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Facilitating the restoration of aquatic plant communities in a Ramsar wetland

Running head: Facilitating wetland restoration

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Author contributions

SJ, TT, HS conceived this research; SG, JN designed the survey method and undertook the surveys; SJ, JN undertook the analysis of the data; SJ wrote the manuscript; JN, SG, HS, TT edited the manuscript; SJ completed subsequent revisions to the manuscript

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Abstract

Human activities such as land clearing and intensive land use around water bodies, particularly wetlands, have a detrimental impact on water quality and quantity, aquatic plant communities and associated wetland fauna. Lake Alexandrina and Lake Albert are internationally significant Ramsarwetlands located at the terminus of the Murray River, Australia's longest river system. Agriculture, water regulation and extraction and droughts have had a detrimental impact on native plant communities in the lakes. We studied the influence of young (<1 - 3 years) and old (8 - 11 years) plantings of a native sedge (bulrush), *Schoenoplectus tabernaemontani*, to facilitate the establishment of aquatic plant communitiesin comparison to remnant and control sites. We also measured how planting structure (height, stand width and stem density) changed with age in comparison to remnant sites. Results suggest that as plantings age they get substantially wider and have a greater maximum height, although do not reach similar stand widths by 11 years of age when compared to remnant areas. However, old plantings do not differ from remnant habitats in relation to aquatic plant species richness, counts of aquatic plants and community composition. Young plantings have substantially less abundant and diverse plant communities, but are developing on a similar trajectory to old plantings. It is likely that planting sedges along lake shorelines causes a breakwater effect that facilitatesthe recolonization of wetland plants between the planted area and the water's edge.Management agencies should consider restoring native sedges to increase aquatic biodiversity, and potentially reduce erosion.

Keywords: Restore, wetlandflora, sedge,river club-rush, community composition, planting structure.

Implications:

- The planting of large aquatic plant species, such as sedges,can assist native aquatic plant communities to recolonize lake shorelines, increasing the resilience of Ramsar listed habitats and other wetlands.
- Sedge plantings are likely to provide greater shoreline protection as they get older as plantings expand and become structurally more complex as they age.
- Practitioners trying to restore aquatic plant communities after wetland degradation, as a result of drought or livestock grazing, should seekto create a breakwater effect between the restored area and the shoreline so sediment can settle and other plants can become established in this low energy zone.

Introduction

The health and persistence of wetlands around the world have continually declined as a result of anthropogenic disturbance in human-dominated landscapes, despite global agreements such as the Ramsar Convention (1971) to protect threatened ecosystems(Matthews 1993; Hettiarachchi et al. 2015).Impacts on wetlands from human influences includeurban development and/or agricultural intensification, pollution, structural degradation, vegetation clearance and excessive water extraction (Bond & Lake 2003; Bond et al. 2008; Pander & Geist 2013). These impacts are often exacerbated by natural processes such as long-term droughts(Bond et al. 2008).

Droughts can cause standing water to reduce or dry up and as water levels recede ecologically diverse littoral communities decline or can become locally extinct(Bond et al. 2008). Grazing by livestock usually intensifies this loss of littoral vegetation (Vanderbosch & Galatowitsch 2011; Pander & Geist 2013). Littoral zones are important because they provide a niche for ecologically diverse communities, and littoral vegetation also functions as a natural breakwater to shorelines by decreasing wave attenuation(Augustin et al. 2009; Gedan et al. 2011).Without this, whenwater levels return wave energy can cause shoreline erosion and reduce the likelihood that aquatic plant species will become established (Vanderbosch & Galatowitsch 2010; Gedan et al. 2011), decreasing potential habitats for fish, invertebrates and birds (Vanderbosch & Galatowitsch 2010). Subsequently, aquatic macrophytes such as sedges may not re-establish naturally and thus require interventions such as replanting (Vanderbosch & Galatowitsch 2010; Pander & Geist 2013).

Restoration of physical habitats has often accompanied the degradation of these aquatic systems and has been used widely as a tool to improve water quality andre-establish biological diversity and thus habitat heterogeneity (Bond & Lake 2003) while also having many positive social outcomes (Bernhardt & Palmer 2011; Pander & Geist 2013). However,little research or monitoring exists to show how effective habitat restoration in aquatic systems is for increasing biodiversity(Chapman & Underwood 2000; Bond & Lake 2003; Bernhardt et al. 2007)and previous studies mainly focus on the use of aquatic plantings for wastewater treatment (Tanner 1994; Zhang et al. 2007; Pollard 2010). This can result in expensive restoration activities that may have limited ecological benefits or social outcomes(Bernhardt et al. 2005). Similarly, little is known about the most effectivemethods to re-establish wetland vegetation to maximise survival of native plant communities(Vanderbosch & Galatowitsch 2011). Re-establishing aquatic vegetation is especially difficult in countries such as Australia, where many water bodies are ephemeral and where droughts can persist for long periods of time (Bond et al. 2008).

This research aimed to assess the benefits or negative impacts of planting a native, non-invasive sedge species, *Schoenoplectustabernaemontani*,on the aquatic plant communities of two large,Ramsar listed freshwater lakes, Lake Alexandrina and Lake Albert, in South Australia. Our project aimed to investigate: (i) how the structure of planted *S. tabernaemontani*stands (stand width, maximum and average stem height and stem density) of different ages (young plantings and old plantings) changed over time compared to remnant (naturally occurring) stands, and (ii) the influence of planted *S. tabernaemontani*stands of different ages (young plantings and old plantings) on aquatic plant community composition in comparison to control sites and remnantareas. By making these comparisons we aimed to discover whether young plantings had a different structure to old plantings, how the structure of young and old plantings compared to that of remnant stands, and how the age of *S. tabernaemontani* stands influenced aquatic plant community composition. This information is valuable for determining the effectiveness of planting native sedge species for maintaining and increasing aquatic plant communities in freshwater lakes and associated water bodies.

