Estuarine, Coastal and Shelf Science 164 (2015) 46–55

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Tidal marsh erosion and accretion trends following invasive species removal, Tamar Estuary, Tasmania

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ARTICLE INFO

Article history:
Received 1 October 2014
Received in revised form
11 June 2015
Accepted 12 June 2015
Available online 23 June 2015

Keywords:
Spartina infestation
Eradication
Salt marsh
Erosion
Accretion

ABSTRACT

The introduction of *Spartina* to intertidal marshes last century in many areas of the world transformed estuarine geomorphology, threatened native species and habitats, and impeded coastal access and use. This study investigated erosion/accretion trends of marsh surfaces following removal of invasive *Spartina* across a substantial intertidal marsh area. Marsh surface changes were monitored within a 0.6 ha experimental site where *Spartina anglica* cover was removed, and compared with surface changes at a comparable control site. Erosion/accretion rates were measured for over two years using a grid transect network, creek cross sectional profiles, and seaward edge delineation. Results showed that a significant erosion of the marsh surface occurred at the experimental site relative to the control site, using two different statistical analyses. Analysis of mean monthly change found erosion rates at the experimental site to be 13.2 mm a^{-1} relative to 2.0 mm a^{-1} at the control site, a rate that was six times greater. Analysis of overall change from the beginning to the end of the study showed that erosion was significantly more pronounced at the experimental site relative to the control site, and increased from the landward edge to the seaward edge at both sites. This study demonstrates the need for consideration of geomorphic processes when managing invasive plants in dynamic environments, and indicates that large scale *Spartina* removal will cause coastal erosion, bringing potential consequences to adjacent near shore waters and ecosystems.

1. Introduction

Invasive species are recognised as a key threat to biodiversity and ecosystem functioning worldwide (Pejchar and Mooney, 2009), and in environments of high sediment mobility such as dune systems and estuarine intertidal zones, they also influence morphology, through change to processes of accretion and erosion. *Spartina* species are such ecosystem engineers (Jones et al., 1994, 1997; Gutiérrez et al., 2000; Strong and Ayres, 2013), in that they influence the environment in which they live by creating or modifying habitat. *Spartina* species are monocotyledonous plants with stiff, robust stems and fleshy interwoven leaves which slow the velocity of tidal waters, and trap suspended sediment at a higher rate than other marsh species (Thompson, 1991; Li and Yang, 2009). Reduced flow velocities in dense *Spartina* promotes sediment deposition (Bouma et al., 2005a), leading to high sedimentation rates (Ranwell, 1964). Subsurface components of *Spartina anglica* consist of extensive systems of robust rhizomatous roots, which protect inter-tidal mud from erosion (Van Eerdt, 1985; Brown, 1998; Brown et al., 1999), and contribute organic matter to sediment accumulation (Li and Gao, 2013).

Spartina was intentionally introduced in Europe, the USA, China, Australia and New Zealand last century (Strong and Ayres, 2009), for the benefits of marsh sediment accretion leading to land extension and coastal protection (Chung, 2006; Strong and Ayres, 2009; Wan et al., 2009; Zuo et al., 2012). Negative consequences subsequently became recognised, such as replacement of native vegetation, and destruction of important migratory shorebird and waterfowl habitat (Hedge and Kriwoken, 2000; Strong and Ayres, 2013; Boon et al., 2014). In China, three introduced *Spartina* species are estimated to have caused annual economic losses of US 2000 million dollars, and are now declared as notorious invasive species (Sun et al., 2015). In Australia, *S. anglica* became considered to be an invasive species (Laegdsgaard, 2006; Boon et al., 2014), threatening the ecological integrity of estuarine wetlands of international importance (Wells, 1995; Doody, 2008). Furthermore,

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http://dx.doi.org/10.1016/j.ecss.2015.06.013

sediment buildup and dense vegetation impacted coastal access, brought threats to coastal infrastructure and aquaculture (Kriwoken and Hedge, 2000; Doody, 2008), and resulted in estuarine morphological transformation (Sheehan and Ellison, 2014).

As perceptions changed, control programs commenced of infestations of *S. anglica* and other invasive *Spartina* species (Roberts and Pullin, 2008; Doody, 2008; Sheng et al., 2014). Chemical control using herbicides has been most widespread (Doody, 2008; Sheng et al., 2014), however mechanical removal to control smaller infestations has also been used in areas such as San Francisco Bay and Puget Sound, USA, the north east coast of England, and Port Gawler, South Australia (Hedge et al., 2003; Doody, 2008; Strong and Ayres, 2013). Some infestations in Washington State, San Francisco Bay and parts of Europe, New Zealand and Australia approached eradication, while others proved too large (Guenegou et al., 1991; Kriwoken and Hedge, 2000; Reeder and Hacker, 2004; Taylor and Hastings, 2004; Grevstad, 2005; Bortolus, 2006; Sheng et al., 2014).

While investigations have been conducted to determine the direct effects of control methods on sediments and biota (Frid et al., 1999; Hammond and Cooper, 2002; Roberts and Pullin, 2008; Buckley and Han, 2014; Lampert et al., 2014), little research has been carried out to investigate any undesirable long-term geomorphological impacts of large scale *Spartina* removal. There may be potential for sediment erosion following marsh defoliation (Kirby, 1994), which could have negative consequences in increased estuarine turbidity and changes to sediment budgets.

This study undertook an investigation into the consequences of large scale *Spartina* removal on the erosion or accretion trends of marsh surfaces across the intertidal zone, in an estuary where its introduction half a century ago led to profound estuarine intertidal morphology change (Sheehan and Ellison, 2014). The aim of this study was to remove a substantial area of invasive *Spartina* across a land to sea sector of marsh, and investigate the subsequent consequences to the marsh surface morphology, as a result of erosion/accretion trends.

2. Methods

2.1. Study area

The Tamar Estuary is the largest estuary in North Tasmania (100 km^2), with the majority of freshwater inflow at the estuary head from the North and South Esk Rivers, and extends 70 km along a bedrock confined drowned river valley (Pirzl and Coughanowr, 1997; Ellison and Sheehan, 2014). Mean daily temperatures are between 5 and 25 °C, annual rainfall about 675 mm, and wind directions dominantly north–westerly through to northerly, with some south-easterly influences. The tidal range in the central part of the estuary is 2.6 m (Ellison and Sheehan, 2014).

