

The Facilitative Role of *Kosteletzkya pentacarpos* in Transitioning Coastal Agricultural Land to Wetland During Sea Level Rise

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Abstract Rising sea level is increasing soil salinity and flooding frequency, directly impacting low-lying coastal farmlands. In response to reduced production of traditional crops, dikes may be built which will prevent inland migration of wetlands. Seashore mallow, *Kosteletzkya pentacarpos*, is being developed as an alternative crop for such areas. Eventually, when the crop can no longer be harvested because of flooding, we hypothesize that seashore mallow will facilitate the establishment of desirable wetland species by acting as a nurse crop through this transitional period. Four treatments were planted in flood-irrigated plots at an upland field site adjacent to a salt marsh. The control, *Spartina patens*, *K. pentacarpos*, and combined treatments were laid out in a complete randomized block design with replication. These were sampled for species percent vegetative cover, morphological traits, above-ground biomass, and leaf litter. The presence of seashore mallow enhanced *S. patens* and *Baccharis halimifolia* recruitment and did not negatively impact growth of planted *S. patens*. Communities established around *K. pentacarpos* were both productive and diverse, and leaf litter was increased by *K. pentacarpos*. Our findings support the use of *K. pentacarpos* as a low-cost nurse crop in salinized agro-ecosystems. Such a strategy would prolong agricultural function and minimize loss of ecological services by allowing gradual inland wetland migration.

Keywords Sea level rise · Coastal agriculture and salinization · *Kosteletzkya pentacarpos* · *Spartina patens* · Nurse plant · Wetland

Introduction

Accelerating sea-level rise is a serious threat to coastal wetlands globally (Craft et al. 2009; Nicholls et al. 1999; Nicholls et al. 2007). Changes in vegetation and greater inundation rates have already been observed, particularly in combination with adjacent land use and increased frequency of severe weather events (Donnelly and Bertness 2001; Elsey-Quirk et al. 2011; Hartig et al. 2002; Warren and Niering 1993). Wetland sedimentation rates must match sea-level rise or species must migrate inland to survive (Reed 1995). As coastal development may limit migration, planning ahead for space into which wetland species can migrate could mitigate future losses and ensure vital wetland ecosystem services (Nicholls et al. 1999; Stralberg et al. 2011), valued by Constanza et al. (1997) at \$14,785 per hectare per year.

Coastal agricultural land is also susceptible to the impacts of sea-level rise. Exposure to more frequent severe weather events and higher tidal ranges is expected to progressively salinize soils and contaminate freshwater aquifers (IPCC 2007). Patchy areas of growth will become increasingly common, as traditional crops are not salt-tolerant. Current sea-level rise projections would render many coastal agricultural areas too saline and too wet to cultivate. In Delaware, a 0.5-m rise in sea level would inundate approximately 51 km² of highly productive soils and 31.6 km² of undeveloped groundwater recharge areas (DDNREC 2012). Dikes are already being built around local Delaware farmlands (personal observation). However, such salt-water exclusion practices are an expensive and temporary solution to preserving agricultural land and would further hinder inland migration of wetland species. Many of these agricultural sites could be suitable upland sites into which wetland vegetation could migrate, hence, creating an opportunity to turn the unfortunate loss of one service into a positive gain of another.

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However, if left unmanaged not only is coastal agricultural land at risk of losing productivity but it may also provide a poor habitat for migrating wetlands. For example, agricultural soil has been shaped by the prolonged cultivation of annuals, whose shallow roots and required maintenance diminish soil productivity (Glover et al. 2007; Lal 2004). Unlike wetland soils, where conditions favor wetland species, these soils are low in sodium and oxidized and may thus encourage invasive weeds. For example, the highly invasive *Phragmites australis* (Cav.) Trin. ex Steud., is a costly problem on the East coast of North America and becomes established more easily in lower salinity and more oxidized soils from where it can spread via rhizomes into more typical wetland soils (Seliskar 2007). On the coast of the Chesapeake Bay, its spread has been linked to agricultural activities (Chambers et al. 2008) and its presence diminishes important ecosystem services (Able and Hagan 2003).

A carefully chosen salt-tolerant crop could be a cost-effective tool to mitigate such impacts. The use of nurse plants in restoration of disturbed habitats is a recommended strategy (Padilla and Pugnaire 2006). Nurse plants can improve the potential for productivity in their immediate environment by ameliorating stressful conditions for seedlings, such as direct radiation, fluctuation of air and soil temperatures, salinity or drought, hence creating a microclimate that is ideal for germination and growth (Armas and Pugnaire 2005; Bertness 1991; Hacker and Bertness 1995; see Padilla and Pugnaire 2006 for review). They can improve soil properties, nutrients and texture, and enhance microbial activity (Armas and Pugnaire 2005; Padilla and Pugnaire 2006) while their physical presence can trap seeds (Egerova et al. 2003), protect colonizing species from grazers, and attract pollinators (Padilla and Pugnaire 2006). Therefore, by creating a more suitable environment, a transitional salt-tolerant crop could maximize wetland migration potential while simultaneously prolonging economic yield for farmers.

