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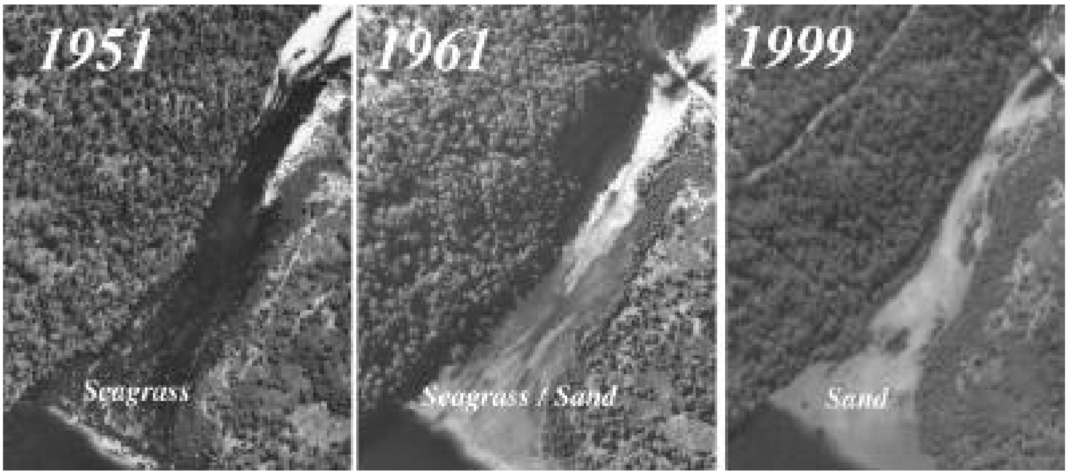
FRDC Project 97/220:

Seagrasses in southern NSW Estuaries - their ecology, conservation, restoration and management

A.J Meehan and R.J West



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FRDC Project 97/220:
Seagrasses in southern NSW Estuaries -
their ecology, conservation, restoration and management

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OBJECTIVES:

1. To investigate the factors affecting the growth and distribution of seagrasses in major estuaries in southern NSW.
2. To provide information on present and past distributions of seagrasses in selected estuaries.
3. To investigate the impacts of resource management decisions on existing seagrass meadows and review the criteria for selecting key seagrass sites for protection.
4. To carry out experimental trials examining the factors influencing successful restoration of damaged sites.

NON-TECHNICAL SUMMARY

A three year study of the seagrasses in southern New South Wales (NSW) estuaries has been carried out by means of a postgraduate scholarship. This research has provided information on the distribution of all species of seagrass in a selection of major estuaries in the region, but has concentrated primarily on *Posidonia australis* Hook. *f.* This is the dominant seagrass species 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. Throughout 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 and management of *P. australis*, as it is thought to be highly vulnerable to development, slow to recover from damage and difficult to transplant.

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 ± 2 $cm.y^{-1}$. 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 ± 1.0 $cm.yr^{-1}$. 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 as 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 that has sustained significant loss of *P. australis* but now appears ready for further restoration.

Overall, a number of significant findings have been made in relation to the long term distribution, status, growth and ecology of seagrasses in the NSW south coast region, particularly with respect to *Posidonia australis*. These data will significantly add to the information available on seagrasses in the region and assist in their better management by local and state government agencies.

OUTCOMES ACHIEVED

1) **The conservation status of the seagrass *Posidonia australis* in southern NSW estuaries has been established through a number of related studies.**

These studies included an assessment of historical changes in distribution through remote sensing and a study of the rate of recolonisation of impacted sites. *Posidonia australis* has suffered a significant loss in distribution in most estuaries and displays an extremely slow rate of recovery.

2) **The life history of *Posidonia australis* has been investigated through a study of seedlings and small plants in southern NSW.**

This study revealed that *Posidonia australis* seedlings are actively colonising bare sand in St Georges Basin leading to formation of new beds. This may be the only estuary in southern NSW where this is occurring. However, this process is slow, with plants requiring up to four years before lateral extension occurs.

3) **Methods for assessing the health of *Posidonia australis* through small scale monitoring programs have been determined.**

Characteristics of *Posidonia australis* meadows were assessed as potential indicators of health, and sampling methods established.

4) **Successful experimental trials to provide a small-scale method for transplanting *Posidonia australis* shoots have been carried out.**

These trials have provided some confidence that some impacted areas may be rehabilitated through transplantation of *Posidonia australis* shoots.

5) **Recommendations have been made which, if adopted, should assist in the management and conservation of the seagrass *Posidonia australis* within the southern NSW region.**

KEYWORDS:

Seagrasses, *Posidonia australis,* recovery, transplants, seagrass management, estuaries.

1. INTRODUCTION

1.1 Background

This report presents an overview and summary of a PhD that investigated the conservation status, ecology and restoration of the seagrass *Posidonia australis* Hook. *f.* in south eastern Australia. A number of studies were conducted using remote sensing techniques, *in situ* monitoring and field based experiments. Each study is presented as a separate chapter in this report. Each chapter consists of a brief introduction, a list of major findings, and a discussion of results. Further details of three of these studies are provided in Appendix 3 and 5, which consist of two published scientific papers, and in Appendix 4, which is a draft manuscript in submission. Further information regarding the remaining studies is available by consulting the PhD thesis:

Meehan, A.J. 2001. Conservation status of the seagrass *Posidonia australis* Hook. *f.* in south eastern Australia. Environmental Science, University of Wollongong. 230 pp.

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). Seagrasses have also been reported to contribute to detrital food chains by their high rates of primary production (King 1981), and to 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, 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).

P. australis meadows 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). This study examined *P. australis* communities found in a region of the south east Australian coastline that stretches from southern Sydney to Eden (Fig. 2.1). 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 Need

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.

In Australia, the extent of these seagrass losses (see Shepherd et al. 1989) 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 project was to assess the conservation status of *P. australis* communities in southern NSW estuaries by addressing the objectives listed below.

1.3 Objectives

Initially, this project had the following broad objectives:

a) To investigate the factors affecting the growth and distribution of seagrasses in major estuaries in southern NSW.

b) To provide information on present and past distributions of seagrasses in selected estuaries.

c) To investigate the impacts of resource management decisions on existing seagrass meadows and review the criteria for selecting key seagrass sites for protection.

d) To carry out experimental trials examining the factors influencing successful restoration of damaged sites.

Objectives (a) and (b) are met by the presentation of data on historical distribution of all seagrass species in Port Hacking, St Georges Basin, Bermagui River and Wagonga Inlet. This historical analysis has produced GIS based maps (available by consulting the PhD thesis) of present and past seagrass distribution in selected estuaries and identified several key processes and impacts that have caused changes in the growth and distribution of seagrass species over the last fifty years. Objective (c) is met with respect *to Posidonia australis* in Section 7, where several key sites are recommended for protection. Objective (d) is met with respect to *Posidonia australis* by the presentation of data on trial transplanting of *Posidonia australis* to a number of damaged sites in Port Hacking.

Due to the broad nature of these original objectives, it was necessary to clarify them and develop new ones to better assess the conservation status of *Posidonia australis* in south eastern Australia. In particular, it was considered necessary to investigate the capacity of *P. australis* to recover from damage, and the feasibility of monitoring *P. australis* communities in order to detect changes in health. Five objectives were formulated for the PhD, and these are listed below.

1) To provide information on present and past distributions of *P. australis* and other seagrass species in southern NSW estuaries, and relate long term changes in seagrass distribution to catchment and waterway development.

2) To examine recovery of *P. australis* in southern NSW estuaries, and measure patch establishment, seedling establishment and regrowth of *P. australis* beds.

3) To monitor the health of *P. australis* in southern NSW estuaries, identify any impacts of present resource management decisions, and assess the feasibility of a monitoring program to prevent future losses.

4) To carry out experimental trials examining the factors influencing successful restoration of damaged *P. australis* sites in southern NSW estuaries.

