2023

Innovative Tidal Control Successfully Promotes Saltmarsh Restoration

Caleb Rankin
University of Newcastle - Australia

Troy Gaston
University of Newcastle - Australia

Mahmood Sadat-Noori
University of New South Wales

William Glamore
University of New South Wales

Jason K. Morton
Avondale University, jason.morton@avondale.edu.au

See next page for additional authors

Recommended Citation

Rankin, C., Gaston, T., Sadat-Noori, M., Glamore, W., Morton, J., & Chalmers, A. (2022). Innovative tidal control successfully promotes saltmarsh restoration. *Restoration Ecology*, e13774.

RESTORATION ECOLOGY
The Journal of the Society for Ecological Restoration
SER SOCIETY FOR ECOLOGICAL RESTORATION

TAILORED RESTORATION RESPONSE: PREDICTIONS AND GUIDELINES FOR WETLAND RENEWAL

RESEARCH ARTICLE

Innovative tidal control successfully promotes saltmarsh restoration

Caleb Rankin[1,2], Troy Gaston[1], Mahmood Sadat-Noori[3], William Glamore[3], Jason Morton[4], Anita Chalmers[1]

The reduction of saltmarsh habitat at a global scale has seen a concomitant loss of associated ecosystem services. As such, there is a need and a push for habitat rehabilitation. This study examined an innovative saltmarsh restoration project in Australia which sought to address the threats of mangrove encroachment and sea level rise. The project was implemented in 2017, using automated hydraulic control gates, termed "SmartGates," to lower the tidal regime over one site, effectively reversing sea level rise at a local level. Measured indicators of saltmarsh cover, number of species, seedling counts, and saltmarsh assemblages all showed significant positive development over time, with trends varying based on saltmarsh zone. The saltmarsh, predominantly *Sarcocornia quinqueflora*, developed from remnant supralittoral (previously high) marsh which remained at 45% cover to achieve over 15% coverage across the cleared habitat after 3 years. Slower development in the low marsh (<5%) compared to other zones contrasts with other saltmarsh restoration studies which may be due to the unique nature of the restoration method or the nature of Australian saltmarsh species which favor higher elevations and drier conditions. The development of saltmarsh at the treatment site was found to track toward that at comparison sites over time, becoming similar to some comparison sites by the studies end. This study highlights the usefulness of the novel restoration method used and of the measured indicators for assessing saltmarsh development. This innovative tidal control method could play an important role in the future of saltmarsh restoration worldwide.

Key words: climate change, mangrove encroachment, novel restoration, restoration evaluation, restoration techniques, sea level rise

Implications for Practice

- Altering the tidal regime of an area using a tidal replicate method can effectively counteract sea level rise to preserve or restore critical saltmarsh where geomorphology allows.
- The tidal replicate method, implemented via an automated gate system, controls the hydrodynamics of a site, allowing for adaptive management of the site. By effectively simulating tidal conditions, this method can enhance restoration and preserve intertidal coastal wetlands in a changing climate.
- The tidal replicate method has practical advantages over other methods, including managed saltmarsh retreat, topsoil addition, and facilitating natural accretion. The method involves minimal physical, chemical, or biological disturbances, can be easily adapted for changing circumstances, replicates natural system response, and effectively combats sea level rise at short time scales.

Introduction

Saltmarshes are being rapidly lost around the world due to sea level rise, coastal development, pollution, invasive species, and mangrove encroachment which displaces saltmarsh habitat in subtropical and temperate regions (Saintilan et al. 2014; Adam 2019). The issue of mangroves displacing saltmarsh is a recent phenomenon, where a few cold tolerant mangrove species are extending past their historic poleward limits displacing previously diverse and extensive saltmarsh habitat (Saintilan et al. 2019). Displacement is restricted to cooler climates as warmer latitudes have a higher diversity of mangroves, limited

Author contributions: CR, AC designed study and wrote manuscript; MS-N, WG, JM, TG reviewed the manuscript; CR, MS-N, AC implemented fieldwork; AC identified species; CR, JM, AC analyzed the data; WG, MS-N designed restoration project; WG, TG secured funding.

[1]School of Environmental and Life Sciences, University of Newcastle, Newcastle, New South Wales 2258, Australia
[2]Address correspondence to C. Rankin, email caleb.rankin@uon.edu.au
[3]School of Civil and Environmental Engineering, Water Research Laboratory, University of New South Wales Sydney, Sydney, New South Wales 2093, Australia
[4]School of Education and Science, Avondale University, Cooranbong, New South Wales 2265, Australia

doi: 10.1111/rec.13774
Supporting information at:
http://onlinelibrary.wiley.com/doi/10.1111/rec.13774/suppinfo

diversity, and extent of saltmarsh, and mangroves are in decline (Woodroffe et al. 2016; Saintilan et al. 2019). In many areas an increase in mangrove abundance is encouraged due to their ecosystem services, including carbon sequestration, and vertical accretion counteracting climate change and sea level rise (Krauss et al. 2014; Huxham et al. 2018; Bertolini & da Mosto 2021).

The rate and extent of saltmarsh loss worldwide is of international concern with widespread impacts on biodiversity, migratory shorebird and fish habitat (Prosser et al. 2017), estuarine and coastal productivity (Svensson et al. 2007; Altieri et al. 2012; Raoult et al. 2018), water quality, flood regulation (Narayan et al. 2017), and carbon sequestration (Poffenbarger et al. 2011; Brittney et al. 2018). To address these issues, governments and nongovernmental organizations around the world are taking steps to preserve and restore saltmarshes. However, saltmarsh loss continues, and with sea level rise projections for the year 2100 ranging from 0.61 to 2 m (Bamber et al. 2019; Horton et al. 2020) and ever-increasing coastal development, there is a reduction in the number of areas where approaches using landward saltmarsh retreat or depending on vertical accretion rates to restore or preserve saltmarsh is feasible.

In response to these issues, a novel restoration experiment is being conducted in Australia utilizing automated hydraulic control gates (powered onsite with solar panels), termed "SmartGates" (Roman & Burdick 2012), to promote restoration and preservation of saltmarsh through dynamically lowering the tidal regime (Sadat-Noori et al. 2021). Various projects around the world have utilized tidal gates and hydrological manipulation to restore saltmarshes, such as Glamore et al. (2021), Masselink et al. (2017), and Smith and Medeiros (2013). However, the majority of these have focused on the reintroduction of tidal flows into reclaimed saltmarsh habitat and have not explored the use of SmartGates to counteract sea level rise through lowering tidal regimes to prevent or reverse the drowning of saltmarsh habitat where landward retreat is not a feasible option. Lowering the tidal regime has wide implications for future saltmarsh restoration and preservation efforts around the world where local geomorphic conditions are suitable, namely, with a channel suitable for installing a SmartGate. This includes over 1,184,000 ha of internationally important Ramsar wetlands around the world (Sadat-Noori et al. 2021). Used effectively, this method could expand the current extent of saltmarsh habitat (Howe et al. 2010).

This method was trialed in the Hunter River estuary of New South Wales, Australia. A SmartGate was installed at the entrance of a historically extensive saltmarsh wetland which had been mostly displaced by the mangrove *Avicennia marina* since the 1990s. The mangroves were removed and the tidal regime dynamically altered to promote maximal saltmarsh growth and limit future mangrove encroachment (Sadat-Noori et al. 2021). Nets placed across the inlet assisted with this by trapping the majority of mangrove propagules while allowing saltmarsh seeds and nekton through. At this site, hereafter known as Area E, Sadat-Noori et al. (2021) found that saltmarsh cover had increased significantly since restoration but, as has been noted elsewhere, saltmarsh growth through reinstating physical and hydrological conditions does not always lead to healthy and diverse functioning ecosystems (Sullivan et al. 2017). Here, we report on the development of saltmarsh vegetation cover, species richness, and composition in Area E up to 3 years post restoration work and note how it has developed in relation to local comparison sites. In this study, we hypothesized that saltmarsh vegetation development between sites, zones, and sampling times would show meaningful ecological trends in support of a novel restoration treatment that could preserve saltmarsh sites of importance against sea level rise.

