REHABILITATION OF THE ORANGE RIVER MOUTH SALT MARSH: SEED, WIND AND SEDIMENT CHARACTERISTICS

By

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Abstract

The Orange River is an important source of freshwater and like many other wetlands in semi-arid regions, supports various social (Spurgeon, 1998), economic (Spurgeon, 1998; Bornman *et al.*, 2005) and ecological functions. The saltmarsh at the Orange River Mouth has become degraded over time following numerous anthropogenic impacts. As a result the Transboundary RAMSAR site was placed on the Montreux record emphasising the importance for rehabilitation. The potential of the marsh for natural rehabilitation was assessed through three physical factors which were considered to have the most influence on the saltmarsh i.e. 1) sediment 2) water 3) wind.

Three sampling areas were chosen to investigate the sediment characteristics of the ORM saltmarsh and the suitability for seed germination and adult survival. Site A was representative of the general marsh area, Site B was thought to have favourable sediment conditions for saltmarsh growth because of the large numbers of seedlings and Site C was prone to inundation by wind blown sediment. The sites were sampled in 2005 (dry conditions) and in 2006 after high rainfall and river flooding. Electrical conductivity (EC) of the sediment throughout the marsh was hypersaline in many instances above the tolerance range for *S. pillansii* (> 80 mS.cm^{-1}) The freshwater event in 2006 lowered salinity significantly in two of the three sites.

Differences in sediment characteristics were also compared for three habitats i.e. driftlines, open sites and under vegetation. Driftlines (*C. coronopifolia* = 872 seedlings m^{-2}; *S. pillansii* = 1296 seedlings m^{-2}) and the microhabitat associated with adult plants (*C. coronopifolia* = 803 seedlings m^{-2}*; S. pillansii* = 721 seedlings m^{-2}) created favourable conditions for seedling growth, however open unvegetated (*C. coronopifolia* = 56 seedlings m^{-2}*; S. pillansii* = 49 seedlings m^{-2}) areas had significantly lower seedling density.

Due to the marsh currently being in a desertified state this study aimed to establish whether the remaining vegetation could produce enough seed to revegetate the marsh. Laboratory studies indicated that seeds of both species germinated best in freshwater (0 psu). The germination of *S. pillansii* seeds was 40 % at 0 psu compared to 5 % at 35 psu. After storage under hypersaline conditions (35 psu) *C. coronopifolia* showed 100 % seed germination when returned to freshwater whereas

storage at 70 psu decreased the viability of *S. pillansii* seeds. The plants are producing adequate seed that will allow for regrowth and rehabilitation if sediment and groundwater characteristics are suitable for seed germination, seedling growth and adult survival. However the increase in bare areas at the Orange River mouth as a result of salt marsh dieback has increased the available sediment source. The wind blown sediment has covered large areas of the remaining adult salt marsh vegetation, particularly in the northern corner at Site C, causing further die-back.

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This has been a challenging experience and I feel my scientific thought process has matured extensively.

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Chapter 1: Introduction

The catchment of the Orange River is the largest in South Africa (1 000 000 km^2) and the lower reaches form the border with Namibia (DWA, 1990) (Figure 1). The river is an important source of freshwater and like many other wetlands in semi-arid regions, supports various social (Spurgeon, 1998), economic (Spurgeon, 1998) and ecological functions. Under natural conditions the Orange River Mouth (ORM) was vegetated with mostly reeds and sedges within and adjacent to the channel. The large floodplain, present south west of the mouth, used to be covered in saltmarsh. The saltmarsh has become degraded over time following mining activities near the mouth, increased abstraction of freshwater and flow regulation upstream, leading to a loss of vegetation. A causeway from Alexander Bay to the beach was also constructed that effectively separated the marsh from the main river channel and prevented river and tidal flow into the marsh area. At present more than 300 ha of the saltmarsh is devoid of vegetation.

Figure 1.1 A map of southern Africa showing the location of the Orange River and catchment.

The Orange River Mouth was designated a RAMSAR site on the 28 June 1991 and placed on the Montreux Record on 26 September 1995 (http://www.ramsar.org/key_montreux_record.htm, 2006). The Montreux Record is used to indicate wetlands of international importance that are threatened and require intervention. For the site to be removed from the Record the marsh needs to be rehabilitated. The degraded marsh has had an impact on migratory water bird populations (Anderson *et al*., 2003). Despite all the descriptions of the impact of abstraction and mining (DWA, 1990; CSIR, 1991, 1997), very little was known of the effect and impact on the saltmarsh. Bornman *et al*. (2004a) described the conditions of the saltmarsh prior to the Working for Wetlands rehabilitation activities. Their study focussed on sediment and groundwater conditions whereas this study includes research on seed characteristics and effects of wind.

Attempts were made by the Alexkor mining company to rehabilitate the degraded marsh in 1997. The rehabilitation effort was partially successful in that observations over the last ten years show that the intertidal marsh has flourished. This occurred naturally, and due to the size of the marsh and the expense of artificial rehabilitation, the objective of this project was to assess the natural rehabilitation potential of the marsh. The assessment included the study of seeds and their dynamics in the marsh, the physico-chemical properties of the marsh sediment and the effect of wind blown sediment on the vegetation. The focus was on water, sediment and wind as these factors were thought to have the greatest influence on the life cycle of the two dominant saltmarsh species namely: *Cotula coronopifolia* L. and S*arcocornia pillansii* A.J.Scott. For rehabilitation to occur naturally all phases of the plant life cycle must be completed namely seed production, seed deposition, germination, seedling establishment and adult vegetation survival (Figure 1.2).

The nature of the sediment ultimately determines the presence, growth rate, reproduction and recruitment success of vegetation (Onaindia & Amezaga, 1999). In this study a suite of sediment characteristics (Chapter 4) were investigated to gain some understanding of the preferences of the two species and to assess the suitability of the sites for rehabilitation. The following parameters were measured; electrical conductivity, sediment moisture, organic and carbonate content, sediment particle size, depth to groundwater and sediment compaction and shear. During the study there was a high rainfall and flooding event in 2006 that allowed for the determination of the effect of freshwater on the sediment characteristics.

Water is important in arid areas as it is often a limiting factor. It is particularly important in this saltmarsh as the marsh is isolated from the main river channel by the causeway and is characterised by hypersaline groundwater. Water also plays a role in the distribution of seed. For natural rehabilitation to occur the ability of the seed to germinate and establish in new areas is necessary. The life cycle phases of the two dominant plant species were investigated in Chapter 5 (Figure 1.2).

Wind blown sediment (Chapter 6) inundates the adult vegetation leading to a loss of vegetation and a loss of a seed source (Figure 1.2). The presence of vegetation influences the sediment characteristics especially the ability of the wind to transport sediment. The properties of the wind blown sediment were investigated to determine if this was a threat to rehabilitation and whether it could be reduced with active management.

A general hypothesis for the study was that the ORM saltmarsh has adequate seed and rehabilitation will occur if the sediment and groundwater characteristics are suitable for seed germination, seedling growth and adult survival.

The objectives of the project were to:

- Measure the sediment characteristics of the saltmarsh to assess suitability for seed germination, seedling establishment and adult survival (Chapter 4);
- Assess the effect of freshwater (2006 rainfall and flooding event) on the sediment characteristics (Chapter 4);
- Establish whether there were adequate seed available to recolonise the site once conditions became favourable by measuring seed production, deposition and seedling density (Chapter 5);
- Determine the effect of salinity and water on seed germination as well as the effect of exposure to hypersaline conditions (Chapter 5);
- Establish whether the seed bank was persistent and whether seeds remained viable after long term storage under different treatments (Chapter 5);
- Investigate the source, properties and accumulation of wind blown sediment and the effect on the marsh (Chapter 6); and
- Investigate the potential to reduce wind blown sediment through the use of wind breaks such as shade cloth (Chapter 6).

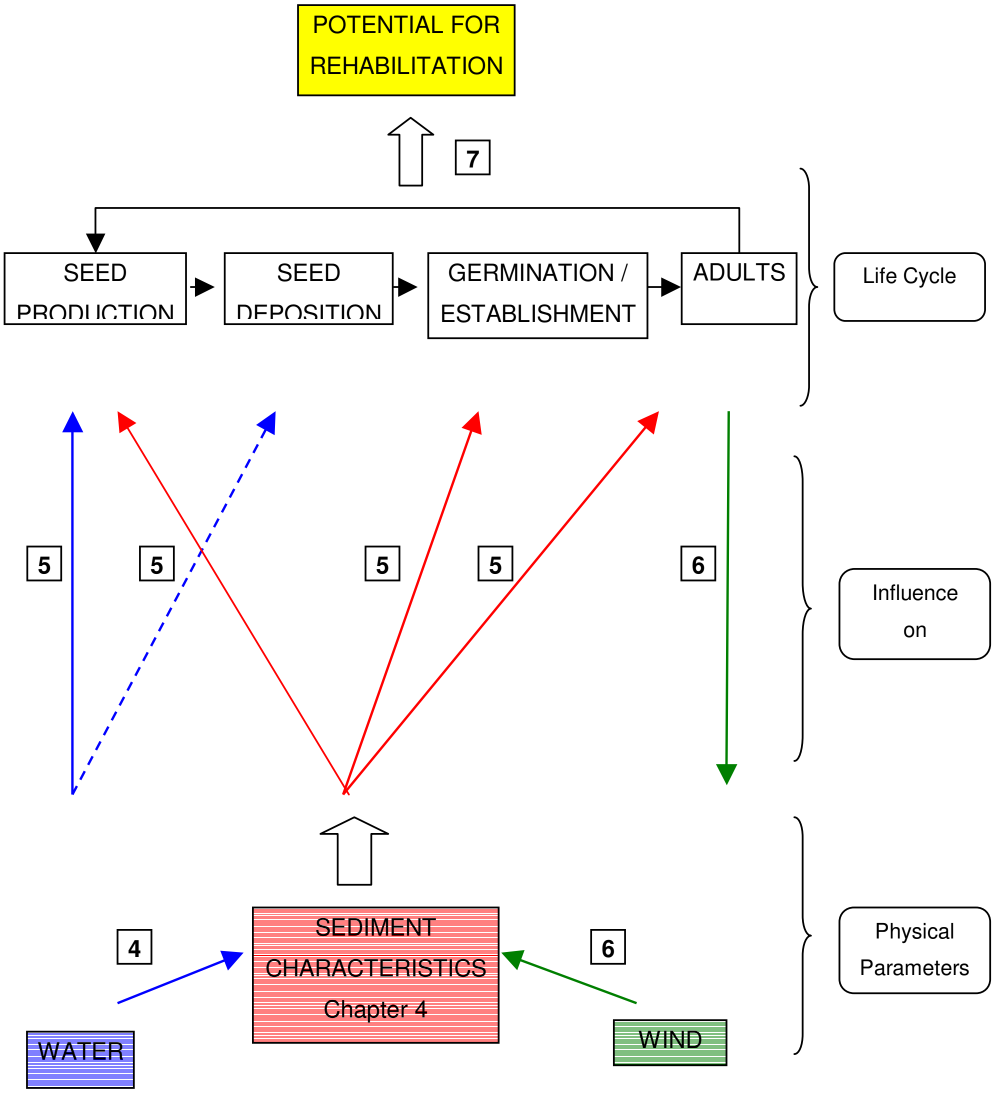


Figure 1.2 The structure of the thesis showing the relationships between the physical characteristics of the Orange River Estuary and the life cycle of the vegetation. Numbers in the figure represent the chapter numbers where the data are presented.

Chapter 2: Literature Review

2.1 Saltmarshes

Saltmarshes have an ecological value in primary production, nutrient cycling, as a habitat for fish, birds and other wildlife and in stabilising shorelines (Adams *et al.,* 1999). Most of the 3 000 km of the South African coastline is subjected to moderate to high wave energies. With saltmarsh development restricted to protected estuaries and embayments (O'Callaghan, 1992; Cronk & Fennessy, 2001), 75 % of the 17 000 ha of saltmarsh area occurs in five systems, namely: Langebaan Lagoon, Knysna Estuary, Swartkops Estuary, Berg Estuary and the Olifants Estuary (Adams, *et al.,* 1999).

Saltmarshes are vegetated wetlands that are dominated by herbaceous and small shrubby halophytes (Lubke & de Moor, 1998). Halophytes are species that have the ability to complete their life cycle in salt concentrations from 50 – 250 mM NaCl (Flowers*,* 1985; Cronk & Fennessey, 2001). Halophytes inhabit areas according to the range of conditions they can withstand; conditions dominated by the tide at the lower levels, but almost independent of them at the highest levels (Ranwell, 1972). At the lower level, inundation period limits growth, while at the upper level terrestrial species out-compete the marsh species (Carter, 1988).

2.1.1 The Importance of saltmarshes

In many parts of the world saltmarsh values are being recognised and more widely appreciated, and saltmarsh restoration is a growing science and practice. Saltmarshes are valued for their support for biodiversity (Álvarez-Rogel *et al.*, *In Press*), aesthetically and economically (Spurgeon, 1998). Habitat values are frequently recognised and the protection and restoration is often undertaken for the purpose of conserving birds (Zedler, 2000; Álvarez-Rogel *et al.*, *In Press*), amongst others. This is especially important in arid areas where rainfall is low and standing freshwater is scarce. Saltmarshes support waterbirds, and because people value recreational bird-watching, various governments support the conservation and restoration of coastal wetlands and endorse international agreements such as the Ramsar and Bonn Conventions.

2.1.2 Rehabilitation

There is much debate on the question of how to define restoration/rehabilitation success? Restoration refers to the process of returning an ecosystem or habitat to its original community structure, natural complement of species, and natural functions (Zedler, 2000). This is not possible at the Orange River Mouth due to the size of the degraded marsh and the changes over the years on the natural morphology. In this case the term rehabilitation is appropriate where the effort is in the recovery of specific ecosystem services in a degraded ecosystem or habitat through revegetating an area (Wolters *et al.*, 2005). Restoration and rehabilitation are often used interchangeably and the success of projects are determined on whether the objectives for the project were met.

Zedler and Callaway (1999) propose that new terminology be used for the assessment of wetland rehabilitation projects, focusing on progress towards ecological goals rather than the yes/no alternative of success versus failure. Rehabilitation projects that are not undertaken as mitigation should be termed in reference to progress of the relevant attributes, which include hydrologic, water-quality-improvement, and biodiversity support functions. In addition the ecosystem should be self-sustainable. Other possibilities for assessing success are to compare the ecological structure or functioning of a restored site with one or more reference sites. However comparing conditions with a natural reference system may not be realistic or appropriate because restoration may start on different substrate or different elevation gradients. Defining success focuses on the question of whether the aim should be the restoration of the structure of an ecosystem or its functioning. Zedler and Callaway (1999) point out that the restoration of functionality often takes longer than the rehabilitation of plant communities themselves.

A prerequisite for the successful rehabilitation of saltmarsh communities is the availability of a target species source and the ability of the species to reach the target area. The best results are expected when the target species are still present in the community species pool of the target area, which consists of the established vegetation and the seed bank (Zobel *et al.*, 1998).

Water-borne seed transport during floods is a requirement for successful rehabilitation, provided that seed sources are available and that restoration sites are flooded at least occasionally (Merrit & Whol, 2002). Mitsch *et al.* (*In Press*) and Neff and Baldwin (2005) considered hydrochory to be the main mechanism by which species were introduced into a restored wetland site.

The aim of this study is to predict whether the saltmarsh vegetation will colonise previously vegetated sites when the hydrological function is restored. Thus the saltmarsh has not yet been established and due to the area in question planting of relevant species is not viable. The literature suggests that rehabilitation of the structural component is the first and most important stage in saltmarsh restoration. (Wolters *et al.,* 2005).

The most obvious indication that the physical and chemical properties are becoming favourable is the presence of vegetation and the type of vegetation. Usually plant cover, density, and/or biomass have been measured for assessment. Some studies also measured species number, plant height and total stem length. Only two of the 25 papers reviewed by Zedler (2000), looked at the change in area of different habitat types or vegetation classes.

Compliance is the preferred term for achieving thresholds, or standards, for each criterion specified in the mitigation requirements. To call such projects as successes or failures without an evaluation of functional integrity or sustainability introduces confusion. Compliance to the Montreux record would involve the rehabilitation of primarily the vegetation as this attracts the subsequent water-bird diversity. Only Simenstad and Thom (1996) have evaluated bird use of restored coastal wetlands, despite the fact that bird use of areas is often a goal of projects.

The assessment of sediment in saltmarsh habitats is important due to the extreme conditions of this ecosystem and the constant changes of both physical and chemical properties. Thus organic matter and moisture content, sediment texture and salinity should be evaluated. For some plant studies previous literature measured nutrients, pore water constituents and water potential gradients (Zedler, 2000).

2.2 Dominant species and life-cycle characteristics

2.2.1 Saltmarsh species

In terms of percentage cover, two species can be considered dominant at the Orange River Mouth saltmarsh.

Cotula coronopifolia L.

This is a herbaceous, succulent creeping perennial endemic to South Africa (Lubke & de Moor, 1998). It is common in both fresh and saline waters. It has erect stems with sheathing bases. The leaves are light green often with dissected and lobed margins. The flowers are yellow and carried in a compact head. The seeds of most *Cotula* species have a wide salinity tolerance range of 5 – 24 psu. (practical salinity units) (Zedler & Beare, 1986; Vanderplank, 1999).

Plate 1 *Cotula coronopifolia* found in the intertidal zone of the Orange River Mouth.

Sarcocornia pillansii A.J. Scott

This species is a short gnarled shrub that is found in the zone above the MHWS (Mean Water High Spring), referred to as the supratidal zone. They are succulent stemmed plants with reduced leaves (Plate 2). Under stressful conditions, especially high salinities the plants change colour from green through purple to red, and this was presumed to indicate health of the plant. The sediments at the Orange River Mouth floodplain are saline with little free water available. This is due to occasional flooding by salt water followed by evaporation. Numerous annuals and drought tolerant species from terrestrial origin might also be present in this zone (O'Callaghan, 1992). The roots of *S. pillansii* are thick and woody, penetrating the sediment further than 30 cm (Lithauer, 1989) and up to 150 cm (Bornman *et al.*, 2004b). The fibrous roots are restricted to the top 10 cm of the sediment (Lithauer, 1989). These plants are commonly found in salinities ranging from 28 – 34 psu (O' Callaghan, 1992). *Sarcocornia* spp. can cover large areas in the mid to upper saltmarsh with the majority of species being intertidal. However *S. pillansii* will die if submerged for longer than two weeks (Bornman *et al.*, 2005).

Plate 2 The supratidal and floodplain species, *Sarcocornia pillansii.*

2.2.2 Seed bank

Knowledge of the seed bank composition of the site to be rehabilitated and the seed persistence characteristics of plant species is essential to predict the rehabilitation success (Bossuyt *et al.*, 2005). The seed bank is defined as the reserves of viable seed present in and on the sediment surface (Leck & Graveline, 1979; Roberts, 1981; Hopkins & Parker, 1984) and associated litter (Simpson *et al.,* 1989), capable of replacing adult plants (Baker, 1989). The size of the seed bank, the contribution made to it by the different species and patterns of seed distribution reflect the seed production by the resident plant community. Inputs of seed from distant sources may also contribute to the seed bank (Simpson *et* a*l*., 1989). In addition, the seed bank contains not only seeds from the preceding year but may contain seeds from many previous years (Roberts, 1981), especially seeds of species that form long-term persistent seed banks (Thompson & Grime, 1979). The seed bank, in effect, is a temporal dispersal syndrome in which dispersal over time (with seeds) is a strategy for exploiting favourable habitats that are patchily distributed in time and space (Ma *et al.*, 2006). Persistent sediment seed banks allow species to persist in a dormant state until suitable conditions for their germination and establishment occur (Capon & Brock, 2006). However, the physical conditions on saltmarshes may not normally favour the accumulation of persistent, buried seed banks (Hutchings & Russell, 1989).

A major problem with most studies of seed banks is that estimates of seed numbers are very imprecise. Seeds have a clustered spatial distribution, which produce large sampling variances and imprecise estimates of abundance. The spatial distribution of seeds has ecological ramifications. Seed predation or density-dependant competition among seedlings can result in increased mortality if seeds are clustered and a uniform spatial pattern would minimise mortality. There are other instances, however, when clustering of seeds can increase fitness. For example, shallow depressions tend to be moister than the surrounding soil and therefore more conducive to germination in environments where surface desiccation is a limiting factor (Bigwood & Inouye, 1998).

2.2.2.1 Seed bank classification

The most widely accepted classification system for seed banks was proposed by Thompson and Grime (1979). They identified four functional types based on seed persistence and timing of germination.

Seed bank type I: species with transient seed banks present only during summer. Species from this group release all their seeds during late spring and early summer and germinate all at one time in the cooler and wetter conditions in autumn. Species with a type I seed bank appear to be adapted to environments with predicable low summer rainfall (Thompson & Grime, 1979).

Seed bank type II: species with transient seed banks present only during winter. Species in this group have seeds that remain dormant throughout the winter and germinate in spring. Species that have evolved this strategy are usually from regions where plant growth in late autumn and winter is restricted by the cold (Thompson & Grime, 1979). After germination, no viable seed is present in the sediment from species with transient seed banks (Thompson & Grime, 1979). Species that have evolved these strategies are adapted to take advantage of predictable damage or mortality to the extant vegetation (Thompson & Grime, 1979).

Species with type III or IV seed banks have persistent seed banks and not all viable seed germinates at one time (Thompson & Grime, 1979). They found that many species were capable of forming persistent seed banks and this group of plants is heterogeneous with respect to the form and function of the seed bank. However, species that inhabit environments subject to severe and unpredictable disturbances by cultivation, fire or large water level fluctuations generally form a large persistent seed bank. Leck and Brock (2000) demonstrated that all species in the seed banks of temporary and semi-permanent wetlands on the New South Wales northern tablelands "false start" (Thompson & Grime (1979) type III or IV seed banks). Species that have persistent seed banks would have an advantage over species that do not bet hedge (e.g. staggered germination) because of their ability to persist through "false starts". That is if conditions suitable for germination are followed by unfavourable conditions for growth and survival the seed bank is not replenished and may lead to lower recruitment.

Species with a type III seed bank have many seeds that germinate soon after release from the parent plant. This component of the seed output effectively functions as a type I or II seed bank, however a proportion of the seeds do not germinate and become incorporated into the persistent seed bank (Thompson & Grime, 1979). Species that have evolved a type IV seed bank have very few seeds that germinate in the period immediately following release from the parent plant and the species maintains a seed bank, which does not vary greatly in size from season to season. The seed banks of these species are also large in comparison to the annual production of seeds (Thompson & Grime, 1979).

2.2.2.2 Role of the seed bank in vegetation dynamics

Major and Pyott (1960) stated that the seed bank is part of the flora of a system, as it helps to determine the community even though it is not readily evident. The primary role of the seed bank is to ensure the continuation of species after disturbance or natural mortality has killed the extant vegetation (Simpson *et al*., 1989). Population maintenance (Simpson *et* a*l*., 1989) and a reserve of genetic diversity (Templeton & Levin, 1979; Simpson *et al.,* 1989) are also functions of the seed bank.

Species that have seeds that are dispersed by wind or water can disperse great distances from the parent plant. For wind-dispersed species the chances of landing in a suitable environment are low, hence these species tend to produce large numbers of small seed (Cunningham *et al*., 1981). Seed from species adapted for dispersal by water (hydrochory) are more likely to be deposited in a suitable environment and they therefore generally produce a smaller number of larger seeds (Cunningham *et al*., 1981).

Many species have specific requirements for germination and unless those requirements are met, the seed will remain dormant. Factors such as temperature, light, salinity, oxygen, depth of burial and redox potential can influence seed germination (Haag, 1983; Galinato & van der Valk, 1986; Baskin & Baskin, 1989; Baskin *et al.* 1993). In addition, seeds may require scarifying of the seed coat (Haag, 1983; Baskin & Baskin, 1989) or require multiple wet and dry cycles (Brock & Rogers, 1998) to break dormancy. The duration a seed can remain viable is important. Seeds of some species remain viable in dry storage for several years but lose viability when submerged in a matter of weeks (Nicol & Ganf, 2002), while other species lose viability when desiccated (Farnsworth, 2000). Species with seeds that lose viability after being desiccated are generally from wet areas such as rain forest and permanent wetlands or are viviparous (Farnsworth, 2000). Recovery after disturbance may be dependent on dispersal into the disturbed area for species that have short-lived seeds or transient seed banks. In contrast, species that have long-lived seeds or persistent seed banks do not need to rely on dispersal into a disturbed area.

2.2.2.3 Wetland and floodplain seed

The west coast estuarine systems of South Africa, not unlike Australian wetland and floodplain habitats, are highly variable and subject to both flood and drought disturbances. The seed bank provides one mechanism for persistence through unfavourable conditions and regeneration of plant communities after these disturbances (e.g. van der Valk & Davis, 1976; Casanova & Brock, 1990; Grillas *et al.,* 1993; Bliss & Zedler, 1998; Brock & Rogers, 1998). Systems that are subject to extreme desiccation may rely solely on seed for regeneration because asexual propagules may not survive (Thompson & Grime, 1979). The seed bank strategies employed by the species present may reflect adaptations to a hydraulic regime (Leck & Brock, 2000).

Leck and Brock (2000) compared the seed bank strategies from temporary wetlands in Australia and tidal freshwater wetlands in the U.S.A. The temporary wetlands in Australia are subject to water-level fluctuations on a seasonal or aseasonal basis resulting in substantial areas of the wetland with low sediment moisture at times (Casanova & Brock, 1990; Brock & Rogers, 1998; Leck & Brock, 2000; Warwick & Brock, 2003). Tidal freshwater wetlands are also subject to large water level fluctuations, but levels fluctuate diurnally resulting in saturated or inundated sediment in the intertidal zone (Leck & Graveline, 1979; Simpson *et al.,* 1989; Leck & Simpson, 1987; Leck & Simpson, 1994; Leck & Simpson, 1995; Leck & Brock, 2000; Leck, 2003). Leck and Brock (2000) found that all species present in the seed banks of the Australian wetlands formed persistent seed banks, whereas 37 % of the species present in the tidal freshwater wetland had transient seed banks. They also reported that a further 42 % of species present in the tidal freshwater wetlands had short-term persistent seed banks (less than 5 years), in comparison to 11 % from Australian temporary wetlands (Leck & Brock, 2000). The results from this study suggest that species from Australian temporary wetlands are adapted to persisting through “false starts”, where a favourable period for germination may be followed immediately by unfavourable conditions for growth and survival and the plants are unable to replenish the seed bank (Leck & Brock, 2000).

The seed bank strategies of floodplain species also reflect adaptation to the hydraulic regime. Brock and Rogers (1998) reported that 10 out of 16 species from an ephemeral floodplain in South Africa had persistent seed banks (adapted to false starts) and two species required a second wetting cycle to break dormancy.

2.2.3 Seedling recruitment and germination

The period of germination and seedling establishment is one of the most critical stages in the life cycle of halophytes (Ungar, 1982). Saltmarsh seed banks are generally characterised by single species dominance and low species diversity. Recruitment from the seed bank is seen to be an important mechanism for the rehabilitation of vegetation in saltmarshes (Baldwin *et al.*, 1996). Recruitment of halophytes is affected mainly by abiotic factors that relate largely to topographic and edaphic conditions.

2.2.3.1 Saltmarsh zonation

The saltmarsh can be divided into three different marsh areas that correspond to the tidal influence under which the area is exposed. Different saltmarsh species have adapted differently to the subtidal, intertidal and supratidal regimes. Saltmarshes with distinct tidal inundation patterns and elevation gradients are dominated by zonal patterns of saltmarsh plants (Silvestri *et al.*, 2005). Saltmarsh plant zonation is influenced by biotic interaction and temporal and spatial edaphic gradients (Noe & Zedler, 2001). The lower limits of these zones are set by tolerance to physical stress and the upper limits to the lack of competitive ability against more terrestrial species (Bornman *et al.*, 2005).

2.2.3.2 Saltmarsh succession

Succession is a highly complex process and thus the understanding needs careful analysis of vegetation dynamics and their environmental relations (Roozen & Westhof, 1985; Beeftink, 1985). Mosaic structures are common in developmental stages of plant communities (Beeftink, 1985). A low tidal influence will initiate physical and chemical sediment processes, such as the desiccation of the upper sediment layers, oxidation of metal and sulphur compounds and probably desalination. High tidal influences on the saltmarsh will promote the reverse and thus it is supposed that the low tidal influences promote succession processes in the vegetation while high tidal influences initiate regression phenomena (Beeftink, 1985). However some results of long term monitoring reveal that the dynamics of saltmarsh vegetation does not always proceed according to succession (De Leeuw *et al.*, 1993). The sedimentation rate is an important environmental factor involved in the process. A second group of environmental impacts affecting saltmarsh vegetation is natural disturbances such as periods of extreme coldness or long periods of rainfall resulting in waterlogging or submerged conditions. This disturbance may result in dieback of plant populations sensitive to those conditions leaving bare patches of sediment on which secondary succession will take place (Beeftink, 1985).

2.2.3.3 Limitations to seed germination

Seed germination has a different salinity optimum to that of vegetative growth. Seeds may withstand high salinities but only germinate when the salinity is reduced (Ungar, 1978; Riehl & Ungar, 1982; Bakker *et al.,* 1985; Callaway *et al.*, 1990; Callaway & Subraw, 1994; Espinar *et al.*, 2005). Studies on germination requirements of saltmarsh plants have shown that high sediment salinities inhibit the germination and establishment of seedlings (Baldwin *et al.*, 1998; Wolters *et al.*, 2005). Under typical bare patch conditions seed germination is severely limited by high substrate salinities (Shumway & Bertness, 1992; Lindig-Cisneros & Zedler, 2002). Substrate salinities are markedly higher in bare patches than the surrounding vegetation and may be up to 10 psu higher (Shumway & Bertness, 1992).

It is recognised that saltmarsh plants generally germinate better under low salinity conditions (Ungar, 1978; Riehl & Ungar, 1982; Bakker *et al.*, 1985; Callaway, 1990; Lindig-Cisneros & Zedler, 2002). Naidoo and Naicker (1992) found that the maximum germination for *Triglochin bulbosa* L. and *Triglochin striata* Ruiz. Pav. was attained under non-saline conditions and at salinities less than 250 mol m^{-3} NaCl (25 psu). Shumway and Bertness (1992) showed that most saltmarsh seeds germinate better under conditions of freshwater and full light.

Several saltmarsh species show increased seed germination in the presence of salt (Ungar, 1978), with most seeds able to recover germination potential when returned to freshwater after hypersaline treatments (Ungar, 1978, 1979; Kahn & Gul, 1998; Gul & Weber, 2001).

The annual fluctuations in salinity and the duration of flooding are factors that may provide seeds with accurate signals regarding the potential success in establishing seedlings and could be an important system of regulation (Grime *et al.*, 1981; Schultz, 1997; Espinar *et al.*, 2005). Wind and rain aid the release of some *Sarcocornia* spp. seeds and if the sediment is moist the seeds will germinate within two days and thus the seedlings will establish themselves within the first week (Tölken, 1967). Freshwater input to saltmarshes in the form of high flows, floods and rain therefore serves to stimulate the germination of plants and aid establishment.

2.2.3.4 Limitations on seedling establishment and survival

The spatial variation in species density and seedling establishment could be determined by perennial plant cover (Noe & Zedler, 2001), soil texture and elevation (Noe & Zedler, 2001), physical disturbance (Shumway & Bertness, 1992) and seed dispersal patterns (Gul & Weber, 2001). Seedling survival may be limited by abiotic factors such as nutrient availability (Koch and Reddy, 1992), salinity (Callaway *et al*., 1990; Aziz & Kahn, 2001), temperature (Aziz & Kahn, 2001) and sediment moisture (Ungar *et al.,* 1979; Callaway *et al*., 1990) or biotic factors such as shading and predation (Fenner, 1985; Noe & Zedler, 2001). Seedling survival was found to be very low in the high marsh for halophytes affected by salinity (Hutchings & Russell, 1989).

2.2.4 Driftlines and microhabitats

2.2.4.1 Driftlines as depositional environments

Driftlines are mounds of organic debris deposited on the shorelines of water bodies (Houle & Belleau, 2000), and are formed as water levels decrease (Vogt *et al.,* 2006). At the Orange River Mouth, driftlines form discontinuous rings around the edges of water bodies. Seeds accumulate in driftlines which provide optimal conditions for seed germination and seedling establishment.

Large numbers of seeds (150 000.m^{-2}) in driftlines have been reported from the Vindel River in Sweden (Nilsson & Grelsson, 1990), and from the Schiermonnikoog Saltmarsh in the Netherlands (Wolters & Bakker, 2002). In terms of distribution of seeds along driftlines, Nilsson and Grelsson (1990) demonstrated that species with long floating propagules had higher abundance along lakeshores and tranquil reaches than species with short-floating propagules and suggested that the differences in distribution were due to differences in buoyancy properties. Wolters and Bakker (2002) suggested that floating propagules become stranded by ebb tides and these and other organic debris form strandlines in saltmarshes. Cellot *et al.* (1998) reported that strandlines deposited by spring floods in the Rhone River were composed of a combination of redistributed propagules from the existing propagule bank and new vegetative dispersal units.

2.2.4.2 Microhabitats

Ignaciuk and Lee (1980) suggested that the recruitment of four annual driftline species was the result of the presence of larger numbers of seeds in the driftline providing a more suitable habitat for germination and establishment than the surrounding sediment. However, it is possible that the non-seed component of the driftline could inhibit germination of many species (van der Valk, 1981). Seeds buried under the driftline (especially small seeded species) may not have the carbohydrate reserves to emerge through the debris or receive cues such as light to break physiological dormancy (Simpson *et al.,* 1989). In systems where there are unpredictable changes, and where the sediment has limited water-holding capacity, the non-seed component of the driftline could act as mulch.

Perennial plant cover may increase the success of seedling establishment (Noe & Zedler, 2001). Shading creates lower air temperatures and evaporation rates (sediments remain moist with a lower salinity for a longer time) (Noe & Zedler, 2000). Adult vegetation may also act as a trap for the seeds (Smith & Kadlec, 1985; Aziz & Kahn, 1996; Aguiar & Sala, 1997; Bullock & Moy, 2004) from wind and rainfall (Ma *et al*., 2006) leading to a perceived increase in seedling density. Microhabitats under the existing vegetation also allow for increased periods of suitable conditions. Bare areas, on the other hand, have rapidly fluctuating conditions, where surface salinity may increase rapidly becoming 10 psu higher than the surrounding sediment (Shumway & Bertness, 1992; Lindig-Cisneros & Zedler, 2002).

Distribution patterns and depositional environments of seed banks can directly affect rehabilitation potential (Reichmann, 1984; Ma *et al*., 2006) and an understanding of the relationship between microhabitats and seed bank characteristics is important (Ma *et al*., 2006).

2.3 Effect of physico-chemical factors

The germination of saltmarsh plants responds to salinity (Kingsbury *et al.,* 1976; Ungar, 1978; Woodell, 1985; Callaway *et al*., 1990; Shumway & Bertness, 1992; Kuhn & Zedler, 1997; Noe & Zedler, 2000) and moisture (Kuhn & Zedler, 1997; Noe & Zedler, 2000; Vincente *et al*., 2004).

2.3.1 Stress tolerance

Saltmarshes are stressful environments due to the fluctuation of the physico-chemical regime (Nybakken, 2001). Halophytes have adapted to grow in saline conditions, where competition is low. *S. pillansii* have been recorded to grow in salinities up to 70 psu / 80 mS.cm^{-1} (Bornman *et al.*, 2004a, b). If exposed to freshwater, halophytes will grow profusely in the absence of competition. Stress may include: drought, inundation and salinity (Vicente *et al.*, 2004), temperature, oxygen deficiency, competition and herbivory (Nybakken, 2001). Nutrients are often limiting factors for plant growth, however saltmarsh sediments generally have a high organic content from which nutrients can become available.

In semi-arid and arid regions water is scarce. Many saltmarshes occurring in these regions contain plants that are adapted to the low moisture and high salinity environments. Another dominant feature of saltmarshes is the fluctuation of salinity. Although all the factors above exert stress on the vegetation, the lack of water entering the ORM has the greatest influence on the salinity and moisture content of the marsh. Salinity and moisture have been reported to have a significant effect on seed germination and establishment as well as the distribution of adult vegetation (Flowers, 1977; Ungar *et al.*, 1979; Jerling, 1984; Rozema and Blom, 1985; Beare & Zedler, 1987; Broome *et al.*, 1988; Kahn, 1993; Ungar & Woodell, 1993; Aziz & Kahn, 1996; Kuhn & Zedler, 1997; Aziz & Khan, 2000, Kahn *et al.*, 2000; Aziz & Kahn, 2001; Kahn & Gul, 2006).

2.3.1.1 Moisture stress

The natural hydrological regime of most wetlands have been, and are being altered through water abstraction. The hydrological regime influences the amount of moisture available in the sediment of the wetlands. The establishment of wetland plants from seed banks is influenced by these changes in the hydrological regime (van der Valk & Davis, 1978; Poiani & Johnson, 1989; Neill, 1993; Brock & Rogers, 1998; Baldwin *et al.*, 2001).

Plants are exposed to water stress due to extreme sediment water deficits in arid and semi arid environments (Morgan, 1984). Adaptation to water stress in plants involve the reduction of cell dehydration (leaf-shedding, leaf rolling and low stomatal conductance) and / or tolerance through osmotic adjustment. Osmotic adjustment is an important mechanism in drought tolerance as it enables a continuation of cell expansion, stomatal and photosynthetic adjustments and increases plant growth by lowering their water potential in response to decreasing sediment water availability and salinity (Aziz & Kahn, 2003). To maintain water uptake and turgor under saline conditions halophytes need to maintain a water potential that is more negative than that existing in the sediment (Greenway, 1968, Ungar, 1991). A permanent water potential gradient is maintained and is essential for the transpiration (Bornman, 2002). The solutes within each cell create a water potential which becomes more negative upwards through the plant so that the gradient is maintained (Bornman *et al.*, 2004b). The lowering of the water content as a means of lowering the water potential carries with it an inevitable loss of turgor (Bornman, 2002). If growth is unimportant, then this may be a sufficient means of maintaining a positive inflow of water to the plant. Under drought the volume of flow is likely to be low and so supply could be maintained only by an increase in the concentration of ions (Flowers & Yeo, 1986).

Saltmarsh plants can survive under low osmotic potentials due to the fact that they can accumulate organic and inorganic osmoticants in their cytoplasm and vacuoles (Bertness & Ellison, 1987; Vincente *et al.*, 2004). Proline is one of the most common compatible osmolytes in water stressed plants, does not interfere with normal biochemical reactions, making their survival possible under stress (Stewart and Lee, 1974). Proline levels increase significantly over dry periods (Aziz & Kahn, 2003), and drop dramatically after rainfall events (Aziz & Kahn, 2003).

2.3.1.2 Salinity stress

The rainfall at ORM is low and prevents the flushing of naturally and unnaturally accumulated salts (Bornman *et al.,* 2005). An adaptation to increased salinity levels has evolved at the germination stage of development that permits seeds to remain dormant during these stressful periods (Jerling, 1984; Kahn, 1993; Ungar & Woodell, 1993; Aziz & Kahn, 1996; Kahn & Gul, 2006). Seeds of saltmarsh seed banks can remain dormant and germinate when hypersaline conditions are alleviated (Gul & Weber*,* 2001).

Adult halophytes can accumulate salt in their tissues to high concentrations (Barrett-Lennard, 2002; Kahn & Gul, 2006). Most halophytes show optimum growth in the presence of salt (Flowers, 1977; Ungar *et al.*, 1979; Rozema & Blom, 1985; Ungar, 1991; Kuhn & Zedler, 1997; Aziz & Khan, 2000; Kahn *et al.*, 2000; Aziz & Kahn,

2001) but most are inhibited by high salt concentrations, with none showing optimal growth at 35 psu (Ungar, 1991). Many halophytes attain maximum growth under freshwater conditions (Callaway *et al.*, 1990). An increase in sediment salinity affects halophytes by an increase in fresh weight, dry weight and succulence up to the species' salinity tolerance after which it decreases again (Flowers, 1985; Adams & Bate, 1994; Khan *et al.*, 2000; Cronk & Fennessy, 2001). Growth inhibition under saline conditions (Ungar, 1978; Callaway & Subraw, 1994; Kuhn & Zedler, 1997; Zhu, 2001; Zedler *et al.*, 2003; Vicente *et al.*, 2004) is usually associated with dehydration at high salinity, which is due to increased salinity stress and the resultant loss of cell turgor (Naidoo & Rughunanan, 1990) because of inadequate tissue osmotic adjustment (Ungar, 1991).

As plants mature from seed to rhizome-bearing adults, they can become more tolerant of salt (Beare & Zedler, 1987; Broome *et al.*, 1988). The plants are able to take in water despite low external water potentials through the process of osmotic adjustment or osmo-regulation. The plant increases its internal solute concentration with NaCl or other compounds such as proline (Cronk & Fennessy, 2001; Kahn & Gul, 2006; Strewart & Lee, 1974). Other adaptations include salt avoidance through exclusion, secretion, shedding and succulence. Salt exclusion is the most important means of surviving high salt concentrations and all halophytes exclude most of the salt in their growth medium, but salt exclusion at the roots is not entirely sufficient (Cronk & Fennessy, 2001). Plants in the Chenopodiaceae family can absorb Na^+ in mature parts of the roots, blocking them from advancing into the shoots but this process is only effective at low levels of salinity. Succulence in halophytes dilutes the internal salt water and thereby lessens the negative effects of high salt (Flowers, 1985). Succulent plants often close their stomata during the day and open them at night, minimising daytime water loss (evapotranspiration) (Cronk & Fennessy, 2001).

2.3.2 Responses to freshwater input (flooding and rainfall)

Flood events are necessary to periodically leach accumulated salts from the root-zone caused by evapotranspiration (Jolly *et al.*, 1993; Bertness *et al.*, 2002). During a flood, groundwater levels are temporarily raised near the channel by inflow from the river. Although flooding reduces the salinity of the sediment (Callaway, 1990; Neill, 1993) and groundwater, flooding increases the distribution of seed to areas that may not have been reached previously (Vogt *et al.*, 2006). Water-borne seed dispersal (hydrochory) is important for the re-establishment of vegetation (Vogt *et al.*, 2004) in areas away from the adult population. Isolated single large rainfall events and flooding play an important ecological role, especially in dry regions where precipitation is usually highly variable in quantity and erratic in occurrence (Noe & Zedler, 2001). Extreme floods may especially increase seed dispersal (Capon & Brock, 2006; Vogt *et al.*, 2006). During flooding, large amounts of material are transported and deposited along driftlines, creating important microsites for the germination of seed (Xiong & Nilsson, 1997). Building of dams reduces the flood peak and hence the frequency of overbank floods. The sediment input into the floodplain will also decline as a result. These sediments bring organic matter (Nilsson & Grelsson, 1990; Goodson *et al.*, 2001) and nutrients which are important for the floodplain vegetation of semi-arid areas, especially saltmarshes (Noe & Zedler, 2001).

Relatively large floods and rainfall events are required to increase the moisture of the surface sediments to a level suitable for germination (van der Valk, 1981; Gul & Weber, 2001; Shaukat & Siddiqui, 2004). These suitable window periods are normally of short duration (Zedler, 1983). Flooding also affects other life history stages of wetland plants by influencing growth rates or reproductive vigour of seedlings and adult plants (Zedler, 1983; Warwick & Brock, 2003). Although flooding may induce seed germination, die back can occur due to a lack of sufficient water for establishment termed “false-starting” (Rose, 1996).

In saltmarshes, infiltration of the top centimetres of the sediment by large rainfall events increases sediment moisture content and decreases salt levels by dilution and leaching of salts into lower sediment (Cisneros *et al.*, 1999). Smaller rainfall events do not leach salts into lower sediment depths. Evaporation during extended periods of low rainfall and high temperatures (Broome *et al.*, 1988) decreases sediment moisture content, increases salt concentration and creates a hydrostatic gradient (Serfes, 1991) that brings salt water from the subsurface to the sediment surface. This also occurs in areas with large ranges in tidal amplitude where the marsh is not flooded during neap tide (Broome *et al.*, 1988). This is a characteristic of the ORM saltmarsh as well.

2.3.3 Tidal restriction

Tidal energy is an essential driving force in saltmarshes and the loss thereof alters the role of these coastal wetlands (Roman *et al.,* 1984). Tidal exchange is important because the saltmarsh plants are alternately drained and inundated by tidal action.

Reduced tidal exchange can alter intertidal zonations patterns and may even result in the complete loss of vegetation (Silvestri *et al.*, 2005). The hydrology of tidal wetlands is sensitive to changes in topography (Cronk & Fennessy, 2001). In addition to tidal fluxes, several other factors play an important role in the hydrology of wetlands. These include vegetation, rainfall, seasonal variations in evapotranspiration, extreme tidal or flood events and variations in regional groundwater flow (Huges *et al.*, 1998). Tidal exchange between the sea and the estuary can become restricted due to a build up of sediment in the channel.

2.3.4 Mouth dynamics

Opening and closing of the mouth of an estuary results in changes in nutrient exchange and salinity as well as fluctuations in water depth that may severely impact submerged and emergent macrophyte communities within the estuary. Factors affected by flooding include the inhibition of leaf growth, inhibition of stem extension, inhibition of photosynthesis, promotion of extension of growth, senescence and reduced plant productivity (Jackson & Drew, 1984).

2.3.5 Substrate characteristics

Saltmarshes represent a dynamic habitat that has characteristics of both ocean and terrestrial systems (Vernberg, 1993; De Leeuw *et al.,* 1991). Saltmarshes occur on a wide variety of substrates ranging from coarse sands to fine clays with both high and low concentrations of organic matter. Edaphic factors that influence saltmarsh vegetation includes electrical conductivity, sediment moisture content, sediment organic content, sediment particle size, depth to groundwater, compaction and sediment shear strength (Broome *et al.*, 1988; El-Demerdash *et al.*, 1995).

