Monitoring and assessment of direct seeding revegetation projects in the Goulburn Broken CMA

A Report to the Goulburn Broken Catchment Management Authority

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GOULBURN BROKEN CATCHMENT MANAGEMENT AUTHORITY

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Executive Summary

The aim of this study was to design a monitoring protocol to evaluate direct seeding revegetation undertaken between 1999–2009 in the productive plains and upland slopes of the Goulburn Broken CMA, NE Victoria, and to identify factors that influence the success of conservation outcomes among sites. This information was then used to direct ongoing management of these sites, to guide management of future direct seeding, and to elucidate areas where further research is required to assist adaptive management of the catchment's revegetation projects.

We designed a monitoring protocol that targeted attributes able to indicate a site's relative trajectory along the restoration path and its capacity to be self-regenerating. The data that was collected will be stored for use in comparisons with future long-term monitoring. We analysed the current dataset, using a modelling approach to test hypotheses about the influence of environmental, ecological and management factors on our 'success' indicators (stem density and species survival in furrows, natural regeneration outside furrows, and abundance of weeds). We used an information theoretic approach to identify which factors had the greatest influence on each of our indicators.

The number of species sown was found to increase stem density, suggesting that seed mixes with higher species richness could be used to improve establishment success. However, the majority of sites in this study were found to be too dense, if restoring a site to its pre-disturbance state is the objective of a direct seeding project. To reduce densities, either a decrease in the amount of seed sown or prescribed thinning of young stands would be recommended. Species survival was related to the topographic wetness index of a site, with more species surviving in sites whose position in the landscape permitted greater potential for water retention. Direct seeding has commonly involved seeding a mixture of species simultaneously rather than sowing species individually across a site. Separating species and placing them in a site according to their germination niche could improve species survival rates.

Natural regeneration occurred primarily from seeds dispersing from outside the sites. Thus, at 6–15 years old, these sites are not capable of being self-sustaining, even with a substantial amount of *Acacia* seed deposition. We recommend experimenting with

disturbance options (e.g. scalping, fire) to determine whether they improve regeneration success and should become prescribed management.

Weed cover was high at most direct-seeded sites, and native herbaceous cover was low. The legacy of fertiliser use at most sites was found to be facilitating the dominance of exotic annual weeds over indigenous perennial ground cover species, impeding the establishment of species-rich herbaceous ground layers typical of the original plant communities existing across the study area (e.g. Plains Grassy Woodlands). Interestingly, crash grazing (fast-rotation grazing) was found to have a suppressive effect on weed abundance. Crash grazing is a relatively new method of livestock grazing and thus there is sparse literature on its impacts on native Australian species. Therefore, we recommend investigating the role of crash grazing in weed control, in particular the timing and frequency of grazing events to reduce weed biomass while permitting native seedling growth.

Much uncertainty remains about the future conservation outcomes of direct seeded restoration projects. Long-term monitoring projects can be invaluable for determining the importance of management practices on outcomes, as their influence can fluctuate through time. Long-term monitoring can also assist in our understanding of contextual influences that cannot be entirely accounted for in snapshot studies but that remain of crucial importance to outcomes. During the course of this project, a long-term monitoring project was established with the aim of robustly quantifying the results of direct seeding. In winter 2015, 8 sites were measured–following the monitoring protocol established in this project–prior to direct seeding. These sites will be monitored at regular intervals in the coming years. This project represents a significant contribution to the field of restoration ecology in Australia.

Contents

1. Introduction

The Goulburn Broken Catchment Management Authority's Biodiversity Strategy aims to protect, extend and enhance the quantity and quality of native vegetation in the catchment by 2030 (Miles et al. 2010). Much of the original native vegetation was cleared to make way for agriculture after European settlement. Hence, the majority of remaining floral biodiversity occurs on private land and a key part of the Authority's strategy involves offering incentives to landholders to restore native vegetation to areas of their property. Restoration projects across the catchment have various objectives and associated starting states, for example: augmenting native remnants, waterways or wetlands; revegetation of corridors (i.e. roadsides, riparian buffers, wildlife corridors connecting habitat remnants); shelterbelt creation; and revegetation of cleared and degraded patches >2 ha (Miles et al. 2010; Rumpff et al. 2010).

Since the late 1990s, direct seeding has been increasingly employed for revegetation projects because of its economic and implementation efficiencies compared to tube stock planting. The process involves using a mechanical seeder to scalp the topsoil and deposit seed in the resulting furrows. Since 2000, direct seeding has been used in over 300 revegetation projects in the catchment, with the long-term intention of restoring degraded areas to their pre-clearance ecological vegetation class (EVC) (Miles et al. 2010; Department of Sustainability and Environment 2006). Whether these sites are on-target to reach their biodiversity outcomes is unknown because, like many restoration efforts, they have not yet been systematically monitored. Long-term monitoring and evaluation allows us to assess the trajectory of restoration efforts and to understand how management practices can be changed to improve outcomes (Lindenmayer et al. 2012).

To rectify the uncertainty around progress of directly seeded sites in the Goulburn Broken catchment, we designed a monitoring protocol to assess the current condition of sites aged 6–15 years, that could also be used to re-survey and evaluate their development through time. The CMA were interested in understanding not only what condition the sites were in relative to one another, but also in determining which factors may be the most important in driving a site's 'success'. Deciding which ecosystem attributes to measure to assess revegetation success was based on a brief review of the literature and on consultation with the CMA.

Here we report on the process of designing the monitoring protocol, its implementation and the findings from the first year of survey. Results of monitoring of this first round were used to derive models of the processes that were considered likely to influence the condition of direct seeded sites at 6–15 years of age. Model results are interpreted to inform the CMA of practices that may need to be adjusted in future direct seeding projects, and areas where further research is required to understand management that could rectify identified shortcomings of revegetated sites.

1.1. Designing the monitoring protocol

To determine both how to measure (i) successful outcomes of direct seeding and (ii) factors influencing outcomes, we conducted a review of the direct seeding literature. We included only research articles in our search that had been conducted in temperate regions in Australia and which involved revegetation of native species. The search was undertaken using Google Scholar, EBSCO, Web of Science and Science Direct and produced a total of 32 relevant research articles. Of these, only 16 evaluated the outcomes of direct seeding. Establishment after direct seeding (0–5 years) was the most common phase assessed, with 12 articles solely exploring this period. Only three articles reported results on long-term monitoring, with the oldest site being 17 years. Factors affecting plant establishment are useful to indicate early filters that may be influencing plant composition, however, they may not be as relevant for explaining the variation in vegetation attributes and conditions of more mature sites. With this in mind, we incorporated the theories of 13 published reviews of direct seeding to assist with choosing attributes to measure direct seeding outcomes and included literature on other restoration types (hand planting, natural regeneration) in southeastern Australia to help formulate hypotheses about factors influencing successful outcomes.

In 2005 Ruiz-Jaen & Aide published a review of how researchers and practitioners measured restoration success. They found that most studies measured three main site attributes: (1) diversity (species richness and abundance); (2) vegetation structure (e.g. % cover, plant density); and (3) ecological processes (nutrient cycling, biological interactions that lead to functional integrity, e.g. pollination and dispersal), although ecological processes were only rarely measured (Ruiz-Jaen and Aide 2005). These same patterns of assessment were found for the 16 research articles evaluated, with few studies incorporating all three categories of attributes. While the three categories only represent some of the measures required for a comprehensive assessment ((Ruiz-Jaen and Aide) mention nine), if all three are included in a monitoring program they are considered a robust indicator of a site's trajectory along the restoration path and its capacity to be self-regenerating (Ruiz-Jaen and Aide 2005; Tolsma, Duncan, and Sutter 2013). As such, we included attributes belonging to all three categories in our monitoring design. Ideally, we would also want to compare attribute measures to those of our reference habitat (e.g. a remnant of the appropriate EVC) to allow us to determine the extent to which the site has been restored (Ruiz-Jaen and Aide 2005). Although this was beyond the scope of the present study, it is certainly something to consider in future projects (Section 5).

1.2. Attributes indicating successful outcomes

With any restoration monitoring program, choosing attributes to measure will be shaped by limitations in resources and time. Such was the case in the present study. We chose attributes that were easily observed and measured (i.e. above-ground vegetation), could be sampled and compared at different time periods, were indicative of biological functioning, and were likely to respond predictably to stress (Tolsma, Duncan, and Sutter 2013). To determine the success of germination, growth and survival of sown species and successional stage reached, we measured the plants growing in the furrow lines recording: species richness, abundance, density of plants, life stages and structural attributes. To assess the capacity of sites to perpetuate and the effectiveness of localised seed dispersal we surveyed areas between furrows for any natural regeneration (again recording species and life stage). Finally, we measured native and exotic ground covers to indicate the condition of the soil seed bank, with sites containing a lower abundance of exotic weeds considered to be in better condition. Details of how measurements were made can be found in Section 2 (Methods).

1.3. Factors influencing success

Factors hypothesised to affect restoration outcomes in the direct seeding literature fall under four broad categories: site attributes (e.g. land-use history, weed load); site conditions (soil attributes, rainfall, temperature, topography); site preparation (quantity and choice of herbicide, row width, sowing season, soil preparation (e.g. scalping)); and seed mix (species sown, seed viability, seed treatment). Possibly because many of the direct seeding studies only assessed outcomes during the establishment phase, factors deemed important in the restoration literature such as landscape context (e.g. proximity to native vegetation) and ongoing management after a revegetation effort were not considered.

We collected data for each site on attributes representing each of the four categories explored in the literature (Section 2.4, Table 2). We were limited by the data available, with very little detailed information about the site condition and site preparation prior to direct seeding. However, we also included landscape context (the amount of native woody cover within 150 m radius) to evaluate the potential dispersal effects from outside the site, as well as collecting data on the occurrence and intensity of post-seeding grazing which may negatively influence restoration outcomes (Spooner, Lunt, and Robinson 2002b). Details of data collection and justification of attributes included in analysis are in Section 2.4.

2. Methods

2.1. Study area

The study area includes the Productive Plains and Upward Slopes socio-ecological systems of the Goulburn Broken Catchment Management Area, which encompass areas of the Riverina Plains, Central Uplands and Northern Inland Slopes Victorian biogeographic regions. The area has experienced extensive loss of native vegetation through large-scale clearing and has contributed to the conditions that have led to 13% of the catchment's native plant species and 22% of its native faunal species being listed as threatened (Miles et al. 2010; DSE 2007).

2.2. *Vegetation surveys*

2.2.1. *Site selection*

Fifty sites across the study area were selected for surveys (Fig. 1) representing a subset of 190 directly seeded sites sown in the Productive Plains and Upward Slopes between 1999 and 2009 (before management changes were initiated). All 190 sites had records of species sown, were sown with the aim of biodiverse revegetation (i.e. not shelterbelts) and were fenced prior to seeding to exclude livestock. Weed control was often undertaken before seeding, either by herbicide application within the furrows or across the whole site. On-going herbicide weed control after seeding was rare. A standard density of seeds was sown across sites: 500g per ha (2.5 km linear furrows) with the aim of achieving a density of 1 plant per metre. Species composition of seed mixes was intended to match the pre-clearance ecological vegetation class (EVC) of the area. However, seed mixes were developed iteratively through trial-and-error and thus differed through time as well as among sites. Earlier mixes generally consisted of fewer species and contained more *Eucalyptus* species than later sowings. *Acacia* and *Eucalyptus* species were the most common for all mixes. All seeds were treated prior to sowing, either via mechanical alteration or for *Acacia* species, via inoculation with symbiotic bacteria (*Bradyrhizobium*) or addition of smoke water.

