Predicting the Effects of Salinity on Three Dominant Macrophytes: An Anticipatory Approach to the Restoration of Degraded Coastal Wetlands in NSW, Australia

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Submitted in fulfilment of the requirements of the degree of Doctor of Philosophy

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Declarations

This work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. I give consent to this copy of my thesis, when deposited in the University Library, being made available for loan and photocopying subject to the provisions of the Copyright Act 1968.

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I hereby certify that the work embodied in this Thesis is the result of original research, the greater part of which was completed subsequent to admission to candidature for the degree (except in cases where the Committee has granted approval for credit to be granted from previous candidature at another institution).

Mary E. Greenwood
**Dedication**

*I dedicate this thesis to my late father Charles Stephens (1920-1999), who distilled in me his love of the natural world and to my canine companions who returned that love unconditionally.*

**Acknowledgements**

My father (Charles) was the most compassionate person I have ever known and I could also not have competed this task without the support and eternal optimism shown by my mother Jean and her bevy of friends.

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ABSTRACT

The Hunter Estuary Wetlands (NSW, Australia) are important locally, nationally and internationally. They contain significant breeding and nursery grounds for commercial fisheries and are essential shorebird foraging and roost sites. Originally a mosaic of fresh- and salt-marsh, these wetlands have become degraded due to the erection of flood mitigation structures. Reintroduction of a more natural tidal regime is proposed, which is expected to decrease freshwater macrophytes and increase saltmarsh distribution.

An a priori approach was undertaken to assess the relative salinity tolerance of three macrophytes, prior to restoration commencing. Study species included a glycophyte, *Phragmites australis* (Cav) Trin. ex Steudel, and two closely related estuarine saltmarsh species, the invasive exotic *Juncus acutus* L. and native *Juncus kraussii* Hochst.. Short- and long-term effects of salinity at key life stages were assessed for each species. For *P. australis*, the reliability of physiological and morphological responses to salinity stress was assessed under both laboratory and field conditions as potential indicators for future monitoring of initial restoration progress. Competitive/facilitative interactions between the two *Juncus* species under various salinity regimes were also examined.

Results showed salinity affected viability of *P. australis* but not *Juncus* species seeds. Irrespective of species, cooler temperatures enhanced germination capabilities under saline conditions. *Juncus* species displayed superior germination capabilities ≤ 10 ppt salinity; however, unexpectedly, above 10 ppt germination of *P. australis* was higher.

All three species are highly salt tolerant, although salt adaptation mechanisms were found to differ among species. *P. australis* excluded sodium (Na⁺) where possible, only accumulating Na⁺ to toxic levels beyond particular salinity concentrations (~ 20 ppt) and temporal duration (four months). *Juncus* spp. accumulated Na⁺ in both root and shoot tissue without noticeable
damage. Overtime, *J. acutus* regulated Na\(^+\) uptake at exposure concentrations above 5 ppt salinity, while *J. kraussii* did not commence regulation until concentrations exceed 10 ppt.

A 50% reduction in photosynthesis, biomass, height and density of *P. australis* was apparent at 20 ppt salinity and mortality at 30 ppt. In *P. australis*, although height and density were indicative of salinity stress under laboratory conditions, only density showed potential as an indicator of reduced vigour under field scenarios, providing a valuable potential tool to track initial expected restoration trajectories. Although affected, neither *Juncus* species experienced a 50% reduction in measured endpoints at 40 ppt salinity. However, biomass allocation was asymmetrical. Under stressful conditions, *J. acutus* maintained shoot increase at the expense of root development. Conversely, as salinity rose *J. kraussii* preserved root development rather than shoot growth. *J. acutus* was facilitated by the presence of *J. kraussii* under freshwater conditions, but suffered a competitive response at 10 ppt salinity. *Juncus kraussii* was detrimentally affected by being grown with *J. acutus* at 5 ppt, but unaffected under non-saline and 10 ppt salinity conditions.

All three species possess overlapping salinity tolerances. Creating conditions that favour a particular species is perhaps not realistic, given the limited resources of many restoration initiatives. Flooding duration, depth and waterlogging may modify these results. However, the most plausible scenario is that *P. australis* will continue to dominate marshes after tidal reinstatement. With time, where soil salinity rises above 30 ppt, distribution of *Juncus* species will increase. The relative salinity tolerances of *J. acutus* and *J. kraussii* are analogous. Under mild salinity regimes *J. acutus* is likely to out-compete *J. kraussii*. *Juncus kraussii* is expected to be restricted to areas of high salinity stress.
PREFACE

This thesis was undertaken to elucidate the possible effects of planned restoration programs on dominant marsh vegetation species, prior to commencement of restoration initiatives. The study was funded by the University of Newcastle, with industry support from the Hunter-Central Rivers Catchment Management Authority, while a University of Newcastle Postgraduate Research Scholarship (UNRS External) supported the project. A summary of publications arising from the research to date is provided in Table P.1 (Appendix P).

Individual chapters within the body of the thesis are presented in publication format. They adhere to typical scientific format with summary, introduction, literature review, aims and hypotheses, materials and methods, results and discussion sections included. These chapters are drawn together with a general introduction of the overall project and a synthesis of the information presented in the final chapter. Combined acknowledgement and reference sections are used.

Table -P1 Publications arising from this research. See appendix P for individual publication

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M. E. Greenwood
Salinity effects on three dominant macrophytes xvi
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<td>P7</td>
<td>Greenwood, M.E. and MacFarlane, G.R., 2004. Germination characteristics of two <em>Juncus</em> Species, with regard to salinity and temperature. 25th Anniversary Society of Wetland Scientists Annual Meeting. Seattle, WA, USA, 18-23 July 2004</td>
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CHAPTER 1:
INTRODUCTION

1.1 Thesis rationale and aims

Similar to other countries Australia has an extensive history of modifying tidal flows to estuarine wetlands for various urban, industrial and agricultural activities (Long and Mason 1983; Adam 1990; Boesch et al. 1994; Streever and Genders 1997; Mitsch and Gosselink 2000; Turner et al. 2001; Laegdsgaard 2006). Estuarine wetlands are recognised as important ecosystems with high primary productivity (Clarke and Jacoby, 1994; Streever, 1997). In turn this provides quality habitat and food sources for nekton, avian species and other fauna (Morton et al. 1987; Minello 2000; Weinstein et al. 2000; Hattori and Mae 2001; Saintilan 2004). Estuarine wetlands also provide many ecosystem functions, such as flood attenuation, storm and changing climatic protection, ground water recharge, sediment transport, pollution sinks and stabilisation (Keddy 1999; Zedler et al. 2001; Kulsler 2003).

Increased recognition of the value of estuarine wetlands has generated numerous restoration programs. Often projects include the reintroduction of tidal flows to areas degraded through land reclamation and flood mitigation initiatives. Many of these wetlands are currently pastures or fresh/brackish marshes. Increasing saltmarsh distribution will benefit floral and faunal biodiversity and create habitat for migratory waders (Streever et al. 1996; Wilcox and Whillans 1999; Teal and Weisharb 2005).

Passive restoration initiatives often presuppose that the reinstatement of tidal flows, and anticipated increase in salinity, will result in the return of original saltmarsh communities. To date, few quantitative data are available on short- and long-term effects of increasing soil salinity on resident plant species; or, how salinity tolerance may vary dependant on life stage within and among species. Additionally, it is unclear if reestablishment of
native saltmarsh species is feasible, especially when closely related exotic halophytes exist in neighbouring remnant source populations and compete with natives.

Rarely is a proactive position taken. A priori experimental work can determine salinity tolerance and salinity effects on resident dominant glycophytic macrophytes, or co-existing halophytic species (i.e. relic saltmarsh species still existing at low abundance) that are expected to displace the dominant(s), before the reinstatement of tidal flows.

Vegetation monitoring is often included in restoration activities, taking place before and after commencement of tidal flow modification, with comparison to appropriate reference (i.e. pristine saltmarsh) and control (i.e. unrestored) locations. The focus of these studies is toward the monitoring of desirable restoration endpoints; in other words, the reestablishment and maintenance of saltmarsh abundance and diversity, which may take years to occur.

