reduced erosion have been recorded by several authors. An order of magnitude decrease in suspended sediment concentration and transport was identified in Western Australia (McKergow et al., 2003), 40% in Ohio (Owens et al., 1996), 85% in New Zealand (Williamson et al., 1996) and a 47-87% decrease in Pennsylvania (Carline and Walsh, 2007). These studies have measured the combined effects of vegetation regeneration and the absence of livestock as destructive agents over time. This thesis will take a different approach. Roughness measurements will be used to determine how effective the regenerating vegetation under livestock exclusion is at reducing potential bench sediment transport during a flood event, in comparison to pre livestock exclusion roughness and potential roughness under further revegetation steps.

The response of vegetation to livestock exclusion is generally significant but may be variable among communities, individuals, locations, grazing intensities and time since exclusion (Jansen et al., 2007; Sarr, 2002; Platts, 1979). Some studies have identified reductions in species richness under livestock exclusion due to the decreased disturbance favouring the establishment of exotic species (Green and Kauffman, 1995), while others found no difference (Hoover et al., 2001). Schulz and Leininger (1990) found both significant increases and decreases among individual species in response to livestock exclusion. Hough-Snee et al. (2013) found for a riparian community in Utah an increase in hydrophytic species and a decrease in grazing tolerant species under livestock exclusion after four years. Many authors have noted a lack of regeneration in woody species (Hough-Snee et al., 2013; Williams et al., 2008; Jansen and Robertson, 2001). Few studies have explicitly examined the role of riparian seed banks in the regeneration of riparian plant communities after livestock exclusion. Williams et al. (2008) found the seed bank of cleared and grazed riparian areas to hold greater abundances of germinable seeds, particularly annual grass species, than wooded riparian areas. The authors found that riparian seed banks held good potential for the regeneration of native understory species after livestock exclusion. Nicol et al. (2007) found that the seed bank of an ephemeral wetland in the Murray-Darling Basin, which had been grazed by sheep, to be lower in seed density and species richness than a non-grazed wetland. Disturbance through grazing will increase opportunities for seed incorporation into the seed bank, increasing the concentration of seeds, including exotic species at the top of the seed bank (Williams et al., 2008). This thesis will determine the changes in vegetation composition in response to livestock exclusion to examine whether the technique has been successful in restoring native species. These findings will provide an indirect measurement of the utility of the seed bank for passive revegetation. The effects of three years of livestock exclusion will be assessed using a study reach of Olney Arm Creek, a tributary of Wollombi Brook, in the Lower Hunter Region, NSW, Australia.

Environmental flows

Environmental flows have been used with some success for maintaining and rehabilitating various aspects of river health such as facilitating fish passage, providing spawning ground, inundating seasonal wetlands for bird breeding events, flushing and rejuvenating habitat (Docker and Robinson, 2014; Poff and Zimmerman, 2010; King et al., 2009; Vietz et al., 2005; Dollar, 2004; Chessman et al., 2003; Whiting, 2002). The seed of riparian species may be adapted to hydrological regimes and these adaptations may be reflected in riparian seed bank composition (Greet et al., 2013a; Brock, 2011; Merritt et al., 2010; James et al., 2007). Environmental flows have been used to maintain riparian vegetation in wetland or floodplain coupled river systems. For example Catford et al. (2011) found that environmental flow releases, coincident with spring flooding to River Murray wetlands would reduce terrestrial weed species and facilitate the growth of native macrophytes. Environmental watering may also be used to stimulate the recruitment of species from riparian seed banks. Robertson and James (2007) found ephemeral riparian species to exhibit longevity in the seed banks of a degraded floodplain and that returning flooding regimes would increase their recruitment. However the authors also caution that environmental watering may facilitate the establishment of exotic species. Capon (2007) identified an increase in seedling emergence in a desert floodplain after 4-8 weeks of inundation, and an increase in the proportion of annual monocot species. Greet et al. (2013a) recorded greater seed abundances and richness in riparian seed banks along regulated in comparison to unregulated lowland rivers in northern Victoria. After a floodplain environmental watering event the seed bank of a lowland river in semi-arid Australia was found to become an increasingly important source of richness to the regenerating community when compared to the standing vegetation (Reid and Capon, 2011). Wetzel et al. (2001) found that reconnecting a floodplain wetland of the Kissimmee River in Florida via environmental watering would enhance the regeneration for one of three target communities.

Knowledge gaps such as establishing what the environmental responses to natural and regulated flow variation are have been identified as important obstacles to implementing water management in regulated systems (Naiman et al., 2008). It is unknown whether it is possible to use environmental flows to stimulate germination in desirable riparian species stored in the seed bank of rivers in south-eastern Australia. If possible, environmental water releases may be manipulated in order to modify the depth and durations of inundation to geomorphic units along regulated reaches. Changes in duration, frequency and depth act as an "environmental sieve", restricting the species that are able to germinate from the seed bank (Jensen et al., 2008). Unpublished work by Harris et al. (2011), using a limited number of species, found that inundation depth may not be a key factor in germination from the seed bank of riparian species from the Hunter Valley. This thesis will explore whether varying durations of inundation of bench units may increase the recruitment of desirable species from the seed bank of degraded rivers. The case study site is the Hunter River, in the Upper Hunter Region, NSW, Australia.

3. Thesis aims

This thesis will firstly address the effectiveness of livestock exclusion, an existing passive revegetation technique in degraded rivers, to achieve the revegetation objectives of improving erosion resistance and restoring native vegetation. It is expected that due to the prolonged duration of grazing and the degraded landscape condition the restoration of native vegetation will be poor, although relief from grazing will drastically improve erosion resistance. Then this thesis will examine the potential for environmental flows, a potential passive revegetation technique, to be used to improve the recruitment of desirable species from the seed bank of a degraded river system. It is expected that species which are inundation tolerant will exhibit greater seedling emergence numbers than non-inundation tolerant species. By investigating the efficacy of these passive revegetation techniques further understanding of the utility and limitations of riparian seed banks for river rehabilitation will be provided. These aims will be addressed by answering the following research questions, which are framed under three sections:

1. **Measuring the success of livestock exclusion and environmental flows as passive riparian revegetation techniques**
    i. Did the regenerating vegetation under livestock exclusion increase erosion resistance and restore native species after three years?
    ii. Did seedling emergence significantly increase under varying environmental flow assisted or natural flood inundation durations?

2. **Management considerations**
    i. What further management actions should be taken to improve the effectiveness of livestock exclusion as a passive revegetation technique?
    ii. What steps need to be taken in order to implement environmental flows as a passive revegetation technique in degraded rivers of south-eastern Australia?
3. **Assessing the utility of the seed bank under passive revegetation**
    i. How do the seed bank composition and management objectives limit the effectiveness of livestock exclusion as a passive revegetation technique?
    ii. How may flooding and the use of environmental water releases affect seed bank dynamics within degraded rivers?

Chapter 2: Regional setting

This thesis comprises two studies a livestock exclusion and environmental flow study, which were conducted in the Lower and Upper Hunter Regions of the Hunter Valley, New South Wales respectively. This pair of passive riparian revegetation techniques, livestock exclusion and environmental flows, have been studied together because they are widely used within river systems that have experienced the common processes of degradation through riparian land clearing, grazing and flow regulation. The sites are separated since the livestock exclusion study required data collection over an extended period of time and a data collection survey had been carried out in January 2012. The environmental flow study reach had to be regulated and reserve an environmental water allocation, which is not available in the Lower Hunter Region. The study reaches for both projects are deemed to be representative of the characteristics of many degraded rivers across south-eastern Australia. The position of the study reaches within the Hunter catchment is shown in Appendix 1.

1. Lower Hunter Region (livestock exclusion)

The 0.7 km study reach is located at Olney Arm Creek, an upper tributary of Wollombi Brook, in the Wollombi Brook subcatchment (Figure 2.1). The Wollombi Brook subcatchment drains an area of approximately 470 km^2. The mean annual rainfall is approximately 900 mm (O'Donnell et al., 2015). The rivers within the subcatchment are sand-dominated, derived from Triassic intercalated sand-stone and shale that comprises the catchment (Erskine and Saynor, 1996). The Hunter Region has a high flash flood magnitude index (Erskine and Saynor, 1996). At the study reach, Olney Arm Creek is a partly confined, planform controlled, low sinuosity, sand bed river.

The study reach is divided between two sites, livestock-exclusion and continuously-grazed (Figure 2.1). The livestock-exclusion site was fenced along the top section of bench level two, which is the upper bench level, to prevent grazing within approximately 10 m either side of the low-flow channel. The livestock exclusion fences were established in January 2012. The floodplain vegetation outside of the fences is dominated by pasture grasses, herbaceous weeds and bracken (*Pteridium esculentum*), the regenerating vegetation within the fences also includes sparse numbers of *Acacia parvipinnua* and *Melaleuca linariifolia* and a variety of more obvious sedges and rush species. The vegetation on the valley slopes consists of the Wollombi Redgum-River Oak Woodland, Sheltered Blue Gum Forest communities, Coastal Ranges Open Forest, Hunter Range Grey Gum Forest and Sheltered Rough Barked Apple Forest communities (LHCCREMS, 2000). Intensive grazing and clearing of riparian vegetation has resulted in extensive erosion creating an over-enlarged macrochannel, ranging between 55 and 140 m in width, with two levels of bench units inset within the macrochannel (c.f. Fryirs et al., 2012; Erskine et al., 2010). A geomorphic condition assessment conducted in 2011 described both sites as being in poor geomorphic condition with moderate recovery potential (Hunter-Central Rivers CMA, 2011).

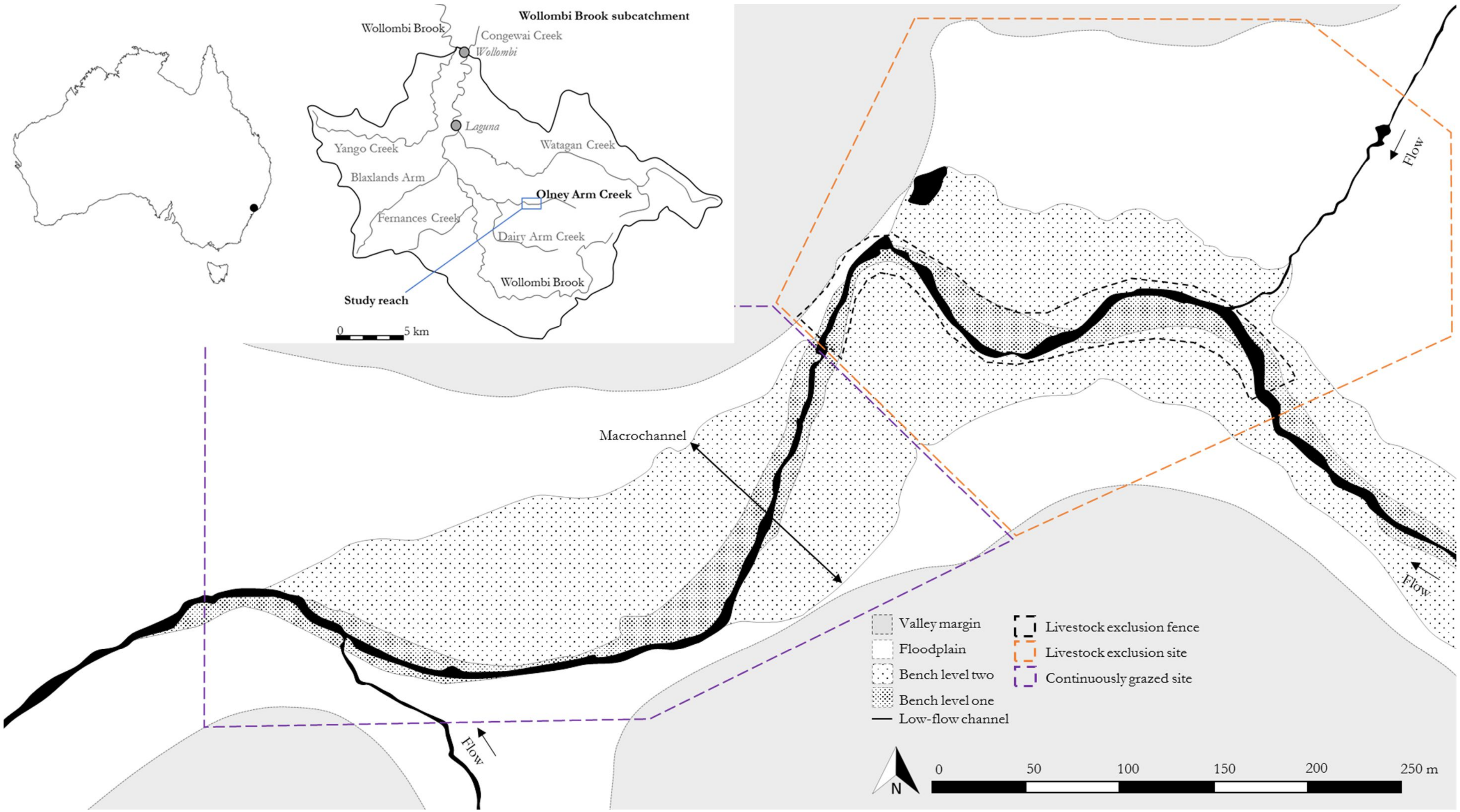


Figure 2.1. Geomorphic map of the study reach at Olney Arm Creek showing the livestock-exclusion and continuously-grazed sites.

2. Upper Hunter Region (environmental flows)

The study reach is shown in Figure 2.2. The Upper Hunter subcatchment drains an area of approximately 4220 km^2 (Hoyle et al., 2007). The reach is approximately 5 km south west of Muswellbrook. Rainfall for the Muswellbrook area averages 600 mm per year, with the upper catchment areas receiving in excess of 1400 mm per year (Kyle and Leishman, 2009). The Muswellbrook bridge gauge (gauge no: 210002) record extends back to 1907 but has been augmented with archival information of significant flood heights and extrapolated data from the Maitland and Singleton gauges back to 1806 (Hoyle et al., 2008). A 1:100 year flood event occurred in 1955, the magnitude of this event may have been exceeded in the 1806 and 1870 flood events when the river was under different channel conditions (Hoyle et al., 2008). The average bankfull discharge is 1700 m^2/s with an average recurrence interval of 14 years. 90% of flows are less than 12 m^2/s, with 10% being less than 1 m^2/s (Hoyle et al., 2008). Glenbawn Dam was completed in 1958, 11 km upstream of Aberdeen, capturing approximately 30% of the catchment upstream of the study reach. The dam resulted in a lowering of large flows and an increase in low flows (Erskine, 1985). The sediment trap at the dam has an efficiency of approximately 98.9% (Erskine, 1992). The Upper Hunter River extends upstream of the Goulburn River confluence. Three major tributaries the Rouchel Brook, Dart Brook and the Pages River join the Hunter River between Glenbawn Dam and the study reach and serve to establish a "high degree of naturalness to the flow regime" (M. Simons 2015, pers. comm., February 9).

The Upper Hunter Region has been subject to intensive land use since European Settlement in the 1820's, initially by clearing, grazing and de-snagging and latterly by flow regulation and coal extraction (Connor et al., 2004, Hoyle et al., 2008). The floodplain has been extensively used for agriculture (Kyle and Leishman, 2009). River training works were implemented in the 1950's in order to stabilise river movement and the 8 km study reach is now a passive meandering, low to moderate sinuosity, gravel bed river (Hoyle et al., 2007). The riparian vegetation community is dominated by exotic herbaceous species and willows (*Salix* spp.) (Brierley et al., 2005). Intermittent and extensive bar and bench units are inset within a macrochannel that ranges in width between 75 to 600 m (Hoyle et al., 2008).

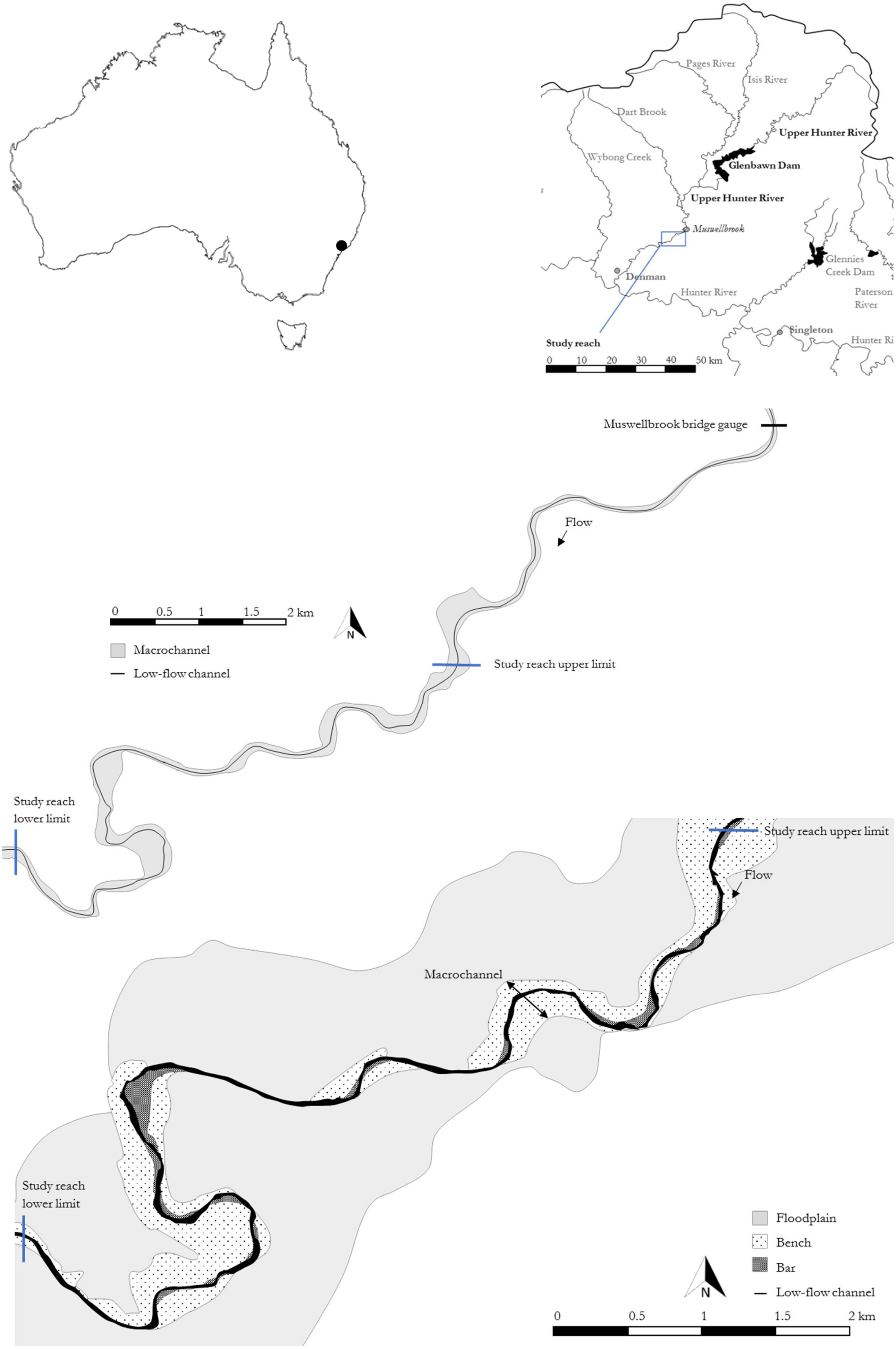


Figure 2.2. Geomorphic maps showing the study reach within the Upper Hunter River subcatchment.

Chapter 3: Methods

1. Livestock exclusion

Data was collected over a three year period from summer 2012 to winter 2015. Three sets of survey methods were designed to ascertain whether passive regeneration from the seed bank under livestock exclusion achieved the dual revegetation objectives of increasing erosion resistance and restoring native vegetation (Hunter-Central Rivers CMA, 2011). To establish whether livestock exclusion was effective, geomorphic and vegetation composition data has been compared between a livestock-exclusion site and a continuously-grazed (control) site, shown in Figure 3.1. Pre livestock exclusion site data was not available and therefore the downstream continuously-grazed site is used in place of pre livestock exclusion data. The site photos in Figure 3.2 show the livestock-exclusion site to be in an approximately equivalent condition to the continuously-grazed site in 2012 before fencing. Additionally the valley wide vegetation survey data has been used as a further control measure to attribute changes in vegetation composition within the fences at the livestock-exclusion site to livestock exclusion. Only a section of the livestock-exclusion site within the top of bench level two has been protected from grazing but for ease of differentiation between the sites it will be referred to as the livestock-exclusion site.

<u>Geomorphic surveys</u>

Cross-section surveys were conducted in 2012 (after 24 months of livestock exclusion) and 2015 (after 42 months of livestock exclusion) to identify any changes in the geomorphic structure of the study reach over time. A Leica TCR-705 total station was used to survey 10 transects across the valley floor, as near to perpendicular to the macrochannel as possible, at both the continuously-grazed and livestock-exclusion sites (Figure 3.1). Manning's n values for both sites were calculated using the component method during winter 2015 using the visual coefficients defined by Arcement & Schneider (1989), to compare the differences in roughness between the sites as a measure of erosion resistance. Sediment samples from the upper 2 cm of the surface were collected from each vegetatation survey quadrat in Summer 2015. Field texture analysis (National Committee on Soil and Terrain, 2009) was undertaken to describe the geomorphic units that occured within the study reach. A visual estimate using a hand lens of the percentage composition of grain sizes for the coarse fraction of sediment was conducted using minimum increments of 5. A total of five samples for bench level one and bench level two at both the livestock-excluded and continuously-grazed sites were analysed and the results averaged for each bench level and site combination.