Maps of candidate sites were developed in QGIS (Quantum GIS Development Team, 2014) using existing information from both the Catchment Activity Management System (CAMS) geodatabase and expert knowledge. Sites were then checked in the field for suitability and satellite information was collected to improve maps *post hoc*.

To best infer relationships between predictor variables (e.g. management, environmental, and ecological factors) and revegetation outcomes, we chose sites from our set of candidates that were >1.5 km apart and were similar in terms of elevation, topography (slopes >20° were excluded), system type (terrestrial rather than riparian), and size (between 1.5–13 ha, mean 3 ha). Of the 190 sites, we also excluded those found in the field to be: mixed with hand planting; sown into remnant vegetation; on properties where the landholder was un-contactable; and arrayed randomly (thus prohibiting standard survey methods). We also excluded 26 sites that had failed (Table A.1).

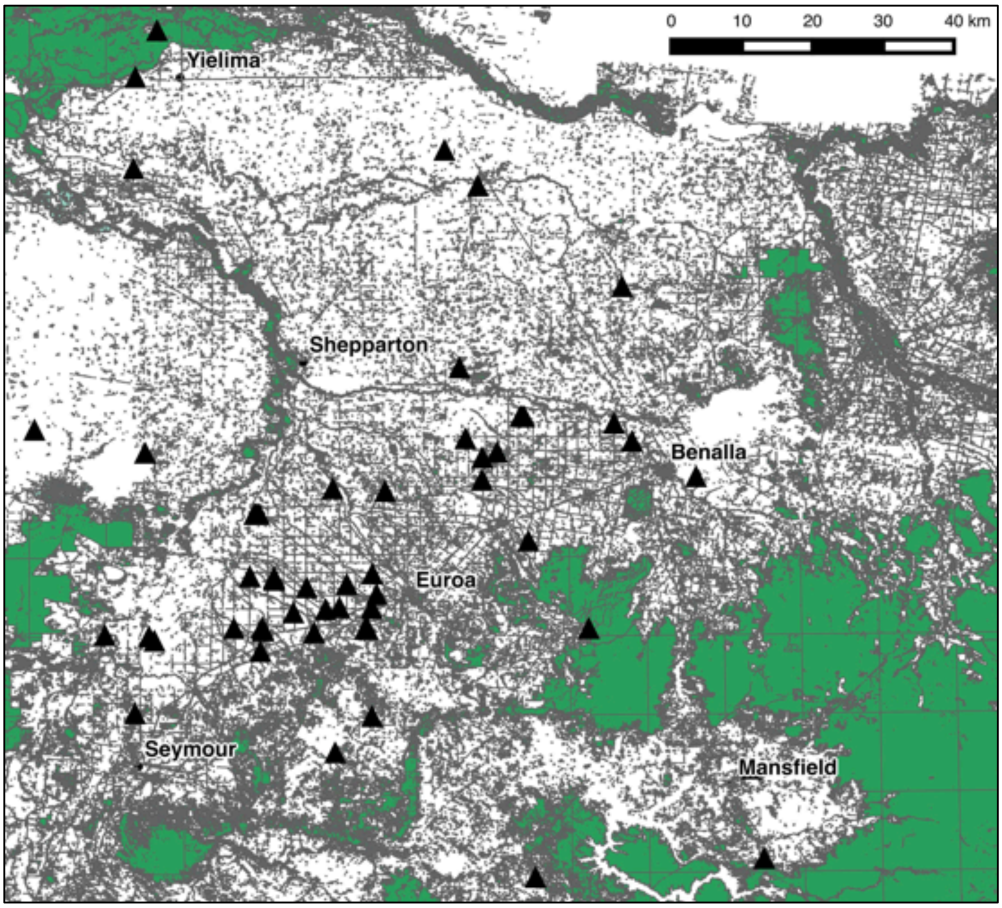


Figure 1 Map of the study area showing survey sites (black triangles) and tree cover (green indicates dense cover and grey more scattered cover).

2.2.2. Monitoring protocol

Field surveys were conducted in January–April 2015 to measure vegetation attributes at each of the study's 50 survey sites. Field officers who had been involved in the direct seeding chain (e.g. seed contractors, seed bank staff, local botanists) were employed to conduct surveys because of their expertise in identification of locally indigenous plants and understanding of the revegetation method.

Vegetation data needed to be collected using a sampling method that would be repeatable across a variety of site arrays because sites differed in their dimensions and configuration. We standardised survey effort for a 1 ha area and surveyed a 1 ha plot for every 3 ha of site area (with maximum of 2 plots at any given site). For each 1 ha plot we established 250 m of transects, stratifying along the furrow lines across any obvious environmental gradient, and excluding the 2 most external furrows adjacent to fence lines (Fig. 2). This survey effort represented 10% of the total furrows sown for a 1 ha area (2.5 km). Transects were used to measure variables describing structure and composition of sown plants. Each stem, its species, and life stage were recorded along transects. Rules for determining life stages for each species can be found in Table A.2. In addition, structural measurements were made for ≤5 individuals of each species and life stage combination found in the furrows (Table 1).

We measured the number of regenerating woody stems within quadrats. Quadrats were positioned alongside transects, and were the length of each transect and 1 m wide starting from the furrow's edge into the inter-row space (250 m^2 per ha).

We measured ground cover using repeated point quadrats along line transects located perpendicular to furrows and totaling 400 points (~200 m) for each 1 ha (e.g. Bullock (1996) and Tolsma and Newell (2003)). At each 50 cm point along the transect, a <0.5 cm diameter rod was placed vertically and any ground surface type (bare ground, rock, crusts, log(s), litter, native scat, pest scat) and any plant part attached to a living plant that touched the rod (up to 1 m height) was scored. Plants were classified as: exotic grass-like annual, exotic grass-like perennial, exotic broadleaf, native grass-like, and native broadleaf. GPS coordinate data for all transects was recorded and stored in a GIS layer allowing for future monitoring to be conducted and measures to be compared through time.

Table 1 Structural measurements made for each species and life stage combination of plants found in the furrows. Only trees and shrubs are shown as few ground cover species were sown or survived. Classifications of species life stages are described in Table A.2.

Life form	Life stage	Measurements
Tree and shrub	Seedling	Height
Tree and shrub	Sapling	Height
Shrub	Juvenile	Height and width at widest point
Shrub	Adult	Height and width at widest point
Tree	Juvenile (<4m height)	Height and DBH
Tree	Juvenile (>4m height)	Height, DBH and crown width
Tree	Adult	Height, DBH and crown width

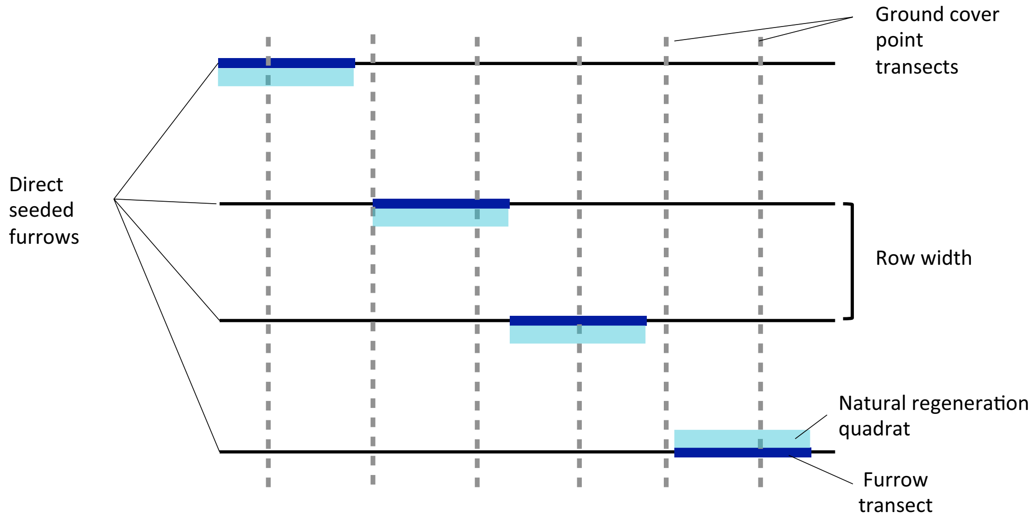


Figure 2 Diagram of the sampling design employed at each survey site.

2.3. Response variables

We chose four response groups to gauge the relative condition of sites surveyed: stem count, species survival, density of naturally regenerating stems, and proportion weed cover.

Stem count was the number of stems counted in the 250 linear meters surveyed at each site. As survey area was equal across all sites this is essentially a measure of stem density. Species survival is the number of species sown that survived in the furrows (were present at survey). The greater the species survival, the better the biodiversity outcome.

Natural regeneration was the number of stems/m^2 detected in between seeded furrows. Natural regeneration is an indicator of a site's ability to be self-sustaining, both in terms of the maturity of overstorey species and in soil condition.

Weed cover was the percent of exotic annual ground cover at a site. It was an indicator of site condition: on the restoration trajectory, sites with higher weed cover are closer to areas of recent agricultural use (Hallett et al. 2014).

2.4. Predictor variables

As established in the direct seeding literature, there are a number of potential explanatory variables that can affect the nominated response variables (stem count, species survival, natural regeneration and weed cover). We developed 13 variables (Table 2) based on the literature and consultation with GBCMA about predictors they wanted included. These variables represent broad hypothesised drivers of direct seeding outcomes. Hypotheses were configured differently depending on the response group being modelled in accordance with ecological theory (see Table 3 for details).

(1) Environmental conditions: rainfall, topographic wetness index, and percent of clay in the soil.

Rainfall is the amount of rain (mm) that fell in the 12 months following direct seeding. This was calculated from modelled rainfall data supplied by the Australian Water Availability Project (AWAP; Raupach, et al. 2009; Raupach, et al. 2011). Water availability in the first year post-seeding can affect a species' ability to germinate (Carr et al. 2009; Barron, Dalton, and Miller 1998).

Topographic wetness index (**TWI**) is an indicator of the potential retention of water at a site based on site topography (i.e. sites located at the base of a catchment or lower down a slope will have higher TWI than those upslope). Each site's TWI was derived from a raster TWI surface created from a 20 m DEM (Moore et al. 1993; Vesk and Dorrough 2006). Water availability and waterlogging can affect the ability of directly seeded seedlings to thrive (Azam et al. 2014)

Soil clay is the percent of clay in the top 5 cm layer of soil for each 1 ha plot and was derived from ASRIS 0–30cm Clay Content GIS data layer (CSIRO Land & Water, 2011). For sites where two values existed, if applicable the predominant value (covering >75% of site) was used, otherwise the mean of the two values was used. High soil clay has been demonstrated to negatively effect emergence of seedlings potentially because of reduced root penetration which may be exacerbated by compaction common in agricultural areas (Hallett et al. 2014).

(2) Site characteristics*:* includes variables that describe the starting state of the survey site; previous land use, weed cover, extent of surrounding tree cover, age.

Land use is a binary variable indicating whether the land was previously used for cropping or not within 5–10 years of direct seeding. Those sites that were not used for raising crops were either exotic or native pastures. All sites had a history of livestock grazing. Legacies of land use can affect the soil condition and weed load and thus impact outcomes of direct seeding and natural regeneration (Dorrough and Scroggie 2008).

Weed cover is the proportion of the survey area that consisted of exotic plants (see section 1.2). High weed cover is thought to impede outcomes of direct seeding through competitive effects and/or by creating a physical barrier between seeds and the soil (Standish et al. 2007).