Spartina anglica was introduced to the Tamar Estuary last century on behalf of the Launceston Port Authority at Windermere (Fig. 1) in an attempt to reduce estuarine siltation issues (Ranwell, 1967; Phillips, 1975; Pringle, 1993). The upper estuary was prone to siltation at a time when the estuary served as a major shipping channel, and it was believed that vegetating the mudflats would promote vertical accretion, to better define the channel and enhance scour, therefore reducing the reliance on expensive dredging operations.

Subsequently, *S. anglica* spread throughout the estuary, particularly seaward of the initial planting sites to areas not subject to siltation. The total area was estimated to be 374 ha in 2006 (Sheehan and Ellison, 2014), and is Australia's largest infestation. Its introduction and continued spread has transformed the intertidal zone from gently grading mudflats, sandy beaches and gravels into laterally extensive *S. anglica* monocultures composed of fine grained sediments (Phillips, 1975; Pringle, 1993; Bird, 2008), that have trapped extensive volumes of sediment to substantially change the estuarine morphology (Sheehan and Ellison, 2014). Since the 1990's government and community programs have worked to control spread and remove smaller *Spartina* infestations around Tasmania (Kriwoken and Hedge, 2000).

2.2. Study design

A study location was selected on the central western coast of the estuary south of Swan Point (Fig. 1), and a control and experimental site each of 0.6 ha was established at the location separated from each other by a distance of 200 m, yet remaining in the same embayment with similar aspect. The location is in a relatively remote area of the estuary, largely out of view and away from roads and dense residential areas, so minimising risks of disturbance during the experiment. The marsh at this location (Latitude 41° 15′54″ S; Longitude 146° 58′9″ E) extends seaward some 120 m from Mean High Water (MHW), with sediment depths of about 50 cm accumulated under *Spartina*, upon a substrate of sands and gravels (Sheehan and Ellison, 2014).

The study location is in a wide section of the Tamar Estuary, flanked by extensive and gently grading sandflats which are exposed during low tide, and extend ca. 1 km seaward to the main channel. The Tamar is a tidally dominated estuary and the gentle grade and width of the sandflats south of Swan Point result in the attenuation of wave energy before reaching the *Spartina* marsh. The seaward margin of the marsh at this part of the estuary was poorly defined, with clumps of *S. anglica* extending seaward onto the sandflat. Typically there was little change in elevation between the marsh and the sand flat, and an absence of erosional scarps or micro-cliffs.

This study compared erosion/accretion rates between an experimental site where *Spartina anglica* cover had been removed, and a control site, where *S. anglica* cover remained intact. Erosion rods were used, which if hammered into the ground below the marsh show accretion/erosion trends of the marsh surface over time (Nolte et al., 2013). Changes are interpreted as either erosion (increase in rod length exposed above the marsh surface) or accretion (decrease in rod length exposed above the marsh surface). To compare erosion/accretion between the two treatments, Acetyl rods were used, with a diameter of 6 mm and lengths ranging between 0.5 and 1 m depending on the thickness of marsh sediments. These were pre-cut in the laboratory, and the top of the rod melted on a hot plate so that the measuring surface was flat. The rods were inserted into the marsh perpendicular to the surface and firmly embedded in the lower sandy substrate by about 50 mm, and leaving 50 ± 2 mm protruding from the marsh surface.

Three erosion/accretion monitoring studies were established within each of the sites (Fig. 2): a transect study to investigate marsh surface erosion/accretion trends, an edge retreat study to focus on change at the seaward edge, and a creek study to investigate changes in tidal creek cross sectional profiles.

2.2.1. Transect study

Given the large scale of the study sites, the number of points required for statistical validity, and the need to complete each remeasurement in one low tide, a grid transect design using erosion rods was used (Kirby et al., 1993; Kirby, 1994; Gilman et al., 2007; Nolte et al., 2013). With the *S. anglica* sediment accumulation at this area of the estuary being <1 m (Sheehan and Ellison, 2014), rods could be inserted firmly into the sand/gravel pre-*Spartina* surface beneath the fine grained marsh muds (Fig. 2).

Six transects perpendicular to the shoreline were established in both the experimental and control sites between the landward

