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Conservation status of the seagrass Posidonia australis Hook f. in south east Australia

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Conservation status of the seagrass *Posidonia australis* Hook f. in south east Australia

By Alexander Meehan

A thesis submitted in partial fulfilment
of the requirements for the award of the degree of

DOCTOR OF PHILOSOPHY

ENVIRONMENTAL SCIENCE
FACULTY OF SCIENCE
THE UNIVERSITY OF WOLLONGONG

2001
DECLARATION OF ORIGINALITY

I, Alexander Meehan, declare that this thesis, submitted in fulfilment of the requirements for the award of Doctor of Philosophy, Environmental Sciences, Faculty of Science, University of Wollongong, is wholly my own work unless otherwise referenced or acknowledged. This thesis has not been submitted for qualifications at any other academic institution.

Alexander Meehan
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My gratitude to Rob Williams and Dr Nick Otway for the advice given during the planning stages of the monitoring study.

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ABSTRACT

*Posidonia australis* Hook.f. is the dominant seagrass in a number of estuaries in south eastern Australia, where it provides habitat and food for a variety of marine fauna, and acts as an important nursery area for juvenile fish, including species of commercial importance. Elsewhere in Australia, there have been numerous reports of significant declines in the abundance of *P. australis*, largely as a result of anthropogenic impacts. These losses have caused great concern about the conservation status of *P. australis*, as it is thought to be slow to recover from damage and difficult to transplant. This study investigated a number of issues related to *P. australis* in southern New South Wales (NSW), an area where *P. australis* has not been previously studied in detail. Research was carried out in six estuaries in the region: Port Hacking, Jervis Bay, St Georges Basin, Wagonga Inlet, Bermagui River and Merimbula Lake.

The status of *P. australis* in southern NSW estuaries is poor. Five of the six estuaries in the region have lost significant areas of *P. australis* over the last fifty years, largely due to anthropogenic impacts, such as shell grit mining, engineering works and deterioration in water quality caused by foreshore development. These losses ranged from about 15% in Bermagui River to 30% in St Georges Basin. *P. australis* increased in area in one estuary, Wagonga Inlet. This increase was probably the result of engineering works at the entrance, which increased tidal range and tidal prism and enabled *P. australis* to expand into deeper waters.

Long term natural recovery of damaged *P. australis* meadows in southern NSW was slow but measurable. In Jervis Bay, seismic survey ‘holes’ created in the late 1960’s have been progressively recolonised by *P. australis* over the last thirty years. *P. australis* rhizomes have spread from the surrounding beds at an average rate of $21 \pm 2 \text{ cm.y}^{-1}$. It was estimated that the ‘holes’ should be completely revegetated at various times over next century, the earliest being 2034 and the latest 2071. In Port Hacking, areas once dredged for shell grit mining have been recolonised by *P. australis*, and a number of small *P. australis* patches now exist in the mined area. Unfortunately, it was not possible to determine whether the patches were the result of
seedling development or regrowth from fragments that were too small to be seen on the historical aerial photographs.

Natural recovery of damaged *P. australis* was also observed in St Georges Basin, NSW. On the edge of surviving *P. australis* beds a large number of plagiotropic (horizontally spreading) rhizomes were observed to be colonising surrounding substrate at an average rate of $21.4 \pm 1.0 \text{ cm.yr}^{-1}$. At four sites, *P. australis* seedlings were found to have recolonised the substrate between these larger beds. For the first time in this region, small *P. australis* plants were also found. However, seedling abundance was low and only two seedlings had matured sufficiently to produce a plagiotropic rhizome. It was estimated that, on average, *P. australis* seedlings may take up to four years before a plagiotropic rhizome is produced. Natural recovery by sexual reproduction in *P. australis* communities is therefore likely to take several decades.

Monitoring *P. australis* communities in southern NSW estuaries was able to detect changes in *P. australis* health due to natural variability, as well as possible anthropogenic impacts. Monitoring *P. australis* shoot density was particularly effective, as small natural variation, and the absence of a strong seasonal trend, meant that sustained declines in health appeared an obvious downward trend in the time series graphs. Generally, natural variation in *P. australis* shoot density fell within 20% of the mean for each site, while more extreme changes in health, possibly due to anthropogenic impacts, fell above 30%. Therefore, a change of $>30\%$ represents a change in *P. australis* health that is of concern with respect to management. This trigger level should be used to monitor the health of *P. australis* in future monitoring studies.

Experimental transplanting of *P. australis* in Port Hacking to restore damaged meadows was successful and increased the amount of *P. australis* habitat in the Port. Three of the five sites survived for the duration of the study, and shoot numbers increased at two of the sites, by 60% at the control site and more than 300% at the Burraneer Bay site. The transplant units (*P. australis* sprigs) also exhibited significant rhizome growth, rhizome branching and shoot growth. Burraneer Bay in Port Hacking
provided a good example of an area which has sustained significant loss of
*P. australis* but now appears ready for further restoration.

Overall, this study confirmed that *P. australis* is vulnerable to a wide variety of
anthropogenic impacts, but that the response to these impacts is complex. *P. australis*
is also extremely slow to recover naturally and is not transplanted easily. However,
*P. australis* in southern NSW estuaries has not declined to the extent that it has in
more developed estuaries, such as Botany Bay, and has a greater capacity to recover
than previously thought. This study also demonstrated that suitable sites for
restoration do exist in southern NSW estuaries, but each site needs to be carefully
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Prior to a feasibility study, restoration needs to be instigated at a number of sites, in
order to compensate for the loss of *P. australis* that has occurred over the last fifty
years. Meanwhile, the remaining areas of *P. australis* in southern NSW estuaries need
to be conserved and protected from waterway and catchment development. A number
of management and research recommendations have been made to assist in this
process.
PUBLICATIONS RELATED TO THIS RESEARCH


Meehan, A.J., and West, R.J. XXXX. Seedling development and patch formation of *Posidonia australis* Hook. f in a south east Australian estuary (in prep.).

Meehan, A.J., and West, R.J. XXXX. The effects of entrance channel modification on the seagrass communities of Wagonga Inlet, NSW, Australia (in prep.).
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Table 7.1. Area of seagrass loss and likely cause of decline in sites within Port Hacking, NSW, since beginning of aerial photography.

Table 7.2. Growth characteristics of *Posidonia australis* shoots transplanted to three surviving sites within Port Hacking, NSW, a CS, RJP and BB.
CHAPTER 1. INTRODUCTION

1.1 General background

Seagrass communities are a key component of coastal ecosystems throughout the world (Kikuchi 1980, Larkum 1976, West 1983, Robblee and Zieman 1984, Walker 1989, Hutomo and Peristiwady 1996). Aside from their intrinsic value, they provide food, habitat and shelter for many commercially important species of fish and crustaceans (Poiner et al. 1989, Gray et al. 1996), contribute to detrital food chains by their high rates of primary production (King 1981), and may also trap and stabilise sediment, thus contributing to the quality of marine and estuarine waters (Gambi et al. 1990).

In Australia, there are over 30 species of seagrass and eight species of which are in the genus *Posidonia* (Walker and Prince 1987). *Posidonia australis* Hook. f. is the most common and widespread species of this genus in Australian waters (Womersley 1981, Walker 1989). *P. australis* is a perennial species of seagrass characterised by long strap-like leaves, erect leaf shoots and a robust rhizome (Den Hartog 1970, West 1983). Like many seagrass species, it forms extensive and ecologically important meadows. For example, *P. australis* leaves are consumed by fish species, such as the fan-bellied leatherjacket (*Monacanthus chinensis*), as well as by sea urchins and various crustacean species (Bell et al. 1978, Conacher et al. 1979, Klumpp and Nichols 1983, West 1983). *P. australis* meadows also function as habitat and shelter by protecting fauna from predation, desiccation and extreme water movements, such as storm waves; as well as promoting sedimentation of local and foreign organic matter (King 1981). These attributes make *P. australis* beds important nursery grounds for various fish species in the south eastern region of Australia. This includes species of commercial importance, such as yellowfin bream (*Acanthopagrus australis*), black bream (*Acanthopagrus butcherii*), sea mullet (*Mugil cephalus*) and luderick (*Girella tricuspidata*) (Burchmore et al. 1984, McNeill et al. 1992, Rotherham 1999, West and Jones 2001).
**P. australis** is the dominant seagrass species in many estuaries in south east Australia (West et al. 1985). Unlike **P. australis** meadows in Western Australia (WA) and South Australia (SA), which are found predominantly in open marine embayments and nearshore areas, distribution of **P. australis** in south east Australia is restricted mainly to estuaries and sheltered embayments by the high energy wave regime along the open coast (Womersley 1981, West et al. 1985). This study examined **P. australis** communities found in a region of the south east Australian coastline that stretches from southern Sydney to Eden (Fig. 1.1). Henceforth, the study area will be referred to as southern NSW. **P. australis** has been found in six estuaries and three marine embayments in southern NSW. The estuaries are Port Hacking, St Georges Basin, Wagonga Inlet, Bermagui River, Merimbula Lake and Pambula Lake (West et al. 1985). The marine embayments are Jervis Bay, Batemans Bay and Twofold Bay (West et al. 1985). In most estuaries in southern NSW, **P. australis** grows from just below the low tide mark to depths of 3 - 5m, except in Jervis Bay, where continuous beds extend to depths of 10m (Larkum 1976, West 1990).

### 1.2 Statement of the problem

In the past, estuaries in central and northern NSW have been focal points for human settlement. The larger estuaries, in particular, provided access to ocean and river shipping and the adjacent lowlands provided suitable land for agriculture (Saenger 1995). In southern NSW, residential and industrial development in estuaries has not yet been as intensive or widespread as it has been in other regions of NSW. However, this trend has changed in recent years. Increasing population in this region has meant that estuaries and catchments in southern NSW are becoming increasingly popular destinations for settlement and recreational usage. For example, population growth in the coastal region of southern NSW was 38% between 1986 and 1996 (EPA 1997).

The distribution of **P. australis** within estuaries in southern NSW, coupled with the recent urban expansion and tourism in this region, makes **P. australis** susceptible to many environmental stresses, such as increased nutrients and sediment, dredging, reclamation and damage from recreational fishing and boating (see Larkum and West 1990, Meehan 1997, Williams and Watford 1999, Wolterding 2000). In several NSW
estuaries, some of these factors have already caused a reduction in the area and health of *P. australis*. For example, in Botany Bay, NSW, the once continuous meadows of *P. australis* have degraded into a number of fragmented beds over the last fifty years, due to poor catchment management, uncontrolled effluent disposal and dredging of the Bay’s entrance. This dredging increased wave height and led to erosion of *P. australis* beds (Larkum and West 1990). Meehan (1997) has also reported that in Merimbula Lake in southern NSW, *P. australis* beds appear to have been damaged by foreshore urbanisation and the construction of oyster leases.

In Australia, the extent of these seagrass losses (see Shepherd et al. 1989 and Kuo et al. 1996) has caused great concern about the management of these important communities and led to recommendations for, and implementation of, protective measures in some regions (Jacoby 1997, NSW Fisheries 1997 and 1999). However, at present there is little information on the conservation status of *P. australis* in southern NSW, as most previous studies of *P. australis* have concentrated on two estuaries within the Sydney Basin, namely Botany Bay and Port Hacking (see Larkum 1976, Kirkman and Reid 1979, West 1983, West et al. 1990, Wolterding 2000). The aim of this thesis is to assess the conservation status of *P. australis* communities in southern NSW estuaries by addressing the following questions:

- What is the present status of *P. australis* in southern NSW estuaries and have changes in area occurred due to catchment and waterway development?
- How fast are *P. australis* communities able to recover from anthropogenic damage and what growth strategies does the species adopt?
- Can the health of *P. australis* in southern NSW estuaries be monitored in such a way that small scale changes are detected quickly and future losses can be prevented?
- Is it feasible to rehabilitate *P. australis* communities by the use of transplants and thereby restore damaged areas?

In addressing these questions, the following gaps in knowledge were identified.
1.3 Present gaps in knowledge

Firstly, there is a lack of information on the status of *P. australis* in southern NSW estuaries. While the area and distribution of all major seagrasses were mapped by West et al. (1985), no follow up study has been carried out. For most estuaries in the region, long term changes in *P. australis* communities in relation to catchment development remain unknown. The exceptions are for two small estuaries, Merimbula Lake and Pambula Lake (Meehan 1997). At present, it is not possible to determine whether losses of *P. australis* communities have also occurred in southern NSW estuaries or, where changes have occurred, to identify the impacts that may have caused them.

Secondly, there is little information on the natural rate of recovery of *P. australis* communities in southern NSW estuaries after damage by anthropogenic and/or natural impacts. Information on natural recovery rates of *P. australis* communities is required to assess whether *P. australis* is at risk of long term or even permanent damage from these impacts. Information on natural recovery would also allow an estimate of the likely time scale before *P. australis* recovers to pre-impact levels. This, in turn, would make it possible to assess the need for artificial planting at damaged sites to accelerate recovery.

Thirdly, there is no information on small scale changes (for example, seasonal changes in abundance) in *P. australis* communities in estuaries outside the Sydney region. In many estuaries in southern NSW, there is no baseline data for any of the community parameters that might be monitored to determine *P. australis* health, such as shoot density. Small scale changes and natural variability in some of the major community parameters of *P. australis* need to be quantified. Since monitoring of *P. australis* health is often concerned with detecting the impacts of catchment development, it is also necessary to distinguish between natural variability and anthropogenic change. Without this information it is not possible to establish a systematic monitoring program aimed at the conservation of *P. australis* habitats in southern NSW estuaries.
Fourthly, a technique to rehabilitate and restore damaged *P. australis* meadows has still not been successfully trialed in this region. This is of considerable importance with regard to present management strategies, as under current NSW legislation (NSWF 1997), development proposals that impact on *P. australis* require compensation in the form of habitat restoration.

The objectives listed below have been formulated to address these gaps in knowledge.

### 1.4 Thesis objectives

1) To determine the present status and document long term changes in area and distribution of *P. australis* (and other seagrass species) in southern NSW estuaries using remote sensing, and to identify the natural and anthropogenic impacts responsible for these changes.

2) To investigate long term natural recovery of *P. australis* in southern NSW estuaries using remote sensing, and predict the likely time-scale of the recovery process.

3) To investigate short term natural recovery of *P. australis* in southern NSW estuaries by examining seedling establishment and patch formation.

4) To measure small scale changes in *P. australis* communities in southern NSW with *in situ* studies, and assess the implications for future monitoring programs.

5) To develop and assess a technique to transplant *P. australis* and rehabilitate damaged *P. australis* habitats in southern NSW.

6) To assess the conservation status of *P. australis* in southern NSW estuaries, assess the implications for management, and, make recommendations for future research.
1.5 Thesis structure

This thesis is comprised of eight chapters.

In Chapter One, a general background is provided regarding *P. australis* in southern NSW estuaries. The remainder of this Chapter introduces the study sites and provides information on estuarine environments in southern NSW. A brief rationale is presented on why particular estuaries were chosen for individual studies.

Chapter Two is a review of relevant literature and is divided into two sections. The first is concerned with previous studies of *P. australis* in southern NSW; long term studies of seagrass ecosystems; seagrass restoration techniques; and the environmental requirements of *P. australis* and other seagrass species. The second is concerned with the methodologies used in this study to investigate the conservation status of *P. australis* in southern NSW. These include, monitoring of seagrasses using remote sensing techniques and *in situ* field techniques, and methods used to age *P. australis* plants.

In Chapter Three, long term changes of *P. australis* in Port Hacking, St Georges Basin, Wagonga Inlet and Bermagui River are presented. The mapping of *P. australis* in these four estuaries will give a complete data set on long term change in *P. australis* in southern NSW estuaries, as Merimbula Lake and Pambula Lake were mapped by Meehan (1997). Historical aerial photographs and a Geographic Information System (GIS) have been used to document changes in area and distribution of *P. australis* (and other major seagrass species), as well as document natural and anthropogenic impacts, such as changes in foreshore landuse and modifications made to entrance channels.

In Chapter Four, natural recovery rates of *P. australis* are examined in two case studies. These two case studies represent the only examples in southern NSW estuaries where remote sensing has been used to examine the recovery processes of *P. australis* communities. The first case study examined recovery of *P. australis* in
meadows damaged by seismic blasting in Jervis Bay. The second case study examined recovery of *P. australis* in areas damaged by shell grit mining in Port Hacking.

In Chapter Five, vegetative and sexual propagation of *P. australis* was investigated in St Georges Basin. This is the only estuary in southern NSW where seedlings have been found *in situ*. The life history of *P. australis* has been reconstructed from seedlings and small plants using ageing techniques, and aspects of plant development have been identified, including rhizome growth rate and shoot production. This study has helped to assess the ability of *P. australis* to naturally recover in the short term through various growth strategies.

In Chapter Six, small scale changes in *P. australis* communities were examined *in situ* at four sites in three estuaries, namely Port Hacking, Wagonga Inlet and Merimbula Lake. These estuaries were chosen for two reasons. Firstly, because they represent the northern (Port Hacking) and southern (Merimbula Lake) extremes of the study area, as well as the approximate midpoint (Wagonga Inlet); and secondly, because they contain large areas of continuous, shallow water *P. australis* beds. Variability in *P. australis* shoot density, leaf area and flowering rate have been examined. The study also attempted to determine whether natural variation can be discriminated from anthropogenic decline.

Chapter Seven assesses the feasibility of transplanting healthy *P. australis* rhizomes attached to an anchoring system, to establish new *P. australis* meadows in Port Hacking. This estuary was chosen because loss of *P. australis* was larger compared to other estuaries in southern NSW and it is closed to commercial fishing. This study examined the effect of site location on survival rate and growth dynamics of the *P. australis* transplants.

In Chapter Eight, the major findings of this thesis are combined to assess the conservation status *P. australis* in southern NSW estuaries. The implications of these findings for seagrass management are discussed, and recommendations for future research are made.
1.6 The study area

1.6.1 Climate

The study area, southern NSW, has been defined as coastline between Sydney and Eden (Fig. 1.1). Near Sydney, the climate is temperate, characterised by mild winters and warm summers. However, towards Merimbula, summer temperatures are milder and winters are slightly colder. For example, in Sydney the average summer (February) maxima and minima are 27°C and 17°C respectively, while the coastal winter (July) maxima and minima are 17°C and 8°C respectively (Bureau of Meteorology 2001). Near Merimbula, the average summer (February) maxima and minima are 25°C and 15°C respectively, while the coastal winter (July) maxima and minima are 16°C and 4°C respectively (Bureau of Meteorology 2001).

In southern NSW, precipitation is distributed fairly evenly throughout the year. The average annual precipitation near Sydney is about 1000-1200mm per year, while near Merimbula it is lower, at 820mm per year (Bureau of Meteorology 2001). Along the entire south east Australian coastline, south-westerly to westerly winds prevail in winter, with summer being dominated by south-easterly winds and some north-easterly sea breezes.

Mean monthly ocean water temperatures in southern NSW are about 17 - 23°C (Womersley 1981). Water temperatures within some southern NSW estuaries can fall to 9°C in winter (Merimbula Lake) and rise to as high as 25°C in summer (Port Hacking). Tides on the NSW coast are semi-diurnal with two low tides and two high tides per lunar cycle of 24 hours and 50 minutes (Womersley 1981). Tidal range is 1.2 - 2m, and at exposed sites wave energy can be moderately high.

1.6.2 Distribution of seagrasses within study area

Six species of seagrass grow in southern NSW: Zostera capricorni, Zostera muelleri, Heterozostera tasmanica, Halophila ovalis, Halophila decipiens, Ruppia spp. and P. australis (West et al. 1985). Zostera spp. and Halophila spp. are the most
widespread seagrasses in southern NSW, occurring in both permanently and intermittently open estuaries (West et al. 1985). Distribution of *P. australis* in southern NSW is thought to be restricted to estuaries where salinity is high and nutrients are low, and it is not found in any estuaries with entrances that periodically close (West et al. 1985). As noted earlier, six estuaries in southern NSW contain *P. australis*: Port Hacking (34° 05'S, 151° 08'E), St Georges Basin (35° 07'S, 150° 40'E), Wagonga Inlet (35° 07'S, 150° 07'E), Bermagui River (36° 26'S, 150° 37'E), Merimbula Lake (36° 54' S, 150° 54'E) and Pambula Lake (36° 54' S, 150° 54'E) (Fig. 1.1). Studies were conducted in each of these estuaries, as outlined in Table 1.1. Major physical and hydrological characteristics of these estuaries have been compiled in Table 1.2. Pambula Lake was not studied due to lack of resources, though long term changes in seagrass distribution and abundance have been previously studied by the author (Meehan 1997).

*P. australis* is also found in three open embayments in southern NSW: Jervis Bay (150° 45'E, 35° 08'S; Fig 1.1), Bateman’s Bay and Twofold Bay. These open embayments were largely excluded from this investigation, as the purpose of this thesis was to investigate *P. australis* in estuarine situations, where they are more likely to be influenced by catchment development. The one exception was Jervis Bay, which provided a unique opportunity to study long term natural recovery of *P. australis* in a near pristine environment (see Chapter Four).

1.6.3 Estuarine environments

According to Roy (1984), estuaries in NSW were excavated into bedrock by rivers during times when sea levels were lower. Southern NSW estuaries are generally located landward of sand barriers deposited during the post-glacial marine transgression, which ceased 6000 to 7000 years ago (Thom and Roy 1983). These estuaries are connected to the sea by inlet channels which may be permanently or intermittently open (West et al. 1985). Seaward of these sand barriers are open embayments, such as Jervis Bay. These large embayments have permanent marine conditions, in contrast to estuaries, whose conditions fluctuate between marine and freshwater according to entrance morphology and degree of fluvial input.
Fig. 1.1. Location of estuaries in southern NSW studied in this thesis: Port Hacking, Jervis Bay, St Georges Basin, Wagonga Inlet, Bermagui River and Merimbula Lake.
Table 1.1: List of estuaries and open embayments in southern NSW that contain *Posidonia australis*; and rationale for their inclusion or exclusion in this study. Asterisk denotes open embayments.

<table>
<thead>
<tr>
<th>Estuary</th>
<th>Remote sensing of long term change in <em>P. australis</em></th>
<th>Remote sensing of long term recovery of <em>P. australis</em></th>
<th>In situ monitoring of <em>P. australis</em> growth</th>
<th>In situ monitoring of <em>P. australis</em> recovery</th>
<th>Restoration of damaged <em>P. australis</em> habitat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Port</td>
<td>this study</td>
<td>This study</td>
<td>this study</td>
<td>insufficient resources</td>
<td>this study</td>
</tr>
<tr>
<td>Hacking</td>
<td>previously investigated</td>
<td>This study</td>
<td>Previously investigated</td>
<td>insufficient resources</td>
<td>insufficient resources</td>
</tr>
<tr>
<td>Jervis Bay*</td>
<td>previously investigated</td>
<td>This study</td>
<td>investigated</td>
<td>insufficient resources</td>
<td>resources</td>
</tr>
<tr>
<td>St Georges Basin</td>
<td>this study</td>
<td>Insufficient resources</td>
<td>this study</td>
<td>this study</td>
<td>this study</td>
</tr>
<tr>
<td>Batemans Bay*</td>
<td>excluded from this thesis as it is an open embayment, rather than an estuary. Also, it contains only one small <em>P. australis</em> bed (West et al. 1985)</td>
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<table>
<thead>
<tr>
<th>Location</th>
<th>Study Type</th>
<th>Applicability</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wagonga Inlet</td>
<td>this study</td>
<td>not applicable to</td>
<td>this study not applicable to</td>
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<td>this estuary</td>
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<tr>
<td>Bermagui River</td>
<td>this study</td>
<td>not applicable to</td>
<td>Insufficient resources</td>
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<td>this estuary</td>
<td>this estuary this estuary</td>
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<tr>
<td>Merimbula Lake</td>
<td>previously</td>
<td>not applicable to</td>
<td>this study insufficient resources</td>
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<td></td>
<td>investigated</td>
<td>this estuary</td>
<td>this estuary low priority - P.</td>
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<td>australis small loss of</td>
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<td></td>
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<td></td>
<td>P. australis</td>
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<tr>
<td>Pambula Lake</td>
<td>previously</td>
<td>not applicable to</td>
<td>Insufficient resources</td>
</tr>
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<td></td>
<td>investigated</td>
<td>this estuary</td>
<td>insufficient low priority -</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>small loss of P. australis</td>
</tr>
<tr>
<td>Twofold Bay*</td>
<td></td>
<td>excluded from this</td>
<td>thesis as it is an open embayment,</td>
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<td></td>
<td></td>
<td>thesis as it is an</td>
<td>rather than an estuary. Also, it</td>
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<td></td>
<td>open embayment,</td>
<td>contains only one small P. australis</td>
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<td></td>
<td></td>
<td>rather than an</td>
<td>bed (West et al. 1985)</td>
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</tbody>
</table>
Table 1.2. Physical, hydrological characteristics of Port Hacking, Jervis Bay, St Georges Basin, Wagonga Inlet, Bermagui River, Merimbula Lake in southern NSW. O - permanently open. I - intermittently closed. T - training walls (data from Bell and Edwards 1980, West and Jones 2001).

<table>
<thead>
<tr>
<th>Estuary</th>
<th>Water area (km²)</th>
<th>Catchment area (km²)</th>
<th>Water temp. (°C)</th>
<th>Salinity (ppt)</th>
<th>Entrance conditions</th>
<th>Commercial fisheries production 1995/96 (kg/yr)</th>
<th>Catchment landuse</th>
</tr>
</thead>
<tbody>
<tr>
<td>Port Hacking</td>
<td>11.3</td>
<td>180</td>
<td>12 - 26</td>
<td>29 - 35</td>
<td>O</td>
<td>not fished commercially</td>
<td>50% cleared. National park and Urban</td>
</tr>
<tr>
<td>Jervis Bay</td>
<td>102.1</td>
<td>410</td>
<td>14 - 25</td>
<td>32 - 36</td>
<td>O</td>
<td>297 000</td>
<td>Low to moderate use. Freehold, Crown and National Park</td>
</tr>
<tr>
<td>St Georges Basin</td>
<td>38.9</td>
<td>390</td>
<td>9 - 28</td>
<td>22 - 37</td>
<td>O</td>
<td>106 751</td>
<td>Low to moderate use. Freehold, Crown and National Park</td>
</tr>
</tbody>
</table>


<table>
<thead>
<tr>
<th>Location</th>
<th>H</th>
<th>Flora 100</th>
<th>Flora</th>
<th>Fauna</th>
<th>Condition</th>
<th>Clearing Percentage</th>
<th>Type of Land Use</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wagonga Inlet</td>
<td>6.3</td>
<td>97</td>
<td>11 - 25</td>
<td>32 - 36</td>
<td>O/T</td>
<td>Considerable oyster production</td>
<td>25 - 50% cleared. Mainly State forest</td>
</tr>
<tr>
<td>Bermagui River</td>
<td>1.4</td>
<td>94</td>
<td>11 - 25</td>
<td>n/a</td>
<td>O/T</td>
<td>4612</td>
<td>50 - 75% cleared. Mainly freehold and State forest</td>
</tr>
<tr>
<td>Merimbula Lake</td>
<td>4.6</td>
<td>48</td>
<td>9 - 25</td>
<td>17 - 36</td>
<td>O</td>
<td>Considerable oyster production</td>
<td>50 - 75% cleared. Mainly State forest</td>
</tr>
</tbody>
</table>
The distribution of seagrasses within estuaries in southern NSW depends on the type of estuary and the degree of infilling that the estuary has undergone through catchment erosion and marine sand incursion (Roy 1984). These factors are discussed below.

1.6.4 Estuary types in the study area

On the basis of geomorphology, Roy (1984) classified estuaries in southern NSW into three basic estuary types: (1) Drowned river valley estuaries; (2) Barrier estuaries; (3) Saline coastal lakes. With the exception of Port Hacking, which is a drowned river valley, all the estuaries studied in this thesis are classified as barrier estuaries, and have permanently open entrances (West et al. 1985). *P. australis* is not found in barrier estuaries with intermittently open entrances, or in saline coastal lakes, possibly due to fluctuating salinity and water levels found in these situations (West et al. 1989).

Drowned river valley estuaries are characterised by open entrances and a full tidal range (Roy 1984). Marine tidal deltas form at the estuary mouth and extend into the lower estuary, where they are reworked by wave action and tidal currents, while fluvial sediments accumulate in deep mud basins behind these deltas (Roy 1984). As fluvial deltas move into these basins, inter-tidal and shallow water environments increase (Roy 1984). In NSW, seagrasses are often associated with drowned river valleys when they have undergone a large degree of infilling.

Barrier estuaries are characterised by narrow entrance channels, situated within barrier sand flats (Roy 1984). Tidal range is attenuated to a degree determined by the entrance channel morphology and entrance size relative to the estuary (Roy 1984). Some barrier estuaries, such as Merimbula Lake, have large tidal range due to large, wide entrance channels and relatively small mud basins; while other barrier estuaries, such as St Georges Basin, have less tidal range, due to long and narrow entrance channels and large mud basins (Bega Valley Shire Council 1997). West et al. (1989) noted that in barrier estuaries with large tidal range, *P. australis* was found seaward of *Zostera* spp., whereas in estuaries with a small tidal range, this situation was reversed. In any case, marine deltas in barrier estuaries are well developed and form shallow, low energy environments ideal for *P. australis* communities (Roy 1984).
According to Roy et al. (2001) each estuary in southern NSW, regardless of type, consists of four main depositional environments. These environments are (seaward to landward): the Marine Tidal Delta zone (MTD); the Central Mud Basin zone (CMB); the Fluvial Delta zone (FD) and Riverine Channels/Alluvial Plain. This study has used these characteristic zones to divide estuaries into separate units for mapping long term change (see Chapter Three). This characterisation of depositional environments has enabled a systematic and consistent method of examining seagrass change at the sub-estuary level to be undertaken in this research (see Chapter Three). The characteristics of only the first three zones will be discussed briefly, as *P. australis* is not found in riverine channels (West et al. 1985).

**Overview of estuary zonation**

The marine tidal delta zone (Fig. 1.2) is composed of quartzose sand and may be heavily influenced by tidal currents and wave action. This zone is composed of high energy environments, such as deep tidal channels and shoaling bay beds, and low energy environments, such as sub-tidal and intertidal sand flat (Roy et al. 2001). These flats are often colonised by seagrasses (Roy et al. 2001).

The central mud basin zone (Fig. 1.2) is characterised by low energy environments composed of fine clay sized silt and river sediment (Roy et al. 2001). Salinity gradients may exist in these basins, and landward sectors may be more brackish and nutrient rich than seaward sectors (Roy et al. 2001). The greater depth and lower light levels means that seagrasses are usually found only on the sandy shoreline facies around the sides of the basin (Roy et al. 2001).
Fig. 1.2. Typical distribution of the marine tidal delta zone, central mud basin zone, and fluvial delta zone in southern NSW estuaries. From Roy et al. (2001).
The fluvial delta zone (Fig. 1.2) is where rivers enter estuarine waters and deposit their sediment load (Roy et al. in prep). It is a more complex association of sediment types, and ranges from clean fluvial sand and gravel on the channel shoals to organic rich mud in brackish swamps (Roy et al. 2001). Salinities, temperatures and tidal exchange fluctuate in response to river flow. Environments include distributary channels, delta mouth bars, levee banks and delta front slopes (Roy et al. 2001). Seagrasses can occur on stable parts of the delta surface, but do not grow to the same depths in this zone as in other zones due to higher turbidity (Roy et al. 2001). *P. australis* is generally absent from this zone of the estuary (West et al. 1985).
CHAPTER 2. LITERATURE SURVEY

2.1 Introduction

This literature survey is divided into two sections. The first section deals with ecological studies of *P. australis* and other seagrass species. The second deals with the methodologies used in this thesis to investigate the conservation status of *P. australis* in southern NSW estuaries.

In the first section, literature relating to the following topics are discussed:

- Previous studies of *P. australis* in southern NSW.
- Long term changes in *P. australis* and other seagrass species in relation to natural and anthropogenic impacts.
- Natural recovery of *P. australis* and other seagrass species following damage.
- Sexual and vegetative propagation in *P. australis* and other seagrass species.
- Techniques used to transplant seagrasses, and case studies illustrating the success that *Posidonia* transplants have had in restoring damaged habitats.
- Environmental factors influencing the distribution and growth of *P. australis* and other seagrass species.

In the second section, literature relating to the following topics are discussed:

- The application of remote sensing to examine long term changes in seagrass communities.
- Techniques used to monitor small scale changes in seagrass communities.
- Estimating age and growth rates of seagrasses using the concept of the Plastochrone Interval (P.I.).
2.2 Section 1: Ecological studies of *Posidonia australis* and other seagrass species

2.2.1 Previous studies of *Posidonia australis* in southern NSW

Early investigations of *P. australis* in southern NSW were carried out by Wood (1959). This involved a qualitative description of *P. australis* morphology, followed by some observations about the environmental requirements of this species. Since then, research has been carried out on a variety of topics, including: the ecology and growth dynamics of *P. australis*, particularly around Sydney (Larkum 1976, West and Larkum 1979, Kirkman and Reid 1979, West 1980, Wolterding 2000); structure and morphology (West 1983, West 1990); the area and distribution along the NSW coastline (West et al. 1985); long term changes in selected NSW estuaries (Larkum and West 1990, Meehan 1997, Williams and Watford 1999); response to reduced light (Fitzpatrick and Kirkman 1995), use as a fisheries habitat (Burchmore et al. 1984, Rotherham 1999, West and Jones 2001) and potential for rehabilitation (West et al. 1990).