5) To assess the conservation status of *P. australis* in southern NSW estuaries, recommend key sites for protection, and suggest further research.

1.4 Transfer of information

Information from this project has been provided for use in a number of management plans and reports. These include estuary processes studies, estuary management plans, and state of the environment reporting.

2. STUDY AREA

Six species of seagrass grow in southern NSW: *Zostera capricorni*, *Zostera muelleri*, *Heterozostera tasmanica*, *Halophila ovalis*, *Halophila decipiens* 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).

Six estuaries in southern NSW contain *P. australis*: Port Hacking (34^{o} 05’S, 151^{o} 08’E), St Georges Basin (35^{o} 07’S, 150^{o} 40’E), Wagonga Inlet (35^{o} 07’S, 150^{o} 07’ E), Bermagui River (36^{o}26’S, 150^{o} 37’E), Merimbula Lake (36^{o} 54’ S, 150^{o} 54’E) and Pambula Lake (36^{o} 54’ S, 150^{o} 54’E) (Fig. 2.1). Studies were conducted in each of these estuaries, with the exception of Pambula Lake, which 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^{o} 45’E, 35^{o} 08’S; Fig 2.1), Bateman’s Bay and Twofold Bay. These open embayments were largely excluded from this investigation, as the purpose of this study was to investigate *P. australis* in estuarine situations, where they are more likely to be influenced by catchment development. The 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 Five).

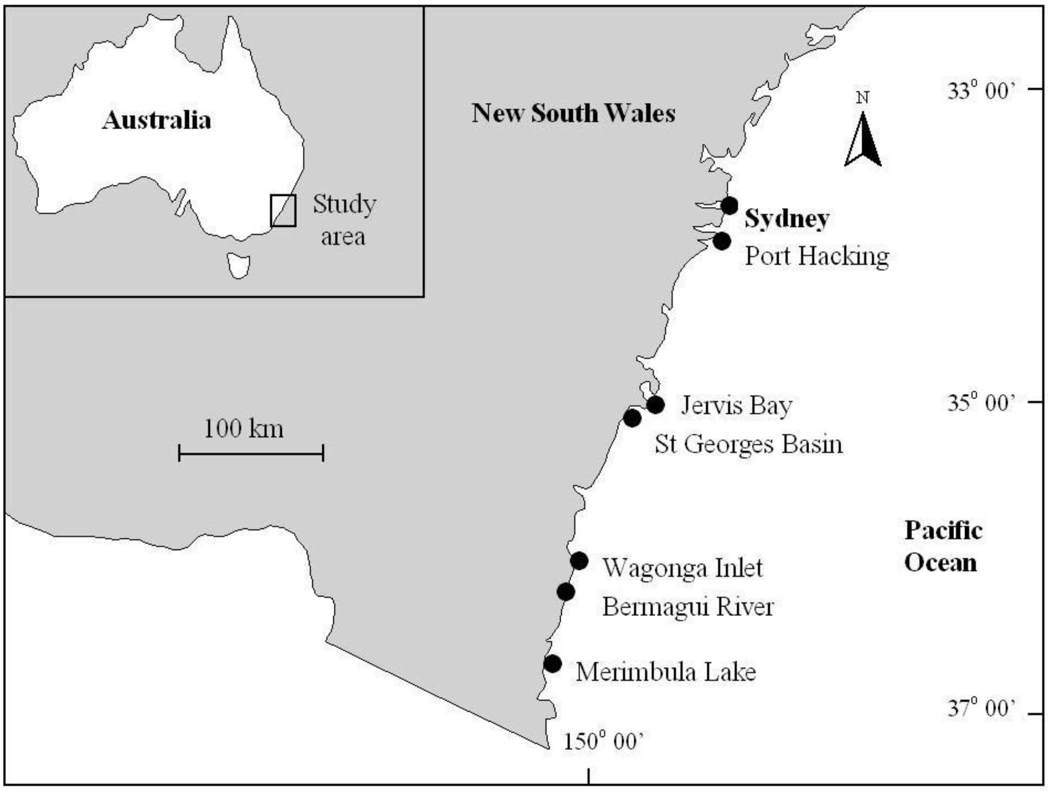

Figure 2.1. Location of estuaries in southern NSW studied in this thesis: Port Hacking, Jervis Bay, St Georges Basin, Wagonga Inlet, Bermagui River and Merimbula Lake.

3. METHODS

As noted, this project used remote sensing techniques, *in situ* monitoring and field based experiments to investigate seagrasses in southern NSW estuaries. This chapter provides an overview of the methods used to map historical seagrass distributions, and to monitor the health of *P. australis* in three estuaries in southern NSW. A detailed methodology for the other studies is provided in Appendix 3, 4 and 5, as part of each paper.

3.1 GIS mapping of present and past seagrass distribution

3.1.1 Digital acquisition and rectification of aerial photographs

Aerial photographs 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.

3.1.2 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). The delineation of these zones provided 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 on their environments is available.

Visual interpretation was used to map and discriminate between 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; Fig. 3.1). In cases where aerial photographs were compromised by solar reflectance, 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* from 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.

Distributions of seagrasses were mapped manually by one operator. Numerous polygons were traced using a mouse and computer screen at a scale of 1: 2000. At this scale individual pixels could 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 15m^2 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.

3.2 Measuring natural recovery of *Posidonia australis* at two sites in southern NSW.

3.2.1 Measuring recovery of Posidonia australis in Jervis Bay

To assess whether there has been any re-colonisation of the seismic blast holes in Jervis Bay, 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.

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.

Community Type	Observable Characteristics
Posidonia australis	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 *Zostera*. 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)
Zostera spp.	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 *P. australis*, or on deep edge of *P. australis* bed.
Halophila spp.	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 *Zostera*. Communities more often continuous than discrete beds.