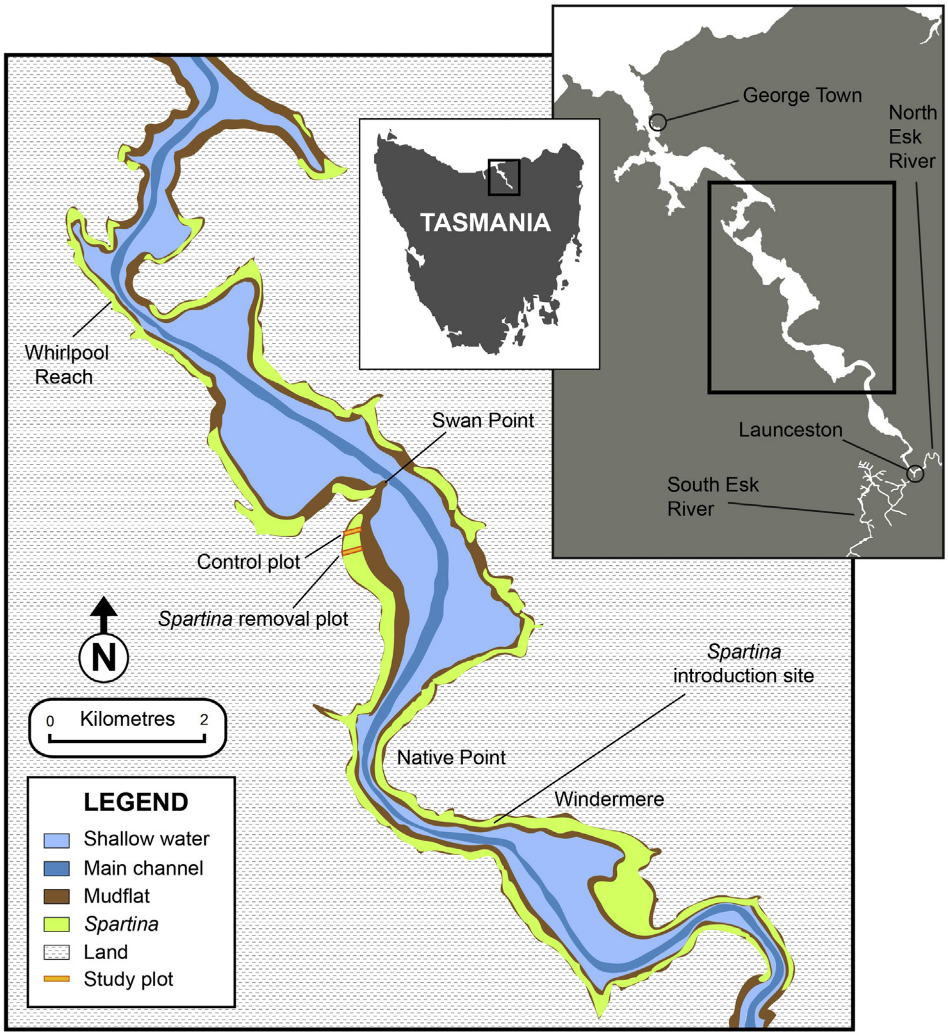

Fig. 1. Map of the Tamar Estuary, Tasmania, showing location of the study area.

bank and the sand flats seaward of the *S. anglica* marsh (Fig. 2). Acetyl rods were placed every 10 m along each transect to a distance of 100 m from MHW (Fig. 2). Thereafter rods were placed every 5 m to within 3 m of the seaward marsh edge, and then at 1 m intervals to the seaward edge. In areas such as the upper and lower banks where micro-cliffing was anticipated, rods were placed at each minor break in slope. For the outer marsh edge, this involved placing a rod at the edge on the marsh surface, and another on the sandflat surface. Both the experimental and control sites had isolated *Spartina* clumps extending onto the sand bank seaward of the laterally extensive sward. Where transects intercepted these clumps, rods were placed at both the upper clump and lower sandflat surfaces. Rods were classified as either landward bank rods, marsh rods, sandflat rods or clump rods (Fig. 2) to allow determination of erosion/accretion rates for each of these landforms.

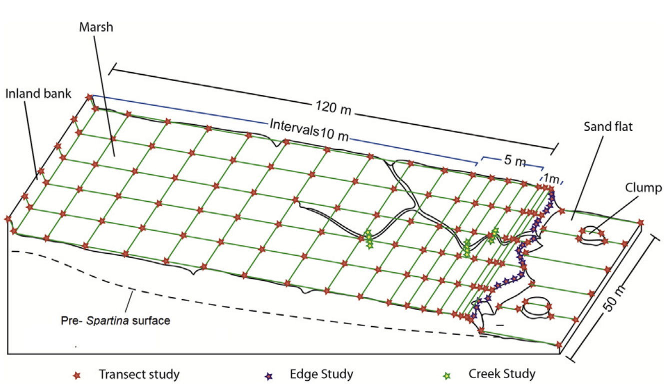

Fig. 2. Diagrammatic representation of the experimental and control study areas' sample design, with grid transect based, seaward edge and creek cross section studies.

2.2.2. Edge study

The seaward margin of the marsh of both sites had isolated *Spartina* clumps offshore of the main marsh (Fig. 2). The edge was therefore defined as the seaward-most extent of the continuous sward, and rods were placed every 2 m along this margin to determine marsh retreat or progradation. Rods were also placed around the margins of offshore clumps as part of the transect study.

2.2.3. Creek study

Three tidal creek cross sections were monitored in major creeks of the lower, mid and the upper marsh at both the experimental and control sites. At each cross section, nine erosion rods were installed across the creek, placed at each major break in slope, and across the creek channel base. Marsh surface rods were placed 1 m either side of the channel edge, top edge rods were placed 0 cm and

10 cm from the creek top edge, and channel rods were placed at the lower edges and central thalweg.

For replication, mud surface elevation across each creek transect was recorded using a surveyors level at the beginning and end of the study to determine change in cross sectional shape over this period. An aluminium plate of 1 mm thickness was placed at each point when surveyed, to prevent the survey pole base from sinking into the mud.

2.3. Removal of *Spartina*

The experimental site was treated with the monocot herbicide Fusilade to cause *Spartina* mortality. Permits were obtained from the State Government, and their staff assisted in the treatment following experience gained from successful control programs of small infestations in other estuaries (Hedge and Kriwoken, 2000; Department of Primary Industries, Water and Environment, 2002). Three follow-up treatments were required within 6 months of the initial treatment for total removal. Maintenance sprays were carried out each subsequent summer to control seed set and vegetative spread from the surrounding marsh.

2.4. Measurement

All rods from both the experimental and control sites (Fig. 2) were measured monthly during the same low tide period for over two years. A small, thin stainless steel ruler was placed on the mud surface at the base of the rod in order to give an average surface in the event that rods had caused localised scouring or sedimentation. The protruding rods were consistently measured on the south side, from the top of the rod down to the ruler, using a stainless steel carpenter's square to ensure horizontality. Measurements of individual rods were rejected if observations suggested the rod had been influenced by local processes such as coning or mounding (Nolte et al., 2013), or if rods had been bent or lost.

2.5. Statistical analysis

2.5.1. Transect study

Changes over time to marsh levels and sandflat surfaces seaward of the marsh were analysed from all monthly rod measurements along transects from both the control and experimental sites. The first analysis was of yearly erosion rates calculated for each rod using the difference in the length of the exposed rod between two consecutive measurements (i.e. T1 and T2), and applying the following equation:

$$Erosion\ rate\ \left(mm\ a^{-1}\right) = \frac{(Rod\ length\ T2 - rod\ length\ T1)}{\left[\frac{date\ T2 - date\ T1}{365}\right]}$$

Scatter plots of mean monthly erosion/accretion rates against time were plotted for each treatment, and linear trend lines fitted. As this study was principally concerned with the overall difference in erosion rates between treatments with time, trend lines were considered to give the best estimate of general linear trends. Separate regression lines were fitted to the estimated average rate of erosion per month for each rod.

An Analysis of Variance (ANOVA) was then used to compare trend line slopes between control and experimental treatments. The ANOVA also included the covariate distance from shore (offset), which adjusted for the observed variability known to be caused by the offset, removing residual error. In this analysis, each rod was treated as an independent measurement due to there being no treatment replication, and given the lack of replication, *P*-values and 95% CIs must be interpreted with caution.

