SALTMARSHES ON THE FRINGE: RESTORING THE DEGRADED SHORELINE OF THE EDEN ESTUARY, SCOTLAND

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Scottish Oceans Institute, School of Biology

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Restoring the degraded shoreline of the Eden Estuary, Scotland

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This thesis is submitted in partial fulfilment for the degree of PhD at the University of St Andrews

September 2014

With love to my mother, Brenda and
in memory of my late father, Geoff Maynard

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ABSTRACT

Saltmarshes are highly valued habitats but the majority of the Eden Estuary's saltmarsh was buried under sea defences and *ad hoc* rubbish dumps during the last century. Without saltmarsh the degraded shoreline may be even more vulnerable to rising sea levels and increased wave and tidal energy. This study investigated planting native saltmarsh species, common in the estuaries of Eastern Scotland, to restore saltmarsh development and sedimentation to the Eden Estuary's shoreline.

The survival and growth of the sedge *Bolboschoenus maritimus* (Sea Club-rush) and the grasses *Phragmites australis* (Common Reed) and *Puccinellia maritima* (Common Saltmarsh Grass) were compared in planting trials. These were seeded or transplanted onto unvegetated upper mudflats in front of eroded *P. maritima* saltmarsh and a disused rubbish dump. The longer term sustainability of this practice was assessed by comparing sediment deposition and surface elevation in the transplant sites, natural saltmarsh and upper unvegetated mudflats.

B. maritimus outperformed *P. australis and P. maritima.* Springtime, high density planting was successful, whereas seeds, planting in autumn and low density planting failed. Growth in the transplanted *B. maritimus* sites was relatively slow for the first three years but subsequently overtook growth of the seaward edge of natural *B. maritimus* marsh. Sediment was not deposited on natural *P. maritima* and was low on upper unvegetated mudflats and in young transplant sites. Most deposition occurred in four year old sites of *B. maritimus*. Sediment surface elevation in natural *P. maritima* remained constant throughout the year, but increased in all the other sites during the summer. The upper mudflat was the only site to erode during winter. A significant, positive association was found between tide height and sediment deposition, while winds from the south-east were associated with significantly more deposition than winds from the south-west.

The direct planting of saltmarsh vegetation has restored a valuable and rapidly disappearing habitat to the degraded shoreline of the Eden Estuary. The low-cost and simplicity of this restoration practice give it great potential as a sustainable coastal management option that should be explored in other Scottish estuaries. This form of restoration could help to increase the resilience and reduce the vulnerability of degraded shorelines to climate change and rising sea levels.

DECLARATION

I, Clare Maynard, hereby certify that this thesis, which is approximately 47,000 words in length, has been written by me, that it is the record of work carried out by me and that it has not been submitted in any previous application for a higher degree.

I was admitted as a research student in September 1999 and as a candidate for the degree of PhD in September 2000; the higher study for which this is a record was carried out in the University of St Andrews between 1999 and 2012.

Signature of candidate:

Date: September 2014

I hereby certify that the candidate has fulfilled the conditions of the Resolution and Regulations appropriate for the degree of PhD in the University of St Andrews and that the candidate is qualified to submit this thesis in application for that degree.

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CHAPTER 1 INTRODUCTION

1.1 Study motivation

Many of the saltmarsh communities in the Eden Estuary, in the region of Fife on the east coast of Scotland, have suffered considerably in the past from the dumping of waste materials along the foot of low cliffs and on to the top of the tidal flats. Parts of the coastline have been particularly disfigured and are littered with largely industrial rubbish. Low cliffs have been generated by earth and building site debris. Tidal flats frequently reveal quantities of broken china, glass bottles and items of military hardware. Other sections of the Eden's shoreline have been defended by sea walls, gabion baskets, rip rap and soft, low embankments. Most of these artificial shorelines have little in the way of fronting saltmarsh, and any that is present tends to be severely eroded and fragmented.

Scotland's mild and wet oceanic climate however, favours also the development of brackish swamp saltmarsh communities. Within the Eden Estuary these communities are apparently healthy; though the stands are discontinuous and limited in extent they grow onto the bare muds and during calm conditions mud may accumulate and cover the detritus. The stems of the plants serve to damp out motion of the waves in the waters during high tide and encourage deposition of sediment, building slightly raised zones of mud within and immediately adjacent to the marshes. These muds effectively bury stranded debris of all sorts and can return an area to a natural condition over a relatively short period. The wider the

belt of successful plants the greater the build-up of sediment and the greater the protection for the cliffs of sand and clay which lie behind them. The marsh plants propagate through seeding and also through vegetative growth, many developing a strong and massive system of rhizomes below the surface, which serve to resist erosive attack by the waves.

This study seeks to stimulate the extension of these healthy marsh stands along the reaches of shore where the most disfigurement occurs, in order to cover existing debris on or near the surface, repair and/or replace other, eroded saltmarsh, and thereby also to provide additional protection for the low cliffs which back the shoreline. Plant growth may be stimulated through seeding and transplantation. The optimum methods to ensure success and to stimulate sedimentation between them and the established marsh margin will need to be determined.

If successful in the Eden the methods could be applied at sites in other estuarine coasts in Scotland to help cover the results of tipping, enhance saltmarsh communities and protect defence structures using low cost methods and soft engineering techniques. With rising sea levels the need to develop such alternative systems of coastal protection are of increasing importance.

1.2 Background

Coastal barriers such as seawalls, embankments and rubbish tips were built around the upper shores of estuaries extensively in the past in order

to reclaim intertidal land for agriculture and development. The Eden Estuary Local Nature Reserve, a small pocket estuary on the east coast of Scotland, is a case in point because more than 60% of the shore is ring fenced by hard coastal defence structures (Fife Council, 2008). Unfortunately, this practice buried much of the saltmarsh habitat that once would have surrounded the estuary (Crawford, 2008). While the invertebrate-rich mudflats of the Eden Estuary sustain a wide variety of globally-important populations of waders and wildfowl, its saltmarsh habitat is generally classed as being in an 'unfavourable condition' by the statutory conservation authority, Scottish Natural Heritage, not only because upper marsh communities are poorly represented, but also because many of the remaining fringing saltmarsh communities are extremely eroded at their seaward edges and in the process of dying back (SNH, 2008). The extensive erosion of one of the most dominant communities, Common Saltmarsh Grass (*Puccinellia maritima* (Hudson) Parl.), is particularly worrying because this key species is the backbone upon which many other saltmarsh species depend (Gray & Mogg, 2001).

Globally and here in the UK, approximately 40 -50% of saltmarsh habitat was lost during land reclamation practices in the 20th century (Deegan et al, 2012). However, the saltmarsh habitat that remains on the fringes of estuaries is also eroding and dying back (Allen, 1992). The erosion is thought to be caused by rising sea levels and the associated increase in wave and tidal energy, in a process known as coastal squeeze, where the inland retreat of saltmarsh communities from increasing tidal

inundation is prevented by the hard sea defences which increase wave and tidal energy erosion at the seaward edge of a fringing saltmarsh (Morris et al, 2004). In turn this causes the death of the aboveground vegetation and therefore exposure of the underlying sediment to even more tidal scour. Lowered rates of sedimentation deposited onto a saltmarsh surface by incoming tidal waters can also cause saltmarsh die back and erosion (Adam, 2002). This is partly because saltmarsh communities rely on sedimentation to provide the nutrients necessary for growth. However, a lack of sedimentation also lowers the shore profile because the position, or elevation, of the marsh in the tidal frame can no longer be maintained. The combining factors of less sediment, lowered shore levels, hard sea defences, increasing tidal inundation and increasing wave and tidal energy are thought responsible for the current saltmarsh erosion and a reduction in the further colonisation, growth and development of saltmarsh habitat in the estuaries of the UK.

Despite the abundance of hard sea defences and widespread saltmarsh erosion there are a number of caveats relating to the east coast of Scotland, and in the Eden Estuary, which need to be considered. First, the rate of sea level rise has been slower around the coast of Scotland compared to other parts of the UK, largely because the land is still rebounding after the last glaciation event (Shennan & Horton, 2002). For example, values for the east coast of Scotland give an annual increase in sea levels in the order of 0.7mm per year (Ball et al, 2008; Werritty, 2012). These relatively low rates suggest that provided there is an

adequate supply of sediment being delivered to the saltmarsh surface, saltmarsh growth and development should be able to keep pace with rising sea levels, as they did in the geological past (Crawford, 2001). Second, the sand wash throughout parts of the estuary in addition to the fine muds and silts in the estuary's channel (personal observation) would suggest there should be an adequate sediment supply available for the growth and development of the saltmarsh communities. This is despite the reduction in sediment entry and lowering of shore level that occurred when a sand spit expanded during the last two decades and narrowed the channel at the mouth of the estuary (Crawford, 1998).

Even more importantly, the Eden Estuary has other saltmarsh communities that are not dying back. Like other estuaries on the east Coast of Scotland, the Eden has numerous though discontinuous stands of brackish swamp and reedbed communities (Hill, 1997). These stands extend from the upper marsh zones seawards to the low marsh zone and therefore share the same tidal range and environmental conditions as the eroding saltmarsh beds. Although limited in extent, mono-dominant stands of Sea Club Rush (*Bolboschoenus maritimus* (L.) Palla, formerly known as *Scirpus maritimus* L.) and Common Reed (*Phragmites australis* (Cav.) Trin. ex Steudel) are apparently healthy and have no signs of erosion. Swamp and reedbed communities are often not considered as common saltmarsh habitat because they require the brackish conditions that tend to occur only upriver within an estuary. These communities however, can thrive within the lower parts of estuaries on the east coast

of Scotland because sediment salinity is lowered by freshwater input from higher rainfall in the catchment area, and the early morning summer haar that is common to the east coast of Scotland (Burd, 1989; Hill, 1997). Records of increasing precipitation in Scotland (Jenkins et al, 2009) and the increasing flow of freshwater into the Eden Estuary (Chocholek, 2013) would suggest that the range and extent of these brackish saltmarsh communities may increase in the future.

In addition, there may be an increased threat to native saltmarsh communities from the expansion of invasive plant species. The Common Cordgrass (*Spartina anglica* C. E. Hubb.) was planted in the mid-20th century in the Eden Estuary to stabilise the upper shore where it expanded and flourished (Crawford, 2008). Nationwide concerns over its invasive nature caused a policy change and from the 1980s onwards there has been a systematic programme of eradication within the Eden Estuary (R. Strachan, personal communication) and elsewhere. However, the reproductive and regeneration capacity of this species means that eradication meets with limited success and only when it is intensively undertaken. Warmer temperatures associated with climate change will favour the spread of these communities (Gray & Mogg, 2001) and there is increasing concern that native saltmarsh species will be further threatened and outcompeted by *S. anglica* in the future.

Despite the apparent health of brackish saltmarsh habitat and the expansion capacity of *S. anglica*, most of the Eden's shoreline remains degraded; both the unvegetated sections and those fronted by eroded

saltmarsh. Since the early 1980s, approximately 32 ha of saltmarsh have declined to 12 ha (Fife Council, 2008). With sea level rise and coastal squeeze, and without a fronting saltmarsh, the shoreline and hinterland will become ever more vulnerable to erosion and flooding. The usual solutions to this involve engineering works such as raising embankments or replacing them with higher seawalls or gabion baskets. The continuing erosion of hazardous waste from coastal rubbish dumps may be halted by encasing the dump in concrete, or protecting it behind a seawall, or removing the waste altogether. However, hard engineering measures have a tendency to concentrate wave energy and aggravate erosion elsewhere (Brampton, 1992). They are also expensive, biologically impoverished and do not address the loss of saltmarsh habitat.

Managed realignment, the deliberate setting back of the coastline, is considered to be a solution to rising sea level and coastal squeeze (Garbutt, 2009). The practice has a high conservation merit because it can restore large areas of saltmarsh within a relatively short space of time. Conversely, it is costly, not feasible for rubbish dumps, because it is not known precisely what the rubbish contains, and is not practical when the hinterland is highly developed, such as that which surrounds the Eden Estuary, with the world famous St Andrews Links Golf Course, a RAF base and highly productive agricultural land.

This study therefore investigated an alternative and sustainable method of coastal protection by using the healthy and non-eroding brackish marsh communities to create new saltmarsh habitat in sections

of the Eden's degraded shoreline. Regeneration of the existing saltmarsh by transplantation would ensure the survival of the habitat, whilst remedial saltmarsh planting of entirely new reaches of saltmarsh habitat will enhance the overall diversity and functionality of the estuarine ecosystem. It may be possible to restore the natural process of saltmarsh development and sedimentation through the creation of young, actively developing marshes. This would go some way to building in resilience and reducing the vulnerability of the shoreline to rising sea levels. Planting a belt of vegetation even some two meters wide could reverse the erosion on the disfigured stretch of coast and protect the remaining saltmarsh from being colonised by invasive species. Also, it will be important for birds, increase overall biodiversity and work towards British Action Plan (BAP) objectives and the EU Water Framework Directive (WFD).

Though there is a wealth of information available about saltmarsh restoration during managed realignment schemes (Brooke et al, 1999; Garbutt, 2009), there is little information about saltmarsh creation on unvegetated and eroded shorelines. Findings from other studies provided some insights but could not serve to expand the knowledge base as effectively as planting trials in the field.

1.3 Study objectives

The overall objective of this study was to attempt to restore saltmarsh vegetation and stimulate sedimentation in the degraded sections of the Eden Estuary's shoreline. The following questions were therefore posed:

1. Would it be possible to transplant native saltmarsh species into degraded sections of the Eden Estuary's shoreline?
2. What methods of planting would be the most successful?
3. Would the environmental conditions at the selected sites have any effect on the success or failure?
4. If restoration were successful, how would the subsequent growth in the transplanted marsh compare to growth in natural marsh?
5. How would sedimentation, as measured by deposition and accretion, in the transplant sites compare to that in natural marsh and upper unvegetated mudflats?

1.4 Thesis structure

A review of the Eden Estuary's saltmarsh, in terms of status, function and management is considered in greater detail in the next chapter, along with greater detail of the issues of climate change, such as sea level rise and coastal squeeze, pollution, sedimentation and invasive species. The third chapter describes experimental and statistical design and provides more detail on each selected study site, in addition to the methods used

in the planting trials, the measurement of the environmental conditions at each site, and how vegetative growth and sedimentation were recorded. The fourth and fifth chapters present the data from the vegetation and sedimentation studies respectively, while the sixth chapter discusses these findings, provides recommendations and makes suggestions for the direction of further work. A conclusions chapter provides a final and clear summary of the study.

CHAPTER 2 THE SALTMARSHES OF THE EDEN ESTUARY

2.1 Introduction

Similar to other saltmarsh habitat in the estuaries of Eastern Scotland, the saltmarshes of the Eden Estuary are botanically unique within the UK because saltmarsh communities from northern and southern parts of the UK meet (Crawford, 1998). While the northern, brackish swamp community type within the estuary appears stable, the southern, more Mediterranean type of community that is also present has all the hallmarks of the erosion and die-back of saltmarsh habitat both globally and in the UK. Like all estuaries where human development has been significant around the shores, the Eden is ring fenced by a mixture of old and crumbling or new sea defences that protect valuable hinterland (Figures 2.1 and 2.2).

These sea defences have little in the way of fronting saltmarsh and most of these communities have symptoms of extreme die-back, such as fragmentation and erosion at its seaward edge. The erosion was recorded in some areas during the 1990s as being in the order of approximately one metre per year (Fife Council, 1995). Not all of the Eden's saltmarsh is degenerating however, with some stands being stable; if not actively developing, they are neither eroding nor dying back. This presents a dichotomy and raises the question that it may be possible to resurrect a former practice of saltmarsh restoration through the direct planting of saltmarsh vegetation into the areas that are eroding.

To set the study in context, this chapter provides an overview of the Eden Estuary, describes the past and current status of the estuary's saltmarsh, outlines the importance of saltmarsh habitat and discusses saltmarsh formation and sedimentation. Saltmarsh loss and the processes that have affected saltmarsh development will also be described, followed by review of the saltmarsh restoration techniques currently or formerly practiced to halt shoreline erosion and saltmarsh loss and degradation.

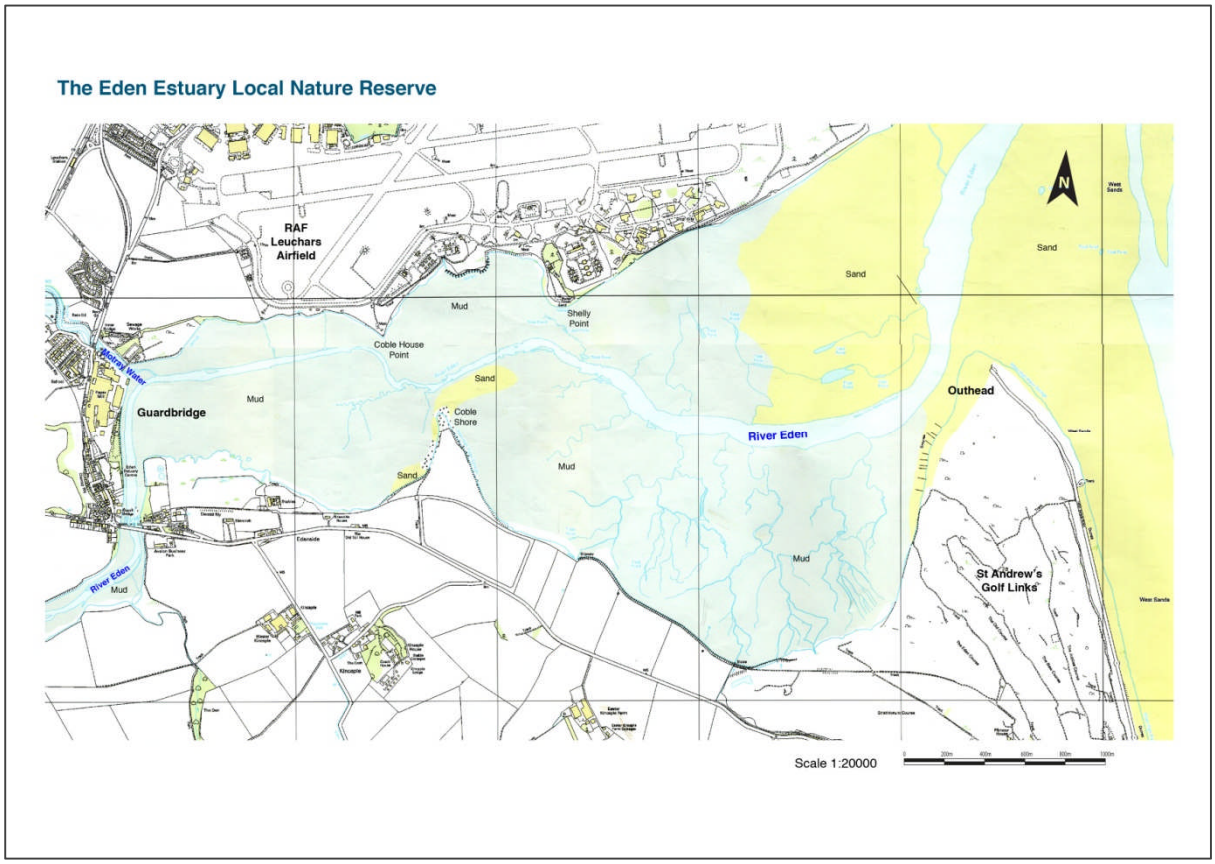


Figure 2.1 Map of the Eden Estuary Local Nature Reserve and surrounding area.

Figure 2.2 Aerial view of the Eden Estuary Local Nature Reserve (courtesy of D. Paterson & RAF Leuchars).

2.2 An overview of the Eden Estuary

The Eden Estuary is a small and comparatively wide and shallow pocket estuary on the east coast of Scotland, located between St Andrews to the south and The Tay Estuary to the north (central grid reference NO4701 95; Figure 2.1). The Eden Estuary was designated as a Site of Special Scientific Interest (SSSI) in 1971 and as a Local Nature Reserve (LNR) in 1978. It is also part of the Firth of Tay & Eden Estuary Special Protection Area (SPA) under the Birds Directive and the Firth of Tay & Eden Estuary Special Area of Conservation (SAC) under the Habitats Directive. It is therefore a Natura 2000 site, which are made up of SPAs and SACs. It is

also part of The Firth of Tay & Eden Estuary RAMSAR site for its wintering wildfowl and waders and has Class A (excellent) status under the EU Water Framework Directive in the estuary classification scheme.

The Eden is a well-mixed estuary but freshwater input is low compared to tidal flood water (SEPA, 1998). Salinities range from a uniform 28 ppt at high tide (Johnston et al, 1979) to 20-30 in the sand and mudflats when exposed at low tide (Bates et al, 2004). Freshwater input comes from the River Eden, Motray Water and the Moonzie Burn, which drain approximately 320 km^2. The gradients within the estuary vary and in the central parts are between 1:200m and 1:300m. Sediment grain size generally increases towards the mouth of the estuary and decreases towards Guardbridge (Eastwood, 1976). The finest sediments are on either side of the river channel and the channel bed is covered in gravel (McManus and Green, 1976). Wave heights between 0.4-1.0 metres have been recorded within the estuary (Posford Duvivier, 2000). Tidal currents between <0.1-0.15 m/s were recorded at Outhead (HR Wallingford, 1992) and between 0.24m/s and 1m/s in the river channel (McManus & Green, 1976). Wave and tidal energy is weak on the upper shoreline during calm conditions and wave energy tends to dominate (personal observation).

2.3 Saltmarsh distribution in the UK

Saltmarsh is a rare habitat and in the UK accounts for only 44,500 ha (Jones et al, 2013) compared to ancient semi-natural woodland, another

rare habitat of which the UK has only 350, 000 ha. It is also one of the UK's most natural ecosystems if enclosure and/or grazing have not occurred (Burd, 1989), even compared to semi-ancient woodlands or raised bogs. Most saltmarsh in the UK is found in the low-lying soft shores of the southeast and the northwest of England (Figure 2.3). Scotland has very little, largely due to its mainly hard, rocky and exposed coastline. Scotland's largest concentrations of saltmarsh therefore are found in the low lying land of the south-eastern and south-western parts of the country (Burd, 1989) such as the larger firths of the Forth, the Tay and the Solway. The Solway Firth in the south-west of Scotland is the largest single expanse of saltmarsh in the whole of the UK (May & Hansom, 2003) and the rest of Scotland's saltmarsh, approximately two-thirds, is in the east coast firths, such as the Tay Estuary and the Firth of Forth (Burd, 1989).

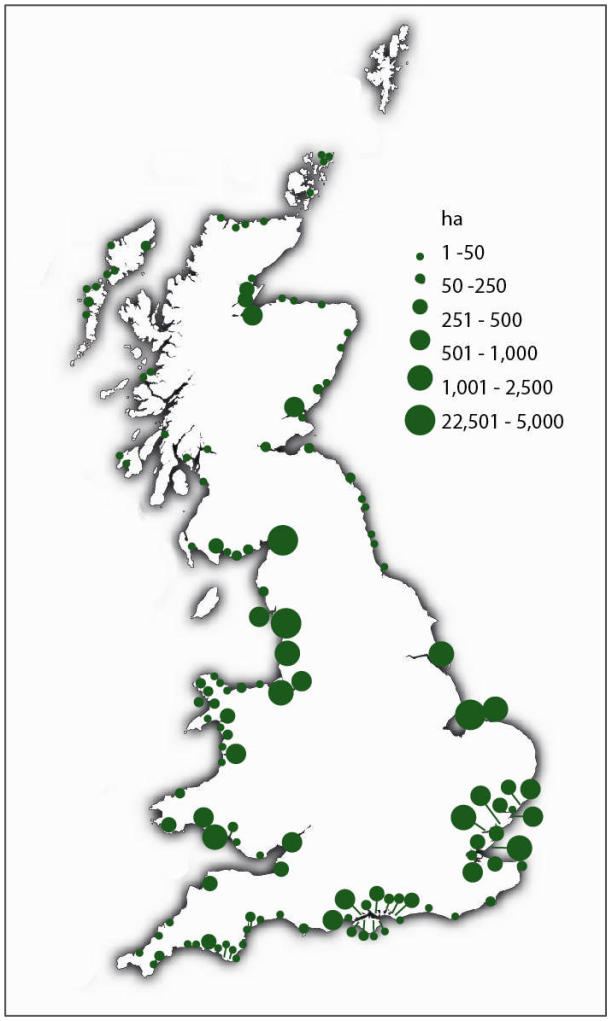


Figure 2.3 The distribution of saltmarsh in the UK (redrawn from Davidson et al, 1991). The size of the symbol shows the approximate area of saltmarsh.

2.4 Saltmarsh communities of the Eden Estuary

Most UK saltmarsh communities are represented in the Eden Estuary These range from pioneer communities of Glasswort (*Salicornia europaea*), low-mid marsh communities of the Common Saltmarsh Grass (*P. maritima)* to upper marsh communities of Saltmarsh Rush (*Juncus gerardi)* (Fife Council, 2008). The Eelgrasses *Zostera*

augustifolia, Z. noltii, and *Z. marina* are nationally scarce but all have been recorded within the Eden Estuary. A transition of zones based on tidal inundation is apparent (Figure 2.4) though most communities are confined to a narrow ribbon along the shoreline.

Figure 2.4 A transitional saltmarsh community at Coble Shore, the Eden Estuary. Note the narrow fragments of P. maritima between the shingle and terrestrial vegetation.

In common with other UK saltmarsh and indeed most temperate saltmarsh around the globe, more than 50% of the Eden Estuary's saltmarsh is composed of *P. maritima* communities (Table 2.1), fronted by the annual pioneer species *S. europeae*, which is a

common sight in most estuarine saltmarsh around the UK (Figure 2.5).

Table 2.1 The saltmarsh & swamp communities of the Eden Estuary (The Eden Estuary Management Plan 2008-2013).

Community Type	**Total Area (ha)**	**Percentage**
Spartina pioneer marsh	0.3	1
Salicornia/Suaeda pioneer marsh	3.0	10
Puccinellia low-mid marsh	1.4	5
Puccinellia mid-upper marsh	17.0	53
Festuca/Agrostis/J.gerardii mid-upper marsh	4.1	13
Blysmus/E.uniglumis communities in mid - upper marsh	0.009	0.03
Upper marsh/swamp communities: a) *Agropyron repens* dominant	0.6	2
b) *Bolboschoenus maritimus* dominant	3.1	10
c) *Phragmites* dominant	1.8	6
d) *S. tabernaemontani* dominant	0.007	0.02
e) Phalaris dominant	0.14	0.4
f) Oenanthe dominant	0.01	0.03
g) *Glyceria* dominant	0.44	1
Total	32.0	100

Figure 2.5 An actively developing bed of the pioneer S. europaea, expanding over the upper mudflat in front of a narrow strip of P. maritima. An embankment of terrestrial vegetation is visible in the background.

The majority of saltmarsh in the southern UK is mainly composed of strictly halophytic species such as *P. maritima* and *S. europeae* whereas other brackish swamp saltmarsh communities such as reedbed and brackish swamp communities are occasional in distribution, comprising only 5% of the upper marsh zone (Doody, 1997). However, saltmarsh communities in the northwest of the UK have a high predominance of reedbed and swamp communities because of higher rainfall, early morning summer haar and higher freshwater input from river catchments and therefore can occupy over 50% of the upper marsh

zone. The saltmarsh communities of the firths of Eastern Scotland (the Cromarty, the Montrose Basin, the Tay, the Eden and the Forth) are therefore distinctly intermediate in character between these southern and northern types (Leach & Phillipson, 1985; Proctor, 1987; Hill, 1997). Here in Scotland reedbed and swamp communities are often referred to as 'tidal reed' because they form the majority of the saltmarsh communities of the middle and inner parts of the Tay and Forth estuaries, for example. Indeed, the Firth of Tay is home to the largest single expanse of both tidal reed and saltmarsh in Eastern Scotland (Leach & Phillipson, 1985).

The saltmarshes on the Eden Estuary are relatively small (32 ha) compared to those of the larger firths of Eastern Scotland, but nevertheless have a high botanical importance because the transition between the southeast and the northwest so clearly meets around its shores (Leach & Phillipson, 1985; Crawford, 1998). Brackish swamp *B. maritimus* (Figure 2.6) and *P. australis* (Figure 2.7) are locally extensive in the Eden Estuary (approximately 16% of the total).

Figure 2.6 B. maritimus marsh at Edenside extending from landward upper marsh to seaward lower marsh (Guardbridge in the background).

In some areas around the Eden's shore these stands form transitional to freshwater swamp or fen, as they would in southern UK saltmarshes, but in other places they occupy low and mid marsh zones, being completely or partially inundated during high tide, respectively. This is especially true on the northern shore of the Eden Estuary where the saltmarsh habitat is largely dominated by tidal reed and swamp, though there are some stands of *P. maritima* marsh. Sediment salinity on the northern shore is likely reduced by a number of local factors, e.g., a high proportion of surficial clay channelling freshwater seepage along the estuary's margins, lower farming activity on the adjacent land and the local

haar in the morning that shrouds the estuary in summer. Marsh stands extend well into the pioneer zone; a zone more usually associated with higher salinities and wave action.

Figure 2.7 A stand of P. australis extending from a sheltered inlet seawards into the low marsh zone.

On the Eden's southern shore these brackish communities are only present in the inner reaches of the estuary, where soil salinity is reduced by the entry of freshwater from the River Eden and the Motray Water (see Figure 2.1). The typical zonation of *P. maritima* and *S. europeaa* communities in the central estuarine area of the southern shore more closely resembles saltmarshes in the south-east of the UK, and especially moving towards the towards the mouth of the estuary, the presence of

the St Andrews Links Golf Courses indicates that the natural habitat landward of the marsh is sand dune and therefore in keeping with southerly and Mediterranean types of saltmarsh as defined by Adam (1978).

2.5 The importance of saltmarsh

Approximately 80% of the UK's saltmarsh has been granted SSSI status (Davidson et al, 1991) and saltmarsh habitat is further protected by the EU Birds Directive and Habitats Directive. This protection is because it is increasingly valued for the ecosystem services it provides such as coastal protection, pollution filtration, nutrient turnover, resource production and carbon fixation (Boorman, 1999; Chmura, 2009), especially in light of climate change and sea level rise. The Eden Estuary and its saltmarshes are highly specialized and dynamic ecosystems that have a high ecological and economic significance to the surrounding area (Table 2.2).

Saltmarshes support a wide range of marine organisms, often commercially valuable species, which depend on the habitat at some point during development. Saltmarshes sustain many bird populations by providing a high tide refuge for birds that feed on the mudflats, as a breeding site for waders, feeding grounds for geese during winter, as well as supporting a wide range of passerines and birds of prey (Davidson et al, 1991). Saltmarsh also supports a wide range of terrestrial invertebrates, such insects and arachnids inhabiting the vegetation (Foster, 2000). The richest areas for terrestrial invertebrates tend to be

where saltmarsh grades into other terrestrial habitats (Kinnear, 1996) because of the high floristic and structural diversity (Adam, 1990). The continued survival of many specialist, salt-adapted plant species are dependent on saltmarsh (Crawford, 2001).

Saltmarshes break down organic matter, filter sediment and nutrients from upland waters and there have a key role in the cycling of organic material and nutrients important for the marine food chain (Nedwell, 2000; Boorman, 2000; Laffoley & Grimsditch, 2009). Saltmarsh therefore benefits society indirectly through the food chain and in the conservation of wildlife.

Saltmarsh also has recreational and aesthetic appeal, providing walking, fishing, wildfowling and bird watching. The vegetation can capture and remove pollutants from the water column. More directly, saltmarshes absorb floodwater and dissipate storm surges. The presence of saltmarsh stabilises the shoreline, especially when human development has been significant. Saltmarsh has the ability to promote sediment accretion, resist wave energy, withstand storms and help prevent erosion (Brooke et al, 1999). These abilities mean the entire marsh acts as a buffer to the coastline (Brampton, 1992).

Table 2.2 Saltmarsh services either of direct benefit to wildlife or as part of the estuarine ecosystem.

Direct wildlife benefits	**Wider estuarine functions**
High tide refuge for waders	Shoreline stability
Breeding sites for range of birds	Sediment accretion
Feeding ground for geese	Wave attenuation
Inshore fish nurseries	Nutrient source
Marine invertebrate habitat	Carbon sink
Specialist plants	Flood plain
Insect/amphibian habitat	Pollution trap
Grazing for terrestrial mammals	Amenity value

The importance of the role of saltmarsh habitat in flood defence to the coast has been recognised within the UK flood risk management plans (DEFRA, 2009). King and Lester (1995) for example, estimated that if the saltmarsh along the entire coast of Essex were removed, rebuilding sea defence walls to replace their function would cost £600 million. It was further argued that the presence of a saltmarsh can reduce the overall costs of sea defence because of their natural ability to dissipate wave energy (Table 2.3; Möller et al, 2001). As can be clearly seen, the wider the saltmarsh, the lower the height and cost of a seawall. This saving is directly attributed to the presence of the vegetation acting like a baffle to reduce the force of wave and tidal energy against a seawall.

Table 2.3 The relationship between the width of a saltmarsh and the height and cost of a sea wall (Möller et al, 2001).

Width of marsh (m)	Height of seawall (m)	Cost (per m)
80	3.0	£400
60	4.0	£500
30	5.0	£800
6	6.0	£1,500
0	12.0	£5,000

The economic value of Scotland's saltmarshes is not yet available, although research is underway at both Glasgow and St. Andrews universities (personal communications). Scotland's saltmarsh habitat is relatively small in comparison to those in England (3% and 24% of the coastline, respectively), but the majority are correspondingly located in heavily populated and developed estuaries (Jones et al, 2013), and it is probable that the cost of replacing them with flood defences would be considerable also. As the table of costs above shows (from currency values more than a decade ago) it would also be expensive to raise the height of any existing seawall in order to reduce the incidence of coastal flooding as a result of rising sea level.

2.6 Saltmarsh formation and development

Saltmarsh generally forms in quiet, wave-sheltered areas, to such an extent that more than 90% of saltmarsh habitat occurs in estuaries

(Davidson *et a!,* 1991), and saltmarshes are the natural habitat of upper tidal flats in temperate regions around the world (Adam, 1990). Complex interactions between climate, tidal inundation, salinity, sediment type and availability determine the composition of saltmarsh vegetation, though models relating to saltmarsh formation are very general (Figure 2.8). Intertidal sand and mudflats are initially stabilised by the binding action of surface algae, e.g. diatoms and *Enteromorpha* spp., but the first flowering plant, the Eelgrass (*Zostera* spp.), only colonises the mudflat when the height of the sediment exceeds mean high water neap tides, where its essentially aquatic nature can tolerate the high salinity and physical movement of each tide (Adam, 1990). Flowering plants with a terrestrial form, like Cordgrass (*Spartina* spp.) and Glasswort (*Salicornia* spp.) colonise slightly higher up the tidal frame than Eelgrass.

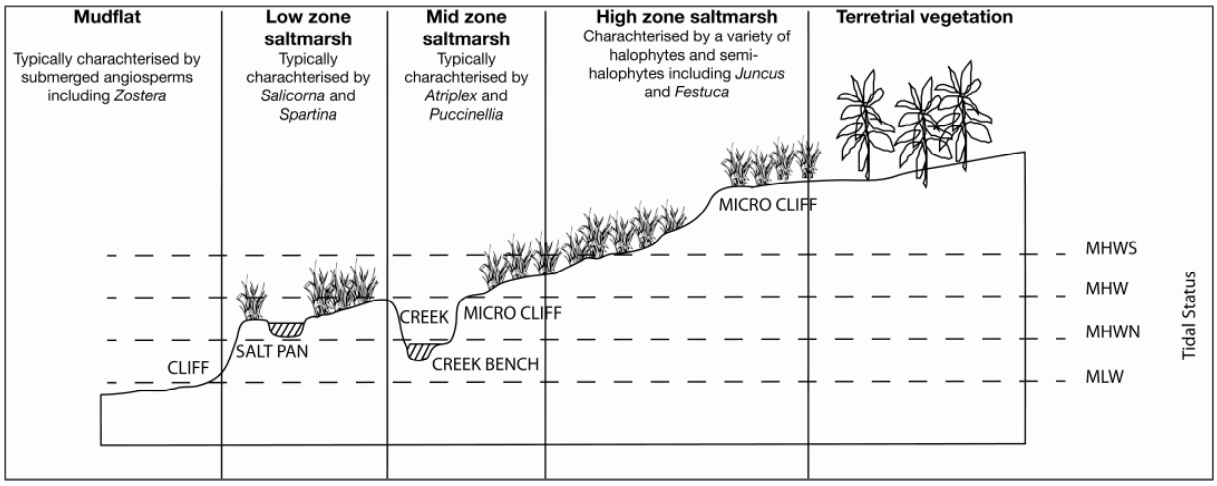


Figure 2.8 Generalised saltmarsh profile showing the main saltmarsh vegetation zones in relation to the stage of the tide (redrawn from Brooke et al, 1999).

These colonists maintain surface stability through root mats and by anchoring the sediment (Adam, 1990) while the plant stems reduce water velocity (Moller, 1999) and cause sediment to drop out of suspension from the inundating tidal waters (Reed, 1999). Saltmarsh sedimentation increases over time, and slowly raises the height of the vegetated zone, thus reducing the frequency and duration of tidal inundation. As this accretion process continues, transitional zones to terrestrial habitats develop (Adam, 1990). The reduction in tidal inundation in the upper marsh means that the opportunity for sedimentation to occur decreases and a change of plant community occurs. Total species richness generally increases with elevation (Doody, 2008) and a characteristic zonation of the vegetation becomes apparent (Figure 2.9).

Figure 2.9 Saltmarsh zonation on the south shore of the Eden Estuary. A low, pioneering marsh of S. europeae and a mid-marsh of P. maritima (covered with Sea Aster) occupy the zones between the sea (background) and terrestrial vegetation (foreground).