Kosteletzkya pentacarpos (L.) Ledeb., formerly *K. virginica* (L.) Persl. (Blanchard 2008), is a perennial dicot (Malvaceae) that has been proposed for use as a halophytic crop (Gallagher 1985; Gallagher and Seliskar 1993; Gallagher 1995; He et al. 2003; Sommers 1979). It yields multiple products which are currently under development; specifically, oil (Moser et al. 2013; Ruan et al. 2008), biodegradable bio-absorbent fiber products, such as animal bedding, hydromulch, and cat litter (Vaughn et al. 2013), thread (Fan et al. 2011), and animal feed (Islam et al. 1982). *K. pentacarpos* has many desirable traits for sustainable agriculture, such as perennial growth, self-pollination (Ruan et al. 2009), and long seed viability (Poljakoff-Mayber et al. 1992; Poljakoff-Mayber et al. 1994). Also, protocols are available for its improvement through tissue culture techniques (Cook et al. 1989) and molecular transformation (Li et al. 2006). Multiyear pilot projects have successfully cultivated and harvested *K. pentacarpos* in the United States of America

(Gallagher 1985; Halchak et al. 2011) and in China (He et al. 2003; Ruan et al. 2008). This halophyte has the properties necessary for becoming a salt-tolerant crop and for prolonging the productivity of coastal agricultural land.

Several qualities of *K. pentacarpos* theoretically could also make it an excellent nurse crop. It is a perennial, and the cultivation of a perennial crop is suggested to improve soil conditions through nutrient storage, carbon sequestration, increasing exchange of water and air between soil layers, and reducing erosion (Glover et al. 2007). A good candidate to survive the unstable environmental conditions that may arise during the transition, the plant's deep roots maintain the plants through periods of abiotic stress, such as drought or waterlogged conditions (Halchak et al. 2011). For at least 6 years, the plant produces more stems than the previous year (Gallagher 1985; Halchak 2009), thus could provide a dense physical barrier against invasion and a habitat for wildlife. Its dense stems could also increase sedimentation rates, by slowing tidal currents passing through the stand, as has been shown for other species (Gleason et al. 1979; Van Hulzen et al. 2007; Widdows and Brinsley 2002). It is native to brackish marshes of the Atlantic and Gulf coasts of the United States of America (Radford et al. 1968), with sparse populations noted in Europe (Pino et al. 2007), and does not exhibit invasive character. Instead, it is usually found intermittently dispersed amongst the dominant upper marsh species and is not likely to compete with or hinder the progress of migrating wetland halophytes when salinity and waterlogging increase.

In this study, we sought to better understand the ecology of *K. pentacarpos* and to investigate whether it could be applied as a tool for promoting desirable wetland plant establishment in upland locations during sea-level rise. In order to evaluate the suitability of *K. pentacarpos* for this purpose, we asked three questions. (1) What species naturally colonize a plant community started around *K. pentacarpos*? (2) What is the impact of *K. pentacarpos* on the species diversity and productivity of its surrounding community? (3) How does *K. pentacarpos* interact with the common coastal colonizer *Spartina patens* (Aiton) Muhl? The goal of these questions was to determine whether the cultivation of *K. pentacarpos* is more beneficial than allowing sea-level rise to change the land without further management and to identify what these benefits are.

Methods

Experimental Design and Maintenance

The experiment was a complete randomized block design with replication conducted on the grounds of the University of Delaware H. R. Sharp campus in Lewes, Delaware (38°46' 46.68" N, 75°9'48.11" W). Situated upland to a natural salt

marsh (approximately 15 m), the site was used as agricultural land before being converted to experimental plots in the mid-1970s. Approximately 6 years prior to this study, the site was abandoned to a natural state and was not exposed to salt water. The land was tilled, cleared of above and below ground plant matter, and leveled, remaining fallow for a year before the study began. The soil was sandy loam, with 73 % sand, 19 % silt, and 8 % clay particles. Table 1 describes soil conditions at the start of the study.

Four treatments were replicated three times randomly within each of two blocks (Fig. 1). Replicate plots were 3 m × 4 m in size. The treatments consisted of the following: (1) fallow/untreated control (C), (2) planted with *K. pentacarpus* (K), (3) planted with *S. patens* (S), and (4) both planted in combination (KS). Suitable combinations of these four treatments were used to answer the project questions. Specifically, when exploring the impact of *K. pentacarpus* on *S. patens* we compared variables describing the planted *S. patens* in treatments S and KS. When investigating the impact of *K. pentacarpus* on the recruitment of *S. patens* to the area, we monitored the colonization of *S. patens* in treatments C and K. To study the structure, productivity and diversity of a community started around *K. pentacarpus*, we compared the community composition of treatment K with C and S.