A second analysis investigated the net change in elevation using rod data from the first measurement (November Year 1) and the last measurement (February Year 3) to compare overall change to bed level both within and between sites. Net change analysis also allowed a visual assessment of relative bed level change and the determination of the degree of homogeneity in accretion/erosion rates across sites and between treatments. Net change data were used to produce contour maps and 3D relief models. While these models do not represent actual marsh topography, they provide a representation of the magnitude and direction of change. An independent sample t-test was performed to compare net change in relative elevation between the control and experimental site. Assumptions for use of a t-test were checked and met.

A General Linear Model (GLM) was used to assess the effect of the site, transect and distance from shore (offset) on relative elevation change, using the experimental unit of the rod. The model simultaneously assessed the combined effects of site, position in the grid (Fig. 2) and distance from shore so that the spatial variability in relative elevation change could be quantified. A GLM of this type has the underlying assumptions of:

1. All observations are independent;
2. The random variation (or residual) after fitting the model is approximately normally distributed, and
3. The residuals display constant variance across the range of values of the explanatory variables in the model (i.e. site, transect and offset).

A scatter plot showing change in relative elevation versus distance from shore concluded that variance of the residuals increased as the distance from shore increased, therefore violating assumption 3 of the GLM. This was addressed by performing a natural log transformation, which can be used when there is evidence that the residual variability increases with increasing (fitted) values of the outcome (Welham et al., 2014). Such a log transformation more appropriately models a multiplicative relationship such as a percent increase rather than an additive one such as unit increase (Welham et al., 2014). This is a mathematical feature of logarithms, and by taking a log transformation, the residual assumptions were satisfied, whereas without they were not. In doing this, however, it was necessary to base the analysis on the original rod lengths rather than on relative elevation, as the latter contained a number of negative values for which a natural logarithm does not exist. Since the relative elevation values were derived directly from the subsequent change in rod lengths relative to the original length, it is the equivalent analysis.

The rod lengths at T2 (which were all positive) were transformed using the natural logarithm and modelled as a function of the following explanatory variables:

- the natural logarithm of the rod lengths at T1 (continuous variable)
- Site (a factor with 2 levels: control and experimental)
- Distance from shore in metres (continuous variable)
- transect (a factor with 6 levels: T1, T2, T3, T4, T5, T6)

The factor ‘Site’ tested for overall differences in rod length between the two treatments, which could presumably be attributed to the removal of vegetative cover. The variable ‘Distance from shore’ tested the relationship between rod length at time 2 and distance from the shore at time 1. The factor ‘Transect’ compared individual transects to determine if rod length varied laterally within and across study sites. The factor Transect had six levels; therefore pair-wise comparisons were conducted to identify where the statistically significant differences occurred.

2.5.2. Edge study

Temporal change of the outer marsh margin was investigated as for the transect study, by fitting regression lines to estimate the average rate of erosion per month at each rod. An independent sample t-test was used to compare average monthly erosion rates between control and experimental treatments. Assumptions for use of a t-test were checked and met.

2.5.3. Creek study

The data set for the creek study, comprising of 9 pins across 3 transects at each site, was not considered sufficient to apply statistical analysis. Changes at each site were compared using a geomorphic assessment of creek cross sectional shape, supplemented by the scatter plots and trend lines of mean monthly erosion/accretion rates for the four landform types identified, of marsh surface, top edge, channel edge and channel centre.

3. Results

This study provided a large data set of results from across the entire marsh surface from the landward edge to sand flats offshore, with each study site encompassing an area of 6000 m^2 of marsh. Monthly measurement of erosion rods at the control and experimental sites for over two years provided a total of 9306 measurements from 394 rods, yielding a total of 8898 rate determinations. This extensive data set allowed the determination of depositional and erosional trends for over two years following *Spartina* removal.

3.1. Marsh surface erosion/accretion trends

Time series of mean monthly erosion/accretion rates from the *Spartina* marsh surface of the experimental Fig. 3 and control sites are shown in Fig. 4A, including trend lines. Monthly measurement results showed erosion rates across the entire marsh surface of the experimental site to be 13.3 mm a^{-1}, while the control site showed a mean monthly erosion rate of 2.0 mm a^{-1}. Trend line analysis (Fig. 4B) showed that the values at both treatments were normally

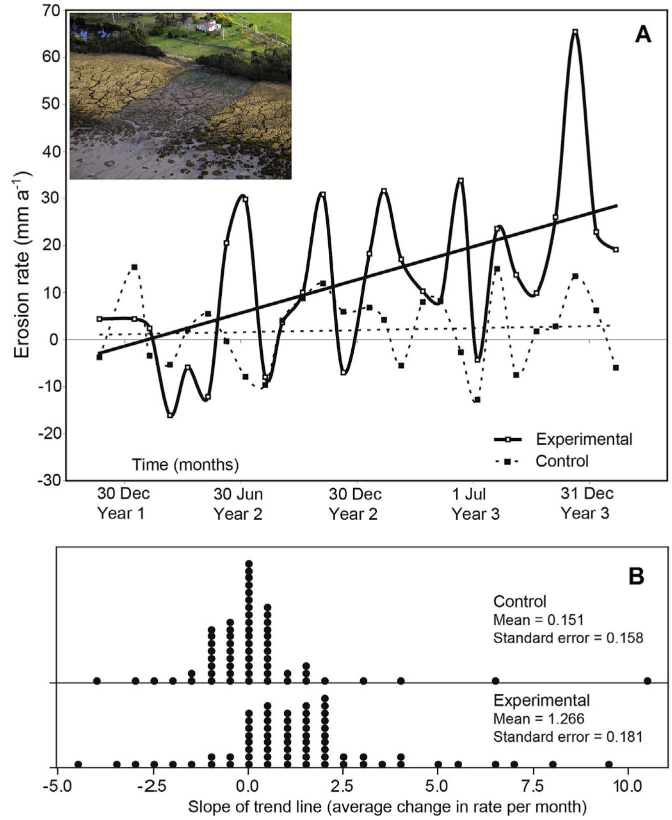

Fig. 4. **A**. Comparison of mean monthly erosion rates of the marsh surfaces of the experimental and control sites, showing trend lines and smooth connecting lines. Inset: Experimental site in June of year 2. **B**. Dot plots of the slope of trend line (average change in rate per month) for the marsh surface, with each point representing up to 2 observations.