A moderate flood event occurred during April 22nd and 23rd 2015, inundating bench level one at the flood peak but not exceeding the height of bench level two. There is no flood guage in close proximity to the site but debris left by the flood event was used as a proxy for flood height at each cross-section. The

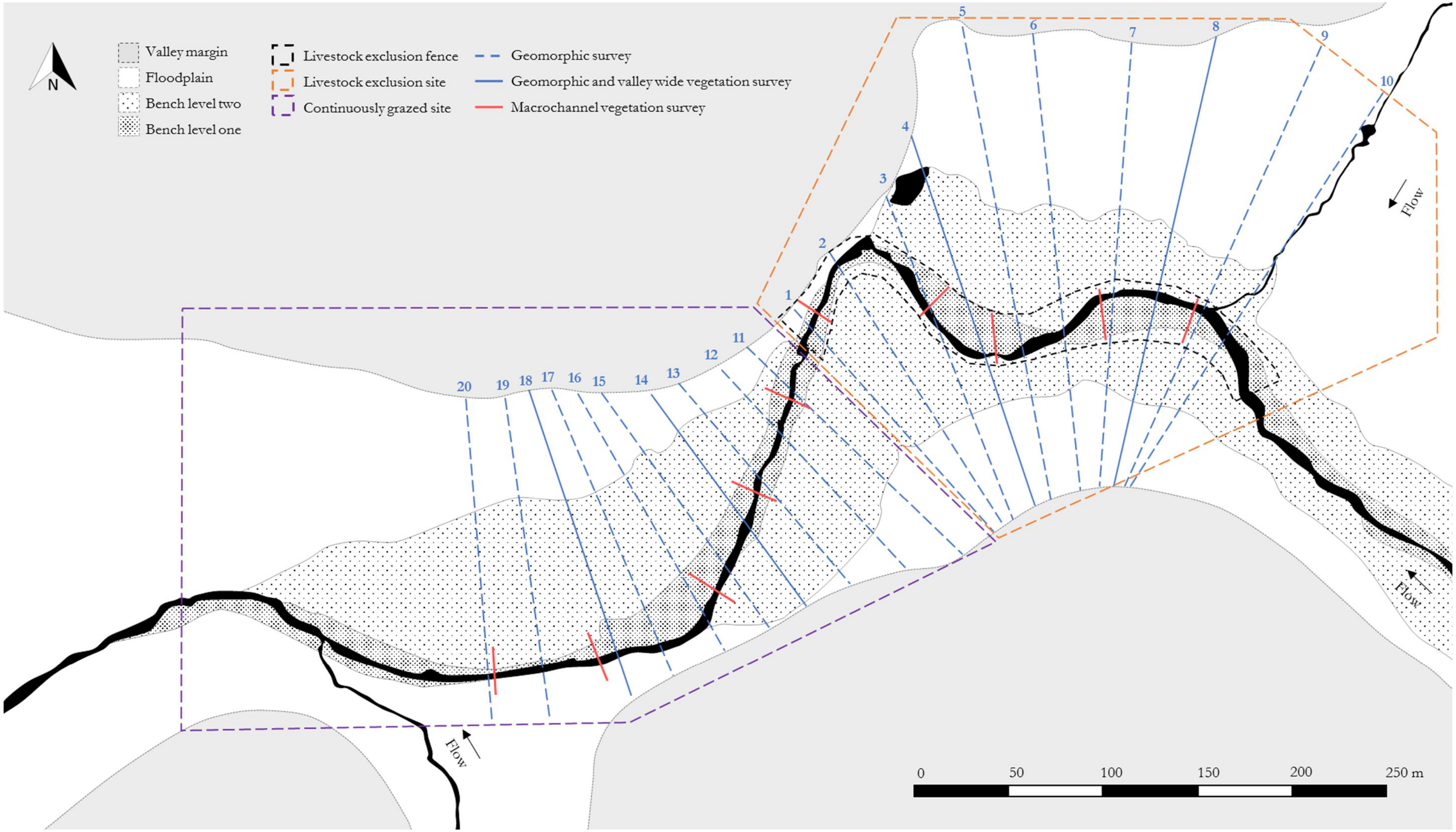


Figure 3.1. Geomorphic map of the livestock exclusion and continuously grazed study sites and survey transects at Olney Arm Creek.

flood height data was then used to calculate discharge values during the flood event for each cross-section. The discharge and grain size data was then used to simulate how different management scenarios, creating different degrees of roughness, would increase or decrease unit stream power and potential bench sediment transport for each cross-section using Geomorphic Assesor (Parfait, 1998). The Ackers-White method for calculating potential sediment transport was used with the D_{35} particle diameter being 0.21 mm for the livestock-exclusion site and 0.26 for the continuously-grazed site. The roughness coefficients scored at the continuously-grazed site were used as a 'pre' revegetation roughness scenario and the coefficients scored at the livestock-exclusion site was used as a 'passive' revegetation scenario. An 'active' re-vegetation scenario was scored a roughness coefficient of 1.0 representing 3 year old plantings of trees and shrubs equivalent to 'medium-dense brush' (Arcement and Schneider, 1989). These simulated roughness values were applied to the section within the fences at the livestock-exclusion site and to an equivalent area at the continuously-grazed site (10 m either side of the low-flow channel). Roughness values were assigned the grazed floodplain and bench sections (0.025), the revegetation simulation bench sections (pre: 0.025, passive: 0.040, active: 0.100), and to the low-flow channel section itself (0.045). The roughness values for the low-flow channel and sections outside of the fences remained constant for all simulations. The slope data used in the discharge and sediment transport calculations were derived by graphing the relative height of the lowest point of the channel bed, excluding pool sections, of each cross-section at each site and the distance between each point between the transects (Figure 3.3).

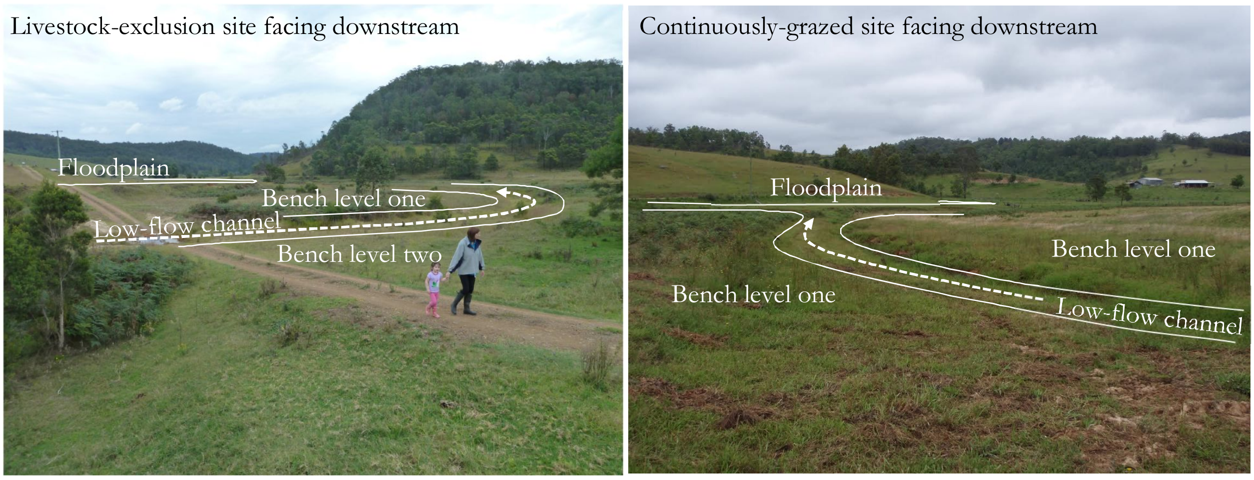


Figure 3.2. A visual comparison of the condition of the livestock-exclusion and continuously-grazed sites before the livestock exclusion fences were established. Photos taken in summer 2012 facing downstream.

<u>Valley floor vegetation surveys</u>

Vegetation surveys extending from either side of the valley margin were conducted in summer 2012 (pre livestock exclusion), autumn 2014 (after 28 months of livestock exclusion), summer 2015 (36 months) and autumn 2015 (38 months). These surveys were designed to identify any changes in the vegetation composition outside of the fences over time. The data from these surveys were compared over time, not between sites. Confirming that there were no changes in the vegetation composition outside of the

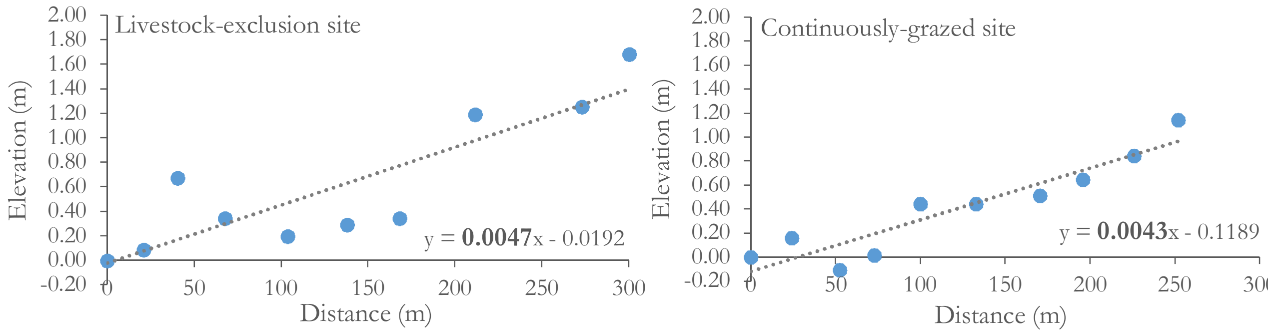


Figure 3.3. Slope calculation results for Olney Arm Creek at the livestock-exclusion and continuously-grazed sites. Data points display the lowest point of the channel at each transect. Axis show relative differences in height and distance between the points.

livestock exclusion fences during the study period allows any changes in vegetation composition within the fences to be attributed to the effects of livestock exclusion. These surveys were carried out across two transects at both the control and livestock-exclusion site, shown in Figure 3.1. A 1 m^2 quadrat was used to visually estimate percentage cover to the species level, every 10 metres across each distinct geomorphic unit on each transect, excluding the fenced section at the livestock-exclusion site and an equivalent area at the continuously-grazed site. A total of twenty-four and twenty-five grazed quadrats were sampled at the continuously-grazed and livestock-exclusion sites respectively. The results of these analysis are provided in Appendix 2. Any significant differences between time periods were attributed to valley wide effects, e.g. rainfall, since both sites share the same pattern of increase or decrease in critera over time. The only exception is exotic species richness between the autumn 2014 and 2015 samples, therefore care has been taken when interpreting the results for exotic species.

Macrochannel vegetation surveys

A second set of vegetation surveys were conducted to compare differences in the vegetation composition of the bench units within the livestock exclusion fences and a comparable area at the continuously-grazed site. These surveys were conducted in winter 2014 (after 30 months of livestock exclusion), summer 2015 (36 months) and winter 2015 (42 months). The percentage cover of vegetation per quadrat is an indicator of the degree of sediment protected by vegetation. Five 20 m transects were surveyed at both the livestock-exclusion and continuoulsy-grazed sites, perpendicular to the low-flow channel. A 1 m^2 quadrat was sampled every second metre along the transect, excluding the low-flow channel, so that five quadrats were taken either side of the low-flow channel. Where the distance from the low-flow channel to the edge of the fence was too short to sample five quadrats every second meter, extra quadrats were added to the opposite side to ensure a total of ten quadrats were sampled per transect. The data were grouped so that fifty quadrats were sampled per site. The quadrats were sampled by visually estimating vegetation percentage cover to the species level for each layer of vegetation, so that the total was not limited to one hundred percent. The percentage cover of exposed sediment per quadrat was also visually estimated as an independent measure of susceptibility to erosion.

Statistical analyses

The species level data were split into origin, either native or exotic, and their growth form according to the National Committee on Soil and Terrain (2009). The identification website Plantnet (Royal Botanical Gardens Sydney, 2015) was used to define origin and growth forms. Naturalised species were defined as exotic for this study in order to separate the desirable natives from the undesirable exotics and pasture species. The valley wide vegetation survey data were compared to identify changes in vegetation compostion over time either between summer or autumn, not between sites. This approach prevented any confounding differences due to seasonal change. Data from the macrochannel surveys was compared between the livestock-exclusion and continuously-grazed site using independent t-tests and Mann Whitney U tests, where data was non-normal and could not be transformed to normality, for each survey period. The comparisons used percentage cover, species richness and the native and exotic components of each for the comparisons. The criteria for comparison under both vegetation survey methods was percentage cover and species richness. A presence or absence comparison was also made between the species identified in the vegetation surveys and the surrounding extant communities, according to the CMA site report (2011) and LHCCREMS (2000) vegetation community profiles. Minitab version 17 was used to conduct these statistical analyses (Minitab 17 Statistical Software, 2010). PERMANOVA and SIMPER analyses using Bray-Curtis similarity were also applied to the macrochannel survay data to identify species level differences in vegetation composition between the sites and bench levels, determining which species accounted for fifty percent of the difference. These multivariate statistics were conducted using PAST statistical software (Hammer et al., 2001).

2. Environmental flows

Geomorphic modelling

A seedling emergence experiment was conducted to examine the effects of flood durations on the seedling emergence of ten native riparian species that are considered desirable for river rehabilitation. This information was then used to ascertain whether environmental water allocations could be used for passive riparian re-vegetation along degraded rivers. Light Detection and Ranging (LiDAR) data was used to construct fourty-eight cross-sections of the channel of the Hunter River (Figure 3.4) along the study reach. The natural and environmental water release asssisted flood levels required to inundate the bench units at each cross-section were calculated using Geomorphic Assessor (Parfait, 1998). Discharge data from the flow guage at Muswellbrook (gauge number: 210002), approximately 5 km upstream of the study reach, was obtained from PINEENA CM and used to model 1:1, 1:2, 1:5, 1:10, 1:20 and 1:25 average return intervals (hereafter ARI's) for each cross-section. The minimum ARI required to fully inundate benches was found to be the 1:5 ARI. A 1:5 ARI along the study reach is unlikely to result in significant bench erosion (Hoyle et al., 2012) and so represents an appropriate interval for environmental flows for passive re-vegetation purposes. Flood exceeding the 1:5 ARI were then identified in the historic record at Muswellbrook, which extends back to 1913, and the duration that those floods persisted above the 1:5 ARI were recorded. The experiemental inundation durations identified were 12 hours, 24 hours, 48 hours and 72 hours. These flood durations were then used as the simulated inundation durations in a seedling emergence experiment with control treatment of 0 hours inundation.

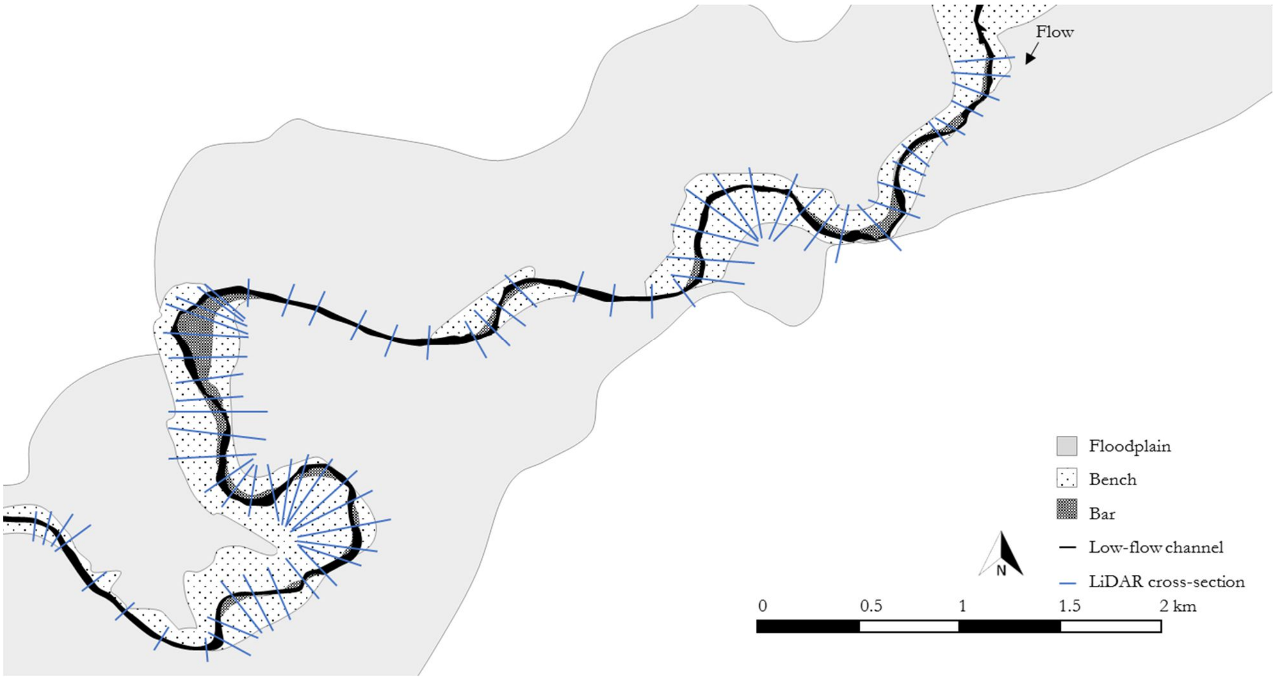


Figure 3.4. Geomorphic map of the Upper Hunter study reach showing LiDAR derived cross-sections.

Seedling emergence experiment

The ten species used in the seedling emergence experiment were chosen based on their occurrence along the Hunter River in the Upper Hunter Region and desirable rehabilitation status (UHRRI, 2004, Schneider, 2007). The tree growth form was represented by *Eucalyptus camaldulensis* and *Casuarina cunninghamiana*, shrubs by *Acacia parvipiunnula* and *Acacia salicina,* grasses by *Microlaena stipoides* and *Bothriochloa macra*, sedge by *Carex appressa* and rush by *Schoenoplectus validus* and *Bolboshoenus caldwellii. Lomandra longifolia* was included and although it is classified as a herb, in the riparian zone it performs a functional role similar to that of a grass or sedge due to it's tussocky growth. No other herb species were included in the study due to their comparitively low geomorphic value for soil binding and erosion resistance (Abernathy and Rutherfurd, 1999, Lyons et al. 2000). The importance of herbs for ecological succession, providing food or habitat was not a consideration for this study. Although woody species are generally poorly represented in sediment seed banks (O'Donnell, 2014, et al.; Hopfensperger, 2007), they have been included since it was desirable to test seedling emergence responses to inundation durations across a range of growth forms. Addittionally both the tree species used have seed that is spread by hydrochory, under which inundation inevitably occurs (Price & Lovett, 1999). In this way the study may more effectively direct further investigation and examine responses among the growth forms most desirable for river rehabilitation. Due to the low numers of species representing each growth form group no statistical comparisons can be made among growth form groups. The species can be broadly classified as inundation or non-inundation tolerant. Inundation tolerant species include *E. camaldulensis*, *C. cunninghamiana*, *C. appressa*, *S. validus* and *B. caldwellii.* Seeds with an Eastern States providence were selected wherever possible, these included *E. camaldulensis*, *M. stipoides*, *C. appressa*, *S. validus* and *B. caldwellii.* Pre-treatments were applied to the seeds as recommended, it was reasoned that the seeds within bench units would be scarified during incorporation into the sediment seed bank and by subsequent wetting and drying. This study assumed the seeds in the field would be ready to germinate and awaiting appropriate conditions to do so. A pilot study comparing pre-treatment to non pre-treatment found *A. parvipinnula* seeds to require pre-treatment to germinate and that *A. salicina* seeds germinated much more successfully with pre-treatments. Both Acacia species were placed in boiling water for 1 minute prior to planting. The *L. longifolia* seeds were soaked in water for ten days prior to planting, with the water replaced every 24 hours.

Seeds were planted in rectangular punnets to 5 mm depth in a medium of washed coarse river sand. This medium was chosen to closely represent in physical terms the bench sediment at the study reach. Ten seeds of a single species were planted in each punnet, forming one replicate. The seeds were planted in two rows of five. Ten replicates of each species were used for each inundation treatment, totalling one hundred seeds tested per species per treatment. The punnets were covered with a layer of muslin cloth and secured with wire around the rim, to prevent seeds floating and escaping the punnets during inundation. Punnets containing *E. camaldulensis* seeds were covered with a double layer of cloth due to their small size. The punnets were then lowered into containers of water so that the top of the sediment was inundated to a depth of 5 cm. The containers were left in a glasshouse at the Macquarie University plant growth facility to maintain a water temperature between 15 and 20 ˚C throughout inundation. Once the inundation period ended the punnets were removed from the containers and left to drain in seed trays in the glasshouse. Once drained, the muslin cloth was removed and carefully examined for any seeds which may have been caught. Any seeds that were found in the cloth or that had risen to the surface of the sediment were replanted in the centre to ensure the punnet represented an in situ sediment seed bank. The punnets were randomly distributed into seed trays throughout the glasshouse. The glasshouse temperature was maintained at approximately 25 ˚C during the day and 15 ˚C during the night. The seeds were watered by misting four times a day for 5 minutes each time. At day 36 of the experiment misting was reduced to 2 minutes each time due to a reduction in hot conditions as the experiment moved into winter. Newly emerged seedlings and new deaths were recorded every second day for seventy days. Emergence was recorded as the first day of shoot or root emergence from the sediment.