2.3.5.1 Sediment salinity

Sediment salinity plays an important role in determining the vertical and horizontal distribution of saltmarsh species (Wolters *et al.*, 2005). The depth of the water table, rainfall and evaporation on the marsh, groundwater seepage from adjacent land and the salinity of the tidal water that inundates the marsh controls the sediment salinity (Cisneros *et al.*, 1999). The importance of these factors depends on the location within the estuary and the marsh (Haw, 1984). Evaporation of the surface moisture concentrates salts (Thorburn *et al.*, 1992; Jolly *et al.,* 1993; Ridd & Sam, 1996; Gul & Weber, 2001; Bertness *et al.*, 2004). As a result, sediment salinity is spatially and temporally highly variable within saltmarshes. The use of groundwater by perennial vegetation leads to a gradual accumulation of salt in the root zone and this can severely impact the growth and long term survival of the vegetation (Barrett-Lennard, 2002).

High salinity is found in soils and groundwater of arid climates where leaching by rainwater is not effective in diluting the salt solutions (Hillel, 1971). However, freshwater inflow into an estuary is important because it recharges groundwater sources and dilutes the tidal water that inundates the marsh. Lack of freshwater can result in a high salinity that may cause salt accumulation in the intertidal saltmarshes (Adams *et al.*, 1992). In arid areas, most of the moisture in the intertidal saltmarsh comes form tidal flushing (Zedler, 1980) with interstitial water in the saltmarsh sediments influenced by the frequency and duration of tidal flushing, freshwater input in the form of rain or groundwater, evaporation and sediment properties affecting the rate of exchange across the sediment-water interface (Snow & Vince, 1984). Monitoring of sediment salinity during saltmarsh restoration is helpful for the evaluation of success (Wolters *et al.*, 2005).

2.3.5.2 Sediment moisture and organic content

Coastal saltmarshes are characterised by moist sediments and groundwater near the sediment surface (Bornman *et al.*, 2004a). Sediment moisture content is variable both spatially and termporally and is influenced by a number of factors, such as topography, vegetation cover, depth to watertable and rainfall (Gomez-Plaza *et al.*, 2001). In some cases sediment texture is the most important factor influencing sediment moisture (Gomez-Plaza *et al.*, 2001). Moisture can induce seed germination and is a requirement for plant growth. Sediment moisture content is contained in the interstitial pore spaces of sediment particles. The size of the pores depends on the size of the sediment grains. Large sediment grains can accumulate large amounts of water but due to their irregular shape the water is easily transferable through a hydraulic gradient. Clay particles retain the most moisture of all the sediments well but the moisture is often not available to plants due to the small particle size of the sediment (Gomez-Plaza *et al.,* 2001).

Organic matter is important in the sediment as it increases the sediments water holding capacity of the sediments. It also provides the sediment with nutrients.

These nutrients are important for the functioning of the ecosystem as low organic matter can lead to nutrient deficient sediment (Bai *et al.*, 2005). The proportion of sediment organic content depends on the stabilizing capacity of different size fractions, both clay and silt fractions play an important role in semi arid sediments (Gomez-Plaza *et al.*, 2001). Most of the sediment organic matter can be found in the clay and silt fractions (Caravaca *et al.*, 1999; Jacobson *et al.*, 2000). Bornman *et al.* (2005) suggested that large quantities of topsoil, consisting mostly of the silt and clay fraction, were lost to wind erosion from the desertified marsh at the Orange River. They also suggest that an important component of the rehabilitation of the area would be the reintroduction of fine sediment into the marsh.

2.3.5.3 Sediment texture

Sediment characteristics influence saltmarsh colonization and growth. The texture of the soil plays an important role in seed germination, seedling establishment, root growth and distribution of the saltmarsh vegetation (Bornman *et al.,* 2004a). Plant biomass may vary with sediment texture but this has not been related linearly to sediment grain size (Pye, 1987). Sediment particle size also affects the drawdown of seeds, with a larger particle size facilitating more drawdown than smaller sizes (Ma *et al.*, 2006).

Two main sediment sources are found in estuaries, i.e. fluvial and marine. The deposition of these sediments are controlled by the size of the particle and the speed of the water current. The very small particles of silt and clay are deposited at low water flow, with the larger particles settling out at stronger flow. Once silt and clays have consolidated and considerably reduced their water content, it becomes more difficult to erode them.

The water table of the ORM is situated in coarse marine sand, similar to the Olifants River (West coast) (Bornman *et al.*, 2004a, b). A sandy sediment with large pores will have a greater hydraulic conductivity than clay sediment with narrow pores even though the total porosity of clay is greater than that of sandy sediment (Hillel, 1971). In sediment with large pores water is more rapidly conducted, but empties quickly when these sediments become unsaturated. In sediment with small pores, most of the pores remain full and conductive even at times when evaporation causes movement of water to the surface. The hydraulic conductivity does not decrease as steeply and may actually be greater than that of sediment with larger pores (Hillel, 1971). Low unsaturated hydraulic conductivity of the surface sediment layers appears to be a common feature in saltmarshes (Thorburn *et al*., 1992). It also reduces sediment permeability, which in turn would retain rainwater for longer periods at the sediment surface. The upper layer will determine the infiltration of water into a profile with a fine-textured layer over a coarse-textured one (Hillel, 1971). In arid areas, coarse-textured sediments generally support more vegetation than finer textured sediments, due to deeper percolation of moisture. *S. pillansii* occurs on sediments with a high silt content (Bornman, 2002). Although clay retains moisture for longer, most of the moisture is not available to the plant (Gomez-Plaza *et al.,* 2001). The mechanical damage caused by the shrinking of clay sediments during drought, prevents the successful establishment of the plants (Bornman, 2002).

2.3.5.4 Sediment compaction and shear strength

Sediment compaction is an important indicator of sediment moisture content (Amacher & O'Neill, 2004) and can have a variety of effects on productivity within the sediment resulting from changes in both physical and chemical properties. Reduction in pore space resulting from sediment compaction can constrain the size and extent of root systems, reduce infiltration rates (Greacen & Sands, 1980) and the flow of air and gases through the sediment (Cannell, 1977), all of which can limit the ability of the roots to absorb water, nutrients and oxygen. Sediment compaction tends to be higher in open areas than under adjacent vegetation (Amacher and O'Niell, 2004).

Sediment shear strength is also dependent on sediment moisture content. Shear strength measures the consistency of the sediment below the surface (> 3 cm) and it represents compaction of the subsurface sediments. In sediments with high moisture content shear strength is low and increases as moisture decreases. Sediment texture may also play a role in shear strength as large unsorted sediment grains will have lower shear strength than well sorted smaller ones. In saltmarsh surfaces that from a crust, shear strength remains similar from open areas to areas under adjacent vegetation due to the crust limiting the evaporation (Watts *et al*., 2003).

2.4 Depth to watertable and groundwater salinity

The water table can vary in depth, because of variation in elevation and the location of the infiltration water supply (Hillel, 1971). Groundwater is an important source of moisture to plants inhabiting semi-arid saltmarsh environments (Riehl & Ungar, 1982;

Jolly *et al.*, 1993; Pan *et al.,* 1998; Cisneros *et al.*, 1999) and can influence the surface sediment through the capillary rise of chlorides from the water table to the surface (Greenway, 1973; Cisneros *et al.*, 1999). Hypersaline surface conditions, that exist at the Orange River Mouth are unsuitable for plant growth, therefore halophytes extend their roots down to the less saline groundwater. However groundwater of arid climates, where leaching by rainwater is not effective in diluting the salt solution, tend to have high a salinity (Hillel, 1971).

The position of the water table affects the sediment moisture profile (Hillel, 1971). Below the surface, sediment salinity is influenced by the water table salinity for both vegetated and unvegetated sediments (Cisneros *et al.,* 1999). In sediments without vegetation cover, capillary rise is greater than the infiltration rate, resulting in a concentration of salts at the surface (Cisneros *et al.*, 1999). In vegetated sediments the salinity of the sediment is similar or less than the electrical conductivity of the groundwater (Cisneros *et al.,* 1999). Coastal groundwater levels in contact with the ocean fluctuate in response to tides (Serfes, 1991). Diurnal and higher frequency tidal water table fluctuations decrease with distance inland (Serfes, 1991), with fluctuations during spring-neap tidal cycles normally less attenuated.

2.5 Wind and aeolian sand movement

Windblown dust is a common feature of arid ecosystems where sparsely vegetated sediments act as a major source of small particulate matter (Grantz *et al.*, 2003). Dust may settle on leaves, twigs and bark surfaces of plants for extended periods of time (Mctainsh and Strong, *In Press*). This is especially true in desert environments where low rainfall frequency prevents the removal of deposited dust particles from leaves and other plant surfaces (Pye, 1987). Dust deposition on above ground plant organs may induce various chemical or physical effects (Grantz *et al.*, 2003). Excessive dust deposition along unpaved roads, is known to cause defoliation, shoot death and lowered primary production in shrubs from the Namib Desert (van Heerden *et al.*, 2006)

Deposits of windblown dust, commonly referred to as loess, cover significant parts of the surface of the earth (Greeley & Iversen, 1985). The dominant particle size of loess grains is silt (10 - 50 µm in diameter and making up 40 – 50 % of the total), but up to 30 % clay (smaller than 5 µm) and up to 10 % find sand (> 250 µm) may also be present (Greeley & Iversen, 1985). These size distributions may be as a result of sorting during fluid transport (air or water), or the coagulation-aggregation of fine particles such as clay. Quartz is the primary mineral of the coarse grains in most loess deposits, averaging 65 % in abundance with feldspars and various calcium and magnesium carbonates make up the remainder (Pye, 1987). Loess deposits occur primarily in semiarid and temperate zones between latitude 24º and 55º N and 30º and 40º S (Greeley & Iversen, 1985).

Silt and sand size pellets can form by aggregation of clay particles in playa lakes and similar desert environments. The mineral composition of the pellets varies considerably and may include smectite, illite, kaolinite, carbonate, sulphate and chlorides (Pye, 1987). The pellets originate by wind erosion of mud curls and salt-mud efflorescence on dry lake beds.

Aeolian dust transport involves three stages i.e. entrainment, dispersion and deposition.

2.5.1 Entrainment

The nature of wind sediment transport is controlled both by the nature of the airflow near the ground and by the properties of the ground surface over which the flow occurs. The threshold velocity is the strength of the wind needed to move a particle. Factors that increase the threshold velocity of sediments are the presence of water and the concentration of salt (Pye, 1987). Soil moisture is one of the most important factors influencing resistance to wind erosion and the threshold velocity needed for sediment particle movement by wind increases with increasing soil moisture (Azizov, 1977). The intrinsic factor in the increase in soil resistance due to moisture content is the cohesive force of soil water (Weinan *et al.*, 1996). Salinity in surface sediments increases the threshold velocity due to the stronger cohesive forces than moisture alone (Pye, 1987). Nickling and Eccelstone (1981) found that 3 - 4 % is the critical moisture content of surface sediment and that sediment particles were not affected by surface tension if moisture content was below this level. In the case of crusted sediments, abrasion by impacting particles has been observed to be important in breaking up sediment aggregates and releasing fine particles into the air stream (Chepil, 1945; Hagen, 1984). Aeolian action on many natural sediments and sediments, other than well sorted dune sands which contain few fines, involves simultaneous saltation and suspension transport (Pye, 1987). The relative importance of each transport mode depends on the particle size characteristics and structure of the deflated material.

2.5.2 Dispersion

Once dislodged from the bed, a particle may move by sliding, rolling, bouncing (saltation) or in suspension. Sliding and rolling together are known as surface creep (Pye, 1987). When a grain is dislodged it can either return to the surface almost immediately or remain in suspension. The settling of a particle depends on its mass and shape (Pye, 1987). The coarse and medium silt grain size which makes up most of typical loess, is mainly confined to low-level, relatively short-range transport. The nature of the surface over which the dust moves has an important control on the pattern of dispersion. Surfaces formed by dry, featureless sediments can be regarded as reflective in the sense that most of the settling dust particles will be rapidly re-suspended if the wind continues (Chepil, 1945). If the wind containing dust passes over an erodible surface containing fine material, the number of particles leaving the surface may exceed the number of particle settling on it, leading to an increase of suspended particles. On the other hand, water bodies and marshes are highly retentive with regard to settling dust and do not act as sources of newly deflated material (Pye, 1987).

2.5.3 Deposition

Deposition of dust can occur in one of four ways: (1) if there is a reduction in wind velocity and turbulence; (2) the particles are ‘captured’ by collision with rough, moist or electrically charged surfaces; (3) the particles become charged and form aggregates that settle back to the ground; (4) the particles are washed out of atmospheric suspension by precipitation. Deposition can occur where there is a local or regional reduction in wind velocity due to meteorological or topographical factors. Rapid deposition of dust in short-term suspension frequently occurs as wind containing dust crosses a roughness boundary, e.g. between bare ground and a vegetated area (Pye, 1987). The wind velocity above the vegetation canopy is reduced and the threshold velocity may fall below the critical value for re-suspension of settling grains. Settling grains can also be deposited and avoid re-suspension in the lee of topographic obstacles where wind velocity and turbulence near the ground are reduced. Accumulation of dust rarely occurs on the upwind sides of obstacles where an increase in wind velocity and turbulence is usually observed (Pye, 1987). In this study this phenomenon was referred to as ‘tails’. Moist surfaces are capable of permanently trapping dust which comes into contact with them. In some desert and semi-arid areas, the water table is maintained at sufficiently shallow depth to keep the ground surface permanently moist.

Coarse and medium silt grains travelling at relatively low levels in the atmosphere either settle back to the surface or are trapped by obstacles. Part of the fine silt and clay are less affected by changes in surface roughness and are deposited only if aggregation occurs or it they are washed out by rain or other precipitation (e.g. fog and mist) (Zobeck and Popham, 1998). Collision and aggregation of fine particles can occur simply as a result of Brownian motion, laminar shear or turbulent motions, or through the build up of bipolar electrostatic charges (Chepil & Woodruff, 1957).

The grain size distribution of sediment transported above a wind-eroded surface is strongly influenced by the grain size and structure of the surface material and by the strength of the wind (Pye, 1987). It is probable that the smaller size particles are derived from the exposed soils by sandblasting during wind erosion. Clay is removed form the sediment and from the surface of quartz grains. The finer-grained sediment has a higher percentage of clay than the coarse-grained sediment and yields a higher percentage of fine airborne particles (Gillette & Walker, 1977). In areas very close to a dust source particles typically show a wide range of particle sizes (Nickling, 1983) with the mean dust size decreasing with height above the surface (Chepil & Woodruff, 1957). For far-travelled dust particles, the size ranges from 1 – 25 µm with local dust size normally > 25 µm. Dust particles as large as 200 µm have been recorded (Pye, 1987).

Chapter 3: Study Site Description

The Orange River Mouth Wetland was designated as a Ramsar site on the 28th of June 1991 (Cowan, 1995). The Orange River Mouth RAMSAR site (Figure 3.1) was placed on the Montreux Record on 26 September 1995 (http://www.ramsar.org/key_montreux_record.htm, 2006). The Montreux Record is a register of wetland sites on the List of Wetlands of International Importance where

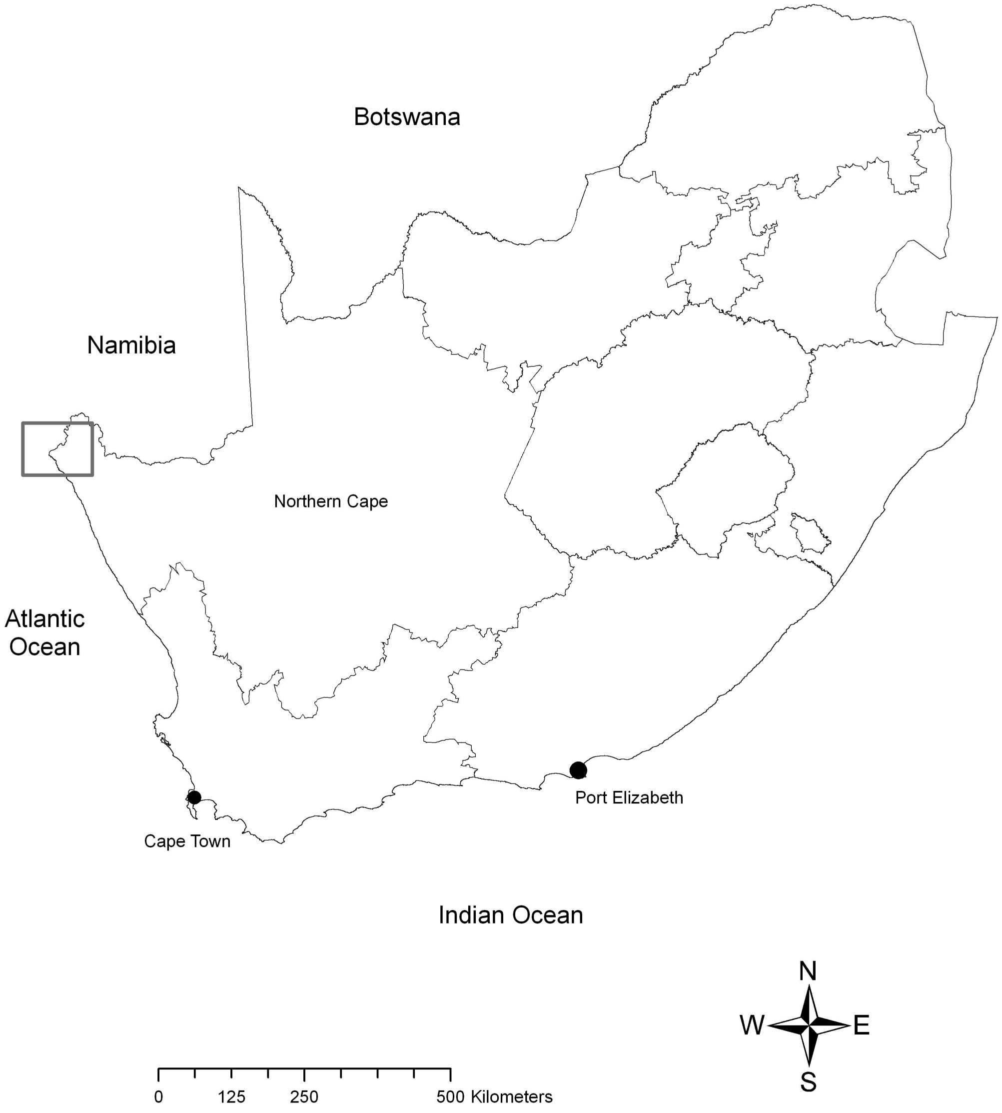


Figure 3.1 A map of South Africa showing the location of the Orange River Estuary

changes in ecological character have occurred, are occurring, or are likely to occur as a result of technological developments, pollution or other human interference. The purpose of the record is to draw national and international attention to the need for rehabilitation of the site.

The study site is situated at the mouth of the South Africa's largest river on the Atlantic coast and forms the border with Namibia. The Ramsar site has an area of about 298 ha (Plate 3). The nearest towns are Alexander Bay in the Northern Cape Province, South Africa and Oranjemund in Namibia. The wetland plays an important role as one of a limited number of wetlands along the arid Atlantic coastline of southern Africa. The wetlands are situated between the north and south flood margins of the Orange River, extending from the Sir Ernest Oppenheimer Bridge to the Atlantic Ocean, a distance of about 10 km. The wetland to the west of Alexander Bay comprises an area of approximately 500 ha, of which at present more than 300 ha is devoid of vegetation. This study concentrated on the desertified saltmarsh to the south and immediately to the north of the causeway.

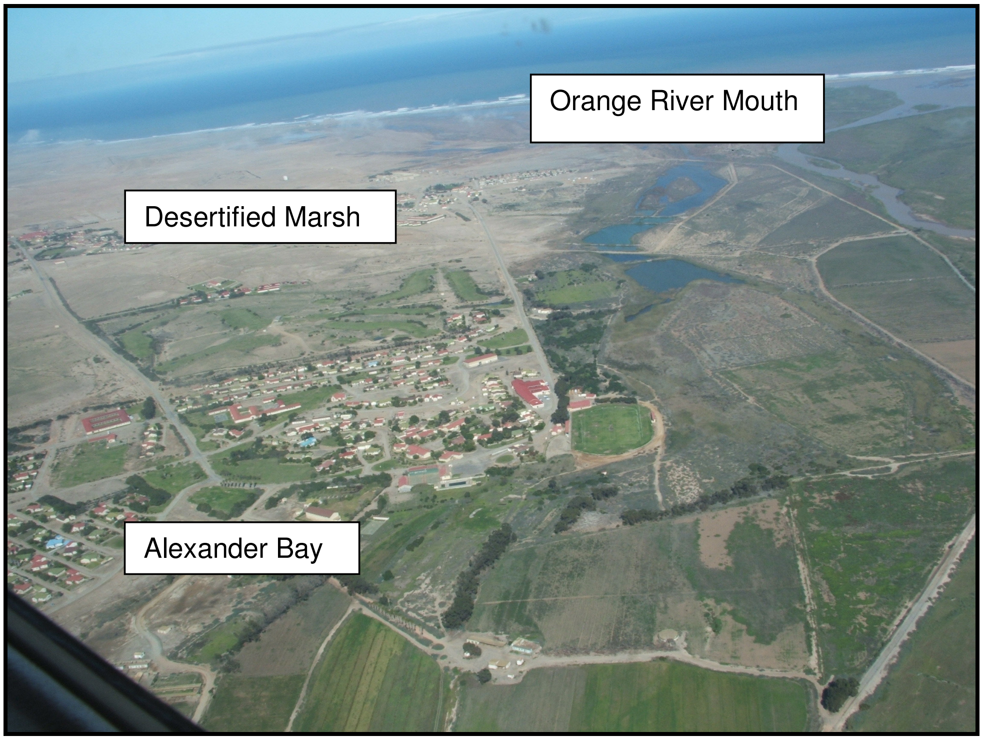


Plate 3 Aerial photograph of Alexander Bay and the Orange River Mouth saltmarsh.

Plate 4 Two hectares of rehabilitated intertidal habitat. *Cotula coronopifola* (bright green) recolonizing the banks of the intertidal channel after the breach in 1997. *Sarcocornia pillansii* (purplish red) forms monospecific stands at higher elevation away from the channel.

3.1 History

Before 1960 there was little damming of the river. The natural flow regime was characterised by strong floods that occurred in the high rainfall summer months, peaking in March, followed by periods of very low flow during the winter (low rainfall) months. The ORM periodically closed prior to river flow regulation, due to the action of longshore movement of beach sands and settling out of river sediments at the mouth. The ORM was a very dynamic system under the natural flow regime as low flows resulted in an increase in the salinity of the estuary whereas closed mouth conditions caused backflooding of the saltmarshes with relatively freshwater. This would have prevented the development of hypersaline sediment conditions in the floodplain.

The decline of the vegetation in the saltmarsh started in 1929 when the backflooding was reduced due to the mouth not being allowed to close (Figure 3.2). In the 1960's a beach access road was constructed that prevented tidal exchange as well as freshwater entering the marsh. In 1979 the marsh was in a similar state to what it is currently, with large desertified areas. In 1986, seven years later the marsh had recovered with areas devoid of vegetation (42 %) lower than in 1979 (63 %). This recovery of the marsh was under natural conditions with the casueway still intact. However anthropogenic impacts continued to place the vegetation at the ORM under pressure until events in 1988, 1993 and 1995 lead to the destruction of the marsh. In 1988 a large flood breached the embankments of the marsh and flooded the floodplain. The presence of the causeway prevented the floodwater from draining and the marsh was covered in water. The beach berm was breached to allow the floodwater to drain but the mouth was left open open for a month allowing seawater to enter the marsh and the salinity of the sediment and groundwater increased. The flood also caused scouring of vegetation resulting in large dieback. The plants that survived the scour were killed off by the standing water and increased sediment and groundwater salinity. In 1993, the mouth closed and backflooding occurred. This event would have the reduced salinity but because there was no way for the water to drain, evaporation concentrated the salts on the floodplain. The combination of standing water and increased sediment and groundwater salinity related to the flood event caused the latest die back in saltmarsh vegetation (Bornman *et al.,* 2005).

The river flow has been controlled since the construction of more than 23 major dams (especially the Vanderkloof Dam and the Gariep Dam) and numerous weirs (Venter & van Veelen, 1996). It is estimated that by 1989 the Mean Annual Runoff (MAR) had been reduced to 50 % of the natural MAR (DWA 1990). The sustained release of water from the large dams even in winter (mainly for hydro-electricity generation and agriculture) meant that the river mouth now remains open almost permanently. The dams have also caused a considerable reduction in the occurrence and severity of small to medium sized floods with the result that the ORM is only properly flushed during severe flood events. The sustained flows in winter prevent mouth closure and backflooding, which is considered to be a crucial part of the ecological functioning of the saltmarsh at the ORM. The future objective of flow regulation in the Orange River is to maintain a low flow during winter, which will allow the mouth to close and through backflooding the saltmarsh will be inundated (Venter & van Veelen, 1996). The minimum flow that is required to keep the mouth open is 12 $m^3.s^{-1}$ and the maximum flow, to ensure the mouth closes in winter, is 5 $m^3.s^{-1}$ (Venter & van Veelen, 1996). Floods are vital to the normal functioning of the Orange River Mouth. Major floods occurred during summer in March 1967, February 1974, February/March 1988, March 2000 and February 2006.

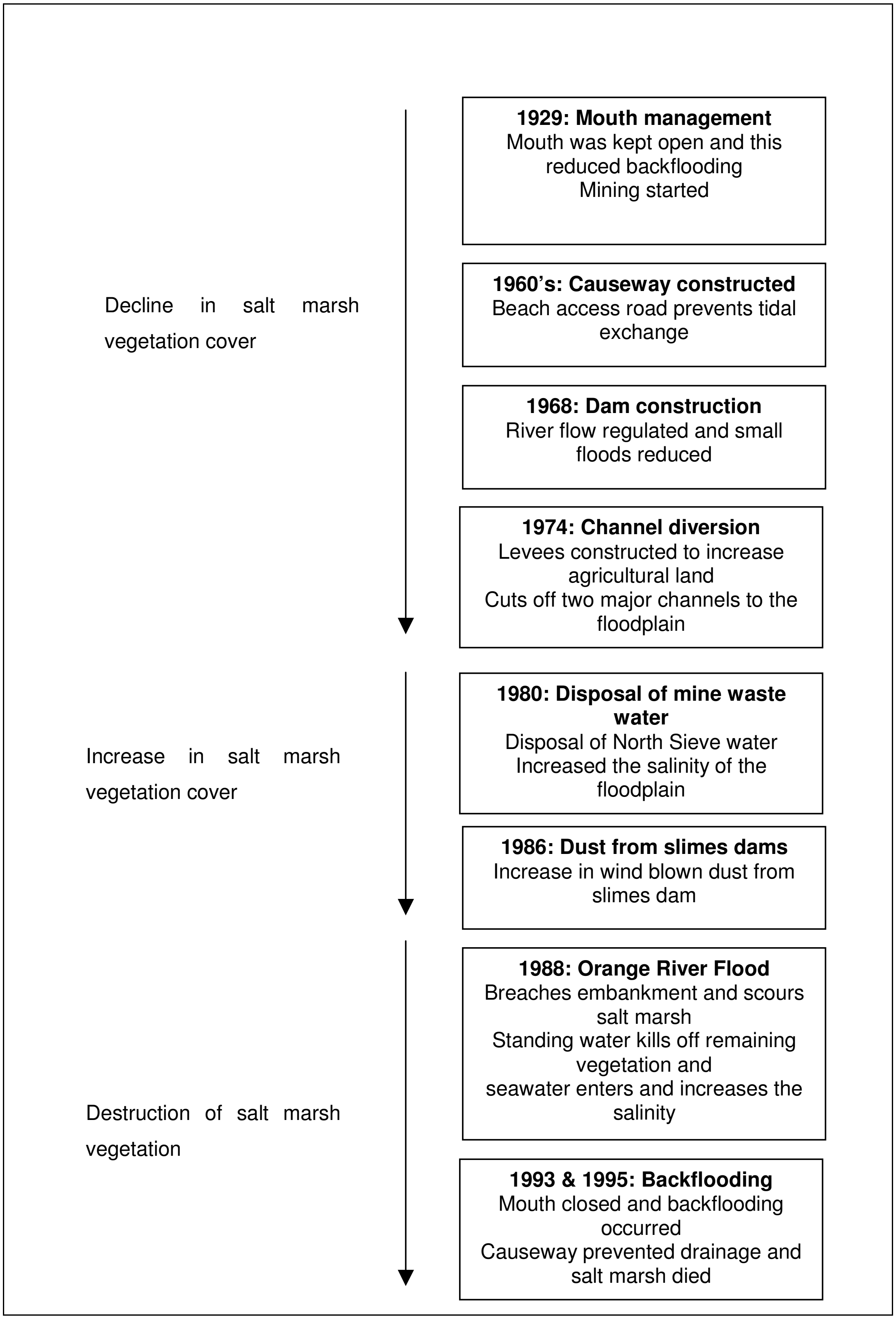


Figure 3.2 History of the degradation of the Orange River Mouth (Bornman *et al.*, 2005).

3.2 Working for wetlands

Working for Wetlands rehabilitation projects are intended to produce sustainable environmental outcomes, using implementation models that simultaneously contribute to the employment creation and skills transfer objectives of government's Expanded Public Works Programme. Funding for these activities is provided by the Department of Environmental Affairs and Tourism to the South African National Biodiversity Institute (SANBI), the home of the programme. All rehabilitation interventions are undertaken within the context of improving the integrity and functioning of the ecosystem, and include measures that address both causes and effects of degradation (http://www.sanbi.org/research/wetlandprog.htm#prog).

Rehabilitation at the ORM was first initiated through the removal of a section of causeway near the mouth by the mining company, in 1997 which allowed water to permanently return to a small section of the saltmarsh (Plate 6). The intertidal saltmarsh species, *Cotula coronopifolia,* re-colonised the new area subjected to tidal flows. *Sarcocornia pillansii* responded to the increased tidal flushing through increased cover abundance, growth and seed production in the supratidal zone surrounding the newly re-created intertidal area. Although water was also able to flow into the remainder of the wetland through the breach during times of flooding and backflooding, the outflow was restricted to the same small breach resulting in long periods of standing water in the desertified marsh. This meant that instead of the water flowing easily through the system and flushing out the salts leached from the sediment, it receded slowly re-depositing most of the salt as the water evaporated and the problem of hypersalinity persisted.

The first phase of the Working for Wetlands project began at the Orange River Mouth in June 2005 and the objective was to breach the causeway at strategic places to allow drainage of the marsh after a backflooding event (Plate 5). The ability of the breaches to achieve their desired goal was demonstrated during 2006 when the wetland was subjected to two instances of flooding. The breaches facilitated drainage of the marsh. The marsh was completely covered by water and although the breaches were successful, two more sites on the causeway were identified as requiring further breaching (Working for Wetlands, 2006).

Plate 5 An example of a break through the causeway allowing water to flow into the desertified marsh from the main Orange River channel.

Plate 6 An oblique aerial photograph showing an example of a recent breakthrough in the causeway by the Working for Wetlands programme.

3.3 Climate

All climate data presented in this section for Alexander Bay was provided by the South African Weather Services.

3.3.1 Rainfall

Average annual precipitation at the Orange River Mouth is only 41.4 ± 2.9 mm with an average annual evaporation of over 2000 mm.y^{-1}. Rain generally falls in gentle showers spread over the season from May to August, but can, in some years, fall with a single, heavy shower as occurred in 2004 and 2006. High flows in February 2006 were the result of a rainfall event in the catchment of the Fish River (a tributary of the Orange) in Namibia as well as flooding of the Orange River itself. This together with the constricted mouth resulted in backflooding occurring. In April 2006, 80 % of the mean annual precipitation for Alexander Bay occurred (Figure 3.3 and 3.4). This coupled with heavy rainfall in the catchments of the Orange River resulted in a higher than usual discharge of water from the dams upstream and backflooding occurred during spring tides. This was an unusual event as the marsh is generally dry (Figure 3.3) with no standing water present. Sampling for this study took place in June 2005 and April/May 2006, capturing both a dry and a wet period.

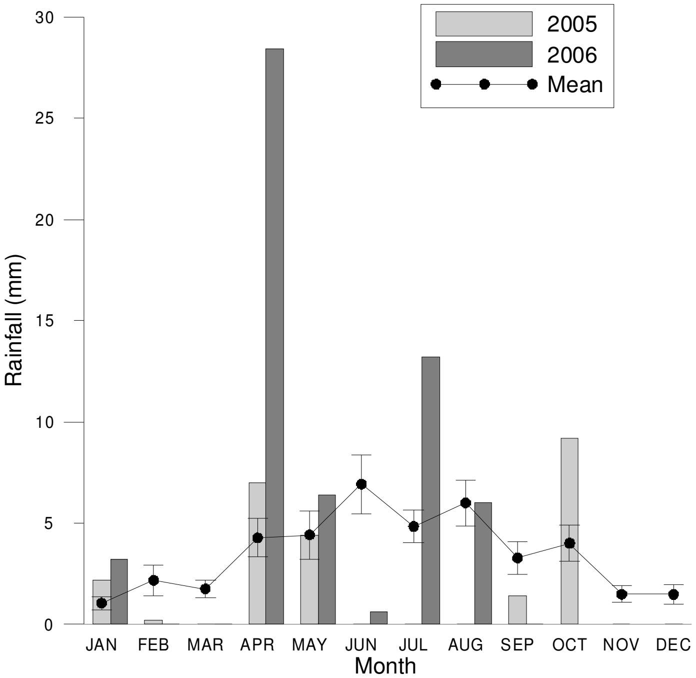


Figure 3.3 Monthly rainfall for 2005 and 2006, against the mean rainfall (n = 42 years).

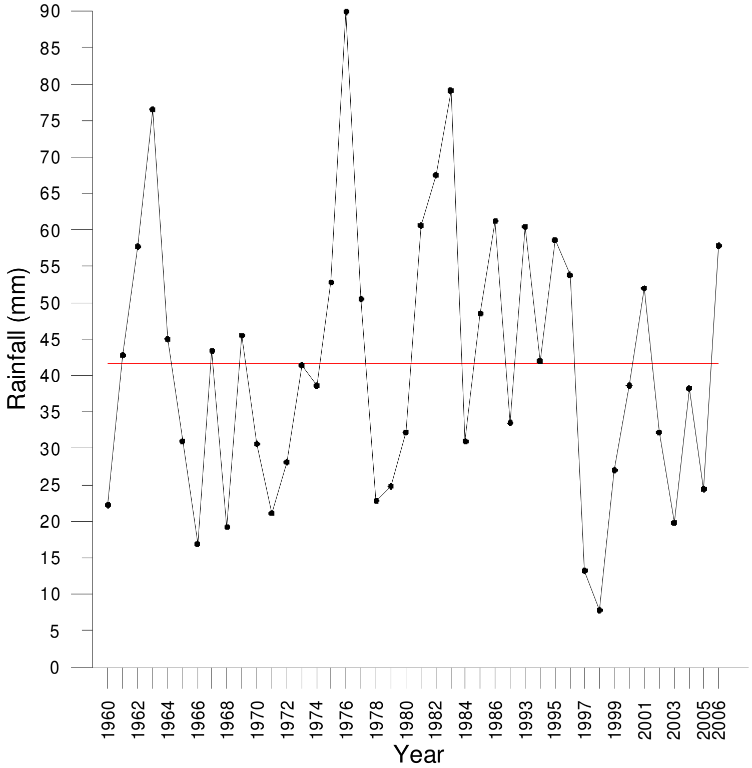


Figure 3.4 The annual rainfall from 1960 to present. The horizontal line represents the mean (41.4 ± 2.9 mm) (n = 42 years).

3.3.2 Wind

Winds are a dominant element of the climate along the west coast of Southern Africa with southerly sea breezes occurring during most of the year (Figure 3.4). These southerly winds exhibit daily variations in direction and strength, but are usually strongest during the afternoon (Figure 3.5b, c). In winter, high pressures over the interior of southern Africa cause strong south-easterly winds, also known as berg winds. These hot winds are responsible for the area's warmest weather. The wind from the Atlantic Ocean is important as it brings in cool moist sea air that supports the vegetation in the western part of Namaqualand. The southerly winds have deflated the desertified marsh surface, transporting the fine sediment to the river, and smothering the saltmarsh at the north-eastern extreme of the floodplain.

Wind speed is highest in the summer months when rainfall is low and the marsh is dry (Figure 3.5). It is also strongest at 14:00 during the day and these strong winds persist until evening (Figure 3.6). This long period of strong wind allows for large amounts of sediment to be transported into the north-eastern corner of the marsh.

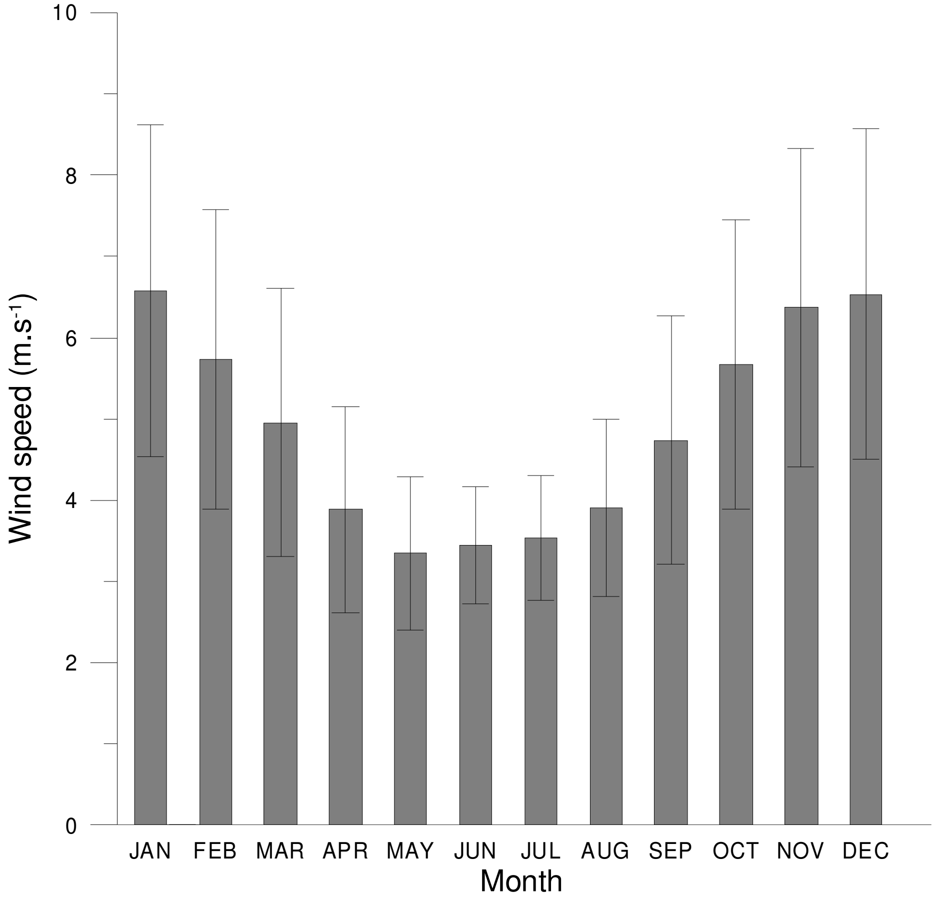


Figure 3.5 The wind strength during the year (bars = ± SE) (n = 42 years).

Table 3.1 Wind direction ranges applicable to Figure 3.6

Direction	Degrees	Direction	Degrees
N	348.75 to 11.25	S	168.75 to 191.24
NNE	11.25 to 33.74	SSW	191.25 to 213.74
NE	33.75 to 56.24	SW	213.75 to 236.24
ENE	56.25 to 78.74	WSW	236.25 to 258.74
E	78.75 to 101.24	W	258.75 to 281.24
ESE	101.25 to 123.74	WNW	281.25 to 303.74
SE	123.75 to 146.24	NW	303.75 to 326.24
SSE	146.25 to 168.74	NNW	326.25 to 348.74

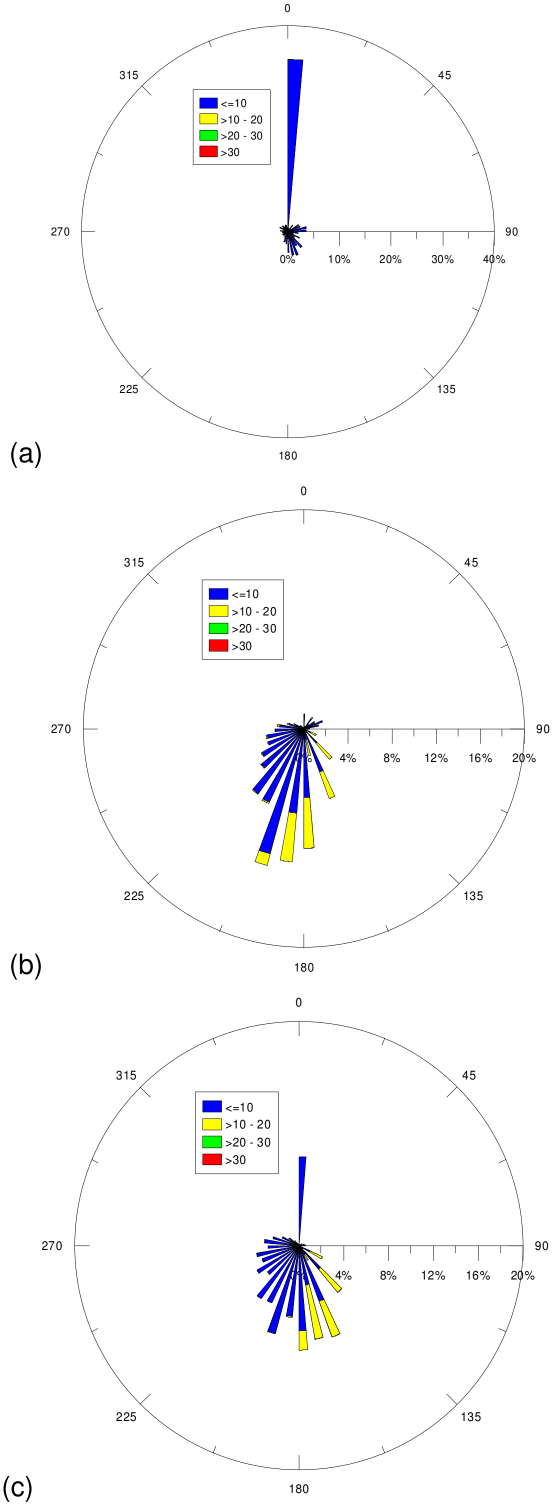


Figure 3.6 Wind speed mean during the year at (a) 08:00, (b) 14:00 and (c) 20:00.

3.3.3 Temperature

During the summer months the minimum and maximum temperatures vary by ~ 8 ºC. While in winter (May to October) the temperature difference is markedly higher with minimum temperatures reaching 3 ºC and maximums around 20 ºC.

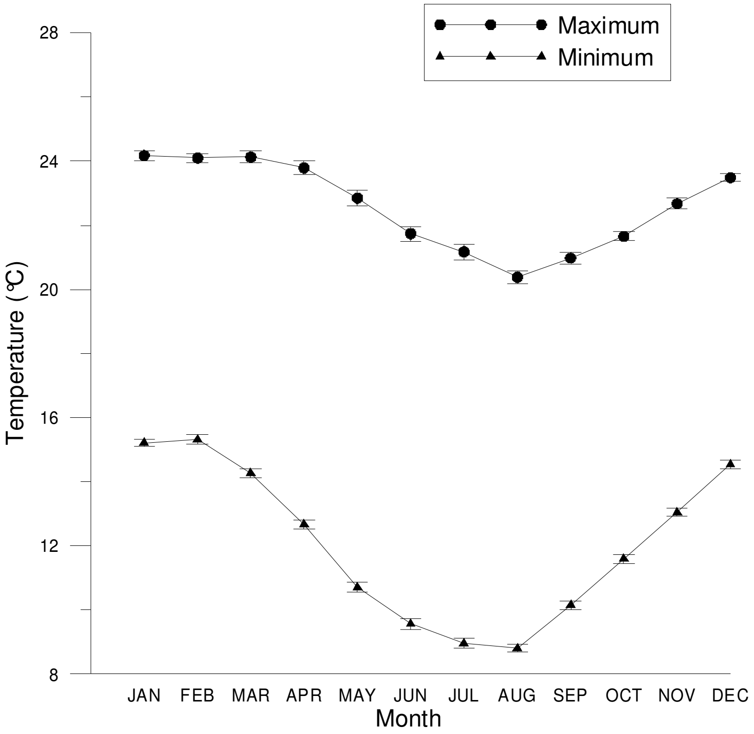


Figure 3.7 Minimum and maximum temperatures for Alexander Bay.

3.3.4 Fog

The importance of fog in the water relations of saltmarsh plants in semi-arid environments have not been quantified, but it is expected to be significant. Fog occurs on average for 89 days of the year mostly from March to May (Bornman *et al.,* 2005).

Chapter 4: Sediment Characteristics

4.1 Abstract

Three sampling areas were chosen to investigate the sediment characteristics of the ORM saltmarsh and the suitability for seed germination and adult survival. Site A was representative of the general marsh area, Site B was thought to have favourable sediment conditions for saltmarsh growth because of the large numbers of seedlings present and Site C was prone to inundation by wind blown sediment. The sites were sampled in 2005 (dry conditions) and in 2006 following high rainfall and river flooding. Sediment samples were collected from the surface and close to the water table.

In 2005 the marsh was characterised by hypersaline sediment due to the saline groundwater. Electrical conductivity (EC) was significantly higher in the surface sediment in Site A (121 ± 4.8 $mS.cm^{-1}$) and C (107 ± 22.2 $mS.cm^{-1}$), while Site B (69 ± 6.4 $mS.cm^{-1}$) had uniform values between the surface and bottom samples. High rainfall and river flooding in 2006 lowered the EC in Site A (51 ± 4.9 $mS.cm^{-1}$) and B (44 ± 3 $mS.cm^{-1}$) and resulted in no difference between surface and water table values. However for Site C, the surface sediment EC (125.9 ± 3.1 $mS.cm^{-1}$) was still significantly higher than the water table sediment (41.8 ± 2.4 $mS.cm^{-1}$) due to wind blown sediment. Moisture content for Site A, B and C was significantly lower at the surface (7.8 ± 1.5 %) than at depth (10.9 ± 0.6 %), while organic matter showed no significant difference between surface (2.3 ± 0.5 %) and water table (1.5 ± 0.2 %) sediments.