Methods

Species description

Schoenoplectus tabernaemontani (C.C.Gmel.) Palla (syn. *Schoenoplectus validus* (Vahl) A. Love & D. Love; *Scirpus tabernaemontani* Vahl), locally known as the river club-rush,is a large, perennial, rhizomatous sedge (bulrush) that grows 2 - 3 m in height (up to 5 m in very favorable conditions) in water up to 1.5 m depth (Sainty & Jacobs 2003). It is native to Australia and many other parts of the world. Ecosystem services provided by *S. tabernaemontani* include erosion control, waterbird habitat, fish habitat, sediment and water column aeration and water quality improvement (Sainty & Jacobs 2003). It is a common emergent species around the edges of freshwater and brackish lakes, but unlike other large emergent plant species such as *Phragmitesaustralis* and *Typha domingensis* it does not form dense monospecific stands (Hudon et al. 2005; Gehrig et al. 2012). *Schoenoplectus tabernaemontani* usually grows in deeper water than *T. domingensis* and *P. australis*(Sainty & Jacobs 2003) and is often associated with submergent taxa such as

Myriophyllum spp., *Potamogeton* spp., *Ceratophyllum demersum* and *Vallisneriaaustralis*(Gehrig & Nicol 2010; Nicol et al. 2013).

Site description

Lake Alexandrina and Lake Albert are located in South Australia at the end of Australia's longest river system, the Murray River (Fig 1). The lakes are large and shallow and contain fresh to brackish water. They have been listed, along with the Coorong, as a wetland of international importance under the Ramsar Convention(DEH 2000). The lakes originally joined the Coorong, a shallow saline lagoon stretching for more than 100km along the South Australian coast, and the sea. However, due to excessive water extraction along the Murray River, barrages and locks were erected in the 1930s to maintain water levels and reduce saline water incursions into the lakes. These barriers kept water levels relatively static, influencing the vegetation growing in the lakes and increasing shoreline erosion. Several droughts, increasing salinity and ongoing water extraction along the river system had caused large losses of native plants and animals in the lakes and surrounding water bodies (van Dijk et al. 2013).

The Millennium Drought (1995 - 2009) greatly exacerbated these environmental impacts and also caused dramatic socioeconomic problems. The Millennium Drought was the worst hydrological drought since records began in the late 19th century and caused surface water storages along the Murray-Darling Basin to decline by 73% and flows into Lake Alexandrina to cease(van Dijk et al. 2013). The impact of the drought were arguably most severe in the lower lakes region as water levels in Lake Alexandrina declined by approximately 2m, salinity levels rose to seven times higher than normal (7000 μS/cm) and the exposed lake bed started to release sulfuric acid(van Dijk et al. 2013). These impacts greatly reduced the number of aquatic plants around Lake Alexandrina and Lake Albert, and many of these shoreline communities have not recovered since the water started to return in 2010(K. Strother*pers. comm.*).

*Schoenoplectus tabernaemontani*occurs naturally along the shorelines of Lake Alexandrina and Lake Albert and had been planted by community organisations since 1996 to reduce shoreline erosion (K. Strother*pers.comm.*). After the Millennium Drought the Department of Environment, Water and Natural Resources (DEWNR) continued this work (2011 onwards) as *S. tabernaemontani*could be planted in relatively deep water (up to 1 m depth), did not form dense homogenous stands and was thought to reduce erosion risk and increase shoreline biodiversity (Nicol et al. 2013). Initially *S. tabernaemontani*was propagated by taking remnant plant clumpsinto a nursery, dividing the clumpsinto individual stems and growing them in hessian pots. More recently nurseries germinated the plants from seed whichwere grown in individual pots, ready for transplanting when there were between 8 -15 stems present. *Schoenoplectustabernaemontani*wasthen planted from November to Marcharound the lakes at 1 m intervalsin two offset rows in 50 - 80 cm of water at least 10 m from the shoreline; these offset rows extend for up to 1,000 m adjacent to the shoreline.

We surveyed 11 locations around Lake Alexandrina and Lake Albert between April - May in 2013, 2014 and 2015. At each location there were one or more survey sites and each site consisted of one of four treatment types. These four treatments were: (i) old plantings; (ii) young plantings; (iii) remnant areas and (iv) control areas.In total there were four old plantings;six young plantings;three remnant areas and eight control areas(Table S1, Fig 1). Old plantings were planted 8 - 11 years ago with one sedge species, *S. tabernaemontani,* and were planted as described above. Young plantings similarly contained *S. tabernaemontani* and were planted <1 - 3 years ago. Control sites were areas along the shoreline that had not been planted and lacked large sedge species. Control sites were on a similar aspect and substrate to old and young plantings and were usually more than 200 m away from planted sites. Generally one planted site was paired with a control site, although sites that contained both young and old plantings (Wellington Lodge and NurraNurra) shared a control site (Table S1). Remnant areas contained naturally occurring stands of *S. tabernaemontani*and other sedge species and were generally considered to be 20 years of age or greater.A large proportion of the shorelines around Lake Alexandrina and Lake Albert had been fenced between 2010 and 2012, due to government incentives, to reduce livestock access on exposed sections of the lakebed. Therefore, all of the planted, control and remnant sites were fenced from livestock, reducing the possibility of these areas being disturbed or detrimentally impacted by grazing.