These studies have shown that *P. australis* contributes greatly to the productivity of estuaries in NSW, particularly through the production of leaf material (Kirkman and Reid 1979, West and Larkum 1979, West and Larkum 1983). For example, annual primary production of *P. australis* leaf material in Botany Bay was 0.3 - 2 tonnes/ha/yr (West and Larkum 1979) and in Port Hacking was 2.4 tonnes/ha/yr (Kirkman and Reid 1979). *P. australis* leaf material is thought to decompose and be consumed by a rich variety of detrital feeders and macrofauna. Collett et al. (1984) found 323 species of polychaetes, crustaceans and molluscs inhabiting *P. australis* communities at five NSW estuaries. This macrofauna is, in turn, consumed by larger animals (West 1983). *P. australis* meadows are also directly grazed by larger species of marine fauna. As an example, Bell et al. (1978) found that *P. australis* consisted of about 40% by volume of the food eaten by three species of leatherjackets. Similarly, Burchmore et al. (1984) found that *P. australis* leaves, and their attached epiphytes,
represented about 11% of the total food volume for a number of fish species in Port Hacking.

Seagrass meadows have been shown to serve as important nursery areas for numerous fish species, including those of commercial importance (Gray et al. 1996, Bell and Pollard 1989, West and King 1996). This also appears to be true for *P. australis*. For example, Ferrell et al. (1993) found 103 species of fish inhabiting the *P. australis* beds in Port Hacking, Jervis Bay and Wagonga Inlet. In Port Hacking alone, Burchmore et al. (1984) captured 30 species of fish from *P. australis* beds, and eleven of these species were of some economic importance. Not only do *P. australis* communities support a diverse array of fish species, they may support different assemblages of fish compared to other seagrass species. For example, Middleton et al. (1984) found that in Botany Bay, NSW, *P. australis* communities supported larger sized fish than *Z. capricorni* communities. While *Z. capricorni* satisfied the shelter requirements of juvenile fish in the early stage of their development, the fish soon migrated to the *P. australis* beds, where the longer leaved meadows and associated epiphytes were considered to provide more food (Middleton et al. 1984). Rotherham (1999) found that a similar situation occurred in *P. australis* beds and *Z. capricorni* beds in Port Hacking and St Georges Basin. Two fish species in particular, luderick (*Girella tricuspidata*) and tarwhine (*Rhabdosargus sarba*), appeared to move to *P. australis* after spending initial stages in *Z. capricorni* meadows.

The above studies of *P. australis* in southern NSW have demonstrated its importance in the estuarine environment. However, *P. australis* is vulnerable to human disturbance in this region. The estimated rates of rhizome growth and sexual propagation of *P. australis* suggest that it is an extremely slow growing species. Flowering of *P. australis* is reported to be rare and irregular, rhizome extension amongst the slowest recorded (less than 20cm.yr\(^{-1}\)) and sexual propagation almost nil (Larkum 1976, West and Larkum 1983, Inglis and Smith 1998, Wolterding 2000). For example, Larkum (1976) stated that *P. australis* did not successfully seed and did not show any ability to regrow in cleared areas. Likewise, West (1990) found that, at one site in Jervis Bay, only 1 out of 600 randomly collected *P. australis* rhizomes displayed active growth. Although little experimental work has been done on sexual
reproduction in *P. australis* in southern NSW, studies from elsewhere in Australia have confirmed observations about its low recruitment from seed, although flowering and seed production appears to be higher in southern Australia (see Waycott 1995 and Waycott et al. 1997). For example, Kuo and Kirkman (1996) observed that many *Posidonia* seedlings in south-west Australia failed to survive winter storms, and those that survived were soon covered with epiphytic growth and eventually disappeared. They also noted that small plants of *Posidonia* spp. had yet to be found in south-west Australia, suggesting that expansion of *Posidonia* meadows through seedling colonisation was a rare event. This means that recovery of *P. australis* communities following damage may take many decades, possibly centuries. This is of particular significance considering that many marine and estuarine environments in Australia have lost *P. australis* due to coastal development (see below).

2.2.2 Long term changes in seagrass communities in relation to natural and anthropogenic impacts.

Long term changes in seagrass communities have been documented in a number of studies over the last two decades, both in Australia and around the world (Cambridge 1975, Larkum and West 1990, Pergent-Martini and Pergent 1996). These studies have been initiated by a need to quantify human impact on these important vegetation communities. In many cases they have documented significant losses (Orth and Moore 1983, Cambridge et al. 1986, Short et al. 1996) that have been attributed to various factors, including disease (Den Hartog 1987), pollution (Cambridge 1975, Larkum and West 1990, Short et al. 1996), reclamation and dredging (Giesen et al. 1990, Onuf 1994, Long et al. 1996), mariculture (Everett et al. 1995), sand movement (Kirkman 1978, Vidondo et al. 1997) grazing by marine animals (Supanawid 1996) and storm activity (Clarke and Kirkman 1989, Preen et al. 1995).

In the majority of documented cases, long term changes have been attributed to anthropogenic factors. For example, in Botany Bay (NSW) mapping from aerial photographs taken between 1942 to 1984 showed the distribution of *P. australis* to have sustained large losses (about 57%) over the 42 year period (Larkum and West 1990). Increased turbidity brought about by foreshore and catchment development
appeared to reduce the depth limit of *P. australis* in the Bay (Larkum 1976, Larkum and West 1990). In addition, the construction of airport runways and Port Facilities also caused *P. australis* to decline in area, as the associated dredging increased wave heights and caused erosion of already damaged *P. australis* beds.

Similarly, Cambridge (1975) documented the loss of *P. australis* from Cockburn Sound, a sheltered marine embayment in Western Australia. Examination of aerial photographs pre-1952 indicated that *P. australis* formed a continuous cover around the perimeter of the Sound at 1-6m depth. However, in 1952 much of the eastern shore of the sound was zoned for heavy industry and port facilities, and by 1974 seagrass beds adjacent to these industrial complexes had fragmented or disappeared. Further research found that increased nutrient loading was a major cause of *P. australis* depletion (Cambridge et al. 1986). Increased nitrogen from a sewage treatment plant and fertiliser factory is thought to have caused an increase in epiphytic growth on *P. australis* leaves, which in turn reduced light intensity so that the deeper limit of *P. australis* retreated. The thinning of the leaf canopy may have allowed the seagrass beds to become vulnerable overgrazing by sea urchins, which decimated *P. australis* plants in some areas (Cambridge et al. 1986).

In another example, Pergent and Pergent Martini (1996) studied long term change in *Posidonia oceanica* meadows near Marseilles, France. Documentary evidence suggested that the position of the beds changed little between 1898 and 1975. However, between 1975 and 1987 the area of the beds declined by 41%, with the lower limit of the seagrass bed decreasing from a depth of 20m to 10m. Sewage outfall was considered to be largely responsible for this loss. This phenomena was noted earlier in other parts of the Gulf of Marseilles (Perez and Picard 1975). In this instance, harnessing of the Rhone for hydro-electric power may also have contributed to seagrass decline. The fine particulates in the Rhone River were normally dispersed out to sea by strong and regular floods, but these floods were much reduced after the hydro-electric system had been constructed. The reduction in river flow meant that the particulates were not dispersed and were able to be swept along the coast by countercurrents and settle on the *P. oceanica* beds. This reduced light available for photosynthesis (Perez and Picard 1975).
There are also examples of seagrass decline being related to natural phenomena. For example, seagrasses in the Gulf of Carpentaria sustained considerable damage due to the passage of Cyclone Sandy in 1985. The cyclone removed, undermined or smothered 70% (128km²) of seagrasses in the area, and scouring and smothering eventually destroyed the remainder of the bed. Overall, this represented a 20% loss of the seagrasses in the Gulf (Poiner et al. 1989). In another example, Seddon et al. (2000) documented a loss of 12,717 ha of intertidal and shallow subtidal *Amphibolis antarctica* and *Zostera* spp. in northern Spencer Gulf, South Australia. They attributed this dieback to a series of negative tides in combination with a hot El Nino summer, which led to desiccation and eventual death of the intertidal seagrass.

In other cases, both anthropogenic and natural factors have acted in tandem to cause long term decline. For example, Preen et al. (1995) documented flood and cyclone related loss of more than 1000km² of seagrass from Hervey Bay, Queensland. They found that seagrass loss in shallow water (<10m) could be attributed to uprooting during heavy seas; however, seagrass loss in deeper water was probably due to a persistent plume of turbid water resulting from flooding of developed catchments. The authors also noted that, without the cyclonic seas, re-suspension of sediment would not have occurred, the plume would have dissipated sooner and seagrass loss may not have been as great.

The above studies include clear examples of anthropogenic related seagrass decline, which have occurred as a result of large scale industrial and residential development. In other cases, the cause of the decline is not as clear and the reaction of seagrass to stress can be difficult to explain. For example, Meehan (1997) studied historical change in seagrass communities in Merimbula Lake by mapping aerial photographs from 1948 to 1994. Over the 46 year period, shallow beds comprised of *Zostera* spp., *Halophila* spp. and sparse *P. australis* were reduced in area, while deeper *P. australis* beds were stable. Extensive foreshore development, oyster leases and stormwater drains along the northern shore may have caused an influx of sediment which temporarily reduced water quality in the intertidal zone and caused the loss of intertidal seagrass communities. However, intertidal communities on the southern
shore decreased as well, even though development was minimal and stormwater drains were absent.

King and Hodgson (1986) documented changes in seagrasses from 1980 to 1985 in Tuggerah Lake, NSW, and found that the area occupied by the seagrass ranged from 13.13 km² to 19.11 km² but without any apparent trend. They concluded that marked fluctuations in area and distribution are an inherent characteristic of seagrass communities, and that even large changes in seagrass distribution should be interpreted with caution, especially in areas where baseline data or control points were unavailable. This appears to be particularly true in areas subject to low level, indirect impacts, such as Merimbula Lake. However, some sites which have had large scale industrial development do appear to show reasonably clear patterns of decline.

2.2.3 Long term recovery of seagrass ecosystems

As discussed above, declines in the areas of seagrass communities have been recorded in many parts of the world and have been attributed to various natural and anthropogenic impacts (see above). However, there have been relatively few attempts to estimate the recovery time for damaged seagrass beds and little information is available concerning spreading rates for Australian seagrasses, either in terms of rhizome growth, patch formation or seedling establishment (for summary, see Clarke and Kirkman 1989 and Kuo et al. 1996). This makes it difficult to assess the impacts of seagrass losses and to predict when partial or full recovery might occur.

The ability to recover after a disturbance varies greatly between seagrass species. For the species of seagrass in southern NSW, observations over many years have indicated that *Halophila* spp. are relatively quick to respond to change and may recover from damage within months, while *Zostera* spp. may take several years to recolonise denuded areas (see Larkum and West 1983, West 1990, Larkum and West 1990). Studies from outside southern NSW suggest that these observations are accurate with respect to *Halophila* spp. For example, Supanwanid (1996) conducted trials on the recovery of *H. ovalis* after grazing by dugong. The trials found that the biomass of *H. ovalis* in and beside dugong grazing trails was equal after two months of non-
grazing. Also, Walker et al. (1989), in a study on the effects of boat moorings on seagrass beds in Western Australia, observed that bare areas in a *P. australis* bed were colonised by *H. ovalis* and *H. tasmanica* within six to eight months of the removal of moorings. However, *P. australis* has shown little ability to regrow following damage.

In one of the few studies of the colonisation of a seagrass species in south east Australia, Larkum (1976) cleared from small plots (4 m²) in Botany Bay and assessed regrowth. Despite the length of the experiment, three years, no regrowth was observed and movement of rhizomes into the denuded areas was negligible.

Recently, studies have suggested that some species of *Posidonia* may be able to recover from damage, albeit slowly. For example, Kendrick et al. (2000) documented an increases in seagrass cover on Success and Parmelia Banks, WA, between 1965 and 1995. On Success Bank, seagrass cover increased by 22%, from 507 ha to 1036 ha. On Parmelia Bank, seagrass cover remained constant overall but increased in the western region. In both cases the increase was due to the colonisation and growth of *Posidonia coriacea* and *Amphibolis griffithii* (Kendrick et al. 2000).

The above studies demonstrate that colonising species of seagrass, such as *Halophila* spp., will recolonise damaged areas quickly, but slower growing climax species, such as *P. australis* may take decades to return to pre-impact levels, and may not return at all. To understand why seagrass recovery varies amongst species, it is necessary to examine aspects of the reproductive and vegetative growth of different seagrass species.

### 2.2.4 Sexual and vegetative propagation in seagrass communities

Seagrasses have two possible modes of propagation, vegetative and sexual, and thus two mechanisms by which they can recover from damage (Grey and Moffler 1978).

In seagrasses, vegetative propagation occurs by recurrent growth of rhizome internodes which are formed when leaves arising from growing meristems on the shoot undergo abscission, and are replaced by new leaves (Tomlinson 1974, Duarte et al. 1994). Growth may be horizontal (by plagiotropic rhizomes) or vertical (by
orthotropic rhizomes). In the case of *Posidonia* spp. orthotropic rhizomes grow vertically and slowly, and are characterised by short internodes, long straight leaves and infrequent branching (Duarte et al. 1994). Plagiotropic rhizomes grow horizontally and often faster, and are characterised by longer internodes, shorter, curved leaves, and more frequent branching (Molenaar et al. 2000; Fig. 2.3). Together these two forms of vegetative growth cause an increase in plant biomass and density, and maintain the existing bed (Grey and Moffler 1978, Orth et al. 1994). In some cases (for example, for *H. tasmanica*), vegetative propagation can result in the colonisation of new areas when sections of mature plants break off, float away and settle at a new location (Cambridge et al. 1983). However, this mechanism of dispersal has not been reported for *P. australis*.

Sexual propagation refers to the processes of flowering, germination, seed dispersal and seedling success. Sexual reproduction of seagrasses is often responsible for the colonisation of new areas, and may determine whether unvegetated areas are colonised or large disturbed areas repaired. It also provides adaptability and competitive ability through the maintenance of genetic diversity (Kuo and Kirkman 1996).

In the case of *P. australis*, vegetative growth is extremely slow (Larkum 1976, Kuo and Kirkman 1996, Marba and Walker 1999). For example, West (1990) estimated an average rhizome growth of 2.5 cm.y\(^{-1}\) for *P. australis* at sites along the northern shoreline of Jervis Bay (NSW) and a maximum rate of rhizome growth of about 29 cm.y\(^{-1}\) for horizontal rhizomes. West (1990) suggested that mature beds of *P. australis* often had very few horizontal (active) rhizomes, leading to slow recovery of damaged areas. In Western Australia, Marba and Walker (1999) calculated a rhizome extension rate for three *Posidonia* species using reconstruction techniques. Horizontal rhizome extension rates ranged from 9.3 cm.yr\(^{-1}\) for *P. australis*, 12.3 cm.yr\(^{-1}\) for *P. augustifolia*, and 5.5 cm.yr\(^{-1}\) for *P. sinuosa*. Though slow, these rates of rhizome growth are still considerably faster than that of *P. oceanica* on the Spanish Mediterranean coast, where rhizome growth was amongst the slowest recorded: 2.3 cm.yr\(^{-1}\) (average) and 5.9 cm.yr\(^{-1}\) (maximum) (Marba et al. 1996).
Fig 2.3. Orthotropic and plagiotropic rhizomes of *Posidonia* spp. and axes differentiation (adapted from Molenaar et al. 2000).
The reported rhizome growth rates of other species of seagrass are considerably faster (Table 2.1). For example, Duarte and Sand-Jensen (1990) calculated a rhizome elongation for *Cymodocea nodosa* in north-east Spain rate of 132 cm.yr\(^{-1}\). In another study, Gallegos et al. (1993) examined the dynamics of *Thalassia testudinum* in the Mexican Caribbean. Rhizome growth was calculated by ageing consecutive shoots and measuring the internodal length along the rhizome material, and was found to range from 22.3 cm.yr\(^{-1}\) to 35 cm.yr\(^{-1}\).

Along with slow rates of vegetative growth, successful sexual reproduction in *P. australis* appears to be a rare event, at least in south eastern Australia. Larkum (1976) observed that while flowering of *P. australis* was regular, seedlings were not often observed *in situ*. Larkum and West (1983) noted that the production of an apical meristem did not occur for at least a year, so seedling growth was very slow. In one of the few studies on *P. australis* seedlings, Kuo and Kirkman (1996) found that less than one fifth of seedlings that had colonised a boat mooring scar survived the winter storms. Also, Kirkman and Kuo (1990) observed that young plants of *P. australis* had yet to be found in south-west Australia, suggesting that long term survival of seedlings was also very low.

In view of these life history characteristics, the recovery of *P. australis* communities may require assistance in the form of transplanting of vegetative material.

### 2.2.5 Restoration of seagrass habitats using transplants

Transplantation of seagrasses is a technique that has been used to aid recolonisation of areas which have lost seagrass cover, either due to natural or human disturbances. It may also be used to create new seagrass communities in suitable areas, as a way of increasing habitat for commercially valuable or endangered fauna (Phillips 1982). Artificial restoration of seagrass habitats has been attempted for a number of species, with varying degrees of success (Larkum and West 1982). Lewis (1985) lists several types of planting units used for seagrass restoration: plugs, sprigs and seeds/seedlings.
Table 2.1: Plastochrone Intervals and rhizome/shoot growth rates for various seagrass species. P.I. represents the time interval between the formation of two successive leaves (Duarte et al. 1994).

<table>
<thead>
<tr>
<th>Species</th>
<th>Horizontal Rhizome P.I. (d)</th>
<th>Horizontal rhizome elongation (cm.yr⁻¹)</th>
<th>Vertical rhizome elongation (cm.yr⁻¹)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Posidonia australis</td>
<td>52.8</td>
<td>9.3</td>
<td>1.4</td>
<td>Marba and Walker 1999.</td>
</tr>
<tr>
<td></td>
<td>55</td>
<td>-</td>
<td>2</td>
<td>Wolterding 2000</td>
</tr>
<tr>
<td></td>
<td>28</td>
<td>-</td>
<td>-</td>
<td>West 1980</td>
</tr>
<tr>
<td>Amphibolis antarctica</td>
<td>48.3</td>
<td>5.5</td>
<td>15</td>
<td>Marba and Walker 1999.</td>
</tr>
<tr>
<td>Heterozostera tasmanica</td>
<td>12.8</td>
<td>55.7</td>
<td>8.9</td>
<td>Marba and Walker 1999.</td>
</tr>
<tr>
<td>Thalassia Testudinum</td>
<td>5.5</td>
<td>22.3</td>
<td>-</td>
<td>Gallegos et al. 1993</td>
</tr>
</tbody>
</table>
The first two types of planting units are examples of vegetative transplant techniques (Larkum and West 1982). A plug consists of the seagrass and its surrounding sediment. It is excavated from a donor site and then placed at the transplant site, often into an excavated hole (Lewis 1985). The turf may be anchored using pipes construction rods or steel mesh. Sprigs are vegetative sections of the seagrass consisting of roots, rhizomes, leaves but not sediment (Lewis 1985). They should usually be selected to include an apical meristem, without which they may be unable to spread out (Tomlinson 1974). Sprigs may be anchored by using wire or steel pegs, or attaching them to a wire grid that is then placed on the sediment. Transplant success can be reasonably high when using these vegetative transplant techniques, though this is dependant on species. For instance, *Zostera* spp. spread out more quickly than *Posidonia* spp., and thus may colonise transplanted areas with greater initial success.

The second restoration technique uses seedlings. The primary advantage of using seedlings is that seedlings generate an apical meristem directly - vegetative transplants may not have an active apical meristem and so will not spread until one is generated. However, this technique is more difficult than using vegetative transplants as seedlings can be difficult to find in sufficient quantities. For example, *P. australis* seedlings have rarely been sighted *in situ* in south east Australia. Also, seedlings of species such as *P. australis* may not generate an apical meristem for up to one year (West and Larkum 1982).

Over the last thirty years many attempts have been made to apply these transplant techniques to rehabilitate damaged seagrass communities. A review of Australian seagrass rehabilitation and restoration programs was provided by Lord et al. (1999). Selected studies from Australia and overseas involving *Posidonia* transplants have been summarised below to give an indication of the various techniques and success rates.

The only study that has attempted to transplant *P. australis* in south east Australia was conducted in Botany Bay, NSW, by West et al. (1990). Both *P. australis* and the other major seagrass species in the region, *Z. capricorni*, were transplanted. Transplant
units consisted of single shoots of *P. australis* and plugs of *Z. capricorni* consisting of 20-30 shoots. Transplant units were either unanchored or anchored using wire pegs or steel rods at depths of 1, 2 and 3m. In some cases, Artificial Seagrass Units (ASU’s) were used to determine whether protection increased survival rates of the transplants. After three months of monitoring, it was found that highest survival was obtained for units anchored with wire pegs (60-70%). *P. australis* transplants had a lower survival rate than *Z. capricorni* transplants at 1m, 2m, but not at 3m. *P. australis* also had a lower survival rate when unanchored. Longer term survival was not able to be monitored as severe storms in August 1986 destroyed the transplants. Thus, one of the most important results of this study is the emphasis on site selection for seagrass transplants, and the need to ensure that the factor which caused seagrass decline is no longer operating when restoration is attempted (West et al. 1990).

In a more recent experiment in Western Australia (Paling et al. 2001), an underwater seagrass planting machine (‘ECOSUB1’) was used to transplant large sods of *P. sinuosa*, *P. coriacea* and *A. griffithii*. About 1500 sods measuring 0.25m² in area and 0.5m deep were transplanted. After two years, survival rate was 76.8% and 75.8% for *P. sinuosa* and *P. coriacea*, and 44.3% for *A. griffithii*. The authors noted that survival rate was dependent on the species being transplanted, although winter storm damage affected *A. griffithii* transplants.

A similar rate of survival was obtained by Piazzi et al. (1998). In this study, individual rhizomes of *P. oceanica* were planted near Vada, Italy. Two morphological types of rhizomes were used - plagiotrophic and orthotrophic. Plants were anchored to a wire grid and checked for survival and growth rate at six monthly intervals for three years. Transplant success varied from 59 to 76%, with plagiotrophic rhizomes having the highest rate of success. However, rhizome expansion was very slow, at less than 5 cm. yr⁻¹.

The viability of *P. oceanica* seedlings as a transplant unit was assessed in the same region of Italy (Balestri et al. 1998). Aquarium grown *P. oceanica* seedlings were transplanted to examine the effects of substratum (matt vs pebbles) and herbivore exclusion over a three year period. The survival and growth of seedlings under natural
conditions was also investigated. Seedling survival and development was highest when grown on matt - 70% survival for transplanted seedlings and 66% for seedlings that had colonised naturally, while herbivores did not affect survival.

2.2.6 Environmental factors affecting seagrass growth and distribution

This section of this literature survey discusses many of the environmental factors that affect seagrass growth and distribution. Although these factors have not been directly investigated in this study, an understanding of the possible effects was needed to interpret long term changes of seagrass communities presented in Chapter Three, and small scale changes in *P. australis* communities in Chapter Six.

Distribution of seagrasses is influenced by a number of environmental factors, many of which are a function of the location within the estuary and the type of estuarine system. At a local level, each species of seagrass is distributed according to the interaction of light and temperature, sediment and nutrients, salinity, and, wave and current action (Day et al. 1989). These factors are discussed below.

*Light and Temperature*

Light quantity and quality is of primary importance to photosynthetic processes and therefore in determining seagrass growth rates, distribution and abundance. In most areas, seagrasses live in low and variable light environments (Abal et al. 1994). Some species are found throughout the intertidal zone where as others prefer the subtidal zone (Den Hartog 1970). However, the same species of seagrass can often be found distributed at different depths in different estuaries, in response to variations in turbidity of the waters. For example, Larkum (1976) observed that the lower limit for *P. australis* at Towra Point, Botany Bay, was 3m below Mean Low Water (MLW), whereas at Greenpatch, Jervis Bay, the lower limit was 9m. The lower limit of other seagrass species can also vary greatly (Duarte 1991). For example, *Zostera* spp. are able to survive at much greater depths in clearer waters than they are in highly turbid waters, with depth limits ranging from 2m to 30m (Kennish 1986).
Depth distribution of seagrass has often been related to the amount of Photosynthetically Active Radiation (PAR) available (Onuf 1994). The PAR available is dependant on the light attenuating properties of the water, which is determined by factors such as dissolved substances, phytoplankton and the amount of suspended materials (Onuf 1994). In estuarine situations, these factors are likely to be related to drainage basin characteristics, tidal flushing, eutrophication, estuarine morphology, water circulation, fluvial sedimentation rates and anthropogenic disturbances, such as dredging. For example, dredging has the potential to reduce light reaching the sea bed for periods of many months (Onuf 1994), while eutrophication may result in algal blooms in the water column that block out 99% of surface light (Gordon and McComb 1989). Eutrophication may also increase the density of epiphytic organisms growing on seagrass leaves and lead to a reduction in seagrass health (Larkum 1976, Cambridge et al. 1986).

Due to relationship between seagrass productivity and PAR, any change to the available PAR has the potential to alter the health and extent of seagrass communities (Fitzpatrick and Kirkman 1995). Understanding the effects of reduced PAR is important to allow interpretation of the response of seagrass to human impacts. It is difficult to determine how much light must be available, and for what duration, to allow seagrasses to retain their distribution. However, minimum light requirements for seagrasses appear to be higher than for terrestrial plants (Abal et al. 1994). Seagrasses can withstand several days of reduced light resulting from storms and dense cloud cover (Onuf 1994). A number of studies have demonstrated a decline in seagrass productivity in response to reduced light during shading experiments. The response of seagrass to shading varies according a number of factors, such as the light requirements of the individual species, the intensity of shading, and length period of shading.

Two studies illustrate the negative effects that shading has on species of Posidonia. Neverauskas (1988) investigated the response of P. sinuosa in Spencer Gulf, SA, to reduction in light. This was achieved by suspending four 50% shadecloths over a bed in 10 - 11m water depth, with one being removed every three months. Standing crop,
leaf density, shoot density and dry weight of epiphytes decreased slightly after six months, then decreased rapidly over the next three months.

Fitzpatrick and Kirkman (1995) measured the effects of prolonged shading stress on *P. australis* in Jervis Bay, NSW. Steel frames (1.5m x 1.5m x 1.5m) covered in 90% shadecloth were suspended over beds at 3m water depth. After 1 month, leaf growth rate was reduced to 44% compared to an unshaded control plot, and remained lower for the following 2 months. Epiphyte biomass decreased after three months of shading. Shoot density also decreased as duration of shading increased. After 3 months shoot counts were zero in some shaded areas; after eight months they were zero in all shaded plots. After nine months dead rhizomes were observed. No recovery was found in the shaded plots, even after 17 months.

Seagrasses are able to respond to changes in PAR by a number of morphological and physiological adjustments, and thus they are not entirely controlled by the daily light regime (Abal et al. 1994). For example, seasonal patterns of biomass, shoot density and leaf growth rate have been observed for a number of seagrass species in south east Australia (Table 2.2). Seasonal changes in above ground biomass, and other parameters, have been linked to changes in light intensity and day length, as well as salinity and temperature. However, light appears to be the most significant variable in determining seasonal abundance. For example, Mellors et al. (1992) showed that seagrass standing crop was significantly correlated with day length, maximum temperature and wind, with the first two components explaining 97.4% of the variance (Mellors et al. 1992). This correlation was interpreted as the dependence of seagrass on high light saturation, and the relationship between temperature and photosynthesis. In another case, Kerr and Strother (1990) showed that biomass of *Zostera muelleri* was significantly correlated with diffuse solar radiation, day length and maximum temperature.
Table 2.2. Seasonal changes in above ground biomass for different seagrass species in south east Australia, taken from a range of Australian studies.

<table>
<thead>
<tr>
<th>Species</th>
<th>Location</th>
<th>Maximum AGB* (g dry wt.m⁻²)</th>
<th>Maximum AGB* (g dry wt.m⁻²)</th>
<th>Author</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Posidonia</em> australis</td>
<td>Botany Bay, NSW.</td>
<td>189 (May)</td>
<td>286 (Apr)</td>
<td>West &amp; Larkum 1979</td>
</tr>
<tr>
<td></td>
<td>Jervis Bay, NSW.</td>
<td>101 (Oct)</td>
<td>135 (Mar)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Spencer Gulf, SA.</td>
<td>294 (May)</td>
<td>453 (Jan)</td>
<td></td>
</tr>
<tr>
<td><em>Zostera</em> capricorni</td>
<td>Port Hacking, NSW.</td>
<td>444</td>
<td>616</td>
<td></td>
</tr>
<tr>
<td><em>Heterozostera</em> tasmanica</td>
<td>Port Phillip Bay, VIC.</td>
<td>71 (Aug)</td>
<td>279 (Jan)</td>
<td>Bulthius &amp; Woerkerling (1983)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>56 (Aug)</td>
<td>226 (Jan)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>99 (Jan)</td>
<td>286 (Apr)</td>
<td></td>
</tr>
<tr>
<td><em>Zostera</em> muelleri</td>
<td>Swan Bay, Victoria</td>
<td>2.2</td>
<td>84.6</td>
<td>Kerr &amp; Strother (1990)</td>
</tr>
</tbody>
</table>

* above ground biomass.
Spatial variation of seagrass abundance in relation to PAR has also been observed. For example, West (1990) found depth-related morphological variations in *P. australis* in Jervis Bay, NSW. Shoot densities, leaf area per shoot and unit frond weight were found to be significantly greater in shallow areas (1 - 2m) when compared with deeper areas (6 - 7m), presumably due to higher PAR in shallow waters. Moreover, shoot densities were found to be linearly related to depth, particularly between -2m and -8m (A.H.D). Likewise, Larkum et al. (1984) found that biomass of *Z. capricorni* decreased progressively with depth, probably due to progressive reduction in PAR.

**Sediment and nutrients**

There has been little research on the sediment requirements of most Australian seagrass species. For example, in NSW it has been noted that *P. australis* generally occurs on sandy substratum, whereas *Zostera* spp. grows on a range of substrates (Harris et al. 1979, West and Larkum 1983). The role of sediments has been studied largely in relation to the effects of nutrients on growth.

It is of considerable importance to understand the effects of increased nutrients on seagrass communities, as nutrient enrichment is often associated with human development. The transport of nutrients from soils to waterbodies is a natural part of the hydrological cycle and is needed to sustain estuarine ecosystems (Gabric and Bell 1993). However, anthropogenic activities have burdened waters with excessive nutrients, and have the potential to affect seagrass communities (EPA 1997). The response of seagrasses to these increases will depend on factors such as species and nutrient status of the substrate.

Nutrient availability can have a major effect on growth of seagrass communities, in terms of distribution, morphology and seasonal cycle (Short 1987). This may be particularly true for shallow water seagrasses, as they are not likely to be affected by light regime, since PAR is above saturation point (Dennison et al. 1987).

Nutrient type is also of importance in determining seagrass growth. For instance, eelgrass standing crop has been shown to be dependent on the availability of nitrogen
resources, particularly in terrigenous environments (Short 1987). However, in the carbonate environments, seagrass growth may be limited by available phosphorus. In Wilson Inlet, in south west Australia, large increases in the biomass of *Ruppia megacarpa* are thought to be the result of phosphatic fertiliser application to agricultural lands upstream in the catchment (Lukatelich 1987).

Some studies have attempted to differentiate the effects of individual nutrients on seagrass growth. For example, Bulthius and Woekerling (1981) conducted *in situ* additions to *H. tasmanica* beds with nitrogen and phosphorus in Western Port, Australia. No significant increase in density or biomass was exhibited by *H. tasmanica* in this instance. It is thought that the difference in response was due to much higher nutrient concentrations in the interstitial water of the sediments in Western Port than Port Phillip Bay. Thus, nutrient status of sediments is likely to determine whether the seagrass community is nutrient limited (Bulthius et al. 1992).

**Salinity**

Salinity tolerance varies greatly between different species of seagrass. For example, Tyerman et al. (1984) found that *P. australis* was able to withstand salinities as low as 19 ppt, without any affect on net photosynthesis or leaf growth. However, West (1980) found that exposure of *P. australis* to freshwater under laboratory conditions resulted in high mortality of shoots. This may account for that the fact that, on the NSW coast (see West et al. 1985), *P. australis* is found only in permanently open estuaries in which salinity is close to marine levels, and only drops for short periods following rainfall. The other major seagrass in this region of NSW, *Z. capricorni*, is found in estuaries with both open entrances and intermittently open entrances, as well as in riverine channels, suggesting a much greater tolerance to varying and low salinity levels.

Along with PAR, substratum and nutrients, salinity influences the species of seagrass species found in an estuary, and also the distribution of those species within the estuary. For example, in Merimbula Lake, *P. australis* and *Zostera* spp. co-exist in the Marine Tidal Delta zone and the Central Mud Basin zone where salinity rarely falls
below 35ppt. However, *P. australis* is less abundant in the Fluvial Delta zone, and is absent from the Riverine Channel, areas where *Zostera* spp. grow readily and where salinity fluctuates according to riverine flow (Roy et al. 2001).

In one study, Montague and Ley (1993) assessed salinity levels in the canal networks in north-eastern Florida Bay (USA). They found that, at some stations, widely fluctuating salinities caused physiological stress that impaired growth and survival of marine vegetation. Also, because salinity levels did not remain within the range of tolerance of any one species for a long period of time, no species was able to permanently colonise the area.

In other cases, changes in salinity may actually benefit certain species of marine vegetation. For example, in 1987 in the Coorong Lagoons, South Australia, an increase in the flow of the Murray River caused salinity to fall in the north lagoon and water to become brackish (Geddes 1987). In response to this change in salinity, *Ruppia megacarpa* flowered profusely and *Ruppia tuberosa* expanded throughout the lagoon.