Tree cover is the proportion of land within a 150 m radius of each site's boundaries that contained native wooded vegetation. This was based on data from the NV2010–Extent version 2 GIS layer (DELWP, 2014). Dispersal of seeds from neighbouring remnant vegetation can assist with restoration outcomes (Vesk and Dorrough 2006).

Age is the cumulative number of months since direct seeding occurred until January 2015.

(3) Management: row width, weed control, post-sowing grazing.

Row width is the mean distance between directly seeded furrows at a given site. Row width is thought to influence natural regeneration (Schneemann and McElhinny 2012) and weed cover (Jonson 2010a).

Weed control is a categorical variable describing the scale of weed control conducted prior to direct seeding. There are three levels: no weed control, weed control in furrows only, and whole site weed control. Data about the herbicide applied or the number of applications was not available. Effective weed control in the first year of seeding has been shown to improve seed germination (Knight, Beale, and Dalton 1997; Barron, Dalton, and Miller 1998).

Grazing is a categorical variable describing the intensity of livestock grazing that occurred after direct seeding. Although fences are erected to exclude livestock, some landholders have allowed various levels of grazing to occur on their directly seeded sites. Grazing levels include: High (regular, ongoing grazing), Medium (annual or bi-annual crash grazing), Low (grazing on rare occasion), None. Grazing can negatively impact both directly seeded plant growth and natural regeneration (Spooner, Lunt, and Robinson 2002a; Prober and Smith 2009).

(4) Seed input: species mix (for stem count); stem density, age and tree cover (for natural regeneration).

Species mix is the number of species sown at a survey site. This predictor was only included in the models with stem count as the response variable. Seed mix was determined from records kept by the Goulburn Broken Indigenous Seedbank, from personal communication with seed contractors who carried out the direct seeding, from interviews with landholders (whether they added seed), and from records held by the regional DELWP offices.

Stem density is the number of stems per metre in the furrows. This was calculated by dividing stem counts by 250 m (section 2.2.2). Stem density was only modelled as an explanatory variable in natural regeneration models. For these models, hypothesised seed input did not include the species mix but rather consisted of variables describing seed deposited from mature furrow plants (Stem density + Age) and seed that may have dispersed to the site from surrounding remnant vegetation (Tree cover, Table 3).

(5) Germination barriers: grazing and weed cover

'Germination barriers' includes explanatory variables that are suspected of inhibiting natural regeneration in SE Australian landscapes (Weinberg et al. 2011; Spooner, Lunt, and Robinson 2002b; Standish et al. 2007).

(6) Starting state: was described by a single predictor variable – fertiliser.

Fertiliser is a categorical variable indicating the level of fertiliser in the soil prior to direct seeding. Fertiliser levels are: High (high levels applied within a year of seeding), Low (low levels applied within a year of seeding or high levels but not applied within 5–10 years of seeding), and None (fertiliser used more than 10 years prior to seeding). This predictor was only included in weed cover models because fertiliser levels may influence weed load (Dorrough and Scroggie 2008; Fischer et al.

2009). For models of the other three response variables, weed cover was included as an explanatory variable but not fertiliser because of potential correlation between the two (highly correlated predictor variables should not co-occur in statistical models (Zuur and Ieno 2010)).

Table 2 Summary of predictor variables used in statistical models. Details of each and their derivation can be found in section 2.4.

Variable	Description
Rainfall	Total rainfall (mm) for the first 12 months after direct seeding
TWI	Topographic wetness index (an indicator of relative waterlogging)
Soil clay	Percent of clay in top 5 cm of soil
Land use	Binary variable – was the site used for crops prior to seeding (yes or no).
Weed cover	Proportion of surveyed area covered by exotic plants.
Tree cover	Proportion of area containing native woody vegetation within a 150 m radius of each site's boundaries.
Age	Time (no. of months) between direct seeding activity and January 2015.
Row width	Mean distance between directly seeded furrows
Weed control	Scale of application of weed control (herbicide). Categorical predictor with levels: None, Furrow, Whole site
Grazing	Categorical variable describing the intensity of livestock grazing after direct seeding: High, medium, Low, None.
Seed mix	Number of species sown
Stem density	Number of stems per metre in furrows.
Fertiliser	Categorical variable of fertiliser load: High, Low, None.

2.5. Data Analysis

Scatterplots of predictors against response variables were made to determine the nature of relationships. All relationships were found to be approximately linear. Continuous explanatory variables were standardised (centered and divided by two standard deviations, (Gelman 2008)) prior to inclusion in models and categorical variables were left unchanged, so that predictor effects could be compared on the same scale (Grueber et al. 2011).

Table 3 Model hypotheses and representative variables.

Response variable	Model hypotheses	Predictor variables	Key differences in model sets among response variables
Stem count	Environmental conditions	Rainfall + TWI + soil clay	The interaction term weed cover × age was **not** found to be a significant influence on stem count, however, its inclusion noticeably improved the global model's AIC. The interaction describes the condition of weed cover tending to decrease with increasing age, likely because of increased stem density (which increases levels of shade and litter).
	Site characteristics	Land use + weed cover + tree cover + age + weed cover × age	
	Management	Row width + weed control + grazing	
	Seed input	Species mix	
Species survival	Environmental conditions	Rainfall + TWI + soil clay	Row width was not included under the 'management' category because it was unlikely to influence species survival. Age was not considered as an explanatory variable because it was unlikely to influence species presence at these post-establishment sites (6–15 years old).
	Site characteristics	Land use + weed cover + tree cover	
	Management	Weed control + grazing	
Natural regeneration	Environmental conditions	Rainfall + TWI + soil clay	Hypotheses differed for natural regeneration (NR) compared to furrow response groups because it is an unmanaged process and thus influenced by a different suite of attributes. Seed input for NR consisted of input by mature plants (stem density and site age) and abundance of seed trees within 150 m radius. 'Germination barriers' includes explanatory variables that are suspected of inhibiting natural regeneration in SE Australian landscapes.
	Initial management	Row width + weed control	
	Germination barriers	Grazing + weed cover	
	Seed input	Tree cover + stem density + age	
Weed cover	Environmental conditions	Rainfall + TWI + soil clay	Weed cover involves a different suite of processes compared to managed and unmanaged native plant growth at direct seeded sites. GBCMA were particularly interested in how management actions influenced weed control and so these were established as pre- and post-management categories. Fertiliser load was included because of its potential to encourage favourable conditions for exotic weedy plants over native covers.
	Initial management	Row width + weed control	
	Post management	Age + grazing	
	Starting state	Fertiliser	

2.4.2. Model development and selection

To identify the determinants of our response variables (stem count, species survival, natural regeneration and weed cover) we constructed multiple models that represented our competing hypotheses (environmental conditions, site characteristics, management, seed input, germination barriers and starting state) to assess their influence on each response type. Each single-hypothesis model contained all variables associated with the relevant hypothesis (Table 3). We also built models that represented all possible combinations of our hypotheses plus a null model (total set of 16 models for stem count, natural regeneration and weed cover, and 8 models for species survival).

We modelled relationships using generalised linear models (GLMs). Stem counts were modelled with a Poisson distribution since ecological count data can only be zero or positive and has a tendency towards higher density of low count records. Species survival and weed cover were modelled with a binomial distribution as these both represented proportion data, taking the form of a two-vector response: the total number of presences (species germinated, points weeds present), and the total number of absences (species not recorded, points non-weedy ground cover present) for each site (Warton and Hui 2011). The density of naturally regenerating stems was log-transformed and modelled with a Gaussian distribution as this method resulted in residuals of global models (including all predictors) being the most normally distributed (Zuur and Ieno 2010). Prior to log transformation, the value of the smallest non-zero observation in the dataset (0.004 stems/m^2) was added to each y-value to correct for zero values, following Warton (2011). We tested the global models for each response group for overdispersion, autocorrelation (using Moran's *I* statistic), and for collinearity between predictors. Autocorrelation was not detected for any response group, and all collinearity factors were <2. Overdispersion was found for models of stem count, species survival and weed cover, so stem count was modeled with a quasi-Poisson error distribution while species survival and weed cover were modelled with a quasi-binomial distribution.

For each response group, an information theoretic approach was used to compare the relative support for each of our hypotheses (across all combinations of models) and to assist with model selection (Burnham and Anderson 2002; Grueber et al. 2011)). For natural regeneration, model support was determined by calculating Akaike's information criterion corrected for small sample size, AIC_c (Burnham, Anderson, and Huyvaert 2011) and $QAIC_c$ (quasi-AIC_c) for stem counts, species survival and weed cover. $QAIC_c$ was calculated according to methods outlined in Bolker (2009) for the 'MuMIn' package in the R software programme. The difference between each model's (Q)AIC_c value and that of the best fitting model (Δ_i) were also calculated. Only models with (Q)AIC_c values ≤2 are considered to have substantial support (Burnham and Anderson 2002).

Akaike weights (w_i) were calculated to determine the relative likelihood of each model in the candidate set being the most parsimonious: a combination of the fit of the data and number of parameters included (with penalties for increasing complexity) (Burnham & Anderson 2002). Akaike weights (w_i) were then used to rank the groups of 16 models (8 models for species survival). In the absence of a standout model (w_i ≥0.90, Burnham & Anderson 2002), we calculated model-averaged parameter estimates and standard errors for all predictors contained in models ranked higher than the null models to account for model-selection uncertainty (Burnham and Anderson 2002; Richards, Stephens, and Whittingham 2011). These are derived by calculating the weighted average of regression coefficients of each predictor over all subset models, with weights corresponding to the w_i for models that include the predictor of interest (Burnham and Anderson 2002, 150-155).

To test the fit of the 'best' model for each response variable, we calculated the percentage of null deviance explained (d^2) (Zuur et al., 2011). All statistical analyses were conducted in the R software package (R Core Team 2015). GLMs were constructed using the 'stats' package (R Core Team 2015). Choice of error distribution for overdispersed models (e.g. between beta-binomial and quasi-binomial, or negative binomial and quasi-Poisson models) was assessed via half normal plots of global model residuals in the 'hnp' package (de Andrade Moral et al., 2011) according to Hinde & Demetrio (2007). Model selection, model averaging and model fit were carried out using the 'MuMIn' package (Barton 2013). Autocorrelation analyses were made using the 'spdep' package (Bivand 2014) and multicollinearity variance inflation factors (and generalised VIF's for categorical predictors) were assessed in the 'car' package (Fox and Weisberg 2013).

2.5.1. Principal components analysis of ground covers

We conducted principal component analysis (PCA) to quantify the relative abundance of ground covers across sites, in the R 'stats' package, using function 'prcomp' (R Core Team 2013). Variables consisted of the percent cover of each ground cover type: bare ground, rock, crusts, log(s), litter, native scat, pest scat, weeds (sum of exotic grass-like annual, exotic grass-like perennial, and exotic broadleaf plants) and native plants (sum of native grass-like and native broadleaf plants). Pearson correlation analyses were subsequently run between all variables to assist with PCA interpretation.

3. Results

3.1. Monitoring results

A total of 47 species were recorded in the furrows across the survey sites. Species richness ranged from 4–14 species, however, on average each site contained ~1 species in the furrows which was not recorded in the species list. These species may have germinated via dispersal of seed into the site by wind or animal vectors, from seed stored in the seed bank, or added by landholders but not recalled (section 2.4, *seed mix*). It may also be possible that some records were not accurate or were only recorded in part. Median density of live stems in furrows was 0.56 stems/m (1390 stems/ha) and ranged between 0.06–3.40 stems/m (150–8460 stems/ha). Median mortality was 0.02 stems/m (range 0–0.97 stems/m).