Statistical analysis

The most successful treatment was defined as that which produced the greatest seedling emergence after seventy days. One way ANOVA's were used to compare seedling emergence success under the different treatments. Independent t-tests were then used to compare the most successful treatment against the least successful and control treatment. Minitab version 17 was used to conduct these analyses (Minitab 17 Statistical Software, 2010). The timing of seedling emergence was compared among treatments for each species using Kaplan-Meier survival analyses in SPSS statistical software (IBM Corp, 2013), which uses log-rank, Breslow and Tarone-Ware tests.

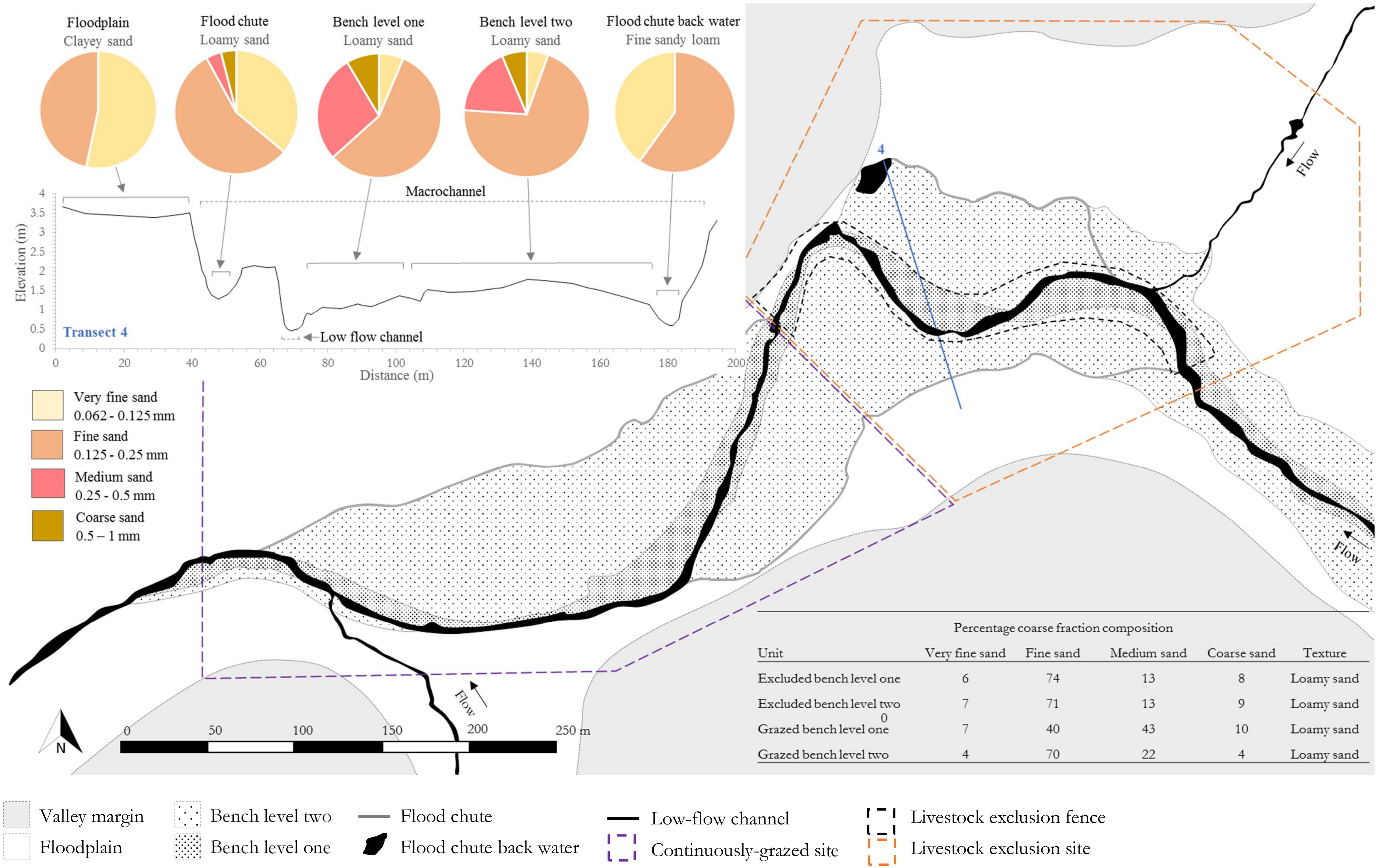
Chapter 4: Results

1. Livestock exclusion

Geomorphic site description

The study reach and the geomorphic units within the reach are shown in Figure 4.1. The two bench levels at the study reach indicate two stages of channel contraction. Bench level one is the lower level and is inset within the bench level two suggesting that it was formed at a later stage and that channel contraction processes are ongoing. Bench level one is the smaller of the two levels ranging in width between 5 and 26 m, while bench level two is more extensive ranging between 17 and 93 m in width along the study reach. Figure 4.1 shows the sedimentary characteristics of the coarse fraction of sediment for each of the common geomorphic units. Both levels are comprised of loamy sand with the coarse fraction dominated by fine sand, the distinguishing factor being that bench level one has a greater percentage of medium sand. This difference is likely due to the frequency of flows that have the energy to move medium sand onto the surface of this units. Since bench level two is higher and further from the channel, these flows occur less frequently and therefore less medium sand is deposited on the surface of this unit. The inset table in Figure 4.1 displays sedimentological differences between bench levels one and two and between the livestock excluded and continuously-grazed sites. There is no substantial difference in composition between bench levels one and two at the livestock excluded site and bench level two is very similar between the livestock excluded and continuously-grazed site. There is a clear increase in medium sand at bench level one when compared to bench level two at the continuously-grazed site and when compared to bench level one at the livestock-exclusion site.

Flood chutes, which form when flood waters are realigned over the benches and short-circuit channel bends (Brierley & Fryirs, 2005), occur at both sites on the distal edges of bench level two. Flood chutes are found abutting the macrochannel bank on at least one side of the channel at any location. The



Percentage coarse fraction composition

Unit	Very fine sand	Fine sand	Medium sand	Coarse sand	Texture
Excluded bench level one	6	74	13	8	Loamy sand
Excluded bench level two	7	71	13	9	Loamy sand
Grazed bench level one	7	40	43	10	Loamy sand
Grazed bench level two	4	70	22	4	Loamy sand

Figure 4.1. Geomorphic map displaying the location of common geomorphic units within the study reach and coarse fraction sedimentlological descriptions of geomorphic units. Pie charts show percentages based on mean average values across the study reach. The inset table shows percentages based on mean average values for bench levels one and two for the livestock excluded and continuously-grazed site individually.

sediment of these units are classed as loamy sand and are dominated by fine sand. In these units there is a greater proportion of very fine sand with less medium and coarse sand than for the bench levels. A flood chute back water occurs where a flood chute loses energy such that it cannot return to the low-flow channel creating a semi-permanent area of standing water. Since this unit is at the most distal edge of bench level two and at the furthest point along the flood chute it is unsurprising that the coarse fraction of the sediment is composed of very fine and fine sand, the latter being the coarsest calibre of sediment at this unit. The field texture of fine sandy loam indicates a greater degree of clay is deposited here during times when the water is standing in the flood chutes. At the study reach the flood chute back water sits against bedrock and water can only re-enter the low-flow channel if it has the energy to be forced around the valley margin. The floodplain sediment is classed as clayey sand and is dominated by very fine sand. Being the infrequently and carries the finer calibre materials to the floodplain when it does.

<u>Comparing erosion resistance between the livestock-exclusion and continuously-grazed sites</u>

Two key erosion resistance criteria, the total percentage cover of vegetation per quadrat and the percentage of exposed sediment per quadrat, were compared between the livestock-exclusion and continuously-grazed sites. Table 4.1 shows that the livestock excluded site had significantly less exposed sediment than the continuously-grazed site in winter 2014 ($P = 0.0004$), summer 2015 ($P = 0.0038$) and winter 2015 ($P = 0.0046$). The mean total percentage cover of vegetation was greater at the livestock excluded site for each time period, although not significantly for any time period. The livestock excluded site scored a greater Manning's n value of 0.067 compared to 0.040 for the continuously-grazed site (see Table 4.2). The key components driving this increase are amount of vegetation and effect of obstructions, despite the continuously-grazed site having a higher degree of irregularity. A visual comparison of the vegetation structure at the livestock excluded and the continuously-grazed site is shown in Figure 4.2. The photos demonstrate that there is a greater amount of vegetation and structural diversity at the livestock excluded site, as early as spring 2011, which was six months after livestock exclusion. This is due to the increased size of the vegetation and the regenerating tree species at the livestock-exclusion site, which were not captured in the quadrat sampling due to their patchy distribution. A greater amount of instream vegetation can also be observed at the livestock excluded site. Additionally a seasonal difference can also be observed, with much of the visible vegetation during winter 2014 and to a lesser extent winter 2015, being deceased at the study reach. This is compared to the spring and summer photographs which are covered by living vegetation.

Table 4.1. Results of independent t-tests and Mann-Whitney U tests for differences in percentage cover per quadrat between continuously-grazed and livestock excluded sites at three time periods.

			Winter 2014			Summer 2015			Winter 2015		
Criteria	Site	N	Mean	T-value	P-value	Mean	T-value	P-value	Mean	T-value	P-value
Total percentage cover of vegetation	Grazed	50	48.1	0.54	0.587	85	1.6	0.112	62.4	0.85	0.407
	Excluded		51.7			95.4			67.3		
			Median	W	P-value*	Median	W	P-value*	Median	W	P-value*
Percentage cover of exposed sediment	Grazed	50	20	1956.5	0.0004**	10	2137.5	0.0038**	12.5	2126.5	0.0046**
	Excluded		0			0			2.5		

*P-value is adjusted for ties.
**P-values < 0.05 are significant.

Table 4.2. Manning's n values for each site with the visual coefficients used in the calculations.

Site	n_b	n_1	n_2	n_3	n_4	m	Manning's n
Grazed	0.030	0.005	0.001	0.002	0.002	1.000	0.040
Excluded	0.030	0.001	0.001	0.010	0.025	1.000	0.067

Visual coefficients defined by Arcement & Scneider (1989). Manning's n formula: $n = (n_b + n_1 + n_2 + n_3 + n_4)m$.
nb = base value, n1 = degree of irregularity, n2 = variation in channel cross section, n3 = effects of obstruction, n4 = amount of vegetation, m = degree of meandering.

The cross-sections in Figure 4.3 (livestock-exclusion) and 4.4 (continuously-grazed) identify only minor geomorphic change between 2012 and 2015. At the livestock excluded site transect four is unchanged, while transect six experienced minor incision of the low-flow channel and deposition of sediment within a flood chute on bench level two. The morphology of the low-flow channel at transect six suggests that the flat bed identified at this section in 2012 may currently be transitioning to a pool section in 2015. At the continuously-grazed site transect twelve is unchanged, with erosion of sediment from bench level one occurring at transect sixteen. A knickpoint formed downstream of the transect twenty at the continuously-grazed site during the flood event in April 2015 resulting in a bed lowering of 0.6 m, shown in Figure 4.5.

<u>Comparing the effects of simulated re-vegetation scenarios on flood processes</u>

Debris left by the moderate April 2015 flood event provided a proxy for flood height. Based on the reconstructed flood heights unit stream power and potential bench sediment transport values for three management scenarios were simulated using manipulated roughness values. These scenarios were pre (continuously-grazed site), passive (livestock-exclusion site) and active (livestock exclusion with directly planted woody species) re-vegetation scenarios. The discharge values used to calculate the unit stream power and sediment transport potential was that which was calculated from the reconstructed flood height for each transect, with the average unit stream power being 20 m^3/s for the livestock-exclusion

Figure 4.2. Photographic comparison of the livestock excluded and continuously-grazed sites over time. Photo A was taken facing upstream while stood on bench level one. All other photos are taken facing downstream, while standing on bench level one at both sites.

site and 16 m^3/s for the continuously-grazed site. A comparison of unit stream power and potential bench sediment transport averaged across the study reach is shown in Figure 4.6, demonstrating that the pre revegetation simulation experienced the greatest unit stream power and potential bench sediment transport. The pattern was followed for the individual sites. Livestock exclusion resulted in a reduction of potential bench sediment transport by 7434 tonnes per day (18%) when averaged across the livestock-exclusion site, compared to the pre revegetation condition. Active revegetation could reduce potential bench sediment transport by 9714 tonnes per day (22%), when averaged across the livestock-exclusion site, when compared to the pre management condition. Implementing livestock exclusion at the countinuously-grazed site could reduce potential bench sediment transport by 2280 tonnes per day (9%) and active revegetation by 2236 tonnes per day (9%), averaged across the site.

The majority of individual cross-sections displayed reductions in unit stream power and potential bench sediment transport under passive revegetation and further reduction under active revegetation (Table 4.3). Exceptions to this pattern are cross-sections six, ten, twelve and fourteen, which have the same values for each simulation criteria despite the differences in roughness. It is expected that at these cross-sections a threshold in discharge has been exceeded whereby the effect of the roughness values tested have no appreciable effect on reducing unit stream power or potential sediment transport. Another exception is the bench sediment transport value for the active simulation at transect eighteen which is greater than the passive value, despite the reduction in unit stream power for the active simulation. This anomaly is due to the channel bed slope being locally steeper in this section. In this case more vegetation (roughness) is required to have a significant effect on sediment transport capacity. The site of maximum reduction in potential bench sediment transport under livestock exclusion for the livestock-exclusion site was transect eight with 31286 tonnes per day (78%). At the continuously-grazed site the maximum reduction in potential bench sediment transport was transect sixteen which could have experienced a reduction of 5255 tonnes per day (93%) in potential bench sediment transport using livestock exclusion.

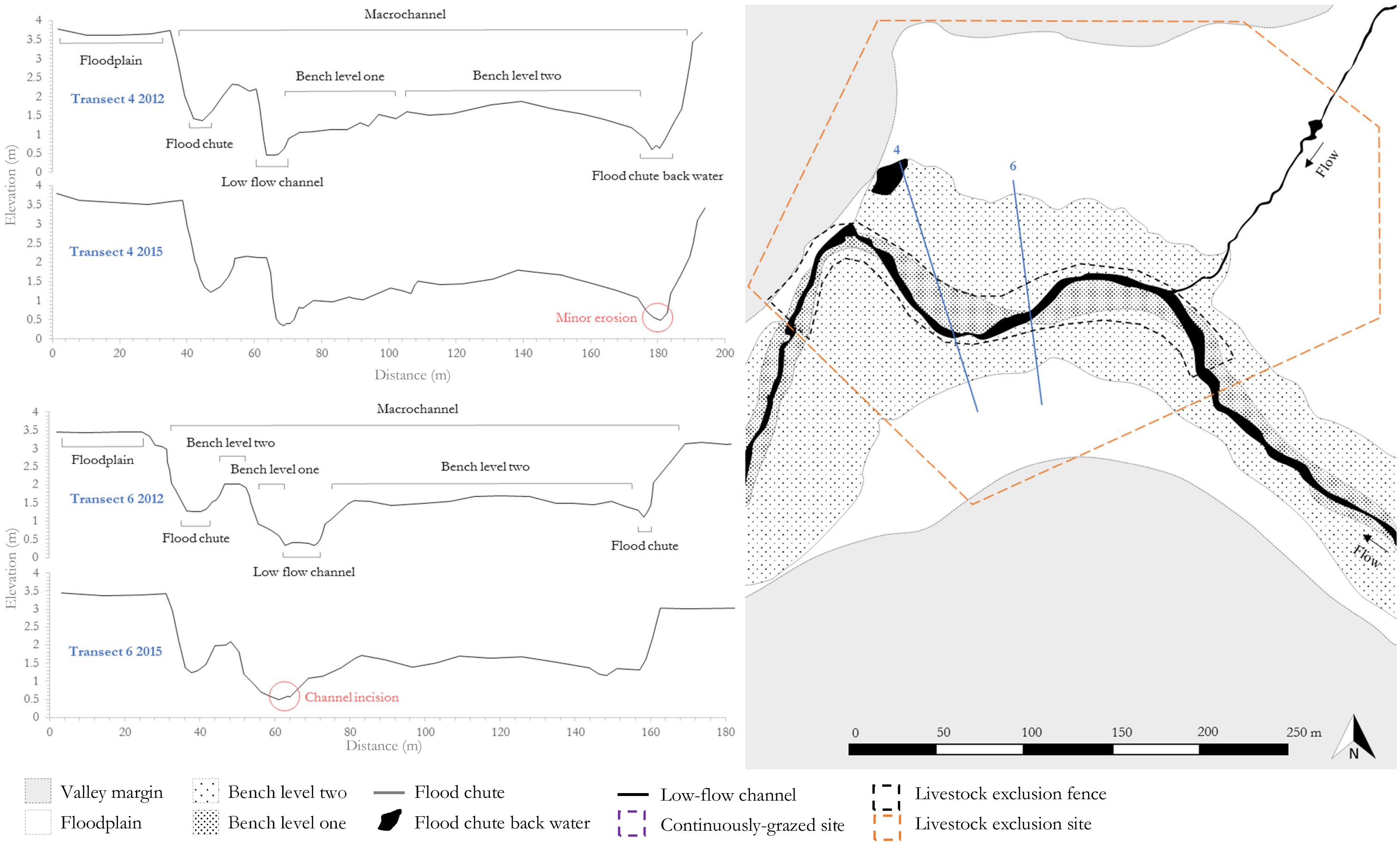


Figure 4.3. A comparison of representative transect cross-sections in 2012 and 2015 for the livestock excluded site.

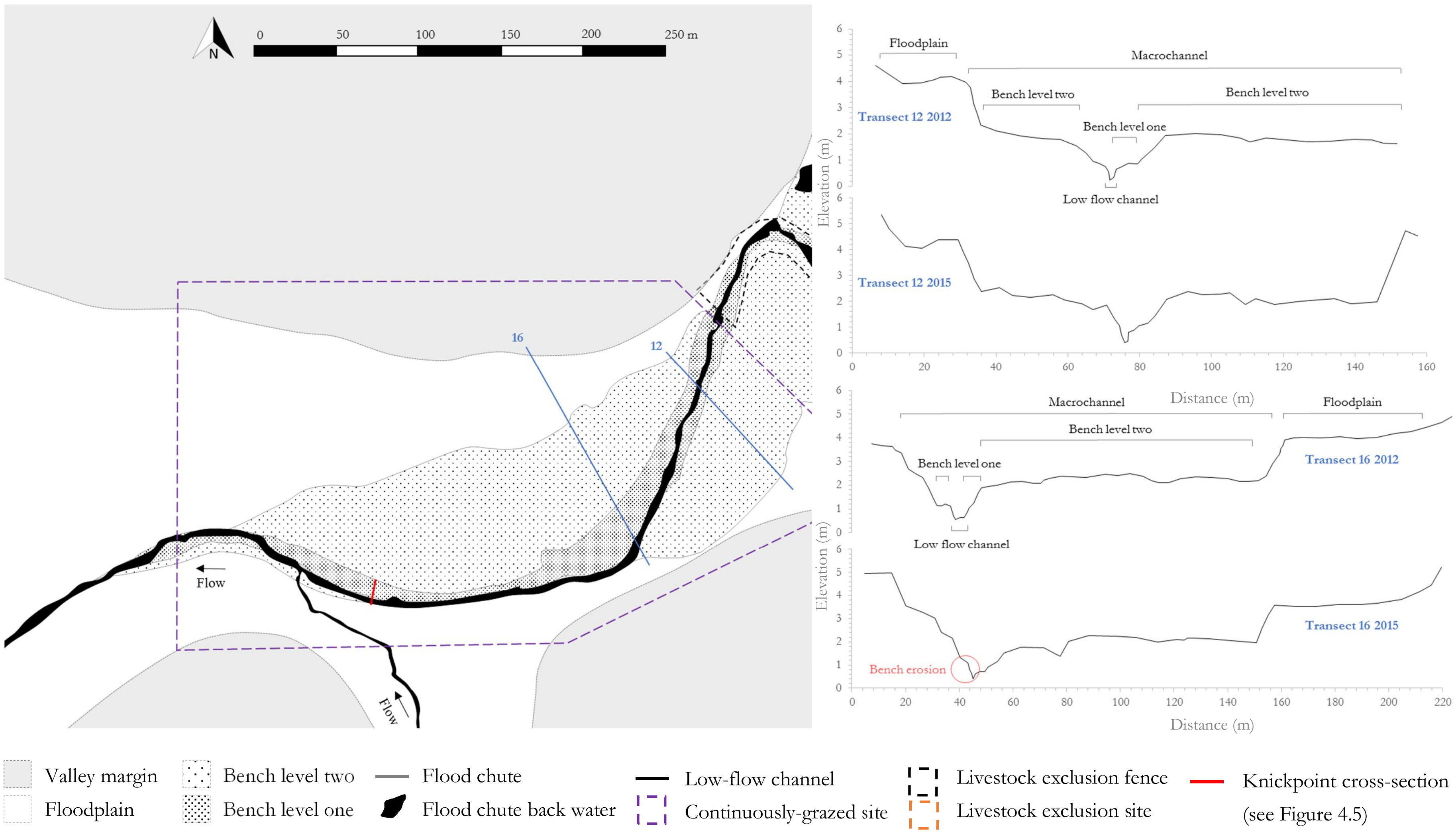


Figure 4.4. A comparison of representative transect cross-sections in 2012 and 2015 for the continuously-grazed site.

