An evaluation of riparian restoration:

A case study from the Upper Murrumbidgee Catchment, NSW, Australia.

William Higgisson

Bachelor of Environmental Science

Institute for Applied Ecology

University of Canberra

A thesis submitted in partial fulfilment of the requirements for the degree of Bachelor of Applied Science (Honours) at the University of Canberra.

December 2014

Abstract

Riparian zones are highly productive and dynamic habitats, which possess a diverse range of ecological processes. The riparian zone mediates the movement of water, sediment and nutrients between terrestrial and aquatic ecosystems. This process is vital to the health of freshwater ecosystems. Many riparian areas across the world are now in a degraded condition, impacting on their ability to function effectively. Riparian restoration has therefore become an important part of water resource management. Riparian restoration has focused on improving and enhancing riparian vegetation. The underlying assumption has been that on-ground works will automatically improve the ecological functioning of the riparian zone. As long-term monitoring and assessment of ecological restoration projects is rare, this assumption has not been well tested.

This study assessed the riparian and geomorphological condition of sites restored as part of a large scale restoration project undertaken in the Upper Murrumbidgee Catchment between 2000 and 2003. Sites were restored using different methods (fencing to exclude livestock, planting tubestock, and direct seeding) with the primary dual objectives of reducing sediment and nutrient delivery into the Murrumbidgee River by controlling erosion and protecting and enhancing riparian vegetation.

The objectives of this study were: 1) To determine how effective common riparian restoration methods are at: enhancing and protecting the riparian vegetation and reducing stream bank erosion, 2) to determine how different riparian restoration methods influence different features of the riparian zone, and 3) to determine the factors that have affected the outcomes of riparian restoration.

A geomorphological assessment and a riparian vegetation assessment were performed at sites that had undergone different restoration methods and unrestored control sites. Aerial imagery was also used to compare width of riparian canopy vegetation and projective foliage cover before restoration commenced and ten years after.

Restoration has led to significant improvements in total riparian vegetation condition and a range of riparian attributes. Width of riparian canopy vegetation, native mid-storey cover, native canopy cover and seedling recruitment of mid-storey species were significantly better in sites that had undergone fencing and tubestock planting or fencing and direct seeding compared to the control sites. Analysis of remotely sensed data demonstrated improvements in both the width of riparian canopy vegetation and projected foliage cover, in sites in all restoration methods but especially sites in active restoration methods. After ten years bank condition was found to be significantly better in sites in all restoration methods compared to the unrestored sites. Remnant vegetation was found to have a significant influence on the abundance of seedlings of both canopy and mid-storey species and the amount of debris on a site. Native groundcover was also found to influence seedling recruitment. The species used on the actively revegetated sites was found to vary in survival rate (occurrence probability) with *Acacia*, *Eucalyptus* and *Casuarina* species performing the best. Out of the actively revegetated species, eight *Acacia* species and one *Leptospermum* species had successfully recruited seedlings. This study demonstrated that after ten years restoration has led to improvements in riparian vegetation condition and bank condition and provides evidence that the restoration project has met its initial project objectives.

Certificate of Authorship of Thesis

Except as specifically indicated in footnotes and quotations, I certify that I am the sole author of the thesis submitted today entitled: *An evaluation of riparian restoration: A case study from the Upper Murrumbidgee, Catchment, NSW, Australia*, in terms of the Statement of Requirements for a thesis issued by the University Research Degrees Committee.

Signature ____________________________

Date ____________________________

Acknowledgements

My Supervisors:

Dr. Fiona Dyer (Senior Research Fellow, Institute for Applied Ecology) for all your encouragement, patients, and guidance.

Dr. Paul Downey (Associate Professor of Environmental Science, University of Canberra) for your support, guidance and rigorous editing.

Special thanks to Lori Gould and Greening Australia for your assistance, information and funding of this project.

Thanks to Dr. Sue Briggs (Adjunct Professor, University of Canberra) for your much appreciated assistance with the project.

Thanks to Dr. Maria Boyle (Educational Designer, University of Canberra) for assisting with the data analysis.

Thanks to Alica Tschierschke for assistance with GIS and mapping.

Thanks to all the landholders I visited, for allowing me access to their properties and giving up their valuable time to meet with me.

Table of Contents

List of tables

List of figures

1 Introduction

1.1 Context

The human requirements of freshwater coupled with the immense pressure put on freshwater ecosystems (Gleick 2003), has led to the growing recognition of the social and ecological importance of water resource management (Brooks & Lake 2007). The riparian zone and its associated vegetation influences the condition of freshwater ecosystems (Naiman, *et al.*, 2010), as riparian vegetation mediates the movement of materials such as sediment and nutrients from the terrestrial environment to the aquatic environment (Naiman & Decamps 1997). As low lying points in the landscape the riparian zone and its associated vegetation are strongly influenced by changes to the landscape (Palmer, *et al.*, 2007). Changes in the landscape such as those attributed to agricultural land-use have led to many riparian areas currently being in a poor condition, reducing their ability to maintain the condition of aquatic environments (Patten 1998). Riparian restoration, which is essentially protecting and enhancing riparian vegetation has been a common strategy tasked to recover the normal functioning of the riparian zone in a hope that this will improve river system health and water quality. The implementation of riparian restoration has increased exponentially (Bernhardt, *et al.*, 2005) along with an increased public investment (Brooks & Lake 2007). Unfortunately there is a lack of information on the implementation and outcomes of most restoration projects (Bernhardt, *et al.*, 2005), as monitoring and post-project assessment in ecological restoration is rare, especially long-term > 5 years (Wohl, *et al.*, 2005). This has led to a lack of scientific data on the response of the riparian zone and river geomorphology to riparian restoration actions (Allan 2004; Williams, *et al.*, 1998).

1.2 Thesis outline

The first chapter of this thesis will introduce the riparian zone, its definition, function within the landscape, the importance of riparian vegetation to the functioning of the riparian zone, and the implications of agriculture for the riparian zone. The chapter then discusses ecological restoration as the solution to rectifying degraded or damaged riparian areas, and monitoring and assessment of riparian restoration. This background provides context that establishes a set of null hypotheses that were developed for the purpose of this study, and outlined at the end of chapter 1. Chapter 2 outlines the study site, the restoration project that was monitored, study design, and the methods used for data collection and analysis. Chapter 3 presents the results of the study following the set of outlined hypothesis. Chapter 4 discusses the main findings and conclusions, in relation to the current literature, and the implications for management.

1.3 The riparian zone

1.3.1 Definition

The term riparian refers to biotic communities living on the shore of streams and lakes (Naiman & Decamps 1997). The riparian zone comprises the stream channel and the adjacent area of terrestrial landscape where the vegetation may be influenced by elevated water tables or flooding and by the ability of soils to hold water (Naiman, *et al.*, 1993). Riparian zones are typically highly productive (Décamps, *et al.*, 2004), extremely diverse, dynamic and complex biophysical habitats (Naiman, *et al.*, 1993). As a consequence the riparian zone is extremely important to catchment wide ecological function (Naiman, *et al.*, 2010) and is recognised as a significant landscape component in maintaining regional biodiversity (Naiman, *et al.*, 1993).

Riparian areas are transition zones (or interfaces/eco-tones) in the landscape separating patches of aquatic and terrestrial elements (Planty-Tabacchi, *et al.*, 1996; Naiman & Decamps 1997; Ewel, *et al.*, 2001). They serve as a conduit for fluxes of material and energy from one element to another in the ecosystem (i.e. from terrestrial to aquatic and vice versa). The movement of water, sediment and nutrients from the aquatic parts of the riparian zone, and the deposition of sediments and nutrients from the terrestrial areas influence the form and function of the riparian zone (Ewel, *et al.*, 2001). The riparian zone of a given area is the product of water and material interactions in three dimensions (longitudinal, lateral and vertical) (Brinson, *et al.*, 2002).

Riparian zones are also characterised by the large number of hydrological disturbances associated with water-level fluctuations and availability (Planty-Tabacchi, *et al.*, 1996). Hydrological disturbances regulate population size, species diversity (Lytle & Poff 2004) and community structure. These disturbances place riparian organisms under several challenging environmental conditions including: floods, erosion, abrasion, and drought, in addition to normal biotic challenges such as predation, competition and resource availability (Naiman & Decamps 1997). Thus organisms inhabiting the riparian zone need to develop life-history strategies and adaptations to survive and exploit hydrological disturbances (Lytle & Poff 2004).

The character and value of riparian zones develop as a result of an infinite number of complex interactions among ecosystem features spanning geomorphology, hydrology, and biota (Kauffman, *et al.*, 1997). The flow patterns of water, delivery of sediment and presence and placement of woody debris have been identified as the key processes that regulate the ecological characteristics of the riparian zone and alterations to the catchment can directly affect the input of these materials (Naiman, *et al.*, 1993).

1.3.2 Riparian vegetation

Riparian plant communities exhibit a high degree of structural and compositional diversity (Gregory, *et al.*, 1991). This diversity is thought to be caused by the intensity and frequency of floods, small scale variations in topography and soils, variations in climate, and disturbance regimes imposed on the riparian corridor (Naiman, *et al.*, 1993). Species composition and community characteristics of the riparian zone are shaped by flood velocity and frequency (Naiman & Decamps 1997), substrate characteristics, light, water temperature, water nutrient content (Bornette & Puijalon 2011) and the availability of water from the water-table (Richardson, *et al.*, 2007). Ecological influences such as competition, herbivory, soils and disease also contribute (Naiman & Decamps 1997).

The history of flooding significantly influences the distribution and composition of riparian plant communities (Gregory, *et al.*, 1991; Hupp & Osterkamp 1996). Flooding removes vegetation which creates space for new plant colonisation, and increases the availability of resources, such as nutrients and light (Richardson, *et al.*, 2007). Connections along the riparian zone and with adjacent ecosystems regulate species immigration and emigration (Nilsson & Svedmark 2002). The life-history strategies of riparian plant species will determine whether, where and when a riparian plant may colonise a site (Richardson, *et al.*, 2007).

1.3.3 The importance of riparian vegetation

Rivers move vast amounts of water, but rivers move far more than just water. Soil erosion, sediment movement and deposition are natural features of river systems, and rivers expand and contract in response to seasonal changes in runoff (Naiman, *et al.*, 2010). The sediment movement or cut-and-fill alluviation is the primary formative process of riverine landscape diversity including riparian features (Naiman, *et al.*, 2010). Different flow rates are responsible for different riverine and riparian habitat features (Poff, *et al.*, 1997). The incision of river channels is one of the primary sources of sediment in streams and this occurs where the sheer stress of the river exceeds the sheer strength of the river bed or bank (Naiman, *et al.*, 2010). Riparian vegetation is the key moderator of cut-and-fill alluviation as vegetation slows runoff velocity and increases infiltration (Naiman, *et al.*, 2010). As such the presence of riparian vegetation determines the pathway by which precipitation reaches the channel (Poff, *et al.*, 1997).

The reduced flow rates and increased infiltration from riparian vegetation prevents river erosion and reduces downstream flooding (Patten 1998). Macrophytes take up nutrients, prevent scouring of sediment during high flows, and reduce sediment mobilisation (Naiman, *et al.*, 2010). The root system of woody vegetation reinforces the bank and the enhanced evapotranspiration and reduced infiltration reduces soil moisture level (Darby 1999; Simon & Collison 2002). The influence of riparian vegetation on bank stability is dependent on factors such as the type and density of vegetation, its age and its health (Thorne 1990). For example, grasses and shrubs are effective bank stabilisers at low velocity stream-flow, while woody plants can remain effective at higher velocities (Thorne 1990). Individual or small groups of trees cause turbulence which can increase bank erosion (Darby 1999), while dense vegetation minimises the isolated turbulence (Thorne 1990). The ability of riparian vegetation (living and dead) to obstruct, divert or facilitate water flow (Tabacchi, *et al.*, 2000), influences in-stream hydraulic processes such as flow routing and turbulence (Tabacchi, *et al.*, 2000), which in turn strongly influences large-scale fluvial and morphological river processes such as channel form and bank deposition (Hickin 1984). The level of control that streamside vegetation has on the stream environment is related to stream size, the hydrologic regime, and the local geomorphology (Naiman, *et al.*, 1993). In general the larger the river, the wider and more complex the riparian zone (Nilsson & Svedmark 2002).

Riparian vegetation influences aquatic and terrestrial fauna (Naiman, *et al.*, 1993). Macroinvertebrate (Cummins, *et al.*, 1989; Arnaiz, *et al.*, 2011), and fish communities (Pusey & Arthington 2003) are provided shelter, habitat and food by riparian vegetation (Balcombe, *et al.*, 2011; Naiman, *et al.*, 1993). Woody debris produced by riparian vegetation, acts as habitat for fish and macroinvertebrates (Naiman & Decamps 1997), and is consumed by a few specialised aquatic insects (Gregory, *et al.*, 1991). It also provides protection for small mammals and birds (Naiman & Decamps 1997). The riparian zone operates as a wildlife corridor, which organisms such as fish, birds and mammals use for dispersal, migration and movement (Naiman, *et al.*, 2010). The diversity within many fauna groups such as spiders, birds and mammals is disproportionately higher in the riparian zone than the surrounding landscape (Naiman, *et al.*, 2010; Bennett, *et al.*, 2014). The riparian zone operates as a refuge for biota within a landscape dominated by human land-use or during times of drought and for this reason it constitutes important habitat for rare or uncommon species (Naiman, *et al.*, 2010).

Naiman and Decamps (1997) provides a useful summary of the main physical and ecological functions of riparian vegetation, and these are as follows:

1. Control of the movement of materials and channel morphology,
2. Production of organic matter,
3. Provision of habitat in both aquatic and terrestrial portions of the system,
4. Control of stream microclimate,
5. Maintenance of an enhanced level of biodiversity,
6. Provision of an ecological corridor for species movement and
7. Filtration of nutrients.

1.3.3 Human requirements of the riparian zone

The position of the riparian zone as an eco-tone between the terrestrial and aquatic environments places it on the front line to deal with the stresses caused by food production of the land (the problem) and the preservation of freshwater (the importance). Riparian zones have the ability to retain a significant proportion of water, sediment and nutrients and return chemically more pure water to the stream or river (Naiman 2010) as a result of ecosystem processes such as sediment trapping, nutrient cycling and flood mitigation (Ewel, *et al.*, 2001). This ability is an economically valuable service to society because of the dependence of humans on clean water supplies. The ability of riparian areas to perform these duties is closely related to their condition and presence of vegetation. The riparian zone is recognised as a key component of fresh-water management (Naiman, *et al.*, 1993) vital to the human dependence on freshwater and the benefits that clean water supplies (Arthington, *et al.*, 2010).

1.4 Impacts on the riparian zone in an agricultural landscape

Many riparian areas across the world are now in a degraded condition (Brinson, *et al.*, 2002), impacting on their ability to function effectively. In Australia the removal of riparian vegetation has been common practice (Jansen, *et al.*, 2007) resulting in a number of adverse impacts. These impacts include elevated rates of channel and bank erosion, increased sedimentation, reduced water quality and a loss of biodiversity (Growns, *et al.*, 2003; Burger, *et al.*, 2010). Within agricultural landscapes the riparian zone is prone to further pressures such as livestock grazing, forestry and cropping all of which exacerbate issues such as erosion, sedimentation and water degradation through their effects on the structure, productivity and functional integrity of riparian zones (Patten 1998). Agriculture is a major cause of riparian damage (Paul & Meyer 2001), with the impacts varying depending on the type of agriculture (Lester & Boulton 2008). The context of this thesis will be specifically focused on the impacts of agricultural land-use on the riparian zone.

Agriculture is the primary land-use in Australia, with livestock grazing the most common agricultural activity (National Land and Water Resources Audit 2001). Since European settlement, rivers and wetlands have been used for watering livestock (Jansen & Robertson 2001) which tend to concentrate in riparian areas for comfort, energy conservation and availability of food (Bryant 1982). The presence of livestock can impact on the morphology (Trimble & Mendel 1995), and structure and function (Robertson & Rowling 2000) of the riparian zone.

1.4.1 Livestock grazing

Riparian areas are particularly susceptible to damage by livestock (Robertson 1997). Grazing changes the community structure and dynamics of riparian vegetation (Crosslé & Brock 2002) and reduces vegetation cover (Armour, *et al.*, 1991). The selective nature of herbivore grazing decreases the density of individual species and reduces the species richness (Fleischner 1994), resulting in the dominance of species that are less palatable and more resilient to grazing pressures.

The movement of livestock along the riparian zone results in the compaction of soil and removal of ground cover, reducing soil permeability and increasing surface run-off (Belsky, *et al.*, 1999). These factors lead to increased erosion, sediment delivery to streams and reduced fertility of riparian soils (Sovell, *et al.*, 2000; Belsky, *et al.*, 1999). The increased runoff rates and erosion resulting from livestock movement leads to larger more intense flood events (Belsky, *et al.*, 1999), amplifying the erosion and sedimentation. Costin (1980) showed that runoff rates and soil loss are inversely proportional to percentage groundcover.

The effects of livestock on riparian zones are well documented and research has consistently identified that livestock have major negative impacts on the vegetation and soils of the riparian zone (Trimble & Mendel 1995; Robertson & Rowling 2000; Jansen & Robertson 2001). Studies have identified that livestock grazing can result in a reduction in groundcover, abundance of woody debris and leaf litter (Robertson & Rowling 2000). Jansen and Robertson (2001) identified a reduction in riparian condition associated with increased stocking rates and Trimble and Mendel (1995) identified changes in geomorphological condition resulting from livestock grazing.

In addition to the impacts directly associated with livestock grazing there are often indirect impacts on the riparian zone resulting from land clearing, pasture improvement and weed invasion. These impacts will be discussed briefly in the following sections, as they have influenced the current condition of the riparian zone of south east Australia and would likely be a factor in the outcomes of riparian management.

1.4.2 Land clearing

Clearing for grazing land has accounted for more than 50% of the total land cleared in Australia (Barson, *et al.*, 2000). Land clearing is a contributing factor to rising saline water tables (dryland salinity) in some areas (Lambers 2003) and increased surface runoff rates (Siriwardena, *et al.*, 2006). Robertson and Rowling (2000) identified that in addition to grazing by livestock, clearing of trees on floodplains and riverbanks is potentially the most significant factor that influences the structure of riparian vegetation communities.

1.4.3 Pasture improvement

To provide adequate feed for livestock, pastoralists of southern Australia improved their pastures by introducing exotic grasses and using fertilisers such as superphosphate (Starr, *et al.*, 1999). Exotic grasses have become widespread, supported by the use of nutrient rich fertilisers which have given them a competitive edge particularly in the riparian zone, with the availability of moisture (McIver & Starr 2001).

1.4.4 Invasion of the riparian zone by weeds

The riparian zone is prone to invasion by weed species (Planty-Tabacchi, *et al.*, 1996), because of factors such as the transport of propagules via water movement, flooding disturbance and water availability (Naiman, *et al.*, 2010; Ede, *et al.*, 2010). Plant invasions are increased by human induced disturbances to the riparian zone (Richardson, *et al.*, 2007) such as land clearing, livestock grazing and river regulation (Ede, *et al.*, 2010). Livestock aid in the spread and establishment of exotic species; by spreading seeds in their faeces and on their fur, reducing competition by native species by selective foraging and by creating open areas which provides opportunity for weed species to colonise (Fleischner 1994).

Weeds adversely affect normal ecological function of the riparian zone. For example; certain weed species aggressively colonise banks due to their resistance to water flow, which can cause issues such as channel narrowing (Tickner, *et al.*, 2001). The high water consumption requirements of some exotic species can lead to a reduction in local water availability for the native species (Tickner, *et al.*, 2001). Spooner and Briggs (2008) found that recruitment of native seedlings negatively correlated with cover of exotic annual grasses and forbs.

1.4.5 Flow manipulation

The construction of dams and weirs along with the subsequent regulation of flow impacts ecosystems both upstream and downstream of the regulation point (Nilsson, *et al.*, 2005). Flow manipulation alters downstream channel form, sedimentation, (Patten 1998; Nilsson, *et al.*, 2005) and riverine seed-bank dynamics (Greet, *et al.*, 2013). As fluvial and hydrological processes are key determinants in the distribution patterns of riparian plant communities (Hupp & Osterkamp 1996), flow manipulation can result in a decline in riparian plant diversity and cover (Stromberg, *et al.*, 2007), sometimes affecting riparian vegetation for hundreds of kilometres downstream (Goodwin, *et al.*, 1997). The effects of flow modification are amplified in regions with highly variable rainfall, caused by the establishment of more consistent water regimes (Dudgeon, *et al.*, 2006). Changes in flow regimes and water extractions will likely contribute to the geographic boundaries where plant species and communities will exist (Johnston, *et al.*, 2009).

1.5 Ecological restoration

1.5.1 The need for ecological restoration

During the past 50 years, humans have changed the structure and function of natural ecosystems more rapidly and extensively than at any other time in human history (Millennium Ecosystem Assessment 2005). The growth of the world's population since

1950, the increased intensity of economic activity, and the rising per capita consumption of energy and material (Wackernagel & Rees 2013; Millennium Ecosystem Assessment 2005) has dramatically increased the demand for ecosystem services (Alcamo, *et al.*, 2005).

All ecosystems are now influenced by human activity (Vitousek, *et al.*, 1997), often resulting in substantial degradation and a loss of biodiversity (Aronson, *et al.*, 2006). Ecological restoration offers a potential solution (Dobson, *et al.*, 1997; Brudvig 2011), to restore the provision of ecosystem services or mitigate some of the implications of human land-use (Naiman & Decamps 1997). Ecological restoration was hailed as a new environmental paradigm for a healthy, mutually beneficial relationship between humans and the natural landscape (Jordan III 1994). It was described as a new strategy in conserving biological diversity by Jordan, *et al.*, (1988), who predicted that the role of restoration in conservation will be crucial as pristine areas become rarer. Ecological restoration has undergone dramatic growth as an academic discipline (Young 2000), and is an essential component of both the management of production systems and the conservation of biodiversity (Hobbs & Norton 1996).

1.5.2 The aim of ecological restoration

Ecological restoration is the process of assisting or initiating the recovery of an ecosystem that has been degraded, damaged, or destroyed (SER 2004). Ecological restoration aims to safeguard and repair nature (ecosystems and biodiversity) (van Andel & Aronson 2012), or re-establish the links between organisms and their environment (Kauffman, *et al.*, 1997). The common target of restoration is therefore a predisturbance or “natural” state (Jackson & Hobbs 2009). Thus the goal is to restore the historical features of the system (Suding, *et al.*, 2004). In Australia, the “natural” state refers to the condition pre-European colonisation (Jackson & Hobbs 2009).

It has now become apparent that the environment has drifted because of human actions and so too should the targets of restoration (Jackson & Hobbs 2009). The target of a natural state has little consideration of feedbacks between biotic and abiotic factors that may have developed in the degraded state. Attempts to maintain or restore past conditions could create ecosystems that are not adapted to current conditions and more susceptible to undesirable changes (Millar, *et al.*, 2007). This has led to some unexpected restoration outcomes (Suding, *et al.*, 2004). As a consequence there is growing recognition that restoration has to be undertaken in a context of rapid and ongoing environmental change (Hobbs, *et al.*, 2011).

It is often assumed that a return to the natural state is possible simply by removing the stresses (such as livestock), and allowing natural recovery (Aronson, *et al.*, 1993). Rutherfurd, *et al*., (2000) explains that restoration of streams and riparian areas in Australia to a pre-European condition is often impossible as it involves changing all the inputs and outputs (water quality and quantity, sediment and organisms) from upstream, downstream and the riparian zone. For these reasons restoration objectives are often targeted.

There is a variety of motivations for restoring ecosystems. These include:

- The preservation and conservation of biodiversity,
- The recovery of social values that were once provided by ecosystems,
- Human attachment to wild areas, and
- The restoration of natural capital and compensating for anthropogenic climate change (Clewell & Aronson 2006).

1.5.3 Restoration ecology of the riparian zone

Riparian restoration aims to produce self-sustaining natural processes and links between terrestrial, riparian, and aquatic ecosystems (Kauffman, *et al.*, 1997). Naiman, *et al.*, (2010) outlines several common justifications for riparian restoration; these are bank stability, habitat diversity, fish production, biodiversity, and buffering diffuse pollution.

Interest in riparian issues has grown rapidly since 1970 (Goodwin, *et al.*, 1997). In 2005, a synthesis of river and stream restoration projects implemented in USA with information gathered from 37099 restoration projects revealed that riparian management was one of the most commonly cited reasons for river restoration (26.5%) yet it was documented as a primary project goal in only 8% of projects (Bernhardt, *et al.*, 2005). These findings were supported in a similar study in Victoria, Australia, with investment in the riparian zone being the most common form of river restoration activity as well as the cheapest on a per-project basis (Brooks & Lake 2007). These findings demonstrate the importance attributed to the riparian zone in the management of river system health and function. Riparian restoration is rarely done specifically for riparian management and is seen as a solution to improve other features of the river such as water quality or bank stabilisation (Bernhardt, *et al.*, 2007). Bash and Ryan (2002) identified bank stabilisation as the most common project type in a survey on stream restoration.

In line with the move to a landscape scale approach to riparian and riverine management (Steel, *et al.,* 2010), ecological restoration of riparian zones requires consideration of the surrounding landscape. This includes an appreciation of current and past land-use, the condition of the landscape and how it has affected the riparian area (Kauffman, *et al.,* 1997). From a natural resource management perspective riparian zones should be recognised as extending into the groundwater, up above the canopy, outward across the floodplain, up the near-slopes that drain to the water and laterally into the terrestrial ecosystem (Ilhardt, *et al.,* 2000). The complex hydrological regime faced by the riparian zone means achieving effective riparian restoration is not easy and often expensive (Zedler 2000).

The importance of an intact riparian zone is acknowledged by Naiman, *et al*., (1993) who states that many of the ecological issues related to land-use could be ameliorated with the effective management of the riparian zone.

1.6 Riparian restoration

1.6.1 Background

In Australia, riparian restoration has adapted as the outcomes of past management actions have become apparent. Over the past few decades there has been a shift from government implemented river engineering programs designed primarily for flood mitigation, water resource development and erosion control to a more community based ecological approach (Brooks, *et al.,* 2006). Prior to 1990 it was common practice to clear and straighten river channels, remove native trees, and woody debris, with the aims of protecting buildings and infrastructure from inundation and to minimise the disruption to public services caused by erosion (Erskine 2001). One example is the Williams River in NSW, where between 1954 and 1986, the river channel was straightened, and willows, poplars, and privets were planted to address channel instability (Erskine 2001). This attempt at early river restoration had bio-geomorphic and ecological consequences which included bed erosion, removal of habitat and colonisation of an array of exotic species (Erskine 2001). There are now regulations on species selection for river and riparian restoration (ACT Government 2003), along with growing knowledge on suitable species selection (Webb & Erskine 2003). There is now a greater focus on appropriate species selection for river and riparian restoration (Webb &

Erskine 2003), with some states such as the ACT Government (2003) regulating in favour of it.

In riparian restoration it is accepted that the planting or regeneration of riparian vegetation will prevent or reduce river-bank failure (Hubble 2004; Docker & Hubble 2008). A study by Abernethy and Rutherford (2000) found that vegetated banks can stand nearly four metres higher than their bare counterparts. There is a general perception that the reintroduction of plants into the riparian zone will automatically increase the remaining ecosystem components (Hobbs & Norton 1996).

Riparian restoration in Australia commonly consists of actions such as the construction of fences (to exclude livestock), planting, direct seeding and assisted natural regeneration of native vegetation, along with weed reduction or removal (Brooks & Lake 2007). The costs of riparian restoration can vary significantly depending on the methods used (Schirmer & Field 2002) and different restoration regimes are appropriate under different scenarios (McIver & Starr 2001). Identifying the most appropriate action for a given scenario is vital to ensure that the project objectives are met.

1.6.2 Assisted natural regeneration (passive restoration)

Assisted natural regeneration (passive restoration) is where no seeds or seedlings are added to the site, but the remnant trees and shrubs are protected and seed stored in the seed bank is encouraged to germinate (Schirmer & Field 2002). This involves the exclusion of native/feral animals by fencing off the area (Correll 2005; Rutherfurd, *et al.*, 2000). Excluding livestock from the riparian zone has been used as a tool for restoring and maintaining water quality and hydrologic function (Brinson, *et al.*, 2002) as well as promoting vegetation, increasing bank stability (Carline & Walsh 2007), reducing bare-ground and increasing species richness (Briggs, *et al.*, 2008).

The response of riparian vegetation to exclusion can vary depending on factors such as prior adaptation of the vegetation to grazing by livestock, availability of seed sources for recruitment and the extent of degradation of the vegetation (Jansen, *et al.*, 2007) and climatic conditions (West 1993). In Australia the response of vegetation to livestock exclusion is well documented. Jansen and Robertson (2001) provide strong evidence of the benefits associated with excluding livestock from the riparian zone. The exclusion of livestock has been found to result in a shift away from disturbance-tolerant pasture species, with herbaceous riparian vegetation recovering quickly after livestock exclusion and riparian woody species recovering at a slower rate (Hough-Snee, *et al.*, 2013). Spooner and Briggs (2008) showed significantly more eucalypt trees and shrub cover in fenced areas than unfenced. Lunt, *et al.*, (2007) showed that livestock exclusion increases the richness, cover and composition of herbaceous plant communities in a riparian forest. The establishment of plant species in passive restoration is largely determined by their occurrence in the surroundings and the presence of their seed in the seed bank (Prach & Hobbs 2008).

Passive restoration is the cheapest form of restoration in Australia (Schirmer & Field 2002), and when successful ensures that there will not be an artificial quality to the composition or spatial configuration of the vegetation (Middleton 1999). The revegetation of the riparian zone following restoration can occur spontaneously but in other cases may require intervention (Middleton 1999), such as the inclusion of vegetation.

1.6.3 Active restoration methods

Typically active riparian restoration consists of the inclusion of grasses, shrubs and trees between the normal bank-full level and the actively farmed land (Anbumozhi, *et al.*, 2005). It is often used to accelerate and influence the successional trajectory of recovery (Holl & Aide 2011). Active restoration is done by either direct seeding the site or planting tubestock (Rutherfurd, *et al.*, 2000). Of these, planting tubestock is the most frequently used active restoration method (56%) in Australia, followed by direct seeding (31%) and a combination of both techniques (13%) (Ruiz-Jaen & Mitchell Aide 2005).

Active riparian restoration is appropriate in riparian areas that are not likely to experience natural regrowth, although on occasion active methods are misused in situations where either the natural vegetation is capable of coming back or where plantings cannot survive (Briggs 1996).

1.6.3.1 Revegetation using tubestock

Revegetation of the riparian zone using tubestock has been found to offer higher survival rates than direct seeding and an increased ability to plan the final density of plants than direct seeding (Young & Evans 2000; Schirmer & Field 2002). From a social perspective the use of tubestock is often preferred as stakeholders like to see large plants as soon as possible (Young and Evans 2001).

1.6.3.2 Revegetation using direct seeding

Revegetation by direct seeding is generally a cheaper method of revegetation than tubestock planting (Schirmer & Field 2002). A study by Palmerlee and Young (2010) demonstrated that although direct seeding had a lower survival rate than planted tubestock, this was offset by the high costs of purchasing and planting tubestock, concluding that direct seeding is more than twice as cost effective as planting tubestock. Direct seeding results in plants with a better developed root system and no chance of the plants being pot bound or top heavy (Schirmer & Field 2002). Once established, woody species grown through direct seeding perform better than individuals transplanted from pots (Young & Evans 2000). However, the success of revegetation by direct seeding has been found to be highly variable and unpredictable (Middleton 1999).

1.6.4 Post-restoration maintenance

Riparian restoration is not simply the process of planting or establishing vegetation at a site. Once restoration is complete the ecosystem needs to be managed to ensure the on-going health of the restored ecosystem (SER 2004). Post restoration maintenance is needed to achieve restoration goals, and can comprise of maintaining fences, pest management (of varying intensity) and irrigation (Shafroth, *et al.*, 2008). The success of riparian restoration is likely to be influenced by on-going site maintenance. Shafroth, *et al*., (2008) says that the maintenance of seeded or planted vegetation for the first two growing seasons is critical. Seedling survivorship has been found to be significantly increased with the inclusion of weed control (herbicide and weed mat) (Sweeney, *et al.*, 2002). Controlled livestock grazing at certain times of the year has been recognised as an effective method for controlling certain weed species (Dorrough, *et al.*, 2004).

Given that landholders are a major stakeholder involved in riparian restoration in south east Australia, the amount of follow up site maintenance will vary depending on the level of enthusiasm for the project and the resources available to the landholder. Often landholders have to see the benefits of a restoration action to build confidence and adopt long-term commitments, if not they may decide not to pursue the project (Mendham, *et al.*, 2007).

1.7 Monitoring and assessment of riparian restoration

Livestock exclusion and revegetation has been standard practice for improving riparian condition in Australia for at least 30 years (Breckwoldt 1983). During this time millions of dollars have been invested into these activities. Given such a high level of investment and effort, the question is, have our riparian restoration efforts led to improvements in riparian and bank condition and met our long term restoration objectives?

Streambank erosion has been found to be greater in grazed areas than un-grazed areas (Kauffman, *et al.*, 1983), and livestock exclusion and the establishment of riparian vegetation has been found to effectively improve channel stability (Parkyn, *et al.*, 2003), and reduce sediment and nutrient export (Line, *et al.*, 2000), which reduces the sediment concentration in stream (Owens, *et al.*, 1996). No comparable Australian studies that monitored the geomorphological responses to riparian restoration were located. Further, there have been no studies performed specifically to compare the outcomes of different riparian restoration methods. Understanding the riparian and geomorphological responses to restoration actions is vital to identify if we are meeting our objectives, and how appropriate the restoration methods used are for a given scenario. Young & Evans (2000) and Schirmer & Field (2002) recommend that further research is done on the relative success of different restoration methods.

1.7.1 The Implementation of restoration ecology

There is a resounding call for reforms to better connect the science and practice of restoration (Dickens & Suding 2013), from both the restoration practitioners and the restoration ecologists (Cabin, *et al.*, 2010). Ideally, restoration ecologists provide ideas, guidance, and data that benefits restoration practitioners, while practitioners put the science in to practice, exchange insights with scientists and make their sites available to develop and test theories (Cabin, *et al.*, 2010). It is reported that restoration practitioners receive little input from the scientific community (Palmer, *et al.*, 1997), contributing to a lack of guidance on what type of restoration is appropriate (Roni, *et al.*, 2002). Spooner and Briggs (2008) say that a better understanding of the ecological outcomes of restoration is needed to determine the most appropriate action to take for a restoration objective.

To date most restoration projects have not been subject to objective post-project evaluation, monitoring or assessment; (Kondolf 1998; Erskine 2001; Wohl, *et al.*, 2005; Bernhardt, *et al.*, 2005). It is suggested that the current lack of monitoring and assessment has led to a lack of reliable estimates of the environmental response to different restoration actions (Allan 2004; Williams, *et al.,* 1998). Hobbs & Norton (1996) describe that restoration ecology has progressed on an *ad hoc* site and situation-specific basis, with little development of general theory or principles that would allow the transfer of methods.

From what we know of the methods used, the few studies available and general theory, it is hypothesised that there will be differences in the success of different riparian restoration methods (passive restoration, active restoration through planting tubestock and active restoration through direct seeding). This is expected to manifest in differences in riparian condition and bank condition. While the differing methods are likely to result in different outcomes, variables such as site characteristics (geology, slope, vegetation community), initial state of the site (erosion, remnant trees, vegetation cover), and (post project), and maintenance effort (watering, weeding, re-planting) are also expected to influence the outcomes of riparian restoration.

1.7.2 The need for monitoring in restoration

Monitoring in ecological restoration is done for a number of reasons. Broadly; project level monitoring can determine if restoration actions have been effective, and broad scale monitoring can assess the success of integrated restoration actions in relation to achieving biological goals (Roni & Quimby 2005).

A post-project appraisal is essentially an evaluation of the effectiveness of a restoration output (Downs & Kondolf 2002). The learning potential from post-project appraisal is increased by including a baseline survey, a period of pre-project monitoring, and post-project monitoring (Downs & Kondolf 2002). Baseline data are critical in enabling project managers to measure progress over the life of a restoration project (Bash & Ryan 2002), and evaluate project success (Kondolf 1995).

Restoration provides an opportunity for ecological research (Jordan & Gilpin 1990). From a scientific perspective a restoration project can be seen as an experiment (Kondolf 1995), conducted to ensure that maximum information can be gained to improve future efforts regardless of project outcome (Bernhardt, *et al.,* 2007). A failed project could be more valuable than a successful one if the reasons for failure are understood and the information used so the mistake does not happen again (Kondolf 1995).

Project monitoring and evaluation can detect flaws in project design and enable adaptive management (Woolsey, *et al.,* 2007) and ultimately improve the management of the resource. Most riparian restoration plans now contain at least reference to the need for an adaptive approach (Walters 1997). Adaptive management is a structured process of experiential learning (Walters 1997), grounded in the admission that humans know how to manage ecosystems in some way; but just don't necessarily do it well (Lee 1999). Adaptive management acknowledges that managed resources will change as a result of human intervention and that flexibility is required to adapt to these changes (Gunderson 1999). This adaptive approach requires on-going project monitoring to track changes in a system in response to an action.

1.7.3 Determining Successful Restoration

It is often assumed that restoration projects are beneficial (Kondolf 1998), and there is a predisposition to regard restoration as good (Kondolf 1995). This assumption is often supported by reporting positive outputs such as the number of plants planted or kilometres of river restored as well as the secondary benefits of restoration such as the social aspects (community engagement, education and awareness). As with any public investment there is a need to report the return on investment. Empirical assessments of restoration success are critical to justify the inclusion of ecological restoration in natural resource policies (Wortley, *et al.,* 2013). If success cannot be proven, there is a great risk that public support for restoration projects will decline (Woolsey, *et al.,* 2007).

While there is global agreement of the importance of restoration, there is extensive debate around what characterises successful restoration (Palmer, *et al.,* 2005; Wortley, *et al.,* 2013). Defining success often depends on perspective, goals and time (Zedler 2007). Hobbs and Harris (2001) and Zedler (2007) recommend avoiding success or failure as descriptors and using adequate measures of progress toward agreed restoration goals. Higgs (1997) argues that successful restoration is an ecologically effective restoration accomplished in the least amount of time with the least input of labour, resources and materials. Bullock, *et al.,* (2011) discusses that success could be considered based on the economic benefits attributed by the increased ecosystem services recovered through restoration.

The Society for Ecological Restoration (SER) international primer on ecological restoration (2004) states that an ecosystem has recovered once it contains sufficient biotic and abiotic resources to continue development without further assistance. The SER (2004) contains a list of nine attributes of restored ecosystems that can form a basis for determining when restoration has been accomplished, these are:

1. The restored ecosystem contains a characteristic assemblage of the species that occur in the reference ecosystem and that provide appropriate community structure.
2. The restored ecosystem consists of indigenous species to the greatest practicable extent.
3. All functional groups necessary for the continued development and/or stability of the restored ecosystem are represented or have the potential to colonize by natural means.
4. The physical environment of the restored ecosystem is capable of sustaining reproducing populations of the species necessary for its continued stability.
5. The restored ecosystem apparently functions normally for its ecological stage of development, and signs of dysfunction are absent.
6. The restored ecosystem is suitably integrated into a larger ecological matrix or landscape, with which it interacts through abiotic and biotic flows and exchanges.
7. Potential threats to the health and integrity of the restored ecosystem from the surrounding landscape have been eliminated or reduced as much as possible.
8. The restored ecosystem is sufficiently resilient to endure the normal periodic stress events in the local environment that serve to maintain the integrity of the ecosystem.
9. The restored ecosystem is self-sustaining to the same degree as its reference ecosystem, and has the potential to persist indefinitely under existing conditions.

Restoration which is deemed a success should not be assumed to be an ecological success, as there are often economic or social objectives driving restoration projects (Palmer, *et al.*, 2005). For this reason Palmer, *et al.*, (2005) proposed five criteria for ecological success, referred to as the standards for ecologically successful river restoration. These are:

1. Articulation of a guiding image of dynamic state, using historical information such as aerial photographs, ground photographs, maps and biological survey records to establish prior conditions. Or using an undisturbed or already recovered reference site to frame restoration goals or the use of stream classification systems.
2. Ecosystems are improved: there are measureable changes or demonstrated improvements in physicochemical and biological components of the agreed upon guiding image.
3. Resilience is increased: ecologically successful river restoration should allow the restored river to be a resilient self-sustainable system.
4. No lasting harm: ecologically successful restoration should be done with minimal long term impacts caused.
5. Ecological assessment is completed: undertake monitoring and evaluation, regardless of success assessments should be shared to further knowledge.

Guides to effective restoration such as the SER (2004) nine attributes of restored ecosystems and the criteria for ecological success (Palmer, *et al.*, 2005) are important yet only provide broad overarching statements and as such are fairly unusable by restoration practitioners. Zedler (2007) describes that terms such as 'ecosystems are improved and 'do not harm' defined by Palmer (2005) provide no objective measurement. This observation is one of the reasons why environmental condition assessments are performed which use indicators to reliably estimate the current condition of a site.

1.7.4 Assessment of ecological condition

A key component of any restoration activity is the (pre and post restoration) assessment of ecological integrity (Innis, *et al.*, 2000). Ecological monitoring generally refers to sampling something in an effort to detect a change in a physical, chemical, or biological parameter (Roni & Quimby 2005). Ecological monitoring is seen as the first step in restoring an ecosystem (Karr & Chu 1998), and provides a solid foundation for making resource and management decisions (Naiman, *et al.*, 2010). Riparian condition assessments provide a simple and effective way to monitor the condition of the riparian zone. Riparian condition assessments along with all biological assessments use indicators to describe environmental condition. Indicators are physical, chemical, biological or socio-economic measures that best represent the key elements of a complex ecosystem or environmental issue (Fairweather &

Napier 1998). Indicators estimate the condition of ecological resources, magnitude of stress, exposure of a biological component to stress, or the amount of change in condition (Breckenridge, *et al.*, 1995). Effective indicators focus on the attributes of living systems that give the clearest signals of human impacts (Karr & Chu 1998) and should be the minimum set of indicators that will provide rigorous data (Fairweather & Napier 1998). The use of indicators in biological condition assessments relies on the assumption that a few simple measurements can say something about the condition that will aid scientific understanding and management decisions (Norris & Hawkins 2000). For this reason indicators must be put through a rigorous evaluation process.

Worldwide there are numerous examples of methods for assessing the ecological integrity of the riparian zone. The Riparian Forest Quality (QBR) was developed in the Mediterranean (Munné, *et al.*, 2003), the Riparian, Channel, and Environment (RCE) Inventory was developed in Sweden (Petersen 1992), and the Riparian Evaluation and Site Assessment (RESA) method (Fry, *et al.*, 1994) in Arizona, USA. In Australia, the Index of Stream Condition (ISC) (Ladson, *et al.*, 1999) was developed in Victoria for waterway management and the Tasmanian River Condition Index is a framework developed for assessing Tasmanian river condition (NRM South 2009). Jansen and Robertson (2001) developed a rapid appraisal index of the ecological condition of floodplain riparian habitats, as part of a study on the effects of livestock on riparian habitats in New South Wales (NSW) Australia. Jansen and Robertson's (2001) riparian condition assessment was later developed in to the Rapid Appraisal of Riparian Condition (RARC) in 2004 (Jansen 2004), and is widely used in NSW.

An alternative approach to river assessment is an assessment of geomorphological condition. This approach will usually focus on stream channels rather than the adjacent banks, and provide useful information on the condition of the drainage-line and bank (Naiman, *et al.*, 2010). As the aim of riparian restoration is often to reduce erosion and increase bank stability, the use of a geomorphic assessment can be very useful in gauging project outcomes. Rosgen's (1997) method attempts to identify the morphological features of a river's stable state to identify the best long-term stabilisation/ restoration management option for a degraded river.

Using indicators determined by quality ecological condition assessments, are useful for setting objectives and planning restoration actions and likely to lead to an efficient increase in riparian and bank condition. For example the RARC (Jansen 2004) assessment tool, scores sites according to attributes such as the width of the riparian zone (points increasing with riparian width), vegetation cover, (canopy, understorey and ground cover) and presence of debris, tussock grass and reeds. Basing project actions on increasing these attributes will likely result in an increased RARC score, indicating improved riparian condition.