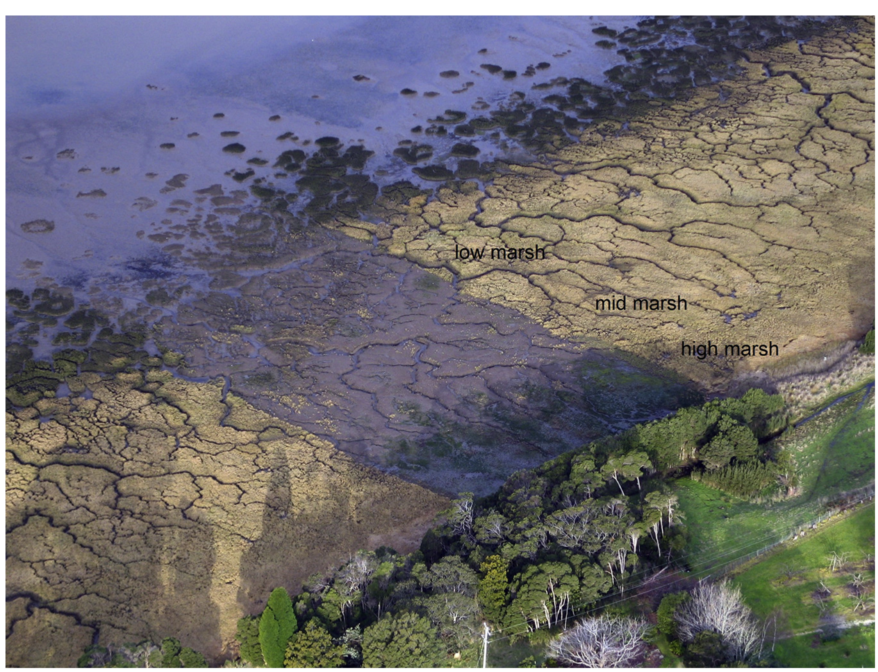

Fig. 3. Experimental site following *Spartina* removal as a result of treatment with herbicide (Photo: M. Sheehan).

distributed around the mean with a number of extreme values. The control site was centred over a slope of 0.0, while the majority of observations at the experimental site fell between a slope of 0.0 and 2.4 indicating a more erosional regime. Treatments were compared using an analysis of variance, which compared the two sites while also considering the effect of distance from shore. This found a difference in slope between the two treatments of 1.120 (standard error = 0.242), demonstrating that erosion rates at the experimental site were statistically significantly higher [$P = F_{1,225}(21.37) < 0.001$] than those at the control, and showing that marsh surface erosion occurred at a statistically significantly greater rate at the experimental site relative to the control.

Cumulative erosion or net change results over the duration of the experiment were used to produce contour maps of marsh surface morphology, providing a spatial representation of the magnitude and direction of overall change (Fig. 5A). Erosion of 40–50 mm occurred over the intertidal marsh surface at the experimental site, giving an erosion rate of 30–37 mm a^{-1}. Erosion was more pronounced in the outer 40 m of the marsh, with erosion of up to 100 mm at the seaward edge. In contrast, this analysis showed the control site to experience overall marsh surface accretion across the landward and central marsh of 10–20 mm (7–15 mm a^{-1}), with net erosion of 10–20 mm also occurring along the seaward 20 m of the control site.

Dot plots of overall net change in marsh surface morphology (Fig. 5B) support the contour map assessment (Fig. 5A), despite both treatments being slightly positively skewed by some outliers. The experimental site showed a higher net change value on average than the control (Fig. 5B), indicating a more pronounced erosional regime at the experimental site. An independent samples t-test confirmed this, showing that on average the experimental site was 0.03 lower than the control (P-value < 0.001). A scatter plot of vertical change against distance (Fig. 6) showed that both sites become increasingly erosional towards the seaward edge.

This observation was investigated further by fitting a GLM which tested the statistical significance of the difference in net erosion between sites as well as the within-site spatial variable of distance from shore simultaneously. The GLM model showed that the effect of distance from shore on the erosion was significant (*P*-value = <0.001) for both treatments on average, and estimated that with every 10 m increase in distance from the shore, rod length at T2 increases by a factor of 1.06 or 6 percent, where increasing rod