Differences in sediment characteristics were also compared for three microhabitats i.e. driftlines, open sites and under vegetation. Sediment characteristics under the vegetation were most favourable for the germination and establishment of seedlings, while conditions fluctuated in the open unvegetated areas in response to evaporation and tidal changes (i.e. spring vs. neap). Sediment compaction and shear were higher for the open areas compared to under the vegetation. A hard crust formed in the open areas reducing the ability of the seed to germinate.

The electrical conductivity of the sediment was in many instances above the tolerance range of the dominant species *Sarcocornia pillansii* (> 80 $mS.cm^{-1}$). This study showed that rainfall and flooding were important in lowering sediment electrical

conductivity, which would promote the growth of adult vegetation. The survival of these plants would ensure seed production and provide suitable microhabitats for seedling survival.

4.2 Introduction

Sediment is a basic component of ecosystems and its role extends beyond being a substrate for plant growth. Although edaphic characteristics are partly influenced by plant cover it is ultimately the sediment that largely determine the plant species composition (Álvarez-Rogel *et al.*, 2006). Understanding the edaphic factors would make it possible to assess where natural rehabilitation will take place especially with regard to seed germination and establishment.

Bornman *et al.* (2004a) found that the sediment of the desertified floodplain was characterised by high surface salinity. Sediment salinisation occurs in semi-arid wetlands where there is capillary movement of water from shallow groundwater to the surface and where evaporation of the salts are concentrated within the upper sections of the sediment profile (Jacobson *et al.,* 2000; Kahn & Gul, 2006). The aim of this study was to investigate some of the changes that may have occurred since the study in 2004 (Bornman *et al*., 2004b). In 2005 the marsh was dry with only a small amount of rain falling in that year. Rainfall in 2006 was above average causing the Orange River to flood and large areas of standing water were present in the marsh. The effect of the rainfall event on the hypersaline conditions of the marsh was investigated. The two sampling trips (2005 and 2006) allowed for a comparison of the behaviour of the marsh and vegetation response under dry and wet conditions.

Due to the size of the marsh area (> 300 ha), three sites (A, B and C) were chosen to represent the different characteristics present within the marsh. Site A was representative of the general marsh area, Site B was thought to have favourable sediment conditions for saltmarsh growth because of the large numbers of seedlings present and Site C was prone to inundation by wind blown sediment.

The sediment characteristics of the microhabiats (driftlines, unvegetated and vegetated sites) were sampled to assess suitability for seedling establishment and survival. The microhabitat sampling sites occurred in the vicinity of Site A and were affected by spring and neap tides. At spring high tide the lower most break through in the causeway allowed water to push into the marsh and the extent is shown in Figure 4.1 During the neap tide the water did not extend as far as the spring tide and the marsh dried out rapidly (Figure 4.1).

The overall objective of this chapter was to assess the suitability of the study site for seed germination, seedling establishment and adult survival. This would provide an understanding of the rehabilitation potential of the salt marsh from remaining saltmarsh vegetation namely the two dominant halophyte species, *Cotula coronopifolia* L. and *Sarcocornia pillansii* A.J.Scott.

4.3 Materials and Method

The same sampling procedure was used in 2005 and 2006, however the 2006 survey included sediment characteristics of the microhabitats provided by existing vegetation, which potentially created a suitable environment for the germination of seeds and the establishment of seedlings. Sediment taken from each site was analysed for electrical conductivity, sediment moisture, organic and carbonate content, sediment particle size, depth to groundwater and sediment compaction and shear. The samples were taken back to the laboratory at the Nelson Mandela Metropolitan University (NMMU) for analysis.

4.3.1 Site description

Three main study sites were selected due to their different characteristics. Sites A, B and C were assessed in terms of suitability for germination and establishment of seedlings as well as survival of mature vegetation. This in turn determines the rehabilitation potential of the marsh. Site A represented the largest portion of the marsh and the prevailing sediment conditions. Site B had seedlings and was thought to have favourable environmental conditions for seedling growth. Site C represented an area affected by wind blown sediment. All sites included supratidal and floodplain habitat. Site A was the only site that had intertidal habitat. These sites are described further in terms of their physical characteristics.

Plate 7 Site A with large bare areas and sparse vegetation cover.

Plate 8 Healthy vegetation was found in close proximity to the tidal channel in Site A.

Site A, represented the relatively intact saltmarsh and was characterised by vegetation clumps of the dominant supratidal and floodplain species, *Sarcocornia pillansii* (Plate 7). Site A also had intertidal habitat dominated by dense stands of *Cotula coronopifolia*. Seven sites in this area were chosen, two located in the intertidal and the remainder in the supratidal / floodplain areas. The largest area of the marsh consists of supratidal and floodplain habitat and therefore the emphasis was placed on the sediment characteristics of these sites (Plate 8).

Sites A1 and A2 were located within the supratidal zone with some intertidal habitat, where the vegetation cover was high (Figure 4.1). Some intertidal habitat was present in this area due to a break in the causeway created in 1997 to introduce freshwater to the desertified marsh. Sites A3, A4 and A5 were located along spring tide driftlines of the main tidal channel flowing into the desertified marsh. Site A6 was inland and represented a floodplain habitat. Site A7 was adjacent to an inlet that formed part of the main channel of the river and flowed in from the break in the causeway during times of high flow.

The second site (Site B) occurred to the west of the causeway (Figure 4.1) (Plate 9). The presence of numerous seedlings growing along driftlines was the main feature of the site. Three locations were sampled (B1, B2 and B3) in this site. The site was inundated from overtopping events where estuarine water would breach the natural sand berm protecting the site (Figure 4.1) (Plate 10). This site borders the river and the source of freshwater provides favourable conditions for growth.

Plate 9 Large stands of *Sarcocornia pillansii* in Site B. Culverts were originally installed to facilitate the movement of water through the causeway with little success.

Plate 10 The berm separating the tidal channel on the right, from Site B on the left.

The third site (Site C) was located west of the causeway furthest from the sea (Figure 4.1). This site was characterised by the accumulation of sediment on the vegetation (Plate 11). The mature vegetation was smothered by the sediment transported across the marsh by the prevailing wind (Plate 12). Three locations were chosen within this site. The first site (C1) was characterised by vegetation completely covered by sediment. The second site (C2) was located at the edge of the vegetation where sediment accumulation was visible on the vegetation (monospecific stand of *S. pillansii*). This vegetation was gradually being smothered (covered 50 % of the vegetation) by wind blown sediment. The third site (C3) was at the centre of the monospecific stand where the influence of the sediment was limited due to the protection offered by the plants on the periphery.

Plate 11 Accumulation of wind blown sediment covering the vegetation at Site C.

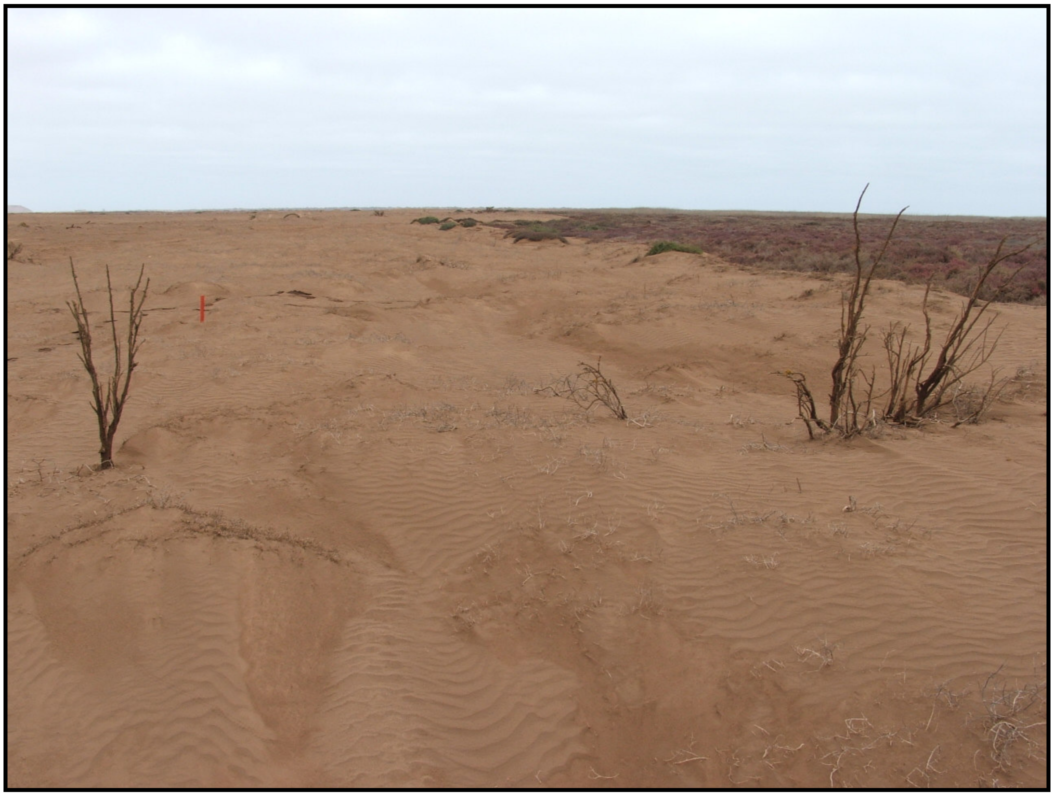


Plate 12 Loss of vegetation at Site C due to smothering by wind blown sediment.

Sediment was collected from the surface and near the water table using a hand auger at each site during both sampling sessions. The sediment was analysed for electrical conductivity, sediment moisture content, sediment organic content, sediment carbonate content and particle size. Depth to groundwater was only recorded in 2005 due to extensive flooding and a shallow water table in 2006. Sediment compaction and shear were only recorded in 2006.

4.3.2 Electrical Conductivity

Sediment electrical conductivity ($mS.cm^{-1}$) rather than salinity (measured in practical salinity units or psu) was measured because, although seawater and saline groundwater are dominated by sodium and chloride, salts normally precipitate out as calcium sulphate and sodium chloride, both contributing to the ion concentration of the sediment (Bornman, 2002). The term salinity will be used interchangeably with electrical conductivity.

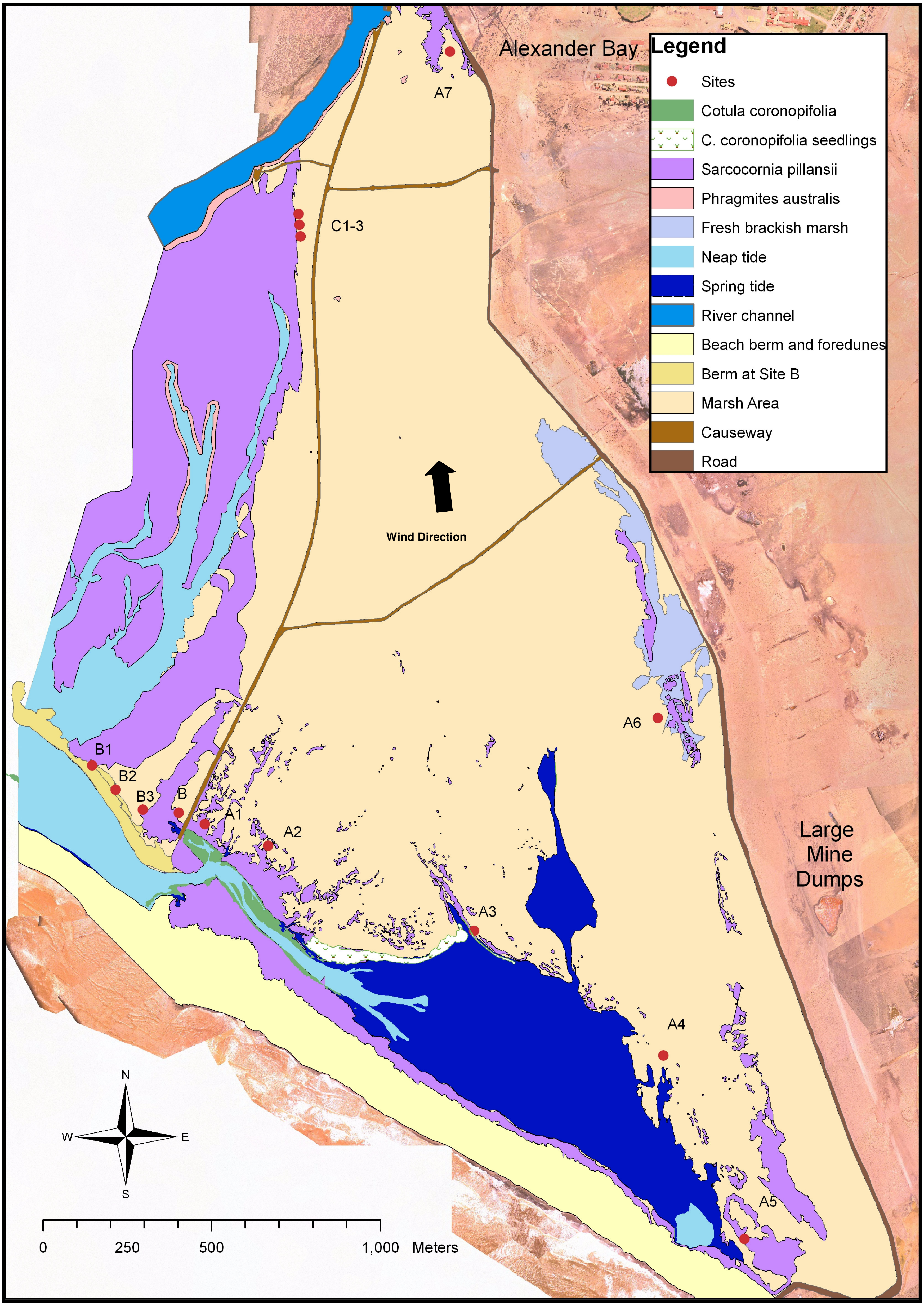


Figure 4.1 A detailed map of the Orange River Mouth an dthe location of the sites sampled. A smaller pullout map is appended in Appendix 2 (Figure A.2.1) for ease of use.

Sediment electrical conductivity of a saturated paste was determined according to the method by the Non-Affiliated Soil Analyses Working Committee (1990). An air-dried sediment sample (250 g) was placed in a beaker and de-ionised water was added until saturated. The sediment was tested for the properties of a saturated paste, i.e. no excess water on the surface of the sediment, and if too dry more de-ionised water was added. The amount of de-ionised water used to attain saturation for each sample was recorded. The saturated paste was allowed to stand for at least one hour. The saturated paste was then filtered through a Bachman filter using Whatman No. 40 filter paper. The electrical conductivity of the filtrate was measured using an YSI 30M/10FT hand held conductivity meter.

4.3.3 Sediment moisture content

Sediment moisture content was determined according to the methods set out by Black (1965). Into a crucible, 10 g of each sediment sample was weighed and placed in a drying oven for 48 hours at 100 °C. The samples were reweighed and the moisture content was determined using the following equation:

$$\left(\frac{Mw-Md}{Mw}\right)*100$$

Where *Mw* is the initial wet mass and *Md* is the mass after drying.

4.3.4 Sediment organic content

The sediment organic content was determined according to the method of Briggs (1977). The dried sediment samples from the sediment moisture content experiment were placed in a muffle furnace (ashing oven) for 8 hours at 550 °C. The crucibles were removed from the ashing oven and placed in a desiccator containing anhydrous silica crystals until cool. The percentage organic matter was calculated as a loss of mass during ashing as a percentage of the initial mass using the following equation:

$$\left(\frac{Md-Ma}{Md}\right)*100$$

Where *Md* is the initial dry mass and *Ma* is the mass after ashing.

4.3.5 Sediment carbonate content

These analyses did not follow the site descriptions as carbonates were not seen as being present or having a significant affect on the distribution of the vegetation. If high levels of carbonates were recorded they may have had an effect on the solute content of the sediment and changed the sediment properties in terms of pH. Random sediment samples were used from the sediment collected for the other analyses. In 2005, 18 samples were analysed and 21 in 2006.

Sequential loss on ignition (LOI) is a common and widely used method to estimate the organic and carbonate content of sediments (Dean, 1974, Heiri *et al.*, 2001). After oven-drying the sediment to a constant weight, organic matter was combusted in a first step to ash and carbon dioxide at a temperature of 550 ºC. The LOI was then calculated using the following equation:

$$LOI_{550} = ((DW_{105} - DW_{550})/DW_{105})*100 \qquad (1)$$

Where LOI_{550} represents LOI at 550 ºC (as a percentage), DW_{105} represents the dry weight of the sample before combustion and DW_{550} the dry weight of the sample after heating to 550 ºC(both in g). In the second step, carbon dioxide evolved from carbonate, leaving oxide and LOI was calculated as:

$$LOI_{950} = ((DW_{550} - DW_{950})/DW_{105})*100 \qquad (2)$$

Where LOI_{950} is the LOI at 950 ºC (as a percentage), DW_{550} is the dry weight of the sample after combustion of organic matter at 550 ºC, DW_{950} represents the dry weight of the sample after heating to 950 ºC, and DW_{105} is again the initial dry weight of the sample before the organic carbon combustion (all in g). Assuming a weight of 44 g mol^{-1} for carbon dioxide and 60 g mol^{-1} for carbonate ($CO_3{}^{2-}$)(Heiri *et al.*, 2001), the weight loss by LOI at 950 ºC multiplied by 1.36 should then theoretically equal the weight of the carbonate in the original sample (Heiri *et al.*, 2001).

4.3.6 Sediment particle size

The hydrometer method, as set out by Gee and Bauder (1986) was used. Approximately 40 g of sediment, previously air-dried, was accurately weighed out in a pre-weighed beaker and allowed to equilibrate with the atmosphere overnight. To this sample was added 100 ml of a 50 $g.l^{-1}$ solution of Sodium hexametaphosphate

$(Na_3PO)_6$ and 250 ml distilled water. The beakers were placed on a mechanical shaker for 1 hour prior to the start of the experiment. The sediment mixture was then placed in a 1 litre measuring cylinder and the volume made up to 1 litre with distilled water. The cylinder was closed off at the mouth and shaken by hand for at least 1 minute. Two drops of amyl alcohol were added to remove the foam on top of the sample. A hydrometer was inserted after 30 seconds, 60 seconds, 3 minutes, 1.5 hours and 24 hours. A blank containing a similar solution was also prepared but without sediment. The temperature of the solutions was taken using a mercury thermometer. The readings were then used in the following equations to calculate the percentage size fractions in the sample.

Determine C, the concentration of sediment in suspension in $g.l^{-1}$, using:

$$C = R - RL$$

Where R is the uncorrected hydrometer reading (in $g.l^{-1}$) and RL is the hydrometer reading of the blank solution.

The determination of P, the summation percentage for the given time interval, using:

$$P = \left(\frac{C}{Co}\right)*100$$

Where *Co* is the oven dried weight of the sample.

Determine *X*, the mean particle diameter in suspension in µm at time *t* using:

$$X = \theta t^{-1/2}$$

Where θ is the sedimentation parameter (µm min $^{1/2}$) and is a function of the hydrometer settling depth, solution viscosity and particle and solution density.

$$\theta = \left(\frac{18\eta h'}{[g(\rho_s - \rho_l)]^{1/2}}\right)$$

Where h' is the hydrometer settling depth (cm), p_s = sediment particle density (g.cm^3), p_l = solution density (g.cm^{-3}), g = gravitational constant (cm.s^{-2}) and η = fluid viscosity in poise (g.cm^{-1}.s^{-1}).

The relationship of the settling depth to the hydrometer dimensions were approximated by:

$$h' = -0.164R + 16.3$$

Where R is the uncorrected hydrometer reading.

The summation percentage was calculated as follows:

$$P_{2\mu m} = m \ln\left(\frac{2}{X_{24}}\right) + P_{24}$$

Where X_{24} is the mean particle diameter in suspension at 24 hours, P_{24} is the summation percentage at 24 hours, and m was determined using the following equation:

$$m = \frac{P_{1.5} - P_{24}}{\ln(X_{1.5} - X_{24})}$$

Where m is the slope of the summation percentage curve between X at 1.5 hours and X at 24 hours. $X_{1.5}$ is the particle diameter in suspension at 1.5 hours, and $P_{1.5}$ is the summation percentage at 1.5 hours. This procedure was repeated for the 30 second and 60 second readings.

The particle size ranges were: Sand > 50 µm, Silt 2 – 50 µm and Clay < 2 µm (Singer, 2006).

4.3.7 Compaction and Shear

Compaction and shear were measured at seven locations in Site A to determine the differences in the micro-climates/habitats that occurred within the site in 2006 and the suitability of the sites for seedling establishment. The three microhabitats were the driftlines, open areas and under the vegetation. The open areas were considered to be more than 1 m away from any vegetation. Compaction measures the compressive strength of the sediment. Twenty replicate samples were taken for each of the microhabitats in the locations at Site A. The microhabitats were sampled during spring and neap conditions. Only Site A7 remained wet, while the other sites experienced drying during neap tide. The MATEST Pocket Penetrometer (S070-S071) was used to test surface sediment. Shear strength is used to determine the shear strength of clay sediments having low consistency, especially when soft. The shear strength test was carried out in close proximity to that of compaction tests. For shear strength the MATEST Pocket Vane Shear Apparatus (S075-S076) was used.

4.3.8 Depth to groundwater

Depth to groundwater was determined by augering with the aid of a hand auger to at least 30 cm below the water table. The watertable was allowed to stabilise over night. Depth to groundwater was measured using a graduated meter stick. Depth readings were taken during spring high and low tide.

4.3.9 Statistical analysis

One-way ANOVAs were used to determine significant differences among means. A Tukey post – hoc test was carried out where significant differences among means were found. All statistical analyses were conducted using Statistica (Version 7, 2003), StatSoft, Inc.

4.4 Results

4.4.1 Electrical Conductivity

4.4.1.1 Site A

In 2005, A2 was the only location with salinity lower than the tolerance range for the supratidal species (< 80 $mS.cm^{-1}$) (Figure 4.2a). The sediment electrical conductivity in most of the marsh area was higher than 100 $mS.cm^{-1}$ (Figure 4.2a).

4.4.1.2 Site B

Site B (Figure 4.2b) had electrical conductivity readings that ranged from 41 $mS.cm^{-1}$ for groundwater to 75 $mS.cm^{-1}$ for the sediment surface. There were no significant differences between B1 - B3 ($p > 0.05$; $n = 18$).

4.4.1.3 Site C

Electrical conductivity (Figure 4.2c) in watertable sediments at Site C ranged from 26.49 ± 13.6 $mS.cm^{-1}$ to 52.2 ± 0.17 $mS.cm^{-1}$ ($n = 9$), with the surface sediment of C1 reaching a high electrical conductivity of 93 ± 11.9 $mS.cm^{-1}$ ($n = 3$). However there was no significant difference between the sites ($p < 0.05$, $n = 18$). Large standard errors indicated the spatial variability in sediment characteristics.

4.4.1.4 Site and Year Comparisons

Site B had significantly lower electrical conductivity ($p < 0.05$, $n = 60$) than Site A for both surface and watertable sediments (Figure 4.3a) but were not significantly different to watertable sediments in Site C ($p > 0.05$, $n = 60$). Overall there was no significant difference between the surface sediments of Site A (120 ± 4.7 $mS.cm^{-1}$) and C (105 ± 6.7 $mS.cm^{-1}$). Site C was the only site where the surface sediments were significantly ($p < 0.05$, $n = 18$) higher (106.5 ± 22.2 $mS.cm^{-1}$) than its associated groundwater sediments (45.4 ± 2.4 $mS.cm^{-1}$). In 2006 (Figure 4.3b) the surface sediments in Site C were still significantly higher (125.9 ± 3.1 $mS.cm^{-1}$) than the associated watertable (41.8 ± 2.4 $mS.cm^{-1}$) sediments ($p < 0.05$, $n = 18$) and were also significantly higher than all the other sites in 2006 and Site B in 2005 ($p < 0.05$, $n = 36$). The differences between sites for the two years showed that the conductivity had decreased across all sites.

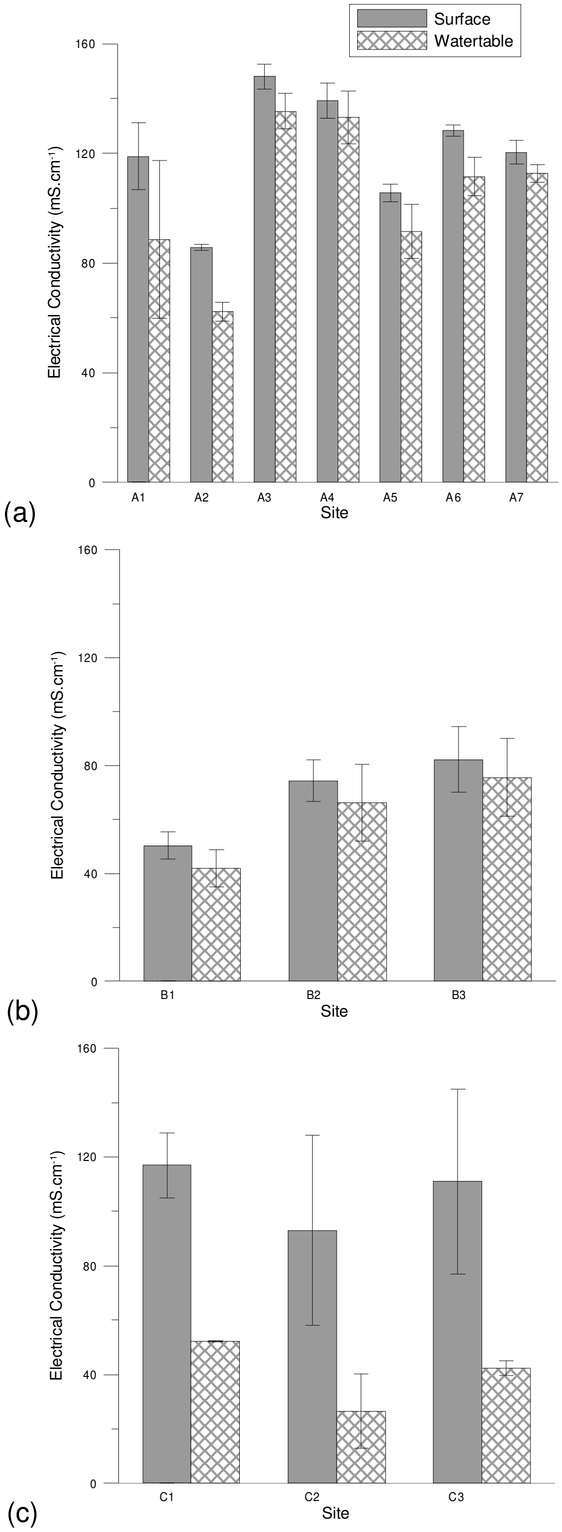
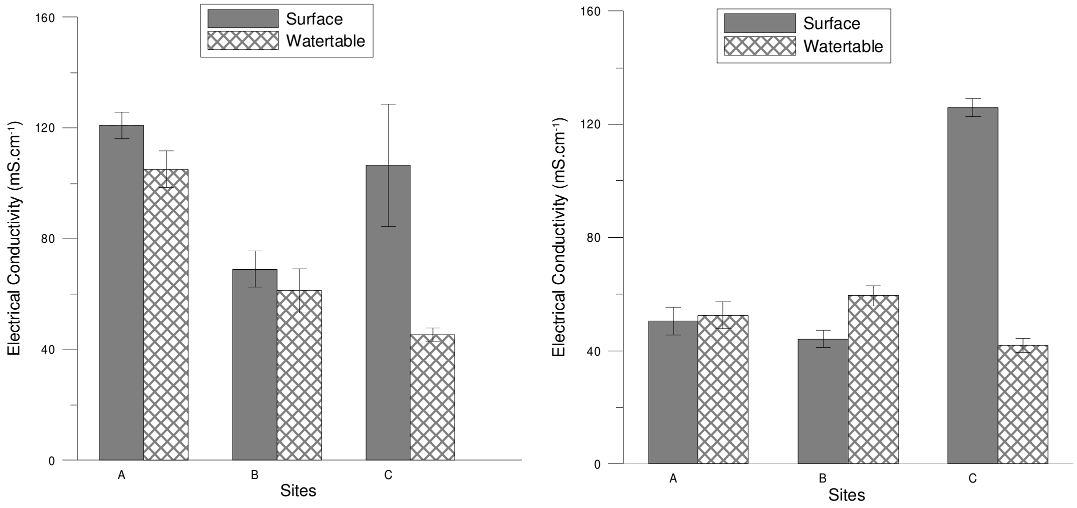


Figure 4.2 Sediment electrical conductivity for surface and watertable sediments within (a) Site A, (b) Site B and (c) Site C (bars = ± SE).



(a)

(b)

Figure 4.3 Sediment electrical conductivity for surface and watertable sediments for sites A, B and C in (a) 2005 and (b) 2006 (bars = ± SE).

4.4.1.5 Microhabitats

Figure 4.4 shows the relationships between spring and neap tide conditions. There were no driftlines in A1 (Figure 4.4a). The sediment electrical conductivity was highly variable. The driftlines and vegetated microclimate at A7 showed significantly lower values of 4.5 ± 0.5 mS.cm^{-1} and 2.5 ± 0.3 mS.cm^{-1} respectively ($p < 0.05$, $n = 74$), compared to the unvegetated areas (26.6 ± 18.8). The driftline electrical conductivity was lower than 80 mS.cm^{-1} in A2, A3, A5 and A7 and were significantly lower than A4 and A6 ($p < 0.05$, $n = 74$). In the open unvegetated areas (Figure 4.4b) the surface sediment electrical conductivity was higher than 80 mS.cm^{-1} during the neap tide and were significantly higher than the watertable sediment in sites A1, A4, A5 and A6 ($p < 0.05$, $n = 74$). All sites except A6 had a sediment EC below 80 mS.cm^{-1} in the areas under the vegetation (Figure 4.4c).

Figure 4.5. is a repetition of the data displayed in Figure 4.4 presented differently to more easily compare the different microhabitats. Electrical conductivity (EC) (Figure 4.5) was significantly higher ($p < 0.05$, $n = 74$) under neap conditions in the driftlines (80.4 ± 17.3 mS.cm^{-1}) and in the open unvegetated areas (83.5 ± 7.6 mS.cm^{-1}) compared to the spring tide. EC was lowest under the vegetation and showed no significant change during both spring and neap tide. The electrical conductivity in the driftlines remained constant during both tides ($p > 0.05$, $n = 74$).

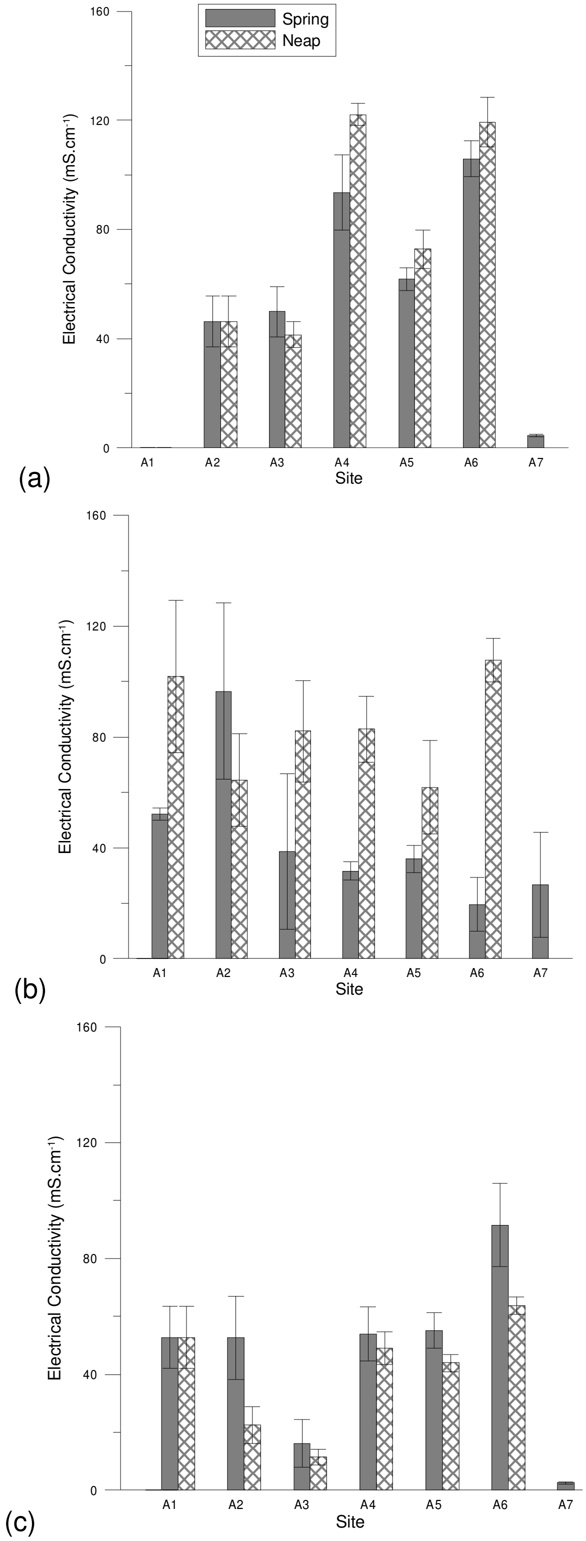


Figure 4.4 Electrical conductivity for the sediment in (a) driftlines, (b) open unvegetated areas and (c) under the vegetation for the locations within Site A in 2006 (bars = ± SE).

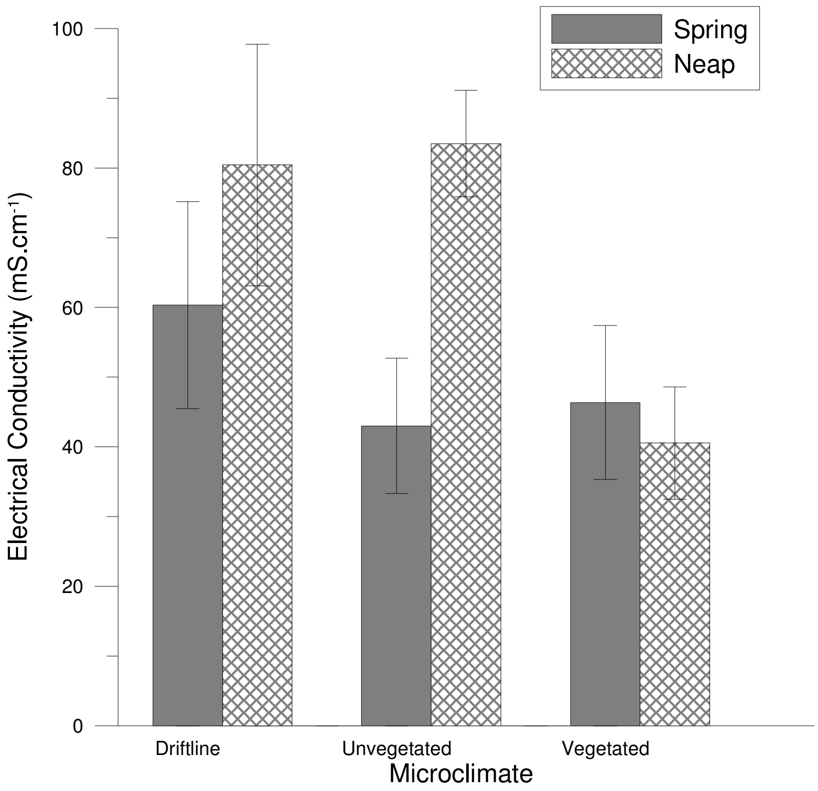


Figure 4.5 Mean electrical conductivity for the microhabitats within Site A in 2006 (bars = ± SE).

4.4.2 Sediment moisture content

4.4.2.1 Site A

Sediment moisture content ranged from 7.53 ± 0.93 % to 12.77 ± 0.6 % in Site A (Figure 4.6a), with no significant differences recorded between the sites or between surface and watertable samples ($p < 0.05$, $n = 45$).

4.4.2.2 Site B

The surface of B1 (Figure 4.6b) was significantly lower compared to all the other sites ($p < 0.05$; $n = 18$). This was probably because of the sandy nature of the sediment. The watertable sediment samples had a significantly higher moisture content than the surface samples with a mean of 11.57 ± 0.34 % compared to the surface mean of 5.62 ± 2.52 % ($p < 0.05$, $n = 18$).

4.4.2.3 Site C

The surface of C2 was significantly lower (Figure 4.6c) than the surface sediment of sites C1 and C3 ($p < 0.05$; $n = 18$). The moisture content increased significantly with depth, with a mean of 4.86 ± 1.27 % at the surface and 10 ± 0.8 % at the bottom ($p < 0.05$, n = 18).

4.4.2.4 Site and Year Comparisons

In 2006 there were no significant differences between the moisture content of all sites ($p > 0.05$, n = 80) (Figure 4.7). In 2005 (Figure 4.7a) the surface sediments in Sites B (6.3 ± 1.7 %) and C (6.3 ± 1 %) were significantly lower than that of Site A (10.7 ± 0.5 %) ($p < 0.05$, n = 80). There was less variability between surface and watertable sediment moisture values in 2006 ($p > 0.05$, n = 18) (Figure 4.7b).

4.4.2.5 Microhabitats

Sediment moisture content was uniform throughout the marsh. The microhabitat moisture did not differ significantly between microhabitats or during spring and neap tides ($p < 0.05$, n = 74) (Figure 4.8)

Although significant differences occurred between sites within each of the different microhabitats, for the purposes of understanding the role of microhabitats they were not discussed in detail. These results are included in Appendix 1 (Figure A.1.1).

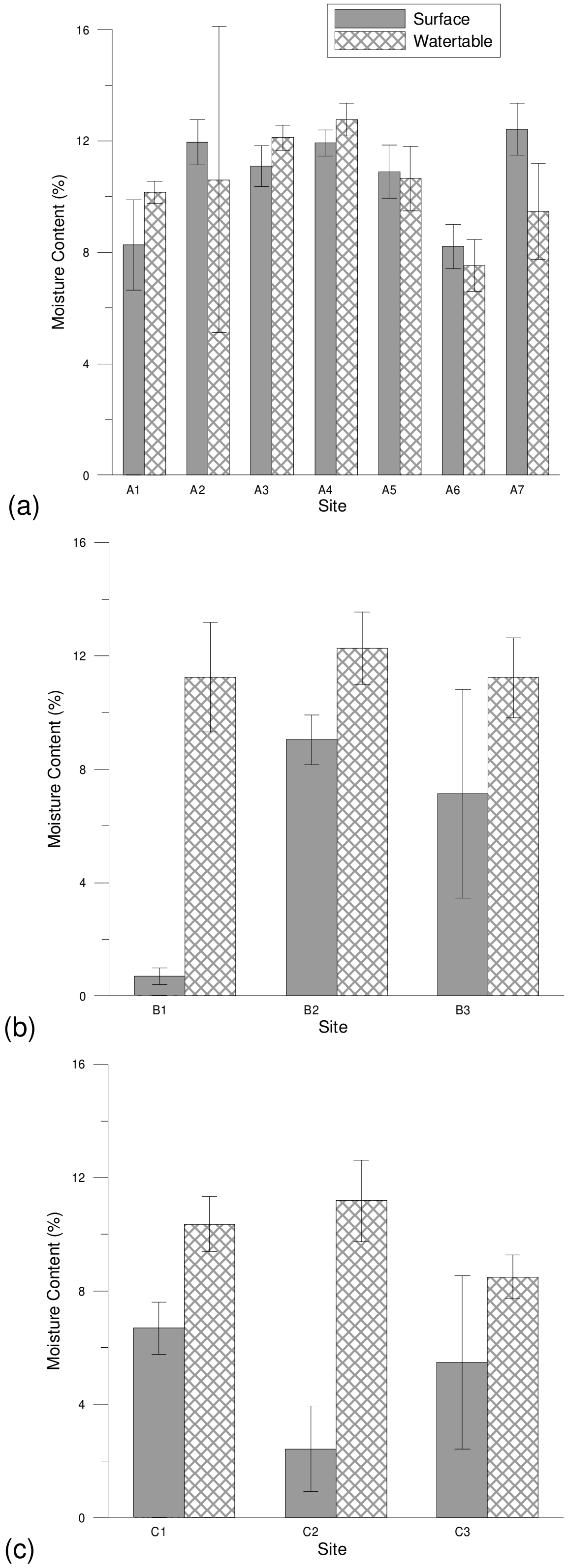
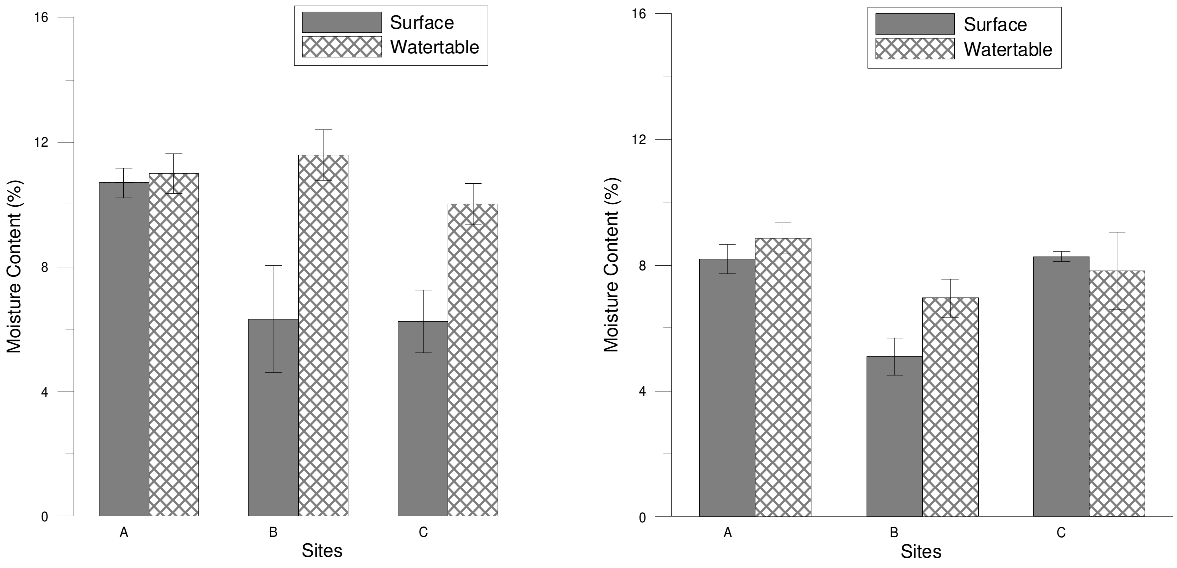


Figure 4.6 Moisture content of the surface and watertable sediment within (a) Site A, (b) Site B and (c) Site C (bars = ± SE).



(a) (b)

Figure 4.7 Sediment moisture content for surface and watertable sediments for Sites A, B and C in (a) 2005 and (b) 2006 (bars = ± SE).

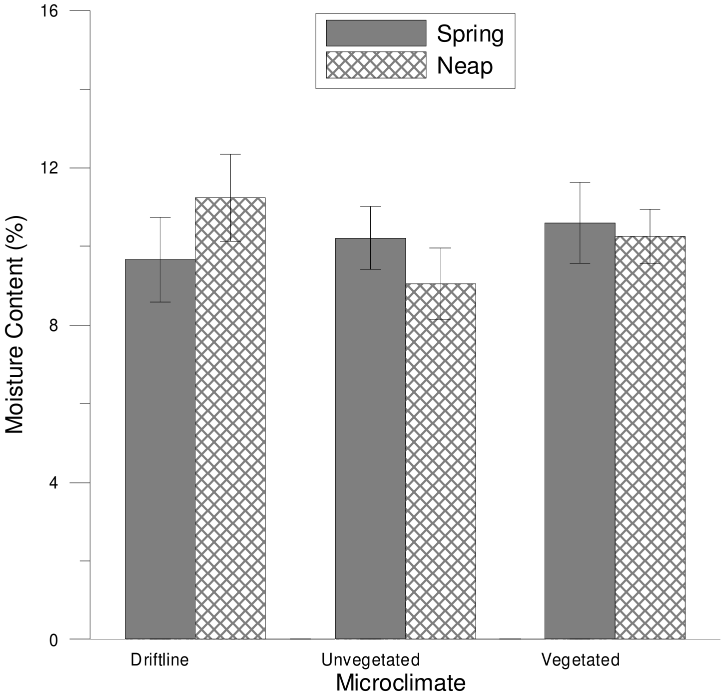


Figure 4.8 Mean sediment moisture content for the microhabitats of the saltmarsh during spring and neap tide in 2006 (bars = ± SE).

4.4.3 Sediment organic content

4.4.3.1 Site A

Site A (Figure 4.9a) had a similar sediment organic content for the surface sediments, with the watertable sediments significantly lower in Site A5, A6 and A7 ($p < 0.05$, $n = 42$). Site A2 had a higher organic content in the watertable sediments but it was not significant ($p > 0.05$, $n = 42$).

4.4.3.2 Site B

Sediment organic content was uniform throughout Site B (Figure 4.9b) and ranged from 0.53 ± 0.04 % at the surface of B1 to a maximum of 1.68 ± 0.09 % at the surface of B2 ($p > 0.05$, $n = 18$). The mean for the site including both surface and watertable sediment was 1.37 ± 0.17 %.

4.4.3.3 Site C

In Site C (Figure 4.9c) the surface sediment organic content was significantly higher than the watertable sediment ($p < 0.05$, $n = 18$). Site C showed two extremes in terms of sediment organic content, where the surface of C1 had significantly higher ($p < 0.05$; $n = 18$) organic matter (3.19 ± 0.22 %) than all three Sites (C1, C2, C3) and the bottom of the C3 had the lowest value (0.69 ± 0.12 %).