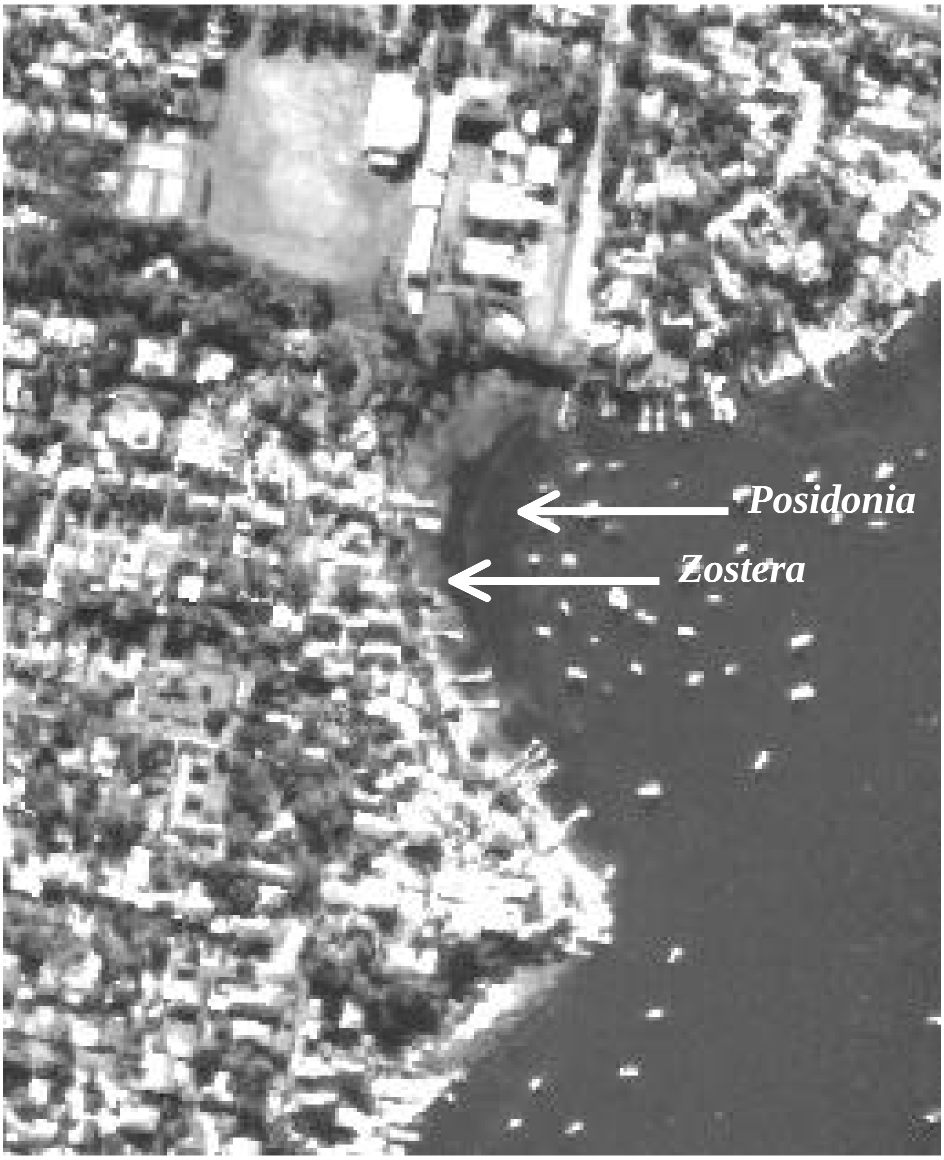

Figure 3.1. An aerial photograph of Gunnamatta Bay, Port Hacking, showing darker *Zostera* spp. in the interidal zone, and lighter *Posidonia australis* in the subtidal zone.

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 ($r = (A/\pi^{0.5})$; (3) spreading rates ($cm.yr^{-1}$) 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 each hole to have a radius of zero based on the mean spreading rate.

3.2.1 Measuring recovery of Posidonia australis in Port Hacking

Unlike Jervis Bay, it was not possible to map the recovery of *P. australis* in Port Hacking 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:

$$\text{Patch age (y)} = \text{Long axis (m)} \div 0.2 \text{ m}$$

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.

3.3 Assessing seedling development and patch formation of *Posidonia australis* in St Georges Basin, NSW.

3.3.1 Vegetative growth of Posidonia australis

At one site in St Georges Basin, 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.

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:

Rhizome elongation ($cm.yr^{-1}$) = 365 x new rhizome material (cm) ÷ time interval (days).
Plastochrone interval (days), horizontal shoots = time interval (days) ÷ 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:

P.I. (days), vertical shoot = (nos.. rhizome internodes ÷ nos. shoot internodes) x rhizome P.I.
Shoot elongation ($cm.yr^{-1}$) = (365 ÷ Shoot P.I. (days)) x 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.

3.3.2 Reproductive growth of Posidonia australis

Seedling abundance was assessed at the four randomly chosen sites. 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 orthotropic 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.

3.3.3 Regenerative growth of Posidonia australis

Initial site inspections at one of the sites 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 one site, 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.

3.4 Monitoring small scale changes in *Posidonia australis* in southern NSW estuaries as an indicator of health.

3.4.1 Pilot study

As there are no well established protocols for monitoring *P. australis* communities in the region 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).

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 0.0625m^2 quadrat (see Neverauskas 1987, Mellors et al. 1992, Wolterding 2000). The 0.5m x 0.5m quadrat (0.25m^2) 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 0.25m^2 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 $1m^2$ 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 buoy at each corner. Ten samples of a $1m^2$ quadrat, $0.25m^2$ quadrat and $0.0625m^2$ 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 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 $1m^2$ 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.

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. The C.V was found to be lowest for the 1.0m^2 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 1m^2 quadrat.

Results - 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 1m^2 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 1m^2 quadrats was required.

Results - 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.

3.4.2 Monitoring health of Posidonia australis

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 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 $1m^2$ 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.

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.

3.5 Experimental transplanting of *Posidonia australis* in Port hacking, NSW, to assess the feasibility of restoration.

3.5.1 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 Chapter Four) 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 1951was overlayed onto the GIS layer showing the distribution of *P. australis* in 1999, making it possible to identify areas where *P. australis* had declined.

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). 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 some existing *P. australis* beds. All sites were characterised by shallow water (<lm) at Mean Low Water (MLW)), sand substrate and the presence of remnant *P. australis* beds. *Z. capricorni* was present at RJP.

3.5.2 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.

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 400m^2, and then stored in seawater for a maximum of 2 hrs. 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. 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 and CS). 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.

3.5.3 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 lengths on both rhizomes and shoots were measured and the following formulas used to calculate growth (see Duarte et al. 1994):

Rhizome elongation ($cm.yr^{-1}$) = 365 x new rhizome material (cm) ÷ time interval (days).
Plastochrone interval (days), horizontal shoots = time interval (days) ÷ 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:

P.I. (days), vertical shoot = (nos.. rhizome internodes ÷ nos. shoot internodes) x rhizome P.I.
Shoot elongation ($cm.yr^{-1}$) = (365 ÷ Shoot P.I. (days)) x 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 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 ($\log(x + 1)$).

4. RESULTS AND DISCUSSION: STATUS OF *POSIDONIA AUSTRALIS* IN SOUTHERN NSW ESTUARIES

4.1 Introduction

This Chapter assesses the present status of *P. australis* in southern NSW estuaries and examines 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 (Meehan 1997) of Merimbula Lake and Pambula Lake in order to assess the present status of *P. australis* in the region.

4.2 Major findings

- In Port Hacking, seagrass cover declined by 54%, from 183.4 ha (1951) to 84.4 ha (1999) (Table 4.1). *P. australis* communities declined by 21%, from 41.2 ha (1951) to 32.7 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 23%, from 378 ha (1961) to 292 ha (1998) (Table 4.2). The CMB zone lost 27% (79.7 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 83.5 ha (1957) to 75.1 ha (1994) (Table 4.3). Seagrass cover actually increased substantially in the Central Mud Basin zone by 14.6 ha (38%). *P. australis* cover increased significantly due to the spread of established shoreline communities into deeper waters. By contrast, *Zostera* spp. and *Halophila* spp. decreased 14.9 ha (97%). 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 34% (14.8 ha), from 42.9 ha (1957) to 28.1 ha (1998) (Table 4.4). 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 48% (7.5 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.

Table 4.1. 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.

REGION	**YEAR**				
	1951 (ha)	**1961 (ha)**	**1975 (ha)**	**1985 (ha)**	**1999 (ha)**
Marine Tidal Delta zone					
P. australis:	34.7	40.2	28.7	29.4	25.9
Zostera spp:	64.4	43.5	14.3	39.2	31.9
Sparse *Zostera* spp:	11.1	20.4	1.0	0	3.8
Zostera spp. & *P. australis*:	5.6	5.4	6.0	0.9	2.6
P. australis, *Zostera* spp. & *Halophila*. Spp:	34.5	8.6	11.2	13.8	8.9
Subtotal:	150.3	118.1	61.1	82.2	73.1
Central Mud Basin zone					
P. australis:	6.5	9.3	5.2	7.5	6.8
Zostera spp:	2.1	2.0	0.9	1.3	0.9
Zostera spp. & *P. australis*:	6.1	0	0	0	0
P. australis, *Zostera* spp. & *Halophila* spp:	0.6	0	0	0	0
Subtotal:	15.3	11.3	6.1	8.8	7.7

Table 4.1 continued,

REGION	**YEAR**				
	1951 (ha)	**1961 (ha)**	**1975 (ha)**	**1985 (ha)**	**1999 (ha)**
Fluvial Delta zone					
Zostera spp:	17.9	11.1	7.8	8.8	3.7
Subtotal:	17.9	11.1	7.8	8.8	3.7
Combined areas					
P. australis:	41.2	49.5	32.9	35.8	32.7
Zostera spp:	83.9	56.6	23.0	49.2	36.4
Sparse Zostera spp.	11.5	20.4	1.0	0	3.8
Zostera spp. & *P. australis*:	11.7	5.4	6.0	0.9	2.6
P. australis, Zostera spp. & *Halophila* spp:	35.1	8.6	11.2	13.8	8.9
Total:	183.4	140.5	74.1	99.7	84.4

Table 4.2. 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.

