Minimizing the opportunity costs of carbon mitigation on farmland via restoration of salinized soils

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Abstract

Mass conversion of native vegetation to agricultural land-use triggered secondary salinity, a hydrological imbalance, which has damaged more than 1.75 million ha of farmland in south-western Australia. Various types of reforestation have been proposed and tested to restore the hydrological balance, however the economic returns from these cannot compete with existing farm practice and land-holders thus have a reluctance to adopt. An alternative approach has been to reforest abandoned saline areas with salinity and/or water-logging tolerant trees to avoid displacement of farming activities and to minimize opportunity costs.

We propose an improved evaluation methodology for saline site reforestation and in this paper test its feasibility. Three eucalypts species (*Eucalyptus salubris*, *E. camaldulensis* and *E. sargentii*) were planted adjacent to a salt scald in Wickepin, Western Australia, and their survival and growth were monitored under increasing soil salinity ranging from moderate (ECe: 400-800 mS m^{-1}), high (800-1,600 mS m^{-1}) or extreme (>1,600 mS m^{-1}) with shallow (< 1 m depth) saline ground water (EC: 2,000-3,000 mS m^{-1}). Survival and growth of *E. sargentii* and *E. salubris* in the saline discharge areas were comparable to those in the arable area, and reforestation by these species can thus avoid land competition and opportunity costs.

1. Introduction

The extensive dryland farming area of southern Australia has been damaged by secondary salinity (Harper et al. 2021), with estimates that 1.75+/-0.34 million ha has been affected in Western Australia, alone (Cacetta et al. 2022). Secondary salinity has been triggered by the extensive deforestation of native vegetation for agriculture which has resulted in a hydrological imbalance through rising and increased discharge of saline ground water (Turner and Ward, 2002). Ground water recharge under native forest, was normally around 0 mm $year^{-1}$ with a maximum of 5 mm $year^{-1}$ (Dunin, 2002), but after deforestation such recharge increased markedly (George et al., 1997). This has resulted in an increase in pressure in groundwater systems contained within the deep regolith, and the dissolution of stored salts. These salt stores have accumulated over long periods in the regolith and are estimated to be from 50 to 5000 Mg ha^{-1} (McFarlane and Williamson, 2002). The discharge of the saline groundwater in lower parts of the landscape results in a decrease in crop yields, or in extreme cases remnant vegetation death (Turner and Ward, 2002) and the abandonment of farmland.

Various measures to manage secondary salinity have been proposed and investigated, with reforestation across the farmed landscape a major methodology (Lefroy and Stirzaker, 1999; Harper et al. 2017) in an attempt to restore the landscape's hydrological balance. Such displacement has a lower impact on the surrounding environment than civil engineering approaches such as drainage. Drainage has been shown to be ineffective in this region due to both the volumes of water that need to be removed and the cost of establishment (Horn, 2017). Various estimates of the amount of farmland that would need to be displaced by reforestation to resolve the water imbalance have been reported in Western Australia. For example, 24% (Lefroy and Stirzaker, 1999), 16% (White et al., 2002), 22% (Dunin, 2002), and at between 16 and 80% (George et al., 2001).

To treat secondary salinity, therefore, there are various difficulties to be overcome (Lefroy and Stirzaker, 1999; Flugge and Abadi, 2006). Among these difficulties, the largest is considered to be the opportunity costs associated with the displacement from farmland by reforestation, *e.g.* farmers no longer earn income from crop production or pasture grazing from the displaced farmland. When analysed, the business-as-usual scenario brought much larger profit to land-holders over more than half century compared with reforestation scenarios, and this resulted in a low incentive for land-holders to adopt such measures (Flugge and Abadi, 2006). To solve this challenge, the development of approaches that minimize such opportunity costs and thus increase the incentives to land-holders is necessary.

In previous studies, compensation of opportunity costs by utilizing carbon mitigation investment was proposed. Approaches included reforestation as a secondary salinity measure as a co-benefit of carbon sequestration projects (Harper et al., 2007) or woody biomass obtained from reforestation as a bio-energy source (Sochacki et al., 2007; Harper et al., 2010, 2014). However, the reduction of opportunity costs is strongly dependent on the value of carbon credits, bio-energy (Flugge and Abadi, 2006), the value (*e.g.* rent) of the land being treated, and thus realization of such measures fully depends on external factors. The externalities associated with salinization of landscapes, such as the loss of biodiversity or damage to built public infrastructure also need to be taken into account.

To reduce opportunity costs to nearly zero without depending on external factors, reforestation in abandoned farmland damaged by secondary salinity (*e.g.*, marginal area of salt scalds and saline seeps), has been investigated in south-western Australia. From these research trials, available woody species with saline tolerance and relatively high growth were screened, and appropriate planting density (trees ha^{-1}) were investigated for several species (Archibald et al., 2006; Biddiscombe et al., 1985, 1989; Zohar et al., 2010; Harper et al., 2012; Sochacki et al., 2012). These studies suggested that reforestation of salinized lands, both as a treatment of secondary salinity measure and to mitigate carbon emissions would reduce the opportunity costs substantially and be highly feasible. Most of the previous studies, however, did not compare biomass production between such saline reforestation and farmland displacement, and also did not consider the risk of failure. In addition, some studies did not evaluate biomass or carbon sequestration yields but only height or crown volume of trees. The water use of trees is directly related to their growth rate in water-limited environments even under saline conditions (Hatton et al., 1998; Morris and Collopy, 1999; Benyon *et al*., 2001), and thus evaluation of biomass productivity is essential.

To make improved secondary salinity treatment measures that resolve water imbalance, mitigate carbon and also reduce opportunity costs to nearly zero we investigated the following:

1: Selection of woody species, with high salinity and water-logging tolerance, with comparable survival and growth in non-saline areas.

2: Exclusion of woody species which cannot survive with shallow ground water and/or highly saline conditions, so as to minimize the risk of failure of reforestation.

3: Evaluation of the biomass productivity of the selected woody species.