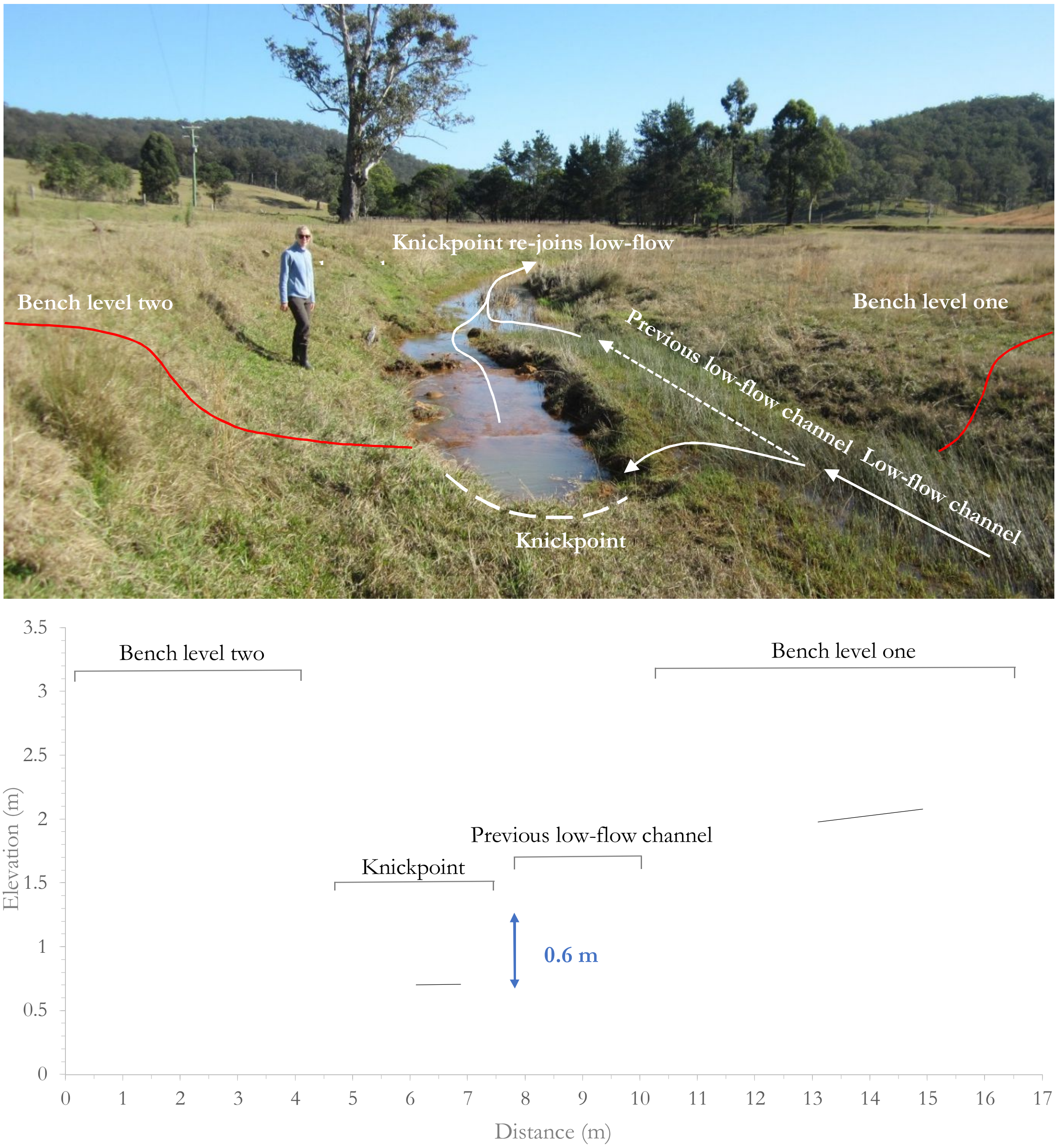


Figure 4.5. Annotated photograph and cross-sectional diagram of the Knickpoint formed in the April 2015 flood event.

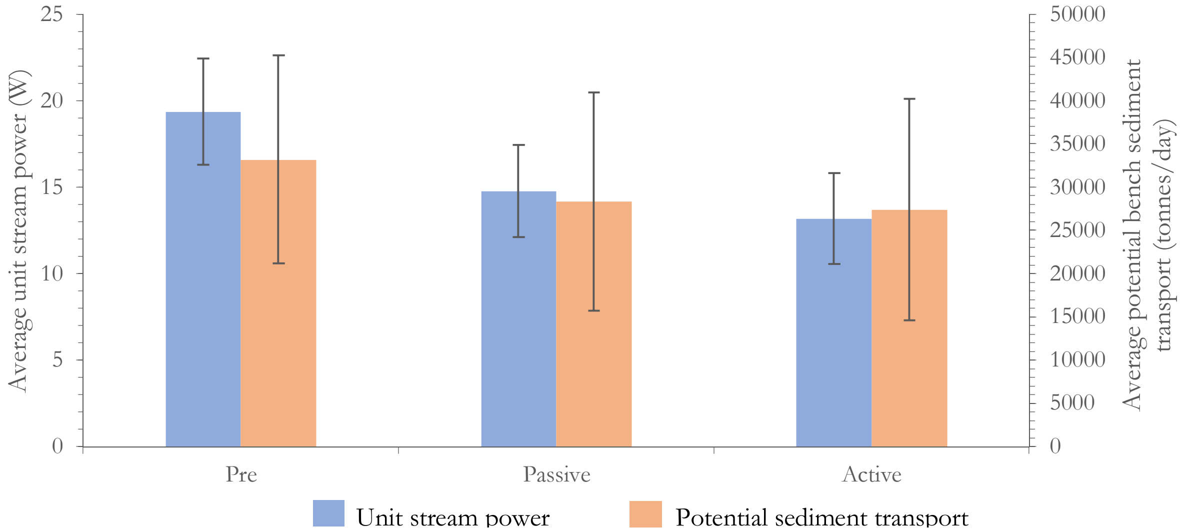


Figure 4.6. Comparison of unit stream power and potential bench sediment transport the three re-vegetation scenario simulations of pre, passive and active re-vegetation averaged across the study reach.

Table 4.3. Results of revegetation simulations on unit stream power and potential bench sediment transport for the livestock-exclusion and continuously-grazed sites.

Site	Cross-section	Revegetation scenario	Discharge (m^3/s)	Unit stream power (W)	Potential bench sediment transport (tonnes per day)
Livestock-exclusion	10	Pre	46.5	27.9	123605
		Passive	46.5	27.9	123605
		Active	46.5	27.9	123605
	8	Pre	24.9	18.1	40000
		Passive	24.9	17.7	8714
		Active	24.9	17.5	4600
	6	Pre	17.6	7.6	32600
		Passive	17.6	7.6	32600
		Active	17.6	7.6	32600
	4	Pre	4.3	1.7	3200
		Passive	4.3	1.7	920
		Active	4.3	1.7	325
	2	Pre	8.7	24.9	9250
		Passive	8.7	23.7	5644
		Active	8.7	20.3	1300
Continuously-grazed	12	Pre	30	15.7	58546
		Passive	30	15.7	58546
		Active	30	15.7	58546
	14	Pre	31.2	17.3	49792
		Passive	31.2	17.3	49792
		Active	31.2	17.3	49792
	16	Pre	7.2	35.3	7300
		Passive	7.2	3.2	2045
		Active	7.2	3.1	530
	18	Pre	4.6	20.9	3620
		Passive	4.6	14.3	778
		Active	4.6	14.2	2500
	20	Pre	4.6	24.3	4180
		Passive	4.6	18.7	875
		Active	4.6	6.5	390

Comparing vegetation composition between the livestock excluded and continuously-grazed sites

Vegetation composition data was compared between sites to identify differences associated with livestock exclusion. The livestock-exclusion site was found to have a significantly greater percentage cover of native species than the continuously-grazed site at all time periods (winter 2014: P = 0.0042, summer 2015: P = 0.0049, winter 2015: P = 0.0157), shown in Table 4.4. No significant differences between the sites were identified for the other test criteria, except for a significant increase in the percentage cover of forb species at the livestock-exclusion site in summer 2015 (P = 0.0126). Figures 4.6 and 4.7 illustrates some non-significant trends within the vegetation communities. The exotic proportion of percentage cover and species richness is generally much greater than the native proportions at both the livestock excluded and continuously-grazed sites. Only in summer 2015 is the native proportion of percentage cover greater than the exotic proportion. Native percentage cover and species richness was greatest during the summer 2015 time period, whereas exotic percentage cover and species richness was generally greater in the winter time periods.

Table 4.4. Results of Mann Whitney U tests for differences between the continuously-grazed and livestock-exclusion sites at three time periods.

Criteria	Site	N	Winter 2014 Median	Winter 2014 W	Winter 2014 P-value*	Summer 2015 Median	Summer 2015 W	Summer 2015 P-value*	Winter 2015 Median	Winter 2015 W	Winter 2015 P-value*
Native % cover	Grazed	50	0.0	2909.50	0.0042**	23.0	2932.00	0.0049**	3.0	2869.50	0.0157**
	Excluded		0.0			57.5			10.5		
Exotic % cover	Grazed	50	40.0	2503.00	0.8821	58.5	2274.50	0.084	50.5	24.57.5	0.6440
	Excluded		41.5			25.0			50.0		
Forb % cover	Grazed	50	15.5	2710.00	0.2028	10.0	2885.50	0.0126**	17.0	2699.50	0.2293
	Excluded		20.5			20.0			26.0		
Graminoid % cover	Grazed	50	20.0	2417.50	0.4579	68.0	2594.00	0.6363	35.0	2442.50	0.5711
	Excluded		20.0			73.5			30.5		
Species richness	Grazed	50	4.0	2567.00	0.7692	4.0	2413.00	0.4371	5.0	2767.00	0.0911
	Excluded		4.0			4.0			6.0		
Native species richness	Grazed	50	0.0	2808.00	0.0043**	2.0	2651.50	0.3588	1.0	2986.50	0.0006**
	Excluded		0.0			2.0			1.0		
Exotic species richness	Grazed	50	4.0	2434.50	0.5234	4.0	2316.00	0.1437	4.0	2549.50	0.8669
	Excluded		4.0			3.0			5.0		

*P-value is adjusted for ties.
**P-values < 0.05 are significant.

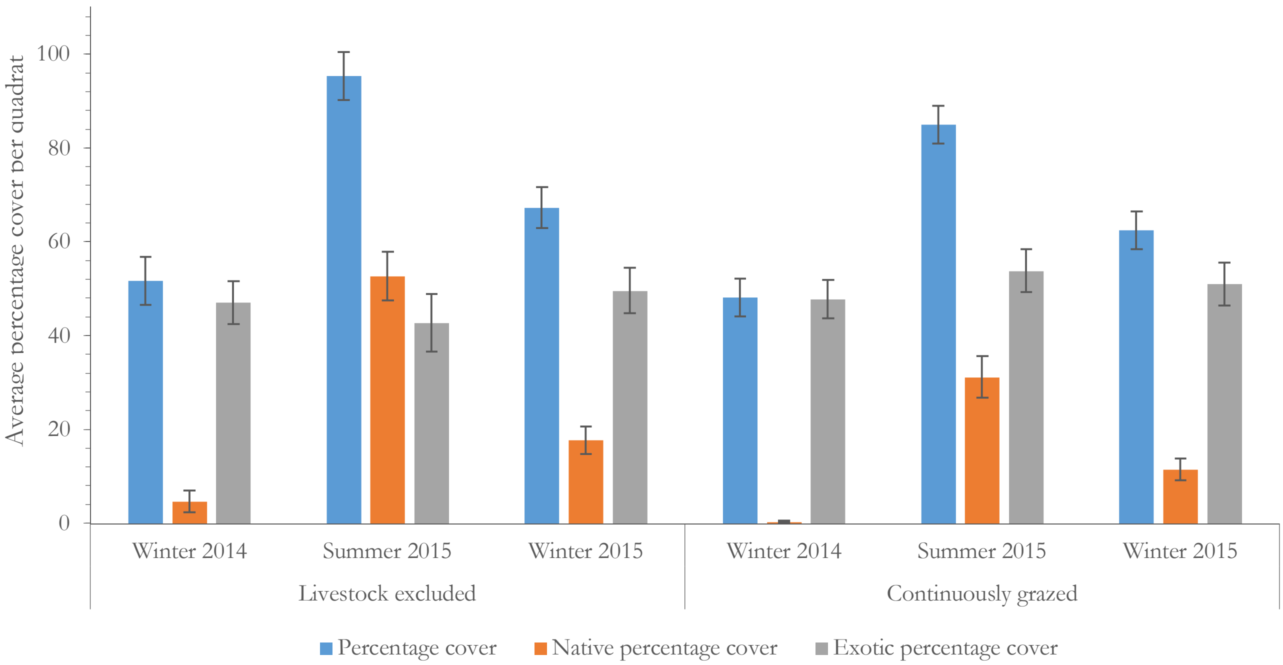


Figure 4.6. Comparisons of mean average percentage cover per quadrat between the livestock-exclusion and continuously-grazed sites for three time slices. Error bars show standard error.

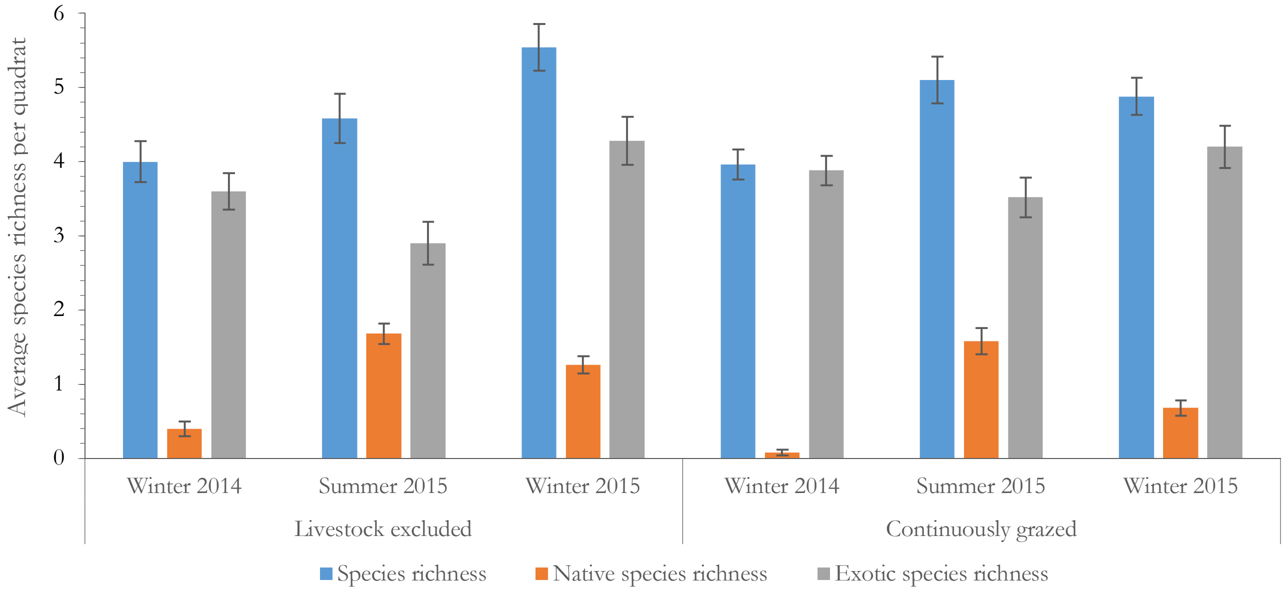


Figure 4.7. Comparisons of mean average species richness per quadrat between the livestock-exclusion and continuously-grazed sites for three time slices. Error bars show standard error.

A breakdown of the vegetation community data into growth form groups found that the pattern of changes in percentage cover over time for each group is very similar across both sites. Appendix 3 shows that grasses were the most dominant growth form, followed by herb species, at both sites across all time periods. The other growth form groups fluctuate in percentage cover across the time periods but contribute only small amounts to the total percentage cover. There is a distinct lack of woody species in the data. The *Acacia parvipinnula* and *Melaleuca linariifolia* species shown in Figure 4.2 are the only woody species regenerating within the riparian zone and have not been captured in the data due to their patchy distribution. These shrub or small tree species have only regenerated on benches at the livestock-exclusion site. The species richness comparisons shown in Appendix 4 demonstrate exotic species to be more dominant for each growth form group except for the sedge and fern groups. Comparisons of native and exotic species show that native species richness is greater at the livestock-exclusion site, for each group except rush. The groundcover and herb species richness was found to be greater in the winter than the summer time periods. Fern species richness is greater for the native species in the summer time period. For the grass species, native species are more numerous in the summer period with the inverse true for exotic species.

Table 4.5 lists the species identified in the macrochannel surveys which are representative of the surrounding extant vegetation communities at the site, according to (Hunter-Central Rivers CMA, 2011). These are the Wollombi Redgum-River Oak Woodland, Sheltered Blue Gum Forest communities, Coastal Ranges Open Forest, Hunter Range Grey Gum Forest and Sheltered Rough Barked Apple Forest communities (LHCCREMS, 2000). Only five species were identified as representative of these extant communities from a total of fourty-six.

Table 4.5. Species identified at the livestock exclusion site in the macrochannel surveys that are representative of the extant vegetation communities, surrounding the livestock exclusion fences at the study reach and within the intact upstream riparian zone.

	Wollombi Redgum-River Oak Woodland	Sheltered Blue Gum Forest	Coastal Ranges Open Forest	Hunter Range Grey Gum Forest	Sheltered Rough Barked Apple Forest
Entolasia marginata		Yes			Yes
Microlaena stipoides	Yes		Yes	Yes	
Pteridium esculentum	Yes	Yes	Yes		Yes
Melaleuca linariifolia	Yes				
Acacia parvipinnula		Yes		Yes	

Vegetation community profiles defined by (LHCCREMS, 2000)

PERMANOVA tests found significant differences in vegetation composition at the species level between the livestock-exclusion and continuously-grazed sites at all time periods (winter 2014: P = 0.0001, summer 2015: P = 0.0001, winter 2015: P = 0.0001), shown in Table 4.6. The vegetation composition of bench levels one and two were also found to be significantly different for each time period (winter 2014: P = 0.0001, summer 2015: P = 0.0001, winter 2015: P = 0.0001), with no significant interaction found between site and bench level (winter 2014: P = 0.9841, summer 2015: P = 9998, winter 2015: P = 0.2293). Table 4.7 lists the species identified by SIMPER analysis that as a group contribute to at least 50 percent of the difference between the vegetation communities of the livestock-exclusion and continuously-grazed sites. For winter 2014 all species are classed as exotic, with all but *Pennisetum clandestinum* having greater mean average percentage cover at the livestock-exclusion site. At summer 2015 there were two native and two exotic species. *P. clandestium* was found to be have greater average percentage cover at the livestock-exclusion site. Both native species for summer 2015 had greater average percentage cover at the livestock-exclusion site, while the exotic grass *Paspalum dilitatum* was greater at the continuously-grazed site. For winter 2015 all species had greater average percentage cover at the livestock-exclusion site apart from *P. dilitatum*. This time period was again dominated by exotic species.

Table 4.6. Results of two-way PERMANOVA tests for differences in vegetation communities between the livestock excluded and continuously-grazed sites and also between bench level one and two.

Time period	Comparison	Sum of squares	df	Mean square	F-value	P-value
Winter 2014	Site	1.7387	1	1.7387	5.9905	0.0001**
	Bench level	2.2828	1	2.2828	7.865	0.0001**
	Interaction	-5.2767	1	-5.2767	-18.18	0.9841
Summer 2015	Site	2.022	1	2.022	6.1357	0.0001**
	Bench level	2.8754	1	2.8754	8.7255	0.0001**
	Interaction	-6.1262	1	-6.1262	-18.59	0.9998
Winter 2015	Site	1.4252	1	1.4252	4.622	0.0001**
	Bench level	1.4587	1	1.4587	4.7308	0.0001**
	Interaction	-4.8665	1	-4.8665	-15.782	0.2293

**P-values < 0.05 are significant.

Table 4.7. Results of SIMPER analysis identifying species that collectively were found to cause over 50 percent of the difference between the livestock excluded and continuously-grazed site at the three time periods.

