# Seagrass Rehabilitation in Metropolitan Adelaide

## I. Transplantation from Donor Beds

**Prepared for**

**Coast al Protection Board**

**Department for Environment and Heritage**

**by**

**Stephanie Seddon, David Miller, Sonja Venema and Jason Tanner**

**South Australian Research and Development Institute,**

**Aquatic Sciences**

**May 2004**

**SARDI Aquatic Sciences Publication No. RD04/0038**

Department
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**South Australian Research and Development Institute**

SARDI Aquatic Sciences
2 Hamra Avenue
West Beach SA 5024

Telephone: 8200 2400
Facsimile: 8200 2406
http://www.sardi.sa.gov.au

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# EXECUTIVE SUMMARY

The Adelaide metropolitan coast has lost over 5,000 ha of seagrass habitat between Bolivar and Aldinga (over recent decades), due to increasing levels of anthropogenic pollution and coastal development. As a consequence, the Adelaide foreshore and marine environment have undergone considerable and ongoing erosion. However, recent remedial action aimed at reducing the nutrient load discharged into Gulf St Vincent is predicted to greatly improve water quality and hence seagrass growth and survival in the future. As a result, research into seagrass rehabilitation in the metropolitan region is now a worthwhile exercise.

Following the Seagrass Restoration Workshop held in Adelaide in 2001, a long-term three-phase program of research and development over 9+ years for seagrass rehabilitation was developed for the Adelaide metropolitan coast. This program aims to examine a number of approaches to seagrass rehabilitation, including:

1) Donor bed dependant methods, involving the collection of mature seagrass from a donor meadow (e.g. cores and sprigs) for transplanting to areas of seagrass loss.

2) Donor bed independent methods, involving the collection, successful germination, growout and planting of seedlings.

3) Recruitment facilitation methods aimed at maximising the success of natural recruitment events in addition to facilitation of the spread of fast growing species in preparation of sites for in-planting cultured seedlings of slow growing species.

This report documents experiments on the transplantation of mature plants, and is the first of a series of reports examining the potential of these approaches for seagrass rehabilitation along the Adelaide metropolitan coastline. Subsequent reports will cover the culture and propagation of seedlings and the facilitation of natural recruitment, in addition to a literature review of seagrass rehabilitation efforts worldwide (in particular focussing on research since the Seagrass Workshop in 2001).

Fieldwork for the donor dependent trials started in February of 2003. Trial planting sites were set-up off Henley Beach and West Beach for comparing two commonly used transplant methods (cores and sprigs) using two species of seagrass, *Amphibolis antarctica* and *Heterozostera tasmanica*, sourced from mature donor beds.

Of the planting sites selected for these initial trials, West Beach was less suitable than first expected, with poor survival of both species. This was attributed to one or more of the following factors; higher wave energy, poor water quality (possibly due to proximity to the

Barcoo and Torrens outlets), large rafts of drift seagrass and algae, and a possible net loss of sediment from the area. This highlights the need to carefully assess potential planting sites on a number of levels (e.g. water quality, sediment stability and hydrodynamic regime) a fact that was continually stressed by participants at the Seagrass Restoration Workshop (Seddon & Murray-Jones, 2002). The difficulty for selecting sites along the Adelaide metropolitan coast is that, with the exception of sediment movement (Fotheringham, 2002), very little fine scale site-specific data exist regarding water quality, light availability and hydrodynamics with which to make informed decisions.

Due to the preliminary nature of this study, it was not possible to conclude whether the use of biodegradable hessian mats was a successful sediment stabilisation strategy. At West Beach this strategy was unsuccessful, as many of the mats appeared to have washed away, causing the loss of associated transplants, particularly sprigs. However, it was noted that there was a significant amount of natural recruitment of *A. antarctica* seedlings to the mats during winter. This finding will be followed up in 2004/05 with a detailed study of methods to enhance natural recruitment. Thus, hessian mats (that are useful for securely attaching sprigs), may only be suitable for seagrass rehabilitation in lower wave energy environments.

Interestingly, while the core method provided for higher survival (due in part to loss of mats with the sprigs), the surviving sprigs appeared to be healthier, indicating the potential for further refining of the sprig method. Over the first nine months of monitoring, there were no obvious signs of lateral expansion of transplants. However, in March 2004, one year after planting, lateral expansion of *H. tasmanica* transplants (both cores and sprigs) was evident. It is clear that determining the success of transplant trials requires a long-term commitment, and further monitoring is recommended to document the rate of expansion of both cores and sprigs to assess whether this changes following the summer growth period.

An evaluation of the costs indicated that transplanting *H. tasmanica* was more economical than *A. antarctica,* and that sprigs were more cost effective to plant. However, in assessing the relative merits of species and methods, this needs to be weighed up against the potential for both species to spread into neighbouring areas. Further monitoring of the planting sites over time should allow us to determine the rates of lateral expansion and how well the transplants survive their second winter.