2. Materials And Methods

2.1. Research area

The research area is on the Tremayne farm in the Shire of Wickepin located approximately 200 km south-east from Perth, the capital of Western Australia. This area has a typical semi-arid warm Mediterranean climate with a seasonal drought (October to May) and winter rain (June to September) (Environment Australia, 2000). Mean, maximum and minimum values of annual rainfall for the past 30 years (1986–2015) were 375 mm (*S.E.* = 12.2 mm), 501 mm (2011) and 236 mm (2010), respectively (Australian Bureau of Meteorology data (weather station No. 10654; http://www.bom.gov.au/climate/data/stations/). The mean annual rainfall during the observation period (2006–2015) was similar (366 mm, *S.E.* = 26.2 mm), and the maximum and minimum values were the same.

The topography of this area is gently undulating with a landscape of low relief formed on the Archaean rocks of the Yilgarn Craton, these being deeply weathered in the Tertiary, with subsequent erosion and deposition (Environment Australia, 2000). Salt accumulated in the deep weathering profiles and had been subsequently been mobilized following the disruption of the hydrological balance caused by deforestation for agriculture.

The land-use of the research area is the typical dryland-farming in south-western Australia with rotation cropping of wheat or canola and pasture grazing.

2.2. Reforestation sites

A salt scald of about 0.8 km^2 (117º40’03.02”E, 32º44’03.67”S) previously reported by Sochacki et al. (2012) was selected. This area of the farm was formerly arable but has been salinized since farm development. Its surrounding area (more than 1 km^2) had been abandoned as a result of secondary salinity. In 2006, within this abandoned marginal area, two experimental sites of about 0.2 ha each were established in the upstream (117º40’22.14”E, 32º44’13.77”S) and downstream (117º39’59.13”E, 32º43’15.53”S) areas, which were named “Upper salt scald site” and “Lower salt scald site”, respectively. As a control, one experimental site of about 0.15 ha was established in an upper slope of the adjoining arable area (117º40’47.85”E, 32º43’59.20”S), which was named “Arable area site”. This site is 27 m higher in elevation compared to the sites in the valley floor. By comparing tree survival and biomass between the salinized and arable sites appropriate tree species can be selected.

The ground water level of the Upper & Lower salt scald sites was shallow (0–1 m deep), but frequently rose to the ground surface after sustained rainfall. In addition, the ground water level observed in 2007 and 2016 at each site had not changed. Thus, the soil moisture content at 1 m depth at each site was saturated, or moistened through capillary rise throughout the year. Ground water at the Upper & Lower salt scald sites was extremely saline with EC values measured during March to September in 2007 of 2,760-2,960 mS m^{-1} and 1,930-2,920 mS m^{-1}, respectively.

On the other hand, in the Arable area site the ground water table was estimated at about 5 m depth, which was estimated from the terrain and ground water table of monitoring bore about 300 m distant. Soon after establishment of the experiment, the soil moisture content at 1 m and 2 m depth was 10%, and ground water never reached the ground surface.

Soil salinity of each site was measured in 2006 (soon after establishment of the trees) and 2015 or 2017 from collected soil samples. Soil samples were collected at 10 cm intervals from the surface to 50 cm deep by a corer. Samples were taken from 5 points in the Arable site (center of site and center of four sides). The Upper & Lower salt scald sites both had a slight slope (less than 1 degrees), and samples were taken in a center line along this slope. Soil samples were collected from 6 points in the Upper salt scald site and 9 points in the Lower salt scald site. Collected samples were, air dried, and values of $EC_{1:5}$ (mS m^{-1}) were measured according to the 1:5 (soil : water suspension) method (Rayment and Higginson, 1992). Measured values of $EC_{1:5}$ were transformed into ECe (mS m^{-1}) using the following equation from George and Wren (1985):

$$ECe = (364/SP) \times EC_{1:5}$$

where SP denotes saturation percentage of the soil. The observed porosity of the soil samples varied from 30–40% and thus 35% was used as a SP value in this study.

2.3. Reforestation experimentation

Tree seedlings were in June 2006 planted in Arable area site (0.15 ha), Upper salt scald site (0.21 ha) and Lower salt scald site (0.22 ha). Planted tree seedlings consisted of three tree species: *Eucalyptus salubris* F. Muell. (gimlet), *E. camaldulensis* Dehnh. (river gum) and *E. sargentii* Maiden (salt river gum). *E. salubris* is widespread across dry and semi-arid areas of south-western Australia (Gosper et al., 2013) and a native species of Avon Wheatbelt bio-region (FloraBase: https://florabase.dpaw.wa.gov.au/). Thus, *E. salubris* was regarded as a control species in this experimentation. *E. camaldulensis* is regarded as a primary candidate for remediating saline discharge areas in landscapes of low relief where shallow ground water has risen close to the soil surface under semi-arid conditions (Bush et al., 2013). In addition, with reports of its high saline and water-logging tolerance (Marcar et al., 1995) and reforestation success under highly saline conditions (more than 500 mS m^{-1} of $EC_{1:5}$ value) (Morris and Collopy, 1999), this species were selected although it was not a native species in this bio-region (FloraBase). *E. sargentii* was a native species in this bio-region (FloraBase), with a higher drought- and salt-tolerance than *E.*

camaldulensis (Marcar et al., 1995) and reports of reforestation success in saline seep areas (Biddiscombe et al., 1985, 1989).

Effects of provenance and/or seedlot of elite trees of *Eucalyptus* species on seedling growth and survival under various conditions are well known and reported (*e.g.* Marcar et al., 2002; Zohar et al., 2010; Bush et al., 2013), however because of seedling availability from the local nursery, selected lines were not used in this experimentation.

In each reforestation site, four types of planting treatment were applied: direct planting, ripping + mounding, hole drilling and pipe setting. At right angle to the contours at each site, paralleled line blocks with a 2 m interval were set, and the four planting treatments were allocated to random line blocks. Each treatment was replicated 3 times. A set of three neighboring plants along a line block formed a species replicate with 2 m planting interval, and each replicate was allocated randomly within each line block. Among the four planting treatments, two (hole drilling and pipe setting) were excluded from further investigation because they failed.

Seedlings that died within one year after planting were replaced by new seedlings, which would be used for making allometric equations. Such replaced seedlings were excluded from further investigation (*i.e.*, not in use for calculations of survival ratio and biomass).