It is uncommon for monitoring programs to examine the progression of restoration activities from the perspective of studying the initial demise of unwanted dominant species. A priori experimental work on the effects of increasing salinity on undesirable glycophytes may quantify the early sub-lethal effects of saline exposure and longer-term effects on growth, productivity and ultimately survival. Assessment of such endpoints may provide robust biomonitoring tools in field based scenarios. Knowledge of the progression of initial demise in real restoration initiatives can increase adaptive management capabilities and therefore the likelihood of suitable restoration outcomes.

1.1.1 Aims, scope and contribution of the research

This research was undertaken to assess the effects of salinity on three macrophytes of degraded Australian coastal marshes. Study species were a dominant glycophyte, *Phragmites australis* (Cav) Trin. ex Steudel, and two closely related estuarine saltmarsh species, the invasive exotic *Juncus acutus* L. and native *Juncus kraussii* Hochst.. Effects of salinity on each species germination and initial growth phase are elucidated. Short-and long-term (twenty-four hours to four-month period) exposure are considered. Assessment
of a number of known stress indicators is made, to determine if they are reliable parameters of short/long-term response to salinity stress. Relationships between salinity and stress indicators, reported under laboratory conditions, are compared with field situations data to determine if results are transferable to natural conditions. Finally, interspecific competition between two closely related saltmarsh species (native vs. invasive exotic Juncus spp.) is investigated to determine if competitive ability is modified by a salinity gradient.

1.2 Plant community dynamics

The distribution of plant communities within any given area are a result of complex interactions among individuals, populations, communities and the environment (Emery et al. 2001; Tilman and Lehman 2001; Pennings et al. 2003; Crain et al. 2004; Ewanchuk and Bertness 2004; Hill et al. 2005). Historically, there has been a general decline in biodiversity within plant communities worldwide (Mack et al. 2000; Dybas 2004; Houlaian and Findlay 2004; McNeely 2004; Yurkonis et al. 2005). Invasive native and exotic species invade systems, competing with and possibly replacing indigenous species (Yurkonis et al. 2005). Pressures, resulting largely from anthropogenic activities and urbanisation mean that few, if any, ecosystems are now considered pristine (Mack et al. 2000). Increased urbanisation results in land changes and fragmentation of habitats. Clearing and burning land changes biotic communities and environments, altering nutrient cycling and modifying soil structure and erosion rates (Isacch et al. 2004; Hill et al. 2005). Added to this, climate change is now recognised as a major threat to the potential distribution of many species (Donnelly and Bertness 2001; Higgins et al. 2003).

1.2.1 Restoration research

Community concerns and conservation legislation have resulted in restoration and rehabilitation of many degraded ecosystems, including restoration of habitats modified for mitigation purposes, gaining considerable interest in the preceding five decades (Adam 1990; Wali 1999; Laegdsgaard 2006). This has spawned a plethora of related ecological research including ecosystem resilience/resistance/stability, ecological engineering, risk analysis and resource and adaptive restoration management techniques.
(Bradshaw 1996; Pastorok et al. 1997; Keddy 1999; Zedler and Callaway 1999; Davis and Slobodkin 2004; Teal and Weisharb 2005). Systems studied include rangelands (Page and Bork 2005), mining sites (Venter and van Vuren 1997) and old growth forests (Gurnell et al. 2004). However, due in part to the proximity of, and pressure from, urban development the large majority of research relates to wetlands and associated systems (Turner and Lewis 1997; Zedler and Callaway 1999; Zedler et al. 2003; Callaway 2005; Laegdsgaard 2006).

1.3 **Saltmarsh**

Saltmarsh are common tidal wetlands habitats. Generally located in temperate mid-latitudes they occupy intertidal environments, surviving in a niche created by saline conditions that restrict even mangroves (Adam 1997). Generally, saltmarsh is situated in areas protected by adjoining estuaries, or sub estuaries (Mitsch and Gosselink 2000). In Australia, saltmarsh is typically found in the upper tidal zone of estuaries, often behind mangrove systems (Saintilan and Williams 1999). This dynamic and successional habitat is often characterized by low floral diversity and high primary productivity, tidal floods having significant influence on the function and health of the saltmarsh (Congdon and McComb 1980; Adam 1990; Clarke and Jacoby 1994; Zedler et al. 2001; Adam 2002).

1.3.1 **Saltmarsh values**

Estuarine, saltmarsh is a major contributor to shoreline stabilisation, often being the first terrestrial land encountered. Saltmarshes help control floods, improve water quality and contribute significantly to coastal food systems. Saltmarsh aids in the storage, dilution and stabilisation of pollutants, along with transformation and exportation of nutrients into the marine environment (Long and Mason 1983; Adam 1990; Streever and Genders 1998; Turner et al. 2001; Wolters et al. 2005; Laegdsgaard 2006).

1.3.1.1 **Nekton use**

A number of commercially important fish species utilise saltmarsh and surrounding mangrove forests at some stage in their life cycle. At Botany Bay (NSW), more than 40
fish species were recorded over a two-year sampling period. Of these, 13 economically important species were observed in intertidal-marsh and mangrove-swamps (Richardson 1983). More recently, stable isotopes ($\delta^{13}$C, $\delta^{15}$N, $\delta^{34}$S) have been used to track trophic interactions between marine fauna and vegetation species (Weinstein et al. 2000; Connolly 2004). Values of $\delta^{13}$C in the American mummichog (Fundulus heteroclitus) suggest reducing tidal restrictions can improve trophic support functions, through increasing creek sinuosity and facilitate movement of individuals (Woznzk et al. 2006).

### 1.3.1.2 Avian use

Shorebirds, such as plovers, stilts and curlews, depend upon various wetland types (Smith 1991; Straw 2000). Migratory shorebirds are particularly vulnerable to habitat change, as they require a continuation of habitat type (Commonwealth Government 2004; Ling et al. 2006). Along coastal fly-ways saltmarsh affords safe feeding, resting and over-wintering habitat for shorebirds (Saintilan and Williams 1999). Since insects and soft bodied invertebrates make up the majority of saltmarsh fauna, saltmarsh is particularly favoured by shorebirds, rather than mangrove forests or tall reed communities (Laegdsgaard 2006). Additionally, the low vegetation of saltmarsh enables rapid identification of predators by birds while foresting and roosting (Straw 2000; Laegdsgaard 2006).

### 1.3.1.3 Species Refuge

Saltmarsh provides habitat for species which, at present have little or no economic worth, thereby increasing diversity at the regional scale (Bridgewater and Cresswell 1999). Increasingly, saltmarsh habitat encourages ecotourism, recreation and education, such as research into coastal and climate influences, vegetation interactions and avian ecology.

### 1.3.2 Saltmarsh Loss

Globally, there has been an extensive loss of saltmarsh habitat (Clarke and Hannon 1967; El-Demerdash et al. 1990; Burchett et al. 1998; Allen 2000; Mitsch and Gosselink 2000; Adam 2002; Wolters et al. 2005; Laegdsgaard 2006). Although found around inland lakes and salt-flats, saltmarsh is predominantly considered coastal habitat. Many countries, including Australia, are highly urbanised along the southern coastal fringe (Fagan and Webber 1996). Saltmarsh ecosystems are fragile and easily disturbed by small
hydrological or topographical changes. Land reclamation and flood mitigation works such as dredging, the instillation of levees, culverts and floodgates, modify tidal flow, as do industrial infrastructures such as power transmission lines and access roads (Hughes 1998; Crain et al. 2004; Laegdsgaard 2006). At the individual marsh scale, saltmarsh can be damaged by trampling, grazing, rubbish dumping and the effects of storms (Silliman and Bertness 2002; Florentine and Westbrooke 2005; Mack et al. 2000). The result is saltmarsh habitat being lost, reduced in size, or having altered ecosystem structure and function (Streever and Genders 1998; Williams et al. 2000; Adam 2002). Additionally, and increasingly, changes in saltmarsh communities and shifts within saltmarsh assemblages (low marsh species encroaching into high marsh areas) are often considered a response to altered conditions, such as rising sea-levels and climatic changes (Hartig et al. 1997; Donnelly and Bertness 2001).

