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Title: Replanting agricultural landscapes: How well do plants survive after habitat restoration?

Running Head: Restoration survival in agricultural landscapes

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Abstract:

Landscape-scale habitat restoration has the potential to return ecosystem functions and services and mitigate the loss of native flora and fauna. However, restoration projects rarely monitor the effectiveness of restoration efforts, such as quantifying the establishment success (survival) of the planted species. We monitored a landscape-scale revegetation program in south-eastern Australia that planted five million plants representing 35 native species over a four-year period (2012 - 2015). We assessed the restoration effectiveness across years to evaluate how different lifeforms survived over time and the factors that influenced the differential survival of lifeforms and individual plant species 3 months (spring), and 9 months (after summer) post planting. Establishment success varied across years with survival lowest in the 2015 planting season. Survival of different lifeforms after summer were associated with site-level variables (e.g. mean maximum temperature, rainfall and soil type) with survival generally declining due to high temperatures, low rainfall, and for species planted on sandy or saline soils. Maximum temperature, rainfall and soil type were the most important predictors of compositional change in the 20 species commonly planted across years, with two saltbush species (*Atriplex paludosa, Enchylaena tomentosa*) and one eucalypt species (*Eucalyptus fasciculosa*) having the highest survival, while one sedge species (*Juncus kraussii*) and two grass species (*Poa poiformis* and *Puccinellia stricta*) had among the lowest observed survival. These results highlight the importance of monitoring establishment success through survival to detect changes in the composition of lifeforms and species to guide future re-plantings aimed at returning the desired plant diversity.

Implications:

- The survival of revegetated species and lifeforms is negatively influenced by high temperatures and low rainfall during the first 9 months after planting, resulting in the loss of almost one third of restored plants. Plants in saline or sandy soils are likely to have lower survival rates.
- Climate change may influence the timing and location of future restoration. In this study species with the highest potential for restoration were two saltbush species (*Atriplex*

paludosa and *Enchylaena tomentosa*), as well as a eucalypt species (*Eucalyptus fasciculosa*).

- Monitoring of restoration plantings is vital to understand how the composition of species and lifeforms changes over time, and to inform adaptive management strategies aimed at resetting the trajectory of plantings towards a reference state.

Introduction:

Landscape degradation through land-use change (clearing and conversion) and habitat fragmentation are major factors associated with the loss of biodiversity throughout the world (Abensperg-Traun et al. 2004; Fahrig 2003; Fischer & Lindenmayer 2007), with only 31% of the Earth's primary forests remaining (FAO 2016). To maintain biodiversity and mitigate on-going change, extensive habitat restoration via methods such as revegetation are necessary (Bell et al. 1997; Hobbs 1993; Hobbs et al. 2014; Lindenmayer et al. 2010). Revegetation can buffer core areas of remnant vegetation and provide linear strips or stepping stones of habitat across an agricultural matrix, with the goal of enabling animal and plant movement and dispersal (Vesk & Mac Nally 2006). Space, time and available funds usually constrain the extent and quality of revegetation in a given landscape (Chazdon 2008; Merriam & Saunders 1993).

Habitat restoration shows potential to enhance and maintain native animal and plant communities in natural systems (Lindenmayer et al. 2010; Wortley et al. 2013). However, the biodiversity outcomes of revegetation are poorly known for some habitats (Jellinek et al. 2014; Lindenmayer et al. 2010), and socio-ecological outcomes are seldom studied (Sacha Jellinek et al. 2019). While studies into the effectiveness of restoration are increasing, landscape-scale restoration programs are seldom studied because funding is usually focussed towards on-ground outcomes rather than monitoring how effective restoration has been in restoring biodiverse habitats (Chapman & Underwood 2000). This means that there is limited interpretation of restoration results and little follow-up to determine replanting success or failure (Atyeo & Thackway 2009; Chapman & Underwood 2000; Godefroid et al. 2011). This lack of information may lead to a substantial expenditure of resources on restoration activities without the expected biodiversity benefits (Jellinek et al. 2014; Rumpff et al. 2011).

As environmental stressors such as climate change increasingly impact biodiversity (Prober et al. 2019), and as conservation schemes become increasingly important to maintain our remaining native plant and animal communities, there is a need to understand what proportion of plants survive in restoration projects, and which species persist (Belder et al. 2018; Hobbs 2018). Understanding factors that influence plant survival after the first year of planting is critical because initial planting success will impact the future effectiveness of the restoration, and because initial survival will alter as climate becomes increasingly unpredictable (Ruthrof et al. 2013). Previous studies suggest that the survival of replanted vegetation and the structure and function of restored systems may be impacted by abiotic, biotic and anthropogenic processes (Hallett et al. 2014; Middleton et al. 2010; Standish et al. 2012). For example, Hallett et al. (2014) showed that survival after direct seeding was usually limited by variables such as soil type and climate, while Middleton et al. (2010) found that planting technique and competition from exotic species influenced planting success. There are a variety of guidelines that outline the required steps to properly plan and implement restoration activities (Breed et al. 2018; Hancock et al. 2018; Harrison et al. 2017;

McDonald et al. 2016), however, few provide guidelines to adequately assess restoration success through time and space (Camarretta et al. 2019).

To maximise the benefits of habitat restoration, it is necessary to gain an understanding of what factors influence vegetation survival in restored areas. Our project assessed a landscape-scale restoration project in south-eastern Australia that planted in excess of five million native plants over a four-year period. Through monitoring plant survival in restored habitats, we sought to: (i) quantify how plant survival changed directly after planting (spring), after the first summer (autumn), and between planting years, (ii) determine how factors such as soil type, mean rainfall and temperature influenced plant survivorship, and (iii) assess how survival of different life-forms and individual species is likely to influence community composition of restored sites. Our results have important implications for natural resource managers and practitioners undertaking restoration activities, as they provide a reference of restoration success in the first nine months after planting, outline the potential variables that are likely to influence plant establishment and how this may influence community composition over the long-term.

Methods:

Study Area

Our project focussed on assessing the survivorship of native plants that had been planted in an agricultural landscape in south-eastern Australia. Our study landscape surrounded a Ramsar listed wetland of ecological and cultural significance - the Coorong, Lower Lakes and Murray Mouth region (CLLMM). This Ramsar wetland is located at the terminus of the

Murray-Darling Basin, which covers approximately 14% of Australia (Cann et al. 2000) and contains the Murray River, the world's third longest navigable river.