2.4. Evaluation of tree survival and biomass

In September 2007, surviving trees were counted, and in September 2015, surviving trees were counted and their tree height and trunk diameter at 0.3 m height were measured. Percentage survival was calculated based on this tree counts.

For biomass calculation, allometric equations of each tree species were developed from destructive sampling of aerial parts cut from replaced trees or trees from other reforestation sites inside the catchment. Form of allometric equations was a power function with tree parameters D^2H or H as independent variables and dry-biomass (W_{s+b} or W_l) as dependent variables (Table 1). D, H, W_{s+b} and W_l denotes diameter [m] at 0.3 m high, tree height [m], stem and branch biomass [kg] and leaf biomass [kg], respectively. From the 2015 tree measurements and allometric equations in Table 1, mean tree biomass [kg tree^{-1}] was calculated. Equations of their independent variables as D^2H (CAM1, CAM2, SAL1, SAL2, SAR1 and SAR2) were used for biomass calculation of large trees, and those as H (CAM3, CAM4, SAL3, SAL4, SAR3 and SAR4) were used for small trees.

By comparing significant difference of survival ratio and mean tree biomass among three experimental sites within each tree species, and when such data in Upper & Lower salt scald sites had no significant difference with those in Arable area site, such tree species meets objective of this study.

Tukey's multiple comparison test of population ratio ($\alpha = 0.05$) was applied to evaluate tree survival. This can test significant differences of survival values and then rank them.

For evaluation of biomass difference, firstly ANOVA ($\alpha = 0.05$) was adopted and only after significant difference of mean tree biomass was found, Tukey's multiple comparison test of population mean ($\alpha = 0.05$) was used. If homoscedasticity of grouped data was denied by Bartlett test, Steel-Dwass multiple comparison method (nonparametric rank test) was applied instead of this Tukey's test.

Details are described in section 3.2, but there was a case that only a single tree survived in the Lower salt scald site. In such case, biomass difference of that species in Arable area site and Upper salt scald site were tested by Student's one side *t*-test ($\alpha = 0.05$).

At last, among selected tree species and planting methods in Upper & Lower salt scald sites, each survival and mean tree biomass was compared by using above mentioned statistical analyses.

3. Results

3.1 Site salinity

Figure 1 shows measured ECe values in each site, and changes of ECe from 2006 to 2015 or 2017. During the period of observation in this study, the Arable area site was free from salinity damage, on the other hand, salinity increased at both the Upper & Lower salt scald sites. At the start of the experiment, ECe values of both sites were from 400 to 800 mS m^{-1} categorized as moderate soil salinity (Marcar and Crawford, 2004). But after only about 10 years, the salinity of the Upper & Lower salt scald sites became severe and was categorized as high (800-1,600 mS m^{-1}) and extreme (more than 1,600 mS m^{-1}) soil salinity (Marcar and Crawford. 2004) in most measurement depth.

3.2. Selection of feasible tree species

Figure 2 shows mean tree biomass and survival ratio observed in September 2015 (9.25 years after planting) at the three reforestation sites. Figure 2A shows the results of the direct planting method and Fig. 2B the results of the ripping + mounding planting method.

There was no significant difference on survival ratio and mean tree biomass among Arable area site and Upper & Lower salt scald sites for *E. salubris*, planted as control, and *E. sargentii*, planted as promising species. On the other hand, *E. camaldulensis*, planted as primary candidate for secondary salinity measure, had significantly a lower survival ratio in the Lower salt scald site and lower mean tree biomass in the Upper & Lower salt scald sites. Both planting treatments (direct planting and ripping + mounding) had similar results. As a result, *E. salubris* and *E. sargentii* met the objective of this study.

3.3. Comparison of survival and biomass between selected tree species

Figure 3 compares the survival ratio and mean tree biomass for *E. salubris* and *E. sargentii* and planting methods in the Upper & Lower salt scald sites. At both sites, mean tree biomass of *E. sargentii* was significantly larger than that of *E. salubris*, but the survival ratio had no significant difference. In addition, there was no significant difference between planting methodologies.

Table 2 summarizes observed results of this study consisting of survival ratio of 2007 and 2015, mean tree biomass of 2015, stand biomass in 2015 [Mg ha^{-1}], stand biomass increment during 9.25 years [Mg ha^{-1} $year^{-1}$], initially planted tree number and initial stand density [trees ha^{-1}].

4. Discussion

Through this feasibility study, *E. sargentii* and *E. salubris* met the objective of this study, because these species showed statistically the same survival ratio and mean tree biomass between agricultural area (Arable area site) and saline discharge area (Upper & Lower salt scald sites) as shown in Fig. 2. Considering biomass productivity, in other word land use efficiency, *E. sargentii* planted by direct planting or the ripping + mounding method in marginal area of salt scald is the recommended reforestation methodology. Climatic condition (rainfall) during observation period of this study (from 2006 to 2015) were similar to that during the recent 30 years, and thus observed data in this study were were considered representative values of this region.

The experimental location of Upper & Lower salt scald sites in this study had quite severe saline and water-logged condition, and such condition was considered one of the severest among previous studies reported in south-western Australia as described in the following three points.

The first point was the occurrence of a quite shallow ground water table at around 1 m deep throughout a year. This was not limited to the experimental sites but was more widespread across at least more than half of the area. Such shallow ground water table (1 m deep) was rare case. For example, reported ground water depth by Archibald et al. (2006) and Biddiscombe et al. (1985, 1989) were more than 2 m deep, that by Zohar et al. (2010) varied from 1 to 4 m deep, and that by Bush et al. (2013) was from 1.5 to 4.5 m deep. In addition, this ground water frequently rose up to the ground surface (nearly zero) after consecutive rainfall in winter (during Jun. to Sep.), which is the common situation in this region.

Second point was extremely high EC value of ground water of Upper & Lower salt scald sites ranging from 2,000 to 3,000 mS m^{-1}. Reported EC values of ground water was at most 2,000 mS m^{-1} in Boundain which is close to our research area (Wickepin). In other cases (Archibald et al., 2006; Bush et al., 2013) salinity was around 1,000 mS m^{-1} or less.