Time	Species	Growth form	Origin	Average dissimilarity	Contributing percent	Average percentage cover	
						Excluded	Grazed
Winter 2014	*Andropogon virginicus*	Grass	Exotic	15.03	20.48	11	11.7
	Pennisetum clandestinum	Grass	Exotic	13	17.72	11.3	9.9
	Hypochoeris radicata	Herb	Exotic	7.416	10.11	7.93	6.5
	Conyza spp.	Herb	Exotic	5.791	7.893	4.64	4.34
Summer 2015	*Isachne globosa*	Grass	Native	15.82	20.11	24.8	15.9
	Paspalum dilatatum	Grass	Exotic	11.04	14.03	7.73	19.8
	Pennisetum clandestinum	Grass	Exotic	9.518	12.1	14.1	10.3
	Pteridium esculentum	Fern	Native	5.966	7.584	9.89	4.42
Winter 2015	*Paspalum dilatatum*	Grass	Exotic	13.22	17.82	12.6	16.7
	Carex appressa	Sedge	Native	8.213	11.07	9.36	4.1
	Pennisetum clandestinum	Grass	Exotic	8.199	11.05	8.07	7.6
	Hypochoeris radicata	Herb	Exotic	5.125	6.908	7.07	6.56
	Romulea rosea	Herb	Exotic	4.897	6.6	5.7	3.72

Table 4.8 shows that the vegetation composition is significantly different between bench levels one and two at both the livestock-exclusion (winter 2014: $P = 0.0109$, summer 2015: $P = 0.0012$, winter 2015: $P = 0.0035$) and continuously-grazed sites at each time period (winter 2014: $P = 0.0001$, summer 2015: $P = 0.0002$, winter 2015: $P = 0.0001$). Figure 4.8 demonstrates that the difference between the communities is greater at the continuously-grazed site, with the groupings of the bench level vegetation data points being much tighter at the livestock-exclusion site for each time period. In fact the majority of data points for bench level two can be found within that of bench level one for the livestock-exclusion site. This is compared to the continuously-grazed site which is much more distinct. SIMPER analysis of the vegetation composition between bench levels one and two show that only a few species are responsible for driving the differences. Table 4.9 shows that the species identified as important experienced varying responses to livestock exclusion with *Andropogon virginicus* and *Paspalum dilatatum* having greater abundance at bench level two and *Carex appressa* and *Isachne globosa* having greater abundance at bench level one. *Pennisetum clandestinum* and *Hypochoeris radicata* were more abundant at different bench levels in winter 2014 and 2015.

Table 4.8. Results of one-way PERMANOVA tests for differences in vegetation composition between bench levels one and two for the three time periods.

Time period	Site	Total sum of squares	Within-group sum of squares	F-value	P-value
Winter 2014	Livestock excluded	13.15	12.37	2.651	0.0109**
	Continuously grazed	15.01	11.85	12.83	0.0001**
Summer 2015	Livestock excluded	14.41	13.09	4.254	0.0012**
	Continuously grazed	15.15	13.47	5.978	0.0002**
Winter 2015	Livestock excluded	12.28	11.45	3.041	0.0035**
	Continuously grazed	16.28	14.65	5.323	0.0001**

**P-values < 0.05 are significant.

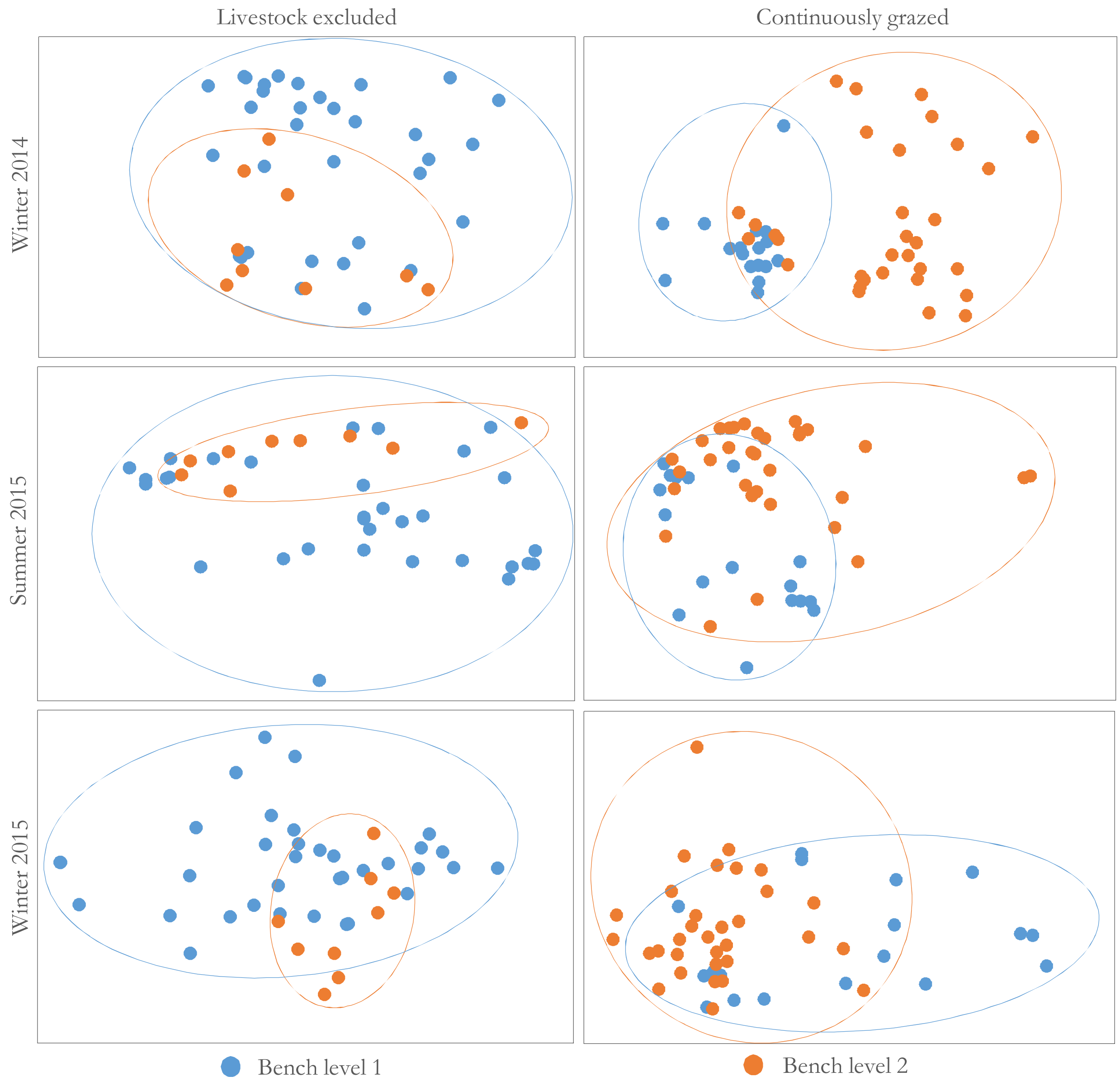


Figure 4.8. MDS plots using Bray-Curtis similarity, comparing the vegetation communities of bench level one and two at the livestock excluded and continuously grazed sites.

Table 4.9. Results of SIMPER analysis identifying species that collectively were found to cause over 50 percent of the difference between bench levels one and two at the three time periods.

Time	Species	Average dissimilarity	Contributing percent	Average percentage cover	
				Bench level 1	Bench level 2
Winter 2014	*Andropogon virginicus*	17.28	22	6.49	15.2
	Pennisetum clandestinum	16.12	20	16.1	6.23
	Hypochoeris radicata	8.813	11	9.54	5.34
Summer 2015	*Isachne globosa*	16.77	20	29.3	8.07
	Paspalum dilatatum	14.08	17	5.85	24.9
	Pennisetum clandestinum	10.94	13	9.21	15.9
Winter 2015	*Paspalum dilatatum*	14.46	19	13.8	16
	Pennisetum clandestinum	10.49	13	2.45	14.8
	Carex appressa	8.275	11	8.43	4.15
	Hypochoeris radicata	5.653	7	5.81	8.07

**P-values < 0.05 are significant.

2. Environmental flows

Geomorphic site description

The eight kilometer study reach along the Hunter River is shown in Figure 4.9. The study reach is a passive meandering, low to moderate sinuosity, gravel bed river (Hoyle et al., 2007). The reach is characterised by the occurrence of intermittent bench and bar units of considerable but variable size, inset within an extensive macrochannel. The macrochannel ranges in width between 75 to 600 m (Hoyle et al., 2008). The bench units are elevated approximately 3 m above the low-flow channel and are composed of poorly sorted non-cohesive sands and gravels (Hoyle et al., 2012). These units are inundated very infrequently (Hoyle et al., 2012). The bench units are densly vegetated with flooding, groundwater access and substrate being the localised geomorphic controls on vegetation distibution (Hoyle et al., 2012). The roughness coefficients, following Arcement and Schneider (1989), assigned to the vegetation of the bench units range between 0.045-0.001 (Hoyle et al., 2012).

Bench unit inundation durations

The bench units at the study reach are inundated by the 1:5 flood average return interval (hereafter ARI), with surface sediment mobilised by the 1:10 ARI and bench sediment fully mobile in the 1:50 ARI (Hoyle et al., 2007, Hoyle et al., 2012). Figure 4.10 illustrates the flood heights associated with each ARI up to 1:20, demonstrating that the 1:5 ARI is the lowest flood magnitude able to inundate bench units. Table 4.10 shows the discharge of each ARI able to inundate the bench units along the study reach and the associated bench inundation durations identified from the historic flow record from the Muswellbrook gauge (gauge number: 210002). The minimum duration was one day, maximum three days and the average two days. These durations determined the 12, 24, 48 and 72 hour inundation treatments in the seedling emergence experiment. Only daily data is available from the Muswellbrook gauge in the PINEENA CM database, therefore a twelve hour treatment was included to represent flows that reached the 1:5 ARI magnitude but lasted less than one day.

Table 4.10. ARI's able to inundate bench units at the study reach and their corresponding durations.

ARI	Discharge (m^3/s)	Flood duration at or above return interval (days)		
		Minimum	Average	Maximum
1:5	999	1	2	2
1:10	1658	2	2	3
1:20	2025	2	2	2
1:25	2760	2	2	2

Historic flood durations were derived from the PINEENA water database managed by the NSW office of Water.

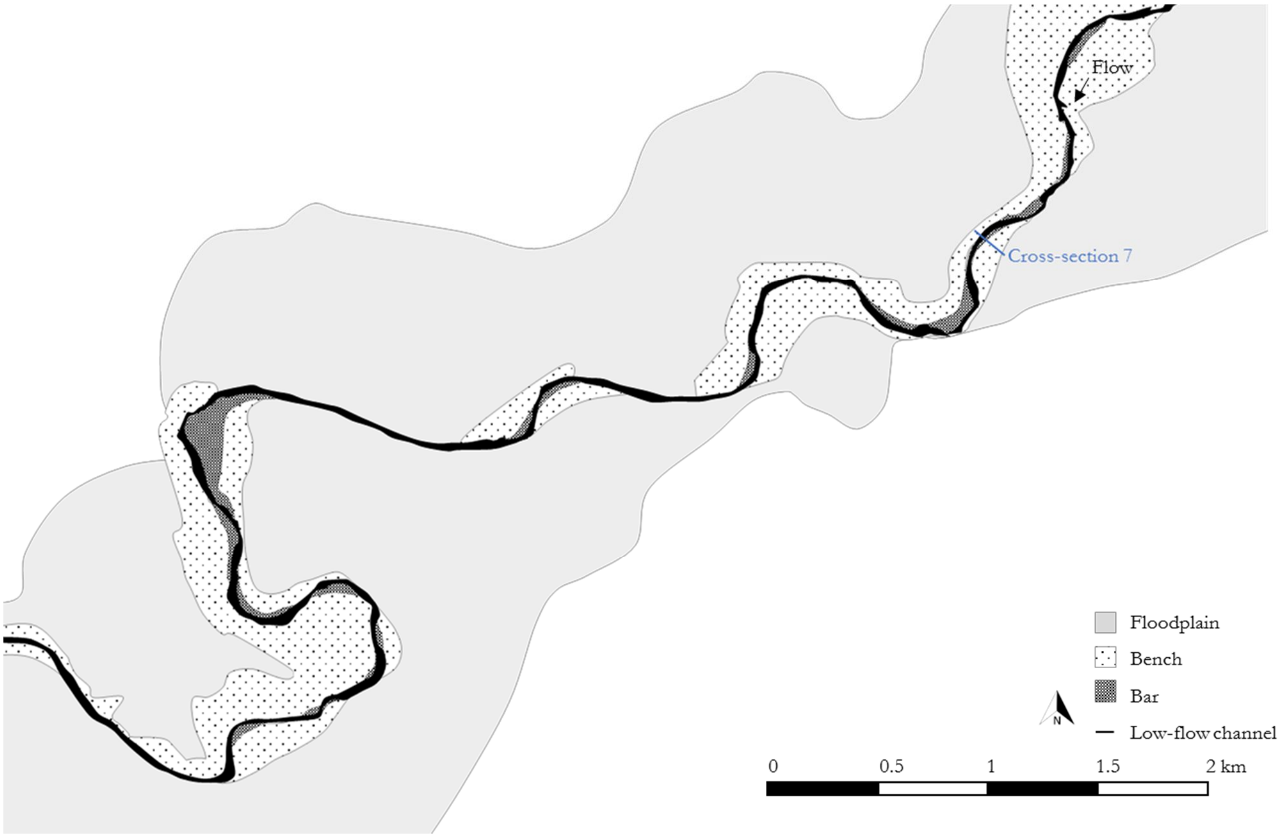


Figure 4.9. Geomorphic map of the Upper Hunter study reach showing geomorphic units and the location of cross-section 7, see Figure 4.10.

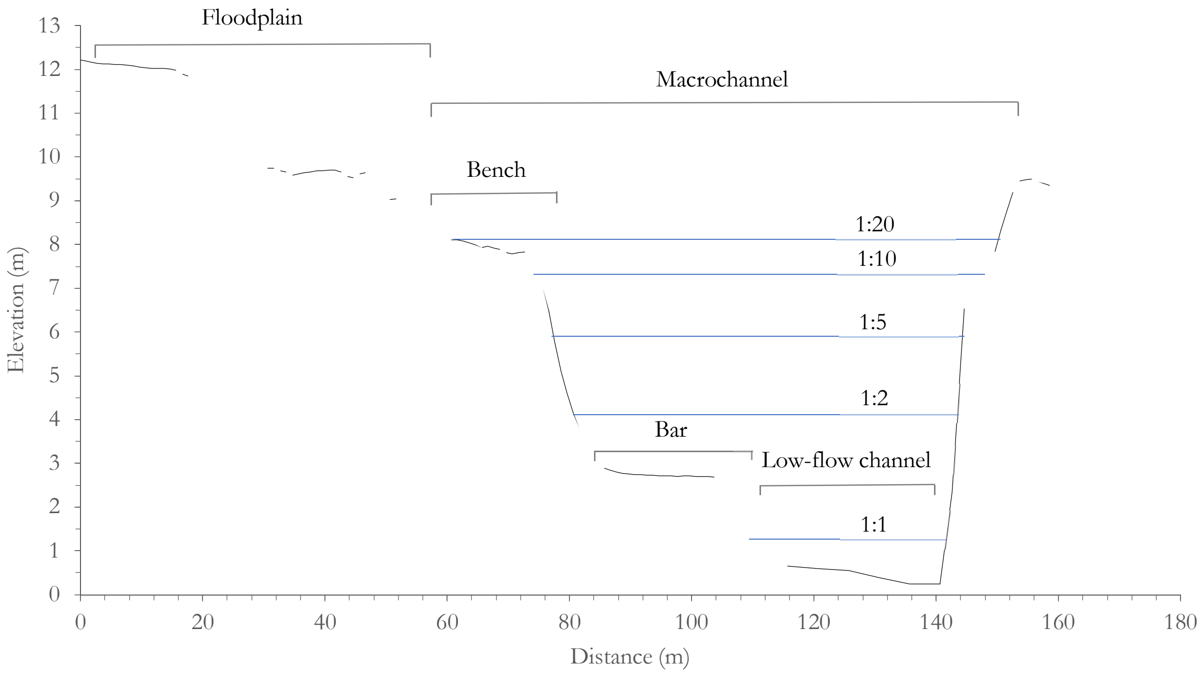


Figure 4.10. An illustration of the flood heights under the modelled ARI's, using cross–section 7. The 1:25 ARI overtops the banks of the macrochannel at this cross-section and is therefore not shown.

An environmental water allocation of 20,000 ML is reseved for the Upper Hunter catchment between Glenbawn and Glennies Creek dams (NSW Department of Water and Energy, 2009). This allocation has not been used in the period between 2004 to 2015 (NSW Office of Water, 2013). The only environmental water release occurred over three days during September 2001, releasing 5,000 ML per day, totalling 15,000 ML of water released (Hancock and Boulton, 2005). Since the allocation has not been utilised, an environmental water release could potentially be used to boost a natural flow up to the magnitude of the 1:5 ARI, in order to provide water to the infrequently inundated bench units. Table 4.11 displays the increases in discharge that could be achieved by releasing environmental water concurrent with natural flows. The table shows that only by releasing the entire environmental water allocation is it possible to boost a natural flow of 1:4 magnitude up to that of the 1:5 ARI. The release of 5,000 ML per day recorded by (Hancock and Boulton, 2005) would not be effective in providing environmental water to bench units, only raising the water level by approximately 38 cm at Aberdeen, which is approximately 25 km upstream of the study reach. In order to boost a 1:2 or 1:3 magnitude flow to that of a 1:5 ARI, which is the most likely scenario, a release of more than double the currently reserved environmental water allocation would be required. The current environmental water allocation is therefore insufficient for the purposes of providing environmental water to bench units during small flood events. However the current allocation may be of use to extend flood durations above that of the 1:5 ARI which naturally inundate bench units.

<u>Effects of simulated flood durations on seedling emergence</u>

The results of one way ANOVA tests of the effectiveness of inundation treatments on seed species is shown in Table 4.12. Only *Acacia parvipinnula* ($P < 0.001$) and *Bothriochloa macra* ($P = 0.003$) showed a significant difference in seedling emergence among the treatments. Figure 4.11 shows no pattern of preference within growth form groups and shows varibility in emergence success among treatments for many of the species. There was no pattern of treatment preference for *Eucalyptus camaldulensis*. There is a general reduction in percentage seedling emergence with increasing inundation duration for *Casaurina cunninghamiana,* except for the 48 hour treatment which was the most successful. Seedling emergence for *A. parvipinnula* was greatest under the 72 hour inundation treatment, which was significantly different to the other treatments. Generally percentage seedling emergence increased with inundation duration for *Acacia parvipinnula*. *Acacia salicina* was most successful under the 12 hour treatment but seedling emergence then declined with increasing inundation duration. There was no pattern of preference for *Lomandra longifolia* with the 0 and 72 hour treatments being most successful by a small margin.

Table 4.11. Possible increases to discharge using environmental water releases.

ARI	Discharge m^3/s	Discharge ML/day	Environmental water release (ML/day)	Total discharge (m^3/s) (ARI + environmental water release)
1:1	11	950	5000	69
			10000	127
			15000	185
			20000	242
1:2	350	30218	5000	408
			10000	466
			15000	524
			20000	581
1:3	566	48916	5000	624
			10000	682
			15000	740
			20000	798
1:4	783	783	5000	840
			10000	898
			15000	956
			20000	1014
1:5	999	86312	5000	1057
			10000	1115
			15000	1173
			20000	1230
1:10	1658	143214	5000	1716
			10000	1774
			15000	1832
			20000	1889

Historic flood durations were derived from the PINEENA water database managed by the NSW office of Water.

Both grass species *Microlaena stipoides* and *Bothriochloa macra* showed generally decreasing percentage seedling emergence with increasing inundation duration. Seedling emergence was significantly greater for *B. macra* under the 0 hour control treatment when compared to the 48 and 72 hour treatments. *Carex appressa* had the most successful seedling emergence overall, with a trend of decreasing percentage seedling emergence with increasing seedling emergence time. *Schoenoplectus validus* had generally greater seedling emergence with increasing inundation durations while *Bolboschoenus caldwelli* favoured the 12 hour duration with the other treatments being approximately equal.

Table 4.12. Results of ANOVA tests for differences between inundation treatments for each species.

Treatment	0 hours			12 hours			24 hours			48 hours			72 hours					
Species	n	Mean	St dev.	n	Mean	St dev.	n	Mean	St dev.	n	Mean	St dev.	n	Mean	St dev.	df	F	P-value
E. camaldulensis	10	0.7	0.675	10	0.3	0.483	10	0.5	0.707	10	0.4	0.516	10	0.4	0.876	4	2.25	0.079
C. cunninghamiana	10	3.1	1.449	10	3.0	1.414	10	2.8	1.549	10	4.0	0.816	10	2.5	1.179	4	1.85	0.135
A. parvipinnula	10	1.3	1.494	10	1.3	1.567	10	1.0	0.667	10	2.3	1.160	10	4.1	1.792	4	8.36	0.000**
A. salicina	10	1.4	2.221	10	1.9	1.663	10	0.9	0.738	0.9	0.9	0.994	10	0.6	0.843	4	0.28	0.276
L. longifolia	10	4.2	1.751	10	3.3	1.829	10	3.3	1.252	10	2.9	1.101	10	4.9	1.912	4	2.58	0.050
M. stipoides	10	2.7	1.337	10	1.9	1.524	10	1.3	0.823	10	1.3	1.636	10	1.2	1.135	4	2.30	0.074
B. macra	10	2.8	0.919	10	2.1	1.370	10	1.5	0.972	10	1.0	0.816	10	1.3	1.059	4	4.70	0.003**
C. appressa	10	8.2	1.229	10	8.0	2.944	10	8.2	1.619	10	8.0	1.633	10	7.5	1.509	4	0.23	0.919
S. validus	10	1.2	1.135	10	1.6	1.266	10	1.4	0.966	10	2.5	1.716	10	1.7	1.252	4	1.48	0.224
B. caldwellii	10	0.3	0.675	10	1.2	2.251	10	0.2	0.422	10	0.2	0.422	10	0.3	0.675	4	1.44	0.235

**P-values < 0.05 are significant.

Table 4.13. Results of independent T-tests comparing differences between the most successful and least successful and between the most successful and control treatments per species.