There are few developing marshes left in the UK (Adam, 2002), though there are notable exceptions, for example, in parts of Morecambe Bay on the northwest coast of England. The lack of unimpeded succession is the underlying cause responsible for the lack of marsh development in the face of increasing pressure from climate change and sea level rise. The continued survival of many estuarine saltmarshes therefore has been called into question (Adam, 2002).

2.7 Saltmarsh loss

The importance of saltmarsh is recognised by current protection measures but in former times, and in the most populated estuaries, land reclamation greatly reduced the amount of UK saltmarsh, and what remains on the seaward side of enclosed land has since undergone rapid erosion and die-back, with the greatest losses recorded in south-east England (Pye & Allen, 2000). Given that most human development has been along the banks of rivers and estuaries, it is no surprise that the greatest losses to the habitat have occurred within estuaries. Historically, saltmarsh habitat occupied a large portion of the sheltered and low-lying land that surrounds estuaries (Adam, 1990) and the mudflats and

saltmarshes were generally resilient over the course of human history (Crawford, 2008). The direct loss of saltmarsh habitat was a common occurrence in the past when embankments were raised and sea defences were built around the fertile land that surrounds estuaries. Upper marsh zones were reclaimed and converted to farmland and all that remains of far greater populations are isolated and fragmented saltmarsh communities on the seaward side of defences and embankments (Allen, 1992). The saltmarshes in the Eden may be botanically unique but like all other developed estuaries, the impact of land claim on the majority of the saltmarsh communities was severe. Biological diversity is a necessary prerequisite to the continuing selection and adaptation to changing environmental conditions, but gene flow between increasingly smaller and isolated plant communities is greatly compromised by fragmentation (Crawford, 2008).

2.7.1 Land reclamation in the Eden Estuary

Small-scale losses to saltmarsh have occurred due to increases in farming activity ever since medieval times but, over the last 200 years, land reclamation and industry has substantially increased the loss (Crawford, 2001). Documents relating to the earlier history of the Eden's saltmarshes are scarce, but maps provide some evidence that large-scale changes to the surrounding area took place as early as the 1850s. For example, the Leuchars to St. Andrews railway was constructed in the

mid-1800s on an artificial embankment approximately 4.0 m above sea level, which cut off the low-lying hinterland on the south shore (formerly the upper marsh zone) from tidal influence and allowed increases in farming activity.

Maps dating from the 1880s also show a number of tile works located inland at the head and around the inner parts of the estuary, implying the drainage and removal of marsh to extract the underlying clay. A great part of the natural landscape however, must have been preserved at the start of the 20th century, as Wilson (1910) noted that reed formed 'a jungle of considerable extent' around the confluence of the Motray Water, and that Eelgrass (or grasswrack) 'clothes the mudflats in summer'. In describing its use in the thatching, packing and stuffing industries, Wilson implied a far greater abundance than is present today. Referring to the estuary's southern shore, he described 'vast acres of salt-grass flats occupying a large space on solid, dense, blue clay a foot or two above the mudflats', with 'minimal erosion because of the shelter afforded by the estuary'.

It was shortly after this period that the higher ground above the north shore of the estuary began to be used as an airfield during the First World War. This is today RAF Leuchars. Presently the land between Tentsmuir Forest and the airfield, called Earlshall Muir, is a rich and complex mosaic of wetland and dune habitat and it is likely that the land the airfield was built upon would also have been a mosaic of habitats. During the 20th century this land was developed and the landward edge

of the estuary's shoreline consolidated. The old runway was replaced during the Second World War, and the concrete waste from the old runway was tipped along a significant stretch of the shore in the area known as the Coble Flats (Figure 2.10), burying the existing saltmarsh in the process (Crawford, 2001). The loss of saltmarsh through direct burial or reclamation is relatively rare at present, although some still occurs during the development of marinas, for example.

Figure 2.10 The use of rubble as a sea defence tip at the RAF Leuchars boundary.

2.7.2 Sea level rise and climate change

Global sea levels have risen between 10-20 cm during the 20th Century, and appear to have accelerated since the late 1990s (IPCC, 2007). Long term measurements at Aberdeen show a rise of around 0.7 mm per year (Ball et al, 2008; Werritty, 2012) but increases in sea level have been offset by the rise in the Scottish landmass that has occurred since the weight of the ice sheets at the end of the last ice age has been removed in a process known as isostatic rebound. This rise has been measured at about 1 mm per year (Shennan & Horton, 2002; Figure 2.11) and therefore the Eden Estuary is in a zone of relative sea-level fall. Whether the land will continue to rise or has come to a halt creates some uncertainty for the future for Scotland's coasts.

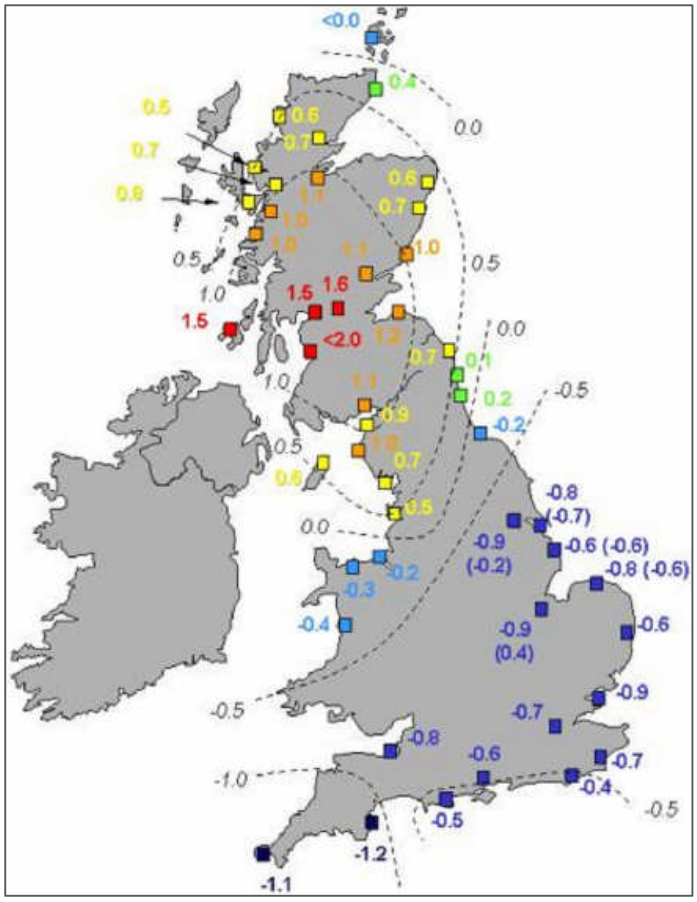


Figure 2.11 Late Holocene relative land/sea-level changes (mm/yr) in Great Britain. Positive values indicate relative land uplift or sea-level fall, negative values are relative land subsidence or sea-level rise (from Shennan & Horton, 2002).

Increased storm frequency associate with climate change can also create severe storm surges that can overtop and damage sea defences. SEPA's flood map showed that some 60-70% of the Links Trust golf courses will be extensively damaged in the event of a 1 in 200 year storm surge whereas Jarvis (2007) showed that a 10-year storm surge of 3.7m (OD) would not greatly affect those parts of the golf course that are protected by gabions, while those parts of the golf course whose

boundaries lie within the Eden Estuary and are protected by saltmarsh and low embankments are at risk (Defew & Paterson, 2009). However, according to Townend & Pethick (2002) the biggest threat from storm surge is at the head of an estuary. The head of the Eden Estuary is at the Guardbridge former papermill (see Figure 2.1) which would appear to have been an abundant wetland area in the past (Wilson, 1910). Wave conditions are also likely to be affected through changes in weather patterns and increased storminess, leading to an increased risk of inundation and coastal erosion and changing patterns of sedimentation.

Saltmarsh has in the past responded to flooding and rising sea levels by expanding in sheltered estuaries (Crawford, 2008). Marsh accretion rates in the Thames Estuary (French & Burningham, 2003) and Essex (van der Wal & Pye, 2004) show that the growth of saltmarsh vegetation should be able to keep pace with the current levels of rising seas, and furthermore should match the projected rates of mean sea level rise. Allen and Pye (1992) have shown that the medium and long-term evolution of saltmarshes can be influenced by relative sea level rise and that it mostly affects their vertical growth. Therefore the real concern with regards the future of saltmarsh habitats and sea level rise will be in the ability of saltmarsh habitats, in the Eden or elsewhere, to maintain vertical growth through sedimentation and therefore stay ahead of increasing tidal inundation. Furthermore, though the increase in wave and tidal energy, whether from sea level rise or increased storminess can reduce the horizontal growth of saltmarshes due to increased scouring,

this very scouring has been shown to release sediment from the leading seaward edge of saltmarsh and deliver it into the main body of marsh in order to maintain vertical growth (Morris et al, 2004).

Climate change effects such as sea level rise and increased storminess also have to be balanced with other effects such as increased precipitation and therefore freshwater input to estuaries. Scotland's climate is oceanic and distinctly mild and wet, and records show that the average annual rainfall has increased over the last 45 years (Jenkins et al, 2009). More specifically, summer rainfall has decreased and winter rainfall increased in the east of Scotland, but the overall long term trend of increasing freshwater input into the Eden Estuary (Figure 2.12) has been correlated to monthly rainfall averages (Chocholek, 2011).

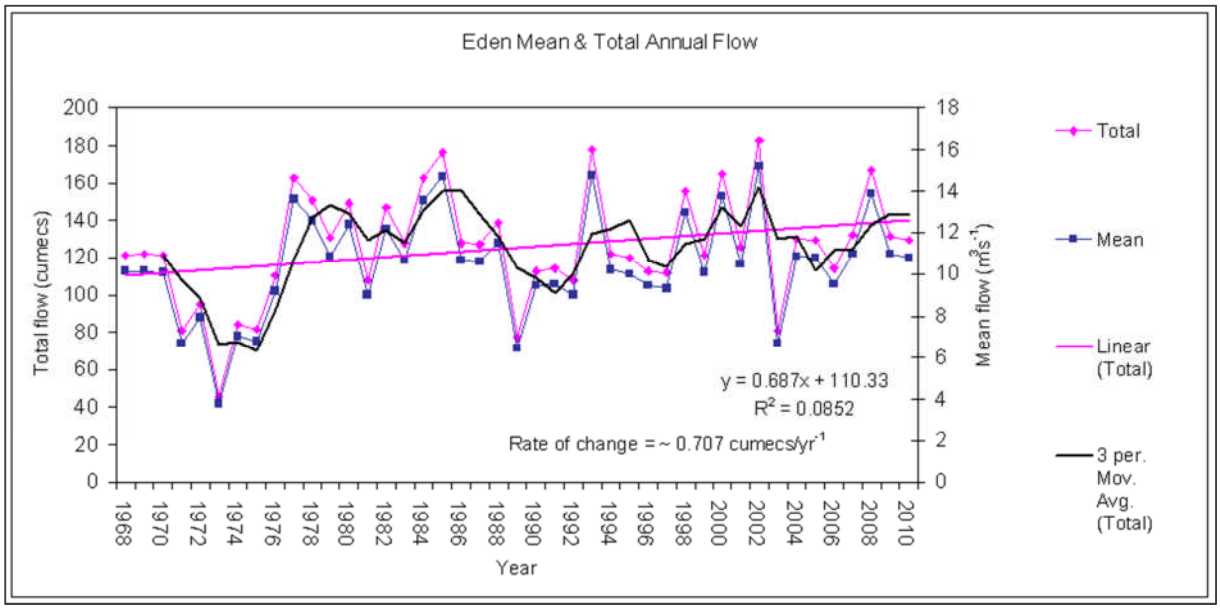


Figure 2.12 River Eden mean flow (ms-1) and total annual flow in cumecs (Chocholek, 2011).

This may have opposing effects upon the saltmarsh communities within the Eden. First, the increase in water depth over the saltmarsh surface could lead to vegetation die back, but conversely, there would be a corresponding reduction in sediment salinity due to the increasing input of freshwater, which would benefit the expansion of brackish and swamp communities, suggesting that the inimical effect of sea level rise on some saltmarsh communities may be balanced with a beneficial effect on swamp communities, and thus potentially changing the vegetation composition and distribution within the Eden's saltmarsh habitat.

2.7.3 Coastal squeeze

Coastal squeeze is a process that describes how the building of sea walls and/or embankments for land reclamation prevent marsh communities from migrating inland, away from the increasing water levels associated with rising sea levels (Wolters et al, 2005b). This process is most pronounced in the south-east of England because the isostatic tilting of the UK landmass increases the rate of relative sea level rise (Pye & Allen, 2000). Relative sea levels raise the low water mark while the high water mark is held in place by defences resulting in a reduction in the width of the intertidal zone.

However, current saltmarsh loss in the UK is caused by tidally induced erosion to saltmarsh communities on the seaward side of man-made structures. In relatively large expanses (or width) of saltmarsh with the presence of an upper marsh, such as those at Aberlady Bay in the

Firth of Forth, and where there are no hard sea defences, the erosion damage to saltmarsh communities is minimal. Though some saltmarsh erosion may occur as natural senescence takes place, these tend to be replaced by young, active communities (personal observation).

Within the Eden Estuary however, approximately 62% of the estuary's shoreline has been artificially reinforced and wherever these occur, the adjacent saltmarsh has become eroded. In 1989, 32 ha of saltmarsh remained. By 2008, *P. maritima* saltmarsh habitat in the Eden was reduced to 12 ha, constituting about 50% habitat disappearance in just 20 years (Fife Council, 2008) There is little in the way of natural saltmarsh regeneration at the forefront of the degraded marshes (Figure 2.13) and it is possible that coastal squeeze, and increasing wind and wave activity acting on the seaward edge of the marsh, may be the cause of this die-back.

Figure 2.13 A compacted sediment platform stripped of vegetation on a stand of P. maritima on the Eden's southern shoreline. The raised area in the middle of the image represents the former height of the vegetation.

2.7.4 Sediment availability

Sediment being carried onto a saltmarsh by tidal waters can be a pre-requisite for saltmarsh development (Fragoso, 2001; Adam, 2002), but there is evidence to suggest that there has been a reduction in sediment availability during the last century. Vast quantities of sediment were deposited offshore during, and at the end of, the last Ice Age but over time this sediment has been reworked and deposited to form sand dunes, spits and barriers both around the coast and at the mouths of major estuaries (Dyer, 1997). It is thought that this supply is becoming exhausted (Jones et al 2013) but St. Andrews Bay is still replete in

sediment (McManus, 1998). Within the Eden basin however, Crawford (2008) suggested that the die-back of the saltmarsh within the Eden Estuary is the on-going effect of a lowered shore profile, caused by a reduction in the supply of sediment entering the estuary when the river channel at the mouth became narrowed due to the growth of a municipal rubbish dump at the Outhead sand spit. In addition, the straightening and entraining of watercourses throughout the upland river basin may also have resulted in less silt being transported downstream. However, sand wash is apparent in many areas within the Eden, most noticeably at Shelly Spit on the northern shore, whilst both the Eden and Motray rivers run brown with silt during every spring and autumn (personal observations).

The Eden Estuary's saltmarsh habitats present anomalies with regards the effect of species response and distribution to climate change and sea level rise, or sedimentation and the ability of the saltmarshes to trap sediment and correct a falling shore level to reduce the impact of coastal squeeze. However, the Eden's saltmarshes have also been impacted by pollution and the introduction of invasive species and these are discussed in the next section.

2.7.5 Pollution

The Eden Estuary has suffered from pollution in the past including high fish mortality, the closure of mussel beds as unfit for human consumption and increased metal content near former landfill sites

(Defew & Paterson, 2008). It is likely that pollution also impacted the Eden's saltmarsh communities though it is not known whether the saltmarsh sediments contain large concentrations of heavy metals. In southern estuaries, heavy metals found in saltmarsh sediments dating from the 1900s (Adam, 1990) and herbicides (Mason et al, 2003) have also been linked to saltmarsh die-back. Saltmarsh erosion is particularly worrying because trapped metals can be released back into circulating waters (Boorman, 2000). Toxins can weaken underground plant organs and therefore resistance to tidal scour.

Saltmarsh die-back has been correlated with eutrophication and excessive algal growth in estuaries (Turner et al, 2004). The Eden Estuary was declared a nitrate-vulnerable zone in the early 1990s because of the pollution caused by increasing industrial practices, intensive farming and population expansion (Clelland, 1997). The Eden was given protection from potential eutrophication in 1991 by the EU Nitrates Directive, and The Urban Waste Water Treatment Directive. Diffuse forms of pollution around the Eden therefore have been greatly reduced in recent times, such as the closure of a pig farm and associated slurry effluent, a relatively new sewage treatment works at Guardbridge and elsewhere in the Eden's catchment.

Farming however, still contributes relatively large amounts of nutrients to the system and for the Eden catchment 98kg of nitrogen per hectare per year and 10kg of phosphorus per hectare per year have been recorded (TIDE, 2005). High nitrate levels encourage excessive above

ground biomass during the growth period which leads to reduced carbohydrate reserves below ground necessary for survival during dormancy (Darby & Turner, 2008; Deegan et al, 2012). This has been shown to be particularly the case for reed-beds and may be responsible for the over-toppling that occurs as enhanced growth increases the height of plant stems while the below ground root and rhizome system weakens (Crawford, 1998).

2.7.6 Invasive species

Another concern for the Eden's saltmarshes was the former introduction of the invasive plant species *S. anglica,* which is now considered a pest species (Lacambra *et al.,* 2004). It was planted on the Eden in 1948 using rhizomes and was considered to be forming a stable shoreline (Crawford, 1998; Figure 2.14). At one stage this species occupied an area 20x300m on the edge of the saltmarsh, and started to spread within the lagoon formed by Shelly Spit. It was also introduced to the Solway Firth (Harvey & Allen, 1998) and the Cromarty Firth (Smith, 1982) in order to stabilise shorelines. This hybrid species is a vigorous grower and colonises mudflats lower on the tidal frame than any other native plant species. These abilities led to it being extensively planted as a prelude to land-claim in many English estuaries during the first half of the 20th century also.

Figure 2.14 S. anglica flourishing during the 1970s on the southern shore of the Eden Estuary and prior to its eradication from the 1980s onwards. The narrow mouth of the estuary is visible in the background. (Image courtesy of RMM Crawford.)

However, within the south of the UK its rapid growth and spread earned it a reputation for encroaching on wader and wildfowl feeding grounds and out-competing native pioneer plant communities; as a result, it is systematically removed on an annual basis from most nature reserves (Lacambra et al, 2004). The species tends to form mono-dominant stands to the exclusion of all else, whereas native species not only co-exist alongside other species, but the structural diversity enhances floral and invertebrate diversity of a saltmarsh. It was also suggested that its presence could reduce wader numbers by reducing the area of mudflat available to feeding birds (Raybould, 2005).

This information spread throughout the UK and led to the decision to uproot the plant from the Eden Estuary, even though growth rate calculations by Crawford (personal communication) in the 1980s did not suggest the species dominated. Crawford argued that the species was only present because it had been actively planted, and at its northern limits the cooler temperatures would reduce its fertility and leave it to spread by the slow process of vegetative means. The further natural spread of this species into other Scottish estuaries may be restricted because of the effect that cooler temperatures have on seed set (Crawford, 1998). However, within the Eden Estuary, those colonies on the south shore have colonised the marsh area behind Shelly Spit on the north shore (Figure 2.15; personal observation and R. Strachan, personal communication).

Figure 2.15 Clumps of S. anglica growing in the summer of 2008 on patches of saltmarsh. This area is located behind the shelter of Shelly Spit (RAF Leuchars in the background).

Reciprocal transplant experiments by Gray & Mogg (2001) showed that increased temperatures and atmospheric carbon dioxide favoured the growth and spread of *S. anglica* over *P. maritima*, as the former uses the C4 metabolic pathway in photosynthesis**.** Furthermore, the die-back of *P. maritima* within estuaries has left an environmental niche (and space) on the low to mid marsh zones that *S. anglica* will find relatively easy to occupy. For example, along the south shore of the estuary are a few advanced clumps that have grown into the present saltmarsh (Figure 2.16). These clumps are not currently removed since they are assisting in

preventing edge erosion and the damage that digging out could cause would be detrimental to the ribbon marsh.

Figure 2.16 S. anglica growing on clumps of P. maritima on the southern shore of the Eden Estuary.

Spartina may yet extend its range into more northerly latitudes and has the potential to spread, not just in the Eden Estuary, but further north throughout other Scottish estuaries, such as the Cromarty and the Tay, if current trends in warmer temperatures due to climate change continue.

2.8 Saltmarsh management practices

According to the requirements of the UK Biodiversity Action Plan for coastal saltmarsh, it will be necessary to create some 100 ha of new saltmarsh per year to avoid further net loss (Huggett, 1999; BAP, 2008). It is difficult to see how this will be possible, given that saltmarsh losses continue to exceed gains (Rupp-Armstrong & Nicholls, 2007). One of the aims of managed realignment is to fulfil this goal directly by restoring habitat on previously reclaimed land. Another option is to halt or reverse the process of erosion, and regenerate existing marsh, using marsh creation techniques developed in other parts of the world (King & Lester, 1995).

2.8.1 Managed realignment

Managed realignment restores former saltmarsh on reclaimed land and can also help to meet flood defence requirements (Burd, 1995). The first deliberately breached sites in the UK were on the Essex coast in 1991 (Northy Island) and 1995 (Tollesbury and Orplands) where former reclaimed lands have since reverted to saltmarsh habitat. In Scotland, a seawall protecting low-lying land, a bird sanctuary belonging to the RSPB, at Nigg Bay in the Cromarty Firth was breached in 2003. Although plant colonisation was initially slow, viable saltmarsh communities have since returned (Crowther, 2007). There has also been a proposal in recent years for a managed realignment site in the Firth of Forth, although as is the case with many realignment sites, there has been a great deal of

public opposition and from the farming community as giving land back to the sea can be perceived as a backward step.

The early realignment schemes throughout the UK (Figure 2.17) have demonstrated that saltmarsh can naturally colonise breached sites where seeds and viable fragments are transported by high tides, especially during peak dispersal times in early autumn (Burd, 1995). The speed of recolonisation can depend on a local source of marsh propagules available for dispersal but according to Wolters et al (2005a) there is no direct relationship to long-term successful habitat restoration. However, direct planting of saltmarsh vegetation is recommended if the initial vegetation cover is low and rapid cover is necessary.

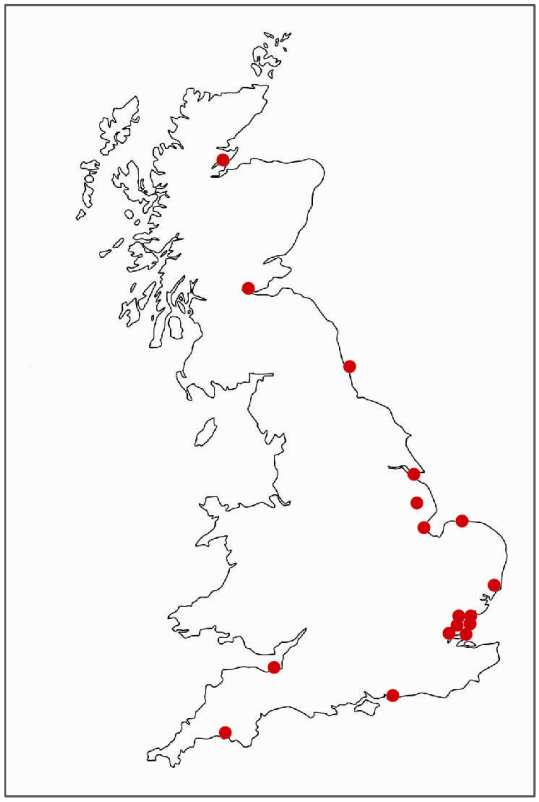


Figure 2.17 Managed realignment sites in Great Britain (redrawn from Garbutt, 2009).

Managed realignment in the Eden has been proposed for some time but given the value of the hinterland it may not be possible nor desired by the public. There is, therefore, a strong need to develop other methods to prevent further deterioration and to enhance the buffering capacity of estuarine fringe saltmarsh.

2.8.2 Direct planting of saltmarsh habitat

The first known written account of land reclamation through the deliberate spreading of rhizomes onto upper mudflats was in the 12th century Anglo-Saxon Chronicles (R. Crawford, personal communication). Later in the 19th century, as mentioned previously, the planting of *S. anglica* as a prelude to land claim was a commonplace practice in the UK. In an analogous period of economic growth, the population expansion in China has led to a deliberate policy of Spartina planting to convert mudflats prior to land claim for agriculture on the Dongtai Peninsula (Chung, 2004).

However, the main body of work in the direct replanting of saltmarsh habitat has come from the many decades of experience by the Corps of Engineers (US Army) where they achieved a high rate of success creating marshes composed of *Spartina alterniflora* Loisel. (the Smooth, or Saltmarsh Cordgrass) in the 1970s especially, on the hurricane-prone east coast of the United States (Knutson et al, 1990). *S. alterniflora* was planted directly to stabilise dredged spoil and create new marshes on both the Eastern and Gulf coasts of the United States

(Lewis, 1982, Craft et al, 1999; Figure 2.18). Also, more recent mitigation legislation in the USA requires that wherever saltmarsh habitat is destroyed, the same amount, if not quality, must be replaced elsewhere.

Figure 2.18 The Pine Knoll Shores, North Carolina, USA. A constructed saltmarsh shortly after planting with S. alterniflora transplants in 1974 (top) and after three years (1977; middle) and 21 years (1995; bottom). Image from Craft et al, 1999.

Saltmarsh restoration through the direct planting of vegetation onto shorelines appears to be an unexplored practice in the UK, which may be due to the conservation issues that arose after the widespread planting of *S. anglica* during the 20th century (Lacambra et al, 2004). However, in other parts of the world the practice of direct planting of saltmarsh vegetation has advanced dramatically to encompass saltmarsh creation in areas of high wave energy. For example, filtration enhancement devices (FEDS) are simple rectangular baffles made from geotextile fabrics, stuffed with biodegradable straw, that are designed to reduce wave action at the seaward edge of marshes and have been employed with good success on the marshes of New England (Burke, 1998).

The Riley-encased methodology is another simple technique whereby a transplant of a mangrove species is planted and encased with a tube, much like the tubes used to protect young tree saplings. This method is used to restore mangroves on high-energy shorelines and revetments where natural recruitment is not possible and conventional planting methods ineffective (Riley, 1999). Re-vegetation trials on hypersaline mudflats in the Arctic saltmarshes of Hudson Bay have also improved plant establishment and growth with fertiliser and peat mulch (Handa, 2000; Handa et al, 2002).

Greenhouse-based studies using tissue-cultured salt marsh species are also attempting to find species with desirable genetic characteristics for use where ideal planting environments are not achievable, with the potential to make previously un-restorable areas more productive (Wang

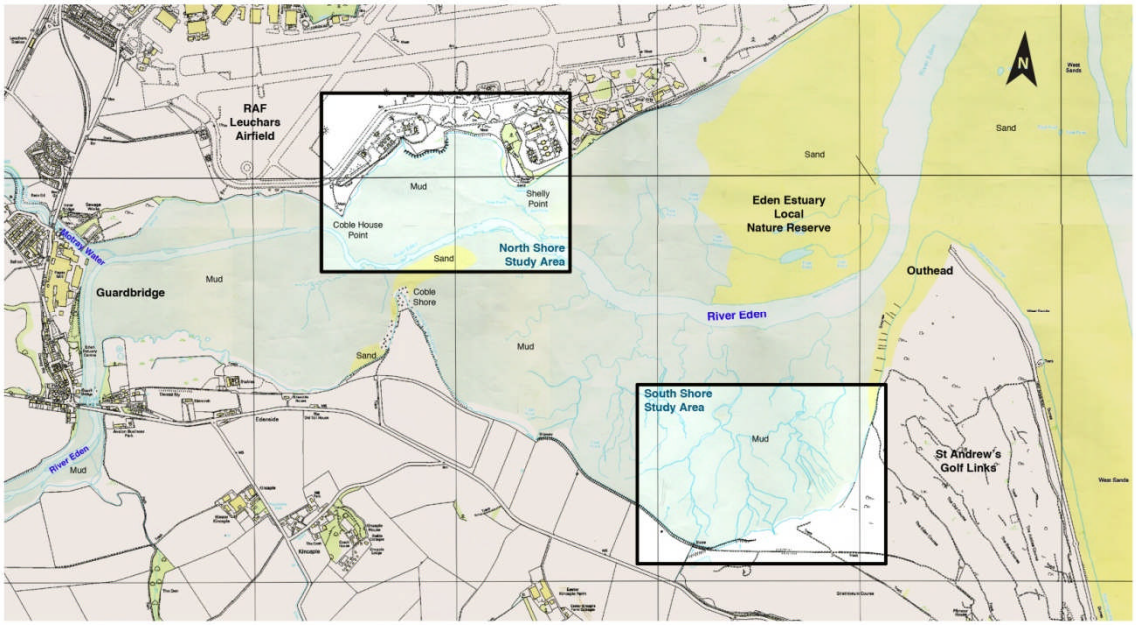
et al, 2007). These methods are as yet unexplored in the UK, but as they are protracted and expensive it was considered necessary to explore initially at least, more basic methods.

CHAPTER 3 METHODS

This chapter provides a more detailed description of the various planting sites on each shore of the Eden Estuary (Figure 3.1). It explains the experimental design and the methods that were employed to establish saltmarsh. It also describes the collection of data for growth rates, environmental conditions and sedimentation studies.

3.1 Study locations

Though the north and south shores of the Eden Estuary share some similarities they are also significantly different in character (see Chapter 2). A more detailed description, sketch maps and photographic records of each planting site therefore follows.



Eden Estuary showing
North & South shore study sites

Figure 3.1 Map of the Eden Estuary showing areas of study on the north and south shores.

3.1.1 Sites on the north shore

Planting sites (1 – 5) on the north shore were located at the base of the cliff that forms the perimeter to land belonging to RAF Leuchars (Figure 3.2). This section of the north shore is relatively sheltered from south-westerly winds by Coble House Point and from easterly winds by Shelly Point. The natural stands of saltmarsh (A and B) represent controls and harvesting sites (also referred to as donor marsh).

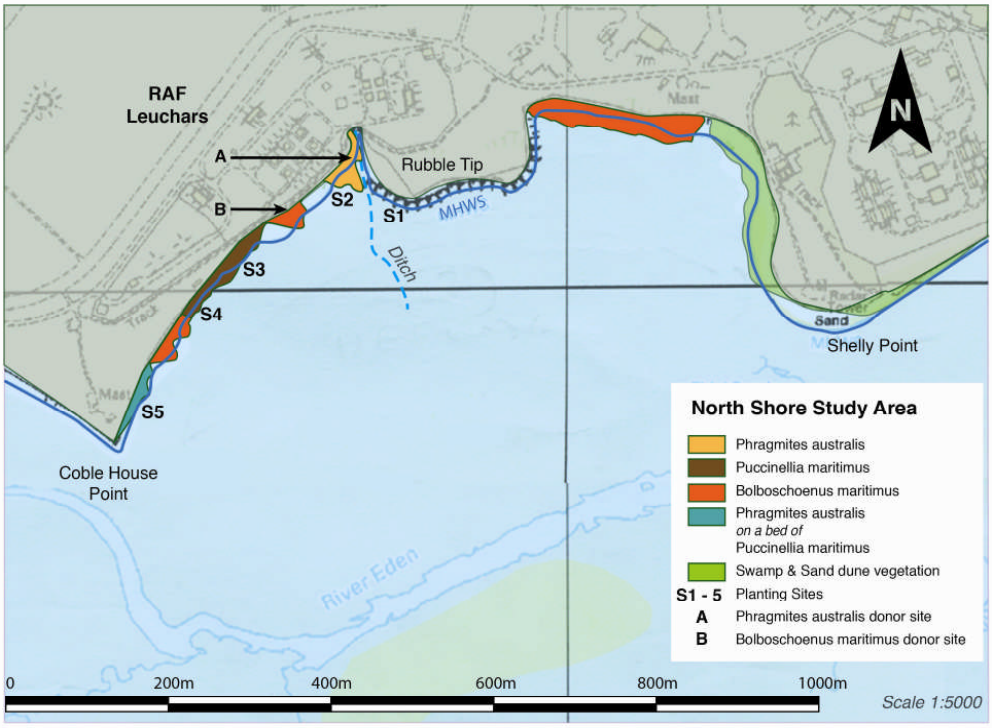


Figure 3.2 The north shore study area showing the locations of planting sites 1 to 5 and natural marsh stands.

Site 1 at the foot of the rubbish tip lies to a freshwater ditch that drains from RAF Leuchars land into the main River Eden channel. The face of the rubbish tip below the high water mark is not colonised by terrestrial vegetation and the mudflats at the foot of the tip are mainly sandy silt sediments with many scoured features and waste such as tyres, metal, crockery and glass were numerous (Figure 3.3). Diatomaceous mats, Eelgrass and indicators of benthic invertebrates such as worm casts were also absent.

Figure 3.3 Planting site 1 at the base of the rubble tip on the north shore of the Eden Estuary.

Site A, a mono-dominant and natural *P. australis* stand (Figure 3.4), was used as a control and as a donor marsh for the harvesting of plant material. It was a relatively sheltered stand located within a small inlet of the coastline. There was a low density of stems at the outer edge of the marsh that increased with an increasingly thick bed of plant detritus into the middle of the marsh. There were no signs of erosion and the stand appeared healthy.

Figure 3.4 Donor reedbed (site A) on the north shore of the estuary below the RAF Leuchars perimeter fence in the background.

Site 2 (Figure 3.5), immediately adjacent to site A, was similar to Site 1 in that the cliff behind the site was only sparsely colonised by vegetation and was covered by concrete waste material. The sediment surface was less eroded and had no waste materials but was devoid of plant life and benthic invertebrates.

Figure 3.5 Planting site 2 on the north shore of the Eden Estuary.

Site B was a natural and small stand of *B. maritimus* (Figure 3.6) and similar to Site A increased in stem density and thickness of the underlying plant detritus towards the base of the cliff. **Sites 3 and 4** (Figure 3.7) were located in front of an eroding *P. maritima* saltmarsh that measured app. 25 m from the seaward edge to the base of the cliff, which was completely colonised by terrestrial vegetation. The seaward edge of the *P. maritima* saltmarsh was severely eroded and had a low and vertical step down onto the mudflats. Planting site 3 was high in water and silt content but devoid of other organisms, whereas the eroded *P. maritima* marsh behind Site 4 was only 15 m wide and the mudflats appeared sandier and drier than all the other sites.

Figure 3.6 Donor brackish swamp marsh of B. maritimus (Site B) on the north shore of the estuary.

Figure 3.7 Planting sites 3 (foreground) and 4 (background) located in front of a narrowing fringe of P. maritima marsh on the north shore. The perimeter fence of RAF Leuchars is visible at the top of the embankment.

Site 5 was adjacent to Coble House Point and on the mudflats directly below another natural reedbed below the RAF Leuchars cliff face. The seaward edge of the reedbed was an eroded scarp approximately half a metre above the mudflats (Figure 3.8). This site was the closest to the Eden river channel and silt and water content was high.

Figure 3.8 Planting site 5 on the mudflats below a reedbed encroaching over former P. maritima marsh.

3.1.2 Sites on the south shore

The south shore planting **Sites 6 – 8** were located on the mudflats directly in front of a natural *P. maritima* marsh in front of the soft embankment that bounds the Eden Golf Course (Figure 3.9). The natural *P. maritima* saltmarsh represented the only saltmarsh community on this part of the coast and in width measured approximately 40 m from below the golf course embankment to the seaward edge of the marsh. The first two to three metres of the marsh seaward edge was highly fragmented and eroded. This natural saltmarsh was used as the source of *P. maritima* propagules for the south shore experiments.

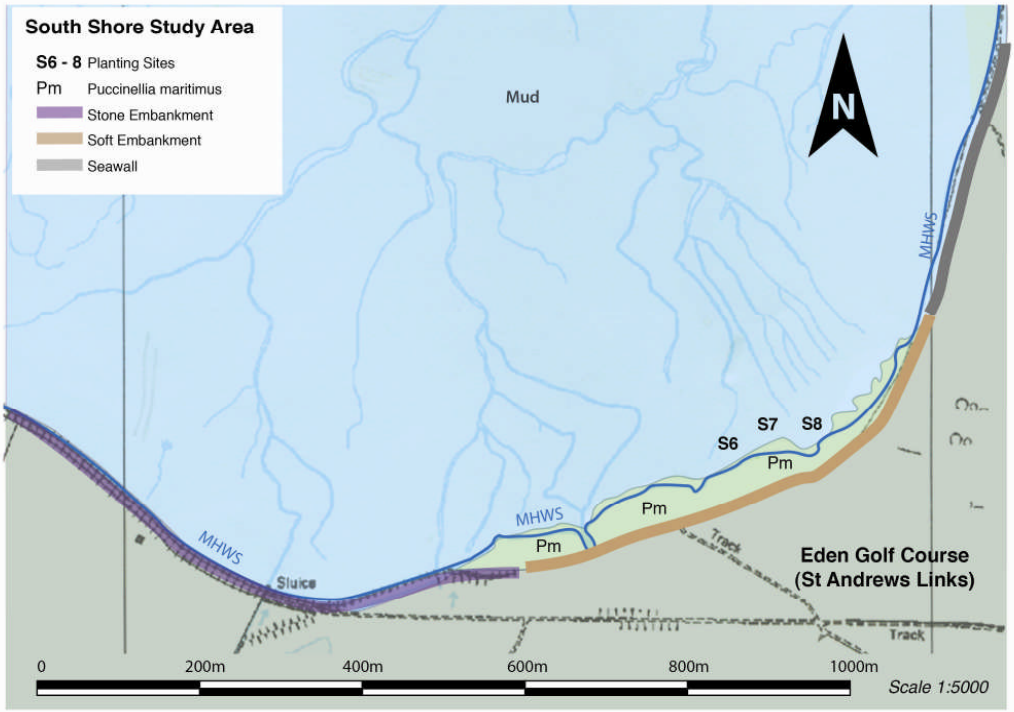


Figure 3.9 The south shore study area showing the locations of planting sites 6 to 8 and the natural P. maritima saltmarsh.