4.4.3.4 Site and Year Comparisons

A comparison of sites in 2005 (Figure 4.10a) showed that the sediment organic content of the surface of C was significantly higher than the other sites ($p < 0.05$; $n = 80$). In 2006, Site C still contained the highest organic content ($p < 0.05$; $n = 80$). The sediment organic content decreased from 2005 (Figure 4.10a) to 2006 (Figure 4.10b), but not significantly ($p > 0.05$, $n = 80$).

4.4.3.5 Microhabitats

The sediment organic content (Figure 4.11) for the microhabitats sampled showed similar patterns to that of sediment moisture content in terms of variability. Under spring tide conditions the sediment organic content was not significantly different for the microhabitats ($p > 0.05$, $n = 74$). The individual site data for Site A appears in Appendix 1 as the data may be used in the future for monitoring the status of the marsh (Figure A.1.2).

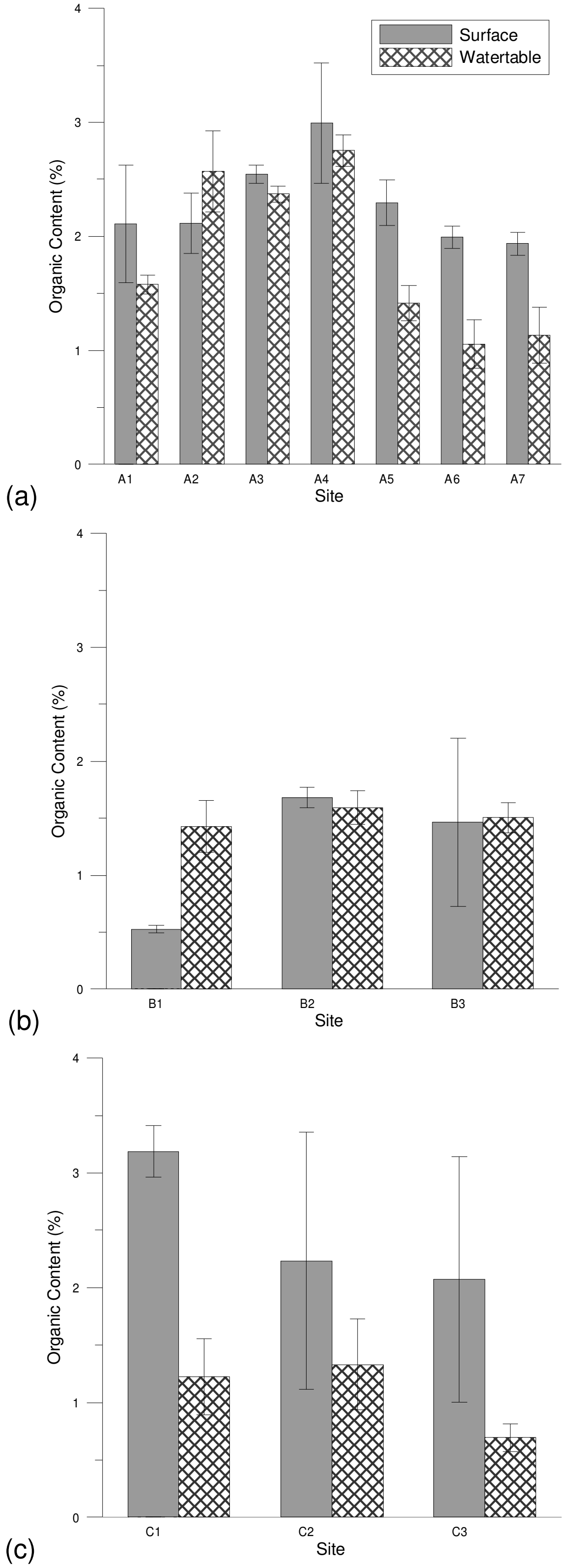
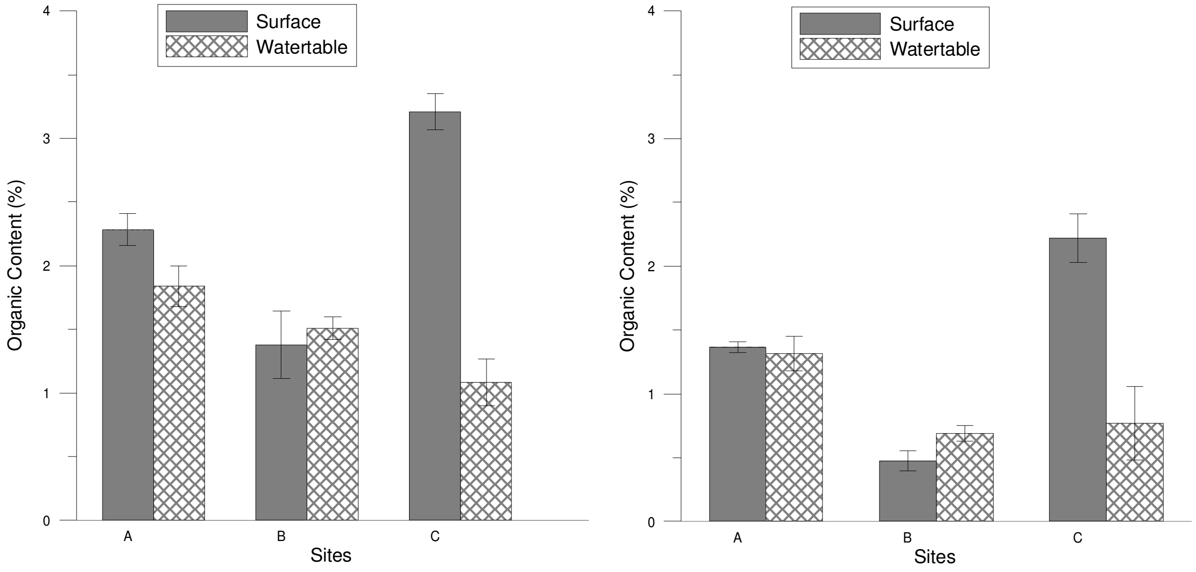


Figure 4.9 Organic content of surface and watertable sediments in (a) Site A, (b) Site B and (c) Site C (bars = ± SE).



(a) (b)

Figure 4.10 Sediment organic content for surface and watertable sediments for Sites A, B and C in (a) 2005 and (b) 2006 (bars = ± SE).

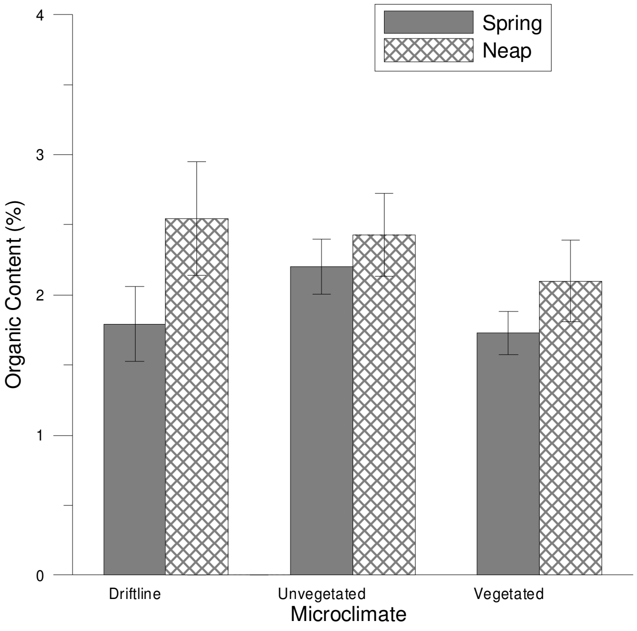


Figure 4.11 The mean sediment organic content of the microclimate for the marsh under spring and neap tides (bars = ± SE).

4.4.4 Sediment carbonate content

Sediment carbonate content was sampled in 2005 and 2006. The samples were taken to investigate the carbonate content of the site and to test whether it would change from year to year. The mean carbonate content (Figure 4.12) in 2005 (0.8 ± 0.13 %, n = 18) was not significantly different ($p > 0.05$, n = 39) from the carbonate content found in 2006 (1.2 ± 0.11 %, n = 21).

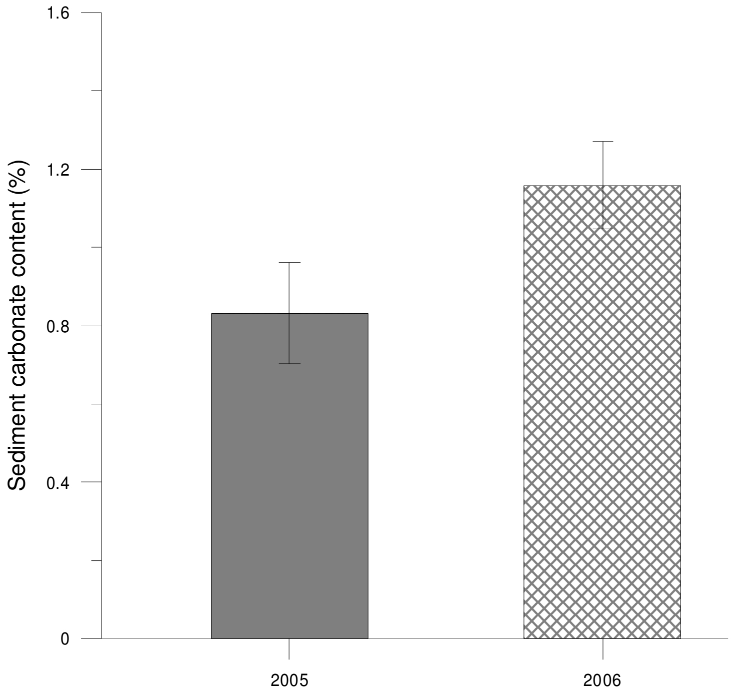


Figure 4.12 The mean sediment carbonate content of 2005 and 2006 (bars = ± SE).

4.4.5 Sediment particle size

4.4.5.1 Site A

Silt (61.7 ± 4 %) was significantly higher ($p < 0.05$, n = 42) in all sites except in the surface sediments of A6 and the watertable sediments of A7 ($p > 0.05$, n = 42). Surface sediments of A6 had a significantly lower ($p < 0.05$, n = 42) contribution of silt with no significant differences between the sand and clay fraction ($p > 0.05$, n = 42). Sand was significantly higher ($p < 0.05$, n = 42) in A7, with no significant differences between silt and clay ($p > 0.05$, n = 42)

4.4.5.2 Site B

Site B (Figure 4.13b) showed variation among the sites. Site B1 and B2 had significantly less silt at the surface than in the watertable sediment ($p < 0.05$, $n = 18$). Site B3 contained no silt at all and had a significantly higher contribution of sand ($p < 0.05$, $n = 18$). Clay content was only significantly higher at the surface of B2 compared to the other sites ($p < 0.05$, $n = 18$).

4.4.5.3 Site C

The clay size fraction was significantly lower in sites C2 and C3 and the watertable sediments of C1 ($p < 0.05$, $n = 18$). The surface sediment of C1 had no significant differences between size fractions ($p > 0.05$, $n = 18$). Silt made up a significantly larger ($p < 0.05$, $n = 18$) percentage of the sediment at C2, in the watertable sediments of C1 and the surface of C3. There was no significant difference between sand and silt in C3 ($p > 0.05$, $n =18$), although there was significantly more sand than in the other sites ($p < 0.05$, $n = 18$).

4.4.5.4 Site and Year Comparisons

The sand contribution to the sediment in 2005 in the surface sediments (Figure 4.14a) of Site A was significantly lower than that of Site B ($p < 0.05$, $n = 78$). Site B had significantly lower amounts of silt in its surface sediment than the watertable sediments and Site A and C ($p < 0.05$, $n = 78$). The watertable sediments of Site C had a significantly lower contribution of clay than all the other sites ($p < 0.05$, $n = 78$).

Site A in 2006 (Figure 4.14b) was homogenous in surface and watertable sediments ($p > 0.05$, $n = 18$). The surface sediments in Site B consisted of large amounts of silt, which were significantly higher than the silt fraction in the watertable sediments ($p < 0.05$, $n = 18$). The watertable sediments in Site B had significantly more sand than the surface ($p < 0.05$, $n = 18$). Site C had a significantly higher clay content in the surface sediments compared to the other sites in 2006 ($p < 0.05$, $n = 18$).

The influence of the flood in February 2006 can be assessed by comparing samples collected prior to (2005) and after the flood (2006) (Figure 4.14). Site A was not significantly influenced by the flood ($p > 0.05$, $n = 96$). Site B showed significant changes with the silt fraction increasing significantly ($p < 0.05$, $n = 96$) from 4.84 ± 1.9 % in 2005 to 74.9 ± 9.4 % in 2006. The clay fraction in Site C increased significantly in the surface sediments ($p < 0.05$, $n = 96$). This was the site most affected by the flood as it was the closest to the main river channel.

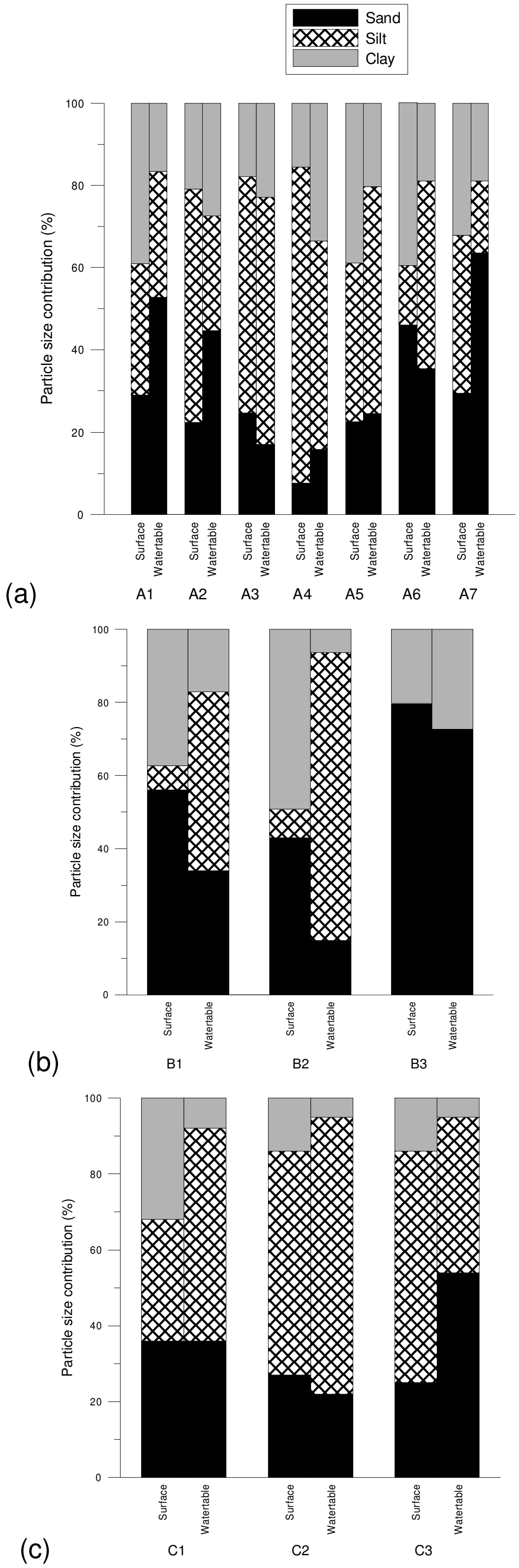
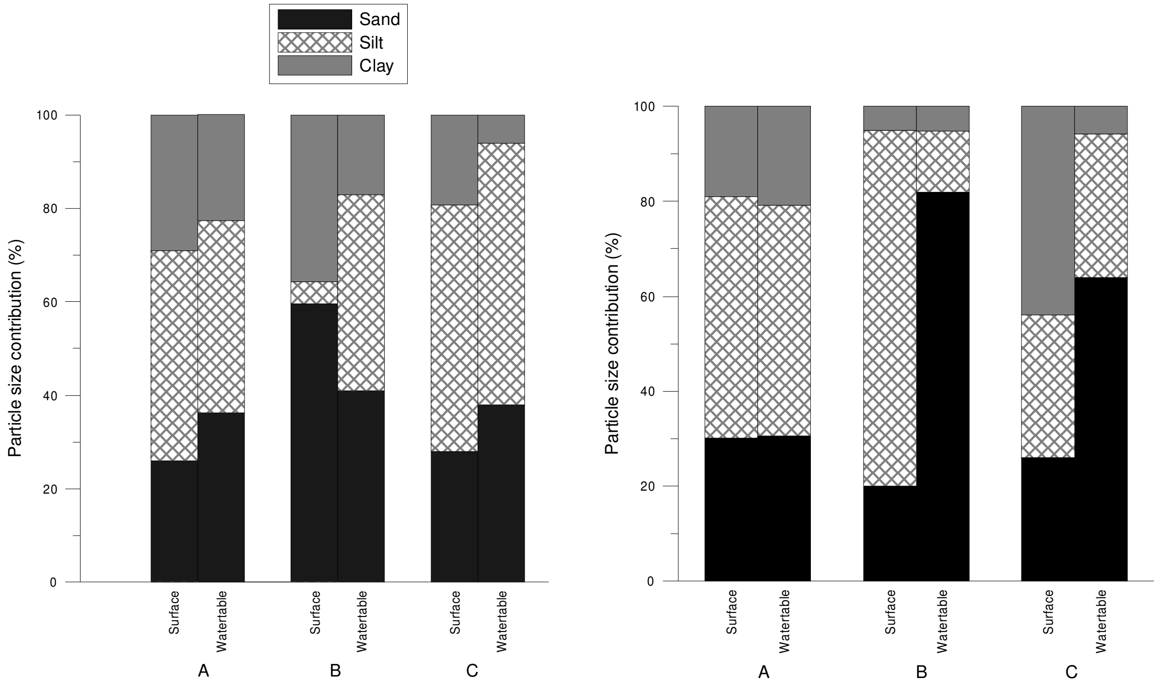


Figure 4.13 Sediment particle size for (a) Site A, (b) Site B and (c) Site C.



(a) (b)

Figure 4.14 Sediment particle size of the three sites within the marsh in (a) 2005 and (b) 2006.

4.4.5.5 Microhabitats

Sediment particle size of the microhabitats were similar ($p > 0.05$, n = 24) (Figure 4.15). Silt made up most of the sediment in the open unvegetated areas (48.4 ± 1.9 %) and under the vegetation (54.2 ± 2 %). There were no significant differences between the percentage sand and clay ($p > 0.05$, n = 24) composition. Driftlines were not measured because they are sites of organic matter accumulation and it was postulated that it had similar sediment characteristics to the other microhabitats.

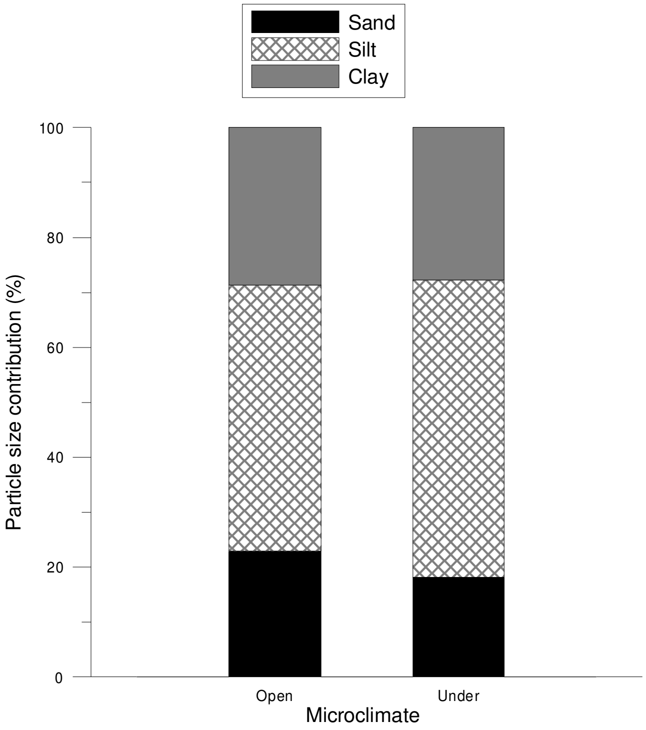


Figure 4.15 Mean sediment particle size of the microhabitats within Site A.

4.4.6 Compaction and Shear

Under spring tide conditions (Figure 4.16a) the compaction of the sediment in the open unvegetated areas was significantly higher than compaction under the vegetation ($p < 0.05$, n = 516). The mean compaction in the open areas was 0.79 ± 0.2 kg.m^{-2}, while it was 0.25 ± 0.06 kg.m^{-2} under the vegetation. The trend during the neap conditions (Figure 4.16a) was similar to that of spring conditions where the compaction was significantly higher in the open areas ($p < 0.05$, n = 516). The compaction increased although not significantly under the neap conditions to 1.08 ± 0.2 kg.m^{-2} ($p > 0.05$, n = 516) from 0.79 ± 0.23 kg.m^{-2} in the open areas and remained similar under the vegetation. Shear strength readings at spring tide (Figure 4.16b) showed that open areas (0.13 ± 0.03 kg.m^{-2}) had a significantly higher shear strength ($p < 0.05$, n = 516) than that found under the vegetation (0.1 ± 0.02 kg.m^{-2}). Shear strength showed no significant difference ($p > 0.05$, n = 516) from spring (0.38 ± 0.13 kg.m^{-2}) to (0.25 ± 0.06 kg.m^{-2}) neap tide.

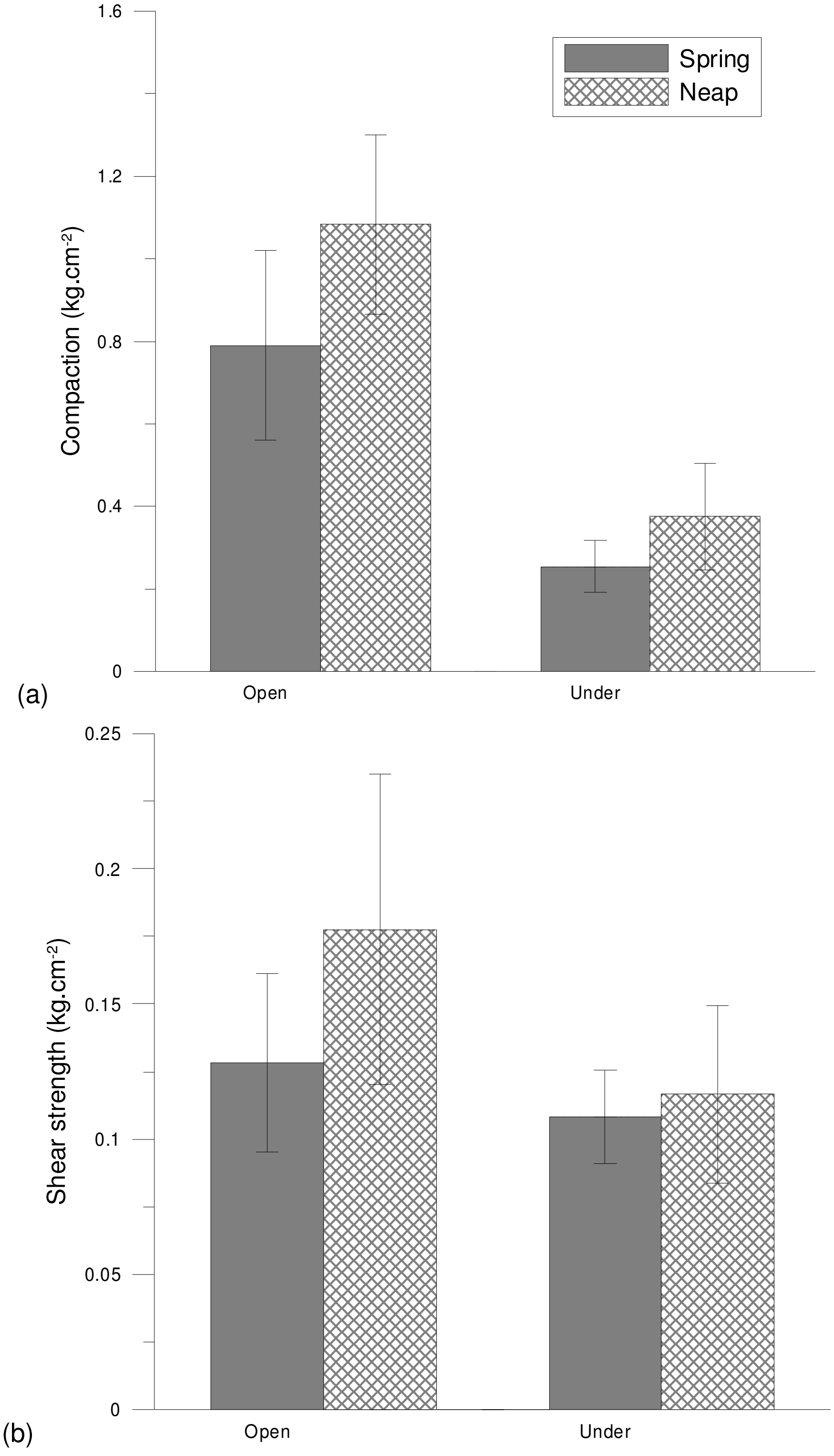


Figure 4.16 The (a) compaction and (b) shear of the sediment for the open areas and under the vegetation during both spring and neap tide (bars = ± SE).

4.4.7 Depth to groundwater

4.4.7.1 Site A

The depth to groundwater of Site A (Figure 4.17a) ranged from 22.50 ± 5.63 cm in A4 to 85.30 ± 2.03 cm in A2. The mean depth to groundwater was 41.50 ± 5.83 cm. There was no significant difference between spring high tide and low tide ($p > 0.05$; $n = 14$).

4.4.7.2 Site B

Site B (Figure 4.17b) had a mean depth to groundwater of 58.98 ± 4.23 cm, with no significant difference ($p > 0.05$; $n = 18$) between high tide and low tide or between sites. Depth to groundwater ranged from 44.21 ± 7.38 cm in B3 to 67.58 ± 2.33 cm in B1.

4.4.7.3 Site C

With a mean depth to groundwater of 144 ± 7.60 cm, Site C (Figure 4.17c) had a significantly ($p < 0.05$; $n = 26$) deeper groundwater than Sites A and B. There was no significant difference ($p > 0.05$, $n = 26$) between high and low tide. Depth to groundwater ranged from 120 ± 23.59 cm in Site C3 to 161 ± 8.62 cm in Site C1.

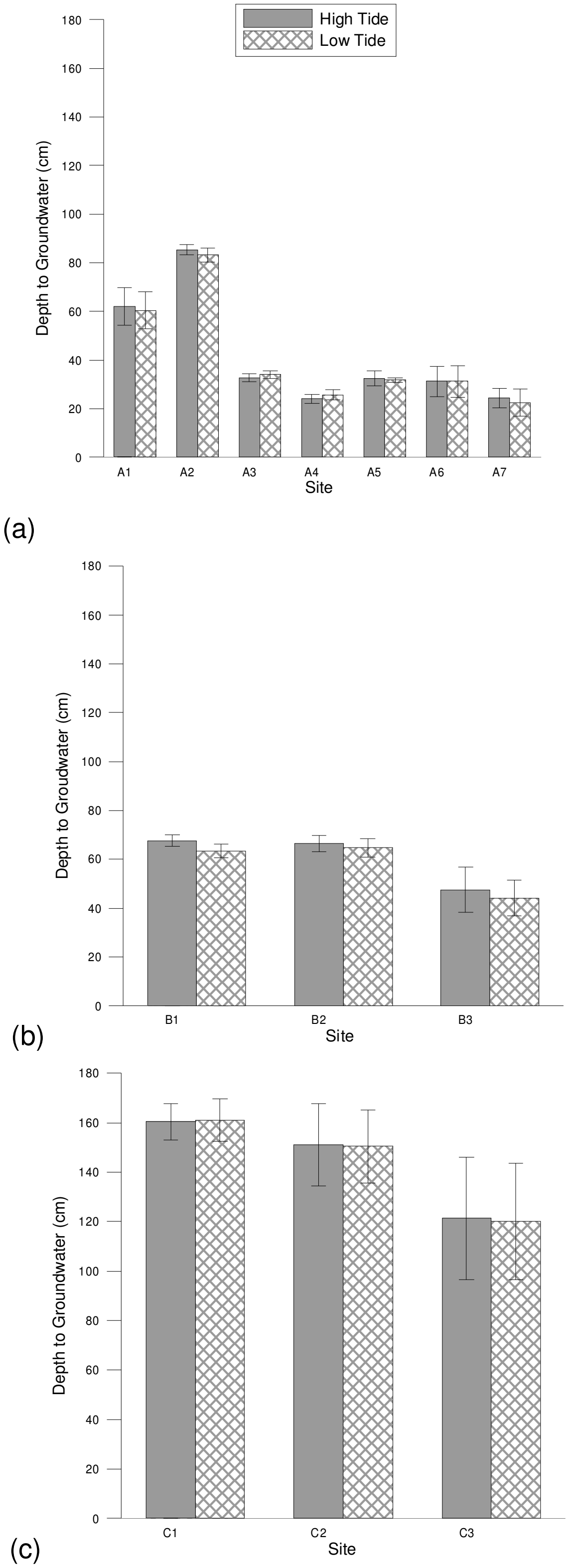


Figure 4.17 Depth to groundwater for (a) Site A, (b) Site B and (c) Site C during spring high and low tide (bars = ± SE).

4.5 Discussion

The measured sediment characteristics at the three different sites of the ORM allows one to make assumptions about the potential of the entire area for rehabilitation. Since the seed bank dynamics have been investigated (Chapter 5), the sediment characteristics will ultimately determine whether the deposited seeds and emerging seedlings will be able to survive to become adults leading to a recolonisation of the desertified marsh. Edaphic factors are important in limiting species to a particular zone in a saltmarsh (Cooper, 1982; Van Diggelen, 1991). Environmental gradients are mainly influenced by sediment moisture and/or flooding regime and/or sediment salinity and the relationships have been intensively studied (Zedler *et al.*, 1999; Álvarez-Rogel *et al.*, 2000, 2001; Bouzillé *et al.*, 2001; Denslow & Battaglia, 2002; Piernik, 2003).

In 2005, Site A was characterised by a high sediment conductivity (> 80 $mS.cm^{-1}$). This was higher than the tolerance range for an adult of the dominant supratidal species, *Sarcocornia pillansii* (Bornman, 2002). There was no significant difference between surface and water table sediments due to the influence of the hypersaline groundwater on the surface layers. High surface salinities are typical of sediments of supratidal and floodplain saltmarshes around South Africa, especially on the west coast (Bornman *et al.,* 2004a). The surface sediment of the supratidal and floodplain areas remains relatively moist throughout the year because of the water potential gradient (Bornman *et al*., 2004a). Organic content ranged between 1 – 3 % which was regarded as low to medium by the US EPA (1991), but similar to values recorded in the Olifants Estuary (Bornman, 2002). The sediment was dominated by silt in all locations with sand and clay making up equal contributions. Because edaphic factors are controlled by groundwater properties it was not surprising that the depth to groundwater was shallow (42 cm). Site A represents ~ 80 % of the marsh in terms of physical characteristics. Rehabilitation could potentially occur in areas with close proximity to the tidal channel that reduces the salinity of the groundwater and in turn creates favourable edaphic conditions with salinity within the tolerance range of the supratidal and floodplain species, *S pillansii.* Site A also contains the intertidal species *C. coronopifolia*, which will re-colonise areas according to tidal penetration and consistency.

Groundwater was shallow in Site B (~ 58 cm) and was characterised by a low electrical conductivity (61.24 $mS.cm^{-1}$) suitable for use by *S. pillansii*. Surface electrical conductivity were slightly higher than the water table sediment and moisture was lower due to evaporation and sediment texture. Sand has a lower hydraulic conductivity (Bornman, 2002), making the movement of water to the surface slower and less effective because of the air spaces. Organic matter was low at 1 %, further decreasing the ability of the sediment to retain moisture. This site was separated from the river by a berm, allowing easier interaction of the groundwater of Site B with the river. The presence of seedlings at the site indicated favourable conditions (Chapter 5). In 2005, 418 $seedlings.m^{2}$ were recorded. This was higher than Site A where seedlings reached a density of 271 $seedlings.m^{-2}$ (Figure 5.9). This site had favourable salinity for seed germination. However, this site is prone to inundation during river flooding or high rainfall. The berm situated at the southwestern end of the site prevents drainage and inundation for long periods of time and results in die-back of seedlings (Bornman *et al.*, 2005).

The lack of seedlings at Site C and presence of unhealthy appearing adults indicated that the conditions were not favourable for either growth form. This was confirmed by the survey of the sediment characteristics where high surface sediment conductivity existed (107 $mS.cm^{-1}$) (Figure 4.3). The watertable sediment was significantly lower at 40 $mS.cm^{-1}$. This site is close to the main river channel allowing the groundwater to be influenced and regulated by river and tidal flow. Sediment moisture and organic content however was low in the surface sediments with the depth to groundwater exceeding the reach of mature plants (> 150 cm) (Riehl & Ungar, 1982; Bornman *et al.*, 2004b). The deeper groundwater is probably due to the deposition of wind blown sediment in that area. The silt was deposited on the mature vegetation and the relatively slow growing plants cannot outgrow the rate of deposition and either becomes smothered or the groundwater becomes too deep to access. Although the salinity of the groundwater is within the tolerance range of the plants, the depth of the groundwater excludes its use as a source of water for the developing seedlings (Bornman, 2002). The deposition of sediment in this site also limits seedling establishment. The profile changes from a well defined layer of sand topped with silt at C3 to a mixed sediment profile where the sediment is loosely arranged in C1. These changes in the sediment profile have occurred due to the influence of the sediment transported by the wind.

In 2006 the marsh had undergone changes due to a large flood and rainfall event in February. Alexander Bay received more than its annual rainfall within a 5 month period (Figure 3.3). The comparison of the marsh between the two years gave an indication of the conditions experienced by the marsh during wet and dry periods. The flood reduced the surface and groundwater electrical conductivity by more than 50 % in Site A, which was the area most affected by hypersalinity in 2005. Moisture content decreased after the flood due to the flushing of salts from the surface sediments (Zedler, 1983; Jolly *et al.*, 1993; Bertness, 2002) allowing more moisture to evaporate. There was no significant import or loss of organic matter following the flood event suggesting that the majority of organic matter brought down by the flood was flushed out of the desertified marsh. The flood therefore succeeded in completely flushing the surface of the desertified marsh. The sediment particle composition did not change in Site A but it did in Site B and C. The berm situated at Site B stopped the movement of water creating a pool of standing water. As the river water entered the floodplain from the north-east, clay particles, being heavier than silt, were deposited first, thereby explaining the increased clay in Site C. As the water spread across the desertified marsh, the velocity of the flood flows decreased until a point where the suspended silt was deposited. Site B is located in an area where flood flows from the mouth and river areas probably met, creating ideal conditions for the settling out of suspended particles.

In general the EC and moisture content decreased between 2005 and 2006. The surface salinity in Site C remained high and it is hypothesised that this is as a result of the salinity of the wind blown sediment that inundates the site.

The presence of seedlings in the marsh indicated that conditions, especially in Site A had become favourable in 2006 for germination (see Chapter 5). Upon further investigation at Site A it was found that the seedlings germinated in three microhabitats, namely: driftlines, open unvegetated areas and under existing vegetation (Chapter 5). The increased rainfall and flooding event increased the number and size of the microhabitats. Before the rain only those microhabitats in close proximity to the tidal channel would have been suitable for seed germination and establishment. Favourable conditions occurred in all the microhabitats during spring tide due to the proximity of the sites to the tidal reach. During neap tide the EC increased significantly in the open areas to within the upper tolerance range of the adult plants. The other two microhabitats showed less fluctuation. The driftlines in A4 (107.87 ± 9 $mS.cm^{-1}$) and A6 (112.67 ± 6 $mS.cm^{-1}$) had consistently high EC over the spring and neap tides and no seedlings were found in these areas (Chapter 5). The microhabitat created by the adult plants had a stable EC with no significant difference between spring and neap tides. The adult plants shaded the sediment surface, reducing the effect of evaporation and retained the moisture in the surface sediments for longer (Callaway, 1994).

Conclusion

Rainfall and flooding significantly reduced the salinity in the desertified marsh. The microhabitats formed by the driftlines and vegetation created a stable environment and sediment conditions did not fluctuate as much as the open unvegetated areas. Tidal inflow created by the breakthrough in the causeway regulated the sediment salinity but this was restricted to areas in close proximity to the tidal reach. Isolation from the main river channel resulted in hypersaline (> 80 $mS.cm^{-1}$) conditions and the driving physical factor for seedling germination and establishment is primarily salinity (Parrondo *et al.*, 1978; Burchill & Kenkel, 1990; Neill, 1993; Krüger & Peinemann, 1996; Bornman, 2002). The edaphic variables in the marsh became favourable after the large flood and rainfall event. However, only the microhabitats created by the adult vegetation and driftlines allowed for any long term survival of seedlings due to their physico-chemical stability. The long term (1 – 5 year) survival of the seedlings will have to be investigated. The sediment characteristics in the unvegetated areas fluctuate considerably and become hypersaline when the freshwater and tidal water recedes. The vegetated areas in close proximity to the tidal influence, or river channel, created the highest potential for rehabilitation in the marsh. They are characterised by low salinity (< 80 $mS.cm^{-1}$), low compaction (0.3 ± 0.05 $kg.cm^{-2}$) and relatively high moisture content (10 ± 0.20 %).

To effectively rehabilitate the ORM it is evident that frequent flooding / backflooding (man made) is required to flush the salts from the surface soils. The initial frequency of the flooding events should be in two concurrent years. The first to flush the sediment of salts and the second to promote seed germination and establishment. A third year of high flow could ensure the survival of the seedlings. As the desertified marsh is progressively colonised the frequency of flooding events can be decreased as the microhabitat provided by the adults will ensure the survival of the seedlings. Interference with the hydrodynamics of the mouth and river flow only need to continue until the entire marsh has been recolonised to the natural cover abundance of *S. pillansii* (similar to what Bornman (2002) recorded at the Olifants Estuary). Thereafter large floods as occurred in 2006 should be sufficient in maintaining the saltmarsh community.

Chapter 5: Seed Production, Deposition and Seedling Germination

5.1 Abstract

The Orange River Estuary is characterized by a large floodplain area (298 ha) near the mouth that used to be colonized by saltmarsh vegetation. Due to various anthropogenic impacts (Chapter 3) 90 % of the saltmarsh was lost and the area became desertified. The objective of this study was to establish whether there were adequate seed available to recolonise the site once conditions became favourable. Revegetation of the saltmarsh will only take place if the seed produced germinate, establish and survive to adulthood. The intertidal species *Cotula coronopifolia* L. produced more seed in 2006, 424 800 m^{-2} compared to 325 792 m^{-2} in 2005. Seed production for *Sarcocornia pillansii* A.J. Scott was 292 984 m^{-2} in 2005 and 134 277 m^{-2} in 2006. However adult stem density was higher in 2006 after the area experienced above average rainfall. *S. pillansii* had a seed output of approximately 40 billion over an area of 29 ha while *C. coronopifolia* produced approximately 8 billion seeds in 2 ha.

Water was important in distributing the seed which were deposited on driftlines. Driftlines (*C. coronopifolia* = 872 seedlings m^{-2}; *S. pillansii* = 1296 seedlings m^{-2}) and the microhabitat associated with adult plants (*C. coronopifolia* = 803 seedlings m^{-2}*; S. pillansii* = 721 seedlings m^{-2}) created favourable conditions for seedling growth, however open unvegetated (*C. coronopifolia* = 56 seedlings m^{-2}*; S. pillansii* = 49 seedlings m^{-2}) areas had significantly lower seedling density.

Laboratory studies indicated that seeds of both species germinated best in freshwater (0 psu). No *C. coronopifolia* seeds germinated at 35 psu and germination was less than 15 % for the 15 psu treatment compared to greater than 60 % for the 0 psu treatment. The germination of *S. pillansii* seeds was 40 % at 0 psu compared to 5% at 35 psu. After storage under hypersaline conditions (35 psu), *C. coronopifolia* showed 100 % seed germination when returned to freshwater, whereas storage at 70 psu decreased the viability of *S. pillansii* seeds. This indicates the importance of freshwater pulses (e.g. rain and floods) for the germination and establishment of the two dominant salt marsh species.

Further laboratory studies on *S. pillansii* indicated that the seeds were viable after long-term storage under warm, cold, light, dark, wet and dry treatments. They were able to germinate once exposed to favourable conditions although germination was higher for warm compared to cold storage treatments. It is important that freshwater inundates the marsh periodically as this leads to seed entering new areas for establishment, allows previous seedling cohorts to continue growing and enables adults to increase cover and set seed. The plants are producing adequate seed that will allow for regrowth and rehabilitation if conditions are favourable. The future health and survival of the adult *S. pillansii* plants is important for the long-term rehabilitation of the marsh as they are the seed source and also provide microhabitats for seedling survival. *S. pillansii* has a persistent seedbank as not all seeds germinate at one time. This ensures the survival of this perennial species as only a few successful seedlings are necessary.

5.2 Introduction

The natural rehabilitation of the ORM will rely on the ability of the plant species in the marsh to complete their life cycles. The adult plants of the dominant species set seed annually to ensure the survival of the species. Seeds facilitate rapid bare area re-establishment following the loss of adult populations through disturbances like flooding, drought or burial (Fenner, 1985). Wetland plants generally rely on water for the dispersal of seed (hydrochory) (Fenner & Thompson, 2005). Although seeds germinate rapidly the seedlings require two to three weeks additional moisture to establish (Fenner 1985). Similar to the adult phase the seedlings then require a reliable source of moisture to grow and mature. The adults are generally tolerant of water and salinity stress and can persist through periods of drought and hypersalinity (Fenner, 1985) Hypersalinity limits the germination of seeds and limits the establishment of seedlings, ultimately influencing the distribution of the adult vegetation (Fenner & Thompson, 2005).

As the seeds are hydrochorous, they are deposited along water lines on the shore referred to as driftlines. The driftlines include a collection of organic matter which creates conditions favourable for germination. The driftlines are important microhabitats that were compared with two other distinct microhabitats at the ORM, i.e. open, unvegetated areas and under the adult vegetation.

The aim of this chapter was to investigate the different phases in the life cycle of the two dominant saltmarsh species, *Cotula coronopifolia* L. and *Sarcocornia pillansii* A.J.Scott. This included the number of seeds produced and deposited, where the seed was deposited and where the seedlings established. The cover of the adults was also mapped to determine the potential of the seeds to colonize new habitat. It was possible to assess the importance of freshwater and tidal flooding in the salt marsh because of the two extreme environmental conditions prevalent in 2005 (drought) and 2006 (flood and high rainfall).

5.3 Materials and Method

The life cycle measurements were taken in the field but seed viability, the effect of hypersalinity and transient/persistent seed bank trials were performed under controlled conditions in the laboratory.

5.3.1 Study Site

Three main sites were selected in the ORM wetland that represented the main marsh (A), seedling site (B) and dust site (C) (Figure 5.1.). Site A was largely devoid of vegetation and represented the largest portion of the ORM wetland. The seedling site (B) had a high cover abundance of adult plants as well as numerous seedlings and represented favourable condition for saltmarsh growth. The dust site (C) represented a degraded area different to Site A in that dieback occurred not as a result of high salinity but because of sediment inundation. The study sites described above are the same used in the edaphic study (see Chapter 4) (Figure 4.1).

In 2005, no distinct differences in adult plants were visually evident and sampling for seed production was conducted in only one area believed to represent the entire marsh. Following the good rains in 2006, clear colour differences (green, purple and red) were evident in individual *S. pillansii* plants and as a result these individuals were separately analysed to determine if there were any differences in seed production. In 2005 the production was measured on adult vegetation in a specific area close to the intertidal channel. The production was measured on individuals with a high flowering percentage. In 2006 the sampling included a range of different adult plants in order to capture the effect of perceived health and cover abundance on the production of seed. The individual plants measured in 2006 were located at various stations within the marsh as indicated in Figure 4.1.

The distribution and cover of the dominant vegetation was mapped in the field (Figure 5.1). Spatial data were captured using a handheld GPS with ArcPad version 7.0 software. ArcGIS version 9 was used to create the maps.

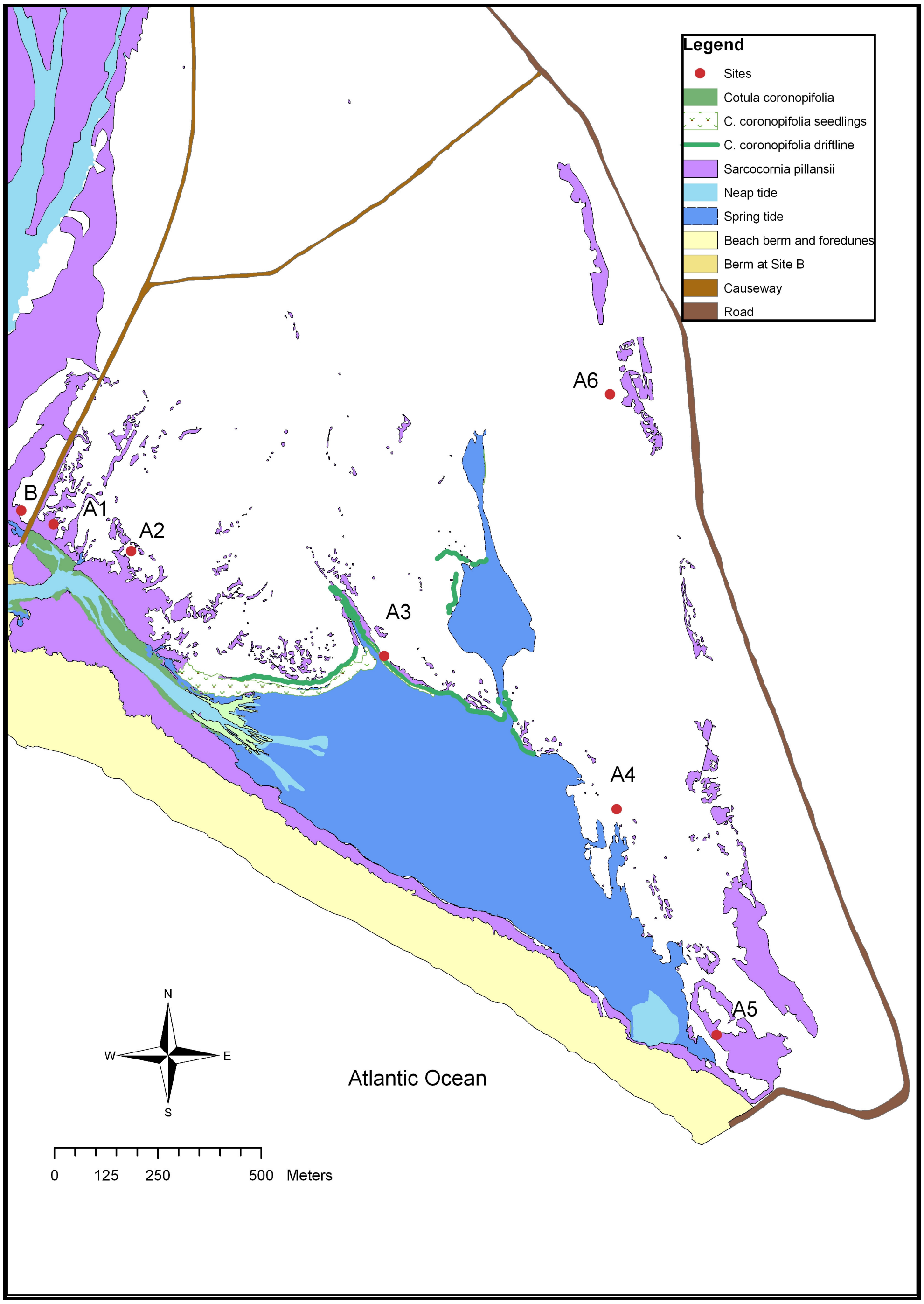


Figure 5.1 Locality map of Site A where production was determined and microhabitats were investigated.