Methods

Study Area and Novel Restoration Method

The Hunter River estuary is one of Australia's most highly modified estuaries with approximately 90% of saltmarsh lost in the past century (Williams et al. 2000). With the Hunter River estuary containing Ramsar wetlands of international importance, primarily due to providing important shorebird habitat and supporting estuarine productivity (Ramsar 2012), the protection and restoration of these saltmarshes is a high priority. At Area E (Fig. 1), state government and nongovernmental organizations, assisted by local volunteers, are implementing mangrove removal, strategic tidal control, and managed saltmarsh retreat to restore and conserve saltmarsh in the face of sea level rise over a 24 ha site (NSW National Parks and Wildlife Service 2015; Sadat-Noori et al. 2021). Before the early 2000s, Area E was predominantly saltmarsh with some pasture grasses, but has since experienced sea level rise and flooding events after the removal of culverts and widening of the creek entrance. The associated increase in tidal inundation has aided rapid mangrove encroachment into the saltmarsh habitat (Boys & Williams 2012). This has reduced the use of the saltmarsh as roosting habitat for migratory shore birds which is just one of the important roles of these wetlands, supporting 45 species of migratory birds listed under international agreements as well as several waterbirds listed as endangered (Ramsar 2012). Two of the comparison sites in this study (i.e. Cobbans Creek and Crabhole Flats; Fig. 1) have had similar histories of culvert removal and mangrove encroachment, but it is Area E that has subsequently been the subject of various studies reviewing the impact of mangrove encroachment and the need for their removal (Howe et al. 2010; Boys & Williams 2012).

Restoration efforts at Area E focusing on mangrove removal, utilizing heavy machinery and follow-up hand removal of seedlings by volunteers, was undertaken in 2016 and completed in 2017 (Sadat-Noori et al. 2021). Heavy machinery was constrained to access roads and parked off site to minimize damage to the landscape. The SmartGate was installed over the inlet with the assistance of local contractors and, after 2 months of construction, began operation by February 2017 (Sadat-Noori et al. 2021) costing approximately AU$200,000 including construction, initialization, and maintenance. Similar to the nearby Tomago Wetlands (Glamore et al. 2021), and as previously described by Sadat-Noori et al. (2021) for Area E, the local saltmarsh elevation range, tidal data, and hydrodynamic modeling was used to determine a hydrological regime that supports saltmarsh growth which was then implemented by the SmartGates. This modeling found that local saltmarsh grew between the

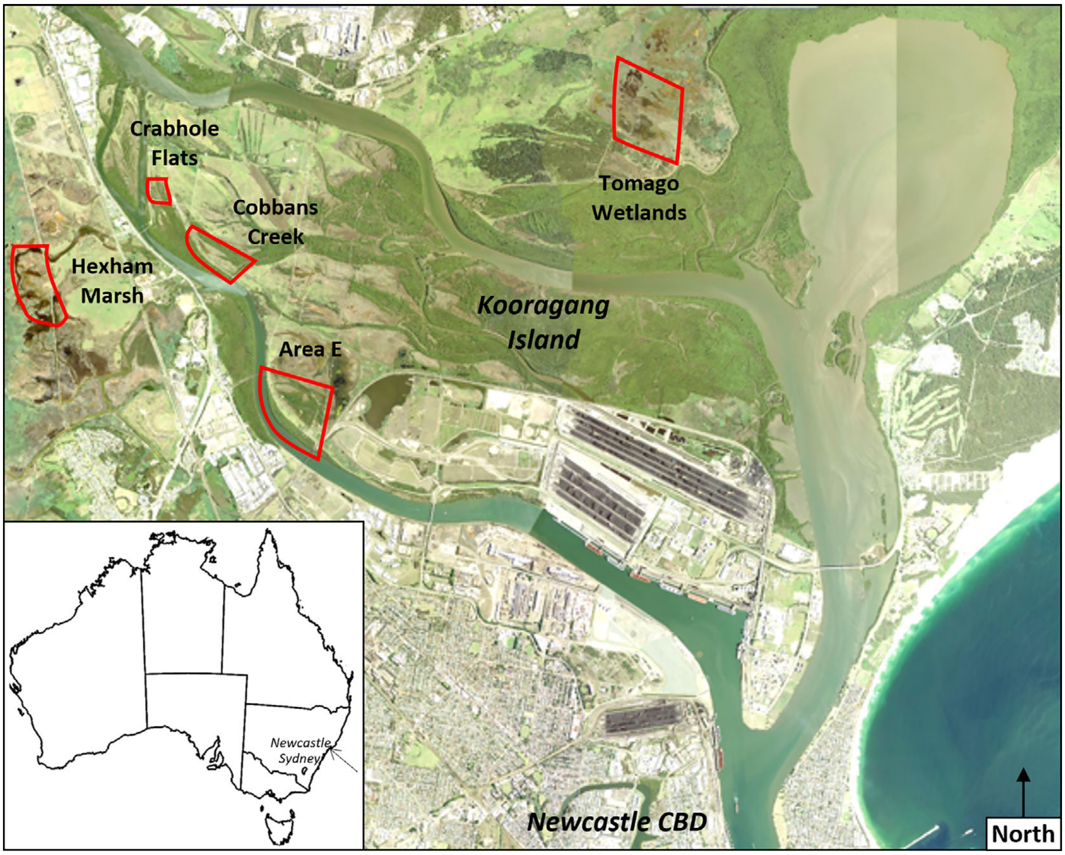

Figure 1. Location of treatment (Area E) and comparison sites in the Hunter River estuary, New South Wales, Australia. Map credit: NSW SIX Maps (2019).

mean high-water levels, around 0.75 m Australia Height Datum (AHD) locally, to the highest astronomical tides.

Based on 14 sites near Area E in the lower Hunter River estuary, surveys using real-time kinematics-GPS identified the elevation range of local saltmarsh habitat with the median saltmarsh level being 0.77 m AHD. Local tidal data and hydrodynamic models were then used to understand the inundation patterns and frequency over established saltmarsh habitat. This frequency and pattern of inundation was then replicated at Area E and a synthetic tide lowered by 0.45 m was then developed to optimize saltmarsh growth (and minimize mangrove favorable inundation patterns) based on high accuracy surveys of the site's topography (Sadat-Noori et al. 2021). The synthetic tide applied onsite included king, spring and neap tidal variations in efforts to maintain the remnant high marsh (and eliminate exotic grass incursions) present before restoration while maximizing saltmarsh habitat as proposed by Howe et al. (2010). This tidal restoration method is novel and, if successful, has wider implications around the world for saltmarsh at risk from sea level rise. Full details of the tidal replicate method can be found in Sadat-Noori et al. (2021).

Four comparison sites were used to compare the development of vegetation at Area E with established saltmarsh sites (Table 1; Fig. 1). Historical information in the form of past studies, monitoring and land use reports, historic imagery, and maps, were consulted to inform site selection. All sites had some similarities in disturbance histories, both direct (e.g. grazing, invasive species, and vehicle use) and indirect (e.g. culverts and eutrophication). Tomago Wetlands and Hexham Marsh are both large saltmarsh sites (in excess of 400 ha) where tidal inundation has been reestablished (Boys 2015; Glamore et al. 2021; Rayner et al. 2021). Cobbans Creek and Crabhole Flats are both smaller saltmarsh sites with similar hydrology to Area E. These were both included in the 14 sites on which Area E's hydrological modeling was based. Given the differences in geography, hydrology, historical impacts and saltmarsh restoration history of sites on Kooragang Island and in other parts of the Hunter River estuary, saltmarshes were expected to vary between sites and so Area E was tracked toward a comparison site range informed by multiple sites (White & Walker 1997; Craft 2016; Gann et al. 2019) as opposed to a single comparison site.

Experimental Design and Sampling

Area E was sampled on eight occasions from November 2017 to February 2020 across summer and winter seasons (Table 1). Comparison sites were sampled in November 2018, July 2019, and November 2019. An additional sampling time in June 2018 was conducted for Cobbans Creek and Crabhole Flats as these sites experience similar hydraulic conditions to Area E pretreatment

Table 1. Outline of restoration sampling times at each of the study sites.