**Waves and Currents**

The effects of wave action and current strength on seagrass species are also significant, however, little research has been carried out on these factors. As has been noted by a number of authors (Womersley 1981, West et al. 1989), seagrass communities on the south east coast are mostly restricted to estuaries and sheltered embayments due to the high energy wave regime along the open coastline. However, wave activity, particularly storm wave activity, also impacts on seagrasses within estuarine systems. For example, Larkum et al. (1984) noted that storm waves caused large circular scars in *Z. capricorni* beds in Botany Bay (NSW). Watford et al. (1998) also documented storm damage to *Z. capricorni* and *P. australis* beds in Gunnamatta Bay, Port Hacking (NSW).
2.3 Methods used in this study to investigate the conservation status of *Posidonia australis* in southern NSW

2.3.1 *The application of remote sensing to examine seagrass dynamics*

Remote sensing is a technique that measures the reflectance characteristics (for example, colour, electromagnetic radiation) of the earth’s surface and relates these measurements to vegetation and land use. Imagery produced by remote sensing ranges from the traditional analogue image, the aerial photograph, to digital imagery, such as Landsat and CASI (Compact Airborne Spectrographic Images).

Seagrass meadows form characteristic landscapes that are visible from remotely sensed imagery (Steffenson and McGregor 1976, Kendrick et al. 1999 and 2000, Mumby and Green 2000b). In some cases, different species of seagrass, such as *P. australis* and *Z. capricorni* can exhibit noticeable differences in colour, tone and texture that can be detected on present and past remotely sensed imagery (Larkum and West 1990, Meehan 1997, Williams and Warford 1999). Due to the excellent temporal replication of remotely sensed data at some locations, such as aerial photography, changes in the area, species composition and distribution of seagrass landscapes for many sites can be mapped over several decades, providing information on long term variability in seagrass distribution.

In Chapter Three and Four, aerial photographs have been used to document long term change and recovery of *P. australis* communities in southern NSW. Digital images, such as Landsat images, were not found to offer the spatial or temporal resolution required for this study. For example, in southern NSW, aerial photography spans the last 70 years in metropolitan areas, and 40 to 50 years in most regional areas; while Landsat and SPOT imagery extends back only 15 or 20 years. Seagrass species identification also appears to be beyond the capability of digital imagery at this stage. For example, Jernakoff and Hick (1994) noted that even high resolution CASI could not distinguish between mono-specific beds of different seagrass species. In contrast to this, studies that have used aerial photographs have been able to distinguish...

Another valid way of documenting long term change in seagrass communities is to compare maps made of the study area from previous investigations. However, comparison of maps is often problematic because of differences in scale, map accuracy and interpretation between operators. For example, King (1988) noted that estimates of seagrass area in Lake Illawarra, NSW, probably did not vary over time because of real historical change, but because one method used aerial photographs and did not map sparse seagrass beds, while another used extensive field surveys which were capable of mapping sparse communities. For southern NSW estuaries, historical maps of seagrass distribution are only available for the 1980’s, such as maps produced by West et al. (1985). This does not offer the time frame to assess the impacts of catchment development on seagrass abundance and distribution.

Aerial photographs are the best type of remotely sensed data for this type of study, but they require some degree of processing before they can be used to produce maps of seagrass habitats. Firstly, aerial photographs require correction to remove any geometric errors present in the image due to such factors as panoramic distortion and instrument error. Secondly, a suitable method of interpreting the photographs needs to be chosen, as there are a number of valid ways of converting image data into habitat maps (Green et al. 2000a). These two steps of pre-processing will be discussed below.

*Geometric correction of aerial photography*

Aerial photographs have geometric distortions for a number of reasons, including movement of the aircraft during photograph runs, and imperfections in the optical system of the camera. These distortions can be removed through a number of steps.

Firstly, Ground Control Points (GCP’s) are acquired. GCP’s are specific positions and consist of two pairs of known co-ordinates. GCP’s may be read from a topographic map or acquired using a Global Positioning System (GPS) unit.
Secondly, these GCP's are used to solve a polynomial equation of either first, second or third order (Powley and Drurry 1997). Higher order polynomials require more GCP's but may able to remove more of the geometric distortion (Powley and Drurry 1997). However, Green et al. (2000a) note that higher order polynomials (particularly third order polynomials) may actually increase geometric distortion in some parts of the image, and recommend first or second order polynomials for most applications.

The polynomial equation is then used to rectify and remove distortion. Not all geometric error is removed by rectification. The error remaining in the rectified image is measured by the Root Mean Square (RMS) error (Powley and Drurry 1997). The RMS error for a GCP is the difference between the coordinates of the source GCP and the coordinates of the rectified GCP (Powley and Drurry 1997). A high RMS error indicates an inaccurate GCP. It should be noted that RMS error is a measure of goodness of fit of GCP's and consequently is only a general indication of image accuracy (Green et al. 2000a). Once geometric distortions have been removed from an image, interpretation of the landscape can be carried out.

Photo-interpretation (visual interpretation) of image data

Visual interpretation of aerial photography is a process whereby the photograph is studied by eye and habitats identified on the basis of reflectance characteristics (colour, tone and texture). The operator may also use local knowledge, previous maps made of the study area and a sound knowledge of seagrass ecology, to identify species and draw boundaries between habitats (Green et al. 2000b).

The delineation of habitat boundaries can be achieved by using a tracing overlay and pencil, or using a mouse and computer screen. In both cases the resulting data is called a Vector layer. Each separate habitat that is traced by pencil or digitised by mouse is called a Polygon. The size and shape of the polygons will vary according to a number of factors, including the scale of the photo and the complexity of the habitat (Green et al. 2000b). For example, a fragmented seagrass habitat may need a larger number of smaller polygons than a continuous seagrass bed, but the operator can choose to group all small habitats into one large polygon, and simply label the polygon: Sparse cover
seagrass. Using a mouse, computer and GIS software may enable the operator to magnify the image and therefore locate the boundaries of each habitat type more accurately. Although a GIS may be used, the method does not involve computational effort or statistical processing to aid the operators interpretation. As a result, the process is, to some extent, subjective.

Multispectral classification of image data

Statistical processing and computational effort is used in multispectral classification, and therefore this method might be regarded as less subjective (though not necessarily more accurate). A three stage process is used to group pixels together into different classes (Green et al. 2000b). Firstly, the spectral envelope of each habitat class is defined (the 'signature'). Secondly, signature evaluation checks that the signature is representative of the habitat and that there is not too much overlap between different habitats. Thirdly, mathematical algorithms are used to classify every pixel on the image according to its signature (Green et al. 2000b). The resulting layer is called a Raster layer, where every pixel is a distinct unit.

The main advantage of classification over visual interpretation is that it is quicker and less tedious. However, classification, even when supervised, often confuses spectral signatures and may result in less than accurate maps (Scarpace et al. 1981). For example, Ackleson and Klemas (1987) found that classification confused the spectral characteristics of Zostera marina with deep water. In a study of the Turks and Caicos Islands, Green et al. (2000b) found that visual interpretation was significantly more accurate than multispectral classification, but of a similar accuracy to classification aided by contextual editing.

Visual interpretation has previously been used by this author to map seagrass habitats (see Meehan 1997), and was the preferred method for the present study. Most of the older aerial photographs of southern NSW estuaries are black and white, and of average quality. Classification methods do not work well with these older black and white photographs (Meehan, unpub. data).
2.3.2 Monitoring seagrass communities using in situ techniques

The importance of seagrasses in the coastal environment has meant there is considerable interest in monitoring seagrass health (Thomas et al. 1999). The objective of monitoring is to ensure that any deterioration in seagrass health is quickly detected, and that this action triggers some form of management intervention to prevent large scale losses occurring.

Assessing seagrass health can, in theory, be carried out using both remote sensing and in situ techniques. The advantages and disadvantages of both these techniques are summarised in Table 2.3. However, in southern NSW estuaries, the available remote sensing data is not suitable to detect small scale changes in seagrass health. For example, aerial photographs for many estuaries in southern NSW are only available at scales of 1:25 000 to 1:50 000 and repeated every 3 to 5 years. These scales do not allow small scale changes to be detected. While it is possible to identify seagrass species, and to measure area and distribution, little information can be gathered on other aspects of seagrass health, such as density.

Monitoring the health of seagrass in situ usually requires describing changes in a specific seagrass community parameter. This parameter is taken to represent the health of the seagrass bed being monitored. While the selection of a parameter to monitor is quite subjective (see Wood and Lavery 2000) the parameters that have been widely used to detect small scale changes include biomass, density, percentage cover and productivity (Kirkman 1996). The advantages and disadvantages of these parameters have been summarised in Table 2.4. At present there is no standard set of protocols to monitor seagrass health using any of these parameters, although there have been some attempts to evaluate various monitoring techniques (see Mellors 1991, Inglis and Smith 1995). Many of the studies that have monitored seagrass health in south east Australian estuaries have varied in terms of the type of parameter, the number of samples and the sampling interval (see Larkum 1976, West and Larkum 1979, West 1990, Fletcher 1997, Wolterding 2000). For this study it was considered necessary to review some of the monitoring methods and independently decide on an appropriate
method to monitor small scale changes in *P. australis* communities in southern NSW estuaries.

Due to the extremely slow growth of *P. australis*, destructive sampling measures, such as collecting plants to measure biomass, were not considered suitable. Such destructive sampling is likely to have a long term impact on otherwise healthy seagrass beds. This meant that *in situ* counts of shoot density or estimates of percentage cover were the most appropriate techniques.

Percentage cover is a measure of seagrass health and sociability (Kirkman 1996). It is recorded by subjectively assessing the cover of seagrass at various points in the seagrass bed, and ascribing each point to a predetermined class interval (Kirkman 1996). The type of class interval is not constant in the literature, and different researchers use very different techniques. For example, Evans and Gibbs (1983) used only three intervals to classify seagrass communities on the NSW coast: 0-25% (sparse cover), 25-50% (moderate cover), and >75% (dense cover). King (1988) also used three intervals when mapping seagrasses in Lake Illawarra. However, the class intervals were different, with sparse cover designated as 0-15%, moderate cover as 15-50%, and dense cover as >50% (King 1988). King (1988) also measured the sociability of the seagrass by classifying communities into a) individual clumps; b) patches up to 10m, and c) beds of even distribution. Another study by King and Barclay (1986) used seven intervals to classify seagrass percent cover: <2%, 2-11%, 12-24%, 25-40%, 41-60%, 61-80%, and >80%. These examples are indicative of studies that have used percentage cover to estimate seagrass health, and illustrate the range of methods adopted.
Table 2.3: Advantages and disadvantages of aerial photography versus *in situ* monitoring of seagrass health.

<table>
<thead>
<tr>
<th>Technique</th>
<th>Advantages</th>
<th>Disadvantages</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aerial Photography</td>
<td>Excellent source of historical data.</td>
<td>Cannot readily detect changes in density or biomass of seagrass.</td>
<td>Larkum et al. 1984,</td>
</tr>
<tr>
<td></td>
<td>Large spatial changes easily detected.</td>
<td>Processing of images is slow and final maps may take months to produce.</td>
<td>West et al. 1985,</td>
</tr>
<tr>
<td></td>
<td>Less field work required, which is important in isolated areas.</td>
<td></td>
<td>Larkum and West 1990,</td>
</tr>
<tr>
<td></td>
<td>Ideal way to provide an inventory of seagrass resources.</td>
<td>Photography may be compromised by sun glint, shading, etc.</td>
<td>Short et al. 1996.</td>
</tr>
<tr>
<td><em>In situ</em></td>
<td>Physical sampling of bed yields quick results and short processing times.</td>
<td>Practical problems with ensuring permanency of sampling stations.</td>
<td>Harris et al. 1979,</td>
</tr>
<tr>
<td>monitoring</td>
<td>Can detect small scale changes in seagrass health.</td>
<td>May give false signal in dynamic seagrass environments.</td>
<td>Larkum et al. 1984,</td>
</tr>
<tr>
<td></td>
<td>Output is quick to obtain and may be acted on accordingly.</td>
<td>Labor intensive.</td>
<td>King and Holland 1986,</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>West 1990, Abal and</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Dennison 1996.</td>
</tr>
</tbody>
</table>
Table 2.4: Advantages and disadvantages of various community parameters used to measure seagrass health.

<table>
<thead>
<tr>
<th>Measure</th>
<th>Advantages</th>
<th>Disadvantages</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Responds quickly to perturbation.</td>
<td>Requires SCUBA in deep water, which may be beyond resources of study.</td>
<td></td>
</tr>
<tr>
<td>counts</td>
<td>unhealthy sites</td>
<td>Deeper sites require SCUBA, which may be beyond resources of study.</td>
<td></td>
</tr>
<tr>
<td>of plant</td>
<td>Non destructive.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>parts</td>
<td>Often correlated with biomass.</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Responsive to perturbations.</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Does not require laboratory processing.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Percentage</td>
<td>Quick and easy and may be estimated from water surface.</td>
<td>Is usually estimated visually and so is subjective.</td>
<td>King 1986, King &amp; Barclay 1986, King 1988, Heidelbaugh &amp; Nelson 1996.</td>
</tr>
<tr>
<td>Cover</td>
<td>May be correlated with biomass.</td>
<td>Depending on technique used, may not be very sensitive.</td>
<td></td>
</tr>
<tr>
<td>Productivity</td>
<td>Is very sensitive to perturbations.</td>
<td>May give misleading results. For example, leaf growth rate may increase even as shoot density decreases. Requires repeated site visits and therefore may be impractical</td>
<td>West &amp; Larkum 1979, Kirkman &amp; Cook 1982, Larkum et al. 1984.</td>
</tr>
<tr>
<td>-----------------------</td>
<td>------------------------------------</td>
<td>----------------------------------------------------------------------------------------------------------------</td>
<td>----------------------------------------------------------</td>
</tr>
<tr>
<td>Epiphytes</td>
<td>May be a useful good indicator of eutrophication. Epiphyte species may be good bio-indicators.</td>
<td>Is more a measure of water conditions than actual seagrass health. ‘Unwarranted importance’ - may not differ between health and unhealthy sites (see Wood and Lavery 2000).</td>
<td>Fitzpatrick &amp; Kirkman 1985, Neveraluskas 1987, Williams &amp; Ruckelshaus 1993.</td>
</tr>
</tbody>
</table>
The density of a seagrass community refers to the number of plant parts per unit area. Shoot density and/or leaf density is usually counted when biomass samples are taken, but can also be counted non-destructively in situ (Kirkman 1996). Compared to percentage cover, in situ density estimate require more time and effort and may require the use of SCUBA in deeper waters. However, in situ density estimates is often a more objective measure than percentage cover and, in the literature, the only difference in technique is the choice of quadrat size and sample size. Also, studies by Neverauskas (1988) and Fitzpatrick and Kirkman (1995) have shown that shoot density responds rapidly to shading, and is closely correlated with biomass (Kirkman 1996). This makes it a useful surrogate for biomass, and, probably a more reliable measure than percentage cover.

2.3.3 A description of the method used to estimate age and growth rates of *Posidonia australis*

*P. australis* is a rhizomatous plant that spreads by recurrent growth of rhizome internodes (Duarte et al. 1994). This process leaves scars on the rhizome which are referred to as nodes. The rhizome between two leaves is called an internode, and can represent the time interval between the formation of two successive leaves (Duarte et al. 1994). This time interval is termed the plastochrone interval (P.I.). The use of a rhizome internodal sequences in seagrasses was probably first used by Patriquin (1973), when he estimated the growth rate, age and production of *Thalassia testudinum*.

The 1:1 relationship between production of internodes and leaves is the basis for reconstructing growth rates and demography of seagrass plants (Duarte et al. 1994). Age estimates from plastochrone intervals can be used to interpret various aspects of seagrass ecology, such as rhizome growth rate and shoot production. Direct measurements of leaf P.I. can be obtained by observing the emergence of new leaves on marked shoots, or by placing a tag around the most recent internode and measuring the number of internodes produced after a set time interval. In southern NSW estuaries, this technique was first used by West (1980) to measure rhizome and shoot production in Botany Bay.
Horizontal rhizome growth can be calculated by dividing the length of the rhizome between two shoots, by the difference in ages of the two shoots (Patriquin 1973 in Duarte et al. 1994), or alternatively by regressing the number of internodes between 2 shoots against their age difference. Similarly, vertical growth can be calculated by regressing the length of the short shoots against shoot age.

Seagrass colonisation rates may be estimated using reconstruction techniques. Since seagrass beds expand by horizontal rhizome growth, patch expansion is dictated by the maximum rate of rhizome elongation (which can be estimated from P.I). Determination of age gradients through patches may show colonisation rates even more accurately, as it takes into account branching of rhizomes, and variations in growth rate according to changing environmental conditions (Duarte et al. 1994). Colonisation rates are of considerable value in estimating the time it may take damaged seagrass beds to recover.
CHAPTER 3. STATUS OF *POSIDONIA AUSTRALIS* IN SOUTHERN NSW ESTUARIES

3.1 Introduction

Remote sensing of seagrass communities using aerial photographs has been used to document the status and examine long term changes in seagrass cover in a number of estuaries in southern NSW (see Larkum and West 1990, Meehan 1997). These historical investigations have included estuaries subject to considerable anthropogenic impacts, such as Botany Bay near Sydney (West and Larkum 1990), as well as estuaries that are less disturbed, such as Merimbula Lake on the far south coast of NSW (Meehan 1997). These studies have been able to determine whether seagrass communities have decreased or increased over the last fifty years and, to a lesser extent, correlate these changes in seagrass area with anthropogenic and/or natural impacts.

The objective of this Chapter is to assess the present status of *P. australis* in southern NSW estuaries and examine whether large areas have been lost as a result of estuary and catchment development. Aerial photographs and a GIS have been used to map the area and distribution of *P. australis* and other seagrass species in selected estuaries in this region: Port Hacking, St Georges Basin, Wagonga Inlet and Bermagui River. These have been combined with a previous study by the author of Merimbula Lake and Pambula Lake (Meehan 1997) in order to assess the present status of *P. australis* in the region.

Where possible, any natural and anthropogenic impacts that may have caused a change in *P. australis* abundance and distribution have also been documented. These impacts include changes in foreshore landuse and estuary entrance conditions. By simultaneously documenting change in *P. australis* distribution and level of estuary and catchment development, it has been possible to correlate some of the natural and anthropogenic impacts with changes in *P. australis* distribution, thus identifying the most likely causes of change.
3.2 Methods

3.2.1 Digital acquisition and rectification of aerial photographs

Aerial photographs (see list in Appendix 1) were scanned at 300 dots per inch (dpi), resulting in pixel sizes that ranged from 0.5m to 3m. This resolution was chosen to provide an acceptable compromise between image detail and image file size. Images were saved in 24-bit Tagged Image File Format (TIFF).

Georectification was carried out with DIMPLE software (Process software). In the case of St Georges Basin, Wagonga Inlet and Bermagui River, the Australian Map Grid (AMG) coordinates for a number of Ground Control Points (GCP’s) were read from a 1:25 000 topographic map, accurate to ± 12.5m. In the case of Port Hacking, coordinates were read from a Digital Control Model supplied by Sutherland Shire Council, accurate to ± 1m. Between 6 and 10 GCP’s per image were used in the GCP model. GCP’s with high Root Mean Square (RMS) errors (>15m) were removed from the model and the remaining 4 -12 ground control points were used to rectify each image using a linear algorithm. Where possible a minimum of six GCP’s were used to rectify each image, although on some of the older photographs only four GCP’s could be used, due to the absence of prominent features. A discussion of the errors associated with rectified images and the implications this has for the accuracy of seagrass mapping has been included (see Appendix 2).

3.2.2 Estimating the degree of foreshore and waterway development

Foreshore development (for this study, defined as land within 100 m of the high tide mark) was assessed by mapping the area of developed land in each time series of photographs. Foreshore development was assessed by examining changes in the percentage of foreshore classed as Vegetated, Cleared, Agricultural, Residential and Industrial. These categories were used because the distribution and long term changes in each of these land use categories has the capacity to influence seagrass distribution. For example, activities associated with agricultural and residential land may increase
the supply of nutrients to the estuary, while cleared land may increase sediment supply. Waterway development was assessed by examining large scale engineering works, particularly changes to entrance channels, as well as the construction of marinas, jetties and oyster leases.

3.2.3 Seagrass mapping

To facilitate analyses and discussion of results, each estuary was divided into the three geomorphic zones according to Roy et al. (2001): the Marine Tidal Delta (MTD), the Central Mud Basin (CMB) and the Fluvial Delta (FD) (Figs. 3.1 to 3.4). The characteristics of these zones were discussed previously (Section 1.6.4). The delineation of these zones provides a method of dividing estuaries into smaller units. These units have proved to be better than focusing on individual bays, or dividing the estuary into northern and southern shorelines, as these geomorphic units are consistent between estuaries. Also, some general assumptions can be made about their characteristics in terms of substrate, salinity and wave action. This is important for many estuaries in southern NSW, as little information is available.

Visual interpretation was used to map and discriminate between *P. australis* and other seagrass species on the basis of colour, tone, texture, environment and depth. For example, the three major groups of seagrass in southern NSW (*P. australis*, *Zostera* spp. and *Halophila* spp.) often have distinct differences in colour and tone on aerial photographs (Table 3.1). In cases where aerial photographs were compromised by solar reflectance and/or insufficient light penetration, mapping was not attempted.

For the most recent photographs, *in situ* field surveys were used to identify species and confirm visual interpretation of the aerial photographs. Previous maps, such as West et al. (1985), were used to aid interpretation of older photographs. While *P. australis* communities were of primary interest, other seagrass species were also mapped. However, it was not possible to discriminate between different species of *Zostera, Halophila* and *Ruppia* on the aerial photographs, and so these were only mapped to the genus level. A maximum of eight categories of seagrass were identified in the aerial photographs:
1) *P. australis*.
2) *P. australis* and *Zostera* spp.
3) *P. australis*, *Zostera* spp. and *Halophila* spp.,
4) *Zostera* spp.
5) Sparse *Zostera* spp.
6) *Zostera* spp. and *Halophila* spp.
7) *Zostera* spp., *Halophila* spp., and *Ruppia* spp.
8) *Ruppia* spp.

In cases where the species could not be identified, or where the transition from one species to another could not be clearly delineated, a default category of mixed seagrass was used. For example, in the MTD of Bermagui River *P. australis* formed discrete beds within the *Zostera* meadows. However, this transition could not be delineated from the aerial photographs so the polygons were classified as Category 2: *P. australis* and *Zostera* spp. Situations like this also occurred in Port Hacking, Wagonga Inlet and Bermagui River.

Distribution of seagrasses were mapped manually by the author. Numerous polygons were traced using a mouse and computer screen at a scale of 1: 2000. At this scale individual pixels were able to be seen, which was considered necessary to accurately locate the boundaries of the seagrass communities. Two simple control rules were used to quantify seagrass habitats, both of which would impact on overall areas calculated. Firstly, isolated patches less than 15m² were not mapped. Secondly, clusters of patches were mapped as separate meadows only when the distance between one meadow and the other was greater than 10m.
Table 3.1. Observable characteristics of *Posidonia australis*, *Zostera* spp. and *Halophila* spp., as seen in visual interpretation of aerial photographs of estuaries in southern NSW.

<table>
<thead>
<tr>
<th>Community Type</th>
<th>Observable Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Posidonia australis</em></td>
<td>Colour range black to dark green, though may appear as a light grey on BW photos. Colour varies depending on depth, but usually lighter than <em>Zostera</em>. Beds usually have soft edges and smooth appearance with substrate completely concealed. Communities usually continuous, unless stressed. Generally grows sub-tidally (minimum depth 0.2m)</td>
</tr>
<tr>
<td><em>Zostera</em> spp.</td>
<td>Darker beds, usually black, often smaller and fragmented. Sparse beds may be lighter in appearance with substrate often visible. Can grow in extreme environments, including channels and storm impacted areas. Often found in intertidal areas near <em>P. australis</em>, or on deep edge of <em>P. australis</em> bed.</td>
</tr>
<tr>
<td><em>Halophila</em> spp.</td>
<td>Often difficult to detect due to sparse growth and low canopy cover. Usually light grey or green/brown in appearance, forming ghostlike deposits in intertidal areas, often with <em>Zostera</em>. Communities more often continuous than discrete beds.</td>
</tr>
</tbody>
</table>
Figure 3.1. Port Hacking, NSW, showing the approximate boundaries of the Marine Tidal Delta (MTD), Central Mud Basin (CMB) and Fluvial Delta (FD), and localities referred to in study (see Fig 1.1 for locality map).
Figure 3.2. St Georges Basin, NSW, showing the approximate boundaries of the Marine Tidal Delta (MTD), Central Mud Basin (CMB) and Fluvial Delta (FD), and localities referred to in study (see Fig 1.1 for locality map).
Figure 3.3. Wagonga Inlet, NSW, showing the approximate boundaries of the Marine Tidal Delta (MTD), Central Mud Basin (CMB) and Fluvial Delta (FD), and localities referred to in thesis (see Fig 1.1 for locality map).
Figure 3.4. Bermagui River, NSW, showing the approximate boundaries of the Marine Tidal Delta (MTD), Central Mud Basin (CMB) and Fluvial Delta (FD), and localities referred to in thesis (see Fig 1.1 for locality map).
Digitising accuracy was considered to be ± 3m on well defined boundaries and ± 10m along poorly defined boundaries. To assess intra-operator error, a number of areas in Port Hacking were mapped three times and the resulting values compared. An example of one such analysis is shown in Table 3.2. The small standards errors in this analysis indicate the consistency of visual interpretation.

3.3 Results

The results of this study of long term changes in seagrass areas in Port Hacking, St Georges Basin, Wagonga Inlet and Bermagui River are presented in Section 3.3.1 to 3.3.4 respectively. A summary of findings can be found in Section 3.3.5.

3.3.1 Port Hacking

Degree of estuary and foreshore development

Port Hacking has the most highly urbanised foreshore of the four estuaries studied, with almost the entire northern shoreline occupied by high density residential land. The southern shoreline is relatively undeveloped, as most of it falls within the boundaries of the Royal National Park (RNP). Over the last fifty years, the major change to foreshore landuse in Port Hacking has been the conversion of low density, newly subdivided land to a high density urban landscape (Table 3.3). As noted, this occurred primarily on the northern shore, such that there are now only a few remnant patches of vegetation at the heads of Burraneer Bay and Yowie Bay. The majority of the southern shoreline has remained in the preserve of the RNP and Maianbar was the only new development to occur on the southern shore between 1951 and 1999. As well as the clearing of steeply sloped land to make way for housing at Maianbar, a large water pipeline was constructed from Bundeena to Maianbar (near Fisherman’s Bay) across Cabbage Tree Basin. Vegetated land through Port Hacking decreased slightly to make way for urbanisation; presently 44% of the foreshore is vegetated (Table 3.3). Land classified as Agriculture or Industry was limited in 1951 and has not significantly changed.
Table 3.2. Area of seagrass communities on the eastern shore of Gunnamatta Bay, Port Hacking, as interpreted by same operator on three separate occasions from the 1999 aerial photographs.

<table>
<thead>
<tr>
<th>Gunnamatta Bay</th>
<th>Replicate 1 (ha)</th>
<th>Replicate 2 (ha)</th>
<th>Replicate 3 (ha)</th>
<th>Mean ± S.E (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eastern shore P. australis</td>
<td>0.35</td>
<td>0.43</td>
<td>0.40</td>
<td>0.39 ± 0.02</td>
</tr>
<tr>
<td>Zostera spp.</td>
<td>0.10</td>
<td>0.09</td>
<td>0.10</td>
<td>0.09 ± 0.00</td>
</tr>
<tr>
<td>Zostera spp., Halophila spp. and P. australis</td>
<td>13.83</td>
<td>13.2</td>
<td>15.4</td>
<td>14.14 ± 0.65</td>
</tr>
<tr>
<td>Total seagrass</td>
<td>14.36</td>
<td>13.72</td>
<td>15.9</td>
<td>14.66 ± 0.64</td>
</tr>
</tbody>
</table>
Table 3.3. Percentage of foreshore occupied by Vegetated, Cleared, Agriculture, Residential and Industrial Land in Port Hacking, St Georges Basin, Wagonga Inlet and Bermagui River. Residential Class A refers is scattered housing, Residential Class B is dense housing.

<table>
<thead>
<tr>
<th>Estuary</th>
<th>Vegetated</th>
<th>Cleared</th>
<th>Agriculture</th>
<th>Residential Class A</th>
<th>Residential Class B</th>
<th>Industrial</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>%</td>
<td>%</td>
<td>%</td>
<td></td>
<td></td>
<td>%</td>
</tr>
<tr>
<td>Port Hacking</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1951</td>
<td>51</td>
<td>2</td>
<td>0</td>
<td>45</td>
<td>2</td>
<td>0</td>
</tr>
<tr>
<td>1999</td>
<td>44</td>
<td>12</td>
<td>0</td>
<td>3</td>
<td>41</td>
<td>0</td>
</tr>
<tr>
<td>St Georges Basin</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1961</td>
<td>70</td>
<td>4</td>
<td>0</td>
<td>26</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1998</td>
<td>65</td>
<td>0</td>
<td>0</td>
<td>4</td>
<td>31</td>
<td>0</td>
</tr>
<tr>
<td>Wagonga Inlet</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1957</td>
<td>63</td>
<td>0</td>
<td>23</td>
<td>14</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1994</td>
<td>48</td>
<td>18</td>
<td>15</td>
<td>6</td>
<td>13</td>
<td>1</td>
</tr>
<tr>
<td>Bermagui River</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1957</td>
<td>72</td>
<td>0</td>
<td>16</td>
<td>12</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1998</td>
<td>66</td>
<td>2</td>
<td>20</td>
<td>12</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>
Long term changes in the area and distribution of *Posidonia australis* and other seagrass species

Significant loss of seagrass occurred in the MTD, CMB and FD zones of Port Hacking from 1951 to 1999 (Table 3.4; Fig. 3.5 - 3.9). Overall, seagrass cover declined by 54%, from 181.1 ha (1951) to 83.6 ha (1999) (Table 3.4). Seagrass cover reached its lowest point in 1975 (74.1 ha), increased slightly in 1985, and declined once again in 1999, though not to the same level as 1975 (Table 3.4). The highest percentage loss was in the FD zone (79%), followed by the MTD zone (51%) and the CMB zone (50%).

Relative loss of seagrass was smallest for *P. australis* communities and largest for *Zostera* dominated communities. *P. australis* communities declined by 18%, from 38.9 ha (1951) to 31.9 ha (1999) (Table 3.4). This loss occurred entirely in the MTD zone, particularly between 1951 and 1975, although further losses occurred from 1975 to 1999 (Fig 3.5 - 3.9). *P. australis* fluctuated in the CMB zone but overall did not change significantly.

Large areas of *Zostera* communities and mixed seagrass have been lost throughout the Port. *Zostera* communities declined by 56%, from 83.9 ha (1951) to 36.4 ha (1999), and mixed *Zostera*, *Halophila* and *P. australis* communities declined by 75%, from 35.1 ha (1951) to 8.9 ha (1999). As well as declining to a greater extent, *Zostera* communities also exhibited greater variability than *P. australis* communities. For example, *Zostera* communities in the MTD declined from 43.5 ha in 1961 to 14.3 ha in 1975, but then increased to 39.2 ha in 1985. In the same period, *P. australis* firstly declined then remained stable.
Table 3.4. Area occupied by *Posidonia australis* and other seagrass communities in the Marine Tidal Delta zone, Central Mud Basin zone and Fluvial Delta zone of Port Hacking between 1951 and 1999, as interpreted from aerial photographs.

<table>
<thead>
<tr>
<th>REGION</th>
<th>1951 (ha)</th>
<th>1961 (ha)</th>
<th>1975 (ha)</th>
<th>1985 (ha)</th>
<th>1999 (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marine Tidal Delta zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>34.7</td>
<td>40.2</td>
<td>28.7</td>
<td>29.4</td>
<td>25.9</td>
</tr>
<tr>
<td><em>Zostera</em> spp:</td>
<td>64.4</td>
<td>43.5</td>
<td>14.3</td>
<td>39.2</td>
<td>31.9</td>
</tr>
<tr>
<td>Sparse <em>Zostera</em> spp:</td>
<td>11.1</td>
<td>20.4</td>
<td>1.0</td>
<td>0</td>
<td>3.8</td>
</tr>
<tr>
<td><em>Zostera</em> spp. &amp; <em>P. australis</em>:</td>
<td>5.6</td>
<td>5.4</td>
<td>6.0</td>
<td>0.9</td>
<td>2.6</td>
</tr>
<tr>
<td><em>P. australis</em>, <em>Zostera</em> spp. &amp; <em>Halophila</em>. Spp:</td>
<td>34.5</td>
<td>8.6</td>
<td>11.2</td>
<td>13.8</td>
<td>8.9</td>
</tr>
<tr>
<td>Subtotal:</td>
<td>150.3</td>
<td>118.1</td>
<td>61.1</td>
<td>82.2</td>
<td>73.1</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>REGION</th>
<th>1951 (ha)</th>
<th>1961 (ha)</th>
<th>1975 (ha)</th>
<th>1985 (ha)</th>
<th>1999 (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Central Mud Basin zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>6.5</td>
<td>9.3</td>
<td>5.2</td>
<td>7.5</td>
<td>6.8</td>
</tr>
<tr>
<td><em>Zostera</em> spp:</td>
<td>2.1</td>
<td>2.0</td>
<td>0.9</td>
<td>1.3</td>
<td>0.9</td>
</tr>
<tr>
<td><em>Zostera</em> spp. &amp; <em>P. australis</em>:</td>
<td>6.1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><em>P. australis</em>, <em>Zostera</em> spp. &amp; <em>Halophila</em> spp:</td>
<td>0.6</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Subtotal:</td>
<td>15.3</td>
<td>11.3</td>
<td>6.1</td>
<td>8.8</td>
<td>7.7</td>
</tr>
</tbody>
</table>
Table 3.4 continued,

<table>
<thead>
<tr>
<th>REGION</th>
<th>1951 (ha)</th>
<th>1961 (ha)</th>
<th>1975 (ha)</th>
<th>1985 (ha)</th>
<th>1999 (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Fluvial Delta zone</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Zostera spp.</em></td>
<td>17.9</td>
<td>11.1</td>
<td>7.8</td>
<td>8.8</td>
<td>3.7</td>
</tr>
<tr>
<td><strong>Subtotal:</strong></td>
<td>17.9</td>
<td>11.1</td>
<td>7.8</td>
<td>8.8</td>
<td>3.7</td>
</tr>
<tr>
<td><strong>Combined areas</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis:</em></td>
<td>41.2</td>
<td>49.5</td>
<td>32.9</td>
<td>35.8</td>
<td>32.7</td>
</tr>
<tr>
<td><em>Zostera spp.:</em></td>
<td>83.9</td>
<td>56.6</td>
<td>23.0</td>
<td>49.2</td>
<td>36.4</td>
</tr>
<tr>
<td><em>Sparse Zostera spp.:</em></td>
<td>11.5</td>
<td>20.4</td>
<td>1.0</td>
<td>0</td>
<td>3.8</td>
</tr>
<tr>
<td><em>Zostera spp. &amp; P. australis:</em></td>
<td>11.7</td>
<td>5.4</td>
<td>6.0</td>
<td>0.9</td>
<td>2.6</td>
</tr>
<tr>
<td><em>P. australis, Zostera spp.</em> &amp; <em>Halophila spp.:</em></td>
<td>35.1</td>
<td>8.6</td>
<td>11.2</td>
<td>13.8</td>
<td>8.9</td>
</tr>
<tr>
<td><strong>Total:</strong></td>
<td>183.4</td>
<td>140.5</td>
<td>74.1</td>
<td>99.7</td>
<td>84.4</td>
</tr>
</tbody>
</table>
Figure 3.5. Distribution of *Posidonia australis* and other seagrass species in Port Hacking in 1951.
Figure 3.6. Distribution of *Posidonia australis* and other seagrass species in Port Hacking in 1961.
Figure 3.7. Distribution of *Posidonia australis* and other seagrass species in Port Hacking in 1975.
Figure 3.8. Distribution of *Posidonia australis* and other seagrass species in Port Hacking in 1985.
Figure 3.9. Distribution of *Posidonia australis* and other seagrass species in Port Hacking in 1999.
Recently, *Zostera* spp. has also come under threat by the arrival of an invasive exotic algae, *Caulerpa taxifolia*, at several locations throughout the Port (Appendix 3). This noxious pest has already out-competition considerable areas of *Zostera* spp. in Gunnamatta Bay and Fisherman’s Bay, and has since spread to at least three other locations. The reason for its sudden increase in range is unclear, but *C. taxifolia* is a rapid coloniser of sheltered waters and has been reported to overgrow seagrasses in the Mediterranean Sea (Meinesz 1999). Plans are being developed to monitor its spread within Port Hacking (R. Creese, NSWF, pers. comm., 2001).