In this landscape, mean annual rainfall varies from 255 - 706 mm, with the Mount Lofty Ranges to the north generally having higher rainfall compared to other areas of the Ramsar wetland. Mean annual temperatures range between 15.3 - 16.5 °C, although summer maximums can exceed 40 °C (Bureau of Meteorology 2019). Topographic variation is slight, with a maximum elevation of 180 m on the south-eastern slopes of the Mount Lofty Ranges. The natural vegetation is diverse, ranging from wetland-associated habitats, samphire and terrestrial communities including grassland, coastal heathland, mallee and grassy woodlands (S. Jellinek et al. 2019). Much of the terrestrial landscape has been cleared since European settlement for agriculture, including livestock grazing, cropping and viticulture (Berkinshaw 2009). As a result the majority of the landscape is either fragmented or relictual with an average remnancy of 22% (S. Jellinek et al. 2019).

Plantings

Restoration plantings were undertaken within a 5 km area of the Coorong, Lake Alexandrina and Lake Albert Ramsar wetland (Fig. 1) (S. Jellinek et al. 2019). Native plant species were planted at selected sites as tubestock in winter (June - August) each year from 2012 to 2015. Tubestock were in the form of 'hiko cells' (rigid plastic trays of 40 plants each, 43 mm wide and 100 mm deep) or 'forestry tubes' (square sided individual tubes 50 mm wide and 150 mm deep), grown from local provenance seed and propagated by local plant nurseries. The plants were healthy prior to planting and were 6 - 8 months old when planted and less than half a metre tall. Tubestock were hand planted by community groups and commercial contractors, with approximately 800,000 native plants being planted annually.

On average, 35 native plant species were planted at each site and comprised approximately 11% overstorey, 38% midstorey and 51% understorey species, but overall around 200 species were planted across the landscape (S. Jellinek et al. 2019). These were largely woody species (trees and shrubs), tussock grasses and salt tolerant groundcovers selected to resemble remnant habitats (reference state) that had previously occurred in the landscape - samphire, mallee, sheoak and native pine woodland and eucalypt woodland. Lifeforms planted were defined as: trees - woody vegetation > 3 m tall, shrubs - woody vegetation > 0.5 m tall up to 3 m tall, grasses - non woody plants with long narrow leaves up to 0.5 m tall, herbs - ground plants with a woody or non-woody base and an ephemeral ground stem, vines - plants with a climbing habit, and sedges - tussock forming plants associated with wet areas.

These occurred on a variety of different soil types - saline, sand-hills, sandy loam soils over calcrete and sandy loam soils over clay (Hall et al. 2009). The soils were generally loamy or clay soils over sand, calcrete or clay (B, D and F soils), sandy loams over clay (K soils), sand over clays (G soils) or coastal sands (H soils) or wet soils that were either saline (N2) or freshwater (N3). The Mt Lofty Ranges generally had loam over clay soils with wetter soils (N3) on the lower slopes and depressions while the Coorong and surrounding peninsulas had sandy soils and more saline soils (N2) in seasonally inundated areas (S. Jellinek et al. 2019).

The majority of restored sites were privately owned and had previously been used for agricultural production (grazing and cropping) over the last century. Plants were planted across a site in locations where they were most likely to be found naturally and therefore survive best. For example, species that preferred wet soils were planted in depressions or along drainage lines. All plants were guarded, usually with cardboard cartons. All sites were fenced 3 to 6 months prior to planting to exclude domestic livestock. Fences were permanent and 1.2 metres high, consisting of wooden posts with 4 to 6 electric or plain wires. Sites were sprayed with herbicide at least once - a month prior to planting - to reduce weed competition (S. Jellinek et al. 2019). Where possible, rabbits and other herbivores were controlled.

Survivorship Surveys

To determine the survival rates of restored sites we surveyed multiple transects twice annually; in spring (approximately three months after the start of planting in winter - September to October) and then again in autumn (after the first summer, approximately nine months after planting - March to April). These surveys were undertaken for each planting year from 2012 - 2015 (Fig. 1). Sites were not revisited in subsequent years after the initial spring and summer surveys were undertaken. Transects consisted of surveying 1 m either side of a 50 m line and identifying planted seedlings to species level and noting if they were alive or dead. Plants that were dead but could not be identified were also recorded as ‘dead unknown’. The number of transects undertaken at each site was proportional to the size of the planted area (Table S1).

Vegetation monitoring was conducted at a total of 153 sites (Table S1), with a proportion of these surveyed in spring (61 sites). Spring monitoring was not undertaken in the first year of planting (2012). Surveys were undertaken by two teams of two people who were experienced in plant identification and independent of the planting program. We also calculated the mean rainfall, mean temperature, and mean maximum temperature for the quarter leading up to spring (June - September) and autumn (November - February) obtained from three different weather stations (Milang, Meningie and Wellington) that spanned the planting area (Bureau of Meteorology 2019).

Analysis

To assess whether plant survival varied seasonally (spring and autumn) and between years among lifeforms we fitted the following generalised linear mixed model assuming a binomial error distribution and a logit link function using the *glmmTMB* package (Brooks et al. 2017) in R (R Core Team 2014):

Y = μ + lifeform + season + year + lifeform*season + lifeform*year + season*year + lifeform*season*year + *site* + *lifeform*site* + *season(lifeform*site)* + *year(season(lifeform*site)* + ε

where Y is a vector of alive and dead response values, lifeform corresponds to the broad grouping of species (tree, shrub, herb, grass, sedge and vine), season is the spring and autumn survival assessment periods, year is the planting period excluding the 2012 planting, site is the random effect of planting location and its interaction on the fixed effects, and ε is the random error. Model assumptions of normality and homoscedasticity of residuals as well as overdispersion were assessed using simulated residuals using the *DHARMa* package (Hartig 2019) following Zuur & Leno (2016). The statistical testing of fixed effects was undertaken using Type III Wald chi-squared likelihood-ratio tests performed using the *glmmTMB* package. When significant fixed effects were detected, a Tukey's multiple comparison test on the logit scale and back-transformed estimates of the least-square means and 95% confidence intervals (CI) were obtained using the *emmeans* package (Lenth 2019).