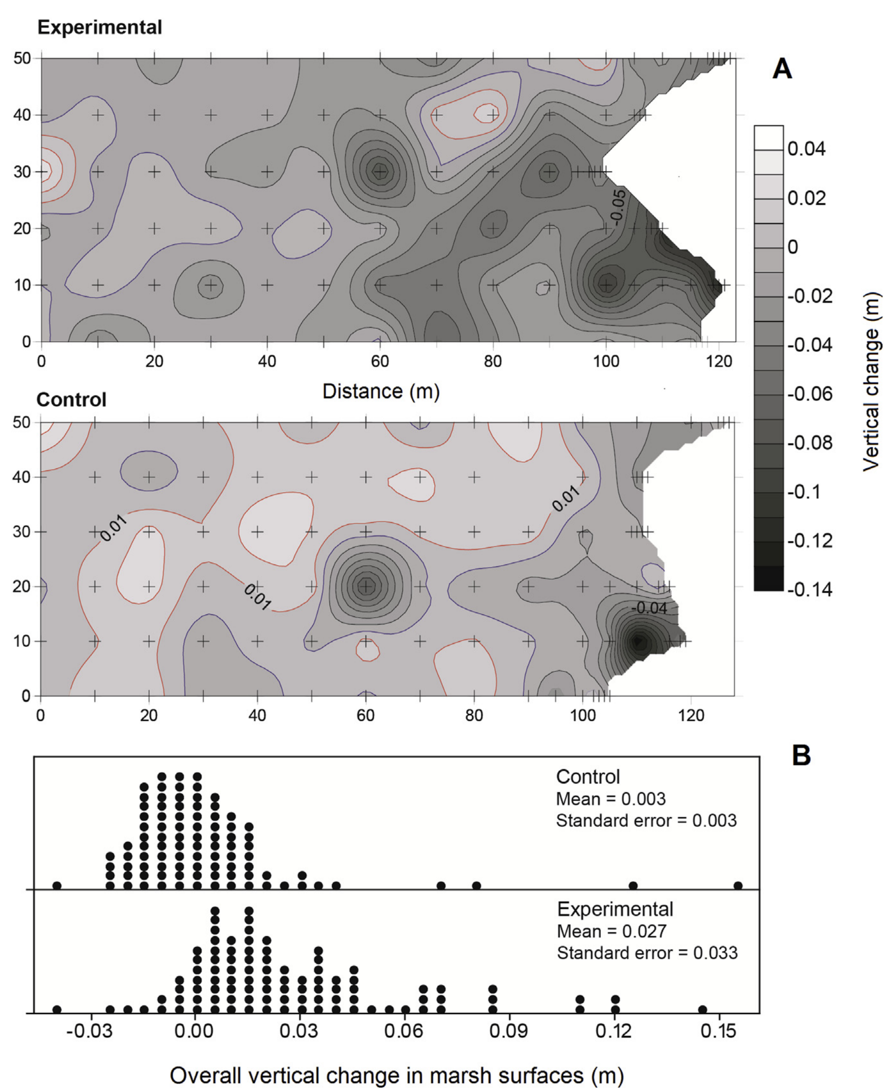

Fig. 5. **A**. Contour maps showing the overall magnitude and direction of marsh surface change over the duration of the removal experiment. **B**. Dot plot of net change in elevation of the control and experimental sites over the duration of the removal experiment. Negative values indicate elevation gain while positive values indicate a lowering of the marsh surface.

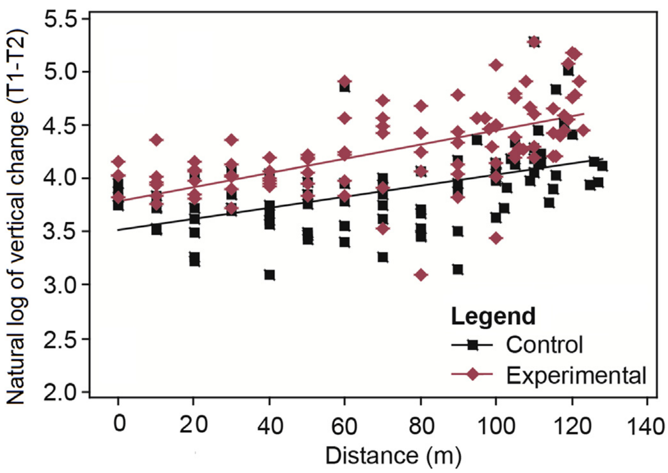

Fig. 6. Scatter plot of vertical change (T1–T2) with distance from shore, using the natural log of rod length data to remove the effect of uneven variance.

length indicates marsh surface lost, assuming T1, site and transect all remained fixed. The site by offset interaction was added to the model and concluded that there was no statistically significant difference (P-value = 0.3) in the effect of distance from shore between sites, further evident by the parallelism of the trend lines fitted to the scatter plot data (Fig. 6).

3.2. Edge study

A summary of erosion rod results for the seaward edge study is given in Table 1, showing a mean erosion rate of 22.7 mm a^{-1} for the experimental site and 13.5 mm a^{-1} for the control site. Trend lines (Fig. 7A) indicate increasing erosion in the experimental site relative to the control site, with a difference in trend slope of 1.140 (Fig. 7B), which was found not to be significantly different ($P = 0.3844$).

3.3. Creek study

Erosion rod results from the creek study (see Supplementary data) found that the only landform of the creek system where erosion rates differed between treatments was at the top of banks. Typical topographic cross sections from each treatment are shown in Fig. 8, showing that the upper bank of the creeks in the experimental site eroded to open the creek cross-sectional profile and reduce the gradient of creek channel banks (Fig. 8). While this has occurred at both sites it was more pronounced at the experimental site, with the bankfull cross-sectional area increased by 14.6% at the experimental site relative to 8.2% at the control site (Fig. 8). Creek depths and lower channel widths remained relatively constant over time both within and between treatments, with no evident erosional trend.

Table 1
Summary of erosion rod data for the seaward edge study. All measurements are given as mm a^{-1}, with positive numbers indicating erosion and negative numbers indicating accretion.

	Experimental	Control
Number giving data	525	425
Number with no change	36	42
Mean rate	22.7	13.5
Minimum rate	−937	−300
Median rate	45.5	183.5
Maximum rate	1028	667
Standard deviation	127.9	66.2

4. Discussion

Results showed that over 2 years after *S. anglica* removal, a significant erosion of the marsh surface occurred at the experimental site relative to the control site (Figs. 4–6), using two different statistical analyses. The mean monthly erosion rate of 13.3 mm a^{-1} at the experimental site relative to 2.0 mm a^{-1} at the control site (Fig. 4), is a rate of six times greater at the experimental site relative to the control site. This is supported by the observation of geomorphic changes which included micro-cliffing at a scale of up to 50 mm of the seaward margin of the marsh and creek banks, and the sapping of fines from within the root zone in the seaward marsh. Following *Spartina* dieback in Georgia (USA), Crawford and Stone (2015) found higher bulk density, lower field capacity and coarser soil textures in dieback patches relative to healthy vegetation patches, similar to our observations of the loss of fines and erosion. The erosion results give a measure of net marsh surface change as shown by rods firmly held into a harder substrate (Gilman et al., 2007; Nolte et al., 2013), and as the rods are shallow with the hard surface about 50 cm under the marsh sediments, they are not measuring compaction or subsidence, they indicate erosion.

Erosion occurred over the entire experimental site, and the magnitude of erosion increased towards the seaward edge (Fig. 5), and these results can be compared with natural marsh dieback elsewhere. Sudden marsh dieback has recently occurred in parts of the U.S.A., owing to causes including fungal pathogens, soil properties and relative sea-level rise (Alber et al., 2008; Temmerman et al., 2012; Elmer et al., 2013; Ganju et al., 2013; Brisson et al., 2014; Crawford and Stone, 2015). *Spartina* dieoff areas in Maine showed higher water velocities, higher erosion and less shoreline stabilisation relative to healthy marsh vegetation areas (Brisson et al., 2014). Increased tidal flow across the marsh platform occurred following loss of marsh vegetation (Temmerman et al., 2012). After *Spartina* died at the experimental site in our study and the marsh surface became bare mudflat, it is likely that higher water velocities (Temmerman et al., 2012; Brisson et al., 2014) contributed to the increased erosion rates observed.