The planting site area, approximately 100 m long, was homogenous in character, the sediment being relatively dry, sandy, scoured and rippled (Figure 3.10). This whole section of coast was relatively exposed to prevailing south westerly and easterly winds compared to the north shore sites.

Figure 3.10 Planting sites 6 – 8 on the south shore of the Eden Estuary in front of a natural P. maritima marsh.

3.2 Vegetation studies

3.2.1 Planting design

The first planting trials conducted were on the north shore of the Eden Estuary in October, 1999 to represent autumn planting and in March, 2000 to represent spring planting. Sites 1 to 5 were subdivided into plots that measured 5 m parallel and 2 m perpendicular to the shore. The plots were marked out with bamboo canes and one planting trial (treatment) was assigned to each plot, and each plot replicated twice (Table 3.1). Two plots at each site were left unplanted to represent mudflat controls. The trials included either the vegetative transplants or the seeds of *P. australis* and *B. maritimus*. These were planted or sown during autumn or spring and at a high or low density.

As a consequence of these trials and at the request of the Eden Estuary Local Nature Reserve Management Committee the study was extended in 2003 to the estuary's south shore. The planting trials continued to test *B. maritimus* but *P. australis* was replaced by *P. maritima*. The trials were refined to include only vegetative transplants at either low or high density. The reasons these refinements and changes were necessary will be discussed in the plant establishment and growth chapter. However the methods for the south shore followed a similar format to the north shore trials. Three sites were selected and each site was divided into plots measuring 5 m by 2 m, marked out by bamboo canes. At each site, plots were assigned treatments and replicated twice. Some plots were left unplanted to represent mudflat

controls.

Monitoring at the planting sites began in the April following planting, i.e., at the start of the growing season. In each plot, the number of the above ground vertical shoots to appear above the sediment surface was recorded at the start of the growing season (April) and the number of these shoots to remain alive at the end of the growing season (September) was determined.

Table 3.1 The planting trials conducted in the Eden Estuary on the north shore in October 1999 and March 2000 and on the south shore in March 2003. See the accompanying text for more detail of the changes to the planting trials between the north and south shores.

Site	**Species**		**Season**		**Propagule**		**Density**	
North shore trials								
1	B. maritimus	P. australis	Autumn	Spring	Sprigs	Seeds	Low	High
2	B. maritimus	P. australis	Autumn	Spring	Sprigs	Seeds	Low	High
3	B. maritimus		Spring		Sprigs	Seeds	Low	High
4	B. maritimus		Spring		Sprigs	Seeds	Low	High
5	B. maritimus	P. australis	Spring		Sprigs	Seeds	Low	High
South shore trials								
6	B. maritimus	P. maritima	Spring		Sprigs		Low	High
7	B. maritimus	P. maritima	Spring		Sprigs		Low	High
8	B. maritimus	P. maritima	Spring		Sprigs		Low	High

3.2.2 Vegetation establishment

3.2.2.1 Seed harvesting, preparation and sowing

Seeds, as close to maturity as possible (Woodhouse, 1974), were harvested from natural marsh stands adjacent to the experimental sites during September 1999. The use of unprocessed seeds is the most rapid method of seeding (Woodhouse, 1974) but seeds can be processed prior to use with procedures such as scarification techniques or imbibing the seeds in a salt solution. For this study, unprocessed, i.e., not treated, seeds were used immediately for the autumn planting session. Seeds to be used the following spring were also unprocessed, but wet-stored in cool, dark conditions, as specified by Lewis (1982).

Seed densities of 100 or 200 seeds per square metre (low and high density, respectively) were sown at the plots within each site. The seeds were first mixed with dry sand to facilitate sowing and the plots were prepared for seeding by tilling the sediment with a garden rake. The sediment was smoothed over after sowing the seeds, so that the seeds were covered to a depth of 2 cm (Lewis, 1982; Clevering, 1995).

3.2.2.2 Vegetative harvesting, preparation and planting

Plugs of soil, measuring approximately 20 x 20 x 30 cm, were dug from natural marsh stands to harvest vegetative transplants. The size of the plug must ensure that all the roots and rhizomes attached to the plant shoots were incorporated. Pieces of rhizome or single stemmed rhizomes

were not recommended as viable (Dunne et al, 1998). As the marsh centres were dense and harder to dig, and have been shown to provide less vigorous plants of smaller and poorer quality (Knutson et al, 1990), most of the transplants were removed from the outer and looser sediment at the seaward edge of the natural marsh stands). Seedlings can also be grown in the controlled environment of a greenhouse (from seeds or transplants) prior to field planting. This practice can be constrained by budget however, even though seedlings can be acclimatised to field conditions.

The plugs were separated into single sprigs, a shoot with a developing bud, a rhizome and associated roots (Figures 3.11 and 3.12), retaining as much soil as possible to minimise root disturbance and transplant shock (a general term to describe the impact of stress and damage that can be caused to vegetation through transplanting). The term sprig is usually used to refer to those species with large underground rhizomes, whereas for common grass species, such as *P. maritima*, the units of transplant are known as 'turfs' (Figure 3.13). However, for simplicity, this study refers to all the vegetative units as sprigs. After separation, the sprigs were immediately planted in the appropriate plots, leaving enough shoot above the ground to allow the developing bud to emerge.

Figure 3.11 The vegetative transplant units (sprigs) of B. maritimus used in the planting trials.

Figure 3.12 The sprigs of P. australis used in the planting trials.

Figure 3.13 The turfs of P. maritima used in the planting trials on the south shore of the estuary.

3.2.3 Plant growth measurements

3.2.3.1 Plant heights

The height of individual plant shoots within each planting site and natural marsh stand were collected at regular intervals throughout the summer months from 2000 – 2008. Ten emerging vertical shoots selected at random were measured to the nearest millimetre with a ruler placed as lightly as possible at the base of the shoot where it emerged from the mudflat and to the tip of the entire shoot. Occasionally the tip of the shoot would have died back (yellowed and dried) but this was disregarded because it is a normal process and common to all the plant stems.

3.2.3.2 Vertical shoot density

A small square quadrat measuring 1 x 1 m was thrown at random into the donor marshes and planted sites in order to count the number of vertical shoots per square metre. This was done three times in each site for replication. However during the first two to three years of growth in the planted sites it was possible to simply count all the stems within the whole plot or site and measure density per square metre by dividing the number of emergent shoots by the size of the plot/site.

3.2.3.3 Lateral expansion

The length and breadth of each marsh was also measured at the end of each growing season from 2000 – 2008. The last living vertical shoot was used as a marker to represent the furthest edge of each planting site and natural marsh. Length and breadth were multiplied to ascertain the areal expansion, or lateral spread, of the planted marshes.

3.2.4 Environmental conditions

3.2.4.1 Shore profiles

Shore profile data were collected using a real time kinematic GPS in 2000 on the north shore and 2003 on the south shore. The GPS base was located at the Irvine Building in the University of St Andrews. The known co-ordinates meant that the position and height above sea level was computed to millimetre accuracy. A transect line from the marsh centre to

20 metres beyond the outer edge of the marsh onto the mudflats was set up for each site and data were plotted as metres above sea level, Ordnance Datum (OD). If necessary, a conversion factor of 2.9 (datum for Leith, Edinburgh) can be added to Ordnance Datum to convert the value to Chart Datum (CD), the height in metres above the lowest astronomical tide.

3.2.4.2 Soil salinity

Surface water salinity readings were measured in the field using a hand-held refractometer. Most measurements were collected within one to two hours after high tide, which meant there was enough surface water available at each site. Measurements were collected prior to planting and in subsequent growth years. Surface water was collected with a pipette and dropped onto the lens of the refractometer with care taken to ensure that the water was free from sediment. An average of six readings was taken per site and the data were expressed in Practical Salinity Units (psu).

3.2.4.3 Water content

Sediment samples were collected to record the average moisture content. A plastic coring tube that measured 20 cm long by 10 cm in diameter was inserted into the sediment at each site. The core containing the sediment was then carefully removed and the contents shaken into a labelled bag

and sealed to retain moisture. The samples were placed in a pre-weighed foil dish, and both the sample and the foil dish weighed prior to placing in an 80°C oven for 48h. The samples were removed and re-weighed to assess moisture content and the data were expressed as the percentage of water to fresh soil weight (wet sediment weight – dry sediment weight / wet sediment weight x 100).

3.2.5 Vegetation study data analysis

3.2.5.1 Vertical shoot emergence between species

For this analysis, the data from each treatment and at each site were pooled to compare the number of sprigs or seeds to develop an above ground vertical shoot, at the start and at the end of the growing season (April and September respectively). The percentage success of each species was tested using a one-way analysis of variance (ANOVA) and post-hoc analysis of means was compared using a Tukey test (α= 0.05). The assumption of homogeneity of variance and normality was met for most of the data and transformation was considered unnecessary.

3.2.5.2 Comparison of planting methods

Comparisons between the different planting methods were made for each species, using one-way ANOVA. No data transformation was necessary to meet the assumptions of least squares analysis. Comparison between the means was performed using a Tukey test

(α= 0.05).

3.2.5.3 Between site comparisons of salinity and water content

One way ANOVAs were used to compare soil salinity and water content between the sites. No data transformation was necessary to meet the assumptions of least squares analysis. Comparison between the means was performed using a Tukey test (α= 0.05).

3.2.5.4 Growth rate comparisons between sites

Normality and homogeneity of variance were adequate for most of the plant growth data and so no transformation was considered necessary. A two-way ANOVA compared the stem height and stem density between sites and between years, and their interaction. Comparison between the means was performed using a Tukey test (α= 0.05).

3.3 Sediment study

3.3.1 Experimental design

The sediment study commenced in July 2004 and investigated the patterns in sediment deposition and accretion at four sites on the north shore and four sites on the south shore of the Eden Estuary. These included an upper mudflat and a natural *P. maritima* marsh on each shore, one year old planting sites of high density *B. maritimus* and *P. maritima* on the south

shore, and a four year old planting site of high density *B. maritimus* on the north shore (Table 3.2).

Table 3.2 The sites selected for sediment studies conducted in July 2004 on the north and south shores in the Eden Estuary.

North shore	South shore
Upper mudflat	Upper mudflat
Natural *P. maritima*	Natural *P. maritima*
Natural *B. maritimus*	*B. maritimus* transplants (1 yr)
B. maritimus transplants (4 yrs)	*P. maritima* transplants (1 yr)

Each site was divided into two replicate plots measuring approximately 10 m parallel by 5 m perpendicular to the shore. The original transplant plots within site 3, the largest of the planting sites on the north shore, had expanded and merged. This site was divided into two plots and represented a four year old *B. maritimus* stand for the sediment study by which time stem density was approximately 150 m^{-2}. The transplant plots within site 8 on the south shore, i.e., one-year-old *B. maritimus* and one-year-old *P. maritima*, formed the replicate plots for the sediment study. At the time of the study, stem density within the one-year-old *B. maritima* plot was only 30 m^{-2}, while growth in the *P. maritimus* plot was extremely limited, each transplant unit retaining its original size of 5 × 5 cm. Natural marsh and upper mudflat reference sites were also

divided into plots for replication and were located immediately behind and adjacent to the planting sites, respectively.

3.3.2 Short-term sediment deposition

Sediment traps were deployed to measure short-term sediment deposition following the protocol first established by Reed (1989). Sediment deposited each day was collected on pre-weighed filter papers (9 cm Whatman GF/C) placed on the sediment surface. The filter papers were secured to plastic discs to prevent adhesion to the sediment surface. Each day (i.e. after two high tides) the discs were lifted to allow the filter paper to be collected and replaced with a fresh paper. The discs and clean filter papers were then returned to the sediment surface in a new and undisturbed area for the following day.

Five filter papers were laid in each plot. After collection, the filter papers were oven-dried overnight at 40 °C and reweighed to 0.1 mg. Brown (1998) corrected for salt weight and the area of the filter paper covered by the paperclips, but on investigation the present study showed only a negligible effect, and corrections were considered unnecessary. Wave activity damaged some of the filter papers and the relative percentage loss was deducted from the overall surface area of the filter papers (636 cm^2).

Sediment deposition rates were measured over the course of one month, between the 21 July and 22 August, and included two neap and spring tidal cycles. However, data collected during 9 days of the study were removed from the analysis, since prolonged rainfall had

caused the filter papers to disintegrate. Sediment deposition was expressed as mg dry weight sediment per unit area (mg 100 cm^{-2}).

3.3.3 Sediment surface level

Metal marker poles were used to measure relative vertical changes in marsh and mudflat elevation. At each plot in each site, a pair of 2 m poles was driven into the sediment 1 m apart so that they extended 1 m above the sediment surface. Each data point was the average of three readings taken as the distance from a builders' level placed on top of the two poles to the bed surface. The zero bed level was established in July 2004, and measurements were taken after the last spring tide in each subsequent month until July 2005. Data were expressed as mm of wet and unconsolidated sediment per month or per annum, and as positive (accretion) or negative (erosion) in relation to the zero bed level.

3.3.4 Sediment study data analysis

3.3.4.1 Sediment deposition

The assumption of homogeneity of variance and normality was met for all but the natural *B. maritimus* site and data transformation was therefore considered unnecessary. The data are first presented as the mean quantity of sediment per filter paper over the whole study period: i.e. the sum total deposited over 23 days divided by the total number of filters laid in each site. There were no significant differences between the plots

within each site and therefore these data were pooled to compare the sites using one-way analysis of variance (ANOVA). Post-hoc analysis of means was compared using a Tukey test ($\alpha = 0.05$).

3.3.4.2 Sediment surface level

No transformation was necessary to meet the assumptions of least squares analysis and the data from the plots within each site were pooled. A two-way ANOVA compared the sediment surface level changes between sites and between months, and their interaction. Comparison between the means was performed using a Tukey test ($\alpha = 0.05$).

3.3.4.3 Tidal height

The temporal variation in the data set appeared to show little pattern (data not presented). In a preliminary study, however, a subset of the data was used to analyse the effect of changing water levels on sediment deposition between sites only on those days when wind speeds were low (below 3 m s^{-1}) and from a south-easterly direction. A Pearson product–moment correlation was used to compare the results for 4 days of neap tides (between 4.2 and 4.6 m), 3 days of average tides (between 4.6 and 5.0 m) and 4 days of spring tides (greater than 5.0 m). These tide heights were used as proxy data and were taken from the tidal height chart for the gauge at the Port of Dundee, in the neighbouring Tay Estuary.

3.3.4.4 Wind direction

The effect of winds from the south-east or the south-west, the two most prevalent wind directions during the month-long study, on sediment deposition was investigated. Three days of each wind direction were available for comparison when tide height was between 4.6 and 5.0 m (above chart datum) and wind speeds were low (below 3 m s^{-1}). A two-way ANOVA was used to compare variation in sediment deposition between sites, wind direction and their interaction. Post-hoc analysis of means was conducted using a Tukey test (α = 0.05). Wind directions were recorded at RAF Leuchars on the northern shore of the estuary (Meteorological Office).

CHAPTER 4 VEGETATION ESTABLISHMENT

4.1 Introduction

Saltmarsh restoration guidelines, especially those from North America, may not be applicable to Scotland. This study evaluated the restoration of saltmarsh communities into degraded sections of shoreline in the Eden Estuary through transplantation trials of *Phragmites australis, Bolboschoenus maritimus* and *Puccinellia maritima*. These three species are all native to the UK and are present in the estuary, though the latter species is severely eroded and dying back, similar to other saltmarsh in the UK. The effect of planting month, planting density and transplant type was compared and the growth of the plants within any successful trial plots was evaluated. Shoreline profiles, salinity and water content at the various sites were also measured to determine the conditions necessary for success.

Saltmarsh restoration through the direct transplantation of vegetation was suggested by King and Lester (1995) as a way to replace, or aid the recovery of, saltmarsh undergoing the process of die-back. Currently, saltmarsh restoration in the UK tends to apply to the natural recolonisation of saltmarsh vegetation during managed realignment schemes; although the process of colonisation has been shown to be enhanced by direct planting methods. A wealth of information on the procedures necessary to ensure the successful establishment of many

saltmarsh species is now available (Brooke et al, 1999; Zedler, 2001). Managed realignment sites are relatively sheltered because the tidal waters enter and exit the site through only one or two breaches in the seawall or embankment. The findings within publications such as 'The Restoration of Vegetation on Saltmarshes' by the Environment Agency may not be applicable to direct planting on degraded shorelines.

Information on the direct planting of saltmarsh vegetation is not common in the literature and it has been considered impractical for saltmarshes on the sinking coastline of the south east of England (Boorman, 2003). Precedents and methods have been set in other parts of the world, however. For example, researchers and coastal managers in North America have experience spanning decades and include saltmarsh creation in areas of mass saltmarsh die-back, such as the Louisianan and Georgian coastlines. Spoil from dredging to keep open shipping routes in Chesapeake Bay was placed, graded for appropriate slope and height above sea level and directly planted with *Spartina alterniflora*. Saltmarsh vegetation was also planted to recreate habitat for mitigation purposes on the hurricane-prone eastern seaboard in several New England states. Recreating wetlands and saltmarsh in the USA is now a commercial enterprise (Knutson et al, 1990; Dunne et al, 1998; Zedler, 2001).

Similar to many estuaries around the UK, the non-native and invasive *S. anglica* was planted directly onto the upper mudflats of the Eden Estuary in 1948. The species flourished and helped to increase shoreline stabilisation. However, few records exist of the methods that

were employed or of the conditions that led to success. Concerns during the 1980s and 1990s of the invasive nature of non-native species finally led to a concerted effort to eradicate the species from the Eden Estuary's shoreline. Since then loss of the native saltmarsh communities has continued and with increasing pressure from the effects of sea level rise there is an urgent need to revisit direct planting using more appropriate native species as a form of erosion control.

The Eden Estuary provided the opportunity to investigate the restoring saltmarsh habitat and direct planting on degraded shorelines as an erosion control method because of the presence of small and relatively healthy pockets of brackish swamp marsh such as *B. maritimus* and *P. australis*. These two latter species are found in most UK estuaries (Burd, 1989) but tend to be associated with high marsh transitional zones, i.e., where freshwater outflow ameliorates the salinity of the flooding tide and wave energy is restricted. *B. maritimus* in particular is one of the most ubiquitous species on temperate marshes in the northern hemisphere and has a wide environmental tolerance (Broome *et al,* 1995; Kantrud, 1996; Yang, 1999). This implied that they had the potential to be used in direct planting trials as studies from the United States have shown that success is more likely when the planted species is from the same or similar environment (Lewis, 1982).

In the first instance, whether the saltmarsh communities in the Eden share a similar environment can be ascertained by comparing shoreline profiles. This provides information such as marsh elevation (OD) in

relation to tidal height and therefore the frequency of flooding and sediment conditions such as salinity and water content. The methods necessary to establish saltmarsh vegetation also needed to be determined. These included propagule type, planting month and planting density. Seeds or vegetative transplants (sprigs) for example, are both considered potential propagules. Seeding tends to be more cost-effective and less labour intensive than sprig planting (Lewis, 1982) but sprigs can be used over a greater variety of conditions and over wider tidal ranges (Dunne et al, 1998). For example, in both *P. australis* (Wijte & Gallagher, 1996) and *B. maritimus* (Lieffers & Shay, 1982) adult plants and vegetative propagules are more resilient to extreme environmental conditions than young plants and seedlings.

Establishment success may also depend on the planting month. Springtime planting gives the plants a chance to take root before winter storms, but seed release occurs in autumn and over-wintering in the sediment may be a necessary precursor to germination. Planting density can be particularly important, depending on site conditions, the availability of propagules (or size of donor marsh) in addition to whether or not rapid plant cover is required. Lewis (1982) recommended 100 seeds per square metre as suitable to overcome natural seedling mortality, but Hughes (1999), referring to managed realignment sites, showed that benthic invertebrates can both damage seedlings and devour large quantities of seeds so a higher density may be necessary. Sprig planting density can

range from 1 per square metre (Lewis, 1982) to 20 per square metre (Clevering, 1997), whilst Garbisch (1994) recommended a high planting density if shoreline stabilisation was the desired goal. This study used these different planting methods to investigate the viability of saltmarsh restoration.

4.2 Results

In the first place the data from the various planting trials on the north shore of the estuary (Oct 1999 and March 2000) were pooled in order to compare the different methods used to establish the marshes. Data from the south shore planting trials (March 2003) are presented separately. Success or failure of the planting sites was assessed and related to individual site conditions, such as shore profiles and the soil water and salinity content. Subsequent results compared growth and expansion of the successful sites with natural saltmarsh.

4.2.1 Comparisons of species and planting methods

In the following comparisons all the data were combined from each planting site in order to present the percentage of the mean number of vertical shoots to have emerged for each species and each treatment as a proportion of the number of sprigs or seeds planted or sown.

4.2.1.1 The effect of propagule type

Seed germination did not occur for either *B. maritimus* or *P. australis*, whether the seeds were sown in autumn or spring, or at high or low density. Further analysis of seed metabolic activity showed that the seeds of *B. maritimus* or *P. australis* were not viable (data not shown). Subsequent comparisons are of vegetative propagules only.

4.2.1.2 Vertical shoot formation in B. maritimus and P. australis

Significant differences between *B. maritimus* and *P. australis* in the mean total number of shoots to have initially emerged were apparent ($F_{1,\ 29}$ = 4.47; P = 0.02). *B. maritimus* had a very high success rate (66%) compared to *P. australis* (32%) (Figure 4.1). By the end of the growing season in September there was a clear reduction in the number of shoots to remain alive in both *B. maritimus* and *P. australis*, 21% and 0% respectively ($F_{1,\ 29}$ = 3.16; P = 0.02), though *P. australis* had declined considerably by July, while those shoots of *B. maritimus* that had formed still thrived.

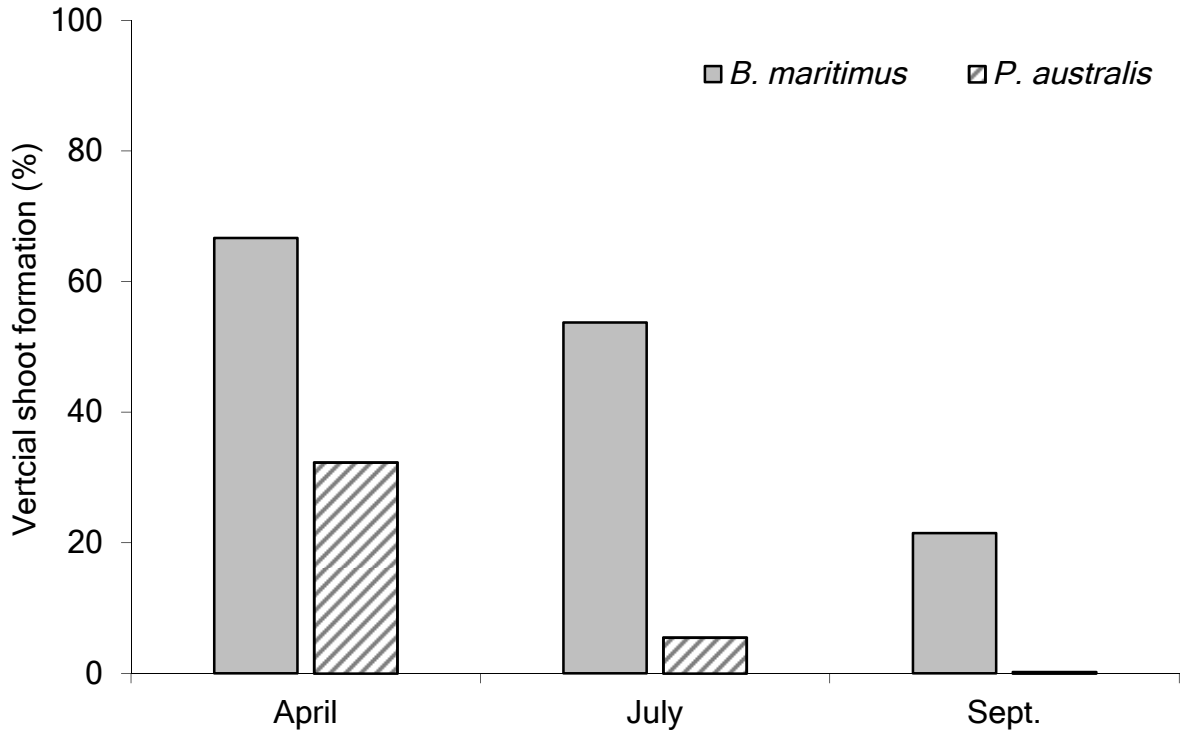


Figure 4.1 The number of vertical shoots (%) to emerge and remain alive in B. maritimus and P. australis (n = 640 and 310, respectively) in the first summer of growth (April to September 2000).

4.2.1.3 The effect of planting season

Comparisons were made between those sprigs planted in autumn (October 1999) and those planted in spring (March 2000) for *B. maritimus* and *P. australis* only (Figure 4.2). A significant difference was found only for *B. maritimus* sprigs ($F_{1,15}$ = 3.84; P = 0.05) with 36% sprig success for autumn planting compared to 74% for spring planting. A significant difference was not found for *P. australis* however (25% and 37%, respectively) ($F_{1,15}$ = 0.63; P = 0.44). By the end of the growing season all *P. australis* sprigs had died, but there was a significant difference between autumn (25%) and spring (86%)

planting for *B. maritimus* sprigs ($F_{1,15}$ = 5.02; P = 0.01).

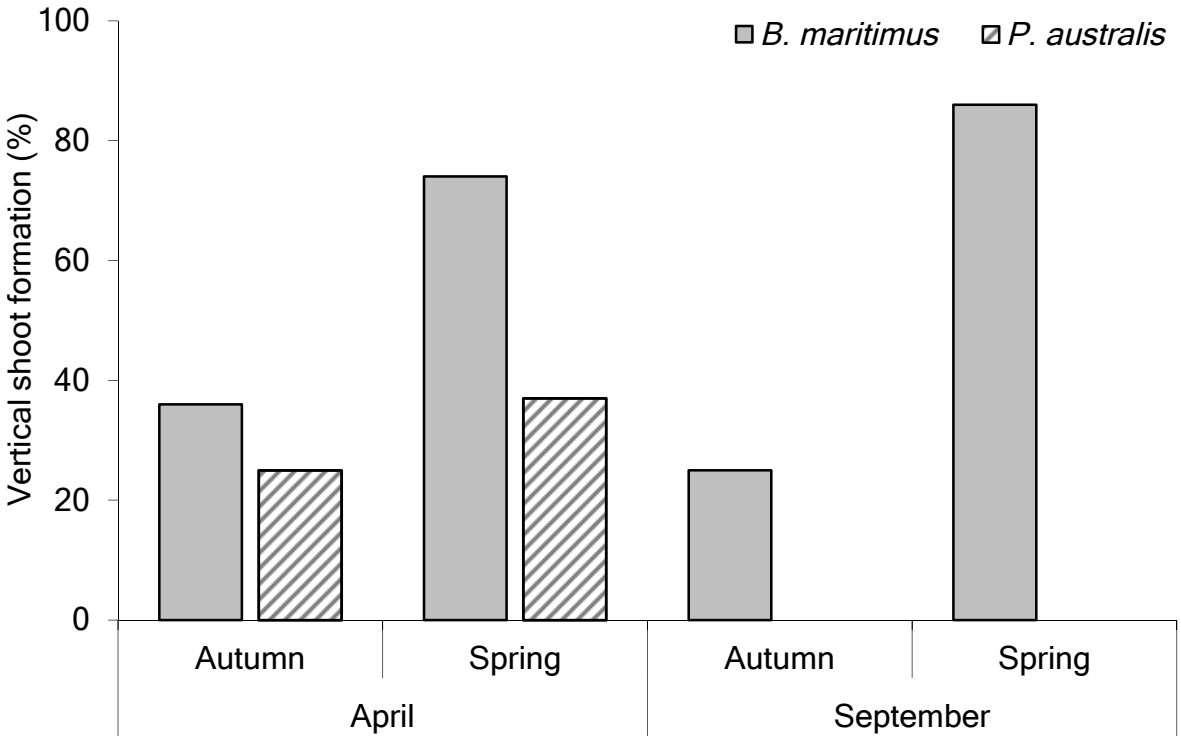


Figure 4.2 Vertical shoot formation (%) in B. maritimus and P. australis for sprigs planted in either autumn or spring (n = 200 for each species in each planting season). Data were collected in April and September 2000.

4.2.1.4 The effect of planting density

Initially, there were no significant differences between high and low density planting for *B. maritimus* having 69% and 65% ($F_{1,\ 15}$ = 1.33; P = 0.44) or *P. australis* with 34% and 33% ($F_{1,\ 11}$ = 1.82; P = 0.214). Clear differences between the two planting densities had emerged at the end of the growing season for *B. maritimus* ($F_{1,\ 15}$ = 19.54; P = 0.05) with 57% of the high density sprigs still alive compared to only 10% of the low

density sprigs (Figure 4.3).

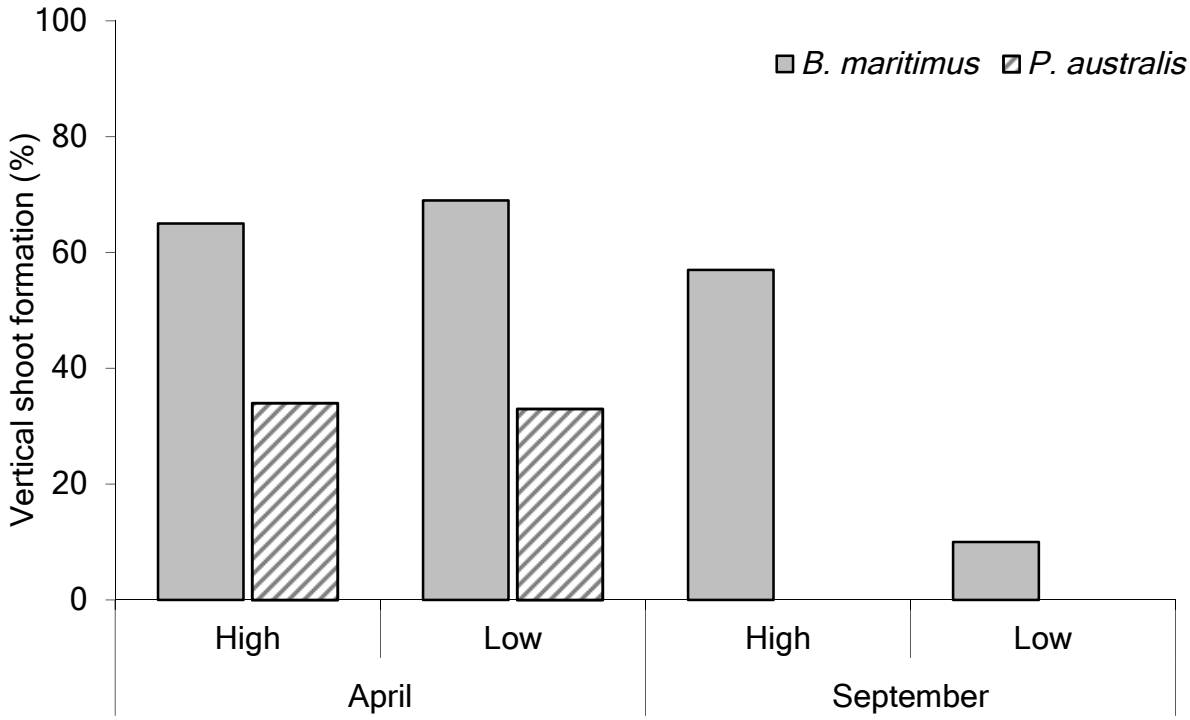


Figure 4.3 The effect of high and low density planting on shoot emergence (%) in B. maritimus and P. australis (n = 800 and 80, 600 and 60, respectively). Low density = 1 sprig per m^2 and high density = 10 sprigs per m^2.

4.2.1.5 Vertical shoot formation in B. maritimus and P. maritima

B. maritimus outperformed *P. maritima* during initial shoot emergence in the south shore planting trials (April, 2003) with 71% compared to 15%, respectively (Figure 4.4) and was significantly different ($F_{1,\ 29}$ = 4.47; P = 0.02). However, by September 2003 there was not a significant difference between *B. maritimus* and *P. maritima* ($F_{1,\ 29}$ = 1.16; P = 0.33) in the number of shoots to remain alive though more shoots had died back in *B. maritimus* (71% - 15%) compared to *P.*

maritima (15% - 5%).

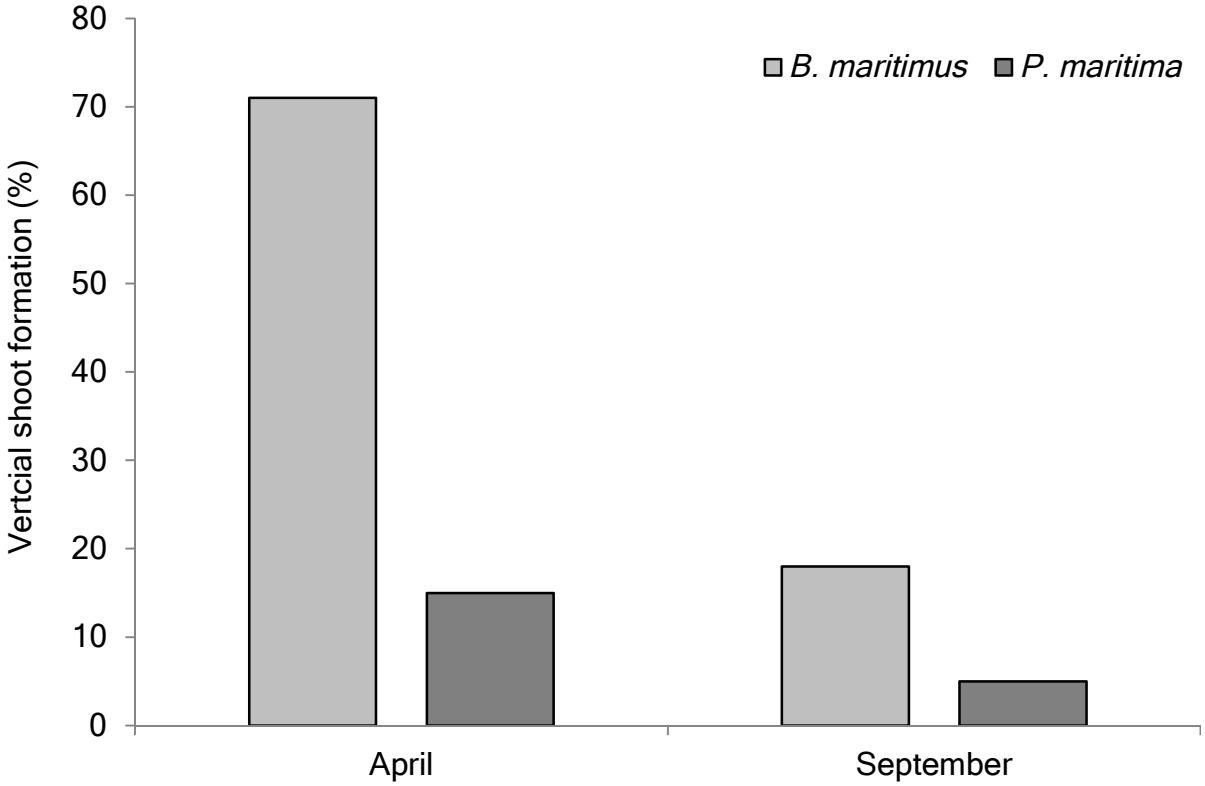


Figure 4.4 Vertical shoot formation (%) in B. maritimus and P. maritima for sprigs planted in March 2003 (n = 200 for each species). Data for comparison were collected in April and September 2003.

4.2.1.6 The effect of planting density

Initially, there were no significant differences between high and low density planting for either *B. maritimus* at 69% and 62% ($F_{1, 15}$ = 1.33; P = 0.44) or *P. maritima* 12% and 10% ($F_{1, 1}$ = 1.19; P = 0.44), respectively. At the end of the growing season there was no significant difference for high and low density planting of *P. maritima* ($F_{1, 1}$ = 1.19; P = 0.44) though 5% of the sprigs were still alive in the high density plot, whereas none had survived in the low density plot. Clear differences

between the two planting densities had emerged at the end of the growing season for *B. maritimus* only ($F_{1,\ 15}$ = 9.54; P = 0.05) with 35% of the high density sprigs still alive compared to only 5% of the low density sprigs (Figure 4.5).