Planting structure

Planting structure in this case is the size, density and height of a distinct patch of planted *S. tabernaemontani*and provides information on the rate of spread and health of the planted area. For this research, we collected data on the stem density, height of the tallest stem and the height of ten randomly selected stems using the method below.To determine the structure of *S. tabernaemontani*stands, we randomly placed five 1 x 1 m quadrats along one 100 m transect at each site of planted or remnant *S. tabernaemontani*(Fig 1, Fig 2). The placement was determined using five random numbers ranging from 1 - 100 (representing the length of a transect) that we randomly generated prior to going into the field. Within each quadrat we measured the stem density (number of stems), height of the tallest stem and the height of ten randomly selected stems. We also recorded thewidth of the *S. tabernaemontani*stand (referred to as 'stand width') at each quadrat by measuring from the outer edge of the quadrattowards the shoreline. Inplanted sections of *S. tabernaemontani*we surveyed along the two planted lines, whereas in remnant areas surveys were undertaken on the stand edge that was furthest from the shoreline.

Community composition

Using methodology developed to assess aquatic vegetation condition (Marsland & Nicol 2008), we assessed community composition in the same 100 m section of shoreline in which the planting structure surveys had been undertaken in planted and remnant *S. tabernaemontani*stands. At control sites a 100 m section of shoreline was randomly selected (Fig 2). Within this 100 m section,three transects were established in the middle and at either end perpendicular to the shoreline. At planted sites we recorded aquatic species that were between theplantings and the shoreline.Along each transect five depths were sampled: +0.8, +0.6, +0.4, +0.2, and 0 m above sea level, based on the Australian Height Datum (AHD). The shorelines of Lake Alexandrina and Lake Albert are approximately 0.75m AHD(Bureau of Meteorology 2015). Lower elevations were not surveyed due to the absence of vegetation. At each depth along each transect, three quadrats were established that were 1 x 3 m in size and separated by 1m (Fig 2). All plants within each quadrat were identified to species level and their percent cover was estimated using a modified Braun-Blanquet scale(Braun-Blanquet 1932).

Data analysis

Planting structure

We analyzed theresponse of *S. tabernaemontani*structureusing several simple linear regressionsfrom the lme4 package (Bates et al. 2014)in R(R Core Team 2014). Each of the four structural attributes: stem density; height of the tallest stem; height of ten randomly selected stems; and stand width were analyzed separately to determine the relationship between plant structure and planting age (young plantings <1 -3 years of age, old plantings 8 - 11 years of age, and remnant areas that were all given an age of 20 years). In order to ensure that individual observations did not influence our results we used Cook's distance (Cook 1977) to examine the influence of outliers on regression coefficients, ensuring observations did not have a Cook's distance > 1.0. The data from the three years of sampling were combined for this analysis and results were presented using ggplot2 (Wickham 2009) and included 95% confidence intervals (CI) of model coefficients.

Community composition

We used R (R Core Team 2014) and lme4(Bates et al. 2014)to perform a generalized linear mixed model with a Poisson distribution to analyze the relationshipbetween treatment type and species richness and counts of aquatic plants. A Poisson distribution was selected because our graphed data were consistent with the dispersal of count data(Hilborn & Mangel

1997). The 2.5th and 97.5th percentiles of the distribution were computed to provide 95% CI for model coefficients. In this model we used treatment type (young planting, old planting, remnant or control) as a fixed effect with a random effect of site. Data were combined over the three years of sampling for each site. Residual plots did not reveal any obvious deviations from normality. Species richness was the total number of species recorded in between 0.6 - 0 m AHD at each site, while counts of aquatic plantswere analogous to abundance and was calculated as the average number of each species recorded in between 0.6 - 0 m AHD at each site. To maximize the likelihood that the species richness and counts of only aquatic plants was analyzed, we excluded species recorded at 0.8 AHDto reduce the number of terrestrial species.

It was considered that independent variables would display 'some evidence' of a difference if the proportion of CI overlap was no more than half the average length of the two overlapping arms ($p < 0.05$), and the two overlapping arms did not differ in length by more than a factor of two(Cumming & Finch 2005). Independent variables were considered to display 'quite strong evidence' of a difference if there was no overlap, or a gap between confidence intervals ($p < 0.01$) (Cumming 2009; 2012).

In order to determine whether community composition of aquatic plant species was a function of site and treatmenttype (young planting, old planting, remnant and control), we used a cluster analysis in R (R Core Team 2014)using the vegan package(Oksanen et al. 2014). The species matrix was constructed using the total cover of each species recorded at a site. The cluster analysis included a Bray-Curtis dissimilarity measure, a widely used tool for analyzing community assemblage data, and a McQuitty similarity analysis(Crawley 2012). We then used the metaMDS function in vegan to undertake non-metric multidimensional scaling (nMDS)(Anderson & Ter Braak 2003).Similarly, this used a Bray-Curtis dissimilarity with 1,000 random starts, ending after two runs. An examination of the stress values showed that a two dimensional axis was sufficient to achieve low levels of stress for aquatic plant community composition (stress = 0.132).The indicator value for each species in a cluster was calculated using the indvalfunction (Dufrêne & Legendre 1997) in the labdsv package (Roberts 2013).

Results

Planting structure

Analysis indicated a substantial increase in the stand width of *S. tabernaemontani* as plantings aged ($R^2 = 0.49$, $p = <0.001$, Fig 3a). We found that young plantings had a significantly lower stand width than old plantings, and that remnant sites were substantially wider than either young or old plantings. Similarly, the maximum height of *S. tabernaemontani*stemswas greatest in remnant areas ($R^2 = 0.24$, $p = <0.001$, Fig 3b). While young plantings generally had lower maximum stem heights than old plantings or remnant areas, plantings around three years of age were more similar to old plantings than to other young planting sites. Eleven year old plantings also had a similar maximum stem height to remnant areas. The average height of *S. tabernaemontani*stems and stem density did not substantially differ between planted stands of different ages or planted stands and remnant areas (Table S2).