To determine how environmental factors differentially effect survival of lifeforms after the first summer (9 months post-planting) within each planting year, we fitted the following model using the procedure mentioned above:

$$Y = \mu + \text{lifeform} + \text{soil} + \text{temperature} + \text{rainfall} + \textit{site} + \textit{lifeform(site)} + \varepsilon$$

where soil is the dominant soil subgroups for each transect (Table S2) using data obtained from the Department of Environment, Water and Natural Resources in ArcGIS (ESRI 2011), and mean maximum temperature and mean rainfall are the standardised covariates.

To ascertain how the survival of individual species impacted species composition over the planting season, we chose twenty species that were the most commonly planted across all the planting sites. These included a mix of trees (n = 6), shrubs (n = 6), grasses (n = 5) and sedges (n = 3). Compositional change was modelled as a function of soil, rainfall and temperature after the first summer growth period using the non-parametric, multivariate random forest implementation of *gradientForest* package (Ellis et al. 2012). The goal of *gradientForest* was to identify important predictors of compositional change and points along the environmental gradient that are important thresholds of this change. Here, we refer to compositional change as the change in species abundance during the first growing season, which was accounted for by fitting a year by season cofactor.

Species abundance was natural log-transformed (log[x + *c*]), using a constant value *c* of 1 pertaining to the minimum non-zero observation for any species, and was treated as the response variable in the fitted gradient forest model. The final model was an ensemble of 2000 regression trees, where each tree was cross-validated using the out-of-bag samples (OOB, the bootstrap subsample of the data not used in the building of a given regression tree), which were conditionally permutated when the pairwise correlation between predictor variables were above the threshold of r = 0.50 following Strobl et al. (2008). The overall importance of a given predictor variable was estimated by the increase in the mean squared OOB error after randomly permuting the predictor variable (Ellis et al. 2012).

Results:

Lifeform survival across years

Survival was initially high (range: 90 - 96%) 3 months post-planting (spring), with vines tending to survive best. After the first summer (autumn), survival decreased (range: 69 - 94%) with grasses showing the greatest mortality. Irrespective of planting year there was a significant interaction between lifeform and season (χ^2_5 = 67.6, P < 0.001), with all lifeforms except vines (P = 0.947) showing a significant (P < 0.05) decrease in the proportion of individuals surviving between seasons (Fig. 2). Survival significantly varied across years (χ^2_2 = 10.5, P = 0.005), with species planted prior to 2015 tended to be three times more likely to survive (odds ratio: 3.5 [2013 vs 2015] and 3.0 [2014 vs 2015]) than those planted in 2015, with only 89% of plants surviving during the 2015 planting season, irrespective of lifeform (Fig. 3). However, the survival of different lifeforms did not significantly differ across years (P = 0.058) or seasons within years (P = 0.134).

Environmental effect on survival of lifeforms

The probability of lifeform survival was most significantly associated with the soil type they were planted into (Table S3, Fig. 4, Fig. S1). While there was a general pattern for lifeform survival to decrease as the mean maximum temperature increased, this was not significant ($P > 0.05$; Table S3) suggesting early age survival was not contingent on maximum temperature. Rainfall had a significant ($P < 0.05$) effect on survival (Table S3) with a positive rainfall coefficient in all models except the 2012 planting season, suggesting increasing lifeform survival with increasing rainfall.

Change in species composition with environment

Species composition after the first summer growth period was modelled using a subset of the data that represented the log-transformed abundance of the 20 most common planted species. Of the plant species most regularly planted during this project, two saltbush species had the highest survival at 96% - marsh saltbush (*Atriplex paludosa*), and ruby saltbush (*Enchylaena tomentosa*), followed by pink gum (*Eucalyptus fasciculosa* - 92%) (Fig. 5). The species that had the lowest survival were sea rush (*Juncus kraussii* - 59%), coast tussock-grass (*Poa poiformis* - 54%) and Australian saltmarsh grass (*Puccinellia stricta* - 54%) (Fig. 5). Trees commonly used in this project that had moderate survival included ridge-fruited mallee (*E. incrassata* - 87%), Murray pine (*Callitris gracilis* - 86%), drooping she-oak (*Allocasuarina verticillata* - 82%) and white mallee (*E. diversifolia* - 70%) (Fig. 5).

The most important predictors of changes in species abundance after the first summer were mean maximum temperature, mean rainfall, sand over clay and deep sand soil types (Fig.

6a), consistent with the models of lifeform survival (see above). Individual species curves showed a strong change point at 30°C indicating a steep change in species composition and another but weaker change point at 40°C (Fig. 6b). Species compositional changes were homogenous under low rainfall (50 mm to 150 mm), but species composition steeply changed monotonically with increasing precipitation (Fig. 6b).

Discussion:

Plant survival varied across years and seasons within a year, with the soil type and rainfall at a site significantly influencing the survival of different lifeforms. After the first summer, lifeform survival decreased to 69% (grass) and 95% (vine), showing a 1% to 23% decrease in survival from the spring assessment. The proportion of individuals surviving were much higher than some other studies, where a global review of 249 plant reintroductions found that 52% of plants survived (Godefroid et al. 2011), and a study on eucalypt woodlands found plant survival declined by 50% in the first 6 months after establishment, and few survived after 5 years (Clarke 2002).

Survival between years substantially differed in our study with maximum temperatures having a negative effect on plant survival. Indeed, extreme climatic events are known to have a negative effect on plant survival and growth (McDowell et al. 2008; Niu et al. 2014; Prober et al. 2015). These negative effects will likely be exacerbated under a hotter and drier future climate in south-eastern Australia (Ruthrof et al. 2013), especially with rainfall predicted to decline over the winter months - when planting is usually undertaken (Jellinek & Bailey 2020). Similarly, multiple abiotic and biotic factors influence the survival of plants during restoration (Hallett et al. 2014; Perring et al. 2015; Standish et al. 2012), such as soil type, which is known to drive the survival of plants in restoration projects (Haan et al. 2012; Hallett et al. 2014; Perring et al. 2015). While we were unable to statistically test the interaction between lifeform and soil type due to our models not converging, soil type was an important predictor of changes in species composition. Indeed, grass species were more frequently observed in loam over clay soils compared to those planted in saline soils, whereas tree species were more frequent in calcareous loam soils compared to those planted in deep sandy soils. This pattern is consistent with previous studies showing that plant survival is higher on sandy and loamy soils over clay, probably due to better water holding capabilities, and lower on sandy (Perring et al. 2015) and saline soils.