Site	*Restoration*	*Size (ha)*	*Sampling Times*								
			Nov 2017	*Feb 2018*	*Jun 2018*	*Nov 2018*	*Mar 2019*	*Jul 2019*	*Nov 2019*	*Feb 2020*	n
Area E	2017 (from mangroves)	48	X	X	X	X	X	X	X	X	8
Tomago	2007 (from pasture)	>300				X		X	X		3
Hexham	2008 (from freshwater wetland and pasture)	>600				X		X	X		3
Cobbans Creek	1990 (from pasture and degraded saltmarsh)	80			X	X		X	X		4
Crabhole Flats	1993 (from pasture)	20			X	X		X	X		4

(Table 1). Saltmarsh cover and diversity is typically recorded using replicate 1-m^2 quadrats (Zedler et al. 1995; Laegdsgaard 2006; Craft 2016). Thus, saltmarsh development in the current study was measured using random replicates of 1-m^2 quadrats, similar to Matthews et al. (2009) and Veldkornet et al. (2015). Strings divided each quadrat into twenty-five 400-cm^2 sections to assist accuracy of visual surveys. Quadrat placement was stratified-random, with an equal number of quadrats in each level of stratification (high, mid, and low marsh). In Area E, a fourth zone comprising the remnant high marsh community, now termed the supralittoral marsh, was measured. Each quadrat was permanently marked for relocation with pegs, and the latitude and longitude of each quadrat was recorded and mapped in QGIS software.

Indicators assessed were total saltmarsh cover per square meter, number of saltmarsh species present, the percentage cover of each saltmarsh and weed species, and saltmarsh seedling counts (for the species in this study the presence of cotyledons increased the certainty that recently germinated seedlings could be distinguished from older seedlings or remnant mature individuals of very small cover). Identifications and nomenclature for plants were based on Harden (1990–2000), and augmented by the National Herbarium of NSW (n.d.) and the Centre for Australian National Biodiversity Research (2017). The usefulness of these measures for assessing restoration trajectories were evaluated based on criteria of results quality (preference for low variability as represented by standard error), sensitivity, and usefulness for target setting (Kurtz et al. 2001; Niemi & McDonald 2004; Jørgensen et al. 2016; Queirós et al. 2016).

The three zones of high, mid and low marsh sampled at each site were identified by hydroperiod and informed by local topography and publicly available 2 m^2 2014 digital elevation models (DEMs) (Digital Elevation Data | Geoscience Australia [https://www.ga.gov.au]). Zones in Area E were identified by 2017 DEMs (1-m resolution) and had lower elevations (−0.45 m AHD) than the comparison sites with equivalent numbers of tidal inundations due to the altered tidal regime from the SmartGate (Fig. 2). Out of 700 annual tides, 45 inundated the high marsh zone (above 0.90 and 0.45 m AHD for local comparison sites and Area E, respectively). The mid marsh was inundated by 50 tides (above 0.81 and 0.36 m AHD for comparison sites and Area E) and the low marsh was inundated by 600 tides annually (above 0.75 and 0.30 m AHD in comparison sites and Area E, respectively). At Area E, the zone comprising the entire remnant saltmarsh before initialization of SmartGates and removal of mangroves was inundated by 0.01% of annual tides (above 0.55 m AHD). Quadrats were placed according to randomly generated numbers within the predetermined saltmarsh zones based on DEM mapping and local topography. At Area E, 25 quadrats were placed in each of the four zones (i.e. a total of 100 quadrats; Fig. 2). In the comparison sites, 10 quadrats were placed in each of the three zones (i.e. a total of 30 quadrats per site).

Statistical Analyses

The null hypotheses of no changes in Area E over time and no differences between sites for the indicators of (1) total saltmarsh cover, (2) number of saltmarsh species, (3) percent weed cover, (4) seedling counts, and (5) saltmarsh assemblage were tested using distance based permutational multivariate analysis of variance (PERMANOVA+ for PRIMER) tests in PRIMERv7 (Anderson 2017).

To identify changes over time at Area E, a two-factor PERMANOVA was used with the factors of sampling time (eight sampling times) (Table 1) and saltmarsh zone (four zones: low, mid, high, and supralittoral) (Fig. 2). Note that the fourth zone, that is, supralittoral, was exclusive to Area E. To assess development at Area E in relation to comparison sites, a three-factor PERMANOVA was used with the factors of sampling time (three for Hexham Marsh and Tomago Wetlands and four for Area E, Cobbans Creek and Crabhole Flats; Table 1), sites (five sites; Fig. 1), and saltmarsh zones (three zones; Fig. 2). Monte Carlo *p*-values were obtained from 999 permutations of residuals. Where the two- and three-factor PERMANOVA analyses found factors or interactions to be significant, pair-wise tests were used to assess where, and to what extent, differences occurred.

Due to the nature of the sampling design as repeated indicators (i.e. fixed quadrats sampled repeatedly over consecutive sampling times), a similarity matrix was created to test for the exchangeability of samples before analysis as recommended by Clarke et al. (2014). This was done using square root transformation of raw data to create a Bray–Curtis similarity matrix for each of the indicators. Saltmarsh assemblages were visualized in two-dimensional space using nonmetric multidimensional-scaling (NMDS) ordination plots. These NMDS plots gave a useful representation of three-dimensional multivariate data with stress values of less than 0.05. The similarity percentages (SIMPER) routine in PRIMERv7 was used to determine the species that typified vegetation communities

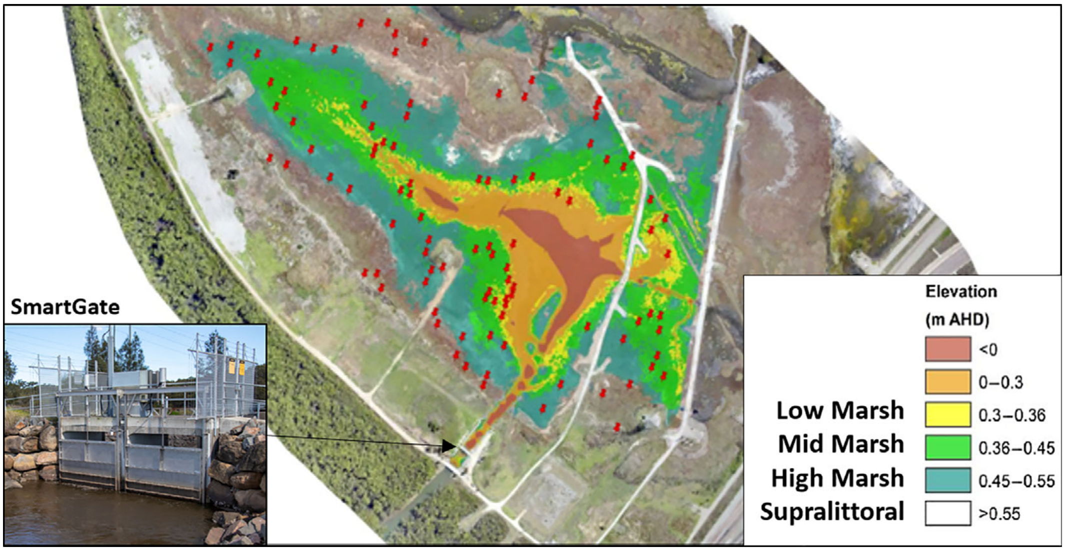

Figure 2. Distribution of quadrats (red pins) at Area E and digital elevation model (DEM) illustrating expected zones of low marsh (yellow), mid marsh (green), and high marsh (blue), and the remnant supralittoral marsh above altered high tides (no color). Arrow shows the location of the SmartGate (photo inset) at the inlet. The central compartment on top of the SmartGate protects all electronics from exposure and prevents any environmental contamination.

within sampling events and sites, and to identify differences in these between Area E and comparison sites.

Results

Total Saltmarsh Cover

Saltmarsh cover changed significantly over the duration of this study. For the two-way interaction between sampling time and zone in Area E (pseudo-$F = 3.57$, $p \le 0.001$) (Table S1A), it was found that the mean mid and high marsh cover changed the most between sampling times, increasing from <1 to 15% (± 4%) ($p \le 0.001$) and 23% (± 4%; $p \le 0.001$), respectively, by the end of the study (i.e. February 2020) (Fig. 3A). Low marsh cover changes were less pronounced, increasing from 0 to 2% mean cover (± 0.7%; $p < 0.002$). The supralittoral marsh cover showed little variation between sampling times with no difference between the beginning and end of the study ($p = 1.0$; Fig. 3A).