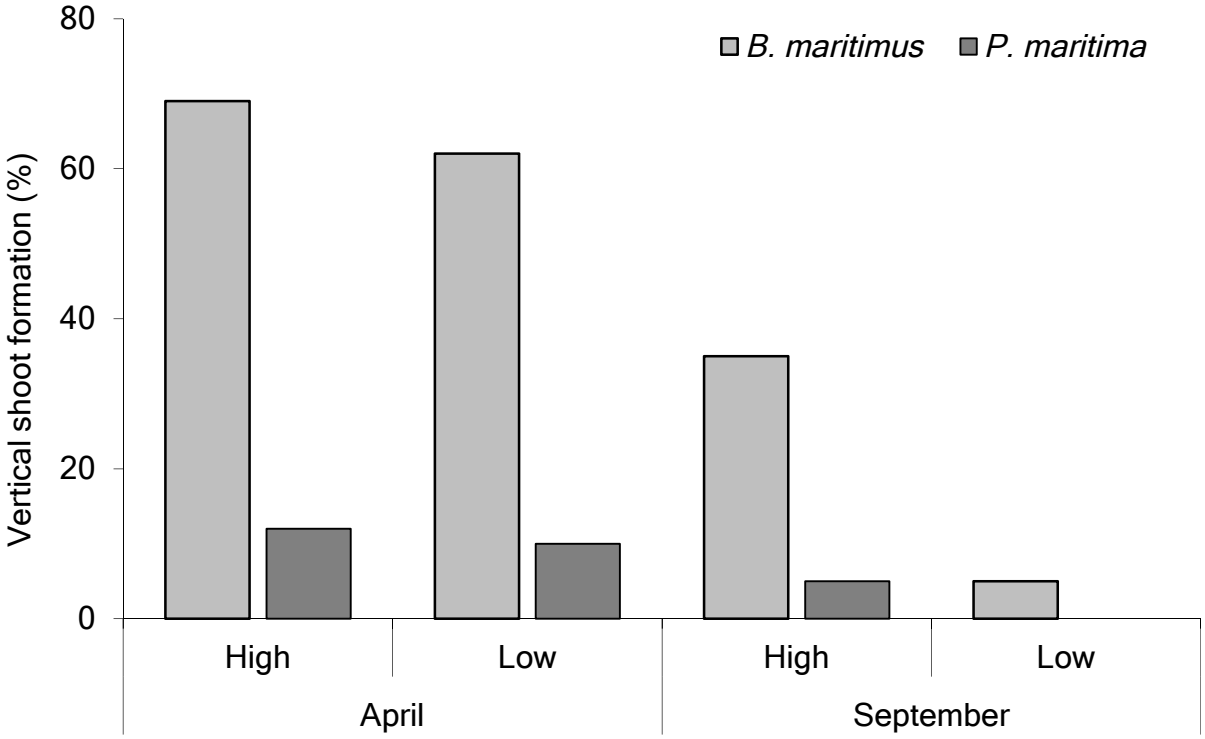


Figure 4.5 The effect of high and low density planting on shoot emergence (%) in B. maritimus and P. maritima (n = 800 and 80, 600 and 60, respectively). Low density = 1 sprig per m^2 and high density = 10 sprigs per m^2

4.2.2 Environmental conditions

4.2.2.1 Shore profiles

The natural marshes and planting sites on the north shore were between 0.6 and 1.3 m above sea level (OD) and marsh elevation therefore was

much lower than the corresponding south shore sites which were between 1.7 and 3.2 m above sea level (OD). Natural marshes A and B (*P. australis* and *B. maritimus*; Figure 4.6) commenced at the base of the cliff just below 1 m OD (zero metres seaward) and gently declined to the last living plant at 0.6 m (10 metres seaward).

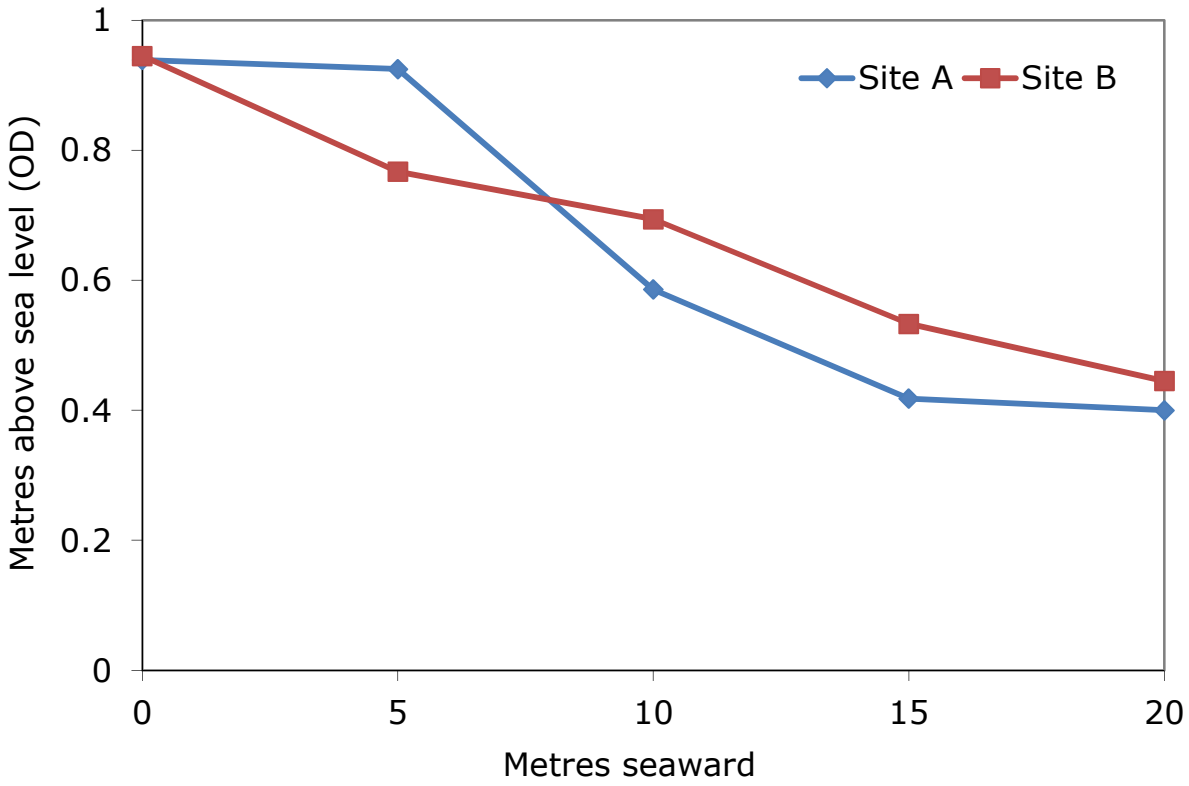


Figure 4.6 Shore profiles for the natural marsh sites A and B on the north shore with height in metres above mean sea level (OD).

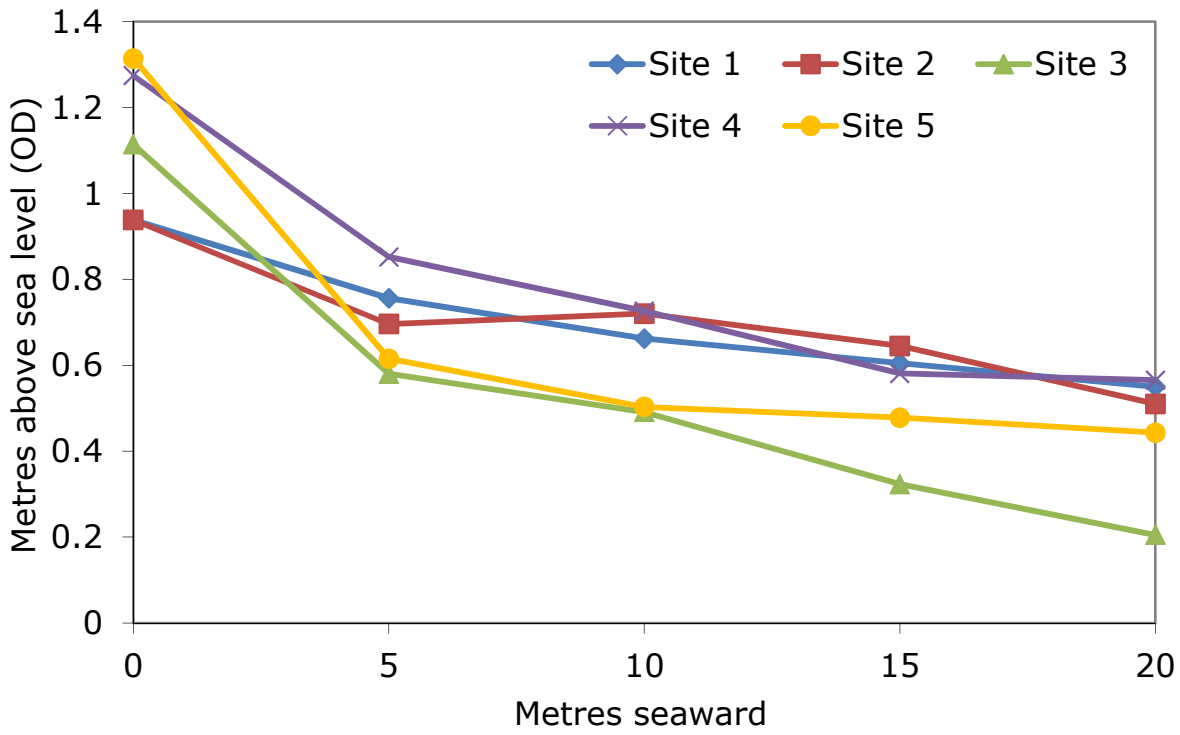


Figure 4.7 Shore profiles for north shore planting sites 1 – 5, with height in metres above mean sea level (OD).

Planting sites 1 and 2 shared similar profiles to the adjacent natural marshes, starting below 1 m OD at the base of the cliff at zero metres seaward. Planting took place within these sites around 5 – 10 metres seaward at 0.7 m OD (Figure 4.7). The shore profiles for Sites 3, 4 and 5 commenced either on the top of eroded *P. maritima* marsh (sites 3 and 4) or a reedbed stand (site 5) and therefore have a slightly higher elevation of 1.1 m – 1.3 m OD at zero metres seaward. However, the actual planting sites were all on the mudflats immediately below the natural marshes and lay between 0.5 m to 0.7 m OD (5 – 10 metres seaward).

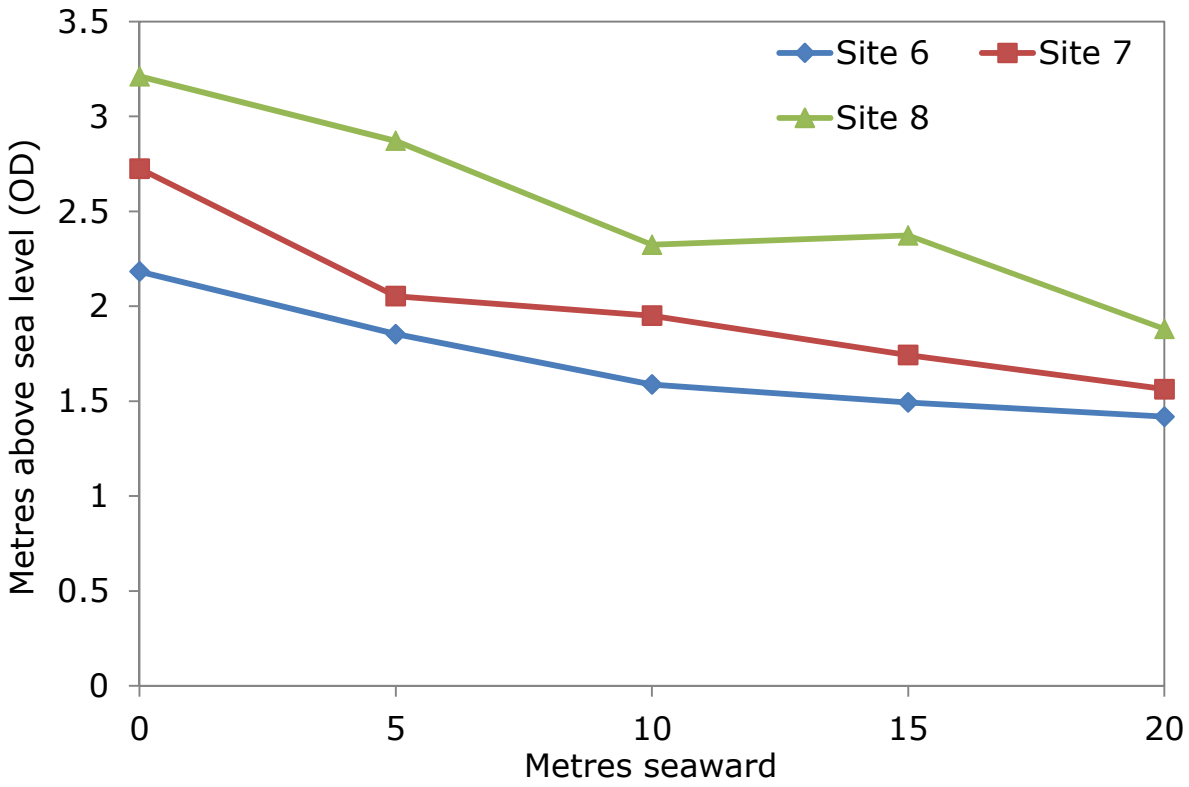


Figure 4.8 Shore profiles for the south shore planting sites 6 – 8, showing height in metres above mean sea level (OD).

The south shore transplant sites 6 to 8 were each located on the mudflats below the seaward edge of an eroded *P. maritimus* marsh. The *P. maritima* marsh edge was between 2 m – 3.2 m above sea level OD (0 – 5 metres seaward) and the tranplant sites 1.6 m – 2. 7 m above sea level OD (5 – 10 metres seaward; Figure 4.8). This means there was almost a metre increase in the elevation above sea level from site 6 to site 8, at 2.2 to 3.2 m OD, respectively.

4.2.2.2 Sediment salinity

The mean and range of salinity in the sediment varied between the sites (Table 4.1) but all the natural marshes and planting sites were brackish. Significant differences were found between the sites on the north shore ($F_{8, 81}$ = 32.57; P < 0.001) but not on the south shore ($F_{2, 27}$ = 2.57; P < 0.2). The two natural marshes A and B (*P. australis* and *B. maritimus*) were similarly low in salinity between 10 – 12 psu and similar to planting Site 3.

However, planting sites 1, 2 and 4 were higher in salinity content (between 16 – 22 psu) and shared similar values to the natural *P. maritima* marsh on the south shore (18 – 20 psu). Average salinity for planting Sites 6 – 8 were between 20 – 28 psu though site 8 had the greatest range of 15 – 27 psu.

Table 4.1 Means, standard errors (SE) and the range of sediment salinity (psu) for all sites.

SITES	Mean	SE	Range
NORTH SHORE			
1	19.2	2.20	18 – 22
2	19.4	3.24	19 - 20
3	10.5	1.40	10 - 12
4	18.0	2.42	18 - 20
5	16.2	4.87	16 - 17
P. australis (A)	10.4	2.50	10 - 12
B. maritimus (B)	12.3	2.54	10 - 12
SOUTH SHORE			
6	24.2	3.40	22 - 27
7	28.86	3.45	23 - 30
8	20.63	2.10	15 - 27
P. maritima	19.76	4.85	18 - 20

4.2.2.3 Sediment water content

Sediment water content between the north sites was significantly different ($F_{8, 81}$ = 17.88; P < 0.001) though the average water content for planting Sites 1 and 4 were between 15 and 17 % and much lower than all the other sites which were between 20 – 25% (Table 4.2). Sites on the south shore were also significantly different ($F_{3, 37}$ = 2.57; P < 0.05) which was due to the lower water content of 16% for the natural *P. maritima* marsh compared to 20 – 23% for sites 6 – 8.

Table 4.2 Means, standard errors (SE) and the range of sediment water content (%) for all sites.

SITES	Mean	SE	Range
NORTH SHORE			
1	16.59	0.17	14.12 – 17.56
2	22.30	0.44	21.33 – 24.45
3	24.62	1.27	23.42 – 25.59
4	15.54	0.33	12.50 – 16.88
5	20.77	0.51	19.30 – 22.49
P. australis (A)	24.14	0.60	23.63 – 25.60
B. maritimus (B)	23.96	0.61	22.48 – 24.58
SOUTH SHORE			
6	23.32	0.34	21.67 – 24.21
7	22.50	1.76	20.53 – 24.26
8	20.63	0.71	21.35 – 24.14
P. maritima	15.85	2.91	13.36 – 16.98

4.2.3 Comparisons of growth and lateral expansion

These results focus on the growth of the successful planting sites of *B. maritimus* and the natural *B. maritimus* marsh (Site B), measured by stem height and density within the marsh and the lateral expansion of the marsh. The results for the north (Sites 1 – 3) and south (Site 8) shores are presented separately. Stem height and density of the seaward edge and central parts of the natural marsh were very different and therefore shown separately.

4.2.3.1 Plant stem height and density on the north shore

Significant differences between the plant stem heights on the north shore sites (Figure 4.9) were due to the inclusion of the stem heights growing within the centre of natural marsh Site B (*B. maritimus*; $F_{4,\ 315} = 442.64$; $P < 0.001$). However, there were also significant differences between years ($F_{6,\ 315} = 90.59$; $P < 0.001$) and the interaction between sites and years ($F_{24,\ 315} = 18.74$; $P < 0.001$). Stem height steadily increased every year in the planted *B. maritimus* sites, apart from those plants in Site 1 that failed to grow beyond a few centimetres during 2001. Stem height in the plants at Site 3 remained below 60 cm between 2003 and 2005 but rapidly increased in the subsequent year to 90 cm. Stem heights within plants at the edge or centre of the natural *B. maritimus* were similar throughout the entire study period, with those at the edge between 150 – 220 cm tall and those in the centre between 400 – 580 cm tall.

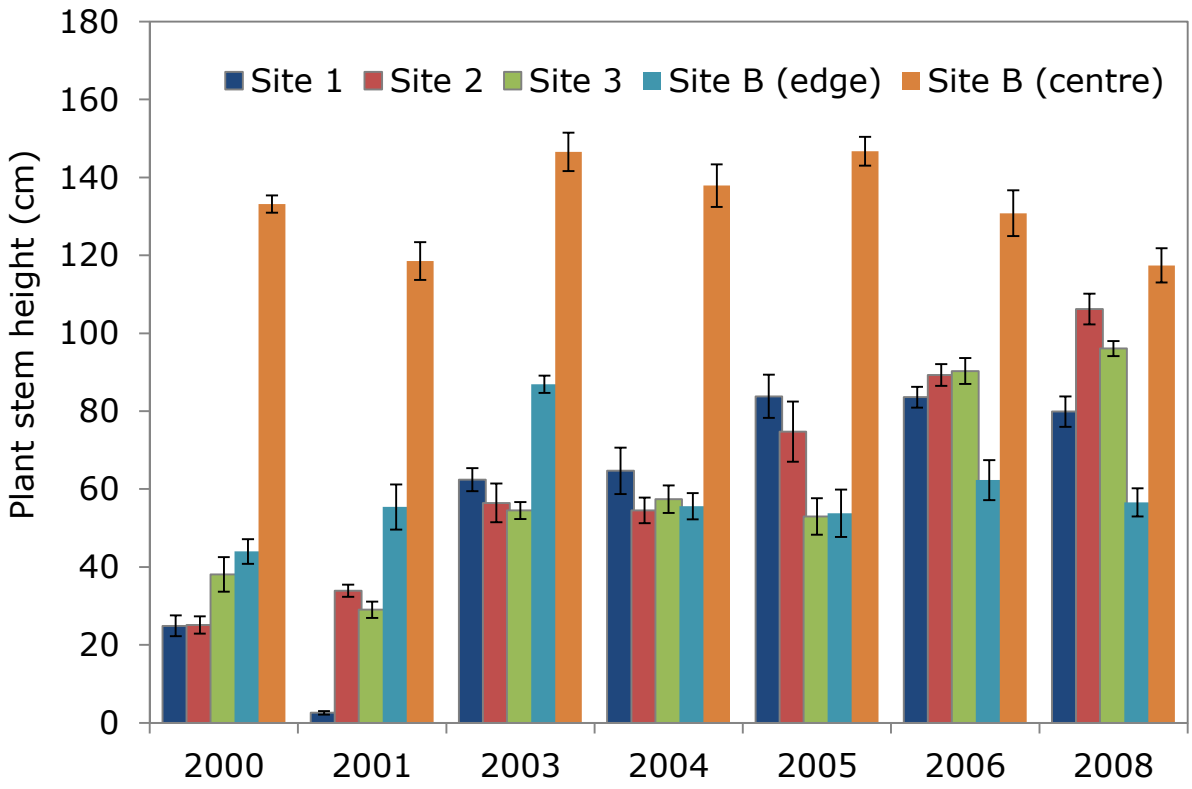


Figure 4.9 Changes in stem height (cm) at planting sites 1 – 3 and natural B. maritima marsh site B between 2000 and 2008..

Similar patterns of change were apparent for stem density in the north shore sites (Figure 4.10). Differences between the two sites ($F_{1,\ 28}$ = 40.04; P < 0.001), years ($F_{1,\ 28}$ = 13.74; P < 0.01) and the interaction between sites and years ($F_{1,\ 28}$ = 19.67; P < 0.001) were also significant. Stem density remained consistent between the study years in the natural marsh for both the edge (120 – 150 per m^2) and the centre (420 – 550 per m^2) of the stand. Change in density is much stronger in the newly planted site and steady throughout for the edge of the natural marsh. Stem density at Sites 1 - 3 steadily increased between 2000 and 2003 from 10 to approximately 100 per square

metre. Between 2004 and 2005 stem density in the planted sites equalled that in the outer edge of the natural marsh (150 – 180 per m^2), but had more than doubled in density during 2006, i.e., six years post-planting. By 2008 stem density in all three planted sites was very similar to the centre stand of the natural marsh (500 and 580 stems per m^2, respectively).

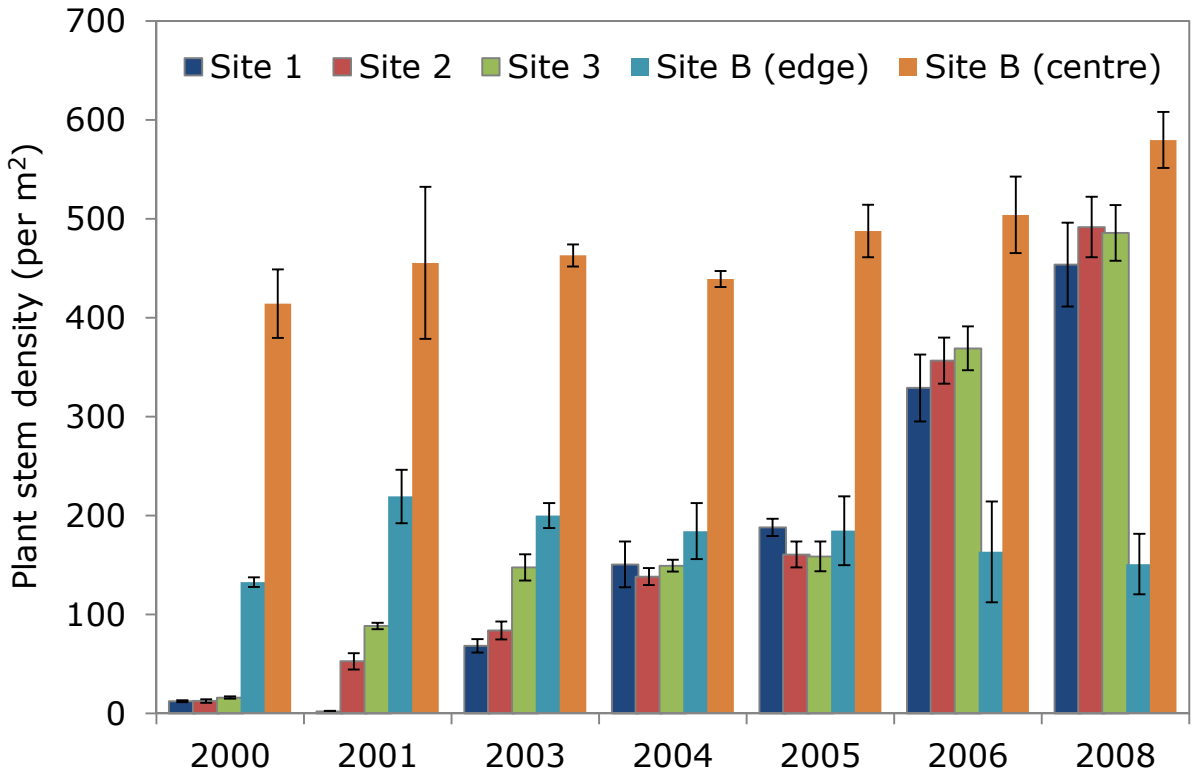


Figure 4.10 Changes in stem density (per m^2) at planting sites 1 – 3 and natural B. maritima marsh site B between 2000 and 2008.

4.2.3.2 Plant stem height and density on the south shore

Sites 6 and 7 had failed by the end of 2003 and as a result stem heights and density at planting Site 8 only were compared to plants at the edge

of the natural marsh *B. maritimus* (north shore) between 2003 to 2008 (Figures 4.11 and 4.12). The sites ($F_{1, 90}$ = 143.74; P < 0.001), years ($F_{4, 90}$ = 3.57; P < 0.01) and the interaction of sites and years ($F_{4, 90}$ = 10.71; P < 0.001) were significantly different. Stem height doubled at site 8 (20 - 40 cm) between 2003 and 2008. Stem height at the edge of the natural marsh had an unusual growth spurt in 2003 but heights were otherwise similar (50 and 60 cm).

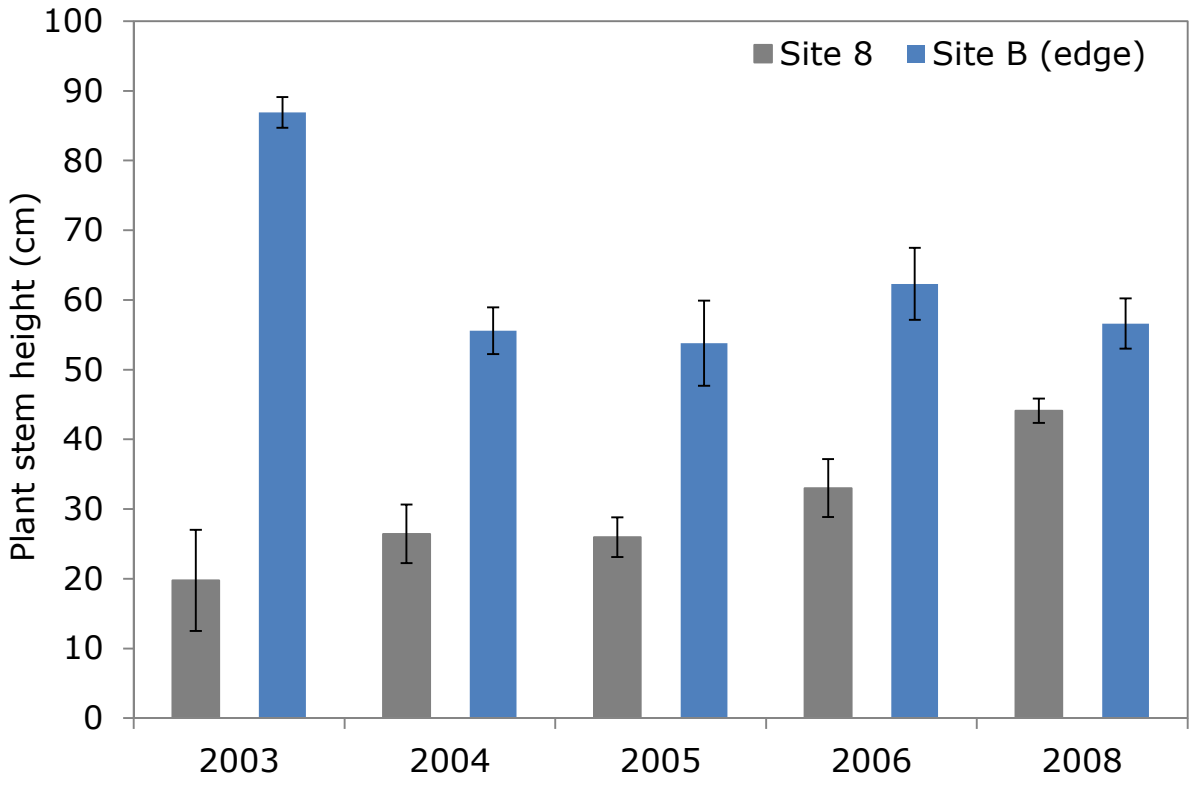


Figure 4.11 Changes in stem height (cm) at planting site 8 on the south shore between 2003 and 2008.

Stem density differences between the Site 8 and the natural *B maritimus* marsh ($F_{1, 20}$ = 6.54; P < 0.01), over the six years ($F_{4, 20}$ = 11.31; P < 0.001) and the interaction between sites and years ($F_{1, 20}$ =

18.40; $P < 0.001$) were significant. Stem density in the planted site steadily increased from 10 m^2 to match stem density in the natural marsh (150 m^2) but then more than doubled between 2006 and 2008 to over 350 stems per m^2 and overtook stem density in the natural marsh.

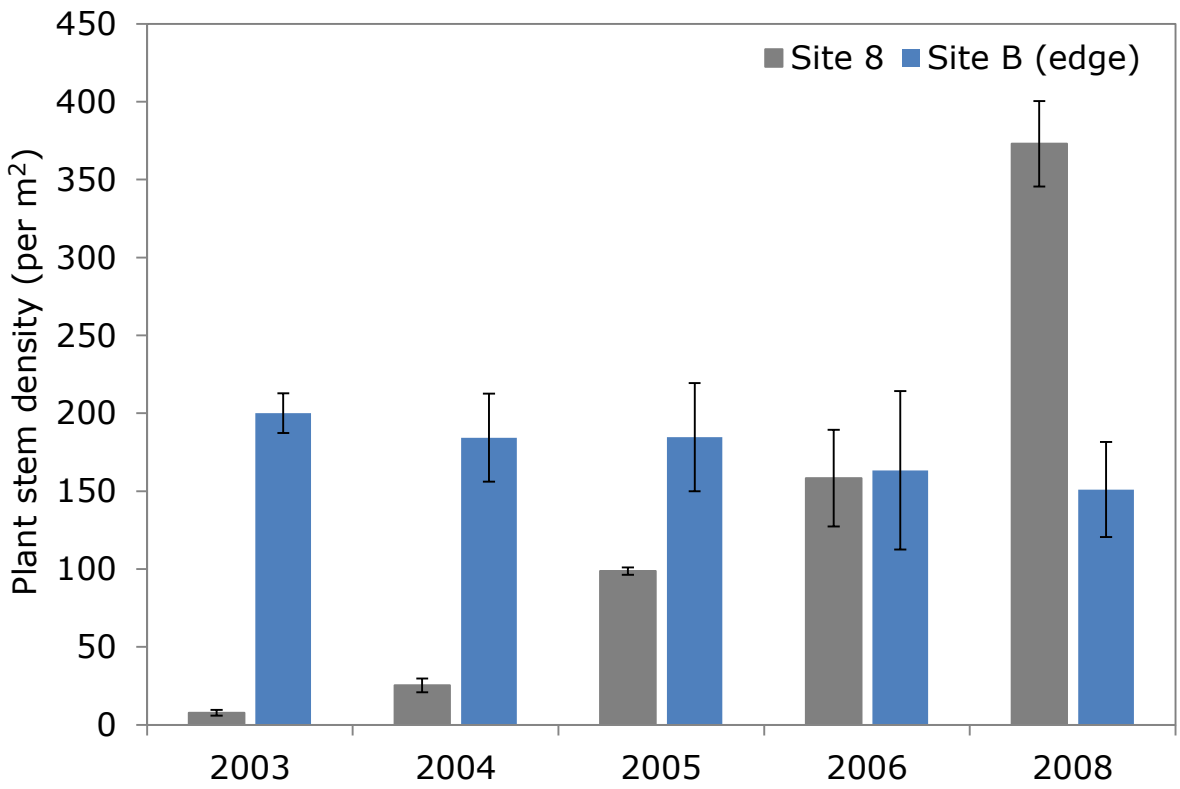


Figure 4.12 Changes in stem density (per m^2) at planting site 8 on the south shore between 2003 and 2008.

4.2.3.3 The lateral expansion of the new marshes

The lateral spread that occurred at all four planted sites between 2003 and 2008 were similarly steady (Figure 4.13). Sites 1 and 2 had grown to

approximately 50 m^2 by 2008 (Figures 4.14 and 4.15), whereas Site 3 was nearly four times larger at 200 m^2 (Figure 4.16). The expansion of planting Site 8 was rapid, despite being planted three years behind those on the north shore, spreading from 5 m^2 in 2003 to 80 m^2 by 2008 (Figure 4.17), and nearly twice the size of Sites 1 and 2.

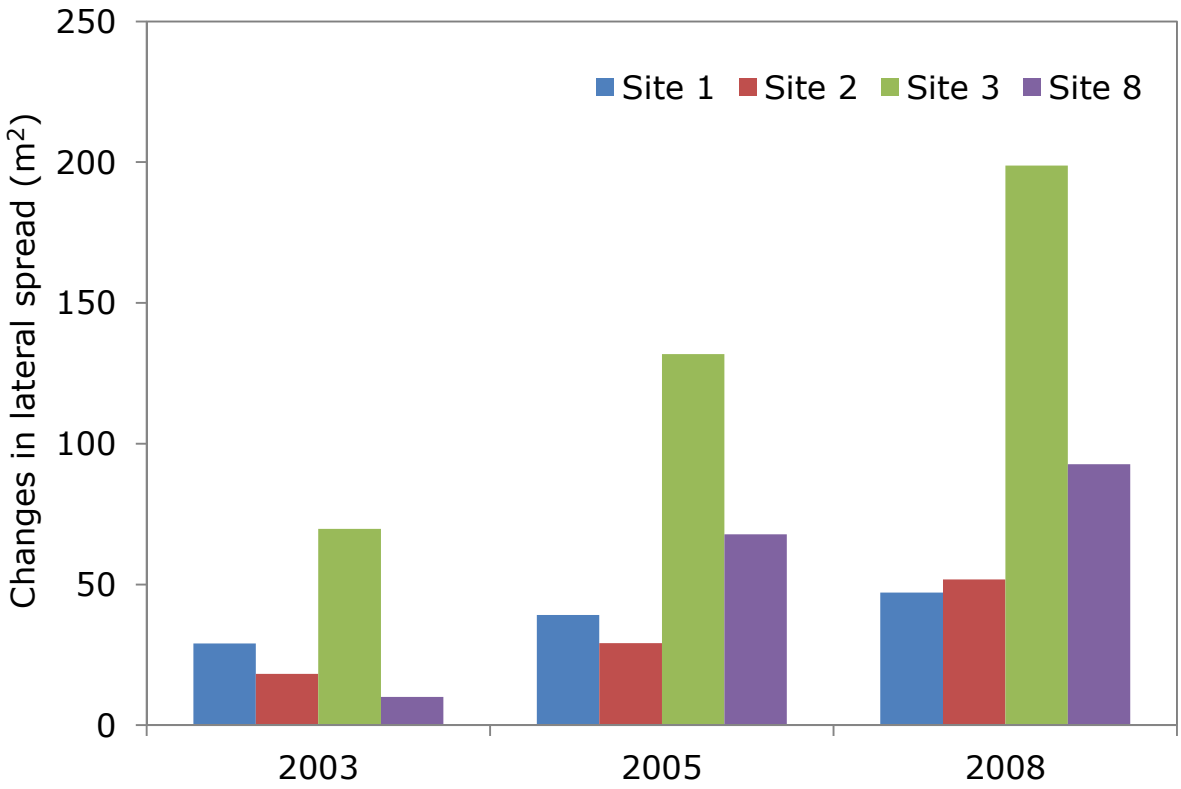


Figure 4.13 The amount of lateral expansion (m^2) in the four planted sites (north shore sites 1, 2 and 3, and south shore site 8) between 2003 and 2008.

Figure 4.14 Changes over time at planting Site 1 between May 2001 (top image) and May 2008 (bottom image). Note the accumulation of fine sediment over the tyres and at the foot of the rubble tip.

Figure 4.15 Changes over time at planting Site 2 from May 2001 (top image) to September 2005 (bottom image).

Figure 4.16 Changes over time at planting Site 3 from May 2000 (top image) and August 2005 (bottom image). The eroded P. maritima marsh behind the new marsh was thriving (not visible in the image).

Figure 4.17 Planting Site 8 on the south shore in Jan 2003 (top image) and November 2009 (bottom image, courtesy of R. Crawford).

4.3 Discussion

New saltmarsh restoration practices are increasingly important given the loss of the habitat and its future regarding sea level rise and climate change. The protection of sea defences from erosion by increased storm activity and valuable hinterland from coastal flooding may be lessened if successful saltmarsh restoration techniques can be found. Within the Eden Estuary, three locally derived marsh species, *P. australis, B. maritimus* and *P. maritima* were compared in transplant trials as potential erosion control agents, under similar field conditions and the growth of the successful trials were monitored.

4.3.1 Propagule type

One of the tenets of saltmarsh restoration is that it is a relatively inexpensive practice compared to reinforcing the shoreline with hard engineering structures. The broadcast sowing of seeds is less labour intensive than vegetative propagation, and, because fewer man-hours are required, is the most cost-effective means to restore saltmarsh vegetation. During this study however, seeding as a form of propagation was wholly unsuccessful. Scattering and burying the seeds during autumn to lay in the sediment over winter imitates natural conditions, but it was possible that the seeds were washed away (Dunne et al, 1998) or devoured by benthic invertebrates (Hughes, 1999).

The seeds that were used during the spring planting phase may also have met similar fates. However, the spring planting seeds were kept in

wet and cold storage over the winter, as suggested by Lewis (1982) and Broome et al (1988). Studies in the USA found that *S. alterniflora* seeds were only successful after being imbibed in seawater (20 – 30 psu) prior to use (Woodhouse et al, 1976). Furthermore, the burial depth of the seeds can also be important as *S. anglica* seeds buried at 1.5 – 3.0 cm in the sediment were more successful than those buried at 0.5 cm (Groenendijk, 1986). Groenendijk (1986) also found that *S. alterniflora* seeds deteriorated within 3 to 4 months of implanting into the sediment.