**Correlation of changes in seagrass distribution and estuary/foreshore development**

Direct modification to Port Hacking’s seabed appears to have had the greatest impact on seagrass distribution. Over the last century extensive dredging activities have been undertaken in Port Hacking, in order to mine shellgrit, establish a fish hatchery at the entrance to Cabbage Tree Basin, and maintain navigation channels. Shell grit mining occurred from 1928 to 1973 in a small section of the MTD zone (Druery and Hurrell 1986). The boundaries of the mining lease coincide perfectly with an area of the MTD that has lost considerable amounts of seagrass. It is therefore reasonable to propose that this mining caused the loss of approximately 40 ha of seagrass (mainly *Zostera* spp.) between 1951 and 1975.

Port Hacking has also been dredged for other reasons. Since dredging began, it is estimated that 200 000 m$^3$ of sand has been removed from the main channel to improve navigation (Druery and Hurrell 1986). This dredging appears to have caused a small loss of *P. australis* between 1951 and 1961. While elsewhere there are no obvious direct impacts of channel dredging, the mobilisation of sand and alteration to the substratum during dredging may have increased turbidity, which would not favour seagrass growth.

One indirect impact of dredging is that the channel dredge sands have been used to augment Deeban Spit, a large sandspit on the southern shoreline (see Fig 3.1). The evolution of Deeban Spit is a complex situation, involving both natural forces and engineering activities. In 1901, about one third of a million tonnes of sand was
dredged from Simpsons Bay to create access to a fish hatchery at Cabbage Tree Basin (Druery and Hurrell 1986). This sand was dumped onto a nearby shoal. By 1942, sand had moved west from this shoal and contributed to a northern and southern section of Deeban Spit, split in two by a southern channel from Cabbage Tree Basin. By 1961 this channel had closed and Deeban Spit was permanently connected to the eastern shore of Cabbage Tree Basin. At this time, it achieved its present day features, but has since been augmented and extended by further addition of dredge material. These modifications, in combination with the installation of a water pipeline to Maianbar, appear to have caused large scale sand migration into Cabbage Tree Basin, resulting in the near complete loss of seagrass (and subsequent expansion of mangroves). The augmentation of Deeban Spit with channel dredge sands may also have forced the main east-west channel across the mouth of Burraneer Bay to migrate into the Bay. This has resulted in sedimentation along the previously stable dropover and the erosion of *P. australis* beds (Druery and Hurrell 1986).

Many other factors have impacted on seagrass health in Port Hacking, including natural impacts. For example, a number of severe storms have occurred along the coast of New South Wales over the last seventy years (Table 3.5). These storms were particularly severe in 1974 and 1975 (Druery and Hurrell 1986). In Port Hacking, large-scale seagrass loss occurred at two exposed zones, the entrance and the southern end of Gunnamatta Bay, between 1961 and 1975, with extensive recovery at the former and only a small recovery at the latter (see Appendix 4 for seagrass changes in the entrance and Gunnamatta Bay). Seagrass species growing in these locations, including fragmented *P. australis*, is presumably in a state of dynamic equilibrium, with species composition and cover varying in relation to storm intensity and recovery time.

One natural process has also affected seagrass cover: the upstream progradation of the marine delta. Since 1930 the shoal at Lilli Pilli has been built up as tidal currents moved sand northward from the lower estuary (Druery and Hurrell 1986). This movement, in combination with channel dredging, may have been responsible for the demise of mixed *P. australis, Zostera* and *Halophila* beds in this area.
Table 3.5. **Summary of large storms along the central coast of NSW.** Category “A” storms are defined as those with a significant wave height (Hs) of between 5-6 m. Category “X” storms have a significant wave height (Hs) >6 m. Intervals constructed to correspond with dates of aerial photos used in this study. Source: NSW Public Works Department, Coast and Flood Branch (1994).

<table>
<thead>
<tr>
<th>Date intervals</th>
<th>Number of storms</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>From</td>
</tr>
<tr>
<td>2/?/1930</td>
<td>12/5/1942</td>
</tr>
<tr>
<td>13/5/1942</td>
<td>12/5/1951</td>
</tr>
<tr>
<td>12/5/1951</td>
<td>25/6/1961</td>
</tr>
</tbody>
</table>

**Total** 78 27 105
In Port Hacking, there is only circumstantial evidence that increased turbidity and nutrients caused the loss of *P. australis* and other seagrasses. In Fisherman’s Bay, which is located in the MTD zone of Port Hacking, the depth limit of *P. australis* appeared to decrease as residential development in Maianbar increased. Intertidal *Zostera* communities also disappeared during this time, possibly due to nearshore sedimentation. In the Hacking River, *Zostera* communities declined by 81% (13 ha) between 1951 and 1999, possibly as a result of increased nutrients and sediment from the expansion of housing development at Grays Point. The expansion of mangroves in this area, from 12 ha in 1951 to 15 ha in 1999 (Williams and Meehan 2001) also supports this hypothesis, as mangrove expansion is thought to result from increased sediment and nutrients (McLoughlin 1985). However, Port Hacking’s small size and high degree of tidal flushing appears to have limited the possible impacts of increased nutrients and sediment on seagrass area to these two areas.

### 3.3.2 St Georges Basin

*Degree of estuary and foreshore development*

Between 1961 and 1998, St Georges Basin underwent a similar style of development to Port Hacking, with all low density subdivided land in 1961 becoming higher density residential areas by 1998 (Table 3.3). Two new housing developments also occurred, at Bream Beach and at the southern end of Sussex Inlet. These two developments resulted in a 9% increase in foreshore residential development. Presently, residential development is restricted to two main areas: the northern shore of the Basin and the eastern shore of Sussex Inlet. The southern shoreline of the Basin and Western shoreline of Sussex Inlet lie adjacent Commonwealth Territory, while the western shoreline of the Basin is undeveloped crown land. There was no foreshore industry or agriculture in either 1961 or 1998.

*Long term changes in the area and distribution of *Posidonia australis* and other seagrass species*
Significant loss of seagrass occurred in the CMB zone of St Georges Basin, but not in the MTD zone or FD zone (Fig. 3.10 - 3.13). Overall, seagrass cover declined by 23%, from 377.9 ha (1961) to 292 ha (1998) (Table 3.6). The CMB zone lost 27% (79.7 ha) of its seagrass communities in two main periods, 1961-1970, and 1979 - 1998 (Table 3.6). Seagrass cover in the MTD zone declined by 9% but increased by 34% (5.0 ha) in the FD zone. However, there was no obvious trend in both the MTD zone and FD zone. The FD zone was highly variable in seagrass cover.

Like Port Hacking, the smallest relative change occurred in pure stands of *P. australis*, which declined by 13% (12.7ha) overall. Decrease in the cover of *P. australis* was restricted to the CMB zone between the years 1979 and 1998 (Table 3.6). Shallow water *P. australis* beds fragmented and declined on the southern shore of this zone (Fig. 3.10 - 3.13). Prior to this, *P. australis* communities were stable, ranging from 52 to 67 ha (though the latter figure was probably related to different classification). Similarly, *P. australis* and *P. australis*/Zostera communities in the MTD zone were also stable, ranging from about 51 to 59 ha. Unlike the CMB zone, however, no significant decline occurred, making the large scale loss of *P. australis* in the CMB zone easy to detect.

It seems likely that pure stands of *P. australis* occurred on the northern shore of the CMB in 1957, 1970 and 1979, but they were unable to be mapped consistently (though they were in 1998, where field investigation allowed species to be verified). A default category of mixed seagrass was used to classify the seagrasses in these areas. These mixed communities of *P. australis*, Zostera spp. and Halophila spp. in the CMB declined over all time intervals. In JewFish Bay (see Fig 3.2), and the area between Island Point and Paradise Beach, this involved the complete loss of seagrass cover from the shallow water shoals (Fig 3.12 and 3.13). This is represented by a total loss of 147 ha of *P. australis*/Zostera/Halophila. In other areas, such as between Sanctuary Point and Tomerong Creek Delta, *P. australis* declined but Zostera spp. and Halophila spp. increased. Losses of mixed *P. australis* communities were replaced by a 53 ha in increase in Zostera/Halophila and a 17 ha increase in Zostera spp. (Table 3.6).
Table 3.6. Area occupied by *Posidonia australis* and other seagrass communities in the Marine Tidal Delta zone, Central Mud Basin zone and Fluvial Delta zone of St Georges Basin between 1961 and 1998, as interpreted from aerial photographs.

<table>
<thead>
<tr>
<th>ZONE</th>
<th>1961 (ha)</th>
<th>1970 (ha)</th>
<th>1979 (ha)</th>
<th>1999 (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marine Tidal Delta zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>45.1</td>
<td>42.0</td>
<td>46.7</td>
<td>51.3</td>
</tr>
<tr>
<td><em>Zostera</em> spp.:</td>
<td>4.8</td>
<td>5.8</td>
<td>3.6</td>
<td>5.7</td>
</tr>
<tr>
<td><em>Zostera</em> spp. &amp; <em>P. australis</em></td>
<td>13.8</td>
<td>9.7</td>
<td>11.4</td>
<td>1.2</td>
</tr>
<tr>
<td>Subtotal:</td>
<td>63.7</td>
<td>57.5</td>
<td>61.7</td>
<td>58.2</td>
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<tr>
<td>Central Mud Basin zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>51.9</td>
<td>53.4</td>
<td>66.8</td>
<td>33.0</td>
</tr>
<tr>
<td><em>Zostera</em> spp.:</td>
<td>21.7</td>
<td>19.9</td>
<td>24.6</td>
<td>38.9</td>
</tr>
<tr>
<td><em>Zostera</em> spp. &amp; <em>Halophila</em>.</td>
<td>14.7</td>
<td>8.1</td>
<td>26.5</td>
<td>68.3</td>
</tr>
<tr>
<td>Spp:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Zostera</em> spp., <em>Halophila</em> spp.</td>
<td>174.9</td>
<td>123.2</td>
<td>109.9</td>
<td>28.3</td>
</tr>
<tr>
<td>and <em>P. australis</em>:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Zostera</em> spp., <em>Halophila</em> spp. &amp; <em>Ruppia</em> spp.:</td>
<td>33.4</td>
<td>55.7</td>
<td>47.3</td>
<td>29.4</td>
</tr>
<tr>
<td><em>Ruppia</em> spp:</td>
<td>3.1</td>
<td>2.6</td>
<td>2.5</td>
<td>19.9</td>
</tr>
<tr>
<td>Subtotal:</td>
<td>299.7</td>
<td>262.9</td>
<td>277.1</td>
<td>220.0</td>
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Table 3.6 continued,

<table>
<thead>
<tr>
<th>ZONE</th>
<th>1961 (ha)</th>
<th>1970 (ha)</th>
<th>1979 (ha)</th>
<th>1998 (ha)</th>
</tr>
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<tr>
<td>Fluvial Delta zone</td>
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<tr>
<td>Zostera spp.:</td>
<td>14.5</td>
<td>21.1</td>
<td>16.4</td>
<td>19.5</td>
</tr>
<tr>
<td>Zostera spp. &amp; Halophila.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spp:</td>
<td>0</td>
<td>1.9</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Subtotal:</td>
<td>14.5</td>
<td>23.0</td>
<td>16.4</td>
<td>19.5</td>
</tr>
<tr>
<td>Combined areas</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P. australis:</td>
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<td>95.4</td>
<td>113.5</td>
<td>84.3</td>
</tr>
<tr>
<td>Zostera spp:</td>
<td>41.0</td>
<td>46.8</td>
<td>44.6</td>
<td>61.8</td>
</tr>
<tr>
<td>Zostera spp. &amp; Halophila.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spp:</td>
<td>14.7</td>
<td>10</td>
<td>26.5</td>
<td>68.3</td>
</tr>
<tr>
<td>Zostera spp., Halophila spp.</td>
<td>188.7</td>
<td>132.9</td>
<td>117.3</td>
<td>28.3</td>
</tr>
<tr>
<td>and P. australis:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zostera spp., Halophila spp.</td>
<td>33.4</td>
<td>55.7</td>
<td>47.3</td>
<td>29.4</td>
</tr>
<tr>
<td>and Ruppia spp.</td>
<td>3.1</td>
<td>2.6</td>
<td>2.5</td>
<td>19.9</td>
</tr>
<tr>
<td>Ruppia spp.:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total:</td>
<td>377.9</td>
<td>343.4</td>
<td>351.7</td>
<td>292.0</td>
</tr>
</tbody>
</table>
Table 3.7. Area occupied by *Posidonia australis* and other seagrass communities in the Marine Tidal Delta zone, Central Mud Basin zone and Fluvial Delta zone of Wagonga Inlet between 1957 and 1994, as interpreted from aerial photographs.

<table>
<thead>
<tr>
<th>REGION</th>
<th>1957 (ha)</th>
<th>1972 (ha)</th>
<th>1977 (ha)</th>
<th>1982 (ha)</th>
<th>1986 (ha)</th>
<th>1994 (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marine Tidal Delta zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>12.6</td>
<td>14.8</td>
<td>16.7</td>
<td>15.6</td>
<td>14.6</td>
<td>13.7</td>
</tr>
<tr>
<td>Zostera spp. &amp; Halophila spp:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis, Zostera spp.</em> &amp; Halophila spp.</td>
<td>11.3</td>
<td>4.7</td>
<td>5.9</td>
<td>5.0</td>
<td>8.1</td>
<td>5.7</td>
</tr>
<tr>
<td>Subtotal:</td>
<td>39.2</td>
<td>33.7</td>
<td>39.4</td>
<td>36.7</td>
<td>25.7</td>
<td>19.8</td>
</tr>
<tr>
<td>Central Mud Basin zone</td>
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<td>n/a</td>
<td>n/a</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>0</td>
<td>15.4</td>
<td>25.0</td>
<td></td>
<td>39.5</td>
<td></td>
</tr>
<tr>
<td>Zostera spp. &amp; Halophila spp:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis, Zostera spp.</em> &amp; Halophila spp.</td>
<td>38.4</td>
<td>18.7</td>
<td>3.9</td>
<td>1.3</td>
<td></td>
<td></td>
</tr>
<tr>
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<td>38.4</td>
<td>47.0</td>
<td>43.3</td>
<td>53.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fluvial Delta zone</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zostera spp. &amp; Halophila spp:</td>
<td>5.9</td>
<td>4.8</td>
<td>2.3</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Subtotal:</td>
<td>5.9</td>
<td>4.8</td>
<td>2.3</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure 3.10. Distribution of *Posidonia australis* and other seagrass species in St Georges Basin in 1961.
Figure 3.11. Distribution of *Posidonia australis* and other seagrass species in St Georges Basin in 1970.
Figure 3.12. Distribution of *Posidonia australis* and other seagrass species in St Georges Basin in 1979.
Figure 3.13. Distribution of *Posidonia australis* and other seagrass species in St Georges Basin in 1998.
The area of *Ruppia* spp. communities increased sevenfold over the same period. As shown in Fig. 3.12 and 3.13, this increase was restricted entirely to Erowal Bay, which has been almost completely overtaken by *Ruppia* spp. since 1979. Field investigation in November 2000 also found sparse occurrences of *Ruppia* spp. growing amongst the *P. australis* beds on the southern shore of the CMB zone. These occurrences were too sparse to map, and would require a separate investigation. However, the co-existence of *P. australis* and *Ruppia* spp. is a highly unusual phenomena that has not been previously reported for estuaries in this area.

The cover of mixed *Zostera, Halophila, and Ruppia* spp. did not decrease or increase overall, although there were large variations between years.

**Correlation of changes in seagrass distribution and estuary/foreshore development**

In St Georges Basin, correlating changes in seagrass area and distribution with anthropogenic and natural impacts was difficult. However, the large increase in *Ruppia* spp. in Erowal Bay, and the presence of *Ruppia* spp. amongst fragmented *P. australis* beds on the southern and northern shore of St Georges Basin, suggests that increased nutrients and increased freshwater runoff is a possible explanation of *P. australis* decline in the CMB of St Georges Basin. A number of studies (Lukatelich et al. 1987, Geddes 1987) have linked increases in *Ruppia* spp. to increased nutrients, and to increased freshwater inputs that cause highly saline water to become brackish. At present, water quality in St Georges Basin is ‘very good’ (Shoalhaven City Council 1998), but was considered to be poor prior to the installation of a sewerage reticulation system, which has eliminated septic system discharge. For example, in the period 1974 to 1978, the level of total nitrogen and phosphorus in the waters of the Basin were regarded as high (SCC 1998). Since *P. australis* favours high salinity, low nutrient, sandy environments, and is intolerant of freshwater (West 1980), *P. australis* communities would be expected to decline as *Ruppia* spp. increase.
Figure 3.14. The foreshore of St Georges Basin, showing the remains of *Posidonia australis* rhizome matt on the shallow water shoals near Sanctuary Point (top) and an isolated *Posidonia australis* bed amongst algae near Picnic Point (above).
Whatever the nature of the impact, it has had an unusual effect on existing seagrass beds which is difficult to explain. Instead of the regression of deep water beds, it is the shallow water *P. australis* communities that appear to have been most affected. For example, on the southern shore the deeper *P. australis* beds show less evidence of fragmentation than adjacent *P. australis* beds in shallow water.

The remnants of isolated *P. australis* plants and *P. australis* rhizome matt (see Fig. 3.14), suggest that *P. australis* once formed extensive beds around almost all the CMB zone, being particularly dominant in the shallow waters of the various bays. As the foreshore has been developed for residential land, *P. australis* communities between Island Point and Paradise Beach (see Fig. 3.2 for locality map) have been decimated and communities between Paradise Beach and Tomerong Creek Delta have declined in area. *P. australis* communities along the entire southern shore have also declined, despite the fact that there was no foreshore development in this area.

Examples of localised fragmentation and die off in the literature (Cambridge et al. 1986) have usually been associated with known source points of pollution, such as industrial effluent discharge. A number of possible scenarios may explain the case of St Georges Basin. For example, shallow water communities, being closer to nutrient and freshwater inputs, may have suffered the greatest effect. However, further research into the mechanics of pollutant dispersal in St Georges Basin would be required to explain this phenomena in any detail.

3.3.3 Wagonga Inlet

*Degree of estuary and foreshore development*

Between 1957 and 1994, foreshore development in Wagonga Inlet was less extensive than either in Port Hacking or St Georges Basin. Urban areas increased from 13% to 19%, with some increase in dense urban areas. Cleared land increased to 18% with the creation of a number of new subdivisions that are presently being developed (Table 3.3). This has involved clearing of vegetated land and a small amount of agricultural
land. Some industrial development occurred between 1957 and 1977, with an oyster farm being built on the northern shore in Barlows Bay (see Fig. 3.3 for locality map). The construction of oyster leases and processing buildings involved some dredging in the intertidal zone as well as foreshore alteration, but it impacted less than 1% of the foreshore.

Long term changes in the area and distribution of Posidonia australis and other seagrass species

In contrast to Port Hacking and St Georges Basin, seagrass area declined only slightly (10%) in Wagonga Inlet, from 83.5 ha (1957) to 75.1 ha (1994) (Table 3.6). The FD zone lost 3.6 ha (61%) and the MTD zone lost 19.4 ha (49%). In both cases, seagrass decrease occurred from 1982 onwards. Seagrass cover increased substantially in the CMB zone by 14.6 ha (38%), and with the exception of 1982, this increase was found to be progressive through all time periods.

P. australis area increased significantly in Wagonga Inlet, particularly in the CMB zone, where it increased from 15.4 ha to 39.5 ha. Some of this increase was due to a reclassification of some mixed P. australis communities to monospecific P. australis stands (Fig. 3.15 - 3.17). Nevertheless there was an overall increase in seagrass cover in the CMB zone, from 38.4 ha (1957) to 53 ha (1994) (Table 3.6). This increase was due mainly to the spread of P. australis into deeper waters, a process that appears to have begun between 1957 and 1977. In addition, communities which were once composed of all three species became more differentiated during the study period, resulting in a distinct banding between Zostera/Halophila beds and P. australis beds (Fig. 3.15 - 3.17). However, this may have also been due to an improvement in aerial photograph quality over time.

By contrast, Zostera and Halophila cover decreased over the same period. In the MTD zone, deeper communities dominated by a P. australis remained reasonably stable or increased, but intertidal communities dominated by sparse Zostera/Halophila beds declined overall by 14.9 ha (97%). This loss of Zostera/Halophila cover started in
1982 and, by 1994, these beds had almost completely disappeared from the area (Fig. 3.15-3.17). *Zostera* cover in the FD zone also decreased, again from 1982 onwards.

**Correlation of changes in seagrass distribution and estuary/foreshore development**

Between 1967 and 1972 two large groynes were constructed at the ocean entrance of Wagonga Inlet to prevent the tidal inlet channel from shoaling and shifting position (PWD 1988). Between 1976 and 1986 mean tidal range in the entrance increased from 0.6m to 0.92m, and mean tidal prism increased from $3.95 \times 10^6$ M$^3$ to $5.44 \times 10^3$ M$^3$ (PWD 1988). It is thought that these changes are responsible for the increase in *P. australis* cover in the CMB zone of Wagonga Inlet, and the decrease in *Zostera* and *Halophila* cover in the intertidal flats of the MTD zone.

The increase in tidal prism between 1976 and 1986 correlates with an increase in the amount of *P. australis* in the CMB zone, and also correlates with a more obvious separation between *Zostera* spp. and *P. australis* communities. It is hypothesised that the changes caused an increase in the amount of PAR available to *P. australis* plants on the deep edge of the meadow, allowing the *P. australis* bed to spread into deeper waters. The change from mixed *P. australis/Zostera* to a distinct *Zostera* and *P. australis* band may also have been facilitated by the entrance modifications, perhaps due to increased salinity which favoured *P. australis* over *Zostera* spp.

The demise of the sparse *Zostera* and *Halophila* beds may also be due to an increased tidal range, as this has resulted in greater exposure of the sand flats. At present, most of these sand flats lack any type of seagrass due to exposure (water level < 0.05m). Only an occasional patch of *Zostera* spp. remains. An increase in tidal range of 0.3m would have lowered the depth of the water level during low tides and may have led to desiccation of the previously extensive beds. However, a similar phenomena of shallow water *Zostera/Halophila* dieoff was documented in Merimbula Lake over the same time period, even though no large structural changes were made to the entrance (Meehan 1997).
Table 3.7. Area occupied by *Posidonia australis* and other seagrass communities in the Marine Tidal Delta zone, Central Mud Basin zone and Fluvial Delta zone of Wagonga Inlet between 1957 and 1994, as interpreted from aerial photographs.

<table>
<thead>
<tr>
<th>REGION</th>
<th>1957 (ha)</th>
<th>1972 (ha)</th>
<th>1977 (ha)</th>
<th>1982 (ha)</th>
<th>1986 (ha)</th>
<th>1994 (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marine Tidal Delta zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>12.6</td>
<td>14.8</td>
<td>16.7</td>
<td>15.6</td>
<td>14.6</td>
<td>13.7</td>
</tr>
<tr>
<td>Zostera spp. &amp;</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Halophila</em> spp:</td>
<td>15.3</td>
<td>14.2</td>
<td>16.8</td>
<td>16.1</td>
<td>3.0</td>
<td>0.4</td>
</tr>
<tr>
<td><em>P. australis, Zostera</em> spp.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>&amp; <em>Halophila</em> spp.</td>
<td>11.3</td>
<td>4.7</td>
<td>5.9</td>
<td>5.0</td>
<td>8.1</td>
<td>5.7</td>
</tr>
<tr>
<td>Subtotal:</td>
<td>39.2</td>
<td>33.7</td>
<td>39.4</td>
<td>36.7</td>
<td>25.7</td>
<td>19.8</td>
</tr>
<tr>
<td>Central Mud Basin zone</td>
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<td></td>
<td>n/a</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td>0</td>
<td>15.4</td>
<td>25.0</td>
<td></td>
<td></td>
<td>39.5</td>
</tr>
<tr>
<td>Zostera spp. &amp;</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Halophila</em> spp:</td>
<td>0</td>
<td>12.9</td>
<td>14.4</td>
<td></td>
<td></td>
<td>12.2</td>
</tr>
<tr>
<td><em>P. australis, Zostera</em> spp.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>&amp; <em>Halophila</em> spp.</td>
<td>38.4</td>
<td>18.7</td>
<td>3.9</td>
<td></td>
<td></td>
<td>1.3</td>
</tr>
<tr>
<td>Subtotal:</td>
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<td>47.0</td>
<td>43.3</td>
<td></td>
<td></td>
<td>53.0</td>
</tr>
<tr>
<td>Fluvial Delta zone</td>
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<td>n/a</td>
<td>n/a</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zostera spp. &amp;</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Halophila</em> spp:</td>
<td>5.9</td>
<td>4.8</td>
<td>2.3</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Subtotal:</td>
<td>5.9</td>
<td>4.8</td>
<td>2.3</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 3.7 continued,

<table>
<thead>
<tr>
<th>REGION</th>
<th>1957 (ha)</th>
<th>1972 (ha)</th>
<th>1977 (ha)</th>
<th>1982 (ha)</th>
<th>1986 (ha)</th>
<th>1999 (ha)</th>
</tr>
</thead>
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<tr>
<td>Combined areas</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis:</em></td>
<td>12.6</td>
<td></td>
<td></td>
<td>40.6</td>
<td></td>
<td>53.2</td>
</tr>
<tr>
<td><em>Zostera</em> spp. &amp;</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Halophila</em> spp:</td>
<td>21.2</td>
<td></td>
<td></td>
<td>35.3</td>
<td></td>
<td>14.9</td>
</tr>
<tr>
<td><em>P. australis, Zostera</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>spp.</em> &amp; <em>Halophila</em></td>
<td>49.7</td>
<td></td>
<td></td>
<td>8.9</td>
<td></td>
<td>7.0</td>
</tr>
<tr>
<td><em>spp.</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td>83.5</td>
<td>84.8</td>
<td></td>
<td></td>
<td></td>
<td>75.1</td>
</tr>
</tbody>
</table>
Figure 3.15. Distribution of *Posidonia australis* and other seagrass species in Wagonga Inlet in 1957.
Figure 3.16. Distribution of *Posidonia australis* and other seagrass species in Wagonga Inlet in 1982.
Figure 3.17. Distribution of *Posidonia australis* and other seagrass species in Wagonga Inlet in 1994.
3.3.4 Bermagui River

Degree of foreshore and estuary development

There was very little change to foreshore landuse in Bermagui River from 1957 to 1998. The percentage of the foreshore vegetated did not decrease appreciably, falling only 6% between 1957 and 1998 (Table 3.3). Agricultural land increased slightly from 16 to 20%, while residential land stayed constant at 12%. However, two major changes were made to the waterway; 1) the construction of Bermagui Harbour in the early 1990's; and 2) the addition of groynes to the entrance in the 1970's. These two changes are clearly visible on Figure 3.18 and 3.20. Approximately 12ha of land/intertidal flats was either dredged or filled to accommodate the harbour and surrounding amenities.

Long term changes in the area and distribution of Posidonia australis and other seagrass species

Significant loss of seagrass occurred in the MTD zone of Bermagui River, and to a lesser extent in the CMB zone (Figure 3.18 - 3.20). Overall, seagrass cover declined by 34% (14.8 ha), from 42.9 ha (1957) to 28.1 ha (1998). The MTD zone lost 13.9 ha (41%) of its seagrass cover and the CMB lost 0.8 ha (15%). Seagrass cover in the FD zone did decrease from 1957 to 1998 but the lowest cover occurred in 1979 (Table 3.7).

Monospecific stands *P. australis* were mapped only in the CMB zone, and decreased by 15% (0.8 ha), mainly between 1979 and 1998 (Table 3.7). While monospecific *P. australis* beds do exist in the MTD zone, they could not be differentiated from surrounding *Zostera* beds, and so were given the default category of mixed *P. australis/Zostera* spp. Overall, mixed *P. australis* and *Zostera* found in the MTD zone (Figure 3.18 - 3.20) declined by 30% (6.5 ha), mainly between 1957 and 1979. It appears likely that most of this loss was *Zostera* communities in shallow, mobile sand environments, but the quality of the photographs prevents further speculation.
Table 3.8. Area occupied by *Posidonia australis* and other seagrass communities in the Marine Tidal Delta zone, Central Mud Basin zone and Fluvial Delta zone of Bermagui River between 1957 and 1998, as interpreted from aerial photographs.

<table>
<thead>
<tr>
<th>REGION</th>
<th>YEAR</th>
<th>1957</th>
<th>1979</th>
<th>1998</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>(ha)</td>
<td>(ha)</td>
<td>(ha)</td>
</tr>
<tr>
<td>Marine Tidal Delta zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Zostera</em> spp:</td>
<td></td>
<td>11.6</td>
<td>4.1</td>
<td>5.6</td>
</tr>
<tr>
<td><em>P. australis</em> &amp; <em>Zostera</em></td>
<td></td>
<td>21.8</td>
<td>17.5</td>
<td>15.3</td>
</tr>
<tr>
<td>Total:</td>
<td></td>
<td>33.4</td>
<td>21.6</td>
<td>19.5</td>
</tr>
<tr>
<td>Central Mud Basin zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td></td>
<td>5.5</td>
<td>5.2</td>
<td>4.7</td>
</tr>
<tr>
<td><em>Zostera</em> spp:</td>
<td></td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><em>P. australis</em> &amp; <em>Zostera</em></td>
<td></td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Total:</td>
<td></td>
<td>5.5</td>
<td>5.2</td>
<td>4.7</td>
</tr>
<tr>
<td>Fluvial Delta zone</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Zostera</em> spp:</td>
<td></td>
<td>4.0</td>
<td>2.0</td>
<td>2.5</td>
</tr>
<tr>
<td>Total:</td>
<td></td>
<td>4.0</td>
<td>2.0</td>
<td>2.5</td>
</tr>
<tr>
<td>Combined areas</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. australis</em>:</td>
<td></td>
<td>5.5</td>
<td>5.2</td>
<td>4.7</td>
</tr>
<tr>
<td><em>Zostera</em> spp:</td>
<td></td>
<td>15.6</td>
<td>6.1</td>
<td>8.1</td>
</tr>
<tr>
<td><em>P. australis</em> &amp; <em>Zostera</em></td>
<td></td>
<td>21.8</td>
<td>17.5</td>
<td>15.3</td>
</tr>
<tr>
<td>Total:</td>
<td></td>
<td>42.9</td>
<td>28.8</td>
<td>28.1</td>
</tr>
</tbody>
</table>
Figure 3.18. Distribution of Posidonia australis and other seagrass species in Bermagui River in 1957.
Figure 3.19. Distribution of *Posidonia australis* and other seagrass species in Bermagui River in 1979.
Figure 3.20. Distribution of *Posidonia australis* and other seagrass species in Bermagui River in 1998.
Zostera communities suffered the greatest loss of cover, with 48% (7.5 ha) of the beds disappearing, mainly between 1957 and 1979. As illustrated by Fig. 3.18 and 3.19, this occurred near the entrance of the MTD zone. Zostera beds in the FD zone also decreased significantly between 1957 and 1979, but recovered in 1998.

_Correlation of changes in seagrass distribution and estuary/foreshore development_

Some of the loss of seagrass can be attributed to reclamation activities, but the majority is associated with the changes in sand distribution. The construction of Bermagui Harbour and the addition of two groynes to maintain the entrance channel may have been responsible for the decline in Zostera cover in the MTD. The construction of the outer harbour wall appears to have forced sand to migrate to the northern shore where it formed a large sand bar and buried extensive areas of Zostera beds. Changes in the flow characteristics may also have contributed to the loss of mixed _P. australis/_Zostera communities further up in the MTD.