Third point was increased soil salinity strength during observation period (within 10 years). Judging from soil salinity criteria reported by Marcar and Crawford (2004), soil salinity of Upper & Lower salt scald sites increased from moderate to high and from moderate to extreme, respectively. Even severe situation, ECe value per se was not the worst condition, because other studies (*e.g.*, Archibald et al., 2006; Harper et al., 2012) reported higher soil ECe values, however increasing salinity strength was not reported.

Because of increasing soil salinity and shallow water table with highly saline ground water as above described, root zone of Upper & Lower salt scald sites was highly damaged by salinity and water-logging. This was considered as reason why all tree species decreased their survival during 2007 to 2015 (Table 2).

Selected tree species in this study (*E. sargentii* and *E. salubris*) which could survive and grow under increasing soil salinity and severe saline water-logging stress were considered to have low failure risk. Considering failure risk of reforestation especially for carbon abatement projects is recently regarded as important (Nolan et al., 2018)

E. sargentii showed results in line with expectations that there was no significant difference on survival ratio and mean tree biomass among Arable area site and Upper & Lower salt scald sites (Fig. 2). In previous studies, availability of *E. sargentii* for secondary salinity measure was controversial. Stolte et al. (1997) stated this tree species was vulnerable to regional rises in water table, and Zohar et al. (2010) reported its salinity tolerance was variable to seed source (tolerant or unknown, but not sensitive). On the contrary, Biddiscombe et al. (1989) and Marcar et al. (1995) stated that this tree species was suitable for planting in saline seep area. Marcar et al. (1995) stated that its survival decreased when soil ECe exceeds 2,000 mS m^{-1}, but Harper et al. (2012) reported good biomass growth (about 4 Mg ha^{-1} $year^{-1}$ of 26 years average) under extremely high soil salinity condition (over 4,000 mS m^{-1}). In this study, *E. sargentii* could survive and grow under increasing soil salinity strength (Fig. 1) with quite shallow saline ground water (0–1 m), and its biomass growth reached about 3–4 Mg ha^{-1} $year^{-1}$ (Table 2), which was comparable to the value reported by Harper et al. (2012). We thus considered *E. sargentii* was not vulnerable to shallow ground water table or extreme soil salinity.

E. salubris was also considered a feasible tree species for reforestation of saline sites for the same reasons as *E. sargentii* (Fig. 2). There are no explicit reports of salinity or water-logging tolerance of *E. salubris* so far. When reforestation of saline sites is required to be conducted not as monoculture planting but as environmental planting consisting of mixture of native tree species (Paul et al., 2013), E. *salubris* was considered important. Environmental reforestation of saline sites has an important potential co-benefit of bio-diversity restoration (George et al., 2012), which may in time attract additional incentive payments (Maraseni and Cockfield, 2015; Andres et al., 2022). However, mean tree biomass of *E. salubris* was significantly lower than that of *E. sargentii* (Fig. 3), and this suggests the need to seek other native tree species which have similar biomass productivity between saline and non-saline land.

The potential of other tree species for reforestation of saline sites has been considered elsewhere. This includes *E. occidentalis* Endl. (Flat-topped Yate) reported by Archibald et al. (2006), Biddiscombe et al. (1989) and Sochacki et al. (2012), E. *astringens* (Maiden) Maiden (Brown Mallet) reported by Biddiscombe et al. (1989)d *spathulata* Hook. (Swamp Mallet) and *E. salicola* Brooker (Salt Gum) reported by Zohar et al. (2010). However, the growth of these species except *E. occidentalis* were reported based on not biomass but tree sizes (*e.g.* height or volume). Therefore, they should be reevaluated based on biomass [kg $tree^{-1}$]. Because the relationship between size and biomass varies species by species and planting density, the same volume does not mean the same biomass. For example, *E. sargentii*, *E. salubris* and *E. camaldulensis* in this study had exactly the same tree size (5.0 m tree height and 0.10 m diameter at 0.3 m high), but quite different biomass calculated based on allometric equations (Table 1) of 24.7 kg tree^{-1}, 23.6 kg tree^{-1} and 12.7 kg tree^{-1}, respectively.

Among these potential candidates, *E. occidentalis* was considered the most promising because of following two reasons. One was that the biomass growth of *E. occidentalis* (around 4 Mg ha^{-1} year^{-1}) at the Wickepin site (Sochacki et al., 2012) was comparable to that of *E. sargentii* in this study. The other was that at another experimental site with highly saline (> 4,000 mS m^{-1}) conditions at Dryandra had similar growth rates (around 4 Mg^{-1} ha^{-1} year^{-1}) for *E. sargentii* and *E. occidentalis* at 26 years of age (Harper et al., 2012), which proved their long-term survival and growth.

For the tree planting methodology, survival ratio and mean tree biomass of two tree species (*E. sargentii* and *E. salubris*) planted by direct planting method was not significantly different to those planted by the ripping + mounding method (Fig. 3), and thus from a cost-effectiveness perspective, direct planting was beneficial due to the reduced initial cost.

5. Concluding Remarks

From this study, we found two tree species (*E. sargentii* and *E. salubris*) which had the same survival ratio and mean tree biomass under highly saline and water-logged condition as those in a non-saline arable area, and thus provides an option for the treatment of secondary salinity. Moreover, these two tree species maintained their high survival ratio and grew well even under severe soil salinity conditions and quite shallow saline ground water. In addition, because more than one tree species native in this bio-region were found feasible, reforestation of saline sites will have an opportunity to obtain bio-diversity restoration co-benefit as environmental planting.

For further studies in future, the transpiration and water use efficiency of selected tree species will be investigated to directly quantify their ability of decreasing saline ground water tables. And additional potential candidate tree species will be tested to determine whether they have comparable biomass productivity between saline and non-saline areas.