Finally, seed metabolic activity in the mature populations of *P. australis* and *B. maritimus* was tested, following the protocol of Kuo *et al* (1996) but the seeds were found to be not viable (data not shown). In summary, seed viability can vary from year to year (Ungar, 1987), seed production can be erratic (Dunne et al, 1998), and seed availability can limit saltmarsh colonisation in newly created saltmarsh (Erfanzadeh et al, 2010). Given that they are a relatively economical means of propagation, seeding as a means to reinstate saltmarsh vegetation requires further research.

4.3.2 Differences in establishment success between species

The tidal regime within an estuary is a critical factor during natural saltmarsh establishment, mainly due to salinity and flooding because of the effect they can have on plant growth (Adams, 2002). Zedler et al (2003) found that high salinity can cause transplant mortality during saltmarsh creation schemes, but the response differed among sites and years.

Waterlogged sediments can also affect plant mortality during saltmarsh creation (Boesch, 1994). In this study, there was a metre of difference in marsh elevation between the north and south shore planting sites and sediment salinity in south shore sites was generally higher. However, given that the natural marsh stands of all three species occupy a broad range of marsh elevations within the Eden Estuary, and all are known to be tolerant of varying degrees of salinity and waterlogging, it might have been predicted that there would be no difference in establishment success; however clear differences emerged. Although *P. australis* sprigs initially budded, these plots then failed completely halfway through the first year of growth.

P. australis and *B. maritimus* are both rhizome producers and therefore the individuals within a natural marsh stand have underground carbohydrate reserves to draw on for the maintenance of respiration during extreme conditions or for shoot emergence in spring. Though *P. australis* is abundant in waterlogged soils, it has a preference for freshwater and is rarely found when soil salinity is higher than 20 psu. The mechanism of shoot elongation is very similar for both these species. Energy resources not utilised during summer growth are stored as complex carbohydrate in the underground stems and rhizomes. This energy supply maintains respiration in the underground parts during winter, and sustains initial shoot development in the spring until photosynthetic tissue develops in the above-ground shoots. Within a single clonal stand the individual shoots are connected via rhizomes underground and this

facilitates access to the energy reserve of the whole stand (Koppitz *et al,* 1997). Individuals located in the more sheltered parts of the stand supply energy to those individuals at the outer edges of the stand where salinity and anoxia may be more prevalent. As the growing season progressed, the decrease in the survival of the planted sprigs may reflect the point where the stored energy of the individual sprig was depleted, and photosynthesis of the new shoots was less than the respiration required by the whole sprig to survive. Thus, having no access to the energy reserve of a whole stand, the individual sprig died although this energy balance was not measured in this study.

4.3.3 The effect of planting month and planting density

The month of planting was found to be crucial in other restoration efforts as the survival rate of planting later in the season can be low (Vanderbosch & Galatowitsch, 2011). In this study, timing of planting did not appear to have an effect at the start of the growing season, but differences became apparent at the end of the growth season when fewer sprigs from the autumn planting session survived. The fragmentation of plant parts, especially rhizomes, during dormancy can be a cue for bud development and growth as a response to perturbation (Charpentier et al, 1998). Autumn planting meant that sprigs were removed from the donor marsh before the onset of dormancy, whereas sprigs planted in spring were removed immediately prior to the end of dormancy, and therefore may have responded to the cue of dormancy

break. Spring planting is also advised to provide transplants with the summer to establish roots before the onset of winter and storm weather (Woodhouse et al, 1976; Burchett et al, 1998). Interestingly, despite the death over the summer of the majority of sprigs planted in autumn, none had been lost during the winter immediately post-planting.

Higher density planting achieved greater success than low density planting. *B. maritimus* can aerate the root zone via the shoots (Kantrud, 1996). Iron hydroxide plaque formation was visible on the rhizomes of the natural *B. maritimus* stands, suggesting they were releasing oxygen. It is suggested that the higher density planting of this species may have resulted in more oxygen in the sediment and as a result, the energy spent in coping with an oxygen deficit was instead reallocated to growth. It was also probable that the increase in plant cover associated with a higher density may have provided greater protection from the physical impact of wave action (Woodhouse et al, 1976; Clevering, 1997).

4.3.4 Species success in relation to environmental conditions

The sprigs of *B. maritimus* were initially far more successful than either of the other species and a greater proportion was alive at the end of the growing season, which confirms that it can occupy a broad range of habitats (Haslam, 1971; Broome *et al,* 1995). It can tolerate hypersalinity (Shay & Shay, 1986) but requires water to reduce the cell damage by excess salt ions. Although anoxia can also reduce growth, the sediment at the transplant sites were not waterlogged, suggesting that

tidal flushing was sufficient to aerate the sediment and increase the survival of this species at least.

Sediment conditions at all of the sites were brackish and not waterlogged, and so it was noteworthy that *P. maritima* initially had a lower rate of success than either *P. australis* or *B. maritimus*, given that it can tolerate high sediment salinities: e.g. at or above open sea strength (i.e., fully marine) (Cooper, 1982). It is a clonal species whose populations can reach maximum potential on stabilised lower marshes (Langlois et al, 2003) though it does not have substantial carbohydrate reserves stored in underground rhizomes like *B. maritimus*.

P. maritima was transplanted only on the south shore of the estuary (Sites 6, 7 and 8). These south shore sites are more exposed than any of the sites on the north shore, and the natural *P. maritima* marsh highly fragmented and eroded. The individual planting units of *P. maritima* were also much smaller and had a very shallow root system compared to the other two species, and it was possible that this limited rooting and therefore the acquisition of nutrients for further growth. However, though limited in growth, some of the transplants at Site 8 were still alive at the end of the growing season, unlike *P. australis*.

Hutchinson (1982) found that *B. maritimus* thrives on silts and suffers competition on sandier sediments; however, within the Eden Estuary the natural stands of this species grow on both silt and sandy sediments. The success of *B. maritimus* over *P. maritima* on the sandier and more exposed south shore sites possibly confirms the wide tolerance of *B. maritimus* to

different environmental conditions. However, *B. maritimus* sprigs failed at Sites 4 and 5 on the north shore also, and at Sites 6 and 7 on the south shore. The proximity of Sites 4 and 5 to Coble House Point may suggest higher wave activity than planting Sites 1 to 3 and therefore responsible for the failure at these former sites. However, two unusual events occurred in the vicinity of Sites 4 and 5 during the summer of 2000. First, there was a high density of the mud snail *Hydrobia ulvae* on the mudflats and the plant stems (Figure 4.18). Paramor & Hughes (2005) suggested that *H. ulvae* grazing on saltmarshes could be a cause of die-back in saltmarshes in the SE of England. It is therefore possible that this may be a factor in the death of the young sprigs during this study also. Second, an unusual quantity of the algae *Enteromorpha* spp. appeared and blanketed both sites in the summer of 2000 (Figure 4.19) and again in the summer of 2003, though by this time these sites had been removed from the study. Ostendorp (1992) showed that algal wash, mainly *Chara* spp, was a major cause of reed decline in Lake Constance, Germany. However, plant mortality caused by algal smothering was observed in 2011 (as part of continuing studies in the Eden Estuary) in an eight year old planted site in 2011 and will be discussed in Chapter 6.

Figure 4.18 Hydrobia ulvae grazing on young B. maritimus shoots and mudflats at Site 5 in July 2000 on the north shore.

Figure 4.19 Smothering by Enteromorpha spp. may have been the cause of plant death at planting Sites 4 (foreground) and 5 (background). This image was taken in September 2000 but the algae had been present at these sites for most of the summer.

Studies to investigate the impact of invertebrate grazing on saltmarsh planting may need further investigation. However, the greater exposure to wave activity at Sites 4 and 5 is unlikely to have been the cause of failure because Site 8 on the south shore was as equally exposed, and yet transplanted *B. maritimus* sprigs were as successful as those at Sites 1, 2 and 3 on the north shore. Exposure to wave and tidal energy can also not be the cause of failure of plant growth at Sites 6 and 7 on the south shore and may have been due to the higher elevation (nearly a metre) compared to Site 8. However, sediment salinity and water content varied little between these three sites and therefore the cause of failure is not clear.

4.3.5 Growth comparisons in the planted marshes

If all the sites on the south shore had failed then it would easy to suggest that the cause of mortality was due to the higher elevation of these sites compared to the successful north shore sites (2.5 – 3.0 m compared to 0.5 – 0.7 m above sea level, respectively). It would also have been understandable if the greater exposure and sandier sites on the south shore could be implicated. However, site differences such as these appeared to have no relevance when it came to growth and expansion of the sites. For example, Site 8 was higher in elevation and therefore had fewer tidal inundation events, and yet throughout the study had similar water content to those sites on the south shore while at the same time higher sediment salinity. It was also planted three years after those sites on the north shore and yet despite this grew and expanded at a much faster rate than Sites 1

and 2, and within just a few years had exceeded the growth of the plants at the outer seaward edge of the natural marsh. The rapid growth and expansion of Site 3 may be due to lower sediment salinity (10 psu) than even that found in the natural marshes (12 psu). The water content of the sediment at Site 3 was also relatively high and marsh elevation suitably low and this combination possibly kept the sediment moist, especially during the drier, summer growth period.

It was apparent that the growth and expansion of the successful planted sites had a positive effect on sediment accretion as the older sites appeared raised in comparison to the upper, unvegetated mudflats immediately adjacent. Increasing plant density increases the rate of sedimentation on a developing marsh (Hall and Freeman, 1994) and as sediment deposition and accretion is the desired consequence of the practice of the direct planting of saltmarsh vegetation, this matter is explored in the next chapter.

CHAPTER 5 SALTMARSH SEDIMENTATION

5.1 Introduction

Estuarine fringing saltmarsh absorbs wave and tidal energy and strengthens upper shorelines by capturing and retaining sediment (Brooke et al, 1999). The remnant saltmarsh populations on the seaward side of the seawalls and embankments in the Eden Estuary have suffered extensive die-back and erosion over the last twenty years (Fife Council, 2008). The erosional features include lateral stripping of the vegetation and deeply eroded incisions into the marsh body, whereby the underlying sediment bed becomes fragmented and the fragments slump on to the upper mudflats.

Coastal squeeze is often evaluated as being the cause of saltmarsh erosion and die-back. This hypothesis appears to make sense, given that rises in sea level increase both the water depth and wave and tidal energy, with many saltmarsh species being sensitive to such hydrological changes. When the overlying tidal waters become too deep for survival, saltmarsh habitats have 'rolled back' in the past. As many estuaries are now ringed by immovable defences, the marsh has nowhere to retreat and the Eden is no different, given that more than 60% of its shoreline has been hard engineered.

However, the concept of coastal squeeze as the driving force behind saltmarsh erosion is overly simplistic and fails to take into account that, given a sufficient supply of sediment for accretion, historic rises in sea

levels have been beneficial to the creation of new marsh (Cundy and Croudace, 1996; Crawford, 2008). It has also been proposed that there is a shortage of sediment available for saltmarsh accretion but this fails to take into account that in some other UK saltmarshes the rate of sediment accretion has been found to be equal to, or greater than, the current rate of sea level rise (Cahoon et al, 2000; van der Wal and Pye, 2004), suggesting that factors other than sea level rise and a lack of available sediment could be responsible for saltmarsh decline.

For example, the loss of the upper marsh zone and subsequent embanking increases the reflected wave and tidal energy over a saltmarsh (Reed et al, 1999). There is also greater sediment movement in general but less fine-grained sediment in front of sea walls, suggesting increased wave reflection (Bozek and Burdick, 2005; Airoldi et al, 2005). The sediment deposition considered necessary for natural salt-marsh development (Adam, 2002) is possibly reduced by the increased physical exposure associated with these regime changes.

Plant health and growth, especially in the underground biomass, can be impaired by sediment starvation (Fragoso, 2001) and hydrological changes (Turner et al, 2001). It has been suggested that less sediment enters the Eden basin than in former times, when the mouth of the estuary became partly occluded by the growth of a spit over a disused town rubbish tip (Crawford, 2008). In addition, the capture and retention of sediment by a marsh that is fragmented and eroded at its seaward edge is unlikely (Ranwell, 1964; Brown et al, 1998) and turbulence can be further

increased without the hydrodynamic protection provided by the vegetation (Leonard and Croft, 2006; Neumeier, 2007).

Vegetation and root die-back from the effects of pollution (Mason et al, 2003) and eutrophication (Darby and Turner, 2008) also increase the vulnerability of saltmarsh to erosion. The *P. maritima* marsh is also relatively old (Wilson, 1910) and, being relatively high on the tidal frame, the main body of the marsh is usually only completely inundated during spring high tides. Young, actively developing marshes lower on the tidal frame tend to have relatively high rates of sediment accretion (Pethick, 1981; Langlois et al, 2003) which decrease with increasing marsh elevation (Temmerman et al, 2003).

More importantly, the extensive literature on the importance of sedimentation in saltmarsh formation and resilience (Reed, 1989; Allen and Duffy, 1998; Boorman, 1998; Brown et al, 1998; Temmerman et al, 2003; Leonard and Croft, 2006; Murphy and Voulgaris, 2006) tends not to include information regarding the influence of transplanted vegetation on sedimentary processes, whilst the potential value of *B. maritimus* in trapping sediment has been overlooked. Whilst many studies have characterised sediment deposition and accretion in the natural marshes around the UK, few sedimentation values are available for the saltmarshes on the east coast of Scotland and none have been conducted on transplanted marsh and there are no studies for the comparison of the rates of accretion that can be achieved and sustained by marsh creation at the forefront of denuded marshes.

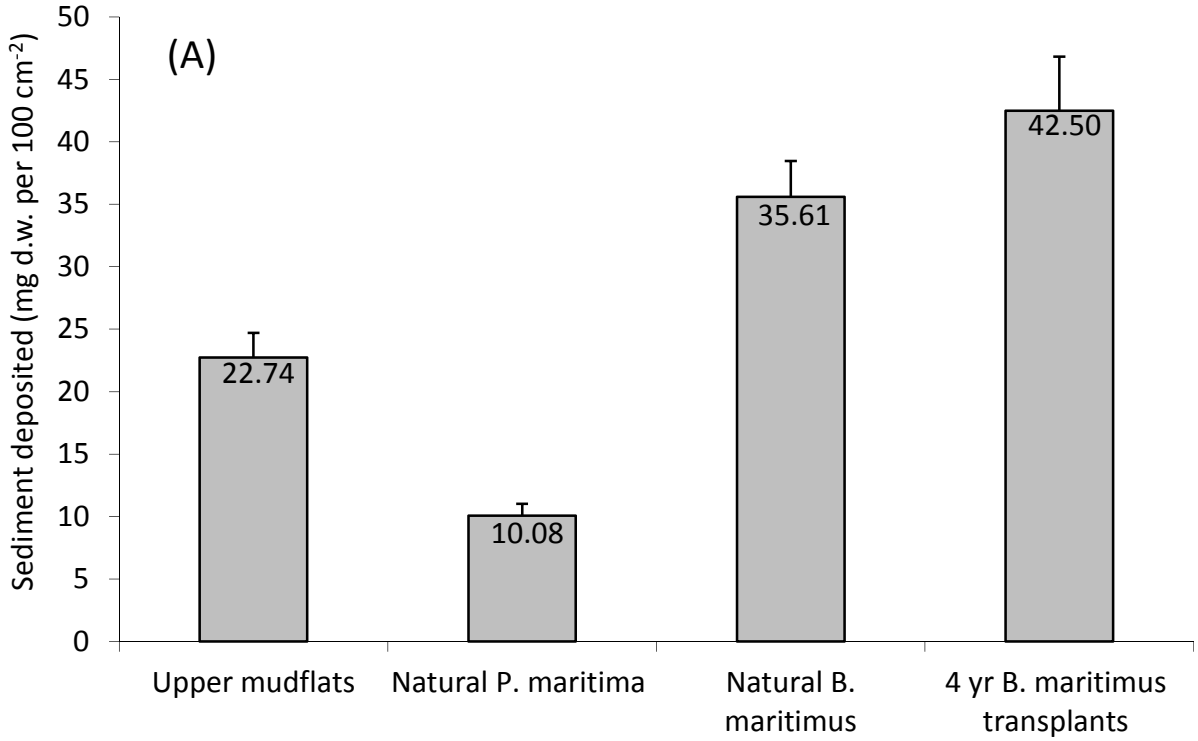
This chapter presents the short-term sediment deposition and accretion patterns in transplants of *P. maritima* and *B. maritimus* and compares them to those in upper, unvegetated mudflats; natural but eroded *P. maritima* marsh; and relatively healthy stands of natural *B. maritimus.* Changes in sediment deposition in these different systems in relation to tidal height and wind direction, and therefore to increasing sea level and climate change, have also been investigated. Sediment deposited each day was collected by means of pre-weighed filter papers placed on the sediment surface whilst the total sediment accreted each month was calculated in relation to the zero bed level by using a bar placed across marker poles.

5.2 Results

5.2.1 Mean total sediment deposition

There was no significant difference between the mean total sediment deposited (35.61 and 42.50 mg cm^{-2}, respectively) in the natural stand and in the four-year-old transplants of *B. maritimus* (Figure 5.1A), although both of these sites gave significantly higher values than the corresponding mudflat (22.74 mg cm^{-2}) or the natural *P. maritima* stand (10.08 mg cm^{-2}; $F_{3,\ 914} = 26.05$; $P = 0.001$). The amount of sediment deposited on the south shore (Figure 5.1B) was not significantly different between the mudflat (20.84 mg cm^{-2}) or the one-year-old transplants of either *B. maritimus* or *P. maritima* (26.94 and 25.78 mg cm^{-2},

respectively), but was significantly lower in the natural *P. maritima* stand (2.6 mg cm^{-2}; $F_{3, 901}$ = 70.65; P = 0.001).



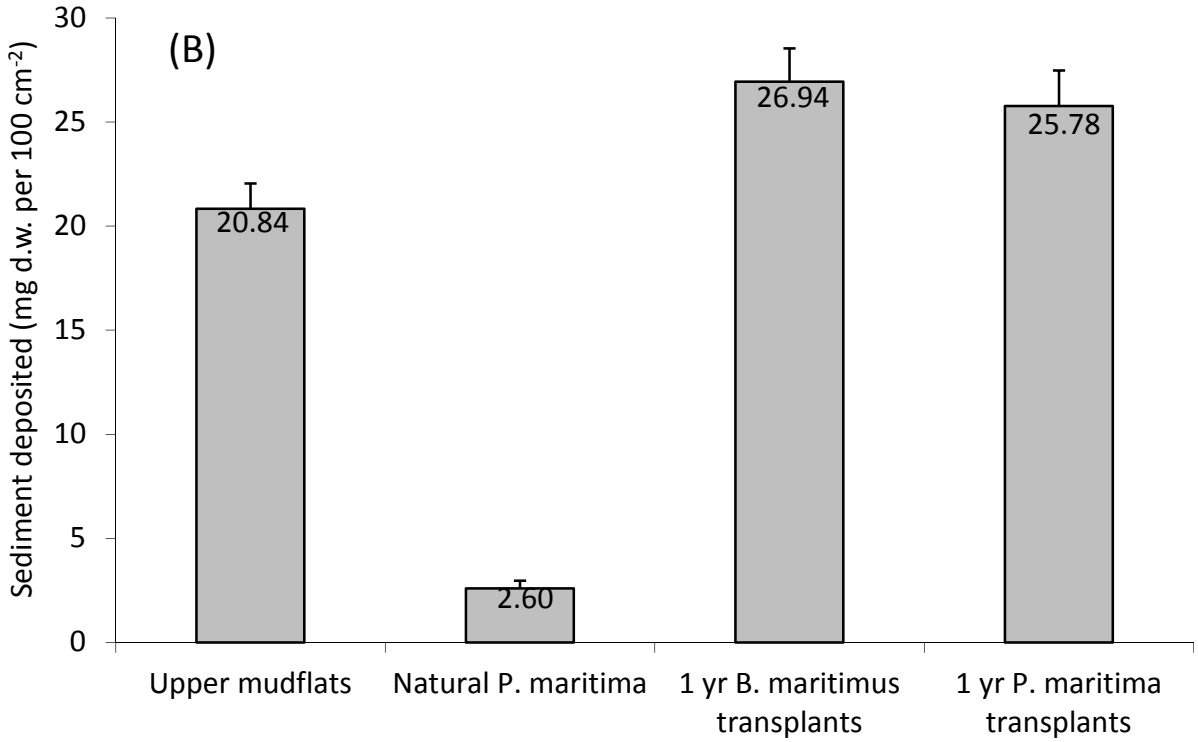


Figure 5.1 Sediment deposited during July and August 2004 for (A) the north shore and (B) the south shore. Results are shown in dry weight (mg 100 cm^{-2}) and relate to the average (+ standard error) of the total amount of sediment deposited over 23 days.

5.2.2 Sediment surface level

At the end of the one-year study, the mudflat, and the natural and transplanted *B. maritimus* had accreted 1.2, 3.4 and 3.8 mm of sediment, respectively, compared to the zero bed level maintained by natural *P. maritima* (Figure 5.2). The differences between the sites were significant ($F_{3,104}$ = 71.63; P = 0.001) and post-hoc analysis showed that the *B. maritimus* sites had significantly more accretion than either the mudflat or the natural *P. maritima* sites.

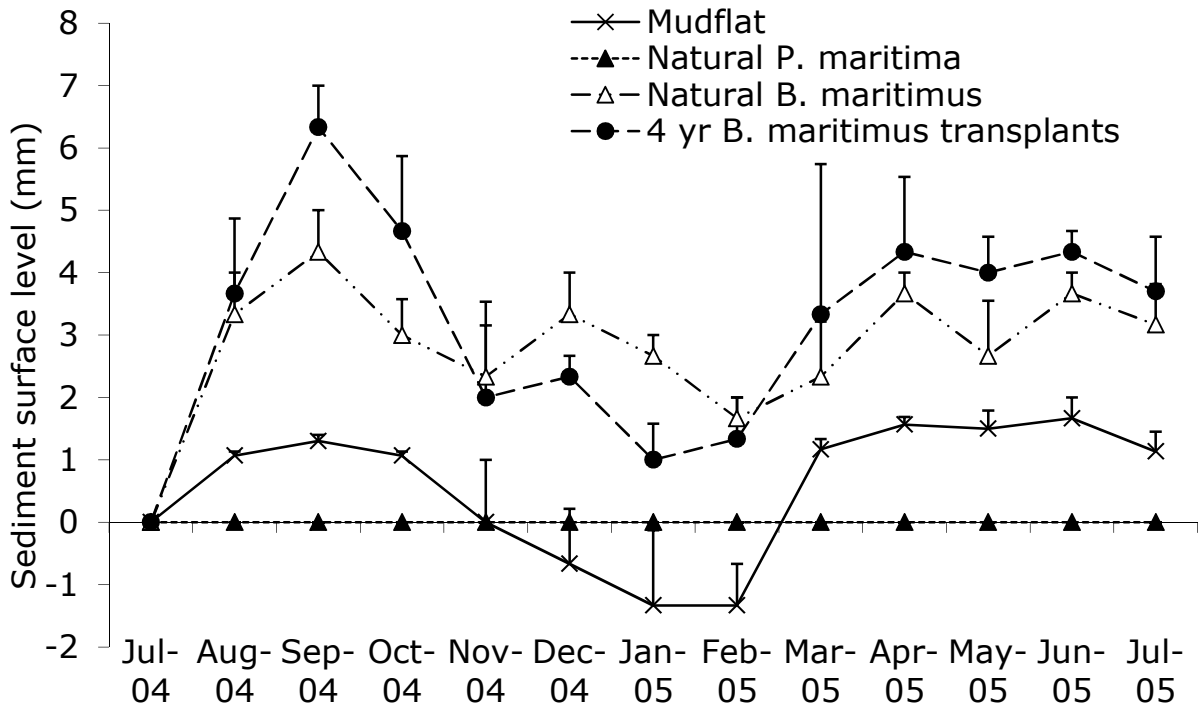


Figure 5.2 Changes in soil level (mm) during 1 year on the north shore of the Eden Estuary. The zero soil level was established in July 2004, and measurements were recorded after a spring tide in each subsequent month until July 2005. Each data point is the average (+ standard error) of six readings taken from below a builder's level placed between two poles.

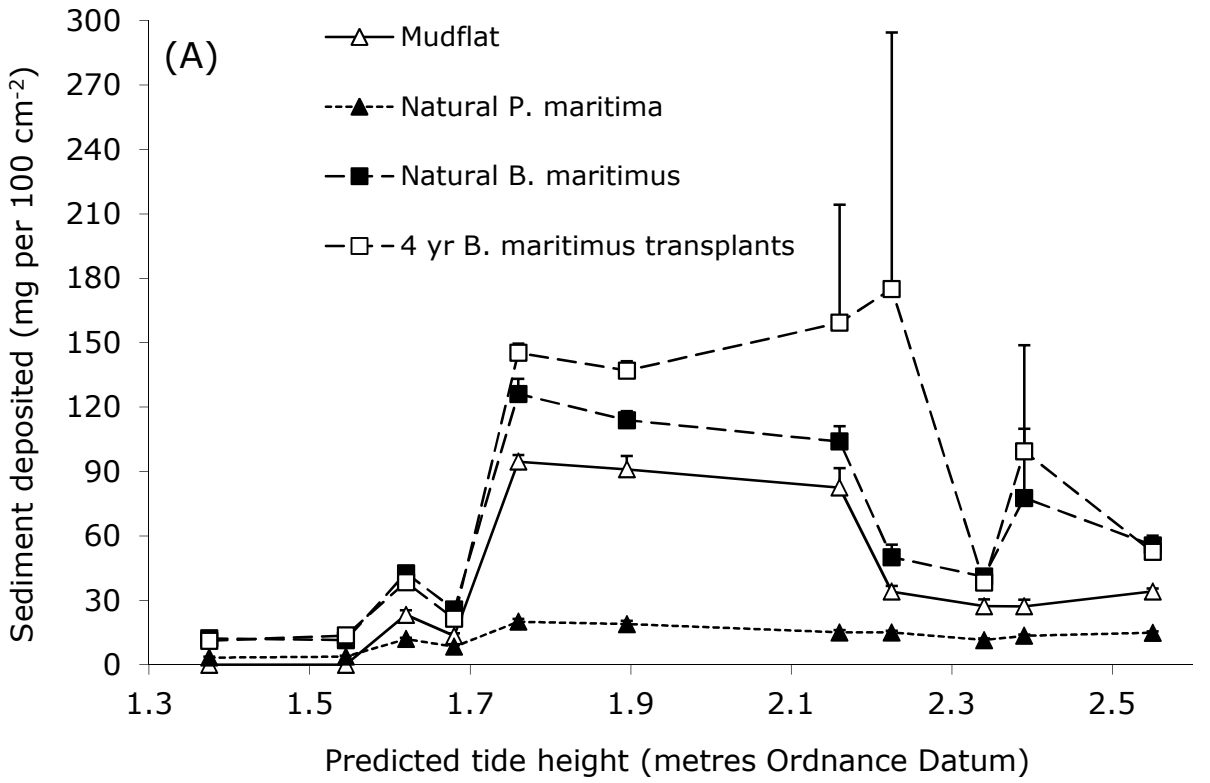
The differences between the months were significant ($F_{12,104}$ = 6.99; P = 0.001), and in general more accretion occurred between April and October than between November and March. The interaction between site and month was also significant ($F_{36,104}$ = 1.64; P = 0.027), as the mudflat and *B. maritimus* sites accreted sediment during the summer months but eroded over the following winter. However, the mudflat site was the only site to erode below zero bed level during winter, losing nearly half of the sediment gained during the previous summer.

The natural and planted *B. maritimus* sites demonstrated a similar pattern throughout the study, but whereas the latter had higher peaks around the high tides associated with March and September, the former retained more sediment over the winter months. On the other hand, by the end of the study there was no significant difference in accretion between the natural or planted *B. maritimus* sites, and even the mudflat sites had recovered the sediment lost over the winter.

5.2.3 Relationship between tidal height and sediment deposition

A significant positive relationship was found between tidal height and

sediment deposition for all the sites on both shores. The strength of the association differed between the sites (r = 0.42 to 0.57, $P < 0.001$, n = 110 per site). However, a curvilinear relationship was apparent, and more sediment was deposited during 'normal' tidal heights, as opposed to either neap or spring tides (Figures 5.3A and B). For example, mean sediment deposited was below 20 mg 100 cm^{-2} for all the sites during the lowest tidal events (1.3 to 1.7 m Ordnance Datum) but a sharp increase became apparent in both the natural and four-year-old transplant sites of *B. maritimus* and the upper mudflats on the north shore once the tide height rose above 1.7 m OD. Although sediment deposition increased with increasing tide height in the natural stands of *P. maritima* on both shores, the effect was not dramatic, and on the south shore in particular it was minimal until spring tides higher than 2.2 m OD were reached.


(A)
Mudflat
Natural P. maritima
Natural B. maritimus
4 yr B. maritimus transplants
Sediment deposited (mg per 100 cm-2)
300
270
240
210
180
150
120
90
60
30
0
1.3
1.5
1.7
1.9
2.1
2.3
2.5
Predicted tide height (metres Ordnance Datum)

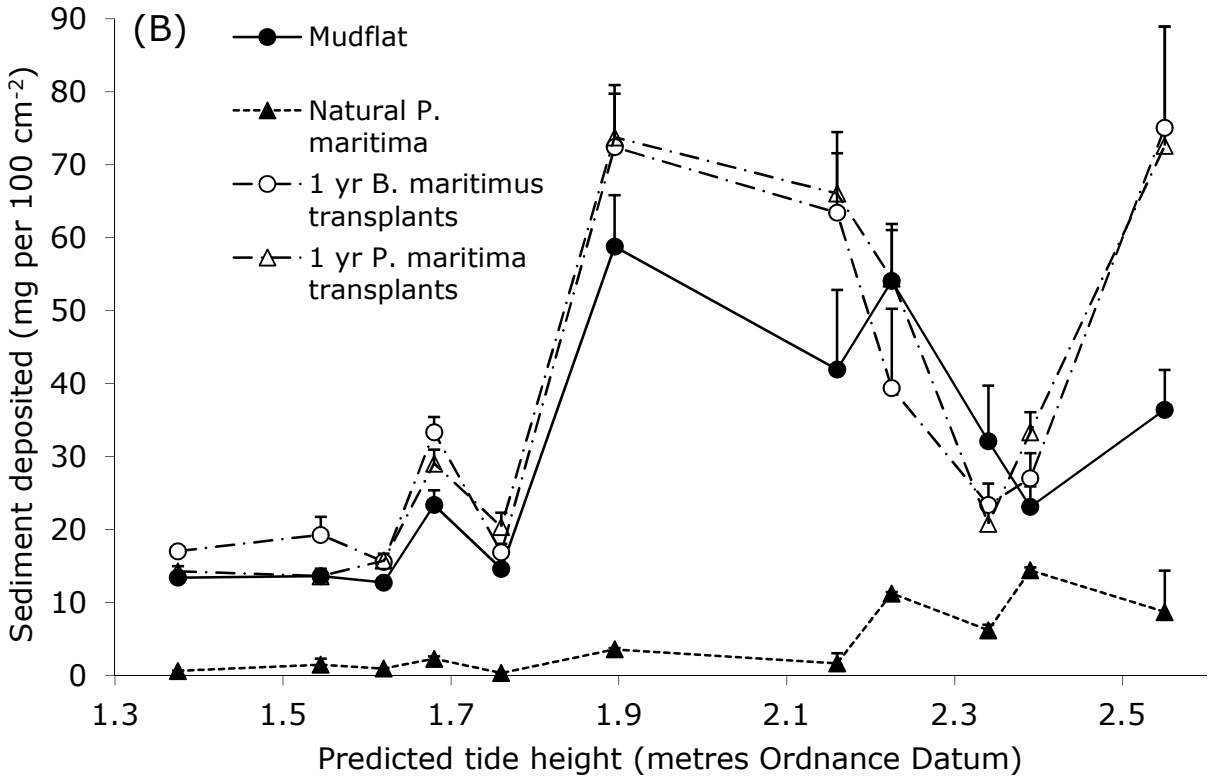

(B)
Mudflat
Natural P. maritima
1 yr B. maritimus transplants
1 yr P. maritima transplants
Sediment deposited (mg per 100 cm-2)
90
80
70
60
50
40
30
20
10
0
1.3
1.5
1.7
1.9
2.1
2.3
2.5
Predicted tide height (metres Ordnance Datum)

Figure 5.3 The effect of predicted tide height (metres above sea level Ordnance Datum) on short-term sediment deposition in the Eden Estuary (means + standard error). The tide heights are the average of two high tides per day: (A) the north shore and (B) the south shore.

5.2.4 Relationship between wind direction and sediment deposition

Winds from the south-east caused significantly more sediment deposition than winds from the south-west ($F_{1,384}$ = 196.21; P = 0.001). The differences between the sites were also significant ($F_{7,384}$ = 32.36; P = 0.001), as was the interaction between site and wind direction ($F_{7,384}$ = 20.49; P = 0.001). However, post-hoc analysis showed that wind direction had the greatest effect on the upper mudflats and the natural and transplanted *B. maritimus* sites on the north shore, but little effect on the mudflats and younger transplants of *P. maritima* and *B. maritimus* sites on the southern shore. There was no effect on the natural *P. maritima* sites on either shore (Figures 5.4A and 5.4B).

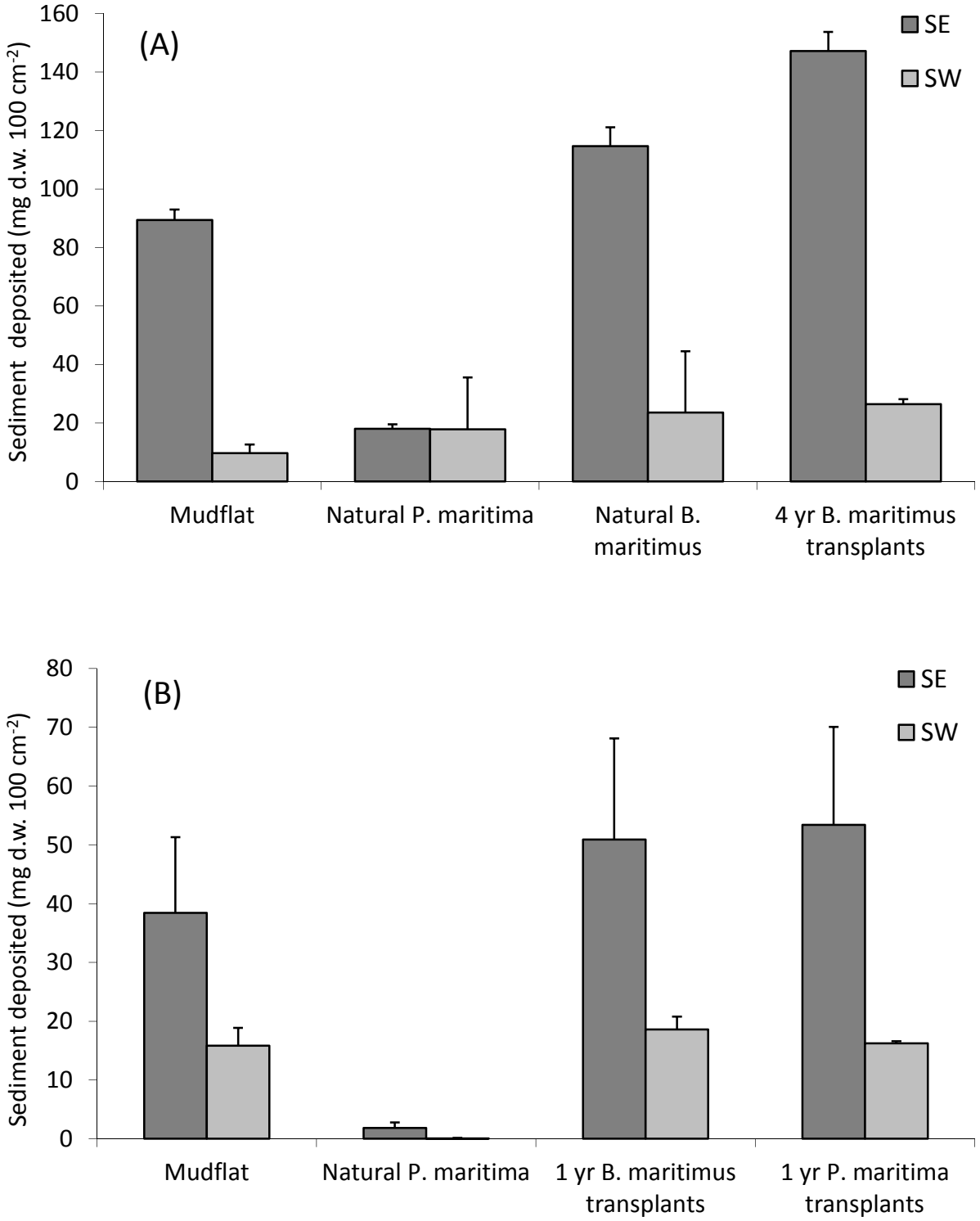


Figure 5.4 The effect of wind direction on short-term sediment deposition (means + standard error): (A) the north shore and (B) the south shore (SE, south-easterly and SW, south-westerly)

5.3 Discussion

5.3.1 Mean sediment deposition and surface levels

The natural marshes of *P. maritima* had lower quantities of deposited sediment compared to any other site, which could be the result of the increase in local turbulence and scour associated with a patchy and fragmented marsh front (Boorman et al, 1998; Brown, 1998; Leonard and Croft, 2006). The four-year-old transplant site of *B. maritimus* had double the quantity of deposited sediment relative to the adjacent, unvegetated mudflat site, and since these two sites share the same elevation this can be directly attributed to the sediment trapping action of plant stems and leaves. The vegetated sites, whether transplants or natural marsh, also had less erosion than the upper mudflats, suggesting that the vegetation conferred some degree of stabilisation and protection to the underlying sediment bed.