3.3.5 _A summary of long term changes in the area and distribution of Posidonia australis in southern NSW estuaries_

Over the last forty to fifty years, five out the six estuaries in southern NSW have lost significant amounts of _P. australis_ (Table 3.9). Loss of _P. australis_ was small in Bermagui River, Merimbula Lake and Pambula Lake, moderate in Port Hacking and large in St Georges Basin (Table 3.9). Most of the estuaries also lost considerable areas of other seagrass species, with the exception of St Georges Basin, where _Zostera_ spp., _Halophila_ spp. and _Ruppia_ spp. all increased (Table 3.9). Wagonga Inlet was the only estuary where _P. australis_ increased. However, _Zostera_ spp. and _Halophila_ spp. declined as _P. australis_ increased, resulting in a small loss of seagrass overall.
Table 3.9. A summary of changes in the area of *Posidonia australis* and other seagrass species in Port Hacking, St Georges Basin, Wagonga Inlet, Bermagui River, Merimbula Lake and Pambula Lake in southern NSW. Key: - < 10% loss, - - 10 to 30% loss, - - - 30 to 50% loss, - - - - > 50% loss; + < 10% increase, ++ 10-30% increase, +++ 30 to 50% increase.

<table>
<thead>
<tr>
<th>Estuary</th>
<th>Change in <em>P. australis</em> communities</th>
<th>Change in other seagrass species</th>
<th>Likely cause of change</th>
</tr>
</thead>
<tbody>
<tr>
<td>Port Hacking</td>
<td>- - -</td>
<td>- - -</td>
<td>Shell grit mining, engineering works, storm damage and foreshore development</td>
</tr>
<tr>
<td>St Georges Basin</td>
<td>- - -</td>
<td>+ + +</td>
<td>Possibly due to increased nutrients and freshwater runoff</td>
</tr>
<tr>
<td>Wagonga Inlet</td>
<td>+ + +</td>
<td>-</td>
<td>Entrance modifications leading to an increase in tidal range and tidal prism</td>
</tr>
<tr>
<td>Bermagui River</td>
<td>-</td>
<td>- - -</td>
<td>Harbour construction</td>
</tr>
<tr>
<td>Merimbula Lake*</td>
<td>-</td>
<td>- - -</td>
<td>Foreshore development, oyster leases though possibly a natural fluctuation</td>
</tr>
<tr>
<td>Pambula Lake*</td>
<td>-</td>
<td>-</td>
<td>Possibly due to increased sedimentation</td>
</tr>
</tbody>
</table>

* data from Meehan (1997).
A variety of anthropogenic and natural impacts could be correlated with the changes in seagrass area and distribution in southern NSW estuaries (Table 3.9). Anthropogenic impacts appeared to cause a much greater loss of both *P. australis* and other seagrass species, while a natural impacts, such as storm damage and natural sand movement, were only observed to impact seagrass distribution in Port Hacking.

### 3.4 Discussion

Overall, the status of *P. australis* communities in southern NSW estuaries is poor, with significant amounts of *P. australis* communities lost from five of the six estuaries in this region. This consistent decline of seagrasses in a wide range of estuaries in southern NSW is of concern, as it illustrates once again the negative effects that estuary and catchment development have on seagrass area and distribution.

A variety of anthropogenic impacts have caused the declines in *P. australis* and other seagrass species documented in this study. While other long term studies have documented loss of seagrass in relation to a single process (for example, landuse, Bulthuis 1983; pollution, Cambridge and McComb 1984) or a single instant in time, such as a cyclone (Poiner et al. 1989) many of the estuaries in southern NSW represent a microcosm of processes operating over the last 50 years. For example, in Port Hacking, shell grit mining, engineering works and foreshore development appear to be responsible for the losses of *P. australis* and other seagrass species. In Wagonga Inlet and Bermagui River, entrance modifications appear to have had the most noticeable impact on seagrass abundance. In St Georges Basin, it was difficult to determine the cause of *P. australis* decline, but the increase in *Ruppia* spp. over the last forty years suggests that increased nutrients and freshwater inputs may be responsible for the loss, as *Ruppia* spp. is thought to increase under these conditions (Lukatelich et al. 1987, Geddes 1987).

Natural impacts have also been responsible for some of the declines in *P. australis* communities. For example, the exposed entrance of Port Hacking means that some seagrass communities in this estuary are susceptible to storm damage. Also, the
progradation of the marine tidal delta in Port Hacking appears to have caused a
decline in \textit{P. australis} distribution. Overall, the effects of natural impacts on the area
of \textit{P. australis} was much less than that of anthropogenic impacts. However, any
strategy designed to conserve and manage \textit{P. australis} needs to recognise that
significant changes to \textit{P. australis} area and distribution can be caused by natural
processes. For example, there would be little point in rehabilitating either the entrance
or the upstream seagrass beds in Port Hacking, if storm waves and natural sand
movement are likely to quickly destroy any transplanted seagrass. This point is further
investigated in Chapter Seven.

Aside from documenting the poor status of \textit{P. australis} in southern NSW estuaries, an
important outcome of this study is in regard to the complexity of long term change in
\textit{P. australis} and other seagrass species. Many studies throughout the world have
documented seagrass decline due to human population growth and accompanying
development activities (see Section 2.2.2). Along with observational and experimental
evidence (Abal and Dennison 1996, Dennison and Kirkman 1996, Moore 1996), two
hypotheses have emerged to explain cases of seagrass regression; 1) that nutrient and
sediment loading from anthropogenic inputs are a major factor in reducing water
quality and causing seagrass decline; and 2) that the reduction in incident light due to
deteriorating water quality causes a reduction in the depth limit of seagrasses. The
patterns of long term change in \textit{P. australis} communities documented in this study
verify that the depth limit of seagrass may respond to anthropogenic impacts.
However, many of the results of this study contradict these two views. In southern
NSW estuaries, the factors responsible for loss of \textit{P. australis} and other seagrass
species are more complex, and the reaction of \textit{P. australis} to stress has varied within
and between estuaries.

For example, in St Georges Basin almost all loss of seagrass occurred in the shallow
waters, while many of the deeper \textit{P. australis} beds, particularly on the southern shore,
appear to have remained in excellent condition, and \textit{P. australis} beds in the entrance
channel of the basin remained in reasonable health. This was also the case for
Merimbula Lake, where deeper \textit{P. australis} showed no overall change in area, even as
shallow \textit{P. australis} communities declined (Meehan 1997). In Port Hacking,
*P. australis* responded to natural and anthropogenic stress in a variety of fashions. For example, in the entrance to Gunnamatta Bay *P. australis* beds fragmented over time; in Fishermen’s Bay the deep *P. australis* beds contracted shoreward, and in the entrance to Burraneer Bay, shallow and deep *P. australis* communities vanished completely from the impacted area. The complex nature of these changes means that monitoring the health of *P. australis* in the future will need to adopt a number of strategies. For example, monitoring the depth limit of seagrass as an indicator of seagrass health may not detect change in some zones of southern NSW estuaries, as many of the impacts documented in this study affected only shallow water communities. Overall, the *P. australis* communities in southern NSW estuaries do not support the two hypotheses outlined above. If nutrient/sediment loading is impacting on water quality in southern NSW estuaries, the relationship between the two is complex. A study by Wilzbach et al. (2000) of three seagrass species in Tarpon Bay, Florida, also emphasised the complex relationship between seagrass health and PAR. They noted that there was no significant relationship between PAR and seagrass shoot density and concluded that the interaction of nutrients, salinity, grazing and mechanical damage greatly influences seagrass health and distribution. This situation also appears to be true for estuaries in southern NSW, where variations in salinity, tidal range and sand movement may cause unpredictable long term changes in *P. australis* communities.

The increase of *P. australis* in Wagonga Inlet over the last forty years is against the trend found for all other estuaries in this region. Reports of seagrass increase due to anthropogenic impacts are rare, but not unreported. For example, West (1993) documented an increase in seagrass cover in the Macleay River, NSW, in a situation similar to that of Wagonga Inlet, where structural modifications to the entrance channel resulted in improved conditions for seagrass growth. It seems feasible that the construction of groynes at the entrance to Wagonga Inlet resulted in an increase in *P. australis* communities, and a decrease in *Zostera* communities. As seagrass species could not be differentiated on some of the earlier aerial photographs, it is difficult to determine by exactly how much *P. australis* has increased in Wagonga Inlet. The dynamics of *P. australis* in Wagonga Inlet should be further investigated to see whether this process of expansion is still continuing.
CHAPTER 4. NATURAL RATES OF RECOVERY FOR POSIDONIA AUSTRALIS AT TWO SITES IN SOUTHERN NSW

4.1 Introduction

Chapter Three has shown a consistent decline in *P. australis* communities in southern NSW estuaries as a result of anthropogenic and natural impacts. This Chapter examines two examples where natural recovery of *P. australis* can be measured after such impacts have been curtailed. The first case study involves a site in Jervis Bay, where large ‘holes’ were created within a continuous *P. australis* bed by seismic blasting in 1969. The second example occurs in Port Hacking, where shell grit mining removed both *P. australis* and *Z. capricorni* communities from a small bay on the southern shore prior to 1973.

The objective of this Chapter is to measure any natural recovery of *P. australis* at these two sites that has occurred over the last thirty years. Particular attention is given to identifying the dominant process leading to recovery of the *P. australis* community (for example, vegetative regrowth or seeding establishment) and attempting to predict the likely time-scale of full recovery.

It should be pointed out that the mechanism and rate of regrowth was not determined for the Port Hacking study. Nevertheless, the study has been included as it a rare example of *P. australis* recovery. Any information on recovery of damaged *P. australis* meadows is valuable in the assessment of the conservation status of *P. australis* in southern NSW estuaries.
4.2 Case Study 1: The natural recovery of *Posidonia australis* in Jervis Bay, NSW, following seismic blasting

4.2.1 Introduction

Jervis Bay is a large open embayment in southern NSW (see Fig. 1.1 for location). It contains significant areas of seagrass and includes beds of at least six species: *P. australis*, *Z. capricorni*, *Z. muelleri*, *H. ovalis*, *H. decipiens* and *H. tasmanica* (West 1990). *P. australis* is the dominant species of seagrass in Jervis Bay, and extensive beds, possibly the largest continuous areas along the east Australian coast, occupy the rim of marine sands that compose the shore (West et al. 1985). Unlike many coastal environments in NSW, the foreshores of Jervis Bay are relatively undeveloped and the seagrass beds remain in near-pristine conditions (Larkum 1976). However, a section of the *P. australis* beds along the southern shoreline of Jervis Bay was damaged in the late 1960’s by the detonation of explosives during a seismic survey. The purpose of the survey was to rate the stability of the area and assess it as a suitable site for a nuclear power facility. Due to public opposition, the power station did not go ahead, however circular areas of bare sand or ‘holes’ measuring approximately 15 - 20 metres in radius had been created in the *P. australis* beds by the survey work. These holes remain to the present day and are clearly visible in aerial photographs of the site (Fig. 4.1). Field investigations of the site have been carried out a number of times over the past 20 years (Ron West, unpub. data) but have not recorded seedling establishment in the bare areas.

4.2.2 Methods

To assess whether there has been any re-colonisation of the denuded sands, the eleven blast holes were mapped from a series of historical aerial photographs using a G.I.S (ArcView Version 3.0). Photographs were available for the years 1972, 1981, 1989, 1993 and 1997. Each photograph was scanned at a resolution of 600 dots per inch (dpi), resulting in a pixel sizes corresponding to ground measurements of 0.63 to 1.71 m. These photos were rectified using a 3 point GCP model using co-ordinates read
from a topographic map. Spreading rates were estimated from average rates of change in the radii of the holes, which were approximately circular.

The steps involved in this estimation were: (1) blast holes were digitised and GIS used to calculate the area \(A\) for each hole in each year; (2) the radius \(r\) for each hole in each year was calculated \(r = \sqrt[0.5]{A/\pi}\); (3) spreading rates \(\text{cm.yr}^{-1}\) were calculated from change in radius of the holes divided by years between aerial photographs; and, (4) mean spreading rate and standard error were estimated by averaging all available spreading rates. Finally, expected recovery times for \(P. australis\) were predicted by calculating the years required for the holes to have a radius of zero based on the mean spreading rate.

\textit{Estimation of errors}

Before presenting the results, it is important to be aware of errors involved in high resolution mapping. There are three main sources of errors associated with the process of mapping features on aerial photos: 1) Digitising error; 2) Image distortion error; and 3) Boundary estimation error.

Digitising error was assessed by mapping each blast hole three times in order to examine the consistency of interpretation. The variation in area of the three estimates was found to be small (<1% by area). Initially, image distortion error was assessed by examining the RMS errors for the rectified image. Unfortunately, RMS errors were not an accurate indication of image distortion error, as only three GCP’s were used to rectify each image. Image distortion errors were further assessed by examining distances between the same features on each aerial photograph, which allowed the distortion still present after rectification to be assessed. This error was found to be moderate (<4% by area).
Figure 4.1. Aerial photograph of the southern shoreline of Jervis Bay (NSW) showing the 11 blast holes as they appear in 1997.
Boundary error refers to the accuracy with which the edge of each blast hole could be located and represents the most significant error in this study. Scanning an aerial photograph at a resolution of 600 dpi does not yield a clear line between the seagrass bed and bare sand, but rather a gradation of pixels from seagrass to sand. This colour gradation is usually three pixels wide, and moves from dark green (interpreted as dense *P. australis*) to light green (interpreted as sparse or colonising *P. australis*) to white (interpreted as bare sand). The true edge of each blast hole could not be located precisely, as the operator could locate the boundary either on either the dark green/light green boundary, or the light green/white boundary. This meant that the boundary could only be located to within one pixel. Therefore, boundary estimation error was calculated by examining the difference in area between two circles that differed in radius by 1 pixel. The size of the pixel depended on the photo. For example, on the 1989 aerial photograph Hole 1 had a radius of 13.2m. The pixel size of the 1989 photo corresponded to a ground measurement of 0.60m. The boundary estimation error was calculated by comparing the area of a circle with radius 13.2m to the area of a circle with radius 13.8m and 12.6m (or 13.2 ± 0.6m). In this case, error was about 9% by area.

Area estimates from remote sensing do not normally present errors, however, in this case the total error for the mapping process is likely to be approximately 15% of the final calculations.

**4.2.3 Results**

Between 1972 and 1997, all 11 blast holes decreased in area due to the spread of *P. australis* rhizomes from the surrounding bed (Fig. 4.2). During the 25 year period, the total area occupied by the 11 holes decreased by 57%, from 7600 m² (1972) to 3300 m² (1997). The decrease in area of individual holes has varied considerably, with a maximum of 71% (448 m²) for Hole 7, and a minimum of 38% (408 m²) for Hole 2 (Table 4.1). Mapping from the aerial photographs suggested that this encroachment of *P. australis* into the holes was a gradual and progressive process. Field sampling during 1993/94 supports the notion of gradual encroachment. Rhizome growth
patterns of shoot clusters collected from the boundary of the blast holes indicates a moderate but steady growth of rhizomes into the cleared areas (Martin Wolterding, University of Sydney, personal communication). However, the recovery of *P. australis* by rhizome growth into the bared areas was not consistent around the perimeter of the holes. Comparison of north-south and east-west axis of the 11 holes indicated that while some sections of the *P. australis* beds have spread consistently into the holes, others have remained relatively static. In addition, some recession of seagrass was observed in 1989, with Holes 1, 2 and 4 increasing in area slightly, before decreasing again in 1993. Storm activity may have been responsible for this increase in estimated area or it may simply reflect the degree of error in high resolution mapping.

Mean spreading rate (± SE), or encroachment rate, of the *P. australis* community over the 25 year period averaged across all conditions and all 11 holes was 21 ± 2 cm.yr⁻¹. However, spreading rates of *P. australis* varied between time periods and between holes. For example, estimated spreading rates were negative for Holes 1, 2 and 4 during the period of recession (1981 to 1989) and greatest for Hole 11 between 1993 and 1997 (58 cm.yr⁻¹).

While it is difficult to predict what might happen to these sites in the future, it is important, particularly for management purposes, to provide some indication of the estimated time of recovery of these damaged sites. For this reason, an estimate has been made of the expected time of recovery for each of the holes, assuming that the average rate of recolonisation of each of the holes remains unchanged. Accepting this assumption, the 11 blast holes should become completely revegetated at various times in the next two centuries, the earliest in the year 2034 (Holes 7 and 8) and the latest in 2071 (Hole 2) (Table 4.1 and Fig. 4.2).
Fig. 4.2. Map of *Posidonia australis* seagrass beds (shown in grey) located along the southern shoreline of Jervis Bay indicating the extent of the 11 blast holes in 1972 and in 1997.
Table 4.1. Estimated area (m$^2$) in 1972 and 1997, and predicted year of recovery for each of the 11 holes created as a result of a seismic survey of the southern shoreline of Jervis Bay (NSW) during the late 1960's. Note that holes have been numbered for convenience.

<table>
<thead>
<tr>
<th>Hole number</th>
<th>Mean area estimated in 1972 (m$^2$)</th>
<th>Mean area estimated in 1997 (m$^2$)</th>
<th>Predicted Year of Recovery$^1$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>716 ± 16</td>
<td>423 ± 6</td>
<td>2053</td>
</tr>
<tr>
<td>2</td>
<td>1180 ± 15</td>
<td>772 ± 8</td>
<td>2071</td>
</tr>
<tr>
<td>3</td>
<td>475 ± 10</td>
<td>263 ± 7</td>
<td>2040</td>
</tr>
<tr>
<td>4</td>
<td>581 ± 2</td>
<td>252 ± 5</td>
<td>2039</td>
</tr>
<tr>
<td>5</td>
<td>562 ± 13</td>
<td>263 ± 9</td>
<td>2041</td>
</tr>
<tr>
<td>6</td>
<td>666 ± 9</td>
<td>259 ± 2</td>
<td>2040</td>
</tr>
<tr>
<td>7</td>
<td>632 ± 11</td>
<td>184 ± 5</td>
<td>2034</td>
</tr>
<tr>
<td>8</td>
<td>632 ± 7</td>
<td>186 ± 4</td>
<td>2034</td>
</tr>
<tr>
<td>9</td>
<td>794 ± 8</td>
<td>274 ± 11</td>
<td>2043</td>
</tr>
<tr>
<td>10</td>
<td>579 ± 26</td>
<td>214 ± 4</td>
<td>2036</td>
</tr>
<tr>
<td>11</td>
<td>734 ± 22</td>
<td>268 ± 8</td>
<td>2042</td>
</tr>
</tbody>
</table>

$^1$Assumes that rhizomes continue to encroach at a rate of 21 cm.yr$^{-1}$. 
4.2.4 Discussion

This study represents one of the first attempts to estimate recovery rates of the seagrass *P. australis* and does so using a method not previously employed (aerial photography). The availability of historical aerial photographs taken at regular time intervals over the past thirty years has provided evidence of the recovery of the damaged *P. australis* bed in Jervis Bay and allowed an estimation of an average spreading rate for the species under these conditions. Many of the holes left by the seismic blasting are less than half their original size, due to the encroachment of *P. australis* rhizomes at an overall average rate of $21 \pm 2 \text{ cm.yr}^{-1}$. Should the process of encroachment continue at the same rate, the *P. australis* community should be fully recovered by the middle of next century. While this is one of the first attempts to measure the recovery rate of *P. australis* after damage, it also demonstrates that recovery of this species can be very slow. In this instance, the *P. australis* community has taken over 25 years to recolonise a total area of only 0.4 hectares (ha). This recolonisation appears to be entirely by vegetative regrowth, with no seedlings found during field inspections.

This calculation of the rate of recovery of *P. australis* represents only one series of estimates at one location. The damaged areas were within a long established, stable, mature seagrass bed of *P. australis* which was likely to have few horizontal rhizomes when originally impacted (West 1990). The location also has excellent water quality and conditions ideal for recovery. At other sites, it is likely that the recovery rate of *P. australis* will be a function of the growth perimeter, the number of horizontal shoots or tillers and the surrounding environmental conditions. For example, a *P. australis* bed in an area where high numbers of horizontal shoots occur, may recover at a faster rate. In the next case study, the natural recovery of *P. australis* is assessed at Port Hacking, an estuary which has lost considerable areas of seagrass.
4.3. Case Study 2: Natural recovery of *Posidonia australis* in Port Hacking, NSW, following shell grit mining

4.3.1 Background

In Port Hacking, shell grit mining occurred from 1928 to 1973 (Druery and Hurrell 1986). The lease area for this activity was located on the marine tidal delta adjacent the Royal National Park, in between Fisherman’s Bay and Red Jacks Point (see Fig. 3.1 for locality map). Mining for shell grit involves the removal of the upper layers of sediment and it is likely that all aquatic vegetation in the mining lease area was destroyed. This included about 40 ha of seagrass between 1961 and 1975 (see Appendix 4 - middle port). *Z. capricorni* was most affected by the mining, but a small area of *P. australis* adjacent the Royal National Park was also removed. Since 1975, *Z. capricorni* has recolonised most of the area where water depth is suitable, but aerial photographs do not suggest a similar recolonisation by *P. australis* (Fig. 4.3). However, field investigation in 1999 found a number of small *P. australis* patches in the once denuded area.

4.3.2 Methods

Unlike Jervis Bay, it was not possible to map the recovery of *P. australis* using aerial photographs as the small *P. australis* beds could not be discriminated from the more extensive *Z. capricorni* communities. Instead, the dimensions of each *P. australis* patch (± 0.1m) were measured by placing a transect through the approximate centre and another transect perpendicular to the first. These formed the long and short axis of each patch. The age of each patch was estimated by assuming that the patches had grown at a rate equal to that of fast growing *P. australis* rhizomes (about 20 cm.yr$^{-1}$), such that:

\[
\text{Patch age (y)} = \frac{\text{Long axis (m)}}{0.2 \text{ m}}
\]
In order to identify the likely origin of these patches, some preliminary sampling was attempted. Along the long axis, shoots were sampled at 1m intervals from patch centre to patch edge. The purpose of this sampling was to age each shoot by counting the number of internodes, and then calculate an age gradient across each patch.

4.3.3 Results

Eleven discrete *P. australis* beds were found in the study area, in amongst dense and continuous *Z. capricorni* communities. Within the perimeter of each patch, *P. australis* has displaced all other seagrass species. The density of *P. australis* shoots was quite low, at around about 100 shoots per m². This density was similar to established beds nearby, where shoot numbers appear to be kept low by exposure during low tides.

The dimensions of the *P. australis* patches ranged from 3.0 x 2.5m to 10.7 x 8.1m. Assuming a yearly spreading rate of 0.2m, these dimensions corresponded to an approximate age of 8 yrs and 27 yrs respectively (Table 4.2). Coincidentally, 27 years have elapsed since shell grit mining was terminated in Port Hacking. The predicted year of formation suggests that patch formation has been occurring since 1973, and continued irregularly through to 1992 (Table 4.2).

Preliminary sampling of shoots in the patch centre and patch edge did not provide useful results. It was not possible to trace the origin of individual shoots back to a single rhizome. In addition, many of the shoots appeared to be growing from deep beneath the sediment surface (> 0.5m). It was not feasible to sample these shoots without causing significant damage to the bed.
Fig. 4.3. Distribution of seagrass in shell grit lease area in 1951, 1975, 1982 and 1999, showing the effects of shell grit mining on seagrass beds, and eventual recovery of *Zostera capricorni* and *Posidonia australis* (see Fig. 3.5 for legend).
Table 4.2. Dimensions of *Posidonia australis* patches in area mined for shell grit in Port Hacking, NSW, and estimated age and time of formation.

<table>
<thead>
<tr>
<th>Patch No.</th>
<th>Long axis (m)</th>
<th>Short axis (m)</th>
<th>Age (y)</th>
<th>Estimated time of formation</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>4.3</td>
<td>2.9</td>
<td>10.75</td>
<td>1989</td>
</tr>
<tr>
<td>2</td>
<td>8.7</td>
<td>7.2</td>
<td>21.75</td>
<td>1978</td>
</tr>
<tr>
<td>3</td>
<td>4.3</td>
<td>3.8</td>
<td>10.75</td>
<td>1989</td>
</tr>
<tr>
<td>4</td>
<td>4.1</td>
<td>3.6</td>
<td>10.25</td>
<td>1990</td>
</tr>
<tr>
<td>5</td>
<td>7.0</td>
<td>6.1</td>
<td>17.5</td>
<td>1982</td>
</tr>
<tr>
<td>6</td>
<td>10.7</td>
<td>8.1</td>
<td>26.75</td>
<td>1973</td>
</tr>
<tr>
<td>7</td>
<td>4.7</td>
<td>3.8</td>
<td>11.75</td>
<td>1988</td>
</tr>
<tr>
<td>8</td>
<td>6.0</td>
<td>5.6</td>
<td>15</td>
<td>1985</td>
</tr>
<tr>
<td>9</td>
<td>10.4</td>
<td>7.3</td>
<td>26</td>
<td>1974</td>
</tr>
<tr>
<td>10</td>
<td>9.1</td>
<td>8.0</td>
<td>22.75</td>
<td>1977</td>
</tr>
<tr>
<td>11</td>
<td>3.0</td>
<td>2.5</td>
<td>7.5</td>
<td>1992</td>
</tr>
</tbody>
</table>
4.3.4 Discussion

This short study demonstrates that there has been some recolonisation by *P. australis* since the termination of shell grit mining in 1973. Unlike the blast holes in Jervis Bay, this recolonisation has not been by vegetative regrowth from a large pre-existing meadow. Rather, discrete and isolated patches of *P. australis* have grown in an otherwise continuous *Z. capricorni* meadow. There were no differences in depth or substratum between the *P. australis* beds and the adjacent *Z. capricorni* meadow. This type of growth has not been reported before for any seagrass communities growing in this region.

The most important aspect of these results is establishing the likely origin of these *P. australis* patches, whether they are the result of seedling establishment and subsequent growth, or whether they are vegetative re-growth from pre-existing beds too small to be seen on the 1975 photographs. The presence of an age gradient may have provided evidence for seedling establishment, however an age gradient could not be established. This was due to the depth of rhizomes and the difficulty in locating the first order rhizome from which patch development must have started.

4.4 Summary

Rates of natural recovery of *P. australis* were investigated at two sites in southern NSW. At the Jervis Bay location, recovery came about through the spread of *P. australis* rhizomes onto the bare sands. The investigation at the Port Hacking site was less clear, with no obvious mechanism of recovery. The latter site should be monitored to see whether *P. australis* seedlings colonise and produce more isolated patches in the years to come.
CHAPTER 5. SEEDLING DEVELOPMENT AND PATCH FORMATION OF *POSIDONIA AUSTRALIS* IN ST GEORGES BASIN, NSW.

5.1 Introduction

This chapter examines recovery of *P. australis* communities in St Georges Basin, as this estuary appears to be one of the few sites in southern NSW where *P. australis* seedlings are found *in situ*. In contrast to Chapter Four, which documented long term recovery of *P. australis* mainly by vegetative regrowth, this chapter attempts to elucidate aspects of sexual propagation leading to natural recovery of damaged *P. australis* communities in southern NSW estuaries.

The rarity of *P. australis* seedlings, and the absence of small plants of the species (see Kuo and Kirkman 1999) has meant that it is has been impossible to study vital aspects of the species’ life history. This includes seedling abundance, rate of patch formation and time taken to produce a plagiotropic rhizome. Information concerning these aspects of sexual propagation, in combination with information on vegetative regrowth documented in Chapter Four, may enable an estimation of the time scale involved in the natural recovery of *P. australis* ecosystems. This, in turn, will make it possible to better assess the vulnerability of *P. australis* to anthropogenic impacts, as well as identify which areas are most in need of protection and/or restoration.

The objective of this study is to measure some attributes of sexual and vegetative propagation in *P. australis* communities in St Georges Basin, such as rhizome growth, seedling growth and patch formation. These data may enable the life history of *P. australis* to be reconstructed and thus identify some of the missing aspects of the development of this species, such as seedling growth rate, production of orthotropic shoots and production of a plagiotropic rhizome.
5.2 Methods

5.2.1 Study area

St Georges Basin (see Figure 1.1 for location map) consists of a deep lake basin with an average depth of nine metres that is linked to the ocean by a narrow inlet channel. The inlet is permanently open and mean tidal range is 0.78m (SCC 1998). Tidal flushing is limited in the upper reaches of the Basin, with tidal range being only 0.036m (SCC 1998). *P. australis* communities occupy most of the inlet channel of St Georges Basin, but in the central mud basin are restricted to cut-off lagoons and embayments (see Figure 3.13 for map of present seagrass distribution). These shallow embayments (depth < 3m) support patchy and mixed seagrass meadows that are dominated by *Zostera* spp. and *Halophila* spp. in the deeper waters (1.5m to 3m) and *P. australis* in the shallower, inshore waters (0.5m to 2m).

In early 1999, fieldwork for a separate study revealed that one of these embayments supported a small number of *P. australis* seedlings. On the basis of this find, a larger scale investigation was conducted at four randomly chosen sites within St Georges Basin: two sites on the southern shore (SGBA and SGBB); one on the northern shore (SGBC), and one on the western shore (SGBD) (Fig. 5.1). All these sites were located in areas that had supported extensive *P. australis* communities in the past (see Section 3.3.2).

5.2.2 Vegetative growth of *Posidonia australis*

At SGBC, 50 rhizomes located on the edge of large *P. australis* patches were tagged by placing a plastic electrical tie around the most recently produced internode. These rhizomes were harvested after approximately 500 days.
Fig. 5.1. Location of sites in St Georges Basin, NSW, surveyed to investigate the occurrence of *Posidonia australis* seedlings and small plants.
The following growth parameters were measured:

- number of new internodes produced since tagging;
- length of each new rhizome internode (mm);
- number of new orthotropic and transitional shoots;
- length of each new shoot internode; and,
- number of second order rhizomes produced.

The following formulas (modified from Duarte et al. 1994) were used to calculate vegetative growth:

\[ \text{Rhizome elongation (cm yr}^{-1}) = 365 \times \text{new rhizome material (cm) \times time interval (days)}. \]

\[ \text{Plastochrone interval (days), horizontal shoots} = \text{time interval (days) \div nos. of new internodes}. \]

To calculate plastochrone interval (P.I.) for vertical shoots, the production ratio of rhizome internodes was compared to shoot internodes, such that:

\[ \text{P.I. (days), vertical shoot} = (\text{nos. rhizome internodes} \div \text{nos. shoot internodes}) \times \text{rhizome P.I.} \]

\[ \text{Shoot elongation (cm yr}^{-1}) = (365 \div \text{Shoot P.I. (days)}) \times \text{Mean shoot internode length (cm)}. \]

Growth architecture was documented using a system of numbering similar to Molenaar et al. (2000). An Order-1 axis was the principal plagiotropic rhizome tagged at the beginning of the experiment. Order-2 axes were the lateral branches produced by an Order-1 axis. Order-3 axes the lateral branches produced by Order-2 axis system. Orthotropic shoots were the vertical, slow growth units, while transitional shoots were shoots in intermediate state between plagiotropic and orthotropic. From each tagged plagiotropic rhizome the following was recorded:

- number of Order-1, Order-2 and Order-3 rhizomes produced; and,
- number of orthotropic and transitional shoots produced.

Ageing techniques were used to calculate the time of shoot production and of second order rhizome production.
5.2.3 Reproductive growth of *Posidonia australis*

Seedling abundance was assessed at the four randomly chosen sites (SGBA, SGBB, SGBC and SGBD). At each site an area measuring 30m x 60m was located randomly within a fragmented *P. australis* community. This area was snorkelled and all isolated *P. australis* shoots or small patches with a diameter < 1m were tagged and counted. Each isolated shoot was then excavated to check for presence of a seed case. In the case of larger patches where the seed case was likely to have disintegrated, the sediment around the base of the plant was removed to see whether the plant was free standing or whether it was connected to a buried rhizome matt. This procedure allowed the origins of each patch to be identified.

A number of seedlings and small plants of different sizes were destructively sampled at each site, and the following morphological and growth characteristics were measured:

- number of shoots and rhizomes per plant;
- length (mm) of each shoot and rhizome internode; and,
- leaf area per shoot.

Seedling age and growth rate were estimated by assuming that each internode represented a fixed time interval. The time interval used was either the mean Shoot P.I. or mean Rhizome P.I. of mature *P. australis* plants, depending on what type of shoot was being examined. Shoot growth was calculated using a mean internode length for each shoot and the average number of internodes produced by mature orthotrophic shoots to estimate yearly extension rates. Rhizome growth was calculated similarly, but the average number of internodes produced by mature plagiotropic rhizomes was used instead. By counting back along the internodes, it was possible to calculate the time (or age of the plant) when new shoots and rhizomes were produced.

5.2.4 Regenerative growth of *Posidonia australis*
Initial site inspections of SGBC suggested that some of the isolated *P. australis* patches were not the result of seedling establishment and patch formation. Instead, they were more likely the fragmented remains of a once more extensive *P. australis* meadow, or were shoots that had regenerated from buried rhizome matt.

To assess the possibility of regenerative growth, the number of patches in each of the four sites not derived from seedlings were counted. At SGBC, the rhizome matt between *P. australis* patches was monitored for production of new regenerative shoots over an 18 month period. Any new regenerative shoots were noted and monitored for survival and development.

To give an indication of the extent of old rhizome deposits, and its relationship to living beds, a section of SGBC area measuring 30 m x 30 m was mapped using a series of parallel transects spaced every 2m. Where the transects intersected a seagrass patch or an old rhizome matt, the location was noted. This enabled a map to be drawn showing the position and approximate area of the present seagrass community, and its likely extent in the past.