Declarations

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References

1. Andres SE, Standish RJ, Lieurance PE, Mills CH, Harper RJ, Butler DW, Adams VM, Lehmann C, Tetu SG, Cuneo P, Offord CA, Gallagher RV (2022) Defining biodiverse reforestation: Why it matters for climate change mitigation and biodiversity. PLANTS, PEOPLE, PLANET. https://doi.org/10.1002/ppp3.10329
2. Archibald RD, Harper RJ, Fox JED, Silberstein RP (2006) Tree performance and root-zone salt accumulation in three dryland Australian plantations. Agrofor. Syst. 66:191-204. https://doi.org/10.1007/s10457-005-6644-2
3. Benyon RG, Marcar NE, Crawford DF, Nicholson AT (1999) Growth and water use of *Eucalyptus camaldulensis* and *E. occidentalis* on a saline discharge site near Wellington, NSW, Australia. Agric. Water Manag. 39:229-244. https://doi.org/10.1016/S0378-3774(98)00080-8
4. Biddiscombe EF, de Bore ES, Rogers AL (1989) Growth of tree species on ploughed mounds near a saline seep. Landsc. Urban Plan. 17:189-196. https://doi.org/10.1016/0169-2046(89)90026-1
5. Biddiscombe EF, Rogers AL, Greenwood EAN, De Boer ES (1985) Growth of tree species near salt seeps, as estimated by leaf area, crown volume and height. Aust. For. Res. 15:141-154.
6. Bush D, Marcar N, Arnold R, Crowford D (2013) Assessing genetic variation within *Eucalyptus camaldulensis* for survival and growth on two spatially variable saline sites in southern Australia. For. Ecol. Manag. 306:68-78. https://doi.org/10.1016/j.foreco.2013.06.008
7. Caccetta PA, Simons J, Furby S, Wright N, George R (2022) Mapping salt-affected land in the South-West of Western Australia using satellite remote sensing, CSIRO Report Number EP2022-0724, CSIRO, Australia, accessed 11 January 2023.
8. Dunin FX (2002) Integrating agroforestry and perennial pastures to mitigate water logging and secondary salinity. Agric. Water Manag. 53:259-270. https://doi.org/10.1016/S0378-3774(01)00168-8
9. Environment Australia (2000) Revision of the Interim Bio-geographic Regionalisation of Australia (IBRA) and the development of version 5.1. Summary report. Department of Environment and Heritage, Canberra.
10. Flugge F, Abadi A (2006) Farming carbon: an economic analysis of agroforestry for carbon sequestration and dryland salinity reduction in Western Australia. Agrofor. Syst. 68:181-192. https://doi.org/10.1007/s10457-006-9008-7
11. George RJ, Clarke CJ, Hatton T (2001) Computer-modelled groundwater response to recharge management for dryland salinity control in Western Australia. Advances in Environmental Monitoring and Modelling, 2:3-35.

12. George SJ, Harper RJ, Hobbs RJ, Tibbett M (2012) A sustainable agricultural landscape for Australia: A review of interlacing carbon sequestration, biodiversity and salinity management in agroforestry systems. Agric. Ecosyst. Environ. 163:28-36. https://doi.org/10.1016/j.agee.2012.06.022
13. George R, McFarlane D, Nulsen B (1997) Salinity threatens the viability of agriculture and ecosystems in Western Australia. Hydrogeol. J. 5:6-21. https://doi.org/10.1007/s100400050103
14. George PR, Wren BA (1985) Crop tolerance to soil salinity. Western Australian Department of Agriculture Technote 6/85.
15. Gosper CR, Prober SM, Yates CJ, Wiehl G (2013) Estimating the time since fire of long-unburnt *Eucalyptus salubris* (Myrtaceae) stands in the Great Western Woodlands. Aust. J. Bot. 61:11-21. https://doi.org/10.1071/BT12212
16. Harper RJ, Dell B, Ruprecht JK, Sochacki SJ, Smettem, KRJ (2021) Salinity and the reclamation of salinized lands. pp.193-208 in “Soils and Landscape Restoration” Stanturf JA and Callaham MA Eds. (Academic Press, London). https://doi.org/10.1016/B978-0-12-813193-0.00007-2
17. Harper RJ, Beck AC, Ritson P, Hill MJ, Mitchell D, Barrett J, Smettem KRJ, Mann SS (2007) The potential of greenhouse sinks to underwrite improved land management. Ecol. Eng. 29:329-341. https://doi.org/10.1016/j.ecoleng.2006.09.025
18. Harper RJ, Okom AEA., Stilwell AT, Tibbett M, Dean C, George SJ, Sochacki SJ, Mitchell CD, Mann SS, Dods K (2012) Reforesting degraded agricultural landscapes with Eucalypts: Effects on carbon storage and soil fertility after 26 years. Agric. Ecosyst. Environ. 163:3-13. https://doi.org/10.1016/j.agee.2012.03.013
19. Harper, RJ, Sochacki SJ, McGrath JF (2017) The development of reforestation options for dryland farmland in south-western Australia: a review. Southern Forests 79:185-196 http://dx.doi.org/10.2989/20702620.2016.1255417
20. Harper RJ, Sochacki SJ, Smettem KRJ, Robinson N (2010) Bioenergy feedstock potential from short-rotation woody crops in a dryland environment. Energy Fuels 24:225-231. https://doi.org/10.1021/ef9005687
21. Harper RJ, Sochacki SJ, Smettem KRJ, Robinson N (2014) Managing water in agricultural landscapes with short-rotation biomass plantations. Glob. Change Biol. Bioenergy 6:544-555. https://doi.org/10.1111/gcbb.12090
22. Hatton TJ, McEwan K, Taylor P (1998) Does leaf water efficiency vary among eucalypts in water limited environments? Tree Physiol. 18:529-536. https://doi.org/10.1093/treephys/18.8-9.529
23. Horn D (2017) Review of inland drainage research (2003-2015). Perth, Australia: GHD Consultants.
24. Lefroy EC, Stirzaker RJ (1999) Agroforestry for water management in the cropping zone of southern Australia. Agroforest Syst. 45:277-302. https://doi.org/10.1023/A:1006241503888
25. Maraseni TN, Cockfield G (2015) The financial implications of converting farmland to state-supported environmental plantings in the Darling Downs region, Queensland. Agric. Syst. 135:57-65. https://doi.org/10.1016/j.agsy.2014.12.004
26. Marcar NE, Crawford DF (2004) Trees for saline landscapes. RIRDC, Canberra, Australia, pp.246.