The findings in the present study appear to confirm that high marsh elevation can result in lower rates of deposition (Temmerman et al, 2003) and accretion (Pethick, 1981; Stoddart et al, 1989), since the southern shore sites, being almost a metre higher on the tidal frame than the northern shore sites, had significantly less deposition. Other factors, however, such as exposure (McManus and Alizai, 1987) and proximity to river channels (Temmerman et al, 2003), may also have had an effect. The low sediment deposition in the natural *P. maritima* sites may be due to fragmentation of the marsh front as well as reduced tidal inundation because of the high elevation of the marsh, but irrespective of the cause,

sediment starvation will be a key factor in its decline (Fragoso, 2001).

Species-specific effects may also be responsible for differences in sediment deposition, as the natural stands of *P. maritima* and *B. maritimus* on the north shore shared the same elevation, and yet considerable differences in sediment deposition and accretion were apparent. For example, the shorter, stiffer stems of *P. maritima* may have accounted for its lower rate of deposition compared to the metre-high stems of *B. maritimus,* as vegetation with taller and more flexible stems can impede the flow of water more effectively and capture sediment to a greater extent during conditions of low flow (Boorman et al, 1998). Conversely, Boorman et al (1998) also found that during high velocity flow, shorter and stiffer stems impeded water flow to a greater extent than taller vegetation. This would imply that *P. maritima* should have captured more sediment than *B. maritimus* during the faster and more turbulent flow conditions of winter, but the winter accretion rates recorded in the present study showed this not to be the case, possibly because stiff shoots caused more scouring than flexible shoots (Bouma, 2009). It is also possible that the sloping marsh edge of natural *B. maritimus* was more effective in dampening wave energy than the fragmented and near-vertical cliff edge of natural *P. maritima* (van Eerdt, 1985).

The similarity in sediment deposition between natural and four-year-old transplants of *B. maritimus* suggests that the expected differences due to marsh elevation (4.0 m and 3.1 m, respectively) may have been counteracted by differences in stem density (respectively, 400 m^{-2} and

150 m^{-2}) (Gleason et al, 1979; Hall and Freeman, 1994). In addition, similar sediment deposition rates between the one-year-old transplant and mudflat sites on the south shore may have been the result of low stem density in the former sites, and differences could emerge as stem density increases in the young transplants over time.

Plant litter and debris reduce the exposure of a sediment bed to erosion (Boorman et al, 1998), and this may explain why the natural stand of *B. maritimus* retained more sediment during winter than the adjacent *B. maritimus* transplant site. Conversely, the enhanced accretion in the transplant site during the summer may have been a consequence of its low stem density and greater substrate exposure, thereby encouraging the growth of diatomaceous mats (Anderson, 2001). Further investigation of the diatom communities within these sediments may be warranted.

5.3.2 Environmental controls on sedimentation

The effect of tidal height on sediment deposition was clear. Spring tides (those above 2 m OD, or 4.9 m Chart Datum) caused more deposition than neap tides (below 1.7 m OD or 4.6 m CD), and yet the highest quantities of sediment were deposited by tides between 1.7 – 1.9 m OD (4.6 and 4.8 m CD). The sediment load in tidal waters tends to be increased by spring tides because of the associated higher current velocities (Murphy and Voulgaris, 2006) and because more sediment in the river channel is captured during a low tide and carried on to the saltmarsh surface by

the following high tide (Ranwell, 1964). The present study showed that sediment deposition peaked during normal tides, suggesting that greater sediment availability in the water column may be cancelled out by higher current velocity. However, it appears that tidal height and marsh elevation exert a strong control on sedimentation in the salt marshes of the Eden and that sea level rise may become a driving force for an increase in accretion and sedimentation rates, as found in the Solent marshes, southern England by Cundy and Croudace (1996).

Marsh restoration efforts should also take account of changing weather patterns, as an increase in storms due to climate change could increase saltmarsh erosion. However, episodic storms and hurricanes can deposit a greater quantity of sediment in saltmarsh than regular tidal inundation due to the increased stirring effect on sediment in the water column (Reed, 1989). Changes in wind regime also can cause a response in marsh sedimentation (Allen and Duffy, 1998). Wind speed and direction, for example, were shown to have a significant effect on sediment deposition in the saltmarsh and reed bed communities in the neighbouring Tay Estuary (McManus and Alizai, 1987). Similarly, low rates of sediment deposition were apparent in the present study when winds blew from a south-westerly direction and may be a consequence of the downward force on the height of the high tide, providing less opportunity for sediment to be deposited on the marsh surface.

The higher rate of deposition associated with winds from the south-east could have been a reflection of the open aspect of the mouth of

the Eden Estuary, suggesting that although exposure is normally considered inimical to saltmarsh development, an increase in sediment deposition from rising sea levels when winds are from the south-east may actually benefit marsh development. However, wave-generated turbulence from increasing wind speed can also affect sediment deposition (Alizai and McManus, 1980), but as wind speeds were low in summer during this study, significant trends were not apparent.

CHAPTER 6 GENERAL DISCUSSION

6.1 Establishing saltmarsh through direct planting

Saltmarsh restoration guidelines developed through experiences in North America suggest that in high energy sites where erosion or die-back is prevalent, saltmarsh can be restored, and even encouraged to prograde seawards, provided the appropriate species and techniques are selected. The physical conditions within a potential restoration site need to be similar to those found in natural saltmarsh habitats, with parameters such as exposure and elevation on the tidal frame being critical (Brooke et al, 1999). If these conditions are met, then why saltmarsh vegetation has not naturally colonised the area is a valid question to ask prior to any restoration effort.

Natural colonisation of saltmarsh species occurs in three ways: by seeding, by vegetative clonal expansion, or by the rooting of vegetative fragments after they become separated from a marsh stand: e.g. by storm action or foraging activity. Not only can seed viability be extremely erratic, but for seeds to germinate, certain conditions during springtime must be met, such as a prolonged drawdown of the tide and an input of freshwater to ameliorate soil salinity. The seeds or seedlings may then be vulnerable to damage through wave energy or herbivory from benthic invertebrates. Clonal expansion through vegetative growth can also be a relatively slow process as growth at the seaward edge of a marsh is often compromised by increased salinity and wave action

(Adam, 2002). Fragments of vegetation can also be carried into new areas by tidal action and is not uncommon in Arctic marshes (Handa, 2000). Floating rafts of tidal debris with fragments of *P. australis* have been observed in the mid to outer parts of the Tay Estuary also and are likely to have come from further up river (personal observation).

Despite these methods of expansion available to clonal species, the opportunity for them to occur has probably been significantly affected by shoreline development (Deegan et al, 2012) and because of the isolation and fragmentation of natural populations of coastal saltmarshes. This study transplanted three saltmarsh species prevalent in estuaries in the east coast of Scotland into areas of the shoreline within the Eden Estuary where natural colonisation and regeneration was not occurring and erosion was significant. *B. maritimus* planting was extremely successful although planting density appeared to be critical to the speed of expansion over the growing season and in subsequent years. Two of the three species planted were not successful however, though a few of the propagules of *P. maritima* have survived but remain extremely limited.

Saltmarsh vegetation was established at the forefront of eroded marsh and degraded upper mudflats, which suggests that there was nothing inherently wrong with the environment, despite the dominance of erosional features in the sediment bed. The results suggest that direct planting of an appropriate saltmarsh species may be a viable option to

restore saltmarsh habitat on degraded estuarine shorelines and, in this study, enhanced sedimentation was a direct consequence of planting.

6.2 Saltmarsh sedimentation and sea level rise

A lack of sediment may be responsible for saltmarsh decline (Jones et al, 2012), but the condition of the marsh most probably compromises its sediment-trapping ability, rather than a general absence of sediment in the estuarine waters. The natural but eroded stands of *P. maritima* measured during this study clearly did not function as a sediment trap, yet the dearth of accretion may have a strong influence on whether the marsh will be able to respond positively to rising sea levels. Accretion rates in other UK areas of saltmarsh are generally less than 10 mm per annum (Table 6.1). However, French and Burningham (2003) calculated that an accretion rate of 3 to 5 mm per annum was enough to offset regional subsidence of 1 to 2 mm per year and the current rise in relative sea level of 2.4 mm per year in the south-east of England.

Table 6.1 Annual accretion rates (mm) reported in the literature regarding British salt marshes.

Location	Position/type	Annual accretion (mm)	References
Eden Estuary	Natural B. maritimus Natural *P. maritima* *B. maritimus* transplants	3.4 0 3.8	Maynard et al. (this study)
Tay Estuary	Reed beds	0.58	Alizai & McManus (1980)
Humber Estuary	Horseshoe Point	9.7 – 11.3	Mohn-Lokman & Pethick (2001)
Humber Estuary	Skeffling	4 – 5	Brown (1998)
Norfolk	Backbarrier	3.9	French & Spencer (1993)
Norfolk	Stiffkey	3.08	Boorman et al. (1998)
Essex	Tollesbury	4.27	Boorman et al. (1998)
Thames Estuary	Mature marsh Immature marsh	2 – 3 3 – 5	van der Wal & Pye (2004)
The Solent		4 – 5	Cundy & Croudace (1996)
Severn Estuary	Mid marsh	4.65	Allen & Duffy (1998)

The stands of *B. maritimus,* either natural or transplanted, in this study accreted sufficient sediment in one year (3.4 and 3.8 mm, respectively) to keep pace with the more recent and general estimates of 0.9 to 1.2 mm annual sea level rise around the British coast, although mean sea level rise in Scotland may not continue to be moderated by isostatic uplift (Ball et al, 2008; Werritty, 2012)**.** The accretion rate should also be more than adequate to match the 0.7 mm annual rise observed for the east coast of Scotland. However, matching sea level rise may not be possible for the natural *P. maritima* communities, even though they were subject to a similar wind and wave climate and shared the same elevation as the natural *B. maritimus* stands. Importantly, transplanted *B. maritimus,* especially after 4 years' growth, enhanced sediment deposition and demonstrated that the sediment trapping function of the saltmarsh habitat within the estuary has the potential to be restored.

Sea level rise and climate change will have an effect on saltmarsh distribution, through increased depth, frequency and duration of flooding. Changes to wind and wave conditions could also increase wave and tidal energy impacting upon the marsh, in addition to an increase in storm incidence and severity. However, this may not be because saltmarsh vegetation cannot adapt to a changing environment but because local conditions at eroded marsh fronts or unvegetated mudflats in front of hard sea defences may not present the opportunity for natural colonisation through seeding because of the increased wave and tidal energy associated with such degraded shorelines. Most saltmarsh seeds require a

substantial drawdown of tidal flooding during springtime for germination to occur. The young seedlings then require sedimentation and stability; these conditions for colonisation appeared not to be met in the sites studied. Clonal expansion, being a slow process anyway, also would be hindered.

Because of the partial closure of the mouth of the Eden Estuary at Outhead, caused by the dumping of municipal waste in former times, shore levels within the estuary have fallen (Crawford, 2008). During the late 1940s, Professor Graham attempted to correct this fall by the planting of *S. anglica*, and by 1970, there was a noticeable raise in the shore level (Crawford, personal communication). These original beds of *S. anglica* have subsequently been removed. While this current study has limitations, it has shown that other native plants, such as *B. maritimus*, may be able to take the place of the invasive species *S. anglica* to prevent further spread predicted to occur with a warmer climate. The direct planting of *B. maritimus* will also repair a degraded but highly valued British Action Plan habitat and enhance the wildlife carrying capacity of the entire estuarine ecosystem. More importantly perhaps, from the coastal landowners' perspective of coastal flooding and erosion, the study has shown that despite active erosion and a lack of sedimentation it has been possible to stimulate sedimentation and saltmarsh development in degraded areas. This will buffer unprotected shorelines and help to raise shore levels, a process promoted in the past by Professors Graham, Crawford and McManus as vitally necessary if the impact of sea level rise and increased

storm activity in the Eden Estuary is to be counterbalanced in the future.

6.3 From small trials to long sections of coastline

It became evident that the findings from this study could be applied on a larger scale, partly because of the unanticipated longevity of the study but also because of the particular success of planting site 1 in front of the rubble tip on the north shore and planting site 8 on the south shore. The growth and sediment accretion in these sites rapidly increased during 2008 and 2009, the south shore site especially developing a very distinct raised mound of sediment under the vegetation compared to the adjacent, flatter and still eroding mudflats. The eroded seaward front of the *P. maritima* marsh behind planting site 8 had resumed growth over the bare sediment and active sedimentation was occurring between the broken and scattered fragments of marsh. Instead of an eroded step up onto the marsh body, a gentle incline formed, especially compared to those parts of the eroded saltmarsh without a planting site to afford protection from wave and tidal energy.

A joint effort got underway in 2010 to apply the findings from this study into larger sections of degraded shoreline, thereby tackling the joint concerns of nature conservation and coastal erosion and thus taking direct action using natural materials and resources being increasingly promoted by the Scottish Government, especially in the Climate Change

Act and in UK flood defence plans. Stakeholders such as the St Andrews Links Trust, RAF Leuchars and the Scottish Environment Protection Agency, funded the establishment of approximately two kilometres of *B. maritimus* saltmarsh in front of the entire length of rubble tip on the Eden's northern shore (planting site 1) and a seawall on the estuary's southern shore. These newly created marshes (Figure 6.1) appear to be following a similar trajectory to the original planting sites. For example, plant height growth and expansion rapidly increased during the second and third year post planting, and sedimentation lagged behind plant growth but increased steadily also. Benthic invertebrate abundance and the natural colonisation of other plant species such as *Zostera* spp. and *Plantago* spp. within the newly planted areas also increased over time. In keeping with other studies that show that young, actively developing saltmarshes have a higher rate of carbon sequestration than older and mature marshes, primary productivity measurements ongoing within the new sites appears to be greater than that in the adjacent, mature marshes and bare mudflats (C. Gollety, personal communication).

Figure 6.1 Two linear kilometres of saltmarsh were planted in front of a seawall (top image) and the rubble site (bottom image) between 2010 and 2014.

A storm in March 2010 demonstrated how urgently this practice needs both further implementation and greater exploration when a combination of onshore winds and extreme high tides (over 6.0 m Chart Datum) breached two 20 m wide sections in one of the Eden Estuary's old sea defences (Figure 6.2). The storm however, did no measureable harm to the natural marshes or to the created marshes adjacent to the embankment. Moreover, the storm failed to uproot plantings that had been planted only a few days prior.

Figure 6.2 One of the two breaches created by a storm in March 2010 to the old embankment on the south shore of the Eden Estuary (photo courtesy of RMM Crawford).

Assuming a cost of £5,000 per metre for a new seawall, without a buffer of saltmarsh (Table 2.3; Möller et al, 2001), then rebuilding just one of these breaches can be calculated at costing £100,000. Protecting shorelines with hard sea defences is therefore expensive, and estimates can currently cost between £3 and £5 million per kilometre. It is also likely that these costs will rise in the future. Building newer and higher defence structures would also result in continued coastal squeeze, and would impact on the nature conservation assets, landscape and wildfowling interests within the Eden Estuary.

By working with landowners, land managers and statutory agencies, the ongoing restoration initiative within the Eden Estuary has shown that the majority of the hard defences around its shores can be protected by a buffer of saltmarsh, using native marsh species already present. This practice may complement the natural expansion of *B. maritimus* that may accelerate with increased rainfall, freshwater input and warming temperatures predicted in the future. It is likely that a marsh buffer planted in front of the old embankment that was breached would have gone some way to reducing storm damage, and further damage in the future. It also establishes an integrated approach to coastal zone management that combines the necessary evil of hard sea defences, with those of soft engineering and better use of the natural resources already extant in the estuary.

6.4 Future research and directions

There were limitations to this study and many lessons were learnt. First, longer term monitoring of sediment accretion rates should be in place at the outset. The sediment study here encompassed only one year but there can be a great deal of variation in sediment accretion rates between years and during stormy weather. For example, on one such occasion during this study, heavy rainfall that occurred during a low tide caused the exposed sediment surface at one planting site to become 'pockmarked'. However, this loose sediment was then transported directly onto the eroding saltmarsh behind the planting site (personal observation) and suggested that an extreme weather event may accrete and not only erode sediment. The process of sedimentation on a saltmarsh surface is therefore complex and requires further elucidation, especially here on the east coast of Scotland where there is a shortfall of research into saltmarsh sedimentation. In order to try to correct this dearth, ongoing research within the Eden Estuary has observed both plant growth and sediment accretion from the outset.

Other planting techniques should be explored, both in the field and greenhouse, because substantial areas of donor marsh may not be available for the intense harvesting that would be required for longer lengths of shore. The practice therefore may not be sustainable. Marsh creation in other parts of the world are relatively advanced by comparison, e.g., commercially-grown greenhouse propagules, acclimatised to appropriate field conditions, are used during saltmarsh creation schemes

as standard practice, where the cost of greenhouse propagation is offset by the increase in the number of propagules available for planting out in the field. The use of well-established and acclimatised cuttings also means that the planting window is not limited to the winter dormancy period.

The expansion of the natural populations of the brackish saltmarsh communities within the Eden should be monitored also. The spread of *B. maritimus* in recent years over eroded *P, maritima* saltmarsh on both the north and south shores has been noticed in particular (Figure 6.3). This also appears to be occurring in some places on the southern shore of the Tay Estuary (personal observation). The marked increase in rainfall on the east coast of Scotland over the last few years (Jenkins et al, 2009) may well be responsible for this change in species distribution. The natural expansion of *B. maritimus* is not only altering species distribution but may help to limit the spread of the invasive *S. anglica* because the shading effects caused by the height and density of *B. maritimus* stands appear to limit the colonisation of *S. anglica*. However, specific vegetation area sizes were last reported during a survey in 1984. Any future work should repeat this survey to measure the extent of change in these major habitats. The character, extent and morphology of the eroded saltmarsh habitats in the estuary should also be monitored. The percentage coverage and species composition could be assessed with the use of annual surveys during the height of the growth season, supported by fixed-point photography and quadrat sampling taken in the same place each year. The extent of at least two of the larger saltmarsh sites on each shore within

the estuary should be monitored, defined from fixed points that could be used to monitor erosion or accretion rates in important areas.

Figure 6.3 The natural spread of B. maritimus encroaching over eroded P. maritima habitat on the southern shore of the Eden Estuary.

P. maritima die-back and its possible regeneration also needs to be explored further, because concern has been raised that the Habitats Directive, in terms of replacing 'like for like' saltmarsh habitat, where the quality of the existing resource in terms of community and species diversity should be maintained, is not being met in many managed realignment sites (Mossman et al, 2012). This criticism may be valid for the mono-culture planting of *B. maritimus* because, compared to *P.*

maritima, the natural stands of *B. maritimus* exclude other higher plants through shading effects and so floristic diversity may be greatly reduced (personal observation). However, Eelgrass (*Zostera* spp.), a priority BAP species whose abundance was greatly affected in the Eden through pollution and disease, has established in the new *B. maritima* stands, though the extent and duration of these mats has yet to be determined. Though *B. maritimus* may reduce overall floristic diversity, the new stands provide a habitat for increased marine invertebrate diversity, and will provide a high tide refuge for the waders and wildfowl that feed and roost on the estuary, which may act as a counterbalance to some extent.

One of the great attractions of the successful planting strategy used in the Eden to such good effect (Figure 6.4) is its wider applications in other coastal systems such as the Tay and Cromarty Firths, and the Montrose Basin. According to the UK BAP for coastal saltmarsh, there should be no further net loss of saltmarsh from the amount that existed prior to 1992. Enhancement of saltmarsh habitats can help contribute to targets set by the UK Biodiversity Action Plan (BAP), and to establish natural ecosystems and compensate for loss of conservation areas on a like-for-like basis as required by the EU Water Framework Directive. Like the Eden, most saltmarsh habitat in the estuaries of east Scotland is degraded. Letting this continue will affect the wildlife carrying capacity of these estuaries, such as bird feeding and numbers, but will also lead to a failure to adopt and work towards the Water Framework and Habitat Directives, British Action Plan targets, flood defence legislation and

Climate Change Act, and do nothing to further the use of natural resources to prevent coastal flooding and erosion of valuable land around the shores of estuaries.

Figure 6.4 A thick stand of B maritimus (2012) has raised the shore level, repaired an eroded saltmarsh and thickened the saltmarsh buffer in front of the world-famous golf links of St Andrews.

CHAPTER 7 CONCLUSIONS

In the Eden Estuary land reclamation, rubbish dumps, seawall construction, pollution and the introduction of invasive species have had a negative effect on the saltmarsh communities and likely to have interrupted saltmarsh colonisation and development. Rising sea levels against a coastline surrounded by man-made structures may further increase the loss of saltmarsh habitat and without the saltmarsh buffer even more pressure will be placed on the hinterland.

Brackish swamp communities are also prevalent in the estuaries of Eastern Scotland and within the Eden these communities are limited in range but apparently healthy. Increased amounts of freshwater into the Eden due to increasing rainfall suggested it could be possible to directly plant these species into sections of degraded shoreline and re-introduce the process of marsh development and sedimentation.

Vegetative propagules of *B. maritimus* planted at high density in springtime were successful. Sediment deposition and accretion was stimulated and began to bury waste materials and repair other, eroded saltmarsh. Previously eroded and degraded sites were transformed into actively accreting and developing marsh areas. Areas of research that still need to be addressed are donor marsh size and the limits of harvesting, the use of other species to enhance biodiversity and different methods that could be employed in more wave exposed shores. Adopting the practice has brought environmental

improvements to the Eden Estuary's shoreline such as increased carbon storage and the restoration and enhancement of a BAP habitat, currently classified as 'unfavourable'. The reduction in the erosion of hazardous waste will increase public and wildlife safety around the estuary, while longer term economic benefits will be gained by protecting the existing defences and rubbish tips with a thriving and actively developing saltmarsh.

The direct planting of saltmarsh vegetation on degraded shorelines within estuaries has the potential to be a rapid, relatively inexpensive and sustainable management option that integrates the needs of wildlife conservation with coastal protection strategies, while balancing any negative effect of further sea level rise by raising shore levels and increasing shoreline resilience. The strategy has wider applications within other estuaries such as the Tay and Cromarty Firths, and the Montrose Basin.

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Accelerating saltmarsh formation on the Eden Estuary, Fife.

C. E. Maynard

Soft coastal environments such as saltmarshes and mudflats form frontline protection of estuarine margins but their integrity is under threat as rising sea-levels cause and exacerbate erosion and flooding events. Marsh planting is a relatively inexpensive, flexible and self-sustaining solution, but to realise its potential there is a need to develop methods of encouraging native pioneer vegetation to establish and ensure optimal sediment entrapment.

The common reed *Phragmites australis* and the sea Club-rush *Scirpus maritimus* were seeded and transplanted in a range of sites on the north shore of the Eden estuary to compare differences in establishment and growth. Several clear outcomes were defined in the first growth season. *S. maritimus* outperformed *P. australis* and although both species were successfully propagated from sprigs (cuttings), neither species succeeded in germinating from locally produced seed. Comparisons were also made between planting date and density; sprigs planted in spring were more successful than those planted in autumn, and high density planting had more success than low density planting.

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Introduction

In an ideal world saltmarshes should provide protection around estuarine margins by dissipating wave and tidal energy, stabilising the substrate and accumulating sediment to build soil structure and volume (Brampton, 1992). Formerly, the north shore of the Eden estuary supported a large and thriving marsh, dominated by the common reed (*Phragmites australis* (Cav.) Trin. ex Steudel) and the sea club-rush (*Scirpus maritimus* L.) (Wilson, 1910). Unfortunately, marsh development was disrupted and an overall reduction occurred when waste was tipped onto the marshes and the upper tidal flats by land reclamation in the 1940s (Crawford, 2001). As a result, the present marsh is only a narrow and fragmented buffer to the low-lying cliffs of sand and clay that form the hinterland.

Although small-scale growth and recovery is apparent in some of the stands that form the marsh area, active erosion is causing other stands to retreat and expose the industrial and military waste. Unfortunately natural regeneration appears insufficient to overcome the physical damage, and allowing the marsh to degenerate further will compromise its role in protecting the cliffs. In other areas, gaps between the marsh stands are leaving the cliffs open to wave attack and the unstable waste and rubble is being carried onto the marsh and mudflats. Given predicted sea-level rise and associated increase in erosion and flooding events (Smith, 2001), these problems will be exacerbated and any further losses to the marsh will result in the need to protect the coastline.

The most apparent solutions in coastal protection however are often not viable. For example, both hard and soft engineering solutions, such as gabions and sediment recharge, can be cost-prohibitive (Brampton, 1992; Valverde *et al*, 1999). Managed retreat, currently being developed in the south of England

(Brooke, 1992, Burd, 1995), is not possible on the north shore of the Eden because no land is available for the retreating plant communities. A more practical solution is marsh creation whereby pioneer vegetation is physically transplanted in front of an eroding marsh either to aid its recovery or replace the marsh (King & Lester, 1995). A relatively simple and low-cost practice, marsh creation has been implemented in other parts of the world, e.g., large-scale, mechanically-planted *Spartina* meadows have been created in the Gulf marshes of the U.S.A. (Boesch *et al*, 1994). However, in Britain this form of coastal defence is commonly overlooked and so very little is known about the effectiveness of planting different, native pioneer plants to accelerate colonisation and halt or reverse erosion (Boorman *et al*, 1989).

The marshes on the Eden estuary provide the ideal opportunity to redress the balance for two reasons. First, the two species that form the pioneer plants are common in many other British estuaries (Burd, 1989); indeed *S. maritimus* is one of the most ubiquitous species on temperate marshes in the northern hemisphere (Broome *et al*, 1995, Kantrud, 1996, Yang, 1999). Secondly, lessons from the United States have shown that success is more likely when the planted species is from the same or similar environment (Lewis, 1982). That the currently thriving stands provided a local source of material could have increased the chance of success in these experiments.

From a logistical standpoint the marsh creation techniques used in the USA are not necessarily applicable in Britain, and the range of factors that need to be determined include propagule type, planting season and planting densities. For example, seeds or sprigs (cuttings) can be used as a propagule but seeding is more cost-effective and less labour intensive than sprig planting (Lewis, 1982).

Conversely, while sprig planting is labour intensive, it has been shown that adult plants and vegetative propagules are more resilient to extreme environmental conditions than their respective seedlings in both *P. australis* (Wijte & Gallagher, 1996) and *S. maritimus* (Lieffers & Shay, 1982) and success is therefore more likely with sprigs. Planting success may also depend on whether planting is undertaken in autumn or spring because, although planting of sprigs and seeds in spring may allow the plants to take root before winter storms, seed release occurs in autumn and over-wintering in the sediment may be a necessary precursor of germination.

Planting density is also important for optimum plant growth and sedimentation. Lewis (1982) recommended 100 seeds per square metre as suitable to overcome natural seedling mortality, but recent work by Hughes (1999) has shown that benthic invertebrates can damage and/or devour large quantities of seeds and seedlings, suggesting a higher density may be more appropriate. Sprig density can range from 1 per square metre (Lewis, 1982) to 20 per square metre (Clevering, 1997) but none of these studies were related to sedimentation rates or the speed and success of plant colonisation.

To assess the use of marsh creation as a coastal defence option in the Eden estuary, it was first necessary to develop the methods required to stimulate marsh growth. The aims of the project therefore were to investigate the effect that propagule, season and density had on planting success in the two species, *P. australis* and *S. maritimus,* to determine which treatment or combination of treatments provides the most suitable means by which to promote marsh regeneration.

Methods

Trial sites on the north shore embayment of the Eden estuary in SE Scotland were chosen at the seaward edge of eroding marshes and in the gaps between the stands of remnant marsh (Fig. 1). All the sites and the existing marsh were at a similar tidal elevation and subjected to flooding during mean high water.

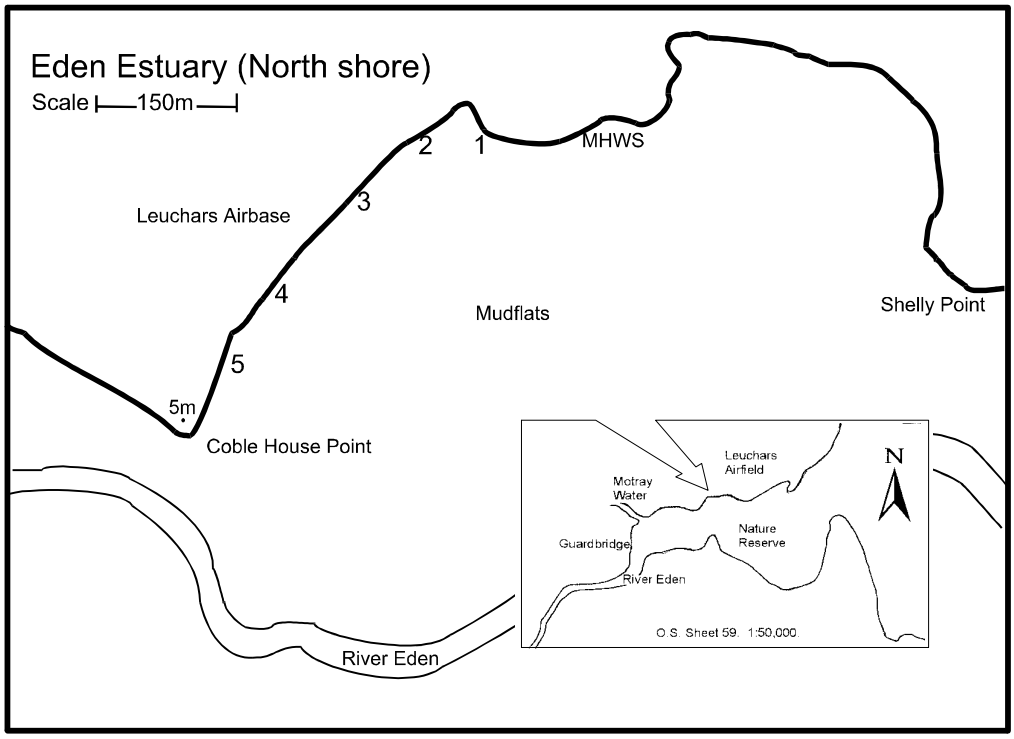

Fig. 1. *The section on the north shore embayment of the Eden estuary, SE Scotland, where planting experiments are underway. Mean High Water Springs (MHWS) and sites 1 – 5 are shown. 5m corresponds to the height of the cliff face.*

At each site, plots measuring 5 by 2 metres were marked out with bamboo stakes and a single treatment was assigned to each plot. The treatments included the sprigs or the seeds of *P. australis* or *S. maritimus*. These were planted or sown in autumn (October 1999) and the following spring (March 2000) at high and

low density (200 seeds/12 sprigs per square metre and 100 seeds/3 sprigs per square metre, respectively) for both planting seasons (see Table 1).

For each treatment the number of sprigs to develop a shoot, or seedlings to appear above the sediment surface, was compared to the total number of sprigs planted or seeds sown. Monitoring began in April 2000 and, over the course of the growing season, the number of buds to develop leaves and subsequently remain alive was determined.

Table 1. *The different treatments tested at sites 1 - 5. Note, P. australis was not tested at sites 3 and 4 and only spring planting was conducted at sites 3, 4 and 5.*

Site no.	**Species**	**Season**		**Propagule**		**Density**	
1	*S. maritimus*, *P. australis*	Autumn	Spring	Seeds	Sprigs	High	Low
2	*S. maritimus*, *P. australis*	Autumn	Spring	Seeds	Sprigs	High	Low
3	*S. maritimus*	-	Spring	Seeds	Sprigs	High	Low
4	*S. maritimus*	-	Spring	Seeds	Sprigs	High	Low
5	*S. maritimus*, *P. australis*	-	Spring	Seeds	Sprigs	High	Low

Seed sowing

Seeds were harvested from plant populations adjacent to the experimental sites during September 1999 and some used the following month. The remainder were dry-stored in cool, dark conditions until required (Lewis, 1982). To achieve a density of 100 or 200 seeds per square metre in 10m^2 plots required 1000 or 2000 seeds respectively. The seeds were mixed with dry sand to facilitate sowing and the plots were prepared by tilling the sediment to 2cm deep. After sowing, the

sediment was smoothed over so that the seeds were covered to the depth recommended for these species (Lewis, 1982; Clevering, 1995).

Vegetative planting

Sprigs were removed from plant populations local to the vicinity by excavating plugs of soil measuring approximately 20 x 20 x 30 cm. This depth incorporates all the roots and rhizomes. The plugs were then separated into single sprigs (a shoot with a developing bud, a rhizome and associated roots) retaining as much soil as possible to minimise transplantation shock. Immediately after separating, sprigs were replanted in the appropriate plots, leaving enough shoot above the ground to allow the developing bud to emerge.

Results & Discussion

Data for each experimental plot were combined to compare the percentage success between the different treatments. In the first instance seed success was compared to sprig success; Fig. 2 shows that for both species seedlings failed to appear and therefore had zero budding success. These results supported an earlier test undertaken to investigate metabolic activity in the seeds (Kuo *et al*, 1996) which showed that the seeds that were sown were not viable. However, seed viability can vary from year to year (Ungar, 1987) and, because seeds are a relatively economical means of propagation, seed viability (in the mature populations) will continue to be monitored over the duration of the project.

Sprig planting was successful for both species, but where bud regeneration in *P. australis* reached 34%, *S. maritimus* had nearly double the success at 64% (Fig. 2). *S. maritimus* occupies a broader range of habitats than does *P. australis*

(Haslam, 1971) which implies that *S. maritimus* is the more tolerant of differences in environmental conditions (Broome *et al*, 1995). Furthermore, other studies have confirmed that *S. maritimus* has a greater ability than *P. australis* to regenerate buds under a broader range of salt and anoxia regimes (Maynard, unpublished data).

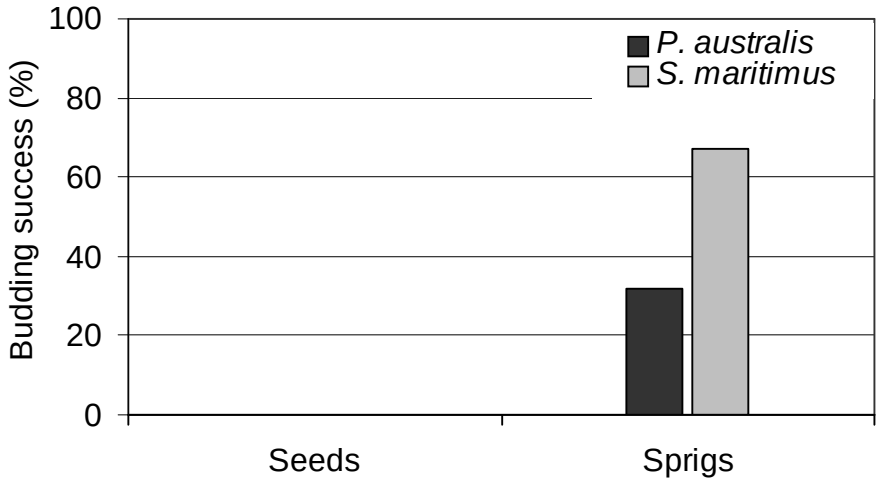

Fig. 2. *The results from autumn planting (Oct. 1999) and spring planting (Mar. 2000) were combined in order to compare the percentage budding success between seed and sprig propagules of P. australis and S. maritimus. Data collection commenced when the first bud emerged in April 2000.*

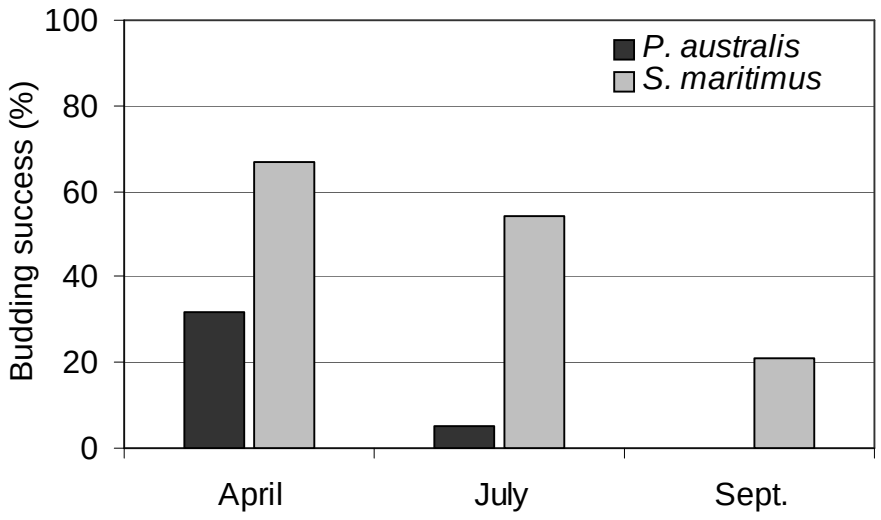

Fig. 3. *The results from both autumn and spring planted sprigs were again combined for each species to compare the differences in budding success (%) over the length of the growing season (April to September, 2000).*

Fig. 3 shows a downward trend in the percentage of successful sprigs to have remained alive over the course of the growing season. Buds of *P. australis* that had initially developed in April had died by September, whereas the number of successful *S. maritimus* sprigs was only reduced to 21% over the same time period. The mechanism of shoot elongation is very similar for both species; excess resources from summer growth are stored as starch in the underground stems and rhizomes. This food supply sustains initial shoot development after winter dormancy until photosynthetic tissue is developed. However, within a single stand individual shoots can be connected via rhizomes and it is thought that this facilitates access to the energy reserve of the whole stand (Koppitz *et al*, 1997). The decrease in the survival of the planted sprigs may reflect the point when the stored energy of the individual sprig was depleted, and photosynthesis of the new shoots was less than the respiration required by the whole sprig to survive. Thus, having no access to the energy reserve of a whole stand, the individual sprig dies.