1.7.5 The current lack of monitoring in ecological restoration

The importance of monitoring and assessment in restoration is unquestioned (Aguiar, *et al.*, 2011) yet there is a reported lack of monitoring or evaluation of performance or outcome of restoration projects (Wohl, *et al.*, 2005; Bernhardt, *et al.*, 2005; Kondolf & Micheli 1995; Erskine 2001; Ewel, *et al.*, 2001). Bernhardt (2005) undertook a study looking at the common elements of a successful river restoration in the USA. Of the 37,099 river restoration projects assessed only 10% of project records indicated that any form of assessment or monitoring occurred. The current lack of monitoring is often a result of poor planning and lack of allocated funds (Purcell, *et al.*, 2002). In a study by Bash and Ryan (2002) respondents were asked to identify barriers to monitoring or evaluation of their project, with funding (34%) being the most common reason. Smokorowski, *et al.*, (1998) reported a lack of information by which to value the expenditure for restoration projects; identifying a need for improvements in assessments, monitoring and reporting. When monitoring does take place, quantifiable success criteria are rarely defined (Golet, *et al.*, 2008). A study by Bernhardt, *et al.*, (2007) reported that nearly half (173/317) the restoration project managers interviewed stated that their project was a success, yet 29 of these projects simply restated project design plans e.g. “plant 700 trees” and 95 project objectives were nonquantitative e.g. “establish a natural channel”.

There are examples of monitoring that have taken place on riparian restoration projects that have deemed the project successful (Rood, *et al.*, 2003; Carline & Walsh 2007; Purcell, *et al.*, 2002; Parkyn, *et al.*, 2003). However there seems to be a lack of documentation of unsuccessful river and riparian restoration projects (Watts & Wilson 2004). Zedler (2007) reviewed literature on ecological restoration and found 116 papers that used the term ‘success’ to describe a restoration outcome and only 10 papers that used ‘failure’. This biased representation of restoration outcomes may, in part, be because of the benefits of positive judgement (Zedler 2007), and an unwillingness to acknowledge failure because of the stigma attached to a failed project (Kondolf 1995). Scientists need to be objective when evaluating restoration outcomes (Zedler 2007).

Riparian restoration can take a relatively long time to be ecologically effective (such as the time it takes to create debris, habitat or canopy cover), therefore project monitoring requires a long-term commitment (Bash & Ryan 2002). Kondolf (1995) recommends 10 years as a reasonable timeframe, and Klein, *et al.,* (2007) estimates that a few decades are needed. As riparian restoration has occurred (in some form) for some time (>30 years) there is an apparent opportunity for long term monitoring, despite this, long-term monitoring of riparian restoration is rare (Kondolf 1995).

Restoration ecologists have been criticised for failing to effectively communicate their work to non-scientists (Cabin, *et al.,* 2010). Many of the rapid biological assessments are designed to be done quickly and with little training. In most cases project monitoring and assessment is done by the restoration practitioners themselves, and although in theory they should be armed with the knowledge, funding and facilities to undertake monitoring effectively this often isn't the case. Frequently, practitioners have not got access to the necessary resources, or another reason maybe that not enough emphasis is placed on monitoring (due to a lack of funding or a lack of apparent importance during project planning).

1.8 Summary

Understanding the outcomes of restoration actions allows restoration actions to be refined and targeted. The science and practice of restoration can be significantly improved by greater assessment of ecological effectiveness (Palmer, *et al.,* 2007) and it is this premise that underpins the purpose of this report.

1.9 Aims and Objectives

Overall project aim:

The main aim of this study was to evaluate the outcomes of common riparian restoration methods.

Specific research objectives

The specific research objectives of this study were:

1. To determine how effective common riparian restoration methods are at:
   a) Enhancing and protecting riparian vegetation.
   b) Reducing stream bank erosion.
1. To determine how different riparian restoration methods influence different features of the riparian zone.
2. To determine the factors that affect the outcomes of riparian restoration.

A priori null hypotheses of study

1. Riparian restoration has not produced improvements in riparian condition.
2. Riparian restoration has not produced improvements in bank condition.

2. Methods

2.1 Study area

2.1.1 The Upper Murrumbidgee Catchment

The present study was conducted in the Upper Murrumbidgee Catchment which is located in the Southern Tablelands of New South Wales (NSW) and the Australian Capital Territory (ACT) (Olley & Wasson 2003). The Upper Murrumbidgee Catchment area is about 14,087 sq km (Gilmore 2008) with 13,144 sq km draining in to the Murrumbidgee River (NSW Department of Land and Water Conservation 1999) (DLWC). The Upper Murrumbidgee River flows from its headwaters above Tantangara Dam in the south west of the catchment (Gilmore 2008), through the alpine region of Kosciusko National Park and the Monaro High Plains in New South Wales (NSW), then through the ACT (MDBA 2014), before reaching Burrinjuck Reservoir to the north-west of the catchment which divides the upper and mid Murrumbidgee Catchments (figure 2.1) (Gilmore 2008; Olley & Wasson 2003). The Murrumbidgee River is one of Australia's largest inland rivers and forms part of the Murray Darling Basin (Gilmore 2008).

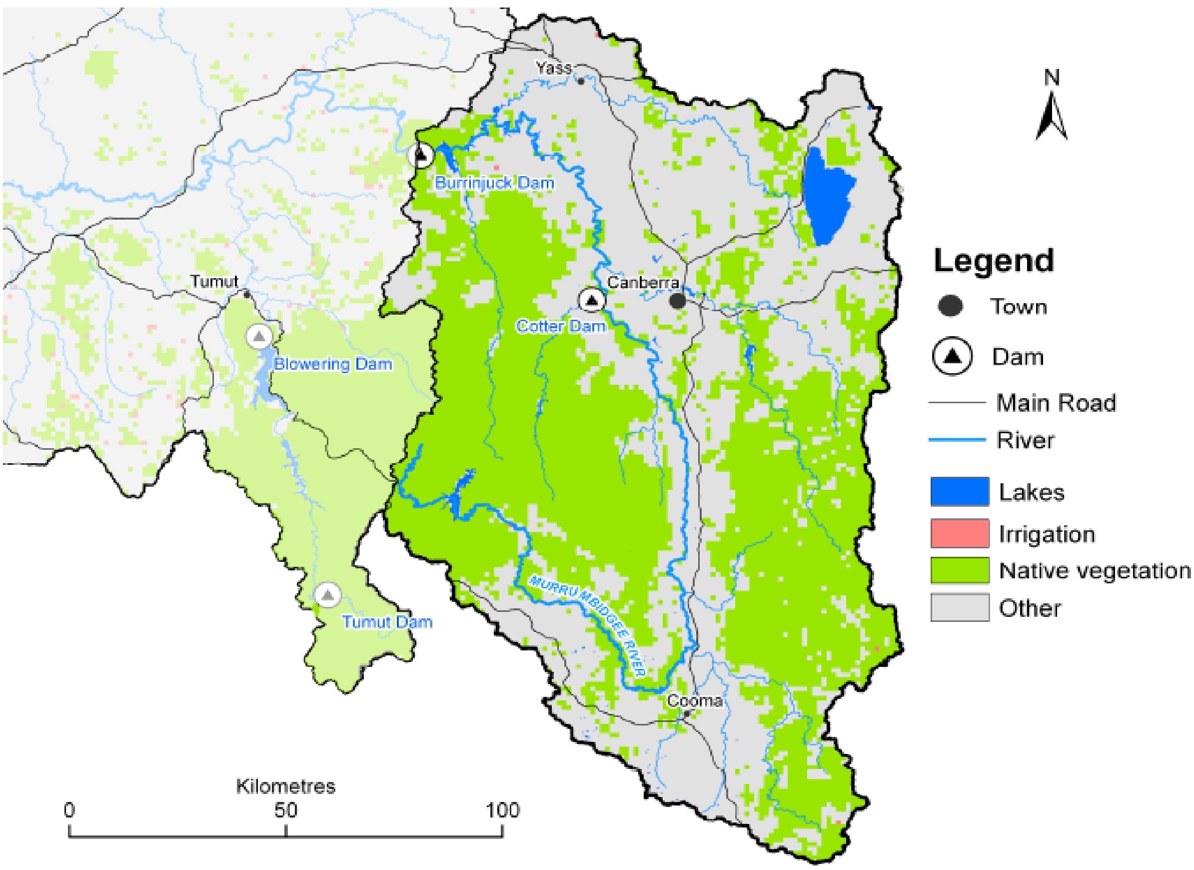


Figure 2.1: The Upper Murrumbidgee Catchment (Gilmore 2008).

Climate in the Upper Murrumbidgee Catchment is temperate with cold winters and hot dry summers. There is significant variation in the climate of the Upper Murrumbidgee Catchment (DLWC 1999). Rainfall varies across the catchment, with mean annual rainfall ranging from around 500 mm/year to more than 1000 mm/year (Gilmore 2008). Generally there is an increase in rainfall from west to east of the catchment (DLWC 1999), and greater rainfall in areas with higher elevation (Edwards & Johnston 1978).

The vegetation community assemblages and their distribution along the Murrumbidgee River within the Upper Murrumbidgee Catchment are diverse (Johnston, *et al.*, 2009). Uncleared areas are commonly dominated by She-oak Tableland Riparian Woodland and remnant Ribbon Gum Tableland Riparian Woodland (Johnston, *et al.*, 2009). Common plant species include: *Eucalyptus blakelyi*, *Eucalyptus melliodora* on lower slopes, *Eucalyptus macrorhycha*, *Eucalyptus rossii*, and *Eucalyptus nortonii* on ridges and areas of *Eucalyptus bridgesiana* and *Eucalyptus mannifera* (Armstrong, *et al.*, 2011).

2.1.2 Land-use in the Upper Murrumbidgee

Land-use in the Upper Murrumbidgee Catchment has predominantly been pastoral, with grazing being the dominant activity (Starr, *et al.*, 1999). Livestock grazing has occurred in the Upper Murrumbidgee Catchment since European arrival in the 1820’s, and increased rapidly as settlers secured land (Starr, *et al.*, 1999). The current condition of much of the riparian zone of the Upper Murrumbidgee Catchment is attributed to past land management (Starr, *et al.*, 1999), with farming practices having major impacts on riparian structure and function (Robertson & Rowling 2000). Currently, large areas of the Murrumbidgee River and its tributaries have less than 40% tree cover along the riparian zone, with low levels of riparian vegetation contributing to high rates of predicted bank erosion (Wilkinson, *et al.*, 2004). Most of the Murrumbidgee River flows through private property between Gundagai and Hay and the riparian zone has been assessed as being in very poor condition (Jansen & Robertson 2001). Similarly; Johnson, *et al.*, (2009) identified that the structural integrity of the riparian zone within the Upper Murrumbidgee Catchment has been compromised in many ways.

The level of impact of land-use is broadly related to stocking rates and stock management, along with other factors such as the rate and extent of vegetation clearing, the development of improved pastures, fertiliser application and invasive species (Starr, *et al.*, 1999). Past land-use has resulted in changes to erosion rates, salinity, and stream-channel widening (Olley & Wasson 2003; Starr*, et al.,* 1999). On-site implications include loss of soil and soil structure, changes to drainage patterns, loss of vegetation, decreased land productivity and reduced property value (Starr*, et al.,* 1999). Offsite implications (mainly downstream) include increased sedimentation in streams and water storages, reduced water quality and increased turbidity and nutrients (Starr*, et al.,* 1999).

Water quality issues across the Murrumbidgee Catchment were assessed and ranked as part of the 'Stressed Rivers' assessment report for the Murrumbidgee Catchment by the then Department of Land and Water Conservation (DLWC 1999). This assessment also provided a strategic focus for managing the identified water quality issues. Stressed Rivers (DLWC 1999) identified that there were high amounts of particulate phosphorus in the Burrinjuck Reservoir which stimulates the growth of algae. The majority of the source of phosphorus (82%) in the Reservoir (Starr 2000) was recognised as originating predominantly from subsoils, which were becoming mobile through gully and stream-channel erosion in the Upper and Mid Murrumbidgee Catchments (Starr, *et al.*, 1999; Olley & Wasson 2003). A series of sites in the Upper and Mid Murrumbidgee catchments were identified (Starr 2000) as the source of sediment and nutrient in the catchment. These sites included:

- Sites of active streambank erosion.
- Active bed and wall erosion of connected tributary gullies.
- Beds of valley floor gullies and streams where deposition and vegetation entrapment of fine sediment is occurring or may occur as the result of channel form, gradient and hydrologic regimes.

2.1.3 The Bidgee Banks Restoration Project

Based on the assessment undertaken by the DLWC (1999), and the work of Starr, *et al.*, (1999) and Starr (2000), a large scale community based river restoration project was initiated in 2000, here after referred by its project name The Bidgee Banks Restoration Project.

The Bidgee Banks Restoration Project aimed to address deteriorating water quality and vegetation loss in the Murrumbidgee Catchment by protecting and rehabilitating degraded riparian areas. The two aims of the Bidgee Banks Restoration Project were to reduce the delivery of sediment and nutrient in to the Murrumbidgee River through erosion control and to protect and enhance native riparian vegetation. The Bidgee Banks Restoration Project involved a relatively new approach to natural resource management as it focused solely on the investment in riparian areas. The Bidgee Banks Restoration Project was undertaken as a partnership between the community, Greening Australia ACT, Greening Australia South East NSW, the NSW Department of Land and Water Conservation (DLWC) and private land-holders (BBPSC 2003).

To deliver the Bidgee Banks Restoration Project; 230 sub-projects were undertaken across 262 properties in the Upper and Mid Murrumbidgee Catchments (Figure 2.3), 104 projects within the Upper Murrumbidgee Catchment and 126 in the Mid Murrumbidgee Catchment. The Bidgee Banks Restoration Project used a range of riparian restoration methods including: fencing of the riparian zone to exclude livestock, planting and direct seeding of native vegetation, alternative stock watering systems (such as the construction of dams), earth works and weed control (BBPSC 2003).

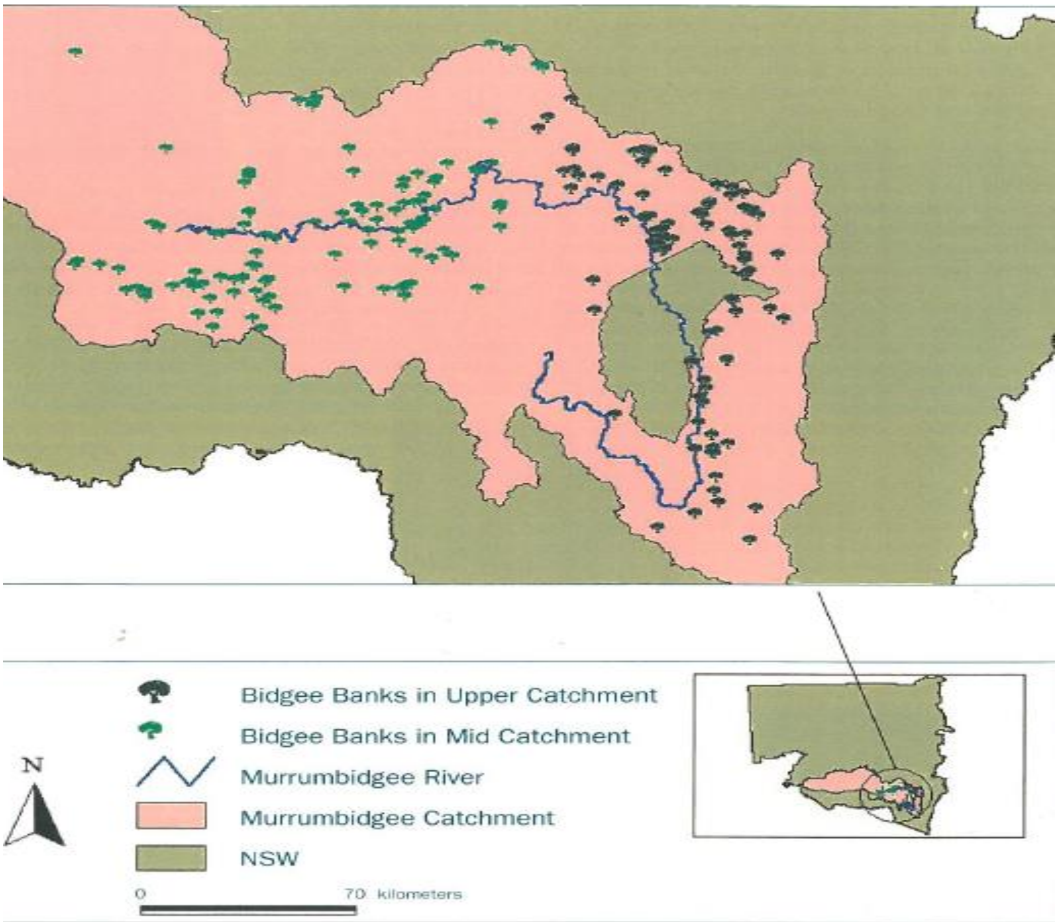


Figure 2.2: Distribution of Project Sites across the Mid and Upper Murrumbidgee Catchments undertaken through the Bidgee Banks Restoration Project (BBPSC 2003).

The final report on the Bidgee Banks Restoration Project outlined a range of outputs that were achieved during the three year project, including 1340 ha of river rehabilitated; 263 km of riparian zone fenced; 698 ha of vegetation protected and enhanced; 198,000 tubestock planted and 215 km of direct seeding (BBPSC 2003). The project won a Banksia award in the Bush, Land and Waterways category in 2002 and a United Nations Association of Australia award for Excellence in Land Management in 2003 (BBPSC 2003).

At least ten years has passed since the implementation of the Bidgee Banks Restoration Project. The ten year period since project implementation provided an opportunity to investigate the outcomes of the restoration actions, and identify whether the initial project objectives have been met. The large scale of the project (230 sub-projects) and time since restoration occurred (>10 years) made it a suitable opportunity to assess the effectiveness of riparian restoration, for improving bank and riparian condition.

2.2 Study design

The study involved sampling a subset of the 104 sites restored between 2000 and 2003 across the Upper Murrumbidgee Catchment as part of the Bidgee Banks Restoration Project. Of these 104 sites, four restoration methods formed the focus of the current project:

Fencing (natural passive recovery),
Fencing and revegetation using tubestock,
Fencing and revegetation using direct seeding,
Fencing and revegetation using both tubestock and direct seeding.

This study focused on the influence of riparian restoration methods on riparian and bank condition. The Rapid Appraisal of Riparian Condition as described by Jansen (2004) was undertaken to assess riparian vegetation condition and individual riparian attributes. The erosion state and bank condition of each site was assessed by conducting an ephemeral stream assessment as outlined by Machiori, *et al*., (2003). Factors that may influence restoration success (e.g. such as site characteristics like the presence of remnant vegetation, non-native species, and climate) and post-project management effort (e.g. grazing pressures, watering and weed removal) were also evaluated.

Several other potentially influential factors have been controlled (figure 2.3, in red), such as restoration organisation, time since restoration, and rainfall. All projects were undertaken around the same time (2000 until 2003), managed by the same organisation (Greening

Australia), and encompassed the same project objectives. The Bidgee Banks Restoration Project sites selected for monitoring were in areas with an annual rainfall of between 600 mm and 700 mm, to reduce the influences of soil moisture and rainfall (Figure 2.4).

Figure 2.3: Potential influences of riparian restoration outcomes, influential factors shown in red were controlled in the current project.

2.3 Site selection

The Bidgee Banks Restoration Project was implemented across both the mid and upper Murrumbidgee Catchments. This study examined sites in the Upper Murrumbidgee Catchment only. The main reason for this was the availability of original project information for these sites from Greening Australia. There were 104 projects undertaken on 126 properties within the Upper Murrumbidgee Catchment (figure 2.2).

It was not possible to sample all 104 restoration sites for the following reasons: (i) many sites have changed ownership in the ten year period since the Bidgee Banks Restoration Project, or the contact details of the land-holder have changed making it very difficult to gain access to the sites, (ii) the time constraints of an honours project did not allow time to sample

all 104 sites, (iii) some sites were never finished, destroyed, or altered, and (iv) some sites did not fit in to any of the four categories of restoration methods being investigated.
Site selection was initially based on availability and willingness of landholders to participate in the project. Properties were strategically selected on the basis of restoration method they received, with the view to achieving the maximum number of each restoration method as possible.

Table 2.1: Number of sites sampled within each restoration method and control sites.

Riparian Restoration Method	Number of Sites Assessed from each method
Fence Only	6
Fence and Tubestock Planting	10
Fence and Direct seeding	5
Fence, Tubestock, and Direct seeding	8
Control sites	9

Rainfall varies within the Upper Murrumbidgee Catchment, with more rain in the higher elevation areas. Sites were chosen only in areas with an annual rainfall of between 600 mm and 700 mm (defined roughly by the oval in figure 2.4), to reduce the influence of soil moisture level.

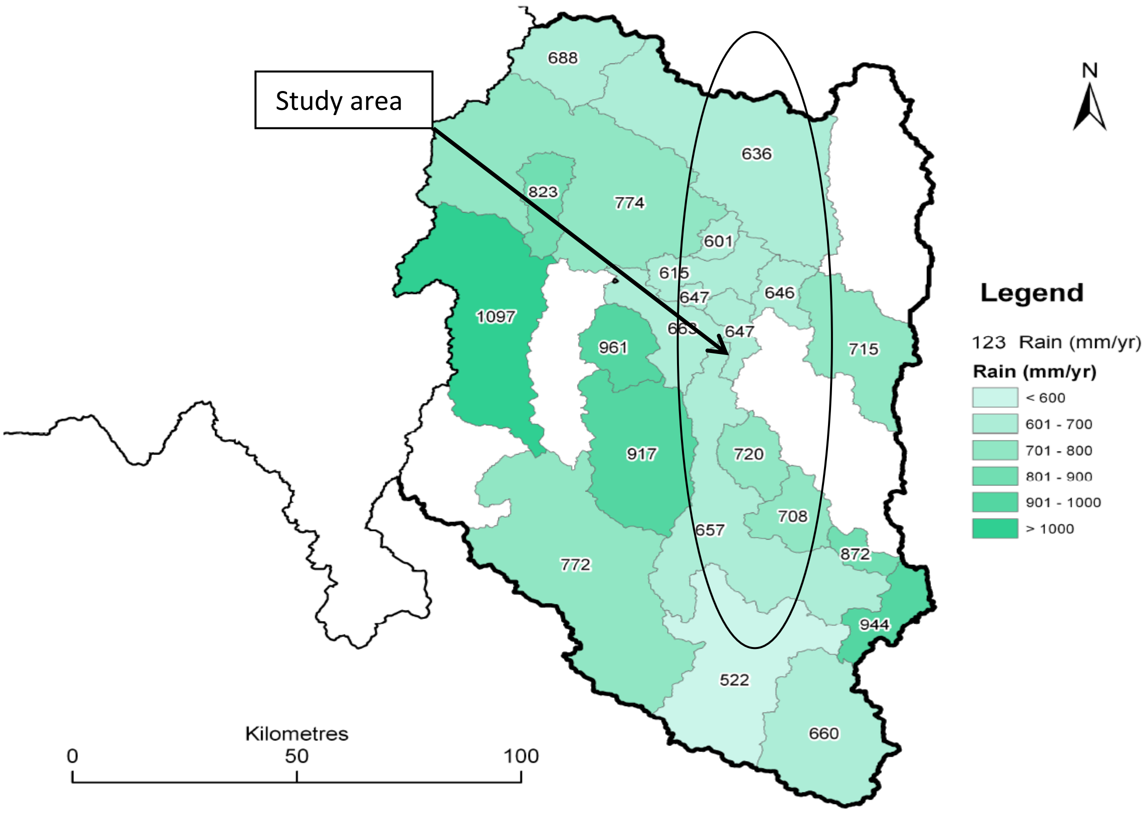


Figure 2.4: The average annual rainfall by sub-catchment in the Upper Murrumbidgee Catchment using the rainfall categories of Gilmore (2008). The area marked by the oval roughly represents the current study area which equates to an average annual rainfall of between 600-700 mm.

Control sites

Control sites were selected on properties where livestock were able to access the riparian zone and there were signs of bank and gully erosion present. The positioning of control sites were always at least 1 km away from restoration sites and were always upstream from restoration sites to reduce the influence of restoration activities on the control site characteristics. The distribution of control sites were spread over the whole sampling area (Figure 2.5). As it was unknown in many cases what type of livestock had grazed the restoration sites before exclusion occurred, the selection of control sites considered the type of livestock, to achieve a relatively even number of sites with cattle (4 sites) and sheep grazing (5 sites). As 18 out of the 29 restoration sites visited had (at least) some remnant vegetation present, this was taken in to consideration, with 5 out of the 9 control sites having some remnant vegetation present. Data were collected from these sites using the same methods as those used to collect data on the Bidgee Banks Restoration Project sites (see- Methods and Justifications). These control sites enabled restored and unrestored sites to be compared.

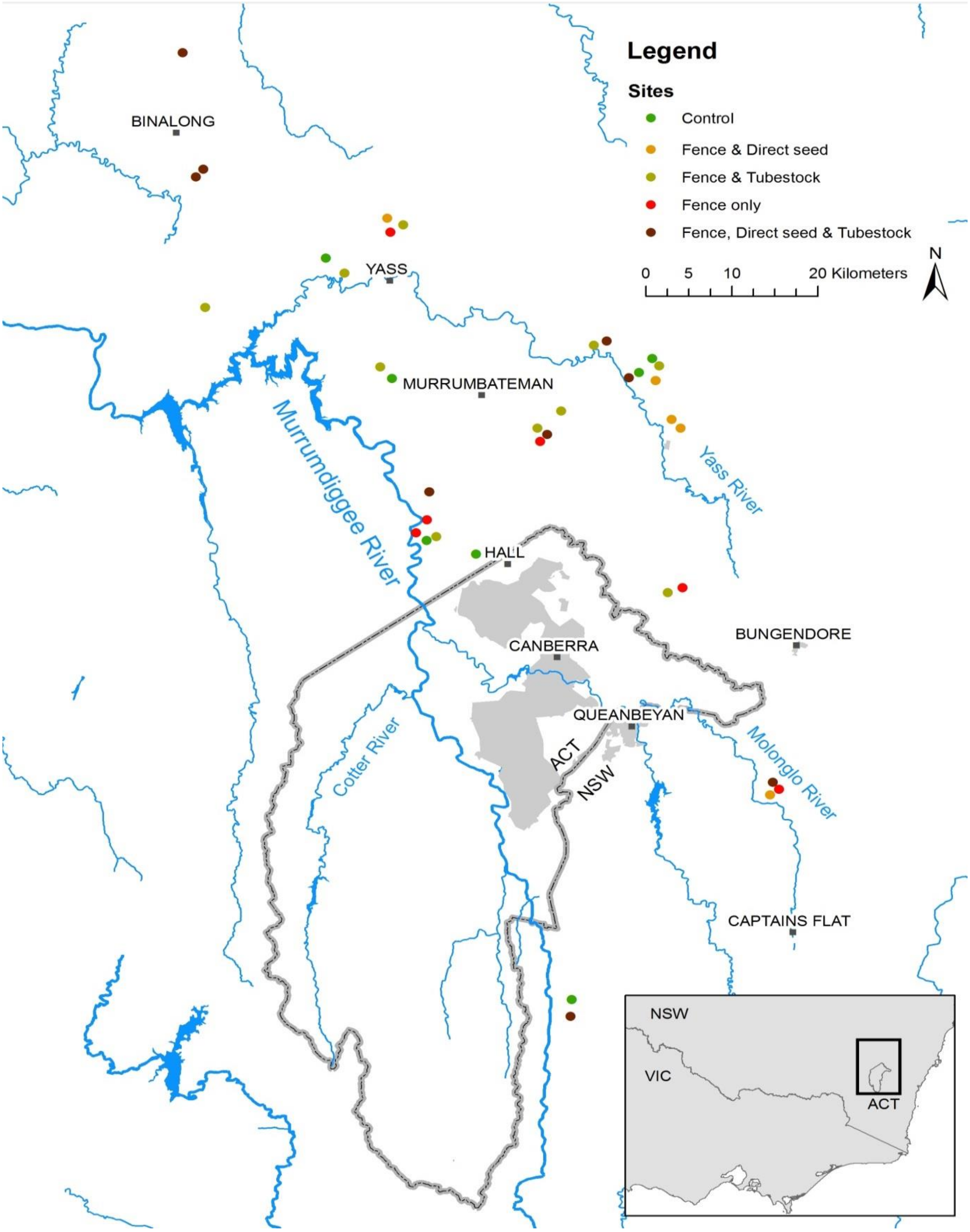


Figure 2.5: Location map of the Bidgee Banks Restoration sites and control sites assessed as part of the current study.

2.4 Methods and Justifications

The Bidgee Banks Restoration Project like many riparian restoration projects worked on the assumption that an improvement in the condition of the riparian zone would lead to an improvement in geomorphic condition and ecological function. The current study assessed both the condition of the riparian vegetation and the condition of the gully and stream-banks of sites restored as part of Bidgee Banks Restoration Project.

2.4.1 Riparian Condition Assessment

Justification

There are a range of options for assessing the condition of the riparian vegetation, with most methods designed for a specific vegetation type or location. The method chosen for this study was the Rapid Appraisal of Riparian Condition (RARC), developed by Jansen (2004). RARC was initially developed as a tool to determine the impacts of grazing management practices on riparian condition in NSW (Jansen & Robertson 2001) and has been widely used to determine the condition of riparian vegetation (Johnston, *et al.*, 2009; Jansen, *et al.*, 2007; Jansen & Robertson 2001; Wilkinson, *et al.*, 2004). The RARC is currently used by Greening Australia as part of their monitoring program to assess riparian restoration sites before and after restoration. It has previously been used as part of a survey of vegetation and habitat in key riparian zones of the Murrumbidgee River, ACT (Johnston, *et al.*, 2009) and in a study on the prioritisation of riparian areas for protection and restoration in the Murrumbidgee Catchment (Wilkinson, *et al.*, 2004). The RARC method is well suited to the study area of the current project (the Upper Murrumbidgee Catchment) and the project objectives as it was specifically designed for agricultural landscapes.

Method

The RARC method involves riparian condition assessment using indicators that reflect functional aspects of the physical, community and landscape features of the riparian zone. The RARC index is made up of five sub-indices, each with indicator variables. The five sub-indices are *habitat* (habitat continuity and extent), *cover* (groundcover, mid-storey cover, canopy cover and structural complexity), *natives* (proportion of native species within each vegetation layer), *debris* (course woody debris, hollow bearing trees, standing dead trees, and leaf litter), and *features* (reeds, tussock grasses, and seedling recruitment). The RARC method gives each variable a score, which are tallied to form a score for each sub-index and these are tallied to produce a total score for riparian vegetation condition. For this study total scores for each index were recorded as well as the cover or count of each indicator to enable each variable to be analysed separately.

At each site, transects (10 metres wide) were established perpendicular to the stream or gully every 100 metres along each individual study site. The first transect was established ten metres inside the fence that crosses the headwaters of the stream or gully. Where the site was less than 400 metres long, the first transect was placed ten metres in from the fence then another three transects were evenly spaced across the site. Where both sides of the riparian zone were treated (fenced), transects were set on both sides, where only one side was treated, only that side was assessed. For calculating all cover (%) (canopy, understorey, groundcover and leaf litter), visual estimates were conducted within 5 metre by 5 metre quadrats at ten metre intervals along each transect, stopping at the outer edge of the riparian canopy vegetation. All presence/absence and abundance data (standing dead trees, hollow-bearing logs, native species regeneration, tussock grasses and reeds) were recorded by assessing the entire transect (within the 10 metre wide span). Table 2.2 outlines the attributes recorded.

Table 2.2: Sub-indices and indicators from the Rapid Appraisal of Riparian Condition, the range and method of scoring each indicator, and the maximum possible total for each sub-index (Jansen 2005).

Sub-index	Indicator	Range	Method of scoring	Total
Habitat				11
	Longitudinal continuity of riparian vegetation (≥ 5 m wide)	0-4	0 = 50%, 1 = 50–64%, 2 = 65–79%, 3 = 80–94%, 4 = ≥ 95% vegetated bank; with 1/2 point subtracted for each significant discontinuity (> 50 m long)	
	Width of riparian vegetation (scored differently for channels < or ≥ 10 m wide)	0 - 4	Channel ≤ 10 m wide: 0 = VW , 5 m, 1 = VW 5–9 m , 2 = VW 10–29m 3 = VW 30–39 m, 4 = VW ≥ 40 m Channel > 10 m wide: 0 = VW/CW 0.5, 1 = VW/CW 0.5–0.9, 2 = VW/CW 1–1.9, 3 = VW/CW 2–3.9, 4 = VW/CW 4, where CW = channel width and VW = vegetation width	
	Proximity to nearest patch of	0 - 3	0 = > 1 km, 1 = 200 m -1 Km, 2 = contiguous, 3 = contiguous with patch > 50	

	intact native vegetation > 10 ha		ha	
Cover				12
	Canopy (> 5 m tall)	0 - 3	0 = absent, 1 = 1-30%, 2 = 31-60%, 3= >60% cover	
	Understorey (1–5 m tall)	0 – 3	0 = absent, 1 = 1-5%, 2 = 6-30%, 3= >30% cover	
	Ground (< 1 m tall)	0 – 3	0 = absent, 1 = 1-30%, 2 = 31-60%, 3= >60% cover	
	Number of layers	0 – 3	0 = no vegetation layers to 3 = ground cover, understorey and canopy layers	
Natives				9
	Canopy (> 5 m tall)	0 - 3	0 = absent, 1 = 1-30%, 2 = 31-60%, 3= >60% cover	
	Understorey (1–5 m tall)	0 – 3	0 = absent, 1 = 1-5%, 2 = 6-30%, 3= >30% cover	
	Ground (< 1 m tall)	0 – 3	0 = absent, 1 = 1-30%, 2 = 31-60%, 3= >60% cover	
Debris				10
	Leaf litter	0 – 3	0 = absent, 1 = 1-30%, 2 = 31-60%, 3= >60% cover	
	Native leaf litter	0 - 3	0 = absent, 1 = 1-30%, 2 = 31-60%, 3= >60% cover	
	Standing dead trees (> 20 cm dbh)	0 – 1	0 = absent, 1 = present	
	Hollow-bearing trees	0 - 1	0 = absent, 1 = present	
	Fallen logs (> 10 cm diameter)	0 -2	0 =none, 1 = small quantities, 2 = abundant	
Features				8
	Native canopy species regeneration (< 1 m tall)	0 – 2	0 = none, 1 = scattered, 2 = abundant, with ½ point subtracted for grazing damage	
	Native understorey regeneration	0 - 2	0 = none, 1 = scattered, 2 = abundant, with ½ point subtracted for grazing damage	
	Large native tussock grasses	0 - 2	0 = none, 1 = scattered, 2 = abundant	
	Reeds	0 - 2	0 = none, 1 = scattered, 2 = abundant	

2.4.2 Gully and stream-bank erosion

Justification

Gully and streambank erosion were assessed using the ephemeral stream assessment (Machiori, *et al.,* 2003). This method is appropriate for this study as it estimates bank stability as an indicator of erosion activity, and can assist in identifying the cause of the erosion.

Method

Eight visual indicators were used to assess the geomorphic condition of the drainage-line within each study site. Transects were established every 25 to 100 metres over the entire site, with a minimum of eight transects established at each site. Where both sides of the stream were treated (restored) both sides were assessed, where only one side was treated only that side was assessed. The indicators produce a rating for each assessment that ranges from very actively eroding through to very stable. The indicators used are shown in Table 2.4.

Table 2.3: Indicators from the Ephemeral Stream Assessment, the range for each indicator, and method of scoring each indicator (Machiori, *et al*., 2003).

Indicators	Range	Method of scoring
Vegetation on the drainage-line floor	1 - 3	1 = little or no vegetation growing on drainage line floor. 2 = Any vegetation present is annual or short lived: partial burial of plants by recent deposited sediment evident. 3 = Dense perennial plant cover, similar to vegetation on the bank of the drainage line: characteristic wetland species composition.
Vegetation on the drainage-line walls	1 - 3	1 = little or no vegetation growing on drainage-line floor. 2 = any vegetation present is annual or short lived: partial burial of plants by recent deposited sediment evident. 3 = Dense perennial plant cover, similar to vegetation on the bank of the drainage-line: characteristic wetland species composition.
Shape and aspect ratio of drainage-line cross-section	1 - 5	1 = very actively eroding: caving, mass wasting and/ or tunnelling present, 2 = Actively eroding: slight undercutting, near vertical walls, alluvial fans also eroding. 3 = Potentially stabilising: side walls become rounded and crusted alluvial fan. 4 = Stabilising: wall angle less than 65˚, small inactive alluvial fan at foot of side walls.

		5 = Stable: gently sloping walls, generally low, “S” shaped bed width > depth.
Longitudinal morphology of drainage line	1 - 4	1 = Currently incising bed in pre-existing loose sediment. 2 = Flat continuous, loose sediment with signs of recent/ frequent movement. 3 = Flat with a cohesive fine textured “soil-like” bed. 4 = Non-cascading pools or ponds, with non-slaking, non-dispersive clay base implying low energy.
Particle size of materials on drainage line floor – material availbe for erosion	1 - 3	1 = Material on floor is similar or smaller in particle size and/or density than material in the walls. 2 = Material on floor is slightly larger in particle size and/or denser than material on walls. 3 = Material on floor is much larger in particle size and/or denser than material on walls.
Nature of drainage line materials	1 - 4	1 = Dispersive material is exposed for greater than 1 m of wall height. 2 = Materials that slake rapidly are exposed on greater than 0.3 m and less than 1 m. 3 = Materials that slake and/or disperse are exposed on less than 0.3 m of wall height. 4 = Materials that do not slake or disperse are exposed on wall surface.
Shape of stream-bordering flats and/or slopes	1 - 5	1 = very steep slope, >30˚ creating high velocity flows. 2 = steep bank, 10 -30˚, permitting moderate to high velocity flows. 3 = Moderately sloped bank, 5-10˚. 4 = Gently sloped bank/floodplain, laterally extensive. <5˚, 5 = Flat bank/floodplain, laterally extensive.
Nature of lateral flow regulation into drainage line	1 - 5	1 = Side arm channel inflow: very high inflow rates. 2= bare bank, laterally extensive. 3 = Sparse grassland/woodland with bare soil bank lip: moderate flow rate, some highly focused inflow locations. 4 = Dense grassland: inflow rate, mostly diffuse. 5 = Woodland with dense litter: very low, diffuse inflow rate.

2.4.3 Site attributes

Site attributes were documented at each site from visual observations. Site attributes include the following:

- The abundance of remnant vegetation (0 =none, 1= scattered, 2 = abundant), and the identification of remnant species present,
- Identification of weed species present,

- Identification of all planted or direct seeded plants present at each site (through on site identification, photo records, and taking samples of each species).
- Document and identify all seedlings (< 1 m tall) of any native species present.

2.4.4 Post project site management

The level of post-project site management and actions undertaken by individual landholders were assessed through discussions with landholders and observations of each site.
These included:

- Watering effort.
- Re-planting effort.
- Weeding effort.
- Vermin control.
- Livestock access on to restoration site.

2.4.5 Field procedure

All field work was conducted in April and May 2014 to reduce any bias associated with seasonal variation. All field work was undertaken by one individual to eliminate sampling bias. Photos were taken at each transect and the surrounding area and a sketch was done of each site. A GPS was used to mark the first transect at each site, to provide points from which future research may be undertaken, and to enable the creation of a study map.

2.4.6 Materials for field work

A measuring wheel was used to mark out each transect, a clinometer was used to measure the angle of the drainage-line wall and surrounding landscape. A rangefinder was used to measure the width of the channel. Transect tape was used to outline each transect area.

2.4.7 Data collection through remote sensing

Justification

As this study did not have the benefit of base-line data taken pre-restoration, comparisons have been made comparing restored sites with control sites. In order to understand if the magnitude of difference between restored sites and controls sites provided a true reflection of the restoration outcomes, remote imagery was used to compare project site condition prior to works taking place, and the current condition of sites (ten years after restoration). Measurements included projected foliage cover and width of riparian vegetation. This enables comparisons to be made in the absence of post project monitoring. Wilkinson, *et al.*, (2004) determined that three indicators from the RARC method could be measured from remote sensing using aerial imagery: canopy cover, width of riparian vegetation and continuity of riparian vegetation. Wilkinson, *et al.*, (2004) identified that 90% of the variance of total RARC scores for the Murrumbidgee tributaries was accounted for by these three components. Jansen and Robertson (2001) showed similar results with width of riparian vegetation and longitudinal continuity scores relating strongly to overall RARC scores.

Methods

In calculating the projected foliage cover of each site from aerial imagery, image recognition software was used to increase accuracy and reduce the time taken to conduct the analysis. The image recognition software WinDIAS 3.2 was used to calculate the total projected foliage cover of each site, before restoration and 10 years after restoration. The boundary fence was located on each site from the post-restoration imagery, and this was transferred to the pre-restoration imagery. The projected foliage cover was calculated for the area within the exclusion fence at each site.

The width of riparian canopy vegetation was calculated using a digital ruler. Transects were established within the aerial images using the same protocols as for the RARC field methods (see field procedure).

2.5 Data analysis

Differences in the total riparian condition, riparian sub-indices, and individual riparian indicators were compared between sites restored using the four different restoration methods and control sites. Differences in ephemeral stream assessment scores and individual indicators of bank stability and erosion state were compared between sites restored using the four different restoration methods and control sites. Data for all variables was tested for assumptions of normality (Shapiro-wilks) and homogeneity of variance (Levene's) (Appendix 1). If the data set was normally distributed then it was tested using a factorial analysis of variance (ANOVA) in R studio. If the data set was non-normally distributed, initially a Box-Cox (1964) best fit analysis was run in R studio to identify a suitable transformation. If possible the data set was transformed to meet the assumptions of an

ANOVA, prior to conducting the ANOVA. Transformations that were used in the current study were Log transformations, Square root transformations, and Asin transformations. If transformation did not result in normality and homogeneity of variance, a Non-parametric Kruskel-Wallis analysis of variance was run in R studio. Significance levels were set at a p value of <0.05, any significant results were tested with a Tukey-Kramer multiple comparison test in R studio, to identify where the significance lied. Linear regressions were run in Microsoft Excel to compare relationships between different data sets.

3. Results

3.1 Outline of results

3.2 Riparian vegetation condition

3.3 Bank and channel condition

3.4 Exploratory analysis, potential influential factors and relationships between variables

3.1 Outline of results

The aim of this study was to evaluate the outcomes of riparian restoration methods. The Upper Murrumbidgee Catchment was an ideal site to test the study hypotheses, as the Bidgee Banks Restoration Project involved a large investment in to riparian restoration in early 2000's across a large number of properties.

To test the hypotheses outlined at the end of Chapter 1, riparian and geomorphological data were collected from 38 sites across the Upper Murrumbidgee Catchment, 10 years post restoration. This chapter presents these data in three parts designed to address each of the study objectives.

In Section 3.2 the changes in riparian vegetation characteristics as a consequence of the different restoration methods are described. This addresses changes in:

- Total riparian condition
- Structural vegetation measures
- Native/non-native composition
- Vegetation habitat and debris measures
- Seedling recruitment

The data used to investigate these changes are from field investigation and remote sensing.

In section 3.3 the second section of this chapter the changes in bank condition and erosion state as a consequence of the different restoration methods are described. This addresses changes in:

- Erosion activity
- Bank stability
- Drainage-line condition
- Lateral flow regulation

In Section 3.4 the final section explores some of the potentially influential factors, and relationships between the field variables, including:

- Remnant vegetation
- Native groundcover
- Vegetation on the drainage-line wall and floor

- Non-native species
- Diversity/species measures – survival/presence
- Influence of climatic variables
- The relative importance of riparian attributes for bank stability and erosion control

3.2 Riparian vegetation condition.

3.2.1 Total riparian condition

Ten years since restoration, restored sites are in better condition than sites that had not been restored, with the mean total riparian vegetation condition score (as determined using RARC) across all restoration sites being 21.3 ±1.3, compared to the sites without restoration (i.e. control sites) which had a mean score of 12.4 ±1.3. The riparian vegetation condition scores across all restoration sites ranged from 6.8 to 33.1 (maximum possible score = 50), while scores from the unrestored sites ranged from 5.1 to 18. All restored sites are now in better condition than unrestored sites irrespective of the restoration method employed (Figure 3.1). Despite overall results indicating improvements in riparian vegetation condition, two restoration sites had low scores (6.8 and 8.5), which implies that at these sites restoration efforts had failed.