Overall, while both the core and sprig techniques are possible, the success rates in these initial trails were low (similar to those found in W.A.); however, these methods may be suitable for less energetic sections of the metropolitan coast. Given their poor success, relatively labour intensive nature and issues of potential donor bed damage, the use of cores and sprigs is not likely to be viable for large scale rehabilitation in Adelaide metropolitan waters. Accordingly, the primary focus for future work investigating seagrass rehabilitation

along the Adelaide metropolitan coast should be on developing seedling growout and transplanting techniques, in addition to the development of techniques to enhance the success of natural recruitment events.

# 1 Introduction

There has been a major decline of inshore seagrass meadows (predominantly *Posidonia* spp and *Amphibolis* spp) along the Adelaide metropolitan coast in recent decades. The loss of over 5,000 ha of seagrass habitat between Bolivar and Aldinga (Hart, 1997; Cameron, 1999) has been linked to increased anthropogenic pollution and coastal development. The general pattern of loss is unusual in that the shallow edge of the seagrass meadows has progressively receded seaward (Hart, 1997). This is in contrast to the commonly held model of seagrass loss in response to declines in water quality in which sedimentation and epiphyte growth lead to light limitation and loss of seagrass at the deeper margins (Bastyan 1986, Hillman *et al* 1990, Larkum 1976, Larkum and West 1990, Ralph *et.al.* in press and Shepherd *et. al.* 1989). This seaward regression of seagrass has been compounded by increasing meadow fragmentation since the late 1960s (EPA, 1998). As a consequence, the Adelaide foreshore and marine environment have undergone considerable and ongoing erosion (Fotheringham, 2002).

Sewage and stormwater discharges into Gulf St Vincent have been implicated as a major cause of seagrass loss along the Adelaide coast (Neverauskas, 1987; Shepherd *et al*., 1989; EPA, 1998; Seddon 2002). The greatest rates of seagrass loss occurred during the 1970s, coinciding with high nutrient and suspended solid loads associated with increased stormwater and sewage discharge along the metropolitan coast (Neverauskas, 1989; Shepherd *et al*., 1989; EPA, 1998). Another possible factor contributing to seagrass decline is the large volume of freshwater discharged from storm water outfalls during major rainfall events, especially given that the seaward recession of seagrass followed the installation of stormwater outfalls during the 1960s (Seddon, 2002). Additionally, high inshore turbidity associated with stormwater runoff and sediment resuspension, particularly during frequent winter storms, is very likely to contribute to seagrass decline (Seddon, 2002).

Fortunately, both the South Australian EPA and SA Water have recognized the need for remedial action to reduce the nutrient load discharged into Gulf St Vincent. SA Water, which is responsible for Adelaide's waste water treatment plants, are in the midst of an environmental improvement program aimed at reducing their discharge of nitrogen into Gulf St Vincent by 71% (Hamilton, 2002). Such initiatives are predicted to greatly improve coastal water quality and hence seagrass growth and survival in the future. For instance, there is evidence of seagrass recolonisation on bare sand in the vicinity of the decommissioned Port Adelaide sludge outfall (Bryars & Neverauskas, 2002). It is largely as a result of these anticipated improvements in water quality that research into seagrass rehabilitation in the metropolitan region is now worthwhile.

## 1.1 Seagrass Rehabilitation R&D Program

In May 2001, SARDI Aquatic Sciences and the Coastal Protection Branch (formally Coast and Marine Branch) initiated and held Australia's first workshop on seagrass restoration to review the current status and knowledge in Australia and overseas (Seddon & Murray-Jones, 2002). Following the workshop, we developed a three phase R&D program over 9 years for seagrass rehabilitation (**Figure 1.1**) along the lines of that proposed by Seddon and Cheshire (1999), with the ultimate objective being to develop reliable techniques for seagrass restoration and rehabilitation suitable for application along the Adelaide metropolitan coast.

Phase 1, which started in November 2002, was originally intended to focus on developing methods for establishing a seagrass nursery to supply large numbers of seedlings for wide-scale planting. Due to the lack of fruiting by the key species over the 2002/03 summer, elements of Phase 2 were initiated in early 2003. In particular, the utility of transplanting mature seagrasses was investigated in 2003, and this is the focus of this report. Some work on facilitating natural recruitment has also been undertaken. Phase 2 proper was proposed to start in November 2004 and originally aimed to develop methodologies to facilitate and augment seagrass recolonisation, but has been postponed owing to the extension of Phase 1 and a lack of funding. Finally, it is hoped that these seagrass rehabilitation methodologies, if shown to be successful on a wider-scale, will be incorporated into a program designed to be run and implemented by the community and stakeholders (Phase 3; **Figure 1.1**).