### 5.3 Results

#### 5.3.1 Vegetative growth of *Posidonia australis*

Over the period of study, almost all plagiotropic rhizomes exhibited substantial growth, and produced new orthotropic shoots, transitional shoots and second order plagiotropic rhizomes. Only two of the tagged rhizomes did not show any horizontal growth; these two rhizomes reverted to orthotropic morphology. Mean rhizome elongation was $21.4 \pm 1.0$ cm yr$^{-1}$; and ranged from 7.0 cm yr$^{-1}$ to 37.9 cm yr$^{-1}$ (Table 5.1). Rhizome internode length ranged from 3mm to 44mm, with a mean of $20.2 \pm 0.9$ mm (from a random sample of 100 internodes). Approximately $11.4 \pm 0.3$ internodes yr$^{-1}$ were produced, which equated to a mean Rhizome Plastochrone Interval of $33 \pm 0.8$ days (Table 5.1).
Table 5.1. Vegetative growth rates of *Posidonia australis* at SGBC in Saint Georges Basin, southern NSW. Fifty plagiotropic rhizomes were tagged for approximately 500 days.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Minimum</th>
<th>Mean</th>
<th>Maximum</th>
<th>(± S.E)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rhizome elongation (cm.yr(^{-1}))</td>
<td>7.0</td>
<td>21.4 ± 1.0</td>
<td>37.9</td>
<td></td>
</tr>
<tr>
<td>Rhizome Internodes (yr(^{-1}))</td>
<td>6.3</td>
<td>11.4 ± 0.3</td>
<td>16.3</td>
<td></td>
</tr>
<tr>
<td>Rhizome P.I. (d)</td>
<td>22.4</td>
<td>33.0 ± 0.8</td>
<td>57.9</td>
<td></td>
</tr>
<tr>
<td>Rhizome internode length (mm), n = 100</td>
<td>3</td>
<td>20.2 ± 0.9</td>
<td>44</td>
<td></td>
</tr>
<tr>
<td>Shoot elongation (cm.yr(^{-1}))</td>
<td>1.3</td>
<td>4.4 ± 0.3</td>
<td>11.7</td>
<td></td>
</tr>
<tr>
<td>Shoot Internodes (yr(^{-1}))</td>
<td>12.1</td>
<td>7.9 ± 0.3</td>
<td>5.6</td>
<td></td>
</tr>
<tr>
<td>Shoot P.I. (d)</td>
<td>30.1</td>
<td>48 ± 1.4</td>
<td>64.9</td>
<td></td>
</tr>
<tr>
<td>Shoot internode length (mm), n = 100</td>
<td>0.5</td>
<td>4.3 ± 0.2</td>
<td>18</td>
<td></td>
</tr>
<tr>
<td>Shoot production. Rhizome.yr(^{-1})</td>
<td>0</td>
<td>0.9 ± 0.1</td>
<td>2.55</td>
<td></td>
</tr>
</tbody>
</table>
Shoot elongation was substantially slower than rhizome elongation, with a mean of 4.4 ± 0.3 cm.yr⁻¹, and a minimum and maximum of 1.3 and 11.7 cm.yr⁻¹ respectively (Table 5.1). Shoot internode length ranged from 0.5mm to 18mm, with a mean of 4.3 ± 0.2 mm (from a random sample of 100 internodes). On average, shoots produced 7.0 ± 0.3 internodes.yr⁻¹, which equated to a mean shoot Plastochrone Interval of 48.0 ± 1.4 days (Table 5.1).

Some aspects of *P. australis* growth exhibited a distinct seasonal pattern. For example, production of new rhizomes, which involved the splitting of the primary Order-1 rhizome to produce an Order-2 rhizome, occurred mainly in Spring and early Summer (Fig. 5.2). The majority of tagged rhizomes produced only Order-2 rhizomes, though a small number produced Order-3 rhizomes.

Shoot production exhibited a similar seasonal pattern to rhizome production, with most new shoot production occurring in Spring and early Summer (Fig. 5.3). The most common growth pattern was for the Order-1 rhizome to split and produce 1-2 second Order-2 rhizomes and 1-2 transitional shoots. However, unlike new rhizome production, new shoots were also produced in early to mid Autumn. Mean shoot production was 0.9 ± 0.1 shoots per rhizome per year (Table 5.1).

Flowering and fruiting were observed at site SGBC in November, 1999. At this time, about 4% of the total number of shoots had produced fruit.
Fig. 5.2. Time of year that Order-2 rhizomes were produced at site SGB in St Georges Basin, NSW (n=46).
5.3.2 Reproductive growth of *Posidonia australis*

*P. australis* seedlings were found at all four sites surveyed (Table 5.2). Seedling abundance ranged from 0.001 seedlings.m\(^{-2}\) (ten seedlings per ha) to 0.014 seedlings.m\(^{-2}\) (140 seedlings per ha). The ages of the collected samples (calculated using a P.I. of 48 days for each internode) ranged from 0.7 years, which consisted of a seedling bearing a single orthotropic shoot, to 8.2 years, which consisted of a small plant bearing four orthotropic shoots and one plagiotropic rhizome. The mean number of shoots per seedling was around 1.5 to 2.5 shoots at SGBB, SGBC and SGBD; though was considerably higher at SGBA (3.7). Estimated growth rate of seedling shoots ranged from 1.1 ± 0.1 to 1.9 ± 0.3 cm yr\(^{-1}\), considerably slower than that of mature orthotropic shoots measured at SGBC (Table 5.1 and 5.2).

Both the number of shoots and leaf area were related to seedling age, with both increasing as seedlings developed through time (Fig. 5.4 and 5.5). Increases in leaf areas and number of shoots were particularly evident between 2 and 5 yrs of age and resulted in a clustering of data points around these ages (Fig. 5.4 and 5.5). There was also clear evidence of an increase in the overall size of *P. australis* plants between 1 yr and 9 yrs (Fig. 5.6).

Small plants, consisting of a number of orthotropic shoots and one plagiotropic rhizome, were found at three of the four sites: SGBA, SGBC and SGBD (Table 5.2). However, these small plants were rare, with only two being found at each site. Each plant had produced one rhizome only, with growth rates ranging from 4.6 cm yr\(^{-1}\) to 18.3 cm yr\(^{-1}\). Only plants at SGBD exhibited growth rates within the range of mature plagiotropic rhizomes; at SGBA and SGBC growth rates of rhizomes was half to a quarter of mature plants. As each plant had produced one rhizome only, spreading was uni-directional. Multi-directional spreading was evident in some of the larger patches; however, none of these were appeared to be derived from seedlings.
Fig. 5.3. Time of year that new orthotropic shoots were produced at site SGBC in St Georges Basin, NSW (n= 53).
Table 5.2. Summary of *Posidonia australis* reproductive growth variables at four sites in St Georges Basin, NSW (SGBA, SGBB, SGBC, SGBD).

<table>
<thead>
<tr>
<th>Variable</th>
<th>SGBA</th>
<th>SGBB</th>
<th>SGBC</th>
<th>SGBD</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total no. of seedlings</td>
<td>26</td>
<td>2</td>
<td>12</td>
<td>17</td>
</tr>
<tr>
<td>No. regenerative patches</td>
<td>5</td>
<td>3</td>
<td>6</td>
<td>9</td>
</tr>
<tr>
<td>Seedling abundance (seedlings.m(^{-2}))</td>
<td>0.014</td>
<td>0.001</td>
<td>0.007</td>
<td>0.009</td>
</tr>
<tr>
<td>Seedling age range (yrs)</td>
<td>3.0 - 4.9</td>
<td>0.7 - 4.3</td>
<td>0.8 - 4.6</td>
<td>0.9 - 8.2</td>
</tr>
<tr>
<td>Mean no. shoots.seedling(^{-1})</td>
<td>3.7</td>
<td>1.5</td>
<td>2.1</td>
<td>2.4</td>
</tr>
<tr>
<td>No. seedlings with plagiotropic rhizome</td>
<td>2</td>
<td>0</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>Shoot elongation ± S.E. (cm.yr(^{-1}))</td>
<td>1.9 ± 0.3</td>
<td>n/a</td>
<td>1.4 ± 0.2</td>
<td>1.1 ± 0.1</td>
</tr>
<tr>
<td>Rhizome elongation (cm.yr(^{-1}))</td>
<td>5.5</td>
<td>n/a</td>
<td>7.9</td>
<td>18.3</td>
</tr>
<tr>
<td></td>
<td>4.6</td>
<td>9.8</td>
<td>15.8</td>
<td></td>
</tr>
</tbody>
</table>
Fig. 5.4. Scatter plot of mean leaf area per shoot and seedling age for *Posidonia australis* seedlings and small plants in St Georges Basin, NSW.
Fig. 5.5. Scatter plot of number of shoots and seedling age for *Posidonia australis* seedlings and small plants in St Georges Basin, NSW.
Fig. 5.6. *Posidonia australis* seedlings and small plants harvested from SGBD, showing the increase in leaf size, number of shoots, and production of first plagiotropic rhizome. From left to right, the ages of the samples are: 0.9, 4.0, 4.9, and 8.2 yrs.

Fig. 5.7. The oldest *Posidonia australis* plant found during study, illustrating production of first plagiotropic rhizome. In this example, it is estimated that it took approximately 7.6 yrs (assuming that each internode represented 48 days).
Table 5.3. Minimum, mean and maximum times (years) taken for *Posidonia australis* seedlings to produce 2\(^{\text{nd}}\) to 5\(^{\text{th}}\) shoot, and plagiotropic rhizome. The P.I. used was 48 days, that of mature *Posidonia australis* plants.

<table>
<thead>
<tr>
<th></th>
<th>Minimum</th>
<th>Mean ± S.E</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>Production of second shoot (yrs)</td>
<td>0.7</td>
<td>2.7 ± 0.3</td>
<td>6.7</td>
</tr>
<tr>
<td></td>
<td>n = 22</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Production of third shoot (yrs)</td>
<td>1.0</td>
<td>3.4 ± 0.3</td>
<td>7.6</td>
</tr>
<tr>
<td></td>
<td>n = 18</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Production of fourth shoot (yrs)</td>
<td>2.2</td>
<td>4.3 ± 0.7</td>
<td>7.9</td>
</tr>
<tr>
<td></td>
<td>n = 9</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Production of fifth shoot (yrs)</td>
<td>3</td>
<td>4.5 ± 0.6</td>
<td>7.6</td>
</tr>
<tr>
<td></td>
<td>n = 7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Production of first rhizome (yrs)</td>
<td>3.0</td>
<td>4.3 ± 0.7</td>
<td>7.6</td>
</tr>
<tr>
<td></td>
<td>n = 6</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Despite intensive sampling, both at the four sites and at other randomly chosen areas around the northern and western shoreline of the basin, the small plant shown in Fig. 5.6 and 5.7 was the oldest that could be found. Although there were larger plants, with diameters of 0.5 to 1m, none of these appeared to be derived from seedlings. The larger patches had very thick rhizomes, longer internodes, and there were remains of rhizome matt at the base, suggesting that these large patches were actually fragmented remains of a once more extensive bed. The extent of the rhizome matt in this area can be seen in Fig. 5.8.

Despite the lack of older samples, an estimated time-line of morphological development was constructed for *P. australis* for the first nine years of seedling growth and patch formation based on the data collected. As Table 5.3 illustrates, production of a second shoot occurs at about 2.7 ± 0.3 yrs, although it could occur as early as 0.7 yrs and as late as 6.7 yrs. The production of the third shoot exhibited a similar age range, although on average occurs later, at 3.4 ± 0.3 yrs. The production of the fourth and fifth shoot occurs at approximately similar times, 4.3 ± 0.7 and 4.5 ± 0.9 yrs respectively, although 2 -3 yrs minimum was required. At the time the fourth and fifth shoots were being produced, a plagiotropic rhizome had begun. On average this took 4.3 ± 0.7 yrs, but occurs as early as 3.0 yrs and as late as 7.6 yrs (Fig. 5.7).

None of the *P. australis* seedlings samples collected had produced more than one rhizome; nor had the rhizome split into a second order rhizome, so it was not possible to accurately model future growth. It cannot be assumed that plants will spread out evenly and in all directions from a single point. Obviously, until the production of a second rhizome, *P. australis* will expand in one direction only. In *Cymodosa nodosa* it has been noted by Gallegos et al. (1996) that location of initial patch formation is offset from patch centre, with patch expansion considerably quicker in one direction. This is likely to be the case in *P. australis*, as well, considering that only one rhizome was produced.
Fig. 5.8. Present distribution of *Posidonia australis* beds and surface rhizome matt in a randomly chosen section of site SGBC, in St Georges Basin, NSW (see Fig. 5.1 for locality map).
5.3.3 Regenerative growth of *Posidonia australis*

At each site, a number of small plants were observed to be growing out of rhizome deposits buried under the surface. Generally these patches were less abundant than seedlings, with numbers ranging from 3 at SGBB to 9 at SGBD. It was difficult to assess whether these plants were fragmented remains of a once continuous bed, or had actually regenerated from the rhizome matt after the initial damage.

Monitoring of the rhizome matt at SGBC did not help in determining the regeneration rates of *P. australis* growing from old matts. No new shoots were produced from the rhizome matt within the eighteen months of monitoring. However, mapping of the study area did demonstrate how extensive the surface deposits were, and the degree of damage these *P. australis* communities had sustained in the past (Fig. 5.8). Regrowth of *P. australis* by regeneration of rhizome matt has important implications for recovery rates of damaged *P. australis* stands, as it allows isolated areas to regrow without sexual reproduction. Such isolated *P. australis* stands occur throughout St Georges Basin and can be seen in Fig. 5.8.

5.4 Discussion

Over the years a significant body of observational evidence has accumulated concerning the poor reproductive success and slow rhizome growth of *P. australis* in south eastern Australia (for example, Larkum 1976, West 1980, West 1990, Wolterding 2000). Estuaries in this region which have lost *P. australis* in the past have shown very limited signs of recovery, even though many anthropogenic impacts have been curtailed and conditions for regrowth appear to exist. While remote sensing of *P. australis* recovery in Jervis Bay and Port Hacking suggests that recovery of *P. australis* communities is possible over long time periods (see Section 4.2 and 4.3), there was no conclusive evidence of seedling recruitment and patch formation in these areas. Indeed, the rarity of *P. australis* seedlings and small plants has generally meant it has not been possible to examine sexual propagation in *P. australis* communities in southern NSW. Vegetative growth, such as rhizome elongation and orthotropic shoot production, has also not been studied in detail.
A such, the *P. australis* communities of St Georges Basin are an important find, as short term natural recovery is occurring in this estuary through both vegetative and sexual propagation. The dynamics of natural recovery in these *P. australis* communities suggests that while short term natural recovery of *P. australis* in the region is slow, it is faster than previously thought. Growth strategies are also diverse, with vegetative growth, sexual propagation and regenerative growth (from old matt) all playing a role.

The expansion of remnant *P. australis* patches through rhizome growth was substantial and widespread over the period of study, with almost all tagged rhizomes colonising bare substrate. This pattern of regrowth, where *P. australis* rhizomes move into bare substrate, diverge, and produce secondary shoots, has been observed before by West (1980) and Wolterding (2000) in Botany Bay. However, Larkum (1976) cleared *P. australis* in Botany Bay and observed no regrowth after three years. The capacity of *P. australis* to colonise bare substrate is largely related to the number of actively growing rhizomes at any one site (West 1990), and numbers of active shoots can vary greatly between sites. In the study by Larkum (1976) it may have been that the number of actively spreading shoots was very low. For example, West (1990) found that only 1 out of 600 randomly collected shoots in Jervis Bay showed active growth. In another example, Meehan (unpublished data) found that at one site in Port Hacking (PHS4; for location see Fig. 6.1), the proportion of plagiotropic shoots (actively spreading) was 3%, while at one site in Merimbula Lake (MLS3; for location see Fig. 6.1), it was 45%. It has also been observed elsewhere, for example, in Wagonga Inlet and Port Hacking, that many bed margins do not have a high proportion of plagiotropic rhizomes and appear to be very stable. This is not only evident on margins where growth is limited by environmental factors, such as exposure (on the shallow margin of the bed) or lack of light (on the deep margin of the bed) but is also evident in cleared areas within the *P. australis* community, such as old anchor scars. The large number of plagiotropic rhizomes on the edges of *P. australis* patches in St Georges Basin, and their high rates of survival over eighteen months, suggests that conditions for regrowth may be optimal. This vegetative growth was surprising, as *P. australis* transplants placed at this sites died in less than two months.
The average rate of *P. australis* rhizome elongation in St Georges Basin of 21 cm.yr\(^{-1}\) was very similar to the rate of rhizome incursion estimated for the blast holes in Jervis Bay (see Chapter Four). It was also within the range of previous studies, which calculated an approximate rhizome extension rate of 20 cm.yr\(^{-1}\) and a shoot extension rate of 2-3 cm.yr\(^{-1}\) (West 1990, Wolterding 2000). However, this rate was considerably faster than rhizome elongation of *P. australis* calculated at other sites. For example, Marba and Walker (1999) calculated a rhizome elongation and shoot elongation for *P. australis* of 9.3 cm.yr\(^{-1}\) and 1.4 cm.yr\(^{-1}\) respectively, approximately half that of this study.

While vegetative growth of *P. australis* in St Georges Basin was similar to, or greater than, previous studies, the rate of *P. australis* rhizome growth is still slow compared to many other species. The seagrass species *Heterozostera tasmanica*, *Zostera marina* and *Cymodocea nodosa* all exhibit rhizome elongation rates 3 - 6 times faster than that of *P. australis* rhizomes in this study (Gallegos et al. 1993, Marba and Walker 1999). For example, *C. nodosa* has an rhizome elongation rate of 160 cm. yr\(^{-1}\) (Duarte and Sand-Jensen 1990). Slow rhizome growth means that fragmented *P. australis* communities in St Georges Basin are likely to take many years, possibly decades, before isolated patches coalesce to form extensive meadows. This is a similar time-scale to that estimated for the recovery of damaged *P. australis* beds in Jervis Bay (see Section 4.2).

Successful recruitment of seedlings was observed at all four sites in St Georges Basin and two sites yielded small *P. australis* plants. The discovery of these *P. australis* plants, which were in various stages of morphological development, enabled the life history of *P. australis* to be reconstructed using ageing techniques, and the recovery of a damaged *P. australis* community to be assessed, with respect to seedling recruitment, patch formation and rhizome growth.

The range of ages of *P. australis* seedlings suggests that successful sexual reproduction has occurred regularly in *P. australis* populations in St Georges Basin over the last decade. These seedlings have survived longer term stresses, including storm waves and damage from commercial fishing. The regularity of flowering and
fruiting in other *P. australis* populations (Inglis and Smith 1998, Wolterding 2000; see also Section 6.4.4) along with the conspicuous absence of seedlings, suggest that seedling survival is probably the most critical factor in determining successful sexual reproduction, and not flowering intensity. However, it is not known why *P. australis* seedlings are abundant in St Georges Basin and so rare elsewhere. The main difference between St Georges Basin and other estuaries in southern NSW is that *P. australis* is distributed shoreward of all other seagrass species, due to the very low tidal range. At present, *P. australis* grows along the seaward edge of the shoals, and its leaves often reach the water surface. It thus forms a type of ‘wall’ between the shore and deeper waters. In between the *P. australis* beds and the shore are many hectares of bare sand. Together, these factors may encourage *P. australis* seeds to settle on the bare substratum, and therefore account for the higher rate of sexual propagation in the shallow areas of St Georges Basin. West et al. (1985) have documented a similar situation in the central coast estuary of Wallis Lake, and this warrants further investigation.

However, the abundance of *P. australis* seedlings and small plants was considerably less than that of other species. For example, the patch formation rate of *Cymodocea nodosa* in Spain, was 0.047 patches m\(^{-2}\) (Duarte and Sand-Jensen 1990) which is many times higher than that of *P. australis* at these sites in St Georges Basin.

Despite low abundance, the recruitment of *P. australis* seedlings and successful patch formation have important implications for the recovery of *P. australis* communities in St Georges Basin. Vegetative propagation has not been the only mechanism of short term natural recovery in this estuary. This means that *P. australis* communities in St Georges Basin have the capacity to form new beds within the fragmented remains of older meadows, as well as colonise areas where damage was so considerable that all seagrass was destroyed. A study by Kendrick et al. (1999) showed that the number of patches positively influences the increase in seagrass area over time, and so it is reasonable to suggest that patch formation in this estuary will substantially decrease the time taken for fragmented communities to become unbroken beds. It is also the only way in which denuded areas can start to generate new *P. australis* communities, without the reliance on mechanical movement of *P. australis* by transplantation.
Nevertheless, sexual propagation still appears to play a minor role in natural recovery of *P. australis* communities over the short term. The production of new shoots on *P. australis* seedlings was slow and the formation of a single plagiotropic rhizome took at least four years. The production of plagiotropic rhizomes, their growth rates and pattern of divergence, dictate how fast *P. australis* patches can grow to form small beds. The oldest plant collected in this study was nine years old and had formed a patch barely half a metre in diameter. Therefore, it seems reasonable to suggest that several decades are likely to elapse before sexual reproduction begins to create *P. australis* habitat with similar functions to established beds, such as forming nursery habitat for fish, contributing to detrital food chains and preventing erosion.

However, the *P. australis* habitats in St Georges Basin do have considerable value as they may represent a rare stock of genetic variability for *P. australis* populations in southern NSW. Little is known about genetic variation of *P. australis* in southern NSW (though see Waycott 1995, Waycott et al. 1997). Since clonal growth through rhizome extension dominates, genetic diversity amongst *P. australis* populations might be expected to be poor (Alberte et al. 1994, Waycott et al. 1997). The effects of low genetic diversity on the capacity of *P. australis* to cope with stress are not known. It is therefore recommended that a study be undertaken to investigate the genetics of *P. australis* populations in St Georges Basin and other estuaries in NSW where sexual propagation has not been recorded. The possible implications this has for the management of *P. australis* populations along the east coast of Australia should also be investigated.
CHAPTER 6: MONITORING SMALL SCALE CHANGES IN _POSIDONIA AUSTRALIS_ COMMUNITIES IN SOUTHERN NSW ESTUARIES AS AN INDICATOR OF HEALTH

6.1 Introduction

In Chapter Three, the status of _P. australis_ in southern NSW estuaries was investigated and it was concluded that, overall, significant areas had been lost from five of the six estuaries. As a result, a program to monitor the health of _P. australis_ communities is required to ensure that this seagrass does not continue to decline in southern NSW estuaries. Such a monitoring program would enable the health of _P. australis_ to be accurately described on a regular basis. This is of considerable importance, given that State of the Environment reporting has become mandatory in NSW (EPA 1997) in order to ensure better management of natural resources. However, in order to be effective, a monitoring program needs to select the best parameters to assess _P. australis_ health and to have sufficient replication to detect small changes, and, to be able to discriminate between natural variability and anthropogenic change. In order to address these three issues, this Chapter has examined small scale changes in _P. australis_ communities in three southern NSW estuaries. Although small scale changes in _P. australis_ communities have been monitored previously, particularly in the Sydney region (Larkum 1976, West 1980, Wolterding 2000), baseline data has not been collected for many of the estuaries in southern NSW. Unlike most previous studies, which have focused on the ecology of _P. australis_, this study investigates the utility of a monitoring program to detect small scale changes in _P. australis_ health.

The objectives of this Chapter are: to review sampling methods that have previously been used to monitor _P. australis_ in southern NSW; to design a monitoring study with high sensitivity and adequate statistical power; to quantify the natural variability of some of the major characteristics of _P. australis_ communities from a wide variety of
spatial and temporal scales; and finally, to determine whether anthropogenic impacts can be separated from a background of natural variability. This final objective is of particular importance. If anthropogenic activities cause changes in *P. australis* health that are greater than those caused by natural variability, then future monitoring programs can be designed specifically to detect these larger changes.

A small scale pilot study was conducted in order to assist in preparing a statistically robust study. A larger scale study was then conducted to examine changes in *P. australis* shoot density, leaf area and flowering shoot density over a two year period at four sites in Port Hacking (PH), Wagonga Inlet (WI) and Merimbula Lake (ML). Shoot density and leaf area were monitored as they are two major characteristics believed to represent the relative health of the seagrass bed (West 1990, Wood and Lavery 2000). Flowering shoot density was monitored in order to gather baseline data for a range of sites in Port Hacking, Wagonga Inlet and Merimbula Lake. As noted in Section 1.5, these estuaries were chosen because they represent the northern (PH) and southern (ML) extremes of the study area, as well as the approximate midpoint (WI); and secondly, they contain large areas of continuous, shallow water *P. australis* beds that can be monitored without use of SCUBA.

### 6.2 Pilot study

#### 6.2.1 Introduction

As there are no well established protocols for monitoring *P. australis* communities in the region (see Section 2.3.2) it was necessary to conduct pilot sampling to choose a suitable monitoring technique, as well as design a statistically robust study. Pilot sampling was conducted in Jervis Bay to determine three important variables: the optimum spatial scale at which to take measurements of *P. australis* communities (quadrat size), the degree of change the monitoring program would be able to detect (effect size), and the number of measurements needed to detect these changes (sample size).
6.2.2 Methods

Shoot density and quadrat size

In previous studies, the quadrat size used to measure *P. australis* shoot density has varied. The most commonly used is a 0.25m x 0.25m, or 0.0625m² quadrat (see Neverauskas 1987, Mellors et al. 1992, Wolterding 2000). The 0.5m x 0.5m quadrat (0.25m²) has also been used in a number of studies (West and Larkum 1979, West 1990, Mellors 1991), while larger quadrats (1m x 1m) have rarely been used (Larkum 1976). Both West (1990) and Mellors (1991) stated that the 0.25m² quadrat is the largest quadrat that can be conveniently used underwater, due to time constraints when using SCUBA. Practical considerations notwithstanding, Larkum (1976) recommended the use of a 1m² quadrat when sampling *P. australis* beds to ensure sampling at an adequate spatial scale. Likewise, a study of the effect of quadrat size and observer bias in estimating seagrass shoots showed that larger sized quadrats gave greater precision than smaller when sampling *P. australis* beds (Inglis and Smith 1995). Given the differing opinions in the literature, it was considered necessary to independently determine the best quadrat size to measure *P. australis* shoot density in this study.

For this pilot work, shallow *P. australis* beds (0.5 - 0.8m depth at low tide) were sampled in Jervis Bay, NSW. A square section of seagrass meadow measuring 10m by 10m was marked off using a weighted buoy at each corner. Ten samples of a 1m² quadrat, 0.25m² quadrat and 0.0625m² quadrat were thrown randomly within the marked area. The total number of *P. australis* shoots was counted in each quadrat.

Leaf area

Ten shoots were randomly collected from within the marked off area and taken back to the laboratory for processing. Leaf height and width were measured for each leaf. Leaf area per shoot was calculated by multiplying total leaf length per shoot by leaf width.
Determining sample size and effect size using power analysis

*A priori* determination of the power of a study makes it possible to make best estimates of the number of replicates required to detect changes of given magnitudes (Bernstein and Zalinski 1983, Green 1993, Heidelbaugh and Nelson 1996). As recommended by Fairweather (1991), the pilot study was used to determine optimum sample size for a range of mean shoot densities and leaf area per shoot. Shoot densities collected for the 1m² quadrat were entered into a spreadsheet and manipulated to obtain a series of ‘modelled’ data sets. These modelled data were created to have similar variability to the real data, but had mean shoot densities that were 5%, 10%, 15%...and up to 50% higher than the mean of the original field collected data. Power analysis was performed on these modelled data and minimum sample size calculated. This analysis was also performed on the leaf area data. This technique is useful in determining the minimum change in the parameter (effect size) that can be determined in different sampling regimes.

6.2.3 Results

**Quadrat size**

To gain a measure of the variability in the data set, the coefficient of variation (C.V) was calculated for each quadrat (Table 6.1). The C.V was found to be lowest for the 1.0m² frame, suggesting it to be the best sized frame to use in sampling beds of *P. australis*, as it has the least variability. This is in agreement with Larkum (1976), though contrasts the recommendation of Inglis and Smith (1995). The latter may have taken into account the time taken to count shoots in the 1m² quadrat.

**Sample size and effect size**

The analysis indicated that a large number of samples were required to detect a 5%, 10% and 15% change in *P. australis* shoot density. For example, 26 replicates (13 per site) were needed to detect a 15% change in mean shoot density. This sample size was beyond the resources of this study. Up to seven minutes was required to count shoot
density in each 1m² quadrat, meaning that 13 replicates would require 91 minutes. However, to detect a 20% change in *P. australis* shoot density only 9.06 samples were required (or about five per site). To achieve a power of at least 0.8 (recommended by Fairweather 1991), 16 replicates were required (or eight per site). It was decided that in this short term monitoring study the effect size that could be practically determined for shoot density of *P. australis* was 20%. To achieve this, a sample size of eight 1m² quadrats was required.

*Leaf area*

With respect to leaf area per shoot, 12 replicates per site were required to detect a 20% difference in mean leaf area per shoot between samples.

**6.3 Methods**

**6.3.1 Data collection**

Areas of *P. australis* in Port Hacking, Wagonga Inlet and Merimbula Lake were monitored for changes in health on the basis of shoot densities and leaf areas. Four sites in each estuary (see Fig. 6.1) were selected randomly and marked out with plastic poles. Each site measured approximately 15m x 10m. Random samples were taken from within these permanent sites every two months, from August 1998 until August 2000. On each occasion:

- Shoot density and flowering shoot density were measured *in situ* in eight 1m² quadrats thrown haphazardly within each site.
- Twelve shoots were randomly harvested, stored on ice for no more than 3 days, and taken back to the laboratory for processing. Leaf height and width were measured for each leaf. Leaf area per shoot was calculated by multiplying total leaf length per shoot by leaf width.
6.3.2 Data analysis

A one-way ANOVA was carried out at the end of the experiment to detect significant differences between sampling intervals at each site for shoot density and leaf area. In some cases, leaf area data were not normal (Shapiro-Wilks test, p=0.05). The data passed some tests for normality but failed others. Due to the robustness of a one-way ANOVA, some degree of non-normality is acceptable and the analysis was carried out without transformation.
Table 6.1. Mean, standard deviation and coefficient of variation for three quadrat sizes used to sample number of shoots in a *Posidonia australis* beds in Jervis Bay, NSW. (n = 10).

<table>
<thead>
<tr>
<th>Quadrat size (m²)</th>
<th>Mean shoots. quadrat</th>
<th>Standard deviation</th>
<th>Coefficient of variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.0</td>
<td>278.3</td>
<td>70.9</td>
<td>25.5</td>
</tr>
<tr>
<td>0.25</td>
<td>67.7</td>
<td>28.6</td>
<td>42.3</td>
</tr>
<tr>
<td>0.0625</td>
<td>17.6</td>
<td>6.0</td>
<td>34.1</td>
</tr>
</tbody>
</table>
Fig. 6.1. Location of sampling sites in (A) Port Hacking, (B) Wagonga Inlet and (C) Merimbula Lake.
6.4 Results

6.4.1 Short term changes in *Posidonia australis* shoot density in Port Hacking, Wagonga Inlet and Merimbula Lake

In Port Hacking, significant differences (p = 0.05) were found between time periods at PHS1, PHS2 and PHS3, though not at PHS4. The minimum difference detected in mean shoot densities ranged from 13.6% at PHS2 to 16.6% at PHS1 (Table 6.2). This exceeded the expectations of the study, which was designed to detect 20% change. At PHS4, the maximum difference in shoot density between sampling intervals was 15%, but this was not statistically significant (power, 0.55).

Shoot densities at PHS1, PHS2 and PHS3 exhibited a weak seasonal pattern for the first year (Fig. 6.2). For example, at PHS1 shoot densities in October and December 1998, and February 1999, were significantly higher than April and June 1999 (Fig. 6.2). A second peak occurred in August 1999, but after this there was no seasonal pattern. This August 1999 peak also occurred at PHS2, PHS3 and PHS4, and was followed by a gradual decline. With the exception of PHS1, shoot density was at its lowest in the final month of monitoring. For example, shoot densities were significantly lower in August 2000 than August 1998 at both PHS2 and PHS3. At PHS3, this decline may be related to a winter dieback of shoots, although the progressive reduction in shoot density, without obvious peak and trough fluctuation, suggests a deterioration in the health of the meadows, unrelated to season, from December 1999 onwards. This deterioration may have been caused by sand movement from an adjacent mobile shoal. Historical aerial photographs of this site suggest that *P. australis* cover in this area has not been stable over the last fifty years (see Section 3.3.1). However, the variability in shoot density at PHS1 and PHS2 made it difficult to detect a progressive downward trend. For example, at PHS2 there was no significant difference in shoot density between August 2000 and the lowest density in the first year of monitoring, April 1999.

In Wagonga Inlet, significant differences (p = 0.05) were found between sampling intervals at WIS1, WIS3 and WIS4, though not at WIS2. The minimum change
detected ranged from 14.1% at WIS4 to 20.1% at WIS1 (Table 6.2). A change in mean shoot densities of 20.7% at WIS2 was not detected as significant (power, 0.55).

Shoot densities at all four sites in Wagonga Inlet exhibited temporal patterns that were irregular and unrelated to season. For example, at WIS1 shoot densities fluctuated in the first 6 months, declined sharply, and then went through a period stability for 12 months (Fig 6.3). This pattern also occurred at WIS3, though there was no overall significant decline because shoot densities were low when monitoring started (Fig. 6.3). At WIS2, despite large variation over time, there was no significant difference in mean shoot density between sampling intervals. At WIS4, shoot densities fluctuated markedly but did not change overall (Fig. 6.3). Only February 1999 was significantly different (p=0.05) from some of the other sampling times.

In Merimbula Lake, significant differences (p = 0.05) were found between sampling intervals at all sites. The minimum change detected in mean shoot densities ranged from 18.3% at MLS2 to 27.4% at MLS4 (Table 6.2).