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

Table 4.2 continued,

ZONE	**YEAR**			
	1961 (ha)	**1970 (ha)**	**1979 (ha)**	**1998 (ha)**
Fluvial Delta zone				
Zostera spp:	14.5	21.1	16.4	19.5
Zostera spp. & *Halophila*. Spp:	0	1.9	0	0
Subtotal:	14.5	23.0	16.4	19.5
Combined areas				
P. australis:	97.0	95.4	113.5	84.3
Zostera spp:	41.0	46.8	44.6	61.8
Zostera spp. & *Halophila*. Spp:	14.7	10	26.5	68.3
Zostera spp., *Halophila* spp. and *P. australis:*	188.7	132.9	117.3	28.3
Zostera spp., *Halophila* spp. and *Ruppia* spp.	33.4	55.7	47.3	29.4
Ruppia spp:	3.1	2.6	2.5	19.9
Total:	377.9	343.4	351.7	292.0

Table 4.3. 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.

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

Table 4.3 continued,

REGION	YEAR					
	1957 (ha)	**1972 (ha)**	**1977 (ha)**	**1982 (ha)**	**1986 (ha)**	**1999 (ha)**
Combined areas		n/a	n/a		n/a	
P. australis:	12.6			40.6		53.2
Zostera spp. & *Halophila* spp:	21.2			35.3		14.9
P. australis, Zostera spp. & *Halophila* spp:	49.7			8.9		7.0
Total:	83.5			84.8		75.1

Table 4.4. Area occupied by *Posidonia australis* and other seagrass communities in the Marine Tidal Delta zone, Central Mud Basin zone and Fluvial Delta zone of Bermagui River between 1957 and 1998, as interpreted from aerial photographs.

REGION	**YEAR**		
	1957 (ha)	**1979 (ha)**	**1998 (ha)**
Marine Tidal Delta zone			
Zostera spp:	11.6	4.1	5.6
P. australis & *Zostera* spp:	21.8	17.5	15.3
Total:	33.4	21.6	19.5
Central Mud Basin zone			
P. australis:	5.5	5.2	4.7
Zostera spp:	0	0	0
P. australis & *Zostera* spp:	0	0	0
Total:	5.5	5.2	4.7
Fluvial Delta zone			
Zostera spp:	4.0	2.0	2.5
Total:	4.0	2.0	2.5
Combined areas			
P. australis:	5.5	5.2	4.7
Zostera spp:	15.6	6.1	8.1
P. australis & *Zostera* spp:	21.8	17.5	15.3
Total:	42.9	28.8	28.1

4.3 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 4.5). Losses of *P. australis* was small in Bermagui River, Merimbula Lake and Pambula Lake, moderate in Port Hacking and large in St Georges Basin (Table 4.5). 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. 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.

A variety of anthropogenic and natural impacts could be correlated with the changes in seagrass area and distribution in southern NSW estuaries (Table 4.5). Anthropogenic impacts appeared to cause a much greater loss of both *P. australis* and other seagrass species, while natural impacts, such as storm damage and natural sand movement, were only observed to impact seagrass distribution in Port Hacking.

4.4 Discussion

Overall, the status of *P. australis* communities in southern NSW estuaries is poor, with significant areas 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.

Table 4.5. 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.

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

* data from Meehan (1997).

A variety of anthropogenic impacts are likely to 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; nutrient 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 distribution. 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. are thought to prefer such 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 nature of the entrance of Port Hacking means that some seagrass communities in this area are very susceptible to storm damage. Also, the progradation of the marine tidal delta in Port Hacking appears to have caused a decline in *P. australis* distribution. Overall, the effects of natural impacts on the area of *P. australis* were much less than that of anthropogenic impacts. However, any strategy designed to conserve and manage *P. australis* needs to recognise that significant changes to *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, as storm waves and natural sand movements are likely to quickly destroy any transplanted seagrass. This point is further investigated in Chapter Eight.

Aside from documenting the poor status of *P. australis* in southern NSW estuaries, an important outcome of this study is in regard to the complexity of long term change in *P. australis* and other seagrass species. Many studies throughout the world have documented seagrass decline due to human population growth and accompanying development activities (Orth and Moore 1983, Larkum and West 1990, Pergent-Martini and Pergent 1996, Cambridge et al. 1986, Short et al. 1996). Along with observational and experimental evidence (Abal and Dennison 1996, Dennison and Kirkman 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 *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 *P. australis* and other seagrass species are more complex, and the reaction of *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 *P. australis* beds, particularly on the southern shore, appear to have remained in excellent condition. *P. australis* beds in the entrance channel of the basin also remained in reasonable health. This was also the case for Merimbula Lake, where deeper *P. australis* showed no overall change in area, even as shallow *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 water quality, in particular its relationship to Photosynthetically Available Radiation (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.

5. RESULTS AND DISCUSSION: NATURAL RATES OF RECOVERY FOR *POSIDONIA AUSTRALIS* AT TWO SITES IN SOUTHERN NSW

5.1 Introduction

In Chapter Four, a consistent decline in *P. australis* communities in southern NSW estuaries as a result of anthropogenic and natural impacts was demonstrated. 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 (for a full report, see Appendix 3). 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.

5.2 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. Over 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 in Jervis Bay over the 25 year period averaged across all conditions and all 11 holes was 21 ± 2 cmy^{-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 century, 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.

5.3 Discussion

This study of Jervis Bay 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 ± 2 $cm.yr^{-1}$. 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 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.

This short study of Port Hacking 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 such 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.

6. RESULTS AND DISCUSSION: SEEDLING DEVELOPMENT AND PATCH FORMATION OF *POSIDONIA AUSTRALIS* IN ST GEORGES BASIN, NSW.

6.1 Introduction

In this chapter, the recovery of *P. australis* communities in St Georges Basin has been examined, 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 Five, 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 1996) 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 Five, 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 was 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. For a full report on this study, see Appendix 4.

6.2 Major findings

- 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 ± 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 ± 0.8 days. 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. Mean shoot Plastochrone Interval was 48 ± 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. 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. 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 ± 0.3 yrs, the third shoot was 3.4 ± 0.3 yrs, and the fourth and fifth shoot occurred at approximately similar times, 4.3 ± 0.7 and 4.5 ± 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 ± 0.7 yrs, but occurred as early as 3.0 yrs and as late as 7.6 yrs.

6.3 Discussion

Over the past twenty-five years, a significant body of observational evidence has accumulated concerning the poor reproductive success and slow rhizome growth of *P. australis* in southeast 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. For example, remote sensing of *P. australis* in Jervis Bay, suggests that recovery of *P. australis* communities is possible over long time periods, but there was no conclusive evidence of seedling recruitment and patch formation in this area.

As such, the *P. australis* communities of St Georges Basin are extremely interesting in that 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 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 possibly regenerative growth (from old matt) all playing a role.

The average rate of *P. australis* rhizome elongation in St Georges Basin was 21 $cm.yr^{-1}$, which is similar to the rate of rhizome incursion estimated for seismic blast holes in Jervis Bay. 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.

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 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 artificial transplantation of *P. australis*.

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.

7. RESULTS AND DISCUSSION: MONITORING SMALL SCALE CHANGES IN *POSIDONIA AUSTRALIS* COMMUNITIES IN SOUTHERN NSW ESTUARIES AS AN INDICATOR OF HEALTH

7.1 Introduction

In Chapter Four, 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 investigated the utility of a monitoring program to detect small-scale changes in *P. australis* health.