Significant differences in saltmarsh cover from the start of the study first occurred in the high and mid marsh at the fourth sampling time ($p \le 0.001$). Cover in both zones increased consistently thereafter until the study's end. However, cover remained lower than in the supralittoral marsh at the end of the study ($p < 0.02$). Cover increases in the low marsh became significant in the fifth sampling event (March 2019) ($p < 0.002$; Fig. 3A). Saltmarsh cover in the low marsh increased slower than in the high and mid marsh and remained consistently lower than in other zones ($p \le 0.001$).

Three-factor PERMANOVAs (Table S2A) showed two-way interactions between sampling time and site ($F = 4.04$, $p \le 0.001$) and sampling site and zone ($F = 3.86$, $p \le 0.001$). Site comparisons for total saltmarsh cover showed Area E trending toward the comparison sites for all zones and that there was considerable variation between comparison sites (Fig. 4). By February 2020 (i.e. 3 years after restoration commenced), cover in the high marsh of Area E was the least different to comparison sites but still significantly lower ($p < 0.01$; Fig. 4B). Saltmarsh cover in the low marsh remained the most different to the comparison sites throughout the study period ($p \le 0.001$) (Fig. 4D).

Number of Saltmarsh Species

The saltmarsh species *Sporobolus virginicus*, *Suaeda australis*, *Sarcocornia quinqueflora* ssp. *quinqueflora*, *Spergularia marina*, *Triglochin striata*, *Atriplex semibaccata*, *Cotula coronopifolia*, and *Juncus kraussii* were recorded at Area E during this study. Similar to saltmarsh cover, pairwise analysis on the time-zone interaction (pseudo-$F = 3.23$, $p \le 0.001$) (Table S1B) confirmed that the number of species changed the most in the mid and high marsh zones. The number of species increased from a mean of 0.15–1.30/m^2 ($p \le 0.001$) and from 0.19 to 1.69/m^2 ($p \le 0.001$) in the high and mid marsh, respectively (Fig. 3B). There were no significant differences over time in the number of species for the supralittoral marsh. The number of saltmarsh species in the low marsh did not increase as fast as in the high and mid marsh, increasing from 0.00 to 0.48/m^2 ($p \le 0.001$) by November 2019.

The number of species in the high and mid marsh first increased significantly from the first sampling event between the third and fourth (July and November 2018) sampling events ($p < 0.01$), while a significant low marsh change was observed in the fifth (March 2019) sampling event ($p < 0.01$) (Fig. 3B). Again, values in the low marsh remained lower than in the high and mid marsh. However by November 2019, 2 years after sampling commenced at Area E, the number of species in the high and mid marsh were not significantly different from the supralittoral marsh ($p > 0.05$) (Fig. 3B).

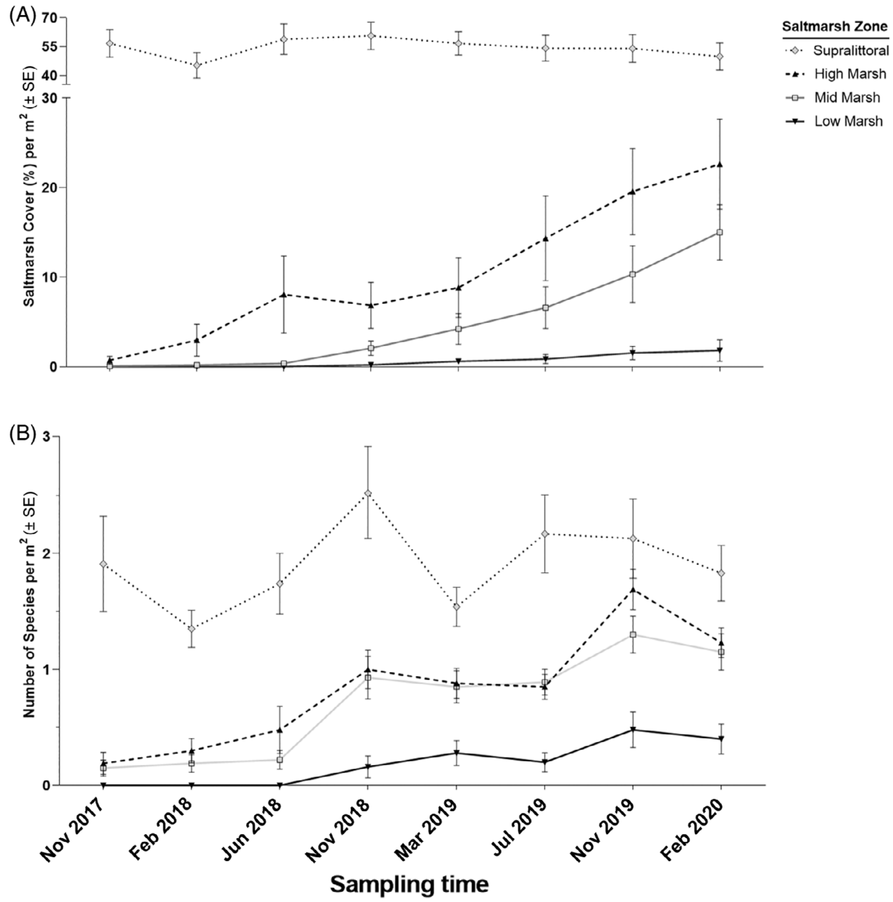

Figure 3. Area E changes in mean percent total saltmarsh cover (A) and number of saltmarsh species (B) per square meter (± SE) separated by saltmarsh zone (supralittoral, high, mid, and low marsh). Measurements are from 25 permanent random quadrats per zone within each sampling time (see Fig. 2).

Two-way interactions between sampling time and site ($F = 6.10$, $p \leq 0.001$) and sampling site and zone ($F = 4.85$, $p \leq 0.001$) were evident in the three-factor PERMANOVA for number of saltmarsh species (Table S2B). The number of species in Area E approached the comparison sites range faster than saltmarsh cover (Fig. 5). By November 2019, the number of species in the high marsh were comparable to those in comparison sites ($p > 0.1$; Fig. 5B). Area E's mid marsh was not significantly different to Hexham Marsh and Tomago Wetlands ($p > 0.1$; Fig. 5C), while the number of species in the low marsh remained significantly lower than in all comparison sites ($p \leq 0.001$; Fig. 5C). Within the comparison sites, differences were limited to Tomago Wetlands having lower counts than Cobbans Creek in November 2019 ($p < 0.005$). In the supralittoral marsh of Area E, the number of saltmarsh species was similar to comparison sites throughout the study (Fig. 5A).

Weeds and Seedlings

Little to no variation in percentage weed cover was observed between sampling times at Area E. Weeds found consisted primarily of salt tolerant pasture grasses, including the introduced *Pennisetum clandestinum*, *Lolium* spp., and *Ehrharta* spp. Members of the Asteraceae family, including *Aster subulatus* (synonym *Symphotrichum subulatum*), *Taraxacum officinale*, *Lactuca serriola*, *Senecio madagascariensis*, and *Sonchus oleraceus* were also present. Most weeds were constrained to the supralittoral marsh, with weed growth in the high marsh limited to February and June 2018 (mean < 0.2%/m^2) and consisting primarily of *Juncus acutus*.

In contrast, the number of seedlings, mostly comprising of succulents, was variable for Area E with a significant two-way interaction between zone and sampling time (pseudo-$F = 4.12$, $p \leq 0.001$) (Table S4A). Seedling counts were highest (upwards of 100/m^2) in July 2019 (i.e. winter), 3 years after the restoration treatment at Area E began (Fig. 6). Other significant seedling recruitment events occurred in June 2018 (winter) and November 2018 (spring) (Fig. 6). Low marsh mean seedling counts were only lower than the high and mid marsh in July 2019 ($p < 0.01$). Mean seedling counts in the supralittoral, high and mid marsh were not significantly different to each other throughout sampling times ($p > 0.05$) (Fig. 6). Fewer than 25 mangrove seedlings were observed growing in Area E. These were outside the quadrats and thus were not recorded for analyses. These were entirely restricted to the low marsh and mudflat zones with no recruitment observed in the developing mid and high marsh.