Community composition

Our results suggested that species richness and counts of aquatic plants in control sites and young plantings were considerably lower than old plantings and remnant areas (Fig 4a, 4b). Aquatic plant species richness showed some evidence of being lower in control sites compared to old plantings (mean expected difference 0.71, 0.7 - 0.74 95% CI) and quite strong evidence of being lower in control sites and young plantings compared to remnant sites (control sites - mean expected difference 1.06, 1.02 - 1.1 95% CI; young plantings - mean expected difference 0.88, 0.86 - 0.89 95% CI). Counts of aquatic plants also showed quite strong evidence of being lower in control sites and young plantings compared to old plantings and remnant sites (control sites and old plantings - mean expected difference 1.18, 1.17 - 1.18 95% CI; control and remnant sites - mean expected difference 1.51, 1.27 - 1.76 95% CI; young and old plantings - mean expected difference 0.99, 0.98 - 1.00 95% CI; young plantings and remnant sites - mean expected difference 1.33, 1.08 - 1.59 95% CI) (Fig 4a, 4b,Table S3, Table S4). Variance for the random effect of site was small for both species richness (variance = 0.01) and counts of aquatic plants (variance = 0.06).

Community composition clustered into two main groups (Fig S1). The first cluster was composed of the control sites and young plantings, and one of the old plantings (NurraNurra). The second group was composed of remnant sites (Hindmarsh Island, Bremer River and Loveday Bay), old plantings (Dumandang, Raukkan and Wellington Lodge) and one young planting (Meningie Foreshore).Community composition of remnant areas was more similar to old plantings than either young plantings or control sites under a two-dimensionalsolution (stress = 0.132, Fig 5). There was a gradient of community composition complexity in the sites that we surveyed that ranged from common wetland plants in control sites and young plantings,to old plantings and remnant stands of *S. tabernaemontani* that contained a more diverse array of aquatic plant species (Fig 5, Table 1). This is supported by our analysis of the plant species most commonly associated with these clusters. This indicatedthat aquatic plants associated with early successional stages were found in the control sites and young plantings (cluster 1). These included semi-aquatic species such as *Crassulahelmsii, Juncususitatus*and *Limosellaaustralis*(Table 1). Plants associated with more established aquatic plant communities were found in old plantings and remnant areas (cluster 2) and consisted of species favoring calm water environments such as *Azolla filiculoides, Myriophyllum salsugineum* and *Rumex bidens*(Fig S1, Table 1).

Discussion

This study is novel in its research into planted sedge species and their importance in degraded wetland environments. Importantly, we found evidence that*Schoenoplectus tabernaemontani*allows otheraquatic plant species to colonize the zone between the planted area and the shoreline. This occurs quite rapidly, and after 11years the community composition in this zone starts to resemble remnant stands that have been established for 20 or more years. Planted *S. tabernaemontani* stands also expand and multiply over time, and the widths of these stands are likely toexpand exponentiallywhere suitable habitats exist as time since planting increases. However, linear plantings of *S. tabernaemontani*are unlikely to gain the width of remnant stands after 11 years of growth, even though the maximum height, average height and density of stems after the first few years of establishment is likely to be similar to remnant stands.

Planting establishment and expansion

Our results indicated that *S.tabernaemontani*established relatively rapidly after planting, possibly because they were planted at a consistent depth (50 - 80cm) over the summer months (November to March). This is consistent with other research on *Schoenoplectus* spp., which suggested that the timing of planting and planting depth were important factors to consider when restoring these sedge communities (Tanner 1996; Vanderbosch & Galatowitsch 2010; 2011).High grazing pressure is also known to have a detrimental impact on the establishment of *S. tabernaemontani*(Tanner 1996)and persistence of soil seed bank(Erkkilä 1998), although this species may be able to re-establish after moderate grazing (Hayball & Pearce 2004). All of our sites were fenced from livestock, reducing the likelihood that these plantings would be disturbed or detrimentally impacted (Vanderbosch & Galatowitsch 2010).

The expansion of planted *S. tabernaemontani*suggests that restoration activities that include the planting of this species are likely to provide shoreline protection in a relatively short amount of time, depending on the environmental factors present at a site. For example, Tanner (1996) found that *S. tabernaemontani*had highly seasonal growth patterns with long periods of active growth compared to other aquatic plant species. Other studies havesuggested that environmental factors likely to influence reed bed expansion include seasonal fluctuations in water levels and nutrient influxes from the lake bottom and surrounding river systems(Liira et al. 2010; Catford et al. 2014). However, based on the response of other wetland plants and other *Schoenoplectus* species to seasonal water fluctuations and nutrient increases (Escutia-Lara & Lindig-Cisneros 2012; Vivian et al. 2014), it is likely that planted and remnant *S. tabernaemontani*stands would increase in size and extent if lake flows were more variable. A study by Catford et al.,(2014) supports this, suggesting that hydrological regulation in the Murray River system limits the establishment of native plants adapted to variable flows while benefiting more invasive species.

It is unclear from our study how competition from other rapidly growing species such as *P. australis*and *T. domingensis*would affect the establishment of new stands of *S. tabernaemontani*or the potential expansion of planted stands(Tanner 1996; Sainty & Jacobs 2003; Rogers & Ralph 2011). For example, *P. australis*is known to expand rapidly in lake environments at a rate of 2.2 m per year (Liira et al. 2010). However, increased water depth is likely to be a limiting factor in the establishment of *P. australis*,as submergence is known to limit its growth (Mauchamp et al. 2001; Hayball & Pearce 2004), whereas *S. tabernaemontani*prefers permanent inundation and deeper water environments (Hayball & Pearce 2004). Therefore, if *S. tabernaemontani*is either planted or naturally establishes in deeper water (50 - 80 cm), it is likely to outcompete faster growing species such as *P. australis*and *T. domingensis*(Sainty & Jacobs 2003).