Significant differences in the total riparian vegetation condition were observed between sites that had received different restoration methods ($F = 4.24$; $df = 4, 33$; $p = <0.01$) (figure 3.1). Restoration method fence and tubestock had sites with significantly better total riparian vegetation condition scores than unrestored sites (Figure 3.1). Fence and tubestock, was the most reliable method to increase total riparian condition, with the highest mean total riparian condition score (23.8), the two highest total riparian condition site scores (33.1, 30.6), and the highest scoring minimum total riparian condition score (16.7) of all restoration methods and control sites. The restoration method fence only, displayed the greatest spread of data (Figure 3.1). Fence only sites varied considerably in the amount of remnant vegetation present on site, with some sites including intact remnant woodlands and others primarily groundcover. Sites restored using fencing and direct seeding also varied considerably in total riparian condition.

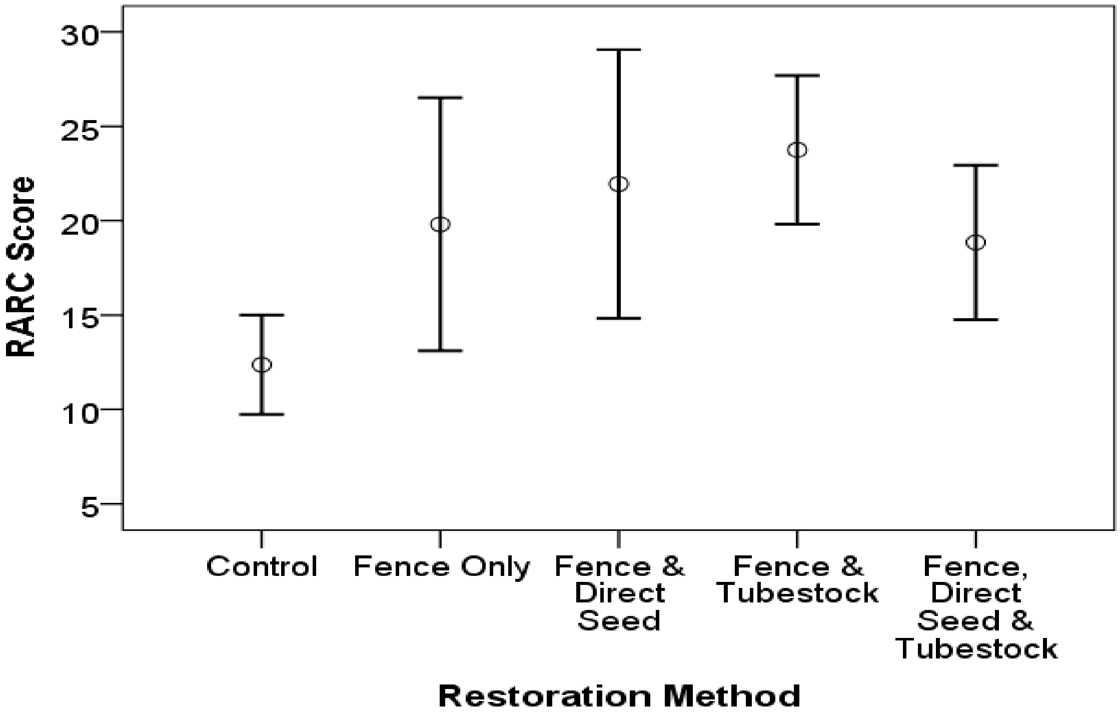


Figure 3.1: Riparian vegetation condition (RARC) scores for the sites within each restoration method: (control, fence only, fence and direct seed, fence and tubestock, and fence, direct seed and tubestock approximately 10 years after riparian restoration actions were implemented. Error bars represent +/- standard error about the mean.

Analysis of riparian vegetation condition sub-indices demonstrated that some attributes of riparian condition explain a large amount of the variance in total riparian vegetation condition scores. For all restoration sites combined, the *natives* sub-index explained at least 82% of the variance in the total riparian vegetation condition score (Table 3.1), whilst the *features* sub-index accounted for very little of the variance (30%), implying that this sub-index is not altered by restoration actions (at least in the first ten years) (Table 3.1).

Differences were observed between different restoration methods and the attribute of riparian condition that accounted for most of the variation in total riparian vegetation condition scores (Table 3.1). At sites that employed fence only, the RARC sub-indices *habitat* and *debris* explained 98% and 93% respectively of the variance in total riparian vegetation condition scores. Within all active restoration methods the RARC sub-indices *cover* and *natives* accounted for more variance in total riparian vegetation condition scores then in fence only sites. Within the control sites the *debris* RARC sub-index explained the most variance in total riparian vegetation condition (81%) and the *features* sub-index the least (10%) (Table 3.1).

Table 3.1: Proportion of variance in the total RARC score explained by each RARC sub-index within each restoration method. R^2 values were calculated from a regression of the values of each sub-index against the total RARC score.

RARC sub-indices	Control sites	Fence only	Fence and direct seed	Fence and tubestock	Fence, direct seed and tubestock	All restoration sites combined
	Df=1, 7	Df=1,4	Df=1, 3	Df=1, 8	Df=1, 6	Df=1, 27
Habitat	0.63	0.98	0.85	0.77	0.66	0.80
Cover	0.63	0.60	0.88	0.78	0.74	0.75
Natives	0.27	0.71	0.98	0.79	0.90	0.83
Debris	0.81	0.93	0.80	0.84	0.44	0.75
Features	0.10	0.27	0.17	0.49	0.19	0.30

3.2.2 RARC sub-indices

Ten years after restoration, restoration method was observed to have had differing effects on the attributes of riparian vegetation condition (sub-index scores). Restoration method had a significant effect on the RARC sub-index *habitat* score (Table 3.2). All restoration methods had higher *habitat* scores than the control sites (Table 3.2a) and very similar mean scores. Fence only had the most variation in *habitat* scores (Figure 3.2a). Restoration method had a significant effect on the RARC sub-index *cover* (Table 3.2). Fence and direct-seeding, and fence and tubestock had significantly greater *cover* scores than the control sites (Figure 3.2b), illustrating the value of active restoration for increasing vegetation cover. Similar results were found in the RARC sub-index *natives* scores, where the sites treated by fencing and planting tubestock were found to have significantly greater RARC sub-index *native* scores than the control sites. All other restoration methods had higher *native* sub-index scores than the control sites (Figure 3.2c). Ten years after restoration occurred, the restoration method used did not have a significant effect on the *debris* or *features* RARC sub-indices (Table 3.2). Although not significant, the control sites had the lowest mean scores for both *debris* and *features* RARC sub-indices (Figure 3.2d and 3.2e).

Table 3.2: Results of statistical analysis of the influence of restoration method on RARC sub-index scores (* = significant at P<0.05). All statistical tests used were ANOVA except for RARC sub-index *habitat* and RARC sub-index *debris* where a Kruskel-Wallis non-parametric test was used. F values are displayed for ANOVA analysis and X^2 values for Kruskel-Wallis analysis.

RARC Sub- Index	**df**	**F values and X^2 Values**	**P Value**
Habitat	4	x^2= 11.1968	0.02*
Cover	4, 33	F = 5.055	0.01*
Natives	4, 33	F = 2.997	0.03*
Debris	4	x^2= 8.6094	0.07
Features	4, 33	F = 1.42	0.25

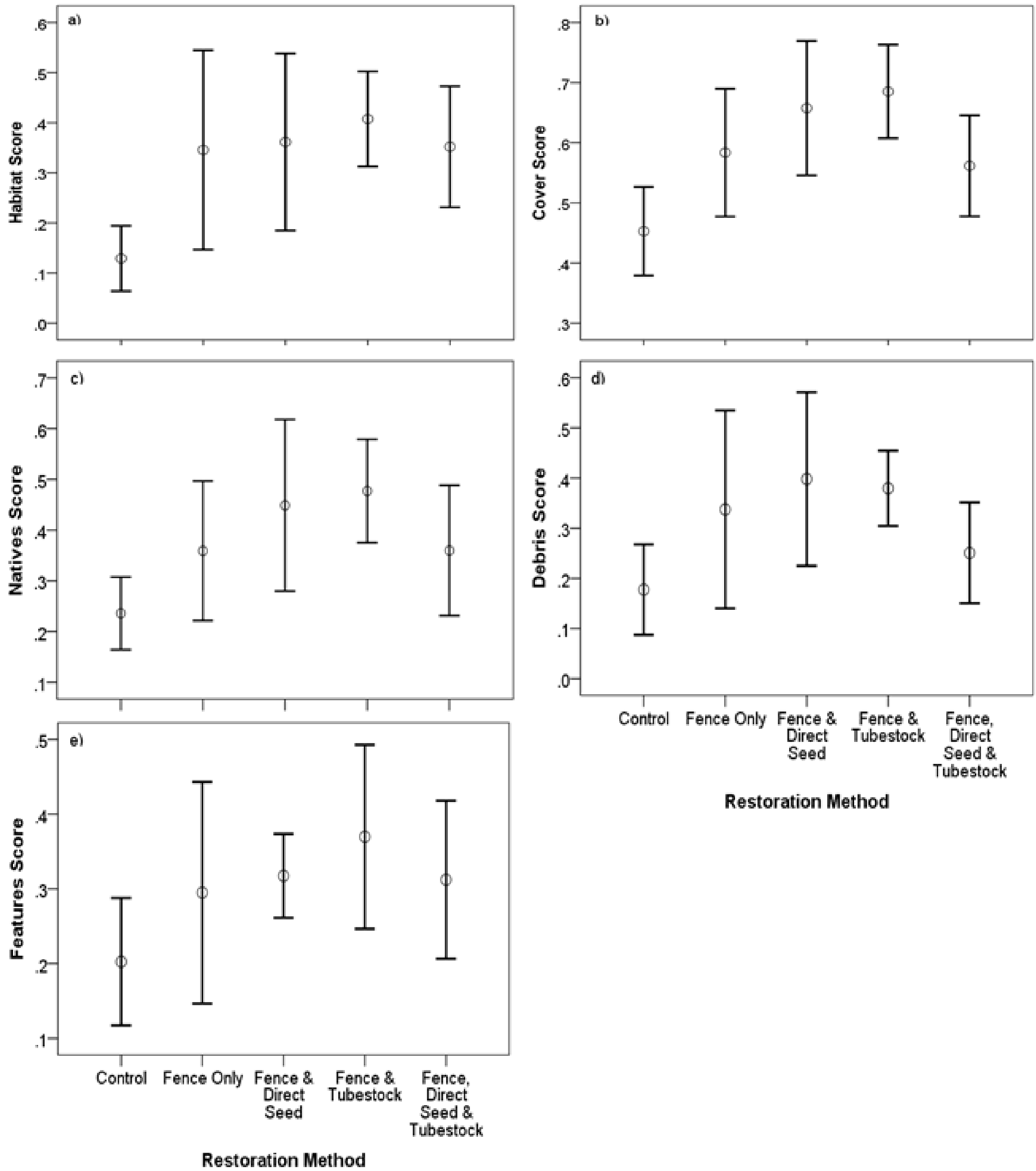


Figure 3.2: Mean riparian (RARC) sub-index scores from sites restored in the Bidgee Banks Restoration Project by restoration method for a) habitat, b) cover, c) natives, d) debris, and e) features from 38 sites in the Upper Murrumbidgee Catchment 10 years after restoration. Error bars represent +/- standard error about the mean.

3.2.3 RARC indicators

3.2.3.1 Field observations

The restoration methods varied considerably in their influence on different RARC indicators. Ten years after restoration occurred, active restoration was an important factor in increasing riparian width with restoration methods: fence and direct-seed, and fence and tubestock displaying a significantly wider canopy than the control sites (Figure 3.3a).

Restoration method had no effect on groundcover (Table 3.3), with all sites sampled having very similar mean groundcover irrespective of restoration method (Figures 3.3b). In spite of this, it was noted that the site with the highest groundcover (100%) had only been fenced, and the lowest mean groundcover (23.8%) was one of the control sites. The cover of mid-storey vegetation varied considerably between restored sites and control sites (Figure 3.3c). The control sites had very little mid-storey cover ranging from 0% to 5%. The restoration method used did not significantly affect the cover of mid-storey species (Table 3.3), but restoration has potentially resulted in an increase in the cover of mid-storey species ten years after restoration. Ten years after restoration, the restoration method used had a significant effect on canopy cover (Table 3.3). Sites restored using fence and direct-seed, and fence and tubestock had significantly greater canopy cover than the control sites (Figure 3.3d). Sites treated with fence, tubestock and direct seeding had relatively low canopy cover (in relation to the other actively restored sites) as a result of a few failed revegetation attempts at problematic sites.

Restoration method had no effect on native groundcover (Table 3.3). The site with the lowest native groundcover (1.5%) had undergone restoration method fence, tubestock and direct seed and the highest (77.5%) was a fence only site. Restoration method had a significant effect on cover of native mid-storey species (Figure 3f). The control sites had very little native mid-storey cover with seven of the nine control sites having no native mid-storey, and the two control sites with native mid-storey having only 0.5% and 1.8%. Active restoration was an important factor in increasing native mid-storey cover with sites treated with fencing, planting with tubestock and direct seeding having significantly more native mid-storey than the control sites, and all other active restoration methods displaying more native mid-storey cover than the control sites (Figure 3.3f). The fence only sites had considerably less native mid-storey cover compared with the active restoration methods (Figure 3.3f). There was very little difference between the overall canopy cover and native canopy cover, with non-native canopy species only present at a few sites (Figure 3.15c). For this reason very similar results were found for native canopy cover as total canopy cover, with active restoration an important factor in increasing native canopy cover (Figure 3.3g).

The percentage of leaf litter on the ground was not significantly affected by restoration method (Table 3.3), but there were notable improvements in leaf litter associated with active restoration (tube-stock and direct seeding) (Figure 3h). Sites that were fenced and direct seeded had the highest mean leaf litter (33.6% ±12), followed by sites that were fenced and planted with tubestock (26.1%±3.7).

Ten years after restoration, restoration method had no effect on the presence of hollow bearing trees or abundance of fallen logs (course woody debris >10cm) (Table 3.3). Restoration method had a significant effect on the abundance of seedling recruitment of mid-storey species (Table 3.3) and a non-significant effect on seedling recruitment of canopy species (Table 3.3). Restoration methods fence and tubestock and fence and direct seed resulted in the highest abundance of seedling recruitment in both mid-storey (Figure 3.3k) and canopy species (Figure 3.3l). All control sites had no seedlings of mid-storey species present and only two control sites had seedlings from canopy species present, suggesting that the presence of livestock have a negative effect on seedling abundance and fencing sites to exclude livestock has a positive effect on seedling recruitment (Figure 3.3k and Figure 3.3l).

Table 3.3: Results of ANOVA or Kruskel-Wallace non-parametric tests on significance of the restoration method on individual riparian characteristics. For all riparian attributes tested, df=4, 33. * = significant at $P<0.05$. All statistical tests used were ANOVA except for mid-storey seedling recruitment and canopy seedling recruitment where a Kruskel-Wallis non-parametric test was used. F values are displayed for ANOVA analysis and X^2 values for Kruskel-Wallis analysis.

RARC indicators	F values and X^2 values	P Value
Width of riparian canopy vegetation	F = 5.60	<0.01*
Groundcover	F = 0.45	0.77
Mid-storey	F = 1.99	0.112
Canopy cover	F = 4.40	<0.01*
Native ground cover	F = 0.55	0.70
Native mid-storey	F = 3.00	0.03*
Native canopy cover	F = 3.76	0.01*
Leaf litter	F = 1.70	0.17
Presence of hollow bearing trees	F = 0.31	0.87
Abundance of fallen logs	F = 2.28	0.08
mid-storey species recruitment	X^2=10.2217	0.04*
canopy species recruitment	X^2= 5.3374	0.25

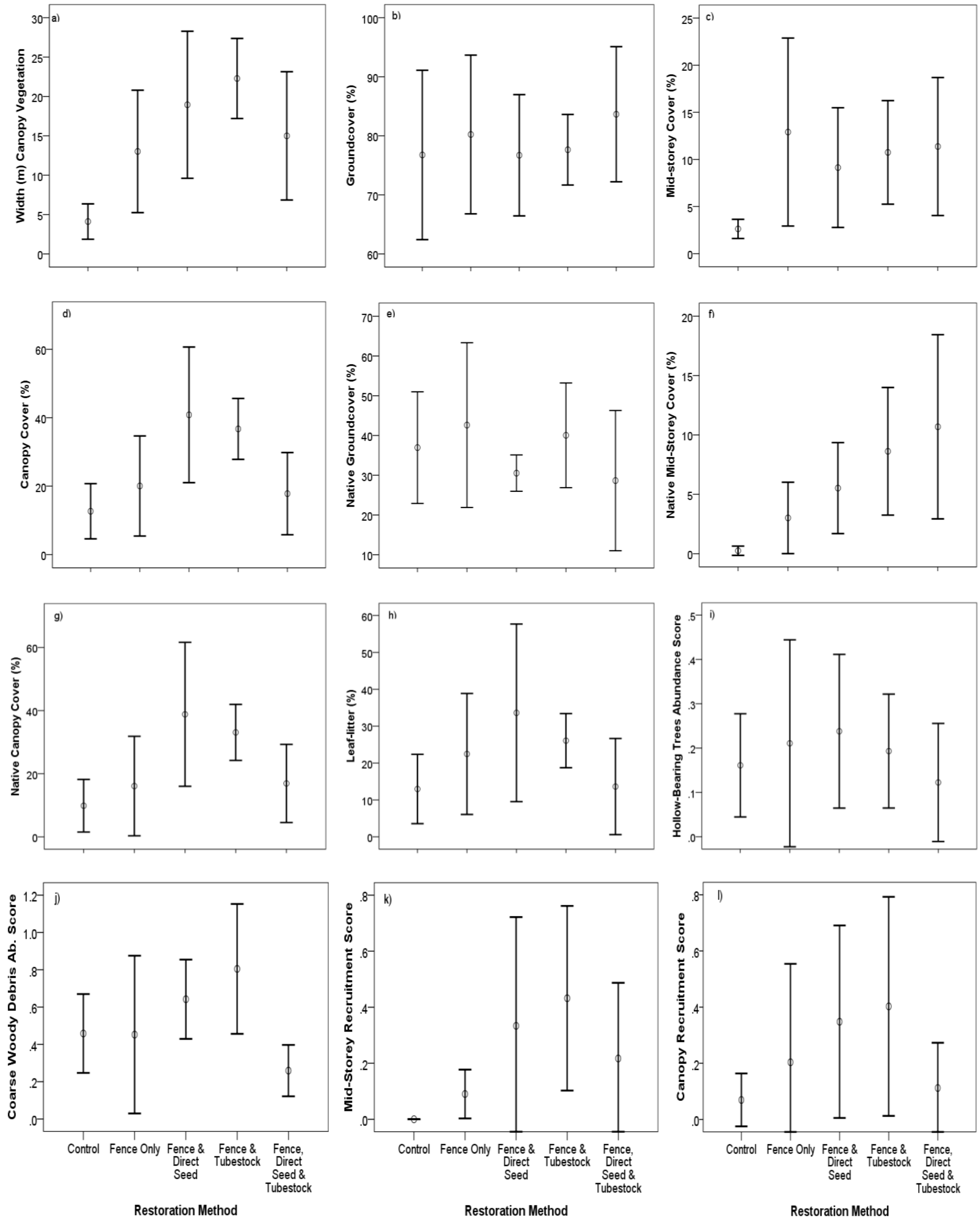


Figure 3.3: Mean riparian vegetation attributes from sites restored in the Bidgee Banks Restoration Project by restoration method a) width of riparian canopy vegetation, b) groundcover, c) Mid-storey cover, d) Canopy cover, e) native groundcover, f) native mid-storey cover, g) native canopy cover, h) leaf litter, i) hollow bearing trees, j) coarse woody debris, k) Seedling (<1 m tall) recruitment of mid-storey species, and l) seedling recruitment of canopy species. Error bars represent +/- standard error about the mean.

3.2.3.2 Before and after restoration (remote sensing).

Aerial imagery was used to compare sites before restoration occurred and ten years after restoration. The width of riparian canopy vegetation and projective foliage cover were calculated at sites with available aerial imagery.

3.2.3.2.1 Width of riparian canopy vegetation.

Overall improvements in riparian canopy width was observed for all restoration sites (regardless of restoration method) over the 10 years since restoration occurred, with the current mean riparian canopy width 19.2 m almost double that prior to restoration 10.8 m (Figure 3.4).

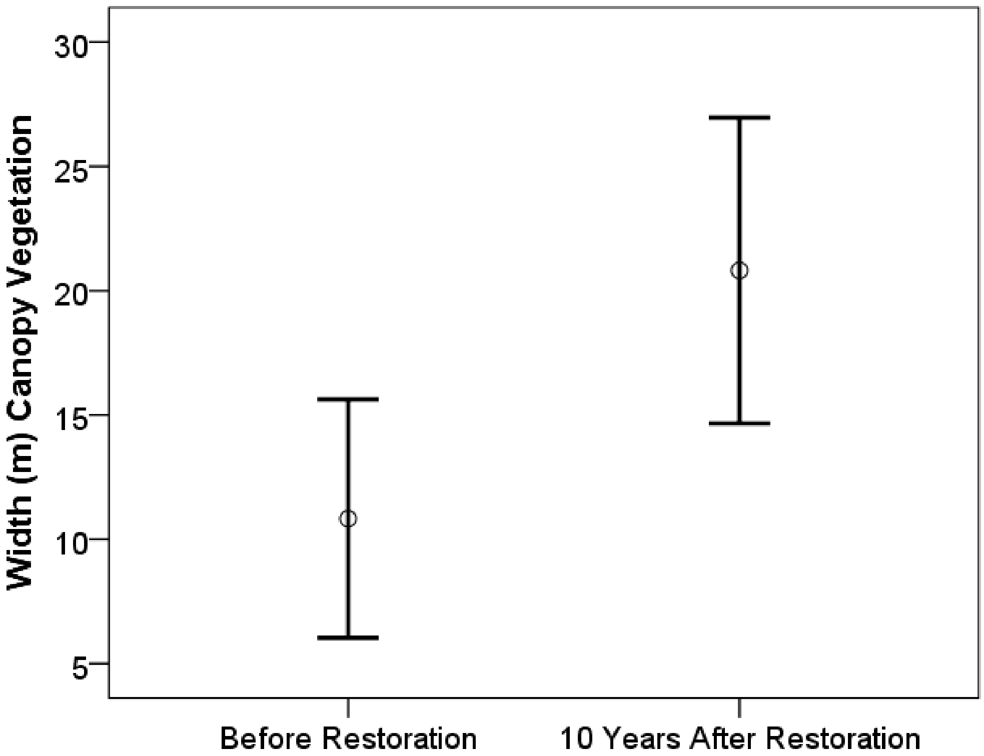


Figure 3.4: Width of the riparian canopy for all restoration sites around the time of restoration (2003) and approximately 10 years after restoration (2014). Error bars represent +/- standard error about the mean.

The change in width of riparian canopy varied between restoration methods over the ten years since restoration (Figure 3.5). Sites that were fenced, fenced and planted with tubestock, and fenced and direct seeded displayed an increased width of riparian canopy cover after ten years. The width of riparian canopy cover in the control sites did not change over the ten year period. The width of riparian canopy before restoration occurred varied between restoration methods, fence only sites displayed the highest 24.7 m and sites that were fenced, planted with tubestock and direct seeded displayed the lowest (4.7 m) (Figure 3.5). The control sites showed no change in width of canopy vegetation in the ten years since restoration (Figure 3.5).

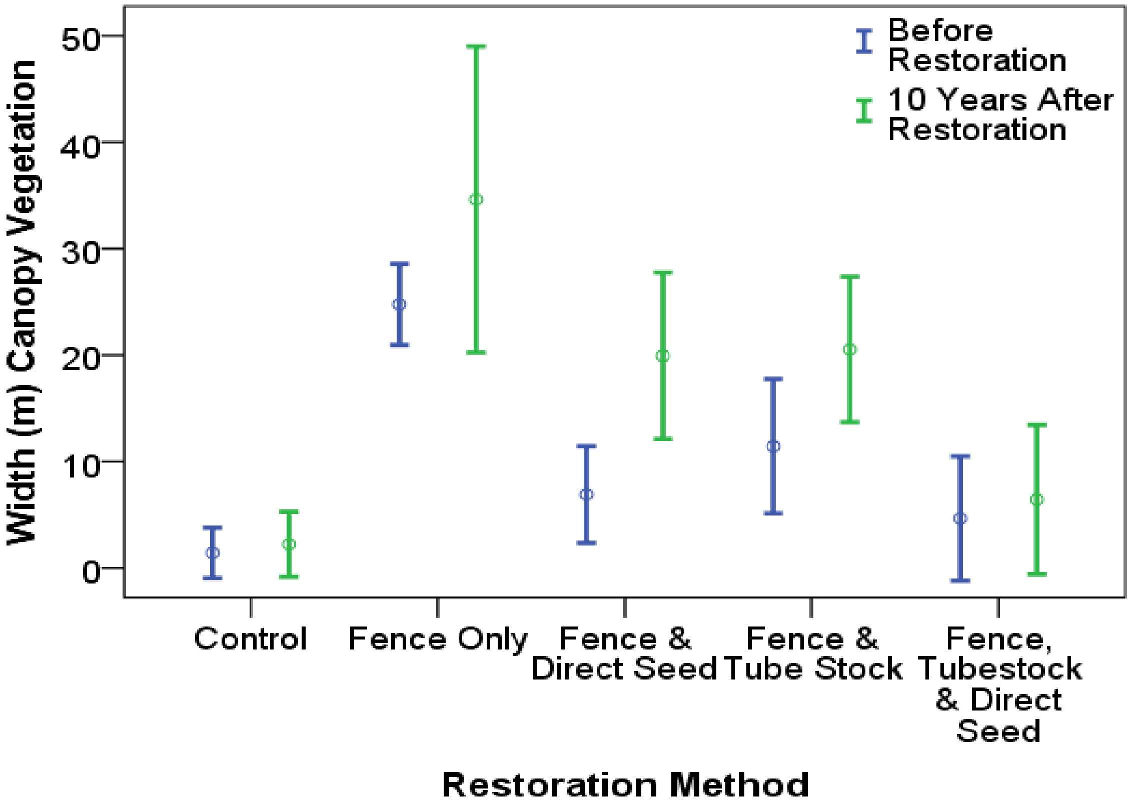


Figure 3.5: Mean width of the riparian canopy vegetation from sites restored using the different restoration methods in 2003, and 2014. Error bars represent +/- standard error about the mean.

Improvements in width of riparian canopy vegetation differed between individual sites (Figure 3.6). Some sites have had marked increases in canopy width and a few sites have had little or no change since restoration.

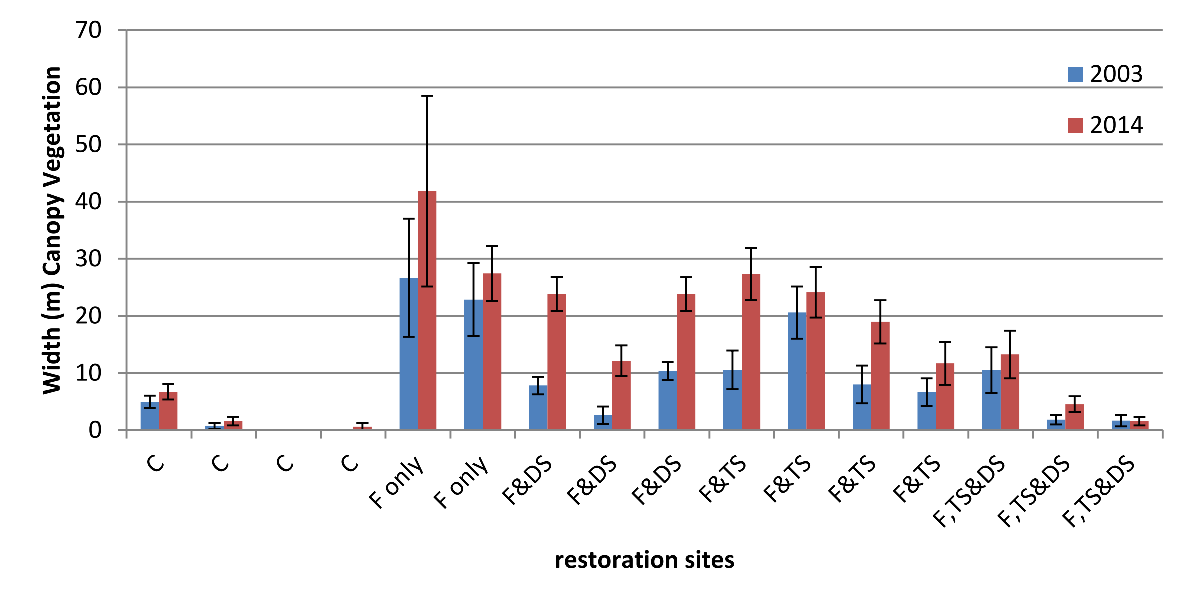


Figure 3.6: Mean width of riparian canopy vegetation at individual sites in 2003 and 2014. Error bars represent +/- standard error about the mean.

3.2.3.2.2 Projected foliage cover

Overall improvements in projected foliage cover were identified for all restoration methods combined (Figure 3.7). In 2014 the projected foliage cover was 20.6% compared to that pre-restoration which was 10.4%.

Figure 3.7: Mean projected foliage cover calculated using image recognition software (WinDias), for 20 restoration sites before restoration (around 2000) and 10 years after restoration (2014). Error bars represent +/- standard error about the mean.

The restoration method used varied in the amount of increase in projected foliage cover since restoration. Active restoration was an important factor in increasing projected canopy cover with restoration method, fence and tubestock displaying the largest change in canopy cover, followed by fence and direct seeding (Figure 3.8). Sites treated with fence only, displayed a small change in canopy cover since restoration occurred. It is interesting to note that different restoration methods varied in the amount of initial projected foliage cover pre-restoration (Figure 3.8), with restoration method fence only having the most initial projected foliage cover and fence, direct seed and tubestock having the least. The control sites did not display any change in projected foliage cover, which demonstrates that the improvements in projective foliage cover seen in restored sites can be attributed to restoration actions.

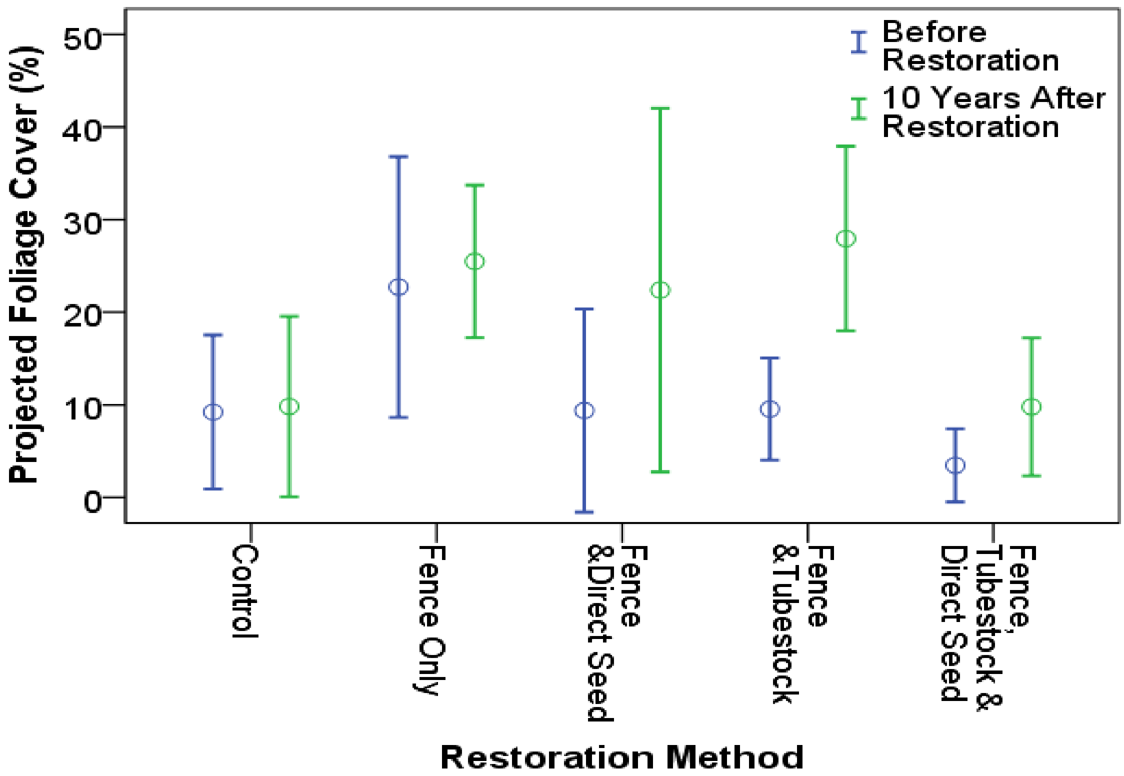


Figure 3.8: Projected foliage cover within each restoration method before restoration (around 2000), and 2014. Error bars represent +/- standard error about the mean.

There is considerable variation, in the change in projected foliage cover since restoration at individual sites (Figure 3.9). Some sites have had large improvements in projected foliage cover and some have had very little (Figure 3.9).

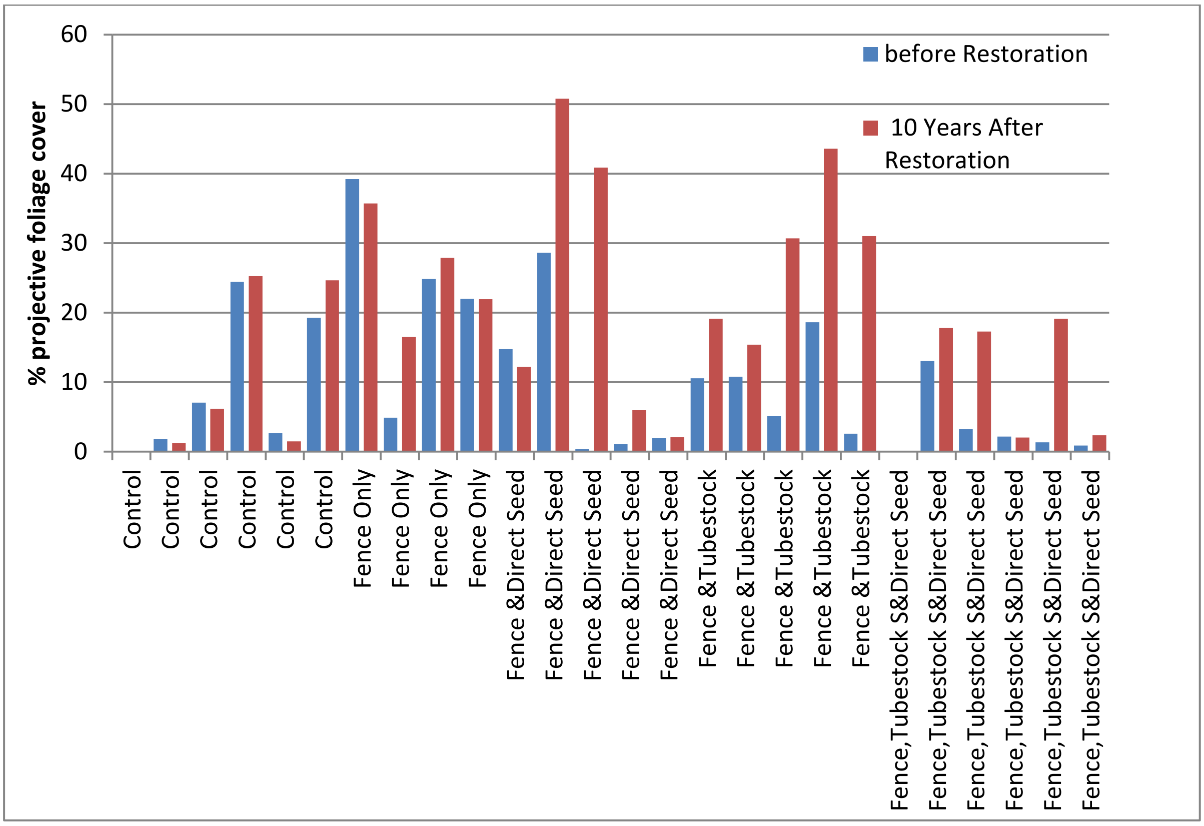


Figure 3.9: Projected foliage cover for each site assessed before restoration and in 2014.

3.3 Bank and channel condition

To identify a) if there have been improvements in bank and channel condition ten years after restoration occurred, and b) if restoration methods have differed in the level of improvements in bank and channel condition, an ephemeral stream assessment was performed at each site. Ten years after riparian restoration occurred, all restoration methods had a significantly greater mean ephemeral stream assessment score than the control sites ($F (4, 33), = 9.92, P = <0.01$), and all restoration methods demonstrated very similar scores (Figure 3.10). This result demonstrates evidence of overall improvements in bank and channel condition attributed to the Bidgee Banks Restoration Project.

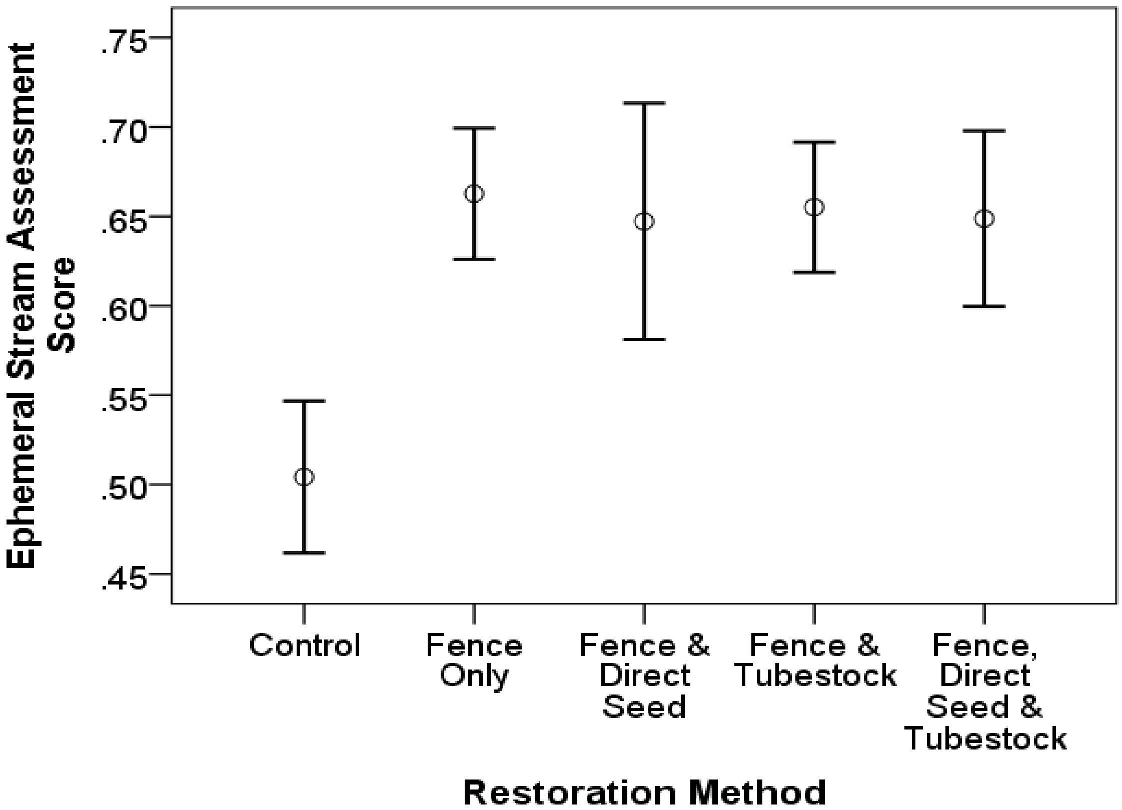


Figure 3.10: Ephemeral stream assessment scores for each treatment type: control, Fence only, Fence and Direct seed, Fence and tubestock, and Fence, direct seed and tubestock from 38 sites in the Upper Murrumbidgee Catchment approximately 10 years after riparian restoration was implemented. Error bars represent +/- standard error about the mean.

Indicators from the ephemeral stream assessment were analysed separately to investigate how restoration methods influence geomorphological characteristics of the stream. The shape of cross section (essentially a bank stability assessment) was significantly affected by restoration method ($F_{(4, 33)}$, = 5.156, P = <0.01), with significant improvements in sites within all restoration methods compared to the control sites (Figure 3.11a). Ten years after restoration occured, restoration method did not have a significant effect on the longitudinal morphology of the drainage-line ($F_{(4, 33)}$, = 1.79, P = 0.15). In spite of this, it was noted that the control sites had the two lowest longitudinal morphology (drainage line condition) scores (1.33 and 1.75/4) but also had the greatest range of data (Figure 3.11b). All restoration methods demonstrated vast improvements in lateral flow regulation, with sites within all restoration methods having significantly greater lateral flow regulation than the control sites ($F_{(4, 33)}$, = 14.94, P = 4.55e-07) (Figure 3.11c).

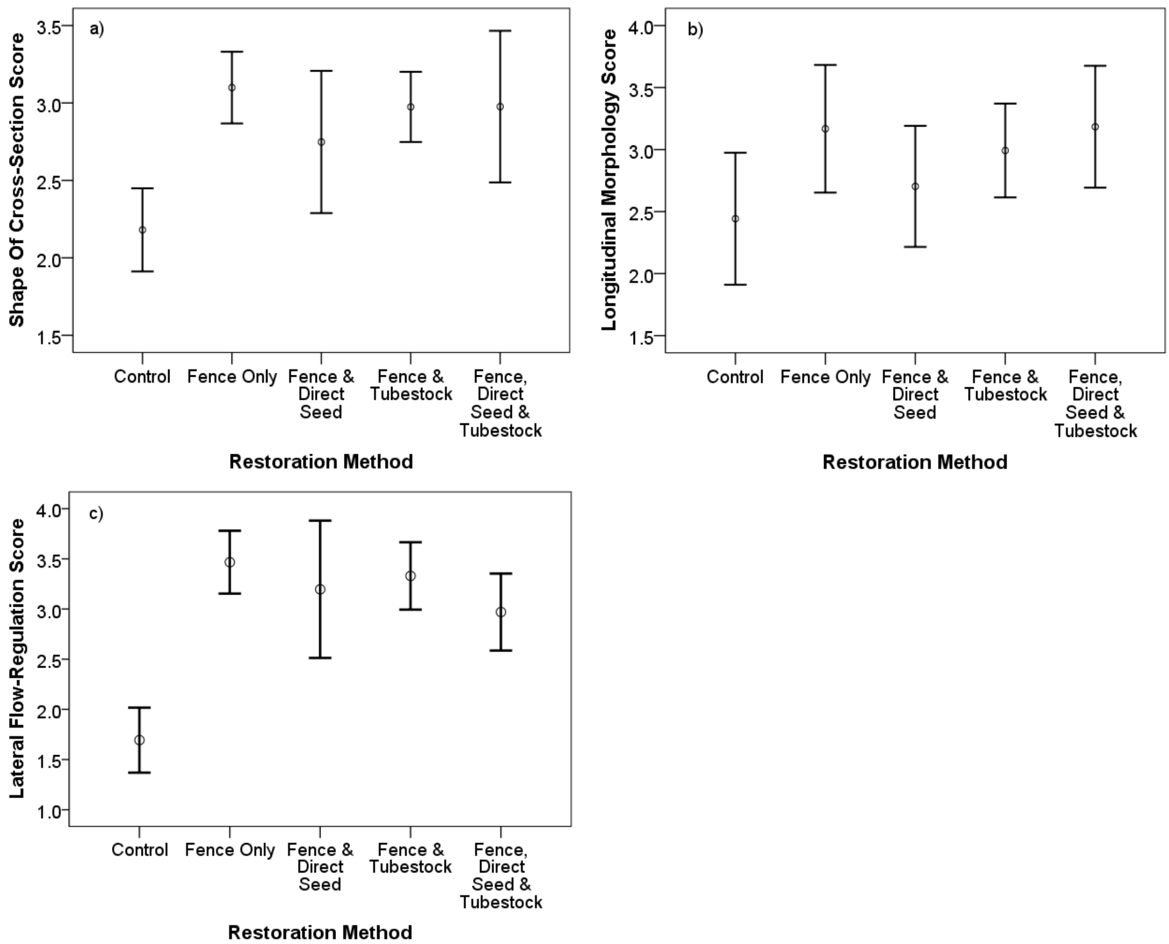


Figure 3.11: Mean **s**cores for bank attributes a) shape of cross section, b) Longitudinal morphology, and c) Lateral flow regulation, within each restoration method. Error bars represent +/- standard error about the mean.