K. pentacarpus seed, originating in Delaware (Sussex County, 38°44'01" N, 75°07'22" W), was planted in the K

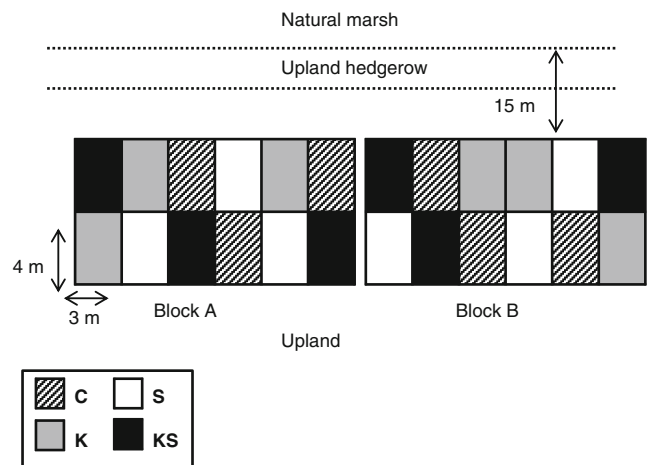


Fig. 1 Illustration of the experimental design and set-up. Treatments: C Control, K planted with *K. pentacarpus*, S planted with *S. patens*, KS planted with both

and KS plots in May of 2009 (Year 0) and was allowed to grow for a year prior to the experiment to simulate a salinized coastal agricultural setting where *K. pentacarpus* was grown in place of traditional crops. In the spring of 2010 (Year 1), *S. patens* was planted in the S and KS treatments in the form of young plant plugs, purchased from Environmental Concern Inc. nursery and originated from seed collected in Talbot County, Maryland (38°46'49" N, 76°07'55" W). Nine plugs were planted evenly within each plot and nine square quadrats of 50 cm by 50 cm were flagged around them for observation. Similar quadrats were marked in C and K plots. These nine permanent quadrats, within each of the twenty four plots, were established as an efficient way to monitor vegetation in the greater replicate plot and their mean value was used to represent that plot in statistical analyses.

During the growing season, the plots were flood irrigated twice weekly from the neighboring tidal creek to simulate salt-water intrusion. The irrigation system was also linked to a freshwater source which allowed for salinity of the flooding water to be manipulated to a target salinity of 22 parts per thousand (ppt). To inform irrigation decisions, pore water salinity was monitored frequently via six stopper-sealed PVC pipes ending in a porous porcelain cup at a depth of 15 cm. Water samples were read with a refractometer. One HOBO temperature data logger was installed amongst the plants in each block and monitored air and soil temperatures, 6 cm above and below ground, throughout the experiment. A rain gage was installed on site and additional local rainfall data was used to validate records (Joseph R. Scudlark, University of Delaware, personal communication).

Data Collection

Natural colonization was allowed to occur in all plots, beginning in Year 1. Species and percent vegetative cover were

Table 1 Soil nutrient content, pH, and other soil descriptors for the experimental site as sampled at the beginning of the study (Year 0)

Soil Variable	Year 0	Unit
pH	8.1 (±0.1)	
Phosphorus	157.6 (±17.8)	mg/kg
Potassium	104.4 (±6.5)	mg/kg
Calcium	191.9 (±22.0)	mg/kg
Magnesium	147.0 (±14.2)	mg/kg
Manganese	8.8 (±2.8)	mg/kg
Zinc	5.9 (±0.6)	mg/kg
Copper	9.2 (±0.9)	mg/kg
Iron	233.6 (±29.3)	mg/kg
Boron	0.6 (±0.1)	mg/kg
Sulfur	17.2 (±1.5)	mg/kg
Aluminum	1,271.4 (±118.8)	mg/kg
Soluble Salts	0.2 (±0.01)	mmhos/cm
Estimated CEC	4.2 (±0.1)	meq/100 g
Base saturation	57.7 (±1.5)	%
Total nitrogen	0.1 (±0.01)	%
Total carbon	0.8 (±0.1)	%
Organic matter (loss on ignition)	3.1 (±0.2)	%
Phosphorus saturation ratio	29.3 (±1.8)	

Values are means (±SE); *n*=24

recorded monthly during the growing season for 2 consecutive years. Of particular interest was the natural recruitment of *S. patens* in C and K plots. The specific locations, where new *S. patens* clumps were identified, were cross-checked for consistency throughout the dataset.

Establishment of planted *S. patens* was monitored via vegetative cover surveys, as mentioned above. Additionally, morphological characteristics of planted *S. patens* clones were recorded at the beginning and end of the first growing season. Traits monitored were shoot height, number of shoots per clone, and clone circumference. Clone circumference included two measurements: an inner circumference, defined as the circumference around the central clump of the plant where highest shoot density occurred; and an outer measurement indicative of rhizome spread. Concurrently, stem height and number were also taken for *K. pentacarpus*. Change in morphological traits was defined as the difference between the June and September values for that trait and used for further analysis.