While *S. tabernaemontani*appears to be relatively resilient to drought conditions, having survived the Millennium Drought and subsequent low water levels byre-sprouting from rhizomes after normal water levels returned to Lake Alexandrina and Lake Albert (Nicol et al. 2013), a variety of factors may influence the persistence of this species. These include nutrient increases (Escutia-Lara & Lindig-Cisneros 2012), droughts and subsequent wetland drying (Catford et al. 2014) and access to the shoreline by livestock and livestock grazing(Vanderbosch & Galatowitsch 2010). Greater research into how factors such as climate variability, water levels, soil nutrients and lake substrates influence the establishment of sedge species may increase our ability to more effectively restore important wetland systems.

Planting facilitated diversity

This study suggests that as planted *S. tabernaemontani*stands age, species diversity of associated littoral aquatic plants is likely to increase as time since planting increases. This is supported by other studies that show that plant - plant facilitation can reduce abiotic and biotic stresses in restored habitats, increasing plant survival, recruitment and growth (Zhang & Shao 2013). In our study it is most likely that planted *S. tabernaemontani*caused a breakwater effect, reducing wave energy into the shore andallowing suspended sediment to settle between the planted vegetation and the shoreline (Gedan et al. 2011; Fairweather et al. 2013). This then provided a low energy environment for aquatic plantsto recolonize(Vanderbosch & Galatowitsch 2010; Nicol et al. 2013).

The species we found associated with control sites and young plantings versus old plantings and remnant areas supports this interpretation. For example, many of the plant species associated with old plantings and remnant areas such as *Typha domingensis, Azolla filiculoides, Myriophyllum salsugineum*, *Ceratophyllum demersum* and *Rumex bidens* require calm water to become established(Romanowski 1998; Sainty & Jacobs 2003; Berkinshaw 2009). Similarly, *Calystegia sepium* requires larger plants to twine around, so it is likely to be recorded in older wetland communities, while *Hydrocotyle verticillata* prefers calm shallow water environments with a silty substrate (Sainty & Jacobs 2003). These calm water environments are most likely to occur behind established wetland communities (Berkinshaw 2009) such as those reported here.

In contrast, aquatic plants associated with young plantings and control sites were usually semi-aquatic, and were more adapted to open habitats(Sainty & Jacobs 2003). The high incidence of open water in these habitats also suggests that it takes longer than three years for *S. tabernaemontani*plantings to form a breakwater effect (Nicol et al. 2013). Thus, old plantings are able to create a low wave energy environment between *S. tabernaemontani*and the shoreline that allows native emergent, submergent, floating and amphibious species to become established (Sainty & Jacobs 2003; Nicol et al. 2013). Although young plantings (<3 years) appear to be on a trajectory to providing adequate habitat, it may take up to 8 - 11 years before these habitats start to resemble remnant areas.

Two of the study sites we sampled did not follow the same trajectory as other sites in similar age-classes. The young planting at Meningie Foreshore contained plant communities more similar to old plantings and remnant areas than control sites or other young plantings. There are two possible reasons for this. Firstly, MeningieForeshore was planted adjacent to an urban area and is likely to have benefited from increased nutrient rich runoff from fertilized recreation areas and private lawns (Tanner 1994; Zhang et al. 2007). Secondly, it is one of the older sites in the young planted treatments (3 years old), allowing greater time for other plants to recolonize the area between the plantings and shoreline. The old planting at NurraNurra(9 years old) showed the opposite effect, whereby its associated plant community was most similar to young plantings and control sites. This site, as with other old plantings and remnant areas, survived and regrew from rhizomes after the Millennium Drought (K. Strother, *pers. comm*.). However, it is thought that NurraNurra remained more exposed during the drought than either Raukkan, Wellington Lodge or Dumandang and thus is taking a longer time to recover (J. Nicol *pers. comm.*). Although wind-caused seiches may initially influence plant growth and establishment, wind direction and intensity varies (Bureau of Meteorology 2015), making it unlikely that certain sites would be more affected than others.

There are likely to be other benefits of planting *S. tabernaemontani*along shorelines that were not reported in this study. Anecdotal evidence suggests that *S. tabernaemontani*plantings reduce shoreline erosion(Nicol et al. 2013), and this is supported by other research that shows wetland vegetation is vital in reducing wave attenuation(Gedan et al. 2011). This protects shorelines from erosion resulting from wave damage and increases sediment build-up between wetland vegetation and the shoreline (Gedan et al. 2011). Studies also suggest that old plantings provide adequate habitat for a variety of native fish species as well as mobile invertebrates, rather than planktonic species that are common in open water habitats (Fairweather et al. 2013). Finally, planting species such as *S. tabernaemontani* is likely to substantially increase carbon uptake in aquatic systems, subsequently reducing the risk of acid sulfate soils during drought years(Moore 2013).

As other studies have indicated, monitoring of restoration projects is usually minimal(Chapman & Underwood 2000), with a large-scale study of river restoration projects in the USA finding only 10% undertook any assessment or monitoring of their restoration activities (Bernhardt et al. 2005). This lack of restoration monitoring may result in a substantial expenditure of funding and resources without the expected benefits of biodiversity or social outcomes. In order to gain a better understanding about how to undertake wetland restoration effectively, and the benefits this restoration has for wetland functionality, a greater amount of funding needs to go into researching and monitoring wetland habitats. This may include increasing links between research institutions, government agencies, community groups and natural resource managers in order to achieve meaningful and cost-effective conservation outcomes.

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Table 1. The aquatic plant species associated with each cluster group: Cluster 1 - young planting, control and one old planting (NurraNurra), Cluster 2 - Remnant sites, old plantings, one young planting (Meningie Foreshore). * Introduced species, V = vulnerable (National Parks and Wildlife Act 1972, Schedule 8), R = rare (National Parks and Wildlife Act 1972, Schedule 9).