Mean shoot densities in Merimbula Lake exhibited a seasonal pattern at two sites, MLS1 and MLS2 (Fig. 6.4). Higher shoot densities occurred in the summer months in both 1998 and 1999, and lower shoot densities in the winter months of 1998, 1999 and 2000. (Fig. 6.4). At MLS1, there was a significant difference between February 1999 and June 2000, while at MLS2 there were significant differences between summer and winter months in both years. There was also significant variation between other months. For example, at both MLS1 and MLS2 shoot densities declined in February 2000 to levels similar to that of winter months, but then recovered quickly, before declining again (Fig. 6.4). This decline is February was possibly related to desiccation of *P. australis* leaves during low tides.
Table 6.2. Minimum change detected as significant, and maximum change not detected as significant, in one way ANOVA of sampling intervals at each of four sites in Port Hacking, Wagonga Inlet and Merimbula Lake. Monitoring program was designed to detect a 20% change in mean *Posidonia australis* shoot density and leaf area. Statistical power is presented in parentheses for non-significant results.

<table>
<thead>
<tr>
<th>Site</th>
<th>Shoot density Minimum change detected (%)</th>
<th>Shoot density Maximum change not detected (%)</th>
<th>Leaf area Minimum change detected (%)</th>
<th>Leaf area Maximum change not detected (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>PHS1</td>
<td>16.6</td>
<td></td>
<td>20.0</td>
<td></td>
</tr>
<tr>
<td>PHS2</td>
<td>13.6</td>
<td></td>
<td>24.6</td>
<td></td>
</tr>
<tr>
<td>PHS3</td>
<td>14.8</td>
<td></td>
<td>26.1</td>
<td></td>
</tr>
<tr>
<td>PHS4</td>
<td>15.0 (0.74)</td>
<td></td>
<td>30.7</td>
<td></td>
</tr>
<tr>
<td>WIS1</td>
<td>20.1</td>
<td></td>
<td>34.4</td>
<td></td>
</tr>
<tr>
<td>WIS2</td>
<td>20.7 (0.55)</td>
<td></td>
<td>33.5 (0.89)</td>
<td></td>
</tr>
<tr>
<td>WIS3</td>
<td>15.9</td>
<td></td>
<td>36.5</td>
<td></td>
</tr>
<tr>
<td>WIS4</td>
<td>14.1</td>
<td></td>
<td>33.0</td>
<td></td>
</tr>
<tr>
<td>MLS1</td>
<td>27.4</td>
<td></td>
<td>34.0 (0.93)</td>
<td></td>
</tr>
<tr>
<td>MLS2</td>
<td>18.3</td>
<td></td>
<td>25.5</td>
<td></td>
</tr>
<tr>
<td>MLS3</td>
<td>21.1</td>
<td></td>
<td>27.3</td>
<td></td>
</tr>
<tr>
<td>MLS4</td>
<td>27.3</td>
<td></td>
<td>26.9 (0.92)</td>
<td></td>
</tr>
</tbody>
</table>
Fig. 6.2. Mean *Posidonia australis* shoot density (± S.E) from October 1998 to August 2000 at PHS1, PHS2, PHS3 and PHS4.
Fig. 6.3. Mean *Posidonia australis* shoot density (± S.E) from August 1998 to August 2000 at WIS1, WIS2, WIS3 and WIS4.
Fig. 6.4. Mean Posidonia australis shoot density (± S.E) from August 1998 to August 2000 at MLS1, MLS2, MLS3 and MLS4.
At MLS3, no seasonal pattern in shoot density was observed, and shoot densities declined by about 40% over the course of the study, from 165 shoots.m\(^{-2}\) to 98 shoots.m\(^{-2}\) (Fig. 6.4). Shoot densities in the first eight months were lower than shoot densities in the last 14 months, with the largest loss of shoots occurring between April and August 1999 (Fig. 6.4). The exact cause of this decline in shoot density is unknown, though Meehan (1997) noted that seagrass communities composed of \textit{P. australis}, \textit{Zostera} spp. and \textit{Halophila} spp. adjacent MLS3 have disappeared over the last fifty years, possibly as a result of foreshore development.

A major decline in shoot density also occurred at MLS4. This appeared to be due to propeller damage throughout the study site (the site is adjacent a small boat ramp). This damage occurred mainly between October 1999 and February 2000, though a significant change was only detected between June 1999 and June 2000. Prior to this damage, shoot densities were extremely stable. From February 2000 through to August 2000 there was some evidence of recovery, although there were also large fluctuations in density (Fig 6.4).

6.4.2 Short term changes in \textit{Posidonia australis} leaf area in Port Hacking, Wagonga Inlet and Merimbula Lake

In Port Hacking, significant differences (\(p = 0.05\)) were found between sampling intervals at all sites. The minimum change detected ranged from 20.0% at PHS1 to 30.6% at PHS4 (Table 6.2).

Over the two year period of monitoring, leaf area exhibited a seasonal pattern (Fig. 6.5). In the first year, leaf area at all four sites was significantly higher in winter and early spring, and lower in summer. For example, at PHS1, leaf area October 1998, and June, August and October of 1999, were significantly higher than all other sampling times. Similarly, at PHS2 leaf area in October 1998, June, August and October 1999, and February 2000, were significantly higher than December 1998 and 1999 (Fig. 6.5). However, in the second year of monitoring, leaf area was again low in December at all four sites, but increased in late summer, and then decreased in the winter months. At PHS2 and PHS3 there was a significant fall in leaf area between February
2000 and June 2000; at PHS2, there was also a significant rise in leaf area between December 1999 and February 2000. This inverted seasonal trend in the first year of monitoring is unusual, though leaf area was highest in late winter and early spring, when water temperatures are rising and may be at the optimum temperature for leaf growth.

In Wagonga Inlet, significant differences (p = 0.05) were found between sampling intervals at WIS1, WIS3 and WIS4. The minimum change detected ranged from 33.0 to 36.5% at WIS3, considerably higher than the intention of the study, which was designed to detect a 20% change in mean (Table 6.2). The maximum change not detected at WIS2 was 33.5% (power, 0.89).

Leaf area in Wagonga Inlet sites exhibited a familiar seasonal pattern. At WIS1, WIS3 and WIS4, leaf area was significantly higher in late spring and summer than winter and early spring months (Fig. 6.6). For example, at WIS1, leaf area was significantly higher in October/December 1998; October/December 1999, and February 2000, than April 1999, April 2000 and June 2000. Seasonal changes were also very large. For example, at WIS4 leaf area fell from about 9500 mm$^2$ in December 1998 to 5400 mm$^2$ in June 1999 (Fig. 6.6). A change of similar magnitude occurred in the 1999/2000 seasonal cycle.

WIS2 was the only site to lack a distinct seasonal pattern. In the first ten months of monitoring, leaf area rose and fell randomly. A winter drop off did occur in June 2000 but this was not detected as significant (Fig. 6.6).

In Merimbula Lake, significant differences (p = 0.05) were found between sampling intervals at MLS2 and MLS3. The minimum change detected was 25.5% at MLS2 and 27.3% at MLS3 (Table 6.2). The maximum change not detected was 34.0% at MLS1 and 26.9% at MLS4 (power, 0.93 and 0.92 respectively).
Fig. 6.5. Mean *Posidonia australis* leaf area (± S.E) from October 1998 to August 2000 at PHS1, PHS2, PHS3 and PHS4.
Fig. 6.6. Mean *Posidonia australis* leaf area (± S.E) from October 1998 to August 2000 at WIS1, WIS2, WIS3 and WIS4.
Fig. 6.7. Mean *Posidonia australis* leaf area (± S.E) from October 1998 to August 2000 at MLS1, MLS2, MLS3 and MLS4.
Sites in Merimbula Lake exhibited a similar seasonal pattern to sites in Wagonga Inlet. At all sites, leaf area peaked during the spring and summer months, and declined over autumn and winter. However, significant differences were only detected at two sites. At MLS2, mean leaf area in December 1999 was significantly different from all other sampling intervals except for December 1998, while at MLS3, mean leaf area in spring and summer months were different from the winter months in both years of monitoring. Timing of seasonal changes did vary. At MLS1, MLS2 and MLS3, leaf area was at its lowest in August 1999 and 2000, though at MLS4, lowest leaf area occurred in August 1999 and June 2000. Peak leaf occurred during December 1999 and April 2000 at MLS1, but at MLS3 it occurred during February 1999 and October 2000 (Fig. 6.7).

6.4.3 Separating natural variability from possible anthropogenic declines in *Posidonia australis* communities in southern NSW estuaries

In this study, several changes in *P. australis* shoot density or leaf area that were statistically significant did not necessarily constitute a deterioration in the health of the *P. australis* community, but simply represented natural variation in response to unknown factors, or perhaps sampling error. In the design of a monitoring program, the issue of biologically significant change often arises. A change in shoot density or leaf area that is statistically significant may be of little biological importance. There are two possible causes for change in these parameters, natural variability and anthropogenic impacts. It would be useful if these two factors could be discriminated.

In order to quantify natural variation in *P. australis* communities, and to help discriminate between natural change and possible anthropogenic change, the patterns of change in shoot density and leaf area were examined in more detail.

To do this, an overall mean value for each site was calculated for both *P. australis* shoot density and leaf area. This mean was calculated using the full range of data points where there was no obvious increase or decreasing trend. Where there was an obvious increasing or decreasing trend, only data points prior to the 'impact' were
used to calculate a mean. In other words, the mean represented the average value for shoot density or leaf area at each site in its ‘natural’ state.

The means were plotted on the time-series graphs for shoot density and leaf area. Percentage change lines were plotted around the mean (for example, 80 and 120% of mean) to represent ‘trigger points’ (Fig. 6.8 - 6.13). Trigger points have not yet been applied to seagrass management in NSW, but are used with respect to fisheries data (NSWF 2001a). NSWF trigger points are used to generate management action when a fishery undergoes an unexpected increase or decrease of predetermined magnitude (NSWF 2001a). For the sensible management of any natural resource, trigger points need to be set at a level where they are sensitive enough to detect an anthropogenic impact, but not so sensitive as to detect natural variability. In the case of monitoring *P. australis* health in southern NSW estuaries, these diagrams were constructed with a number of trigger points to see whether it is possible to set limits on what constitutes natural variation, and to quantify what size change should be of concern in terms of management.

With respect to temporal variation in shoot density in Port Hacking, Wagonga Inlet and Merimbula Lake, the use of time series graphs with percentage change boundaries, or ‘trigger points’, demonstrated that natural variation might be separated from possible anthropogenic changes by the magnitude of change, and to a lesser degree, the time over which it is sustained. For example, at seven of twelve sites shoot density fluctuated within 20% of the mean for the two year monitoring period (Fig. 6.8 - 6.10). At MLS3 and MLS4, where there was a substantial decline in shoot density that may have been caused by anthropogenic impacts, variation in shoot density exceeded 20% of the mean, reaching 30% at MLS4 and nearly 50% at MLS3 (Fig. 6.10). At MLS3, this decline was also continuous and progressive. This can be contrasted with abrupt ‘natural’ declines, where the reduction in shoot density was short lived (no greater than four months) then recovered quickly. However, this pattern is by no means applicable to all sites. For example, shoot density decline at MLS4 was also short lived, even though the reduction appeared to be caused by an anthropogenic impact (boat activity).
At the remaining three sites (PHS3, WIS1 and WIS2) fluctuation in shoot density exceeded 20% but did not exceed 30%. As noted earlier, these three sites were difficult to interpret, because although there did appear to be a general decline in density, there was no significant difference between shoot density in the final monitoring period and in some of the earlier monitoring periods. Therefore, changes in mean shoot density at these sites probably represented natural variation, albeit at a greater level than other sites. However, it seems reasonable to suggest that in most cases natural variation in *P. australis* shoot density fell within 20% of the mean for the site, and in certain cases it may have reached 30%. Therefore, in terms of management, an overall change of >30% in mean shoot density should be of concern, as it may represent an anthropogenic impact. Therefore, a trigger point should be set at 30% of the overall mean of the site.

The situation with respect to fluctuations in *P. australis* leaf area was very different. At only one site, MLS4, did leaf area fluctuate within 20% of the mean. At six out of twelve sites leaf area fluctuated up to and above 40% of the mean, while at the remaining five sites it fluctuated around about 30% of the mean. This high degree of variability in leaf area was largely the result of seasonal change, although there were also cases where leaf area rose and fell irrespective of season. As a result, leaf area does not seem to be as useful as a parameter to monitor changes in *P. australis* communities, as it has high natural variability, which would mask any consistent long term change.

6.4.4 Flowering of *Posidonia australis* communities in Port Hacking, Wagonga Inlet and Merimbula Lake

In Port Hacking, flowering occurred at only 1 site, PHS4 (Table 6.3), in both 1998 and 1999. PHS4 was the deepest and densest bed surveyed in Port Hacking. Flowering shoot density was 26.3 ± 3.8 m\(^2\) in October 1998 (or 11%), but substantially lower in August 1999, at 2.6 ± 1.2 peduncles m\(^2\) (or <1% of total). The remaining three sites are shallower and less dense than PHS4, and are also subject to periodic desiccation during very low tides. Flowering shoots were not observed at any time at these sites.
Fig. 6.8. An assessment of natural variation in *Posidonia australis* shoot density at PHS1, PHS2, PHS3 and PHS4. The overall mean was calculated using all twelve sampling times. With the exception of PHS3, all data points are contained within the 80% and 120% boundaries.
Fig. 6.9. An assessment of variation over time in *Posidonia australis* shoot density at WIS1, WIS2, WIS3 and WIS4. The overall mean was calculated using all twelve sampling times. 70% and 130% boundaries were needed to contain data points for WIS1 and WIS3.
Fig. 6.10. An assessment of variation over time in *Posidonia australis* shoot density at MLS1, MLS2, MLS3 and MLS4. For MLS1 and MLS2, the overall mean was calculated using all twelve sampling times. For MLS3 the overall mean was calculated using the first five data points (as they represented a period of initial stability and were statistically different to the remaining points). For MLS4 the overall mean was calculated using the first eight data points (as they also represented an initial period of stability prior to impact). 70% and 130% boundaries were needed to contain data points for MLS4, while MLS3 fell to almost the 50% boundary.
Fig. 6.11. An assessment of natural variation in *Posidonia australis* leaf area at PHS1, PHS2, PHS3 and PHS4. The overall mean was calculated using all twelve sampling times. The 60% and 140% boundaries were needed to contain data points, although first data point at PHS1, PHS2 and PHS3 exceeded even these boundaries.
Fig. 6.12. An assessment of variation over time in *Posidonia australis* leaf area at WIS1, WIS2, WIS3 and WIS4. The overall mean was calculated using all twelve sampling times. 70% and 130% boundaries were needed to contain data points for WIS1, WIS2 and WIS4, while 50% and 150% were used for WIS3.
Fig. 6.13. An assessment of variation over time in *Posidonia australis* leaf area at MLS1, MLS2, MLS3 and MLS4. The overall mean was calculated using all twelve sampling times. MLS1 and MLS4 were contained by the 80% and 120% boundaries, while MLS2 strayed as high as 140%.
Table 6.3. Mean number of *Posidonia australis* flowering shoots (± S.E.), and flowering shoots as % of total, at sites located in Port Hacking (PH), Wagonga Inlet (WI) and Merimbula Lake (ML) 1998 and 1999. August 1998 and 1999 were the two times when flowering density was at its highest.

<table>
<thead>
<tr>
<th>Site</th>
<th>Mean no. flowering shoots.m⁻² (± SE)</th>
<th>As a % of</th>
<th>Mean no. flowering shoots.m⁻² (± SE)</th>
<th>As a % of</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>in August 1998</td>
<td>in August 1999</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PHS1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>PHS2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>PHS3</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>PHS4</td>
<td>26.3 ± 3.8</td>
<td>11</td>
<td>2.6 ± 1.2</td>
<td>&lt;1</td>
</tr>
<tr>
<td>WIS1</td>
<td>4.3 ± 1.2</td>
<td>3</td>
<td>3.6 ± 1.0</td>
<td>3</td>
</tr>
<tr>
<td>WIS2</td>
<td>15.9 ± 4.2</td>
<td>7</td>
<td>13.4 ± 4.1</td>
<td>5</td>
</tr>
<tr>
<td>WIS3</td>
<td>0.1 ± 0.1</td>
<td>&lt;1</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>WIS4</td>
<td>2.1 ± 1.1</td>
<td>1</td>
<td>1.3 ± 0.5</td>
<td>1</td>
</tr>
<tr>
<td>MLS1</td>
<td>8.9 ± 2.0</td>
<td>6</td>
<td>1 ± 0.5</td>
<td>1</td>
</tr>
<tr>
<td>MLS2</td>
<td>23 ± 6.4</td>
<td>13</td>
<td>1.3 ± 0.5</td>
<td>1</td>
</tr>
<tr>
<td>MLS3</td>
<td>22.9 ± 2.3</td>
<td>14</td>
<td>4.5 ± 0.5</td>
<td>4</td>
</tr>
<tr>
<td>MLS4</td>
<td>26.3 ± 5.9</td>
<td>8</td>
<td>13.8 ± 2.8</td>
<td>4</td>
</tr>
</tbody>
</table>
In Wagonga Inlet, flowering shoots were found at all four sites in August 1998, but only at WIS1 and WIS2 in August 1999 (Table 6.3). The mean number of flowering shoots was highest at WIS2 in both years, ranging from about 13 to 16 shoots.m\(^{-2}\) (or 5 - 7% of total). At WIS1 mean no. of flowering shoots was about 4.0 shoots.m\(^{-2}\) (or 3%), but at WIS3 was 0.1 shoot.m\(^{-2}\).

In Merimbula Lake, flowering occurred at all four sites in both August 1998 and 1999. In August 1998, the mean number of flowering shoots was high at all four sites, ranging from 8.9 ± 2.0 shoots.m\(^{-2}\) (6%) to 26.3 ± 5.9 shoots.m\(^{-2}\) (8%). At MLS3 percentage of shoots flowering was 14%. Similar to Wagonga Inlet and Port Hacking, the mean number of flowering shoots was substantially lower in August 1999, ranging from 1.0 ± 0.5 shoots.m\(^{-2}\) (1%) to 13.8 ± 2.8 shoots.m\(^{-2}\) (4%).

P. australis seedlings were not observed during in situ counts of P. australis shoots, despite the high rate of flowering and fruiting.

6.5 Discussion

In situ monitoring of small scale changes in several characteristics of P. australis communities in three southern NSW estuaries, found considerable variability. Some of the variability in P. australis shoot density and leaf area appeared to be related to natural processes, such as changes of season, and anthropogenic impacts, such as boat damage, while the remainder did not relate to any obvious environmental factors. Monitoring of P. australis communities in these southern NSW estuaries was able to detect small changes in P. australis health, and also identify what size change should act as a trigger point to initiate management action, provided that changes in mean shoot density, not leaf area, were used.

The results of the pilot study suggested that previous monitoring studies in south east Australian waters, with the exception of Larkum (1976), have not used the optimum sized quadrat to monitor P. australis shoot density. These studies may also have had insufficient replication to detect significant changes in P. australis health. In this
study, eight replicates of the 1m² quadrat was able to detect changes in mean shoot density of about 20% at eight out of a possible twelve sites. Consequently, the use of eight 1m² quadrats is recommended for future monitoring programs of *P. australis* in southern NSW estuaries.

Measurements of *P. australis* shoot density were an effective measure of seagrass health. Small natural variation meant that changes of 15 to 20% were able to be detected as statistically significant, and the absence of a strong seasonal trend meant that any consistent decline appeared as an obvious downward trend in the time series graphs. This is particularly evident at two sites, MLS3 and MLS4. The low natural variation in *P. australis* shoot density also meant that when a mean was calculate for the data set, and percentage change lines (trigger points) were placed around that mean, most natural variation in shoot density fell within 20% of the mean, while larger changes that may have been the result of anthropogenic impacts, fell above 30%.

Therefore, it is recommended that a trigger point be placed at 30% of the overall mean shoot density for a site, in future monitoring programs. To be clear, this degree of change is a 30% change over a number of sampling intervals. A change of this magnitude may not represent the effects of an anthropogenic impact, but it is a large enough change to be of concern with respect to management. Such changes require further exploration or review to identify the causes. However, longer term studies at monitoring stations are needed to better set the frequency of sampling of the trigger points.

Unlike *P. australis* shoot density, *P. australis* leaf area did not prove to be a suitable measure to detect changes resulting from possible anthropogenic impacts. Large variation in leaf area meant that only changes as large as 25 to 35% were able to be detected. In addition, large seasonal change, up to 30 or 40% over a two month period, meant that progressive downward or upward trends could not be easily identified. Perhaps most importantly, deterioration in shoot density did not show a corresponding change in leaf area. For example, the loss of the shoots at MLS4 did not appear as a decline in leaf area; and at MLS3, despite the 40% reduction in shoot density, leaf
area exhibited a strong and consistent seasonal trend in both years. *P. australis* leaf area is obviously responsive to natural change, as evidenced by its seasonal change, but does not appear to be particularly sensitive to possible anthropogenic impacts. West (1990) also noted that *P. australis* leaf area was highly variable, mainly because *P. australis* leaves were susceptible to storm damage.
CHAPTER 7: EXPERIMENTAL TRANSPLANTING OF *POSIDONIA AUSTRALIS* IN PORT HACKING, NSW, TO ASSESS THE FEASIBILITY OF RESTORATION

7.1 Introduction

There has been a large decline in *P. australis* communities in southern NSW estuaries over the last fifty years (see Section 3.3.5). However, changes in resource use and improved management of estuarine environments over the last decade have meant that many of the causes of *P. australis* decline no longer pose a threat. For example, shell grit mining in Port Hacking, which directly caused losses of *P. australis* meadows, ceased in 1973 and is very unlikely to occur again. Likewise, the small townships on the foreshore of St Georges Basin have recently been placed on reticulated sewerage, thereby eliminating septic system overflow, which is the most likely cause of declines in *P. australis* in this estuary (SCC 1998). Yet despite an improvement in conditions for *P. australis* growth, natural recovery has been slow. It was demonstrated that *P. australis* communities in Jervis Bay may take up to 100 years to fully recover, even though initial damage was small and localised. Also, the data on *P. australis* rhizome growth rate, seedling establishment and patch development presented in Chapter Five, demonstrated the slow rate of sexual and vegetative growth of *P. australis* communities. Damaged *P. australis* beds are unlikely to re-establish naturally, to pre-development levels, in the medium term (less than ten years). As a result, restoration of *P. australis* meadows may be promoted with the use of transplanted shoots and rhizomes, taken from healthy sites.

Unfortunately, *P. australis* has proven difficult to transplant. In Australia, Kirkman (1997) reported that no trial restoration program had yet succeeded in increasing the amount of seagrass habitat. The one transplant study conducted in southern NSW was unsuccessful (West et al. 1990). Similarly, a trial transplant program conducted in St Georges Basin recorded transplant mortality of >90% in less than two months, due to an inadequate anchoring system (plastic mesh), heavy swell conditions and commercial hauling across the transplant site (Meehan, unpublished data). Higher
rates of survival have been achieved for *Posidonia* spp. elsewhere. For example, *P. sinuosa* and *P. coriaceae* were transplanted in Western Australia (Paling et al. 2001) using an underwater turf cutter to harvest large ‘sods’ of *Posidonia* from the sea floor and transplant them in denuded areas. Survival rate of between 75% and 77% after two years was reported. While this method may be appropriate for some sites, for example, where large-scale impacts are expected on existing beds, it is not useful for small-scale transplanting, such as attempts to restore degraded areas without damaging existing beds. In such cases, the process of removing large ‘planting units’ from existing *Posidonia* beds is likely to cause considerable long term damage.

For the purposes of feasibility trials and small-scale rehabilitation projects, a reliable and successful method to transplant *P. australis* is needed. This Chapter deals with an assessment of the feasibility of transplanting healthy *P. australis* rhizomes, attached to an anchoring system, at a range of sites in Port Hacking (NSW). Sites were chosen to assess the effect of location on survival rate and growth dynamics, in particular the ability of the transplants to colonise surrounding substrate. In addition, the method was designed to ensure minimal impact at the donor site.

### 7.2 Methods

#### 7.2.1 Choosing sites for *Posidonia australis* transplants in Port Hacking

The seagrass transplant study was carried out in Port Hacking. Historical mapping of seagrass beds in Port Hacking (see Section 3.3.1) documented an 18% (7 ha) loss of *P. australis* meadows and a 75% (26 ha) loss of mixed *P. australis*, *Zostera* and *Halophila* beds. The areas that had sustained damage to *P. australis* communities were identified by analysing the historical maps. The GIS vector layer showing the distribution of *P. australis* in 1951 was overlayed onto the GIS layer showing the distribution of *P. australis* in 1999, making it possible to identify areas where *P. australis* had declined. Once these areas had been identified, the boundary of the affected area was delineated, and the overall loss of *P. australis* was measured. This was carried out by re-analysing the vector data layers for each site in the earliest and most recent maps.
Five main areas were identified as having lost significant amounts of *P. australis* since 1951: Gunnamatta Bay (GB), Burraneer Bay (BB), Lilli Pilli Point (LPP), Red Jacks Point (RJP) and Cabbage Tree Basin (CTB) (Table 7.1, Fig 7.1 and 7.2). As noted by Fonseca et al. (1988), site quality is a major problem in successfully transplanting seagrass, and rehabilitation should only be attempted at sites where the original impact has ceased. Prior to transplanting, the suitability of each Port Hacking site for seagrass rehabilitation was assessed in light of the environmental impact that had caused the original loss. All sites were found to be suitable, except for CTB, which was found to have shallowed too much to support *P. australis*. In the remaining four areas, a transplant site was chosen randomly. A control site (CS) was chosen on a bare patch of sand between existing *P. australis* beds (Fig. 7.2). All sites were characterised by shallow water (<1m) at Mean Low Water (MLW), sand substrate and the presence of remnant *P. australis* beds. *Z. capricorni* was present at RJP.

### 7.2.2 Choosing a methodology to transplant *Posidonia australis* in Port Hacking

As noted by Lord et al. (1999) the technology for transplanting seagrass using sprigs and turfs has been well tested. Turfs were not deemed suitable in this experiment, as the weight of even small turfs of *P. australis* prohibits the use of manual planting. Removing large sections of the *P. australis* meadow was also viewed as undesirable, as bare patches would have been likely to exist for several years, possibly making the community vulnerable to erosion. Instead, small vegetative sections of *P. australis* rhizome were used.
Table 7.1. Area of seagrass loss and likely cause of decline in sites within Port Hacking, NSW, since beginning of aerial photography.

<table>
<thead>
<tr>
<th>Site</th>
<th>Area Lost</th>
<th>Likely cause of decline</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gunnamatta Bay (GB)</td>
<td>64% (16 ha) of <em>P. australis</em> and <em>Z. capricorni</em> since 1930.</td>
<td>Severe storm activity in early 1960’s and 1970’s. Bait digging may also have contributed to seagrass loss.</td>
</tr>
<tr>
<td>Burraneer Bay (BB)</td>
<td>67% (5 ha) of <em>P. australis</em> and <em>Z. capricorni</em> since 1942.</td>
<td>Dumping of dredge sands onto nearby spit which caused channel migration across entrance of Bay and erosion of seagrass.</td>
</tr>
<tr>
<td>Lilli Pilli Point (LPP)</td>
<td>83% (7 ha) of <em>P. australis</em> and <em>Z. capricorni</em> since 1951.</td>
<td>Natural progradation of marine tidal delta, in combination with channel dredging, causing erosion of seagrass.</td>
</tr>
<tr>
<td>Red Jacks Point (RJP)</td>
<td><em>P. australis</em> appears to have been out-competed by <em>Z. capricorni</em>.</td>
<td><em>P. australis</em> loss possibly due to nearby shell grit mining.</td>
</tr>
<tr>
<td>Cabbage Tree Basin (CTB)</td>
<td>92% (12 ha) of mixed <em>P. australis</em>, <em>Z. capricorni</em>, <em>Halophila</em> spp. since 1942.</td>
<td>Large-scale sand migration into the Basin smothered seagrass beds.</td>
</tr>
</tbody>
</table>
Figure 7.1. Comparative maps of *Posidonia australis* and other seagrass communities in Gunnamatta Bay and Burraneer Bay, Port Hacking, in 1951 and 1999. Transplant sites have been located in areas of previous *Posidonia australis* growth.
Figure 7.2. Comparative maps of *Posidonia australis* and other seagrass communities near Lilli Pilli Point, Port Hacking, in 1951 and 1999. Transplant sites have been located in areas of previous *Posidonia australis* growth.
In July 1999, sections of *P. australis* rhizome were collected from a long established bed in the marine tidal delta of Port Hacking. This bed was chosen because a high proportion of the shoots (>20%) exhibited plagiotropic morphology. Rhizomes were loosened from the sediment then broken from surrounding plants. Each rhizome section was approximately 20 to 30 cm in length and consisted of 1 to 3 plagiotropic shoots and 2 to 4 orthotropic shoots. Rhizome sections were selected on the basis of fast growth rates, as demonstrated by long internodes and robust physiology, following recommendations made by Piazzi et al. (1998) for *P. oceanica*. About 175 of these sections were collected randomly from an area measuring 400m², and then stored in seawater for a maximum of 2 hrs (Fig. 7.3). It was estimated that less than 0.5% of the total number of shoots in the area were used and random selection over a large area meant that percent cover did not measurably change, thus ensuring there were negligible impacts on the existing beds.

Between 6 and 8 of these sections were attached to steel mesh quadrats (0.5m x 0.5m), using plastic electrical ties, and oriented such that the rhizomes could spread into the surrounding bare areas (Fig. 7.4). Four plagiotropic shoots on each quadrat were tagged by placing a small electrical tie around the most recent internode. Five of these quadrats were then placed at each of the five sites (GB, BB, LPP, RJP, CS; Fig. 7.1 and 7.2). The transplant units were positioned on bare substrate (except at RJP where it was placed over existing *Z. capricorni* beds), so that the metal mesh was above the rhizomes, thereby protecting them from damage.

### 7.2.3 Data analysis

After transplantation, the number of shoots on each quadrat was counted. Thereafter, the number of living shoots was recorded on a bi-monthly basis and percent survival calculated for each unit. After 16 months, 10 tagged rhizomes were harvested from the CS, BB and RJP sites. Internode length on both rhizomes and shoots were measured and the following formulas used to calculate growth (see Duarte et al. 1994):

\[
\text{Rhizome elongation (cm yr}^{-1}\text{)} = 365 \times \text{new rhizome material (cm)} + \text{time interval (days)}.
\]

\[
\text{Plastochrone interval (days), horizontal shoots} = \frac{\text{time interval (days)}}{\text{nos. of new internodes}}.
\]
To calculate plastochrone interval (P.I.) for vertical shoots, the production ratio of rhizome internodes was compared to shoot internodes, such that:

$$\text{P.I. (days), vertical shoot} = \frac{\text{nos. rhizome internodes} \times \text{nos. shoot internodes}}{\text{rhizome P.I.}}$$

$$\text{Shoot elongation (cm.yr}^{-1}\text{)} = \frac{365 \times \text{Shoot P.I. (days)}}{\text{Mean shoot internode length (cm)}}.$$ 

Growth architecture was documented using a system of numbering similar to Molenaar et al. (2000) (see Section 2.2.4). An Order-1 axis was the principal plagiotropic rhizome tagged at the beginning of the experiment. Order-2 axes were the lateral branches produced by an Order-1 axis; Order-3 axes the lateral branches produced by Order-2 axis Orthotropic shoots were the vertical, slow growth units, while transitional shoots were shoots in intermediate state between plagiotropic and orthotropic. From each tagged plagiotropic rhizome the following was recorded: number of Order-1, -2 and -3 rhizomes produced; number of orthotropic shoots produced; and, number of transitional shoots produced.

A one-way ANOVA was carried out data collected at the end of the experiment to detect significant differences between sites for the following variables: nos. of shoots; rhizome elongation rate; rhizome P.I.; shoot elongation rate; and, shoot P.I. In some cases, data were not normal (Shapiro-Wilks test, p=0.05) and required transformation (log(x + 1)).
Figure 7.3. Transplant unit prior to transplanting.

Figure 7.4. Transplant units *in situ* at Burraneer Bay (BB), Port Hacking, July 1999.
7.3 Results

7.3.1 Survival of *Posidonia australis* transplants

Experimental transplanting of *P. australis* in Port Hacking was successful at most sites. Transplants at four of the five sites exhibited high survival rates in the short term (less than 6 months) and transplants at three of the five sites exhibited high percentage survival in the long term (greater than 12 months) (Fig. 7.5). A total of 575 shoots were transplanted to five sites and, despite the total loss of seagrass at two sites, 650 shoots were present at the end of the study. This was due to good growth of *P. australis* shoots transplanted to the BB site (an increase of 253 shoots) and fair growth of shoots transplanted to the CS (an increase of 67 shoots). At the RJP site, survival rate of shoots was 78%.

Transplants at the GB site died after 8 months (Fig. 7.5). After initial losses in the first 2 months, the transplants appeared to recover slightly and show some signs of growth after 4 months (Fig. 7.5). However after 6 months, many of the shoots were less than half their original length, and some of the longer rhizomes had come loose from the sediment. After 8 months the transplants at GB were buried by sand and the area had been colonised by *Z. capricorni* and *Halophila* spp. The LPP site also suffered complete mortality of transplants, although this did not occur until after 10 months (Fig. 7.5). Prior to this, the number of surviving shoots was relatively high and stable and at eight months had actually increased by an average of 3 shoots per quadrat (Fig. 7.5). Some undermining of the steel mesh was observed at 6 months, but this did not appear to affect the transplants. However, the migration of an adjacent shoal eventually eroded the transplants, as well as destroying all *Z. capricorni* beds in the vicinity of the transplant site.
Figure 7.5. Mean number of *Posidonia australis* shoots (± S.E) at the five transplant sites in Port Hacking: Gunnamatta Bay, Burraneer Bay, Lilli Pilli Point, Red Jacks Point and control site between July 1999 and November 2000.
By contrast, survival of the *P. australis* transplants at the BB site was consistently high and never fell below the initial planting density. Total number of shoots increased steadily and had more than tripled after 16 months (Fig. 7.5). Overall, the mean number of shoots per quadrat increased from 22.4 to 73.0. Growth was particularly strong in the last two months, when the mean number of shoots per quadrat increased from 50.2 to 73.0, the largest increase over a 2 month period.