While not studied here, livestock grazing can also negatively affect soil structure, fertility and microtopography (Yates, Norton, et al. 2000), while adjacent land-use practices such as intensive cropping can increase chemical runoff and similarly alter soil structure, causing restoration efforts to be less effective (Bourgeois, Vanasse, Rivest, et al. 2016). Anecdotal evidence suggests that competition from weeds is likely to be a major factor influencing plant survival, as they compete for water, light and soil nutrients (England et al. 2013; Hallett et al. 2014; Middleton et al. 2010), and competition from existing trees may also impact the establishment of ground layers through similar processes (Bourgeois, Vanasse, & Poulin 2016).

Plant survival initially after planting (spring) was most probably a response to planting shock (i.e., planting check or transplant shock) (Close et al. 2005; South & Zwolinski 1997). Planting shock occurs when a plant is stressed, and can be caused by factors such as within-plant characteristics (acclimatisation and nutrient stress), abiotic influences (drought, frost, mechanical damage by wind etc.), and environmental interactions (soil type and compaction, and competition) (Close et al. 2005). In our study plants were likely to be influenced by several factors: the initial condition and age of the nursery stock; planting and guarding techniques used; the amount of water (soil moisture) prior to and during planting; competition from weeds; mechanical damage from wind and sand movement; non-wetting soils on sandy sites and previous land-use practices. While not assessed here, other factors such as ecophysiological traits including fungal associations, soil bacteria and nutrient cycling may play a role in the survival of the different species and lifeforms studied here (Gehring et al. 2017; Macdonald et al. 2019). For example, anthropogenic influences are likely to have altered the soil microbiome of the agricultural sites we studied, negatively influencing mutualistic relationships between plants and soil bacteria (Gehring et al. 2017). Regardless, determining which combination of these factors, amongst other variables, are key future research areas to ensure better restoration outcomes.

Shrubs, herbs, trees and vines were the most successful lifeforms to establish during this study, while grass and sedge establishment was substantially lower. Previous studies in similar landscapes suggest that establishment success is increased with increasing levels of management intervention (Yates, Hobbs, et al. 2000), such as scalping the top soil layer when establishing native grasslands to remove weed seedbanks and nutrient enriched soils (Cuneo et al. 2018; Lindsay & Cunningham 2011). However, even without scalping some species we studied such as tussock-grass (*Poa labillardieri)*, common wallaby grass (*Rytidosperma caespitosum*) and speargrass (*Austrostipa sp*.) survived well after the first summer. This may be related to the soil type some grass species were planted on, having higher survival on loam soils and loamy and sandy soils over clay. Tree species such as pink gum (*Eucalyptus fasciculosa*), ridge-fruited mallee (*E. incrassata*), white mallee (*E. diversifolia*), Murray pine (*Callitris gracilis*), and drooping sheoak (*Allocasuarina verticillata*) survived well during our study (over 70% survival). This was somewhat unexpected, as animal grazing is known to have a major impact on revegetation survival (Forsyth et al. 2015). Greater research should be undertaken to determine which species are most susceptible to animal grazing and modify management actions accordingly to better protect these species.

Change in species composition was best predicted by maximum temperature, rainfall, and soil type. As suggested above, temperature extremes and factors such as soil type are recognised as major factors determining the distribution and survival of plant communities (Reyer et al. 2013; Zimmermann et al. 2009). While the original planted communities were quite diverse, comprising native trees, shrubs and grasses, plant mortality seems to have a large impact on plant community composition, resulting in a community more homogenous to those that were initially planted, making them less similar to the remnant reference ecosystems they were meant to resemble (Marin-Spiotta et al. 2007). These planted communities may be in alternative ecosystem states (Hobbs et al. 2014; Suding et al. 2004), potentially altering the vegetation communities occurring in the landscape in the future and their associated faunal communities (Jellinek et al. 2013).

Limitations

There are several factors that may have influenced the results of this study. We only monitored plant survival during the first year after planting, which is an insufficient amount of time to make inferences about long-term efficacy of habitat restoration. Having long-term monitoring sites is an important aspect of learning from restoration outcomes. Anecdotally we can say that many of the sites and species were surviving and growing well five years after these surveys were undertaken, but this cannot be quantified without further monitoring.

Similarly, the survey technique used (randomly placed 50 m transects) may not have picked-up species that were planted in low densities in specific locations, and due to the nature of restoration plantings, multiple plant communities could have been present at a site, making it difficult to provide specific results for distinct communities. For example, samphire habitats would generally be planted on the edge of a wetland habitats, with eucalypt woodland planted on higher land adjacent to these areas. Similarly, individual species are likely to prefer specific soil types, but due to the field nature of this project it was difficult to assess how different plants rely upon microhabitats to survive and grow. An experimental design that would allow for the separation of the different plant communities may better identify those communities most at risk of species loss. A baseline monitoring method to assess restoration survival is an important future research area, as it would allow management agencies to compare their restoration results over multiple sites and landscapes into the future.

Management Interventions

A range of management interventions could be implemented to increase plant survival during the first year of establishment. To reduce the impact of planting shock, monitoring the condition of nursery stock prior to planting, and ensuring that plants are planted correctly (i.e., appropriate planting depth, guards installed correctly, species planted in suitable locations, etc) could ensure the quality of plants and planting activities. Weed management prior to planting and during plant establishment is also likely to have a beneficial impact on initial plant survival (Close & Davidson 2003).

Reinstating a diversity of species and lifeforms in conservation programs is the most effective way to ensure that a functional ecosystem likely returns after restoration (McDonald et al. 2016). As plants naturally regenerate at different rates and successional stages, planting seedlings amongst existing vegetation and having 'nurse' plants can increase plant survival in the first few years of establishment (Gómez-Aparicio 2009). These natural successional stages are usually not followed during revegetation programs due to time and budget constraints, but establishment may be more effective if areas are restored over multiple years so these nurse plants can establish.