At the end of the second growing season, above-ground biomass was harvested. In order to preserve the study site for future work, only one representative quadrat from each plot was harvested. Samples were dried to constant weight at 60 °C and weighed. As a linear relationship has been identified between herbaceous vegetative cover and biomass (Röttgermann et al. 2000), the biomass measurement was then matched to the recent vegetative cover measurement for the samples harvested and a biomass weight per unit of vegetative cover was calculated. This plot-specific relationship was then used to assign a biomass weight to each vegetative cover measurement of that species in the particular plot for each of the eight quadrats not harvested. For example, if within the harvested quadrat of a replicate plot we found that *B. halimifolia*'s vegetative cover was 20 % and its dry weight 5 g, then for every one percent of vegetative cover *B. halimifolia* weighed 0.25 g, and in another quadrat of the same plot for which *B. halimifolia* had vegetative cover of 15 %, we could assign to it a biomass of approximately 3.75 g. The process, described in the example above, was completed for all plots independently to maximize accuracy and adjust for immediate environmental conditions.

In order to evaluate the effect of the treatments on plant community diversity, the Shannon–Wiener diversity index (Shannon 1948) was calculated using both species above-ground biomass and vegetative cover. Species counts were used to evaluate changes in community richness.

Soil surface leaf litter was measured in two ways during the second growing season (Year 2). A percent litter cover was estimated during vegetative cover surveying and litter was collected during the biomass sampling described above.

Statistical Analyses

Mixed model analysis of variance (ANOVA) was used to test differences ($\alpha \leq 0.05$) for variables sampled. Treatment was

treated as a fixed factor. Block was treated as a random factor. Two-way mixed model ANOVA comparisons tested significance of the effect of treatments and blocks, and their interaction. For *K. pentacarpus* cover in August (Year 1 and Year 2), a three-way mixed model ANOVA comparisons tested significance of the effect of treatments, blocks, year, and possible interactions. All data were examined for normality and appropriately transformed when necessary to meet ANOVA assumptions. Tukey's HSD test was used for further comparison when appropriate.

All statistical analyses and graphs were completed in R, version 2.13.2 (R Development Core Team 2011) and SigmaPlot Version 12.2. R scripts created for mixed model ANOVAs were tested for errors against example data sets.