To date, the techniques used in seagrass restoration are incredibly labour intensive and are only suitable for small areas (several hectares at most), or require expensive machinery designed for major dredging operations where large areas of seagrass need to be salvaged and transplanted (e.g. van Keulen & Paling, 2002). These methods are not suitable or affordable for large scale seagrass rehabilitation. Hence the need for a dedicated program of research directed toward the development of cost effective techniques suitable for application along the Adelaide metropolitan coast.

**SEAGRASS REHABILITATION R&D PLAN**

**PHASE 1**

Establish seagrass nursery methodologies to provide propagules for future wide-scale planting. The focus is on developing techniques for the collection and germination of seeds, grow-out and subsequent planting of seedlings (mainly *Posidonia* spp.).

**Duration:** 1 year.

**Funding:** State government (Coastal Protection Branch, SARDI Aquatic Sciences).

**PHASE 2**

Develop planting methodologies for seagrass rehabilitation and facilitation of natural recolonisation suitable for application over large areas. Assess the success of various techniques and monitor for the return of ecosystems function (fish and other macrofauna).

**Duration:** Up to 3 years

**Funding:** Commonwealth FRDC, with contributions from SARDI Aquatic Sciences, the Coast & Marine Branch, Victorian EPA and Murdoch University. (FRDC proposal submitted in December 2002).

**PHASE 3**

Implementation of methodologies developed during Phase 1 and 2 on a wide-scale by community groups.

**Duration:** 5+ years (ongoing)

**Funding options:** Commonwealth government (National Heritage Trust II), State government (CMB), local councils and community groups (e.g. wide-scale rehabilitation by dive clubs etc. coordination and training by the Conservation Council of SA's Reef Watch Program).

**Figure 1.1 The 9+ year Seagrass Rehabilitation program that was submitted as part of an unsuccessful SARDI proposal for FRDC funding in 2003.**

## 1.2 Phase 1

The two approaches being investigated in Phase 1 are:

1. Donor bed dependant methods, involving the collection of mature seagrass from a donor meadow (e.g. cores and sprigs) for transplanting to areas of seagrass loss; and
2. Donor bed independent methods, involving the collection, successful germination, growout and planting of seedlings.

This document reports on the first part of Phase 1, that is, donor dependent methods of seagrass rehabilitation. It documents a field trial of two methods of transplanting seagrass (cores and sprigs) of two species, *Amphibolis antarctica* and *Heterozostera tasmanica*, which commenced in early March 2003. These species were chosen for initial trials because they are known to successfully recolonise bare sand following seagrass loss (e.g. blowouts).

The second part of Phase 1, donor bed independent methods of seagrass rehabilitation, is running concurrently with this study, with ongoing work to continue through 2004. The second report of this series will document initial work on the seedling nursery techniques in 2002 and progress to date on seedling culture for the 2003/2004 season. The results of the completed initial seedling trials will be presented as a final report in late 2004. Studies on the facilitation of natural recruitment commenced in 2003 and will be the focus of work in 2004/05 with a report to be submitted in late 2005.

## 1.3 Donor Bed Dependent Methods

Transplanting mature seagrass sourced from donor meadows, either as cores or sprigs, is a common method utilised in seagrass restoration and rehabilitation (e.g. Fonseca *et al.*, 1998). Cores (also referred to as plugs) are advantageous in that they minimise disruption of the roots and rhizomes, although they typically require more time than sprigs for collection and transport. On the other hand, sprigs (sections of rhizomes with shoots intact) are relatively quick to collect and have a lesser impact on the donor bed. The main drawback of sprigs is that they are easily washed away in higher wave energy environments. For example, in Western Australia, where the wave energy is very high, only cores are used as sprigs are quickly dislodged, even when pegged into the substratum. The greater below-ground mass of cores enables the transplant to withstand greater water movement. Paling *et al*. (2002) found that the greater the diameter of the core, the higher the wave energy that can be endured before being dislodged. At very high wave energies, sods (55 cm x 44 cm x 35-50 cm) are used. These have sufficient mass that mechanical equipment must be used to transplant them making this a very specialised transplantation technique.

Overall, in terms of time and resources required to collect the seagrass from donor meadows and then transport and plant it at the recipient planting sites, cores and sprigs have their

advantages and disadvantages. Both techniques rely on the presence of healthy seagrass meadows within a reasonable distance from the sites to be rehabilitated; and both have some degree of impact on the donor bed, which may or may not be acceptable depending on its size and the amount of seagrass to be removed. Nevertheless, these methods have proven reasonably successful elsewhere in Australia (Lord *et al.* 1999) and it is important to establish whether they can be successfully applied in South Australia. The limitations of these techniques also need to be assessed in a local context and potential improvements to existing methods investigated. It is also important to develop a skill and knowledge base within South Australia to facilitate future seagrass rehabilitation. Hence the objectives of the initial phase of the seagrass rehabilitation program focus on the application of donor bed dependent methods along the Adelaide metropolitan coast. The two main objectives that are examined in this report are:

1) To assess the relative ease and cost of transplanting cores and sprigs from mature meadows for seagrass rehabilitation.