### 1.3.2.1 An Australian perspective

Table 1-1 provides an overview of saltmarsh loss globally and within Australia. Australia has over 970 estuaries, of these only around 50 estuaries have been studied in detail (Turner et al. 2001). However, the decline of saltmarsh throughout south-eastern Australia is well documented (Saintilana and Williams 1999; Saintilana and Wilton 2001; Rogers et al. 2005). While much of this decline has been attributed to land reclamation and freshwater inputs, mangrove incursion into saltmarsh has also been implicated (Wilton 2002; Rogers et al. 2005). Wilton (2002) found that between 1940 and 1990 saltmarsh habitat at Towra (Sydney region) declined by 50%, with a corresponding increase in mangrove forests. During the same period, over 60% of saltmarsh vanished at Cararma Inlet (eastern coast), due

Between 1954 and 2000 the installation of tidal gates and other human activities has caused over 200 ha of saltmarsh to be degraded at Tomago Wetlands (MacDonald 2001). During the same period 67% of saltmarsh has been lost at Hexham Swamp (Williams et al. 2000), and 99% on Kooragang Island (Morrison 2001). In 2004, Coastal Saltmarsh in New South Wales (NSW) was listed on the Threatened Species Conservation Act 1995 (amended 2002) as an endangered ecological community (DEC 2004).
Table 1-2 Estermates and reasons given for saltmarsh loss.

<table>
<thead>
<tr>
<th>Location</th>
<th>% Loss</th>
<th>Reason for loss</th>
<th>Reference</th>
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<tbody>
<tr>
<td>Globally</td>
<td>Up to 75%</td>
<td>Human intervention</td>
<td>(Mitsch and Gosselink 2000)</td>
</tr>
<tr>
<td>Australia SE coast</td>
<td>39% 1940-2000</td>
<td>Sea level rise</td>
<td>(Wilton 2002)</td>
</tr>
<tr>
<td>NSW</td>
<td>30-100% 1790-1990</td>
<td>Mangrove encroachment</td>
<td>(Saintilan and Williams 1999)</td>
</tr>
<tr>
<td>Hunter NSW</td>
<td>67% 1954-2000</td>
<td>Floodgates</td>
<td>(Williams et al. 2000)</td>
</tr>
<tr>
<td>Hexham Swamp</td>
<td>99.4% 1938-2001</td>
<td>Filling/dredging</td>
<td>(Morrison 2001)</td>
</tr>
<tr>
<td>Hunter NSW</td>
<td>212 ha 1954-2000</td>
<td>Floodgates/levees</td>
<td>(MacDonald 2001)</td>
</tr>
</tbody>
</table>

1.3.3 Land reclamation and tidal restriction

Due to increased recognition of saltmarsh attributes and current legislation in many countries, the threat of large-scale reclamation has been reduced, although some small-scale infilling still occurs (Harty 2004). The Threatened Species Conservation Act (1995) identifies key threatening processes such as the invasion of native plant communities, as well as listing species and communities at risk (DEC 2004). Land reclamation projects affect surrounding saltmarsh, often fragmenting and degrading remaining marsh sites. Increased nutrient and agricultural chemical inputs occur because of stormwater discharge into saltmarsh. Additionally, runoff from urban areas alters natural salinity regimes, promoting the spread of freshwater and brackish plant species (Allison 1996; Levine et al. 1998b; Bart and Hartman 2003; Crain et al. 2004; Wolters et al. 2005; Laegdsgaard 2006). Along roadsides, ditches and drains facilitate the spread of *P. australis* and exotic weeds, while increasing the risk of grass-fires (Long and Mason 1983; Levine et al. 1998b; Adam 2002; Harty 2004; Mack et al. 2000; Bart et al. 2006; Laegdsgaard 2006). In Australia, over 28,000 foreign plants have been introduced in the past 200 years, with around 2500 species establishing in natural habitat (Dybas 2004).
The erection of artificial structures for flood mitigation strategies protects the urban landscape. Many of these structures operate as a one-way valve, allowing freshwater to be released downstream while blocking saltwater from entering the marsh system (MacDonald 2001). This results in altered salinity regimes, sediment structure, water depth and hydoperiods. Typically, saltmarsh sediments exhibit high salinity levels and are often waterlogged, creating anoxic conditions. Disturbing sediments and altering hydrological regimes can induce oxic sediment conditions, resulting in a potential source of acid sulphate soils and the leaching of sulphuric acid into coastal waterways (Clarke and Hannon 1967; Dickinson and Mark 1999; Allen 2000; Adam 2002). Over time, soil salinity upstream decreases, allowing aggressive freshwater and exotic species to invade the saltmarsh. Ultimately, this may lead to the displacement of saltmarsh vegetation, which are considered competitively inferior (Odum 1988; Streever and Genders 1997; Konisky and Burdick 2004; Pennings et al. 2005).

1.4 Hunter Estuary Wetlands

Some of the largest and most important wetlands along the Australian south-eastern coastline are situated within the Hunter River estuary (Hunter Estuary Wetlands, Ramsar No.287) close to the city of Newcastle, NSW (Williams et al. 2000). The area is shown in Figure 1-1. Recognised for its conservation importance as being the most significant site for migratory wading birds in the state of New South Wales, the wetlands form part of the East-Asian-Australasian Shorebird Reserve Network (Streever and Genders 1998; Straw 2000; Svoboda 2004). Wetlands within the estuary consist of a combination of mangrove and saltmarsh communities, along with freshwater pastures and Casuarina woodlands. Standing open water, mudflats, sandy beaches and rock retaining walls complete the habitat composition. The area is botanically diverse with more than 190 plant species recorded, two of which are listed as threatened (Svoboda 2004; Ling et al. 2006). The wetlands comprise of three major sections in various need of restoration, Kooragang Island, Tomago Wetlands and Hexham Swamp, along with smaller marsh sites. As such, they represent an ideal area for the study of restoration ecology.
1.4.1 Hunter River Estuary: History

The Hunter River Estuary has undergone many physical changes over the past 200 years, for a complete history see MacDonald (2001), Ling et al. (2006) and Winning (2006). Early European settlers used the numerous islands at the mouth of the river for timber, coal and lime (oyster shell) removal. Farming and grazing took place in the mid 1800s, along with initial industrial activity such as a saltworks. With the erection of a breakwater and dredging of Newcastle Harbour, the original flow pattern of the river changed. Due to spoil dumping and changing hydrological patterns many small islands began to merge, creating one main land mass (Kooragang Island). Development of Newcastle Harbour continued into the 20th Century and major industrial usage commenced around 1915. Between 1951 and 1976, each year an average of 500,000 m³ of dredged material was used to infill the islands. During this period construction of roads, bridges and rail lines, across Hexham Swamp and Kooragang Island were completed (MacDonald 2001).

Being the largest public/private marsh complex on the NSW coast (6,500 hectares), the Hunter Estuary Wetlands provide a core of protected land (approximately 2,900 hectares) (Hughes 1998; Williams et al. 2000; Svoboda 2004). The wetlands are located in the estuary of the Hunter River, approximately 7 kilometres north of Newcastle on the coast of NSW (32° 51’S, 151° 46’E). Elevation is below 10 metres above sea level and the area receives an average annual rainfall of 1145 mm (Ling et al. 2006). The migratory wader assemblages for these wetlands are estimated at over 10,000 birds, most arriving in October (Gessel 1983). Unfortunately, the number of roosts within the estuary has been greatly reduced over the past three decades and residential waders presently occupy only three sites, two of which are saltmarsh (Martindale 1998). 1.4.1.1 Kooragang Island

1.4.1.1 Kooragang Island

Kooragang Island contains three distinct areas, heavy industrial infrastructure (south), pastoral lands (west) and Nature Reserve (north). Much of the reserve has never undergone tidal restriction and therefore provides a relatively pristine environment (Svoboda 2004). The majority of the western area is under the control of the Kooragang Wetland Restoration Project (1993), which aims to reverse the impact of tidal restrictions.
on the pastoral lands. This project covers around 1,000 hectares and is principally engaged in wetland rehabilitation and educating the wider public on the importance of estuaries.