3.4 Influential factors on riparian restoration outcomes.

The next section of the results is dedicated to investigating some of the potential factors that may have contributed to the outcomes of the Bidgee Banks Restoration Project, and changes that may have occurred in response to restoration actions.

3.4.1 Remnant vegetation's influence on seedling recruitment and debris.

The abundance of remnant vegetation at each site appears to have a positive effect on numerous features of the restored riparian vegetation. The greater the amount of riparian vegetation at the site, the greater the observed seedling recruitment (both mid-storey and canopy species) and debris (Figure 3.12a to c). Mid-storey seedling recruitment (χ^2= 0.76, df= 2, p=0.02) (Figure 3.12a) canopy species seedling recruitment (χ^2= 18.48, df= 2, p=9.715e-05) (Figure 3.12b), and the RARC sub-index *debris* (χ^2= 15.10, df= 2, p=<0.01) (Figure 3.12c) were all significantly effected by the abundance of remnant vegetation.

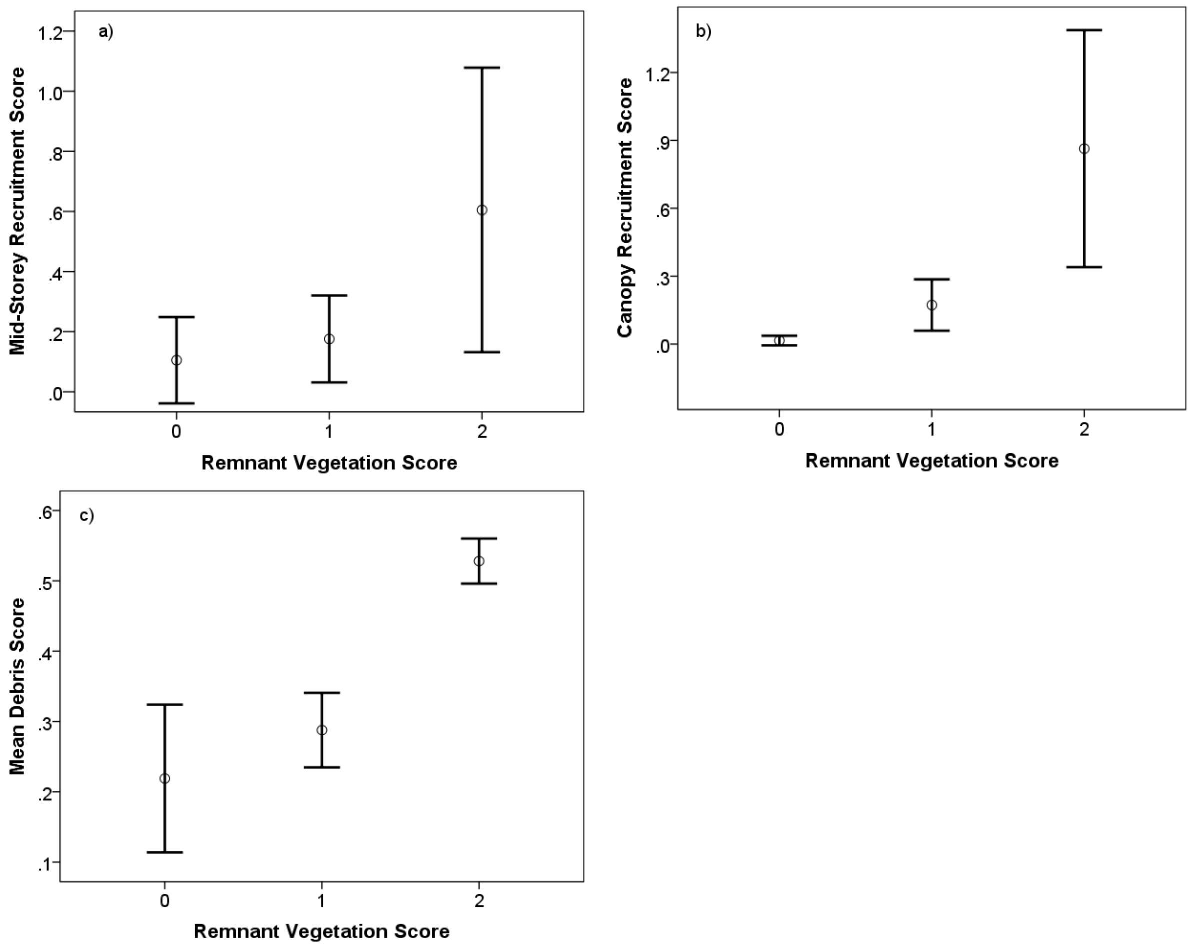


Figure 3.12: Relationship between abundance of remnant vegetation and: a) seedling recruitment of mid-storey species, b) seedling recruitment of canopy species, and, c) RARC sub-index *debris*.

3.4.2 Native groundcover on seedling recruitment

The proportion of native groundcover was compared to the abundance of seedling recruitment of understorey and canopy species within each transect. All transects from control sites were excluded to remove the confounding factor of livestock herbivory. Native groundcover had a significant effect on the seedling recruitment of both mid-storey species ($X^2 = 46.4153$; df = 28; p = 0.016), and canopy species ($X^2 = 52.58$; df = 28; p = <0.01) (Figure 3.13a and Figure 3.13b), with abundance of seedling recruitment increasing with increasing native groundcover.

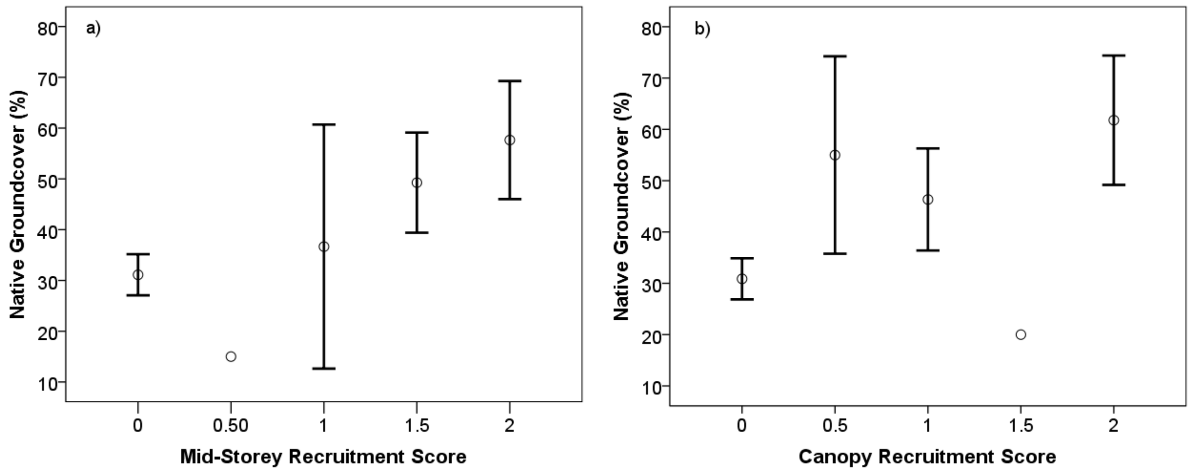


Figure 3.13: Relationship between the percentage of native ground cover and abundance of a) mid-storey species seedling recruitment, and; b) canopy species seedling recruitment. Error bars represent +/- standard error about the mean.

3.4.3 Vegetation on the drainage-line wall and floor and the condition of the drainage-line.

To investigate if vegetation on the drainage-line floor and drainage-line wall influences the condition of the drainage-line, longitudinal morphology of the drainage-line was compared with the abundance of vegetation on the drainage-line floor and wall. As the slope of the surrounding landscape is an influential factor that determines the source and direction of water flow, sites were grouped on the basis of the slope of the surrounding landscape.

The abundance of vegetation on the floor of the drainage-line was found to be associated with the condition of the drainage-line in flat to moderately sloped areas <10˚, and this association reduced with increased slope (Figure 3.14). The abundance of vegetation on the drainage-line wall was found to be associated with drainage-line condition in drainage-lines surrounded by very steep slopes >30˚ (r=0.85), with this relationship reducing with the reduction in slope (Figure 3.15).

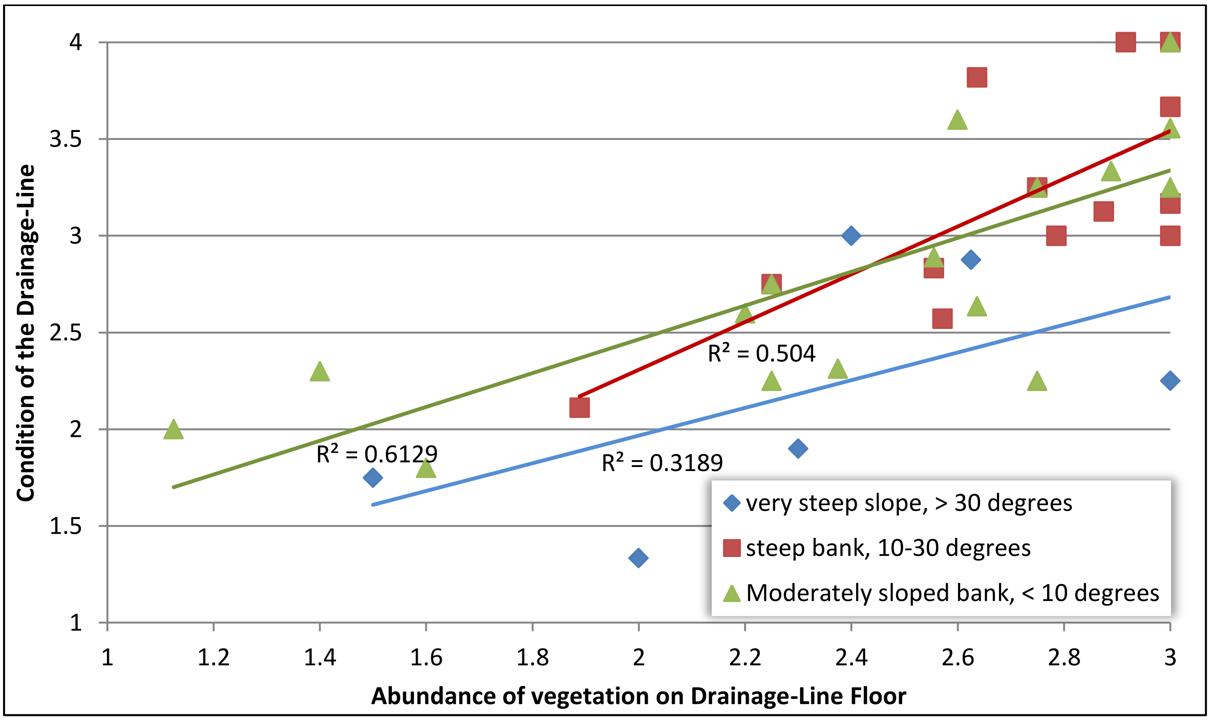


Figure 3.14: Regression analysis between the score representing the longitudinal morphology of the drainage-line and the abundance of vegetation on the drainage-line floor for all 38 sites, with sites put into three categories based on the shape of the adjacent floodplain.

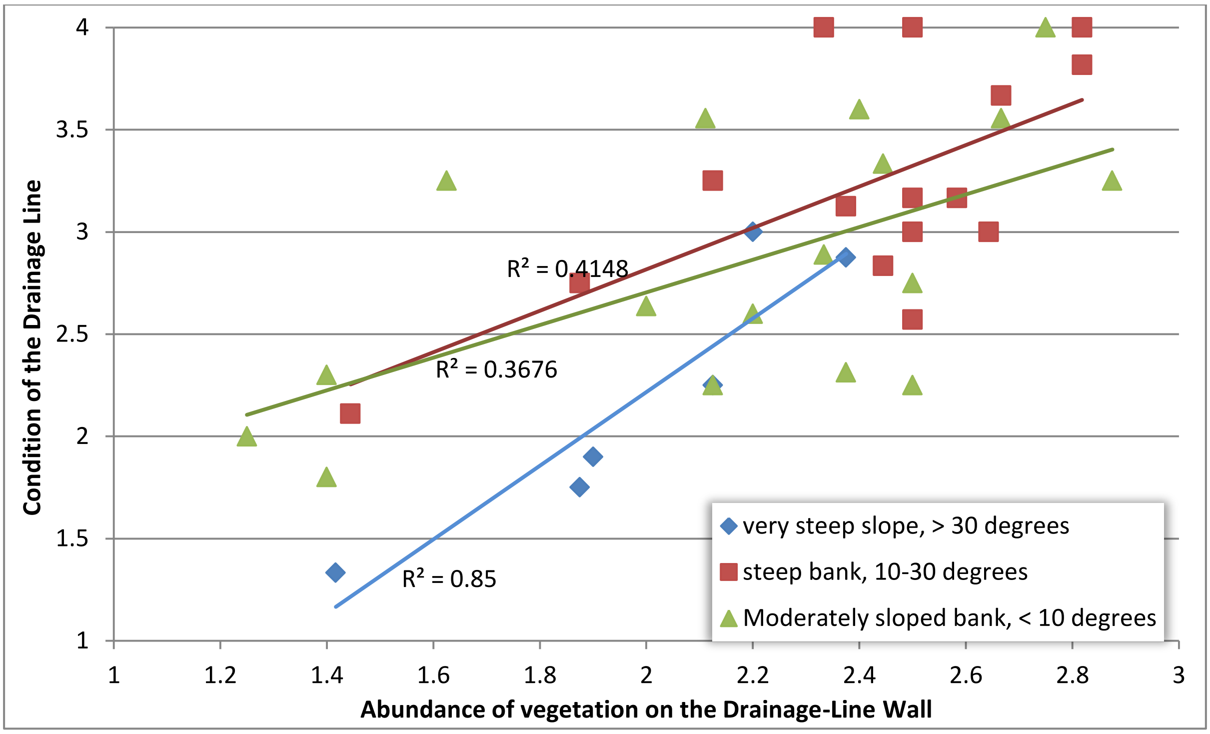


Figure 3.15: Regression analysis between the score representing the longitudinal morphology of the drainage-line and the vegetation on the drainage-line wall for all 38 sites, with sites put into three categories based on the shape of adjacent floodplain.

3.4.4 Restoration method used and cover of non-native species.

The influence of restoration method and cover of non-native species was investigated for each vegetation layer. Ten years after restoration occurred, restoration method did not have a significant effect on non-native groundcover (F = 0.862; df =4, 33; p = 0.497) (Figure 3.16a). All sites had (at least some) non-native groundcover species present, and all restoration methods and control sites had similar non-native groundcover (%). Restoration method had no significant effect on non-native mid-storey cover (F= 1.938; df =4, 33; p=0.127), and no significant effect on non-native canopy cover (χ^2= 0.333, df= 4, p =0.9876). Although not considered to be statistically significantly different, Figure 3.16b illustrates the large amount of non-native mid-storey cover in at least some of the sites treated with fence only compared to all other restoration methods and the control sites.

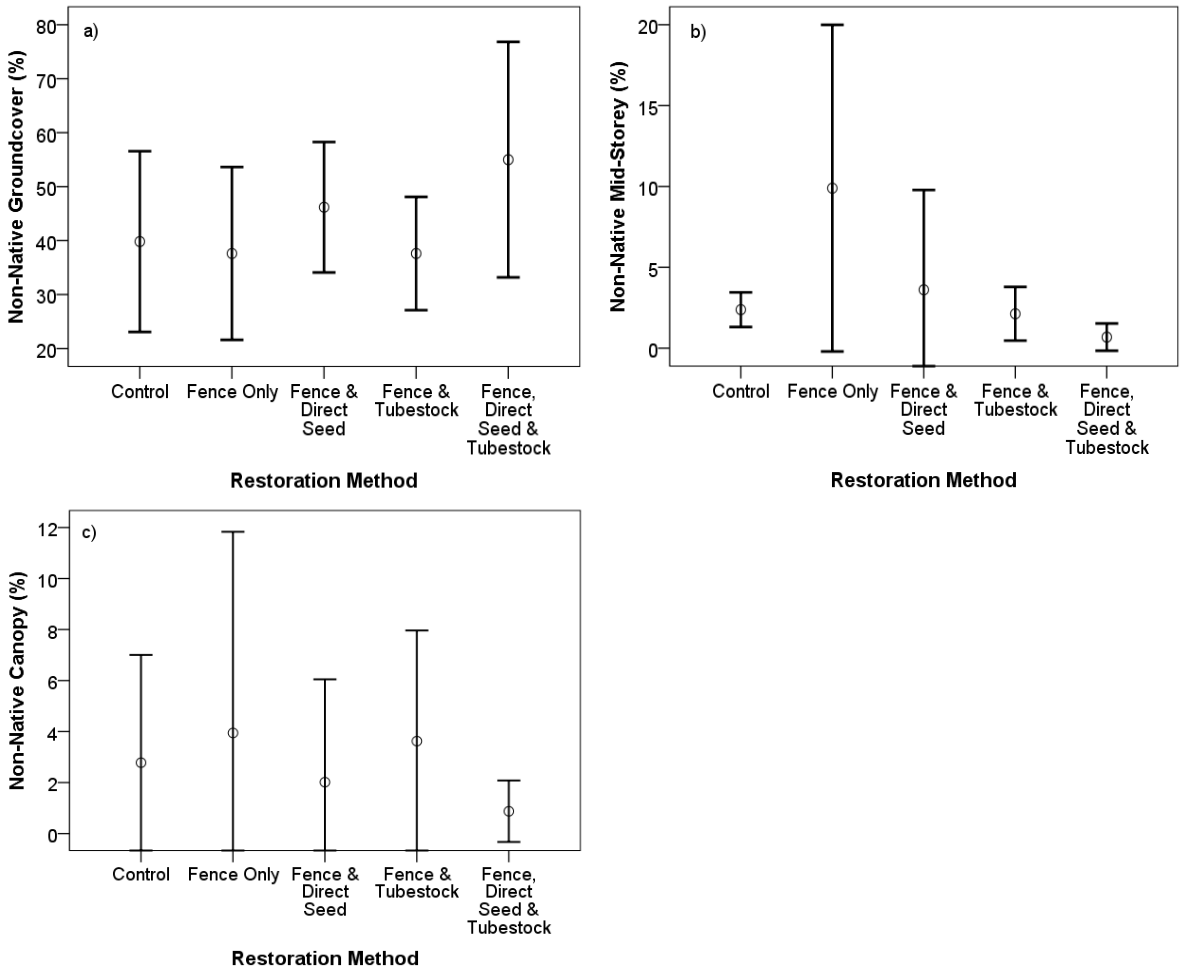


Figure 3.16: Cover of a) non-native groundcover species, b) non-native mid-storey species, and c) non-native canopy species for sites within each restoration method. Error bars represent +/- standard error about the mean.

3.4.5 Species used in riparian restoration

3.4.5.1 The occurrence probability of actively revegetated species

The occurrence probability of plant species used on the actively revegetated sites was investigated. Sites with available data on the species initially planted or direct seeded were compared to the species currently present at each site. The number of sites with a species present (observed) was divided by the number of sites at which that species was planted or direct seeded (expected) to give a % of sites with that species still present. The occurrence probability of individual species (Appendix 2) demonstrated that plant species within a genus perform similarly, with the odd exception. For this reason the species were grouped according to their genus, except for those with only one or two species, which were classified as ‘all other species’. Figure 3.17 clearly illustrates that some genus groups have a far higher occurrence probability than others. The *Acacia* group was found to have the highest occurrence probability with 67% of sites containing the expected *Acacia* species, and a number of *Acacia* species displaying 100% occurrence probability, such as *Acacia boormanii* and *Acacia cultriformis* (Appendix 2). *Casuarina* was the second most successful genus with 55% occurrence probability, followed by *Eucalyptus* with 53% occurrence probability. The category of all other species had a survival rate of only 12%, demonstrating that only a little more than one in every ten species in this category were found at sites at which they were planted (Figure 3.17). This category included some species that completely failed, not being present at any sites that they were expected (Appendix 2), such as *Hardenburgia violacea* which was not observed at the 6 sites it was expected.

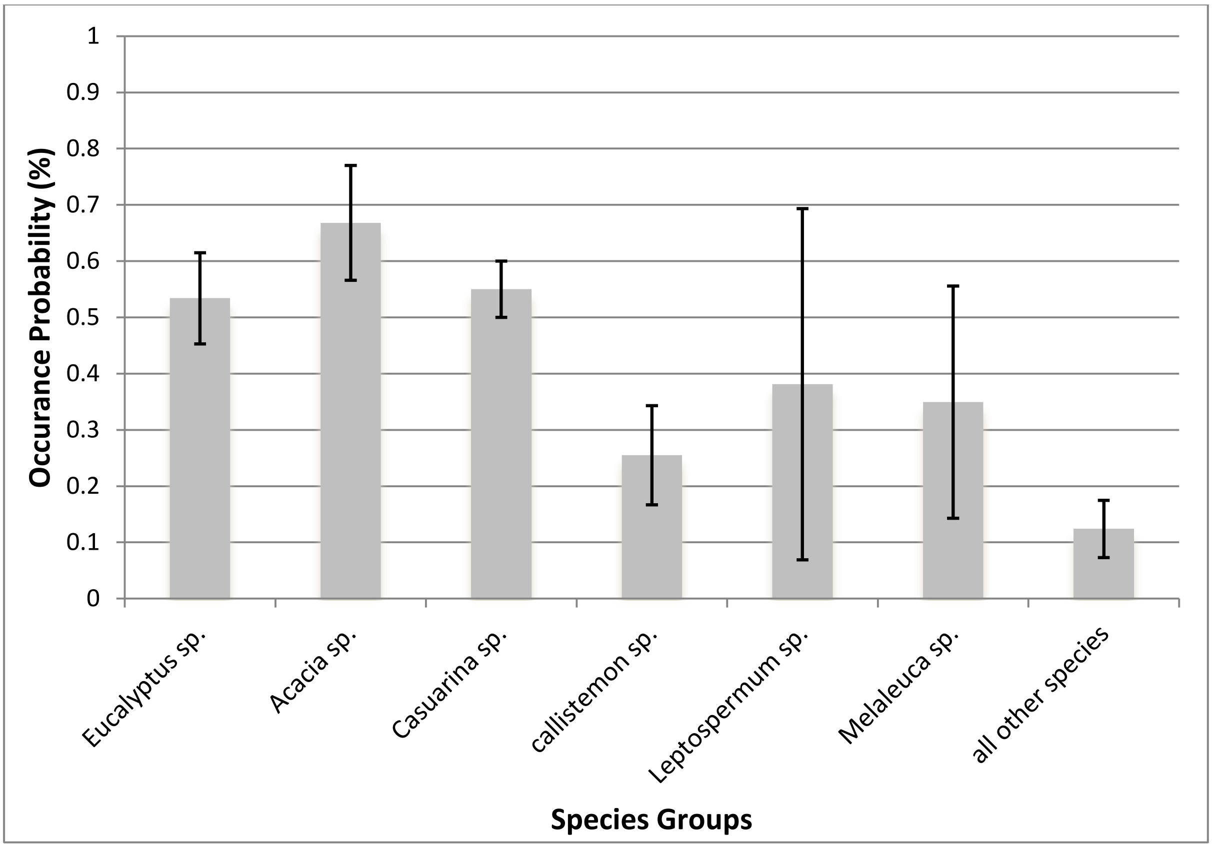


Figure 3.17: The occurrence probability (observed/expected) for species at actively revegetated sites, all species grouped by their genus except for all species with <2 in a genus, which were grouped as, all other species. Error bars represent +/- standard error about the mean.

3.4.5.2 Seedling recruitment from revegetated species

To further investigate the survival rates of plant species used in riparian restoration, the long-term persistence of species was investigated by identifying all seedlings from actively revegetated species. Seedlings were identified at sites with no remnant vegetation present of that species. A total of nine species had successfully reproduced in the ten years since restoration was implemented, with the majority belonging to the genus *Acacia* and one *Leptospermum lanigerum*. Seedlings of *Acacia dealbata* were present at three different restoration sites (Table 3.4). Table 3.4 demonstrates the overwhelming success of *Acacia* at successfully establishing seedlings ten years after restoration.

Table 3.4: Actively revegetated species which have seedlings present, and the number of sites they were present.

Species with seedlings present	Number of sites
Acacia boormanii	1
Acacia dealbata	3
Acacia decurrens	1
Acacia mearnsii	1
Acacia pravisimma	1
Acacia rubida	1
Acacia longifolia	1
Acacia melanoxylon	1
Leptospermum lanigerum	1

3.4.6 The influence of climate on the outcomes of restoration.

3.4.6.1 Influence of rainfall in the first year after revegetation on native canopy cover (as a function of the survival of plantings).

Rainfall was investigated as a possible influence on the survival and growth of plants used in restoration. All planted or direct seeded sites for which the month of planting/seeding was known were assessed. The average rainfall over one month, three months, six months and twelve months after planting or direct seeding were obtained from historical climate records (BOM). These data had no effect on the current native-canopy cover (table 3.5).

Table 3.5: Results from regression analysis of average rainfall for the 1st, 3rd, 6th, and 12th months after revegetation occurred against native canopy cover.

Mean monthly rainfall over:	Degrees of freedom	R^2	F value	P value	Significant effect
1 month	1,12	.05	0.62	0.45	No
3 months	1,12	.04	0.48	0.50	No
6 months	1,12	.04	0.49	0.50	No
12 months	1,12	.10	1.48	0.25	No

3.4.6.2.1 Influence of Temperature in the first three months after revegetation on native canopy cover

Temperature was investigated as a possible contributor to the survival and growth of plants used in restoration. All planted or direct seeded sites for which the month of planting/seeding was known were assessed. The average monthly, minimum and maximum temperatures in the three months after planting or direct seeding were obtained and assessed. There was no significant relationship between the current native canopy cover and the maximum, minimum and average temperature in the three months after planting or seeding (table 3.6). It should be noted that the majority of planting associated with restoration activities occurred in spring time, to minimise the effects of temperature on survival rates.

Table 3.6: Results from a regression analysis of mean, minimum, and maximum temperature recorded over the three months after revegetation occurred against current native canopy cover.

Temperature category	degrees of freedom	R^2	F value	P value	Significant effect
Mean temp.	1,14	<0.01	0.05	0.83	No
Minimum temp.	1,14	0.16	2.56	0.13	No
Maximum temp.	1,14	<0.01	0.12	0.73	No

3.4.6.2.2 Influence of temperature on the diversity of species present on each site

The mean, minimum, and maximum daily temperature in the three months after revegetation took place was compared to the number of observed/ the number expected species at each site. Mean and maximum temperature had no effect and minimum temperature could have had a small effect on the current number of species found at a site (Table 3.7).

Table 3.7: Results from a regression analysis of mean, minimum, and maximum temperature recorded over the three months after revegetation occurred and the number of species observed/ the number expected at each site.

Temperature categories	degrees of freedom	R^2	F value	P value	Significant effect
Mean temp.	1,7	0.115	0.913	0.371	No
Minimum temp.	1,7	0.290	2.910	0.132	No
Maximum temp.	1,7	0.067	0.501	0.502	No

3.4.7 Relative importance of overall riparian condition, riparian vegetation and other riparian features for bank stability and erosion control.

Overall riparian condition and riparian attributes were compared with the ephemeral stream assessment scores, to identify the contribution of different riparian attributes at maintaining bank stability and erosion state. There was no strong relationship between any riparian vegetation attributes and the ephemeral stream assessment scores. Although not significant the width of riparian canopy vegetation had the highest affiliation with the ephemeral stream assessments scores with an R^2 value of .24, demonstrating some association between riparian vegetation width and bank stability. Of interest is the difference in R^2 between the RARC sub-index *cover*, 0.21 and sub-index *natives* 0.08, demonstrating that all vegetation plays a similar role regardless of its providence.

Table 3.8: Regression analysis between total riparian condition, riparian sub-indices, and individual riparian attributes (indicators) against ephemeral stream assessment scores. Degrees of freedom for all regressions was 1 & 36.

Importance of riparian attributes for Ephemeral Stream Assessment Score	**R^2**	**F value**	**P value**
Overall RARC scores (riparian condition)	0.211	9.614	0.004
Habitat sub-index	0.169	7.301	0.010
Cover sub-index	0.211	9.631	0.004
Natives sub-index	0.078	3.065	0.089
Debris sub-index	0.140	5.868	0.021
Features sub-index	0.238	11.255	0.002
Width of riparian canopy	0.240	11.366	0.002
Groundcover	0.066	2.532	0.120
Mid-storey cover	0.068	2.629	0.114
Canopy cover	0.094	3.738	0.061
Native groundcover	0.012	0.423	0.520
Native Mid-storey cover	0.038	1.434	0.239
Native canopy cover	0.073	2.823	0.102
Fallen Logs (> 10 cm in diameter)	0.064	2.442	0.127
Leaf litter	0.053	1.996	0.166
Native tussock grasses	0.050	1.898	0.177
reeds	0.153	6.480	0.015

4. Discussion

4.1 Overview of discussion

In an effort to understand how different riparian restoration methods contribute to improving riparian and geomorphological condition, a sub-set of sites restored as part of a large scale restoration project were assessed. Long-term studies on the influences of excluding livestock and revegetating the riparian zone to improve geomorphological condition are rare. Such studies are vital, as they allow us to assess the outcomes of riparian restoration activities, and to make comparisons between different restoration methods under a variety of climatic and hydrologic conditions (Larsen, *et al.*, 1998). At least ten years has passed since the implementation of the Bidgee Banks Restoration Project, making it an ideal time to revisit them and determine the outcomes.

This chapter will discuss the results of this study in relation to the main aim and objectives described at the end of chapter 1. The discussion has been set out to answer each of the research objectives and null hypotheses. Section 4.2 of this chapter will discuss the research objective 1.a), evaluating how effective common riparian restoration methods are at enhancing and protecting the riparian vegetation.
This section will use the results of the riparian vegetation assessment to address the first hypothesis of this study:

Riparian restoration has not produced improvements in riparian condition.

Section 4.2 will also discuss the research objective 2. To determine how different riparian restoration methods influence different features of the riparian zone.
Section 4.3 will discuss the research objective 1.b), evaluating how effective common riparian restoration methods are at reducing stream bank erosion.
This section will use the results of the geomorphological assessment to address the second hypothesis of this study:

Riparian restoration has not produced improvements in bank condition.

This section will discuss the specific research objective 2. To determine how different riparian restoration methods influence different geomorphological attributes.

The next section (4.4) of this chapter will discuss the third research objective: To determine the factors that have affected the outcomes of riparian restoration. The conclusions will be drawn from the results of the present study and relevant literature from previous studies of interest. The final section (4.5) will discuss some of the social aspects involved in riparian restoration, post restoration maintenance and monitoring and assessment.

4.2 Riparian vegetation condition

4.2.1 Riparian Vegetation Condition

Understanding the impacts of land-use on the riparian zone is a pre-requisite to managing it. One of the management issues in many riparian zones in Australia is the presence of livestock. There have been many studies looking at the implications of livestock on the riparian zone (Robertson & Rowling 2000; Jansen, *et al.*, 2007), which have unanimously agreed that the presence of livestock cause significant damage to the riparian zone (Belsky, *et al.*, 1999). As a result, excluding livestock from the riparian zone as a means to improve riparian and bank condition has been a common practice in Australia for at least 30 years (Breckwoldt 1983). Thus many riparian restoration projects are based on the assumption that the condition of the riparian zone should improve with livestock exclusion and active revegetation.

This study demonstrated improvements in riparian condition at sites that were restored as part of the Bidgee Banks Restoration Project. After ten years, all four riparian restoration methods showed improvements in riparian condition, with livestock exclusion combined with active revegetation (tubestock planting and direct seeding) found to be the most effective way of improving riparian condition as demonstrated by RARC scores. This demonstrates the role of active revegetation in accelerating recovery and is consistent with other observations (Holl & Aide 2011). Fence and tubestock was found to be the most reliable restoration method to improve riparian condition and was (not surprisingly) the most commonly applied restoration method used in the Bidgee Banks Restoration Project, which is a common finding in Australia (Ruiz-Jaen & Mitchell Aide 2005). The revegetation efforts on a few sites completely failed, which could possibly be attributed to climatic conditions but more likely due to other factors such as lack of maintenance, failed planting/direct seeding attempts or pest animals, since most sites were successful under the same climatic conditions. Overall, however the Bidgee Banks Restoration Project appears to have led to overall improvements in riparian condition.

The success of active restoration using direct seeding varied, with direct seeded sites performing the best and worst overall. The variable nature of direct seeding was predicted by Middleton (1999). The method and timing of direct seeding seems to have contributed to its success. At sites that had successful direct seeding, landholders stressed the importance of removing the top layer of soil (to remove competition) along the lines of direct seeding, using an appropriate seed mix that was locally sourced, and timing the distribution of seed just before a large rain event. Seedling establishment has been found to increase with weed control, the leaf litter and moss removed (Knight, *et al.*, 1997; Špačková, *et al.*, 1998) and coinciding direct seeding with rainfall conditions (Clarke 2002). This thesis argues that the unpredictable nature of direct seeding as a revegetation method could be drastically improved by learning from past attempts (Knight, *et al.*, 1997) and being flexible on the timing of the implementation. As direct seeding is a cheaper revegetation method than planting tubestock (Schirmer & Field 2002), and is potentially a more appropriate method for erosion control (see section 4.2), increasing the reliability and consistency of direct seeding, may lead to better outcomes in riparian restoration at a reduced cost.

The improvements in riparian vegetation condition were found to be attributed to changes in specific RARC sub-indices and riparian attributes. For all 29 restoration sites, restoration has contributed to a change in all riparian sub-indices except sub-index *features*, which has not considerably changed ten years since restoration. Active and passive restoration has led to different changes in riparian condition (Figure 3.1). Sites restored using only fencing were observed to have riparian condition dominated by *habitat* and *debris* features (R=0.9815 and 0.9325 respectively). Sites restored using active restoration methods were observed to have riparian condition dominated by *cover* and *natives* features. These results could be attributed to the initial site conditions (pre-restoration). The majority of sites treated using only fencing, had remnant vegetation present pre-restoration. Consequently, 10 years post restoration, these sites have very high RARC sub-index *habitat* and *debris* scores because the remnant vegetation has been able to provide habitat (hollows etc.) and is of sufficient age to provide notable debris to the site. Ten years after restoration, the active inclusion of native vegetation at active restoration sites has led to improvements in vegetation cover and native vegetation cover. The next section explains how the different restoration methods have improved different characteristics of the riparian zone.

4.2.2 Habitat

The *habitat* sub-index comprises a series of measures designed to represent the features of the riparian zone that are important as habitat (for birds, small animals, etc.). These are the width of riparian canopy vegetation, longitudinal continuity of riparian vegetation and proximity to nearest patch of intact vegetation. The *habitat* features were significantly better in sites that had been restored using fencing and the planting of tubestock compared with sites that had not been restored. Riparian vegetation strongly influences the quality of habitat for a range of aquatic and terrestrial biota (Tubbs 1980; Everest & Meehan 1981). Restored sites generally had a significantly wider canopy cover than the unrestored sites. Through remote sensing improvements in canopy width were found to be greater in active restoration then passive, again demonstrating the accelerated recovery seen with active restoration (Holl & Aide 2011).

Previous research has demonstrated that the width of the riparian canopy is a contributing factor in maintaining stream and bank condition, and this contribution depends on factors such as the local geomorphology and position of the stream within the drainage network (Naiman & Decamps 1997). Naiman, *et al.*, (2010) suggested that a seven metre riparian vegetation buffer strip is adequate to provide bank stability, and Wenger (1999) showed that a 30 metre wide riparian zone is sufficient to trap sediment. The 30 metre riparian buffer width has been commonly adopted in many parts of the world (Richardson, *et al.*, 2012). The average riparian vegetation width was 19.2 metre which is nearly twice that prior to-restoration, but just short of the initial aim of the Bidgee Banks Restoration Project which was to achieve a minimum 20 metre wide riparian buffer. It should be noted that the width was also determined in part by the amount of land the landholder was prepared to 'lock-up'.

4.2.3 Cover

The *cover* sub-index comprises a series of measures designed to represent the vegetation cover of the riparian zone. These are groundcover, mid-storey cover, canopy cover and number of vegetation layers present. The *cover* features were found to be significantly better in sites that had undergone active restoration compared to sites that had been unrestored. Cover of the soil surface is important for the reduction of rainfall-driven and overland flow-driven erosion (Greene & Hairsine 2004). Riparian vegetation slows water movement and traps sediments, and is influential in mediating sediment delivery downriver (Naiman, *et al.*, 2010).

The response of vegetation to exclusion of livestock has been seen to have mixed results due to a number of factors: prior adaptation of the vegetation to grazing, availability of seed sources for recruitment, extent of degradation of the vegetation, and factors such as floods, and weeds (Jansen, *et al.*, 2007). Many studies have found an increase in growth and cover of riparian vegetation following the exclusion of livestock (e.g. Dobkin, *et al.,* 1998). Milchunas & Lauenroth (1993) identified that the species composition on a site is related to the evolutionary history of grazing of that site, with increasing grazing history favouring prostrate growth forms of species adapted to avoid or tolerate grazing. The three vegetation layers (groundcover, mid-storey, and canopy cover) varied in their response to restoration actions.

For vegetation to effectively prevent and control gully erosion the last intercepting vegetation layer must be near the soil surface (Valentin, *et al.,* 2005) and in the absence of groundcover; soil crusting, runoff generation and gully initiation can occur (Valentin, *et al.,* 2005). Ten years after restoration, groundcover did not appear to have been influenced by any of the restoration methods, with all restored sites and the control sites having very similar mean groundcover (around 80%). These findings were not expected as groundcover is removed by livestock trampling and herbivory, with many studies showing a reduction in bare-ground and increased groundcover following livestock exclusion (Wimbush & Costin 1979; Gibson & Kirkpatrick 1989; Maron & Lill 2005; Burger, *et al.,* 2010). Based on previous studies groundcover should increase with the removal of livestock. The ten years since restoration in the current project should have been ample time for improvements in groundcover to be seen given improvements have been observed within 4 years of livestock exclusion (Dobkin, *et al.,* 1998). Perhaps the similar groundcover in un-grazed and grazed sites could be a result of the higher than average rainfall that occurred in the three months prior to and during field-work (February, March, April 2014)(BOM 2014), giving an abundance of pasture for livestock to forage. Groundcover fluctuates seasonally in grazed plots (Mavromihalis, *et al.*, 2013) and year to year variation in groundcover is closely related to rainfall (Lunt, *et al.,* 2007). An alternative explanation for the similar groundcover in restored and unrestored sites could be as a result of the increased number of wild animals such as hares, rabbits and kangaroos found in restoration sites (West, *et al.,* 1984), potentially consuming similar amounts of groundcover to the livestock. The reduced impacts of trampling could have led to improvements in detrital mass which was not tested in the current study. Detrital mass improves the riparian zones nutrient filtering function (Robertson & Rowling 2000).

The mid-storey cover was not significantly affected by restoration method, but figure 3.3c illustrates that all restored sites had considerably more mid-storey cover than the control sites, irrespective of the restoration method used. Herbivory by sheep drastically reduces the cover and abundance of shrub species especially around watering points (Caughley, *et al.*, 1987) and livestock exclusion has been found to increase shrub cover (Lindgren & Sullivan 2012).

The third layer of vegetation structure is the canopy cover which was found to be significantly improved by restoration with the greatest improvements in canopy cover found in sites with active restoration. These results demonstrate that (at least some of) the plants that were planted or direct seeded have survived and after ten years are contributing to canopy cover. As many of the revegetated canopy species were not fully grown at the time of monitoring, the canopy cover is likely to continue to increase. The canopy cover was found to be at least eight times greater after 30 years of livestock exclusion in a study by Schulz & Leininger (1990).

4.2.4 Native vegetation

Native vegetation is usually a key objective of restoration projects, so it is useful to consider how the restoration activities affect native vegetation. This can be explored through the RARC sub-index *natives*, which comprises measures of the native vegetation within the three structural vegetation layers in the riparian zone. The *native* features were better in sites in all restoration methods compared to the unrestored sites, with fencing and the planting of tubestock and fencing and direct seeding leading to the highest RARC sub-index *natives* scores (Figure 3.2c). The amount of native cover present in the three vegetation layers that contribute to the RARC sub-index *natives* varied between restoration method used and the control sites.

There was little difference in native groundcover between restored and unrestored sites. Australian riparian groundcover species are generally not adapted to grazing by hard-hooved grazing animals (Jansen, *et al.*, 2007) and livestock grazing at high densities can eliminate native grasses which are replaced by exotic annuals (Caughley, *et al.*, 1987). Herbivore exclusion has been seen to dramatically improve the survival rates of native grass species (Allcock & Hik 2004). It is possible that many of the sites assessed as part of this study have been heavily grazed for so long that recovery of native groundcover species has not yet occurred, and will either require more time or active intervention (such as the direct seeding of grass species). Lunt, *et al.*, (2007) demonstrated that improvements in native groundcover following livestock exclusion were slow only showing dramatic improvements by the 12th year of sampling. Native forbs and native grasses were found to respond differently to grazing (Stahlheber & D'Antonio 2013), and maybe these two groups should have been assessed separately. In the case of the Bidgee Banks Restoration Project, native groundcover was highest at sites with abundant remnant vegetation present. As seedling recruitment rates were found to be increased with native groundcover, choosing riparian restoration sites with remnant vegetation present would contribute to the speed and extent of recovery.

The control sites had very little native mid-storey cover, with 7 of the 9 control sites having no native mid-storey, and the remaining two having a mean native mid-storey cover of less than 2%. These results demonstrate the negative impact that livestock have on native mid-storey cover. In contrast, the restored sites had a mean native mid-storey cover of 7.5%. Spooner and Briggs (2008) showed similar results with significantly higher mid-storey cover in fenced than unfenced and grazed areas. The active restoration methods resulted in sites with a greater native mid-storey cover than the passive approach suggesting that planting of native mid-storey species is key to restoring this element of the vegetation structure.

Native canopy cover was also found to be significantly improved by active restoration, with fence and tubestock the most reliable method of increasing native canopy cover. The results from the remote sensing analysis confirmed this, with projected foliage cover considerably improved in sites in all restoration methods but especially those that were fenced and planted with tubestock or direct seeded. The control sites showed very little change in projected foliage cover in the ten years since restoration, suggesting that foliage cover will not improve without intervention (restoration). Further this finding shows that improvements seen in canopy cover in all restoration methods can be attributed to restoration actions.

4.2.5 Debris

The *debris* Sub-index comprises a series of measures designed to represent the features of the riparian zone that contribute to debris. These are leaf litter, native leaf litter, coarse woody debris, hollows and standing dead trees. The *debris* features were not significantly different in sites in all restoration methods and unrestored sites (Figure 3.2d). The occurrence of debris (such as coarse woody debris) on the banks of streams reduces sediment and nutrient delivery in to streams as it can slow the flow of water, which increases the opportunity for biological uptake or physical adsorption (Gregory, *et al.*, 1991). The evolution of floodplain landscapes initially requires the trapping of sediment and the formation of terraces where vegetation can become established, and debris contributes to the trapping and isolation of sediments (Abbe & Montgomery 1996; Lovett, *et al.*, 2005). Generally there is a time delay for revegetated species to be effectively contributing debris, depending on the type of debris

this has been estimated to be upwards of 100 years (Manning, *et al.*, 2013). Robertson and Rowling (2000) identified that excluding livestock from the riparian zone dramatically increased the quantities of debris present on river banks, with the most marked differences occurring after 50 years of exclusion. The results of the present study confirm that ten years has not been long enough for riparian restoration to be contributing to debris.
Restoration method had no significant effect on all indicators from the RARC sub-index *debris*. Although not significant there were some differences that were noted between restoration methods. Sites that were fenced and planted with tubestock, and fenced and direct seeded had the highest amount of leaf litter and the control sites had the least leaf litter. Vegetative cover of the riparian zone is the main factor influencing leaf litter input (Naiman & Decamps 1997), and this result (although not significant) gives evidence that the actively revegetated plants are currently at an age where they are contributing to leaf litter production. There was very little difference found between the restoration methods and the control sites in all other debris indicators. These results demonstrating that the vegetation is at an age where it would be rare to produce hollows (Bennett, *et al.*, 1994), large woody debris (Manning, *et al.*, 2013) or other debris attributes.