2) To assess the establishment success (in terms of survival, health and coverage) of seagrass shoots transplanted to areas of seagrass loss.

# 2 Methodology

## 2.1 Selection of Planting and Donor Sites

Planting sites were chosen based on their suitability for seagrass growth, one at West Beach (6-7 m depth) and the other at Henley Beach (5-6 m depth)(**Figure 2.1**). Previous work by Fotheringham (2002) indicated that these sites are subject to relatively low sediment movement so are potentially suitable for rehabilitation. However, it is yet to be established, whether the water quality is adequate over the long term. This is a particular concern with regard to the West Beach site, as rainfall and associated runoff discharged from the recently constructed Barcoo outlet could degrade water quality substantially.

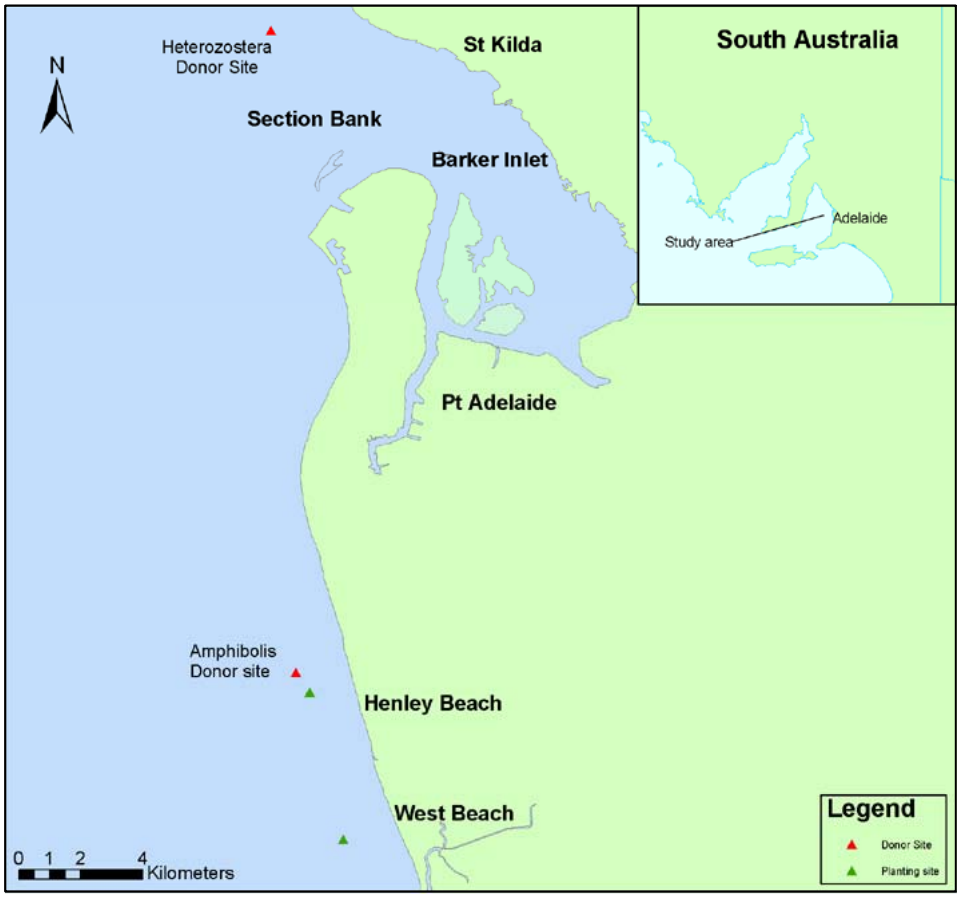


**Figure 2.1 Location of the two experimental planting sites and the two donor sites.**

Donor sites were chosen based on availability of plants and their similarity (depth, water movement and water quality) to the planting sites. *Amphibolis antarctica* was collected from a 6-7 m deep site at Henley Beach (Figure 2.1). Unfortunately, due to the sparse nature of *Heterozostera tasmanica* in the vicinity of the experimental sites, this species had to be collected from a shallower site (2-3 m) further afield near St Kilda.

## 2.2 Transplant Methods

### 2.2.1 Sediment Stabilisation

Sediment instability associated with wave action and tidal currents is a major consideration that could compromise attempts at rehabilitation. To address this issue, biodegradable hessian mats (see **Figure 2.4**) were used in the planting of both the cores and sprigs, both to stabilise the sediment around the transplants and to more effectively secure the sprigs to the substratum. The hessian should also provide a more stable substrate for the rhizomes to attach to and grow around, facilitating their lateral expansion.