In our study, all the plants were locally sourced and grown, and are presumed to be adapted to the local climate. However, climate change will influence the survival of restored species (Prober et al. 2019), especially as conditions are expected to become hotter and drier in south-eastern Australia (CSIRO 2018). This will require practitioners to have a better understanding of the plant provenances that are currently being propagated, including potentially sourcing seed from provenances growing in hotter and drier climatic zones and/or identifying surrogates if some species are unlikely to survive hotter and drier conditions (Breed et al. 2018; Broadhurst et al. 2008; Harrison et al. 2017; Prober et al. 2015). As climate change is likely to cause an increase in extreme weather events, practitioners may need to protect plants from high temperature events (>30 ^{o}C and <38 ^{o}C) and potentially water plants, if feasible, at times of low rainfall. Other management actions could include ensuring species are planted on appropriate soils, ideally those that have the capacity to hold more soil moisture, and adjusting timing of planting if rainfall is likely to be lower over the winter months (Prober et al. 2019). Having better seasonal forecasts to inform the planning and management of restoration plantings may assist practitioners to alter the timing of restoration activities (Hagger et al. 2018).

If the plant communities currently surviving are substantially different to those that were planned, then monitoring is vital in order to learn from restoration outcomes, and to adaptively manage restored areas into the future (Dickinson et al. 2016). Limited rigorous assessments of restored areas can result in inadequate interpretation of restoration results and little follow-up to determine replanting success or failure (Atyeo & Thackway 2009; Chapman & Underwood 2000). This limits the ability of restoration practitioners to learn how to more effectively undertake restoration activities now and in the future (Bernhardt & Palmer 2011; Chapman & Underwood 2000). Understanding how plant communities are changing in the first year, and in subsequent years, is important to allow managers to plant more effectively during restoration, and to re-establish plants that may have been lost in previously restored sites. Having more standardised revegetation monitoring methods, and a centralised database to store monitoring information, would greatly improve our knowledge of revegetation outcomes.

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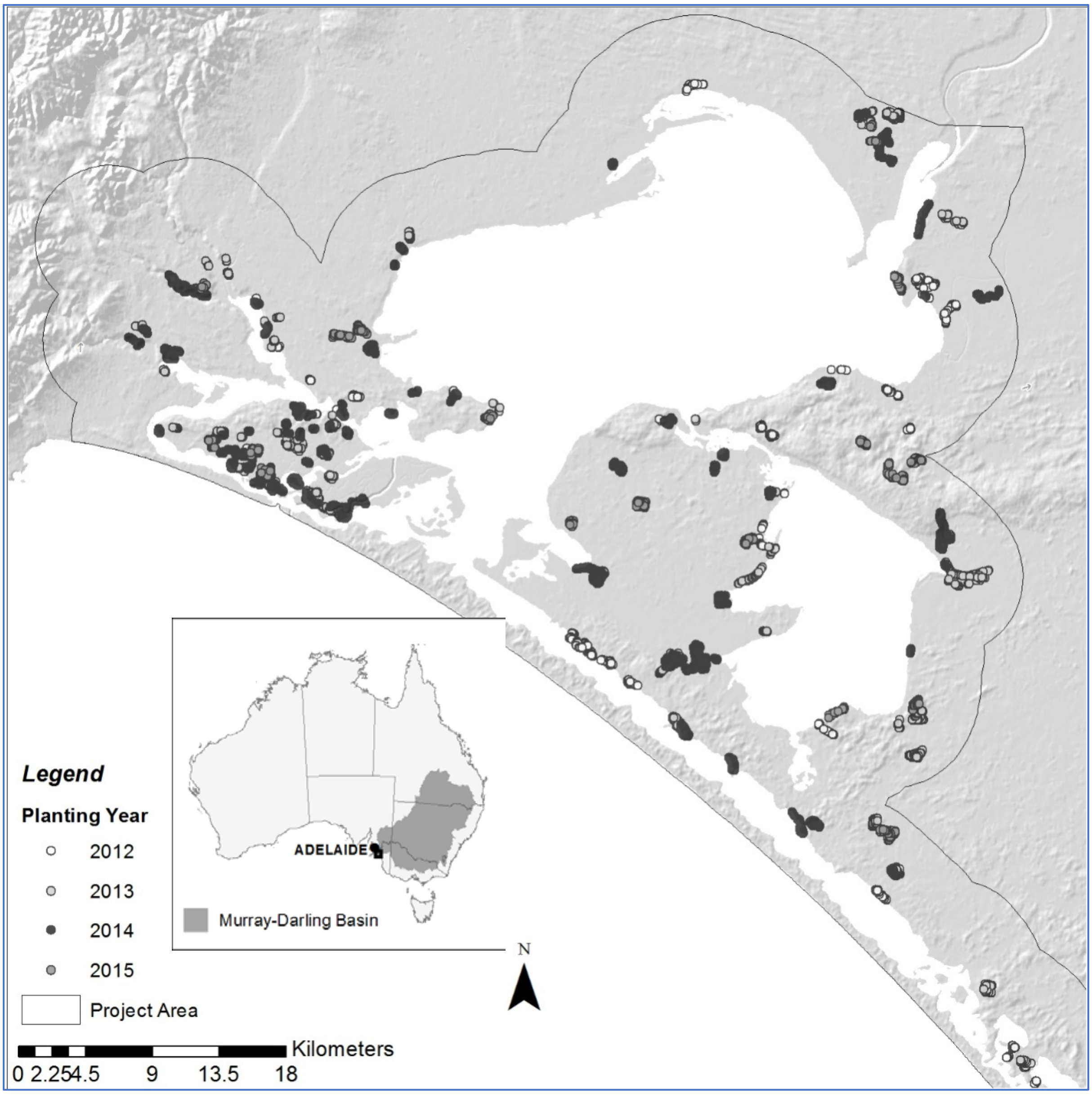

Fig. 1. Study area around the Coorong, Lake Alexandrina and Lake Albert in South Australia showing transect points where restoration survivorship surveys where undertaken between 2012 and 2015. The colour of the circle symbol corresponds to the planting year. Inset map shows the extent of the Murray-Darling Basin (grey shared area) across south-eastern Australia.