The control site lost transplanted shoots in the first 2 months, but then growth was steady. The increase in the mean number of shoots per quadrat was significantly less than at BB, going from 23.0 to 37.0. The overall increase of shoots also varied considerably between quadrats and one quadrat actually exhibited no overall growth. The large increase in mean number of shoots over the last 2 month was also observed at the control site, which increased from 29.8 to 37.0.

The remaining site, RJP, initially showed promising results, with a mean increase of approximately 3 shoots per quadrat after 4 months. However, the total number of shoots decreased quickly over the next 4 months, coinciding with the spring and summer growth of *Z. capricorni*. Number of transplanted shoots at RJP reached a minimum at 8 months (Fig. 7.5), after which survival rate was reasonably stable.

### 7.3.2 Rhizome and shoot growth of *Posidonia australis* transplants

Mean rhizome elongation rate (± SE) of transplanted *P. australis* shoots was 15.2 ± 2.4 cm.yr⁻¹ at the control site (CS), 22.3 ± 1.4 cm.yr⁻¹ at BB, and 9.1 ± 1.0 cm.yr⁻¹ at RJP (Fig. 7.6). Significant differences (p=0.05) were detected between all sites. Rhizome P.I. was 30.0 ± 1.4 days at the control site, 28.4 ± 1.3 days at BB, and 39.9 ± 3.3 days at RJP (Fig. 7.6). Rhizome P.I. was significantly higher (p=0.05) at RJP compared to the other two sites.

Shoot growth rates of transplanted *P. australis* and shoot plastochrone intervals displayed different patterns of variation between sites. Shoot elongation was 2.3 ± 0.2 cm.yr⁻¹ at the control site (CS), 2.8 ± 0.4 cm.yr⁻¹ at BB, and 2.4 ± 0.4 cm.yr⁻¹ at RJP, but there were no significant differences (p=0.05) between these sites (Fig. 7.7). Shoot
P.I. was 38.0 ± 2.6 days at the control site, 41.5 ± 3.4 days at BB and 48.3 ± 3.4 days at RJP (Fig. 7.7). Shoot P.I. at RJP was significantly higher (p=0.05) than the control site but not different from shoot P.I. at BB.

7.3.3 Rhizome and shoot morphology of *Posidonia australis* transplants

In addition to significant rhizome and shoot growth, there was significant branching and shoot production. Transplants at BB produced a total of 15 order-2 rhizomes, compared to 7 at the control at 8 at RJP (Table 7.2). A smaller number of order-3 rhizomes were also produced, with transplants at the control and BB sites producing 4 and 3 respectively, and at RJP producing 1 (Table 7.2). The low production of order-3 shoots at the RJP site was because many of the order-2 rhizomes were damaged during the experiment. Transplants at the BB site were the only ones to produce order-4 rhizomes.

Transplanted *P. australis* at the BB site produced the highest number of orthotropic shoots (23), compared to the transplants at the control site (16) and at RJP (7) (Table 7.2). Transplants at the control site and BB site both produced 6 transitional shoots, while those at the RJP site produced none.
Figure 7.6. (A) *Posidonia australis* rhizome elongation ± S.E (cm.yr⁻¹) for Burraneer Bay, Control and Red Jacks Point. Significant differences detected between all sites (p=0.05). (B) Rhizome Plastochrone Interval ± S.E (d). Red Jacks Point was significantly different (p=0.05). (C) Shoot elongation ± S.E (cm.yr⁻¹). No significant differences were detected between sites (p=0.05). (D) Shoot Plastochrone Interval ± S.E. Red Jacks Point was significantly different from control but not Burraneer Bay (p=0.05).
Table 7.2. Growth characteristics of *Posidonia australis* shoots transplanted to three surviving sites within Port Hacking, NSW, a CS, RJP and BB*.

<table>
<thead>
<tr>
<th></th>
<th>Control</th>
<th>Red</th>
<th>Burraneer</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Jacks Pt</td>
<td>Bay</td>
<td></td>
</tr>
<tr>
<td>No. Order 2 Plagiotropic axis</td>
<td>7</td>
<td>8</td>
<td>15</td>
</tr>
<tr>
<td>No. Order 3 Plagiotropic axis</td>
<td>4</td>
<td>1</td>
<td>3</td>
</tr>
<tr>
<td>No. Order 4 Plagiotropic axis</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>No. Orthotropic shoots produced</td>
<td>16</td>
<td>7</td>
<td>23</td>
</tr>
<tr>
<td>No. Transitional shoots produced</td>
<td>6</td>
<td>0</td>
<td>6</td>
</tr>
</tbody>
</table>

*a* see Figure 7.1 and 7.2 for locations of these sites.
7.4 Discussion

Over the last fifty years, Port Hacking, like many other estuaries in south east Australia, has lost considerable areas of *P. australis* due to anthropogenic impacts (see Chapter Three). Although many of these impacts have ceased in Port Hacking, such as the dredging in seagrass beds, revegetation of damaged areas by *P. australis* has been negligible. Neither small plants nor seedlings have been found in any of the areas studied. Given that natural recovery will be slow (see Chapters Four and Five), the aim of this study was to assess the feasibility of small-scale transplanting as a means of restoration. Despite losses at two sites, experimental transplanting of *P. australis* into damaged areas in Port Hacking successfully increased the amount of seagrass habitat in the Port, with survival at 3 of the 5 sites, and increases in shoot numbers at 2 of the sites. Transplant units exhibited significant rhizome growth, rhizome branching and shoot growth. In the case of Burraneer Bay and the control site, growth dynamics of transplants were similar to that of naturally growing plants (see Kirkman and Reid 1979, West and Larkum 1979, West 1990, Section 5.3.1). Shoot production at these two sites was also high, with the total number of shoots increasing by 60% at the control site and more than 300% at Burraneer Bay.

Ultimately, the survival and growth of transplants was determined by whether the factors that caused original loss were still operating in the study area. As documented in Section 3.3.1, the loss of seagrass communities in the entrance to Gunnamatta Bay was the result of regularly high wave energy and the subsequent migration of sand across the exposed shoal. Bait digging and recreational boating may also have contributed to this loss. Since 1961, the shoal has been characterised by a shifting mosaic of semi-stable *P. australis* beds and opportunist *Z. capricorni* and *Halophila* beds. The latter colonise and expand when conditions are suitable but die off quickly when sand migration re-occurs or heavy storm waves hit the area. The death of the *P. australis* transplants at this site was probably the result of these same factors; the sloughing of longer leaves and the loosening of the rhizomes from the sand supports this hypothesis. Given the degree of natural change at this area, further transplanting is unlikely to be successful.
Large-scale sand movements were also the likely cause of transplant mortality at Lilli Pill Point. Although shoal movement in this area is part of the natural progradation of the marine delta upstream, channel dredging appears to have increased shoal mobility. The changes in substratum mobility were illustrated clearly over the 16-month study. In July 1999, much of the study area was colonised by *Z. capricorni* beds 1 to 5m in diameter. Over time, the sand surrounding these beds was scoured to a depth 0.1 to 0.5m below that of the *Z. capricorni* patches. By May 2000, all *Z. capricorni* beds had been eroded away. This erosion coincided with the destruction of the *P. australis* transplants at this site.

The long term decline of seagrasses at Burraneer Bay was due to the augmentation of nearby Deeban Spit with dredge spoil, which forced the main east-west channel across the mouth of Burraneer Bay to migrate into the Bay. This resulted in sedimentation along the previously stable drop over and the erosion of *P. australis* beds (Druery and Hurrell 1986). The high survival rate and considerable growth of the *P. australis* transplants at the Burraneer Bay site suggests that this site may now be considerably more stable and perhaps suitable for further transplanting experiments. Although the steel mesh used to anchor transplants was buried by 20 to 30 cm of sand by the end of the study, this did not appear to inhibit growth of the transplants at this site. Rhizome elongation, branch formation and shoot production were the highest recorded, and considerably greater than the control site. It would appear that accretion of sand does not inhibit survival or growth of *P. australis*, though as demonstrated earlier, erosion does not favour transplant survival.

The same can be said for the control site, which also had high survival and growth rates of *P. australis* transplants. At the control site, survival rates were lower than Burraneer Bay due to higher initial mortality of shoots, and the unusually high shoot mortality on one of the transplant units. In addition, intermittent sand movement affected the transplants throughout the experiment, possibly limiting the light available for photosynthesis. This may have been responsible for the slower rhizome elongation rate at this site compared to the Burraneer Bay site, although a change in morphology from plagiotropy to orthotropy did occur on some of the control rhizomes, which would also account for the slower growth.
At Red Jacks Point, growth of transplanted *P. australis* shoots was inhibited by desiccation and competition from *Z. capricorni*. In summer at this site, *Z. capricorni* grew to such a density that the *P. australis* transplants could barely be seen. The transplanted *P. australis* shoots appear to have reached an equilibrium at the site and have not expanded further.

The results of these experiments have demonstrated that transplanting *P. australis* shoots, while difficult, can be successful in increasing seagrass habitat. However, candidate sites for rehabilitation should be subject to a feasibility study prior to transplanting, otherwise the likelihood of survival will be low (Fonseca et al. 1988). Suitable sites for further transplantation do exist in Port Hacking, but need to be carefully identified in light of the long term changes that have occurred to seagrass beds in the Port. Despite suitable water depth, substrate conditions and the presence of pioneer seagrass species, some areas of Port Hacking are obviously still subject to environmental impacts, and are presently unsuitable for *P. australis* transplants. West et al. (1990) came to a similar conclusion for denuded sites in Botany Bay, which were still subject to the impacts that had caused the original losses to the seagrass beds. In Port Hacking, Burraneer Bay provides a good example of an area which has sustained significant loss of *P. australis* but now appears ready for restoration. This location would be an ideal site for transplanting *P. australis* shoots at a larger scale.
CHAPTER 8. GENERAL DISCUSSION

8.1 Introduction

This Chapter discusses the major findings of this study with respect to the conservation status of *P. australis* in south east Australia. Firstly, an overview is provided that addresses the questions raised in Section 1.2. Secondly, management implications and future research topics are discussed.

8.2 Overview

8.2.1 *What is the present status of Posidonia australis in southern NSW estuaries and have changes in area occurred due to catchment and waterway development?*

In southern NSW, the area of *P. australis* has declined in five out of six estuaries over the last fifty years. These declines can be correlated with a number of anthropogenic impacts, including engineering works, dredging, and, deterioration in water quality caused by foreshore residential development. Natural impacts, such as storm damage and sand movement, have also reduced the area of *P. australis* at some sites, but these losses were small compared to those caused by anthropogenic impacts. Although the loss of *P. australis* was widespread in southern NSW, the response of *P. australis* to both anthropogenic and natural impacts varied between and within estuaries. For example, in both Port Hacking and St Georges Basin, shallow *P. australis* meadows underwent the largest decline, while deeper beds were impacted less by waterway development. In other instances, *P. australis* remained stable in one geomorphic zone of an estuary, but declined in another. The general hypothesis that seagrass loss is the result of deteriorating water quality, which reduces light intensity (PAR), and causes a reduction in the maximum depth limit that seagrass occurs, is not fully supported by this study.

The status of *P. australis*, in terms of area covered, has improved in only one estuary, Wagonga Inlet. In this estuary, it appeared that modifications made to the entrance
channel resulted in a substantial increase in *P. australis* in the lake basin of the estuary, probably due to the increased tidal prism and tidal range. The greater exchange rate may have increased PAR and salinity and enabled *P. australis* to expand its range.

**Major findings**

- In Port Hacking, seagrass cover declined by 54%, from 181.1 ha (1951) to 83.6 ha (1999). *P. australis* communities declined by 18%, from 38.9 ha (1951) to 31.9 ha (1999). This loss of *P. australis* occurred entirely in the Marine Tidal Delta Zone, particularly between 1951 and 1975. Shell grit mining, channel dredging, engineering works, eutrophication, sedimentation and storm activity all appear to have contributed to this loss.

- In St Georges Basin, seagrass cover declined by 27%, from 376.3 ha (1961) to 278.9 ha (1998). The Central Mud Basin zone lost 32% (95.4 ha) of its seagrass communities in two main periods, 1961-1970, and 1979 - 1998. Like Port Hacking, the smallest relative change occurred in monospecific *P. australis* beds, which declined overall by 13% (12.7ha). Decrease in *P. australis* cover was restricted entirely to the Central Mud Basin zone between the years 1979 and 1998, and was due to the fragmentation and loss of *P. australis* beds growing in shallow water on the northern and southern shore. An increase in freshwater runoff and nutrients from residential areas may be responsible for the loss of *P. australis*. An overall increase in *Ruppia* spp. during this time may substantiate this theory, as *Ruppia* spp. has been shown to increase under these circumstances.

- Seagrass cover declined slightly (10%) in Wagonga Inlet, from 82.8 ha (1957) to 74.7 ha (1994). Seagrass cover actually increased substantially in the Central Mud Basin zone by 15.4 ha (41%). *P. australis* cover increased significantly due to the spread of established shoreline communities into deeper waters. By contrast, *Zostera* spp. and *Halophila* spp. decreased 16.9 ha (98%). The construction of two large groynes at the ocean entrance to Wagonga Inlet was probably responsible for these changes. The increased tidal range and tidal prism may have improved water
clarity, thereby lowering the depth limit of *P. australis*, but also decreased water depth on the intertidal flats, leading to the exposure of *Zostera* spp. and *Halophila* spp.

- In Bermagui River, seagrass cover declined by 33%, from 42.4 ha (1957) to 28.2 ha (1998). Monospecific stands of *P. australis* decreased by 15% (0.8 ha), mainly between 1979 and 1998. *Zostera* communities suffered the greatest loss of cover, with 62% (6.7 ha) of the beds disappearing, mainly between 1957 and 1979. The construction of Bermagui Harbour and the addition of two groynes to maintain the entrance channel appear to be responsible for the decline in *Zostera* cover in the MTD.

8.2.2 How fast are *Posidonia australis* communities able to recover from anthropogenic damage and what growth strategies does the species adopt?

Although damage to *P. australis* caused by catchment development was considerable in many estuaries in southern NSW, *P. australis* communities were able to recover at some sites. In Jervis Bay, seismic blast holes created in 1969 have been recolonised by *P. australis* rhizomes at an average rate of $21 \pm 2$ cm.y$^{-1}$, and by 1997 many of the holes were less than half their original size. In Port Hacking, new *P. australis* patches have developed in areas once mined for shell grit. In St Georges Basin, *P. australis* seedlings were found in situ at a number of sites that had lost seagrass cover over the last forty years. Small *P. australis* patches of varying age and size were also investigated for the first time. However, natural recovery of *P. australis*, whether it be from vegetative regrowth or sexual reproduction and patch formation, is extremely slow. In the medium term (ten to fifteen years), *P. australis* is unlikely to significantly recover and impacted areas may need to be rehabilitated using transplants.

**Major findings**

- In Jervis Bay, seismic blast ‘holes’ have been recolonised by *P. australis* rhizomes over the last thirty years. Between 1972 and 1997, all 11 blast holes decreased in area due to the spread of *P. australis* rhizomes from the surrounding bed. During
the 25 year period, total area occupied by the 11 holes has decreased by 56%. The spreading rate (or encroachment rate) of the *P. australis* community over the 25 year period averaged across all conditions and all 11 holes was $21 \pm 2$ cm yr$^{-1}$, though spreading rates varied between time periods and between holes. Assuming that the average rate of recolonisation of each of the holes remains unchanged, the 11 blast holes should become completely revegetated at various times in the next two centuries, the earliest in the year 2034 and the latest in 2071.

- In Port Hacking, shell grit mining destroyed about 40 ha of seagrass between 1951 and 1975. Since 1975, *Z. capricorni* has recolonised most of the area where water depth is suitable, and eleven discrete *P. australis* beds were found in the study area, nestled amongst dense and continuous *Z. capricorni* communities. These patches ranged in age from 8 yrs to 27 yrs. It could not be established whether these patches had resulted from previous seedling establishment, or whether they were vegetative re-growth from pre-existing beds too small to be seen on the 1975 photographs.

- An *in situ* study of short term natural recovery of damaged *P. australis* bed in St Georges Basin found evidence of vegetative propagation, successful recruitment of seedlings and subsequent development into small plants.

- Plagiotropic rhizomes on the edge of fragmented *P. australis* bed exhibited substantial growth over the 500 days of monitoring, and produced new orthotropic shoots, transitional shoots and Order-2 plagiotropic rhizomes. Mean rhizome elongation was $21.4 \pm 1.0$ cm yr$^{-1}$; and ranged from $7.0$ cm yr$^{-1}$ to $37.9$ cm yr$^{-1}$. The mean Rhizome Plastochrone Interval was $33 \pm 0.8$ days. Shoot elongation was substantially slower than rhizome elongation, with a mean of $4.4 \pm 0.3$ cm yr$^{-1}$, and a minimum and maximum of $1.3$ and $11.7$ cm yr$^{-1}$ respectively. Mean shoot Plastochrone Interval was $48 \pm 1.4$ days. Some aspects of *P. australis* growth exhibited a seasonal pattern. Production of new rhizomes, which involved the splitting of the primary Order-1 rhizome to produce a Order-2 rhizome, occurred mainly from October to December. Shoot production also occurred from October to December. The most common growth pattern for plagiotropic rhizomes was for
the primary Order-1 rhizome to split and produce 1 - 2 Order-2 rhizomes and 1 - 2 transitional shoots.

- *P. australis* seedlings were found at all four sites surveyed, with seedling abundance ranging from 0.001 seedlings.m⁻² to 0.014 seedlings.m⁻². Seedling age ranged from 0.7 years, which consisted of a seedling bearing a single shoot, to 8.2 years, which consisted of a small plant bearing four shoots and one rhizome. Estimated growth rate of seedling shoots ranged from $1.1 \pm 0.1$ to $1.9 \pm 0.3$ cm.yr⁻¹, considerably slower than that of mature orthotropic shoots. Both the number of shoots and leaf area increased as seedlings developed through time. Increase in leaf area and number of shoots was particularly evident between 2 and 5 yrs of age.

- Small *P. australis* plants, consisting of a number of orthotropic shoots and one plagiotropic rhizome, were found at three of the four sites. Each plant had produced one rhizome only, with growth rates ranging from 4.6 cm.yr⁻¹ to 18.3 cm.yr⁻¹. Despite the lack of larger samples, an estimated time-line of morphological development was constructed for *P. australis* for the first nine years of seedling and patch development. The mean production time of a second shoot occurred at about $2.7 \pm 0.3$ yrs, the third shoot was $3.4 \pm 0.3$ yrs, and the fourth and fifth shoot occurred at approximately similar times, $4.3 \pm 0.7$ and $4.5 \pm 0.9$ yrs respectively. At the time the fourth and fifth shoots were being produced, a plagiotropic rhizome also began to grow. On average this took $4.3 \pm 0.7$ yrs, but occurred as early as $3.0$ yrs and as late as $7.6$ yrs.

8.2.3 *Can the health of Posidonia australis in southern NSW estuaries be monitored in such a way that small scale changes are detected quickly and future losses can be prevented?*

In order to conserve remaining *P. australis* communities in southern NSW, the health of existing *P. australis* beds needs to be monitored, and trigger to points need to be established to alert management to possible anthropogenic impacts. Monitoring of *P. australis* communities in three southern NSW estuaries was able to detect small
changes in *P. australis* health. Measurements of *P. australis* shoot density were particularly effective at measuring seagrass health, as small natural variation meant that changes of 15 to 20% were able to be detected as statistically significant, and the absence of a seasonal trend meant that consistent declines appeared as an obvious downward trend in the time series graphs. At most sites, variability in shoot density fell within 20% of the overall mean for each site, while changes that were possibly the result of anthropogenic impacts fell above 30%. It is recommended that future monitoring of *P. australis* in southern NSW use a trigger point of 30% to initiate a study of the possible causes of the change.

**Major findings**

- *P. australis* leaf area did not prove to be a suitable measure to detect anthropogenic impact. Large variation in leaf area meant that only changes as large as 25 to 35% were detected. Most importantly, deterioration in shoot density did not show a corresponding change in leaf area.

- Temporal changes in *P. australis* shoot density were complex and erratic, though some aspects of the variability may have been related to exposure stress, rapid sediment movement and sediment accretion. Only two sites, both in Merimbula Lake, exhibited a seasonal pattern in *P. australis* shoot density.

- Temporal changes in *P. australis* leaf area exhibited a seasonal pattern in Wagonga Inlet and Merimbula Lake, with winter minima and summer maxima. In Port Hacking, *P. australis* leaf area peaked in later spring and winter, but was at its lowest in summer, possibly a result of desiccation or excessively high water temperature.

- Flowering of *P. australis* showed even great variability, with considerable differences between sites, estuary and year. *P. australis* seedlings were not observed during *in situ* counts of *P. australis* shoots, despite the high rate of flowering and fruiting.
• A change in mean *P. australis* shoot density of 30% over a number of sampling intervals is large enough to warrant further investigation, as it may represent an anthropogenic impact.

8.2.4 *Is it feasible to rehabilitate Posidonia australis communities by the use of transplants and thereby restore damaged areas?*

This study has demonstrated that it is feasible to restore damaged *P. australis* habitats using transplants. In Port Hacking, *P. australis* transplants increased the amount of *P. australis* habitat in the Port. However, when considering rehabilitation, candidate sites should be subject to a feasibility study prior to transplanting, in order to increase likelihood of survival. In Port Hacking, Burraneer Bay provides a good example of an area which has sustained significant loss of *P. australis* but now appears ready for rehabilitation. This location would be an ideal site for transplanting *P. australis* shoots at a larger scale, thereby compensating for the substantial loss of *P. australis* that has occurred in Port Hacking and some other estuaries over the last fifty years.

*Major findings*

• A total of 575 shoots were transplanted to five sites and, despite the total loss of seagrass at two sites, 650 shoots were present at the end of the study. This was due to good growth of *P. australis* shoots transplanted to Burraneer Bay (an increase of 253 shoots) and fair growth of shoots transplanted to the Control (an increase of 67 shoots). Transplants at four of the five sites exhibited high survival rates in the short term (less than 6 months) and transplants at three of the five sites exhibited high percentage survival in the long term (greater than 12 months).

• Mean rhizome elongation rate (± SE) of transplanted *P. australis* shoots was 15.2 ± 2.4 cm.yr⁻¹ at the control site (CS), 22.3 ± 1.4 cm.yr⁻¹ at BB, and 9.1 ± 1.0 cm.yr⁻¹ at RJP. Rhizome P.I. was 30.0 ± 1.4 days at the control site, 28.4 ± 1.3 days at BB, and 39.9 ± 3.3 days at RJP. Shoot elongation was 2.3 ± 0.2 cm.yr⁻¹ at the control
site (CS), 2.8 ± 0.4 cm.yr⁻¹ at BB, and 2.4 ± 0.4 cm.yr⁻¹ at RJP. Shoot P.I. was 38.0 ± 2.6 days at the control site, 41.5 ± 3.4 days at BB and 48.3 ± 3.4 days at RJP.

- Transplanted shoot exhibited significant branching and shoot production. Transplants at BB produced a total of 15 Order-2 rhizomes, compared to 7 at the control at 8 at RJP. A smaller number of Order-3 rhizomes were also produced, with transplants at the control and BB sites producing 4 and 3 respectively, and at RJP producing 1. Transplanted *P. australis* at the BB site produced the highest number of orthotropic shoots (23), compared to the transplants at the control site (16) and at RJP (7). Transplants at the control site and BB site both produced 6 transitional shoots, while those at the RJP site produced none.

8.3 Management implications and recommendations for future research

This study has collected a large volume of new information related to the conservation status of *P. australis*, and there are a number of management implications that might now be considered, as well future research studies.

(i) Increased protection of *P. australis* under habitat plans.

The Fisheries Habitat protection plan (NSWF 1997) protects *P. australis* from waterway developments that impact directly on existing beds. However, indirect impacts, such as channel dredging, are not as rigorously controlled, and there is no power within the plan to control catchment development that may cause a decline in the health of *P. australis*. The level of development in southern NSW estuaries is likely to increase substantially in years to come, and *P. australis* beds need to be protected from many of the indirect impacts associated with catchment development. These include: increased channel dredging to accommodate more boats and larger vessels, increased damage from boating activity, possible deterioration in water quality from urban runoff, and the impact of marine pests, such as *Caulerpa taxifolia*. 
Two further research studies that may be of use in this aspect are:

- an investigation into the effects of channel dredging on *P. australis* in Port Hacking (NSW), and,
- a study of the possible effects of *Caulerpa taxifolia* on *P. australis* communities in Port Hacking (NSW). At this stage the distribution of this noxious weed appears confined mainly to *Zostera capricorni* beds. However, it seems likely that it will expand to *P. australis* communities, considering that *P. oceanica* beds in the Mediterranean have been colonised by *Caulerpa taxifolia* (Meinesz 1999).

(ii) Establishment of marine reserves in southern NSW estuaries need to be considered.

There is an opportunity for NSW to create marine reserves in southern NSW on the basis of the data collected in this study. Two keys sites for protection are St Georges Basin and Port Hacking. The *P. australis* communities in the central mud basin of St Georges Basin may constitute the only case of successful sexual propagation in all southern NSW. Not only does this represent an extremely rare case of natural recovery, but, in the long term, it may benefit the health of *P. australis* in southern NSW by acting as a stock of genetic diversity. Therefore, St Georges Basin, particularly the central mud basin zone, is a key site for protection.

Some parts of Port Hacking could also be made into a marine reserve. For example, a substantial amount of healthy *P. australis* exists on the southern shore of Port Hacking, adjacent the Royal National Park. As development activities are already curtailed in this area, it could easily be made a key site for protection. However, other areas, such as the entrance to Gunnamatta Bay, have limited value. Storm damage and anthropogenic impacts have impacted heavily on *P. australis* in this area, and attempts to rehabilitate the site with transplants were unsuccessful. The considerable expanse of bare sand at this site means that the area is now one of the most popular recreational sites in Port Hacking, and it seems evident that its recreational value will outweigh its ecological value.
The estuaries Wagonga Inlet, Bermagui River, and Merimbula Lake may also contain areas of *P. australis* where marine reserves could be established. In Wagonga Inlet, this may be unnecessary, given that *P. australis* has increased. In Bermagui River and Merimbula Lake, much of the *P. australis* communities are covered by oyster leases, and may be in a poor state of health. Further studies are required to assess particular areas of *P. australis* communities that may warrant or require special protection.

(iii) Establishment of a monitoring program.

There are two possible approaches to this issue. The first is to use aerial photography to monitor seagrass health through estimates of area and distribution. NSWFW have taken this approach. At present, NSWFW are remapping the entire NSW coast in order to compare this data to that of West et al. (1985) and assess the large scale changes that have occurred over the last twenty years. The second approach is to establish an *in situ* monitoring program, similar to the one presented in this study, at a number of locations within the region.

It is likely that *in situ* monitoring would be more effective than remote sensing techniques to ensure the health of *P. australis* in southern NSW. At present, remote sensing techniques can only detect a decline in the health of *P. australis* community by detecting a decline in area. *In situ* monitoring is able to detect declines in health before the bed disappears completely, and is therefore more sensitive. However, it is unrealistic to think that regular measurements of *P. australis* shoot density will occur in all southern NSW estuaries. An alternative arrangement may be to use remote sensing to monitor *P. australis* communities (as well as other seagrass species) at regular intervals, perhaps every three to five years. If a decline in area is detected, then *in situ* monitoring should be initiated, perhaps using the concept of trigger levels proposed in Chapter Six.

One possible future study would be to establish monitoring stations at a number of estuaries in NSW to further refine the trigger levels that have been proposed.
(iv) Consideration of *P. australis* as a vulnerable species.

In order to conserve biodiversity and protect threatened aquatic species from human activities, the NSW Fisheries Management Act (1994) allows species to be nominated for protection and conservation, if they are endangered or vulnerable (NSWF 2001b). Nominations for threatened species are considered by the Fisheries Scientific Committee, but there are no specific guidelines available from NSWF to assess whether *P. australis* can be classified as threatened or vulnerable, based on the data collected in this study.

However, the International Union for the Conservation of Nature (IUCN) does list specific guidelines for defining threatened or endangered species. The IUCN Red List criteria for a vulnerable species is an ‘observed, estimated, inferred or suspected reduction of at least 20% over the last ten years’ (IUCN 1999), although recently it has been proposed that this level should be changed to 30%. Using these strict definitions, *P. australis* could not be classified as vulnerable. Firstly, the overall reduction in southern NSW estuaries has been less than 30% over the last decade. Secondly, most of the reduction in area of *P. australis* occurred between twenty and fifty years ago.

However, if *P. australis* populations in individual estuaries can be considered to be distinct populations, either due to geographical isolation, or different genetic components, then *P. australis* in a number of estuaries in NSW, including Botany Bay, Port Hacking and St Georges Basin, may be considered to be vulnerable.

To determine whether this is the case, further research needs to be undertaken:

- A study of the genetic characteristics of *P. australis* communities in St Georges Basin, NSW, compared to other communities were sexual propagation appears to be negligible.

- A survey of estuaries in central and northern NSW estuaries in order to find seedlings and small plants. Wallis Lake, in northern NSW, may be an ideal site for
such a survey, as *P. australis* in this estuary has a similar distribution to that in St Georges Basin.
REFERENCES CITED


Anon. 1994. Summary of large storms along the central coast of NSW. NSW Public Works Department, Coast and Flood Branch (1994).


Appendix 1: Aerial photographs used to map the distribution of seagrass species in Port Hacking, St Georges Basin, Wagonga Inlet and Bermagui River.

**Port Hacking**

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<tr>
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<tr>
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<td></td>
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<td>2/4/1975</td>
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<td>12/3/1985</td>
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<td>35E</td>
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<td>5/7/1999</td>
<td>4482 (M2182)</td>
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<td></td>
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<td>2</td>
<td>9, 10, 11</td>
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<tr>
<td></td>
<td></td>
<td>3</td>
<td>37, 38</td>
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St Georges Basin

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<td>26 150' ASL</td>
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<td>Jervis</td>
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<td>Jervis</td>
<td>5/1/97</td>
<td>NSW 4337</td>
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Wagonga Inlet

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<td>Narooma</td>
<td>22/5/57</td>
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<td>NSW Coastline</td>
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<td>6096m</td>
<td>Excellent</td>
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<td>NSW Coastal Wetlands</td>
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<td>Coastal Wetlands</td>
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<td>Narooma</td>
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## Bermagui River

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<td>Narooma</td>
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<td>NSW 307</td>
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Appendix 2: An assessment of the rectification error in aerial photography used to map the distribution of *P. australis* and other seagrass species in Port Hacking, St Georges Basin, Wagonga Inlet and Bermagui River.

RMS errors on individual images were less than 5m for Port Hacking, and less than 10m for St Georges Basin, Wagonga Inlet and Bermagui River images (Appendix x). However, in some cases more than one aerial photograph was needed to cover the extent of each estuary, so that images had be rectified separately and then joined to form a photographic mosaic. It was therefore necessary to calculate how RMS errors on individual images used in these mosaics interacted when joined. In order to calculate the rectification error between overlapping images, the co-ordinates of randomly selected overlapping points on overlapping photos were compared. These points were not the same GCP's used to rectify the images and, not surprisingly, rectification error on overlapping photos was greater than the RMS errors for individual images. In the case of Port Hacking, mean rectification error ranged from 7 to 18m, reaching a maximum of 29m, even though RMS errors for individual images were less than 5m. It is worth noting that polygons overlayed from different years (particularly in areas where ground control was poor) were often misaligned by up to 30m, even though these polygons represented stable structure which should not have changed position. Obviously this should not be possible if RMS errors were less than 5m, as the maximum misalignment expected would be 10m. This highlights a point made in Chapter Two: that RMS errors are only an estimate of error in the GCP's used, and may not accurately reflect error in parts of the image not covered by the GCP's. In the case of St Georges Basin, where adequate GCP's were the most difficult to find, misalignment was as great as 50m (an average error of 25m for each image). This further suggests that the RMS errors did not give an accurate indication of rectification error still present in the images. A more realistic assessment of rectification error in the images would be approximately 5 - 15m for Port Hacking, and 10 - 25 m for St Georges Basin, Wagonga Inlet and Bermagui River.
These rectification errors influence the accuracy of seagrass mapping. Although they do not appear to greatly influence the size of the polygons, they do influence where they are located. In photographic mosaics, where two or more photos are needed to map the same community, the same polygons may be assigned different coordinates. This means that the final maps may incorrectly display changes in location of beds. At the scale the maps are presented in this study, these errors are not noticeable. However, if the maps were enlarged and co-ordinates used to examine spatial change in individual polygons, then it would significantly influence the results of the analysis. For example, if one was trying to examine depth related regression of seagrass beds then a movement of 25m shoreline may indicate significant regression, but it may simply be rectification error. Ultimately, these maps are designed to show large scale changes in seagrass distribution and are not suitable for a finer scale analysis.
Appendix 4: Area occupied by *Posidonia australis* and other seagrass communities in selected sub-zones of Port Hacking from 1930 to 1999. *solar reflectance obscures part of image.*

<table>
<thead>
<tr>
<th>Zone</th>
<th>1930 (ha)</th>
<th>1942 (ha)</th>
<th>1951 (ha)</th>
<th>1961 (ha)</th>
<th>1975 (ha)</th>
<th>1977 (ha)</th>
<th>1985 (ha)</th>
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<tr>
<td>Zone 1 Entrance</td>
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<tr>
<td><em>P. australis</em></td>
<td>6.1</td>
<td>5.5</td>
<td>5.3</td>
<td>5.7</td>
<td>4.3</td>
<td>3.6</td>
<td>5.4</td>
<td>4.9</td>
</tr>
<tr>
<td><em>Zostera spp.</em></td>
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<td>9.4</td>
<td>6.1</td>
<td>2.7</td>
<td>0.1</td>
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