![Diagram of Hunter Estuary Wetlands](image)

Figure 1-1 Study area. Hunter Estuary Wetlands, Newcastle, NSW, Australia.

1.4.1.2 Tomago Wetlands

Tomago Wetlands include private and public land, the protected area being relatively small (800 ha). In response to ongoing flooding of agricultural land, a drainage network, linked to a major channel and controlled by floodgates, was installed in the 1970’s. Additionally, construction of flood levee banks around Fullerton Cove, which protect the site from tidal inundation (five year frequency) was completed in 1985 (Hughes 1998; MacDonald 2001). The result is that tidal restrictions have degraded and reduced the functionality of the once mainly saltmarsh flats through fragmentation and direct
competition by more aggressive species. Presently, the majority of the Tomago site is freshwater pasture and _Phragmites_ swamp (MacDonald 2001).

**1.4.1.3 Hexham Swamp**

Hexham Swamp incorporates an area of approximately 3800 ha. Over half of the swamp is under private ownership, consisting, for the most part, of freshwater swamp and pasture. Prior to 1950, the site was a mix of fresh, brackish and salt marsh areas. Due, in part, to the reclamation of Kooragang Island and channel dredging in the early part of the twentieth century, severe local flooding occurred around 1950. As a result, a flood mitigation scheme for the Lower Hunter was initiated and eight one-way floodgates were installed at the mouth of the main drainage creek (Ironbark) in 1970-71 (Oceanics 2001). It is estimated that elevations within the marsh have subsided significantly, due to the restriction of normal tidal regimes (Ling et al. 2006). Presently, one of the eight gates restricting Ironbark Creek has been raised approximately 0.25 m, which allows some, although very limited, tidal flow to enter the system (HTMC 1996). Plates 1-1 – 1-4 illustrate types of flood mitigation construction currently within the three study areas.

**1.4.2 Restoration initiatives**

Restoration of the Hunter Estuary Wetlands is a major project of the Hunter-Central Rivers Catchment Management Authority (HCRCMA) (HCRCMA 2007). At Kooragang Island removal or modification of tidal restrictions along two creeks has increased tidal flushing (Streever et al. 1996). Floodgates have been removed along a break in the levee at Tomago Wetlands, while a partial opening of floodgates at Hexham Swamp is proposed for 2008 (HCRCMA 2007). The change to a more natural hydrological regime is expected to increase saltmarsh vegetation and nursery habitat for marine animals. Over time, this would increase the value of habitat for waders and other migratory birds, through increasing roosting areas and feeding grounds.

The importance of hydrology in determining establishment and continuation of particular wetland types and processes is well known (Mitsch and Gosselink 2000). Reestablishment of a tidal hydrological regime has been shown to increase velocity and
low-tide drainage. It may also dissipate floodwaters thereby mitigating the impact of recurring flood events (Streever et al. 1996).

Plate 1-1 New bridge constructed over tidal creek on Kooragang Island
Plate 1-2 Access road with culvert under on an impounded road on Kooragang Island
Plate 1-3 The eight cell floodgates across Ironbark Creek (Hexham Swamp), showing limit of tidal exchange
Plate 1-4 Western floodgates and levee-bank surrounding Tomago Wetlands

In implementing the restoration, a passive approach (opening the area to unrestricted tidal regimes) is to be undertaken. The expectation is that as saline water advances into the marsh, soil salinity levels will increase, heralding the return of original saltmarsh communities and, eventually, creating a stable and functional saltmarsh (Konisky and Burdick 2004; Teal and Weisharb 2005). Unfortunately, although this approach appears
logical and is in line with present day best practice (Callaway 2005), it would seem an overly optimistic view of the situation. Although the hydrology of the area is well documented (Hughes 1998; Williams et al. 2000; Ling et al. 2006), management has little quantitative information on how individual marsh species will respond to the new regime.

1.4.2.1 Hunter River Estuary: Vegetation communities and problematic species

At the present time, *P. australis* is the dominant habitat type in the Hexham Swamp and Tomago Wetlands areas (MacDonald 2001). In 2000, the species was not thought to be problematic on Kooragang Island. However, personal observations indicate a dramatic increase has occurred since 2000. Plates 1-5 – 1-10 illustrate vegetation communities within Kooragang Island, Hexham Swamp and Tomago Wetlands. The dominance of *P. australis* is presently considered the major vegetation threat to the wetlands (Winning 1996; Martindale 1998; Williams et al. 2000; MacDonald 2001; Morrison 2001; Svoboda 2004), although *Typha orientalis* C. Presl., is increasing at freshwater sites.

*Phragmites australis* forms dense, often completely monospecific, stands that exclude all other vegetation. Although *P. australis* provides preferred habitat for the endangered Little and Australasian Bitterns, these stands are often impenetrable to other fauna species, particularly waders (Martindale 1998; MacDonald 2001). Overall, the effect of increased *P. australis* distribution has been a loss of habitat and biodiversity (Briggs 1978; Martindale 1998). Two exotic species are considered particularly problematic, *Cortaderia selloana* (Schult. & Schult.) Asch. & Graebn. and *Juncus acutus*. *Cortaderia selloana* is increasing in areas of higher elevation behind the Tomago Wetland levees. Limited *C. selloana* infestation has been reported in Hexham Swamp (MacDonald 2001); however, the species does not appear to have extend onto Kooragang Island. *Juncus acutus* is widespread and increasing in and around saltmarsh habitat, where it competes with the native brackish rush *J. kraussii*. *Juncus acutus* is considered the most serious threat to saltmarsh community structure and function, with increasing distribution being linked to increased disturbance regimes (Burkett 2000; Adam 2002; Harty 2004; Paul 2007; Laegdsgaard 2006). This reflects the ecological outcome of many estuarine
wetlands subjected to structural flood mitigation along Australia’s eastern coast and strengthens the relevance of these sites as study areas.

1.4.3 Possible effects of restoration initiatives

Restoration ecology acknowledges that degraded ecosystems are often resilient to the restoration process due to changes in the landscape, the nature of feedbacks between abiotic and biotic factors and the availability of native species pools (Bradshaw 1996; Wilcox and Whillans 1999; Davis and Slobodkin 2004; Kulmatiski 2006). Presently, best practice management includes long-term passive restoration initiatives, through implementing a particular environmental trigger that aids the restoration process (Montalto and Steenhuis 2004; Callaway 2005). In the case of estuarine wetlands, restoration is most often achieved through returning hydrological regimes to their previous state. Recent studies on the management of invasive species suggest that full eradication of a pest species is rarely economically or physically possible and that containment and reducing fitness is more achievable (Ranjan 2008). Additionally, conflict may arise due to divergent conservation values, for example two threatened species will be adversely impacted by the demise of freshwater marsh in the Hunter (Little Bitterns and the Green and Golden Bell-Frog) (Streever, 1997; Hamer et al 2002).