27. Marcar N, Crawford D, Leppert P, Jovanovic T, Floyd R, Farrow R (1995) Trees for Saltland: A Guide to Selecting Native Species for Australia. CSIRO Division of Forestry, Canberra.
28. Marcar NE, Crawford DE, Saunders A, Matheson AC, Arnold RA (2002) Genetic variation among and within provenance and families of *Eucalyptus grandis* W. Hill *E. globulus* Labill. subsp. *globulus* seedlings in response to salinity and waterlogging. For. Ecol. Manag. 162:231-249. https://doi.org/10.1016/S0378-1127(01)00518-7
29. Morris JD, Collopy JJ (1999) Water use and salt accumulation by *Eucalyptus camaldulensis* and *Casuarina cunninghamiana* on a site with shallow saline groundwater. Agric. Water Manag. 39:205-227. https://doi.org/10.1016/S0378-3774(98)00079-1
30. National Land and Water Resources Audit (2001) Dryland salinity in Australia. A summary of the National Land and Water Resources Audit's Australian dryland salinity assessment 2000. National Land and Water Resources Audit, Canberra.
31. Nolan RH, Sinclair J, Eldridge DJ, Ramp D (2018) Biophysical risks to carbon sequestration and storage in Australian drylands. J. Environ. Manage. 208:102-111. https://doi.org/10.1016/j.jenvman.2017.12.002
32. Paul KI, Reeson A, Polglase P, Crossman N, Freudenberger D, Hawkins C (2013) Economic and employment implications of a carbon market for integrated farm forestry and biodiverse environmental plantings. Land Use Policy 30:496-506. https://doi.org/10.1016/j.landusepol.2012.04.014
33. Rayment GE, Higginson FR (1992) Australian laboratory handbook of soil and water chemical methods. Inkata Press, Melbourne, pp.330
34. Sochacki SJ, Harper RJ, Smettem KRJ (2007) Estimation of woody biomass production from a short-rotation bio-energy system in semi-arid Australia. Biomass Bioenergy 31:608-616. https://doi.org/10.1016/j.biombioe.2007.06.020
35. Sochacki SJ, Harper RJ, Smettem KRJ (2012) Bio-mitigation of carbon following afforestation of abandoned salinized farmland. Glob. Change Biol. Bioenergy 4:193-201. https://doi.org/10.1111/j.1757-1707.2011.01139.x
36. Stolte WJ, McFarlane DJ, George RJ (1997) Flow systems, tree plantations, and salinisation in a Western Australian catchment. Aust. J. Soil Res. 35:1213-1229. https://doi.org/10.1071/S96066
37. Turner NC, Ward PR (2002) The role of agroforestry and perennial pasture in mitigating water logging and secondary salinity: summary. Agric. Water Manag. 53:271-275. https://doi.org/10.1016/S0378-3774(01)00170-6
38. White DA, Dunin FX, Turner NC, Ward BH, Galbraith JH (2002) Water use by contour-planted belts of trees comprised of four *Eucalyptus* species. Agric. Water Manag. 53:133-152. https://doi.org/10.1016/S0378-3774(01)00161-5
39. Zohar Y, Stefano JD, Bartle J (2010) Strategy for screening eucalypts for saline lands. Agrofor. Syst. 78:127-137. https://doi.org/10.1007/s10457-009-9233-y

Tables

Table 1 Allometric equations of each tree species

Species	Equation number	Allometric equations	R^2	RMSE	n
Eucalyptus camaldulensis	CAM1	$W_{s+b} = 142.3(D^2H)^{0.875}$	0.998	0.37	8
	CAM2	$W_l = 21.9(D^2H)^{0.742}$	0.987	0.05	8
	CAM3	$W_{s+b} = 0.0806H^{3.40}$	0.966	0.71	9
	CAM4	$W_l = 0.0424H^{2.74}$	0.923	0.21	9
Eucalyptus salubris	SAL1	$W_{s+b} = 266.8(D^2H)^{0.896}$	0.995	0.04	7
	SAL2	$W_l = 41.8(D^2H)^{0.682}$	0.987	0.10	7
	SAL3	$W_{s+b} = 0.0659H^{3.97}$	0.996	0.10	7
	SAL4	$W_l = 0.0760H^{3.00}$	0.978	0.09	7
Eucalyptus sargentii	SAR1	$W_{s+b} = 273.9(D^2H)^{0.929}$	0.997	0.15	9
	SAR2	$W_l = 94.4(D^2H)^{0.834}$	0.989	0.28	9
	SAR3	$W_{s+b} = 0.264H^{2.88}$	0.987	0.62	9
	SAR4	$W_l = 0.185H^{2.58}$	0.971	0.25	9

W_{s+b} (kg), W_l (kg), D (m), H (m), RMSE (kg) and n denote stem & branch biomass, leaf biomass, diameter at 0.3 m height, tree height, root mean squared error and destructive tree sample number, respectively.

Table 2 Summary of observed data

	Planted areas and tree species	Survival ratio [-] 2007	Survival ratio [-] 2015	Mean tree biomass [kg tree^{-1}] 2015	Stand biomass [Mg ha^{-1}] 2015	Biomass increment [Mg ha^{-1} year^{-1}] 2006-2015	Planted tree number [trees] 2006	Stand density [trees ha^{-1}] 2006
Direct planting	Arable area							
	E. salubris	0.848	0.457	5.3	6.3	0.68	46	2591
	E. camaldulensis	0.622	0.511	12.0	15.8	1.71	45	
	E. sargentii	0.708	0.625	17.9	29.0	3.14	48	
	Upper salt scald							
	E. salubris	0.695	0.661	7.3	12.6	1.36	59	2612
	E. camaldulensis	0.741	0.370	3.4	3.3	0.36	54	
	E. sargentii	0.762	0.698	15.3	28.0	3.02	63	
	Lower salt scald							
	E. salubris	0.900	0.650	5.0	8.9	0.96	80	2741
	E. camaldulensis	0.720	0.195	2.7	1.5	0.16	82	
	E. sargentii	0.871	0.812	15.8	35.1	3.79	85	
Ripping and mounding	Arable area							
	E. salubris	0.786	0.571	8.6	12.7	1.37	14	2591
	E. camaldulensis	0.600	0.350	9.3	8.5	0.92	20	
	E. sargentii	0.944	0.889	16.6	38.3	4.14	18	
	Upper salt scald							
	E. salubris	0.733	0.567	6.9	10.2	1.11	30	2612
	E. camaldulensis	0.607	0.321	3.4	2.9	0.31	28	
	E. sargentii	0.862	0.690	14.6	26.4	2.85	29	
	Lower salt scald							
	E. salubris	0.935	0.742	4.9	9.9	1.07	31	2741
	E. camaldulensis	0.742	0.032	2.8	0.3	0.03	31	
	E. sargentii	0.903	0.806	16.9	37.4	4.05	31	