Species survival (number of species sown that were present in the survey) differed among sites, with survival of sown species ranging from 18–100%. Survival also differed among species, with *Acacias* having a significantly higher mean rate of survival (0.81, s.e. 0.03) than *Eucalyptus* species (0.47, s.e. 0.06; GLM $p< 0.01$). Species groups other than *Acacia* and *Eucalyptus* could not be statistically compared because they were not sown in enough sites. Survival rates for all species are shown in Appendix table A.3.

Natural regeneration varied widely among sites with a median of 0.028 stems/m^2 (280 stems/ha), range of 0–0.956 stems/m^2 and no regeneration recorded at 18% of sites. Where regeneration was present, median richness of regenerating plants was 2 species (range 1–7 species). A list of all 26 species recorded between furrows in survey sites can be found in appendix (A.1). Very little natural regeneration was found for *Acacia* species relative to the density of mature (seed producing) stems (with the exception of *A. dealbata,* Fig. 3). In contrast, regenerating *Eucalyptus* species had high recruitment relative to the density of mature stems (Fig. 3).

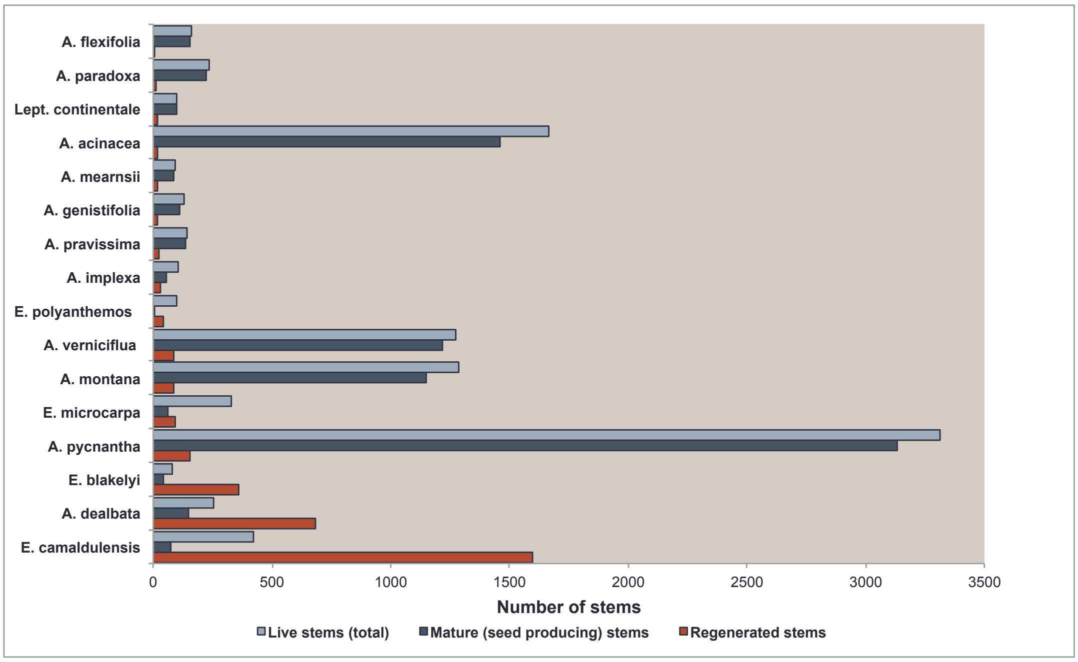


Figure 3 Comparison of the number of total stems, number of mature stems and number of regenerating stems for the 16 species regenerating most commonly in the survey sites. Number of stems is the accumulated number found across all 50 sites.

Weed abundance was generally high, with a median percent cover of 55% (range 7.5–89%). By contrast, native ground covers were often found in low abundance with a median cover of just 6.2% (range 0–48%; absent from 4 sites).

The first two principal components from the ground cover analysis together explained 92% of the variation in ground covers (Table A.4). The first component (PC1) explained 67.5% of the variance and was associated with litter cover and weed cover, with litter cover described by a high positive value (0.6) and weed cover by high negative value (–0.78) indicating a negative relationship between the two. Correlation analyses confirmed a strong negative relationship between weed cover and litter (r = –0.68). The second component (PC2) explained 24% of the variation in ground covers, with commonly present but sparse ground covers (native cover, crusts) taking positive values and commonly present and abundant covers (weed cover, litter) taking negative values. Correlation analyses revealed negative relationships between weed cover and bare ground (r = –0.708) and weed cover and crusts (r = –0.48) but only weak correlations between pairs of remaining variables.

3.2. *Factors influencing furrow stem count*

Table 4 provides the results of multiple regression models explaining the influence of environmental, ecological and management factors on stem count, species survival, natural regeneration and weed cover. No clear standout model was identified for any of the four response groups (w_i ≥0.90, Burnham & Anderson 2002). For stem count, only seed input was found to have strong support (Δ_i <2, Table 4). Summing Akaike weights for all models in the model subset that contained a given model hypothesis gave us a metric of the comparative influence of each hypothesis on the number of stems (Fig. 4). It is clear that seed input is the most influential factor on the number of stems in furrows although its Akaike weight (w_i = 0.61) is still well below the value required to be considered the standout model from the model set (Burnham and Anderson 2002).

Table 4 Model-selection results stem counts, species survival, natural regeneration and weed cover. Included are log-likelihood values (log(L)), degrees of freedom (K), (Q)AICc values, (Q)AICc differences (Δ_i), Akaike weights (w_i), and the percentage of variance explained (d^2). Only models with (Q)AICc differences ≤2.8 are shown.

Model	K	log(L)	(Q)AICc	Δ_i	w_i	d^2
Stem counts						
Seed input	2	-3070.7	60.6	0.0	0.61	0.23
Seed input+ Environmental conditions	5	-2790.9	63.4	2.8	0.15	0.18
Species survival						
Environmental conditions	4	-87.9	109.8	0.0	0.79	0.20
Natural regeneration (stem density)						
Environmental conditions + Seed input	8	-76.2	172.4	0.0	0.40	0.33
Seed input	5	-81.0	173.6	1.2	0.22	0.16
Weed cover						
Starting state	3	-1857.1	63.5	0.0	0.34	0.13
Starting state + Post management	7	-1485.4	63.6	0.1	0.33	0.32
Starting state + Environmental conditions	6	-1665.2	65.9	2.4	0.10	0.23

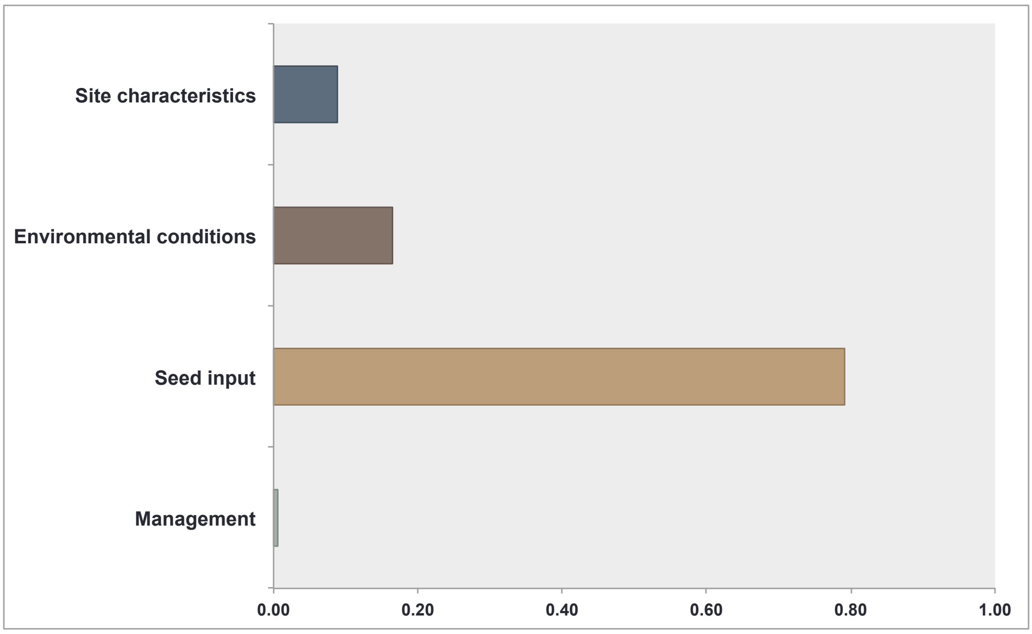


Figure 4 The relative magnitude of importance of candidate models (i.e., hypotheses) on number of existing stems, derived from summing Akaike weights of all model subsets in which the candidate model occurred ($\sum w_i$).

Model-averaging was conducted for all response groups to account for the uncertainty attached to the identifying a 'best' model in the absence of a clear standout model (Burnham and Anderson 2002). Model-averaged estimates of parameter coefficients were calculated for each predictor variable from the subset of models that were ranked higher than the null model (Table 4). Including models ranked lower than the null may lead to spurious results (Grueber et al. 2011). Unconditional standard errors were calculated for the relevant parameter estimates and used to generate confidence intervals (Figs. 5, 8, 11, 14). Given that predictor variables were centered and standardised, those whose confidence intervals do not overlap zero are considered to have an important influence on the response group. Positive coefficients indicate a positive relationship between predictor and response, while negative coefficients indicate the opposite.

For stem count, parameter estimates confirmed the importance of the number of species sown (Fig. 5). They also highlighted a trend between the first 12 months of rainfall and the number of stems in furrows, although rainfall was not considered an important influence because its confidence intervals slightly overlapped zero.

Predictions made from the model-averaged coefficients allow us to better visualise these relationships (Fig. 6). The number of species sown has a positive linear relationship on the number of stems recorded in furrows, although this relationship becomes less certain as more than 15 species are added to the seed mix (as evidenced from the wider confidence intervals about these values (Fig. 6a). Rainfall demonstrates a similar positive linear relationship with stem count until it reaches values around 500 mm, where high relationship uncertainty is indicated by the wide confidence intervals. This uncertainty may be caused by the fact that only three sites had rainfall between 500–800 mm. Regardless, this relationship should be treated as a trend only and subject to cautious interpretation.