The objectives of this Chapter were: 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.

7.2 Major findings

- Temporal changes in *P. australis* shoot density were complex and erratic, though some aspects of the variability may have been related to exposure, 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 greater 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.

- *P. australis* leaf area did not prove to be a suitable measure to detect anthropogenic impact. Large variation in leaf area meant that only change as large as 25 to 35%

were detected. Most importantly, deterioration in shoot density did not show a corresponding change in leaf area.

- A change in mean shoot density *P. australis* of 30% over a number of sampling intervals is considered large enough to warrant further investigation, as it may represent an anthropogenic impact (see Figure 7.1).

7.3 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 1m^2 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 1m^2 quadrats is recommended for future monitoring programs of *P. australis* in southern NSW estuaries.

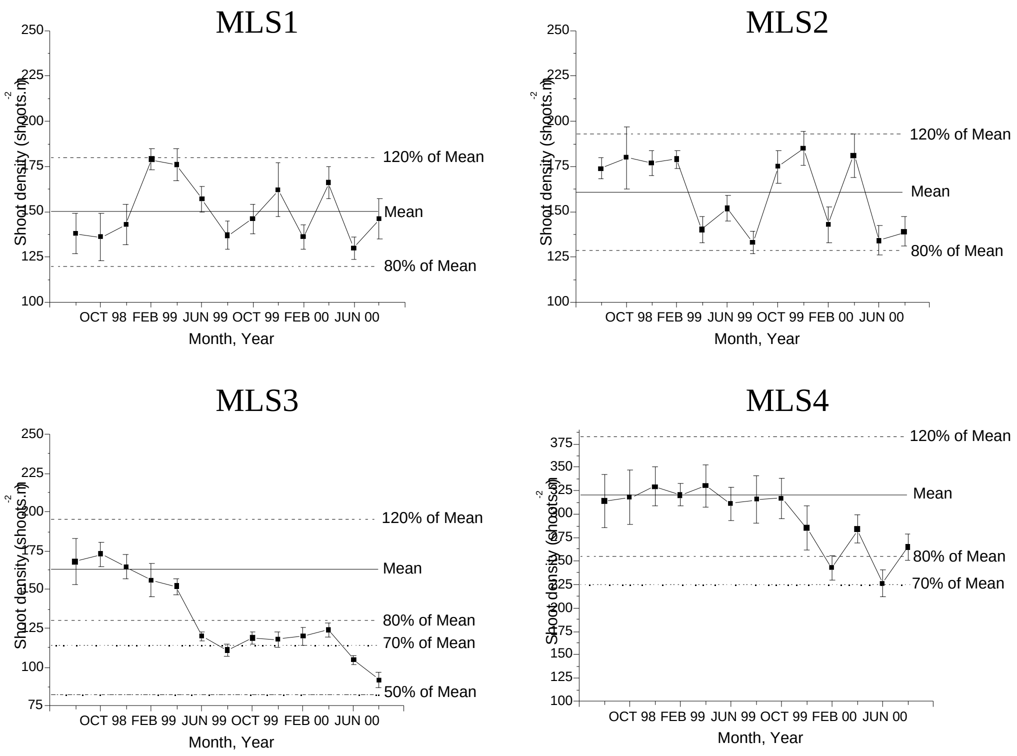

Figure 7.1. An assessment of variation over time in *Posidonia australis* shoot density at Merimbula Lake Site 1 (MLS1), Merimbula Lake Site 2 (MLS2), Merimbula Lake Site 3 (MLS3) and Merimbula Lake Site 4 (MLS4).

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. Both these sites were subject to anthropogenic impacts.

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 calculated 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 change as large as 25 to 35% could 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.

8. RESULTS AND DISCUSSION: EXPERIMENTAL TRANSPLANTING OF *POSIDONIA AUSTRALIS* IN PORT HACKING, NSW, TO ASSESS THE FEASIBILITY OF RESTORATION

8.1 Introduction

There has been a large decline in *P. australis* communities in southern NSW estuaries over the last fifty years. 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. 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 Six, 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.

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. Sites were chosen to assess the affect 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. For full details of this study, see Appendix 5.

8.2 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 (BB) (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 Red Jacks Point (RJP). Rhizome Plastochrone Interval (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 shoots 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 Discussion

Over the last fifty years, Port Hacking, like many other estuaries in southeast Australia, has lost considerable areas of *P. australis* due to anthropogenic impacts (see Chapter Four). 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 Five and Six), 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, see also Chapter Six). 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. For example, 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. The death of the *P. australis* transplants at this site was probably the result of these same factors. 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.

By contrast, 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.

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.

9. CONCLUSIONS

9.1 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.

New South Wales Fisheries (NSWF) habitat protection plans (NSWF 1997) protect *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 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 is possible 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 needs to be considered.

There is an opportunity for NSWF 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 has outweighed its ecological value.

The estuaries Wagonga Inlet, Bermagui River, and Merimbula Lake 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 the issue of establishing a monitoring program for seagrasses in southern NSW. The first is to use aerial photography to monitor seagrass health through estimates of area and distribution. NSWF have taken this approach. At present, NSWF 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 in revealing small changes in the health of *P. australis* in southern NSW. At present, remote sensing techniques can only detect a decline in the health of the *P. australis* communities by detecting quite large changes 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 could be considered as distinct populations, either due to geographical isolation, or different genetic characteristics, 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.

9.2 Benefits

As suggested in the original research proposal, this project has contributed greatly to information that will assist in the improved management of seagrass meadows in NSW, including developing techniques suitable for long and short term monitoring of seagrasses, and the small-scale restoration of areas of *Posidonia australis*. These meadows are critical habitats for a range of economically important fish species.

Results from the research project have already been used in some local management of the estuaries in the south eastern region of NSW, including by agencies such as Local Councils, Regional Catchment Management Boards, NSW Fisheries and the NSW Department of Land and Water Conservation. It is also expected that this project will greatly assist the Endangered Species Committee in determining whether *P. australis* should be listed as a vulnerable species.

9.3 Further development

Future developments in respect to this project will involve the preparation of a number of scientific journal papers and the general interest articles. The results of these studies will be disseminated by the authors to local and state government agencies. Presentations summarising the results and implications have been made at a number of workshops and conferences and will also be made at the NSW Coastal Conference in 2002. In addition, the data will be made available through the Internet.

9.4 Conclusion

The goal of this project was to assess the ecology, conservation, restoration and management of seagrasses in southern NSW estuaries, many of which had not been studied before. This project focused particularly on *Posidonia australis,* as it is one of the major seagrasses in a number of estuaries in southern NSW and is highly vulnerable to human disturbance. The specific objectives of this project were:

1. To investigate the factors affecting the growth and distribution of seagrasses in major estuaries in southern NSW.

This objective has been met fully. Analysis of historical seagrass distribution in several estuaries identified a number of factors causing changes to seagrass distribution. Small scale studies of growth (in terms of leaves, shoots and rhizomes) and large scale studies of re-growth (in terms of spreading rates over the last 50 to 100 years) also addressed this objective.

2. To provide information on present and past distributions of seagrasses in selected estuaries.

This objective has been met fully through historical mapping of seagrasses in a number of estuaries in the region, including Port Hacking, St Georges Basin, Wagonga Inlet and Bermagui River.

3. To investigate the impacts of resource management decisions on existing seagrasses meadows and review the criteria for selecting key seagrass sites for protection.

This objective has been met through a detailed description and investigation of the impact of resource management decisions on *Posidonia australis* in selected estuaries in southern NSW. In addition, there is some information pertaining to the criteria for selection of key seagrass sites for protection and restoration.

4. To carry out experimental trials examining the factors influencing successful restoration of damaged seagrass sites.

This objective has been met fully through a study of experimental transplantation of *Posidonia australis* shoots at a number of sites in Port Hacking.