Species	Cluster	Indicator Value	Probability
Open Water	1	0.539	0.095
Cenchrus sp. *	1	0.163	1
Isolepisproducta(V)	1	0.143	0.522
Nitella sp.	1	0.083	1
Cotulacoronopifolia	1	0.076	1
Crassulahelmsii	1	0.071	1
Limosellaaustralis	1	0.071	1
Juncususitatus	1	0.071	1
Calystegia sepium	2	0.852	0.001
Typha domingensis	2	0.829	0.001
Schoenoplectus tabernaemontani	2	0.731	0.003
Azolla filiculoides	2	0.714	0.004
Paspalumdistichum *	2	0.671	0.024
Myriophyllum salsugineum	2	0.668	0.008
Phragmitesaustralis	2	0.607	0.042
Ceratophyllum demersum(R)	2	0.571	0.006

*Ranunculus trilobus**	2	0.571	0.004
Rumex bidens	2	0.562	0.004
Trifolium sp. *	2	0.429	0.026
Persicarialapathifolia	2	0.429	0.035
Berulaerecta *	2	0.429	0.038
Mentha sp.	2	0.429	0.026
Lemna sp.	2	0.429	0.028
Hydrocotyle verticillata	2	0.429	0.029
Bolboschoenuscaldwellii	2	0.388	0.046

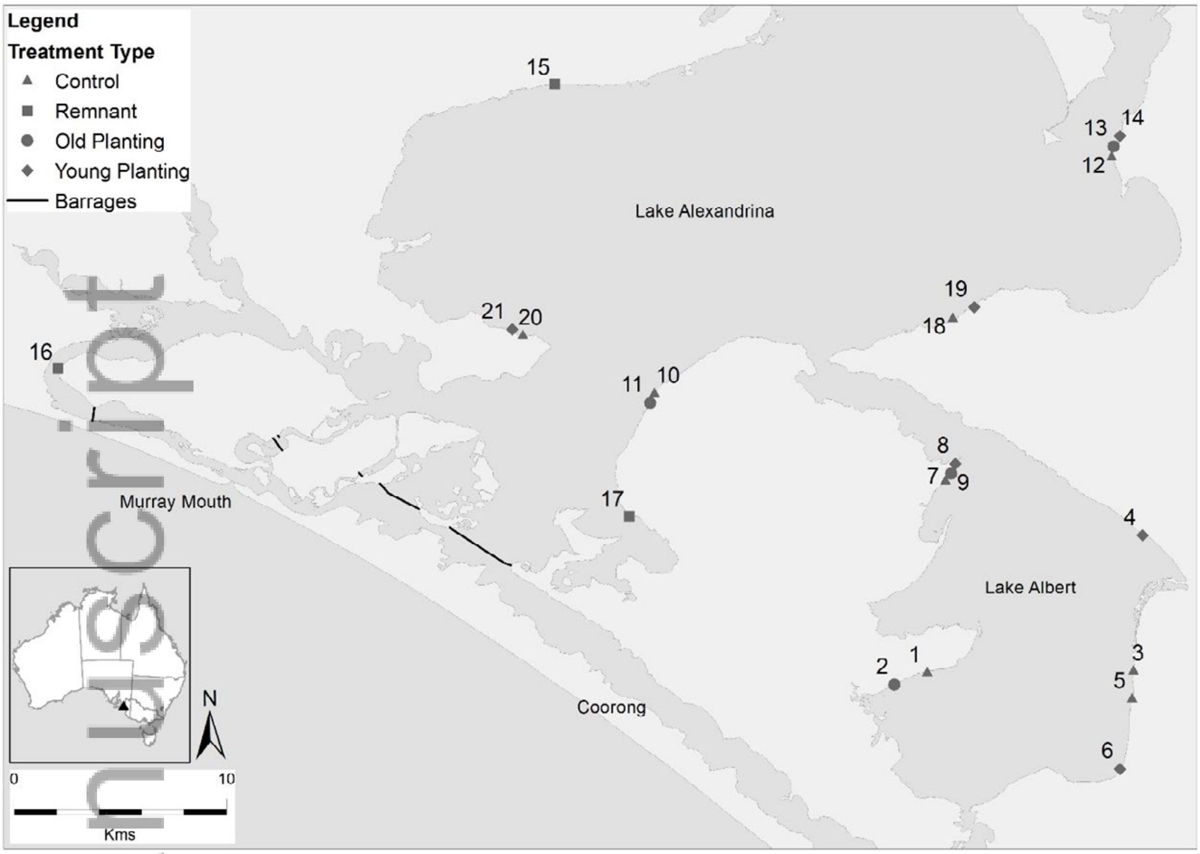


Fig 1. Sites surveyed between 2013 - 2015 in Lake Alexandrina and Lake Albert, South Australia. Numbers represent site treatment IDs (Table S1)