1.4.3.1 Effects of restoration on existing species

Prior to implementation of any restoration project, a better understanding of how major plant species are likely to be affected is required. Questions requiring answers include whether existing problematic species will survive at a particular site/salinity. If not, what is the timeframe for eradicating existing populations? If yes, to what extent will reductions in distribution and/or abundance occur? Knowledge of the extent to which species vary in their relative tolerance to salinity at varying stages in life history is also required; as is, at what point along a particular environmental gradient will desirable species be able to compete with, or out-compete, problematic species. Finally, there is a need to know if existing/new problematic species are likely to return/invade (germinate and establish new populations) within the modified regime.
Plate 1-5 *Juncus acutus* fringing an access road on Kooragang Island
Plate 1-6 *Juncus kraussii* growing in the Nature Reserve
Plate 1-7 Sporadic *Casuarina* trees in an area dominated by *Phragmites australis* in Hexham Swamp
Plate 1-8 A reclaimed area in the Industrial section of Kooragang Island, *J. acutus* (foreground), *J. kraussii* (adjacent) and *P. australis* (background)
Plate 1-9 Saltmarsh on Kooragang Island, showing distinct vegetation boundaries
Plate 1-10 A mosaic of vegetation types in Tomago Wetlands. Remnant saltmarsh is fringed by pasture grasses, and *P. australis* is also present. A narrow stand of *Avicennia marina* borders the river’s edge

M. E. Greenwood

Salinity effects on three dominant macrophytes
Most restoration monitoring programs concentrate on evaluating the outcome of the project through comparing the restored site with both un-restored and desirable sites (pristine saltmarsh). (Chapman 1998; Underwood 2000). However, assessments of restoration success employing these types of approaches may take years to realise. If restoration does not proceed as predicted, opportunities for adaptive management may have long passed. This compromises the ability to quantify the progress of an individual restoration project (Zedler and Callaway 2000; Thom et al. 2005).

Reinstatement of tidal hydrology has an immediate effect on freshwater species and a period of decay is normally experienced. Reducing the vigour of a dominant glycophytic species may occur immediately, or require some period before physiological effects are manifested as morphological impacts. Therefore, as long-term restoration monitoring of a desirable outcome is indisputably linked to the short-term effects on undesirable species, there is a need to set both short- and long-term restoration trajectories.

The ability to proactively predict outcomes is increasingly seen as an essential part of any environmental project (Zedler and Callaway 1999). For example, the release of wastewater into streams or rivers does not commence where the impact of particular waste-loads on benthic fauna is not predictable a priori. Adaptive management allows for continuously changing environmental flows, dependant on the on-ground outcome (Zedler and Callaway 2000; Teal and Weisharb 2005; Thom et al. 2005). However, within Australia, the ability to predict an outcome prior to implementation is crucial, as legal constraints, such as obtaining local development approvals, often prohibit the adoption of a fully adaptive management approach. It is strange therefore that many restoration projects commence without a clear understanding of the consequences to vegetation and therefore, dependent fauna.

Restored saltmarsh sites may take 20-50 years to reach biodiversity and functionality similar to that of natural sites (Onaindia et al. 2001; Laegdsgaard 2006). Therefore, monitoring and comparing restored, control and reference site(s) provides little information on the progression of restoration initiatives in the short term. In order to
predict the effects of tidal restoration projects on the wetlands, a knowledge of short-and long-term salinity tolerances of the major species is required. The ability to monitor sub-lethal salinity effects will enable management to determine if undesirable vegetation is being affected as predicted, and if possible trigger a modification where needed. Longer-term monitoring on the growth and productivity of target species will confirm changes at the community level.

1.5 Study Species

In order for the study to be pertinent to the majority of estuarine systems along the east coast of Australia, species evaluation considered the following criteria as desirable.

1) Dominance; species should be dominant in a particular marsh habitat.
2) Coexistence; species environmental requirements should overlap with each other at some point.
3) Management outcomes; species should address a particular management issue, being either of assistance or hindrance in the restoration process.
4) Historical presence; species should possess the ability to recolonise, for example an existing seed bank or the presence of a nearby population.
5) Morphology; preference was given to species with similar size and or structure, thereby controlling for size dominance.

1.5.1 Selecting appropriate study species

Reports and vegetation maps from the mid 20th century were consulted, for a full description of the vegetation history of the area between 1940 and 1999 see (MacDonald 2001). Ground-truthing concurred with MacDonald (2001) that changes during the previous four decades had resulted in reduced habitat of some species, whilst others increased in dominance. Species considered and their relationship to saline and hydrological gradients are depicted in Table 1-2.
**Table 1-2** Summary of the major literature reviewed for the construction of a model (Figure 1-2) determining the position of marsh vegetation species with relation to salinity and hydrology gradients

<table>
<thead>
<tr>
<th>Species</th>
<th>Reference</th>
<th>Factor studied</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Baumea juncea</em></td>
<td>(MacDonald 2001; Deng et al. 2004)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Bolboschoenus caldwellii</em></td>
<td>(Sainty and Jacobs 1994; Blanch et al. 1999)</td>
<td>Hydrology</td>
</tr>
<tr>
<td><em>Juncus usitatus</em></td>
<td>(Sainty and Jacobs 1994)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Fimbristylis ferruginea</em></td>
<td>(Briggs 1978; Winning 1996)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Juncus acutus</em></td>
<td>(Jones and Richards 1954; Flanagan 1997)</td>
<td>Salinity</td>
</tr>
<tr>
<td><strong>J. acutus</strong></td>
<td>(Auld and Medd 1987; Turner and Streever 1999)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Juncus kraussii</em></td>
<td>(Zedler et al. 1990; Russell 2003)</td>
<td>Salinity</td>
</tr>
<tr>
<td><strong>J. kraussii</strong></td>
<td>(Clarke and Hannon 1970; Deng et al. 2004; Lymbery et al. 2006; Naidoo and Kift 2006)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Phragmites australis</em></td>
<td>(Mauchamp et al. 2001; Mauchamp and Methy 2004)</td>
<td>Hydrology</td>
</tr>
<tr>
<td><em>P. australis</em></td>
<td>(Adams and Bate 1999)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>P. australis</em></td>
<td>(Hootsmans and Wiegman 1998; Mauchamp and Mesleard 2001)</td>
<td>Salinity</td>
</tr>
<tr>
<td><em>Sarcocornia quinqueflora</em></td>
<td>(Clarke and Hannon 1970; MacDonald 2001)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Schoenoplectus maritimus</em></td>
<td>(Sainty and Jacobs 1994; Hamer et al. 2002)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Sporobolus virginicus</em></td>
<td>(Clarke and Hannon 1970; Harden 1993; Hamer et al. 2002)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Triglochin striata</em></td>
<td>(Clarke and Hannon 1970; Sainty and Jacobs 1994)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>Typha orientalis</em></td>
<td>(Deng et al. 2004)</td>
<td>Salinity and Hydrology</td>
</tr>
<tr>
<td><em>T. orientalis</em></td>
<td>(Zedler et al. 1990; Hootsmans and Wiegman 1998)</td>
<td>Salinity</td>
</tr>
</tbody>
</table>

At the present time freshwater and slightly brackish areas are dominated by *Bolboschoenus caldwellii* (V. Cook), *J. acutus*, *Paspalum distichum* L. and *P. australis*, whilst brackish and saline marsh is represented by *J. kraussii*, *Sarcocornia quinqueflora* (Bunge ex Ung.-Sternb.) and *Sporobolus virginicus*. *Fimbristylis ferruginea* (L.) Vahl, once a dominant species, was last reported in the marsh in 1990 (Conroy and Lake 1992). Other species that have decreased in dominance include *J. kraussii*, *S. quinqueflora* and *S. virginicus*, while those that have increased are *P. distichum*, *P. australis* and *J. acutus*. 

M. E. Greenwood

Salinity effects on three dominant macrophytes
Although recorded as a dominant in nearby estuaries, *Baumea juncea* (R. Br.) does not appear to be a primary species in the Hunter estuary. *B. caldwellii* and *T. orientalis* were considered freshwater species and eliminated on the basis of known salinity tolerances. *Schoenoplectus maritimus* L. was considered co-dominant in wet, mildly saline sites and *Triglochin striata* Ruiz & Pav. required almost permanent flooding to become dominant. *Suaeda australis* (R. Br.), *S. quinqueflora* and *S. virginicus* are considered highly salt and drought tolerant, but not suitable for episodic or permanently flooded conditions.

### 1.5.1.1. Estimated salinity and inundation tolerances of possible study species

A model of plant species responses to hydrology and salinity was constructed (Figure 1-2) from the literature shown in Table 1-1 and field observations. Soil salinity (fresh to saline) and hydrology (standing water to upland pastures) were considered. The preferred habitat of each species was plotted along both axes, length of arrow depicting position within the marsh. For example, *Triglochin* spp, may be found in fresh or saline standing water, whereas *B. caldwellii* is only located in wet-marsh and standing water habitat with low salinity.