5.3.2 Seed production

The two dominant saltmarsh species (*Cotula coronopifolia* and *Sarcocornia pillansii*) at the ORM were selected for this study. The number of seed that could potentially be produced by the plant as well as the amount of seed actually produced by the plants at the time of the field survey was calculated by counting the number of seed produced per stem and extrapolating it to the amount of seed produced per unit area. Seed production and percentage cover of the adult plants was measured in the field using a 45 × 45 cm quadrat. Vegetation percentage cover was determined by randomly placing the quadrat in vegetation surrounding the site where production measurements were taken. Within this quadrat the percentage of flowering plants was also determined.

For the intertidal species (*Cotula coronopifolia)*, 16 flower heads were collected randomly within the restored intertidal salt marsh area (Figure 5.1). The number of seeds on each head was counted using a dissection microscope in order to determine the potential seed production. To accurately determine the number of flower heads per square meter, 100 randomly placed (10 × 10 cm) quadrats were sampled. The data were extrapolated to express the number of flowers per square meter. The total area covered by flowering *C. coronopifolia* was mapped in the field to determine total potential seed production.

The seed production of the supratidal and floodplain species, *Sarcocornia pillansii,* was calculated by determining the potential and actual number of seeds at the time of sampling. Samples were collected randomly on the southern side of the causeway near the intertidal channel in 2005 (Figure 5.1, Site A1 and A2). The potential seed production was calculated by counting the number of seeds produced per segment (Plate 13). The number of seed producing segments present on a stem was also counted. This was related to the number of stems found within a quadrat (45 × 45 cm). This value was extrapolated to get a measure of potential seed production per square meter. The assumption for the potential production was that each shoot found on the stem would produce the mean number of seeds.

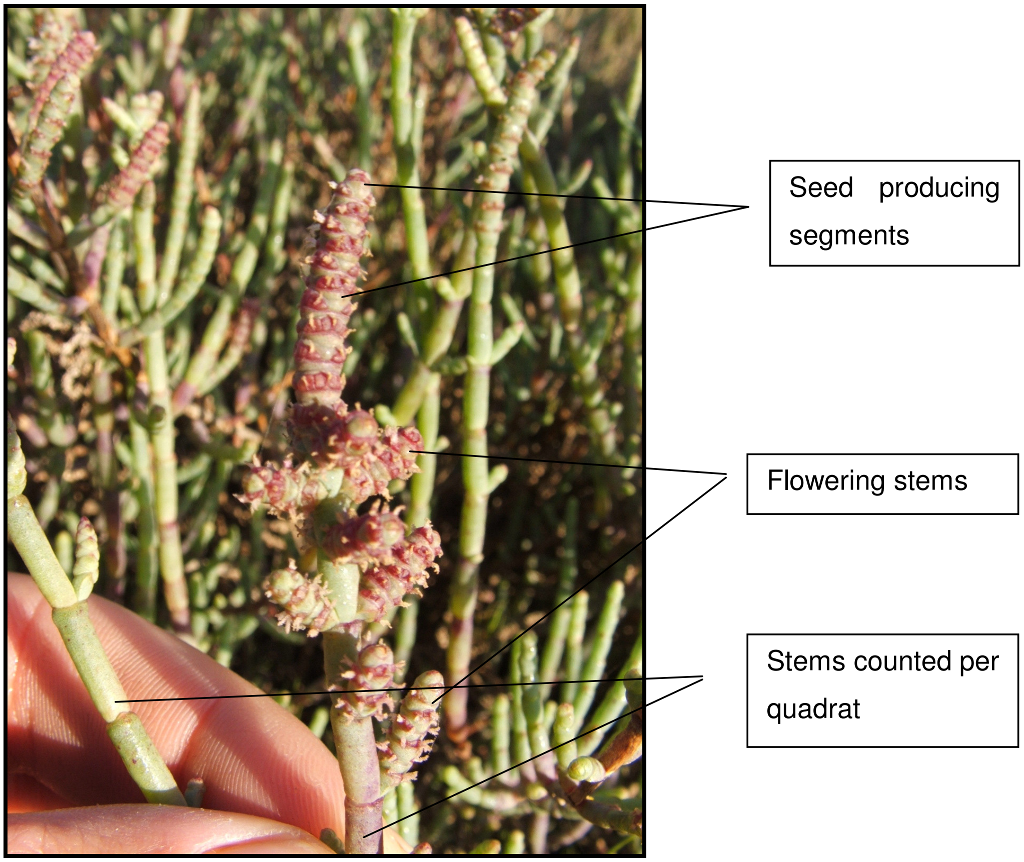


Plate 13 Morphology of *Sarcocornia pillansii* used to determine seed production for the species.

In terms of actual seed production, the health of the plant was first determined as percentage cover of the individual plants and as the percentage of flowering stems. The colour of the vegetation was thought to indicate health of the plant, where purple stems were thought to indicate stress and green plants were healthy. In 2006 replicates of the different colours observed in the field were sampled in order to determine the differences in production. The observed value was replaced by the potential flowering percentage to gain a value in terms of seed produced by the plant. The measurements took place within monospecific stands of *S. pillansii*, where flowering was taking place and seeds occurred on the parent plant. The number of replicates was higher in 2006 due to increased sampling effort. More plants were analysed to test whether perceived health had an influence on seed production.

5.3.3 Seed deposition along driftlines

Driftlines were identified by an accumulation of seeds or seedlings along clearly identifiable water marks (Plate 14). Emergence is used more commonly than direct seed counts to estimate the size of the seed bank because it is less time consuming (Benoit *et al.*, 1989; Brock *et al.*, 1994; ter Heert *et al.*, 1999). Direct counts were used in this study because germination requirements may not be satisfied and because this allowed for an accurate assessment of the size of the seed bank (Brock *et al.*, 1994).

Plate 14 Characteristic driftlines at the Orange River Mouth. Seeds are concentrated in these driftlines that create a favourable microhabitat for the germination and establishment of seedlings. These driftlines were formed during spring high tides.

5.3.3.1 Seed

During the first site visit (August 2005) most seeds were found deposited along driftlines. The seeds were transported by water including spring-high tides (Plate 14) and flooding (Plate 15). In 2006, the width of the driftlines were measured to estimate the number of seed occurring in the different driftlines. A value per square meter was calculated to attempt to quantify total seed deposition in driftlines (Nilsson & Grelsson, 1990; Vogt *et al.*, 2004).

Plate 15 Driftlines formed as a result of freshwater flooding in 2006.

In 2005 eight sampling sites were chosen within driftlines containing seed. Two sites were located in the intertidal zone at A1 and A2. Of the remaining supratidal sites, three were located in Site A (3, 5 and 6) and three in Site B (1, 2, 3) (Figure 5.1.). As far as possible, sampling was repeated at these sites in 2006. However, Site B was inundated by water due to flooding and the driftlines in Site A were located elsewhere in 2006. Site A4 and A7 had no visible tidal driftlines in 2006 because of extensive river flooding.

Quadrats (10 × 10 cm) were placed randomly along the driftlines to determine the variability in deposition of the seeds. Only the top layer of the sediment (1 cm) was sampled for seed because the compaction of the sediment prevented seed infiltrating deeper. Sediment collected on a previous field trip (August 2004) was analysed and no seed was found below the surface crust. A paint brush with a width of 50 mm was used to brush the surface layer into a plastic bag that was sealed and transported back to the NMMU laboratory for analysis. Five replicate samples were collected at each site. The seeds were stored under dry and dark conditions prior to analyses (Gross, 1990; Rhedondo *et al.*, 2004).

In the laboratory the sediment containing the seed was placed in an evaporation basin (125 ml). A Sodium hexametaphosphate $(Na_3PO)_6$ solution of 50 $g.l^{-1}$ was added to the sediment. The dispersal and deflocculating properties allowed the seeds to separate from the sediment and other organic particles. The evaporation basin was placed on a mixing tray and stirred for 10 min. The flotsam, which included most of the floating seeds, was poured off onto paper towelling. The towelling absorbed the water leaving the seeds and other organic matter behind. Seeds of the two dominant species were counted using a KYOWA dissecting microscope at a 40 × magnification. The few seeds that remained behind in the basin were also counted and included in the total count.

5.3.3.2 Seedling counts

Sarcocornia pillansii seedling counts were restricted to the areas where seedlings occurred and included four of the driftlines used for the quantification of seed density. The three driftlines from Site B were used and one from Site A (Figure 5.1). Twenty replicate samples were measured per driftline using a 10 × 10 cm quadrat. The height of the seedlings were measured from the base of the stem to the tip of the main stem. *Cotula coronopifolia* seedlings were counted and measured along a driftline in the intertidal zone. Due to the growth form of the species the radius of the cover of each seedling was measured rather than the height.

5.3.3.3 Flow diagram

A flow diagram was created to represent the various components in the life cycle of the two saltmarsh species.

i) Production

The number of seeds per flower head of *C. coronopifolia* were multiplied by the number of flower heads per quadrat and extrapolated to a value per square meter.

The *Sarcocornia pillansii* production value was obtained by multiplying the number of seed producing segments by the number of shoots per stem. That number was multiplied by the number of stems per quadrat and extrapolated to a value per square meter.

ii) Deposition

Seeds were collected from various driftlines using five replicate quadrats (10 × 10 cm). The seeds were counted in the laboratory. The value from the analysis was multiplied to get a value in square meters. The mean was used for both the intertidal and supratidal species.

iii) Seedlings

The quadrat used for this sampling was identical to the one used for the seed deposition collection. The mean was used as an indication of the overall number of seedlings present within the marsh.

5.3.4 Seed viability

5.3.4.1 Salinity and water treatments

Seeds of the two species were collected in the field and taken back to the laboratory. The seed was stored in plastic bags under ambient temperatures in dark and dry conditions (Gross, 1990; Rhedondo *et al.*, 2004). A factorial experiment was designed. Each treatment consisted of three different water regimes i.e. watered, where the substrate (paper toweling) was damp; waterlogged, where the substrate was soggy with little excess water; and inundated where the seeds were floating in the water. Within these three water regimes, four salinity treatments were used, namely: 0, 15, 35 and 70 psu. Distilled water was used as the 0 psu treatment. The 15 psu solution was diluted with distilled water from seawater at 35 psu. NaCl (70 g) was added to distilled water (1 litre) to obtain the 70 psu treatment. Each different treatment had three replicates of 50 seeds each. The petri dishes containing the seed were placed in a growth cabinet (Conviron E7) with the environmental parameters set to reflect those similar to field conditions found in winter when the seeds normally germinate. The temperature was set at 25 ºC with a day/night cycle of 14/10 hours. The light intensity remained constants at 110 µmol $m^2.s^{-1}$. The seeds were watered every alternate day and germination was recorded. Germination was considered to have taken place once the radical had emerged form the seed coat. Once counted the seedling was removed from the petri dish (Redondo *et al.*, 2004; Naidoo & Kift, 2006). The experiment was conducted over 20 days (no additional seeds germinated after a period of 20 days) after which total viability was calculated for both species.

5.3.4.2 Effect of hypersalinity

To test whether seed could germinate after exposure to hypersaline conditions an experiment was set up with three treatments for each species and three replicates each. Fifty seeds were placed in each replicate dish and treated with saline water. *Cotula coronopifolia* seeds were exposed to water with a salinity of 35 psu and the *Sarcocornia pillansii* seeds to 70 psu. These salinity concentrations were selected because they are at the upper end of the species' tolerance ranges (Bornman *et al.*, 2005). After one, three and six weeks the seeds were removed from the saline treatment and watered with their optimum salinity treatment as determined by the previous experiment (5.3.4.1). The experiment was carried out in a growth cabinet with a temperature of 25 ºC and a day/night cycle of 14/10 h. The light intensity recorded in the growth cabinet was 110.µmol.$m^2.s^{-1}$. The seeds were watered every alternate day and germination was recorded. Germination was considered to have taken place once the radical had emerged from the seed coat. Once counted the seedling was removed from the petri dish because of the influence it may have had on the germination of the other seeds. The experiment was run for 40 days (Kahn & Gul, 2006). Germination in seeds that have been exposed to hypersalinity is delayed due to osmotic imbalance in the seed. Overall viability was calculated as the percentage of seeds that germinated compared to the total number of seeds at the start of the experiment.

5.3.4.3 Storage conditions

To test whether the seed bank of *S. pillansii* was persistent or transient, the viability of the seed after storage under a range of conditions experienced at the ORM was investigated. A factorial experiment was designed that included light, dark, warm, cold, wet and dry as treatments. Three replicate samples of 50 seeds each per treatment were stored under the combination of conditions for three months. The wet sample seed was stored in a plastic bag and a few drops of water were added to the bag and then sealed. The moisture of the bag needed to be such that the seeds did not rot. The dry treatment consisted of seeds stored in an unsealed plastic bag. The light-warm-wet and dry treatment was stored in a growth chamber. The light intensity in the growth chamber was set at 70 µmol m^{-2} s^{-1} and the temperature at 40 ºC. The light-cold-wet and dry samples were placed in a cold room under a desktop lamp where temperatures ranged between 0 and 5 ºC and the light intensity were set at 70 µmol m^{-2} s^{-1}. The dark-warm-wet and dry treatment was placed in a glass container wrapped in aluminium foil. The foil had holes on one side to allow for air circulation within the container. The growth chamber in which the sample was placed had a temperature of 40 ºC. The dark-cold-wet and dry sample was stored in a cold room at temperatures below 5 ºC. These were also placed in a glass container and wrapped in aluminium foil to prevent light reaching the seeds. The seed was stored for three months. Three replicates of 50 seed were used for each storage condition and placed in petri dishes. The seed where watered using the optimum conditions as determined in the viability trials. The experiment ran for 20 days and seeds were considered to have germinated once the radical had emerged. The seedling was then removed from the experiment.

5.4 Results

5.4.1 Seed production

The intertidal species, *Cotula coronopifolia,* produced a large number of seeds (325 792 m^{-2}) in 2005. The average number of seeds per floral head was 231 with a mean of 14 ± 0.99 floral heads counted per m^2 (Table 5.1). In 2006 the number of flower heads increased per square meter to 18 ± 0.5, with 5 more seeds per head to give a value of 424 800 m^{-2}.

Table 5.1 Total seed production (m^{-2}) for the intertidal species, *Cotula coronopifolia.*

Year	Number of flower heads per quadrat (n=100)	Number of seeds per head (n=16)	Total (m^{-2})
2005	14 ± 0.99	231± 11	325 792
2006	18 ± 0.53	236 ± 4	424 800

A mean of 52 seeds were produced per seed producing segment in 2005 for the supratidal species, *Sarcocornia pillansii* (Table 5.2). Only 16 seed producing segments per stem were flowering in the area of study where vegetation cover abundance was 30 %. *S. pillansii* had a density of 72 stems per quadrat and production was calculated at 292 984 seeds m^{-2}. To determine the potential seed production of *S. pillansii,* it was assumed that all the seed producing segments would flower, giving a value of 862 023 seed m^{-2} (Table 5.2). Potential seed production as well as actual production values were lower in 2006.

Table 5.2 The potential and actual seed production of *Sarcocornia pillansii* in a supratidal habitat.

Potential Production				
Year	Number of seed	Seed producing segments per stem	Stems per quadrat	Total (m^{-2})
2005	52.01 ± 0.93 (n = 364)	46 ± 1 (n = 100)	72 ± 2 (n = 30)	862 023
2006	39.26 ± 0.51 (n = 815)	21 ± 0.32 (n = 815)	313 ± 37 (n = 10)	258 056
Actual Production				
Year	Number of seed	Seed producing segments per stem	Stems per quadrat	Total (m^{-2})
2005	52.01± 0.93	16 ± 0.42 (n = 100)	72 ± 2 (n = 30)	292 984 (34 %)
2006	39.26 ± 0.51	11 ± 0.27 (n = 815)	313 ± 37 (n = 10)	134 277 (52 %)

In 2006 the number of seeds per plant were lower than in 2005. The seed producing segments were also lower. But the density of stems was higher than in 2005.

Sarcocornia pillansii covered a greater area of the desertified marsh (298 538 m^2) compared to *Cotula coronopifolia* (20 693 m^2) and therefore produced more seed, approximately 40 billion compared to nearly 9 billion (Table 5.3).

Table 5.3 The area covered by the dominant species, the number of seeds produced (m^{-2}) and the total number of seeds produced in the ORM saltmarsh in 2006.

Species	Area (m^2)	Seed (m^{-2})	Total seed
Cotula coronopifolia	20 693	424 800	8 790 386 400
Sarcocornia pillansii	298 538	134 277	40 086 787 026

The number of flowering segments did not differ significantly from green to purple plants ($p > 0.05$, $n = 815$) (Figure 5.2). The number of flowering branches in the purple vegetation was significantly lower than for vegetation with green foliage and green foliage with purple flowering segments ($p < 0.05$, $n = 815$). There were also no significant differences between the total number of branches in the range of colouration ($p > 0.05$, $n = 815$). Although stress may cause the parent vegetation to seed, the results showed that there was no significant difference in the percentage of plants flowering for the different colours ($p > 0.05$, $n = 815$) (Figure 5.3).

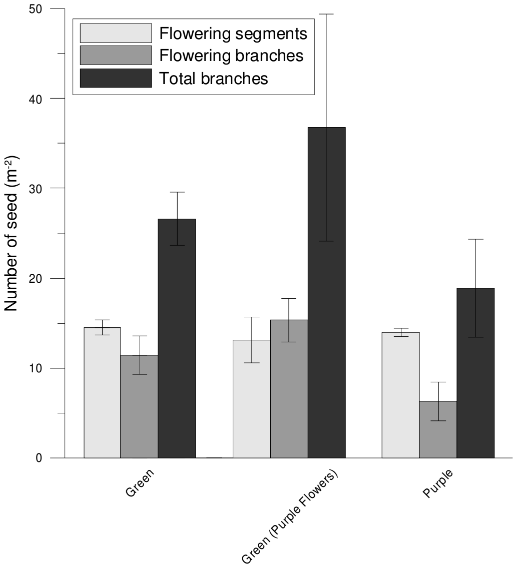


Figure 5.2 Mean number of seeds produced by the three different colour *Sarcocornia pillansii* plants for the three different stem types.

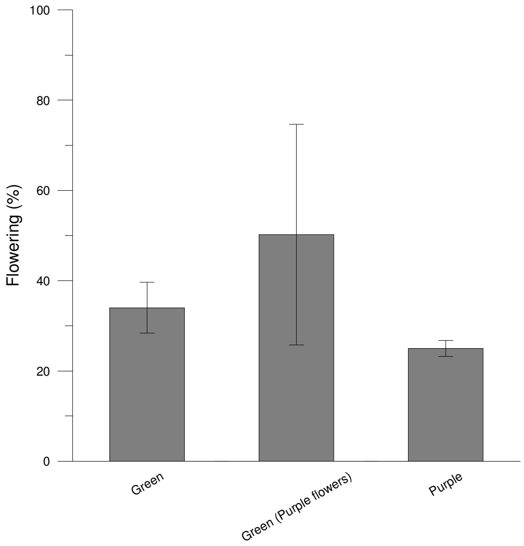


Figure 5.3 Percentage flowering for plants with different colours.

Other indicators of health are percentage cover. Site A4 and A5 had a significantly lower cover than the rest of the sites ($p < 0.05$, $n = 120$). Site A1, A2 and A3 showed the highest cover (> 80 %). Flowering percentage for the locations within Site A showed no significant differences ($p > 0.05$, $n = 120$) other than Site A4 which had the lowest (0.35 ± 0.26 %) flowering percentage ($p < 0.05$, $n = 120$) (Figure 5.4).

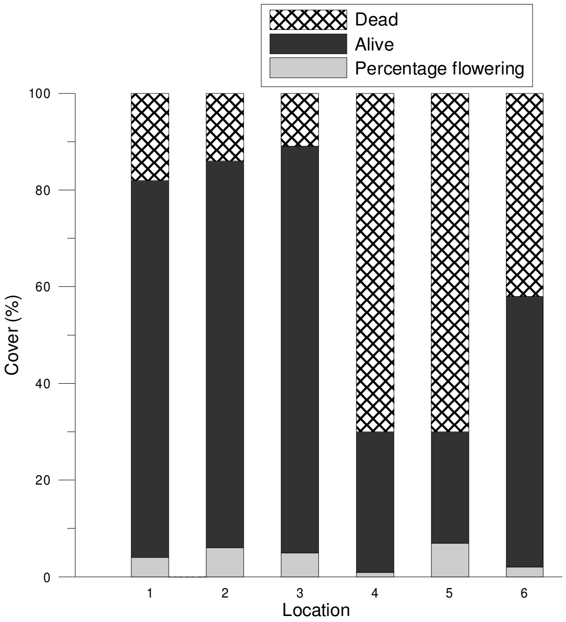


Figure 5.4 An indication of *Sarcocornia pillansii* health in the various locations within Site A.

5.4.2 Seed deposition along driftlines

Sites A4, A6, A7 and B3 did not contain any *C. coronopifolia* seeds in 2005 (Figure 5.5a). The intertidal sites (A1 and A2) had significantly ($p < 0.05$; $n = 54$) higher seed counts (99 400 ± 15 270 seeds m^{-2}; 181 760 ± 7847 seeds m^{-2} respectively) than the other sites where *C. coronopifolia* seeds were found. The intertidal sites at A1 and A2 had significantly higher numbers than any of the other sites sampled for both supratidal and intertidal species ($p < 0.05$; $n = 10$). The driftlines contained significantly less seeds ($p < 0.05$, $n = 89$) in 2006 (Figure 5.5a), but seeds occurred in more driftlines at Site A. Driftlines at Site B were not found in 2006 as Site B was inundated by water. There were no significant differences ($p > 0.05$, $n = 35$) between sites A1 (41 280 ± 9 024 m^{-2}), A2 (47 740 ± 6 615 m^{-2}) and A3 (30 300 ± 5 789 m^{-2}). In driftlines A4 (10 060 ± 3 493 m^{-2}) and A5 (5 260 ± 775 m^{-2}), seed numbers were significantly lower ($p < 0.05$, $n = 35$) than the three previously mentioned sites. Site A6 had no driftline and A7 (580 ± 179 m^{-2}) was significantly lower than all other sites ($p < 0.05$, $n = 35$).

The two driftlines (A3 and A4) analysed for seed deposition showed that seed is abundant with A3 containing 4 500 000 million seeds (Table 5.4).

Table 5.4 indicates the number of seed found along the length of each driftline (2006). The driftlines occurred at different sites and had various widths.

Site location	Species	Length of Driftline (m)	Width of Driftline (cm)	Area of Driftline (m^2)	Seed per Driftline (m^{-2})
A3	*C. coronopifolia*	1356	11 ± 1.4 (n =15)	149.16	4 519 548
A4	*C. coronopifolia*	85	11 ± 1.4 (n = 15)	9.35	94 061

Only sites A3, A6 and A7 contained *S. pillansii* seeds in the driftlines (Figure 5.5b). Site A3 had the lowest density with only 220 ± 156 seeds m^{-2}, while A6 had a significantly greater density of seeds (24 280 ± 9 268 seeds m^{-2}) than the other supratidal sites in 2005 ($p < 0.05$, $n = 30$).

There was a similar pattern for *S. pillansii* in 2006. The seed count was lower in A6 and A7, although not significantly so ($p > 0.05$, $n = 35$), but seeds were found in more driftlines (Figure 5.5b). Site A6 did not differ significantly to any of the other sites ($p > 0.05$, $n = 89$) while Site A3 contained no seeds in 2006. Site A2, A4 and A5 gained a significant amount of seed in 2006 ($p < 0.05$, $n = 89$).

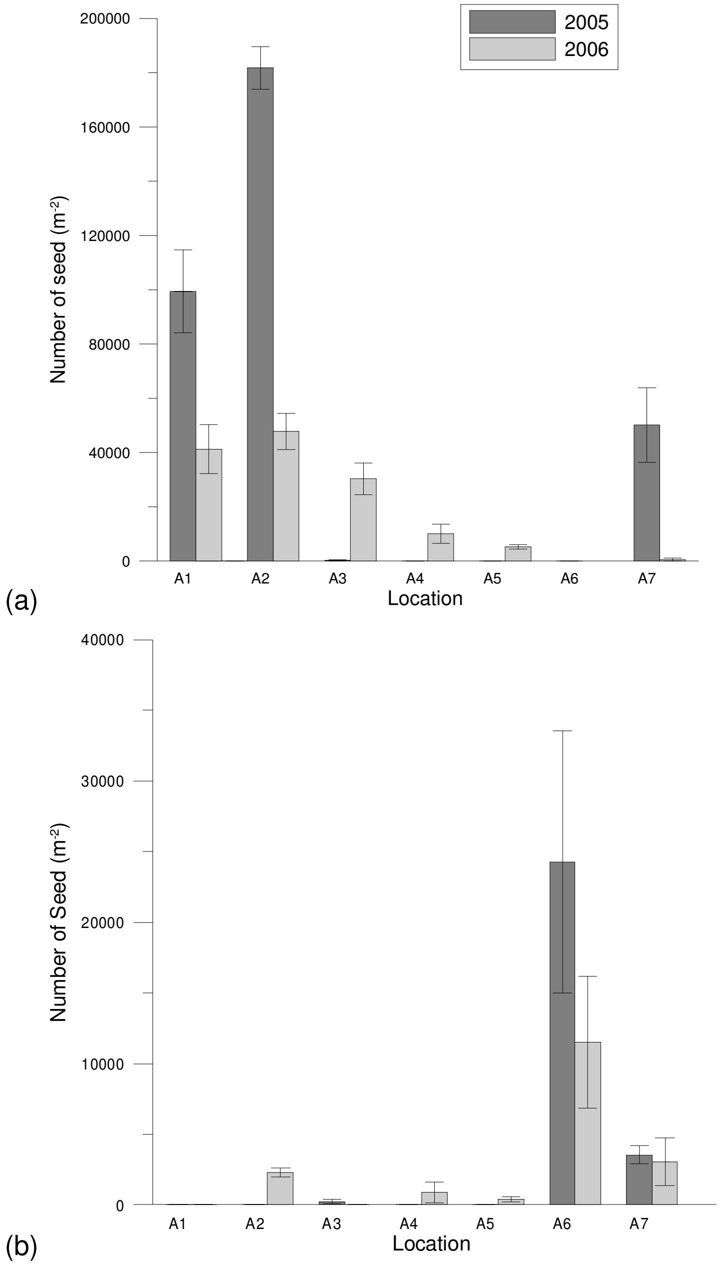
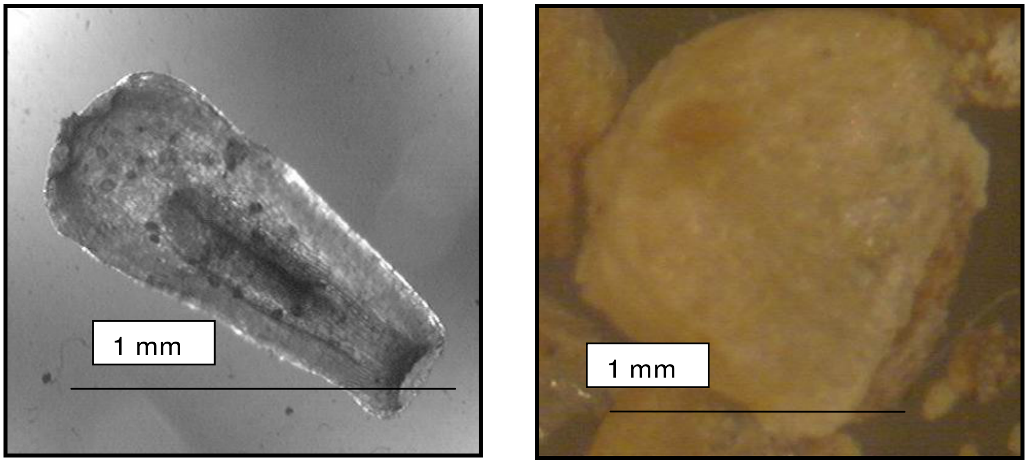


Figure 5.5 The number of (a) *Cotula coronopifolia* and (b) *Sarcocornia pillansii* seeds found along the driftlines of Site A in 2005 and 2006 (bars = ± SE).

5.4.2.1 Seed size



(a) (b)

Plate 16 The seed of the (a) intertidal species *Cotula coronopifolia* and (b) supratidal species *Sarcocornia pillansii*.

The seed of *S. pillansii* was significantly larger (1.3 ± 0.03 mm) than that of *C. coronopifolia* (1.19 ± 0.02 mm) ($p < 0.05$, $n = 50$) (Plate 16).

5.4.2.2 Seedling counts

Figure 5.6a shows the distribution of seedlings for the different sampling sites over the two years. Only one driftline with C*otula coronopifolia* seedlings was found in 2005. This does not mean that there were no other seedlings in the area, but that the seedlings did not grow in driftlines during that particular season. The density of seedlings in 2006 was significantly higher than what was found in 2005 ($p < 0.05$, $n = 100$). Site A2 (585 ± 89 m^{-2}) and A7 (125 ± 64 m^{-2}) had significantly lower seedling density than A3 (1975 ± 381 m^{-2}) in 2006 ($p < 0.05$, $n = 100$).

No *Sarcocornia pillansii* seedlings were found in sites A2, A3, A4, A5 and A6 in 2005 (Figure 5.6b). Site A1 (271 ± 51 m^{-2}) had significantly less seedlings than A7 (1140 ± 266 m^{-2}) in 2005 ($p < 0.05$, $n = 40$). Site A7 had a significantly higher number of seedlings (3830 ± 700 m^{-2}) than all other locations in Site A in 2006 ($p < 0.05$, $n = 100$). Sites A2 (295 ± 60 m^{-2}) and A3 (225 ± 79 m^{-2}) had significantly less seedlings than other locations in 2006 ($p < 0.05$, $n = 100$). No seedlings were found at A4 and A6 in 2006.

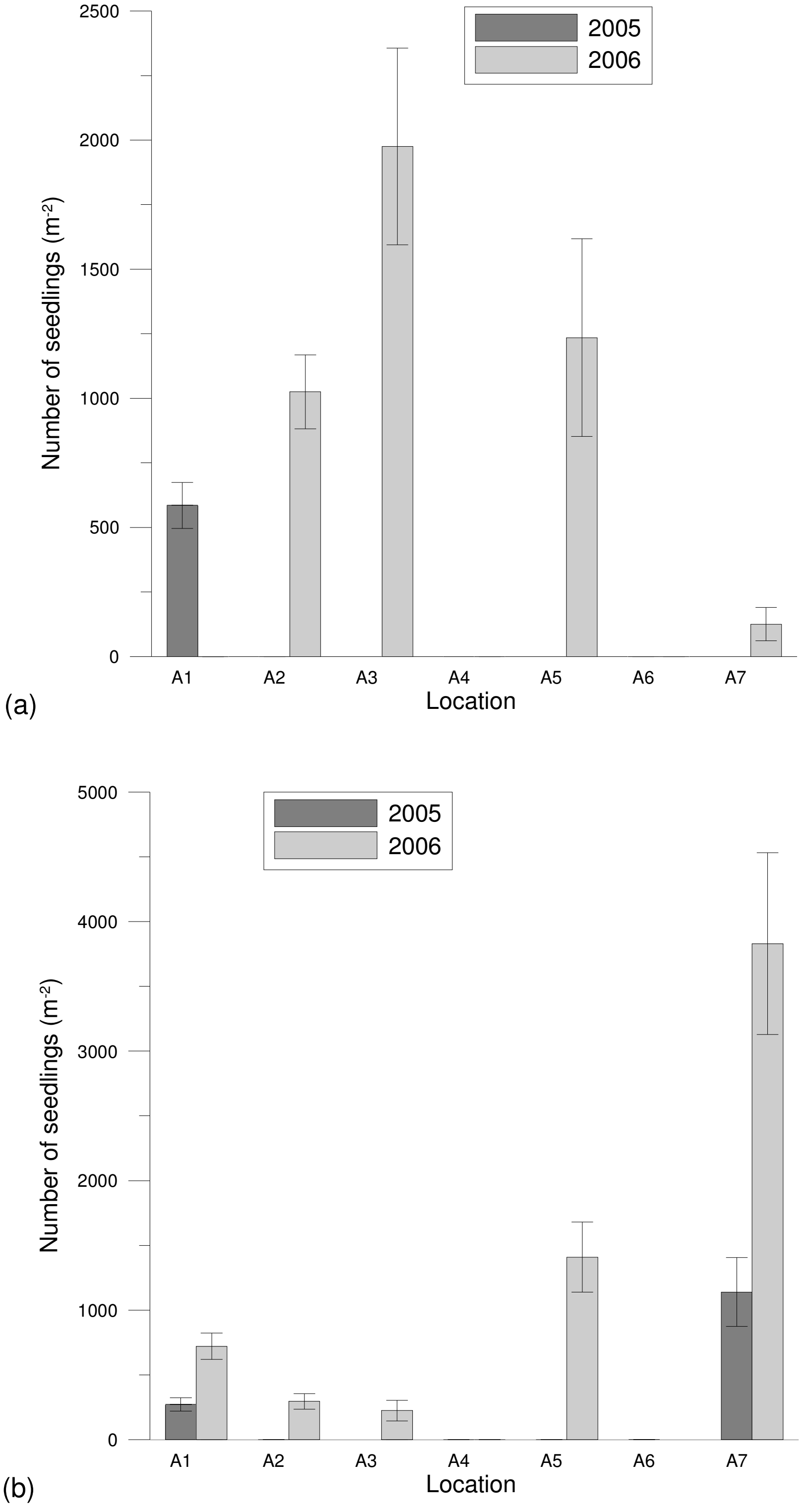


Figure 5.6 The density of (a) *Cotula coronopifolia* and (b) *Sarcocornia pillansii* seedlings found in the driftline in 2005 and in 2006 (bars = ± SE).

Figure 5.7 shows the distribution of *Sarcocornia pillansii* seedlings according to height class. There were significantly more seedlings in 2006 than in 2005 ($p < 0.05$, $n = 120$). In 2006 there were a few individuals that exceeded 80 mm in height whereas the seedlings measured in 2005 were shorter. The radii of the *Cotula coronopifolia* seedlings had a mean of 13.1 ± 1.2 mm in 2006 which were significantly smaller than those measured in 2005 (20 ± 0.7 mm) ($p < 0.05$, $n = 100$).

Rainfall data showed that the rainfall received in 2006 was higher than that for 2005 and the annual mean (Figure 3.4). The highest rainfall (28 mm) occurred in April 2006 (Figure 3.3).

In 2005, 17 % of the seeds produced by *C. coronopifolia* were deposited in driftlines. Of those deposited in the driftline only 1 % germinated and matured to seedlings. This equates to only 0.18 % germination success. The *S. pillansii* driftline contained only 2 % of the seeds produced of which 8 % germinated (Figure 5.8).

After a rainfall event in 2006, 5 % of the seed produced by *C. coronopifolia* appeared in the driftlines and 3 % of the seed germinated. *S. pillansii* showed the larger response to germination when 2.7 % was deposited on the driftline and 36 % germinated. Germination in 2006 was close to 1 % of the total seeds produced for *S. pillansii*. In 2005, germination was only 0.1 % of production for both species (Figure 5.8).

The life cycle of the two dominant species showed a positive response to freshwater in 2006 with a higher germination than was recorded in 2005. Other factors responsible for loss include export by flood and tide, loss to wind, and unfavourable conditions for germination, but these were not investigated in this study (Figure 5.8).

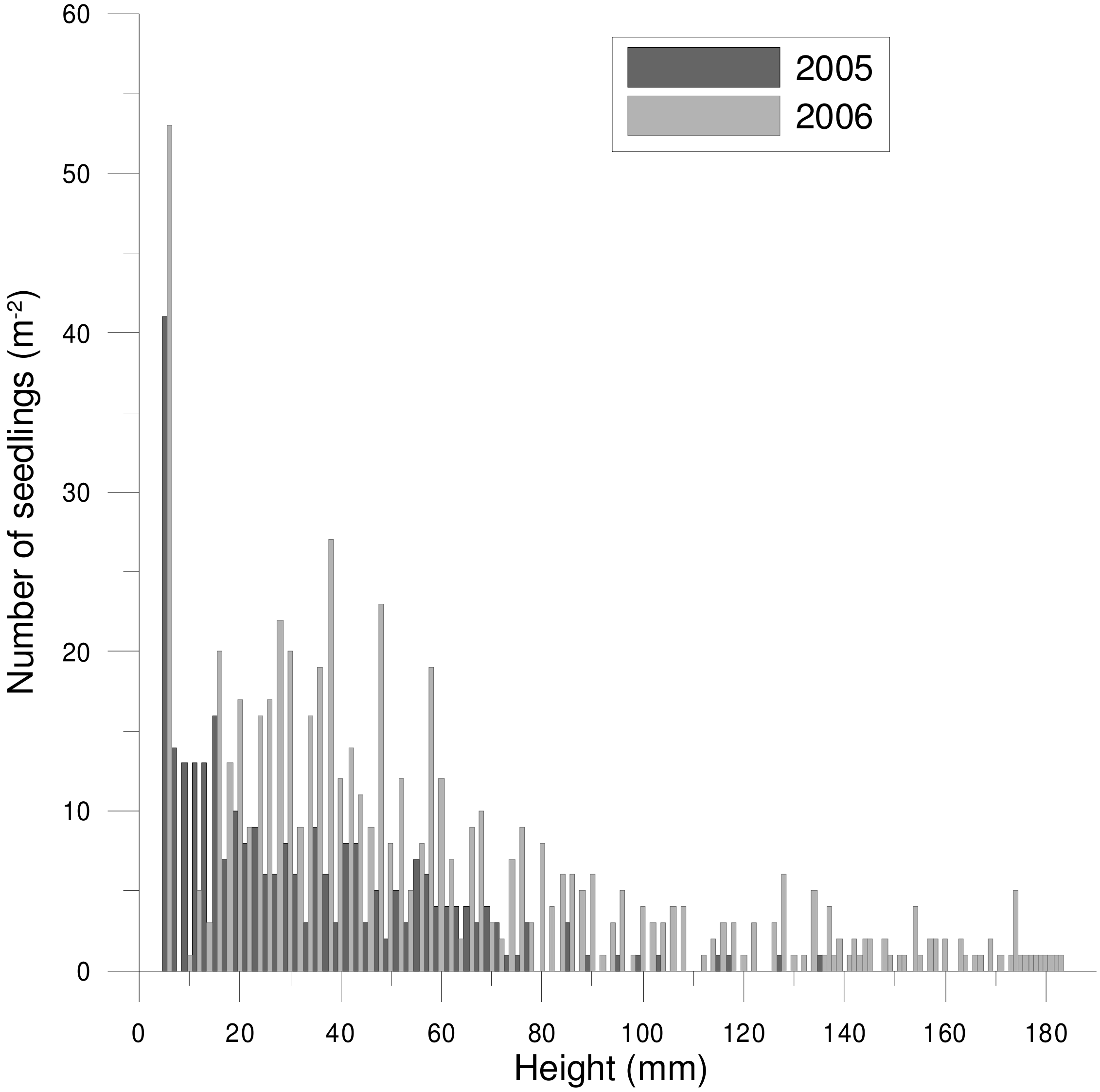


Figure 5.7 The distribution of seedling size classes (*Sarcocornia pillansii*) found in the marsh for two successive years.

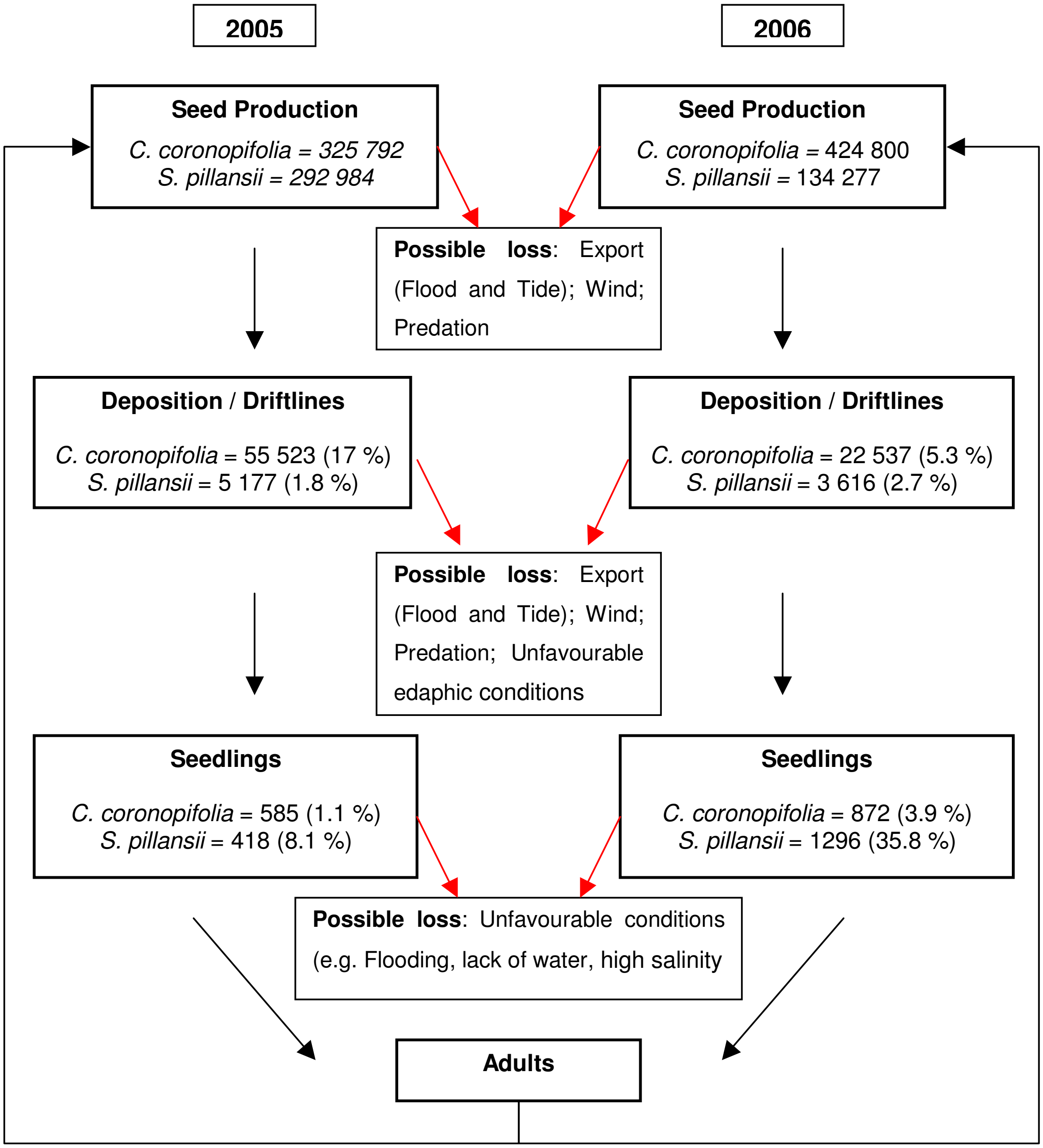
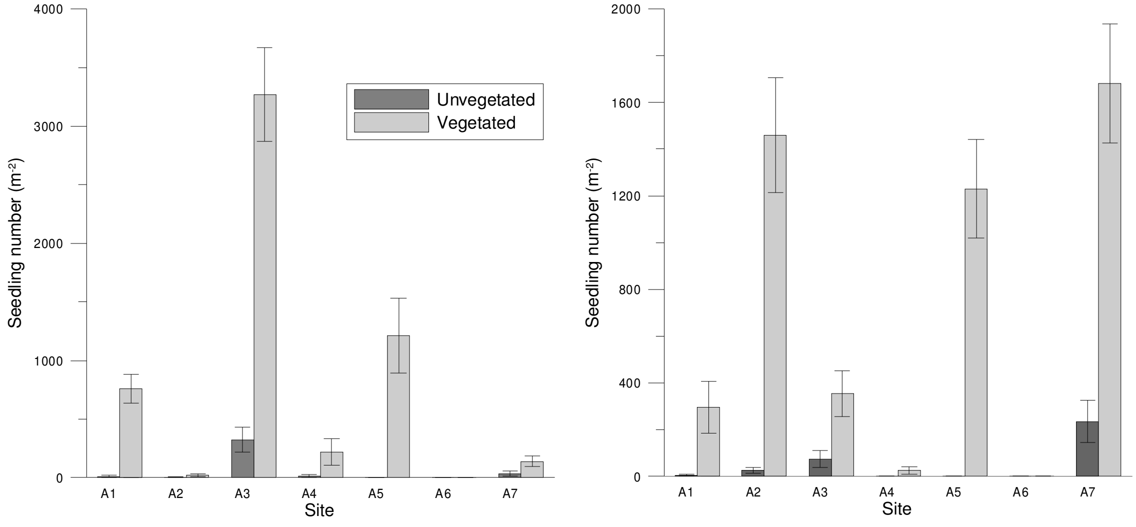


Figure 5.8 Flow chart showing the path from seed to mature plant and the possible losses (not measured) at each life stage. Values are given for the intertidal and supratidal species (Units are .m^{-2}).