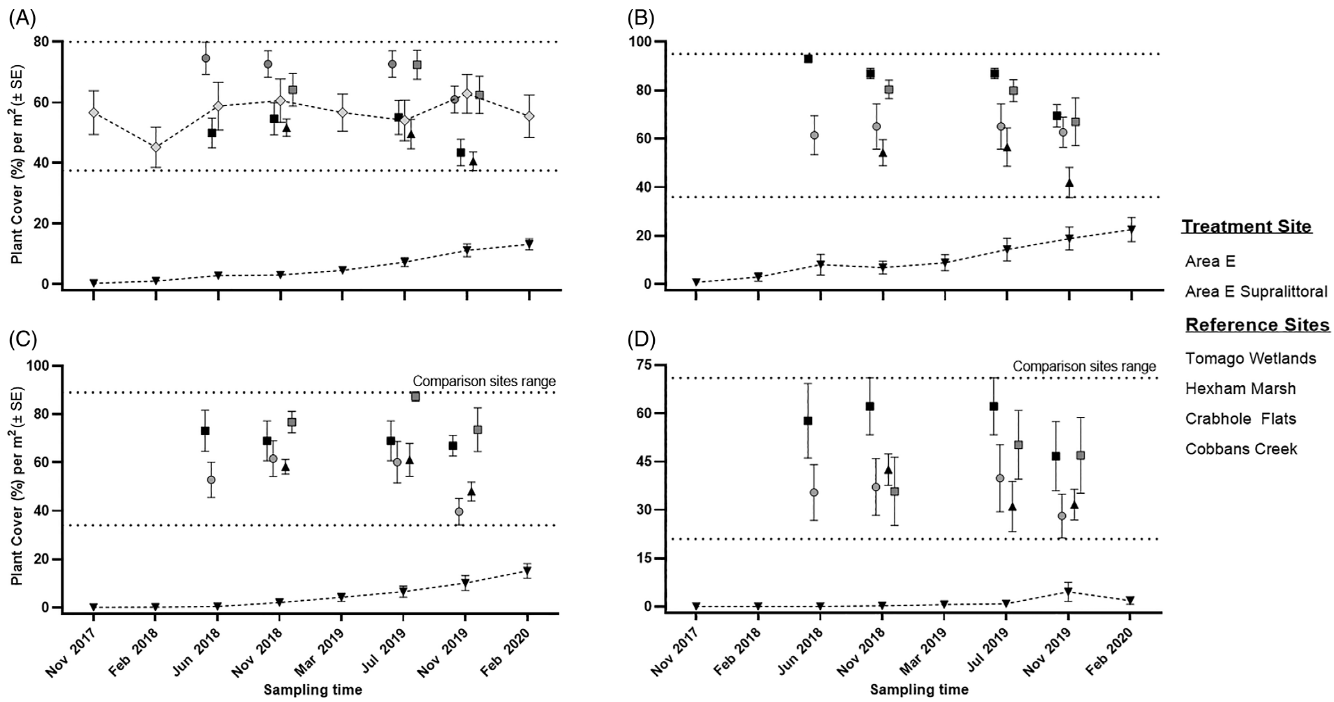

Figure 4. Mean saltmarsh cover per square meter (± SE) in treatment (Area E) and comparison sites from November 2017 to February 2020, separated by zones supralittoral marsh (for Area E) (A), high marsh (B), mid marsh (C), and low marsh (D). Measurements are from 25 (Area E) and 10 (comparison sites) permanent quadrats at each zone on each sampling time (see Fig. 2).

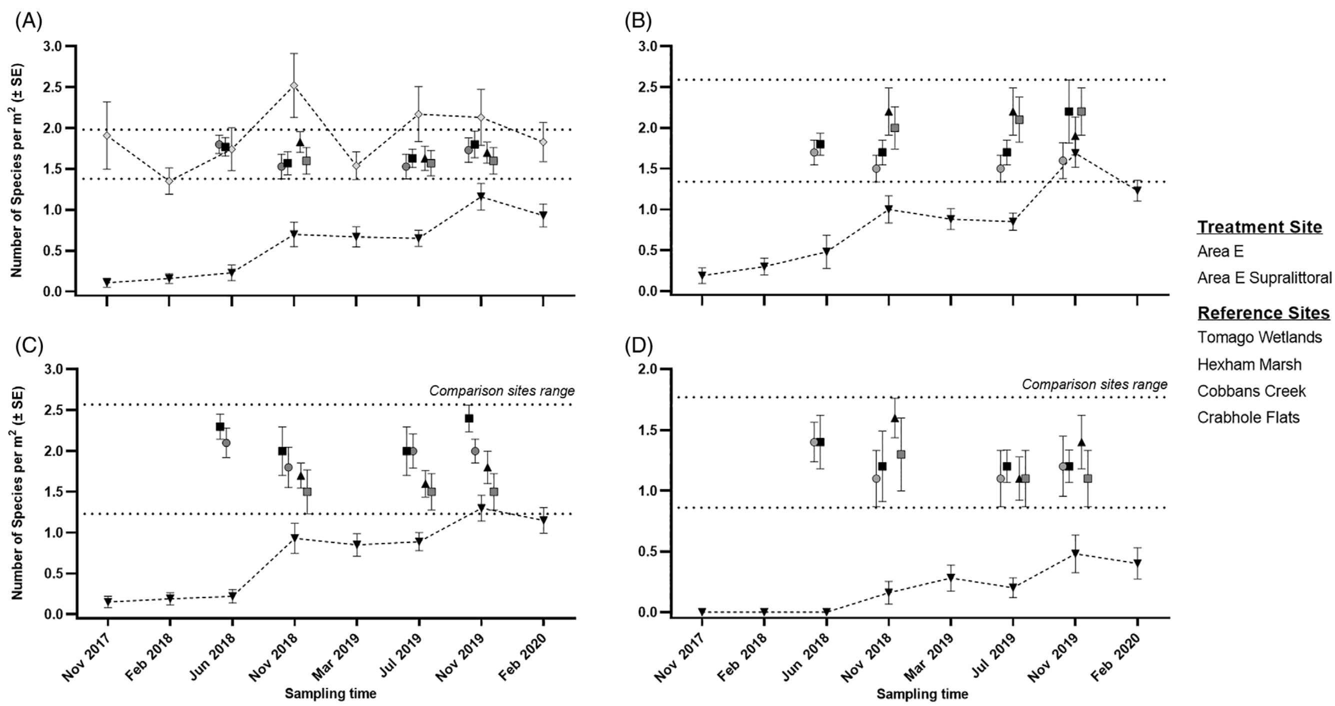

Figure 5. Mean number of saltmarsh species per square meter (± SE) in treatment (Area E) and comparison sites from November 2017 to February 2020, separated by zones supralittoral marsh (for Area E) (A), high marsh (B), mid marsh (C), and low marsh (D). Measurements are from 25 (Area E) and 10 (comparison sites) permanent quadrats at each zone on each sampling time (see Fig. 2).

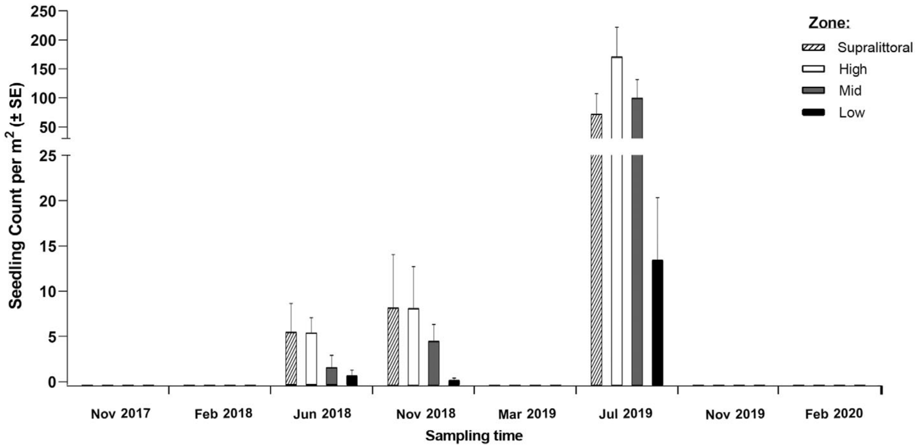

Figure 6. Area E mean seedling counts per square meter (± SE) from November 2017 to February 2020. Measurements are from 25 permanent quadrats for each of the saltmarsh zones in each sampling time (see Fig. 2). Note that only June 2018, November 2018 and July 2019 had mean seedling counts more than 0.1/m^2.