A significant difference was found between higher erosion rates at the experimental site and lower rates at the control site, from analysis of both mean monthly change and overall net change. Erosion occurs as a result of a changing balance between external and internal factors, and locating the study sites on the same type of marsh, alongside, and with the same aspect and fetch attempted to equalise factors of winds, waves and marsh type. Death of *Spartina* foliage would have decreased friction that reduces tidal water movement (Thompson, 1991; Bouma et al., 2005b), so increasing shear stress. As the root mat deteriorated, this would decrease the shear strength that normally protects the mud from erosion (Van Eerdt, 1985; Brown, 1998; Brown et al., 1999). The critical erosion threshold of intertidal sediments of the Humber Estuary was found to decrease away from the shore (Paterson et al., 2000), attributed to the different biological processes occurring with increased distance from the shore and the associated marsh surface forms. Qualitative observations of marsh surface form at the experiment site suggested that biological factors play a significant role in the spatial variability of marsh surface level, and support the findings of the empirical study.

Following removal of *S. anglica,* native salt marsh species such as *Juncus* spp. and *Selliera radicans* expanded from the landward edge by up to 1 m, where they were previously restricted to within 10–40 cm of the shore. It is likely that the expansion of both algae (Fig. 3) and native salt marsh plants have occurred as a result of the

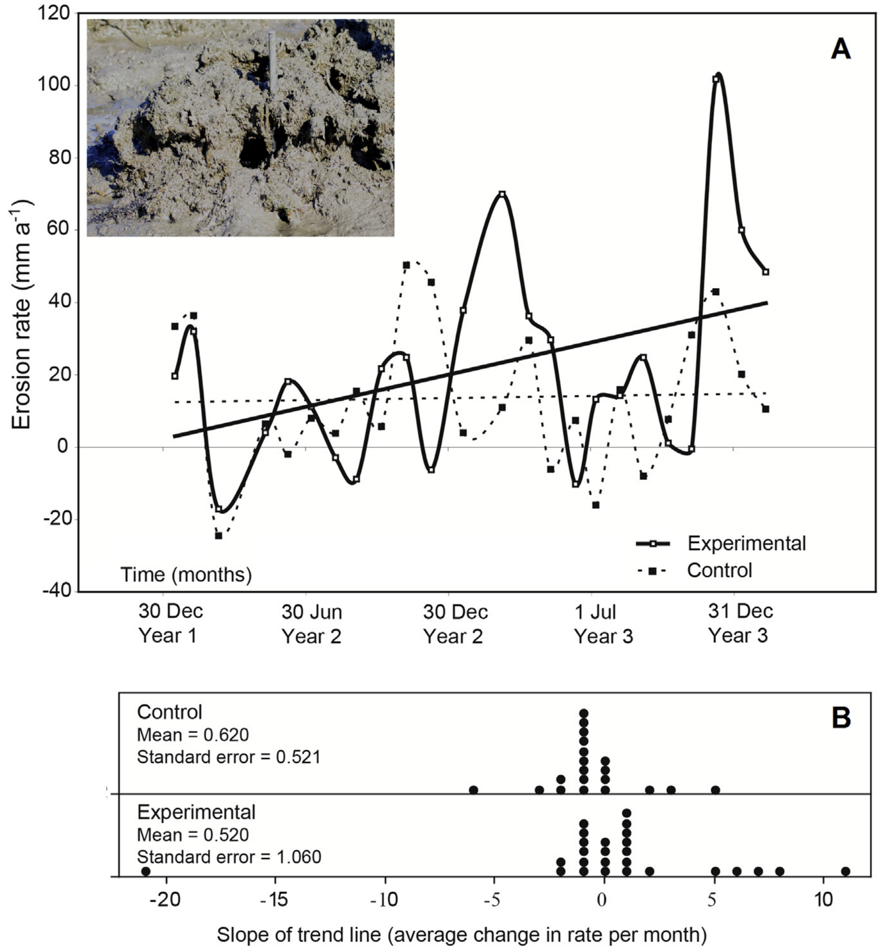

Fig. 7. **A**. Comparisons of mean monthly erosion rates of *Spartina* marsh at the seaward margin of the experimental and control sites, showing trend lines and smooth connecting lines. Trend-lines indicate that there is a greater erosional trend at the experimental site, however this is not statistically significant. Inset: Micro-cliffing and undercutting of the seaward edge. The length of the rod shown is 50 mm. **B**. Dot plots of the slope of trend line (average change in rate per month) for the seaward edge of the *Spartina* marsh.

removal of the competitive and dense canopy of *Spartina*. Vegetation has been shown to be fundamentally important in increasing the erosion threshold, stabilising sediments and reducing flow velocities (Leonard and Luther, 1995; Andersen, 2001; Bouma et al., 2005a). However, before introduction of *Spartina* the intertidal areas of the mid to upper Tamar Estuary lacked native salt marsh vegetation except for a narrow 1–2 m fringe at high water (Phillips, 1975). Native marsh plants are also less tolerant of inundation relative to *S. anglica*, so marsh recolonization is likely to remain limited.

The morphology of the marsh surfaces at both the control and the experimental site oscillated between erosion and accretion throughout the monitoring period (Fig. 4A). The magnitude of the oscillations at the experimental site was notably larger than the control site, and the timing was mostly in association with seasons. Within the experimental site, accretion was the predominant process during summer (December to March) and erosion dominant around winter (July to October), and such seasonal changes were less pronounced at the control site (Fig. 4A). Seasonal variation in tidal mud flat surfaces have been attributed to sediment accretion during calmer summer conditions, and erosion prevailing during windy conditions of winter (Kirby et al., 1993; Kirby, 1994).

Both sites become increasingly erosional towards the seaward edge (Fig. 6) along with a tendency for the upper edge of creek cross sections to widen (Fig. 8), and both changes were more pronounced at the experimental site relative to the control site. While trend lines of erosion at the seaward edge (Fig. 7a) showed increased erosion over time at the seaward edge of the experimental site, lack of significant difference suggested that marsh retreat is occurring at the same rate at both sites and that retreat is caused by a factor other than the removal of *S. anglica*. The results of the edge study may be influenced by the smaller sample size compared to the marsh study, and the large amount of variability in individual rod measurements. Erosion styles exhibited in the lower marsh were indicative of the dissection and decay processes described by Bird (2008), which were empirically and mathematically attributed to ebb-dominated tidal regimes where sediments are exported on the falling tide resulting in the a gradual long-term landward retreat of the marsh (Pritchard et al., 2002; Bird, 2008). Retreat of the seaward edge of marshes throughout the Tamar Estuary has been shown to have occurred in the last few decades, such as a 10 m retreat at Swan Bay (Fig. 1) 1989–2004 (Sheehan and Ellison, 2014).