5.4.2.3 Seedling microhabitat

No seedlings of either species occurred in the open or vegetated areas in A4 and A6 (Figure 5.9). Site A3 contained the highest seedling density in the microhabitat created by the adult vegetation. Seedling densities of *S. pillansii* were lower than *C. coronopifolia* but occurred in more of the sites (Figure 5.9).

Figure 5.10 is the combination of the data in Figure 5.9 presented as the mean. More seedlings germinated and established under the vegetation (Figure 5.10). The seedling density of the two species did not differ significantly ($p > 0.05$, $n = 280$) within the microhabitat but there was a significant difference between the two microhabitats ($p < 0.05$, $n = 280$) (Figure 5.10.).



(a) (b)

Figure 5.9 Seedling density for (a) *Cotula coronopifolia* and (b) *Sarcocornia pillansii* in the microhabitats of open areas and under the vegetation for the locations in Site A in 2006 (bars = ± SE).

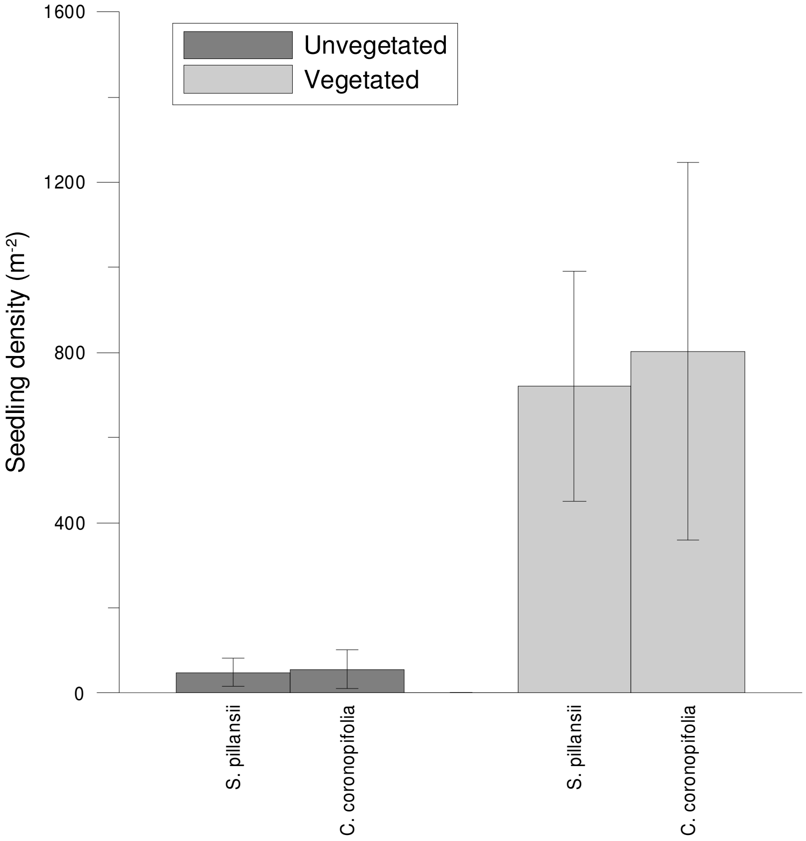
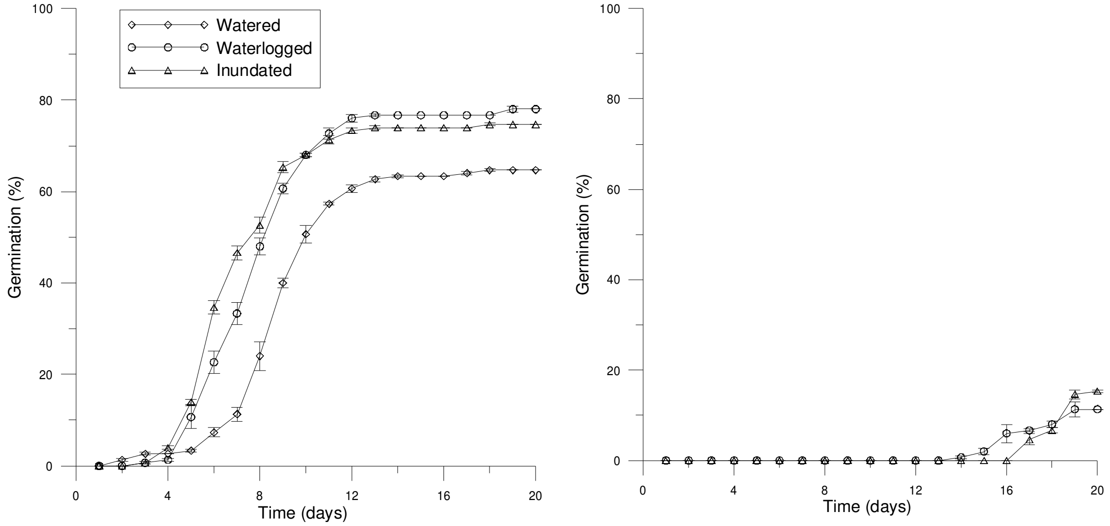


Figure 5.10 The seedling density of the two dominant saltmarsh species in the open areas compared to those under the vegetation (bars = ± SE).

5.4.3 Seed viability

5.4.3.1 Salinity and water treaments

The intertidal species, *Cotula coronopifolia,* had a higher viability than that of the supratidal species *Sarcocornia pillansii*. Under the watered regime, germination of *C. coronopifolia* was only found in the 0 psu treatment. It reached a maximum viability of 78 % and germination started after two days. All treatments at 0 psu had a germination percentage above 60 % compared to 15 % germination for 15 psu treatments (Figure 5.11a). At 15 psu only the watered and inundated regimes showed any germination and only occurred after 14 days (Figure 5.11b). No germination occurred at 35 psu (seawater).



(a)

(b)

Figure 5.11 Viability of *Cotula coronopifolia* at a salinity treatment of (a) 0 psu and (b) 15 psu with different water treatments (bars = ± SE).

S. pillansii seeds had the highest germination percentage (40 %) at 0 psu and this value dropped to 5 % at 35 psu (Figure 5.12). Germination reached its maximum after three to four days. A maximum germination of 25 % was achieved in the waterlogged treatment at 0 psu (Figure 5.12a). Optimal germination occurred for the watered treatment at salinity of 0 psu due to the significantly higher germination than the 15 and 35 psu treatments.

The 15 psu waterlogged treatment had a total germination of 20 % (Figure 5.12b). Again germination increased from day three to four. There was no significant difference between the watered treatments for seeds at 15 psu ($p > 0.05$, $n = 180$).

At 35 psu the waterlogged regime showed germination after 11 days compared to that of the watered regime which germinated only after 13 days (Figure 5.12c). Germination ranged from 5 % in the watered, and waterlogged treatment to 15 % under the inundation treatment.

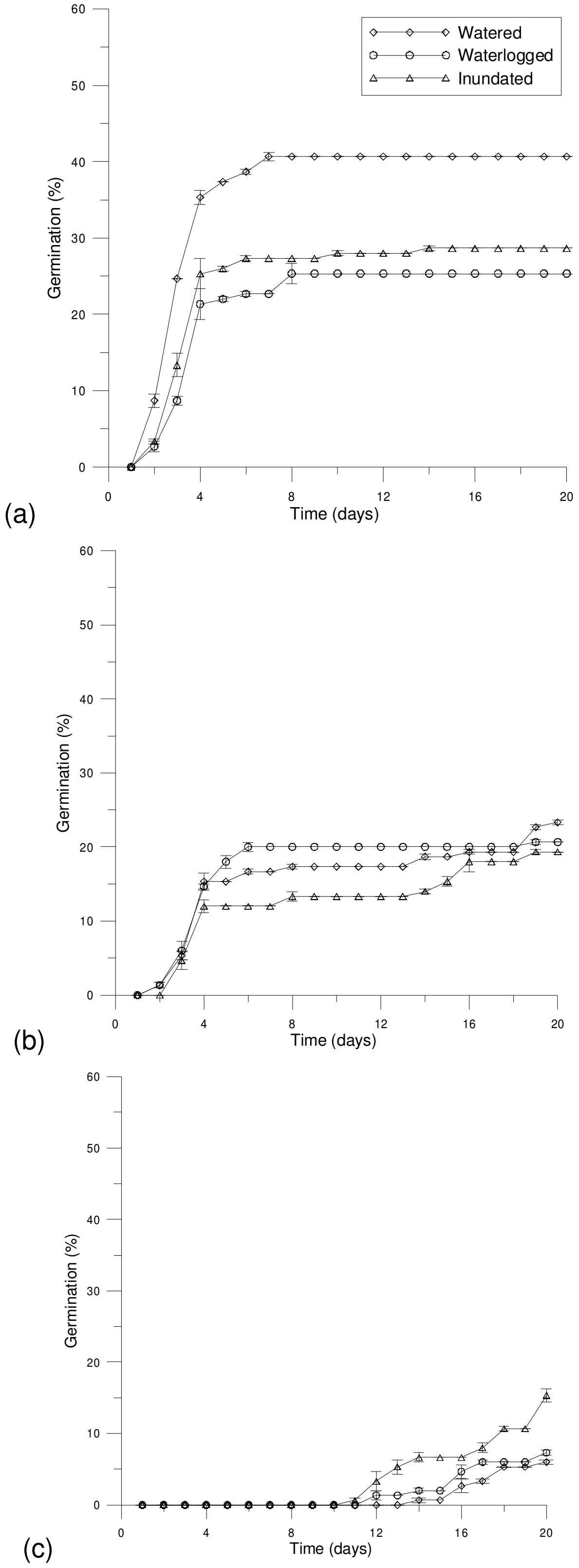


Figure 5.12 The viability of *Sarcocornia pillansii* seed at (a) 0 psu, (b) 15 psu and (c) 35 psu, with different water treatments (bars = ± SE).

5.4.3.2 Effect of hypersalinity

Cotula coronopifolia (Figure 5.13a) showed no difference in germination viability for the three treatments. Germination was maximum at 98 % after 4 days exposure to hypersalinity for three and six weeks, with the seeds treated with one week of hypersalinity germinating to 96 % after 10 days. When comparing this experiment to the investigations for optimum conditions (Section 5.4.3.1), the seeds exposed to hypersaline conditions germinated faster and with more success.

Sarcocornia pillansii (Figure 5.13b) had a seed viability of 20 % that was not significantly different from the optimum condition experiment (Section 5.4.3.1). However only a few seeds germinated every few days extending the germination period by 10 days. The exposure to hypersalinity for 6 weeks did decrease viability and germination ceased on day 15 where it continued in the other two treatments.

5.4.3.3 Storage conditions

There were no significant differences in viability for seeds stored under warm conditions ($p > 0.05$, $n = 24$). Germination reached its maximum (19 – 21 %) after 18 days for the seeds stored under warm conditions. The viability after storage (warm) was 50 % of the optimum viability found in previous experiments (Sections 5.4.6.1 and 5.4.6.2) (Figure 5.14a). Viability was only 10 % after three months of cold storage, except for the dry seed kept in light conditions, which reached the same viability to that of the seed stored under warm conditions i.e. 20 % (Figure 5.14b). Rapid germination occurred on day four for the seed stored under cold storage but little seed germinated thereafter whereas seed germinated consistently after storage in warm conditions.

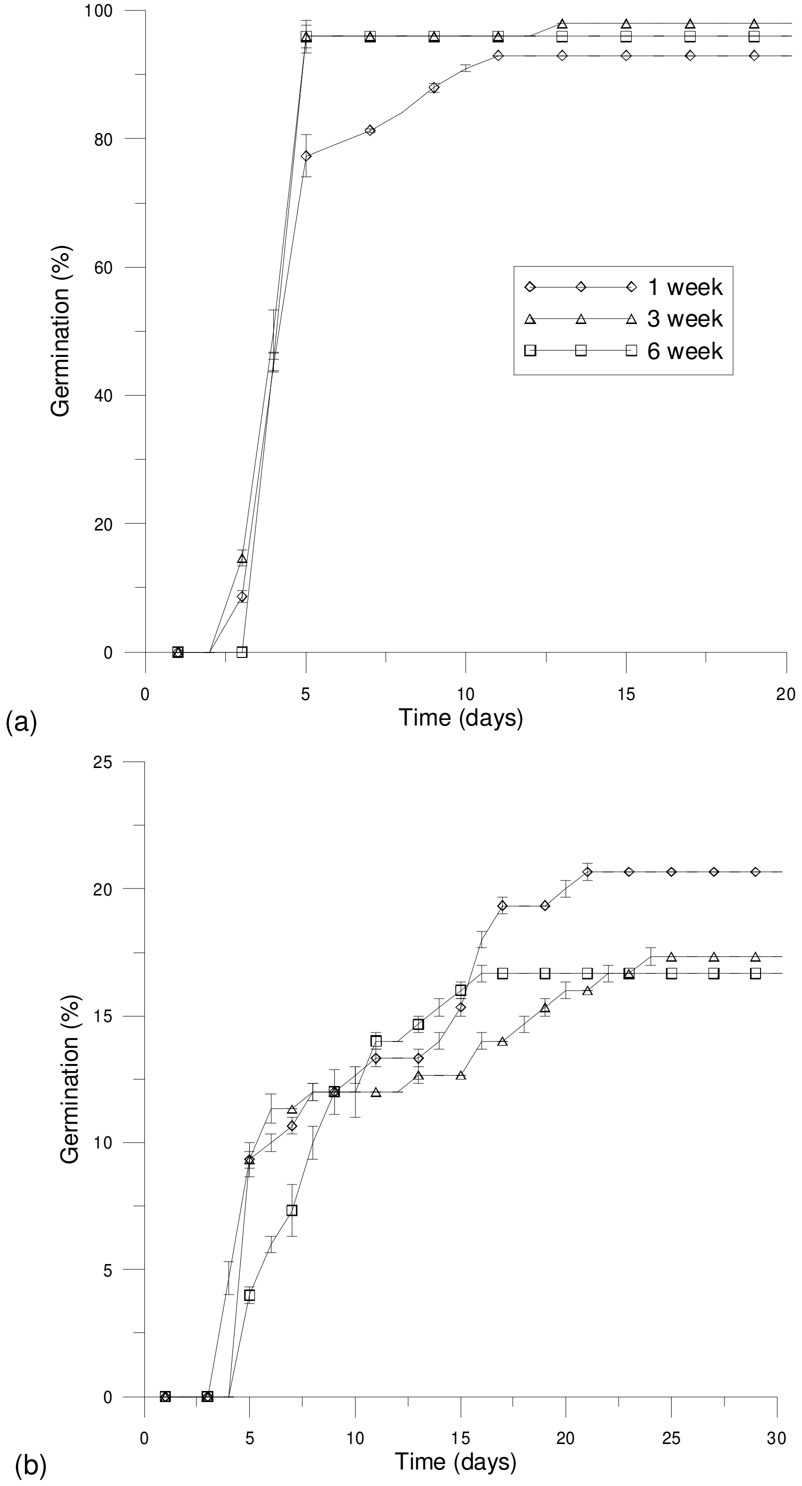


Figure 5.13 The germination of seeds for (a) *Cotula coronopifolia* and (b) S*arcocornia pillansii* after treatment in saline water for one, three and six weeks (bars = ± SE).

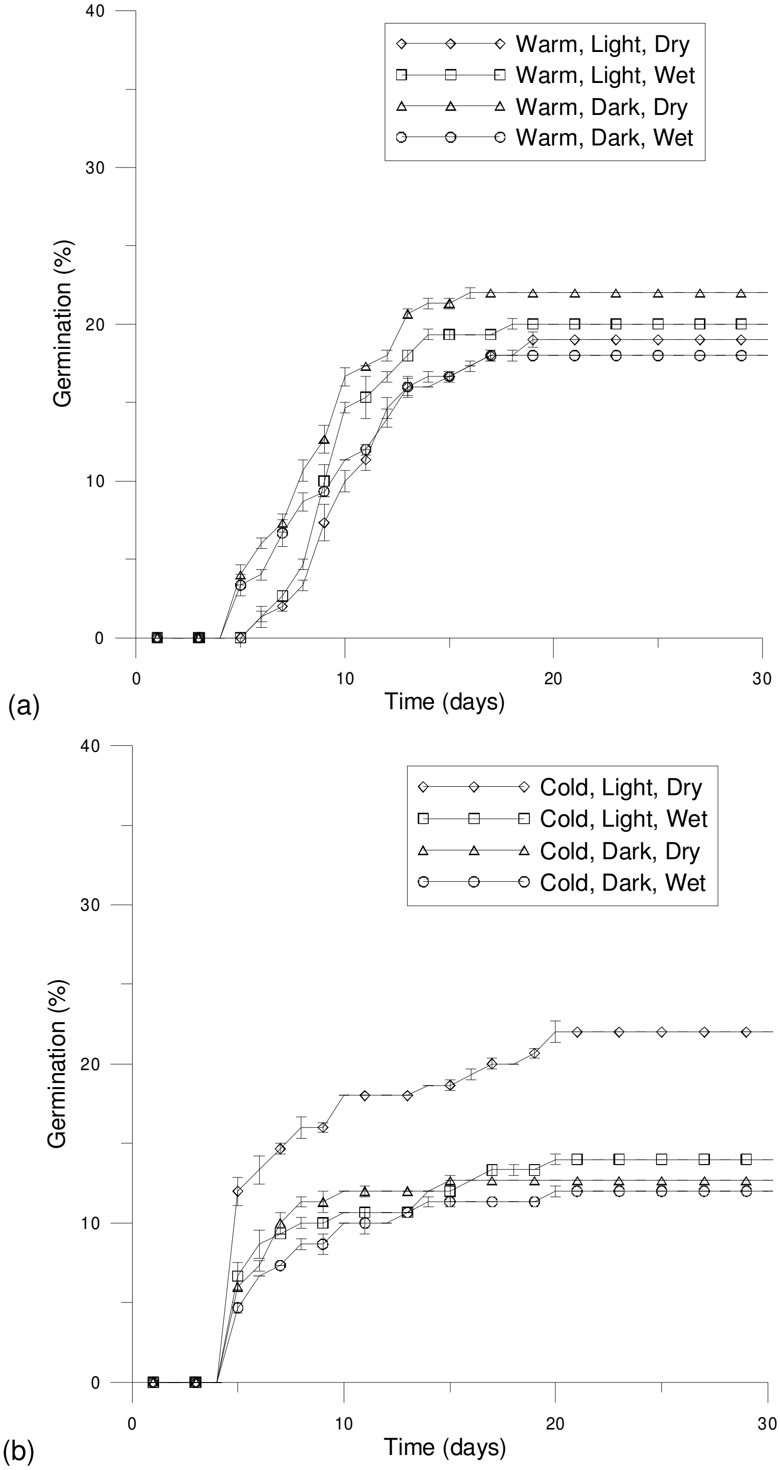


Figure 5.14 The viability of *Sarcocornia pillansii* seeds after three months of storage under (a) warm and (b) cold conditions (bars = ± SE).

5.5 Discussion

Approximately 140 ha of saltmarsh have been lost at the Orange River Estuary because of various anthropogenic causes. These causes mainly relate to the reduction of freshwater entering the marsh. This has resulted in a harsh environment and unfavourable sediment and groundwater salinity make active rehabilitation (transplanting) impossible. Successful natural rehabilitation will only take place if the seed that are produced, by the few remaining adult plants, germinate and survive to adulthood. An understanding of the niche that the species occupy and their salinity and moisture tolerance range allows for the prediction of areas where plants may germinate and establish under favourable conditions. This study investigated seed production and dispersal to determine where there is adequate seed to recolonize the site once conditions become favourable.

It is important to understand the seed dynamics of the two dominant species as they occupy different habitats and their success varies accordingly. Existing vegetation of the study site is closely represented in the seed bank samples which is not uncommon in saltmarshes (Hutchings & Russell, 1989; Ungar & Woodell, 1993; Aziz & Kahn, 1996; Guardia *et al.*, 2000; Gul & Weber, 2001; Shaukat & Siddiqui, 2004). The large amount of seeds produced by these plants is characteristic of saltmarsh plants (van der Valk & Davis, 1979). The intertidal species *Cotula coronopifolia* produced more seed in 2006, 424 800 m^{-2} compared to 325 792 m^{-2} in 2005, due presumably to the rainfall and flooding event providing fresher water. Seed production for *Sarcocornia pillansii* was lower in 2006, after rainfall, due to the plants allocating resources to biomass (stems per m^{2}) (Zedler, 1983; Naidoo & Rughunanan, 1990; Keiffer *et al.*, 1994; Leck & Simpson, 1995; Alexander & Dunton, 2002) rather than to seed production. The timing of the flood and rainfall was also outside of their normal seed production period, i.e. after winter.

At the Orange River Estuary, the number of seeds produced did not correlate to percent health or percentage cover of the adult plants. The assumption was that the distinctive colour of the vegetation indicated the stress the plants were under. The supratidal species shows a purple colour when containing a large amount of solutes stored in the vacuoles of the plant (Stewart & Lee, 1974). These plants increase their solute concentrations to increase the water potential gradient (Stewart & Lee, 1974; Aziz & Kahn, 2000, 2001, 2003). Overall *Sarcocornia pillansii* had a seed output of approximately 40 billion seeds and *Cotula coronopifolia* approximately 8 billion. The larger production by the supratidal species was because of the larger area covered by adult seed producing *S. pillansii* plants. Ungar (1978) hypothesized that large persistent seed banks may exist in hypersaline sediments because seeds germinate only if stressful conditions are alleviated.

Both these species, not unlike many other wetland species, use water as the main means for transportation of seeds. This was evident from the concentration of seeds in driftlines compared to other areas of the marsh.

The seed bank at the ORM saltmarsh was on the surface of the sediment and not at depth. Less than 20 % (55 523 seeds m^{-2}) of the seed produced in 2005 of *C. coronopifolia* were found in driftlines. Seed loss could be attributed to the export of seed to the rest of the estuary and the sea as this species is located in the intertidal habitat. In 2006, only 5 % of the seeds produced (22 537 seeds m^{-2}) were found along driftlines. Seeds were presumed to have been exported from the system due to the flood and the constant tidal exchange that was re-established by the flood.

Seed deposition in driftlines increased in the case of *S. pillansii* from 2 % (2005) to 3 % (2006) of the total estimated production. This can be directly related to the flood where the seed was removed from vegetated areas in the marsh and concentrated in driftlines created by the higher water levels. An additional effect of the increase in water level and water entering the marsh was the dispersion of seeds into new areas. Seed rich driftlines have been recorded in the Vindel River (Sweden) where 189 181 seeds per m^2 were detected in the driftlines following a spring flood (Nilsson & Grelsson, 1990) and in the Schiermonnikoog Saltmarsh (The Netherlands) where a more than twofold increase in the number of seeds was detected in the driftline compared to the surrounding sediment (Wolters & Bakker, 2002). The seedling density data indicated that only 8 - 36 % of seeds deposited germinated into seedlings. When comparing the seedlings found in the marsh to the seeds produced the ratio falls between 0.17 and 0.94. Leck and Simpson (1995) had ratios ranging from 0.004 to 2.9. Although it seems that very little seed recruitment is taking place, the ratios are similar to that found in other studies.

The dispersal of seeds does not mean that the conditions in the new areas are favourable for germination and establishment (Chapter 4). Viability tests showed that germination of the two salt marsh plants at the ORM was affected by salinity. Similar results were recorded in other studies on the influence of salinity (Kingsbury *et al.*, 1976; Ungar, 1978; Woodell, 1985; Callaway *et al.*, 1990; Shumway & Bertness, 1992; Kuhn & Zedler, 1997; Noe & Zedler, 2000; Zhu, 2001; Vincente *et al.*, 2004) and moisture (Kuhn & Zedler, 1997; Noe & Zedler, 2000; Rubio-Casal *et al.*, 2002; Vincente *et al.*, 2004). Freshwater (0 psu) was the most favourable salinity condition for maximum germination (60 – 80 %) of *C. coronopifolia*. At 15 psu viability was significantly lower at 20 % although Partridge and Wilson (1987) reported 40 % at 20 psu for the same species. Moisture availability is more important at higher salinities (Noe & Zedler, 2001), with more seeds germinating under the inundation experiment than under only moist conditions. *S. pillansii* had a lower viability (40 %), but germinated in salinity up to 35 psu. Higher salinities reduced seed germination (Naidoo & Kift, 2006). The viability measured in this study was lower than the recorded viability for other species in this genus. Redondo *et al.* (2004) recorded viability of between 85 and 89 % for other *Sarcocornia* species. Further explanation of low viability could be due to the seeds being collected shortly after seeds were set. At this stage, seeds may be dormant, as has been found for many wetland plants (Cleavering, 1995). The salinity tolerance of the seed of the two species indicated the origin of the adult plants with *C. coronopifolia* restricted to the more brackish intertidal zone and *S. pillansii* to the more elevated, dry, saline supratidal and floodplain areas.

The germination laboratory results were reflected in the field as germination was found in areas where the sediment was moist and had a low salinity. These favourable conditions were associated with microhabitats created by the adult vegetation and driftlines. Although seeds germinate in freshwater, seedlings require an additional two to three weeks of high moisture to survive (Alexander & Dunton, 2002; Kuhn & Zedler, 1997). Driftlines create favourable conditions for seedling growth, however more seedlings were found in microhabitats associated with adult vegetation. Similar results were recorded by Reichman (1984), Hutchings and Russell (1989) and Guo *et al.* (1998).

The higher number of seedlings under the vegetation may also be due to the vegetation acting as a trap for the seeds (Smith & Kadlec, 1985; Aziz & Kahn, 1996; Aguiar & Sala, 1999; Bullock & Moy, 2004) from wind and rainfall (Ma *et al.*, 2006). Microhabitats under the existing vegetation also allow for prolonged periods of suitable environmental conditions. Seedlings in microhabitats were only present in 2006 following germination in response to the lower salinity of the marsh (Chapter 4). Higher seedling densities were found in driftlines and in the microhabitat under the adult vegetation. The open unvegetated areas contained seedlings but in lower numbers and they were not as tall as the individuals in the other two habitats. This was due to the sediment forming a crust making establishment difficult for the seedlings (Brock & Rogers, 1998). Other studies have shown that mean seedling density was variable among wetlands and habitats in the supratidal zone. Seedling densities ranged from 419 seedlings per m^2 in the supratidal vegetated areas of the Tijuana Estuary to 4421 seedlings per m^2 in the intertidal areas in the Sweetwater Marsh, California (Noe & Zedler, 2001). Seedlings at the ORM had densities ranging from 729 seedlings per m^2 in the intertidal zone (Sites A1 and A2) to a mean of 857 seedlings per m^2 in the supratidal zone (Sites A3, A5 and A7).

The sediment characteristics of the vegetated microhabitat were more favourable than the driftline and open areas due to the lower variability, particularly sediment electrical conductivity (Chapter 4). The fluctuation in salinity and moisture in response to evaporation and tidal changes in the driftline and open areas allowed for germination, but the germination rate and seedling density was low. *Sarcocornia pillansii* seedlings increase in height in salinities up to 35 psu but growth ceases at 70 psu (Bornman, 2002).

In 2005, *S. pillansii* seedling height was measured in order to assess the persistence of seedlings over time. However, the site was inundated by water in 2006 and the same seedlings could not be measured. The seedling height recorded in 2006 was in a different location but the sediment characteristics were similar to Site B. The presence of seedlings of different heights occurring in the same area suggests that the seed must have remained viable in hypersaline conditions until the salinity was reduced by the flood and high rainfall. This was confirmed in the laboratory where after being treated under saline conditions and placed in fresh water the seeds germinated. It appears as if the salinity treatment increased the viability (Naidoo & Naicker, 1992; Rubio-Casal *et al.*, 2003, Naidoo & Kift, 2006; Kahn & Gul, 2006) of *C. coronopifolia*, with 98 % of the seeds germinating within three days after treatment (van der Toorn & ten Hove, 1982; Partridge & Wilson, 1987). The seeds appeared to have enforced dormancy when present under saline conditions (Zia & Kahn, 2002) with the seeds recovering when salinity stress is relieved (Kahn & Gul, 1998).

Plate 17 The microhabitat associated with the adult S*arcocornia pillansii* plants creating favourable conditions for seed germination.

Plate 18 Open areas initially create favourable conditions, but due to soil crusting (compaction) and fluctuating edaphic conditions, seedling survival is low.

Sarcocornia pillansii seed did not germinate as quickly with only a few seeds germinating on alternate days. Exposure to high salinity does not only provoke inhibition of germination but also decreases germination speed and rate (Ungar, 1978; Rubio-Casal *et al.*, 2003; Kahn & Gul, 2006). Similar results were obtained by Rozema (1975) studying several species of *Juncus* and Rubio-Casal *et al.* (2003) in the study of two saltmarsh species in a Mediterranean climate.

The results from this study showed that *S pillansii* recovered and reached a viability of 20 %. Other laboratory studies have shown that halophyte seeds can remain viable for long periods under highly saline conditions and germinate when salinity declines (Ungar *et al.,* 1979; Ungar, 1978; Rubio-Casal *et al.*, 2003). This suggests that the seed could persist in the seed bank until conditions favour germination (Hutchings & Russell, 1989). This was confirmed as the seed stored under warm conditions showed higher viability than those under cold conditions. These results reveal a very clear ecological niche for *S. pillansii*. The area in which the species was found had a very marked seasonal cycle and was subjected to environmental extremes, especially during summer. During summer the salt marsh plants are exposed to high sediment salinity (ranging from 40 to 120 $mS.cm^{-1}$ depending on the area).

It is presumed that seeds would remain dormant until the environmental conditions become more favourable, e.g. after a freshwater event (flood / rainfall) decreases the sediment salinity for a short period. Earlier cohorts would therefore have an advantage over later ones (Rubio-Casal *et al.*, 2003). Even though the percentage of seeds that germinate are low it is complemented by the abundant seed bank. A degree of dormancy due to unfavourable conditions (high salinity, low precipitation) and a large seed bank would assure the survival of this species. It can therefore be assumed that the seed bank of the two species is type III as it is made up of mostly seed at the sediment surface that are capable of immediate germination, a few of which will remain viable for more than a year (Thompson & Grime, 1979; Hutchings & Russell, 1989; de Villiers *et al.*, 1994). This can be defined as a persistent seed bank as not all the seed germinate at one time.

Conclusion

Seed banks are important in determining community structure (Kahn & Gul, 2006). Seeds allow establishment at locations distant from neighbours, while vegetative growth is limited to a particular area. Seeds also facilitate the rapid bare area re-establishment following the loss of adult populations (e.g. during hypersaline conditions) (Noe & Zedler, 2001). Water allows seed to be deposited in favourable areas and increases the chances of survival for the species (Smith & Kadlec, 1985; Nilsson & Grelsson, 1990). Large numbers of seed of the intertidal species were found in driftlines in close proximity to the spring high tide mark. The number of seeds produced in the saltmarsh are large with few settling in driftlines and in microhabitats. It is expected that the remainder was transported to other areas by the wind, floods and tide and some may have been lost to predation. Ants were observed in the field and may play a large role in the removal of seed as well as the distribution and viability of the seed bank (Samson *et al.*, 1992). Although the seed production was high in both 2005 and 2006, the freshwater availability in 2006 was higher and led to higher seedling densities. The seedlings are important in the rehabilitation of the marsh and the period of low salinity and inundation are essential for the seedlings to develop into adults (Bornman *et al.*, 2004b). Since the supratidal floodplain species is a perennial, a few successful seedlings could ensure the long-term survival of the species in the community (Hopkins & Parker, 1984). Driftlines are very important in the ORM and the distribution of the extant vegetation indicates that they established along these water levels (Orth *et al.*, 1994). The present distribution of the adult salt marsh plants in the ORM is as a result of the inhibition of seedling recruitment under the present hypersaline sediment conditions (Baldwin *et al.*, 1996). It is important that freshwater (freshwater and tidal floods) inundates the marsh periodically as this leads to seed entering new areas for establishment, allows previous seedling cohorts to continue growing and enables adults to increase cover and set seed.

Chapter 6: Wind and Dust

6.1 Abstract

Wind blown sand is a natural occurrence in coastal wetland habitats. However the increase in bare areas at the Orange River Mouth as a result of salt marsh dieback has increased the available sediment source. The wind blown sediment has covered large areas of the remaining adult saltmarsh vegetation, particularly in the northern corner at Site C, causing further die-back. Ongoing sediment transport is a threat to the natural rehabilitation of the ORM. Sand movement was evident from the formations formed on the leeward side of adult vegetation. These formations (tails) were characteristic of the marsh and indicated the dominant southerly wind direction. The tails had a high sediment electrical conductivity and particle size was similar to that recorded at Site C. The accumulation of wind blown sediment was significantly higher on the vegetation at the edge of a vegetated stand compared to the centre. Between 2004 and 2006 the height of the elevation profile at Site C increased by 0.5 m, which is faster than the plants can grow thus resulting in smothering. Sediment analysis indicated that the source of the sediment was the marsh itself and not the adjacent mine dumps. Freshwater needs to be introduced to the marsh so that vegetation can re-establish within the degraded open marsh area. Once vegetated the amount of wind blown sediment deposited in the northern areas will decrease. The use of wind breaks are not recommended as these will change the topography of the marsh and possibly have further impacts on saltmarsh establishment and growth.

6.2 Introduction

The effect of the wind blown sediment was visually evident at the Orange River Mouth. The ORM is largely devoid of vegetation because of anthropogenic impacts. The main limiting factor to saltmarsh growth is the hypersaline groundwater and surface sediments. This places the vegetation under stress and vulnerable to other negative environmental factors. The additional factors alone may not be detrimental to the survival of saltmarsh plants but combined with sub-optimum conditions can lead to die-back. According to the CSIR (1991), saltmarsh die-back and exposure and mobilization of the marsh sediments by the wind had the most serious environmental consequence. The deflation of the marsh surface and the encroachment of fine sediment from the south-westerly extreme of the marsh towards the river has smothered downwind vegetation, which in turn has died and contributed to the process.

The mine dumps associated with diamond mining have had an impact on the town of Alexander Bay. They are located on the southern side of the town in close proximity to the beach. The sand from the dumps blows directly into the town and deposits on roads, houses and the golf course. Wind blown dust causes structural wear of facilities and buildings (Lain-You *et al.*, 2003). The wind blown sediment also collects on the plants of the saltmarsh, creating a thin layer on the surface of the adult vegetation (CSIR, 1997) (Plate 19). The loss of vegetation is evident from branches protruding from beneath the sediment. The sediment is not deposited in one event but is gradually transported as the wind reaches the threashold velocity. The threshold velocity is the wind velocity at which sediment becomes mobile. It can vary in accordance with particle size, sediment moisture content and salt content. Vegetation also plays an important role in the ability of the wind to transport sediment. The presence of vegetation disrupts the wind flow and reduces sediment transport. The lack of vegetation on the other hand allows for maximum transport. The sediment at the ORM moves across the marsh in depositional structures that develop on the lee side of marsh structures. In the case of the ORM, initially these structures (tails) formed around vegetation, however due to the lack of vegetation in the desertified marsh, these tails accumulate on existing tails. These tails increased in size from the southern side of the marsh to northern corner where deposition smothers vegetation (Site C).

The hypothesis was that the dust occurring on the saltmarsh was from the mine dumps. To test this, the source of the sediment was determined by using two methods, namely: X-ray diffraction and particle size analysis. X-ray diffraction was used to verify the particle size method. Due to the size of the marsh, determination of the source of the sediment will lead to cost-effective mitigation of wind blown sediment. Vegetation wind breaks are often the cheapest and most effective wind breaks but because of the lack of plant material the effectiveness of shade cloth was tested in this study.

The accumulation of sediment in the northern corner of the marsh was obvious and determination of the rate of accumulation was assessed through a profile of the site as well as the measurement of the accumulation of sediment on the vegetation. The accumulation of sediment on the vegetation was also used to test the hypothesis that the presence of vegetation decreases the movement of sediment by wind. The physico-chemical characteristics of the tails were investigated to understand the properties of the wind blown sediment and to compare these to the depositional site (Site C).

The objectives of the study were to:

- Investigate the source and properties of the wind blown sediment (tails) and compare these with the depositional site (Site C);
- Measure the accumulation of wind blown sediment in the northern area of the salt marsh (Site C) and the accumulation of sediment on the plants; and
- Investigate the potential to reduce wind blown sediment through the use of wind breaks such as shade cloth.

6.3 Materials and Method

6.3.1 Site description

The Orange River Mouth has a large desertified area. In the southern corner of the marsh the vegetation is protected form the southerly winds by a beach berm (Figure 6.1). A shallow lagoon also exists in the south of the marsh. It contains water during spring tide while during neap it is drained and reduced to a small pond. Although it is protected from the southerly wind by the berm it is prone to drying out. Sand movement was evident from the formations formed on the leeward side of adult vegetation. These formations were characteristic of the marsh and indicated the dominant wind direction (SSW).

The site used for this chapter was study site C (Chapter 4). It is a depositional environment where sediment transported by the wind is trapped. It has a high elevation and borders a side channel of the Orange River (Figure 4.1). The vegetation at the site is characterised by monospecific stands of *S. pillansii* and no seed or seedlings were found in this area.

Plate 19 Sediment accumulation on the vegetation in Site C.

The wind at the Orange River Mouth had a dominant southerly wind direction and the wind strength (Figure 3.5) peaked in the afternoon (Figure 3.6). The wind is the strongest in summer when conditions are warmest (Figure 3.7).

6.3.2 Sediment tail characteristics

The causeway (Figure 4.1) will influence the effect of the wind and therefore sites were chosen to the north and south of the causeway in order to assess whether the properties of the tails differed. Six sites were chosen along a moisture gradient, from the moist sediment in the south to the dry sediment in the north (Figure 6.1). Site 1 and 2 were located on the northern side of the marsh. Site 3, on the southern side of the marsh was located along the driest part of the moisture gradient and site 6 was the wettest site. The northern sites had the most prominent tails and the southern sites the smallest. Five replicate samples were taken at each site. The top 1 cm of sediment was collected and taken back to the NMMU laboratory for analysis of sediment characteristics. Sediment characteristics measured included particle size, electrical conductivity, moisture and organic content. The methods were the same as those used in Chapter 4. The sediment properties of the tails were compared with Site C described in Chapter 4. Site C1 was completely covered by sediment. The second site (C2) was gradually being smothered (covered 50 % of the vegetation) by wind blown sediment. The third site (C3) was at the centre of the monospecific *S. pillansii* stand.

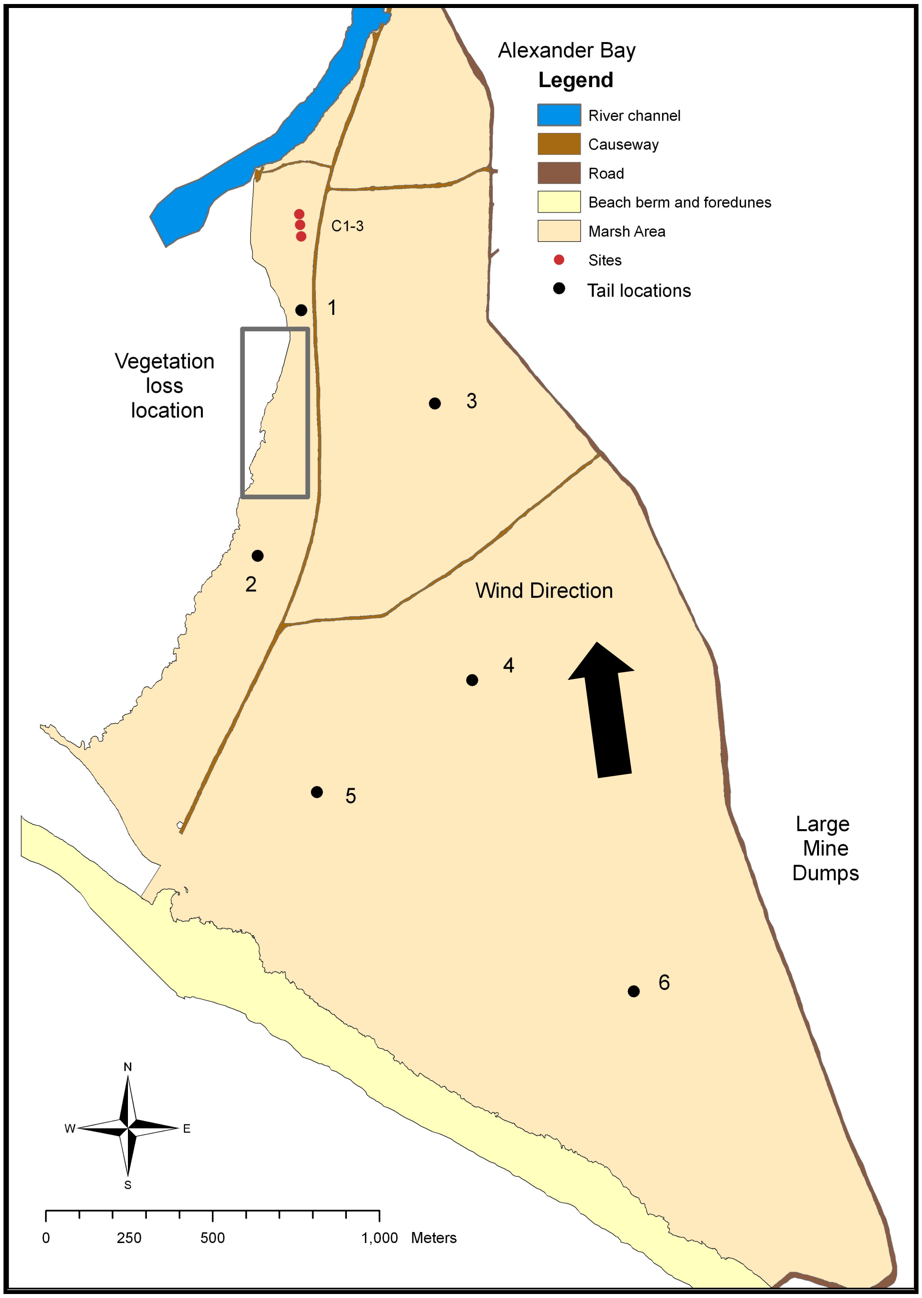


Figure 6.1 Map showing the study sites, path of sediment transport and the orientation of tails in the marsh.

6.3.3 X-ray diffraction

X-ray diffraction is used to determine the mineral composition of materials. A beam of X-rays is focussed on a powdered sample where crystals have a varied and random orientation. Rotating the sample through 120 degrees and using a graphite monochromator to give an incident beam of only the K alpha Cr line, X-rays will be reflected (not fluoresced) at the appropriate Bragg angle. The 2D spacings between the denser layers in the crystal lattice can be measured. XRD can be used to identify minerals, but also to analyse one element in one mineral by selecting a known 2D spacing and counting the intensity. In this case it was used to determine the mineral composition of sediment found within the marsh. This method was used to verify the results obtained from the particle size determination (www.wikipedia.org).

Eight samples were used for this analysis. Four of the samples were ashed and washed in HCl to remove any organic content from the sample as it was thought that the diffractor would be inaccurate if organic matter was present. The other four samples were not washed, but ashed. The samples were taken from four sources within the marsh. Of the four sources two were taken from tails, one from Site C and one from the dried up salt pan, south and upwind of Site C (Figure 6.1).

The samples were placed in a Bruker D8 X-ray powder diffractometer with Cu radiation. The phase identification was done using the ICDD diffraction pattern search from "Powder Diffraction file PDF-2: International centre for diffraction data; Pennsylvania USA, Release 2004".

6.3.4 Sediment accumulation on the plants

The accumulation of sediment on the vegetation was measured in 2005. In 2006 there was no accumulation on the vegetation due to rainfall prior to sampling. Within Site C, thirteen sampling sites were located on the edge of a monospecific stand of the dominant supratidal/floodplain species, *Sarcocornia pillansii*. Another thirteen sites were located in the center of the vegetation. Cover abundance of the plants was measured at these sites using a 10 × 10 cm quadrat (three replicate samples; n = 39). Sediment accumulation on the plants was measured by washing all the sediment from the plants into a sample bottle using distilled water. The amount of sediment accumulated on the plants was determined by filtering the wash water and weighing. The Whatman GFC filter paper was dried at 40 °C for 24 h to remove excess moisture and then weighed. The distilled water from the bottle which contained the wind blown sediment was filtered using this filter paper. The bottles were rinsed with distilled water until all the sediment was removed. The filter paper was then folded and placed in Petri dishes in a drying oven at 40 ºC for 48 h. Once the filter papers were dry they were weighed. The initial mass of the paper was subtracted from the mass after filtration.

The amount of wind blown sediment per unit plant area was determined by measuring the stem surface area. Thirteen stems from the edge of the vegetation and 13 from the center were measured for length and diameter using a vernier caliper. The wind blown sediment samples were collected from the same stems.

6.3.5 Sediment accumulation at Site C

A theodolite (Sokkisha TM 6) was used to determine the elevation profile along a permanent transect at the site. The permanent transect (Figure 6.6) had stakes marked at 20 m intervals. The readings taken in 2006 included the 20 m stakes but additional readings were taken every 5 m between the stakes to more accurately determine the addition of sediment. The profile obtained was compared to that taken by Bornman *et al*. (2004b).