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11. APPENDIX 1: INTELLECTUAL PROPERTY

There is no intellectual property associated with this research project.

12. APPENDIX 2: STAFF

List of staff engaged in this project in either part time or full time capacity:

Alex Meehan

13. APPENDIX 3: RESEARCH PAPER 1

Meehan, A.J., and West, R.J. 2001. Recovery times for a damaged *Posidonia australis* bed in southeastern Australia. *Aquatic Botany* 67, 161-167.

14. APPENDIX 4: DRAFT MANUSCRIPT

Meehan, A.J., and West, R.J. XXXX. Seedling development and patch formation of *Posidonia australis* Hook *f.* in a southeast Australian estuary. *Aquatic Botany* (submitted).

Abstract

The rarity of *Posidonia australis* seedlings and small plants has meant it has not been possible to examine sexual propagation of *P. australis* communities in southeast Australia. Vegetative growth has also not been studied in detail. However, for the first time in southeast Australia, *P. australis* seedlings and small plants were found *in situ* in a small estuary in the region. Samples of seedlings were collected from four sites within the estuary, while mature shoots on existing beds were tagged to measure vegetative growth and provide a Plastochrone Interval (PI) with which to age the seedlings. The mean rhizome elongation of mature plants was 21.4 ± 1.0 $cm.yr^{-1}$ and PI was 33 ± 0.8 days, while shoot elongation was 4.4 ± 0.3 $cm.yr^{-1}$ and PI was 48 ± 1.4 days. The abundance of *P. australis* seedling ranged from 0.001 $seedlings.m^{-2}$ to 0.014 $seedlings.m^{-2}$. The ages of the samples ranged from 0.7 years (a seedling bearing a single orthotropic shoot) to 8.2 years (a small plant bearing four orthotropic shoots and one plagiotropic rhizome). Both numbers of shoots and leaf areas were related to seedling age, with increases in leaf areas and number of shoots particularly evident between 2 and 5 years. Small *P. australis* plants 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 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. On average, production of a second shoot occurred at about 2.7 ± 0.3 years, the third shoot at 3.4 ± 0.3 years, and the production of the fourth and fifth shoot at 4.3 ± 0.7 and 4.5 ± 0.9 years respectively. A plagiotropic rhizome did not begin to grow until 4.3 ± 0.7 years, but one sample occurred as early as 3.0 years and on another, as late as 7.6 years. This data shows that while the capacity of *P. australis* to recolonise denuded areas is greater than previously thought, it is still extremely slow, and remaining areas need to be a conservation priority.

Keywords: Seagrass, *Posidonia australis*, seedling, patch formation, recovery.

1. Introduction

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 the case of *Posidonia australis* Hook. *f*, recovery from anthropogenic impacts has generally been slow or negligible in south east Australia (see Meehan and West 2001). This is partly because the rate of vegetative propagation of *P. australis* is amongst the slowest recorded (Larkum 1976, Kuo and Kirkman 1996, Marba and Walker 1999). For example, West (1990) estimated an average rhizome growth of 2.5 $cm.yr^{-1}$ for *P. australis* at sites along the northern shoreline of Jervis Bay, New South Wales (NSW), and a maximum rate of rhizome growth of about 29 $cm.yr^{-1}$ for horizontal rhizomes. In Port Hacking (NSW), Meehan and West (2002) tagged the rhizomes of *P. australis* transplants and measured growth rates ranging from 9.1 ± 1.0 $cm.yr^{-1}$ to 22.3 ± 1.4 $cm.yr^{-1}$.

However, the major reason for the slow recovery has been the failure of remaining *P. australis* populations in the region to produce viable seedlings that can recolonise damaged areas and expand to form small patches. The rarity of *P. australis* seedlings, and the near total absence of small plants of the species (see Kuo and Kirkman 1999) has meant that it is has been impossible to study important aspects of the species' life history, such as seedling abundance, rate of patch formation, and expansion rate. Information on these aspects of sexual propagation is important to assess the vulnerability of the species, and may also enable an estimation of the time scale involved in the natural recovery of *P. australis* ecosystems.

In this study, sexual and vegetative propagation of *P. australis* has been investigated in St Georges Basin, NSW. This appears to be 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 by estimating the age of plants using the Plastochrone Interval (P.I) (Patriquin 1973, Duarte et al. 1994). Aspects of plant development have been identified, including rhizome growth rate, shoot production rate and time taken to produce a plagiotropic rhizome. The ability of *P. australis* to naturally recover in the short term through various growth strategies has also been assessed.

2. Methods

2.1 Study area

St Georges Basin (35 ^{o} 07'S, 150 ^{o} 40'E) consists of a deep lake basin (average depth 9m) that is linked to the Pacific Ocean by a narrow inlet channel. The inlet channel is permanently open to the sea and mean tidal range is 0.78m throughout the channel (Shoalhaven City Council 1998). However, 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 but in the lake basin are restricted to cut-off lagoons and embayments. 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 eastern shore (SGBD) (Fig. 1). These sites were all located in areas that had supported extensive *P. australis* communities in the past (Meehan 2001).

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. The following growth data was 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:

Rhizome elongation ($cm.yr^{-1}$) = 365 x new rhizome material (cm) ÷ time interval (days).
Plastochrone interval (days), horizontal shoots = time interval (days) ÷ 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:

P.I. (days), vertical shoot = (nos.. rhizome internodes ÷ nos. shoot internodes) x rhizome P.I.
Shoot elongation ($cm.yr^{-1}$) = (365 ÷ Shoot P.I. (days)) x 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.

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 snorkeled 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 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 rates 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 whether shoots or rhizomes being were being examined. Shoot growth was calculated using a mean internode length for each shoot and the average number of internodes produced by mature orthotropic 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.

2.4 Regenerative growth of *Posidonia* *australis*

Initial site inspections at 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 a 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.

3. Results

3.1 Vegetative growth of Posidonia australis

Over the period of study, the majority of 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 ± 1.0 $\mathrm{cm.yr^{-1}}$; and ranged from 7.0 $\mathrm{cm.yr^{-1}}$ to 37.9 $\mathrm{cm.yr^{-1}}$ (Table 1). Rhizome internode length ranged from 3mm to 44mm, with a mean of 20.2 ± 0.9 mm (from a random sample of 100 internodes). Approximately 11.4 ± 0.3 internodes. $\mathrm{yr^{-1}}$ were produced, which equated to a mean Rhizome Plastochrone Interval of 33 ± 0.8 days (Table 1).

Shoot elongation was substantially slower than rhizome elongation, with a mean of 4.4 ± 0.3 $\mathrm{cm.yr^{-1}}$, and a minimum and maximum of 1.3 and 11.7 $\mathrm{cm.yr^{-1}}$ respectively (Table 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 $\mathrm{internodes.yr^{-1}}$, which equated to a mean shoot Plastochrone Interval of 48.0 ± 1.4 days (Table 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. 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. 2). 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 1).

Flowering and fruiting were observed at site SGBC in November, 1999. At this time, 4% of the total number of shoots had produced fruit.

3.2 Reproductive growth of Posidonia australis

P. australis seedlings were found at all four sites surveyed (Table 2). Seedling abundance ranged from 0.001 seedlings.m^{-2} to 0.014 seedlings.m^{-2}. 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 1 and 2).

Both numbers of shoots and leaf areas were related to seedling age, with both increasing as seedlings developed through time (Fig. 3). 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. 3). There was also clear evidence of an increase in the overall size of *P. australis* plants between 1 yr and 9 yrs (Fig. 4).

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 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 were 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 appeared to be derived from seedlings.

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 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.

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 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.

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, regrowth of *P. australis* by regeneration of the rhizome matt has important implications for recovery rates of damaged *P. australis* stands, as it allows isolated patches to regrow without sexual reproduction. Such isolated *P. australis* stands occur throughout St Georges Basin.