5. Conclusions

Few studies have previously investigated the geomorphic response to a reverse restoration, from a full intertidal profile of invasively vegetated marsh back to an unvegetated, intertidal mudflat. This study showed that eradication of *S. anglica* resulted in erosion from the *Spartina* marsh surface at a rate 6 times greater than from the control vegetated marsh. This erosion was caused by erosion of sediments, with inorganic sediments comprising 83% of the sedimentary volume accumulated under *Spartina* since its introduction (Sheehan and Ellison, 2014). Further research could

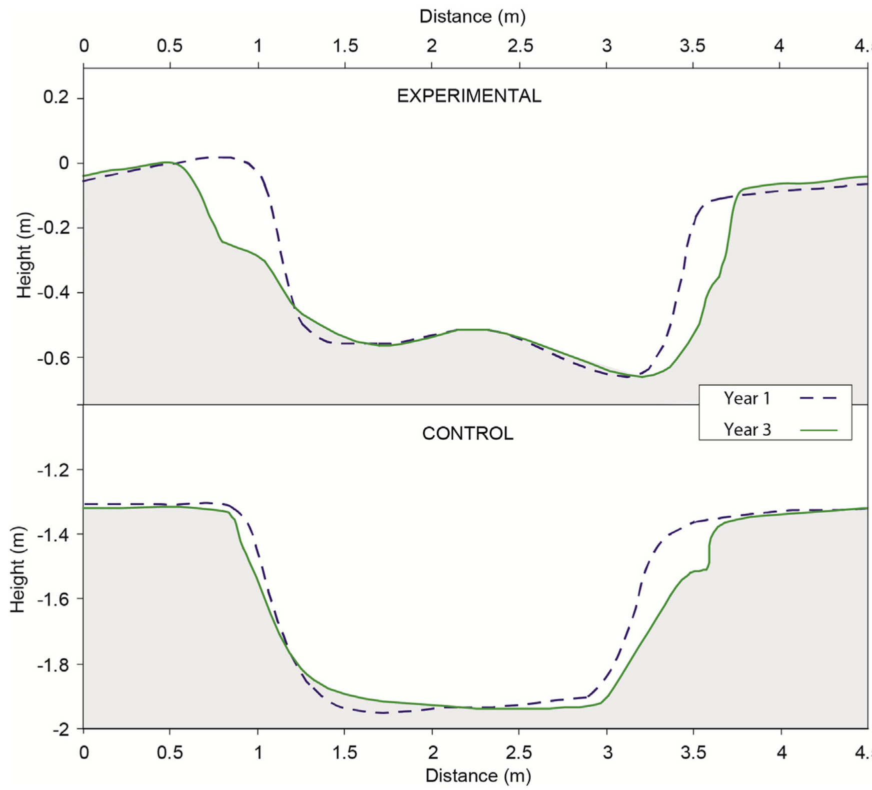

Fig. 8. Creek cross sections of the experimental and control site.

investigate contribution of different sections across the marsh surface, and whether the rate of erosion is further likely to increase once the dead *S. anglica* root mat fully decomposes and the surface cohesion and sediment-binding capacity is diminished. As intertidal wetland soils are an important carbon sink (Howe et al., 2009; Beasy and Ellison, 2013), this may also reduce the role of *Spartina* in carbon sequestration.

The sediment eroded following large scale *Spartina* removal may also increase estuarine turbidity in the short term, and larger scale removal may furthermore pose a threat to water quality or biota of the Tamar Estuary. Heavy metal and other contaminants trapped in sediment deposited decades ago may be of concern at some sites. From ^{210}Pb dating and heavy metal analysis, Seen et al. (2004) showed that a strong correlation existed between contaminant levels in intertidal sediments and mining and industrial activities over the past century, with mining pollution contributing some 10–25 mg kg^{-1} of Pb to estuarine sediments since the 1890s. Local industries such as factories, textile manufacturers, tin smelters, shipping and railway workshops, and other sources such as municipal disposal refuse sites and agriculture historically contributed to sediment contamination of the estuary (Pirzl and Coughanowr, 1997; Seen et al., 2004). There is also potential for *Spartina* removal without associated replanting to result in increased methane emissions (Sheng et al., 2014).

This study increases our knowledge of sediment stability or erosion processes following large scale marsh defoliation. It provides guidance for the future management of *S. anglica* within the Tamar Estuary, and other similar estuarine systems worldwide, and identifies areas where further research would be valuable. The synthesis of the results demonstrates that if large scale removal of *S. anglica* were to occur in the Tamar Estuary, then the sediment trapped is likely to erode in a sheet erosion manner during ebb tide, exacerbated by the widening of tidal creek channels. This is likely to increase as the root mat deteriorates, and further long term study is necessary for evaluation of this effect. Observations indicate that algae may stabilise the surface, and native salt marsh vegetation colonise in the absence of *S. anglica*, allowing for a lesser release of sediment in the short term. However the colonisation of native salt marsh plants is likely to be limited, as they are intolerant of lower elevations in the tidal range. The potential consequences of sediment erosion, particularly if the *Spartina*-trapped sediments are contaminated by historical pollutants, are further areas that need research. For the Tamar Estuary, a precautionary approach would be to continue to limit the current extent of *Spartina* infestation, rather than consider its large-scale removal from shorelines where it is now well-established.

Acknowledgements

This study was funded by the Australian Research Council Linkage Grant LP0214145 and the Department of Primary Industries, Water, Parks and Environment, State Government of Tasmania (DPIPWE). Particular thanks are extended to DPIPWE staff who helped eradicate the *Spartina*, and local residents of Little Swan Point who supported the study by providing access. The comments of two anonymous reviewers allowed improvements to the manuscript, and Rachel Sore of The University of Melbourne provided advice on our use of statistics.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.ecss.2015.06.013.

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