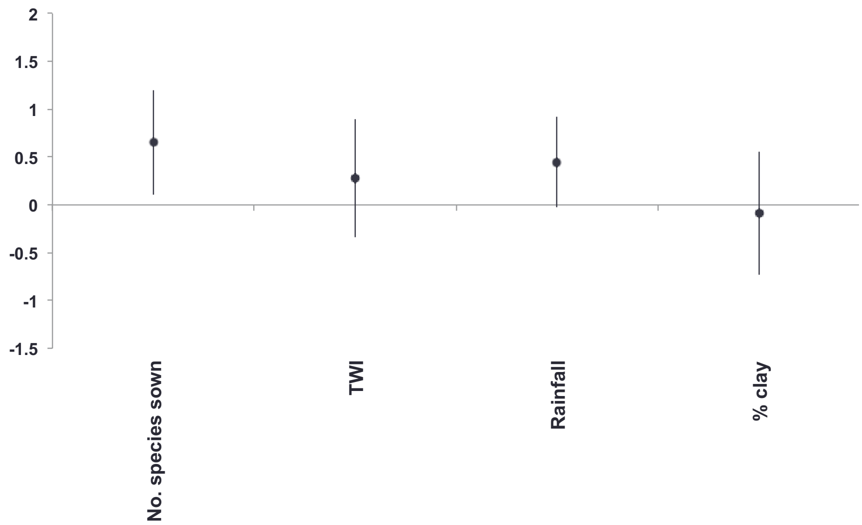
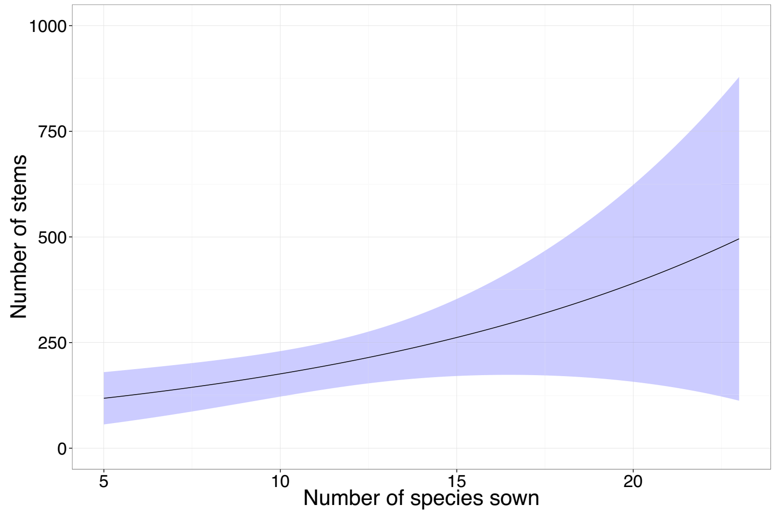


Figure 5 Model-averaged parameter coefficients (and associated 95% confidence intervals) for the number of stems found in furrows. Note that only predictors of model(s) ranked higher than the null model are shown.

(a)



(b)

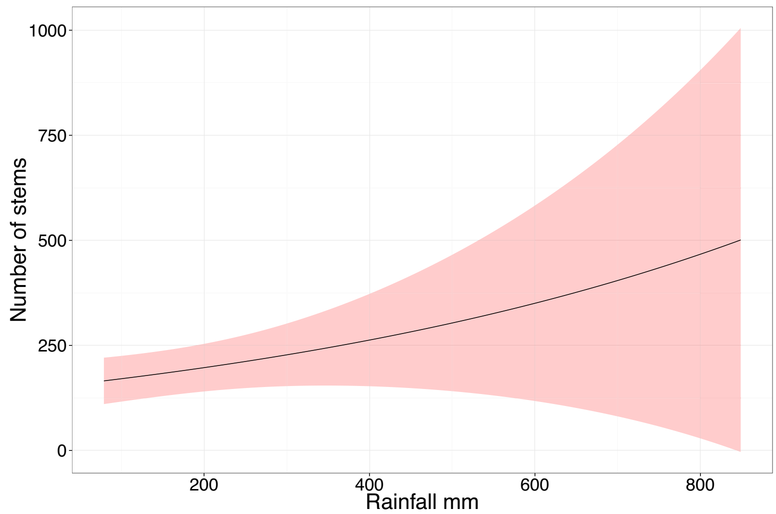


Figure 6 Relationships between stem count and (a) number of species sown, (b) first 12 months of rainfall after direct seeding. Unbroken lines represent the predictions from model-averaged models and coloured shading denotes the 95% confidence intervals. Note that rainfall only demonstrated a relationship trend, likely because of the high uncertainty evident for rainfall above 500 mm.

3.3. Factors influencing species survival

Only the model linking species survival to environmental conditions had strong support (Δ_i<2, Table 4). Summed Akaike weights supported this finding, with environmental conditions far outweighing site characteristics and management hypotheses (Fig. 7).

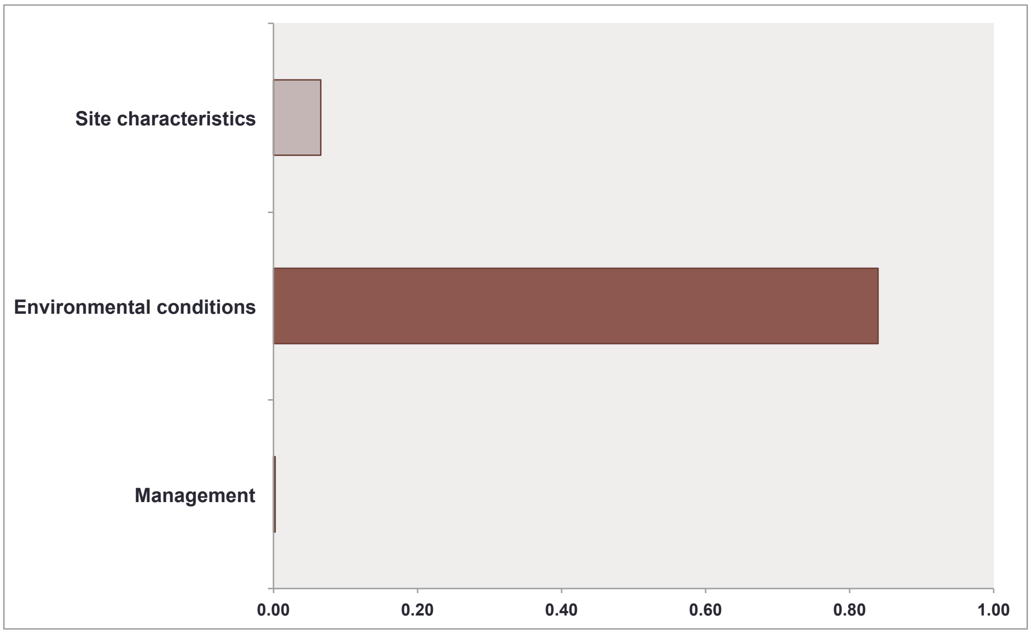


Figure 7 The relative magnitude of importance of model categories (hypotheses) on species survival, derived from summing Akaike weights of all model subsets in which the model category occurred ($\sum w_i$).

Model-averaging revealed that species survival was strongly influenced by the topographic wetness index and that rainfall and percent clay in soils demonstrated comparatively weak effects (Fig. 8). Predictions made from the model-averaged coefficients show that the higher the topographic wetness index (i.e. increased pooling of water during rain events) the greater the probability of survival of sown plant species (Fig 9).

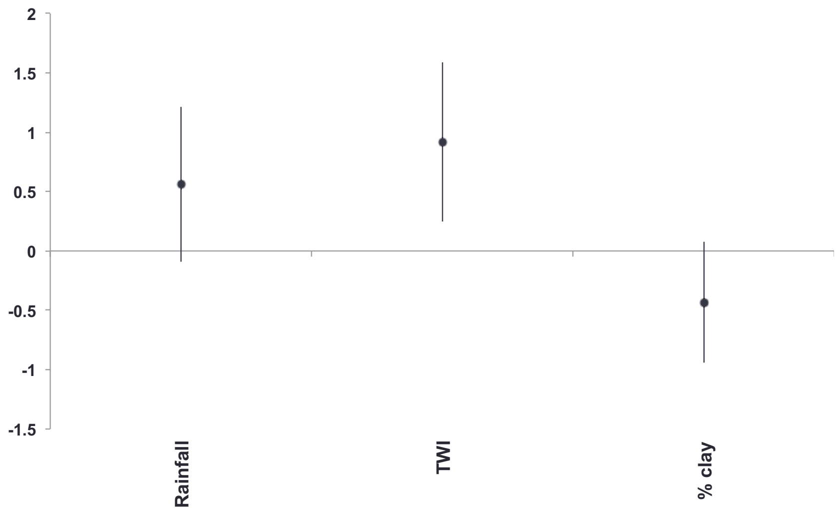


Figure 8 Model-averaged parameter coefficients (and associated 95% confidence intervals) for the survival of sown species. Note that only predictors of model(s) ranked higher than the null model are shown.

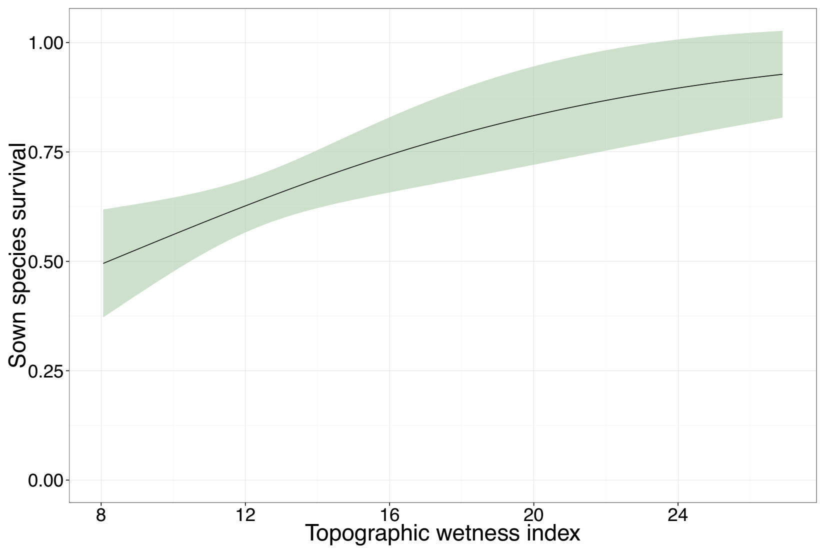


Figure 9 Relationship between species survival and topographic wetness index. Unbroken lines represent the predictions from model-averaged models and coloured shading denotes the 95% confidence intervals.

3.4. Factors influencing natural regeneration

For natural regeneration, two models had strong support (Δ_i<2, Table 4). Summed Akaike weights show that the most influential factors of natural regeneration are seed input and environmental conditions, while hypothesised germination barriers and initial management actions appeared to have little bearing (Fig. 10).

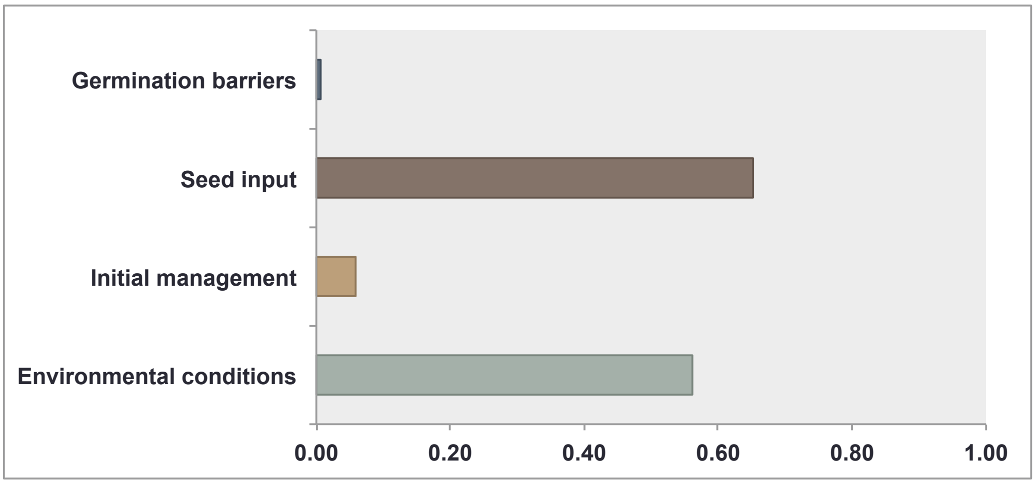


Figure 10 The relative magnitude of importance of model categories (hypotheses) on number of regenerating stems, derived from summing Akaike weights of all model subsets in which the model category occurred ($\sum w_i$).