There have been studies undertaken on biological responses to the inclusion of woody debris (such as reptile abundance) (Shoo, *et al.*, 2014; Manning, *et al.*, 2013). The addition of coarse woody debris would increase habitat and biological diversity, influence channel morphology, and water chemistry (Lester & Boulton 2008). To my knowledge there are no studies on how/if the inclusion of woody debris in to riparian restoration will accelerate improvements in bank stability and erosion. For this reason it is proposed that further studies looks at the benefits of including woody debris in to riparian restoration tasked at improving bank condition.

4.2.6 Features

The RARC sub-index *features* comprise a series of indicators of riparian condition. These are seedling recruitment of mid-storey species, seedling recruitment of canopy species, large native tussock grasses and reeds. The sub-index *features* was found to be unaffected by restoration (Table 3.1 & Figure 3.2e). The indicators from the RARC sub-index *features*, varied in their response to restoration.

The results of this study demonstrate that the presence of livestock have a negative influence on the abundance of seedlings of mid-storey and canopy species, with very few seedlings seen in control sites. There were no mid-storey seedlings present in any of the control sites and seedlings of canopy species present on only two control sites. This demonstrates the importance of livestock exclusion for long-term restoration outcomes, as it re-iterates what others have found on the low rates of seedling survival in the presence of livestock (Jansen & Robertson 2001). In the restored sites mid-storey seedling recruitment was significantly better and canopy species seedling recruitment was not significantly better but more abundant than the unrestored sites. These findings were similar to those of Robertson and Rowling (2000) who found that seedlings were nearly completely absent in grazed sites and present in un-grazed sites. Seedling recruitment has been found to vastly improve with livestock exclusion (Spooner, *et al.*, 2002; Allcock & Hik 2004) with the density of seedlings increasing with time since exclusion commenced (Spooner & Briggs 2008). The ability of livestock to control the density of canopy trees through their impacts on seedlings and saplings has major impacts on the landscape as mature individuals age and die (Caughley, *et al.*, 1987). Obviously seed has to be present either in the seed bank or on existing vegetation for passive restoration to be successful, and comparisons between remnant vegetation and seedling recruitment demonstrates the importance of remnant vegetation to the success of passive restoration. Ten years after restoration the abundance of mid-storey recruitment was associated with active restoration and the presence of remnant vegetation (figures 3.3k and 3.12a), but the recruitment of canopy species was only occurring at sites with remnant vegetation present (Figure 3.12b). These results show that after ten years actively revegetated mid-storey species have successfully reproduced and canopy species have not.

4.2.7 1st A priori null hypotheses

The results demonstrated significant improvements in riparian condition and riparian vegetation characteristics that are attributed to riparian restoration activities. The results of the riparian vegetation assessment as part of this study mean that the first a priori null hypothesis: Riparian restoration has not produced improvements in riparian condition was disproved.

4.3 Bank and channel condition

4.3.1 Total bank condition

Watersheds in agricultural landscapes have been shown to be large contributors of phosphorus and sediment in surface water (Alexander, *et al.*, 2007) with large amounts of phosphorus and sediment lost from eroding stream banks (Schwarte, *et al.*, 2011). It is well recognised that riparian vegetation increases bank stability and reduces bank erosion (Abernethy & Rutherfurd 2000), and conversely that bank erodibilty increases with the removal of riparian vegetation (Micheli, *et al.*, 2004). Muñoz-Robles, *et al.*, (2010) identified two characteristics of a stable gully; these were high groundcover and foliage projective cover of trees and shrubs.

In this study, sites in all restoration methods had significantly better bank condition and erosion state (as evidenced by the ephemeral stream assessment scores) than unrestored sites. The results demonstrate that ten years has been a sufficient time for improvements in bank condition to be seen. Little difference was observed between passive and active restoration methods in mean ephemeral stream assessment scores. The similarity in scores poses an important question: is fencing to exclude livestock sufficient to control erosion and improve bank condition and is revegetation not necessary? Excluding livestock from the riparian zone has been found to lead to less active bank erosion, trampling and tracks compared to sites with continuous livestock access (Burger, *et al.*, 2010; Zaimes, *et al.*, 2004). Potentially, the benefits associated with the revegetation have not been seen yet as the vegetation is not at an age to be contributing to the geomorphological condition. The effectiveness of riparian vegetation in supplying ecological services to the river corridor is determined by the height of the vegetation and its distance from the stream edge (Naiman, *et al.*, 2000). The results from the vegetation assessments demonstrated that ten years after restoration, active restoration significantly increased native canopy cover and width of canopy vegetation, but this does not suggest that the vegetation has reached its maximum height or width of canopy vegetation (and these indicators will likely keep increasing). Further; there were some riparian characteristics which were so far not-significantly affected by restoration such as those within the RARC sub-index: *debris*. It is proposed that all restoration methods have seen the improvements associated with the removal of livestock, such as the reduced trampling, tracks, and soil compression, and as the vegetation matures sites will continue to improve. Improvements such as denitrification and nutrient uptake increase with age of riparian vegetation (Line, *et al.*, 2000). It is recommended that re-monitoring occurs to compare the bank condition of active and passive restoration sites to understand the specific benefits attributed to livestock exclusion and the benefits attributed to revegetation.

Riparian vegetation increases bank stability through the roots providing resistance to shear beyond that at which soil can handle (Docker & Hubble 2008). Docker and Hubble (2008) identified that the roots shear resistance increases when roots are well developed across the entire width of the soil block. Further; Docker and Hubble (2009) identified a rapid reduction in root material with lateral distance from the tree stem. In the Bidgee Banks Restoration Project direct seeding was implemented by ripping trellises longitudinally or laterally in one instance, on the flats near the eroded gully then distributing seed along these trellises. The plants grown from direct seeding have grown very close together in tight bands, and it is likely that the roots are tightly developed over the width of the soil block. It is predicted that this has led to an increase in shear resistance greater than that of the tubestock plantings which were generally more sparsely planted (roughly every 5 metres). For this reason, it is proposed that direct seeding is a more appropriate method then tubestock planting to achieve this. To my knowledge there have been no studies on the most appropriate revegetation method to increase bank stability and future research is recommended to confirm that direct seeding is a more appropriate active restoration method for bank stability then tubestock planting.

In the Upper Murrumbidgee Catchment, over grazing and degradation of the valley bottoms by livestock in the mid 1800's is blamed for triggering much of the gully erosion in the area and increasing sediment in the Murrumbidgee River (Olley & Wasson 2003). Since this time it is thought that many of the eroded gullies have been gradually stabilising (Starr, *et al.,* 1999). A study by Jansen (2001) demonstrated very little difference in bank condition between sites regardless of stocking rates, with most sites having a poor soil structure, which was presumed to be the result of a long history of livestock activity in the riparian zone. The current study demonstrated significant differences between restoration sites and the control sites in bank stability and erosion state using the ephemeral stream assessment (Machiori, *et al.,* 2003). This contrary finding to Jansen (2001) could have been as a result of the bank condition assessment method being more rigorous (8 indicators performed at each transect compared with Jansen's 3), or perhaps it could demonstrate that bank condition does not change unless livestock are completely excluded.

4.3.2 Shape of the cross-section

Crouch (1987) identified that different erosion rates and associated sediment delivery in to streams is associated with different side wall forms. Vertical walls have the highest erosion rate and side-walls start to stabilise as the wall angle becomes <65˚ (Crouch 1987). The scoring system for the ephemeral stream assessment indicator *shape of cross-section* was broadly based on the findings from Crouch (1987). As the current study investigates the reduction in sediment delivery through erosion control this indicator is of importance. The results demonstrated that all restoration methods sites with significantly better *shape of cross-section* scores than the control sites. The result potentially demonstrating that the Bidgee Banks Restoration Project has met its initial project objective of reducing sediment delivery in to the Murrumbidgee River. Although there were differences in the spread of data within restoration methods, all restoration methods had similar mean *shape of cross-section* scores. These findings re-iterate the findings from the overall ephemeral stream assessment in that it is likely that the benefits associated with livestock removal have been seen in sites in all restoration methods and the benefits associated with the vegetation may progressively improve the scores of the indicator *shape of cross-section*.

4.3.3 Condition of the drainage-line (longitudinal morphology)

The results demonstrated very little difference in the condition of the drainage-line between restoration methods and control sites. The relief, slope and valley morphology influences the potential energy of a landscape, the way the energy is used, and the concentration of flow (Brierley & Fryirs 2005). For this reason the primary control on the condition of the drainage-line is the slope of the surrounding landscape. This is not to discount that restoration may potentially lead to improvements in the condition of the drainage-line, but that slope has to be taken in to consideration. The number of sites at which each restoration method was employed was too small to be broken in to slope classes.

4.3.4 2nd A priori null hypotheses

The results demonstrated significant differences in bank and channel condition in restored sites compared to unrestored sites. The results of the geomorphological assessment as part of this study mean that the second a priori null hypothesis: Riparian restoration has not produced improvements in bank condition was disproved.

4.4 Influential factors of riparian restoration outcomes

4.4.1 Presence of remnant vegetation, seedling recruitment and debris

The presence of remnant vegetation in the riparian zone has been found to have many important ecological functions. For example the biomass of non-native annuals were found to be significantly greater in cleared land then uncleared (Hobbs & Atkins 1991).

The results of the present study demonstrated that the abundance of canopy and mid-storey recruitment was strongly associated with the abundance of remnant vegetation. The increased seedling recruitment seen in areas with remnant vegetation could have been attributed directly to the seed dropped by the remnant vegetation, but may be also a result of seed dispersal from frugivorous animals who congregate on or around remnant vegetation (Špačková, *et al.*, 1998; Toh, *et al.*, 1999). This finding demonstrates that passive restoration by simply excluding livestock can be an acceptable restoration method if there is remnant vegetation present on the site.

Comparisons between abundance of remnant vegetation and RARC sub-index *debris* demonstrated that the abundance of remnant vegetation is an important factor in the amount of debris on a site. The predicted time taken for actively revegetated trees to have hollows (Whitford 2002), and to input large woody debris in to the system (Manning, *et al.,* 2013) is a reminder of the importance of protecting and conserving remnant vegetation, and in selecting riparian restoration sites with remnant vegetation present.

4.4.2 Native groundcover and seedling recruitment

The abundance of seedlings of both canopy and mid-storey species was positively related to the amount of native groundcover. Similar results were found by Spooner and Briggs (2008) who showed that tree recruitment was positively correlated to native perennial grasses and negatively correlated with cover of exotic annual grasses. As seedling recruitment is vital to the long-term success of restoration, the abundance of native groundcover is a factor that will determine the extent and speed of recovery. This result would suggest that if native groundcover is relatively low (< 40%) on a future restoration site, management should aim to improve the amount of native groundcover on site.

4.4.3 Non-native cover

Heavy grazing by livestock can push the competitive balance from native to non-native vegetation (Holmgren 2002). For this reason grazed sites have been found to have a higher non-native plant cover compared to un-grazed sites (Hobbs 2001). Non-native species negatively impact on the riparian zone in a range of ways. Non-native vegetation can reduce or eliminate native perennial grasses and forbs (McIver & Starr 2001), reduce the species richness of native herbaceous plants (Abensperg-Traun, *et al.,* 1998), and reduce the abundance of seedlings of native woody species (Hobbs 2001). Grazing exclusion has been found to lead to varying outcomes for non-native species depending on factors such as the condition of the ecosystem (Lunt, *et al.,* 2007). Livestock exclusion can result in the dominance of non-native species in degraded ecosystems on well watered and fertile soils (such as the riparian zone) (Lunt, *et al.,* 2007). As non-native species differ in their influence on ecosystems, we need to differentiate between the ones that require management and those that do not (West 1993).

The present study demonstrated that restoration generally had no effect on non-native groundcover. Forbs and grasses have been found to respond differently to grazing pressures (Stahlheber & D'Antonio 2013), and these were not assessed separately in the present study. Perhaps alternatively 10 years has not been an adequate amount of time for livestock exclusion and revegetation to reduce the non-native groundcover. Perhaps livestock exclusion is not enough and weeding (hand weeding or spraying) is required to remove these species.

Jansen & Robertson (2001) identified that the majority of private property along the Murrumbidgee River including (semi-pristine) reference sites had a mid-storey consisting of mainly non-native species. Through discussions with landholders in the present study, one negative impression of livestock exclusion was that the absence of livestock herbivory increases the abundance of non-native plants and in particular mid-storey species such as *Rubus sp.* (blackberries), *cotoneaster sp*., *Ulex sp.* (gorse), and *Pyracantha sp*. (fire-thorn). It has been found that some landholders think that livestock exclusion increases non-native species (Robertson & Rowling 2000; Jellinek, *et al.,* 2013). The present study demonstrated that livestock exclusion has potentially led to an increase in the non-native mid-storey cover, although this result was not statistically significantly (Figure 3.16b). Interestingly this result was only found at sites at which fence only was the method of restoration. The increase in non-native mid-storey cover could be a result of the past-disturbance (livestock grazing) creating a new trajectory of succession involving native and non-native species, and this result has been found before (Allen 1995). All sites that had been actively restored had proportionally less non-native mid-storey (and conversely proportionally more native mid-storey). This result potentially demonstrates that planting native mid-storey species can fill the niche that would be available for non-native mid-storey species, reducing the abundance of non-native mid-storey species. Hobbs and Atkins (1991) showed that intact shrub canopies effectively prevent the establishment of non-native annual species. This idea was touched on by Kauffman, *et al.,* (1997) who explain how active intervention is often necessary, to avoid exotic competitors.

A contributing factor in the presence of non-native plant species is the historic role they have played in land management. Often native and non-native plants offer similar benefits to the riparian zone by controlling erosion and increasing bank stability (D'Antonio & Meyerson 2002). Historically non-native species have been used in restoration of badly degraded riparian areas because of their fast growth-rate and high survival rates (D'Antonio & Meyerson 2002). In the Bidgee Banks restoration sites it was likely that most of the non-native canopy species found were planted as part of past restoration efforts. The majority of sites had very low non-native canopy cover and there was no significant effect of restoration method.

4.4.4 Rainfall and temperature

Variation in plant productivity can largely be accounted for by rainfall (Caughley, *et al.,* 1987). Since the implementation of the Bidgee Banks Restoration Project, the restoration sites have been subject to fairly harsh climatic conditions, with the restoration coinciding with a long and harsh drought. The east of Australia was in a period of serious to severe rainfall deficiency over most of 2002, and these dry conditions remained with little relief until around 2006 (BOM 2014). Preliminary monitoring of the Bidgee Banks Restoration Project identified that survival rates of tubestock plantings were between 58% - 69%, (Patmore & Davey 2004) which was slightly lower than expected by Greening Australia (about 80%), and this could have been attributed to the dry conditions. The amount of rainfall following restoration did not have a notable effect on the long term restoration outcomes (Table 3.4) particularly given that the short term outcomes (about 2 years) did demonstrate an impact (Patmore & Davey 2004). The influence of drought conditions on plant mortality is increased with increased temperature (Adams, *et al.,* 2009). The temperature following restoration also did not have a notable effect on the long term restoration outcomes (Table 3.5). The non-significant results could have been a result of the analysis, as plant survival rates could not obtained, as it would have been impossible to know how many plants germinated from the direct seeding. This could have led to the influence of climatic conditions not being identified.

4.4.5 Influence of vegetation on the drainage-line wall and floor at maintaining the condition of the drainage-line floor.

Stream channel morphology is influenced by fluvial processes and landforms external to the channel, generally with the gradient of the surrounding hill slopes determining the level of control the surrounding landscape holds over the longitudinal morphology of the drainage-line floor (Grant & Swanson 1995). Valley floor morphology in flat landscapes is primarily shaped by fluvial processes, and in mountainous areas fluvial and non-fluvial processes shape the drainage-line floor (Grant & Swanson 1995). Micheli & Kirchner (2002) identified that the strength and stability of the stream bank increases with the density of sedges and rushes on the drainage-line. Within gully flow is most erosive where flow comes in to direct contact with gully sides (Crouch 1987) and emergent and sub-mergent vegetation prevents scouring of the banks during high flow events (Naiman, *et al.*, 2010).

Vegetation on the drainage-line floor was associated with the condition of the drainage-line in flat landscapes and the strength of this association was found to decrease as the surrounding landscape becomes steeper (Figure 3.14). Conversely the vegetation on the drainage-line wall was found to have the opposite effect. The vegetation on the drainage-line wall was associated with the condition of the drainage-line in very steep areas (slope >30˚), and this association reduced in flat landscapes (Figure 3.15).

In a practical sense, these findings demonstrate that in recommending restoration actions for streams, revegetating the drainage-line floor should be a priority in flat landscapes, whereas revegetating drainage-line walls a priority in mountainous landscapes. The Bidgee Banks Restoration Project did not include revegetation within the channel and these results suggest that the inclusion of aquatic or semi-aquatic species would be advisable where practical. It should also be noted that the timing of planting or seeding of these species is vital, as seedling mortality is very high following flooding episodes (Middleton 1999).

4.4.6 Species used in riparian restoration

The first four **A**ttributes of a Restored Ecosystem described in the SER International primer on ecological restoration (2004)**,** describe species assemblages, indigenous species composition, community structural attributes and how capable the environment is of allowing species reproduction. Many of the sites that involved active restoration contained no remnant vegetation and as such it is important to address these attributes when assessing revegetation success.

Survival rates of revegetated species

The results of the present study demonstrated large differences in the survival rate of different species used in the riparian restoration (Figure 3.17 and appendix 2). Factors such as water availability, channel stability, and soil salinity determine whether or not plantings will survive (Briggs 1996). Most of the sites were initially badly affected by erosion, salinity, and with little or no canopy cover, or native groundcover. It is possible that some of the species selected were simply not suitable for the harsh conditions and consequently certain species were favoured more than others. A solution to increase the number of species surviving on a site could be a staggered approach where only hardy, early succession species are planted/seeded and given time to colonise and make the environment more habitable before planting less hardy species. A similar successional approach has been adopted to rehabilitate degraded mine sites, where suitable plants are initially planted, which over time provide organic matter, lower soil bulk density, and bring minerals to the surface (Bradshaw 1997). In a project such as the Bidgee Banks Restoration Project which aims to reduce sediment delivery through increasing bank stability, and not focused on improving biodiversity a focus on *Acacia*, *Eucalyptus*, and *Casuarina* is likely to have the most beneficial outcomes.

The roots of riparian plant species have been found to vary between species in depth and lateral distance from the stem (Hubble, *et al.*, 2010) and root strength (Abernethy & Rutherfurd 2001). The differences in root architecture between species have been found to determine their ability to re-enforce the soil (Docker & Hubble 2008). Docker and Hubble (2008) found that *Acacia floribunda* exhibit a greater earth reinforcement potential then *Eucalyptus amplifolia* as a result of the species well-spread, highly branched fine roots, with high tensile strength. Abernethy & Rutherfurd (2000) showed that although certain species have greater root reinforcement then others, the combination of two species further increase the bank stability. These findings along with the findings of the present study on the survival rate (Figure 3.17) and recruitment rate (List 3.1) of *Acacia* species, is good evidence that

Acacia species are an appropriate species for riparian restoration directed at increasing bank stability (at least initially, until the site has improved).

Seedling recruitment of revegetated species

From the SER International primer on ecological restoration (2004)**,** one attribute of a restored ecosystem is that the environment is capable of allowing species reproduction. Successful riparian restoration requires the establishment of ecological and physical conditions that will be naturally sustainable (Goodwin, *et al.*, 1997). After ten years, most species planted as part of the Bidgee Banks Restoration Project have not successfully recruited seedlings. Robinson & Handel (1993) showed similar results with revegetated plants contributing very few seedlings in the restoration of landfill sites. Seedling recruitment and survival is generally low, Clarke (2002) found that between 2 and 14% of germinable seed sown emerged, and out of that less than 1% survived to become juvenile plants. The results of the present study demonstrated overwhelmingly that at a ten year old restoration site *Acacia* species have the highest chance of successfully recruiting seedlings. Seedling recruitment rates are species specific (Clarke 2002) and *acacia* species have been seen to recruit well in low-rainfall, Mediterranean type climates (Knight, *et al.*, 1997; Toh, *et al.*, 1999). Potentially other species such as those in the genus *Eucalyptus* may require longer to reach maturity and produce seed and this should be acknowledged when interpreting these results. These results support the recommendations of Robinson & Handel (1993) on choosing species with early reproductive capacity and high seed production (such as *Acacia*) when restoring degraded ecosystems.

4.4.7 Research objective 3: To determine the factors that have affected the outcomes of riparian restoration.

The complex and dynamic nature of riparian ecosystems makes it important to develop management strategies on a site-by-site basis (Briggs 1996). Initial site conditions such as abundance of remnant vegetation, amount of native groundcover, and slope of surrounding landscape where all found to affect the outcomes of riparian restoration. These conditions should be taken into consideration when deciding on initial management strategies and it is likely that some were in the Bidgee Banks Restoration Project. A return to a pre-disturbance state may be achievable simply by removing the anthropic stresses (livestock grazing) in situations where the degradation has not progressed too far (Aronson, *et al.*, 1993) i.e. sites with some remnant vegetation of canopy and mid-storey species, and native groundcover (>

40%). In these situations the results have demonstrated that the removal of livestock will lead to increased seedling recruitment of canopy and mid-storey species, and improvements in bank and drainage-line condition. Pre-restoration many of the restoration sites were in a degraded condition, with no remnant vegetation, and fairly severe erosion occurring on site. As Robinson & Handel (1993) explains the successional process of vegetation reestablishment cannot be expected to occur on these sites. One reason for this is the lack of seed dispersal, as animal dispersers are attracted to trees and shrubs, and without them the seed of woody species will not arrive on site (Robinson & Handel 1993). In cases such as these, often the ecosystem requires active restoration to begin recovery, as they have passed the point where recovery can happen without intervention (Aronson, *et al.*, 1993). The results of the riparian vegetation assessment demonstrated that actively introducing vegetation on to the restoration site leads to improvements in a range of riparian indicators after ten years. The species used in active restoration were found to vary considerably in survival rates and seedling recruitment rates, and this is a major consideration when selecting appropriate species.

4.5 Riparian restoration, monitoring and assessment

4.5.1 Social aspects of riparian restoration

A large majority of the riparian zone in Australia is privately owned and the responsibility for riparian restoration lies primarily with the landholders. In this way riparian restoration and the resource management of clean water are very much social problems (Thomson & Pepperdine 2003). Successful riparian restoration is thus grounded in the integration between the social and natural dimensions of decision making (Naiman, *et al.*, 2010).

The way in which each site was selected for restoration works as part of the Bidgee Banks Restoration Project was initially by the landholder applying to receive assistance. Greening Australia work on the premise that it is difficult to make a landholder do something they do not believe in or approve of, and working with the willing is a priority for the organisation. A major barrier faced in the implementation of riparian restoration is the community mistrust in government organisations, and the information they provide (Thomson & Pepperdine 2003). Gould (2007) identified that funding was often a limiting factor in the Bidgee Banks Restoration Project, and that extra funding could have potentially been used to persuade a landholder to engage in the Bidgee Banks project. Through discussions with landholders in the present study, there were incidences where neighbouring farmers adopted restoration actions as they saw the benefits of restoration unfold.

In a postal survey on the landholder management of river frontages in the Goulburn Broken Catchment undertaken by Curtis and Robertson (2003), when asked if grazing of domestic stock has had a major impact on the existence and diversity of native vegetation on river/ creek frontages: 46% agreed, 18% were not sure, and 37% disagreed. These results would suggest a confliction in the opinion of landholders on the impacts of livestock on native species.

Education and community engagement has become a very useful tool in restoration. As described by Curtis and Robertson (2003) community education of river function and factors that affect river and bank condition will lead to an increase in the number of landholders adopting restoration. Naiman (2013) gives credit to the management and implementation of a river restoration project undertaken in Moreton Bay, Australia, as it demonstrated the power of illustrating the potential financial losses that could be incurred if the situation did not quickly improve.

Overall, landholders seemed very positive about the Bidgee Banks Restoration Project and seem to have become more aware about the implications of livestock on riparian and river condition as a result. Many landholders described personal observations of the improvements they have seen on their restoration site/s including: increased birds to the area, improved groundcover, increased native species (animals and plants), and increased aesthetic value. There seemed to be a relationship between the success of a project and the landholder’s confidence in restoration. The landholder’s increased confidence often leading to the amount they were prepared to maintain the restoration site and further invest into other projects. Curtis and Robertson (2003) found a relationship between the adoption of current recommended practices such as watering stock off-stream and fencing river frontage and the landholder’s confidence in adopting these practices.

The level of financial support for a restoration project is often related to its contribution towards regional priorities such as for natural resource management (NRM) purposes. Riparian restoration undertaken for NRM outcomes, such as the Bidgee Banks Restoration Project often has multiple benefits over and above the main project objectives. As part of a questionnaire to landholders on their attitude to riparian restoration undertaken by Jellinek, *et al.*, (2013), the majority of landholders thought that remnant and revegetated areas reduce wind damage to livestock, increase native animals, and improve the aesthetic value of their property. These on –site benefits potentially have financial gains for landholders and highlighting these might be a good way to promote riparian restoration to unconvinced landholders.

4.5.2 Post-restoration management

4.5.2.1 Grazing of restoration sites

Livestock grazing of restoration sites was predicted as a likely influence of the success of sites restored as part of the Bidgee Banks Restoration Project (Patmore and Davey 2004). Some landholders described allowing intermittent livestock grazing of their restoration sites, to reduce vegetation cover (to reduce the threat of fire), reduce weed species, or as a source of feed. As explained by Spooner and Briggs (2008) the aim of providing fencing incentive funds is to assist landholders to better manage grazing and not necessarily to create a grazing exclosure. Intermittent grazing has been seen to maintain local and regional biodiversity in native grasslands (Dorrough, *et al.*, 2004). Non-native species can be reduced using intermittent grazing (Frost & Launchbaugh 2003) but grazing-sensitive native species may also decline (Mavromihalis, *et al.*, 2013). Westoby, *et al.*, (1989) range successional model predicts that drought conditions affect vegetation in a similar way that grazing pressures do. Therefore; land managers should respond to drought conditions by lowering grazing pressures so the combined effects of these pressures are minimised as much as possible. The reality of land management is that many of the riparian restoration sites were made available for grazing in times of drought due to necessity. This has most likely exacerbated the impacts of the dry conditions on the riparian vegetation and bank condition. There is a lack of essential information on the effects of different grazing strategies on biodiversity in remnant vegetation in south-eastern Australia (Dorrough, *et al.*, 2004). Grazers potentially act as keystone species, maintaining short and open vegetation and preventing the establishment of woody species (WallisDeVries, *et al.*, 1998). The high seedling mortality rates seen in the control sites, is evidence that riparian forests could be transformed in to open grasslands by livestock grazing. It is for this reason that allowing livestock to graze riparian restoration sites (or remnant vegetation) to control weeds or reduce cover should be seen as a last resort.

4.5.2.2 Controlling non-native plant species

In the absence of livestock herbivory, sites may require ongoing site maintenance to control non-native plant species, as non-native species may increase (Mavromihalis, *et al.*, 2013). Site maintenance effort varied considerably, from none through to spraying herbicide, hand weeding, and intermittent livestock grazing (strategically allowing livestock access to the site as a means to reduce non-native cover). As the results demonstrate that some non-native species may increase in the absence of livestock, control measures should be included as part of the riparian restoration strategy. The results showed that including a native mid-storey could reduce the abundance of non-native mid-storey species, and this strategy could reduce the on-going site maintenance required.

4.5.2.3 Livestock exclusion fence; repair and up-keep

As reported by Bernhardt, *et al.*, (2007) the majority of restoration project managers (60%) say that once implemented riparian restoration sites require ongoing site maintenance. Unfortunately there were a few restoration sites where livestock exclusion fences were damaged or degraded, enabling the access of livestock in to the restoration site, and in all of these cases the landholder was aware of this. At these sites there were often obvious signs of damage to the bank or vegetation. In a few instances discussions with landholders prompted them to fix their fence, demonstrating the importance of re-visiting sites.

4.5.3 Benefits of remote sensing

As this study did not have the benefit of pre-restoration data, aerial imagery taken before and ten years after restoration enabled the comparison between before and after restoration. This study demonstrated the effectiveness of using aerial imagery in assessing restoration outcomes. This study demonstrated that two riparian attributes: projected foliage cover and width of riparian vegetation could be monitored from aerial imagery. As field work is expensive and time consuming, monitoring riparian restoration without visiting the site is of great benefit. The use of satellite imagery to survey vegetation has been used as an effective monitoring tool of the riparian zone by others (Wilkinson, *et al.*, 2004; Johansen & Phinn 2006). To my knowledge the use of image recognition software has not been used on aerial imagery to monitor projective foliage cover of riparian restoration sites before and after restoration. This study demonstrated that using image recognition software such as WinDIAS on aerial imagery can dramatically improve precision, and reduce monitoring time.

4.6 Conclusions

Riparian and river restoration is an increasingly common approach to water resource management (Bernhardt, *et al.*, 2007). The Bidgee Banks Restoration Project encompassed very typical riparian restoration objectives (reducing sediment and nutrient delivery in to waterways), which are being tackled worldwide. This study gives encouragement that projects such as the Bidgee Banks Restoration Project are an effective and important component of NRM. The cumulative effect of all the Bidgee Banks Restoration sites has potentially yielded great ecological benefit (Palmer, *et al.*, 2005), hopefully reducing the sediment and nutrient delivery in to the Murrumbidgee River.

The results demonstrated that the impacts and implications of livestock farming on riparian and freshwater ecosystems can be minimised by riparian restoration. The results support others on the importance of excluding livestock from the riparian zone to restore bank and riparian condition. Active restoration was found to significantly increase certain attributes of the riparian zone, such as the width of the riparian canopy vegetation, native canopy cover, native mid-storey cover and seedling recruitment. Certain site conditions were found to be influential over the outcomes of riparian restoration (McIver & Starr 2001), such as the abundance of remnant vegetation, native groundcover and slope of surrounding landscape. These results can be used to develop and adapt management guidelines for riparian restoration. This study did demonstrate that many of the riparian components have not changed dramatically since restoration, demonstrating the often slow nature of ecosystem recovery (Nilsson, *et al.*, 2014) and therefore the importance of long-term monitoring. In the future, the riparian zone will have to accommodate to changes in population density and increased resource consumption, and will likely be expected to fulfil even more roles than today (Naiman & Decamps 1997). These predicted pressures and expectations only increase the importance of meeting riparian restoration objectives and justify investment in to research in riparian restoration ecology.

4.7 Research limitations

The limitations associated with contacting landholders and gaining access to the restoration sites resulted in each restoration method having a relatively small sample size. A larger sample size may have led to restoration methods being more significantly different from each other and the control sites, as there could have been less variation among the mean. A larger sample size could have also allowed more variables to be investigated.

A challenge in a study of this kind is the variation in the condition of sites before restoration commenced. The history of land-use could have led to differences in the condition and response of the site to restoration actions. It was likely that sites varied considerably in the extent of grazing pressures in the years leading up to the livestock exclusion. The number of livestock or years of livestock farming at each site was not obtained as it would have been impossible or very difficult to acquire in most cases as properties have changed hands.

In a few cases the livestock exclusion fences were damaged allowing livestock access to the restoration site. In other cases landholders would allow livestock access for short periods of time. At these sites there were signs of bank damage as a result of livestock movement and vegetation damage as a result of herbivory. This meant that the condition of these sites were the result of partial and not total livestock exclusion.

Grazing by mammals and invertebrates has been seen to increase in livestock exclosures resulting from the increased availability of food and cover (Belsky, *et al.*, 1999). This confounding effect could have reduced the detectability of differences between restored and unrestored sites. There were animals such as rabbits and kangaroos seen in many sites, and these animals could have reduced the cover of vegetation or reduced the condition of the bank.

References

Abbe T.B. & Montgomery D.R. (1996) Large woody debris jams, channel hydraulics and habitat formation in large rivers. *Regulated Rivers Research & Management,* **12,** 201-221.

Abensperg-Traun M., Atkins L., Hobbs R. & Steven D. (1998) Exotic plant invasion and understorey species richness: a comparison of two types of eucalypt woodland in agricultural Western Australia. *Pacific Conservation Biology*, **4,** 21-32.

Abernethy B. & Rutherfurd I.D. (2001) The distribution and strength of riparian tree roots in relation to riverbank reinforcement. *Hydrological Processes,* **15,** 63-79.

Abernethy B. & Rutherfurd I.D. (2000) The effect of riparian tree roots on the mass-stability of riverbanks. *Earth Surface Processes and Landforms*, **25,** 921-937.

ACT Government (2003) *Design standards for urban infrastructure, urban wetlands lakes and ponds*. Territory and Municipal Services, Canberra.

Adams H.D., Guardiola-Claramonte M., Barron-Gafford G.A., Villegas J.C., Breshears D.D., Zou C.B., Troch P.A. & Huxman T.E. (2009) Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought. *Proceedings of the National Academy of Sciences of the United States of America,* **106,** 7063-7066.

Aguiar F.C., Fernandes M.R. & Ferreira M.T. (2011) Riparian vegetation metrics as tools for guiding ecological restoration in riverscapes. *Knowledge and Management of Aquatic Ecosystems,* **402**, 21-33.

Alcamo J., van Vuuren D., Cramer W., Alder J., Bennett E., Carpenter S., Christensen V., Foley J., Maerker M. & Schulze K. (2005) Changes in ecosystem services and their drivers across the scenarios. *Ecosystems and human well-being: Scenarios,* **2,** 297-373.

Alexander R.B., Smith R.A., Schwarz G.E., Boyer E.W., Nolan J.V. & Brakebill J.W. (2007) Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. *Environmental science & technology,* **42,** 822-830.

Allan J.D. (2004) Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics,* ***35,*** 257-284.

Allcock K.G. & Hik D.S. (2004) Survival, growth, and escape from herbivory are determined by habitat and herbivore species for three Australian woodland plants. *Oecologia,* **138,** 231-241.

Allen E.B. (1995) Restoration ecology: limits and possibilities in arid and semiarid lands. *Proceedings of the Wildland Shrub and Arid Land Restoration Symposium*, Las Vegas, Nevada, 7-15.

Anbumozhi V., Radhakrishnan J. & Yamaji E. (2005) Impact of riparian buffer zones on water quality and associated management considerations. *Ecological Engineering,* **24,** 517-523.

Armour C., Duff D. & Elmore W. (1991) The effects of livestock grazing on riparian and stream ecosystems. *Fisheries,* **16,** 7-11.

Armstrong R.C., Turner K.D., McDougall K.L., Rehwinkel R. & Crooks J.I. (2013) Plant communities of the upper Murrumbidgee catchment in New South Wales and the Australian Capital Territory. *Cunninghamia,* **13,** 125-265.

Arnaiz O.L., Wilson A.L., Watts R.J. & Stevens M.M. (2011) Influence of riparian condition on aquatic macroinvertebrate communities in an agricultural catchment in south-eastern Australia. *Ecological Research,* **26,** 123-131.

Aronson J., Floret C., Floc'h E., Ovalle C. & Pontanier R. (1993) Restoration and Rehabilitation of Degraded Ecosystems in Arid and Semi-Arid Lands. I. A View from the South. *Restoration Ecology,* **1,** 8-17.

Aronson J., Blignaut J.N., Milton S.J. & Clewell A.F. (2006) Natural capital: the limiting factor. *Ecological Engineering,* **28**, 1-5.

Arthington A.H., Naiman R.J., McClain M.E. & Nilsson C. (2010) Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. *Freshwater Biology,* **55,** 1-16.

Australian Government, Bureau of Meteorology (2014) *Drought Statement Archive* 2014, Bureau of Meteorology, Melbourne.

Balcombe S.R., Sheldon F., Capon S.J., Bond N.R., Hadwen W.L., Marsh N. & Bernays S.J. (2011) Climate-change threats to native fish in degraded rivers and floodplains of the Murray–Darling Basin, Australia. *Marine and Freshwater Research,* **62,** 1099-1114.

Barson M., Randall L. & Bordas V. (2000) Land cover change in Australia. *Results of the collaborative Bureau of Rural Sciences–State agencies' project on remote sensing of land cover change.* Bureau of Rural Sciences, Canberra .

Bash J.S. & Ryan C.M. (2002) Stream restoration and enhancement projects: is anyone monitoring? *Environmental management,* **29,** 877-885.

Belsky A., Matzke A. & Uselman S. (1999) Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation,* **54,** 419-431.

Bennett A., Lumsden L. & Nicholls A. (1994) Tree hollows as a resource for wildlife in remnant woodlands: spatial and temporal patterns across the northern plains of Victoria, Australia. *Pacific Conservation Biology,* **1,** 222.

Bennett A.F., Nimmo D.G. & Radford J.Q. (2014) Riparian vegetation has disproportionate benefits for landscape-scale conservation of woodland birds in highly modified environments. *Journal of Applied Ecology,***51**, 514-523.

Bernhardt E.S., Palmer M., Allan J., Alexander G., Barnas K., Brooks S., Carr J., Clayton S., Dahm C. & Follstad-Shah J. (2005) Synthesizing U. S. river restoration efforts. *Science,* **308,** 636-637.

Bernhardt E.S., Sudduth E.B., Palmer M.A., Allan J.D., Meyer J.L., Alexander G., Follastad-Shah J., Hassett B., Jenkinson R. & Lave R. (2007) Restoring rivers one reach at a time: results from a survey of US river restoration practitioners. *Restoration Ecology,* **15,** 482-493.

Bidgee Banks Project Steering Committee (2003) Charting a Course for River Restoration. A Summary of Achievements, Problems, Solutions and Lessons Learnt from a Landmark Devolved Grant Project, *Bidgee Banks, Canberra* .

Bornette G. & Puijalon S. (2011) Response of aquatic plants to abiotic factors: a review. *Aquatic Sciences,* **73,** 1-14.

Box G.E. & Cox D.R. (1964) An analysis of transformations. *Journal of the Royal Statistical Society.Series B (Methodological),* 211-252.

Bradshaw A. (1997) Restoration of mined lands—using natural processes. *Ecological Engineering,* **8,** 255-269.

Breckenridge R., Kepner W. & Mouat D. (1995) A process for selecting indicators for monitoring conditions of rangeland health. *Environmental monitoring and assessment,* **36,** 45-60.

Breckwoldt R. (1983) *Wildlife in the home paddock. Nature Conservation for Farmers,* Angus and Robertson, Sydney.

Brierley G.J. & Fryirs K.A. (2005) *Geomorphology and river management: applications of the river styles framework*.Blackwell, Oxford.

Briggs M. (1996) Evaluating degraded riparian ecosystems to determine the potential effectiveness of revegetation. *Proceedings of the Wild Land Shrub and Arid Land Restoration Symposium,* Las Vegas, Nevada, 63.

Briggs M.K. (1996) *Riparian ecosystem recovery in arid lands: strategies and references*. University of Arizona Press, Tucson.

Briggs S.V., Taws N.M., Seddon J.A. & Vanzella B. (2008) Condition of fenced and unfenced remnant vegetation in inland catchments in south-eastern Australia. *Australian Journal of Botany,* **56,** 590-599.

Brooks A.P., Howell T., Abbe T.B. & Arthington A.H. (2006) Confronting hysteresis: wood based river rehabilitation in highly altered riverine landscapes of south-eastern Australia. *Geomorphology,* **79,** 395-422.

Brooks S.S. & Lake P.S. (2007) River restoration in Victoria, Australia: change is in the wind, and none too soon. *Restoration Ecology,* **15,** 584-591.

Brudvig L.A. (2011) The restoration of biodiversity: where has research been and where does it need to go? *American Journal of Botany*, **98,** 549-558.

Bryant L.D. (1982) Response of livestock to riparian zone exclusion. *Journal of Range Management*, **35**, 780-785.

Bullock J.M., Aronson J., Newton A.C., Pywell R.F. & Rey-Benayas J.M. (2011) Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology & Evolution,* **26,** 541-549.

Burger B., Reich P. & Cavagnaro T. (2010) Trajectories of change: riparian vegetation and soil conditions following livestock removal and replanting. *Austral Ecology,* **35,** 980-987.

Cabin R.J., Clewell A., Ingram M., McDonald T. & Temperton V. (2010) Bridging restoration science and practice: results and analysis of a survey from the 2009 society for ecological restoration international meeting. *Restoration Ecology,* **18,** 783-788.

Carline R.F. & Walsh M.C. (2007) Responses to riparian restoration in the Spring Creek watershed, central Pennsylvania. *Restoration Ecology,* **15,** 731-742.

Caughley G., Shepherd N. & Short J. (1987) *Kangaroos: their ecology and management in the sheep rangelands of Australia*. Cambridge University Press, Cambridge.

Clarke P.J. (2002) Experiments on tree and shrub establishment in temperate grassy woodlands: seedling survival. *Austral Ecology,* **27,** 606-615.

Clewell A.F. & Aronson J. (2006) Motivations for the restoration of ecosystems. *Conservation Biology,* **20,** 420-428.

Correll D.L. (2005) Principles of planning and establishment of buffer zones. *Ecological Engineering,* **24,** 433-439.

Costin A. (1980) Runoff and soil and nutrient losses from an improved pasture at Ginninderra, Southern Tablelands, New South Wales. *Crop and Pasture Science,* **31,** 533-546.

Crosslé K. & Brock M.A. (2002) How do water regime and clipping influence wetland plant establishment from seed banks and subsequent reproduction? *Aquatic Botany,* **74,** 43-56.

Crouch R. (1987) The relationship of gully sidewall shape to sediment production. *Soil Research,* **25,** 531-539.

Cummins K.W., Wilzbach M.A., Gates D.M., Perry J.B. & Taliaferro W.B. (1989) Shredders and riparian vegetation. *Bioscience*, **39**, 24-30.

Curtis A. & Robertson A. (2003) Understanding landholder management of river frontages: the Goulburn Broken. *Ecological Management & Restoration,* **4,** 45-54.

D'Antonio C. & Meyerson L.A. (2002) Exotic plant species as problems and solutions in ecological restoration: a synthesis. *Restoration Ecology,* **10,** 703-713.

Darby S.E. (1999) Effect of riparian vegetation on flow resistance and flood potential. *Journal of Hydraulic Engineering,* **125,** 443-454.

Décamps H., Pinay G., Naiman R.J., Petts G.E., McClain M.E., Hillbricht-Ilkowska A., Hanley T.A., Holmes R.M., Quinn J. & Gibert J. (2004) Riparian zones: where biogeochemistry meets biodiversity in management practice. *Polish Journal of Ecology,* **52,** 3-18.

Dickens S.J.M. & Suding K.N. (2013) Spanning the Science-Practice Divide: Why Restoration Scientists Need to be More Involved with Practice. *Ecological Restoration,* **31,** 134-140.

Dobkin D.S., Rich A.C. & Pyle W.H. (1998) Habitat and avifaunal recovery from livestock grazing in a riparian meadow system of the northwestern Great Basin. *Conservation Biology,* **12,** 209-221.

Dobson A.P., Bradshaw A. & Baker A. (1997) Hopes for the future: restoration ecology and conservation biology. *Science,* **277,** 515-522.

Docker B. & Hubble T. (2009) Modelling the distribution of enhanced soil shear strength beneath riparian trees of south-eastern Australia. *Ecological Engineering,* **35,** 921-934.

Docker B. & Hubble T. (2008) Quantifying root-reinforcement of river bank soils by four Australian tree species. *Geomorphology,* **100,** 401-418.

Dorrough J., Yen A., Turner V., Clark S., Crosthwaite J. & Hirth J. (2004) Livestock grazing management and biodiversity conservation in Australian temperate grassy landscapes. *Crop and Pasture Science,* **55,** 279-295.

Downs P.W. & Kondolf G.M. (2002) Post-project appraisals in adaptive management of river channel restoration. *Environmental management,* **29,** 477-496.

Dudgeon D., Arthington A.H., Gessner M.O., Kawabata Z., Knowler D.J., Lévêque C., Naiman R.J., Prieur-Richard A., Soto D. & Stiassny M.L. (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological reviews,* **81,** 163-182.

Ede F.J., Ainsworth N., Hunt T.D. & Zydenbos S. (2010) Managing weeds in riparian zones. *Proceedings of the 17th Australasian Weeds Conference*, Christchurch, 311-314.