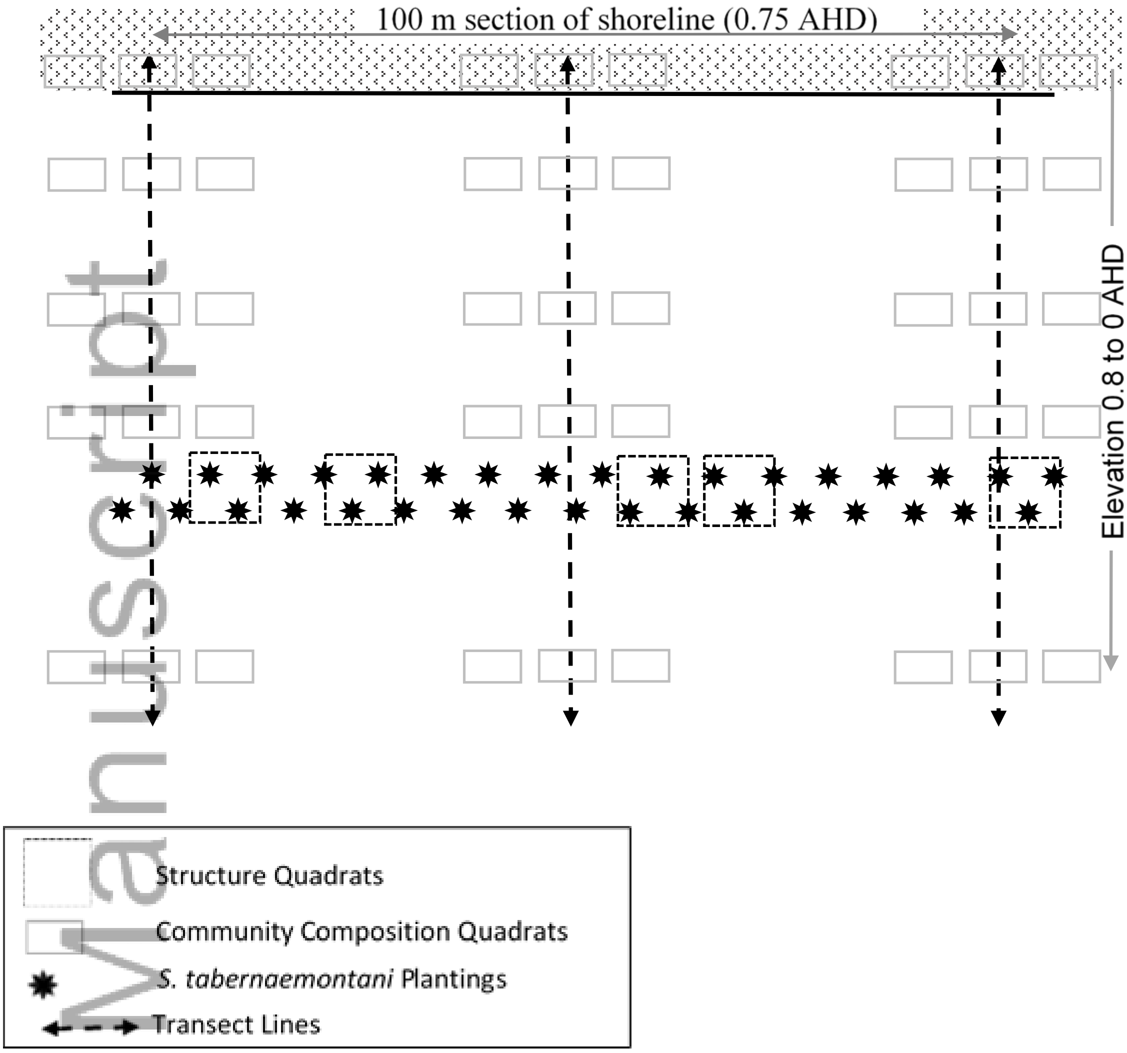
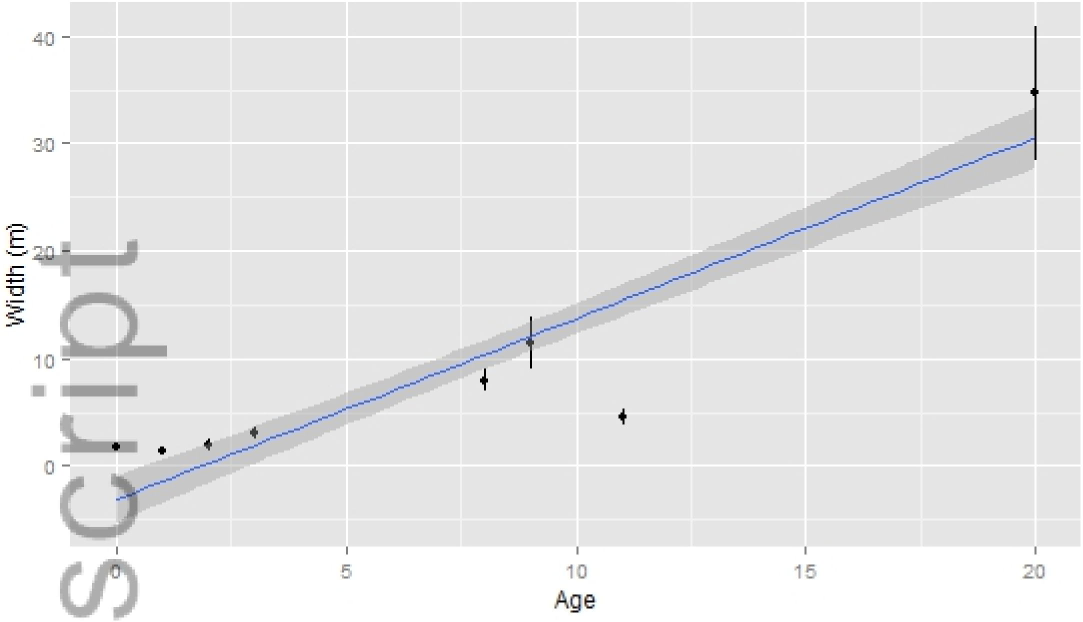
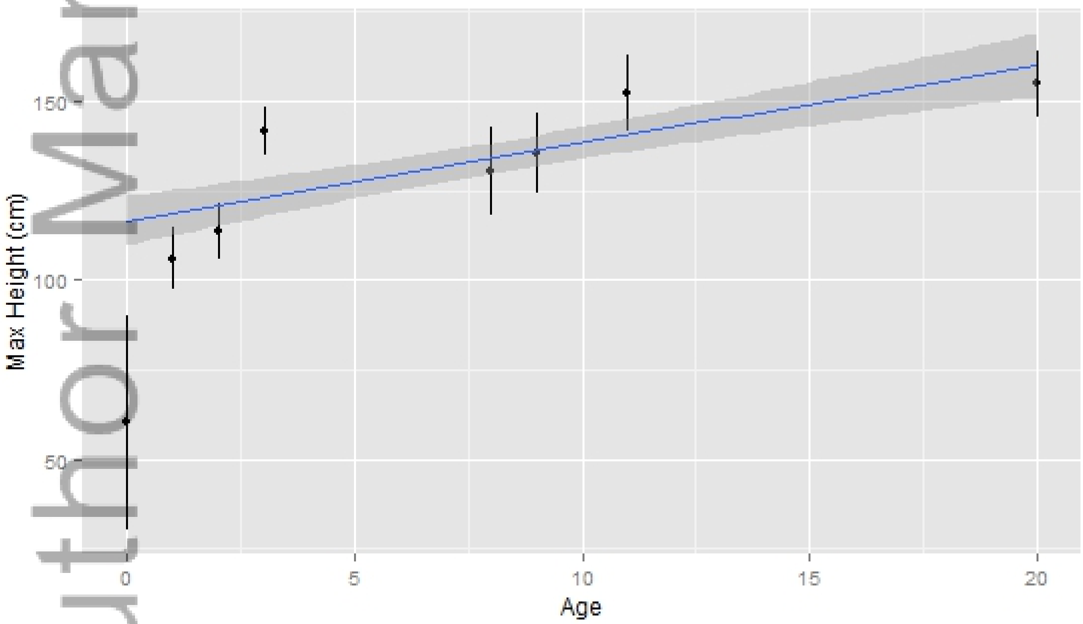