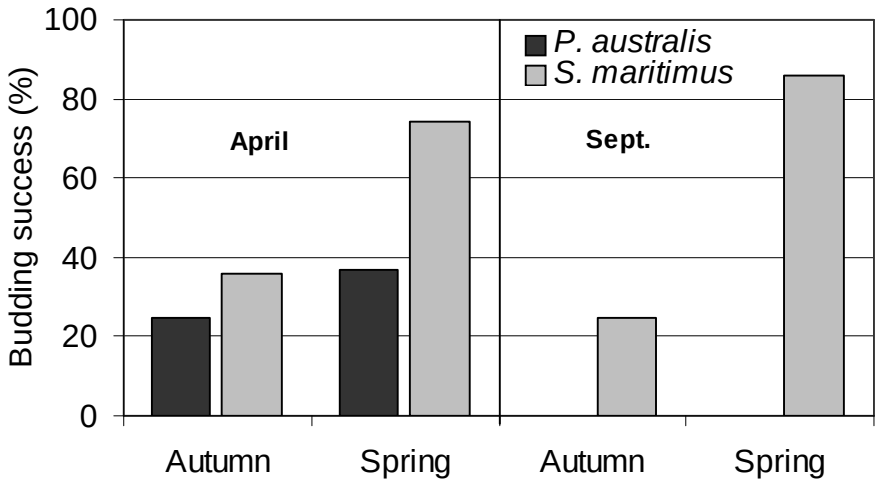

Fig. 4. *The differences in bud regeneration (%) for P. australis and S. maritimus between those sprigs planted in autumn (Oct. 1999) and those planted in spring (Mar. 2000). Initial budding was noted in April 2000 and data collection ended in Sept. 2000.*

Bud emergence for both species occurred during April 2000 but the percentage differed depending on whether the sprigs were planted in the preceding autumn (Oct. 1999) or spring (Mar. 2000) (Fig. 4). At the start of the growing season the autumn-planted sprigs of *P. australis* had 23% success compared to 38% for those sprigs planted in spring. In *S. maritimus* those sprigs planted in autumn showed 38% success as opposed to 77% for those planted in spring. At the end of the growing season however the success of *P. australis* sprigs had degenerated to zero but in *S. maritimus* it was noted that there was a net loss of autumn-planted sprigs to 23%, while those planted in spring had increased to 82%. It is known that breaking fragments off plants during dormancy can be a cue for bud development and growth (Charpentier *et al*, 1998). Sprigs planted in the autumn were removed from the parent populations and replanted before the onset of dormancy. Conversely, those sprigs planted in spring were

removed from the parent plant toward the end of the dormancy period, and could have been responding to this cue.

The effect that planting density had on bud success can be seen in Fig. 5. Neither species showed any difference between sprigs planted at high (12 per m^2) and low (3 per m^2) density during April 2000. At the end of the period of data collection (Sept. 2000) all *P. australis* sprigs had died giving zero success. However, bud success in the high density planting of *S. maritimus* sprigs had only decreased from 64% to 58%, whereas bud success in the low density plots was reduced from 68% to 10% survival.

It is known that *S. maritimus* can aerate the root zone via the shoots (Kantrud, 1996). A higher density of these plants may result in more oxygen in the sediment and the energy spent in coping with an oxygen deficit may instead be reallocated to growth. An increase in plant cover associated with a higher density may also provide greater protection from the physical impact of wave action (Clevering, 1997). The effect of planting density on the survival of the sprigs of *S. maritimus* is clear but ongoing trials hope to establish a more accurate number of sprigs required per square metre to ensure marsh success.

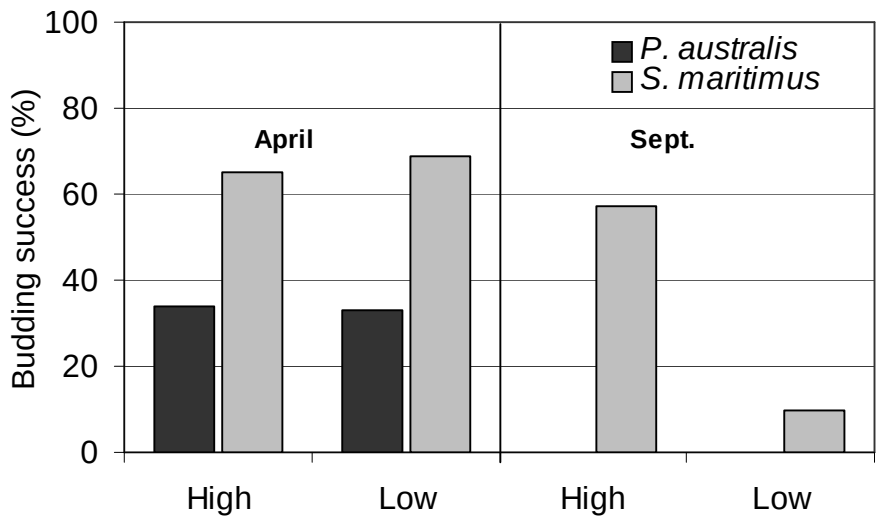

Fig. 5. *The effect of planting density on budding success (%) in P. australis and S. maritimus. Low density represents 3 sprigs planted per* m^2 *and high density 12 sprigs per* m^2*. Data collection commenced during April 2000 and ended in Sept. 2000.*

Conclusions

Within the Eden estuary the fringing and discontinuous saltmarshes vary from stable, flourishing communities to unstable patchy remnants, many of which are undergoing active erosion. However King and Lester (1995) claim that the erosion of a saltmarsh can be arrested if a suitable body of plants is found to attempt regeneration. The work reported here examines the possibility of using locally derived *P. australis* and *S. maritimus* as suitable species and compares the potential of each under similar conditions.

Experiments with these two species have shown that *P. australis* is the least suitable and that *S. maritimus* is the most suitable plant to stimulate marsh regeneration in this area. Finding the best method of propagation was also necessary and in this respect, vegetative planting was more successful than seeding. Timing of planting was also crucial and planting before the winter was less successful than in the spring. These results suggest that to realise the

potential of marsh regeneration on the Eden, planting *S. maritimus* sprigs in the spring is the most effective method. However, higher density planting achieved greater success than low density planting and because this factor also has the greatest bearing on sedimentation rates (Alizai & McManus, 1980; Maynard, unpublished data) further analysis is required.

Natural marsh regeneration after managed retreat programmes in SE England has so far been largely unsuccessful, with notable exceptions (Crawford, 2001). In an attempt to increase marsh area it may be advisable to aid the spread of the plants by transplanting suitable species as a means of speeding up the process. The saltmarshes on the Eden have provided the ideal opportunity for studies designed to increase marsh cover as a means of combating both the physical damage that they have suffered in the past, and any further damage likely to be exacerbated by sea level rise.

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A review of the saltmarshes of the Eden Estuary, Fife, Scotland: status, function and management

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List of figures

1. Introduction

The saltmarshes on the Eden Estuary are relatively small but unique because they represent a transition between northern and southern marsh types found in the UK (Leach & Phillipson, 1985, Crawford, 1998). Botanically important, the Eden's marshes also contribute to the overall functional value of the estuary and like all saltmarshes serve as wildlife refuge, coastal flood defence, pollution filter and carbon sink (Allen & Pye, 1992, Boorman, 1999). The importance of saltmarsh in these respects is recognised by current protection measures. In the past and particularly in the most populated estuaries, land reclamation greatly reduced the amount of UK saltmarsh. What remains on the seaward side of enclosed land has since undergone rapid erosion and die-back, with the greatest losses recorded in south-east England (Pye & Allen, 2000). Likewise, saltmarsh in the Eden was estimated to have eroded by a metre per year during the early 1990's (Hatton, *pers. comm.).* Although at present the rate of erosion is thought to be slower, the majority of the Eden's marshes remain severely damaged (Fig. 1) and it is doubted whether they will be adequate to withstand any increase in tidal energy from accelerated sea level rise (Crawford, 2008). Given the current fragmented state of the marsh, not only will further loss compromise its ecological value, but it will also increase the future cost of coastal protection. This report describes the current status of the Eden's saltmarsh in order to highlight viable and sustainable methods that could reverse the process of erosion and help stimulate marsh development and restoration. The maintenance and if possible restoration of the shore level in the Eden Estuary is also vital for the protection of the West Sands and its world famous golf courses.

Figure 1. Extreme lateral retreat and cliff failure caused by erosion on saltmarsh at Kincaple Flats, south shore of the Eden Estuary.

2. Current status of the Eden saltmarshes

Saltmarshes are the natural habitat of upper tidal flats in temperate regions around the world (Adam, 1990). A relatively rare habitat, the UK has only *c*44, 000 ha of saltmarsh (compare with 350, 000 ha of ancient semi-natural woodland), the majority of which is in the low-lying soft shores of southeast and northwest England (Fig. 2). Only 3% of Scotland's mainly hard, rocky coastline is given over to saltmarsh, with the largest concentrations found in eastern and south-western firths (Burd, 1989).

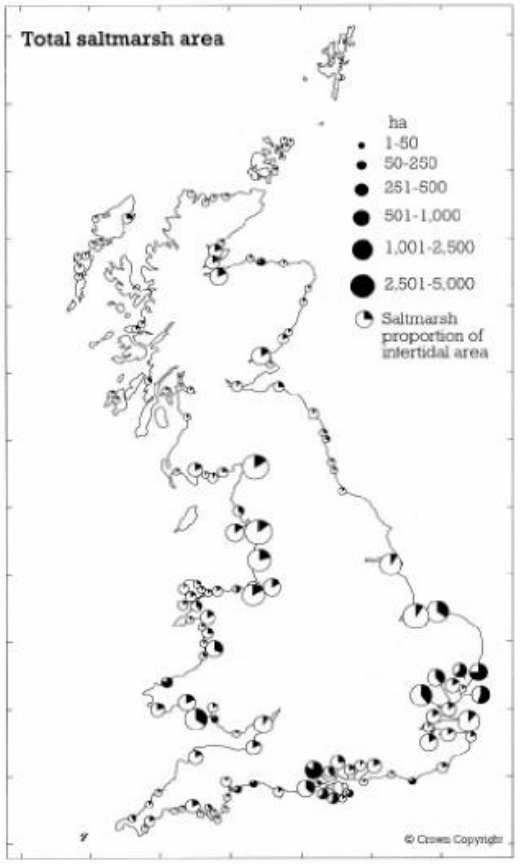

Figure 2. Saltmarsh distribution in the UK (Davidson et al, 1991). The size of the symbol shows the area of saltmarsh and the filled segment is the proportion of saltmarsh within the total intertidal area at each site.

The Eden's saltmarsh contains characteristics of two biogeographical zones in the British Isles; that of the southeast and the northwest (Crawford, 1998). At its widest the estuary is only a mile or so across, and yet the north shore has a greater pre-dominance of the reed bed communities more typical of the wetter northwest, while the south shore has a composition more typical of marshes on the south and east coast (Fig. 3). For example, in south-eastern parts of Britain, Common Reed (*Phragmites australis),* Sea Club Rush (*Scirpus maritimus)* and Bulrush (*Schoenoplectus tabernaemontani)* are high marsh transitional species in brackish to freshwater conditions (Burd, 1989). On the north shore of the Eden these brackish plant communities extend out into bare mudflats, i.e., into the lowest growing pioneer zone. Soil salinity is lower on this shore of the Eden than would normally be expected because of the generally lower farming activity on the adjacent land increasing freshwater drainage, in addition to the high clay composition of the sediment which serves to impede drainage. The local climate also produces an early morning haar over the estuary during the

growth season and further ameliorates soil salinity. On the south shore these reed bed communities are present only in the inner and less saline regions of the estuary whereas the mid estuary saltmarsh resembles marshes in the south east, e.g., the mudflat pioneer zone is occupied by the salt-tolerant Glasswort (*Salicornia europaea),* succeeded landward by saltmarsh grass a foot or two above these mudflats. The presence of the golf links indicates that the natural habitat landward of the marsh is sand dune and therefore in keeping with southerly and Mediterranean types of saltmarsh defined by Adam (1978).

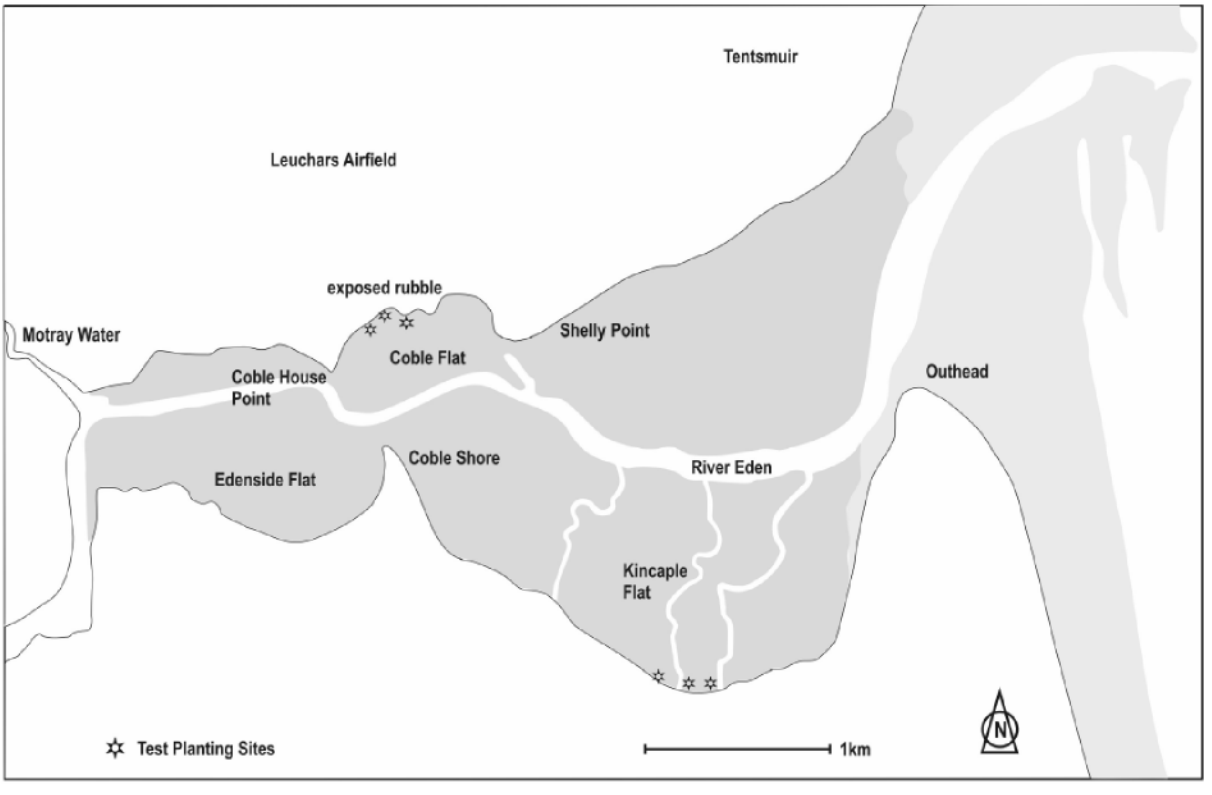

Figure 3. Map of the Eden Estuary showing the location of the trial marsh planting sites.

3. The importance of saltmarsh

More than 80% of saltmarsh in the UK is protected with SSSI status (Davidson *et al*, 1991) as it can be one of the most natural ecosystems when enclosure and intensive grazing have not occurred (Burd, 1989). More recently saltmarsh has been protected by the EU Birds Directive (79/409/EEC) and Habitat Directive (92/43/EEC). Saltmarsh fulfils many functions (Table 1) but particularly sustains many bird populations, providing a high tide refuge for birds that feed on the mudflats, a breeding site for waders, feeding grounds for geese during winter, and

in addition feeding for a wide range of passerines and birds of prey (Davidson *et al*, 1991). It plays a key role in the cycling of organic material and nutrients important for the marine food chain (Nedwell, 2000, Boorman, 2000) and the continued survival of many specialist salt-adapted plant species are dependent on saltmarsh (Crawford, 2001). The saltmarsh also plays host to a wide range of invertebrates, ranging from terrestrial insects and arachnids to marine molluscs and bivalves (Foster, 2000). The richest areas for terrestrial invertebrates tend to be where saltmarsh grades into other terrestrial habitats (Kinnear, 1996) because of the high floristic and structural diversity (Adam, 1990).

The use of saltmarsh for wildlife and conservation indirectly benefits society both through the food chain and in aesthetic appeal. However a more direct feature of saltmarsh is the stability it brings to the estuarine margin, especially when shoreline development is significant. Saltmarsh has the ability to promote sediment accretion, resist wave energy, withstand storms and help prevent erosion (Brooke *et al*, 1999). These abilities mean the entire marsh acts as a buffer to the coastline (Brampton, 1992). King and Lester (1995) estimated that if the saltmarsh along the entire coast of Essex were removed, rebuilding sea defence walls to replace their function would cost £600 million. Although no figures have been calculated for the value of Scotland's saltmarshes in this respect, and despite their relatively smaller total area, it is probable that the cost of replacing them would also be considerable.

Table 1. Saltmarsh services of direct benefit to wildlife or part of the overall function of the wider estuary.

Immediate wildlife benefits	**Wider estuarine function**
High tide refuge for waders	Shoreline stability
Breeding sites for range of birds	Sediment accretion
Feeding ground for geese	Wave attenuation
Fish spawning/nursery	Nutrient/organic matter source
Marine invertebrate habitat	Absorbs excess water run-off
Specialist plants	Pollution trap
Insect/amphibian habitat	Recreation/leisure

4. Saltmarsh formation

Saltmarsh generally forms in quiet, wave-sheltered areas, to such an extent that more than 90% of saltmarsh habitat occurs in estuaries (Davidson *et al*, 1991). Intertidal sand and mudflats are initially stabilised by the binding action of surface algae, e.g. diatoms and *Enteromorpha*, but the first flowering plant, the Eelgrass (*Zostera* spp.), only colonises the mudflat when the height of the sediment exceeds mean high water neaps (Fig. 4), where its essentially aquatic nature can tolerate the high salinity and physical movement of each tide (Adam, 1990). Flowering plants with a terrestrial form, like Cordgrass (*Spartina* spp.) and Glasswort (*Salicornia* spp.) colonise slightly higher up the tidal frame than eelgrass. These colonists maintain surface stability through root mats and by anchoring the sediment (Adam, 1990) while the plant stems reduce water velocity (Moller, 1999) and cause sediment to drop out of suspension from the overlying tidal waters (Reed, 1999). Sedimentation increases over time, thereby increasing the height of the vegetated zone and reducing the frequency and duration of tidal inundation; as this process of accretion continues, transitional zones to terrestrial habitats develop (Adam, 1990). At present, this succession in saltmarsh development is rarely seen in UK estuaries (though there are notable exceptions, for example parts of Morecambe Bay on the northwest coast of England). The fact that there are few developing marshes left in the UK, combined with climate change and accelerated sea level rise, brings the continued survival of estuarine saltmarsh into question (Adam, 2002).

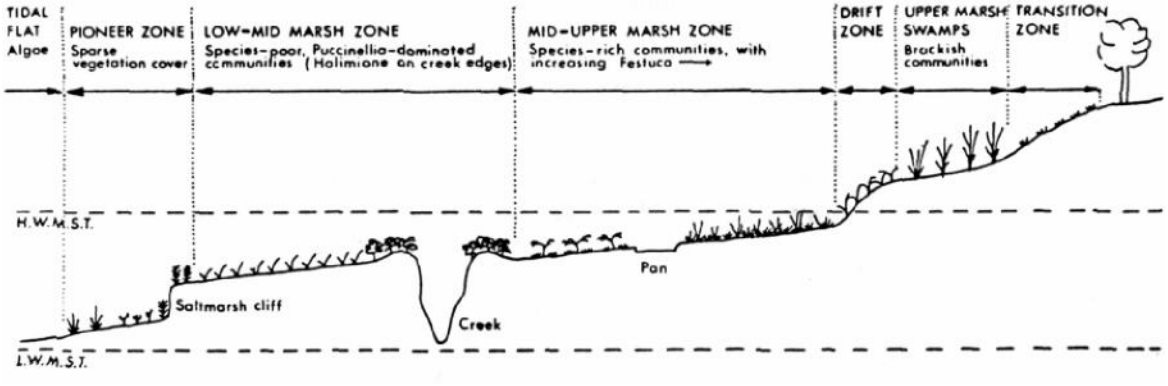

Figure 4. Generalised saltmarsh profile showing saltmarsh succession and main vegetation zones (Burd, 1989).

5. Saltmarsh loss

5.1 Land reclamation

Small-scale losses to saltmarsh have occurred with increasing farming activity since medieval times but over the last 200 years land reclamation and industry substantially increased the loss (Crawford, 2001). Documents relating to the earlier history of the Eden's saltmarshes are scarce, but maps provide some evidence that large scale changes to the surrounding area took place as early as the 1850's. For example, the Leuchars to St. Andrew's railway was constructed in the mid-1800s on an artificial embankment some 4.0 m above sea level, and would have cut off the low-lying hinterland on the south shore from tidal influence and allowed for greater farming activity. Maps dating from the 1880's also show a number of tile works around the head and inner parts of the estuary, implying the drainage and removal of marsh to extract the underlying blue clay. However, much of the natural landscape must have been preserved at the start of the 20th century, as Wilson (1910) noted that reed formed 'a jungle of considerable extent' around the confluence of the Motray Water, and that eelgrass (or grasswrack) 'clothes the mudflats in summer' In describing their use in the thatching, packing and stuffing industries, he implies a far greater abundance than is present today. He also describes 'salt-grass flats occupying a large space on solid dense blue clay a foot or two above the mudflats' on the south shore, with minimal erosion because of the shelter afforded by the estuary. It was shortly after this period that the higher ground of RAF Leuchars, above the north shore of the estuary, began to be used as an airfield during the First World War. The old runway was replaced during the 1940's, and the existing saltmarsh buried when the waste rubble was tipped (Crawford, 2001). Subsequently other sea defences were built around the estuary and currently more than 60% of its shoreline is artificially reinforced and the hinterland enclosed.

5.2 Sea level rise

Hard sea defences prevent inland marsh migration and combined with the post-1964 sea level rise of 2.4 mm per year has been blamed for the current erosion of estuarine saltmarsh, in a process known as coastal squeeze (Wolters *et al*, 2005*b*). This process is most pronounced in the south-east of England because isostatic tilting of the land increases the rate of relative sea level rise (Pye & Allen, 2000) (Fig. 5). Sea level rise *per se* does not necessarily cause saltmarsh

erosion, since saltmarsh has in the past responded to rising sea levels by expanding in sheltered estuaries (Crawford, 2008). In addition the accretion and consolidation of sediment in UK marshes is generally considered to be able to keep pace with current sea level rise (French & Burningham, 2003, van der Wal & Pye, 2004). Current marsh erosion is more likely the ongoing effect of the overall lowering of the shore profile that can occur because of land reclamation (Crawford, 2008). For example the availability of sediment in the Eden Estuary was greatly reduced when the river channel was narrowed by the growth of the rubbish dump at Outhead (Crawford, 2001), despite St. Andrews Bay being replete in sediment (McManus, 1998). The emplacement of sea walls is known also to rebound wave energy and reduce the supply of fine sediments to adjacent marshes (Bozak & Burdick, 2005). Low sedimentation rates on the fringing marshes in the Eden have been confirmed (Maynard *et al*, in review) and could result in restricted marsh development. The reduction in marsh area associated with land reclamation also increases the tidal range and energy impacting on the remaining marsh (Pye, 2000, van der Wal & Pye, 2004) which could further compound the low rate of sedimentation. The diversity of plant life necessary for the continuing selection and adaptation to changing environmental conditions is also greatly reduced in the fragmented marsh populations, and their ability to respond positively to rising sea levels is further compromised (Crawford, 2008).

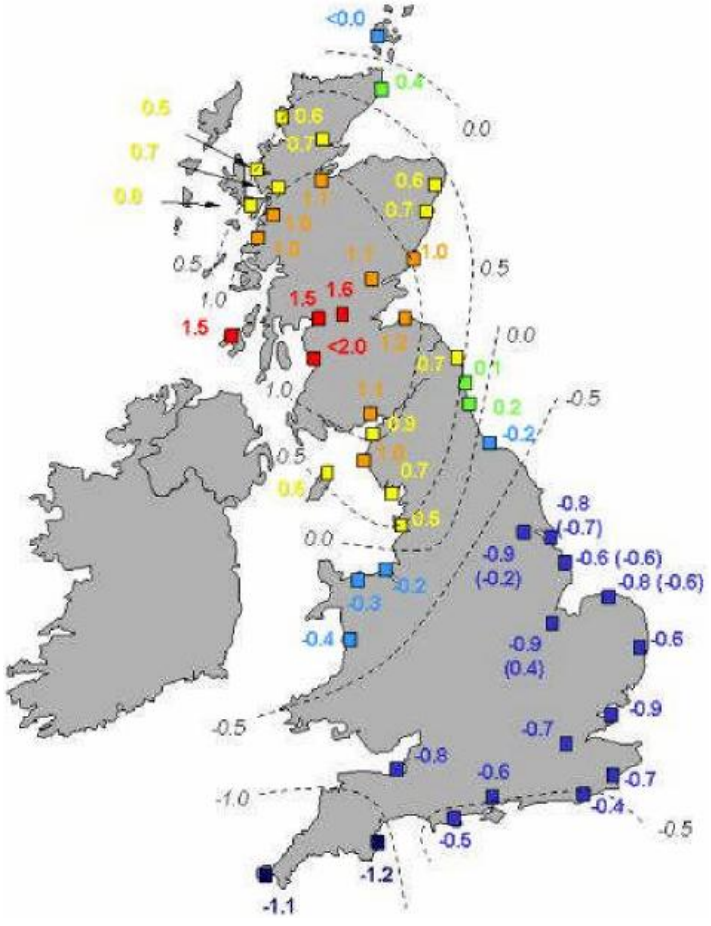

Figure 5. Late Holocene relative land/sea-level changes (mm yr) in Great Britain, positive values indicate relative land uplift or sea-level fall, negative values are relative land subsidence or sea-level rise (Shennan and Horton, 2002).

5.3 Pollution and invasive species

The Eden Estuary was declared a nitrogen vulnerable zone in the early 1990's because of the pollution caused by increasing industrial practices, intensive farming and population expansion (Clelland, 1997). It is likely that pollution also impacted the Eden's saltmarsh communities as saltmarsh die-back has been correlated with eutrophication and excessive algal growth in other estuaries (Turner *et al*, 2004). In southern estuaries heavy metals, found in saltmarsh sediments dating from the 1900's on (Adam, 1990) and herbicides (Mason *et al*, 2003) have also been linked to saltmarsh die-back. It is not known whether the Eden Estuary sediments contain large concentrations of heavy metals, but saltmarsh erosion is particularly worrying because any trapped metals can be released back into circulating waters (Boorman,

2000). The Common Cordgrass (*Spartina anglica)* is considered to be another threat to the long-term survival of native saltmarsh. This hybrid species is a vigorous grower and colonises mudflats lower on the tidal frame than any other native plant species. These abilities led to it being extensively planted as a prelude to land-claim in many English estuaries during the first half of the 20th century. It was also introduced to the Eden Estuary in 1948 and considered to be forming a stable shoreline (Crawford, 2008). Its fast growth and spread earned it a reputation for encroaching on wader and wildfowl feeding grounds and out-competing native pioneer plant communities; as a result it is systematically removed on an annual basis from most nature reserves (Lacambra *et al*, 2004). Although the natural colonisation of this species into other Scottish estuaries appears to be restricted because of the effect of cooler temperatures (Crawford, 1998), the Eden colonies on the south shore have colonised the marsh area behind Shelly Spit on the north shore (Strachan, *pers comm).* Given the warmer temperatures predicted with climate change, this species may yet naturally extend its range into more northerly latitudes (Gray & Mogg, 2001). However, in light of its ability to stabilise shorelines on otherwise rapidly disappearing saltmarshm the feared negative impact of Cordgrass on selected bird populations and native plants may need to be revised.

6. Saltmarsh management

According to the requirements of the UK Biodiversity Action Plan for coastal saltmarsh, in order that there should be no net loss of saltmarsh it will be necessary to create some 100 ha of new marsh per year (Huggett, 1999). Managed realignment aims to fulfil this goal directly by restoring marsh habitat on previously reclaimed land. Another option is to halt or reverse the process of erosion, and regenerate existing marsh, using marsh creation techniques developed in other parts of the world (King & Lester, 1995).

6.1 Saltmarsh restoration with managed realignment

Managed realignment restores former saltmarsh on reclaimed land and can also help to meet flood defence requirements, particularly in areas where coastal squeeze is most apparent (Burd, 1995). The first deliberately breached sites in the UK were on the Essex coast in 1991 (Northy Island) and 1995 (Tollesbury and Orplands) and these former reclaimed lands have since reverted to saltmarsh habitat. In Scotland, a seawall protecting low-lying farmland at

Nigg Bay in the Cromarty Firth was breached in 2003 and although plant colonisation was initially slow, viable saltmarsh communities have since returned (Crowther, 2007). These early realignment schemes have demonstrated that saltmarsh can naturally colonise breached sites as seeds and fragments are transported by high tides, especially during peak dispersal times in early autumn (Burd, 1995). The speed of recolonisation depends on a local source of marsh propagules available for dispersal but according to Wolters *et al* (2005*a*) there is no direct relationship to long-term successful habitat restoration. Therefore saltmarsh planting is recommended if the initial vegetation cover is low and rapid cover necessary. Some sites have also been susceptible to seed and seedling consumption from an over-abundance of soil invertebrates (Hughes & Paramor, 2004). However, Wolters *et al* (2005) argue that the effect of grazing by invertebrates on managed realignment sites has been insufficient to restrict plant succession. The physical attributes of the site are also necessary, for example, the existence or creation of slopes within a site is important to increase habitat diversity, because a uniformly horizontal topography can both lower plant diversity and increase site erosion. It is also recommended that in order to enrich the habitat the landward limit of the site should encompass a succession of transitional, terrestrial zones (Burd, 1995). Managed realignment is no longer in its infancy but despite its success, reinstating saltmarsh on reclaimed land is not always possible, especially when the land may be valuable. There is therefore a strong need to develop other methods to prevent further deterioration and enhance the buffering capacity of estuarine marsh.

6.2 Controlling saltmarsh erosion with replanting

The Corps of Engineers (US Army) used Cordgrass (*Spartina* sp.) to stabilise dredged spoil and create new marshes on both the East and Gulf coasts of the United States during the 1970's (Lewis, 1982, Knutson *et al*, 1990), as did the Chinese when they planted extensive mudflats on the Dongtai Peninsula prior to conversion to agricultural land (Chung, 2004). Here in the UK the deliberate planting of Cordgrass would not be desirable and provides an opportunity to explore the use of native marsh species to control erosion and restore estuarine saltmarsh. With this goal in mind, small-scale planting trials using locally-adapted saltmarsh vegetation were undertaken in the Eden Estuary in 1999 and again in 2003. These studies undertook trial plantings of Common Reed, Sea Club Rush and Saltmarsh Grass on the seaward edge of eroding marshes and at the foot of an eroding rubble cliff (Fig. 3). The initial

study ascertained optimum planting density, timing of planting and propagule type (Maynard, 2003). Further studies compared the sediment trapping ability of the transplants and eroding marsh stands and showed the rate of sedimentation is greater in the planted sites than on the marsh front; former erosion hollows and crevasses within and around the sites also disappeared (Maynard *et al*, in review). Ongoing monitoring at the various transplant sites demonstrates that marsh plants currently present in the estuary are adequate for the task of marsh regeneration. They also represent a more acceptable alternative to Cordgrass for shoreline stabilisation purposes.

6.3 Potential for further planting in the Eden

The rubble cliff on the north shore bounding RAF Leuchars, created during disposal of waste materials in the 1 940s, has been progressively degenerating ever since, exposing the cliffs at the margins of the RAF station to direct wave attack. East of Coble House Point the tidal flats frequently reveal quantities of broken china, glass bottles and items of military hardware. If further planting were carried out and marsh development stimulated at the base of the low cliffs, muds may accumulate and cover the existing debris on or near the surface, thereby providing additional protection and stopping the waste being transported by tidal action onto the mudflats. A small test planting site to the eastern end of the rubble cliff has already ascertained that the physical and chemical conditions of the site are sound (Fig. 6). Additionally the marsh that has developed from a nearby planted site on the western side of the embayment could be used as a source of propagules. Planting a belt of vegetation even some two metres wide could reverse the erosion on the disfigured stretch of coast. The success of other planted sites within the estuary indicate that more planting could be carried out in front of other eroding marsh, particularly on the south shore in the mid estuary. The front of the sea wall that protects the Eden Course, although currently devoid of vegetation, is another potential site. Attempts to regenerate marsh with remedial marsh planting are becoming more common using techniques developed in other parts of the world (Zedler, 2001). The trial plantings conducted in the Eden using native marsh plants were largely successful and employed relatively simple and low cost techniques. The study demonstrates overall the viability of further investigation into this practice.

Figure 6. A small section at the western end of the rubble tip on the north shore of the Eden Estuary; prior to planting in March, 2000 (top) and after the successful replanting trial.

7. Conclusions

The saltmarshes in the Eden are unique because they contain both northern and southern elements of the UK saltmarsh flora. However, like other developed estuaries, the remaining fragmented marsh bears witness to the impacts of land claim and hard sea defences. An acceleration of sea level rise and increased storminess may reduce the eroding marshes even further and the future cost of replacing their buffering function with coastal protection methods could be considerable. It may be possible to keep ahead of the problems without the need to invoke hard engineering to protect land in the upper and mid estuary if action is taken soon to halt the erosion and restore some of the shoreline to a more natural state. Potentially, the two management strategies presented here could protect relatively sheltered coastal systems in which natural materials and processes are manipulated at low cost with no damage to the environment.

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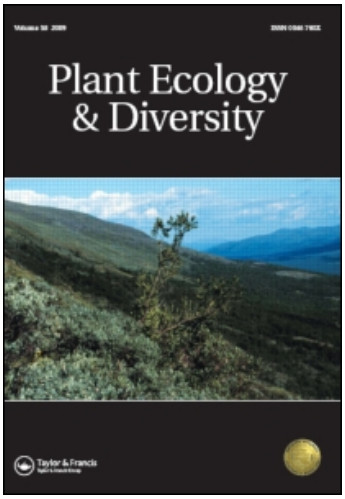

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A comparison of short-term sediment deposition between natural and transplanted saltmarsh after saltmarsh restoration in the Eden Estuary (Scotland)

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SCOTTISH SECTION

A comparison of short-term sediment deposition between natural and transplanted saltmarsh after saltmarsh restoration in the Eden Estuary (Scotland)

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Background: The fringe saltmarsh in the Eden Estuary is suffering severe erosion and its die-back will expose the shoreline to an increase in wave and tidal energy, especially given rising sea levels.
Aims: To investigate the effect of vegetative transplants on saltmarsh sedimentation, the present study aimed to compare short-term sediment deposition and accretion in transplanted sites of *Bolboschoenus maritimus* and *Puccinellia maritima* with those in natural stands of these species and with upper unvegetated mudflats. The effect of tidal height and wind direction on sediment deposition in the different systems was also studied.
Methods: Sediment deposited each day was collected on pre-weighed filter papers placed on the sediment surface and the sediment accreted each month was weighed in relation to the zero bed level by using a bar placed across marker poles.
Results: Older transplants of *B. maritimus* retained significantly higher quantities of sediment than natural *P. maritima* or the upper mudflats, but had similar amounts to that deposited in natural *B. maritimus*. Deposition rates in younger transplants were found to be similar to those on the upper mudflats. Sediment surface elevation in natural *P. maritima* remained constant throughout the year, but increased in all other sites during the summer. The upper mudflat was the only site to erode during winter. A significant positive association was found between tide height and sediment deposition, while winds from the south-east were associated with significantly more deposition than winds from the south-west.
Conclusions: These findings suggest that saltmarsh restoration using vegetative transplants can enhance sedimentation in eroded fringe saltmarsh. This strategy deserves further investigation since it may provide a sustainable management option in the face of rising sea levels.

Keywords: Eden Estuary; erosion; marsh restoration; saltmarsh; sea level rise; sedimentation

Introduction

Estuarine fringing saltmarsh absorbs wave and tidal energy and strengthens upper shorelines by capturing and retaining sediment (Brooke et al. 1999). These structural properties mean that saltmarsh vegetation provides a valuable function in the coastal zone by reducing the cost of protecting the hinterland from flooding and storm surges (Brampton 1992; King and Lester 1995). Furthermore, saltmarsh habitat is an integral part of the wider estuarine ecosystem, being involved in nutrient and sediment transfer to the marine environment, reducing contaminant run-off from entering coastal waters and providing a useful carbon sink to offset carbon dioxide emissions (Boorman 1999). It also provides valuable habitat for both marine and terrestrial organisms and a refuge for specialist salt-adapted plant species.

The Eden Estuary, to the north of St Andrews on the east coast of Scotland, at one time had more extensive saltmarsh vegetation around its shore (Wilson 1910). The subsequent development of farmland to provide an air base and golf links led to the loss of the upper marsh zone as ad hoc coastal defences were put in place, including soil and turf embankments, low cliffs of rubble and inorganic waste, and more recently gabion walls. The remnant populations of saltmarsh grass, *Puccinellia maritima* (Hudson) Parl., on the seaward side of these constructions have since suffered extreme die-back and erosion (Figure 1) especially over the last 20 years (Fife Council 2008). The erosional features are extreme and include lateral stripping of the vegetation and deeply eroded incisions into the marsh body, whereby

Figure 1. Extreme erosion in the natural *Puccinellia maritima* marsh encircling the shoreline of the Eden Estuary.

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the underlying sediment bed becomes fragmented and the fragments slump on to the upper mudflats.