Edwards K. & Johnston W. (1978) Agricultural climatology of the Upper Murrumbidgee River Valley, New South Wales. *Crop and Pasture Science,* **29,** 851-862.

Erskine W.D. (2001) Geomorphic evaluation of past river rehabilitation works on the Williams River, New South Wales. *Ecological Management and Restoration,* **2,** 116-128.

Everest F.H. & Meehan W.R. (1983) *Forest management and anadromous fish habitat productivity*. *Proceedings of the 46th North American Wildlife and Natural Resources Conference*, Washington, 521,- 530.

Ewel K.C., Cressa C., Kneib R.T., Lake P., Levin L.A., Palmer M.A., Snelgrove P. & Wall D.H. (2001) Managing critical transition zones. *Ecosystems,* **4,** 452-460.

Fairweather P.G. & Napier G.M. (1998) *Environmental indicators for national state of the environment reporting: Inland waters*. Department of the Environment, Canberra..

Fleischner T.L. (1994) Ecological costs of livestock grazing in western North America. *Conservation Biology,* **8,** 629-644.

Frost R.A. & Launchbaugh K.L. (2003) Prescription grazing for rangeland weed management: A new look at an old tool. *Rangelands,* **25**, 43-47.

Fry J., Steiner F.R. & Green D.M. (1994) Riparian evaluation and site assessment in Arizona. *Landscape and Urban Planning*, **28,** 179-199.

Gibson N. & Kirkpatrick J. (1989) Effects of the cessation of grazing on the grasslands and grassy woodlands of the Central Plateau, Tasmania. *Australian Journal of Botany,* **37,** 55-63.

Gilmore R. (2008) *The Upper Murrumbidgee IQQM Calibration Report: A Report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project*. CSIRO Publishing, Melbourne.

Gleick P.H. (2003) Global freshwater resources: soft-path solutions for the 21st century. *Science (New York, N.Y.),* **302,** 1524-1528.

Golet G.H., Gardali T., Howell C.A., Hunt J., Luster R.A., Rainey W., Roberts M.D., Silveira J., Swagerty H. & Williams N. (2008) Wildlife response to riparian restoration on the Sacramento River. *San Francisco Estuary and Watershed Science,* **6,** 1-26.

Goodwin C.N., Hawkins C.P. & Kershner J.L. (1997) Riparian restoration in the western United States: overview and perspective. *Restoration Ecology,* **5,** 4-14.

Gould L. (2007) Key ingredients for the successful implementation of riparian rehabilitation programs. *Proceedings of the 5th Australian Stream Management Conference. Australian rivers: making a difference,Thurgoona,* New South Wales, 115-120.

Grant G. & Swanson F. (1995) Morphology and processes of valley floors in mountain streams, western Cascades, Oregon. *Geophysical Monograph,* **89**, 83-101.

Greene R.S. & Hairsine P.B. (2004) Elementary processes of soil–water interaction and thresholds in soil surface dynamics: a review. *Earth Surface Processes and Landforms,* **29,** 1077-1091.

Greet J., Cousens R.D. & Webb J.A. (2013) Flow regulation is associated with riverine soil seed bank composition within an agricultural landscape: potential implications for restoration. *Journal of Vegetation Science,* **24,** 157-167.

Gregory S.V., Swanson F.J., McKee W.A. & Cummins K.W. (1991) An ecosystem perspective of riparian zones. *Bioscience,* **41,** 540-551.

Growns I., Gehrke P., Astles K. & Pollard D. (2003) A comparison of fish assemblages associated with different riparian vegetation types in the Hawkesbury–Nepean River system. *Fisheries Management and Ecology,* **10,** 209-220.

Gunderson L. (1999) Resilience, flexibility and adaptive management—antidotes for spurious certitude. *Conservation Ecology,* **3,** 7-17.

Hickin E.J. (1984) Vegetation and river channel dynamics. *The Canadian Geographer,* **28,** 111-126.

Higgs E.S. (1997) What is good ecological restoration? *Conservation Biology,* **11,** 338-348.

Hobbs R.J. (2001) Synergisms among habitat fragmentation, livestock grazing, and biotic invasions in southwestern Australia. *Conservation Biology,* **15,** 1522-1528.

Hobbs R.J. & Atkins L. (1991) Interactions between annuals and woody perennials in a Western Australian nature reserve. *Journal of Vegetation Science,* **2,** 643-654.

Hobbs R.J., Hallett L.M., Ehrlich P.R. & Mooney H.A. (2011) Intervention ecology: applying ecological science in the twenty-first century. *Bioscience,* **61,** 442-450.

Hobbs R.J. & Harris J.A. (2001) Restoration ecology: repairing the earth's ecosystems in the new millennium. *Restoration Ecology,* **9,** 239-246.

Hobbs R.J. & Norton D.A. (1996) Towards a conceptual framework for restoration ecology. *Restoration Ecology,* **4,** 93-110.

Holl K.D. & Aide T.M. (2011) When and where to actively restore ecosystems? *Forest Ecology and Management,* **261,** 1558-1563.

Holmgren M. (2002) Exotic herbivores as drivers of plant invasion and switch to ecosystem alternative states. *Biological Invasions,* **4,** 25-33.

Hough-Snee N., Roper B.B., Wheaton J.M., Budy P. & Lokteff R.L. (2013) Riparian vegetation communities change rapidly following passive restoration at a northern Utah stream. *Ecological Engineering,* **58,** 371-377.

Hubble T. (2004) Slope stability analysis of potential bank failure as a result of toe erosion on weir-impounded lakes: an example from the Nepean River, New South Wales, Australia. *Marine and freshwater research,* **55,** 57-65.

Hubble T., Docker B. & Rutherfurd I. (2010) The role of riparian trees in maintaining riverbank stability: a review of Australian experience and practice. *Ecological Engineering,* **36,** 292-304.

Hupp C.R. & Osterkamp W. (1996) Riparian vegetation and fluvial geomorphic processes. *Geomorphology,* **14,** 277-295.

Ilhardt B.L., Verry E.S. & Palik B.J. (2000) *Defining riparian areas*. Lewis Publishers,Washington, 23–42.

Innis S.A., Naiman R.J. & Elliott S.R. (2000) Indicators and assessment methods for measuring the ecological integrity of semi-aquatic terrestrial environments. *Hydrobiologia,* **422,** 111-131.

Jackson S.T. & Hobbs R.J. (2009) Ecological restoration in the light of ecological history. *Science,* **325,** 567-569.

Jansen A. (2005) *Rapid Appraisal of Riparian Condition: Version Two*. Land & Water Australia, Canberra.

Jansen A. (2004) *Development and application of a method for the rapid appraisal of riparian condition*. Land & Water Australia, Canberra.

Jansen A., Askey-Doran M., Pettit N. & Price P. (2007) *Impacts of land management practices on riparian land*:*Principles for Riparian Lands Management*, Land & Water Australia, Canberra..

Jansen A. & Robertson A.I. (2001) Relationships between livestock management and the ecological condition of riparian habitats along an Australian floodplain river. *Journal of Applied Ecology,* **38,** 63-75.

Jellinek S., Parris K.M., Driscoll D.A. & Dwyer P.D. (2013) Are incentive programs working? Landowner attitudes to ecological restoration of agricultural landscapes. *Journal of environmental management,* **127,** 69-76.

Johansen K. & Phinn S. (2006) Mapping structural parameters and species composition of riparian vegetation using IKONOS and Landsat ETM data in Australian tropical savannahs. *Photogrammetric Engineering & Remote Sensing,* **72,** 71-80.

Johnston L., Skinner S., Ishiyama L. & sharp S. (2009) *Survey of vegetation and habitat in key riparian zones: Murrumbidgee Rivers, ACT*. Territory and Municipal Services, Canberra.

Jordan III W.R. (1994) *Sunflower Forest': ecological restoration as the basis for a new environmental paradigm*. University of Minnesota Press, Minneapolis. .

Jordan III W.R., Peters II R.L. & Allen E.B. (1988) Ecological restoration as a strategy for conserving biological diversity. *Environmental management,* **12,** 55-72.

Jordan III W.R. & Gilpin M.E. (1990) *Restoration ecology: a synthetic approach to ecological research*. Cambridge University Press, Cambridge.

Karr J.R. & Chu E.W. (1998) *Restoring life in running waters: better biological monitoring*. Island Press, Washington.

Kauffman J.B., Beschta R.L., Otting N. & Lytjen D. (1997) An ecological perspective of riparian and stream restoration in the western United States. *Fisheries,* **22,** 12-24.

Kauffman J.B., Krueger W. & Vavra M. (1983) Impacts of cattle on streambanks in northeastern Oregon. *Journal of Range Management,* **36**, 683-685.

Klein L.R., Clayton S.R., Alldredge J.R. & Goodwin P. (2007) Long-Term Monitoring and Evaluation of the Lower Red River Meadow Restoration Project, Idaho, USA. *Restoration Ecology,* **15,** 223-239.

Knight A., Beale P. & Dalton G. (1997) Direct seeding of native trees and shrubs in low rainfall areas and on non-wetting sands in South Australia. *Agroforestry Systems,* **39,** 225-239.

Kondolf G.M. (1998) Lessons learned from river restoration projects in California. *Aquatic Conservation: Marine and Freshwater Ecosystems,* **8,** 39-52.

Kondolf G.M. (1995) Five elements for effective evaluation of stream restoration. *Restoration Ecology,* **3,** 133-136.

Kondolf G.M. & Micheli E.R. (1995) Evaluating stream restoration projects. *Environmental Management,* **19,** 1-15.

Ladson A.R., White L.J., Doolan J.A., Finlayson B.L., Hart B.T., Lake P.S. & Tilleard J.W. (1999) Development and testing of an Index of Stream Condition for waterway management in Australia. *Freshwater Biology,* **41,** 453-468.

Lambers H. (2003) Introduction, dryland salinity: a key environmental issue in southern Australia. *Plant and Soil,* **257,** 5-7.

Larsen R.E., Krueger W.C., George M.R., Barrington M.R., Buckhouse J.C. & Johnson D.E. (1998) Viewpoint: Livestock influences on riparian zones and fish habitat: Literature classification. *Journal of Range Management,* **51,** 661-664.

Lee K.N. (1999) Appraising adaptive management. *Conservation Ecology,* **3,** 3-16.

Lester R.E. & Boulton A.J. (2008) Rehabilitating agricultural streams in Australia with wood: a review. *Environmental Management,* **42,** 310-326.

Lindgren P.M. & Sullivan T.P. (2012) Response of plant community abundance and diversity during 10 years of cattle exclusion within silvopasture systems. *Canadian Journal of Forest Research,* **42,** 451-462.

Line D., Harman W., Jennings G., Thompson E. & Osmond D. (2000) Nonpoint-source pollutant load reductions associated with livestock exclusion. *Journal of Environmental Quality,* **29,** 1882-1890.

Lovett G.M., Jones C.G., Turner M. G. & Weathers K.C. (2005) *Ecosystem function in heterogeneous landscapes*. Springer Science and Business Media, New York.

Lunt I.D., Eldridge D.J., Morgan J.W. & Witt G.B. (2007) Turner Review No. 13. A framework to predict the effects of livestock grazing and grazing exclusion on conservation values in natural ecosystems in Australia. *Australian Journal of Botany,* **55,** 401-415.

Lunt I.D., Jansen A., Binns D.L. & Kenny S.A. (2007) Long-term effects of exclusion of grazing stock on degraded herbaceous plant communities in a riparian Eucalyptus camaldulensis forest in south-eastern Australia. *Austral Ecology,* **32,** 937-949.

Lytle D.A. & Poff N.L. (2004) Adaptation to natural flow regimes. *Trends in Ecology & Evolution,* **19,** 94-100.

Machiori K., Tongway D. & Loch R. (2003) *A Geomorphic System for Gully Assessment*. Landloch Pty Ltd, Darling Heights and CSIRO Sustainable Ecosystems, Canberra.

Manning A.D., Cunningham R.B. & Lindenmayer D.B. (2013) Bringing forward the benefits of coarse woody debris in ecosystem recovery under different levels of grazing and vegetation density. *Biological Conservation,* **157,** 204-214.

Maron M. & Lill A. (2005) The influence of livestock grazing and weed invasion on habitat use by birds in grassy woodland remnants. *Biological Conservation,* **124,** 439-450.

Mavromihalis J., Dorrough J., Clark S., Turner V. & Moxham C. (2013) Manipulating livestock grazing to enhance native plant diversity and cover in native grasslands. *The Rangeland Journal,* **35,** 95-108.

McIver J. & Starr L. (2001) Restoration of degraded lands in the interior Columbia River basin: passive vs active approaches. *Forest Ecology and Management,* **153,** 15-28.

Mendham E., Millar J. & Curtis A. (2007) Landholder participation in native vegetation management in irrigation areas. *Ecological Management and Restoration,* **8,** 42-48.

Micheli E. & Kirchner J. (2002) Effects of wet meadow riparian vegetation on streambank erosion 2:Measurements of vegetated bank strength and consequences for failure mechanics. *Earth Surface Processes and Landforms,* **27,** 687-697.

Micheli E., Kirchner J. & Larsen E. (2004) Quantifying the effect of riparian forest versus agricultural vegetation on river meander migration rates, Central Sacramento River, California, USA. *River Research and Applications,* **20,** 537-548.

Middleton B. (1999) *Wetland restoration, flood pulsing, and disturbance dynamics*. John Wiley & Sons, New York.

Milchunas D.G. & Lauenroth W.K. (1993) Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs,* **63,** 327-366.

Millar C.I., Stephenson N.L. & Stephens S.L. (2007) Climate change and forests of the future: managing in the face of uncertainty. *Ecological Applications,* **17,** 2145-2151.

Millennium Ecosystem Assessment. (2005) *Ecosystems and human well-being*. Island Press, Washington.

Munné A., Prat N., Sola C., Bonada N. & Rieradevall M. (2003) A simple field method for assessing the ecological quality of riparian habitat in rivers and streams: QBR index. *Aquatic Conservation: Marine and Freshwater Ecosystems,* **13,** 147-163.

Muñoz-Robles C., Reid N., Frazier P., Tighe M., Briggs S.V. & Wilson B. (2010) Factors related to gully erosion in woody encroachment in south-eastern Australia. *Catena,* **83,** 148-157.

Murray-Darling Basin Authority (2014) *How the rivers run: Murrumbidgee.* MDBA, Canberra, viewed 4 May 2014, http://www.mdba.gov.au/about-basin/how-river-runs/murrumbidgee-catchment

Naiman R.J., Bilby R.E. & Bisson P.A. (2000) Riparian ecology and management in the Pacific coastal rain forest. *Bioscience*, **50,** 996-1011.

Naiman R.J., Decamps H. & McClain M.E. (2010) *Riparia: ecology, conservation, and management of streamside communities*. Elsevier, Amsterdam.

Naiman R.J., Decamps H. & Pollock M. (1993) The role of riparian corridors in maintaining regional biodiversity. *Ecological Applications,* **3,** 209-212.

Naiman R.J. (2013) Socio-ecological complexity and the restoration of river ecosystems. *Inland Waters,* **3,** 391-410.

Naiman R., J. & Decamps H. (1997) The Ecology of Interfaces-Riparian Zones. *Annual Review of Ecology and systematics,* **28,** 621-658.

National Land and Water Resources Audit (2001) *Australian agricuture assessment 2001:volume 1,*. National Land and Water Resources Audit, Canberra..

Nilsson C., Polvi L.E., Gardeström J., Hasselquist E.M., Lind L. & Sarneel J.M. (2014) Riparian and in-stream restoration of boreal streams and rivers: success or failure? *Ecohydrology,*in press.

Nilsson C. & Svedmark M. (2002) Basic principles and ecological consequences of changing water regimes: riparian plant communities. *Environmental Management,* **30,** 468-480.

Nilsson C., Reidy C.A., Dynesius M. & Revenga C. (2005) Fragmentation and flow regulation of the world's large river systems. *Science,* **308,** 405-408.

Norris R.H. & Hawkins C.P. (2000) Monitoring river health. *Hydrobiologia,* **435,** 5-17.

NRM South (2009) The Tasmanian River Condition Index, Streamside Zone Field Manual. NRM South, Hobart.

NSW Department of Land and Water Conservation (1999) Stressed Rivers Assessment Report: Murrumbidgee Catchment, Department of Land and Water Conservation, Sydney.

Olley J.M. & Wasson R.J. (2003) Changes in the flux of sediment in the Upper Murrumbidgee catchment, Southeastern Australia, since European settlement. *Hydrological Processes,* **17,** 3307-3320.

Owens L., Edwards W. & Van Keuren R. (1996) Sediment losses from a pastured watershed before and after stream fencing. *Journal of Soil and Water Conservation,* **51,** 90-94.

Palmer M., Bernhardt E., Allan J., Lake P., Alexander G., Brooks S., Carr J., Clayton S., Dahm C. & Follstad Shah J. (2005) Standards for ecologically successful river restoration. *Journal of Applied Ecology,* **42,** 208-217.

Palmer M.A., Ambrose R.F. & Poff N.L. (1997) Ecological theory and community restoration ecology. *Restoration Ecology,* **5,** 291-300.

Palmer M., Allan J.D., Meyer J. & Bernhardt E.S. (2007) River restoration in the twenty-first century: data and experiential knowledge to inform future efforts. *Restoration Ecology,* **15,** 472-481.

Palmerlee A.P. & Young T.P. (2010) Direct seeding is more cost effective than container stock across ten woody species in California. *Native plants journal,* **11,** 89-102.

Parkyn S.M., Davies-Colley R.J., Halliday N.J., Costley K.J. & Croker G.F. (2003) Planted riparian buffer zones in New Zealand: do they live up to expectations?. *Restoration Ecology,* **11,** 436-447.

Patmore S. & Davey A. (2004) *Bidgee Banks Monitoring and Evaluation in the Upper Catchment*: *Report to Greening Australia ACT & SE NSW and the Bidgee Banks Project Steering Committee*. University of Canberra, Canberra.

Patten D.T. (1998) Riparian ecosytems of semi-arid North America: Diversity and human impacts. *Wetlands,* **18,** 498-512.

Paul M.J. & Meyer J.L. (2001) Streams in the urban landscape. *Annual Review of Ecology and Systematics,* **32,** 333-365.

Petersen R.C. (1992) The RCE: a riparian, channel, and environmental inventory for small streams in the agricultural landscape. *Freshwater Biology,* **27,** 295-306.

Planty-Tabacchi A., Tabacchi E., Naiman R.J., Deferrari C. & Decamps H. (1996) Invasibility of species-rich communities in riparian zones. *Conservation Biology,* **10,** 598-607.

Poff N.L., Allan J.D., Bain M.B., Karr J.R., Prestegaard K.L., Richter B.D., Sparks R.E. & Stromberg J.C. (1997) The natural flow regime. *Bioscience,* **47,** 769-784.

Prach K. & Hobbs R.J. (2008) Spontaneous succession versus technical reclamation in the restoration of disturbed sites. *Restoration Ecology,* **16,** 363-366.

Purcell A.H., Friedrich C. & Resh V.H. (2002) An assessment of a small urban stream restoration project in northern California. *Restoration Ecology,* **10,** 685-694.

Pusey B.J. & Arthington A.H. (2003) Importance of the riparian zone to the conservation and management of freshwater fish: a review. *Marine and Freshwater Research,* **54,** 1-16.

Richardson D.M., Holmes P.M., Esler K.J., Galatowitsch S.M., Stromberg J.C., Kirkman S.P., Pyšek P. & Hobbs R.J. (2007) Riparian vegetation: degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions*, **13,** 126-139.

Richardson J.S., Naiman R.J. & Bisson P.A. (2012) How did fixed-width buffers become standard practice for protecting freshwaters and their riparian areas from forest harvest practices? *Freshwater Science,* **31,** 232-238.

Robertson A. (1997) *Land-water linkages in floodplain river systems: the influence of domestic stock*. Frontiers in Ecology: Building the links, Elsevier Scientific, Oxford.

Robertson A. & Rowling R. (2000) Effects of livestock on riparian zone vegetation in an Australian dryland river. *Regulated Rivers: Research & Management,* **16,** 527-541.

Robinson G.R. & Handel S.N. (1993) Forest restoration on a closed landfill: rapid addition of new species by bird dispersal. *Conservation Biology,* **7,** 271-278.

Roni P., Beechie T.J., Bilby R.E., Leonetti F.E., Pollock M.M. & Pess G.R. (2002) A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. *North American Journal of Fisheries Management,* **22,** 1-20.

Roni P. & Quimby E. (2005) *Monitoring stream and watershed restoration*.American Fisheries Society, Bethesda.

Rood S.B., Gourley C.R., Ammon E.M., Heki L.G., Klotz J.R., Morrison M.L., Mosley D., Scoppettone G.G., Swanson S. & Wagner P.L. (2003) Flows for floodplain forests: a successful riparian restoration. *Bioscience,* **53,** 647-656.

Rosgen D.L. (1997) *A geomorphological approach to restoration of incised rivers*. University of Mississippi, Mississippi .

Ruiz-Jaen M.C. & Mitchell Aide T. (2005) Restoration success: how is it being measured? *Restoration Ecology*, **13,** 569-577.

Rutherfurd I.D., Marsh N. & Jerie K. (2000) *A rehabilitation manual for Australian streams*. Land and Water Resources Research and Development Corporation, Canberra..

Schirmer J. & Field J. (2002) *The cost of revegetation*. Environment Australia, Canberra.

Schulz T.T. & Leininger W.C. (1990) Differences in riparian vegetation structure between grazed areas and exclosures. *Journal of Range Management*, **43,** 295-299.

Schwarte K.A., Russell J.R., Kovar J.L., Morrical D.G., Ensley S.M., Yoon K., Cornick N.A. & Cho Y.I. (2011) Grazing management effects on sediment, phosphorus, and pathogen loading of streams in cool-season grass pastures. *Journal of Environmental Quality,* **40,** 1303-1313.

Society for Ecological Restoration International (2004) *The SER International primer on ecological restoration*. Society for Ecological Restoration International Science & Policy Working Group, Tuson.

Shafroth P.B., Beauchamp V.B., Briggs M.K., Lair K., Scott M.L. & Sher A.A. (2008) Planning riparian restoration in the context of Tamarix control in western North America. *Restoration Ecology,* **16,** 97-112.

Shoo L.P., Wilson R., Williams Y.M. & Catterall C.P. (2014) Putting it back: Woody debris in young restoration plantings to stimulate return of reptiles. *Ecological Management & Restoration,* **15,** 84-87.

Simon A. & Collison A.J. (2002) Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. *Earth Surface Processes and Landforms,* **27,** 527-546.

Siriwardena L., Finlayson B.L. & McMahon T.A. (2006) The impact of land use change on catchment hydrology in large catchments: The Comet River, Central Queensland, Australia. *Journal of Hydrology,* **326,** 199-214.

Smokorowski K., Withers K. & Kelso J. (1998) *Does habitat creation contribute to management goals? An evaluation of literature documenting freshwater habitat rehabilitation or enhancement projects*. Fisheries and Oceans Canada, Ontario.

Sovell L.A., Vondracek B., Frost J.A. & Mumford K.G. (2000) Impacts of Rotational Grazing and Riparian Buffers on Physicochemical and Biological Characteristicsof Southeastern Minnesota, USA, Streams. *Environmental Management,* **26,** 629-641.

Špačková I., Kotorová I. & Lepš J. (1998) Sensitivity of seedling recruitment to moss, litter and dominant removal in an oligotrophic wet meadow. *Folia Geobotanica,* **33,** 17-30.

Spooner P.G. & Briggs S.V. (2008) Woodlands on farms in southern New South Wales: A longer-term assessment of vegetation changes after fencing. *Ecological Management & Restoration,* **9,** 33-41.

Spooner P., Lunt I. & Robinson W. (2002) Is fencing enough? The short-term effects of stock exclusion in remnant grassy woodlands in southern NSW. *Ecological Management & Restoration,* **3,** 117-126.

Stahlheber K.A. & D'Antonio C.M. (2013) Using livestock to manage plant composition: A meta-analysis of grazing in California Mediterranean grasslands. *Biological Conservation,* **157,** 300-308.

Starr B. (2000) *Budgeting for Bidgee Banks*: *A report prepared for and on behalf of Greening Australia,*Land and Water Management, Canberra .

Starr B.J., Wasson R.J. & Caitcheon G. (1999) *Soil erosion, phosphorus & dryland salinity in the Upper Murrumbidgee: past change & current findings*. Murrumbidgee Catchment Management Committee, Wagga Wagga.

Steel E.A., Hughes R.M., Fullerton A.H., Schmutz S., Young J.A., Fukushima M., Muhar S., Poppe M., Feist B.E. & Trautwein C. (2010) Are we meeting the challenges of landscape-scale riverine research? a review. *Living Reviews in Landscape Research,* **4,** 1-60.

Stromberg J., Beauchamp V., Dixon M., Lite S. & Paradzick C. (2007) Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid south-western United States. *Freshwater Biology,* **52,** 651-679.

Suding K.N., Gross K.L. & Houseman G.R. (2004) Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology & Evolution,* **19,** 46-53.

Sweeney B.W., Czapka S.J. & Yerkes T. (2002) Riparian forest restoration: increasing success by reducing plant competition and herbivory. *Restoration Ecology,* **10,** 392-400.

Tabacchi E., Lambs L., Guilloy H., Planty-Tabacchi A., Muller E. & Decamps H. (2000) Impacts of riparian vegetation on hydrological processes. *Hydrological Processes,* **14,** 2959-2976.

Thomson D. & Pepperdine S. (2003) *Assessing community capacity for riparian restoration*. Land and Water Australia, Canberra.

Thorne C.R. (1990) *Effects of vegetation on riverbank erosion and stability*: Vegetation and Erosion. Wiley, Chichester.

Tickner D.P., Angold P.G., Gurnell A.M. & Mountford J.O. (2001) Riparian plant invasions: hydrogeomorphological control and ecological impacts. *Progress in Physical Geography,* **25,** 22-52.

Toh I., Gillespie M. & Lamb D. (1999) The role of isolated trees in facilitating tree seedling recruitment at a degraded sub-tropical rainforest site. *Restoration Ecology,* **7,** 288-297.

Trimble S.W. & Mendel A.C. (1995) The cow as a geomorphic agent—a critical review. *Geomorphology,* **13,** 233-253.

Tubbs A.A. (1980) Riparian bird communities of the Great Plains. *Proceedings from Management of Western Forests and Grasslands for Nongame Birds.* United States Forest Service, Ogden.Valentin C., Poesen J. & Li Y. (2005) Gully erosion: impacts, factors and control. *Catena,* **63,** 132-153.

van Andel J. & Aronson J. (2012) *Restoration ecology: the new frontier*. John Wiley & Sons, Hoboken.

Vitousek P.M., Mooney H.A., Lubchenco J. & Melillo J.M. (1997) Human domination of Earth's ecosystems. *Science,* **277,** 494-499.

Wackernagel M. & Rees W.E. (2013) *Our ecological footprint: reducing human impact on the earth*. New Society Publishers, Gabriola Island.

Wallis deVries M.F., Bakker J.P., Bakker J.P. & van Wieren S. (1998) *Grazing and conservation management*. Springer, Sydney.

Walters C. (1997) Challenges in adaptive management of riparian and coastal ecosystems. *Conservation Ecology,* **1,** 2-16.

Brinson M. M., MacDonnell L. J., Austen D. J., Beschta R. L., Dillaha T. A., Donahue D. L. & Stanford, J. A. (2002) *Riparian areas: functions and strategies for mangement*. National Academies Press, Washington.

Watts R.J. & Wilson A.L. (2004) Triage: appropriate for prioritizing community funded river restoration projects, but not for advancing the science of river restoration. *Ecological Management & Restoration,* **5,** 73-75.

Webb A.A. & Erskine W.D. (2003) A practical scientific approach to riparian vegetation rehabilitation in Australia. *Journal of Environmental Management,* **68,** 329-341.

Wenger S. (1999) *A review of the scientific literature on riparian buffer width, extent and vegetation*. Office of Publication Service and Outreach, Athens.

West N.E. (1993) Biodiversity of rangelands. *Journal of Range Management,* **46,** 2-13.

West N.E., Provenza F.D., Johnson P.S. & Owens M.K. (1984) Vegetation change after 13 years of livestock grazing exclusion on sagebrush semidesert in west central Utah. *Journal of Range Management,* ***37,*** 262-264.

Westoby M., Walker B. & Noy-Meir I. (1989) Opportunistic management for rangelands not at equilibrium. *Journal of Range Management,* **42,** 266-274.

Whitford K. (2002) Hollows in jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*) trees: Hollow sizes, tree attributes and ages. *Forest Ecology and Management,* **160,** 201-214.

Wilkinson S., Jansen A., Watts R., Read A. & Miller T. (2004) *Techniques for targeting protection and rehabilitation of riparian vegetation in the Middle and Upper Murrumbidgee Catchment*. CSIROPublishing, Melbourne.

Williams J., Hook R.A. & Gascoigne H.L. (1998) *Farming Action: Catchment Reaction: The Effect of Dryland Farming on the Natural Environment*. CSIROPublishing, Melbourne.

Wimbush D. & Costin A.B. (1979) Trends in Vegetation at Kosciusko: Grazing Trials in the Subalpine Zone, 1957-1971. *Australian Journal of Botany,* **27,** 741-787.

Wohl E., Angermeier P.L., Bledsoe B., Kondolf G.M., MacDonnell L., Merritt D.M., Palmer M.A., Poff N.L. & Tarboton D. (2005) River restoration. *Water Resources Research,* **41,** 176-188.

Woolsey S., Capelli F., Gonser T., Hoehn E., Hostmann M., Junker B., Paetzold A., Roulier C., Schweizer S. & Tiegs S.D. (2007) A strategy to assess river restoration success. *Freshwater Biology,* **52,** 752-769.

Wortley L., Hero J. & Howes M. (2013) Evaluating ecological restoration success: a review of the literature. *Restoration Ecology,* **21,** 537-543.

Young T.P. (2000) Restoration ecology and conservation biology. *Biological Conservation,* **92,** 73-83.

Young T.P. & Evans R.Y. (2000) Container stock versus direct seeding for woody species in restoration sites. *International Plant Propagators Society* **50,** 577-582.

Zaimes G., Schultz R. & Isenhart T. (2004) Stream bank erosion adjacent to riparian forest buffers, row-crop fields, and continuously-grazed pastures along Bear Creek in central Iowa. *Journal of Soil and Water Conservation,* **59,** 19-27.

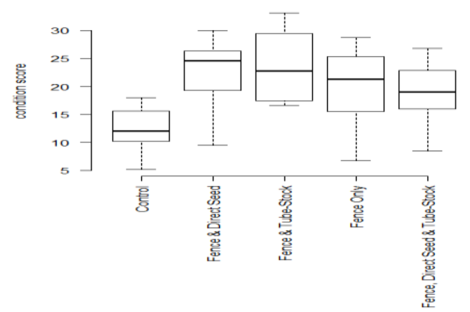
Zedler J.B. (2007) Success: an unclear, subjective descriptor of restoration outcomes. *Ecological Restoration,* **25,** 162-168.

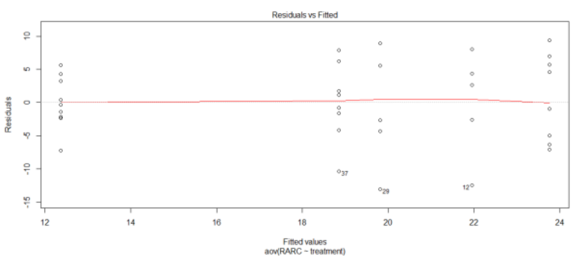
Zedler J.B. (2000) Progress in wetland restoration ecology. *Trends in Ecology & Evolution*, **15,** 402-407.

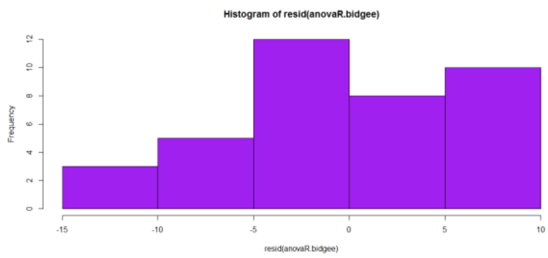
Appendix

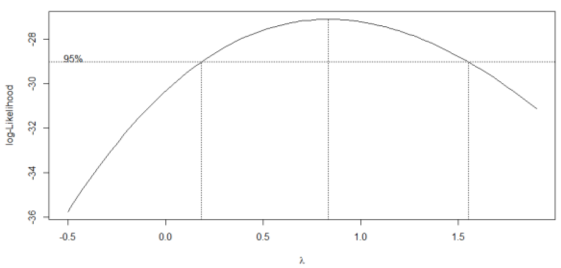
1. **Data analysis- Tests for normality, homogeneity of variance, and outputs of ANOVA, Kruskel-Wallis and post-hoc Tukey-Kramer analysis.**

Rapid Appraisal of Riparian Condition (RARC)



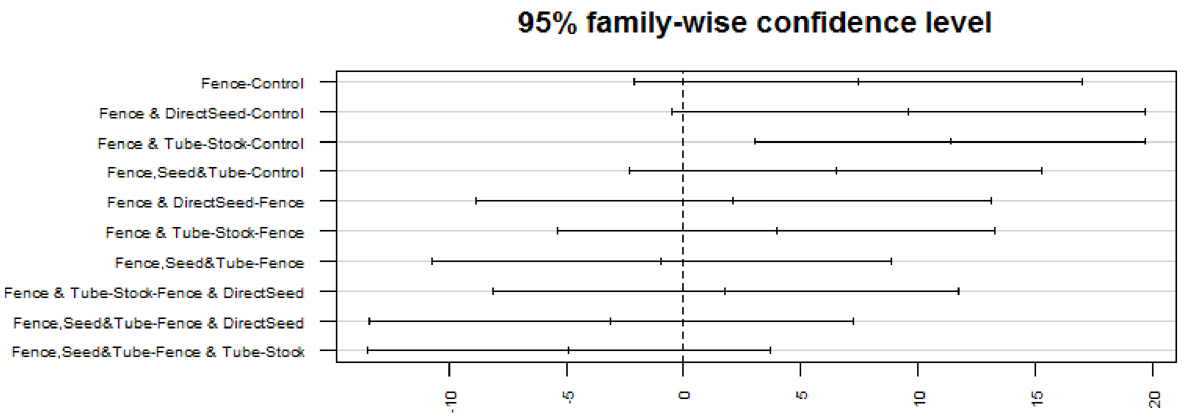


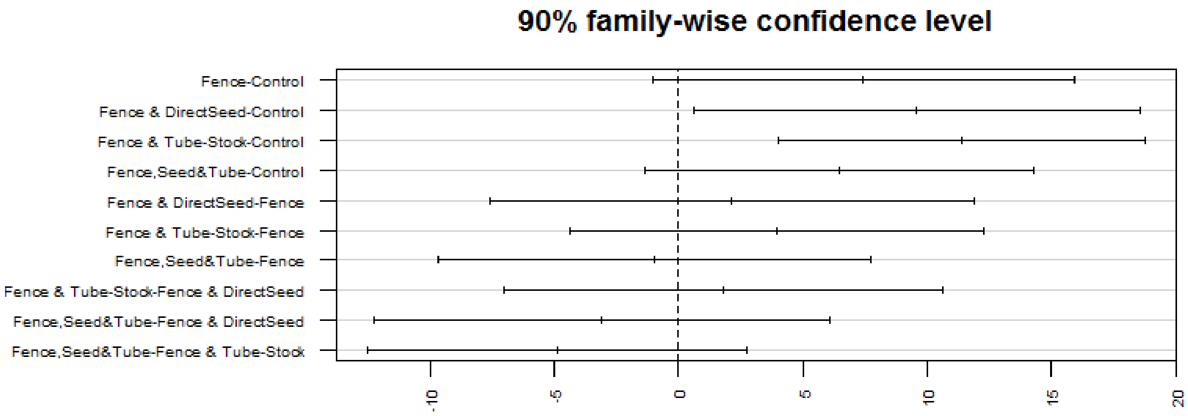




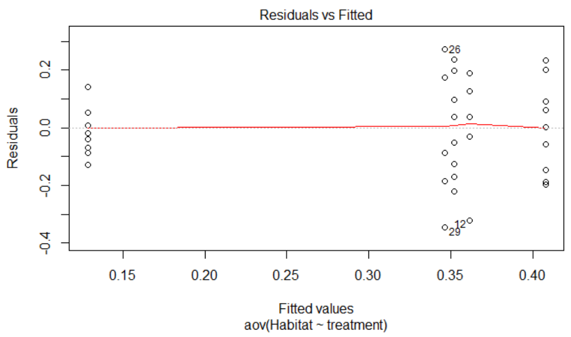

```
> #anova table for RARC
> anovaR.bidgee <- aov(RARC ~ treatment, data=bidgee)
> summary(anovaR.bidgee)
            Df Sum Sq Mean Sq F value  Pr(>F)
treatment    4    668  167.00   4.243 0.00702 **
Residuals   33   1299   39.36
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> kruskal.test(RARC ~ treatment, data=bidgee)

        Kruskal-Wallis rank sum test
```

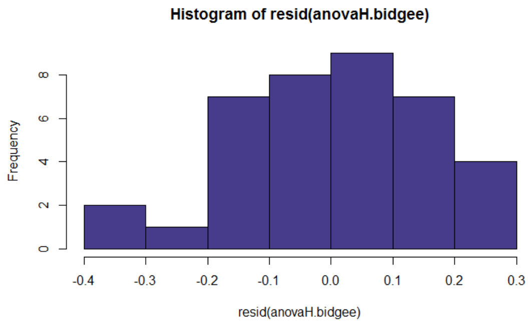
data: RARC by treatment
Kruskal-Wallis chi-squared = 12.9033, df = 4, p-value = 0.01176





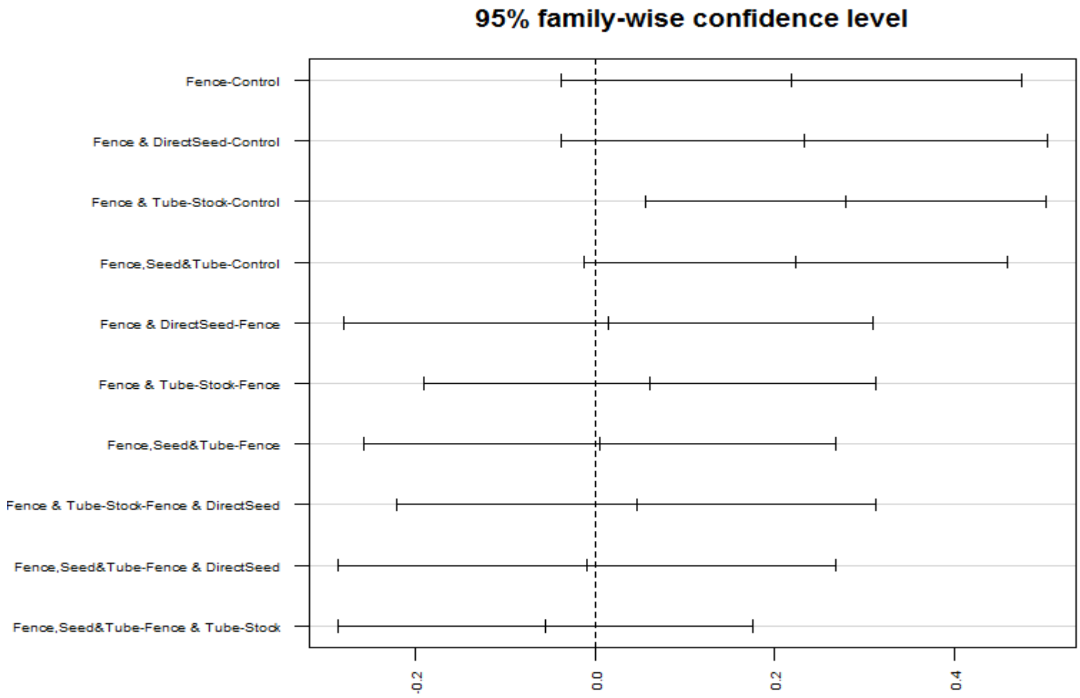
Habitat



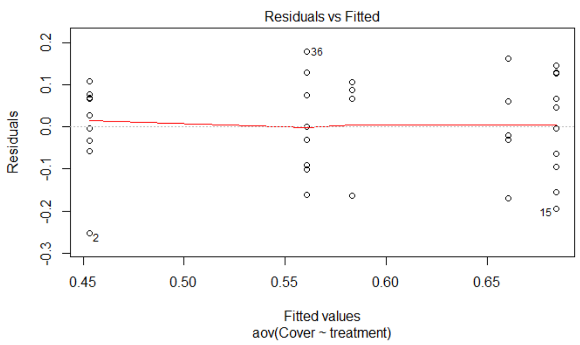


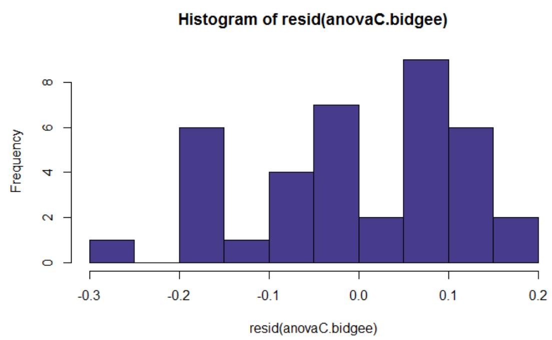

```
> #anova table for Habitat
> anovaH.bidgee <- aov(Habitat ~ treatment, data=bidgee)
> summary(anovaH.bidgee)
           Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.4274 0.10684   3.754 0.0127 *
Residuals   33 0.9393 0.02846
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

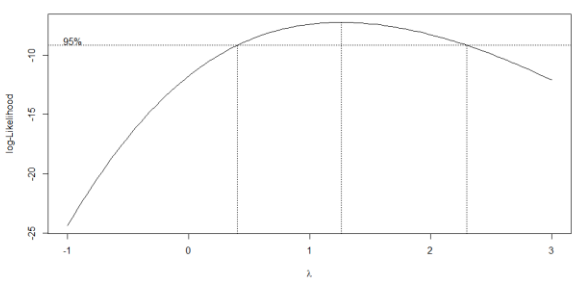
```
> kruskal.test(Habitat ~ treatment, data=bidgee)
            Kruskal-Wallis rank sum test
data:  Habitat by treatment
Kruskal-Wallis chi-squared = 11.1968, df = 4, p-value = 0.02444
```



Cover

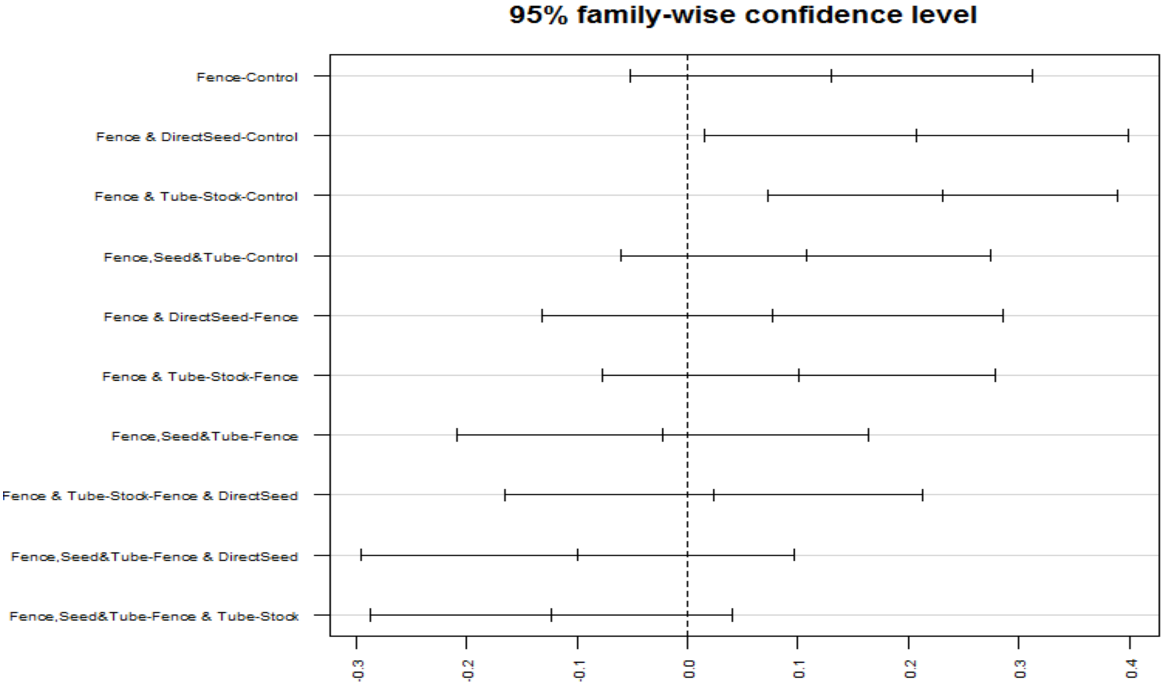




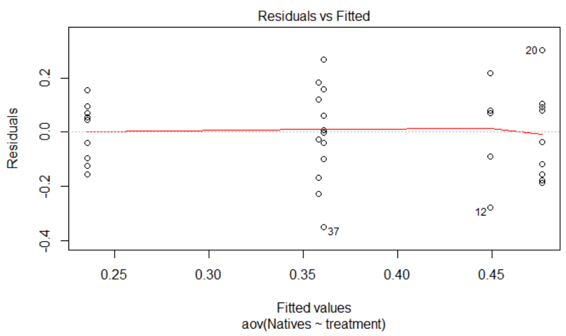


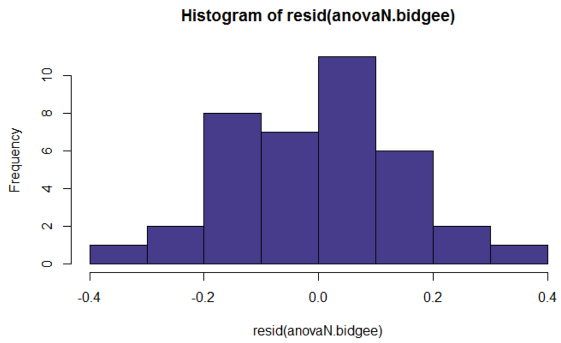