Fig 2. Layout of quadrats to determine the community composition of planted *S. tabernaemontani*sites, remnant *S. tabernaemontani*sites and control areas. Structure quadrats were also undertaken in planted and remnant *S. tabernaemontani* sites.



Width (m)
40
30
20
10
0
0
5
10
15
20
Age

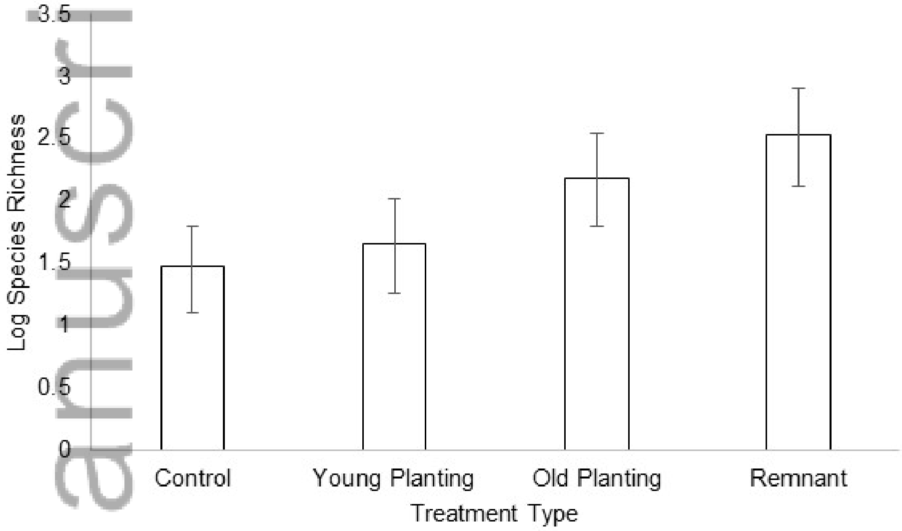
a)



Max Height (cm)
150
100
50
0
5
10
15
20
Age

b)

Fig 3. The influence of planting age (young plantings <1 - 3 years of age, old plantings 8 - 11 years of age, and remnant areas that were all given an age of 20 years) on *Schoenoplectus tabernaemontani*(a) stand width and (b) maximum stem height. Bars represent ±95% confidence intervals. Shaded band around regression line represents a 95% confidence region.



a)

b)

Fig 4. The log species richness (a) and counts of aquatic plant species (b) found in different treatment types around Lakes Alexandrina and Albert. Bars represent ±95% confidence intervals.

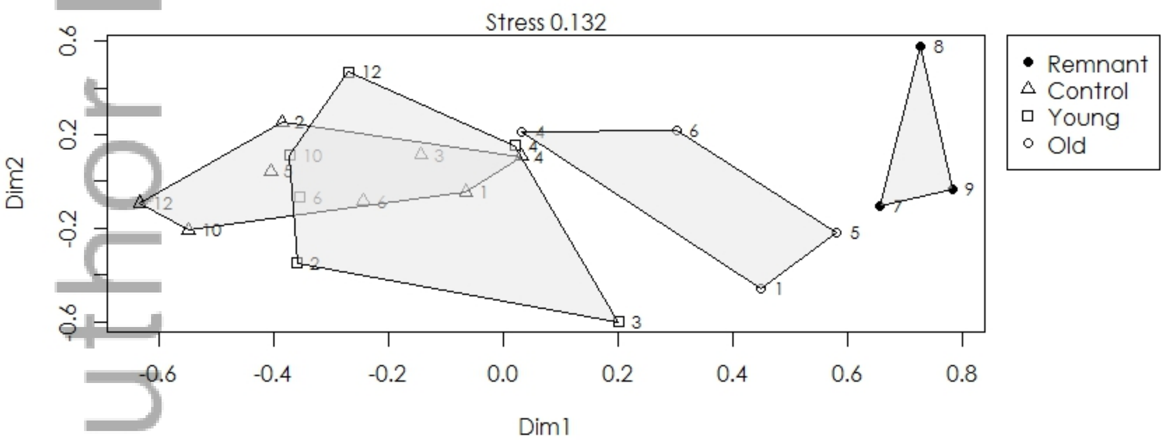


Fig 5. A two dimensional plot of aquatic plant community composition in the four treatments surveyed. Numbers represent Site IDs (Table S1).