Results

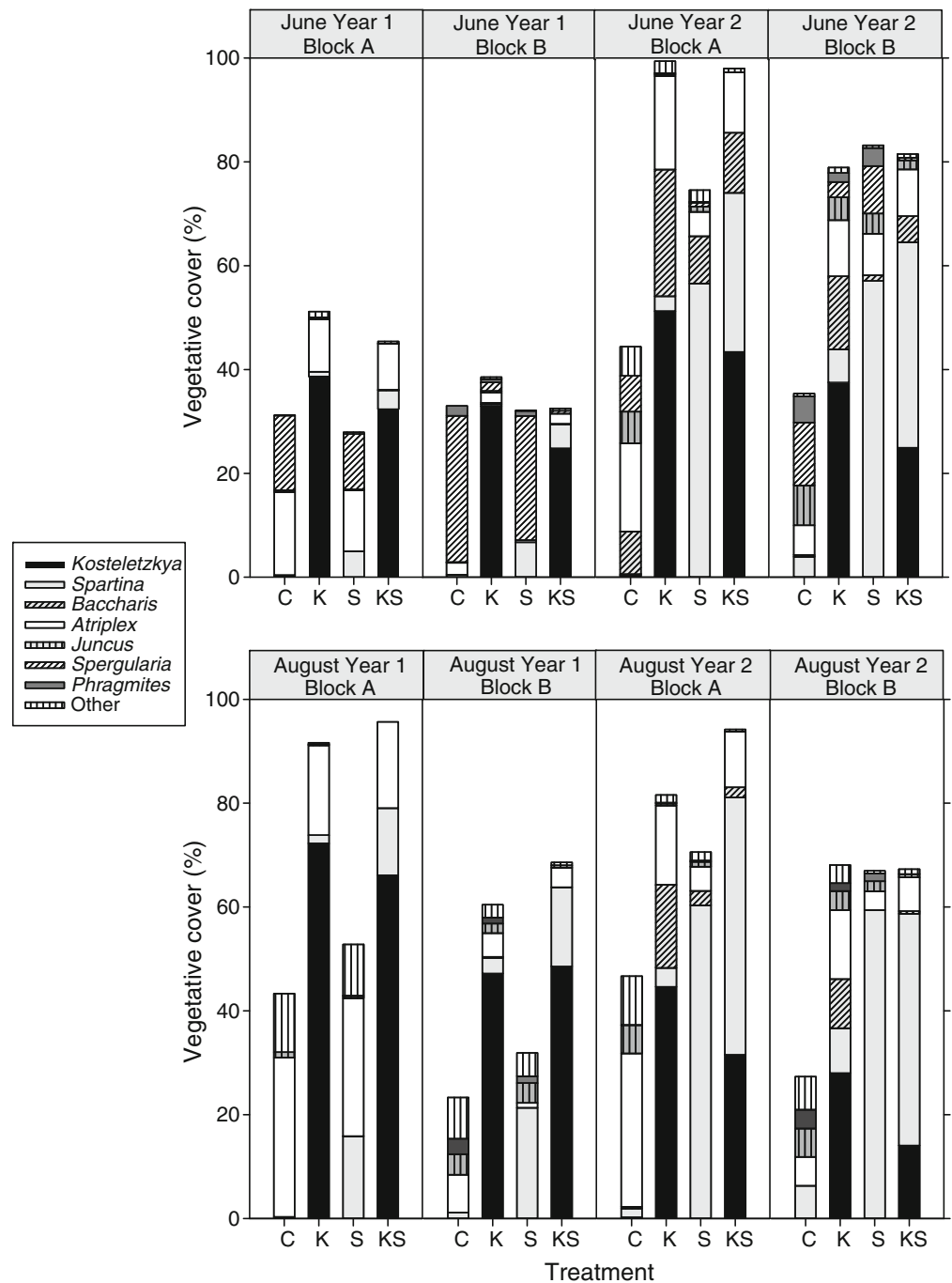
Vegetation Dynamics

In order of decreasing occurrence, the plots were colonized by: *Atriplex triangularis* Willd. (triangle orache), *Juncus gerardii* Loisel. (saltmeadow rush), *Spergularia salina* J. Presl & C. Presl (salt sandspurry), *Baccharis halimifolia* L. (groundsel bush), *S. patens* (saltmeadow cordgrass), *P. australis* (common reed), *Aster tenuifolius* L. (salt marsh aster), *Distichlis spicata* (L.) Greene (saltgrass), and a few seasonal weeds (Fig. 2).

A. triangularis, an annual early-marsh colonizer, was found in all treatments. *B. halimifolia* was rare in Year 1, but germinated broadly during the second summer and establishment was noted by high biomass, particularly in treatment K ($p = 0.03$). Vegetative cover of colonizing *S. patens* (Fig. 3) was significantly higher in treatment K than treatment C in July ($p < 0.0001$), August ($p = 0.03$), and September ($p = 0.009$), but not in June ($p = 0.6$). *P. australis*, an invasive of particular interest, was found in a few replicate plots but its presence may have been linked to previous experiments at the site. *A. tenuifolius* was found in C and K treatments primarily and only in the second summer (Year 2) (included under "Other" label in Fig. 2). *D. spicata* was found in a few small clumps, primarily in open spaces with sparse cover ("Other"). Certain common weeds were observed but remained low overall due to salinity.

Planted *S. patens* vegetative cover in treatments S and KS showed no significant treatment effect in Year 1 (Fig. 4, for September $p = 0.08$) and Year 2 (see treatments S and KS in Fig. 2). Planted *K. pentacarpus* vegetative cover (arcsine transformed) in treatments K and KS for August, the time of its maximum cover, was not different but was higher in Block A ($p = 0.0003$, three-way mixed ANOVA) and Year 1 ($p = 0.03$) (Fig. 2).

Fig. 2 Species composition (mean % vegetative cover) of treatments and blocks in June and August of Year 1 and Year 2. Treatments: *C* Control, *K* planted with *K. pentacarpus*, *S* planted with *S. patens*, *KS* planted with both



Morphology

Morphological measurements were taken for *K. pentacarpus* and *S. patens* in the first growing season (Table 2) and evaluated as the change in each trait across the growing season. Planted *S. patens* maximum shoot height ($p=0.07$), mean shoot height, outer circumference (square-root transformed), and number of shoots per clump were not affected by treatment. There was a significant interaction ($p=0.026$) of treatment and block for the inner circumference (square-root transformed). Further investigation of the interaction showed that planted *S. patens* mean

change in inner circumference was larger in treatment S than in treatment K for Block B only. The mean change in height of *K. pentacarpus* stems for the growing season did not vary by treatment but was higher in Block A than in Block B ($p=0.01$). The increase in mean number of *K. pentacarpus* stems was not different among treatments or between blocks.

Biomass

Above-ground biomass results are shown in Table 3. *K. pentacarpus* biomass was significantly less in Block B

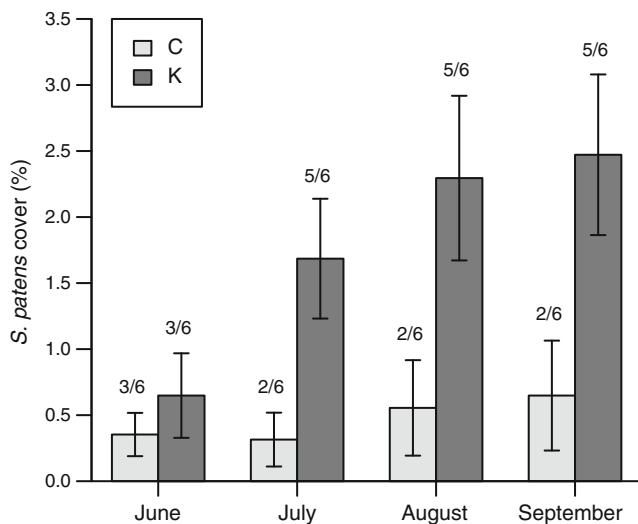


Fig. 3 Natural colonization of *S. patens*, in vegetative cover (%), during the growing season of Year 1 in C and K treatments. Error bars indicate the standard error of the mean; treatments: C Control, K planted with *K. pentacarpus*. Noted above each bar is the number of plots, out of six replicates, in which natural colonization by *S. patens* occurred

than in Block A ($p=0.035$) for both K and KS. Biomass of planted *S. patens* was not significantly different between S and KS treatments. Biomass of colonizing *S. patens* was higher in the K than in the C treatment ($p=0.05$). After subtracting the biomass of *K. pentacarpus* from K sums, total community above-ground biomass productivity was still similar in treatment K and C.