Discussion

Over the past twenty five years, a significant body of observational evidence has accumulated concerning the poor reproductive success and slow rhizome growth of *P. australis* in southeast 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. For example, remote sensing of *P. australis* in Jervis Bay (Meehan and West 2001) suggests that recovery of *P. australis* communities is possible over long time periods, but there was no conclusive evidence of seedling recruitment and patch formation in this area.
As such, the *P. australis* communities of St Georges Basin are extremely interesting in that 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 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 possibly 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. Actively growing bed margins are not common in *P. australis* beds, although have been observed by West (1980) and Wolterding (2000) in Botany Bay. 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), which can vary greatly between sites. 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 the proportion of plagiotropic shoots (actively spreading) ranged from 3% to 45% in a number of sites in south east Australian estuaries. Where bed margins do not have a high proportion of plagiotropic rhizomes, they appear to be very stable. This is evident on bed margins where growth is limited by environmental factors, such as exposure or lack of light, but is also evident in cleared areas within *P. australis* communities, 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 the environmental conditions for regrowth are optimal.

The average rate of *P. australis* rhizome elongation in St Georges Basin was 21 $cm.yr^{-1}$ which is similar to the rate of rhizome incursion estimated for seismic blast holes in Jervis Bay (Meehan and West 2001). 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 a 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.

In the meantime, *P. australis* recovery in St Georges Basin should be increased through seedling colonisation of bare sands. 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. This may be related to the installation of a sewerage reticulation system, which has eliminated septic system discharge and improved water quality (SCC 1998). 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, Meehan 2001) 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) noted this same distribution pattern in the central coast estuary of Wallis Lake. It seems that coastal saline lakes may be the best sites to *find P. australis* seedlings as the large areas of bare sand offer substrate. Other coastal saline lakes in NSW should also be investigated for sexual propagation.

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, and a study investigating this would be invaluable in further assessing the vulnerability of this species.

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List of Figures

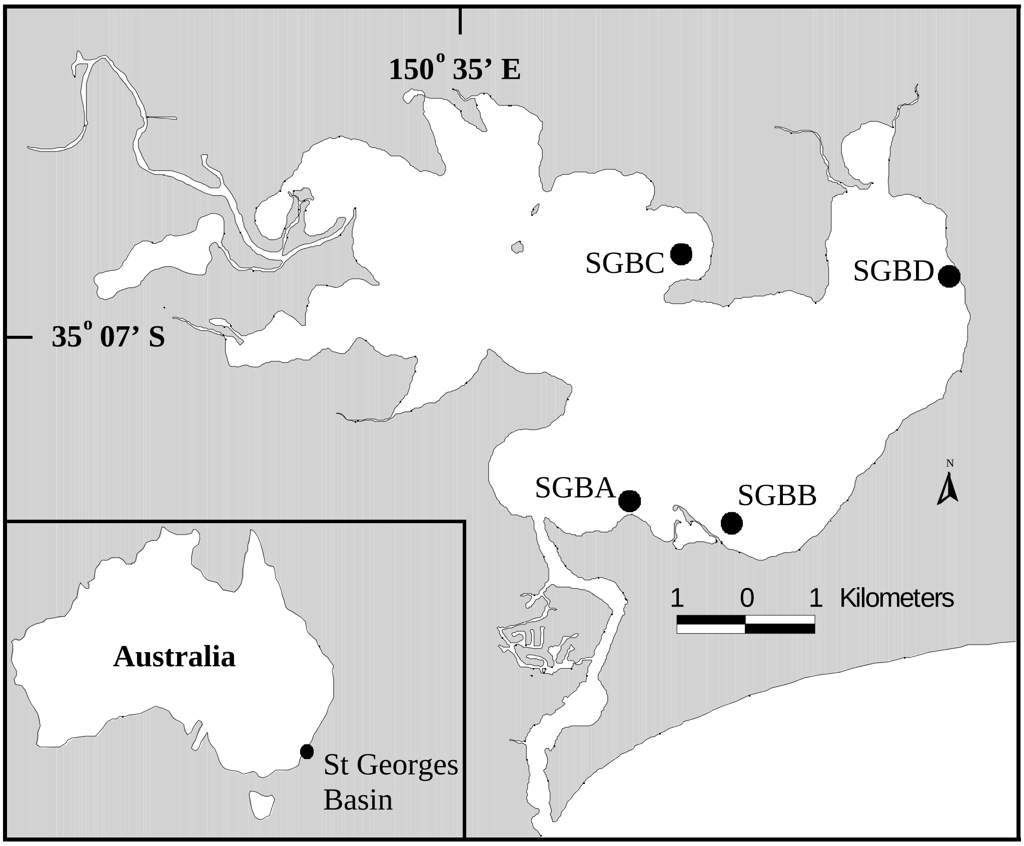

Fig. 1. Location of sites in St Georges Basin, NSW, Australia, surveyed to investigate the occurrence of *Posidonia australis* seedlings and small plants.

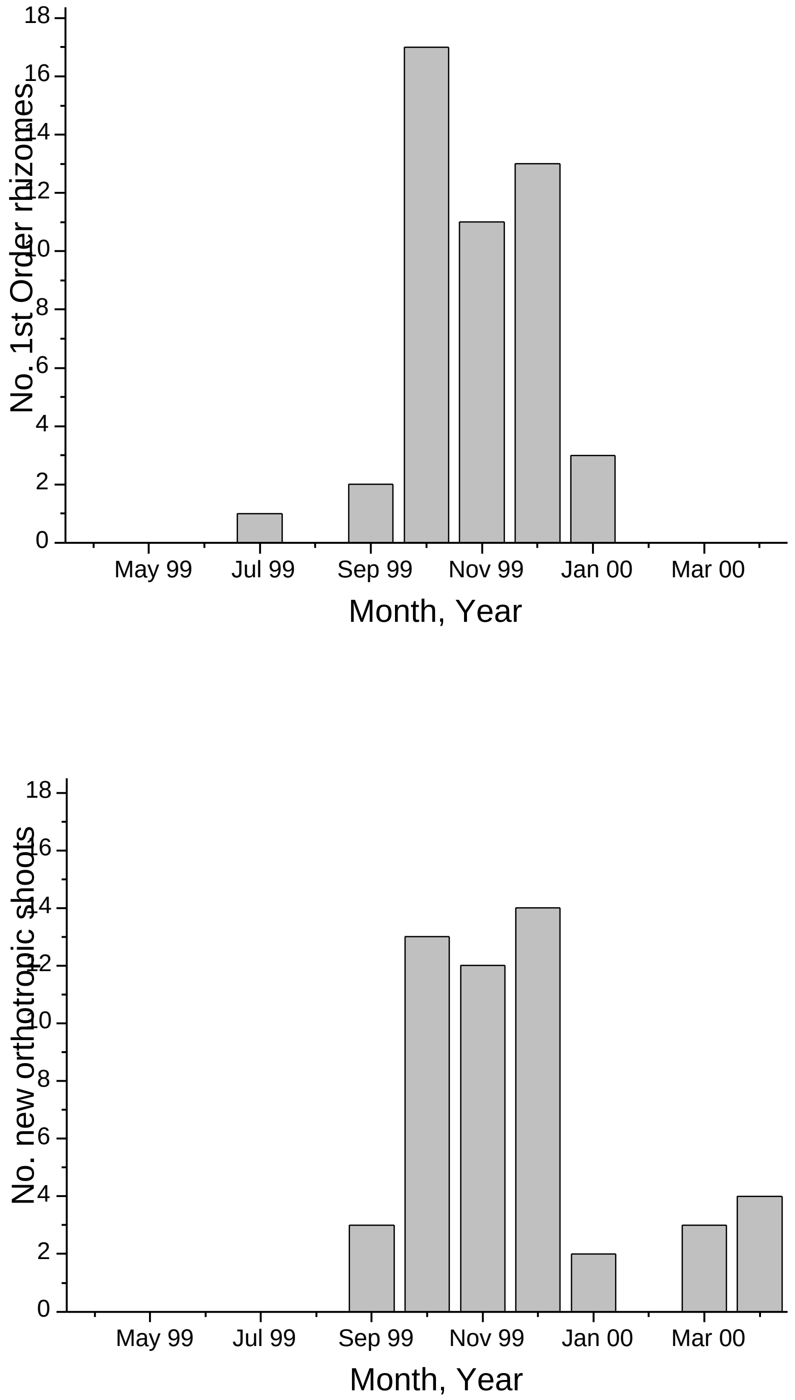

Fig. 2. (A) Time of year that Order-2 rhizomes were produced at site SGBC in St Georges Basin, NSW (n=46), (B) Time of year that new orthotropic shoots were produced at site SGBC in St Georges Basin, NSW (n=53).