		Most successful			Least successful					Control (0 hours)			
Species	n	Treatment	Mean	SE mean	Treatment	Mean	SE mean	T-value	P-value	Mean	SE mean	T-value	P-value
E. camaldulensis	10	72 hours	1.1	0.28	12 hours	0.3	0.15	2.53	0.024**	0.7	0.21	1.14	0.269
C. cunninghamiana	10	48 hours	4.0	0.26	72 hours	2.5	0.37	3.31	0.004**	3.1	0.46	1.71	0.109
A. parvipinnula	10	72 hours	4.1	0.57	24 hours	1.0	0.21	5.13	0.000**	3.1	0.46	1.37	0.188
A. salicina	10	12 hours	1.9	0.53	72 hours	0.6	0.27	2.20	0.046**	1.4	0.70	0.57	0.577
L. longifolia	10	72 hours	4.9	0.6	48 hours	2.9	0.35	2.9	0.012**	4.2	0.55	0.85	0.405
M. stipoides	10	0 hours	2.7	0.42	72 hours	1.2	0.36	2.70	0.015**				
B. macra	10	0 hours	2.8	0.29	48 hours	1.0	0.26	4.63	0.000**				
C. appressa	10	24 hours	8.2	0.51	72 hours	7.5	0.48	1.00	0.331	8.2	0.39	0.00	1.000
S. validus	10	48 hours	2.5	0.54	0 hours	1.2	0.36	2.00	0.064				
B. caldwellii	10	12 hours	1.2	0.71	24 hours	0.2	0.422	1.38	0.201	0.3	0.21	1.21	0.254

**P-values < 0.05 are significant.
Treatment success is based on the final number of emerged seedlings after 70 days.

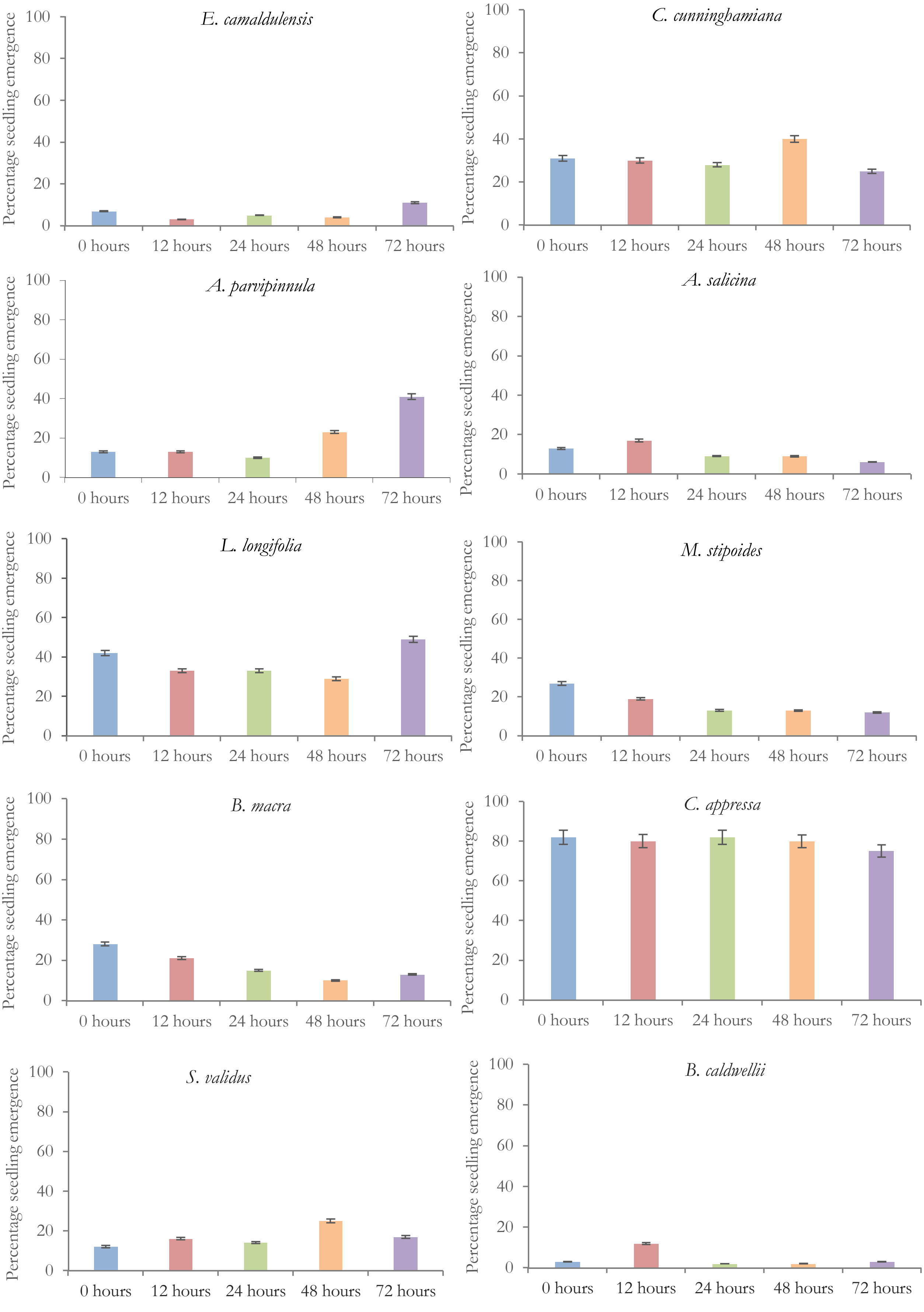


Figure 4.11. Total percentage seedling emergence per species per treatment after 70 days. Error bars show standard error.

The most successful treatment was compared to the least successful treatment and the control treatment using a series of independent t-tests. Table 4.13 shows that the seedling emergence for the most successful treatment was not significantly greater than that of the control treatment for any species. The most successful treatment did significantly increase seedling emergence when compared to the least successful treatment for all species except *C. appressa, S. validus* and *B. caldwellii*. The tree species of *E. camaldulensis* and *C. cunninghamiana* had significantly greater seedling emergence in the long inundation treatments of 72 and 48 hours than the 12 and 72 hour treatments respectively. *A. parvipinnula* had significantly greater seedling emergence under the 72 hour treatment when compared to the 24 hour treatment. *A. salicina* had the significantly greater seedling emergence under the 12 hour treatment when compared to the 72 hour treatment. Seedling emergence for *L. longifolia* was significantly greater under the 72 hour treatment than the 48 hour treatment. For both grass species, the control treatment of non-inundation was significantly more successful than the long inundation treatments of 48 and 72 hours. The sedge *C. appressa* was most successful under the 0 and 24 hour treatments, which was not significantly greater than the least successful 72 hour treament. The rush species *S. validus* and *B. caldwellii* had the greatest seedling emergence for 48 and 12 hours, both not significantly more successful than the 0 and 24 hour treatments which were the least successful for these two species respectively.

The comparisons of most successful and least successful treatments identified some unsusual results. Unusually *L. longifolia* was most successful under the 72 hour treatment and least successful under the 48 hour treatment. The variation between all treatments for this species is small at approximately 20%, with less than half of the seeds planted emerging for all treatments. Additionally seedling emergence did not appear to have fully ceased when the experiment was terminated. These facts combined with a lack of pattern in response to inundation duration may point to an inadequate number of emerged seedlings to draw any meaninful conclusions for this species. The *Acacia* species had varying preferences. *A. parvipinnua* showed a preference for increasing inundation duration while *A. salicina* had the greatest emergence under the shortest inundation treatment. This difference may reflect the differing strengths of the seed coat of these species, a pilot study found that *A. parvipinnula* seeds would not germinate without pre-treatment while *A. salicina* would do so with a reduction in germination success. It is suggested that twelve hours of inundation was sufficient to break down the seed coat of *A. salicina*, while the longer durations left the seeds susceptible to rotting. For *A. parvipinnua,* which has a much more resilient seed coating, it is suggested that the longer inundation treatments were more effective in breaking down the seed coat after providing water to initiate the process of germination. It was expected that *B. caldwellii* would respond more successfully to inundation treatments since this species can establish within instream habitat. The germination of this species is light dependent and if buried generally become part of the seed bank (Halton, 2009). The sensitivity to light may explain the low emergence values since the seeds were buried in order represent an in-situ seed bank. The relative success of the twelve hour treatment may be due to this duration being able to most successfully break down the seed coat, while the longer durations may have prevented too much light from reaching the embryo. It is unlikely that the long durations induced mortaility since the seed of this species is adapted to dispersal by hydrochory and may float for over one hunderd days (Blanch et al., 1999, Halton, 2009). The specific mechanisms behind the varying degrees of success for this species remain unclear.

Comparisons of seedling emergence timing

The results of the Kaplan-Meier survival analysis, comparing the differences in seedling emergence timing among inundation treatments are shown in Table 4.12, with all except *E. camaldulensis, C. cunninghamiana* and *S. validus* showing significant differences in the timing of seedling emergence. Kaplan-Meier curves for percentage seedling emergence over the 70 day study period are shown in Figure 4.12. The difference among treatment distributions for A. *parvipinnula* is that the long treatments of 72 and 48 hours experienced a steeper initial increase and much greater emergence overall. For *A. salicina* the 12 hour treatment is distinct from the others due to its late initial emergence and steady increase rather than experiencing long periods without emergence as occurred under the other treatments. The 72 hour treatment for *L. longifolia* was different to the other treatments in that it began early compared to the 0, 24 and 48 hour treatments and experienced much higher emergence. Each treatment experienced a plateau in emergence around days fourty-six to fifty-six. For *M. stipoides* the 12 and 0 hour treatments were distinct from the others due to their greater emergence. The shape of the treatment curves for *B. macra* are similar except the 72 hour treatment which has a shallower rise than the other treatments and experienced a late increase in emergence. Only the Breslow test found a significant difference in the seedling emergence timing among inundation treatments for *C. appressa*. This difference is due to the emergence start times with the 48 hour treatment beginning far earlier than the other treatments. The 12 hour treatment for *B. caldwellii* was distinct in it's greater emergence values and curved shape, compared to the other treatments which experience very shallow increases with long periods without emergence. The boxplots in Figure 4.13 illustrate the time to first emergence and range of emergence times under different inundation treatments. For the majority of species the variation in the weighted mean seedling emergence time shows little variation while the range of germination times may be more variable. The time to first emergence did not vary substantially for *E. camaldulensis, C. cunninghamiana, A. parvipinnula, M. stipoides* and *B. macra*. Although the seventy-hour treatment for *M. stipoides* and *B. macra* produced poor emergence results, it was the treatment with the first day of emergence for both species. For *B. macra* there is also a general increase in the range of emergence times and later weighted mean emergence time with increasing inundation duration. The range and weighted mean emergence times for *A. salicina* were highly variable, with the twelve and fourty-eight hour treatments having much smaller ranges. The twelve hour treatment for *L. longifolia* is distinct from the other treatments which experienced a general increase in range with inundation duration. There appears to be a decrease in the time to first emergence with increasing inundation duration for *B. caldwellii*, however the low emergence values do not allow for reliable interpretation.

Table 4.13. Kaplan-meier survival analysis test results comparing seedling emergence timing under inundation treatments.

Species	Test	Chi-Square	df	P-value
E. camaldulensis	Log Rank	7.239	4	0.124
	Breslow	7.302	4	0.121
	Tarone-Ware	7.271	4	0.122
C. cunninghamiana	Log Rank	4.935	4	0.294
	Breslow	5.102	4	0.277
	Tarone-Ware	5.039	4	0.283
A. parvipinnula	Log Rank	42.605	4	0.000*
	Breslow	44.87	4	0.000*
	Tarone-Ware	43.782	4	0.000*
A. salicina	Log Rank	11	4	0.027*
	Breslow	10.247	4	0.036*
	Tarone-Ware	10.623	4	0.031*
L. longifolia	Log Rank	13.095	4	0.011*
	Breslow	15.324	4	0.004*
	Tarone-Ware	14.198	4	0.007*
M. stipoides	Log Rank	11.83	4	0.019*
	Breslow	11.776	4	0.019*
	Tarone-Ware	11.814	4	0.019*
B. macra	Log Rank	12.68	4	0.013*
	Breslow	13.082	4	0.011*
	Tarone-Ware	12.896	4	0.012*
C. appressa	Log Rank	4.557	4	0.336
	Breslow	10.308	4	0.036*
	Tarone-Ware	6.39	4	0.172
S. validus	Log Rank	7.962	4	0.093
	Breslow	8.61	4	0.072
	Tarone-Ware	8.284	4	0.082
B. caldwellii	Log Rank	17.791	4	0.001*
	Breslow	17.735	4	0.001*
	Tarone-Ware	17.764	4	0.001*

3. Summary

A summary of the key findings from both studies are provided in table 4.14. The findings are divided into three sections which will form the framework for the discussion chapter under the headings; 1. Passive revegetation success, 2. Management considerations and 3. The utility of the seed bank.

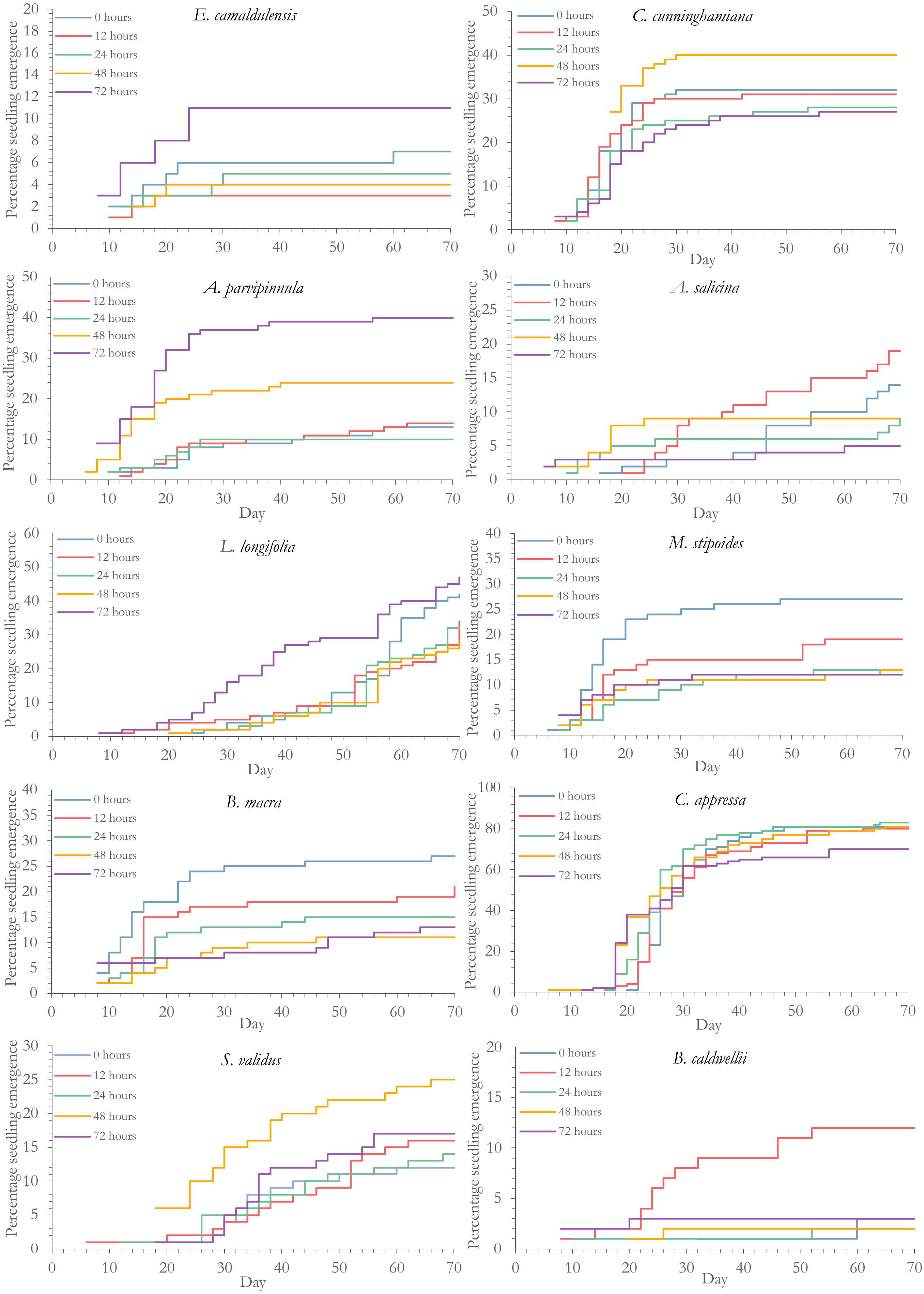


Figure 4.12. Survival curves showing percentage seedling emergence over the 70 day study period.

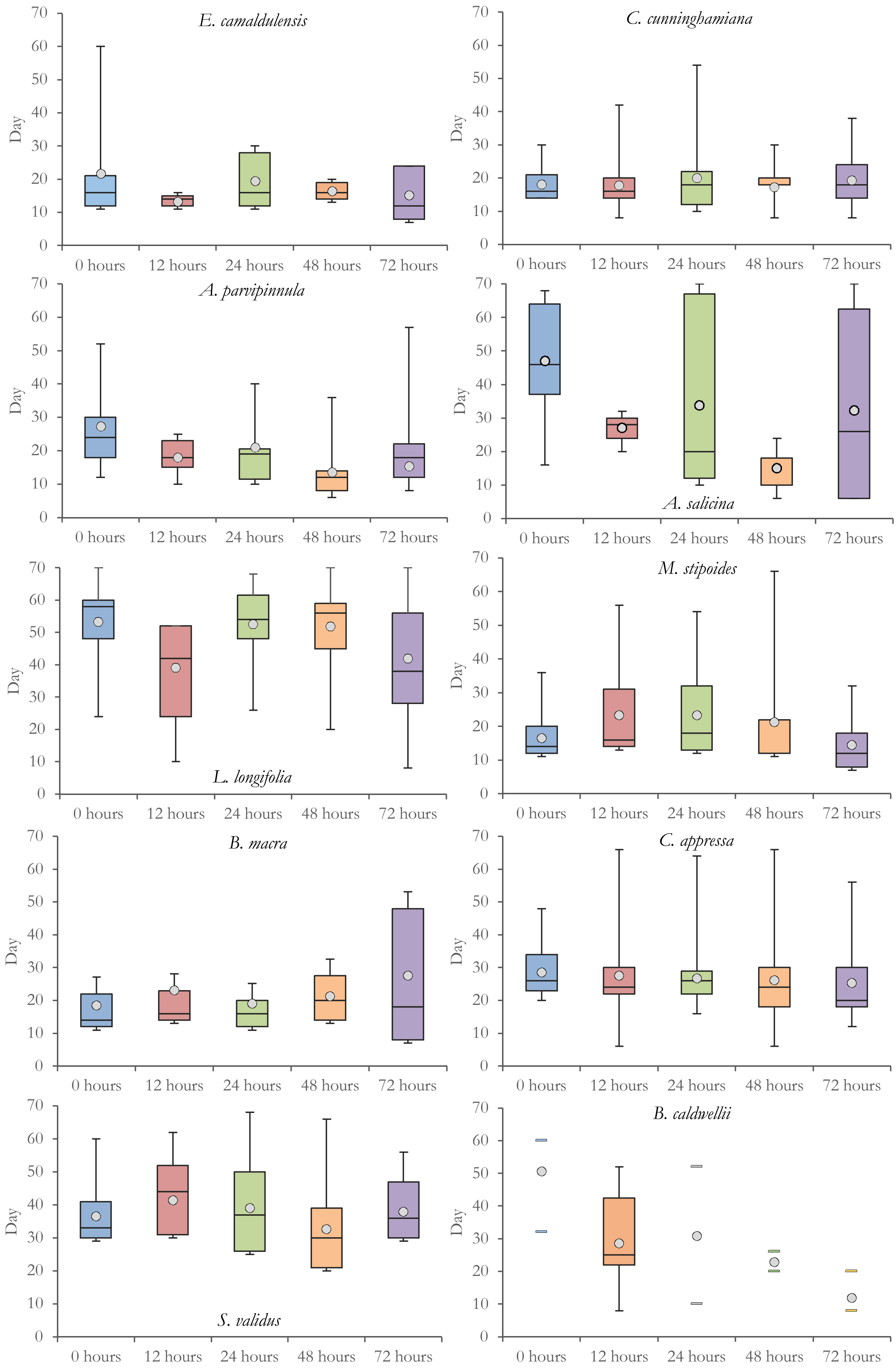


Figure 4.13. Seedling emergence timing over 70 days. Boxplots show median and interquartile ranges. Error bars show the first and last day of seedling emergence. The weighted mean day of seedling emergence is shown by the grey circles.