```
> #anova table for Cover
```

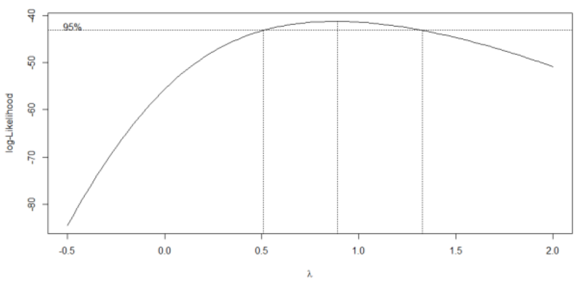
```
> anovaC.bidgee <- aov(Cover ~ treatment, data=bidgee)
> summary(anovaC.bidgee)
            Df Sum Sq Mean Sq F value  Pr(>F)
treatment    4 0.2887 0.07217   5.055 0.00273 **
Residuals   33 0.4712 0.01428
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(Cover ~ treatment, data=bidgee)
             Kruskal-Wallis rank sum test
data:  Cover by treatment
Kruskal-Wallis chi-squared = 13.4118, df = 4, p-value = 0.00943
```



Natives

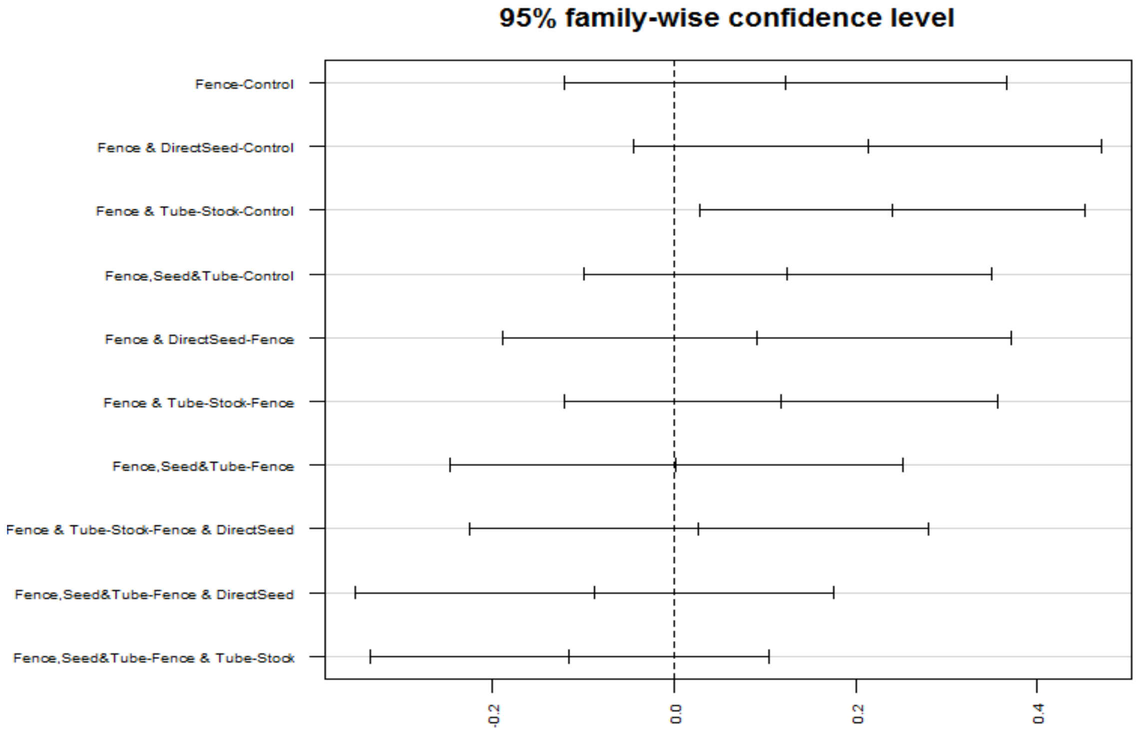




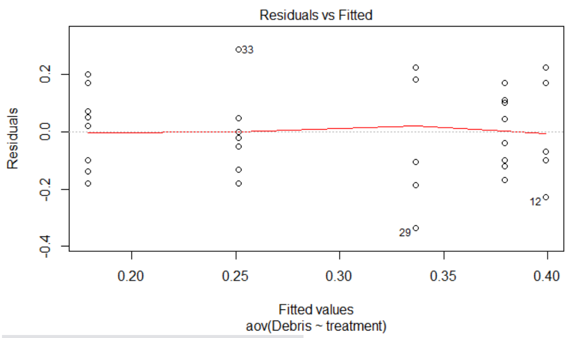


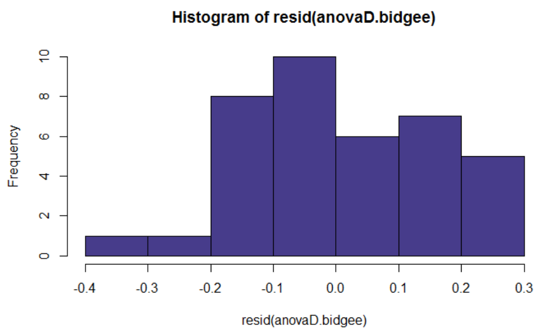
#anova table for Natives

```
> anovaN.bidgee <- aov(Natives ~ treatment, data=bidgee)
> summary(anovaN.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.3089 0.07722   2.997 0.0325 *
Residuals   33 0.8501 0.02576
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> kruskal.test(Natives ~ treatment, data=bidgee)
               Kruskal-Wallis rank sum test
data:  Natives by treatment
Kruskal-Wallis chi-squared = 10.0596, df = 4, p-value = 0.03944
```



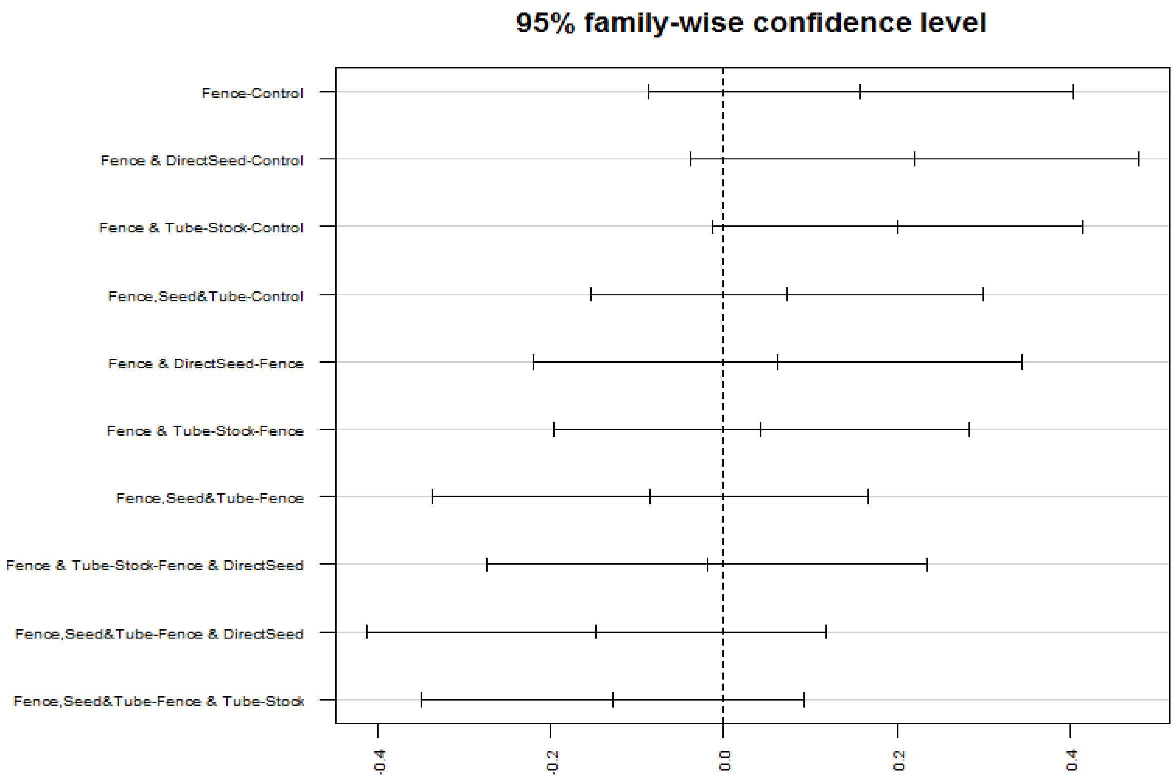
Debris





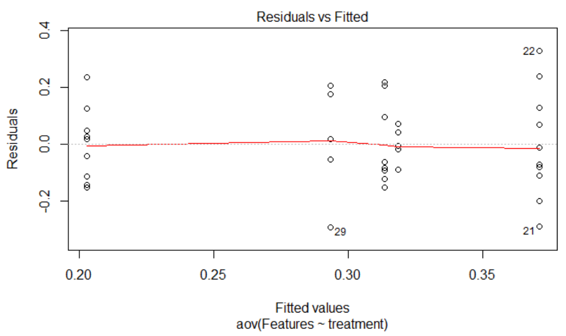

```
#anova table for Debris
> anovaD.bidgee <- aov(Debris ~ treatment, data=bidgee)
> summary(anovaD.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.2713 0.06782   2.608 0.0534 .
Residuals   33 0.8582 0.02600
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

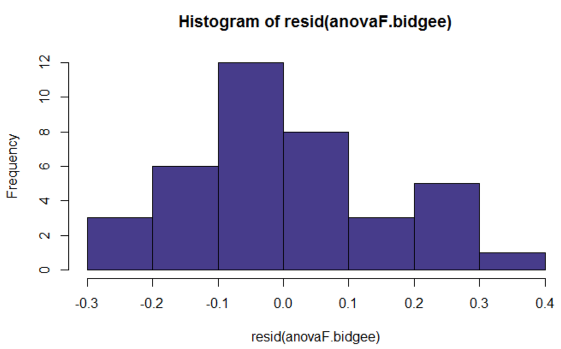
```
> kruskal.test(Debris ~ treatment, data=bidgee)
            Kruskal-Wallis rank sum test
data:  Debris by treatment
Kruskal-Wallis chi-squared = 8.6094, df = 4, p-value = 0.07164
```




```
#anova table for debrisasin
> anovaDA.bidgee <- aov(debrisasin ~ treatment, data=bidgee)
> summary(anovaDA.bidgee)
          Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.4338 0.10846   2.479  0.063 .
Residuals   33 1.4436 0.04375
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

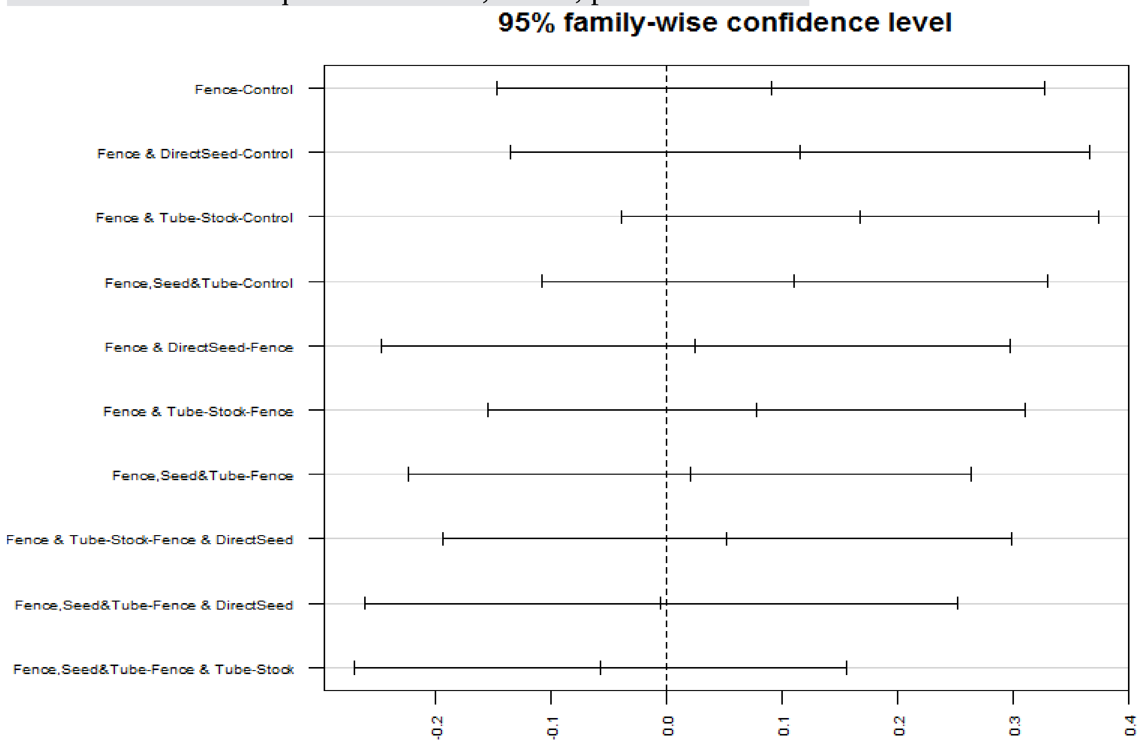
Features



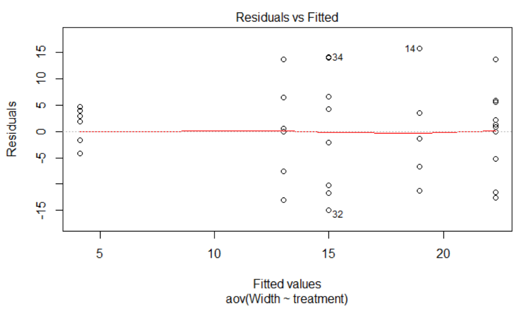


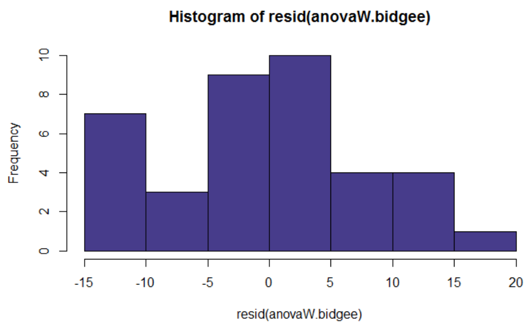

```
#anova table for Features
```

```
> anovaF.bidgee <- aov(Features ~ treatment, data=bidgee)
> summary(anovaF.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.1387 0.03468    1.42  0.249
Residuals   33 0.8058 0.02442
> kruskal.test(Features ~ treatment, data=bidgee)
               Kruskal-Wallis rank sum test
data:  Features by treatment
Kruskal-Wallis chi-squared = 5.5155, df = 4, p-value = 0.2384
```



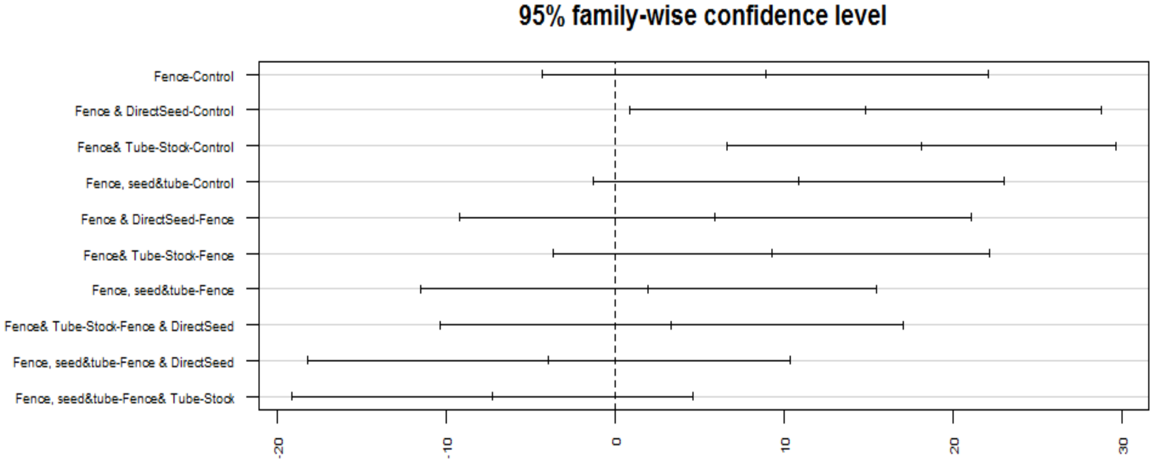
Width of riparian zone

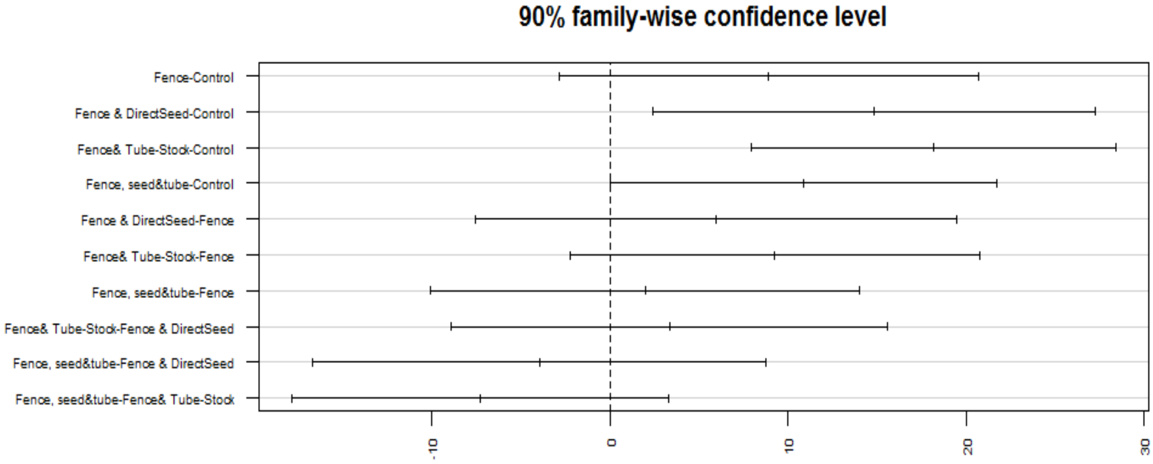




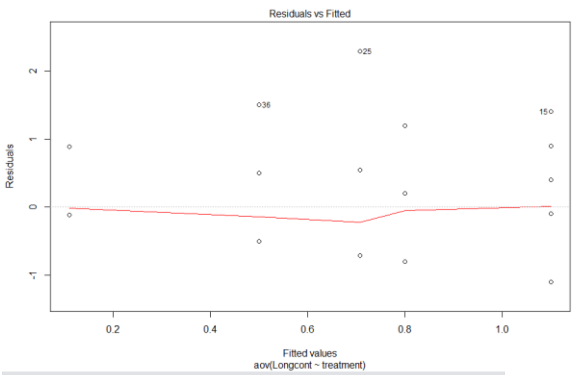

```
#anova table for Width
> anovaW.bidgee <- aov(Width ~ treatment, data=bidgee)
> summary(anovaW.bidgee)
            Df Sum Sq Mean Sq F value  Pr(>F)
treatment    4   1690   422.5   5.598 0.00149 **
Residuals   33   2491    75.5
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> kruskal.test(Width ~ treatment, data=bidgee)
               Kruskal-Wallis rank sum test
```

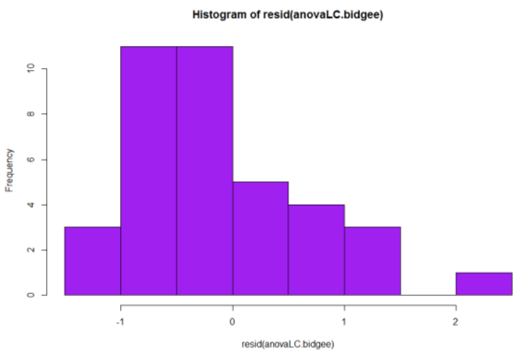
data: Width by treatment
Kruskal-Wallis chi-squared = 15.3664, df = 4, p-value = 0.003999





Longitudinal continuity





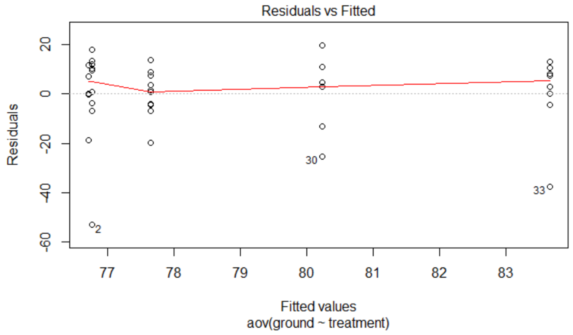
plot(tukeyWidth2, las=2, cex.axis=0.6)
> #anova table for Longcont
> anovaLC.bidgee <- aov(Longcont ~ treatment, data=bidgee)
> summary(anovaLC.bidgee)

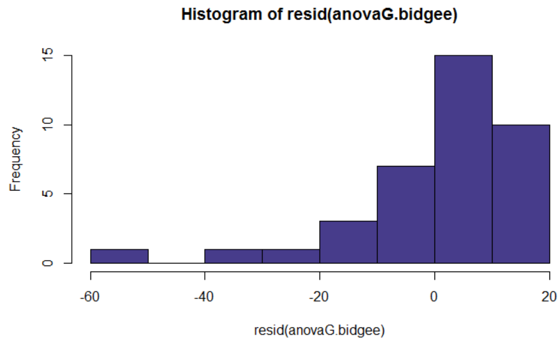
	Df	Sum Sq	Mean Sq	F value	Pr(>F)
treatment	4	4.946	1.2365	1.802	0.152
Residuals	33	22.641	0.6861		

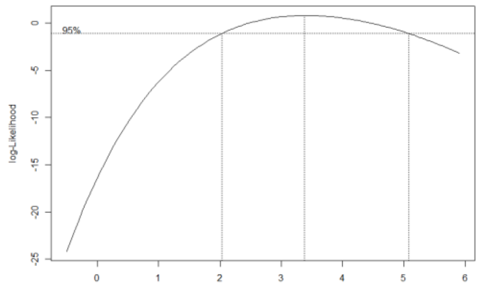
> kruskal.test(Longcont ~ treatment, data=bidgee)
Kruskal-Wallis rank sum test
data: Longcont by treatment

```
Kruskal-Wallis chi-squared = 7.4823, df = 4, p-value = 0.1125
```

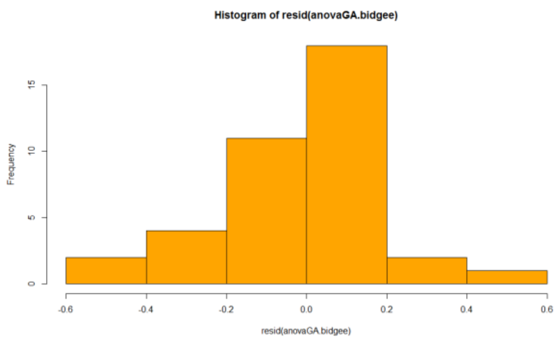
Ground cover by restoration method

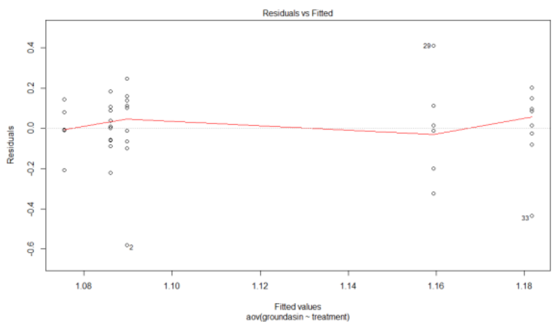




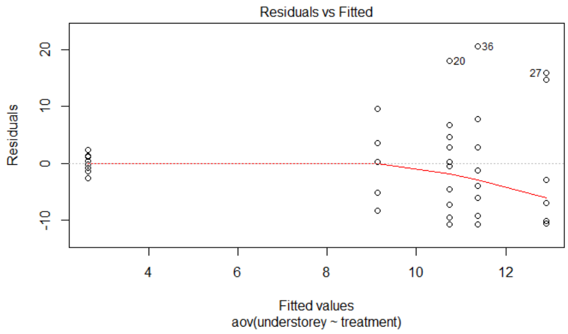


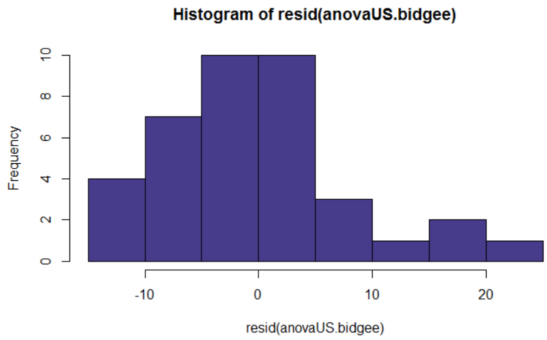

```
#anova table for ground
> anovaG.bidgee <- aov(ground ~ treatment, data=bidgee)
> summary(anovaG.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4    272   68.08   0.273  0.893
Residuals   33   8222  249.14
> kruskal.test(ground ~ treatment, data=bidgee)
               Kruskal-Wallis rank sum test
data:  ground by treatment
Kruskal-Wallis chi-squared = 3.6797, df = 4, p-value = 0.4511
#anova table for groundasin
> anovaGA.bidgee <- aov(groundasin ~ treatment, data=bidgee)
> summary(anovaGA.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  0.069 0.01726   0.448  0.773
Residuals   33  1.271 0.03851
```



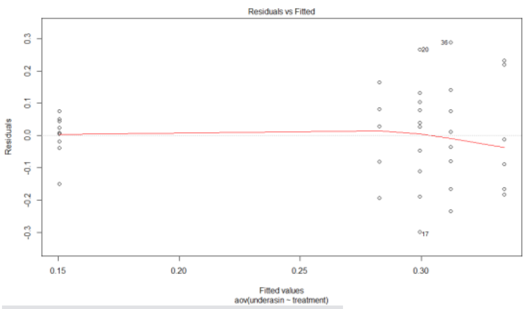


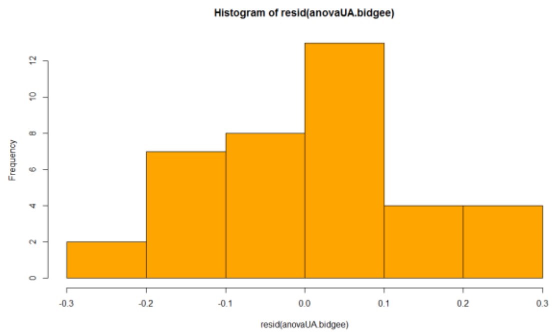
Mid-storey cover by restoration method





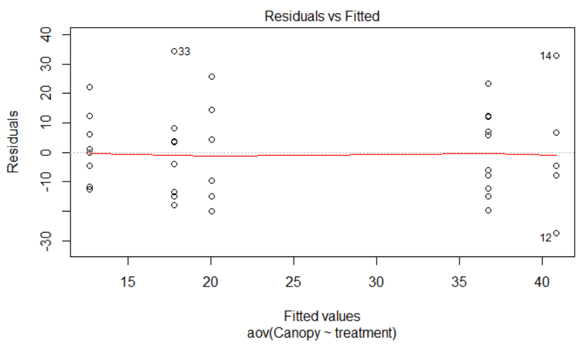

```
#anova table for mid-storey
> anovaUS.bidgee <- aov(understorey ~ treatment, data=bidgee)
> summary(anovaUS.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  532.3  133.08   1.832  0.146
Residuals   33 2396.8   72.63
> kruskal.test(understorey ~ treatment, data=bidgee)
              Kruskal-Wallis rank sum test
data:  mid-storey by treatment
Kruskal-Wallis chi-squared = 7.1722, df = 4, p-value = 0.1271
```

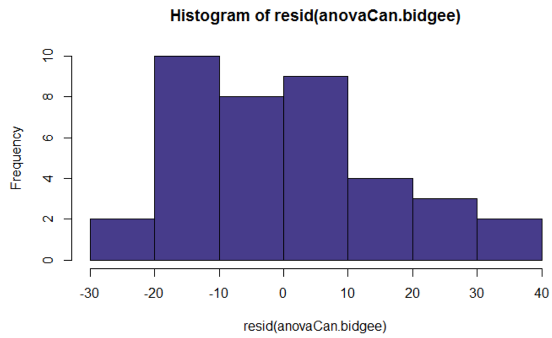




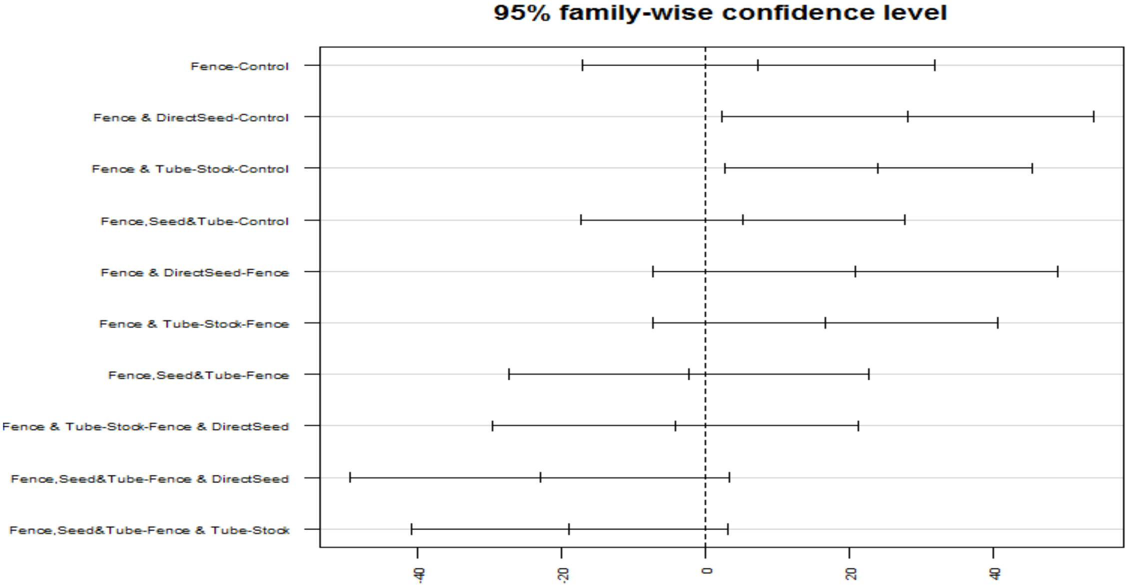

```
#anova table for underasin
> anovaUA.bidgee <- aov(underasin ~ treatment, data=bidgee)
> summary(anovaUA.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.1768 0.04421   1.987  0.119
Residuals   33 0.7344 0.02225
```

Canopy cover by restoration method

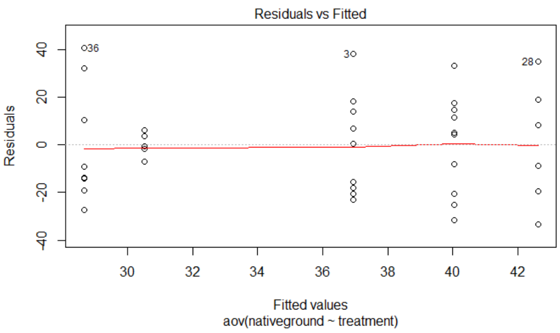


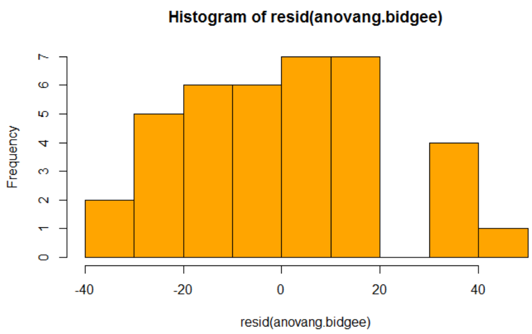


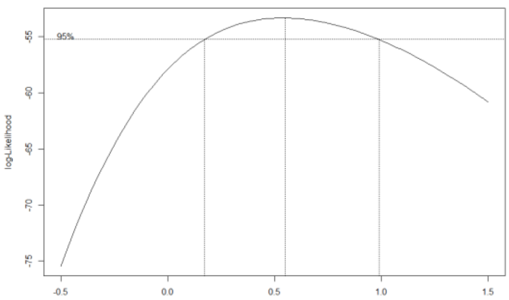
```
#anova table for Canopy
> anovaCan.bidgee <- aov(Canopy ~ treatment, data=bidgee)
> summary(anovaCan.bidgee)
           Df Sum Sq Mean Sq F value Pr(>F)
treatment   4   4563    1141   4.404 0.0058 **
Residuals  33   8547     259
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(Canopy ~ treatment, data=bidgee)
            Kruskal-Wallis rank sum test
data:  Canopy by treatment
Kruskal-Wallis chi-squared = 12.6148, df = 4, p-value = 0.01332
```



Native groundcover by restoration method

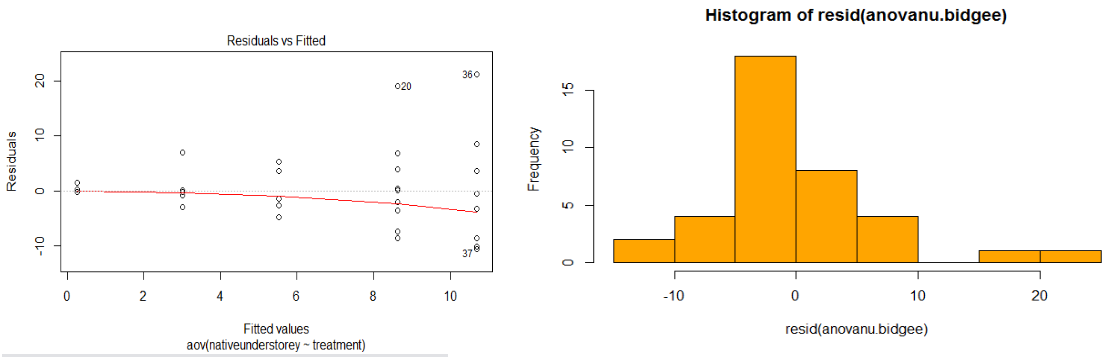




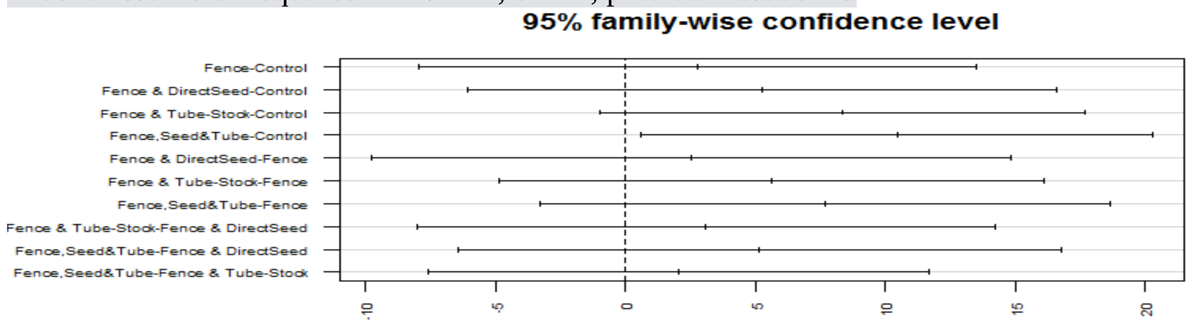


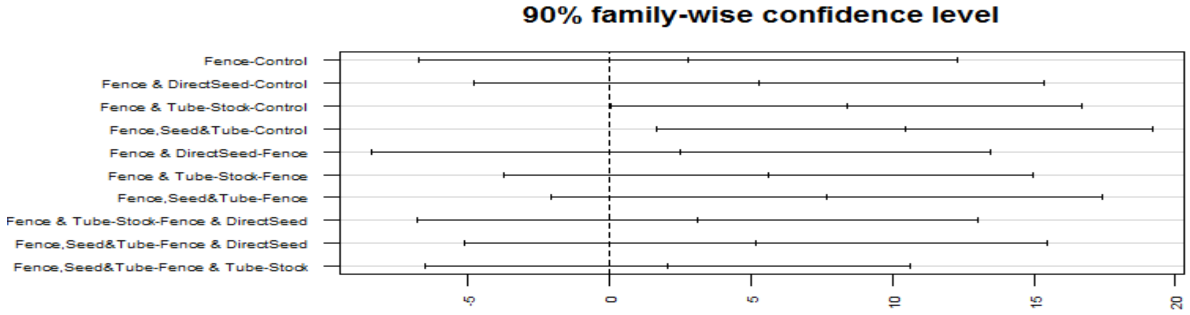
```
#anova table for nativeground
> anovang.bidgee <- aov(nativeground ~ treatment, data=bidgee)
> summary(anovang.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4   1016   254.0   0.554  0.698
Residuals   33  15135   458.6
> kruskal.test(nativeground ~ treatment, data=bidgee)
             Kruskal-Wallis rank sum test
data:  nativeground by treatment
Kruskal-Wallis chi-squared = 2.1445, df = 4, p-value = 0.7092
```

Native mid-storey by restoration method

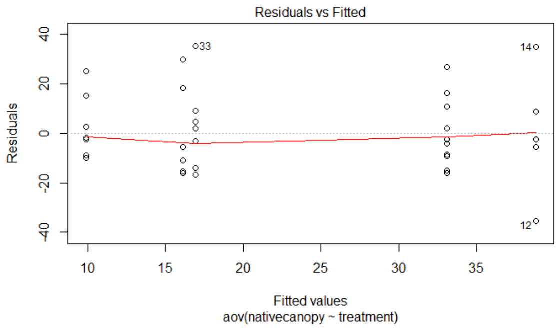


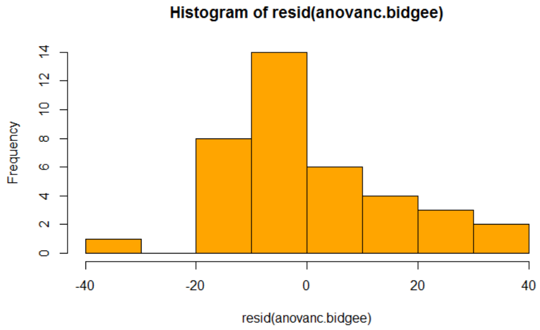

```
#anova table for nativeunderstorey
> anovanu.bidgee <- aov(nativeunderstorey ~ treatment, data=bidgee)
> summary(anovanu.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  594.9  148.73   3.001 0.0323 *
Residuals   33 1635.7   49.57
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(nativeunderstorey ~ treatment, data=bidgee)
             Kruskal-Wallis rank sum test
data:  nativeunderstorey by treatment
Kruskal-Wallis chi-squared = 14.2722, df = 4, p-value = 0.006475
```



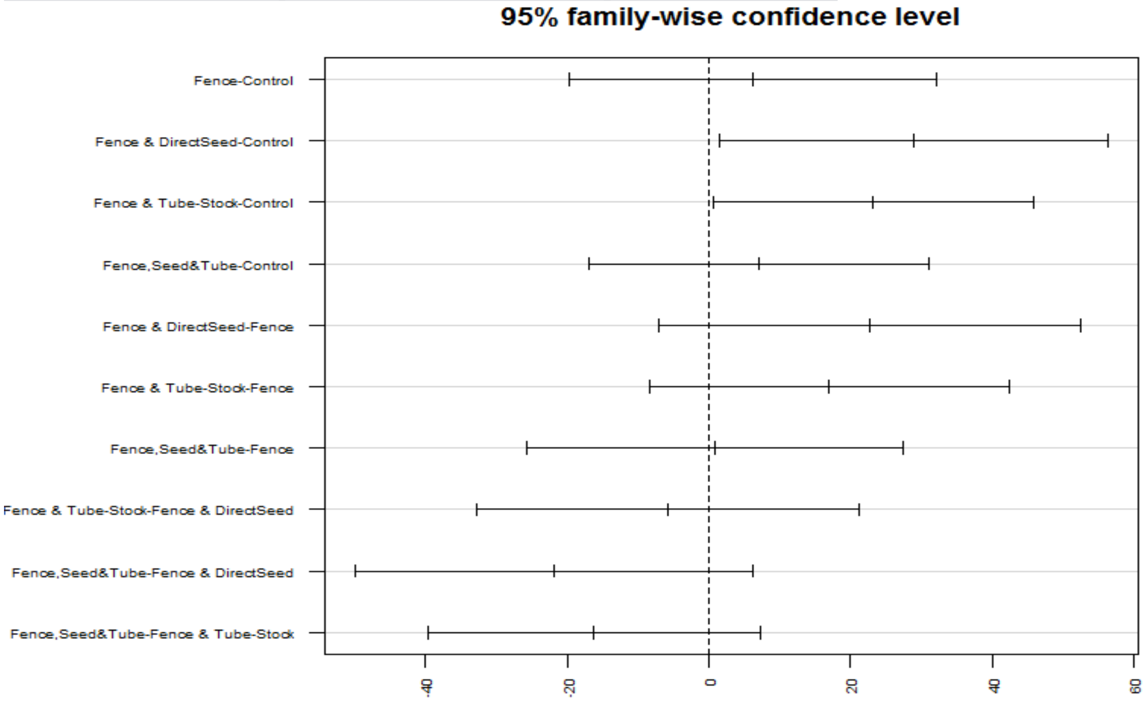


Native canopy cover by restoration method

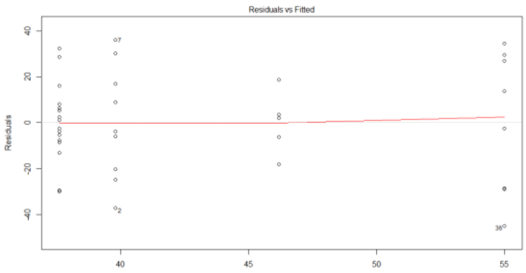


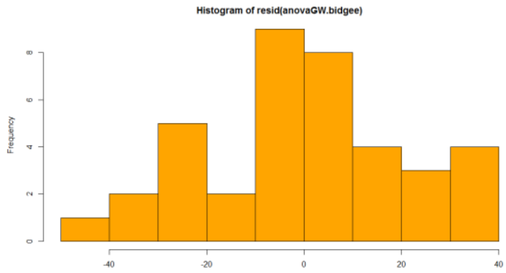


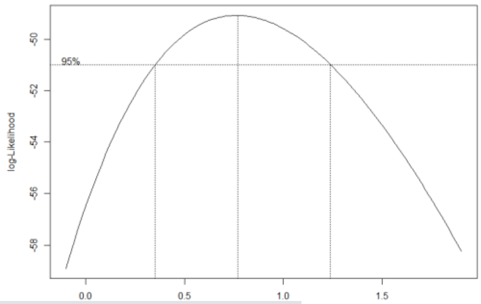