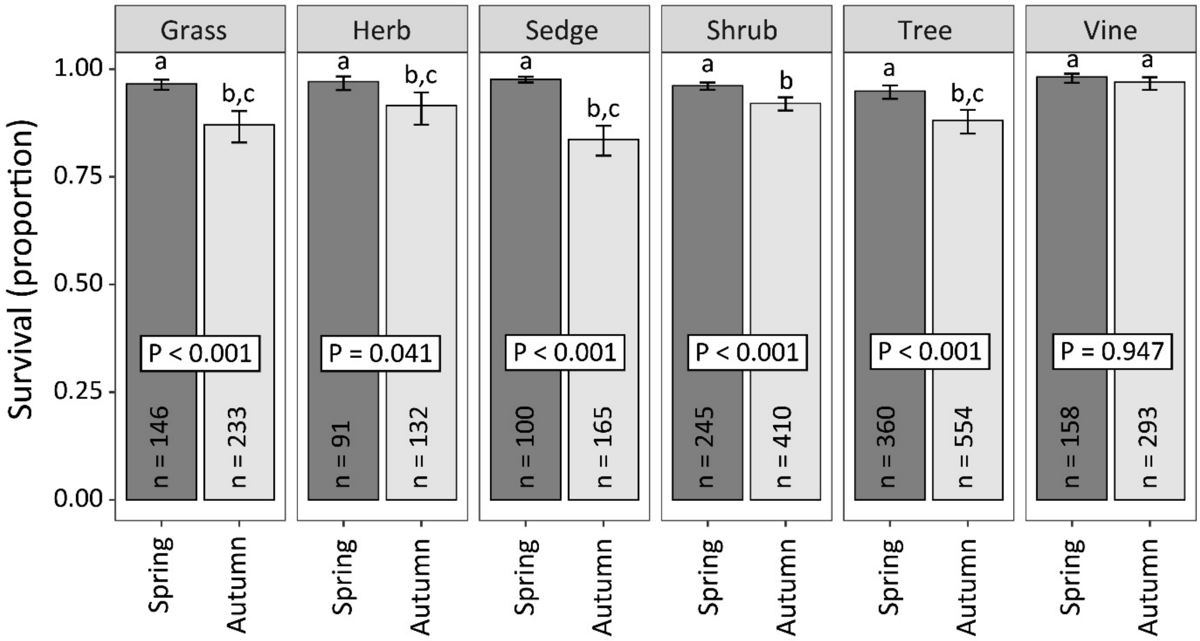

Fig. 2. The least-square mean proportion of survivors and ± 95 % CI for different lifeforms in spring (dark bars) and autumn (light bars). Letters above each bar denote significant differences (P < 0.05) among lifeforms and seasons, while pairwise differences between seasons within lifeforms are also shown, based on a Tukey's multiple comparison test. The sample size (n) for each lifeform by season is shown.

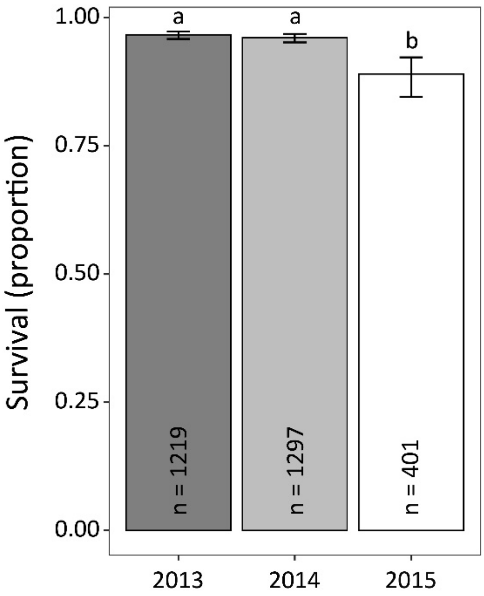

Fig. 3. The least-square mean proportion of survivors and ± 95 % CI for plants across the 2013 to 2015 planting periods. Letters above each bar denote significant differences ($P < 0.05$) among assessed years based on a Tukey's multiple comparison test. The sample size (n) for each year is shown.

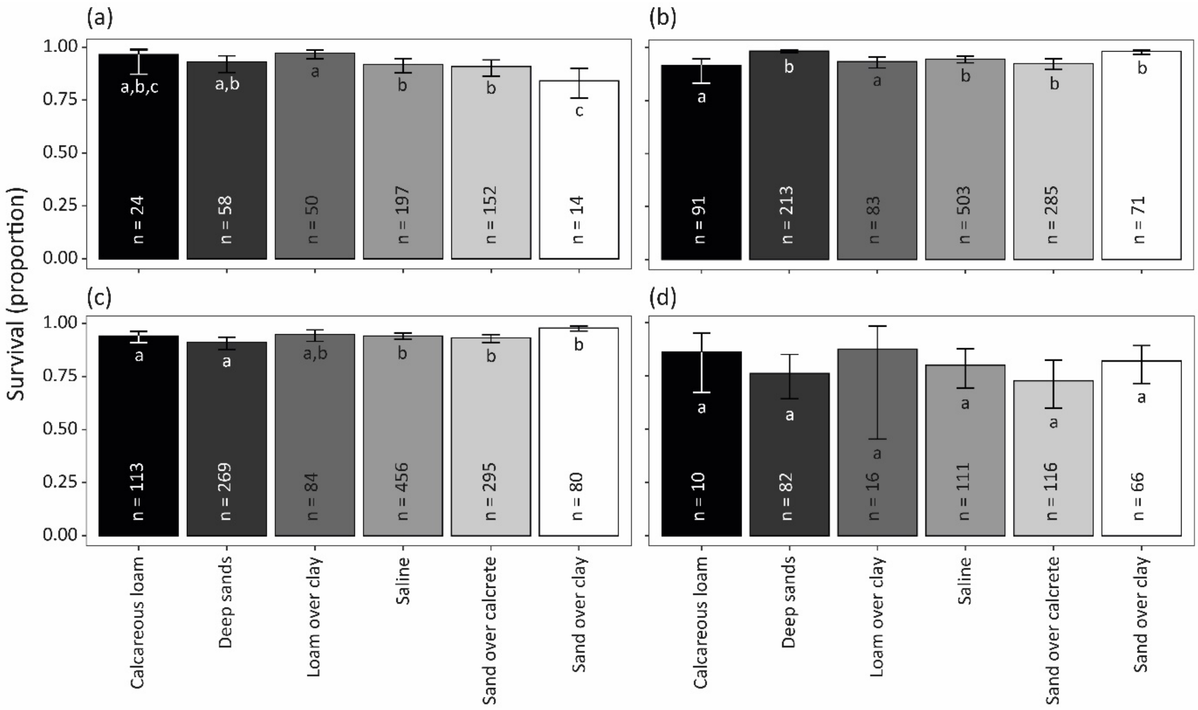

Fig. 4. The least-square mean proportion of plant survival and ± 95 % CI of plants planted in six soil types after the first summer (9 months post-planting) for 2012 (a), 2013 (b), 2014 (c) and 2015 (d). Letters above each bar denote significant differences (P < 0.05) among soil types based on a Tukey's multiple comparison test. The sample size (n) for each soil type is shown.