### 2.2.2 Cores

Transplanting cores of seagrass is a commonly used method in seagrass restoration (e.g. Fonseca *et al.*, 1998) and while the inclusion of sediments means they require more time for collection and transport, they have the advantage of minimising the disruption of the roots and rhizomes (e.g. van Keulen & Paling, 2002). In this study, cores were obtained by inserting a 25 cm length of 10 cm diameter PVC pipe into the sediment surrounding the plants (**Figure 2.2**). Care was taken to ensure that as much of the stems and shoots as possible were inside the PVC core to minimise damage. A small amount of sediment around the core was excavated, the core and its contents lifted out and a cap placed on the bottom, before transportation to the planting sites.

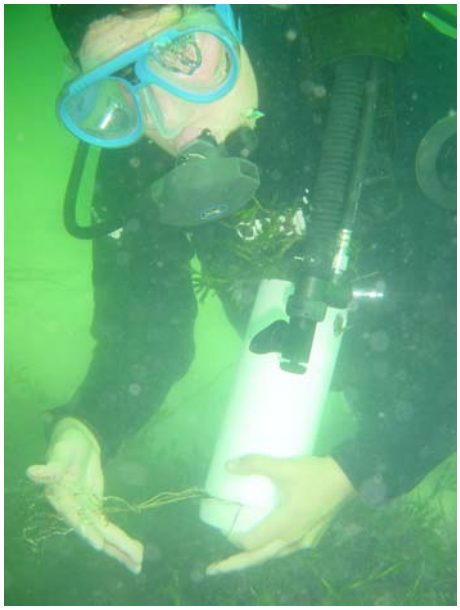

**Figure 2.2 Diver carrying a PVC core of *A. antarctica* collected from the donor meadow at Henley Beach.**

At the planting sites, two replicate hessian mats were installed for each species, with ten cores per mat. These mats (1.2 m × 3 m) were placed over an area where approximately 5 cm of sediment had been excavated. Mats were anchored to the substratum using steel tent pegs in the corners and 20 U-shaped steel wire pegs (wire diameter approx 2 mm) spaced evenly over the mat. In addition, the edges of the mats were buried in a deeper trench. Once firmly secured, the hessian weave was eased apart and sediment excavated in the position that the core was to be planted. The core was then placed in the hole, the PVC sleeve removed, and sediment filled in around the plant. When all ten cores were in place, a layer of sediment approximately 5 cm deep was replaced over the entire mat.

### 2.2.3 Sprigs

Transplanting sprigs (a section of rhizome with roots, shoots and leaves attached) is another method commonly used in seagrass restoration (e.g. Fonseca *et al.*, 1998). Sprigs have the advantage in that they are relatively quick to collect, easy to check for apical meristems and have a lesser impact on the donor bed. In this study, sprigs were collected from the outer edge of the donor bed to ensure the sprig had a growing tip (leading edge) to accelerate the colonisation process. Forty sprigs of approximately 30 cm length were collected by fanning (by hand) the sediment from around the roots and rhizome, while gently lifting the plant rhizome out (**Figure 2.3**).

**Figure 2.3 A typical *A. antarctica* sprig showing four shoots each comprising a woody stem and several leaf clusters attached to an intact section of rhizome complete with roots.**

Sprigs were transported to the boat and then woven into the hessian mats in trays of seawater and loosely attached using jute string (**Figure 2.4**). Sprigs were positioned in three rows approximately 25 cm apart. Each row was made up of sets of two sprigs tied along side each other. Sprig mats were then placed into tubs of water and transported to the planting sites. Two replicate mats per species, each with eighteen sprigs, were planted at each site.

Mats with sprigs attached were installed using the same method as outlined for cores. Once secured in place, sediment was deposited on the mat, covering the rhizomes but leaving the shoots and leaves of the sprigs exposed.

**Figure 2.4 The SARDI / CPB field team weaving *A. antarctica* sprigs into a biodegradable hessian mat.**

### 2.2.4 Installation and Timing

Collection and planting of *A. antarctica* was completed in February 2003 and *H. tasmanica* by mid March 2003 (**Table 2.1**). This process took somewhat longer than expected due to weather conditions, equipment problems and difficulties finding a suitable donor site for *H. tasmanica*.