Figures

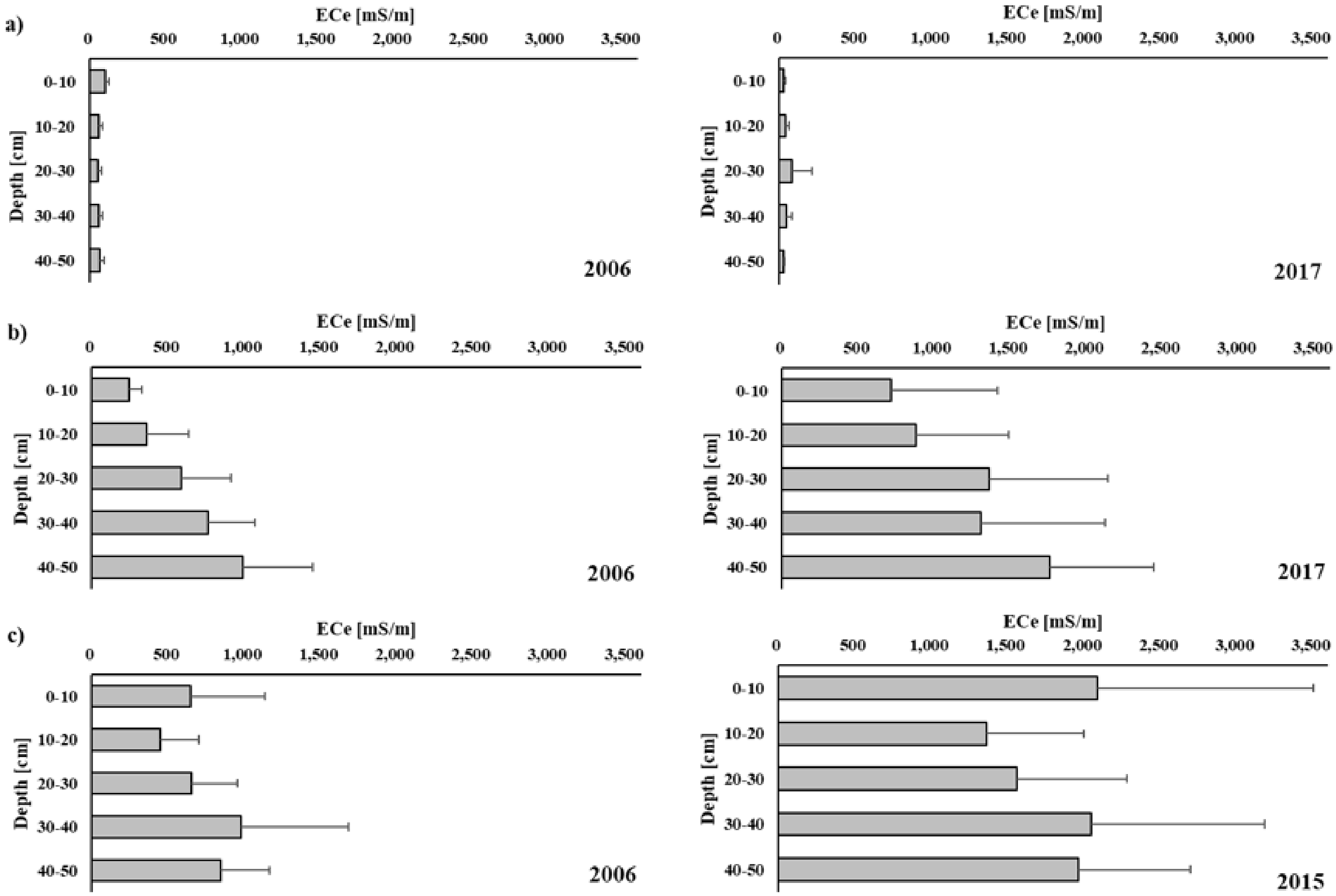

Figure 1

Changes in soil electric conductivity at depths after about 10 years in three sites (a: arable area, b: upper salt scald, c: lower salt scald)