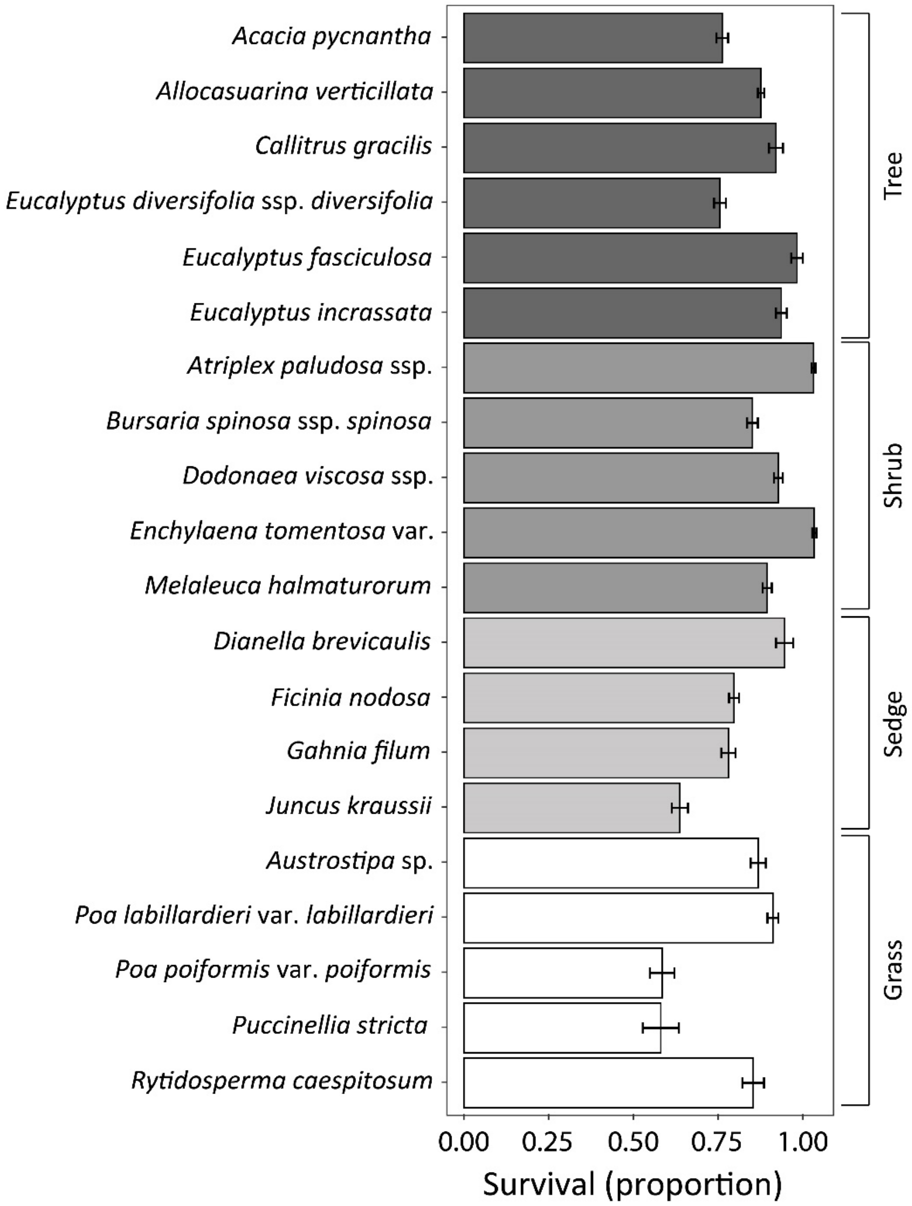

Fig. 5. The airthmetic means (± SE) for the survivorship of the top 20 plant species used in revegetation after the first summer (autumn) pooled across years. Species have been organised alphabetically within the four lifeforms.