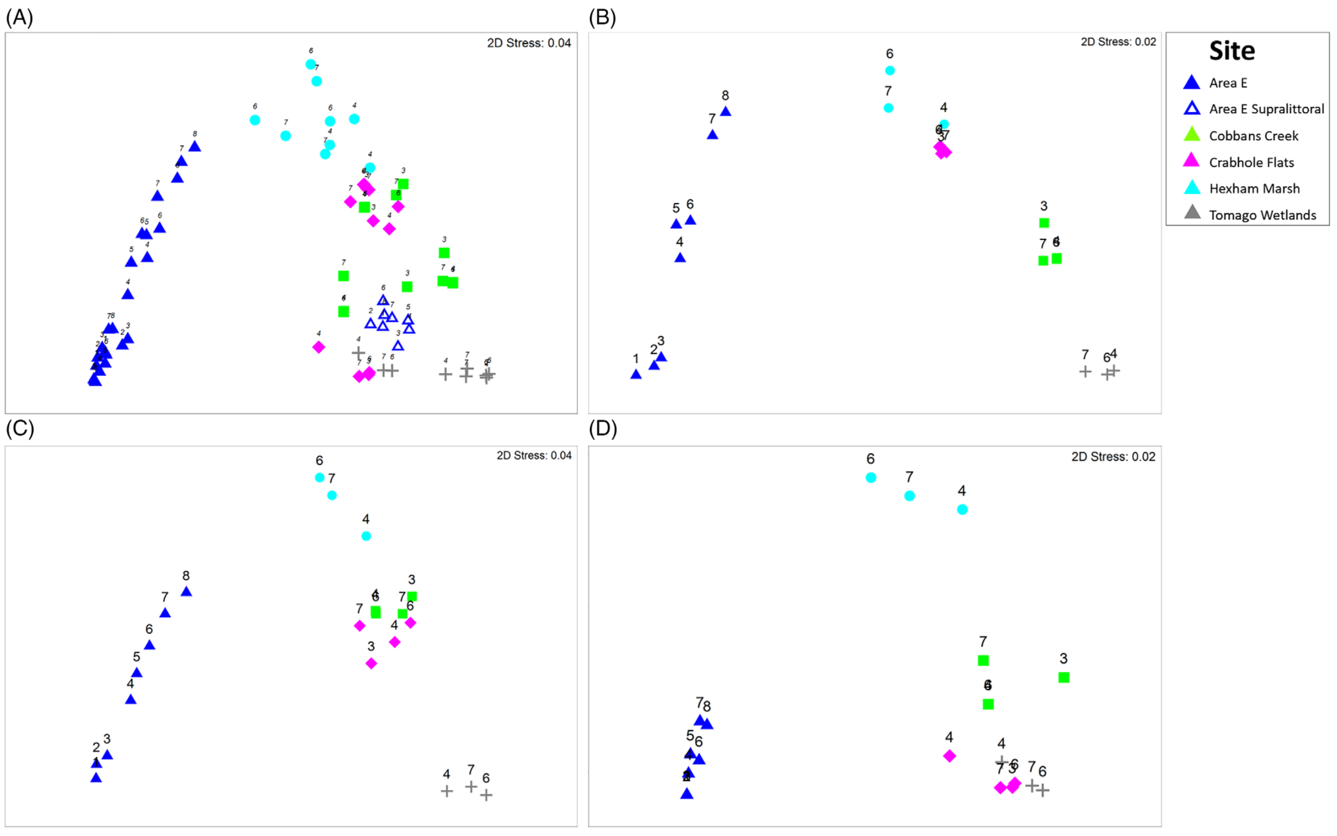

Figure 7. NMDS plots of average similarity of saltmarsh assemblages between the treatment site (Area E) and comparison sites, for all sampling times. Each point represents the average of all quadrats sampled within a sampling time, site, and zone. NMDS plots are separated into combined zones and (for Area E) supralittoral marsh (A), high marsh (B), mid marsh (C), and low marsh (D). Sampling times are denoted by numbers 1 (November 2017) to 8 (February 2020) and sites are distinguished by color and shape. The trajectory of Area E composition over time moves from low similarity compared to all sites (bottom left) to higher similarity with comparison sites.

Between sites, two-way interactions between sampling time and site ($F = 8.23$, $p \leq 0.001$) and sampling site and zone ($F = 6.85$, $p \leq 0.001$) were evident (Table S3A). Seedling counts in comparison sites were consistently under a mean of 1/m^2 through all sites, zones, and sampling times with little variation between sites ($p > 0.1$). Area E seedling counts were significantly higher than in the comparison sites in June 2018 and July 2019 ($p < 0.005$ and 0.001, respectively). The mid marsh in Area E was the most consistently different to comparison sites through all sampling times ($p < 0.01$). However, total Area E seedling counts in November 2018 were not different to Crabhole Flats and Hexham Marsh ($p > 0.1$).

Saltmarsh Assemblages

PERMANOVA analyses on Area E saltmarsh assemblages revealed a significant two-way interaction between zone and sampling time (pseudo-$F = 2.67$, $p \leq 0.001$) (Table S4B). Zones in Area E showed little change in the first three sampling events, remaining clustered on the lower left of the NMDS plot (Fig. 7B & 7D). Changes in the saltmarsh assemblages in Area E were most pronounced between sampling events three and four in the high and mid marsh ($p \leq 0.001$) (Fig. 7B & 7C). In the high marsh of Area E, a pronounced assemblage change occurred again between sampling events six and seven (Fig. 7B). A significant change in the low marsh saltmarsh assemblage only occurred in the fifth sampling event ($p = 0.01$) with little change thereafter.

Between sites, two-way interactions between sampling time and site ($F = 3.18$, $p \leq 0.001$) and sampling time and zone ($F = 6.12$, $p \leq 0.001$) were evident (Table S3B). In the high and mid marsh, Area E showed a trend toward comparison sites from the lower left to the upper middle of the NMDS plots (Fig. 7B & 7C) was closest in similarity to Hexham Marsh. The low marsh showed a similar but significantly reduced trend (Fig. 7D). The supralittoral marsh was, and remained, similar to the comparison sites (Fig. 7A), with pairwise tests showing no change over time for this zone. There were significant assemblage differences between comparison sites (Fig. 7A), most notably (i.e. $p \leq 0.001$ for all sampling times and zones) between Hexham Marsh (top middle of NMDS plots) and Tomago Wetlands (lower right of plot). In contrast, Cobbans Creek and Crabhole Flats remained similar to each other in all sampling times ($p > 0.1$), with the highest similarity shown in the mid marsh (Fig. 7C).

Saltmarsh Species Driving Assemblage Dissimilarity

Three saltmarsh species comprised over 95% of the total saltmarsh cover at each site, zone, and sampling time. These were the perennial succulents *Sarcocornia quinqueflora* and *Suaeda australis*, and the grass *S. virginicus*. Relative cover for the other recorded species, which included *Spergularia marina*, *Triglochin striata*, *Atriplex semibaccata*, *Cotula coronopifolia*, *Juncus kraussii*, and the invasive saltmarsh species *Juncus acutus*, totaled less than 5% in any site, zone, or sampling time.