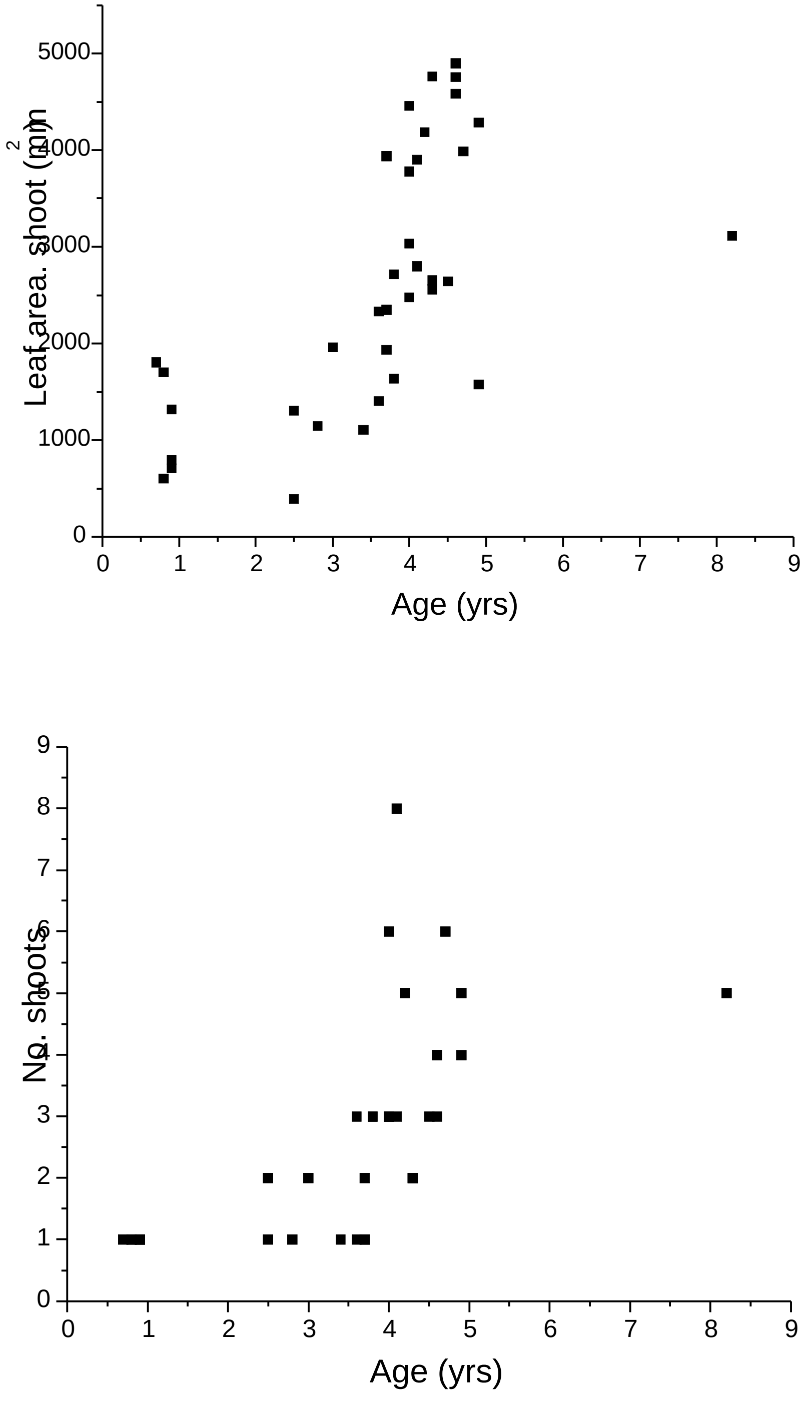

Fig. 3. (A) Scatter plot of mean leaf area per shoot and seedling age for *Posidonia australis* seedlings and small plants in St Georges Basin, NSW, (B) Scatter plot of number of shoots and seedling age for *Posidonia australis* seedlings and small plants in St Georges Basin, NSW.

Fig. 4. *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. 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 1. Vegetative growth rates of *Posidonia australis* at SGBC in Saint Georges Basin, southern NSW. Fifty plagiotropic rhizomes were tagged for approximately 500 days.

Variable	**Minimum**	**Mean (± S.E)**	**Maximum**
Rhizome elongation (cm.yr^{-1})	7.0	21.4 ± 1.0	37.9
Rhizome Internodes (yr^{-1})	6.3	11.4 ± 0.3	16.3
Rhizome P.I. (d)	22.4	33.0 ± 0.8	57.9
Rhizome internode length (mm), n = 100	3	20.2 ± 0.9	44
Shoot elongation (cm.yr^{-1})	1.3	4.4 ± 0.3	11.7
Shoot Internodes (yr^{-1})	12.1	7.9 ± 0.3	5.6
Shoot P.I. (d)	30.1	48 ± 1.4	64.9
Shoot internode length (mm), n = 100	0.5	4.3 ± 0.2	18
Shoot production. Rhizome.yr^{-1}	0	0.9 ± 0.1	2.55

Table 2. Summary of *Posidonia australis* reproductive growth variables at four sites in St Georges Basin, NSW (SGBA, SGBB, SGBC, SGBD).

Variable	SGBA	SGBB	SGBC	SGBD
Total no. of seedlings	26	2	12	17
No. regenerative patches	5	3	6	9
Seedling abundance (seedlings.m^{-2})	0.014	0.001	0.007	0.009
Seedling age range (yrs)	3.0 - 4.9	0.7 - 4.3	0.8 - 4.6	0.9 - 8.2
Mean no. shoots.seedling^{-1}	3.7	1.5	2.1	2.4
No. seedlings with plagiotropic rhizome	2	0	2	2
Shoot elongation ± S.E. (cm.yr^{-1})	1.9 ± 0.3	n/a	1.4 ± 0.2	1.1 ± 0.1
Rhizome elongation (cm.yr^{-1})	5.5 4.6	n/a	7.9 9.8	18.3 15.8

Table 3. Minimum, mean and maximum times (years) taken for *Posidonia australis* seedlings to produce 2nd to 5th shoot, and plagiotropic rhizome. The P.I. used was 48 days, that of mature *Posidonia australis* plants.

	Minimum	**Mean (± S.E)**	**Maximum**
Production of second shoot (yrs) n = 22	0.7	2.7 ± 0.3	6.7
Production of third shoot (yrs) n = 18	1.0	3.4 ± 0.3	7.6
Production of fourth shoot (yrs) n = 9	2.2	4.3 ± 0.7	7.9
Production of fifth shoot (yrs) n = 7	3	4.5 ± 0.6	7.6
Production of first rhizome (yrs) n = 6	3.0	4.3 ± 0.7	7.6

15. APPENDIX 5: RESEARCH PAPER 2

Meehan, A.J., and West, R.J. 2002. Experimental transplanting of *Posidonia australis* in Port Hacking, Australia, to assess the feasibility of restoration. *Marine Pollution Bulletin* 44, 25-31.