**Table 2.1. Timetable of activities for the transplanting experiment during 2003. WB = West Beach, HB = Henley Beach, Section Bank = SB.**

| Date | Activity |
|---|---|
| 17-Jan | Selection of donor and planting sites. |
| 07-Feb | Begin collection and planting at pilot site. |
| 20-Feb | Assess pilot site. |
| 26-Feb | Experimental collection and planting of *A. antarctica* from HB donor site to WB planting site. |
| 27-Feb | Complete planting *A. antarctica* at WB, search for HB planting site. |
| 06-Mar | Experimental collection and planting *A. antarctica* to HB site. |
| 07-Mar | Establish monitoring for *A. antarctica* donor site, replant *A. antarctica* cores at WB* |
| 12-Mar | Search for *H. tasmanica* donor site SB. |
| 17-Mar | *H. tasmanica* collection at SB and planting at WB. |
| 18-Mar | *H. tasmanica* collection at SB and planting at HB. |
| 01-Apr | First monitoring of *A.antarctica* donor site and transplant health at HB, transplant success at both planting sites. |
| 08-Apr | First monitoring of *H.tasmanica* donor site and transplant health at WB. |
| 10-Jun | Second monitoring of *A.antarctica* donor site and transplant health at HB, transplant success at both planting sites. |
| 11-Jun | Second monitoring of *H.tasmanica* donor site and transplant health at WB. |
| 10-Sep | Third monitoring of transplant success at both planting sites. |
| 16-Oct | PAM flourometry monitoring at Henley Beach. |

* Some of the *A.antarctica* cores needed to be replaced as a batch were sourced from an area of deeper sediments, resulting in less roots and rhizomes being extracted with the PVC core.

While there has been extensive inshore seagrass loss along the metropolitan coast, in deeper water further offshore, the seagrass forms continuous dense meadows that appear relatively intact (Seddon, 2002). It is the junction between the area of seagrass loss and surviving healthy meadows that is of most interest for the initial stages of seagrass rehabilitation. By encouraging the expansion of existing meadows inshore to recolonise areas of seagrass loss, we can utilise the protection and stability offered by the existing meadows. For this reason, the planting sites were located on the inshore side of substantial *Posidonia* beds. Alternating core and sprig mats were laid out as closely together as possible, within 1–2 m of existing *Posidonia* beds (or in some cases, *Amphibolis*) and were aligned along the seagrass edge to reduce environmental variability. (**Figure 2.5** and **Figure 2.6**).

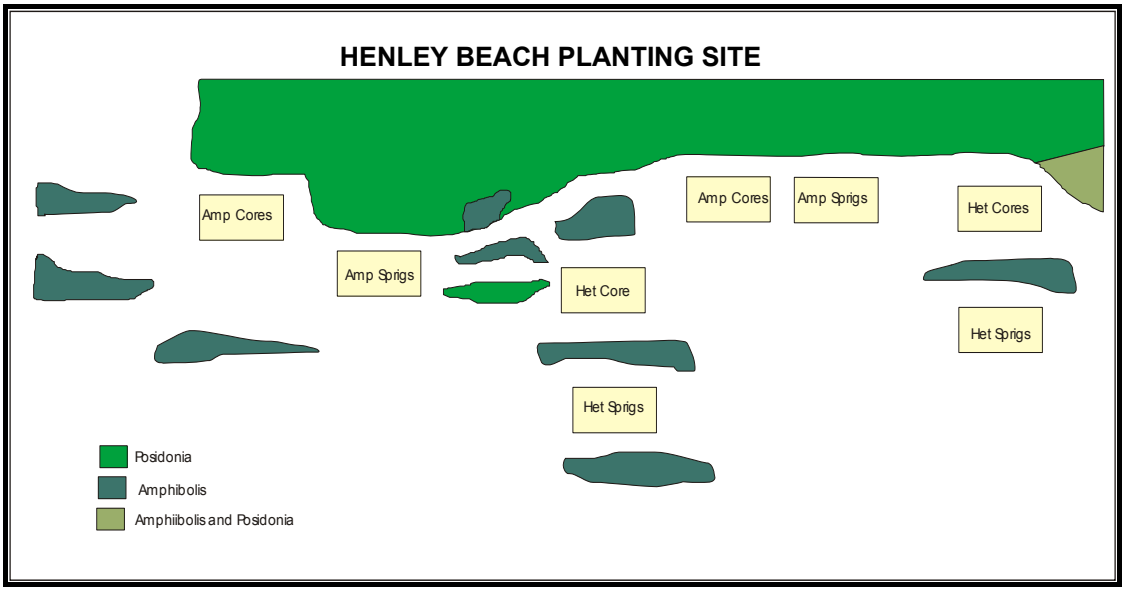


**Figure 2.5 Schema of the Henley Beach planting site, showing placement of hessian mats in relation to the adjacent seagrass meadow. Amp = *Amphibolis antarctica*, Het = *Heterozostera tasmanica*. Not drawn to scale.**