Community Diversity and Richness

Above-ground biomass was used to calculate the Shannon–Wiener diversity index. The highest diversity index was

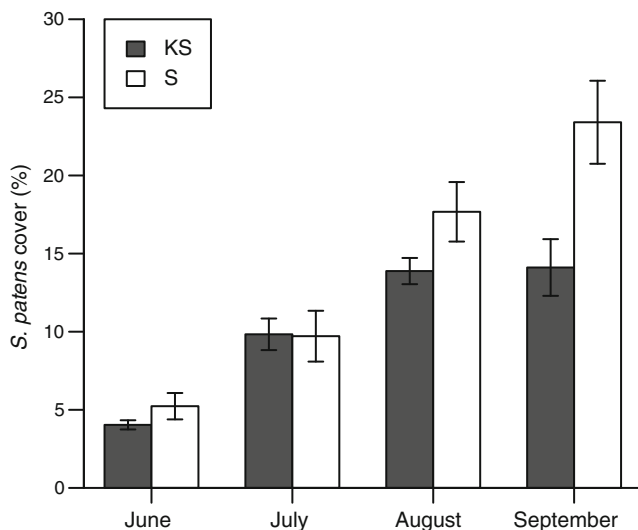


Fig. 4 Mean growth of planted *S. patens*, in vegetative cover (%), during Year 1 in the S and KS treatments. Error bars represent the standard error of the mean; treatments: S planted with *S. patens*, KS planted with both *S. patens* and *K. pentacarpus*

associated with treatments K (1.43 ± 0.08) and C (1.34 ± 0.09), with S (0.33 ± 0.05) and KS (0.58 ± 0.09) being the least diverse ($p=0.03$). The analysis was repeated using percent vegetative cover data and yielded similar results ($p=0.04$), with treatment K and C being the most diverse. Species richness also followed the same pattern. From the initiation of the experiment (June Year 1) to its end (August Year 2), the greatest increase in number of species occurred in K (2.24 ± 0.52) and C (1.65 ± 0.05) treatments, followed by plots containing *S. patens* KS (0.39 ± 0.27) and S (-0.13 ± 0.01).

Leaf Litter

Mean percent cover of leaf litter, sampled in August of Year 2, was 7.24 % (± 1.31) in treatment C, 9.65 % (± 7.02) in S, 49.26 % (± 4.91) in K, and 40.5 % (± 7.02) in KS. The effect of treatment ($p=0.009$) and block ($p=0.0006$) was significant and a Tukey's test showed that treatments with *K. pentacarpus* contained the highest leaf litter cover in Block A. Litter biomass showed the same result (data not shown). Treatments ($p=0.005$) and blocks ($p=0.0002$) were significantly different, with greatest leaf litter biomass in treatments containing *K. pentacarpus* and in Block A.

Discussion

Our findings suggest that the cultivation of *K. pentacarpus* as a transitional crop in salt-affected land has several benefits. The presence of *K. pentacarpus* improved natural recruitment of native halophytes *S. patens* and *B. halimifolia*. *K. pentacarpus* did not negatively impact community diversity, richness, or productivity. In addition, we did not detect a negative effect of *K. pentacarpus* on planted *S. patens* plugs in the immediate growing season or 1 year later. And *K. pentacarpus* leaf litter accumulated heavily on the soil surface after just 1 year of its growth at the site.

Interactions and Recruitment of Key Species

S. patens did not impact the growth of *K. pentacarpus*. In the natural environment, *K. pentacarpus* is not a dominant species and is found scattered throughout brackish marshes (Radford et al. 1968). Although the vegetative cover of *K. pentacarpus* declined significantly in Year 2 (Fig. 2), paralleling strong growth by second year *S. patens* plants, *K. pentacarpus* cover was not greater when growing alone. Patchy regrowth was also observed at a second experimental site that year (8 km from our study site). The reduced reemergence of *K. pentacarpus* in Year 2 may have been caused by five cold periods during December and January (data in Voutsina 2012). The most severe of these occurred in late January when the temperature dropped to an average of -11.8 °C for three

Table 2 Morphological measurements taken at the beginning and end of Year 1 growing season, including *S. patens* mean and maximum shoot height (cm), shoot count, inner and outer clump circumference (cm), and *K. pentacarpus* mean stem height (cm) and stem count within a quadrat