Model-averaging revealed that the density of regenerating stems was strongly influenced by the extent of surrounding tree cover and rainfall (Fig. 11). Seed input from furrow plants (stem density + age) was not an important influence nor were the remaining environmental variables. Predictions made from the model-averaged coefficients demonstrate a strong positive, linear relationship between the extent of tree cover within a 150 m radius of sites and natural regeneration, suggesting that regeneration is driven by seed rain external to the sites (Fig. 12a). Rainfall also shows a positive, linear relationship with the density of regenerating stems, however, the very wide confidence intervals for values higher than 500 mm suggest that this relationship be interpreted with caution (Fig. 12b).

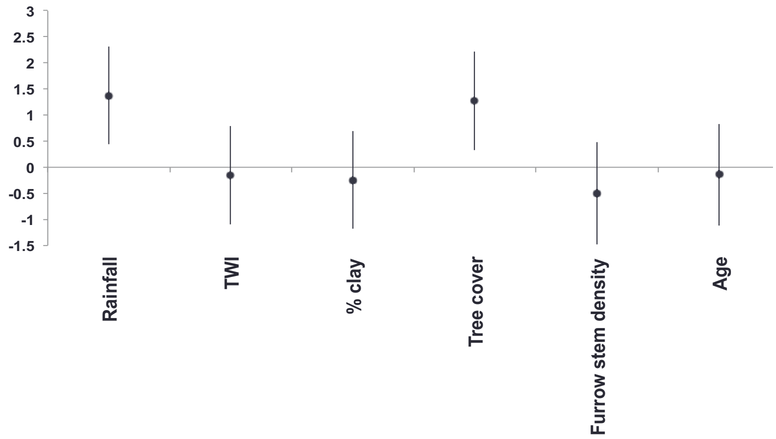
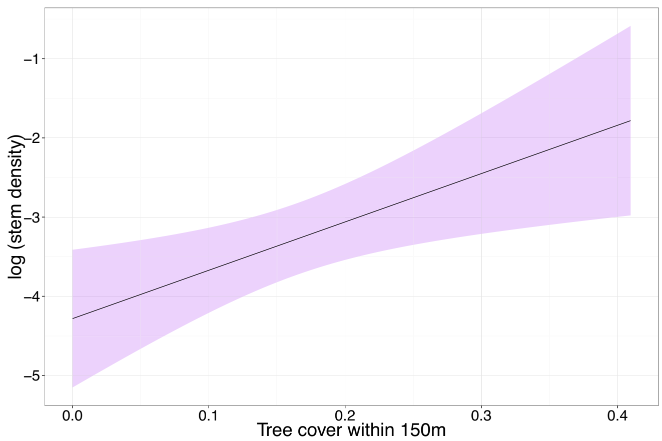


Figure 11 Model-averaged parameter coefficients (and associated 95% confidence intervals) for the number of regenerating stems. Note that only predictors of model(s) ranked higher than the null model are shown.

(a)



(b)

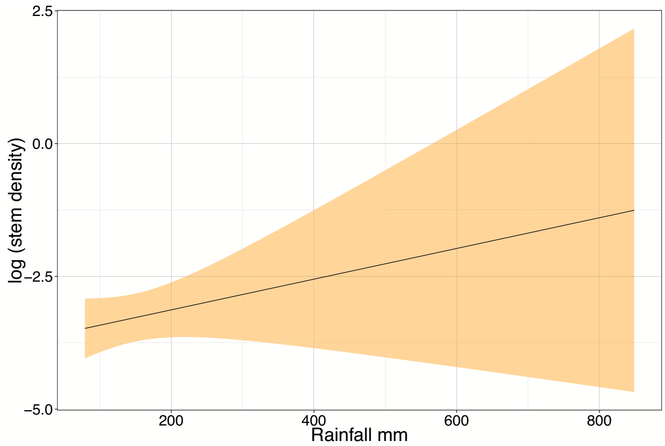


Figure 12 Relationships between natural regeneration and (a) extent of tree cover in surrounding 150 m; (b) first 12 months of rainfall after direct seeding. Unbroken lines represent the predictions from model-averaged models and coloured shading denotes the 95% confidence intervals. Note that

the wide confidence intervals indicate that this relationship should be interpreted with caution.

3.5. Factors influencing weed cover

There were two models with strong support describing the extent of seed cover (Δ_i<2, Table 4). Summed Akaike weights indicate that the most influential factors of weed cover are land use (fertiliser) and post management, with environmental conditions and initial management actions demonstrating comparatively weak influence (Fig. 13).

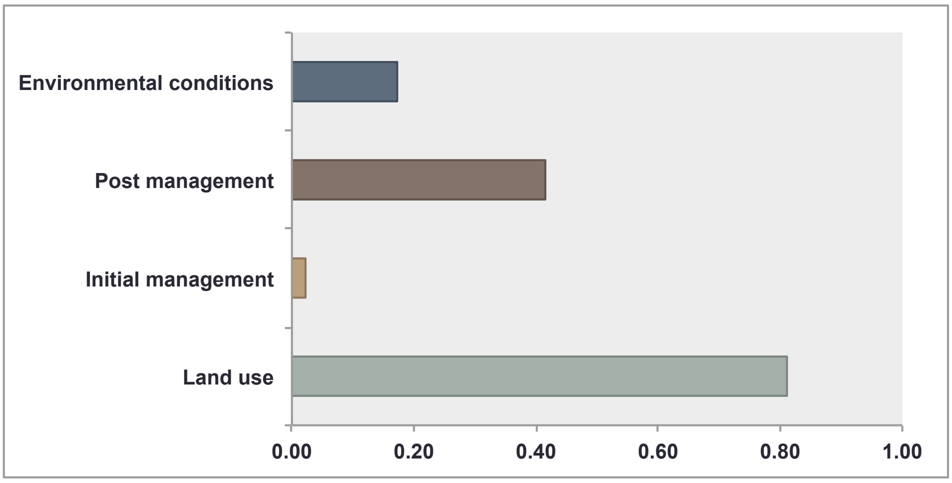


Figure 13 The relative magnitude of importance of model categories (hypotheses) on weed cover, derived from summing Akaike weights of all model subsets in which the model category occurred ($\sum w_i$).

Model-averaging revealed that only medium-level post-seeding grazing was an important influence on weed cover, while low grazing, high fertiliser use and age appeared to have a borderline influence (Fig. 14). Predictions made from the model-averaged coefficients demonstrate a negative linear relationship between weed cover and fertiliser concentrations: higher fertiliser use results in higher weed load (Fig. 15). At all levels of fertiliser use, medium-intensity post-seeding grazing results in the lowest cover of weeds, with low-intensity having a similar effect (Fig. 15). High-intensity post-seeding grazing leads to the highest levels of weed cover (Fig. 15). Age exhibits a negative linear trend, with a mean 10% decrease in weed cover between 5–10 years of age, although again, this relationship should be interpreted with caution because of the high variance indicated by wide confidence intervals around this estimate (Fig. 16).

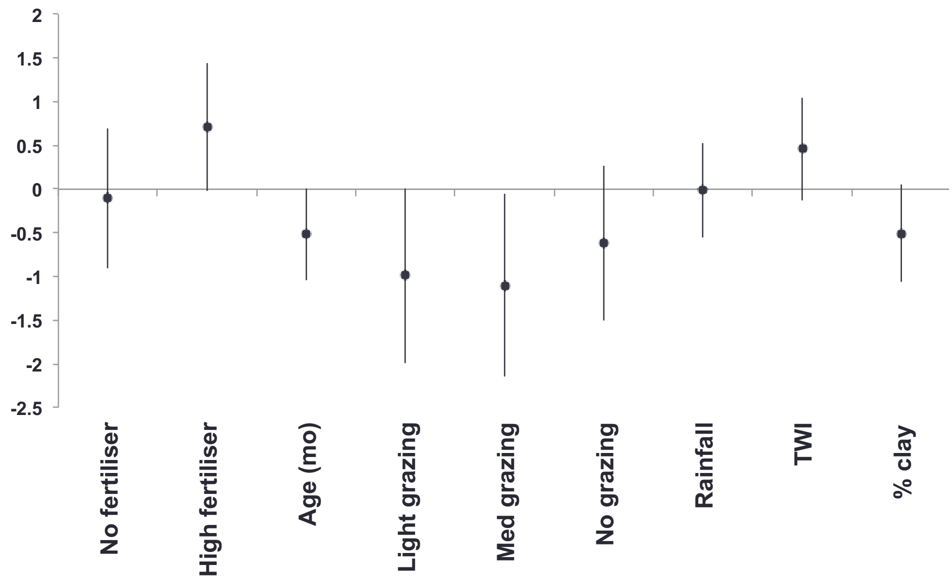


Figure 14 Model-averaged parameter coefficients (and associated 95% confidence intervals) for the number of regenerating stems. Only predictors of model(s) ranked higher than the null model are shown. Note that for each of the categorical variables Fertiliser and Grazing, one variable is missing from the figure because it was set as the reference level ('low fertiliser' and 'high grazing').

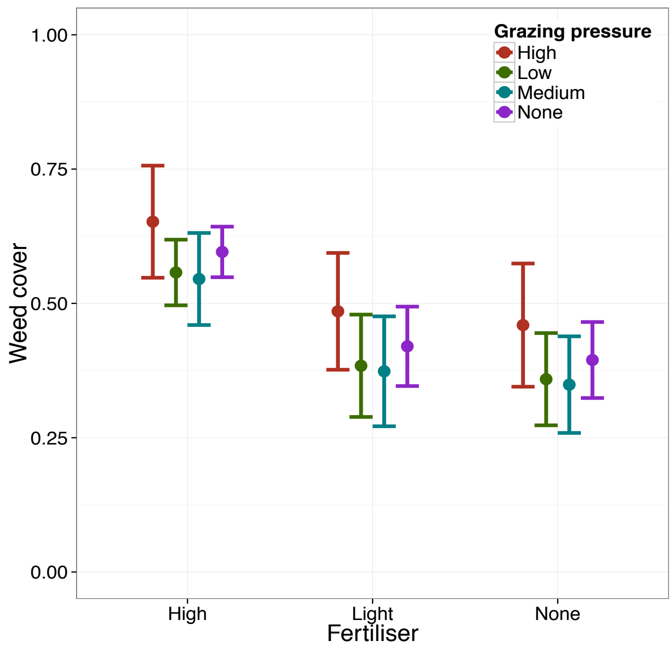


Figure 15 Relationships between weed cover and (i) fertiliser use (x-axis); (ii) grazing pressure (groups). Filled circles represent the predictions from model-averaged models and error bars display the 95% confidence intervals.