Assessment of the developing model revealed three species (*P. australis*, *J. acutus* and *J. kraussii*) to possess overlapping distribution attributes and have important management implications. *Phragmites australis* and *J. kraussii* are species currently/formerly dominant in the area; whereas, it was considered *J. acutus* has the possibly for becoming dominant post-tidal reinstatement. Dependent on the salinity tolerance of each species, the implementation of restoration projects will alter each species distribution and relative contribution to the transforming wetland community.
Figure 1-2 Schematic diagram of plant species in a New South Wales coastal wetland system and their position along two environmental gradients, salinity and elevation. Plants may range between freshwater obligate hydrophytes (bottom left) to upland and highly salt tolerant species (top right).

Pre-flood mitigation *Juncus kraussii* was a dominant species within wet-saltmarsh areas, *Sarcocornia* and *Sporobolus* dominating the dry-saltmarsh. A major change in the floristic pattern of the area occurred during the twentieth century. Historical vegetation mapping of Hexham Swamp clearly shows the increasing encroachment of *P. australis*, into saltmarsh areas (Figures 1-3 – 1-5) (Briggs 1978; Conroy and Lake 1992; Williams et al. 2000; MacDonald 2001; Morrison 2001). In 1938, *P. australis* was probably present in freshwater swamp and *Typha* communities, but not considered a dominant species

M. E. Greenwood

Salinity effects on three dominant macrophytes
until 1954. Between 1954 and 1976 a slight increase in *P. australis* occurred, at the expense of freshwater swamps. After erection of tidal barriers on the three main creeks draining the swamp a dramatic increase in *P. australis* took place. The most noticeable change was the displacement of freshwater swamps by *P. australis*, although areas of saltmarsh and mangrove forests also declined. By 1996 *P. australis* and to some extent pasture grasses had invaded most areas north of Iron Bark Creek. By 2000, only remnant patches of saltmarsh remained. Although less dramatic, both Tomago and Kooragang Island have experienced a loss of saltmarsh and increase of *P. australis* distribution over the past fifty years. Now, communities dominated by *J. kraussii* occur mainly on Kooragang Island, with fragmented patches persisting at both Tomago and Hexham.

Rehabilitation projects relating to Kooragang, Hexham and Tomago recognise the impact of *P. australis* on the ecological character of the wetlands and specifically target the decrease of *P. australis* as a restoration goal (Svoboda 2004; Laegdsgaard 2006). Additionally, *P. australis* is acknowledged as being invasive of estuarine habitats throughout Australia and other countries (Galatowitsch et al. 1999; Pyke and Havens 1999; Roberts 2000; Burdick et al. 2001; MacDonald 2001; Turner et al. 2001; Bart et al. 2006; Laegdsgaard 2006). Therefore, this species is ideal for evaluating initial effects of salinity and investigating whether those effects are similar under field conditions.

*Juncus acutus* is closely related to *J. kraussii* and appears to possess environmental requirements somewhere between those of *P. australis* and *J. kraussii*. *Juncus acutus* is reported to be increasing in distribution (Lamp and Collett 1989; Groves 1998; Parsons and Cuthbertson 2001; Paul et al. 2007; Coutts-Smith and Downey 2006; Laegdsgaard 2006). Within the Hunter, active eradication of the species has been implemented, with varying degrees of success (Flanagan, 1997). Importantly, very little research has been conducted on this species and its potential environmental distribution is not known. Therefore, *J. acutus* presents a possible competitive threat to the reestablishment of *J. kraussii*. 
Figure 1-3 Land cover of Hexham Swamp between 1938 – 1954. Based on aerial photographs. (Morrison 2001)
Figure 1-4 Land cover of Hexham Swamp between 1976 – 1986. Based on aerial photographs (Morrison 2001)
Figure 1-5 Land cover of Hexham Swamp between 1994 – 2000. Based on aerial photographs (Morrison 2001)
1.6 Autecology of target species

1.6.1 Phragmites australis

*Phragmites australis* (Common Reed) (Plate 1-11), is a cosmopolitan species, dominating fresh and brackish wetlands (Galatowitsch et al. 1999; Meyerson et al. 2000; Mauchamp and Mesleard 2001; Bertness et al. 2002). Whether this is due to the introduction of different (more aggressive) genotypes, as has been hypothesized in North America (Saltonstall 2003), or its affiliation with urbanisation advancement (Shaltout et al. 1995; Bart and Hartman 2000; Minchinton and Bertness 2003) is not known. Alternatively, declining distribution and vigour of the species has been documented in many European stands (Van der Putten, 1997). However, once established, *P. australis* has been shown to change the structure of its environment, though decreasing soil density and increasing aeration to lower soil profiles. The species has also been reported to alter primary production output, lower phosphorus availability for competitors and produce near monospecific communities (Hronec and Hajduk 1998; Templer et al. 1998; Bart and Hartman 2000; Minchinton and Bertness 2003; Ravit et al. 2003). The distribution and abundance of *P. australis* in Australia has changed dramatically in the past 200 years. Pre-European settlement *P. australis* covered vast areas of inland marsh, since then its distribution has declined (Roberts 2000). However, the species has increased in brackish and freshwater areas along the Australian coast (Hughes 1998; Roberts 2000; MacDonald 2001). The spread of *P. australis* has, in general, been at the expense of saltmarsh habitat (Hughes 1998; Williams et al. 2000; Laegdsgaard 2006).

1.6.2 Juncus acutus

*Juncus acutus* (Spiny Rush) (Plate 1-12) is an introduced species with wide geographic distribution, including North and South America, Europe and Africa (Lamp and Collett 1989; Harden 1993). Expanding in coastal marsh situations and abandoned mining sites it is regarded as an environmental weed in Australia (Wills 1962). *Juncus acutus* exploits land where salinity and inundation levels are moderate (Auld and Medd 1987) and is increasingly seen as a major problem facing managers of coastal habitat on east and west Australian coasts. *Juncus acutus* and *J. kraussii* belong to the subgenus *Thalassici* of
Juncus, and are the only two species within this major grouping in Australia (Harden 1993). Juncus acutus has been documented as invading estuarine areas dominated by J. kraussii (Wills 1962; Flanagan 1997; Burkett 2000; Paul et al. 2007). Flanagan (1997) compared 1982 and 1997 surveys, concluding that both species occupied similar environmental niches and J. acutus was progressively displacing J. kraussii in the Lower Hunter marsh system.

1.6.3 Juncus kraussii

Originally listed as Juncus maritimus var. australiensis, J. kraussii (Sea Rush) (Plate 1-13) is most often located in brackish and saline wetlands unaffected by urbanization. It is a native salt marsh rush, confined mainly to Australia, New Zealand, South America and South Africa (Harden 1993). Although it appears not to require a saline environment (Russell 2003; Naidoo and Kift 2006), in Australia it is considered endemic of estuarine wetlands (Clarke and Hannon 1967; Congdon and McComb 1980; Clarke and Jacoby 1994; Sainty and Jacobs 1994) and a saltmarsh indicator species (Commonwealth Government 2004; DEC 2004). Being closely related to J. acutus, concern has been shown that cross hybridisation between the two species may be taking place (LCDC 2006).

1.7 Tidal Reinstatement: Salinity as a dominant stressor in plants

Elevation is generally regarded as the most important environmental factor affecting salt-marsh communities. Elevation affects soil salinity, aeration, water depth and flood periods (Armstrong et al 1985; Wolters et al. 2008). These factors act in unison as stressors, affecting plant vigour and shifting community dynamics. This thesis is directed at describing the affects of salinity, although it is acknowledged that inundation and aeration are equally important.