```
#anova table for nativecanopy
> anovanc.bidgee <- aov(nativecanopy ~ treatment, data=bidgee)
> summary(anovanc.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4   4383  1095.7   3.756 0.0126 *
Residuals   33   9626   291.7
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> kruskal.test(nativecanopy ~ treatment, data=bidgee)
              Kruskal-Wallis rank sum test
data:  nativecanopy by treatment
Kruskal-Wallis chi-squared = 11.6027, df = 4, p-value = 0.02056
```



Non-native groundcover (%) by restoration method

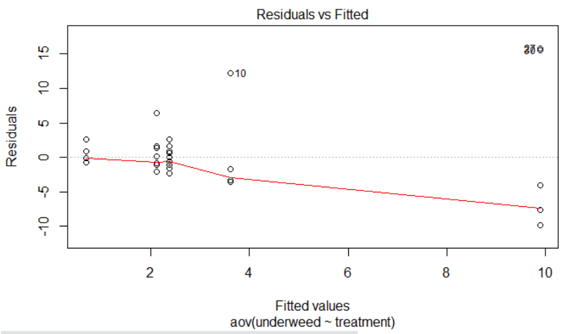


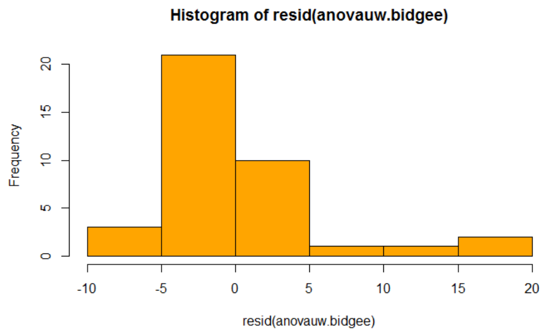





```
#anova table for groundweed
> anovaGW.bidgee <- aov(groundweed ~ treatment, data=bidgee)
> summary(anovaGW.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4   1760   440.0   0.862  0.497
Residuals   33  16852   510.7
> kruskal.test(groundweed ~ treatment, data=bidgee)
              Kruskal-Wallis rank sum test
data:  groundweed by treatment
Kruskal-Wallis chi-squared = 2.2667, df = 4, p-value = 0.6868
```

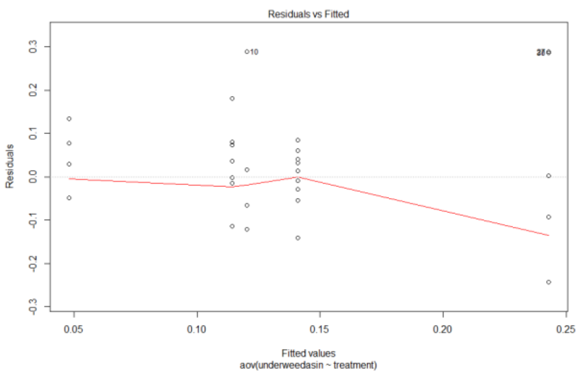
Non-native mid-storey (%) cover by restoration method

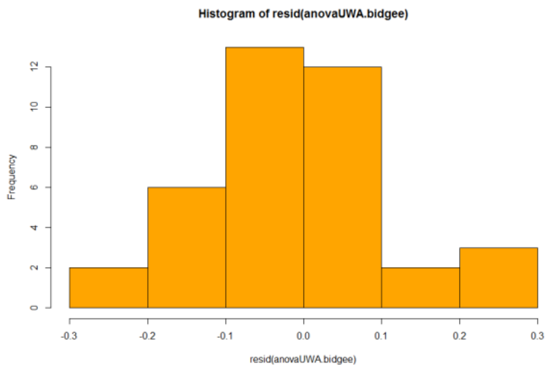


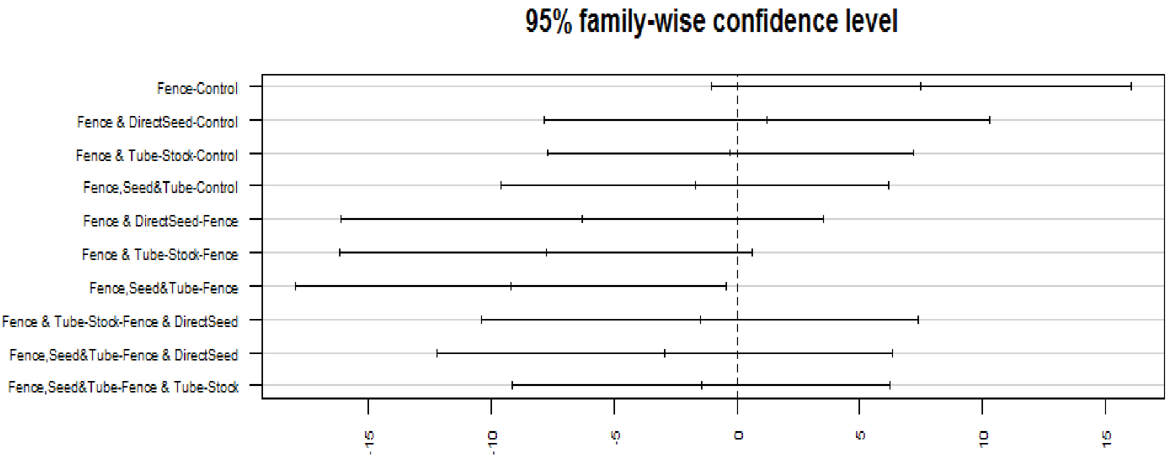


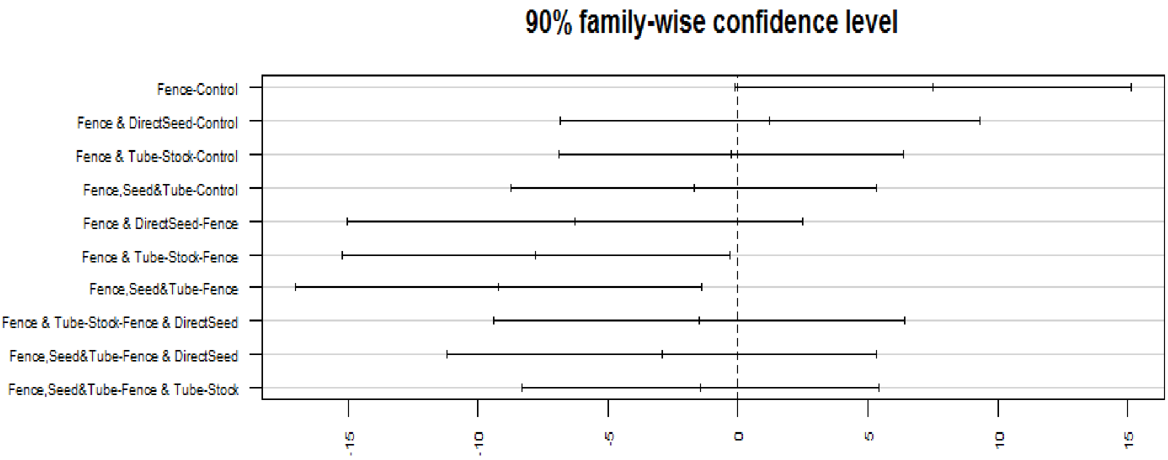

```
#anova table for underweed
> anovauw.bidgee <- aov(underweed ~ treatment, data=bidgee)
> summary(anovauw.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  337.6   84.39   2.657 0.0501 .
Residuals   33 1048.1   31.76
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(underweed ~ treatment, data=bidgee)
              Kruskal-Wallis rank sum test
data:  underweed by treatment
Kruskal-Wallis chi-squared = 5.9499, df = 4, p-value = 0.2029
```

```
#anova table for underweedasin
> anovaUWA.bidgee <- aov(underweedasin ~ treatment, data=bidgee)
> summary(anovaUWA.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.1337 0.03344   1.938  0.127
Residuals   33 0.5693 0.01725
```

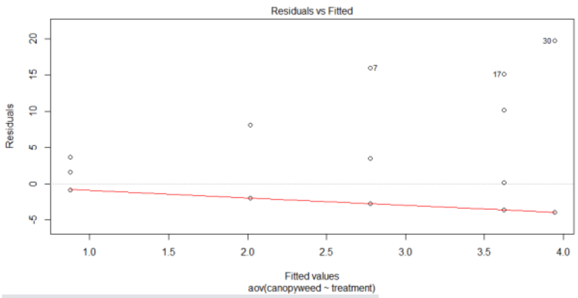


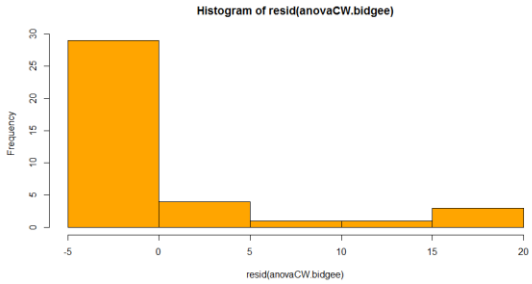






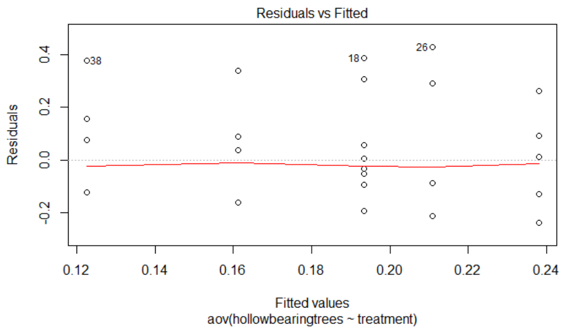
Non-native canopy by restoration method

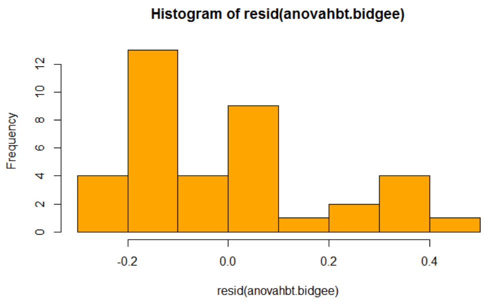





```
#anova table for canopyweed
> anovaCW.bidgee <- aov(canopyweed ~ treatment, data=bidgee)
> summary(anovaCW.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4   46.9   11.72   0.295  0.879
Residuals   33 1312.9   39.78
> kruskal.test(canopyweed ~ treatment, data=bidgee)
              Kruskal-Wallis rank sum test
data:  canopyweed by treatment
Kruskal-Wallis chi-squared = 0.333, df = 4, p-value = 0.9876
```

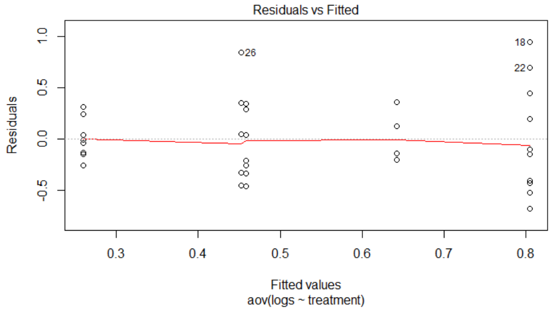
Hollow-bearing trees

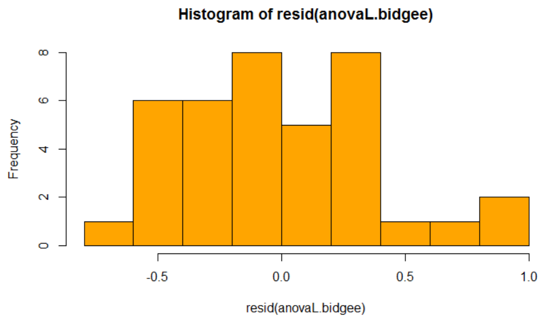





```
#anova table for hollowbearingtrees
> anovahbt.bidgee <- aov(hollowbearingtrees ~ treatment, data=bidgee)
> summary(anovahbt.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.0539 0.01348   0.313  0.867
Residuals   33 1.4224 0.04310
> kruskal.test(hollowbearingtrees ~ treatment, data=bidgee)
              Kruskal-Wallis rank sum test
data:  hollowbearingtrees by treatment
Kruskal-Wallis chi-squared = 1.6392, df = 4, p-value = 0.8017
```

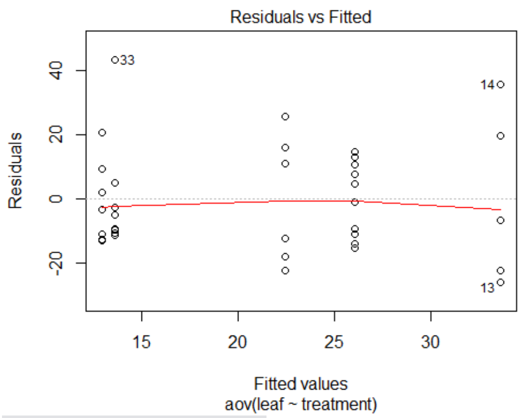
Presence of fallen logs by restoration method

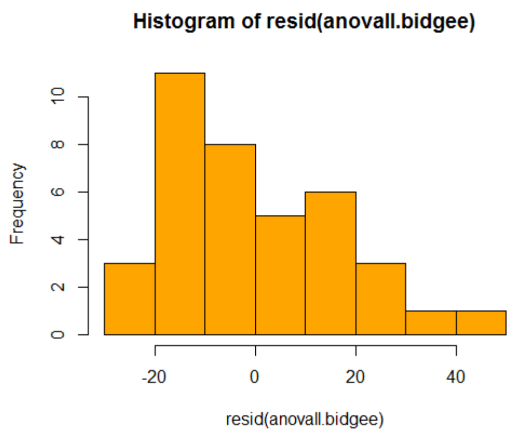





```
#anova table for logs
> anovaL.bidgee <- aov(logs ~ treatment, data=bidgee)
> summary(anovaL.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  1.485  0.3712   2.284 0.0811 .
Residuals   33  5.362  0.1625
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(logs ~ treatment, data=bidgee)

              Kruskal-Wallis rank sum test
data:  logs by treatment
Kruskal-Wallis chi-squared = 7.2229, df = 4, p-value = 0.1246
```

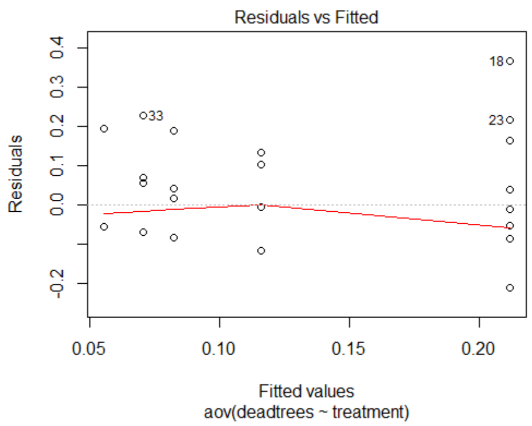
Leaf litter by restoration method

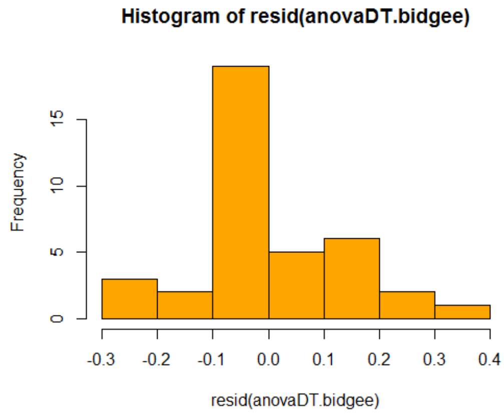





```
#anova table for leaf
> anovall.bidgee <- aov(leaf ~ treatment, data=bidgee)
> summary(anovall.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4   2078   519.5   1.698  0.174
Residuals   33  10095   305.9
> kruskal.test(leaf ~ treatment, data=bidgee)
              Kruskal-Wallis rank sum test
data:  leaf by treatment
Kruskal-Wallis chi-squared = 7.6227, df = 4, p-value = 0.1064
```

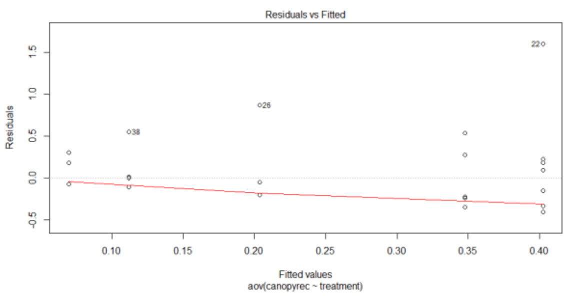
Standing dead trees by restoration method

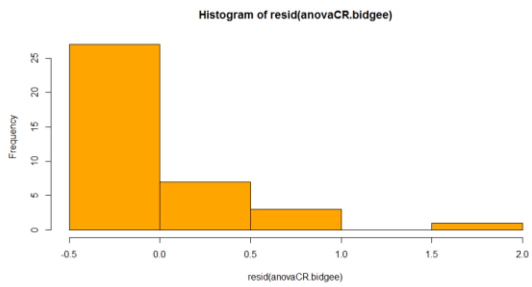


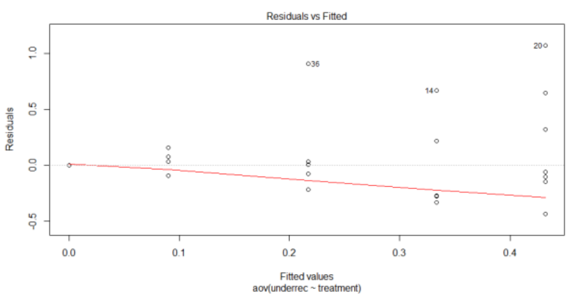


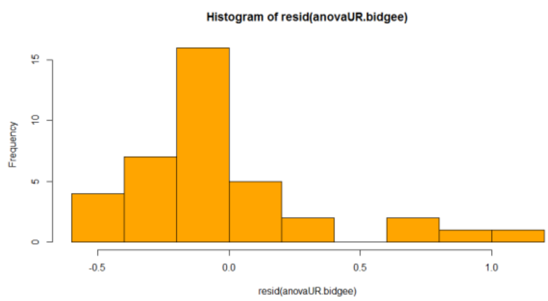

```
#anova table for deadtrees
> anovaDT.bidgee <- aov(deadtrees ~ treatment, data=bidgee)
> summary(anovaDT.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4 0.1472 0.03681   1.865   0.14
Residuals   33 0.6515 0.01974
> kruskal.test(deadtrees ~ treatment, data=bidgee)

            Kruskal-Wallis rank sum test
data:  deadtrees by treatment
Kruskal-Wallis chi-squared = 5.3392, df = 4, p-value = 0.2542
```

Canopy and mid-storey seedling recruitment







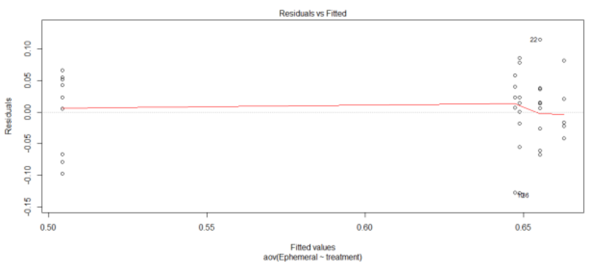


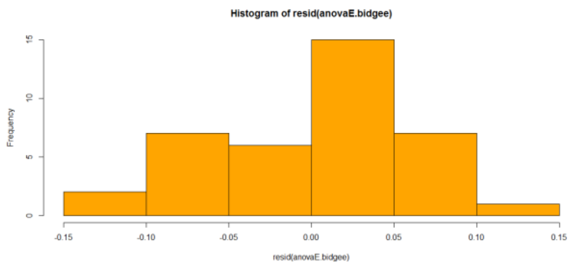

```
#anova table for canopyrec
> anovaCR.bidgee <- aov(canopyrec ~ treatment, data=bidgee)
> summary(anovaCR.bidgee)
```

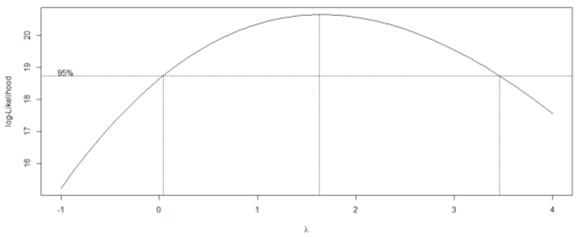
```
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  0.714  0.1785    1.08  0.382
Residuals   33  5.455  0.1653
> kruskal.test(canopyrec ~ treatment, data=bidgee)
                Kruskal-Wallis rank sum test
data:  canopyrec by treatment
Kruskal-Wallis chi-squared = 5.3374, df = 4, p-value = 0.2544

#anova table for underrec
> anovaUR.bidgee <- aov(underrec ~ treatment, data=bidgee)
> summary(anovaUR.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  1.051  0.2627   2.029  0.113
Residuals   33  4.273  0.1295
> kruskal.test(underrec ~ treatment, data=bidgee)
                Kruskal-Wallis rank sum test
data:  underrec by treatment
Kruskal-Wallis chi-squared = 10.2217, df = 4, p-value = 0.03685
```

Erosion state and Bank Stability

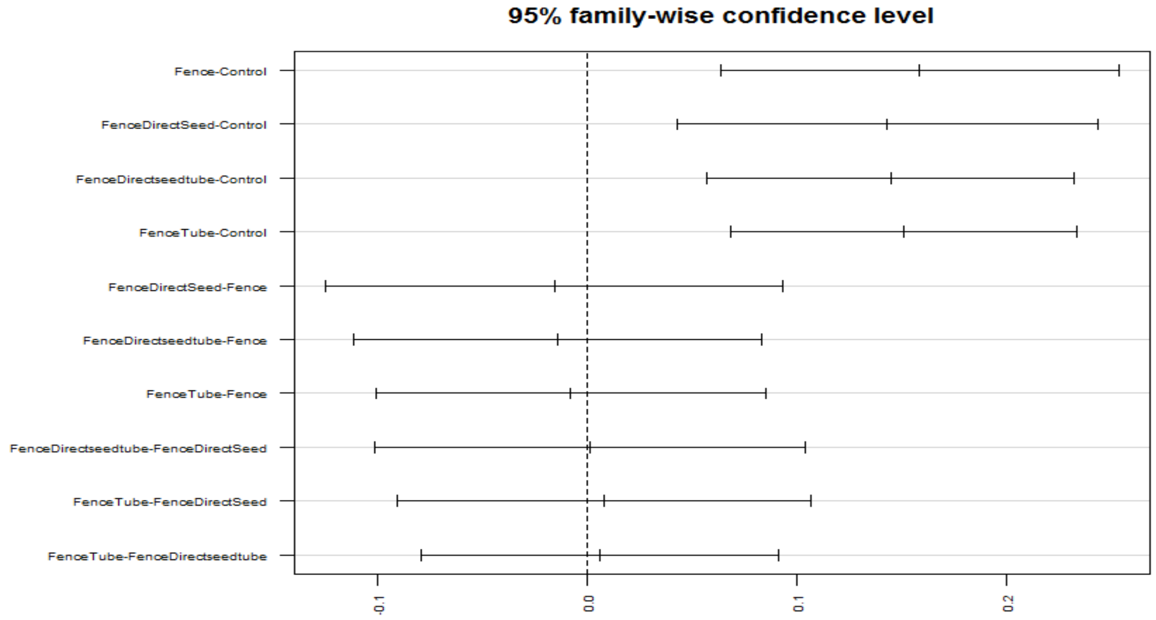
Ephemeral stream assessment



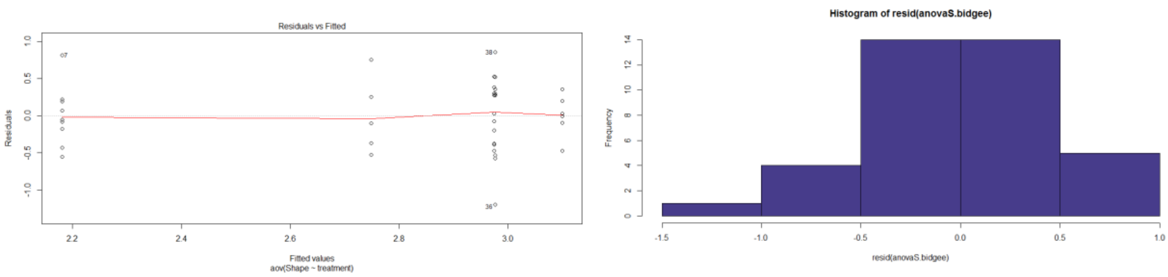


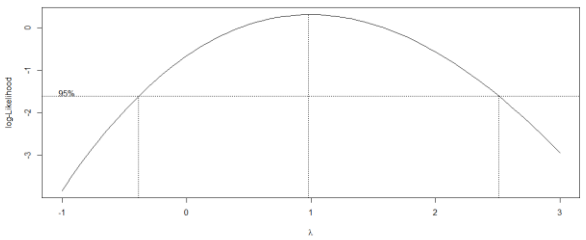



```
> #anova table for Ephemeral
> anovaE.bidgee <- aov(Ephemeral ~ treatment, data=bidgee)
> summary(anovaE.bidgee)
            Df Sum Sq Mean Sq F value  Pr(>F)
treatment    4 0.1540 0.03851   9.919 2.2e-05 ***
Residuals   33 0.1281 0.00388
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(Ephemeral ~ treatment, data=bidgee)
                Kruskal-Wallis rank sum test
data:  Ephemeral by treatment
Kruskal-Wallis chi-squared = 17.1493, df = 4, p-value = 0.001808
```



Shape of cross section

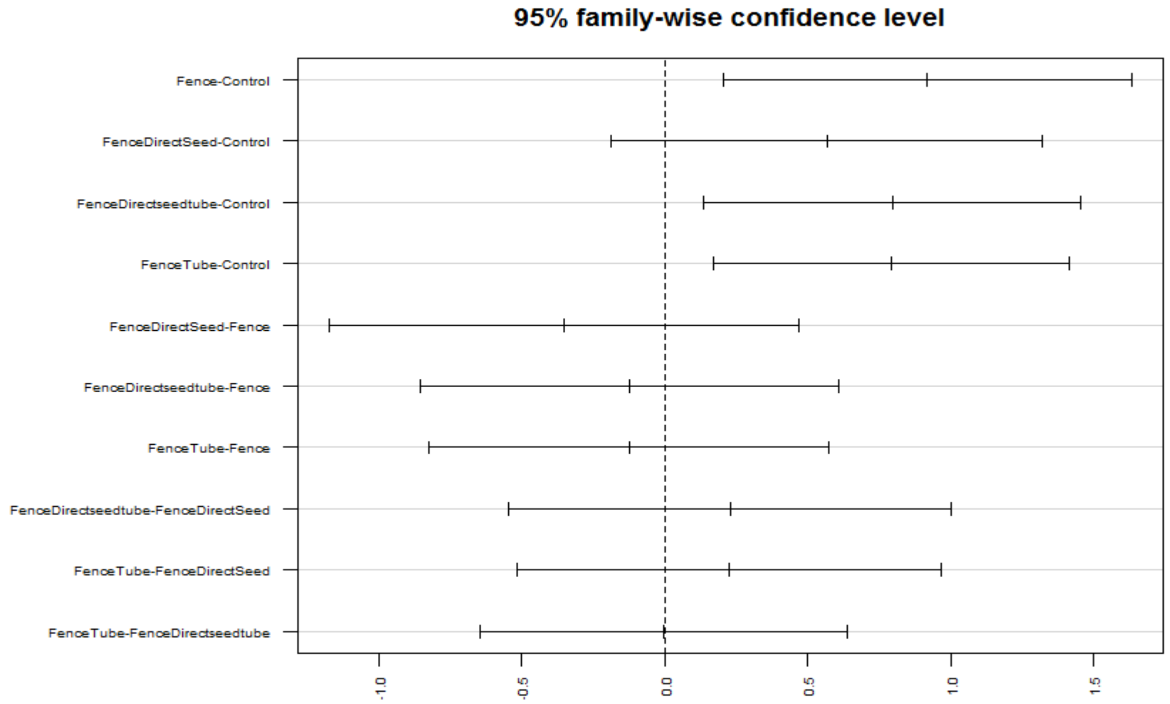




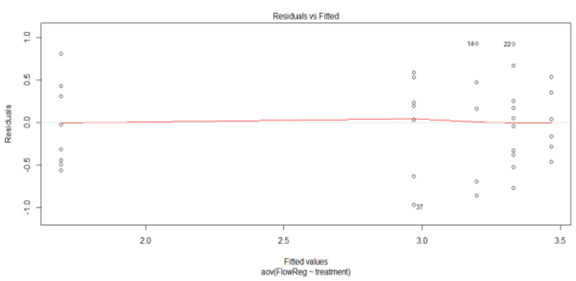
> #anova table for Shape
> anovaS.bidgee <- aov(Shape ~ treatment, data=bidgee)
> summary(anovaS.bidgee)

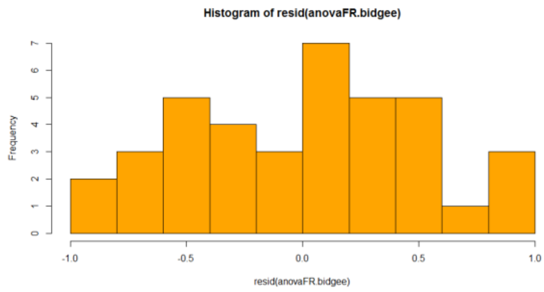
	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
treatment	4	4.538	1.134	5.156	0.00244	**
Residuals	33	7.260	0.220			

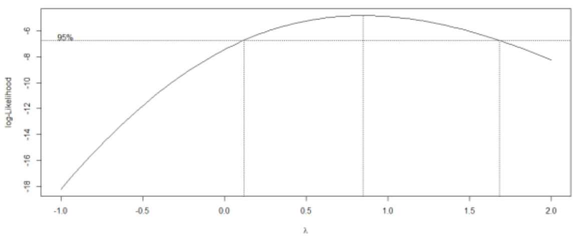
Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> kruskal.test(Shape ~ treatment, data=bidgee)
Kruskal-Wallis rank sum test
data: Shape by treatment
Kruskal-Wallis chi-squared = 14.5083, df = 4, p-value = 0.005838



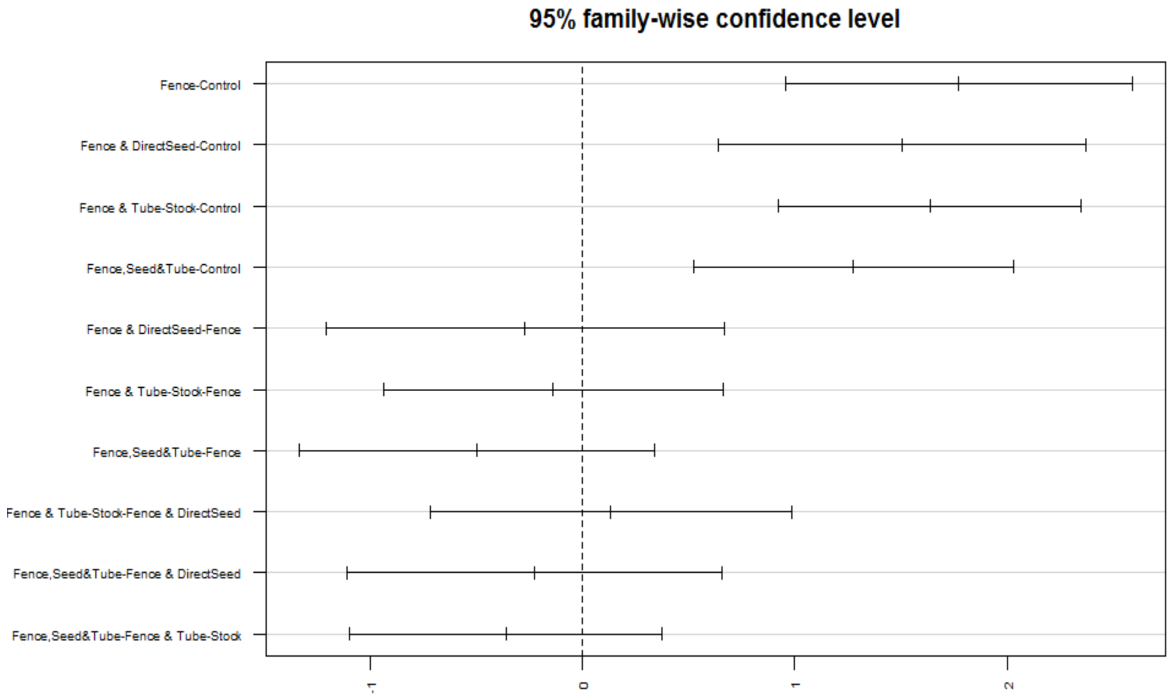
Lateral Flow Regulation



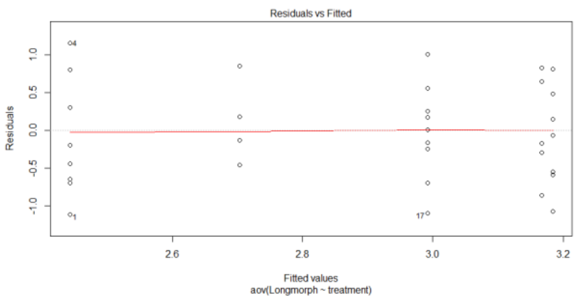


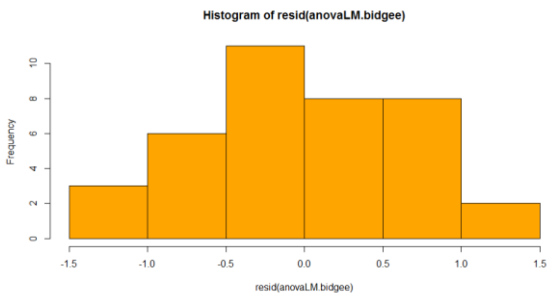


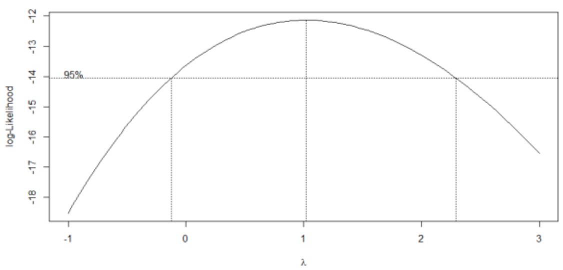

```
#anova table for FlowReg
> anovaFR.bidgee <- aov(FlowReg ~ treatment, data=bidgee)
> summary(anovaFR.bidgee)
            Df Sum Sq Mean Sq F value   Pr(>F)
treatment    4 17.315   4.329   14.94 4.55e-07 ***
Residuals   33  9.559   0.290
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(FlowReg ~ treatment, data=bidgee)
            Kruskal-Wallis rank sum test
data:  FlowReg by treatment
Kruskal-Wallis chi-squared = 20.2011, df = 4, p-value = 0.0004557
```



Longitudinal morphology








```
#anova table for Longmorph
> anovaLM.bidgee <- aov(Longmorph ~ treatment, data=bidgee)
> summary(anovaLM.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
treatment    4  3.235  0.8086   1.794  0.153
Residuals   33 14.871  0.4506
> kruskal.test(Longmorph ~ treatment, data=bidgee)
             Kruskal-Wallis rank sum test
data:  Longmorph by treatment
Kruskal-Wallis chi-squared = 5.6292, df = 4, p-value = 0.2286
```

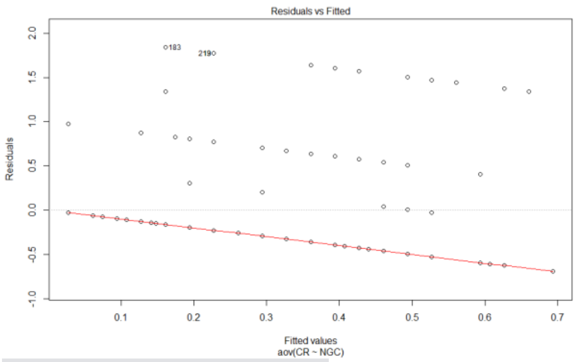
Presence of remnant vegetation

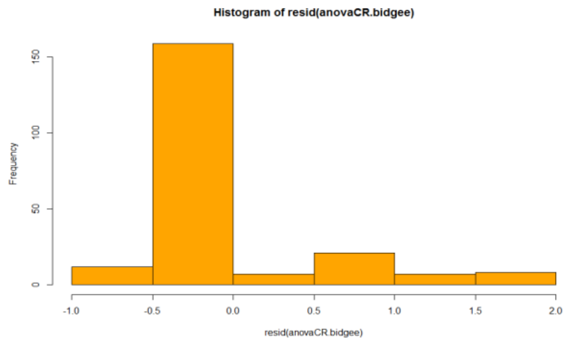
Remnant vegetation and seedling recruitment

```
#anova table for underrec
> anovaURR.bidgee <- aov(underrec ~ remnant, data=bidgee)
> summary(anovaURR.bidgee)
            Df Sum Sq Mean Sq F value Pr(>F)
remnant      1  0.898  0.8982   7.306 0.0104 *
Residuals   36  4.426  0.1229
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(underrec ~ remnant, data=bidgee)
              Kruskal-Wallis rank sum test

data:  underrec by remnant
Kruskal-Wallis chi-squared = 7.6165, df = 2, p-value = 0.02219
```

```
#anova table for canrec
> anovaCRR.bidgee <- aov(canrec ~ remnant, data=bidgee)
> summary(anovaCRR.bidgee)
            Df Sum Sq Mean Sq F value   Pr(>F)
remnant      1  2.579  2.5793   25.87 1.16e-05 ***
Residuals   36  3.590  0.0997
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(canrec ~ remnant, data=bidgee)
              Kruskal-Wallis rank sum test
data:  canrec by remnant
Kruskal-Wallis chi-squared = 18.4786, df = 2, p-value = 9.715e-05
```

Native ground cover influence on recruitment



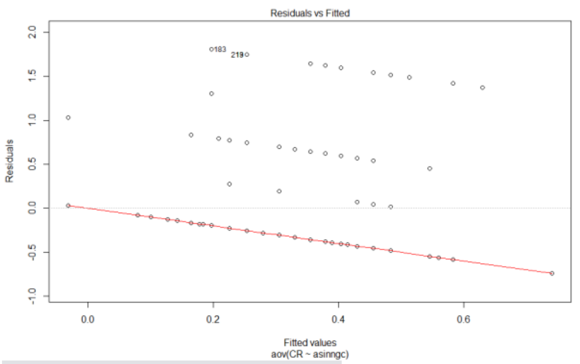


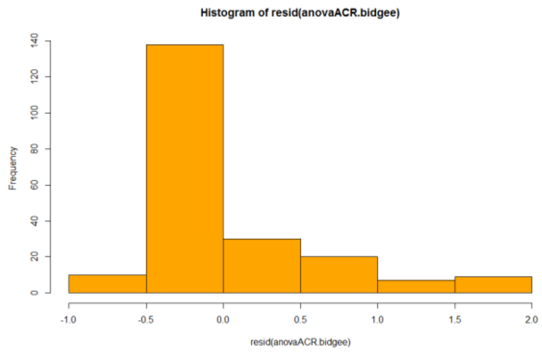

```
#anova table for NGCCR
> anovaCR.bidgee <- aov(CR ~ NGC, data=bidgee)
> summary(anovaCR.bidgee)
             Df Sum Sq Mean Sq F value   Pr(>F)
NGC           1   6.93   6.933   23.74 2.16e-06 ***
Residuals   212  61.91   0.292
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(CR ~ NGC, data=bidgee)
              Kruskal-Wallis rank sum test
data:  CR by NGC
Kruskal-Wallis chi-squared = 52.5761, df = 28, p-value = 0.00329
```

Data was not normal so an asin transformation was done:

```
> #anova table for asinngc
> anovaACR.bidgee <- aov(CR ~ asinngc, data=bidgee)
> summary(anovaACR.bidgee)
             Df Sum Sq Mean Sq F value   Pr(>F)    
asinngc       1   6.29   6.288   21.31 6.76e-06 ***
Residuals   212  62.56   0.295                     
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(CR ~ asinngc, data=bidgee)
              Kruskal-Wallis rank sum test
data:  CR by asinngc
Kruskal-Wallis chi-squared = 52.5761, df = 28, p-value = 0.00329
> #anova table for NGCUR
> anovaUR.bidgee <- aov(UR ~ NGC, data=bidgee)
> summary(anovaUR.bidgee)
             Df Sum Sq Mean Sq F value   Pr(>F)    
NGC           1   6.50   6.499   22.13 4.59e-06 ***
Residuals   212  62.26   0.294                     
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(UR ~ NGC, data=bidgee)
              Kruskal-Wallis rank sum test
data:  UR by NGC
Kruskal-Wallis chi-squared = 46.4153, df = 28, p-value = 0.01578
```

After Asin Data transformation






```
> #anova table for asinngc
> anovaAUR.bidgee <- aov(UR ~ asinngc, data=bidgee)
> summary(anovaAUR.bidgee)
             Df Sum Sq Mean Sq F value   Pr(>F)    
asinngc       1   5.88   5.875   19.81 1.38e-05 ***
Residuals   212  62.89   0.297                     
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> kruskal.test(UR ~ asinngc, data=bidgee)
              Kruskal-Wallis rank sum test
data:  UR by asinngc
Kruskal-Wallis chi-squared = 46.4153, df = 28, p-value = 0.01578
```

2. List of actively revegetated species used in the Bidgee Banks Restoration Project, number of sites were species were used, number of sites were species were observed, and occurrence probability.

Species used	expected	observed	Occurrence prob. %
Acacia boormanii	5	5	100
Acacia cultriformis	5	5	100
Acacia dealbata	11	10	90.91
Acacia deccurens	5	5	100
Acacia genistafolia	3	0	0
Acacia implexa	3	1	33.33
Acacia lanigera	2	2	100
Acacia mearnsii	10	4	40
Acacia melanoxylon	3	3	100
Acacia obliquinervia	1	1	100
Acacia pravissimia	2	2	100
Acacia rubida	11	6	54.56
Acacia siculifolia	1	1	100
Acacia terminalis	1	0	0
Acacia verniciflua	1	0	0
Acacia vestita	2	1	50
Callistemon citrinus	3	1	33.33
Callistemon pallidus	2	0	0
Callistemon pityoides	5	2	40
Callistemon sieberi	7	2	28.57
Carex appresor	1	0	0
Cassinia aculeata	1	0	0
Cassinia longifolia	2	0	0
Casuarina cunninghamiana	5	3	60
Casuarina littoralis	2	1	50
Davesia latifolia	1	0	0
Davesia mimosoides	3	1	33.33
Dodonaea viscosa	3	0	0
Eucalyptus albens	7	4	57.14
Eucalyptus blakelyi	8	7	87.5
Eucalyptus bridgesiana	9	6	66.66
Eucalyptus camaldulensis	6	3	50
Eucalyptus camphora	1	0	0
Eucalyptus cinerea	2	0	0

Eucalyptus globulus	3	2	66.66
Eucalyptus goniocalyx	4	3	75
Eucalyptus macarthrii	1	1	100
Eucalyptus macrorhyncha	3	1	33.33
Eucalyptus mannifera	3	3	100
Eucalyptus melliodora	10	7	70
Eucalyptus microcarpa	1	0	0
Eucalyptus pauciflora	3	1	33.33
Eucalyptus polyanthememos	8	5	62.5
Eucalyptus rossii	2	2	100
Eucalyptus rubida	2	0	0
Eucalyptus saligna	1	1	100
Eucalyptus sideroxylon	2	0	0
Eucalyptus stellulata	3	1	33.33
Eucalyptus viminalis	7	6	85.71
Hardenburgia violaceae	6	0	0
Hovea rosmarinifolia	1	0	0
Indigophera australis	1	0	0
Kunzea ericoides	2	0	0
Leptospermum lanigerum	3	3	100
Leptospermum obavatum	7	1	14.28
Leptospermum polygalifolium	1	0	0
Lomandra longifolia	5	3	60
Melaleuca armillaris	3	1	33.33
Melaleuca decussata	2	0	0
Melaleuca ericafolia	7	5	71.42
Microseris lanceolata	1	0	0
Poa labillarderi	4	0	0