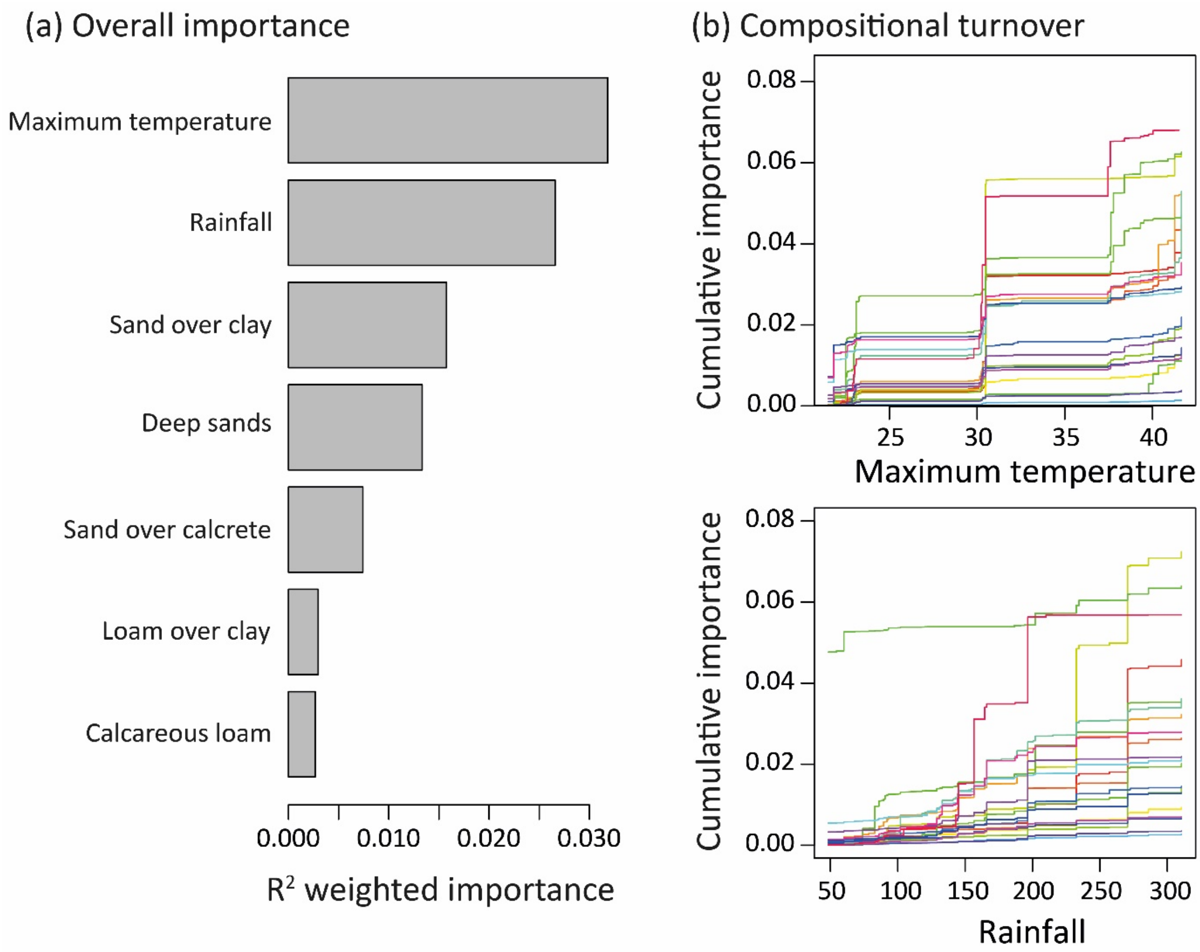

Fig. 6. Overall importance (R^2) for each predictor (maximum temperature, rainfall, soil type) of species composition (a) and the compositional change functions for the two predictors, maximum temperature and rainfall for all 20 species (b). Each line in (b) corresponds to a different species.

Table S1. The number and size of the sites surveyed in each season from 2012 - 2015.

Year	Spring		Autumn	
	Sites	Transects	Sites	Transects
2012			40 (344 ha)	843
2013	27 (217 ha)	387	50 (360 ha)	968
2014	17 (193 ha)	410	46 (396 ha)	1016
2015	17 (244 ha)	428	17 (244 ha)	450

Table S2. The soil type and the number of sites surveyed in each season from 2012 - 2015

Year	Soil Type	Spring	Autumn
2012	Calcareous_Loam		33
2012	Deep_Sands		124
2012	Loam_Over_Clay		45
2012	Saline		331
2012	Sand_Over_Calcrete		284
2012	Sand_Over_Clay		26
2013	Calcareous_Loam	14	88
2013	Deep_Sands	50	179
2013	Loam_Over_Clay	7	60
2013	Saline	212	411
2013	Sand_Over_Calcrete	98	200
2013	Sand_Over_Clay	6	30
2014	Calcareous_Loam	53	110
2014	Deep_Sands	119	222
2014	Loam_Over_Clay	3	56
2014	Saline	150	319
2014	Sand_Over_Calcrete	61	269
2014	Sand_Over_Clay	24	40
2015	Calcareous_Loam	2	6
2015	Deep_Sands	123	115
2015	Loam_Over_Clay	7	5
2015	Saline	84	73
2015	Sand_Over_Calcrete	137	169
2015	Sand_Over_Clay	75	26

Table S3. The analysis of variance (ANOVA) results showing the fixed effect of lifeform, soil, rainfall and maximum temperature on the probability of survival after the first summer (i.e. 9 months post-planting) over the four planting periods (2012-2015). Shown is the chi-square statistic (Chi), the degrees of freedom for the likelihood test (DF), and the probability (Pr) of an observed high chi-square statistic. Max. temp. = Mean Maximum Temperature.

	2012			2013			2014			2015		
	Chi	DF	Pr	Chi	DF	Pr	Chi	DF	Pr	Chi	DF	Pr
Lifeform	33.6	5	< 0.001	62.6	5	< 0.001	124.2	5	< 0.001	6.2	5	0.287
Soil	39.1	5	< 0.001	146.6	5	< 0.001	50.0	5	< 0.001	10.2	5	0.069
Rainfall	65.9	1	< 0.001	17.5	1	< 0.001	4.0	1	0.045	0.0	1	0.925
Max. temp.	0.9	1	0.341	1.9	1	0.172	2.1	1	0.396	0.5	1	0.476

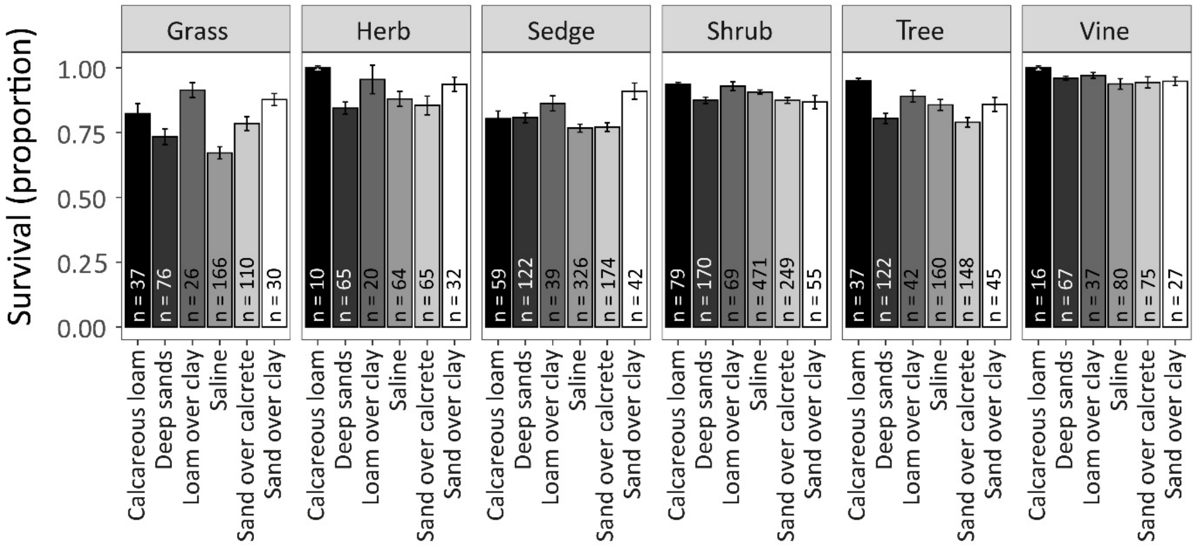

Fig. S1. The arithmetic mean (± SE) for the proportion of surviving lifeforms planted in six soil types pooled across years. The sample size (n) for each lifeform growing on the six soil types is shown.