Increasing tidal flow increases depth and duration of inundation, as well as changing sedimentation, biochemical interactions and, at least short-term, nutrient cycling. Waterlogged conditions restrict oxygen transport to roots, impairing root function in the
low-redox sediment environment. Additionally, sulphidic conditions increase with increased duration of tidal inundation (Long and Mason 1983; Chambers et al. 1998; Mitsch and Gosselink 2000). Sulphide is toxic to some plants and interacts with salinity, decreasing nitrogen uptake as salinity rises (Weisner 1996; Chambers et al. 1998). At present, due to the low elevation and high watertable within the marsh system, many sections of the marsh experience lengthy periods of inundation. Although depth of inundation and frequency will increase, soils and plants already experience some degree of stress related to water-flows. Therefore, changes to the chemical composition of the water are conceivably the most important factors to undergo alteration.

Being affected by hydrology and elevation, soil salinity is considered the foremost stressor in a saltmarsh system and principally responsible for changing vegetation patterns (Armstrong et al. 1985; Bertness and Shumway 1993; Ungar 1998; Huckle et al. 2000). The stress of living in high salinity affects a plant reproduction, establishment, development and productivity. Salinity also modifies interactions between species (Allakhverdiev et al. 2000). The most common ions causing plant stress are Na\(^+\) and Cl\(^-\) (Munns 2002). Relatively few species have developed salt tolerance (Ashraf 2004). Two broad categories of plants are normally recognised, halophytes (salt tolerant) and glycophytes (salt sensitive), depending on their reaction to saline environments (Waisel 1972; Matsumura et al. 1998; Jordan et al. 2002). Salt stress reduces water potential, cause ion imbalance or induce toxicity (Greenway and Munns 1980; Ashraf 2004).

1.7.1 Temporal exposure to salinity affects plant tolerance

Understanding the synergistic relationship between acute (concentration treatment) and chronic (temporal treatment) salinity effects on individual species is important, as stress symptoms may not be apparent in the short term (Munns, 2002) The ability for restoration managers to monitor an immediate reaction and determine possible consequences, under stable conditions, over the life of a project may be of considerable benefit.
1.7.2 Life stage affect on plant salinity tolerance

Research has demonstrated that plants exhibit different sensitivities to environmental stressors at different stages of growth (Zedler et al. 1990; Wijte and Gallagher 1996; Qu and Huang 2005). Similar to other stressors, such as temperature and light, salinity tolerance is highest in dormant seeds (Levitt 1980; Bohnert et al. 1995; Baskin et al. 1998). Conversely, plants tend to be most sensitive to ion stress during the germination and young seedling phase (Wilson et al. 2000). Salt tolerance increases as individuals mature, dependant on the species ability to compartmentalise and/or regulate uptake of Na\(^+\). However, although salinity may not preclude long-term survival, seed viability may be compromised (Greenway and Munns 1980; Katembe et al. 1998; Pujol et al. 2001).

1.7.2.1 Germination and viability

Germination and seedling establishment is normally induced at a favourable time, or place, and crucial in determining species distribution (Baskin et al. 1998). Osmotic effects and ion toxicity are the two main factors causing germination inhibition; although, interaction between salinity and temperature also control germination speed and viability (Khan and Ungar 1984; Khan et al. 2000b). High salinity may inhibit germination of halophytic species but not affect seed viability. Whereas, seeds of glycophytes are often unable to survive similar salinity concentrations (Baskin and Baskin 1998; Khan et al. 2000b; Qu and Huang 2005). For example, halophytic species *Halocnemum strobilaceum* Pall (Pujol et al. 2001) and *Suaeda salsa* L. (Zhao et al. 2003) are able to germinate in salinity as high as 50 ppt, although speed of germination is affected. Conversely, many freshwater species fail to germinate in low to moderate salinity levels (Bewely and Black 1994; Baskin and Baskin 1998).

1.7.2.2 Plant growth

A large amount of literature exists on how salinity adversely affects plant growth. At biochemical and physiological levels affects may include changes in metabolic activity, nutrient and water deficiencies, reduced photosynthesis, impaired respiration and homeostatic ability or, more likely, a combination of these factors (Greenway and Munns...
At the whole plant level, alterations in height, density, and biomass are often recorded (Waisel 1972; Munns and Termatt 1986).

Water, solute and turgor potential are inter-related in plant cells and strongly affected when plants are exposed to salt stress (Parida and Das 2005). Whether due to drought or salinity stress, reductions in turgor pressure have been reported under increased saline conditions (Suarez and Sobrado 2000; Touchette 2006). Accumulation of organic ions in cells results in an increase in solute content, independent of volume changes due to water loss (Taiz and Zeiger 2000).

Plants subjected to salt stress often accumulate high concentrations of inorganic ions or organic solutes (or both). In most salt affected soils, NaCl is the dominant salt, of which Na\(^+\) is the main accumulator in plant cells (Greenway and Munns 1980). Whether glycophyte or halophyte, plants are intolerant to excess sodium in the cytoplasm. In order to maintain metabolic functions plants must exclude salt, compartmentalise excess salts in the vacuole, or redistribute ions to different tissues (Parida and Das 2005). If accumulation is higher than can be regulated in the cytoplasm, toxicity may be induced and growth impaired (Munns and Termatt 1986). Glycophytic species use salt exclusion (excluders) as the major means of salt tolerance, while the typical halophytic response is to maintain a normal water balance through the accumulation (accumulators) of inorganic ions (Taiz and Zeiger 2000; Munns 2002).

In a study of the glycophyte *Populus alba* L, it was found that Na\(^+\) ions were excluded at the root zone (Beritognolo et al. 2007). Conversely, in the halophytes *Atriplex griffithii* Moq (Khan et al. 2000c) and *Suaeda salsa* (Zhao et al. 2003), Na\(^+\) enters the root system and both leaves and shoots accumulate high amounts of Na\(^+\) without loss of biomass. Some halophytes only accumulate Na\(^+\) when exposed to low salinity concentrations, excluding Na\(^+\) ions at concentrations above certain threshold (Greenway and Munns 1980). Therefore, although Na\(^+\) content is often effective in determining between salt tolerant and salt sensitive plants, it can not be considered totally reliable.
Chlorophyll content increases at low salinity (Parida et al. 2002) but decreases and ultimately degrades at high salinity (Jimenez et al. 1997; Ashraf and Harris 2004; Parida and Das 2005). However, salinity does not appear to affect some halophyte species. No difference in the chlorophyll content of saltmarsh species *Sarcocornia virginica* L. (Callaway et al. 1997) or *S. salsa* (Lu et al. 2003) was recorded under a variety of salinity treatments.

In general, salinity reduces net photosynthetic rate and stomatal conductance (Belkhodja et al. 1999; Colom and Vazzana 2002; Lopez et al. 2002; Munns 2002). Salt stress inhibits photosynthesis by reducing water potential. Therefore, the main aim of salt tolerance is to increase water use efficiency under saline conditions (Munns and Termatt 1986). Salt stress may cause short- or long-term effects on photosynthesis. Short-term effects occur immediately and up to 1/2 days exposure and are related to water status; whereas, long-term effects can reduce carbon assimilation (Munns and Termatt 1986).

Halophytic species may increase biomass in low salinity (Khan et al. 2001). However, salinity stress normally results in stunting, decreased stem numbers, low biomass and ultimately mortality (Zedler et al. 1990; Hester et al. 2001; Ashraf 2004). In an evaluation of four wetland species subjected to 4 months exposure at 18ppt salinity, complete mortality was recorded in *Typha latifolia* L. and *Scirpus lacustris* L. and significant shoot and biomass loss in *P. australis* and *S. maritimus* (Hootsmans and Wiegman 1998).

**1.7.3 Use of indicators to determine plant vigour**

Determining an individual species response to tidal reinstatement over time, including immediate and accumulative effects, has the potential to be used by management to monitor progress of restoration outcomes. Of particular interest to restoration managers are salt-tolerant glycophyte species such as *P. australis*, as their responses to the new regime is often uncertain (Short et al. 2000; Svoboda 2004).