In addition to the profile, a GIS analysis was also carried out. The vegetation was mapped from an aerial photograph taken in 2000, using ArcGIS 9. In 2005 and 2006 the edge of the vegetation was mapped in the field to determine if vegetation had been lost using a GPS with ArcPad version 7.0 software. These different boundaries were plotted in ArcGIS and 20 measurements were taken between the three different boundaries to determine distance moved over time.

6.3.6 Wind barrier experiment

Shade cloth experiments were set up in order to assess their effectiveness in limiting wind blown sediment transport. Two different grades of shade cloths (50 and 80 % density) were used and two separate heights above the sediment surface were investigated, namely 0.5 and 1 m. Three locations were chosen in Site C and the shade cloth (1 × 1 m) and (0.5 × 1 m) was secured to brandering timber for rigidity. A total of four 25 cm stakes with a diameter of 4 mm were placed in the sediment surrounding the shade cloth. Two stakes were placed at the sides and one on the windward and one on the leeward side. The shadecloth was left in the field from August 2005 to May 2006. Accurate determination of the effectiveness was not possible because sediment accumulation had been influenced by rainfall and flooding.

6.3.7 Statistics

One-way ANOVAs were used to determine significant differences among means. A Tukey post – hoc test was carried out where significant differences among means were found. All statistical analyses were run using Statistica (version 7, 2003), StatSoft, Inc.

6.4 Results

6.4.1 Sediment tail characteristics and comparison with Site C

6.4.1.1 Sediment particle size

The properties of the tails differed from those of the surrounding sediment (Chapter 4). There was no significant difference in clay content between the tails from the north and south of the causeway ($p > 0.05$, $n = 54$) (Figure 6.2). Site 2 had a significantly larger sand contribution (61.8 ± 6.3 %) ($p < 0.05$, $n = 90$) while the percentage silt was significantly less (5.5 ± 10 %) than those found in Site 1 (39.8 ± 6 %) ($p < 0.05$, $n = 90$).

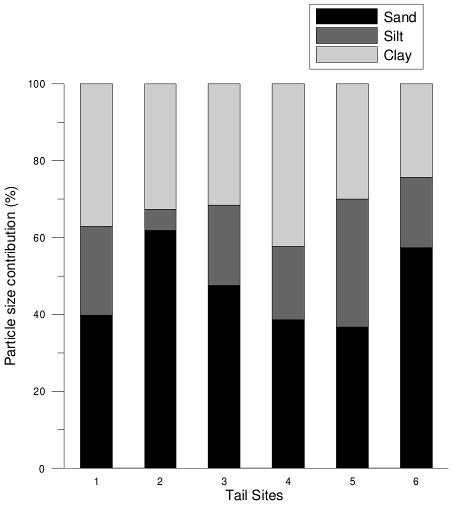


Figure 6.2 Particle size fraction of the sediment in the tails throughout the marsh (n = 5). Sites 1 and 2 were on the northern side of the causeway and Sites 3 – 6 on the southern side.

An increase in the silt fraction of the sediment was recorded from the tails on the desertified marsh through the edge of the vegetation to the centre of the vegetation in Site C (Figure 6.3). This increase in silt was not significant ($p > 0.05$; n = 69), nor was there any significant differences between any of the particle size fractions making up the sediment of the tails and Site C ($p > 0.05$, n = 69) (Figure 6.2.).

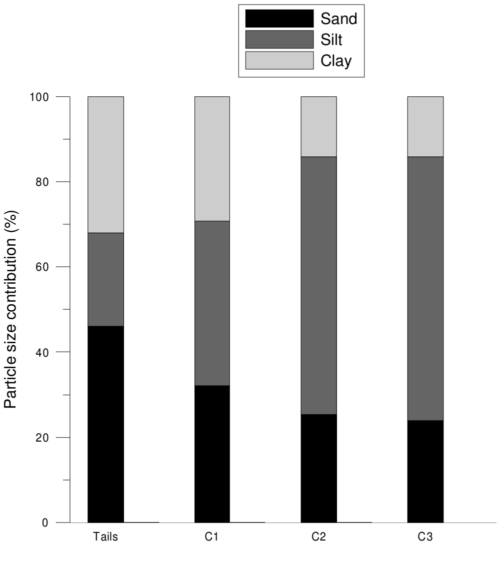


Figure 6.3 The contribution of the different particle size fractions, moving from the tails into Site C.

6.4.1.2 X-Ray diffraction

The results of X-ray diffraction suggest that there are a number of minerals present in all the samples. There are no differences between samples with a few trace minerals occurring in the sample of Site C and the salt pan. Muscovite, illite and quartz were the most abundant minerals (Figure 6.4).

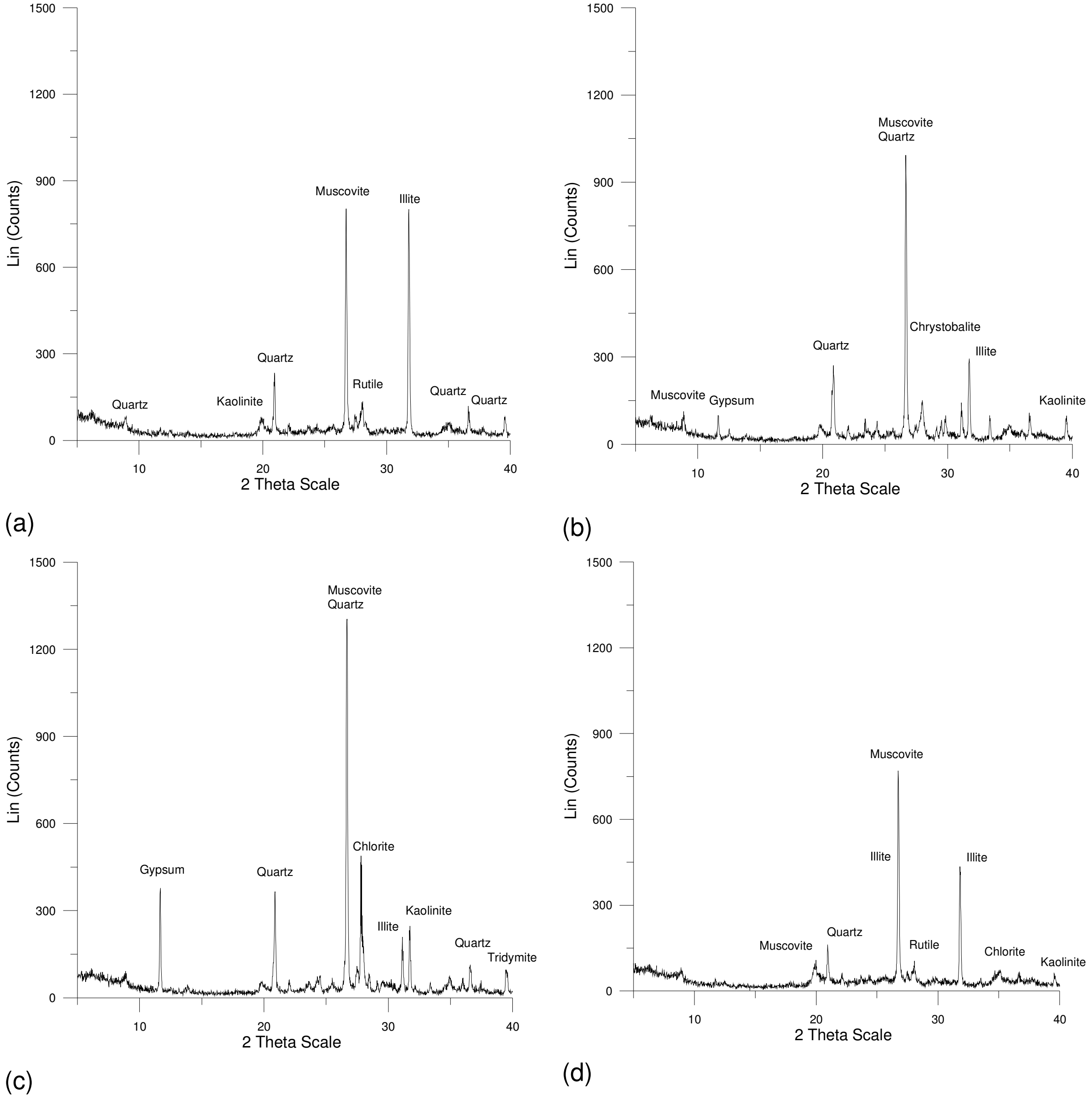


Figure 6.4 X-ray diffraction of sediment from (a) Site C, (b) tails on the southern side, (c) northern side of the causeway and the (d) dry salt pan in 2006.

6.4.1.3 Electrical conductivity, moisture and organic content

Electrical conductivity was not significantly different ($p > 0.05$, $n = 30$) for the two sites on the northern side of the causeway (192.13 ± 2.66 $mS.cm^{-1}$) (Figure 6.5a). The sediment conductivity at Site 4 was significantly lower than all the other sites ($p < 0.05$, $n = 30$). There were no significant differences when comparing the mean for the tails from the north and south ($p > 0.05$, $n = 30$). Site C1 and C2 were significantly lower than the tails ($p < 0.05$, $n = 34$) (Figure 6.4b).

The moisture content in the tail located at Sites 5 and 6 were significantly higher than Sites 1 - 4 ($p < 0.05$, $n = 60$) (Figure 6.5a). There was a significant difference in moisture content in the northern tails with Site 1 significantly lower than Site 2 ($p < 0.05$, $n = 60$). Site 3 had significantly lower moisture than the rest of the southern sites (4 - 6) ($p < 0.05$, $n = 60$). In the comparison of the means (Figure 6.5b), the moisture content of Site C1 was significantly higher than that the tails ($p < 0.05$, $n= 34$), but not significantly different to the rest of the sites ($p > 0.05$, $n = 34$).

The organic content (Figure 6.5a) of the tails was fairy uniform and there was no significant difference between the various sites (1 - 6) ($p > 0.05$, $n = 60$). Site C1 (Figure 6.5b) was significantly higher than the tails ($p < 0.05$, $n = 34$), however not significant different from Site C3 ($p > 0.05$, $n = 34$) (Figure 6.5b).

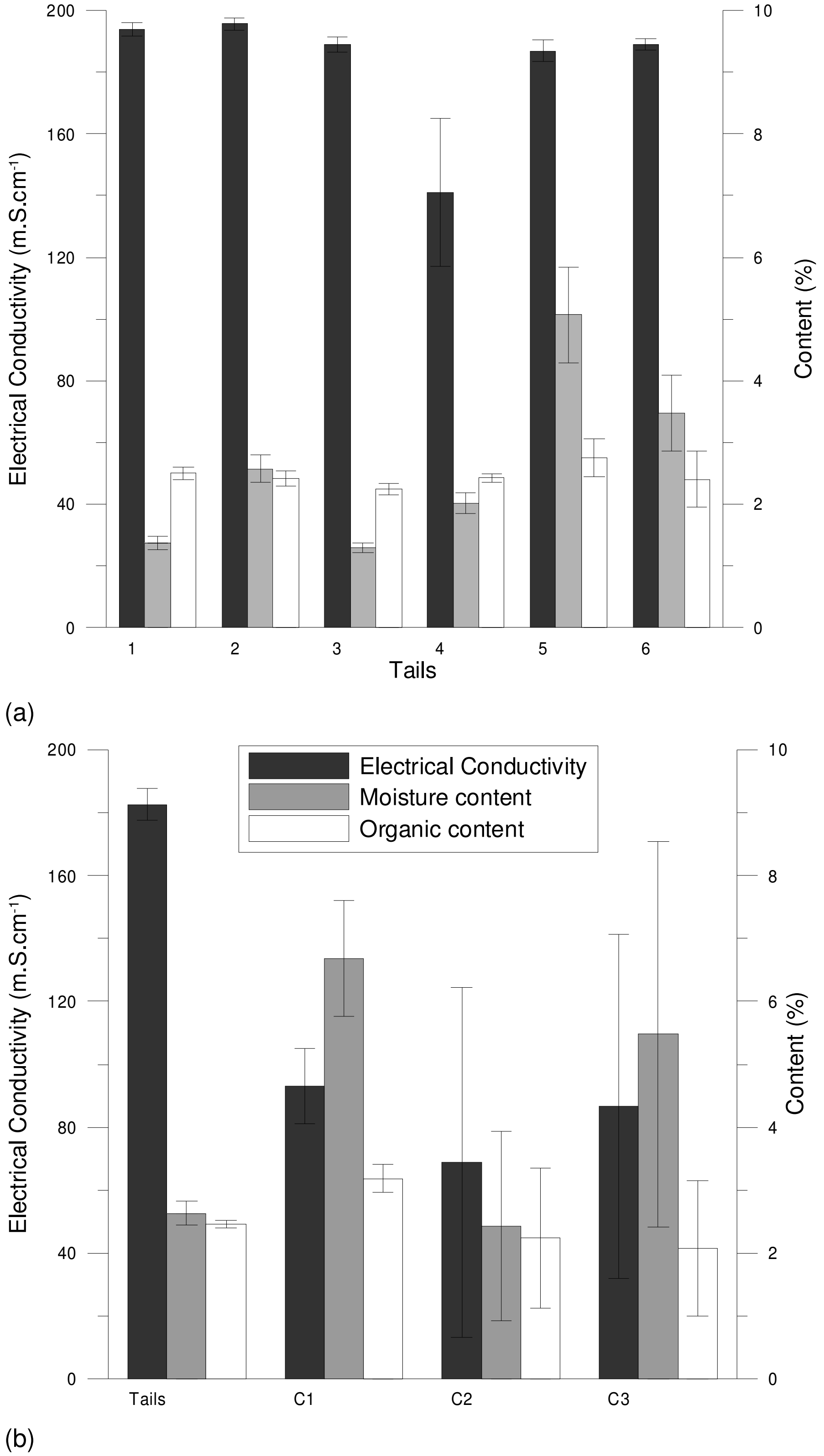


Figure 6.5 Sediment electrical conductivity, moisture and organic content of the (a) tails in 2005, compared to (b) Site C.

6.4.2 Sediment accumulation on the plants

Wind blown sediment accumulation was significantly higher ($p < 0.05$, $n = 78$) on the vegetation at the edge (0.26 ± 0.03 g.m^{-2}, $n = 39$) compared to the center of the vegetation (0.009 ± 0.002 g.m^{-2}, $n = 39$) (Figure 6.6). The cover / health of the vegetation (Figure 6.5) was significantly ($p < 0.05$, $n = 26$) lower at the edge (63 ± 5.48 %, $n = 13$) compared to the center (83.46 ± 4.18 %, $n = 13$).

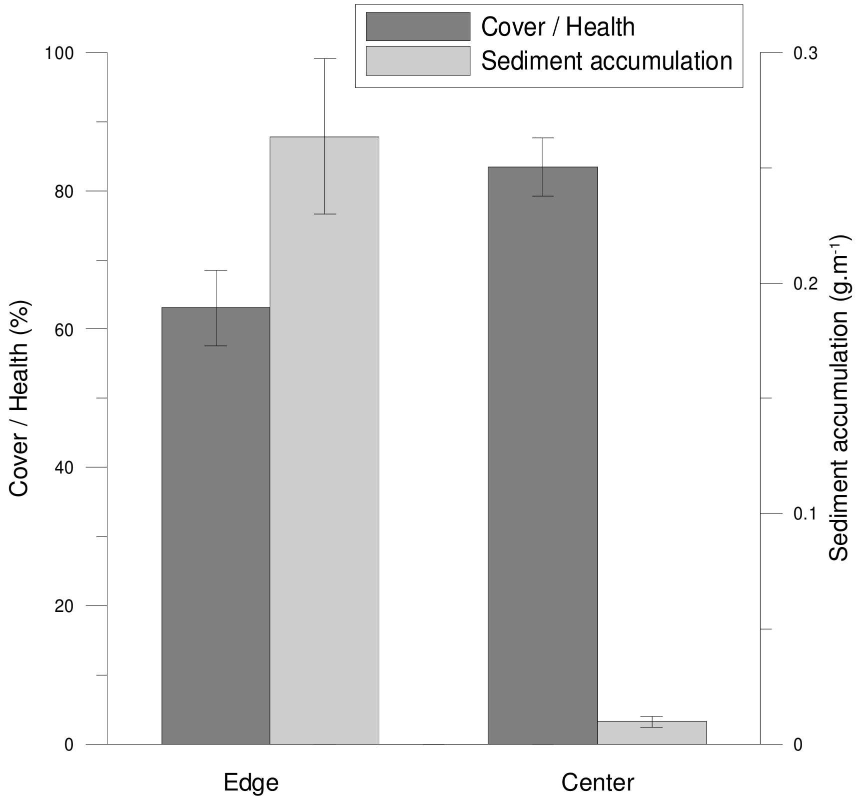


Figure 6.6 Wind blown sediment accumulation on the vegetation in Site C. (bars = ± SE)

6.4.3 Sediment accumulation at Site C

The profile of Site C had changed over a period of two years and there was a 0.5 m increase in elevation throughout the site (Figure 6.7). The vegetation extended to 150 m from the river channel. There was bare ground from 150 m to 240 m.

The edge of the vegetation moved 24.5 ± 3.7 m from 2000 to 2005, while 6.8 ± 0.8 m was lost from 2005 to 2006 (Figure 6.8).

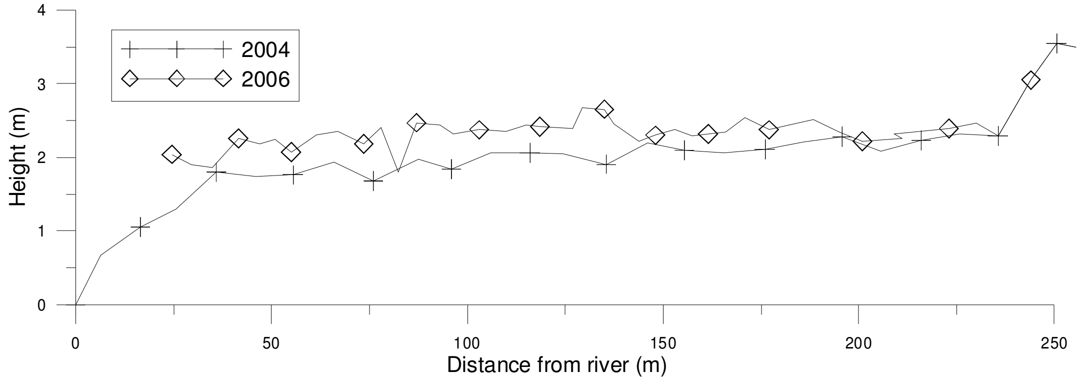


Figure 6.7 Profile of Site C in 2004 and 2006

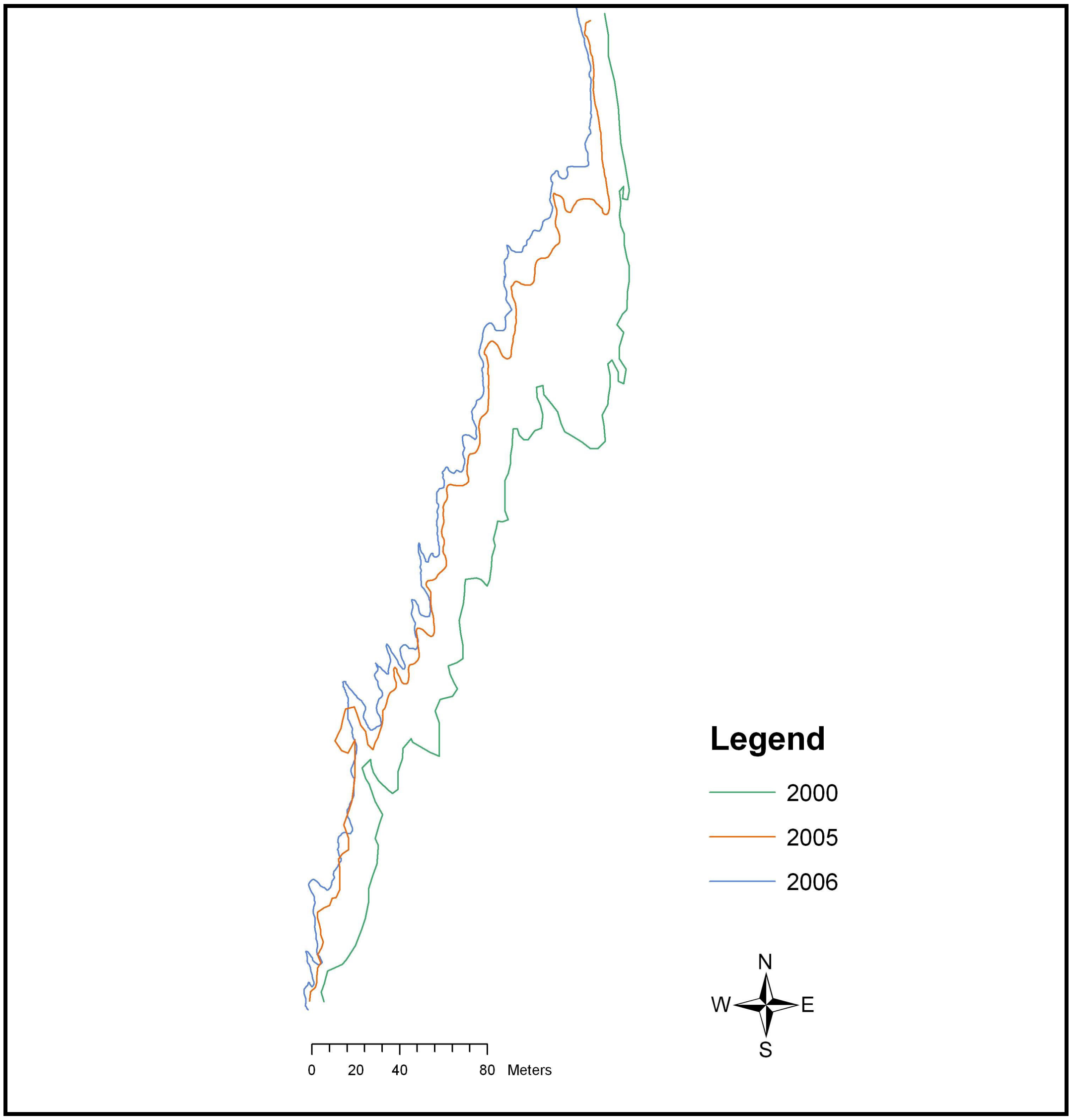


Figure 6.8 Movement of the boundary of the vegetation in Site C indicating a loss of *Sarcocornia pillansii*.

6.4.4 Wind barrier experiment

The effectiveness of the wind barrier was evident in the accumulation of sediment around the structures (Plate 20). However due to the rainfall in 2006, accurate determination of the sediment accumulation was not possible because the profiles had been influenced by water. The only statement that can be made is that there was more sediment around the structures than the surrounding environment and compared to the previous year as the stakes on the windward and leeward sides were completely covered in all experiments. The 0.5 m experiments had more extensive fore and aft dunes compared to the 1 m experiments. There appeared to be no difference in sand formation for the different grades of shade cloth used.

(a) (b)

Plate 20 Shade cloth experiments were set up on site to assess the efficiency in the reduction of wind blown sediment (a) 2005 and (b) 2006.

6.5 Discussion

Many studies worldwide have noted the correlation between dry conditions, consequent reduction in vegetation cover and increased frequency of dust deposition downwind (e.g. Goudie, 1978; Young & Evans, 1986; McTainsh *et al.*, 1998; Prospero & Lamb, 2003). The seasonality and strength of surface winds influence the generation of dust (Chepil & Woodruff, 1957) and rates of dust deposition (Smith *et al.*, 1970; Goossens, 2001). Wind energy is high throughout the region and although threshold velocities near the study sites were not measured, the seasonally high wind speeds between August and January are easily capable of sediment transport.

The dominant wind direction was southerly and was strongest in the summer months when there is little rain and conditions are dry. The dry conditions facilitate the transport of sediment by wind (Reheis, 2006) from the southern side of the marsh to the northern side. This movement of sediment formed sediment tails on the leeward side of the vegetation which increased in size from the southern end of the marsh to the northern side (Plate 21). One of the reasons is due to the larger availability of sediment in the desertified marsh. The particle size of the tails were predominantly of the sand fraction (2 – 50 µm) and were similar throughout the marsh. This particle size (2 – 50 µm) that accumulated at Site C is common in wind blown sediment (McTainsh, *In Press*; Zhenghu *et al.*, 2007). The wind blown sediment contained no seed and some of its physico-chemical properties (e.g. electrical conductivity) differed from the depositional area (Site C).

The shallow groundwater with high surface salinities of the marsh enhanced the evaporative concentration of salts at the surface, killed the vegetation and increased the susceptibility to wind erosion in the desertified marsh (Reheis, 2006). Many studies have found that both alluvial plains and salt pans act as dust sources (e.g. Pye, 1987; Reheis & Kihl, 1995). X-ray diffraction was used in conjunction with particle size to determine the origin of the wind blown sediment. The dominance of muscovite and quartz in the sample are indicative of aeolian sediment deposits (Pye, 1987). The low counts of chlorite suggest that the origin of the sediment is not marine. The sediments of the Orange River Mouth contain an abundance of mica and quartz because of the catchment it drains. In the river itself the most common sediment is fine sand. The lagoon, located on the southern side of the marsh contained no sand, only silt and clay. The fine texture reflects a quiet depositional environment. This together with the similarity of the samples suggests that the sediment transported to Site C had its origins from the lagoon within the marsh. Wind blown sediment smothers vegetation and its physico-chemical properties of the imported sediment negatively affect the surrounding sediment.

Plate 21 The formation of wind blown sediment forming in the lee of the vegetation (tail).

Electrical conductivity of the tails was much higher than that of the surrounding marsh (Chapter 4) as well as the depositional environment at Site C. This was probably because the sediment source of the tails originated from the lower marsh (southern corner) where the surface sediments were more saline (> 80 mS.cm^{-1}). Salts are transported by wind and can be carried great distances (McTainsh & Strong, *In Press*; Reheis, 2006). The effect of the salinity on the plants was not measured, but salinity higher than 80 mS.cm^{-1} is above the tolerance range of *S. pillansii* (Bornman *et al.*, 2004b) see (Chapter 5).

Moisture content of the tails increased from the depositional area in the north to the lagoon in the south. The increase in moisture in the southern corner limits the sediment available for transport. Moisture (Weinan *et al.*, 1996) and surface salinity (Nickling & Ecclestone, 1981) reduces the ability of the wind to move sediment below a certain threshold. The threshold velocity needed for sediment particle movement by wind increases with increasing soil moisture (Azizov, 1977). Nickling (1983) found that 3 – 4 % is the critical moisture content of surface sediment and that sediment particles were not affected by surface tension if water was below this level. The ORM marsh had a surface moisture ranging from 5 – 11 % (Chapter 4). That means that the sediment moisture is above the threshold and there should be no wind blown sediment. Using the moisture of the soil in the saltmarsh is misleading because the wind blown sediment is normally restricted to the top few millimetres that dry out in the heat and wind. Every time the sediment is removed the next layer dries out and is transported. The deflation of the marsh surface did not occur overnight because of the high moisture and salinity values, but rather over many years The lagoon is a depositional environment for fine sediment and the drying of the lagoons increases loss of mineral organic fines from sediment (McTainsh & Strong, *In Press*), explaining the increase of organic matter in C1. The increase of organic matter in C1 may also be due to the inclusion of smothered vegetation in the sediment samples.

The lack of vegetation in the desertified marsh enhanced the effect of the wind allowing more sediment to be transported. A significantly higher amount of sediment accumulated at the edge of the vegetation stand in Site C compared to the centre. The edge of the vegetation formed a buffer from the wind for the rest of the vegetation. Insufficient vegetation results in the deflation and resuspension of fine particles from open areas which eventually accumulate at vegetated desert margins (Fearnehough *et al.*, 1998). Vegetation is one of the key factors affecting wind erosion (Zheng, 2006) and the presence of vegetation decreases wind speed (Bullock & Moy, 2004). However, the sediment accumulated faster than the plants could grow. The elevation profile at Site C indicated up to a 50 cm increase in accumulated sediment between 2004 and 2006. Even if the plants outgrow the sediment build up they do not survive because of the increase in depth to groundwater. In the long term, dust is bound to have adverse effects on health, growth, productivity and mortality of species (Nadioo & Naidoo, 2004). This was evident from the GIS analysis where large amounts of vegetation was lost in a short space of time. The effects of dust accumulation are greater at high salinity because of the combined stress of dust and salinity (Naidoo & Chirkoot, 2004).

Wind is important in the marsh as it probably aids pollination and seed dispersal, however because large areas of the marsh are unvegetated the sediment load is higher than natural and therefore more sediment is available for transportation. Adult vegetation has been lost which is detrimental to the marsh as it will impact on seed production. Intervention is necessary to ensure the survival of the remaining vegetation. It is expected that by re-establishing vegetation within the degraded open marsh area the amount of wind blown sediment deposited in the northern areas will decrease.

The shade cloth caused large amounts of sediment to be trapped on both the windward and leeward side of the barrier. However this method which is commonly used as a sediment stabilizing mechanism is not recommended as it will alter the natural topography of the marsh with secondary impacts. Elevation is important in saltmarsh communities as it determines the zonation (Bertness & Ellison, 1987; Broome *et al*., 1988; Flynn *et al*., 1995; Brewer *et al*., 1998; Houle & Belleau, 2000; Silvestri, *et al*., 2005) and depth to the water table for the groundwater dependant floodplain saltmarsh (Bornman, 2002; *et al*., 2004b).

Conclusion

The wind at the ORM transported sediment, mostly of the sand fraction, which was deposited at Site C. X-ray diffraction showed that the sediment had its origin within the marsh and that the mine dumps were not the source of dust on the marsh itself. The sediment moved through the marsh as tails that differed physico-chemically from Site C. Sediment electrical conductivity was higher than the tolerance limit for *S. pillansii*. This together with the increased depth to groundwater caused by deposition of sediment lead to a loss of already stressed vegetation. It was shown that sediment can accumulate as much as 50 cm in two years (2004 – 2006). This was confirmed by the effectiveness of the shade cloth as a wind barrier where sediment deposition was high. However, windbreaks such as shade cloth are not recommended because this will alter the geomorphology of the marsh. The presence of vegetation promoted deposition of sediment which was shown to collect on the edge of monospecific *S. pillansii* stands.

Ongoing sediment transport poses a threat to the natural rehabilitation of the ORM. Freshwater needs to be introduced to the marsh so that vegetation can re-establish within the degraded open marsh. Once vegetated, the amount of wind blow sediment deposited in the northern areas will decrease.

Chapter 7: Conclusions and Recommendations

7.1 Conclusions

The potential for natural rehabilitation at the Orange River Estuary depends on various physical factors. These include the sediment characteristics, the effect of water on the sediment, the life cycle of the dominant vegetation and the effect of wind blown sediment. The reduction of freshwater entering the marsh has caused the groundwater to become hypersaline. The groundwater ultimately controls the edaphic conditions and because of the hypersalinity of the groundwater the success of germination and seedling establishment was low (Chapter 5). In 2005, Site B had a high density of *Sarcocornia pillansii* seedlings because of favourable sediment conditions, i.e. low salinity and sand dominance that limited the effect of the saline groundwater on the surface sediments. Adult *S. pillansii* vegetation has been lost at Site C as a result of hypersaline conditions and wind blown sediment that smothers the plants (Chapter 4 & 6). Rainfall together with flooding has the potential to reduce the hypersalinity of the groundwater (Bornman, 2004b). However, rainfall alone is insufficient in lowering the surface sediment salinity to a concentration that would allow the natural recruitment of *S. pillansii*.

Reduction of the wind blown sediment at Site C would relieve some of the stress on the plants, however the depth to groundwater is higher than the tolerance range of *S. pillansii* and freshwater flow into this area is therefore necessary. The use of windbreaks to reduce wind blown sediment are not recommended as this will alter the geomorphology of the marsh creating further problems with surface water movement and depth to the water table.

The flood in 2006 and above average rainfall significantly reduced salinity throughout the marsh. This created favourable conditions for seedling germination. However in the open unvegetated areas the salinity fluctuated according to the tide and the survival of seedlings was expected to be low (Chapter 5). If a low salinity is not maintained the marsh may slowly revert back to hypersaline conditions. Microhabitats in the saltmarsh are therefore very important in this semi-arid environment. The microhabitat created by the adult vegetation allowed seedlings to establish because they provided a stable environment. Driftlines were also a suitable habitat for seedling growth but survival would depend on the frequency of freshwater entering the marsh through rainfall and flooding. Tidal action was also important in the marsh as it maintained favourable conditions in the microhabitats. The intertidal species grew and flourished in areas influenced by tidal exchange. This was in contrast to the extreme salinities experienced by the supratidal and floodplain saltmarsh species. Fresher water entering the marsh through tidal and freshwater channels is essential as it reduces hypersaline conditions which is probably the greatest factor limiting seed germination and seedling establishment.

The potential of the Orange River Estuary floodplain to be rehabilitated back to a previous vegetated state will take time as only 2 ha of intertidal vegetation has returned naturally since the last breakthrough in 1997. Previously the saltmarsh recovered naturally in 7 years (1979 to 1986) after flooding. Flooding helps to flush the salts from the marsh and further breakdown of the causeway by the Working for Wetlands programme may speed up the process. The seeds of both species studied are hydrochorous further highlighting the importance of water entering the marsh for natural rehabilitation (Vogt *et al.*, 2004). From a management perspective the freshwater coming down the Orange River is important for the functioning of the remaining vegetation.

This study has shown that the Orange River Mouth saltmarsh has adequate seed for rehabilitation if the sediment and groundwater characteristics are suitable for seed germination, seedling growth and adult survival.

7.2 Recommendations

Recommendations in terms of flow release from impoundments up river are that periodic high releases should be allowed for. These releases would inundate the marsh, leach salts from the sediment and allow for seed to deposit in new areas where they could germinate and establish. Seeds are produced in winter, but only drop in summer and therefore a flood in February as occurred in 2006 is best. Two concurrent years of freshwater would probably be needed to ensure seedling germination and survival.

At times of low flow the mouth should be artificially closed for a short period. This would cause backflooding which allows water to enter areas of the marsh that are elevated above the normal spring tidal reach. The water would leach the surface salts from the sediment. The mouth should remain closed for no longer than necessary to flush the entire desertified marsh and all the standing water must drain thereby exporting salt from the floodplain. The Working for Wetlands project is beneficial to the rehabilitation of the marsh due to the breaches in the causeway allowing for drainage of the marsh from backflooding events. Monitoring should be implemented to assess the overall success of this programme.

Recommendations for further study include investigations of the persistence of seedlings in the marsh. Seedlings are a vital link in the life cycle of the vegetation and the failure of seedling establishment exerts pressure on the extant vegetation which is currently under stress. The survival success of both the seedlings and adults depends on the availability of moisture and low salinity. Flooding was shown to be important in the marsh, however the persistence of the conditions after a flooding event are unknown. If the microhabitats created by the flood and maintained by the tide last for the dry season then rehabilitation potential is higher than if these habitats dry out. The lack of microhabitats in 2005 suggests that these habitats do not persist but more investigation is required.

References

ADAMS, J.B. 1994. The importance of freshwater to the survival of estuarine plants. Unpublished PhD. thesis. University of Port Elizabeth. 177pp.

ADAMS, J.B. and G.C. BATE. 1994. The effect of salinity and inundation on the estuarine macrophyte *Sarcocornia perennis* (Mill.) A.J. Scott. *Aquatic Botany* 47: 341-348.

ADAMS, J.B. and M.M.B. TALBOT. 1992. The influence of river impoundment on the estuarine seagrass *Zostera capensis* Setchell. *Botanica Marina* 35: 69-75.

ADAMS, J.B., KNOOP, W.C. and G.C. BATE. 1992. The distribution of estuarine macrophytes in relation to freshwater. *Botanica Marina* 35: 215-226.

ADAMS, J.B., BATE, G.C. and M. O'CALLAGHAN. 1999. Primary Producers. In: Estuaries of South Africa. ALLANSON, B. and D. BAIRD. (eds.). Cambridge University Press. 340 pp.

AGUIAR, M.R. and O.E. SALA. 1997. Seed distribution constrains the dynamics of the Patagonian steppe. *Ecology* 78: 93-100.

AGUIAR, M.R. and O.E. SALA. 1999. Patch structure, dynamics and implications for the functioning of arid ecosystems. *Tree 14 (7): 273-277.*

ALEXANDER, H.D. and K.H. DUNTON. 2002. Freshwater inundation effects on emergent vegetation of a hypersaline salt marsh. *Estuaries* 25 (6B): 1426-1435.

ÁLVAREZ-ROGEL, J., ALCARAZ, F. and R. ORTIZ. 2000. Soil and moisture gradients and plant zonation in Mediterranean salt marshes of Southeast Spain. *Wetlands* 20: 357-372.

ÁLVAREZ-ROGEL, J., ORTIZ, R. and F. ALCARAZ. 2001. Edaphic characterisation and soil ionic composition influencing plant zonation in a semiarid Mediterranean salt marsh. *Geoderma* 99: 81-98.

ÁLVAREZ-ROGEL, J., CARRASCO, L., MARÍN, C.M. and J.J. MARTÍNEZ-SÁNCHEZ. *In Press*. Soils of a dune coastal salt marsh system in relation to groundwater micro-topography and vegetation under a semiarid Mediterranean climate in SE Spain.

AMACHER, M.C. and K.P. O'NEILL. 2004. Assessing soil compaction on forest inventory & analysis phase: 3 field plots using a pocket penetrometer. Research Paper RMRS-RP-46WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 7pp.

ANDERSON, M.D., KOLBERG, H. ANDERSON, P.C., DINI, J. and A. ABRAHAMS. 2003. Waterbird populations at the Orange River mouth from 1980-2001: a re-assessment of its RAMSAR status. *Ostrich* (3 & 4): 159-172.

AZIZ, I., and M.A. KAHN. 1996. Seed Bank dynamics of a semi-arid coastal shrub community in Pakistan. *Journal of Arid Environments* 34: 81-87.

AZIZ, I., and M.A. KAHN. 2000. Physiological adaptations of *Avicennia marina* to seawater concentrations in the Indus delta, Pakistan. *Pakistan Journal of Botany* 32 (1): 151-169.

AZIZ, I. and M.A. KAHN. 2001. Experimental assessment of salinity tolerance of *Ceriops tagal* seedlings and saplings from the Indus delta, Pakistan. *Aquatic Botany* 70: 259-268.

AZIZ, I. and M.A. KAHN. 2003. Proline and water status of some desert shrubs before and after rains. *Pakistan Journal of Botany* 35 (5): 902-906.

AZIZOV, M.T. 1977. Influence of soil moisture on the resistance of soil to wind erosion. *Soviet Soil Science* 1: 105-108.

BAI, J., OUYANG, H., DENG, W., ZHU, Y., ZHANG, X. and Q. WANG. 2005. Spatial distribution characteristics of organic matter and total nitrogen of marsh soils in river marginal wetlands. *Geoderma* 124: 181-192.

BAKER, H.G. 1989. Some aspects of the natural history of seed banks. In: LECK, M.A., PARKER, V.T. and R.L. SIMPSON (Ed) Ecology of Soil Seed Banks. London. Academic Press. pp 5-19.

BAKKER, J.P., DIJKSTRA, M. and P.T. RUSSCHEN. 1985. Dispersal, germination and early establishment of halophytes and glycophytes on a grazed and abandoned salt marsh gradient. *New Phytologist* 101: 291-308.

BALDWIN, A.H., MCKEE, K.L. and I.A. MENDELSSOHN. 1996. The influence of vegetation, salinity, and inundation on seed banks of oligohaline coastal marshes. *American Journal of Botany* 83 (4): 470-479.

BALDWIN, A.H. and I.A. MENDELSSOHN. 1998. Effects of salinity and water level on coastal marshes and experimental test of disturbance as a catalyst for vegetation change. *Aquatic Botany* 61: 255-268.

BALDWIN, A.H., EGNOTOVICH, M.S. and E. CLARKE. 2001. Hydrologic change and vegetation of tidal freshwater marshes: field, greenhouse, and seed-bank experiments. *The Society of Wetland Scientists* 21 (4): 519-531.

BARRETT-LENNARD, E.G. 2002. Restoration of saline land through revegetation. *Agricultural Water Management* 53: 213-226.

BASKIN, C.C. and J.M. BASKIN. 1989. Physiology of dormancy and germination in relation to seed bank ecology. In: LECK, M.A., PARKER, V.T. and R.L. SIMPSON. (Eds.), Ecology of Soil Seed Banks. Academic Press. San Diego. pp 53-66.

BASKIN, C.C., BASKIN, J.M. and P.L. CHESSON. 1993. Annual seed dormancy cycles in two desert winter annuals. *Journal of Ecology* 81: 551-556.

BEARE, P.A. and J.B. ZEDLER. 1987. Cattail invasion and persistence in a coastal salt marsh: The role of salinity reduction. *Estuaries* 10: 165-170.

BEEFTINK, W.G. 1985. Vegetation study as a generator for population biological and physiological research on salt marshes. *Vegetatio* 62: 469-486.

BENOIT, D.L., KENKEL, N.C. and P.B. CAVERS. 1989. Factors influencing the precision of soil seed bank estimates. *Canadian Journal of Botany* 67: 2833-2840.

BERTNESS, M. and A.M. ELLISON. 1987. Determinants of pattern in a New England salt marsh community. *Ecological Monographs* 57: 129-147.

BERTNESS, M.D., EWANCHUK, P. and B.R. SILLIMAN. 2002. Anthropogenic modification of New England salt marsh landscapes. *Proceedings of the National Academy of Science* 99 (3):1395-1398.

BERTNESS, M., SILLIMAN, B.R. and R. JEFFERIES. 2004. Salt marshes under siege. *American Scientist* 92: 54-61.

BIGWOOD, D.W. and D.W. INOUYE 1998. Spatial pattern analysis of seed banks: an improved method and optimised sampling. *Ecology* 69 (2): 497-507.

BLACK, C.B. 1965. Methods of Soil Analysis. American Society of Agronomy, Inc., Madison, Wisconsin. 1000 pp.

BLISS, S.A. and P.H. ZEDLER. 1998. The germination process in vernal pools: sensitivity to environmental conditions and effects on community structure. *Oecologia* 113: 67-73.

BORNMAN, T.G. 2002. Freshwater requirements of supratidal and floodplain salt marsh on the west coast, South Africa. Unpublished PhD. thesis. University of Port Elizabeth. 243 pp.

BORNMAN. T.G., ADAMS, J.B. and G.C. BATE. 2004a. The influence of floodplain geohydrology on the distribution of *Sarcocornia pillansii* in the Olifants Estuary on the west coast, South Africa. *Journal of Arid Environments* 56: 603-625.

BORNMAN, T.G., ADAMS, J.B. and C. BEZUIDENHOUT. 2004b. Adaptations of salt marsh to semi-arid environments and management implications for the Orange River mouth. *Transactions of the Royal Society of South Africa* 59 (2):85-91.

BORNMAN. T.G., ADAMS, J.B., and C. BEZUIDENHOUT. 2005. Present status of the Orange River mouth wetland and potential for rehabilitation. IECM Report No. 43. 43 pp.

BOSSUYT, B., STICHELMANS, E. and M. HOFFMANN. 2005. The importance of seed bank knowledge for the restoration of coastal plant communities – a case study of salt marshes and dune slacks at the Belgian coast. In: HERRIER, J.L., MEES, J., SALMAN, A., SEYS, J., VAN NIEUWENHUYSE, H. and I. DOBBELAERE (eds.). Proceedings 'Dunes and Estuaries 2005' – International Conference on Nature Restoration Practices in European Coastal habitats, Koksijde, Belgium, 19-23 September 2005 VLIZ Special Publication 19: 685pp.

BOUZILLÉ, J.B., KERNÉIS, E. and B. TOUZARD. 2001. Vegetation and ecological gradients in abandoned salt pans in western France. *Journal of Vegetation Science* 12: 269-278.

BREWER, J.S., LEVINE, J.M. and M.D. BERTNESS. 1998. Interactive effects of elevation and burial with wrack on plant community structure in some Rhode Island salt marshes. *Journal of Ecology* 86: 125-136.

BRIGGS, D. 1977. Sources and Methods in Goegraphy. Butterworths, London. 123 pp.

BROCK, M.A., THEODORE, K. and L. O'DONNELL. 1994. Seed-bank methods for Australian wetlands. *Australian Journal of Marine and Freshwater Resources* 45: 483-493.

BROCK, M.A. and K.H. ROGERS. 1998. The regeneration potential of the seed bank of an ephemeral floodplain in South Africa. *Aquatic Botany* 61: 123-135.

BROOME, S.W., SENECA, E.D. and W.W. WOODHOUSE, Jr. 1988. Tidal salt marsh restoration. *Aquatic Botany* 32: 1-22.

BULLOCK, J.M. and I.L. MOY. 2004. Plants as seed traps: inter-specific interference with dispersal. *Acta Oecologica* 25: 35-41.

BURCHILL, C.A. and N.C. KENKEL. 1990. Vegetation-environment relationships of an inland boreal salt pan. *Canadian Journal of Botany* 69: 722-732.

CALLAWAY, R.M. 1994. Facilitative and interfering effects of *Arthrocnemum subterminale* on winter annuals. *Ecology* 75: 681-686.

CALLAWAY, R.M., JONES, S.J., FERREN, Jr., W.R. and A. PARIKH. 1990. Ecology of a Mediterranean-climate estuarine wetland at Carpinteria, California: Plant distribution and soil salinity in the upper marsh. *Canadian Journal of Botany* 68: 1139-1146.

CALLAWAY, R.M. and C.S. SUBRAW. 1994. Effects of variable precipitation on the structure and diversity of a California salt marsh community. *Journal of Vegetation Science* 5: 433-438.

CANNELL, R.Q. 1977. Soil aeration and compaction in relation to root growth and management. *Applied Biology* 2: 1-86.

CAPON, S.J. and M.A. BROCK. 2006. Flooding, soil seed bank dynamics and vegetation resilience of a hydrologically variable desert floodplain. *Freshwater Biology* 51: 206-223.

CARAVACA, F., LAX, A. and J. ALBALADEJO. 1999. Organic matter, nutrient contents and cation exchange capacity in fine fractions from semiarid calcareous soils. *Geoderma* 93: 161-176.