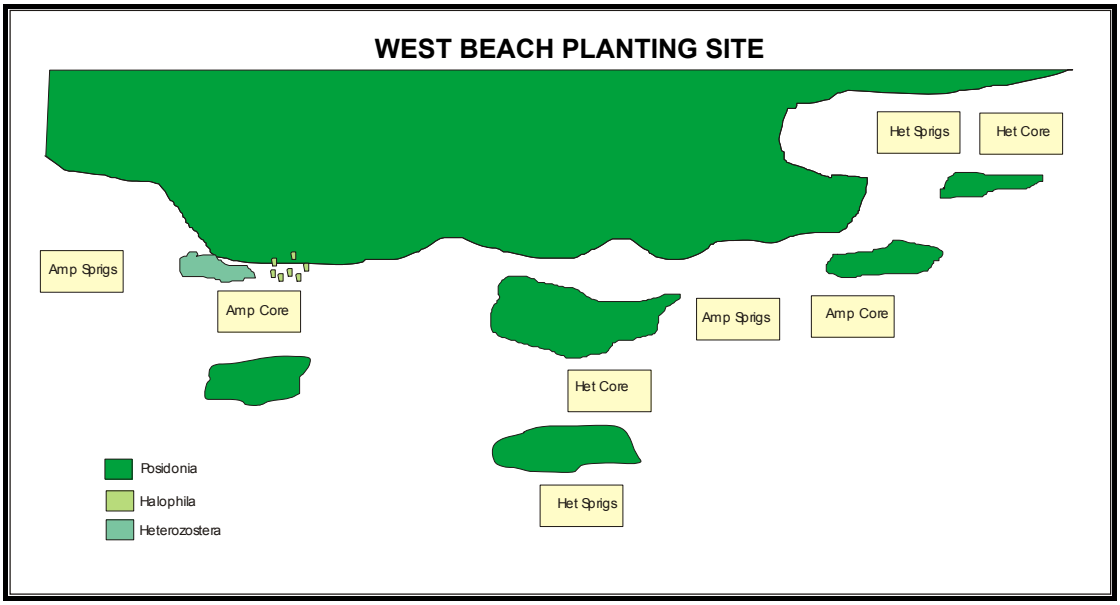


**Figure 2.6 Schema of the West Beach planting site, showing placement of hessian mats in relation to the adjacent seagrass meadow. Amp = *Amphibolis antarctica*, Het = *Heterozostera tasmanica*. Not drawn to scale.**

## 2.3 Monitoring

The monitoring program consisted of three components: 1) Observations of the donor bed from which cores were removed (i.e. checking for seagrass recolonisation or increases in the size of the holes due to erosion: 2) Monitoring transplant success and physiological condition; and 3) monitoring environmental variables at the donor and planting sites.

Monitoring was carried out in April, June and September 2003. While sampling was intended to be quarterly, the last trip was delayed due to adverse weather conditions and logistic constraints.

### 2.3.1 Donor Sites

Observations of the holes left after the removal of cores in the donor beds began in April 2003. Five holes were marked (using flagging tape and a small buoy on a tent peg) and monitored at each of the donor sites. Video recordings were made of the holes and were visually assessed as a qualitative measure of whether holes were being recolonised or eroding and becoming larger. These recordings were carried out concurrent with monitoring of the transplants.

### 2.3.2 Planting Sites

#### *Transplant Survival and Growth*

Transplant survival was assessed immediately after planting, then approximately every three months at both the West Beach and Henley Beach planting sites (**Figure 2.1**). An assessment of each hessian mat was made to ascertain the presence or absence of cores and sprigs and the condition of the mat (whether buried, exposed or missing). The mat as a whole and individual cores and sprigs were recorded on video as a visual reference against which to gauge future spread of the transplants. Shoot survival was assessed by counting the number of shoots present within each core or sprig.

#### *Physiological Condition*

The physiological condition of the transplants was assessed by measuring the photosynthetic efficiency of a sub-sample of the transplants, in addition to control plants at both donor and planting sites, using a diving Pulse Amplitude Modulated (PAM) Fluorometer (Walz, Germany; **Figure 2.7**). Measurements were made before and after 15 minutes of dark adaptation to calculate the maximal quantum yield (Fv/Fm), which is a measure of the potential photosynthetic capacity of a sample on a leaf. The level of activity of photosystem 2 as measured using PAM is widely used as an indicator of plant health/stress in terrestrial and marine environments (Briantais *et al*. 1986, Renger & Shcreiber 1986, Ralph 1999). Thus, when a number of leaves from different shoots within a bed are measured, it is possible to get an estimate of the physiological condition or “health” of seagrass within the bed.