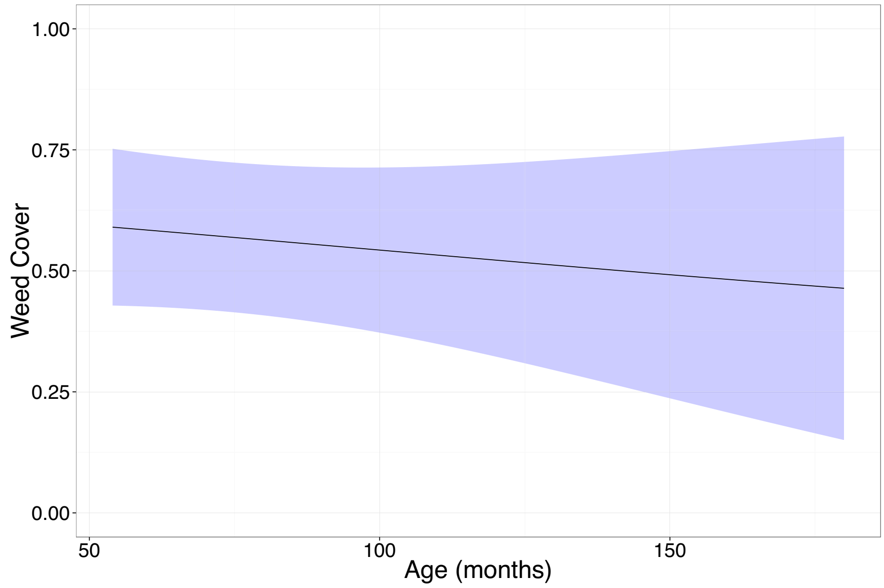


Figure 16 Relationships between weed cover and age. Unbroken lines represent the predictions from model-averaged models and coloured shading denotes the 95% confidence intervals. Note that the wide confidence intervals indicate that this relationship should be interpreted with caution.

4. Discussion and recommendations

The aim of this project was to design a monitoring protocol to assess the progress of direct seeding projects conducted in the Goulburn Broken CMA between 1999–2009, and to determine factors influencing the relative success of sites. We measured attributes that were able to indicate a site's relative trajectory along the restoration path and its capacity to be self-regenerating. We used a modelling approach to test hypotheses about the influence of environmental, ecological and management factors on our 'success' indicators (stem density and species survival in furrows, natural regeneration outside furrows, and abundance of weeds). We used an information theoretic approach to identify which factors had the greatest influence on each of the response groups. Our best models accounted for an intermediate amount of the variation in each response group among sites: 23% of the variance in stem density, 20% of the variance in species survival, 33% of the variance in density of regenerating stems and 32% of the variance in weed cover. There are clearly factors we did not measure (because of resource limitations) that are important influences, and they are considered below in our interpretation of results and recommendations for management and future research.

4.1. Stem density

The abundance of stems growing in furrows increased with an increase in the number of species sown, although the relationship is more uncertain as more than 15 species are included in the seed mix. This uncertainty may reflect the minimal contribution to stem density made by most species outside the genera *Acacia* and *Eucalyptus*. How species richness positively affects stem density is as yet unknown, but it has been similarly observed to lead to an increase in above- and below-ground biomass in a recent direct-seeding study (Perring et al. 2015). It is possible that greater species richness reduces intraspecific competition but this would need to be explored through experimental means. Indeed, it is likely that competition is an important unmeasured driver of stem density, especially given that site characteristics, conditions and management had little influence on stem abundance and that our models explained just 23% of variation in the abundance of stems among sites (Schneemann and McElhinny 2012).

Ecological filters exert a strong influence on stem abundance during establishment (Ede 2014; England et al. 2012). Because these sites are only being monitored for the first time after the establishment phase, it is likely that processes commonly found to affect germination, such as initial weed load (Knight et al. 1997), may have played an important but undetected role. Thus, while weed cover appears to have little effect on the number of stems growing in furrows more than 6 years after seeding, it does not mean that weed control is not an important consideration prior to- and during the first years post-seeding.

We considered higher stem counts to indicate greater success at a site because very low stem densities signify poor recruitment. However, this may not be the case where the purpose of direct seeding is to restore sites to their pre-European settlement EVC condition. Many of the sites in this study are far too highly stocked (median 1390 stems/ha) compared to the mean target density of ~350 stems/ha for a 10 year-old revegetated site (on reclaimed pasture) in the study region (Department of Sustainability and Environment 2006). Such high density can lead to high mortality rates, problems arising from increased competition (reduced species richness and poorly formed plants), and inferior habitat for indigenous fauna (Nardon, Smethurst, and Gibson 2005). Overstocking is quite common in directly seeded sites that have established successfully and may be resolved to some degree by self-thinning (Schneemann and McElhinny 2012; England et al. 2012). However, it is likely that management interventions will need to occur to assist this process. In addition, reducing the amount of seed of those species that consistently recruit well (Table A.3) would assist with overstocking, and experimenting with the rate and consistency of seed flow into the furrows could prove worthwhile (Jonson 2010b).

4.2. Species survival

Of the species sown at a site, the number of species that were successfully recruited was higher at sites with a higher topographic wetness index. That is, sites located in areas that are likely to retain more moisture after rain. It is important to note that because most species other than *Acacia* and *Eucalyptus* genera were sown in comparatively few sites, this relationship applies predominantly to species of these two genera and may also explain why management and site conditions had little influence on the relative survival of these more hardy, wide-ranging species (Florentine et al. 2013). As plant species differ in their tolerances to waterlogging and their water use efficiencies (Azam et al. 2014), it makes sense to sow species in locations where they will best survive. Direct seeding has commonly involved seeding a mixture of species simultaneously rather than sowing species individually across a site. Separating species and placing them in a site according to their germination niche could improve species survival rates (Gibson-Roy, Delpratt, and Moore 2007). This approach will also reduce the potential for the *Acacia*- and *Eucalypt*-rich overstorey to outcompete understorey species (Gibson-Roy et al. 2007; Schneemann and McElhinny 2012).

4.3. Natural regeneration

Natural regeneration occurred primarily from seeds dispersing from outside the sites. Thus, at 6–15 years old, the direct seeded sites were not capable of being self-sustaining. Most of the mature furrow plants at this age were *Acacia* species, and much of their seed was observed in the seed bed. Despite the availability of seed, *A.*

dealbata was the only member of this genus to display successful levels of regeneration. More investigation needs to be made into the apparent germination barriers restricting *Acacia* recruitment. Hypothesised barriers such as weed cover (Semple and Koen 2003) and narrow row width (Schneemann and McElhinny 2012) do not seem to be playing a key role for those species that are recruiting outside the furrows (although this doesn't rule them out as factors for species displaying no natural regeneration). Clearly, many of the *Acacia* species were capable of germinating when their seed was pre-treated and placed in disturbed soil. Therefore, we recommend experimenting with disturbance options (e.g. scalping, fire) to improve regeneration success. Natural regeneration is by its nature patchy and clustered, relying on viable seed to be deposited when suitable edaphic and seasonal conditions are met (Vesk and Dorrough 2006). Thus, when possible, choosing sites for revegetation that are in close proximity to seed trees or native vegetation remnants will improve chances of natural regeneration in the long term (even though management interventions will likely be required for sown plants).

4.4. Weed cover

The majority of directly seeded sites had a high abundance of exotic weed cover and comparatively sparse native herbaceous cover, a common finding in areas with a legacy of agriculture and grazing (Dorrough and Scroggie 2008). Abundance of weed cover was not found to negatively affect the density of stems or survival of species in the furrows (predominantly *Acacia* and *Eucalyptus* species), nor did it appear to have a strong influence on natural regeneration of *Eucalyptus* species. However, the legacy of fertiliser use is clearly facilitating the dominance of exotic annual weeds over indigenous perennial ground cover species (Hallett et al. 2014), inhibiting sites from establishing a species-rich herbaceous ground layer typical of the original plant communities existing across the study area (Department of Sustainability and Environment, n.d.).

In addition to fertiliser use, grazing intensity in the years after seeding also affected the abundance of weeds. Lowest weed cover was found in sites with annual crash grazing post-seeding, while regular livestock grazing resulted in the highest abundance of weeds. This suppressive effect of crash grazing on weed abundance is of great interest because it is a low-cost method of weed control that is likely to be embraced by farmers. However, standard restoration procedures insist on complete exclusion of stock from revegetated areas because of the damage continuous grazing has on seedling survival (Spooner, Lunt, and Robinson 2002a; OShea, n.d.). Fast-rotation grazing (crash grazing) is a relatively new method of livestock grazing and thus there is little literature on its impacts on native species. One study by Fischer *et al* (2009) has found that crash grazing does not negatively impact natural regeneration and allows higher levels of germination than continuous grazing. Therefore, we recommend investigating the role of crash grazing in weed control, in particular the timing and frequency of grazing events to reduce weed biomass while permitting native seedling growth.

4.5. Evaluation of site progress

Overall, direct seeding proved successful in terms of native woody cover for most of the surveyed sites. These sites were abandoned pastures or cropland prior to revegetation and as such the additional structure and plant species resulting from direct seeding after 6–15 years represents a good outcome in terms of biodiversity gains, some weed suppression and provision of habitat for some native fauna (Holland and Bennett 2014). However, long-term trajectories of these sites are uncertain. Firstly, because of the minimal natural regeneration observed and the potential for high-density sites to experience extensive mortality as trees age and competitive exclusion increases, which may lead to the collapse of these communities in the absence of management intervention (Schneemann and McElhinny 2012). Secondly, because of a dearth of understorey species (e.g. non-*Acacia* shrubs, peas, native perennial herbaceous species) either through their non-inclusion in seed mixes prior to 2010, or their low survival rates when sown. These plants make up a significant amount of the species diversity of the grassy woodland habitats that characterised much of the study region pre-European settlement (Department of Sustainability and Environment, n.d.), and their absence in revegetated sites is troubling in terms of biodiversity outcomes and the likelihood that these sites will reach their EVC targets. This is especially the case because of the low levels of natural regeneration of native ground layers on revegetated, agricultural land (Prober and Thiele 2005; Spooner, Lunt, and Robinson 2002b).

5. Future research

Recommendations for future research are outlined in Table 5. During the course of this project we began a new monitoring project aimed at establishing long-term monitoring sites to gain a better understanding of: (i) the successional trajectory of direct seeding through time; (ii) the influence of management at different stages; and (iii) the provision of habitat for native birds over time. The first stage was to conduct monitoring of vegetation, site characteristics, and bird diversity (richness and abundance) prior to direct seeding. Future monitoring will occur during the establishment stage and later stages at regular intervals. These 'starting state' surveys will provide benchmark information to allow us to evaluate the changes occurring as a direct result of the revegetation process. Such monitoring and evaluation is rare in conservation but crucial to determine the difference that revegetation efforts are making, especially when comparing management scenarios (Ferraro and Pattanayak 2006; Lindenmayer et al. 2012). This type of monitoring can also be used to determine the spectrum of revegetation outcomes arising as a result of differences in starting states and landscape context, allowing us an insight into the probabilities of new sites being 'successful' based on these factors. This information can then be used to either assist land managers in choosing the most appropriate location for a given objective, or to re-calibrate the biodiversity objectives of a revegetation project to include a more realistic outcome (Monie, Florentine, and Palmer 2013). Given that the objective of many revegetation efforts in the Goulburn Broken CMA is to restore land to pre-settlement habitat, it is also important for these study sites to be compared to vegetation attributes of reference vegetation (e.g. nearby remnant habitat of the target EVC) to assess whether this goal is achievable. This will represent a substantial contribution to the field of restoration research and direct seeding in particular, and will have ramifications applicable to the many regions and organisations across Australia where direct seeding is used.

In addition to the new monitoring project, Table 5 also outlines future research arising from questions raised during the present study that can lead to improvement of current practices. Some of this research builds on the data collected in the current study that was not used in analyses (requires future monitoring and/or reference site comparison). Other research could be conducted in collaboration with amenable

CMAs or agencies that rely heavily on direct seeding. In the case of establishing a native ground layer via direct seeding, there is potential for collaboration or input from researchers at Charles Sturt University who have developed this field in recent years (e.g. Susan Prober and Ian Lunt).