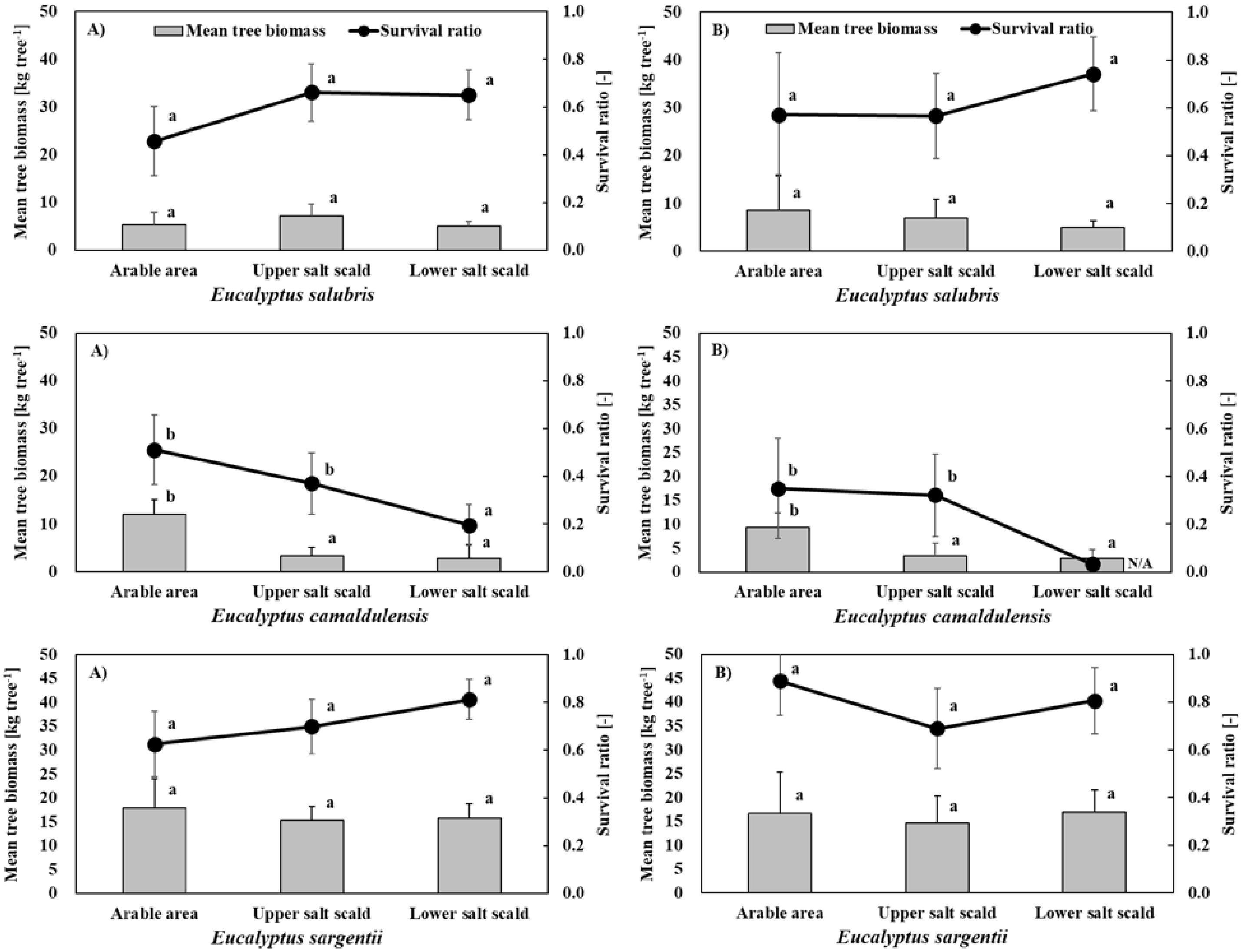

Figure 2

Comparison of survival ratio and mean tree biomass among experimental sites (A: direct planting, B: ripping+mounding)

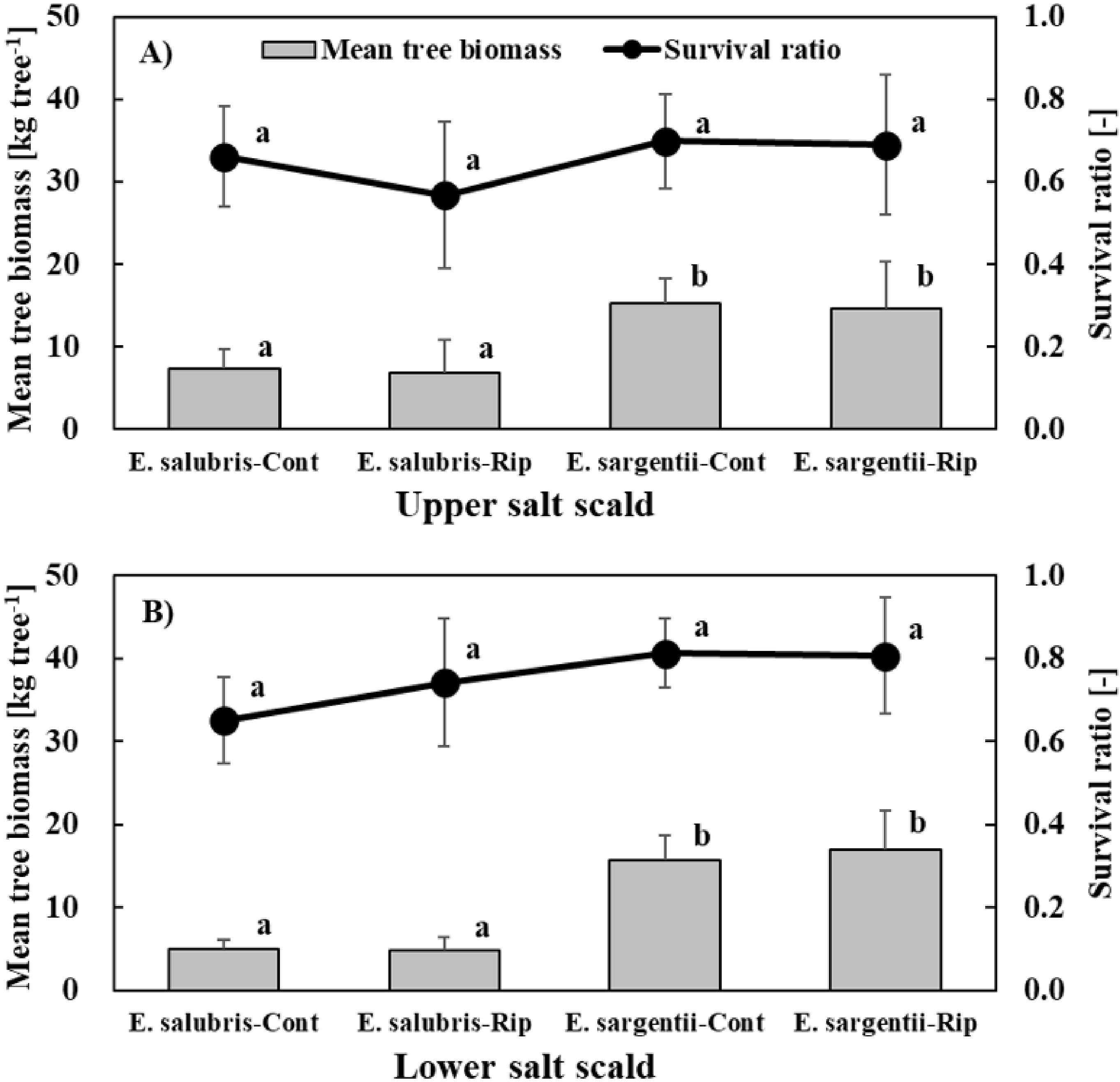

Figure 3

Comparison of survival ratio and mean tree biomass among selected feasible tree species and planting methodology (A: upper salt scald, B: lower salt scald)