CARTER, R.W.G. 1988. Coastal Environments. An Introduction to the Physical, Ecological and Cultural Systems of Coastlines. Academic Press. New York. 617pp.

CASANOVA, M.T. and M.A. BROCK. 1990. Charophyte germination and establishment from the seed bank of an Australian temporary lake. *Aquatic Botany* 36 (33): 247-254.

CELLOT, B., MOUILLOT, F. and C.P. HENRY. 1998. Flood drift and propagule bank of aquatic macrophytes in a riverine wetland. *Journal of Vegetation Science* 9 (5): 631-640.

CHEPIL, W.S. and N.P. WOODRUFF. 1957. Sedimentary characteristics of dust storms. II. Visibility and dust concentration. *American Journal of Science* 255: 104-114.

CHEPIL, W.S. 1945. Dynamics of wind erosion. IV. The translocating and abrasive action of the wind. *Soil Science* 61: 167-177.

CISNEROS, J.M., CANTERO, J.J. and A. CANTERO. 1999. Vegetation, soil hydrophysical properties, and grazing relationships in saline-sodic soils of central Argentina. *Canadian Journal of Soil Science* 79: 399-409.

CLEAVERING, O.A. 1995. Germination and seedling emergence of *Scirpus lacustris* L. and *Scirpus maritimus* L. with special reference to the restoration of wetlands. *Aquatic Botany* 50: 63-78.

COOPER, A. 1982. The effects of salinity and waterlogging on the growth and cation uptake of salt marsh plants. *New Phytologist* 90: 263-275.

COSTA, C.S.B., MARANGONI, J.C. and A.M.G. AZEVEDO, 2003. Plant zonation in irregularly flooded salt marshes: relative importance of stress tolerance and biological interactions. *Journal of Ecology* 91: 951-965.

COWAN, G.I. 1995. South Africa and the Ramsar Convention. In: COWAN, G.I. (ed.) Wetlands of South Africa. South African Wetland Conservation Programme Series. Department of Environmental Affairs and Tourism, Pretoria. 292 pp.

CRONK, J.K. and M.S. FENNESSY. 2001. Wetland Plants: Biology and Ecology. Lewis Publishers. London. 462 pp.

CSIR Report EMA - C 91165. 1991. Environmental Rehabilitation: Orange River Saltmarshes. Council for Scientific and Industrial Research, Stellenbosch. 54 pp.

CSIR Report ENV/S-C 97131. 1997. Orange River Mouth: Saltmarsh Rehabilitation Project. Council for Scientific and Industrial Research, Pretoria. 25 pp.

CUNNINGHAM, G.L., SYVERTSEN, J.P., REYNOLDS, J.F. and J. M. WILLSON. 1981. Some effects of soil-moisture availability on above-ground production and reproductive allocation in Larrea tridentata (DC) Cov. *Oecologia* 40 (2): 113-123.

DE LEEUW, J., VAN DEN DOOL, A., DE MUNCK, W., NIEWENHUIZE, J. and W.G. BEEFTINK. 1991. Factors influencing the soil salinity regime along an intertidal gradient. *Estuarine, Coastal and Shelf Science* 32: 87-97.

DE LEEUW, J., DE MUNCK, W., OLFF, H. and J.P. BAKKER. 1993. Does zonation reflect succession of salt-marsh vegetation? A comparison of an estuarine and a coastal island marsh in The Netherlands. *Acta Botanica Neerlandica* 42: 435-445.

DEAN, W.E. Jr., 1974. Determination of carbonate and organic matter is calcareous sediments and sedimentary rocks by loss on ignition: Comparison with other methods. *Journal of Sedimentation and Petrology*. 44: 242-248.

DENSLOW, J.S. and L.L. BATTAGLIA. 2002. Stand competition and structure across a changing hydrological gradient: Jean Lafitte National Park, Louisiana, USA. *Wetlands* 22: 738-752.

DE VILLIERS, A.J., VAN ROOYEN, M.W. and G.K. THERON. 1994. Comparison of two methods for estimating the size of the viable seed bank of two plant communities in the Strandveld of the west coast, South Africa. *South African Journal of Botany* 60(1): 81-84

DWA (Department of Water Affairs) 1990. Orange River Ecology. Assessment of Environmental Water Requirements for the Orange River Mouth. DWA Report No. V/D400/01/E001. Compiled by Brunette Kruger Stofberg Inc Consulting Engineers.

EL-DEMERDASH, M.A., HEGAZYT, A.K. and A.M. ZILAY. 1995. Vegetation-soil relationships in Tihamah coastal plains of Jazan region, Saudi Arabia. *Journal of Arid Environments* 30: 161-174.

ESPINAR, J.L., GARCIA, L.V. and L. CLEMENTE. 2005. Seed storage conditions change the germination pattern of clonal growth plants in Mediterranean salt marshes. *American Journal of Botany* 92(7): 1094-1101.

FARNSWORTH, E.J. 2000. Ecological and evolutionary perspectives on the physiology of vivipary and seed dormancy in plants. *Annual Review of Ecology and Systematics* 31: 107-138.

FEARNEHOUGH, W., FULLEN, M.A., MITCHELL, D.J., TRUEMAN, I.C. and J. ZHANG. 1998. Aeolian deposition and its effects on soil and vegetation changes on stabilized desert dunes in northern China. *Geomorphology* 23: 171-182.

FENNER, M. 1985. Seed Ecology. Chapman and Hall, London. 214 pp.

FENNER, M. and K. THOMPSON. 2005. The Ecology of Seeds. Cambridge University Press. Cambridge, UK. 250 pp.

FLOWERS, T.J., TROKE, P.F. and A.R. YEO. 1977. The mechanism of salt tolerance in halophytes. *Annual Review of Plant Physiology* 28: 89-121.

FLOWERS, T.J., and A.R. YEO. 1986. Ion relations of plants under drought and salinity. *Australian Journal of Plant Physiology* 13: 75-91.

FLOWERS, T.J. 1985. Physiology of halophytes. *Plant and Soil* 89: 41-56.

FLYNN, K.M., MCKEE, K.L. and I.A. MENDELSSOHN. 1995. Recovery of freshwater marsh vegetation after a saltwater intrusion event. *Oecologia* 103: 63-72.

GALINATO, M.I. and A.G. VAN DER VALK. 1986. Seed germination traits of annuals and emergents recruited during drawdowns in the Delta Marsh, Manitoba, Canada. *Aquatic Botany* 26: 89-102.

GEE, G.W. and J.W. BAUDER. 1986. Particle-size analysis. In: Methods of soil analysis; Part 1 – Physical and mineralogical methods. Second edition. KLUTE, A. (eds.). Agronomy Number 9. Part 1. Madison, Wisconsin, USA. 383-411.

GILLETTE, D.A. and T.R. WALKER. 1977. Characteristics of airborne particles produced by wind erosion of sandy soil, High plains of West Texas. *Soil Science* 123: 97-110

GOMEZ-PLAZA, M., MARTINEZ-MENA, M., ALBALADEJO, J and V.M. CASTILLO. 2001. Factors regulating spatial distribution of soil water content in small semiarid catchments. *Journal of Hydrology* 253: 211-226.

GOODSON, J.M., GURNELL, A.M., ANGOLD, P.G. and I.P. MORRISSEY. 2001. Riparian seed banks: structure, process and implications for riparian management. *Progress in Physical Geography* 25 (3): 301-325.

GOOSSENS, D. 2001. The aeolian dust accumulation curve. *Earth Surface Processes and Landforms* 26: 1213–1219.

GOUDIE, A.S., 1978. Dust storms and their geomorphological implications. *Journal of Arid Environments* 1: 291–310.

GRANTZ, D.A., GARNER, J.H.B. and D.W. JOHNSON. 2003. Ecological effects of particulate matter. *Environment International* 29: 213-239.

GRAECEN, E.L. and R. SANDS. 1980. Compaction of forest soils: A review. *Australian Journal of Soil Research* 18: 163-189.

GREELEY, R. and J.D. IVERSEN. 1985. Wind as a geological process on Earth, Mars, Venus and Titan. Cambridge University Press. London. 333 pp.

GREENWAY, H. 1968. Growth stimulation by chloride concentrations in halophytes. *Israel Journal of Botany* 17: 169-177.

GREENWAY, H. 1973. Salinity, plant growth, and metabolism. *The Journal of the Australian Institute of Agricultural Science* 39: 24-34.

GRILLAS, P., GARCIA-MURILLO, P.,GEERTZ-HANSEN, O., MARBA, N., MONTES, C., DUARTE, C.M., TAN HAM, I. AND A. GROSSMANN. 1993. Submerged macrophyte seed bank in a Mediterranean temporary marsh: abundance and relationship with established vegetation. *Oecologia* 94: 1-6.

GRIME, J.P., MASON, G., CURTIS, A.V., RODMAN, J., BAND, S.R., MOWFORTH, M.A.G., NEAL, A.M. and S. SHAW. 1981. A comparative study of germination characteristics in a local flora. *Journal of Ecology* 69: 1017-1059.

GROSS, K.L. 1990. A comparison of methods for estimating seed numbers in the soil. *Journal of Ecology* 78: 1079-1093.

GUARDIA, R., GALLART, F. and J.M. NINOT. 2000. Soil seed bank and seedling dynamics in badlands of the upper Llobregat basin. *Catena* 40:189-202.

GUL, B. and D.J. WEBER. 2001. Seed bank dynamics in a Great Basin salt playa. *Journal of Arid Environments* 49: 785-794.

GUO, Q., RUNDEL, P.W. and D.W. GOODALL. 1998. Horizontal and vertical distribution of desert seed banks: patterns, causes, and implications. *Journal of Arid Environments* 38: 465-478.

HAAG, R.W. 1983. Emergence of seedlings of aquatic macrophytes from lake sediments. *Canadian Journal of Botany* 61: 148–156.

HAGEN, L. 1984. Soil aggregate abrasion by impacting sand and soil particles. *Transactions of the American Society of Agricultural Engineers* 27: 805-808.

HAW, P.M. 1984. Freshwater requirements of the Knysna Estuary. School of Environmental Studies, University of Cape Town, South Africa. Research report No. 55.

HEIRI, O., LOTTER, A.F., and G. LEMCKE. 2001. Loss on ignition method as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *Journal of Paleolimnology*. 25: 101-110.

HILLEL, D. 1971. Soils and water: Physical principles and processes. In: Physiological Ecology: A series of monographs, texts and treatises. KOZLOWSKI, T.T. (eds.). Academic Press Inc. New York. 288 pp.

HOPKINS, D.R. and V.T. PARKER. 1984. A study of the seed bank of a salt marsh in northern San Francisco Bay. *American Journal of Botany* 71 (3): 348-355.

HOULE, G. and A. BELLEAU. 2000. The effects of drought and waterlogging conditions on performance of an endemic annual plant, *Aster laurentianus. Canadian Journal of Botany* 78: 40-46.

HUGES, C.E., BINNING, P. and G.R. WILLGOOSE. 1998. Characterisation of the hydrology of an estuarine wetland. *Journal of Hydrology* 211: 34-49.

HUTCHINGS, M.J. and P.J. RUSSELL. 1989. The seed regeneration dynamics of an emergent salt marsh. *Journal of Ecology* 77: 615-637.

IGNACIUK, R. and J.A. LEE. 1980. The germination of four annual strandline species. *New Phytologist* 84: 581 – 591.

JACKSON, M.D. and M.C. DREW. 1984. Effects of flooding on herbaceous plants. In: Flooding and Plant Growth. (KOZLOWSKI, T.T., Ed). Academic Press, Inc. New York, London. 112pp.

JACOBSON, P.J., JACOBSON, K.M., ANGERMEIER, P.L. and D.S. CHERRY. 2000. Hydrologic influences on soil properties along ephemeral rivers in the Namib Desert. *Journal of Arid Environments* 45: 21-34.

JERLING, L. 1984. Composition and viability of the seed bank along a successional gradient on a Baltic sea-shore meadow. *Holarctic Ecology* 6: 150-156.

JOLLY, I.D., WALKER, G.R. and P.J. THORBURN. 1993. Salt accumulation in semi-arid floodplain soils with implications for forest health. *Journal of Hydrology* 150: 589-614.

KAHN, M.A. 1993. Relationship of seed bank to plant distribution in saline arid communities. *Pakistan Journal of Botany* 25: 73-82.

KAHN, M.A. and B. GUL. 1998. High salt tolerance in germinating dimorphic seeds of *Arthrocnemum indicum. International Journal of Plant Science* 159 (5): 826-832.

KAHN, M.A., UNGAR, I.A. and A.M. SHOWALTER. 2000. Effects of salinity on growth, water status, and ion content of a leaf succulent perennial halophyte, *Suaeda fruticosa* (L.) Forssk. *Journal of Arid Environments* 45: 73-84.

KAHN, M.A., GUL, B. and D.J. WEBER. 2001. Influence of salinity and temperature on the germination of *Kochia scoparia. Wetland Ecology and Management* 9: 483-489.

KAHN, M.A. and B. GUL. 2006. Halophyte Seed Germination. In: KAHN, M.A and D.J. WEBER (eds.), Ecophysiology of High Salinity Tolerant Plants. Springer. Netherlands. 11-30

KEIFFER, C.H., MCCARTHY, B.C and I.A. UNGAR. 1994. Effect of salinity and waterlogging on growth and survival of *Salicornia europaea* L., an inland halophyte. *Ohio Journal of Science* 94 (3): 70-73.

KEMP, P.R. and G.L. CUNNINGHAM. 1981. Light, temperature and salinity effects on growth, leaf anatomy, and photosynthesis of *Distichlis spicata* (L.) Greene. *American Journal of Botany* 68: 507-516.

KINGSBURY, R.W., RADLOW, A., MUDIE, P.J., RUTHERFORD, J. and R RADLOW. 1976. Salt Stress responses in *Lasthenia glabrata*, a winter annual composite endemic to saline soils. *Canadian Journal of Botany* 54: 1377-1385.

KOCH, M.S., and K.R. REDDY. 1992. Distribution of soils and plant nutrients alsong a trophic gradient in the Florida Everglades. *Soil Science Society of American Processes Journal* 56: 1492-1499.

KRÜGER, H.R. and N. PEINEMANN. 1996. Coastal plain halophytes and their relation to soil ionic composition. *Vegetatio* 122: 143-150.

KUHN, N.L. and J.B. ZEDLER. 1997. Differential effects of salinity and soil saturation on native and exotic plants of coastal salt marsh. *Estuaries* 20: 391-403.

LECK, M.A. 1989. Wetland Seed Banks In LECK, M.A., PARKER V. T. and R.L. SIMPSON. (eds.), Ecology of Soil Seed Banks. San Diego, CA. Academic Press. 283-303 pp.

LECK, M.A. 2003. Seed-bank and vegetation development in a created tidal freshwater wetland on the Delaware River, Trenton, New Jersey, USA. *Wetlands* 23:2.

LECK, M.A. and M.A. BROCK. 2000. Ecological and evolutionary trends in wetlands: Evidence from seeds and seed banks in New South Wales, Australia and New Jersey, USA. *Plant Species Biology* 15 (2): 97–112.

LECK, M.A. and K.J. GRAVELINE. 1979. The seed bank of a freshwater tidal marsh. *American Journal of Botany* 66: 1006-1015.

LECK, M.A. and R.L. SIMPSON. 1987. Seed bank of a freshwater tidal wetland: turnover and relationship to vegetation change. *American Journal of Botany* 74 (3): 360-370.

LECK, M. A., and R.L. SIMPSON. 1994. Tidal freshwater wetland zonation: seed and seedling dynamics. *Aquatic Botany* 47 61–75.

LECK, M.A. and R.L. SIMPSON. 1995. Ten-year seed bank and vegetation dynamics of a tidal freshwater marsh. *American Journal of Botany* 82 (12): 1547-1557.

LIAN-YOU, L., SHANG-YU, G., PEI-JUN, S., XIAO-YAN, L. and D. ZHI-BAO. 2003. Wind tunnel measurements of adobe abrasion by blown sand: profile characteristics in relation to wind velocity and sand flux. *Journal of Arid Environments* 53: 351-363.

LINDIG-CISNEROS, R. and J.B. ZEDLER. 2002. Halophyte recruitment in a salt marsh restoration site. *Estuaries* 25 (6A): 1174-1183.

LITHAUER, P.A. 1989. A study of salt marsh zonation in an intertidal and supratidal saltmarsh in the Gamtoos River Estuary. Unpublished Honours Project, Department of Botany. University of Port Elizabeth. 78pp.

LUBKE, R. AND I. DE MOOR. 1998. Field guide to the Eastern and Southern Cape Coasts. Second Edition. LUBKE and DE MOOR. (eds.) University of Cape Town Press, Rondebosch. 559 pp.

MA, J.Y., REN, J., WANG, G. and F.H. CHEN. 2006. Influence of different microhabitats and stand age on viable soil seed banks of sand-stabilising species. *South African Journal of Botany* 72: 46-50.

MAJOR, J. and W. PYOTT. 1960. Buried viable seeds in two California bunchgrass sites and their bearing on the definitions of a flora. *Vegetation* 13: 253-282.

MCTAINSH, G.H., LYNCH, A.W. and E.K. TEWS. 1998. Climatic controls upon dust storm occurrence in eastern Australia. *Journal of Arid Environments* 39: 457–466.

MCTAINSH, G. and C. STRONG. *In Press*. The role of aeolian dust in ecosystems.

MERRIT, D.M. and E.E., WHOL. 2002. Processes governing hydrochory along rivers: Hydraulics, hydrology, and dispersal phenology. *Ecological Applications* 12:1071-1087.

MITSCH, W.J., WU, X., NAIRN, R.W., WEIHE, P.E., WANG, N. and R. DEAL. *In Press*. Creating and restoring wetlands. A whole ecosystem experiment in self design.

MORGAN, J.M. 1984. Osmoregulation and water stress in higher plants. *Annual Review of Plant Physiology* 35: 299-348.

NAIDOO, G. and D. CHIRKOOT. 2004. The effects of coal dust on photosynthetic performance of the mangrove *Avicennia marina* in Richards Bay. South Africa. *Environmental Pollution* 127: 359-366

NAIDOO, G. and J. KIFT. 2006. Responses of saltmarsh *Juncus kraussii* to salinity and waterlogging. *Aquatic Botany* 84: 217-225.

NAIDOO, G. and K. NAICKER. 1992. Seed germination in the coastal halophytes *Triglochin bulbosa* and *Triglochin striata. Aquatic Botany* 42: 217-229.

NAIDOO, G and Y. NAIDOO. 2004. Coal dust pollution effects on wetland tree species in Richards Bay, South Africa. *Wetlands Ecology and Management* 00: 1-7.

NAIDOO, G. and R. RUGHUNANAN. 1990. Salt tolerance in the succulent, coastal halophyte, S*arcocornia natalensis. Journal of Experimental Botany* 41 (225): 497-502.

NEFF, K.P. and A.H. BALDWIN. 2005. Seed dispersal into wetlands: techniques and results for a restored tidal freshwater marsh. *Wetlands* 25 (2): 392–404.

NEILL, C. 1993. Seasonal flooding, soil salinity and primary production in northern prairie marshes. *Oecologia* 95: 499-505.

NICKLING, W.G. 1983. Grain-size characteristics of sediments transported during dust storms. *Journal of Sedimentation and Petrology* 53: 1011-1024.

NICKLING, W.G. and M. ECCLESTONE. 1981. The effects of soluble salts on the threshold shear velocity of fine sand. *Sedimentology* 28: 505-510.

NICOL, J.M. and G.G GANF. 2002. Water regimes, seedling recruitment and establishment in three wetland plant species. *Marine and Freshwater Research* 51(4): 305 – 309.

NILSSON, C. and G. GRELSSON. 1990. The effects of litter displacement on riverbank vegetation. *Canadian Journal of Botany* 68: 735-741.

NOE, G.B. and J.B. ZEDLER. 2000. Differential effects of four abiotic factors on the germination of salt marsh annuals. *American Journal of Botany* 87: 1679-1692.

NOE, G.B. and J.B. ZEDLER. 2001. Spatio-temporal variation of salt marsh seedling establishment in relation to the abiotic and biotic environment. *Journal of Vegetation Science* 12: 61-74.

NYBAKKEN, J.W. 2001. Marine Biology: An Ecological Approach (5th ed). Benjamin Cummings. New York. 516pp.

O'CALLAGHAN, M. 1992. The ecology and identification of the Southern African Salicorniaceae (Chenopodiaceae*). South African Journal of Botany*. 58 (6): 430-439.

ONAINDIA, M and I. AMEZAGA. 1999. Natural regeneration in salt marshes of Northern Spain. *Annals Botanici Fennici* 36: 59-66.

ORTH, R.J., LUCKENBACH, M. and K.A. MOORE. 1994. Seed dispersal in a marine macrophyte: Implication for colonisation and restoration. *Ecology* 75 (7): 1927-1939.

PAN, D., BOUCHARD, A., LEGENDRE, P. and G. DOMON. 1998. Influence of edaphic factors on the spatial structure of inland halophytic communities: a case study in China. *Journal of Vegetation Science* 9: 797-804.

PARTRIDGE, T.R. and J.B. WILSON. 1987. Germination in relation to salinity in some plants of salt marshes in Otago, New Zealand. *New Zealand Journal of Botany* 25: 255-261.

PARRONDO, R.T., GOSSELINK, J.G. and C.D. HOPKINSON. 1978. Effect of salinity and drainage on the growth of three salt marsh grasses. *Botanical Gazette* 139 (1): 102-107.

PIERNIK, A. 2003. Inland halophilous vegetation as indicator of soil salinity. *Basic and Applied Ecology* 4: 525-536.

POIANI, K.A. and W.C. JOHNSON. 1989. The effect of hydroperiod on seed bank composition in semi-permanent prairie wetlands. *Canadian Journal of Botany* 67: 856-864.

PROSPERO, J.M. and P.J. LAMB. 2003. African droughts and dust transport to the Caribbean: climate change implications. *Science* 302: 1024–1027.

PYE, K. 1987. Aeolian Dust and Dust Deposits. Academic Press. London. 334pp.

Ramsar Information Document. [Internet].
Available from: http://www.ramsar.org/key_montreux_record.htm

RANWELL, D.S. 1972. The Ecology of Salt Marshes and Sand Dunes. Chapman and Hall. London. 258pp.

REDONDO, S., RUBIO-CASAL, A.E., CASTILLO, J.M., LUQUE, C.J., ALVAREZ, A.A., LUQUE, T. and M.E. FIGUEROA. 2004. Influences of salinity and light on germination of three S*arcocornia* taxa with contrasted habitats. *Aquatic Botany* 78: 255-264.

REHEIS, M.C. 2006. A 16-year record of eolian dust in Southern Nevada and California, USA: Controls on dust generation and accumulation. *Journal of Arid Environments* 67:487-520.

REHEIS, M.C. and R. KIHL. 1995. Dust deposition in southern Nevada and California, 1984–1989: Relations to climate, source area, and source lithology. *Journal of Geophysical Restoration* 100: 8893–8918.

REICHMAN, J. 1984. Spatial and temporal variation of seed distributions in Sonoran desert soils. *Journal of Biogeography* 11:1-11.

RIDD, P.V. and R. SAM, 1996. Profiling groundwater salt concentrations in mangrove swamps and tropical salt flats. *Estuarine, Coastal and Shelf Science* 43: 627-635.

RIEHL, E. and I.A. UNGAR. 1982. Growth and ion accumulation in *Salicornia europaea* under saline field conditions. *Oecologia* 54: 193-199.

ROBERTS, H.A. 1981. Seed banks in the soil: Advances in Applied Biology. Academic Press. Cambridge. 55 pp.

ROMAN, C.T., NIERING, W.A. and R.S. WARREN. 1984. Salt marsh vegetation change in response to tidal restriction. *Environmental Management* 2 (8): 141-150.

ROOZEN, A.J.M. and V. WESTHOF. 1985. A study on long-term salt marsh succession using permanent quadrats. In: Ecology of coastal vegetation. BEEFTINK, W.G., ROZEMA, J. AND A.H.L. HUISKES (eds.). *Vegetatio* 59 (60): 429-432.

ROSE, D.A. 1996. The dynamics of soil water following single surface wettings. *European Journal of Soil Science* 47: 21-31.

ROZEMA, J. 1975. The influence of salinity, inundation and temperature on the germination of some halophytes and non-halophytes. *Oecologia Plantarum* 10: 341-353.

ROZEMA, J. and B. BLOM. 1985. Effects of salinity and inundation on the growth of *Agrostis stolonifera* and *Juncus gerardii*. *Journal of Ecology* 65: 213-222.

RUBIO-CASAL, A.E., CASTILLO, J.M., LUQUE, C.J. and M.E. FIGUEROA. 2002. Influence of salinity on germination and seeds viability of two primary colonizers of Mediterranean salt pans. *Journal of Arid Environments* 53: 145-154.

SAMSON, D.A., PHILIPPI, T.E. and D.W. DAVIDSON. 1992. Granivory and competition as determinants of annual plant diversity in the Chihuanhuan Desert. *Oikos* 165: 61-80.

SCHULTZ, W. 1997. Primary dormancy and annual dormancy cycles in seed of six temperate wetland sedges. *Aquatic Botany* 59: 75-85.

SERFES, M.E. 1991. Determining the mean hydraulic gradient of ground water affected by tidal fluctuations. *Ground Water* 29 (4): 549-555.

SHAUKAT, S.S. and I.A. SIDDIQUI. 2004. Spatial pattern analysis of seeds of an arable soil seed bank and its relationship with above-ground vegetation in an arid region. *Journal of Arid Environments* 57: 311-327.

SHUMWAY, S.W. and M.D. BERTNESS. 1992. Salt stress limitation of seedling recruitment in the salt marsh plant community. *Oecologia* 92: 490-497.

SILVESTRI, S., DEFINA, A. and M. MARANI. 2005. Tidal regime, salinity and salt marsh plant zonation. *Estuarine Coastal and Shelf Science* 62: 119-130.

SIMENSTAD, C.A. and R.M. THOM. 1996. Assessing functional equivalency of habitat and food web support in a restored estuarine wetland. *Ecological Applications* 6: 38-56.

SIMPSON, R.L., LECK, M.A. and V.T. PARKER. 1989 Seed banks: General concepts and methodological issues. In: LECK, M.A., PARKER, V.T. and R.L. SIMPSON. (Ed). Ecology of Seed Banks. London. Academic Press. 3-8.

SINGER, M.J. 2006. Solids and Pores. In: SINGER, M.J. and D.N. MUNNS (eds.). Soils: An Introduction. New Jersey. Prentice Hall. 443pp.

SMITH, R.M., TWISS, P.C., KRAUSS, R.K. and M.J. BROWN. 1970. Dust deposition in relation to site, season, and climatic variables. Soil Science Society of America 34: 112–117.

SMITH, L.M., and J.A. KADLEC. 1985. The effects of disturbance on marsh seed banks. *Canadian Journal of Botany* 63: 2133-2137.

SNOW, A.A. and S.W. VINCE. 1984. Plant zonation in an Alaskan salt marsh. II. An experimental study of the role of edaphic conditions. *Journal of Ecology* 72: 669-684.

SPURGEON, J. 1998. The socio-economic costs and benefits of coastal habitat rehabilitation and creation. *Marine Pollution Bulletin* 37 (8-12): 373-382.

SQUIRES, L, and A.G. VAN DER VALK. 1992. Water-depth tolerances of the dominant emergent macrophytes of the Delta Marsh, Manitoba. *Canadian Journal of Botany* 70: 1860-1866.

STEWART, G.R. and J.A. LEE. 1974. The role of proline accumulation in halophytes. *Planta* 120: 279-289.

TEMPLETON, A.R. and D.A. LEVIN. 1979. Evolutionary Consequences of Seed Pools. *The American Naturalist* 114 (2): 232-249.

TER HEERT, G.N.J., SCHUTTER, A. and J.P. BAKKER. 1999. The effect of water supply on seed-bank analysis using the seedling-emergence method. *Functional Ecology* 13: 428-430.

The Non-Affiliated Soil Analyses Working Committee. 1990. Handbook of Standard Soil Testing Methods for Advisory Purposes. Soil Science Society of South Africa, Pretoria. 137 pp.

THOMPSON, K. and J.P. GRIME. 1979. Seasonal variation in the seed banks of herbaceous species in ten contrasting habitats. *Journal of Ecology* 34: 893-921.

THORBURN, P.J., WALKER, G.R. and P.H. WOODS. 1992. Comparison of diffuse discharge from shallow water tables in soils and salt flats. *Journal of Hydrology* 136: 253-274.

TÖLKEN, H.R. 1967. The species of *Arthrocnemum* and *Salicornia* (Chenopodiaceae) in Southern Africa. *Bothalia* 9: 255-307.

UNGAR, I.A. 1978. Halophyte seed germination. *Botanical Review* 44: 233-264.

UNGAR, I.A. 1991. Ecophysiology of Vascular Halophytes. CRC Press, Boca Raton, FL. 209 pp.

UNGAR, I.A., Benner, D.K. and D.C. McGraw. 1979. The distribution and growth of *Salicornia europaea* on an inland salt pan. *Ecology* 60 (2): 329-336.

UNGAR, I.A. and S.R.J. WOODELL. 1993. The relationship between the seed bank and species composition of plant communities in two British salt marshes. *Journal of Vegetation Science* 4: 531-536.

VANDERPLANK, H.J. 1999. Wildflowers of the Port Elizabeth Area. Gamtoos to Swartkops rivers (The coastal bush and Fynbos Region). Bluecliff Publishing, Hunters Retreat. 216 pp.

VAN DER TOORN, J. and H.J. TEN HOVE. 1982. On the ecology of *Cotula coronopifolia* L. and *Ranunculus sceleratus* L.II. – Experiment on germination, seed longevity, and seedling survival. *Acta Oecologica* 3 (17):409-418.

VAN DER VALK, A.G. 1981. Succession in wetlands: a Gleasonian approach. *Ecology* 62: 688-696.

VAN DER VALK, A.G. and C.B. DAVIS. 1976. The seed banks of prairie glacial marshes. *Canadian Journal of Botany* 54: 1832-1838.

VAN DER VALK, A.G. and C.B. DAVIS. 1978. The role of seed banks in the vegetation dynamics of prairie glacial marshes. *Ecology* 59: 322-335.

VAN DER VALK, A.G. and C.B. DAVIS. 1979. A reconstruction of the recent vegetational history of a prairie marsh, Eagle Lake, Iowa, from its seed bank. *Aquatic Botany* 6: 29-51

VAN DER VALK, A.G. and R.L. PEDERSON. 1989. Seed Banks and the Management and Restoration of Natural Vegetation. In: LECK, M.A., PARKER, V.T. and R.L. SIMPSON (Eds). Ecology of Soil Seed Banks. San Diego, CA. Academic Press. 329-344.

VAN DIGGELEN, J. 1991. Effects of Stress on Salt Marsh Halophytes. In: Ecological Responses to Environmental Stresses. ROZEMA, J. and J.A.C. VERKLEIJ (eds.). Kluwer Academic Publishers. 62-73.

VAN HEERDEN, P.D.R., KRUGER, G.H.J. and M. KILBOURNE LOUW. 2006. Dynamic responses of photosystem II in the Namib desert shrub, *Zygophyllum prismatocarpum*, during and after foliar deposition of limestone dust. *Environmental Pollution* 1-12.

VENTER, A. and M. VAN VEELEN. 1996. Refinement of the Instream Flow Requirements for the Orange River and Orange River Mouth. Draft. Prepared for the Department of Water Affairs and Forestry, Pretoria. Project No. P525407.

VERNBERG, F.J. 1993. Salt-marsh processes: A review. *Environmental Toxicology and Chemistry* 12: 1267-2195.

VINCENTE, O., BOSCAIU, M., NARANJO, M.A., ESTRELLES, E., BELLES, J.M. and P. SORIANO. 2004. Responses to salt stress in the halophyte *Plantago crassifolia* (Plantaginaceae). *Journal of Arid Environments* 58: 463-481.

VOGT, K., RASRAN, L. and K. JENSEN. 2004. Water-borne seed transport and seed deposition during flooding in a small river-valley in Northern Germany. *Flora* 199: 377-388.

VOGT, K., RASRAN, L. and K. JENSEN. 2006. Seed deposition in drift lines during an extreme flooding event – Evidence for hydrochorous dispersal? *Basic and Applied Ecology* 7: 422-432.

WARWICK, N.W.M. and M.A. BROCK. 2003. Plant reproduction in temporary wetlands: the effects of seasonal timing, depth, and duration of flooding. *Aquatic Botany* 77: 153-167.

WATTS, C.W., TOLHURST, T.J., BLACK, K.S. and A.P. WHITMORE. 2003. In situ measurements of erosion shear stress and geotechnical shear strength of the intertidal sediments of the experimental managed realignment scheme at Tollesbury, Essex, UK. *Estuarine, Coastal and Shelf Science* 58: 611- 620.

WEINAN, C., ZHIBAO, D., ZHENSHAN, L. and Y. ZUOTAO. 1996. Wind tunnel test of the influence of moisture on the erodability of loessial sandy loam soils by wind. *Journal of Arid Environments* 34: 391-402.

WOLTERS, M. and J.P. BAKKER. 2002. Soil seed bank and driftline composition along a successional gradient on a temperate salt marsh. *Applied Vegetation Science* 5: 55-62.

WOLTERS, M., GARBUTT, A. and J.P. BAKKER. 2005. Salt marsh restoration: evaluating the success of de-embankments in north-west Europe. *Biological Conservation* 123: 249-268.

WOODELL, S.R.J. 1985. Salinity and seed germination patterns in coastal plants. *Vegetatio* 61: 233-229.

Working for Wetlands. 2006. Orange River Mouth Rehabilitation Plan: Phase Two. EcoAfrica. 41 pp.

Working for Wetlands. [Internet]. Working for Wetlands programme.
Available from: http://www.sanbi.org/research/wetlandprog.htm#prog

XIONG, S. and C., NILSSON. 1997. Dynamics of leaf litter accumulation and its effects on riparian vegetation: A review. *The Botanical Review* 63 (3): 240-264.

YOUNG, J.A. and R.A. EVANS. 1986. Erosion and deposition of fine sediments from playas. *Journal of Arid Environments*. 10: 103–115.

ZEDLER, J.B. 1980. Algal mat productivity comparisons in a salt marsh. *Estuaries* 3:122-131.

ZEDLER, J.B. 1983. Freshwater impacts in normally hypersaline marshes. *Estuaries* 6 (4): 346-355.

ZEDLER, J.B. and P. ADAM. 2002. Saltmarshes. In: Handbook of Ecological Restoration. Volume two: Restoration in Practice. M.R. PERROW (ed.). Academic Press. New York. 618pp.

ZEDLER, J.B. and P.A. BEARE. 1986. Estuarine Variability. D. WOLFE (ed.). Academic Press, San Diego. 298pp.

ZEDLER, J.B. and J.C. CALLAWAY. 1999. Tracking wetland functioning: do mitigation sites follow desired trajectories? *Restoration Ecology* 7: 69-73.

ZEDLER, J.B., CALLAWAY, J.C., DESMOND, J.S., VIVIAN-SMITH, G., WILLIAMS, G.D., SULLIVAN, G., BREWSTER, A.E. and B.K Bradshaw. 1999. Californian salt marsh vegetation: an improved model of spatial pattern. *Ecosystems* 2: 19-35.

ZEDLER, J.B., MORZARIA-LUNA, H. and K. WARD. 2003. The challenge of restoring vegetation on tidal, hypersaline substrates. *Plant and Soil* 253: 259-273.

ZEDLER, J.B. and J.C. ZEDLER. 2000. Evaluating the progress of engineered tidal wetlands. *Ecological Engineering* 15: 211-225.

ZHENG, F.L. 2006. Effect of vegetation changes on soil erosion on the loess plateau. *Soil Science Society of China* 16 (4): 420-427.

ZHENGHU, D., HINGLANG, X., ZHIBAO, D., GANG, W. and S. DRAKE. 2007. Morphological, physical and chemical properties of Aeolian sandy soils in northern China. *Journal of Arid Environments* 68: 66-76.

ZHU, J.K. 2001. Plant salt tolerance. *Trends in Plant Science* 6 (2): 66-71.

ZIA, S. and M.A. KAHN. 2002. Comparative effect of NaCl and seawater on seed germination of *Limonium stocksii. Pakistan Journal of Botany* 34 (4): 345-350.

ZOBECK, T.M. and T.W. POPHAM. 1998. Wind erosion roughness index response to observation spacing and measurement distance. *Soil and Tillage Research* 45: 311-324.

ZOBEL, M., VAN DER MAAREL, E. and C. DUPRE. 1998. Species pool: the concept, its determination and significance for community restoration. *Applied Vegetation Science* 1: 55-66.

Appendix 1

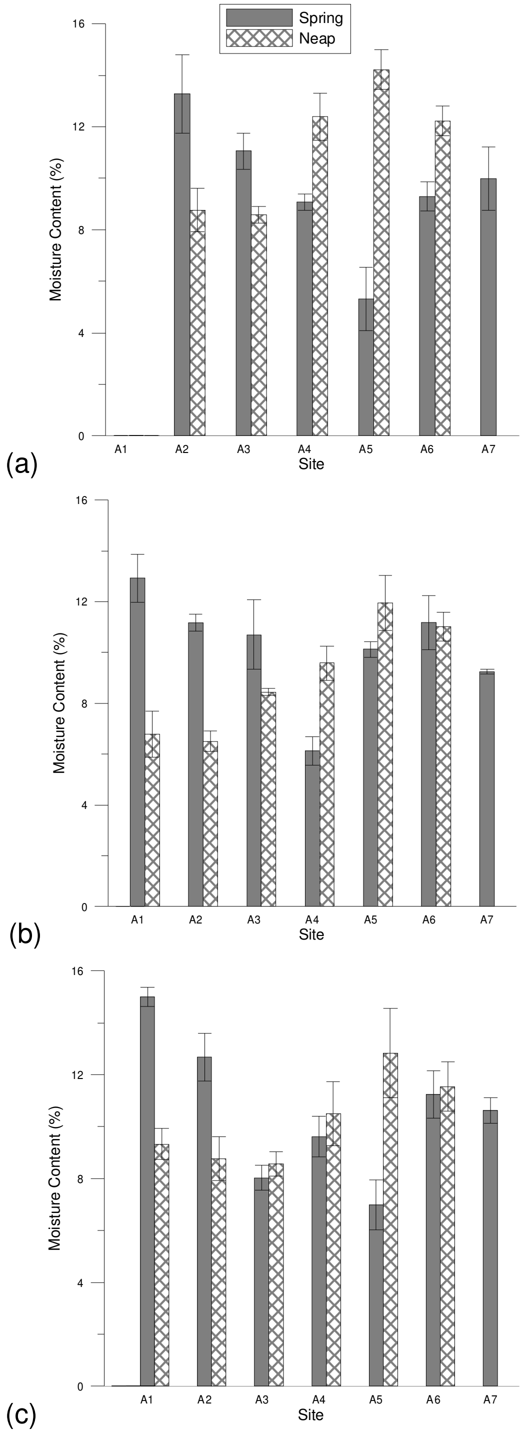


Figure A.1.1 Moisture content in the sediment of the microclimates associated with (a) driftlines, (b) open unvegetated areas and (c) vegetated areas of Site A, 2006.

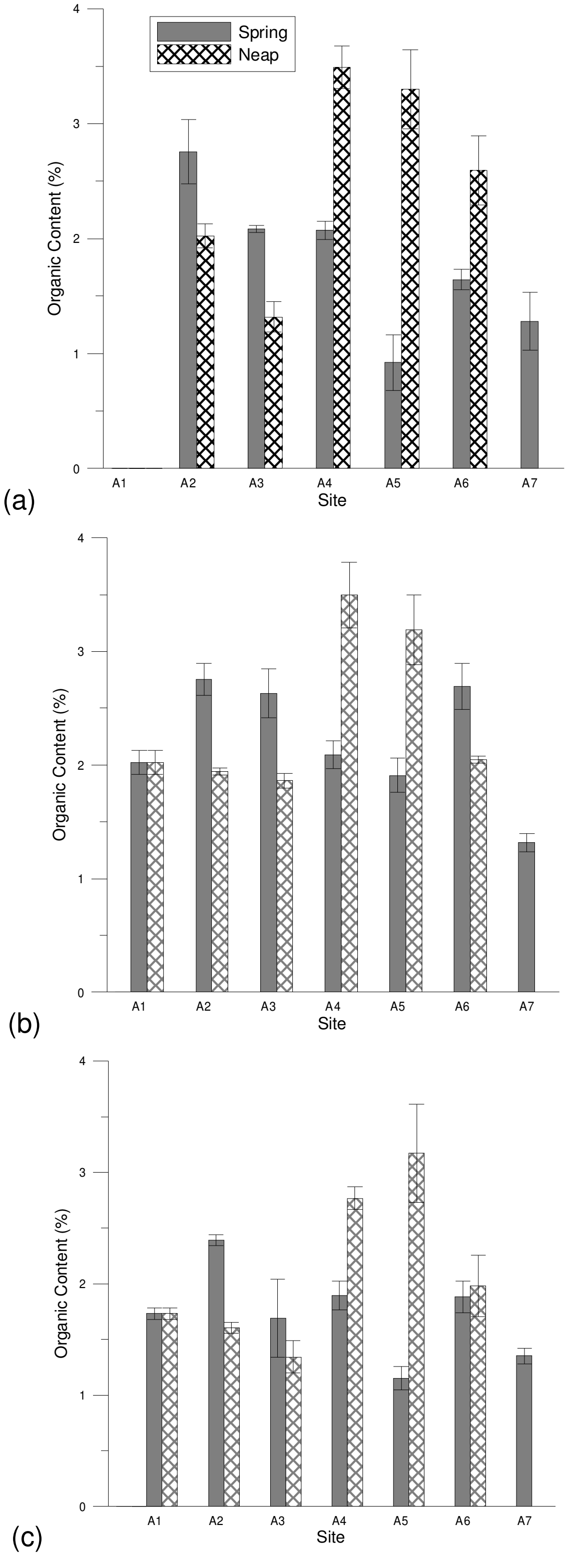


Figure A.1.2 Sediment organic content for the (a) driftline, (b) open unvegetated areas and (c) vegetated areas of site A in 2006.

Appendix 2

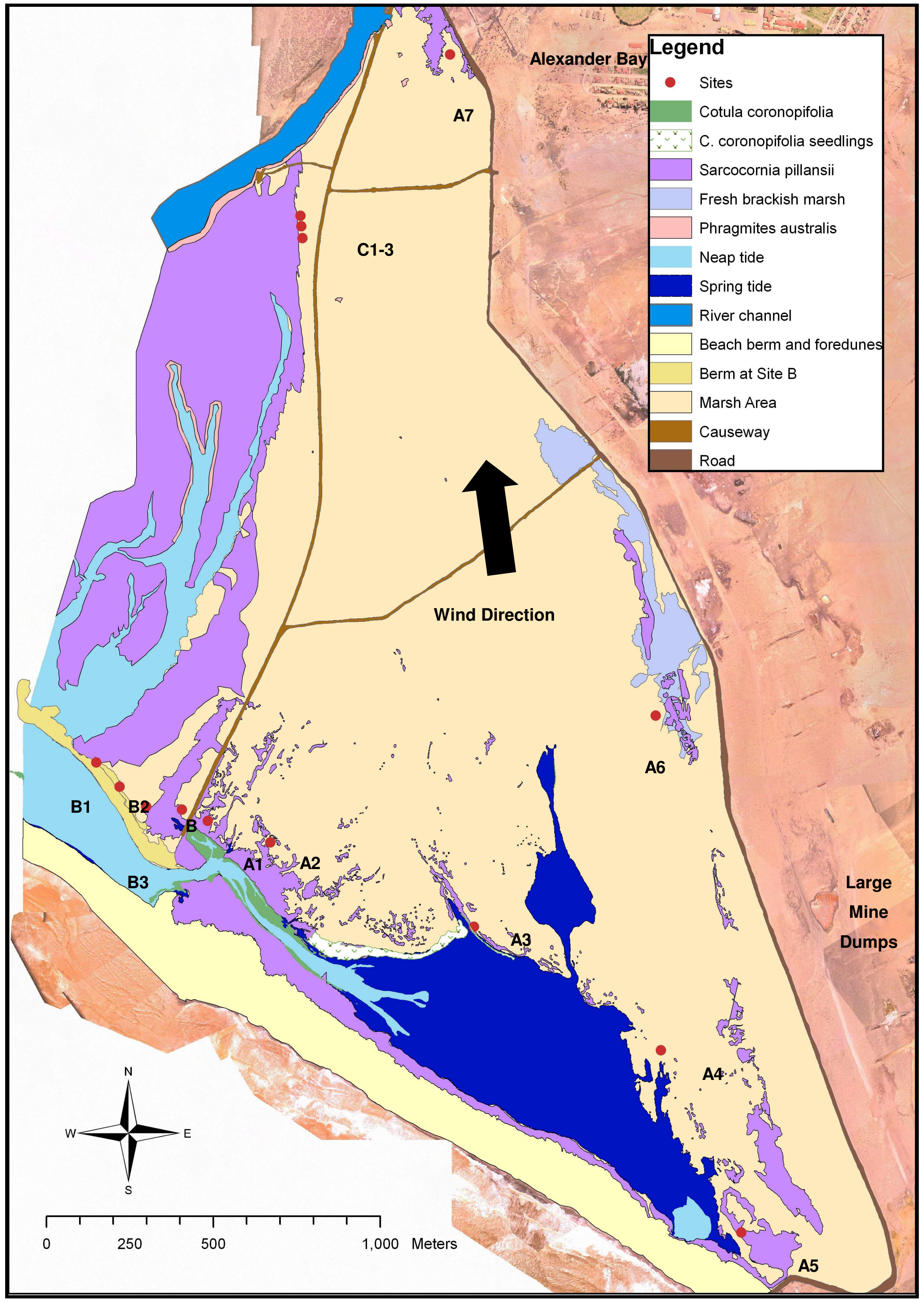


Figure A.2.1 A map of the Orange River Mouth saltmarsh and the locations of the Sites A, B and C.