	K		S		KS	
	June	September	June	September	June	September
<i>S. patens</i>						
Mean shoot height	–	–	45.7 (±0.7)	60.5 (±1.2)	43.0 (±0.8)	70.4 (±1.4)
Max shoot height	–	–	58.9 (±0.8)	83.7 (±1.7)	59.4 (±0.8)	101.7 (±1.7)
Shoot count	–	–	34.7 (±1.1)	116.5 (±5.4)	33.5 (±1.1)	103.1 (±6.2)
Inner circumference	–	–	11.7 (±0.5)	19.3 (±0.8)	12.1 (±0.5)	32.8 (±1.8)
Outer circumference	–	–	13.8 (±0.7)	97.8 (±8.6)	13.4 (±0.6)	53.6 (±4.6)
<i>K. pentacarpus</i>						
Stem height	24.5 (±0.9)	65.9 (±4.2)	–	–	22.9 (±0.4)	68.3 (±4.1)
Stem count	39.2 (±3.2)	39.9 (±3.3)	–	–	36.6 (±3.5)	32.7 (±3.9)

Values are means (±SE); $n=54$

nights which was 8 °C below the average normal low for the month (Delaware Environmental Observatory System, University of Delaware). *K. pentacarpus* cover, biomass and stem growth were also significantly lower in Block B than in Block A. During our study, plots were flood irrigated twice weekly with water pumped from the nearby tidal creek and later drained to remove standing water. Block B did not drain as readily as Block A, resulting in longer periods of water-logged conditions. The difference in *K. pentacarpus* growth could be attributed to this unforeseeable difference in conditions.

Our results indicate that *K. pentacarpus* had negligible impact on the growth of planted *S. patens*. Although expansion of *S. patens* circumference was smaller in KS, this was only true in Block B where *K. pentacarpus* cover was low, hence this result is not likely explained by competition but by

a variation of rhizome behavior in the wetter conditions of Block B. *S. patens* vegetative cover was not found to be different among treatments, and neither was the final above-ground biomass sampled at the end of the second growing season (see *S. patens* biomass of S and KS treatment in Table 3), suggesting that any competition between the two species is overall insignificant to the survival and productivity of the planted *S. patens*. Most importantly, *K. pentacarpus* did not hinder growth or cause mortality of planted *S. patens*. Indeed, by the second year of our study, replicate plots in the KS combined treatment were beginning to mirror the natural hierarchy in which one would find these two species in a natural setting.

Additionally, in this study *K. pentacarpus* facilitated the recruitment of *S. patens*. The natural colonization of *S. patens* was significantly higher in the *K. pentacarpus* treatment than

Table 3 Mean above-ground biomass (g/m²) of species at the end of this study: *S. patens* in S and KS treatments and *K. pentacarpus* in K and KS treatments were planted and all other species occurred naturally

Species	C		K		S		KS	
	Block A	Block B	Block A	Block B	Block A	Block B	Block A	Block B
<i>A. tenuifolius</i>	41.2 (±33.1)	4.7 (±2.6)	1.2 (±0.6)	20.5 (±17.2)	1.6 (±0.7)	0.4 (±0.2)	0.8 (±0.8)	0.4 (±0.2)
<i>A. triangularis</i>	98.2 (±32.1)	33.6 (±14.8)	33.1 (±13.1)	18.1 (±6.4)	1.0 (±0.4)	3.4 (±1.36)	20.0 (±9.1)	7.4 (±2.4)
<i>B. halimifolia</i>	0.4 (±0.4)	–	21.1 (±3.6)	11.5 (±5.9)	0.7 (±0.6)	–	0.4 (±0.2)	0.3 (±0.04)
<i>D. spicata</i>	4.8 (±2.5)	–	0.6 (±0.6)	–	11.3 (±11.3)	–	–	–
<i>J. gerardi</i>	21.2 (±3.1)	12.4 (±4.8)	0.6 (±0.4)	12.1 (±1.7)	3.8 (±1.7)	7.4 (±3.9)	–	1.4 (±1.2)
<i>K. pentacarpus</i>	0.8 (±0.8)	–	137.4 (±28.2)	77.6 (±32.7)	–	–	127.8 (±35.2)	45.4 (±12.7)
<i>P. australis</i>	0.1 (±0.1)	47.4 (±44.4)	1.4 (±0.9)	28.1 (±14.8)	3.4 (±3.4)	19.4 (±10.6)	–	–
<i>S. patens</i>	18.4 (±17.6)	116.4 (±54.6)	53.7 (±25.3)	158.2 (±46.5)	1,244.2 (±292.4)	1,083.0 (±72.3)	1,044.7 (±51.8)	1,222.8 (±203.6)
<i>S. salina</i>	0.5 (±0.2)	1.8 (±1.7)	0.1 (±0.1)	2.2 (±1.9)	<0.1 (±0.1)	0.2 (±0.1)	–	0.1 (±0.1)
<i>C. esculentus</i>	5.5 (±5.5)	–	–	–	–	–	–	–
Total	191.9	216.4	249.0	328.3	1,266.0	1,113.7	1,193.8	1,277.8

Values are means (±SE); $n=3$

in the control treatment. *S. patens* percent cover throughout the first growing season was greater in the K treatment (see Fig. 3), as was the *S. patens* above-ground biomass harvested the following year. Specifically, the biomass of naturally occurring *S. patens* in the K treatment was 2.9 times higher than in the control treatment C for Block A and 1.4 times higher for Block B (Table 3).