Furthermore, Melbourne Water have been collaborating with Melbourne University on monitoring direct seeding in riparian areas and have expressed an interest in building upon GBCMA's current research. There is potential for an ARC linkage proposal in 2016 between Melbourne University, Melbourne Water and GBCMA on research into creating decision tools for restoration in riparian areas, with a focus on direct seeding.

Finally, many restoration projects occur on private land, making the landholder the ultimate manager. Landholders determine the starting state, site preparation, seed mix (often), and whether management interventions are ongoing. Therefore, a great amount of the uncertainty in conservation outcomes of any revegetation project will be driven by the landholder and their values. It makes sense, therefore, to better understand the nature and magnitude of this relationship through research. Particularly as restoring degraded land to its pre-disturbance state is likely to involve significant stewardship in the long-term.

Table 5 Recommendations for future research on direct seeding in the Goulburn Broken CMA. Pink shading indicates data that has already been collected and green shading indicates data that is currently being collected.

Research Question	Data requirements	Research Action	Outcomes
How can we improve natural regeneration in older sites?	1. Natural regeneration levels prior to treatments 2. Experiments of different disturbance options to trigger regeneration (e.g. burning, scalping, etc.).	Set up experiments at sites where natural regeneration data has already been collected. Each site should have a plot of each treatment plus a control plot. Compare experimental treatments.	Management interventions that may be prescribed to improve regeneration outcomes.
Can we reduce competition arising from stem density?	1. Experiments with thinning sites at different ages (6–15 years). 2. Structural data before and after thinning. 3. Experiments with reducing the amount of overstorey seed sown and collection of data on density of sown species through time.	Thin sites of different ages for which there is complete structural data. At each site, thin with a few different intensities (e.g. 0%, 30%, 80%). Collect structural data 1–2 years after thinning and compare with pre-thinning. At DS sites monitored in the 2016 season, experiment with reducing the amount of overstorey seed being sown in a given area.	Information on both early- and later-management options to achieve densities more in-keeping with EVC targets.
Does crash grazing reduce weed load without damaging natural regeneration?	1. Information from landholders whose sites have been crash grazed about the timing and duration of grazing. 2. Experimental data comparing crash grazing to control.	Trial crash grazing according to landholder advice at sites with different levels of weed cover. At each site, only allow grazing in part of the plot so that crash grazing can be compared to no grazing. Best if sites are chosen where no post-DS grazing has yet occurred.	Post-grazing weed management.
How can we increase the species richness and abundance of the native herbaceous layer?	1. Data on the survival of groundstorey and understorey species from year zero onwards and collection of data on possible limiting factors. 2. Experimental data on the effect of either (i) sowing these species at a different time to overstorey species, or (ii) at different locations in the site.	Use the new long-term monitoring project to better track the germination and establishment of these species. Explore the literature on limits or drivers of their recruitment and measure where possible. In new DS sites that will be monitored (2016), trial placing these species in different parts of the site as well as the traditional practice of all species being sown together in the furrows. At sites surveyed in this study, trial direct seeding of understorey and groundstorey species between furrows where landholders are amenable. If possible, experiment with establishing a ground layer first, and adding overstorey in the second or third year. This will be subject to landholder preference and ability to manage sites more intensively.	Determining the best conditions for most effective revegetation of understorey and groundstorey species will improve both biodiversity outcomes, habitat provision, and the trajectory of sites towards reference habitat.
How is habitat provision changing through time?	1. Structural data collected in this study with missing data added. 2. Structural data collected in future monitoring. 3. Structural data collected from new long-term monitoring project (including 'starting state').	Compare structural data through time.	Allows us to ascertain the benefits to indigenous flora and fauna through time which can assist with planning and creating landscape targets for restoration and maintenance of biodiversity.
What factors lead to direct seeding failure?	1. Collect data from landholders after 3 years about failure status, criteria for deciding a site has failed, and reasons for failure when it occurs.	Report on causes of site failure and what failure looks like (this may be very subjective).	Over 60% of causes of site failure in GBCMA are unknown. Site failure is assessed by landholders and their assessment metrics may be subjective. This information can lead to improved management during establishment. It may also identify whether or not site failure is assessed correctly or if more communication on this issue needs to be provided.
Are direct seeding sites going to reach EVC targets?	1. Data from the current study and the long-term monitoring project. 2. Measurements of ecological attributes measured in reference habitat(s).	Establish survey sites in reference habitat (unmodified remnants of appropriate EVC). Measure ecological attributes using the survey methods employed in this study. Compare pre-2010 and post-2014 DS sites with reference sites.	Evaluation of sites against their targets allows us to identify potential management actions that may need to be taken and also help us to re-evaluate our goals. Comparison with pre-2010 and post-2014 DS sites will allow us to be able to see if changes in management practices have resulted in greater restoration success.

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Appendices

Table A.1 Reasons given for failed sites sown from 1999–2009 for which GBCMA has records of species sown.

Reason for failure	Number of sites affected
Fire	3
Abandoned & grazed	11
Flood	5
Drought	1
Weed burden	1
No reason given	5

Table A.2 Rules for determining life stages for each species.

Direct-seeded species	seedling	sapling	juvenile (2–4 yrs), <4m	juvenile (> 4yrs). > 4m	adult (mature)
A. acinacea	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. aspera	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. dealbata	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. flexifolia	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. genistifolia	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. implexa	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. lanigera	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. mearnsii	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. melanoxylon	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. montana	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. paradoxa	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. penninervis	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. pravissima	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. pycnantha	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. rubida	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
A. verniciflua	<30 cm	30 cm–1.5m	NA	NA	> 1.5m and/or reproductive structures
Allocasuarina leuhmannii	<30 cm	31cm–1m	1m–3m		>3m and/or reproductive structures
Busaria Spinosa					
Callistemon sieberi					
Cassinia species	<10 cm	11 cm–1m	NA	NA	~1.3m
Dodonaea viscosa cuneata					
E. albans	<50 cm	50 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. camaldulensis	<50 cm	50–2 m	2–4 m	>2 m	
E. camphora	<50 cm	50 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. dives	<50 cm	51 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. globulus	<50 cm	52 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. goniocalyx	<50 cm	53 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. leucoxylon	<50 cm	54 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. macrorhyncha	<50 cm	55 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	NA – do not mature within 10 yrs
E. melliodora	<50 cm	50–1.5	1.5–2m	>2 m	
E. microcarpa	<50 cm	50–2 m	2–4 m	2–4 m	
E. obliqua	<50 cm	50 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. polyanthemos	<50 cm	51 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. radiata	<50 cm	52 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. rubida	<50 cm	53 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
E. tricarpa	<50 cm	54 cm–1.5	1.5–4m (<4m)	> 4 m and/or reproductive sructures	
Fabaceae species a	<10 cm	11–50cm	NA	NA	50 cm–1m and/or reproductive structures
Hymenanthra denata	<10 cm	10–50 cm	50 cm–2m	NA	>2m and/or reproductive structures
L. lanigerum	<30cm	30 cm–1m	1–2m	NA	2m + and/or reproductive structures

Table A.3 Details on germination rates of sown species among sites.

Species	Germinated	Did not germinate	No. sites sown	Survival
A. acinacea	36	6	42	0.86
A. aspera	3	2	5	0.60
A. bracyobotra	3	0	3	1.00
A. dealbata	13	5	18	0.72
A. flexifolia	6	1	7	0.86
A. genistifolia	17	10	27	0.63
A. hakeoides	3	0	3	1.00
A. implexa	19	7	26	0.73
A. lanigera	1	2	3	0.33
A. mearnsii	10	3	13	0.77
A. melanoxylon	5	2	7	0.71
A. montana	26	2	28	0.93
A. oswaldii	0	2	2	0.00
A. paradoxa	14	4	18	0.78
A. penninervis	0	1	1	0.00
A. pravissima	4	0	4	1.00
A. pycnantha	39	1	40	0.98
A. rubida	2	0	2	1.00
A. salincina	2	1	3	0.67
A. verniciflua	30	5	35	0.86
Allocasuarina leuhmannii	0	2	2	0.00
Arthropodium strictum	0	1	1	0.00
Atriplex semibaccata	0	2	2	0.00
Bursaria spinosa	0	6	6	0.00
Callistemon sieberi	2	2	4	0.50
Callitris glaucphylla	0	2	2	0.00
Calytrix tetragona	0	1	1	0.00
Cassinia arcuata	0	11	11	0.00
Chrysocephalum apiculatum	0	1	1	0.00
Daviesia leptophylla	1	0	1	1.00
Daviesia ulicifolia	0	4	4	0.00
Dianella revoluta	0	1	1	0.00
Dodonaea viscosa cuneata	7	2	9	0.78
E. albens	7	4	11	0.64
E. blakelyi	1	0	1	1.00
E. camaldulensis	7	10	17	0.41
E. camphora	3	2	5	0.60
E. dives	2	1	3	0.67
E. globulus	0	2	2	0.00
E. goniocalyx	1	5	6	0.17
E. leucoxylon	2	1	3	0.67
E. macrorhyncha	1	6	7	0.14
E.melliodora	11	7	18	0.61
E. microcarpa	11	9	20	0.55
E obliqua	0	1	1	0.00
E. polyanthemos	4	5	9	0.44
E. radiata	1	0	1	1.00
E. rubida	0	2	2	0.00
E. tricarpa	0	1	1	0.00
E. viridis	0	1	1	0.00
Enchylaena tomentosa	1	3	4	0.25
Eninuda nutans	0	1	1	0.00
Eutaxia diffusa	1	1	2	0.50
Eutaxia microphylla	0	5	5	0.00
Hymenanthra denata	0	3	3	0.00
Indigofera australis	0	4	4	0.00
Isotoma axillaris	0	1	1	0.00
Kunzea ericoides	0	1	1	0.00
Leptospermum lanigerum	1	1	2	0.50
Leptospermum continentale	3	4	7	0.43
Maireana declavans	0	1	1	0.00
Melaleuca parvistinea	3	0	3	1.00
Mirbelia oxylobioides	0	1	1	0.00
Ozothamnus ferrgineus	0	1	1	0.00
Ozothamnus obcordatos	0	2	2	0.00
Poa labillardieri	0	1	1	0.00
Pultenaea daphnoidies	0	1	1	0.00
Pycnosorus globosus	1	0	1	1.00
Senna artisemoides	2	0	2	1.00
Themeda australis	0	1	1	0.00
Themeda triandra	0	1	1	0.00

A.1 Species found regenerating between directly seeded furrows in survey sites

A. acinacea
A. dealbata
A. montana
A. paradoxa
A. implexa
A. mearnsii
A. pycnantha
A. flexifolia
A. genistifolia
A. pravissima
A. rubida
A. verniciflua
Atriplex semibaccata
Dondonaea cuneata
E. blakelyi
E. camaldulensis
E. microcarpa
E. melliodora
E. microcarpa
E. polyanthemos
E. radiata
Enchylaena tomentosa
Leptospermum continentale
Pimelea linifolia
Pultenaea humilis
Senna artemisioides

Table A.4 Results of principal components analysis of ground cover variables. The first 2 components (PC1 and PC2) only are shown because together they explained 92% of the variance in ground cover relative abundance among sites.

Ground cover	PC1	PC2
Bare ground	0.09310482	0.00275638
Litter	0.60823695	0.49926335
Rock	-0.0030673	-0.0149284
Logs	0.00587688	-0.0004729
Crusts	0.10752097	-0.1441143
Native scat	0.0060242	-0.0043528
Exotic scat	0.00036714	-0.0005582
Other	0.00191369	0.00423074
Weed cover	-0.7797867	0.40918102
Native cover	-0.0408317	-0.7498468

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