In Area E, *S. virginicus* was the dominant saltmarsh species in the supralittoral marsh throughout the study (84.7–95.2% of total saltmarsh cover) but had low cover in the high marsh throughout the study (<1.8% of total cover; Fig. 8A & 8B) and was not recorded in the mid or low marsh. Dissimilarity between assemblages in supralittoral marsh and other Area E zones, as revealed by SIMPER, was in excess of 90% and was driven by higher *S. virginicus* cover in the supralittoral (Fig. 8A) and higher

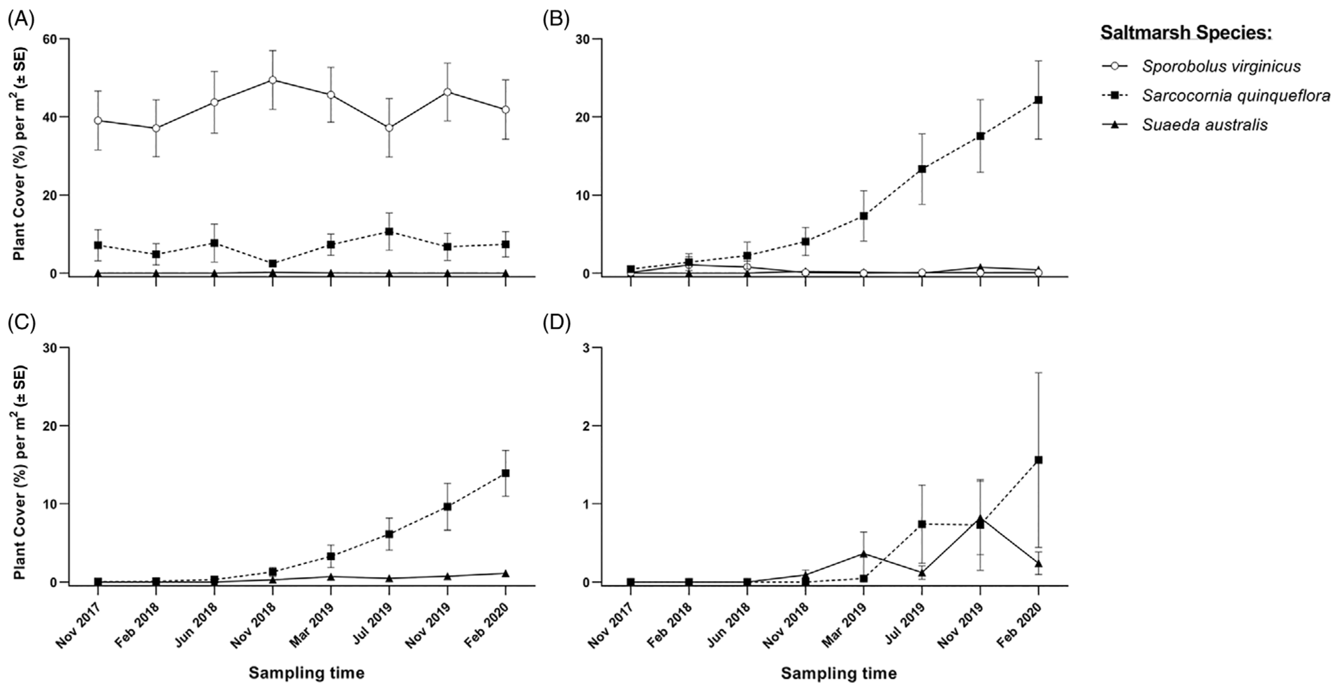

Figure 8. Area E mean percent plant cover per square meter (± SE) for the dominant saltmarsh species from November 2017 to February 2020. Graphs are separated by zones supralittoral marsh (for Area E) (A), high marsh (B), mid marsh (C), and low marsh (D). Note that ranges of the individual *y*-axes differ between graphs to best display trends.

S. quinqueflora cover in the high and mid marsh (Fig. 8B & 8C). In the low marsh, *Suaeda australis* and *S. quinqueflora* had similar proportional covers (6.8–100% and 8.6–93.2%, respectively), with earlier recruitment and growth of *S. australis* (Fig. 8D).

SIMPER identified two species, *S. virginicus* and *S. quinqueflora*, as being most responsible for driving differences between Area E and comparison sites. For identified species, Area E consistently had lower percentage coverage. Both species were identified when comparing Area E to Cobbans Creek for each zone, to Crabhole Flats for the high and mid marsh, and to Hexham Marsh for the high marsh. Only *S. virginicus* was identified as driving differences when comparing Area E with Tomago Wetlands for each zone, and when compared to Crabhole Flats for the low marsh. Conversely, only *S. quinqueflora* was identified as driving differences when comparing Area E to Hexham Marsh for the mid and low marsh.

Discussion

In 2016, a novel restoration method of a lowered synthetic tide and removal of mangroves was implemented to encourage maximal saltmarsh growth. The removal of mangroves at the restoration site of Area E left large areas of bare ground with remnant patches of saltmarsh at higher elevations, including the supralittoral marsh where introduced pasture grasses co-occurred. The use of a SmartGate to lower the tidal regime created a site theoretically optimized for saltmarsh growth through reducing average inundation and maintaining king and spring tides, while nets placed across the inlet canal aimed to reduce the entry of mangrove propagules into the site, preventing most mangrove recruitment with only a few observed recruiting in the low marsh and mudflats. This presented a unique opportunity to measure a range of vegetation indicators to assess saltmarsh establishment over 3 years in an environment optimized, to the best of our knowledge, for maximal saltmarsh development.

This study shows that the novel saltmarsh restoration project has been successful in promoting saltmarsh development. Since November 2017, saltmarsh cover increased consistently toward that of local comparison sites, with expected seasonal variation in the number of species. Interestingly, the two largest comparison sites, Hexham Marsh and Tomago Wetlands which were in excess of 300 ha, were the most and the least different to Area E by the studies end while the similarities of the smaller Cobbans Creek and Crabhole Flats fell between these two. This suggested no relationship between saltmarsh size and similarity in composition. The developing saltmarsh at Area E is predominantly *S. quinqueflora* with very limited *S. virginicus* observed outside the supralittoral marsh. This proportional cover was most similar to Hexham Marsh, whereas other comparison sites and the remnant supralittoral saltmarsh at Area E had higher dominance of *S. virginicus*. This early dominance of *S. quinqueflora* in Area E is likely due to its fast growth rate, rapid recruitment ability, and high seed viability (Zedler et al. 1995; Laegdsgaard 2002; Winning & MacFarlane 2010). This is supported by over 95% of the seedling counts observed in Area E being succulent species, most probably *S. quinqueflora* and *S. australis*.

Regarding *Sarcocornia* spp. dominance, Adam (1990) and Zedler et al. (1995) hypothesized that succession may take place in these cases, with *S. virginicus* becoming dominant later. *Sporobolus virginicus* is known to be a slow recruiter in many instances, spreading primarily through stolons and with a low or nonexistent seed viability (Zedler et al. 1995; Winning & MacFarlane 2010). Implications of the slow recovery of *S. virginicus* to saltmarsh functioning and future restoration projects have not yet been explored. Interestingly, *Triglochin striata* was identified as a codominant species in previous studies in the same area and some of the same sites studied here, namely Cobbans Creek and Area E (Adam 1990; Zedler et al. 1995). However, its presence was scarce in this study at less than 1% at any site, zone or sampling time. The scarcity of *T. striata* in this study may be due to livestock grazing, as Zedler et al. (1995) has reported reduced abundance of this species in saltmarsh disturbed by cattle in the Hunter River estuary, or it may be due to natural fluctuation over time. Since patches of *T. striata* were found to be growing in the low marsh at Area E by the study's end, it will be of interest to monitor whether this previously co-dominant species reestablishes significant cover.

Saltmarsh growth in the Area E low marsh, initially defined as mean low marsh elevation in comparison sites minus the lowered tidal regime implemented in Area E (around 0.30–0.36 m AHD), showed significantly less saltmarsh development than the higher elevations. This could be due to the softer, mudflat-like nature of the substrate observed, but not quantified, in the Area E low marsh. While the low marsh substrate remained mudflat-like throughout the study, periodic saltmarsh recruitment in Area E was observed through seedling counts in 80% of the quadrats. However, only 15% of the low marsh quadrats retained saltmarsh individuals to the study's end. While seedlings are known to have high mortality rates, especially in high-stress environments (Chang et al. 2001; Ungar 2017), exploration of the specific causes of saltmarsh seedling mortality in these and similar conditions could be useful to further understand and promote saltmarsh development. Anecdotally, waterlogging/flooding and burial by fine sediments are proposed as likely causes of mortality based on observations of deceased plants and seedlings in other wetlands (Huiskes et al. 1995). If sediment properties were a significant factor, this would support the suggestion by Ferronato et al. (2018) that soil (or benthos) type and structure, as well as tidal flushing, are important considerations in saltmarsh development. However, further research is needed to explore other factors contributing to the saltmarsh restoration.