Table 4.14. Summary of key findings for the livestock exclusion and environmental flow projects

Project	1. Passive revegetation success	2. Management considerations	3. The utility of the seed bank
Livestock exclusion	*Erosion resistance:* – Non-significant increases in vegetation cover and significant decreases in exposed sediment – Greater Manning's n scores and the regeneration of woody species and growths of *C. appressa* – Achieved a decrease in potential bench sediment transport across the site during the April 2015 flood event *Restoring native species:* – Significant differences between the livestock-exclusion and continuously-grazed site were identified at the species level – Significant increases in the percentage cover and species richness of native species were identified – No substantial restoration of species representative of surrounding extant communities was found	– Identified a lack of woody and late successional species regenerating from the seed bank – Identified a dominance of exotic species within the livestock excluded site after three years	– The composition of the coarse fraction of sediment was found to vary between bench levels one and two – The coarse fraction sediment composition of bench level one at the continuously grazed site was distinct from the same bench level at the livestock excluded site – Significant differences in vegetation composition between bench levels one and two were identified, with the differences more distinct at the continuously-grazed site
Environmental flows	– Seedling emergence was not significantly greater for any species when the most successful inundation treatment was compared to non-inundation – Only *B. macra* and *A. parvipinnula* experienced significant differences in seedling emergence when all treatments were compared – Significant differences were identified between the most and least successful treatments for the majority of species – Significant differences were identified in seedling emergence timing among treatments for the majority of species – No patterns in the amount of seedling emergence or seedling emergence timing among growth form groups were identified	– The current environmental water allocation is inadequate to increase the magnitude of small floods to a magnitude sufficient to inundate bench units – The inundation treatments did not cause significant detrimental impacts on seedling emergence for the majority of species	– Inundation up to 72 hours may not significantly affect seedling emergence from the seed bank of bench units – Seedling emergence timing from the seed banks may be significantly affected by inundation duration

Chapter 5: Discussion

1. Determining the success of livestock exclusion and environmental flows as passive revegetation techniques

Livestock exclusion

Erosion resistance

Vegetation increases erosion resistance by binding sediment and creating drag, lowering the energy of flows and their erosive potential (Weiming and Zhiguo, 2009; Tsujimoto, 1999a; Hickin, 1984). A non-significant increase in vegetation percentage cover was found at the livestock excluded site for each time period. A similar study in Kansas was able to detect a significant increase in percentage cover after two years of livestock exclusion (Hoover et al., 2001), however the majority of other studies take place over much greater time periods e.g. seven years (Lunt et al., 2007), ten years (Green and Kauffman, 1995), thirty years (Schulz and Leininger, 1990). It is expected that over a longer time frame the significance level at the study reach will be achieved. A comparison of the percentage cover of exposed sediment between the sites found it to be significantly lower at the livestock excluded site for each time period. These findings align with Schulz and Leininger (1990) in Colorado, Hoover et al. (2001) in Kansas, Burger et al. (2010) in Victoria and Robertson & Rowling (2000) in southern NSW, using the same or similar test criteria.

The visual comparison between the sites displays a clear increase in the amount and structural diversity of vegetation at the livestock-exclusion site when compared to the continuously-grazed site. Trees and large growths of the sedge *C. appressa* were the most obvious differences. The comparison of growth form groups did not identify significant difference in graminoid percentage cover between the livestock excluded and continuously-grazed sites, which is attributed to their patchy distribution. Hough-Snee et al. (2013) also identified an increase in *Carex* species after four years of livestock exclusion in Utah, USA. Regeneration of these growth forms is an important difference between the sites, since trees and sedges provide greater erosion resistance than grass or herbaceous species during major flows (Rood et al., 2015; Hupp and Osterkamp, 1996; Dunaway et al., 1994). Dense growths of herbaceous species are effective at covering surface sediment, however the depth to which their roots are capable of binding sediment is extremely limited (Abernathy and Rutherfurd, 1999). In contrast trees are excellent at increasing bank strength through root binding and provide important sources of roughness during high flows that herbaceous species cannot provide (Lyons et al., 2000). Grasses and emergent macrophytes are also highly important during high flows as their flattened shape, when submerged, serves to protect surface sediment from bank scour and are also highly effective at trapping sediment during minor flows (Lyons et al., 2000; Abernathy and Rutherford, 1999).

The Manning's n roughness calculations identified greater roughness at the livestock excluded site. The factors driving this increase were the amount of vegetation and effects of obstructions. The size and extent of the vegetation at the livestock excluded site is a critical factor, which by reducing the energy of flows, induces greater deposition of debris from flood waters, potentially also increasing organic matter deposition (Kleeberg et al., 2010; Srivastava et al., 2008; Schulz et al., 2003; Tsujimoto, 1999b). A similar study of floodplains in southern NSW found coarse particulate organic matter and woody debris to be significantly more abundant in livestock excluded sites than grazed sites (Robertson & Rowling, 2000). Woody debris and organic matter are critical components of river health and functioning, playing key roles in creating habitat and structuring macroinvertebrate and fish assemblages (Dosskey et al., 2010; Thorp et al., 2006; Junk et al., 1989; Crook and Robertson, 1999; Bilby and Likens, 1980).

The reconstruced flood conditions of the April 2015 flood event were used to calculate potential sediment transport values for a range of different vegetation conditions that reflect different management scenarios. The regenerated vegetation at the livestock excluded site reduced the average potential bench sediment transport by 7434 tonnes per day. A decrease of 18% on it's pre-livestock exclusion condition in 2011. If the continuously-grazed site was passively revegetated to the standard of the livestock excluded site, the average potential bench sediment transport across the site could be reduced by 2280 tonnes per day, a decrease of 9%. Comparing the maximum difference in potential sediment transport between passive and pre revegetation scenarios for a cross-section at each site reveals an even stronger picture. Transect eight at the livestock excluded site had a reduction of 31286 tonnes per day (78%) of potential bench sediment transport when compared to it's pre revegetation condition. Transect sixteen at the continuously-grazed site could have had a potential bench sediment transport reduction of 5255 tonnes per day (93%) if it were revegetated to the standard of the livestock excluded site. If woody species were introduced to increase the roughness of the study reach, in addition to livestock exclusion, the potential bench sediment transport across the livestock-exclusion site for the April 2015 flood event could have been reduced by a further 1811 tonnes per day. By introducing woody species through direct planting in addition to livestock exclusion at the continuously-grazed site, a further reduction of 56 tonnes per day could be potentially generated. While the addition of woody species in the simulations achieves a relatively small increase in erosion resistance, woody vegetation provides other important functions such as enhanced sediment deposition, provision of large woody debris and valuable habitat (Gurnell et al., 2014; Tabacchi et al., 1998; Naiman & Decamps, 1997; Stauffer et al., 1980).

Improved erosion resistance under livestock exclusion has the potential to arrest knickpoint formation and migration (Erskine et al., 2010; Rutherford et al., 2000; Shields Jr et al., 1995). The knickpoint which formed at the continuously-grazed site has created a plunge pool which is acting to increase channel capacity and concentrate flow. During the April 2015 flood event unit stream power decreased downstream along the livestock excluded reach by 4.2 W/m^2 (15%), compared to the continuously grazed reach where unit stream power increased downstream by 8.6 W/m^2(35%). This contrasting pattern is most likely to be due to the difference in roughness between the sites. The knickpoint was formed just downstream of the last cross-section at the continuously-grazed site where the unit stream power was highest, although not as high as that experienced at the livestock excluded site.

Restoration of native vegetation

The percentage cover and species richness of native species were found to be greater at the livestock excluded site for each time period. These differences were significant at all time periods for percentage cover and significant for winter 2014 and 2015 for species richness. This is in contrast to (Lunt et al., 2007) who found no significant relationship between livestock exclusion and native species richness in a riparian forest in Victoria over a twelve year period. The changes in native species richness identified in this study were significant but small, with the regenerated vegetation at the livestock excluded site being dominated by exotic species. This is unsurprising since riparian zones, particularly in agricultural areas, are highly susceptible to invasion by weed species (Williams et al., 2008; Water and Rivers Commission, 1999). Jansen & Robertson (2001) found weed species to persist in large numbers at sites that have been livestock excluded for over fifty years along the Murrumbidgee River in NSW.

One of the dual aims of livestock exclusion was the restoration of native vegetation (Hunter-Central Rivers CMA, 2011). The term restoration implies returning to a pre-degradation condition (Bradshaw, 1997). A significant difference between the vegetation composition of the sites was identified at the species level however only five out of the fourty-four species identified at the livestock excluded site were representative of the surrounding extant vegetation communities at the study reach. Therefore livestock exclusion has yet to achieve one of its key aims at the study reach. It has been noted that while vegetation communities may respond to livestock exlusion relatively quickly it may take a great deal of time before restoration of extant communities actually occurs and may in some cases not be possible (Hough-Snee et al., 2013; Burger et al., 2010; Williams et al., 2008; Jansen and Robertson, 2001).

<u>Environmental flows</u>

Effects of inundation durations on seedling emergence

The key aim for this project was to establish whether environmental flows could be used as a passive revegetation tool in degraded rivers, such as those that have been used for facilitating fish passage, assisting waterbird breeding, watering vegetation and geomorphic maintenance (Capon et al., 2009; Vietz et al., 2005; Chessman et al., 2003; Whiting, 2002). The criteria used to assess the potential as a tool was whether a set of inundation durations could result in greater seedling emergence than non-inundation. None of the species tested were found to experience significantly greater seedling emergence under any inundation duration when compared to non-inundation. Therefore environmental flows may not represent good tools for improving the recruitment of desirable species from the seed bank in degraded rivers. Additionally Greet et al. (2013b) has noted that in regulated rivers the riparian seed banks are likely to be dominated by understory species, as in non-regulated systems, limiting their potential contribution to community regeneration. The results from this study did provide some significant effects on the amount and timing of seedling emergence.

Significant differences in total seedling emergence among all inundation treatments was identified for only two species, *A. parvipinnua* and *B. macra*. Although the majority of species showed significant differences between the most successful and least successful treatments, only the grass growth forms shared a significant preference for the same treatment, in this case non-inundation. Neither species of grass appear to have seed that tolerates long periods of inundation. This is unsurprising since these species tend to be used at locations on the bank face where inundation is infrequent (Schneider, 2007). The species which may be described as inundation tolerant generally displayed greater emergence under the long inundation treatments. These species are the tree species *E. camaldulensis*, *C. cunninghamiana*, the sedge *C. appressa* and the rush *S. validus*. The rush *B. caldwellii* did not share this pattern and the mechanism behind it's emergence responses to inundation durations are unclear. Interestingly the results suggest that these inundation tolerant species, apart from *C. Appressa,* may require specific inundation durations to significantly increase seedling emergence. For each of the inundation tolerant species the most successful inundation treatment was significantly greater than the least successful inundation treatment but not significantly greater than non-inundation. Duration of inundation has been found to affect species emergence from the seed bank in wetlands (Casanova and Brock, 2000) and so may have a greater effect on species which have some adaptation to seed dispersal by water or are semi aquatic in comparison to terrestrial species. Inundating bench units without regard to identifying the composition of the seed bank and the required inundation durations for each speices is likely to receive variable and not necessarily positive effects on recruitment from the seed bank.

Effects of inundation on seedling emergence timing

The inundation durations resulted in significant differences in the patterns of seedling emergence for the majority of species, although these were variable. Interestingly the species identified as being inundation tolerant, except for *C. appressa* and *B. caldwellii,* did not experience significant differences in emergence timing among inundation treatments. For *C. appressa* only the Breslow test identified a significant difference, due the early and initial increase in emergence for the 48 and 72 hr treatments. These results suggest that the timing of recruitment from the seed bank of bench units for inundation tolerant species may not be significantly impacted by the duration of flooding along the study reach. Each non-inundation tolerant species was found to have significantly different seedling emergence timing patterns among the treatments. Since the difference in earliest emergence is very small among the treatments for most species, further analyses would have to be applied to confirm these are indeed patterns. Other authors have noted the importance of inundation duration in changing the emergence and timing of emergence in species. For example Casanova and Brock (2000) found that inundation duration may be important in segregating plant communities regenerating from the seed bank of wetlands in the Northern Tablelands of New South Wales. Ge et al. (2013) noted a delay in seedling emergence of terrestrial species under inundation from seed banks at Nansi Lake, China.

Summary

This study has demonstrated that within three years, livestock exclusion has signficantly increased erosion resistance, fulfilling one of the dual aims of livestock exclusion (Hunter-Central Rivers CMA, 2011). By actively planting woody species to augment the passively regenerated vegetation, erosion resistance could be further increased, potentially significantly decreasing sediment transport volumes in these degraded sand-bed systems. However livestock exclusion was unsuccessful in restoring species representative of the surrounding extant communities. Despite this livestock exclusion did significantly increase native vegetation cover and species richness, which should be considered a success. Inundation treatments, simulating the effect of environmental or natural flows on bench seed banks were unsuccessful in significantly increasing seedling emergence among a selection of desirable species. In general terms species which were inundation tolerant appeared to display minor increases in seedling emergence under long inundation treatments. Importantly, there were variable responses to individual inundation treatments, with no species significantly preferring the most successful inundation duration over non-inundation, suggesting species specific preferences for treatments. The seedling emergence times of non-inundation tolerant species were found to differ significantly but variably by species. The findings from this study suggest that the provision of environmental water to benches in order to stimulate seedling emergence does not have a high potential as a passive revegetation tool in degraded rivers.

2. Management considerations

Livestock exclusion

Desirable species

A significant limitation of livestock exclusion at the study site was the lack of regeneration of woody and late successional species from the seed bank, a limitation broadly recognised elsewhere (O'Donnell et al., 2014a; Hough-Snee et al., 2013; Goodson et al., 2001; Jansen & Robertson, 2001). These species are desirable for revegetation objectives due to their important roles in maintaining geomorphic function, providing large woody debris, resisting the invasion of exotic species and provision of habitat (Hubble et al., 2010; Corenblit et al., 2007; Tabacchi et al., 1998; Gurnell et al., 1995). Figure 4.2 shows that there has been limited regeneration of woody species from the seed bank, with individuals of *Acacia parvipinulla* and *Melalueca linariifolia* occurring within the fenced sections at the livestock excluded site. Observations of the site over time identified no new regeneration of individuals from these species after autumn 2014, suggesting that regeneration of the woody species may have occurred within a short period after livestock exclusion began, this observation may be attributed to many factors. It has been found that woody species may not be abundant in sediment seed banks and so may not be present in the seed bank at the site (O'Donnell et al., 2014a; Middleton, 2003; Pettit & Froend, 2001a). *A. parvipinnua* individuals, which have been observed to flower profusely each survey year, are abundant on the valley margin and within the upstream macrochannel. It may be that transport from upstream and adjacent sources into the macrochannel of the study reach may be inhibited through a lack of transport vectors, distance, or impediments to dispersal via hydrochory. No significant flood events occurred at the study reach between the April 2015 flood event and the first macrochannel surveys in Winter 2014, it is therefore possible that flows over the bench levels are required to deliver the propagules of woody species to the seed bank via hydrochory or flood debris. For example Pettit and Froend (2001a) found flood debris to be a key source of plant proagules for woody species at two rivers in Western Australia. Seed bank exhaustion occurs where preferential grazing removes individuals or species before they are able to provide seed back to the seed bank, the duration of grazing in the area may have severly depleted the presence of woody species in the seed bank. Additionally herbivory by marsupial species or competition from other regenerated species may be reducing the emergence of woody species (Jansen et al., 2007). Environmental cues or specific conditions such as temperature fluctuations or flooding may also be required to stimulate the germination of some species from the seed bank (Leck et al., 2012; Williams et al., 2008; Corenblit et al., 2007; Jansen et al., 2007). This may be particularly important for species such as *A. parvipinnula* which have hard seed coats. While a key agent of degradation in grazing has been removed, it is unclear which processes, e.g. hydrochory, may need to be replaced or enhanced in order to fully revegetate the study reach in it's post degradation geomorphic context.

Biogeomorphic succession

The regenerated vegetation at the livestock excluded site is dominated by early successional species, with the majority being exotics, a finding shared by other studies (Burger et al., 2010; Williams et al., 2008). After three years of livestock exclusion the regenerating community has changed significantly in species composition, percentage cover, origin and number of species, but only slightly in the relative contributions of different growth form groups. The species composition of the livestock excluded site is indicative of an early secondary successional stage, as it is dominated by pioneer species (Corenblit et al., 2007, Horn, 1974). Desirable late successional and woody species are more likely to utilise methods of regeneration such as lignotubers and seritony than the seed bank (Jensen et al., 2008, Abernethy and Willby, 1999). The lack of desirable species in the seed bank identified in the literature and the lack of regeneration of these species at the study reach, effectively limits the usefulness of livestock exclusion to the regeneration of an understory vegetation community (Bossuyt and Honnay, 2008; Williams et al., 2008). Direct plantings are required to progress the vegetation composition for futher succession, as suggested by (O'Donnell et al., 2014b; Hough-Snee et al., 2013). Aside from the previously discussed benefits that the individuals of desirable species provide, later successional communities are better able to buffer themselves against disturbances and able to re-establish faster when disturbances do occur (Tabacchi et al., 1998). Despite the lack of restoration of the surrounding extant communities, the improvements in erosion resistance, total vegetation cover, native vegetation cover and richness demonstrates that livestock exclusion provides a useful basepoint for further revegetation works.

Active revegetation works

Rather than attempting to restore an native floodplain vegetation community to the bench units through livestock exclusion alone, active plantings of late successional and woody tree species will be required to move the regenerating community towards a managed state that can be sustainably maintained and provide gemorphological and ecological value (O'Donnell et al., 2014b; Holl and Aide, 2011; Dorrough et al., 2008; Vesk and Dorrough, 2006). Several authors have commented that re-establishing natural conditions in degraded rivers is unlikely to be possible and instead suggest focussing on increasing the existing value of a site (Brierley and Fryirs, 2008; Shields et al., 1999; Bradshaw, 1997; Hey, 1997; Hancock et al., 1996). The community must be sustainable in the sense that propagule sources will be provided on a scale that is sufficient to maintain a community structure over time (Groves, 2007). Therefore the identification of important seed sources to the study reach is important to the long term success of the regenerating community, emphasising the need for revegetation projects to work at scales greater than the site level (Cullum et al., 2008; Hillman and Brierley, 2005; Briggs, 2001; Harper et al., 1999).

Weed managment

The number of exotic species and their contribution to vegetation cover at the livestock excluded site across all time periods highlights the need for sustained weed management with passive revegetation projects (Williams et al., 2008). Weed control is especially important in riparian and agricultural areas where sources of weed seed distribution and agents of transport are numerous (Tickner et al., 2001, Price and Lovett, 1999). However, in the absence of any noxious or environmental weed species at the study reach it is suggested that weed management may be counterproductive, particularly at early stages of the passive revegetation process. The cover provided by the regenerating species is more valuable for erosion resistance than community structure (Water and Rivers Commission, 1999). Passive revegetation from the seed bank under livestock exclusion was found to be a highly effective and low cost means of establishing of pioneer species to provide erosion resistance and ground cover (Jensen et al. 2008; Williams et al. 2008).

Environmental flows

Could environmental flows targeting bench units be used for passive revegetation in degraded rivers?

Although this study found that inundation may not be a useful passive revegetation tool in promoting the regeneration of desirable species, the provision of water to extant vegetation in the form of maintenance flows (Siebentritt et al. 2004; Hughes & Rood, 2003), presents another potential passive revegetation use for environmental flows targetting bench units. Providing environmental water to the floodplain is not feasible at the study reach due to the decoupling of the floodplain from the low-flow channel as a result of channel incision and widening (Hoyle et al., 2008). Furthermore the floodplains of the study reach have been extensively utilised for agricultural purposes. Therefore the vegetation communities of the bench units represent the most significant riparian habitat of the study reach and should be the target units for revegetation and protection. Bench units have been used for targets of vegetation inundation and also the recruitment of organic matter elsewhere, for example the Broken catchment in Victoria (Farquharson et al., 2011). This study found that inundation had no significant deleterious effects on seedling emergence of the species tested, suggesting that other passive revegetation applications, such as maintenance flows may be considered appropriate for the study reach. The response of exotic species to inundation was not tested, it has been suggested that environmental flows may be used to increase mortality among exotic species less well adapted to the riparian environment and provide native species with a competitive advantage (Shafroth et al., 2010). Seasonal flows may also be used in regulated systems to assist native and reduce exotic species (Greet et al., 2013a).

The environnmental water allocation reseved for the Hunter River between Glenbawn and Glennies Creek Dams is insufficient to be used as a means of providing environmental flows targeting the bench units of the study site, regardless of the purpose of the release. Passive revegetation is not currently one of the implemented uses of the environmental water allocation, although the use of environmental flows as a tool in maintaining and improving riparian biodiversity is listed as a outcome within the Catchment Action Plan and such a use would fall within a key biodiversity aim of protecting and regenerating streams and wetlands (Hunter-Central Rivers CMA, 2013). The environmental water allocation at the Upper Hunter is relatively small in part because of the three major tributaries that join the Hunter River downstream of Glenbawn Dam and above the study reach, which serve to re-establish a "high degree of naturalness to the flow regime" (Simons 2015, pers. comm., 9 February). In order to use environmental flows as a tool in passive revegetation, the catchment would have to reserve an environmental water allocation of at least double the current amount. Increasing the environmental water allocation may become more important as river managers become more concerned with adaptation to climate change and increasing land use and development pressures (Poff and Matthews, 2013). Setting targets for rehabilitation within regulated systems will involve evaluating which strategies are most effective and can provide ecological and societal outcomes within future parameters of water use pressures which will only increase (George et al., 2011). Utilising the seed bank may not be a viable option for environmental flows but this should not inhibit their use for other purposes.