In common with other estuaries in Great Britain, common cordgrass, *Spartina anglica* (C.E. Hubbard) was introduced to the Eden shoreline in the late 1940s (R. Crawford, pers. comm.). Although the species flourished, it is a non-native species and fears of its invasive and uncontrollable spread over mudflats and other plant communities led to an eradication policy from the late 1960s onwards entailing physical uprooting (Fife Council 2008). In many areas, decayed root mats of both *S. anglica* and *P. maritima* are still apparent some 20 m seaward of their present location and indicate the once-extensive size of the saltmarsh.

Coastal squeeze, when hard sea defences prevent the landward retreat of coastal vegetation due to rising sea levels, is a growing concern. In the past, however, and given a sufficient supply of sediment for accretion, rising sea levels have been beneficial to the creation of new marsh (Cundy and Croudace 1996; Crawford 2008). Moreover, in some other UK saltmarshes the rate of sediment accretion has been found to be equal to, or greater than, the current rate of sea level rise (Cahoon et al. 2000; van der Wal and Pye 2004), suggesting that factors other than sea level rise could be responsible for saltmarsh decline. For example, the loss of the upper marsh zone and subsequent embanking increases the reflected wave and tidal energy over a saltmarsh (Reed et al. 1999). There is also more sediment movement and less fine-grained sediment in front of sea walls, suggesting increased wave reflection (Bozek and Burdick 2005). The sediment deposition necessary for saltmarsh development (Adam 1999) is possibly reduced by the increased physical exposure associated with these regime changes. Furthermore, plant health and growth, especially in the underground biomass, is impaired by sediment starvation (Fragoso 2001) and hydrological changes (Turner et al. 2001). It has been suggested that less sediment enters the Eden basin than in former times, when the mouth of the estuary became partly occluded by the growth of a spit over a disused town rubbish tip (Crawford 2008). In addition, the capture and retention of sediment by a marsh that is fragmented and eroded at its seaward edge is unlikely (Ranwell 1964; Brown et al. 1998) and turbulence can be further increased without the hydrodynamic protection provided by the vegetation (Leonard and Croft 2006; Neumeier 2007).

Vegetation and root die-back from the effects of pollution (Mason et al. 2003) and eutrophication (Darby and Turner 2008) also increase the vulnerability of saltmarsh to erosion. In the past the Eden Estuary was heavily polluted as a result of industrial activity (Clelland 1997) and was affected by eutrophication (Fife Council 2008). The *P. maritima* marsh is also old (Wilson 1910) and, being relatively high on the tidal frame, the main body of the marsh is usually only inundated during high spring tides. Young, actively developing marshes lower on the tidal frame tend to have relatively high rates of sediment accretion (Pethick 1981; Langlois et al. 2003), but in the Eden Estuary new communities of *P. maritima* are no longer developing on the seaward edge of the older marsh. Given the overall condition of the remnant marsh, and noting that it has reduced from 32 ha in 1988 to its current 12 ha (Fife Council 2008), it is thought unlikely to be able to withstand rising sea levels and the increased storm frequency and intensity associated with climate change.

King and Lester (1995) suggested that saltmarsh erosion and die-back can be halted if a suitable body of plants can be transplanted to take its place and act as a buffer. Small stands of the sea club rush, *Bolboschoenus maritimus* (L.) Palla, occur where freshwater drainage enters the estuary and extend into what is usually considered to be the saltmarsh pioneer zone (Figure 2). These native reed-type communities have not been subjected to the same die-back as *P. maritima* and a previous study found that rhizomes of *B. maritimus* successfully formed vertical shoots under a range of salt and anoxic conditions (C. Maynard, unpublished data). This species therefore was considered suitable as a donor for marsh restoration trials (Figure 3).

Figure 2. A natural and relatively healthy stand of *Bolboschoenus maritimus* in the Eden Estuary.

Figure 3. Transplanted *Bolboschoenus maritimus* in the Eden Estuary, planted in 2003 at the seaward edge of an eroded natural *Puccinellia maritima* marsh.

Although marsh restoration using vegetative transplants is relatively common in managed realignment sites (Brooke et al. 1999; Sullivan 2001), the viability of restoring the seaward edge of eroded marsh with transplants has been questioned (Boorman 2003), despite earlier success with *Spartina* plantings. In addition, the extensive literature on the importance of sedimentation in saltmarsh formation and resilience (Reed 1989; Allen and Duffy 1998; Boorman 1998; Brown et al. 1998; Temmerman et al. 2003; Leonard and Croft 2006; Murphy and Voulgaris 2006) tends not to include information regarding the influence of transplanted vegetation on sedimentary processes, and the potential value of *B. maritimus* in trapping sediment has also been overlooked. Although a large-scale marsh restoration project is underway in the Eden Estuary in order to fully assess the methods and environmental parameters necessary for success, the present preliminary study presents the short-term sediment deposition and accretion patterns in transplants of *P. maritima* and *B. maritimus* and compares them to those in upper, unvegetated mudflats, natural but eroded *P. maritima* marsh, and relatively healthy stands of natural *B. maritimus*. Changes in sediment deposition in these different systems in relation to tidal height and wind direction, and therefore to increasing sea level and climate change, have also been investigated.

Methods

Site description

The Eden Estuary (Figure 4), located between St Andrews and the Tay Estuary on the east coast of Scotland, is a small but comparatively wide and shallow estuary with an average diurnal tidal range of 5 m. The estuary is noted for its nationally and internationally important wildfowl populations and forms part of the geomorphologically complex Tay–Eden Estuaries Specially Protected Area and Special Area of Conservation. It is also a Local Nature Reserve (LNR), a designated Site of Special Scientific Interest and a Ramsar site (Fife Council 2008).

Although a relatively small pocket estuary, the northern and southern shorelines are very different in character, partly because the north shore has a lower elevation than the south shore; for example, the north shore upper mudflats and saltmarshes are 3.1 m and 4.0 m, respectively, above chart datum, whereas the corresponding areas on the south shore have elevations of 4.2 m and 5.5 m (C. Maynard, unpublished data). The south shore is also less sheltered from prevailing winds and tidal currents and generally has sandier sediments than the north shore (Eastwood 1976). Additionally, while some horse-grazing of the saltmarsh is permitted on the south shore in the area known as Edenside (Figure 4), livestock grazing has not occurred on any other part of the Reserve's saltmarsh for many years (Fife Council 2008). Rabbit grazing is also limited, as the local populations are controlled by the landowners adjacent to the study areas.

Transplants of *B. maritimus* were established at an eroded marsh front on the north shore in March 2000 (see Maynard 2003 for detail) and again on the south shore in March 2003, with the addition of *P. maritima* transplants. Monitoring the patterns in sediment deposition and accretion commenced in July 2004. The four-year-old transplants of *B. maritimus* had an approximate stem density of 150 m^{-2}, compared to 30 m^{-2} for the one-year-old transplants of *B. maritimus*, whereas growth in the *P. maritima* transplant site was extremely limited, most planted units not spreading beyond their original size of 5 × 5 cm (C. Maynard, unpublished data).

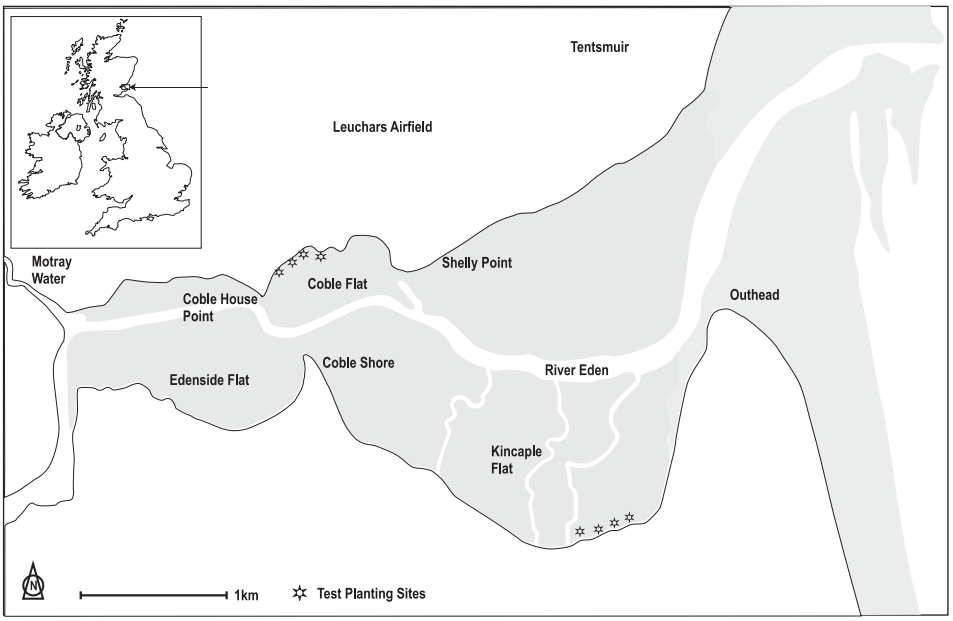

Figure 4. Location map of the Eden Estuary Local Nature Reserve showing the general areas of trial plantings and sedimentation studies (test planting site symbols not to scale).

Table 1. Experimental layout for each shore in the Eden Estuary.

North shore	South shore
Upper mudflat	Upper mudflat
Natural *P. maritima*	Natural *P. maritima*
Natural *B. maritimus*	*B. maritimus* transplants (1 yr)
B. maritimus transplants (4 yrs)	*P. maritima* transplants (1 yr)

Experimental layout

A total of eight sites were investigated, but replication of each site on each shore was not possible (Table 1), even though each vegetated site was monospecific. Two replicate plots, each measuring 5 m × 10 m and at a minimum distance of 10 m apart, were marked out within each site. The natural marsh and upper mudflat sites were located immediately behind and adjacent to the transplant sites, respectively.

Short-term sediment deposition

Sediment traps were deployed to measure short-term sediment deposition following the protocol first established by Reed (1989). Sediment deposited each day was collected on pre-weighed filter papers (9 cm Whatman GF/C) placed on the sediment surface. The filter papers were secured to plastic discs to prevent adhesion to the sediment surface. Each day (i.e. after two high tides) the discs were lifted to allow the filter paper to be collected and replaced with a new one. The discs and clean filter papers were then returned to the sediment surface in a new and undisturbed area for the following day. Five filter papers were laid in each plot. After collection the filter papers were oven-dried overnight at 40 °C and reweighed to 0.1 mg. Brown (1998) corrected for salt weight and the area of the filter paper covered by the paperclips, but on investigation the present study showed only a negligible effect, and corrections were considered unnecessary. Wave activity damaged some of the filter papers and the relative percentage loss was deducted from the overall surface area of the filter papers (636 cm^2).

Sediment deposition rates were collected over the course of one month, between 21 July and 22 August, and included two neap/spring tidal cycles. However, data collected during 9 days of the study were removed from the analysis, since prolonged rainfall had caused the filter papers to disintegrate. Sediment deposition was expressed as mg dry weight sediment per unit area (mg 100 cm^{-2}).

Sediment surface level

Metal marker poles were used to measure relative vertical changes in marsh and mudflat elevation. At each plot in each site, a pair of 2 m poles was driven into the sediment 1 m apart so that they extended 1 m above the sediment surface. Each data point was the average of three readings taken as the distance from a builders' level placed on top of the two poles to the bed surface. The zero bed level was established in July 2004, and measurements were taken after the last spring tide in each subsequent month until July 2005. Data were expressed as mm wet and unconsolidated sediment per month or per annum, and as positive (accretion) or negative (erosion) in relation to the zero bed level.

Data analysis

Sediment deposition

The assumption of homogeneity of variance and normality was met for all but the natural *B. maritimus* site and data transformation was therefore considered unnecessary. The data are first presented as the mean amount of sediment per filter paper over the whole study period, i.e. the sum total deposited over 23 days divided by the total number of filters laid in each site. There were no significant differences between the plots within each site and therefore these data were pooled to compare the sites using one-way analysis of variance (ANOVA). Post-hoc analysis of means was compared using a Tukey test ($\alpha = 0.05$).

Sediment surface level

No transformation was necessary to meet the assumptions of least squares analysis and the data from the plots within each site were pooled. A two-way ANOVA compared the sediment surface level changes between sites and between months, and their interaction. Comparison between the means was performed using a Tukey test ($\alpha = 0.05$).

Tidal height

The temporal variation in the data set appeared to show little pattern (data not presented). In a preliminary study, however, a subset of the data was used to analyse the effect of changing water levels on sediment deposition between sites only on those days when wind speeds were low (below 3 m s^{-1}) and from a south-easterly direction. A Pearson product–moment correlation was used to compare the results for 4 days of neap tides (between 4.2 and 4.6 m), 3 days of average tides (between 4.6 and 5.0 m) and 4 days of spring tides (greater than 5.0 m). These tide heights were proxy data taken from the tidal height chart for the gauge at the port of Dundee, in the neighbouring Tay Estuary.

Wind direction

The effect of winds from the south-east or the south-west, the two most prevalent wind directions during the month-long study, on sediment deposition was investigated.

Three days of each wind direction were available for comparison when tide height was between 4.6 and 5.0 m (above chart datum) and wind speeds were low (below 3 m s^{-1}). A two-way ANOVA was used to compare variation in sediment deposition between sites, wind direction and their interaction. Post hoc analysis of means was conducted using a Tukey test ($\alpha = 0.05$). Wind directions were recorded at RAF Leuchars on the northern shore of the estuary (Meteorological Office).

Results

Mean total sediment deposition

There was no significant difference between the mean total sediment deposited (35.61 and 47.75 mg cm^{-2}, respectively) in the natural stand and in the four-year-old transplants of *B. maritimus* (Figure 5A), although both of these sites gave significantly higher values than the corresponding mudflat (22.74 mg cm^{-2}) or the natural *P. maritima* stand (10.08 mg cm^{-2}) ($F_{3,914} = 26.05$; $P = 0.001$). The amount of sediment deposited on the south shore (Figure 5B) was not significantly different between the mudflat (20.84 mg cm^{-2}) or the one-year-old transplants of either *B. maritimus* or *P. maritima* (26.94 and 25.78 mg cm^{-2}, respectively), but was significantly lower in the natural *P. maritima* stand (2.6 mg cm^{-2}) ($F_{3,901} = 70.65$; $P = 0.001$).

Sediment surface level

At the end of the one-year study, the mudflat, and the natural and transplanted *B. maritimus* had accreted 1.2, 3.4 and 3.8 mm of sediment, respectively, compared to the zero bed level maintained by natural *P. maritima* (Figure 6). The differences between the sites were significant ($F_{3,104} = 71.63$; $P = 0.001$) and post hoc analysis showed that

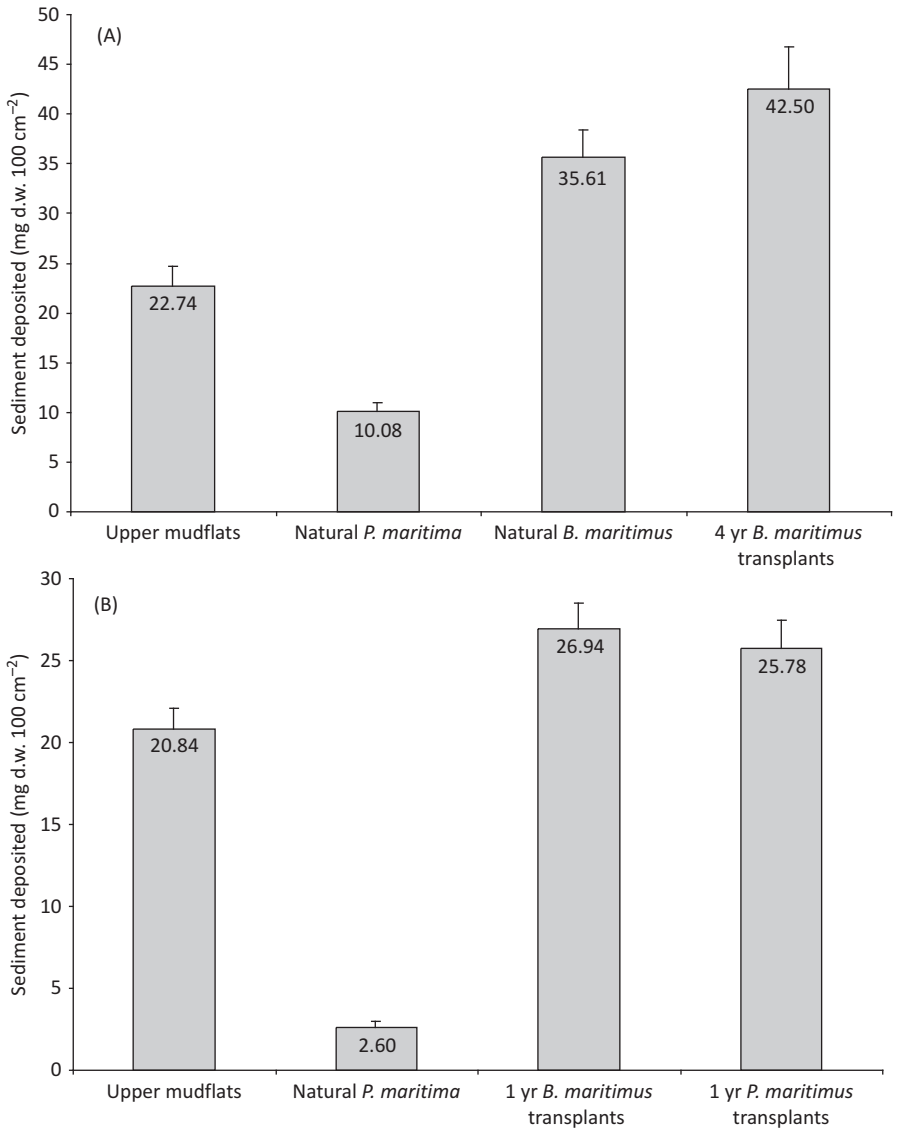

Figure 5. Sediment deposition during July and August 2004 for (A) the north shore and (B) the south shore. Results are shown in dry weight (mg 100 cm^{-2}) and relate to the average (+ standard error) of the total amount of sediment deposited over 23 days.

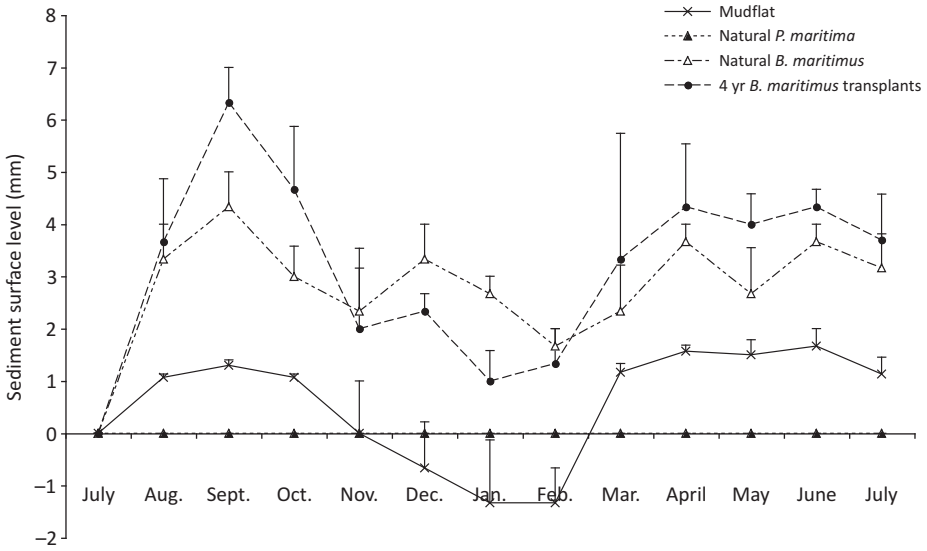

Figure 6. Changes in soil level (mm) during 1 year on the north shore of the Eden Estuary. The zero soil level was established in July 2004, and measurements were recorded after a spring tide in each subsequent month until July 2005. Each data point is the average (+ standard error) of six readings taken from below a builder's level placed between two poles.

the *B. maritimus* sites had significantly more accretion than either the mudflat or the natural *P. maritima* sites.

The differences between the months were significant ($F_{12,104} = 6.99$; $P = 0.001$), and in general more accretion occurred between April and October than between November and March. The interaction between site and month was also significant ($F_{36,104} = 1.64$; $P = 0.027$), as the mudflat and *B. maritimus* sites accreted sediment during the summer months but eroded over the following winter. However, the mudflat site was the only site to erode below zero bed level during winter, losing nearly half of the sediment gained during the previous summer.

The natural and planted *B. maritimus* sites demonstrated a similar pattern throughout the study, but whereas the latter had higher peaks around the high tides associated with March and September, the former retained more sediment over the winter months. On the other hand, by the end of the study there was no significant difference in accretion between the natural or planted *B. maritimus* sites, and even the mudflat sites had recovered the sediment lost over the winter.

Relationship between tidal height and sediment deposition

A significant positive relationship was found between tidal height and sediment deposition for all the sites on both shores. The strength of the association differed between the sites ($r = 0.42$ to 0.57, $P < 0.001$, $n = 110$ per site). However, a curvilinear relationship was apparent, and more sediment was deposited during 'normal' tidal heights, as opposed to either neap or spring tides (Figures 7A and B). For example, mean sediment deposited was below 20 mg 100 cm^{-2} for all the sites during the lowest tidal events (4.2 to 4.6 m chart datum) but a sharp increase became apparent in both the natural and four-year-old transplant sites of *B. maritimus* and the upper mudflats on the north shore once the tide height rose above 4.6 m. Although sediment deposition increased with increasing tide height in the natural stands of *P. maritima* on both shores, the effect was not so dramatic, and on the south shore in particular it was minimal until spring tides higher than 5.1 m were reached.

Relationship between wind direction and sediment deposition

Winds from the south-east caused significantly more sediment deposition than winds from the south-west ($F_{1,384} = 196.21$; $P = 0.001$). The differences between the sites were also significant ($F_{7,384} = 32.36$; $P = 0.001$), as was the interaction between site and wind direction ($F_{7,384} = 20.49$; $P = 0.001$). However, post hoc analysis showed that wind direction had the greatest effect on the upper mudflats and the natural and transplanted *B. maritimus* sites on the north shore, but little effect on the mudflats and younger transplants of *P. maritima* and *B. maritimus* sites on the southern shore. There was no effect on the natural *P. maritima* sites on either shore (Figures 8A and 8B).

Discussion

Mean sediment deposition and surface levels

The natural marshes of *P. maritima* had lower quantities of deposited sediment compared to any other site, which could be the result of the increase in local turbulence and scour associated with a patchy and fragmented marsh front (Boorman et al. 1998; Brown 1998; Leonard and Croft 2006). The four-year-old transplant site of *B. maritimus* had twice the amount of deposited sediment than the adjacent unvegetated mudflat site, and since these two sites share the same elevation this can be directly attributed to the

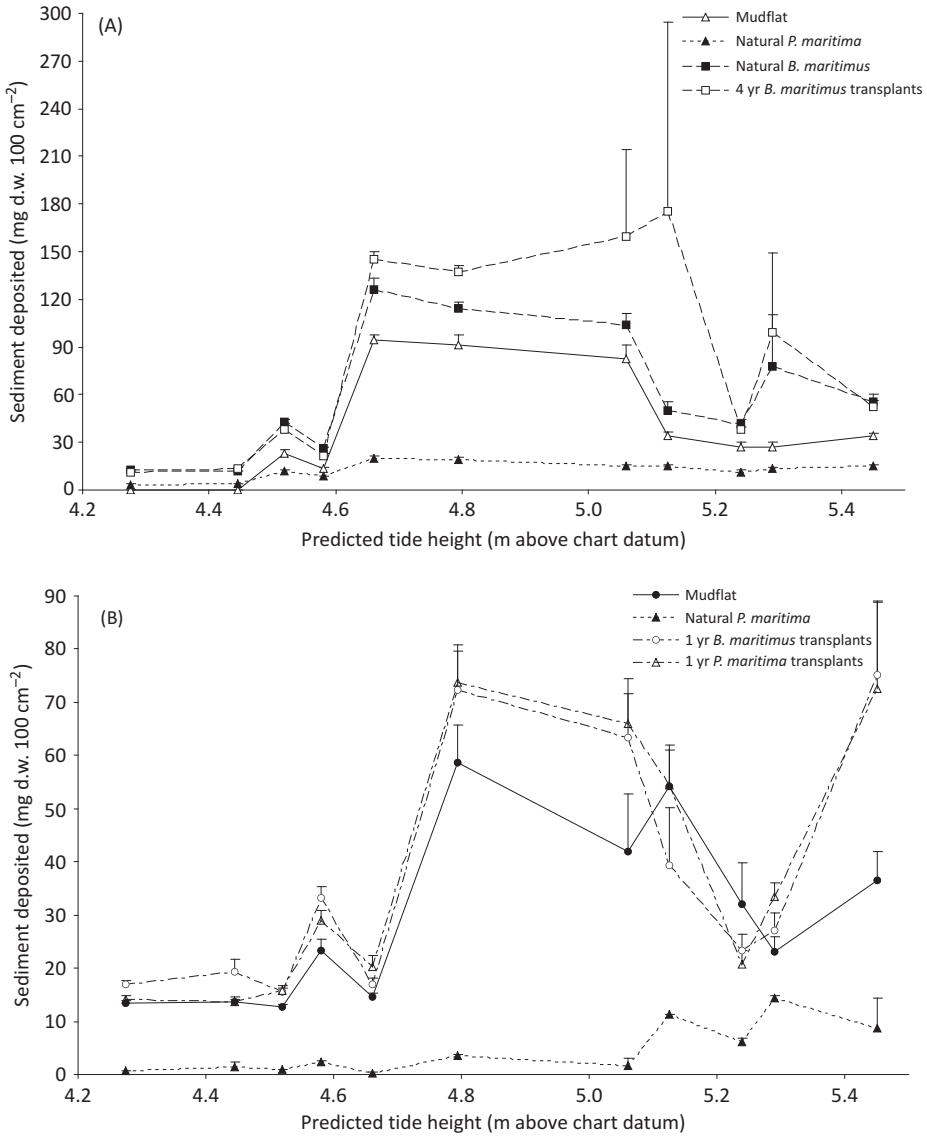

Figure 7. The effect of predicted tide height (metres above chart datum) on short-term sediment deposition in the Eden Estuary (means + standard error). The tide heights are the average of two high tides per day: (A) the north shore and (B) the south shore.

sediment trapping action of plant stems and leaves. The vegetated sites, whether transplants or natural marsh, also had less erosion than the upper mudflats, suggesting that the vegetation conferred some degree of stabilisation and protection to the underlying sediment bed.

The findings in the present study appear to confirm that high marsh elevation can result in lower rates of deposition (Temmerman et al. 2003) and accretion (Pethick 1981; Stoddart et al. 1989), since the southern shore sites, being almost a metre higher on the tidal frame than the northern shore sites, had significantly less deposition. Other factors, however, such as exposure (McManus and Alizai 1987) and proximity to river channels (Temmerman et al. 2003), may also have had an effect. The low sediment deposition in the natural *P. maritima* sites may be due to fragmentation of the marsh front and/or reduced tidal inundation because of the high elevation of the marsh, but no matter what the cause, sediment starvation is possibly a key factor in its decline (Fragoso 2001).

Species-specific effects may also be responsible for differences in sediment deposition, as the natural stands of *P. maritima* and *B. maritimus* on the north shore shared the same elevation, and yet considerable differences in sediment deposition and accretion were apparent. For example, the shorter, stiffer stems of *P. maritima* may have accounted for its lower rate of deposition compared to the metre-high stems of *B. maritimus*, as vegetation with taller and more flexible stems can impede the flow of water more effectively and capture sediment to a greater extent during conditions of low flow (Boorman et al. 1998). However, Boorman et al. (1998) also found that during high velocity flow, shorter and stiffer stems impeded water flow to a greater extent than taller vegetation. This would imply that *P. maritima* should have captured more sediment than *B. maritimus* during the faster and more turbulent flow conditions of winter, but the winter accretion rates recorded in the present study found this not to be the case, possibly because stiff shoots caused more scouring than flexible shoots (Bouma 2009).

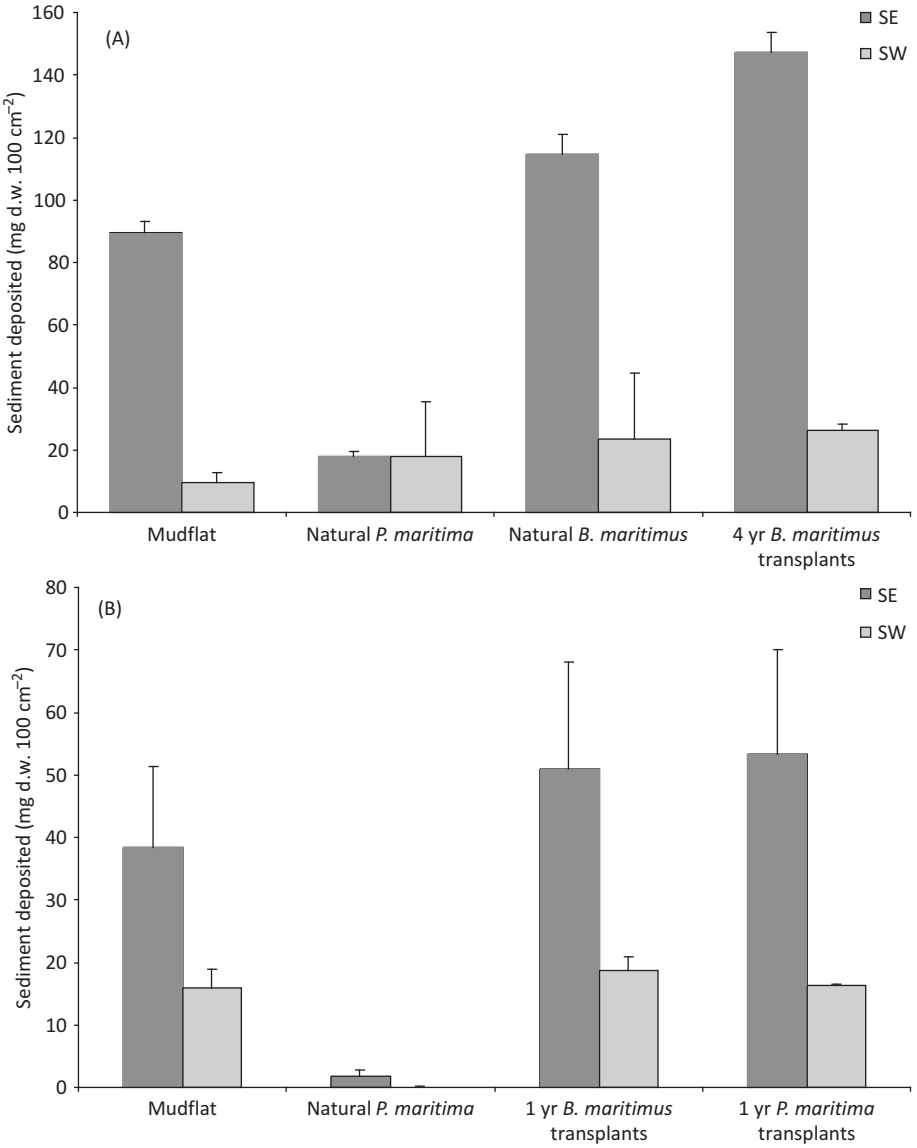

Figure 8. The effect of wind direction on short-term sediment deposition (means + standard error): (A) the north shore and (B) the south shore (SE, south-easterly and SW, south-westerly).

It is also possible that the sloping marsh edge of natural *B. maritimus* was more effective in dampening wave energy than the fragmented and near-vertical cliff edge of natural *P. maritima* (van Eerdt 1985).

The similarity in sediment deposition between natural and four-year-old transplants of *B. maritimus* suggests that the expected differences due to marsh elevation (4.0 m and 3.1 m, respectively) may have been counteracted by differences in stem density (respectively, 400 m^{-2} and 150 m^{-2}) (Gleason et al. 1979; Hall and Freeman 1994). In addition, similar sediment deposition rates between the one-year-old transplant and mudflat sites on the south shore also may have been the result of low stem density in the former sites, and differences may emerge as stem density increased over time in the young transplants.

Plant litter and debris reduce the exposure of a sediment bed to erosion (Boorman et al. 1998), and this may explain why the natural stand of *B. maritimus* retained more sediment during winter than the adjacent *B. maritimus* transplant site. Conversely, the enhanced accretion in the transplant site during the summer may have been a consequence of its low stem density and greater substrate exposure, thereby encouraging the growth of diatomaceous mats (Anderson 2001). Further investigation of the diatom communities within these sediments may be warranted.

Environmental controls on sediment deposition

The effect of tidal height on sediment deposition was clear. Spring tides (those above 5 m) caused more deposition than neap tides (below 4.6 m), and yet the highest quantities of sediment were deposited by tides between 4.6 and 4.8 m. The sediment load in tidal waters tends to be increased by spring tides because of the associated higher current

velocities (Murphy and Voulgaris 2006) and because more sediment in the river channel is captured during a low tide and carried on to the saltmarsh surface by the following high tide (Ranwell 1964). The present study showed that sediment deposition peaked during normal tides, suggesting that greater sediment availability in the water column may be cancelled out by higher current velocity. However, it appears that tidal height and marsh elevation exert a strong control on sedimentation in the salt marshes of the Eden and that sea level rise may become a driving force for an increase in accretion and sedimentation rates, as found in the Solent marshes, southern England by Cundy and Croudace (1996).

Marsh restoration efforts should also take account of changing weather patterns, as an increase in storms due to climate change could increase saltmarsh erosion. However, episodic storms and hurricanes can deposit a greater quantity of sediment in saltmarsh than regular tidal inundation due to the increased stirring effect on sediment in the water column (Reed 1989). Changes in wind regime also can cause a response in marsh sedimentation (Allen and Duffy 1998). Wind speed and direction were shown to have a significant effect on sediment deposition in the saltmarsh and reed bed communities in the neighbouring Tay Estuary (McManus and Alizai 1987). Similarly, low rates of sediment deposition were apparent in the present study when winds blew from a south-westerly direction and may be a consequence of the downward force on the height of the high tide, providing less opportunity for sediment to be deposited on the marsh surface. The higher rate of deposition associated with winds from the south-east is a reflection of the open aspect of the mouth of the Eden Estuary, suggesting that although exposure is normally considered inimical to saltmarsh development, an increase in sediment deposition from rising sea levels when winds are from the south-east may actually benefit marsh development. However, wave-generated turbulence from increasing wind speed can also affect sediment deposition (Alizai and McManus 1980), but as wind speeds were low in summer during the present study significant trends were not apparent.

Conclusions

Although limited both spatially and temporally, the findings of the present study show clearly that the natural eroded *P. maritima* marsh no longer functions as a sediment trap. Accretion rates in some other UK areas of saltmarsh are generally less than 10 mm per annum (Table 2). However, French and Burningham (2003) calculated that an accretion rate of 3–5 mm per annum was enough to offset regional subsidence of 1–2 mm per year and the current rise in sea level of 2.4 mm per year in the south-east of England. The findings in the present study show that *B. maritimus*, either natural or transplanted, accreted enough sediment in one year (3.4 and 3.8 mm, respectively) to theoretically keep pace with the more recent and general estimates of 0.9 to 1.2 mm annual sea level rise around the UK coast, taking into account that mean sea level rise in Scotland is moderated by isostatic uplift (Ball et al. 2008). However, this does not hold true for the natural *P. maritima* communities, even though they are subjected to a similar wind and wave climate and share the same elevation as the natural *B. maritimus* stands. Transplanted *B. maritimus*, especially after 4 years' growth, enhanced sediment deposition and demonstrated that the sediment trapping function of the saltmarsh has the potential to be restored. Furthermore, direct transplantation of a marsh species such as *B. maritimus* will enhance the saltmarsh plant communities of the Eden Estuary and could lead to increased resilience to rising sea levels and storminess.

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Notes on contributors

Clare Maynard is a Ph.D. student at the University of St Andrews, studying the restoration of native saltmarsh communities in the

Table 2. Annual accretion rates (mm) reported in the literature concerning British salt marshes.

Location	Position/Type	Annual accretion (mm)	References
Eden Estuary	Natural *B. maritimus*	3.4	Maynard et al. (this study)
	Natural *P. maritima*	0	
	B. maritimus transplants	3.8	
Tay Estuary	Reed beds	0.58	Alizai and McManus (1980)
Humber Estuary	Horseshoe Point	9.7 – 11.3	Mohn-Lokman and Pethick (2001)
Humber Estuary	Skeffling	4 – 5	Brown (1998)
Norfolk	Backbarrier	3.9	French and Spencer (1993)
Norfolk	Stiffkey	3.08	Boorman et al. (1998)
Essex	Tollesbury	4.27	
Thames Estuary	Mature marsh	2 – 3	van der Wal and Pye (2004)
	Immature marsh	3 – 5	
The Solent		4 – 5	Cundy and Croudace (1996)
Severn Estuary	Mid marsh	4.65	Allen and Duffy (1998)

Eden Estuary. Her main research interests are wetland ecology, habitat restoration and coastal zone management.

John McManus is Professor Emeritus in the University of St Andrews and retains an interest in a number of academic areas. These include sediments and sedimentary processes in rivers, estuaries and lakes, the history of coal mining in Fife, coastal stability and sediment migration.

Robert Crawford is Professor Emeritus in the University of St Andrews and still conducts ecological studies of plants in marginal areas. Polar regions and wetlands in particular continue to absorb his attention.

David Paterson is Professor of Coastal Ecology in the University of St Andrews and the Executive Chair of the Marine Alliance for Science and Technology for Scotland (MASTS). His main research interests lie in the ecology and dynamics of coastal depositional systems, and he is particularly active in aspects of primary productivity, the importance of ecosystem function and the resilience of coastal systems to global change.

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