The use of indicators to measure negative impacts on existing plant communities is relatively novel (Buchsbaum et al. 2006). Establishing reliable dose-response effects of
salinity in a concentration and time dependent manner, under controlled conditions and verifying these in the field, will provide a robust set of indicators for measuring immediate outcomes of restoration initiatives. Negative sub-lethal and/or lethal effects on freshwater vegetation currently dominating these degraded areas are perhaps the most appropriate and rapid indicators to determine if restoration initiatives are following planned trajectories in the short-term. Clearly, such information has application as a management tool, allowing predictions of the likely consequences of an action before it commences (Thom et al. 2005; Laegdsgaard 2006).

1.7.4 Salinity effect on plant-plant interactions

The relative competitive ability of species, especially where environmental tolerances and/or autecology are similar, contributes to the long-term community composition. It is generally accepted that as quality of habitat declines the importance of competition decreases (Grime 1979; Austin et al. 1988; Greiner La Peyre et al. 2001; Craine 2005; Fynn et al. 2005; Lortie and Callaway 2006). For example, studies on saltmarsh species indicate competitive and facilitative outcomes vary along nutrient (Brewer et al. 1998; Levine et al. 1998a; Emery et al. 2001), hydrological (Crain et al. 2004; Deegan et al. 2005; Pennings et al. 2005) and salinity (Bertness 1991; Huckle et al. 2000; Greiner La Peyre et al. 2001; Konisky and Burdick 2004) gradients. The competitive ability of individual species may change over a salinity gradient (Bertness and Callaway 1994; Emery et al. 2001), thereby facilitating the spread of a particular species. Presently, there is a gap in the knowledge of how salinity modifies competitive interactions between the desirable native *J. kraussii* and unwanted exotic *J. acutus*.

1.8 Management context

Monitoring spatial distribution patterns of a restored area is most often achieved by aerial or satellite imagery (MacDonald 2001), and more recently, hyperspatial imaging and LiDAR. These types of monitoring are particularly good for determining changes at the landscape level. However, this scale is not applicable to investigating communities at the finer scale of individual creek catchments, the most common
restoration type project in Australia (Streever 1997). Nor is the approach instructive in assessing immediate responses to alterations in tidal regimes. Information at the individual species level, especially where species are similar in growth patterns, is required to tease out the more delicate patterns of species responses and community development during the restoration process.

Understanding how passive restoration practices may affect these three species, with the ultimate aim of producing desirable self-sustaining saltmarsh communities, is undoubtedly required. Knowledge of relative salinity tolerances of the three target species at various life stages will aid management in making considered predictions on how vegetation composition may change in response to reintroducing estuarine tidal flows. The change will be time dependent, as soil salinity increases in the newly established mid to high saltmarsh and Na$^+$ accumulation takes place. Initially, management needs to know the extent of adverse effects on each species at various salinity concentrations. This will allow predictions as to whether a species can survive, possibly at a reduced growth rate, within the modified regime. Where mortality of a species is likely to take place at a particular salinity concentration the power to predict a time scale of demise under various field conditions becomes key. Conversely, if a species is to remain part of the new self-sustaining vegetation community, germination and establishment must be able to take place at some opportunistic time or season. Having reliable and validated biomonitoring tools, which allow early assessment of the dominant species responses to the altered salinity regime, is paramount to being able to adaptively manage the restoration process. Finally, if restoration initiatives are to succeed, desirable species need to compete successfully in the evolving marsh.

1.9 Thesis Aims Revisited

A general call for additional research on the management, restoration and monitoring of coastal marshes impacted by weed species has long been made (Long and Mason 1983; Adam 1995; Mitsch and Gosselink 2000; Zedler and Callaway 2000; Adam 2002; Callaway 2005; Laegdsgaard 2006). There are numerous reports relating to active
Passive restoration often includes the reintroduction of tidal flows, which reverse effects of flood mitigation initiatives and rejuvenate degraded saltmarsh communities. Detrimental effects of increased soil salinity on existing fresh and brackish plant communities are routinely expected, but rarely evaluated prior to the restoration process. To predict expected outcomes a more proactive approach is required, whereby a priori experimental work determines the salinity tolerance of existing dominant fresh or salt-tolerant macrophytes,

Examining the progression of restoration outcomes, through assessing the initial demise of a salt-intolerant dominant species, could provide managers with a methodology to better predict, monitor and modify estuarine restoration adaptively. At present, quantitative data on initial sub-lethal effects of increasing soil salinity, together with long term effects on growth and survival under controlled and field conditions is unavailable.

To date, no comprehensive work exists on the affects of restoration initiatives on dominant plant species of Australian coastal marsh systems becoming the new founder species, or possessing the ability to out-compete undesirable species. Knowledge of germination requirements and relative salinity tolerance may allow managers to initiate restoration projects during a particular season or under ecologically favourable conditions. Finally, understanding plant-plant interactions of closely related species will help define each species position within the marsh system, reveal possible obstacles to the final restoration scenario and guide future coastal wetland rehabilitation.

1.9.1 Thesis Aims
This research project was designed to answer the following questions:

- Will target species be able to germinate under the new environmental conditions?
What are the sub-lethal physiological and/or morphological responses of the target species to an environmentally relevant salinity gradient, and does the response vary with development stage and/or impact of exposure time?

Are physiological and/or morphological indicators in *Phragmites australis* reliable predictors of salinity stress under both laboratory and field situations, and thus amenable to monitor the early stages of progression of tidal reinstatement initiatives?

How competitive are the two closely related saltmarsh species (*J. kraussii* and *J. acutus*) with each other at various salinity levels?

To address these questions both laboratory and field studies were undertaken. A summary of the research objectives formulated to address the aforementioned management questions are presented below.

**Question 1: Will target species be able to germinate under the new environmental conditions?**

Chapter 2 investigates the effects of salinity and temperature on the germination characteristics of three target species *P. australis*, *J. acutus*, and *J. kraussii*.

Specific aims of this experiment were to:

- Establish germination ability, rate, and viability of the three species across an environmentally relevant salinity gradient.

- Determine if a difference in temperature range influences salinity tolerance or relieves the light response of *Juncus* spp.

- Provide recommendations for estuarine restoration initiatives.

**Question 2: What are the sub-lethal physiological and/or morphological responses of the target species to an environmentally relevant salinity gradient, and does the response vary with development stage and/or impact of exposure time?**

This question was addressed through glasshouse trials into how increasing salinity affected the physiology and survivorship of *P. australis*, *J. acutus* and *J. kraussii*. Results are presented in chapter 3.
Specific aims of this experiment were to:

- Compare the effect of a salinity gradient on the physiology ($\text{Na}^+$ accumulation, photosynthetic function and photosynthetic pigmentation) of target species.
- Compare the effect of a salinity gradient on the morphology (biomass, height and density) of target species.
- Elucidate the effect of salinity on survivorship of the target species, through determining Lowest Effective Concentration (LOEC) and EC$_{50}$ values.
- Provide recommendations for estuarine restoration initiatives.

- **Question 3:** Are physiological and/or morphological indicators in *Phragmites australis* reliable predictors of salinity stress under both laboratory and field situations, and thus amenable to monitor the early stages of progression of tidal reinstatement initiatives?

Chapter 4 reports on the response of salinity-stress indicators in *P. australis* under natural conditions and over a growing season. Field results are then compared to those reported in chapter 3.

Specific aims of this experiment were to:

- Evaluate the response of *P. australis* to a number of stress indicators.
- Determine if indicators are subject to temporal variability.
- Verify if indicators of stress, examined under controlled conditions, are reliable in field situations.

- **Question 4:** How competitive are the two closely related saltmarsh species (*J. kraussii* and *J. acutus*) with each other at various salinity levels?

Chapter 5 reports on a laboratory trial into the effect of salinity on competitive interactions between the *J. acutus* and *J. kraussii*.

Specific aims of this experiment were to:

- Determine the effect of increasing salinity has on seedling establishment of both species.
Investigate whether increasing salinity improved the competitive ability of either species.

Provide recommendations for estuarine restoration initiatives.

Plate 1-11 *Phragmites australis*

Plate 1-12 *Juncus acutus*

Plate 1-13 *Juncus kraussii*