Summary

Livestock exclusion provides a highly effective tool as a low cost means of providing an early successional understory that encourages the regeneration of native species (Williams et al., 2008; Green and Kauffman, 1995; Schulz and Leininger, 1990). However, riparian seed banks do not to provide significant sources for woody and late successional species, reflected in the regenerating vegetation at the livestock-exclusion site, therefore livestock exclusion requires complementary techniques to provide overstorey and key desirable species (O'Donnell et al.a, 2014; Holl and Aide, 2011). Weed management will need to become an essential and ongoing tool once the erosion resistance of the passively regenerating community is sufficient (Water and Rivers Commission, 1999). In order to implement an environmental flow for passive revegetation or maintenance flows the current environmental water allocation for the Upper Hunter catchment would have to be at least doubled.

3. The utility of the seed bank for passive revegetation

Livestock exclusion

Variations in bench level vegetation composition

A significant difference between the species composition of the bench levels at both sites, independent of revegetation condition, was identified. Differences in seed bank composition between geomorphic units has been previously described and attributed to the varying sedimentological characteristics and formation processes acting on geomorphic units (O'Donnell et al., 2015; Goodson et al., 2002). The difference in disturbance frequency through flooding, which created the different sedimentological characteristics between the bench levels, most likely explains the differences in vegetation compostion between the bench level (O'Donnell et al., 2015). The difference in vegetation composition between the bench levels was found to be proportionally greater at the continuously-grazed site where the difference in grain size between the bench levels was also greatest. In the Wollombi subcatchment O'Donnell et al. (2015) found species richness of the seed bank to significantly decrease with increasing average particle size. The authors also identified correllations with other sediment characteriscics for species richness and abundance including the percentage of organics, fine sediment and coarse sediment. The results of this study suggest that differences in vegetation composition due to varying disturbance frequencies may be significant and observable within geomorphic unit types.

An inspection of the key species driving the differencs between the bench levels further suggest disturbance frequency as the key factor behind the differences in vegetation compositions. Two inundation tolerant species *C. appressa* and *Isachne globosa* (RBGSYD, 2015[1], RBGSYD, 2015[2]) had greater percentage cover at bench level one than two. *A. virginicus* and *P. dilatatum* are both terrestrial species and were more abundant at bench level two, where they would receive less disturbance by flows. Both *P. clandestinum* and *H. radicata* had greater percentage cover at bench level one in winter 2014 but bench level two in winter 2015, with *P. clandestinum* also preferring bench level two in summer 2015. In the case of *P. clandestinum,* competition pressure may have been exerted upon *P. clandestinum* at bench level one during summer 2015 when *I. globosa* became more abundant. For *H. radicata* the mechanism is unclear, although this species is an agressive pioneer and it's varying preference for bench level may simply reflect presence of bare substrate (Weeds of Australia, 2015). Disturbance frequency, creating fine scale differences in sediment characteristics, may result in significantly different vegetation compositions regenerating form the seed bank within geomorphic unit types (O'Donnell et al., 2015, McIntyre et al., 1995). Accordingly there are implications for sampling for seed bank assays in order to determine the utility of seed bank of a site for passive revegeation. Land managers should be careful to interpret and account for varying levels of disturbance frequency of landform surfaces within geomorphic unit types when sampling seed bank material and planning revegetation in additon to targeting specific geomorphic units, zones of flood debris and deposits of fine sediment (O'Donnell et al., 2015).

Setting appropriate revegetation goals

The lack of species regenerating under livestock exclusion that are representative of the extant commmunities of the study reach could be considered a failure in terms of the original revegetation objectives (Hunter-Central Rivers CMA, 2011). However, this study, amongst others, demonstrated a lack of understanding of the utility of seed bank in objective setting for revegetation (Bossuyt and Honnay, 2008, Goodson et al., 2001). It has been shown across many locations that the sediment seed bank is often dissimilar to extant vegetation and is principally dominated by early successional species (O'Donnell et al., 2014b; Jensen et al., 2008, Williams et al., 2008, Hopfensperger, 2007).

The floodplain vegetation community that existed prior to clearing and grazing was established in very different geomorphic conditions than that of the contemporary bench units. The floodplain formed through gradual accretion as overbank flows transported and deposited fine sediment on surface of the floodplains. The bench levels at the site formed over a much shorter time period than the floodplain and in few depositional events, leaving a less well sorted and coarser sediment composition. The speed of formation, formation process, erosion processes and sediment characteristics will all exert controls on the composition of seed banks within distinct geomorphic units (O'Donnell et al., 2015, Goodson et al., 2002). Therefore it is unreasonable to expect the seed bank to provide regenerating vegetation that is representative of pre-existing or surrounding extant floodplain communities. Furthermore the individuals regenerating from the bench units will be subject to differing disturbance frequencies and intensities to those of the floodplain communities due to the reduced lateral distance from and vertical elevation above the low-flow channel (O'Donnell et al., 2015; Lite et al., 2005). In light of these considerations, a significant increase in the number and percentage cover of native species within three years is a positive achievement for livestock exclusion at this site. Where the composition of the seed bank is unknown, the protection of regenerating native species is a more suitable objective until it is observed which species or growth form groups are absent and should be directly introduced.

Environmental flows

Possible seed bank responses to inundation

The effects of inundation were negligible for the majority of species. Although grass species may be negatively affected by extended flood durations it is expected that they may be restored by the occupation of disturbed sediment after flooding, since these species are well adapted to disperse and frequently establish in disturbed areas (NSW Department of Primary Industries, 2015). Seven of the ten species were identified as having significantly different patterns of seedling emergence timing over the study period. The effects of inundation on seedling emergence timing may be of some importance in shaping community composition. Capon and Brock (2006) found differences in seed bank compositions along a flood frequency gradient. The flooding of bench units is very infrequent at the study reach (Hoyle et al., 2012), therefore the minor variations in emergence timing are likely to be unimportant. This study used only desirable species for the experiment, it is likely that the seed bank composition of the bench units have been heavily affected by flow regulation. Greet et al. (2013b) found that the seed bank of regulated rivers store high numbers of exotic species and are unlikely to be key sources of plant diversity.

Summary

This study identified significant differences in species composition between bench levels and attributed these to the varying disturbance frequencies experienced by each level. A deficiency in understanding the utility of the seed bank for revegetation and a lack of appreciation of the geomophic change since European settlement lead to the setting of a revegetation objective that is unachievable. The results of the environmental flow study indicate that bench units are unlikely to experience major changes in recruitment from the seed bank under the inundation durations possible in regulated rivers. However small and variable but significant differences in seedling emergence timing were identified for the majority of species among inundation durations. These differences in species composition at the bench levels and responses to inundation underline the importance of fine-scale variations in geomorphic and hydrological process on controlling seed bank composition and recruitment from the seed bank, even within geomorphic unit types.

Chapter 6: Conclusions

Livestock exclusion was found to be an effective passive revegetation technique over a three year period at the degraded study reach in the Lower Hunter Region. Vegetation cover was consistently greater and the amount of exposed sediment was significantly lower at the livestock-exclusion site when compared to the continuously-grazed site, which broadly reflects the findings of other studies. The roughness provided by the regenerating vegetation was found to substantially reduce potential bench sediment transport under a moderate flood event, resulting in an average reduction of 18% across the site and a maximum of 78%, when compared to it's pre livestock exclusion condition. This study found that the reduction in potential bench sediment transport could be further reduced by directly planting woody species which did not regenerate from the seed bank in substantial numbers. The vegetation compostion of the livestock-exclusion site was significantly different to the continuously-grazed site at the species level and experienced significant increases in native species richness and percentage cover, although exotic species dominated. A lack of regeneration of woody species and species representative of extant communities were identified as limitations to the utilisation of the seed bank, as other studies have noted. Livestock exclusion achieved the revegetation objective of increasing erosion resistance but not the restoration of native species. Restoring communities is not a suitable objective for passive revegetation utilising the seed bank nor for sites that have undergone significant geomorphic changes associated with riparian clearing and grazing, for example. A significant difference in the species compositon between two bench levels was identified and attributed to disturbance frequency, suggesting that fine scale differences in geomorphic and hydrological processes may produce significant differences in species composition.

The durations of possible natural or environmental flow assisted inundation durations of bench units at the study reach in the Upper Hunter Region, were not found to significantly improve the seedling emergence of selected riparian species. This study also identified that responses to inundation may be species specific, identifying differences in both total seedling emergence and emergence timing among treatments for multiple species. The inundation durations did not have significant detrimental effects on seedling emergence. While increasing recruitment of desirable species from the seed bank may not be a suitable objective for implementing environmental flows, this should not prevent their implementation for other objectives. The environmental water allocation at the study reach in the Upper Hunter Region, New South Wales, would have to be at least doubled in order to augment minor floods using environmental flows to inundate bench units. Descisions made around future environmental water allocations in the study reach may need to consider increasing the amount of water stored for environmental purposes in the face of changing and increasing pressures and the importance of bench units for river rehabilitation.

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Appendices:

Appendix 1: Location of the study reaches within the Hunter River Catchment

Hunter River Catchment showing major rivers and the locations of the livestock exclusion and environmental flow study reaches.

Appendix 2: Establishing livestock exclusion as the cause of changes in vegetation composition within the livestock exclusion fences

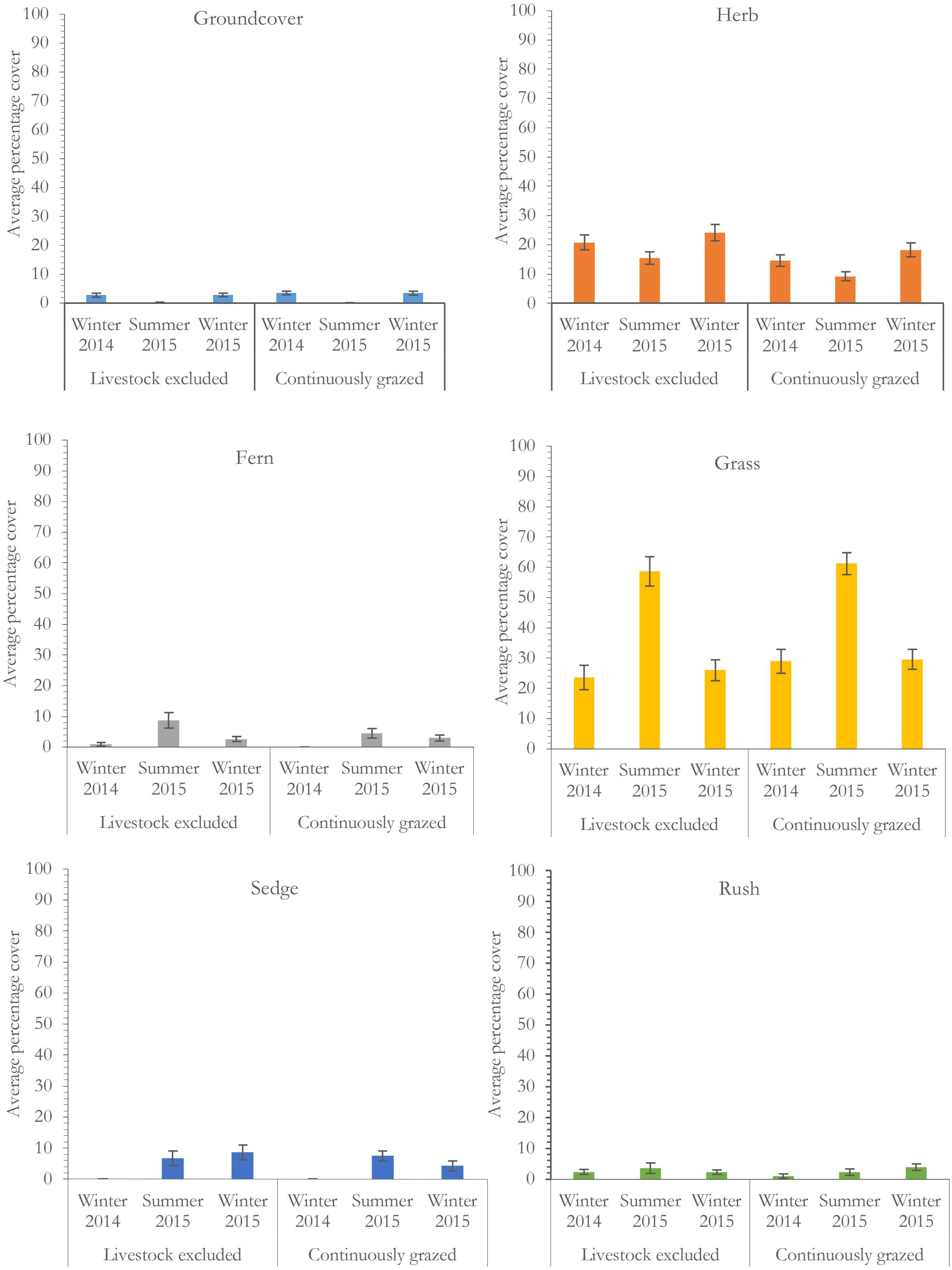
All significant differences over time were found to follow the same pattern of increasing or decreasing mean values across the study reach, except for exotic species richness in the autumn sample dates. Since the same pattern was followed across the study reach the differences were attributed to valley wide environmental factors such as rainfall, therefore the changes in vegetation composition within the fenced zone at the livestock-exclusion site. Care was taken accordingly when interpreting the results of exotic species richness. The difference between the exotic species richness for the autumn samples is small and may be due to several factors that were not able to be studied in this project, for example grazing behaviour. The key information that this table communicates is that there was no change in the other eight factors tested.

Results of paired t-tests for differences in the grazed sections of the continuously grazed and livestock excluded sites over time.

		n	Mean	SE mean	Mean	SE mean	T-value	P-value
Criteria	Site		Summer 2012		Summer 2015			
Percentage cover	Grazed	24	83.75	7.47	82.75	3.12	0.14	0.891
	Excluded	25	82	7.66	93.8	3.83	-1.65	0.112
Native percentage cover	Grazed	24	20.46	5.61	35	5.1	-2.25	0.034**
	Excluded	25	17.16	4.55	30.12	4.64	-1.88	0.073
Exotic percentage cover	Grazed	24	63.29	7.92	47.75	5.82	1.84	0.079
	Excluded	25	64.84	8.49	63.68	5.18	0.19	0.853
Forb percentage cover	Grazed	24	24.21	6	28	5.38	-0.57	0.577
	Excluded	25	17.56	2.51	21.88	3.74	-1.14	0.267
Graminoid percentage cover	Grazed	24	64.5	8.29	64.75	5.05	-0.03	0.976
	Excluded	25	60.08	8.25	73.12	5.3	-1.85	0.128
Species richness	Grazed	24	9.25	0.649	5.958	0.456	5.38	0.000**
	Excluded	25	9.68	0.556	6.24	0.247	5.99	0.000**
Native species richness	Grazed	24	2.375	0.287	2.042	0.221	0.85	0.405
	Excluded	25	2.12	0.218	1.6	0.183	2.06	0.05
Exotic species richness	Grazed	24	6.875	0.575	3.917	0.371	6.17	0.000**
	Excluded	25	7.56	0.529	1.221	0.244	5.77	0.000**
			Autumn 2014		Autumn 2015			
Percentage cover	Grazed	24	83.33	3.41	81.63	3.53	2.47	0.022**
	Excluded	25	87.64	4.26	84.4	3.21	0.68	0.504
Native percentage cover	Grazed	24	19.79	4.54	34.67	5.52	-1.92	0.067
	Excluded	25	6.64	2.05	36.32	4.98	-5.7	0.000**
Exotic percentage cover	Grazed	24	63.54	6.07	46.96	6.14	2.19	0.039**
	Excluded	25	81	5.14	48.08	5.63	4.96	0.000**
Forb percentage cover	Grazed	24	29.58	4.75	12.63	2.18	3.97	0.001**
	Excluded	25	27.92	3.74	16	2.75	3.05	0.005**
Graminoid percentage cover	Grazed	24	61.88	4.92	69	4.44	-1.6	0.122
	Excluded	25	59.52	5.59	67.96	4.27	-1.57	0.131
Species richness	Grazed	24	5.167	0.339	6.5	0.404	-2.78	0.011**
	Excluded	25	5.32	0.39	5.84	0.325	-1.04	0.309
Native species richness	Grazed	24	0.875	0.139	1.292	0.185	-1.79	0.086
	Excluded	25	0.52	0.143	1.44	0.209	-3.76	0.001**
Exotic species richness	Grazed	24	4.167	0.384	5.25	0.362	-2.33	0.029**
	Excluded	25	4.8	0.365	4.4	0.316	0.82	0.422

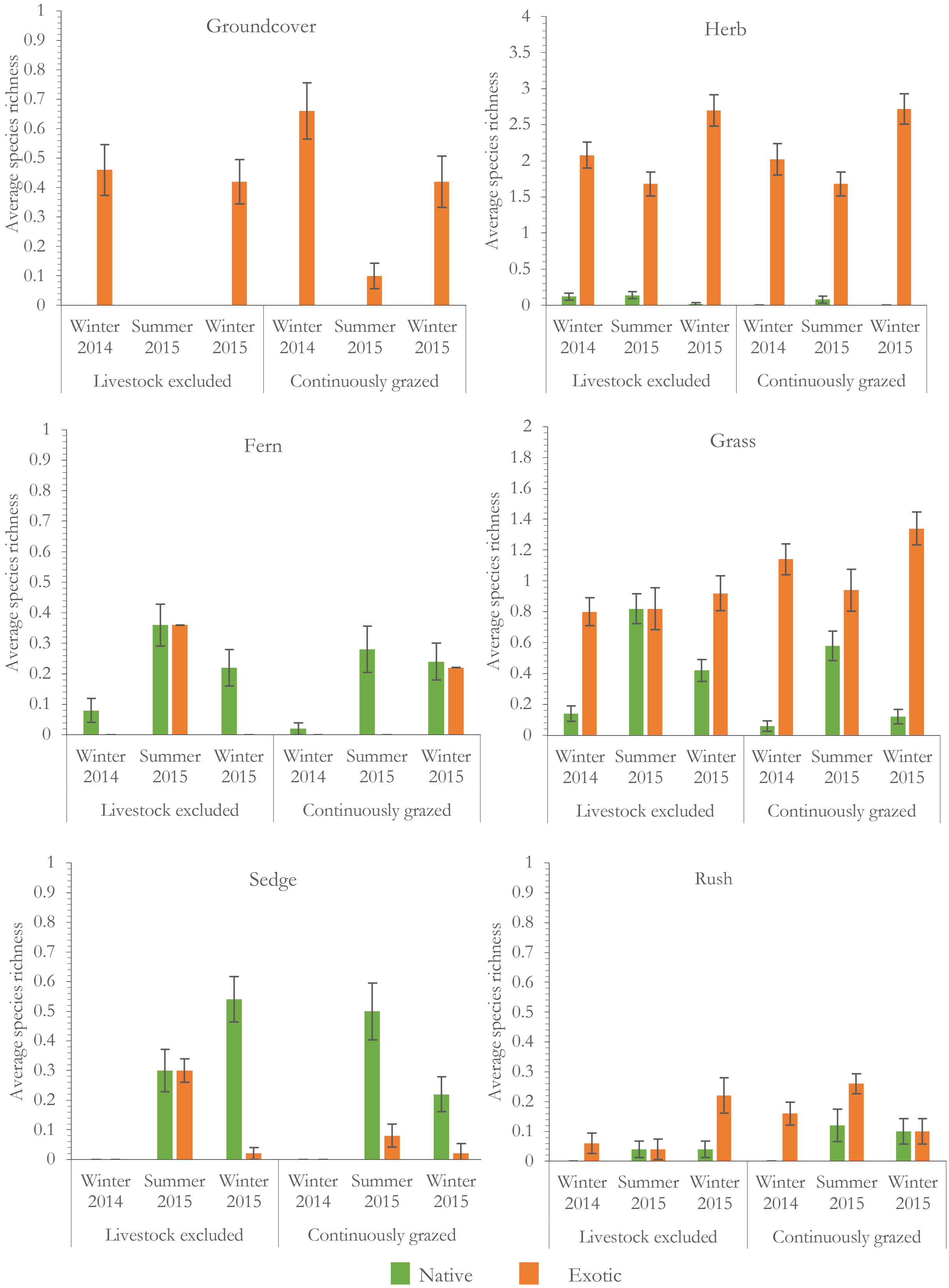
**P-values < 0.05 are significant.

Appendix 3: Changes in growth form group percentage cover after three years of livestock exclusion



Comparisons of mean average percentage cover per quadrat by growth form between the livestock-exclusion and continuously-grazed sites over at three time periods. Error bars show standard error.

Appendix 4: Changes in the native and exotic proportions of growth form group species richness after three years of livestock exclusion



Comparisons of mean average species richness per quadrat by growth form between the livestock-exclusion and continuously-grazed sites over at three time periods. Error bars show standard error.