Research has shown that successful establishment of wetland species is highly related to the availability of seeds, tying a newly created or restored site to neighboring wetlands (Erfanzadeh et al. 2010). The seeds introduced to this study site likely came from two main sources: (1) seeds that were carried in by irrigation water from the marsh creek, and (2) seeds that were wind-dispersed from plants near the experiment. In addition to *S. patens*, our study area was successfully colonized by the annuals *A. triangularis* and *S. salina* in Year 1, and the perennials *B. halimifolia*, *J. gerardi*, and the annual *A. tenuifolius* in Year 2. Control plots were the least consistent in species composition and varied in terms of most dominant species. The robust reemergence of planted *S. patens* individuals during the second growing season increased its dominance in S plots and species richness in these plots decreased (Fig. 2), whereas *K. pentacarpus* plots were rich in *A. triangularis*, *B. halimifolia*, and *S. patens*.

Of particular interest is the dense colonization of *K. pentacarpus* plots by *B. halimifolia* during Year 2 (see Fig. 2 for vegetative cover and Table 3 for biomass). This relationship is desirable as Wang et al. (2006) found that *B. halimifolia* showed promising vegetative blocking of *P. australis* growth potentially due to shading by its thick canopy. Egerova et al. (2003) showed that *Spartina alterniflora* improved *B. halimifolia* recruitment by physically trapping seeds and improving seedling success during early stages of marsh creation. The dense stem canopy of *K. pentacarpus* could be acting in a similar fashion. Overall, our results suggest that cultivation of *K. pentacarpus*, during transition from working land to natural habitat, would create conditions that improve the recruitment and establishment of desirable wetland species.

Management Considerations

For salinized land that can be reclaimed by native vegetation, some might argue for either a direct planting of the project site with desirable species or a hands-off management approach. Direct planting bars an ecologically sound succession from occurring. Garbutt et al. (2006) studied the development of an intertidal system in former agricultural land and found establishment of planted wetland species to be unsuccessful due to unfavorable environmental conditions. On the other hand, a hands-off approach would allow the land to go barren and available to undesirable invasive species and weeds (Baldwin and Mendelssohn 1998). In addition to prolonging the

economic benefit of the land to the farmer, we found that the application of *K. pentacarpus* would provide vegetative cover to salinized land and encourage colonization of desirable species, while it transitions from one state to another during sea level rise, thus encouraging a more competitive transitional community.

In this study, *K. pentacarpus* positively affected soil development. Leaf litter cover and biomass were significantly higher in treatments containing *K. pentacarpus*. In addition, Block A for which *K. pentacarpus* cover was higher was also significantly higher in leaf litter than Block B. The function of leaf litter in nutrient and energy cycling is very crucial (see Facelli and Pickett 1991 for review). Under working conditions, the crop would be sustainably cultivated for multiple years, specifically from the point when traditional crops are killed by the salt and can no longer be economically grown to the time when flooding inhibits working the land. In this time frame, we would expect natural nutrient cycling to be enhanced by the heavy litter deposition of *K. pentacarpus* compared to an agricultural field left fallow.

Ecosystems of higher diversity are less prone to invasion because they are more likely to already have various species exploiting a broad range of niches. For example, Wang et al. (2006) determined that the presence of a “multi-layered wall of plants” was an effective way to reduce growth of the invasive *P. australis*. In our study, replicate plots with *K. pentacarpus* had higher productivity (above-ground biomass and vegetative cover) than control plots and less productivity than *S. patens* plots. However, in *S. patens* plots biomass was primarily composed of *S. patens*. K plots became more diverse than *S. patens*-containing plots and more than or equally as diverse as control plots. This means that a community started around *K. pentacarpus* would be more diverse than a community started directly with *S. patens* and more productive than a community that developed unmanaged. Therefore, the use of *K. pentacarpus* as a transitional crop would maximize diversity and canopy cover, promoting the most desirable combination for a robust plant community.

Whether coastal agricultural fields that become unproductive during sea level rise have the potential to transition into functional wetlands depends on the specific character and hydrology of the site and the time scale of evaluation. However, the growth of *K. pentacarpus* can supply an economic purpose to the land during its transition and has the potential to play a facilitative role, through increasing the recruitment of desirable native vegetation, maximizing the productivity and diversity of the community around it, and contributing organic litter to the soil. Unless the wetland is allowed to migrate upward, this native vegetation is at risk due to increased sea-level rise and coastal development. Therefore, this strategy is a very appropriate and positive use of a native halophytic nurse crop to transition from economically valuable farm land to a coastal ecosystem with economically valuable ecosystem services.

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