Saltmarsh composition is an important contributor to ecosystem function (Laegdsgaard 2006; Friess et al. 2012; Adam 2019). As Lawrence et al. (2018) and Matthews et al. (2009) have highlighted, equivalent plant cover does not mean a saltmarsh has reached equivalent functioning with comparison sites (Mossman et al. 2012). The composition of saltmarsh plants at Area E was different to that of most comparison sites, having a lower representation of *S. virginicus*. As an important food source for many local estuarine species (Taylor et al. 2017; Raoult et al. 2018), this developmental trend could impact its ecological functioning (Li et al. 2014; Lawrence et al. 2018) and further research investigating the response of additional indicators is needed. Given that the vegetation cover was increasing consistently, and the tendency of *S. virginicus* to establish itself over *S. quinqueflora* (Zedler et al. 1995), the developing saltmarsh could prove valuable to the wider estuarine community in the long term.

Findings from this study suggest that the use of fixed 1-m^2 quadrats for vegetation indicators is adequate for measuring and tracking broad-scale changes in developing saltmarsh cover and composition relative to comparison sites. The sampling strategy was able to detect significant differences for most measured indicators within the 27-month sampling time frame, proving useful for community and industry restoration projects where the time and costs allocated for showing progress to goals can have significant limitations (Broome et al. 1988; Kentula 2000; Knight 2018).

Detection of statistically significant differences were found in total saltmarsh cover, the number of species and saltmarsh assemblages. In this study, weed cover showed no variation between sampling times. It should be noted that the number of species, while closest to comparison site equivalency by the study's end, similar to Craft (2016), could only tentatively predict future compositions due to the relative low coverage. The sampling methodology used was also able to detect, with good precision, the variance in saltmarsh composition between Area E and the comparison sites. Given the different functions and roles various saltmarsh taxa provide (Zedler et al. 1995; Winning & MacFarlane 2010), this type of information could prove very useful for management, as well as reporting of saltmarsh indicators against targets.

Saltmarsh total cover showed significant differences within 18 months of beginning restoration, which is consistent with other Australian and overseas studies tracking saltmarsh development (Laegdsgaard 2006; Craft 2016). This trend continued consistently to the end of the study, approaching, but not reaching, equality with the range of values represented by the comparison sites. Thus, based on extrapolation models, saltmarsh could reach an average cover equivalent to that of the comparison sites within 5–6 years, which is well within the 10-year time frame given by other evaluations (Matthews et al. 2009; Craft 2016; Jørgensen et al. 2016). However, future research is needed to confirm continuation of trends.

Contrary to expectations, it was the high marsh that showed the fastest development of plant cover with the low marsh taking the longest to show significant change. This differs to many studies where tidal flows are reintroduced to promote marsh restoration where recruitment is normally fastest in the low marsh (Howard et al. 2015; Craft 2016; Noto & Shurin 2017). The differences seen here are likely unrelated to the hydroperiod imposed by the SmartGates given that the altered hydrology was informed by surveys of multiple local saltmarshes and that hydroperiods are similar to those identified in other recent local saltmarsh studies (Howe et al. 2010; Rodríguez et al. 2017; Sandi et al. 2018). Thus, the reduced recruitment is likely due to infauna, waterlogging, sediment, or chemical properties, all of which have been noted to affect saltmarsh recruitment and development (Paramor & Hughes 2007; Craft 2016; Ferronato et al. 2018). It should be reemphasized that the saltmarsh zones in Area E were around 0.45 m AHD lower than in adjacent saltmarsh sites and that the low marsh especially would be developing in a different bio-geophysical environment. With the limited mangrove seedlings anecdotally observed growing in Area E being restricted to the low marsh and mudflat zones, this supports the hypothesis by Howe et al. (2010) that alteration of hydrology can limit issues of mangrove encroachment while allowing some mangrove habitat to establish at lower zones if desired.

Although sampling in this study occurred only over a short time period (i.e. 27 months), the data is sufficient to inform adaptive management of the site's hydrology. Additionally, sampling occurred during wet and dry seasons which may have affected saltmarsh development. Sampling over a longer period would better inform how patterns and rates of saltmarsh development change over dryer and wetter periods. As such, conclusions should be considered preliminary as sampling over 10 years is often needed to confirm developmental trends (Matthews et al. 2009; Craft 2016; Jørgensen et al. 2016). Future studies could also look at longer term impacts of this restoration method.

This study assessed simple, well-known vegetation indicators to evaluate saltmarsh vegetation response. This increases the studies usefulness to local and community practitioners who may lack the resources and expertise available to researchers. Many other indicators can be, and have been, used to good effect to track saltmarsh vegetation development including leaf and stem density, canopy height, inflorescences, and belowground root biomass. While the usefulness of these indicators is often species-dependent, and their measurement time-consuming, destructive, or requiring specialized equipment or expertise, further research testing the effectiveness of these and similar measures to evaluate the tidal control restoration method may be informative.

The saltmarsh types assessed in this study were limited to those found in the Hunter River estuary of New South Wales, Australia. Given the diversity of saltmarsh forms and functioning both locally and internationally (Adam 1990; Saintilan 2009), replicate large-scale observational experiments on different saltmarsh types and locations, including more studies like the one reported here, would be useful to inform wider restoration practice elsewhere in response to a changing climate. As the necessary ecological indicators have been lacking in some common community-level saltmarsh restoration evaluations (Craft 2016; Knight 2018; Adam 2019), saltmarsh indicators like those described here could be useful for community-level saltmarsh reporting beyond the basic presence and absence of saltmarsh plants often reported (Knight 2018). In conclusion, this study has found that saltmarsh cover and assemblages in Area E developed rapidly within 3 years of the novel restoration method being implemented and, with the rising number of species and high seedling recruitment, is on track to having saltmarsh communities as extensive as those at other local sites.

Acknowledgments

This research project was a joint endeavor between various institutions and people whom the authors wish to thank. Newcastle Coal and Infrastructure Group (NCIG) provided funding and management of Area E. Special thanks to P. Reid and H. Ardagh (NCIG) for overseeing the maintenance and operation of the SmartGates and liaison between parties. C. Drummond flew drones for University of New South Wales Sydney and provided valuable aerial imaging including detailed digital elevation models necessary for hydrological mapping at Area E and other sites. D. Warman and many students (A. Suzzi, B. Yare, J. Desmond, J. Chanachai, J. Beckius, J. Gibb, L. McRae, L. Murphy, L. Mickaill, M. Grew, S. Sanders, S. Ryan, S. Montgomery, and T. Fletcher) at the University of Newcastle and Avondale University provided fieldwork assistance. Thanks to New South Wales National Parks for permission to conduct research at comparison sites. This study was supported by a UON Research Training Program Scholarship to C.R. and conducted under National Parks Scientific License SL102054. Open access publishing facilitated by The University of Newcastle, as part of the Wiley - The University of Newcastle agreement via the Council of Australian University Librarians.

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Supporting Information

The following information may be found in the online version of this article:

Table S1. Total saltmarsh cover and number of saltmarsh species two-factor PERMANOVA table of results.
Table S2. Total saltmarsh cover and number of saltmarsh species three-factor PERMANOVA table of results.
Table S3. Saltmarsh seedling counts and saltmarsh assemblage three-factor PERMANOVA table of results.
Table S4. Saltmarsh seedling counts and saltmarsh assemblage two-factor PERMANOVA table of results.

Guest Coordinating Editor: Emma Asbridge

Received: 13 March, 2022; First decision: 19 April, 2022; Revised: 9 August, 2022; Accepted: 9 August, 2022