Fluorescence measurements for *A. antarctica* were made on a single healthy leaf, by positioning the leaf clip (Diving-LC, Walz, Germany) on the fourth or fifth leaf blade from the base of one of the uppermost leaf clusters. Measurements for *H. tasmanica* were made using two healthy blades from the same shoot positioned side by side in the leaf clip (due to the

narrow blades of *H. tasmanica*, two leaf blades were required to cover the leaf clip aperture to provide a strong fluorescence signal).

At the donor sites, 10 replicate measurements were made for each species ("donor site controls"). At the planting sites, 10 replicate measurements were made for both species on cores (one plant per core) and 10 on sprigs (one plant per sprig). In addition, 10 replicate measurements were recorded for sprigs of both species from plants growing in natural beds adjacent to the experimental mats ("planting site controls"). In all cases, replicates were measured from separate plants.

**Figure 2.7 Diver using the PAM fluorometer on *A. antarctica* leaves to assess plant health.**

## *Physical conditions*

Limited sampling of a range of water quality parameters was carried out opportunistically in conjunction with monitoring trips. Measurements of temperature, pH, salinity turbidity and dissolved oxygen were made using a Horiba W22XD water quality meter. Light (PAR) was measured using a Licor underwater light sensor. Measurements of all parameters were made just below the surface (approx. 0.5m) and at the bottom.

## *Data Analysis*

The mean number of shoots and maximal photochemical yield (Fv/Fm) for each species and treatment were calculated. For shoot numbers a repeated measures ANOVA was used to assess changes over time while a one-way ANOVA was used to determine significant differences in physiological health between the treatments for each monitoring survey (a Tukey's test was applied *post hoc* to determine which treatment means were significantly different).

# 3 Results and Discussion

## 3.1 Monitoring

### 3.1.1 Donor Sites

Visual observations of the donor sites indicated that there was little evidence of further impact following the initial core removal from the *Amphibolis antarctica* donor bed. Video footage of a subsample of core holes immediately after collection and at the end of the study showed no obvious changes (**Figure 3.1** and **Appendix 1**). However, it is not possible to make a definitive conclusion without a more rigorous assessment method (e.g. standardised photography and measurement of hole dimensions *in situ* and/or digitally). Sediment quickly accumulated in the holes created by core extraction to form a level surface with no obvious erosion. Nevertheless, seagrass around the monitored holes did not show any clear signs of recolonisation into the holes within the 9 month period of monitoring although monitoring did not extend into the summer peak growth period.

Poor visibility and weather conditions during the June 2003 monitoring period made locating the *Heterozostera tasmanica* donor bed to assess coring impact impossible. Further searches in good conditions during September 2003 were also unsuccessful and it is assumed that the markers (pegs and flagging tape) at this shallower site were dislodged during storms.

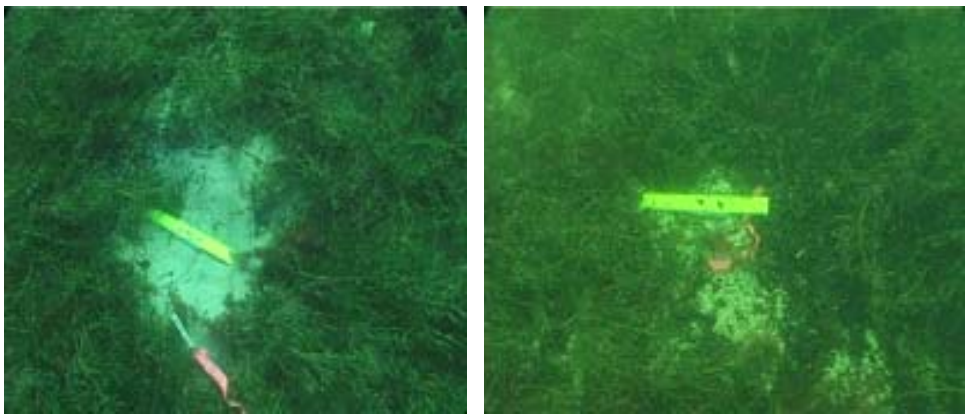

**Figure 3.1 An example of recovery of the holes produced from coring *A. antarctica*. Field observations suggest the holes have accumulated sediment. However, as of September 2003, there was no evidence of seagrass regrowth into these holes (see appendix 1 for more examples).**

### 3.1.2 Planting Sites

#### *Transplant Survival and Growth*

Overall the survival rate of the transplants was lower at West Beach than at Henley Beach (**Figure 3.2**). At West Beach the sprig mats had washed away by the second monitoring trip in June 2003, approximately 3 months after the experiment was initiated, whereas three of the four sprig mats remained at Henley Beach. Stronger surge was observed at West Beach, which is likely to be exposed to a slightly higher level of wave energy relative to Henley Beach, however, this needs to be verified. This probably contributed to the early loss of hessian mats (and therefore some of the transplants) during storm events (discussed further in the environmental monitoring section).

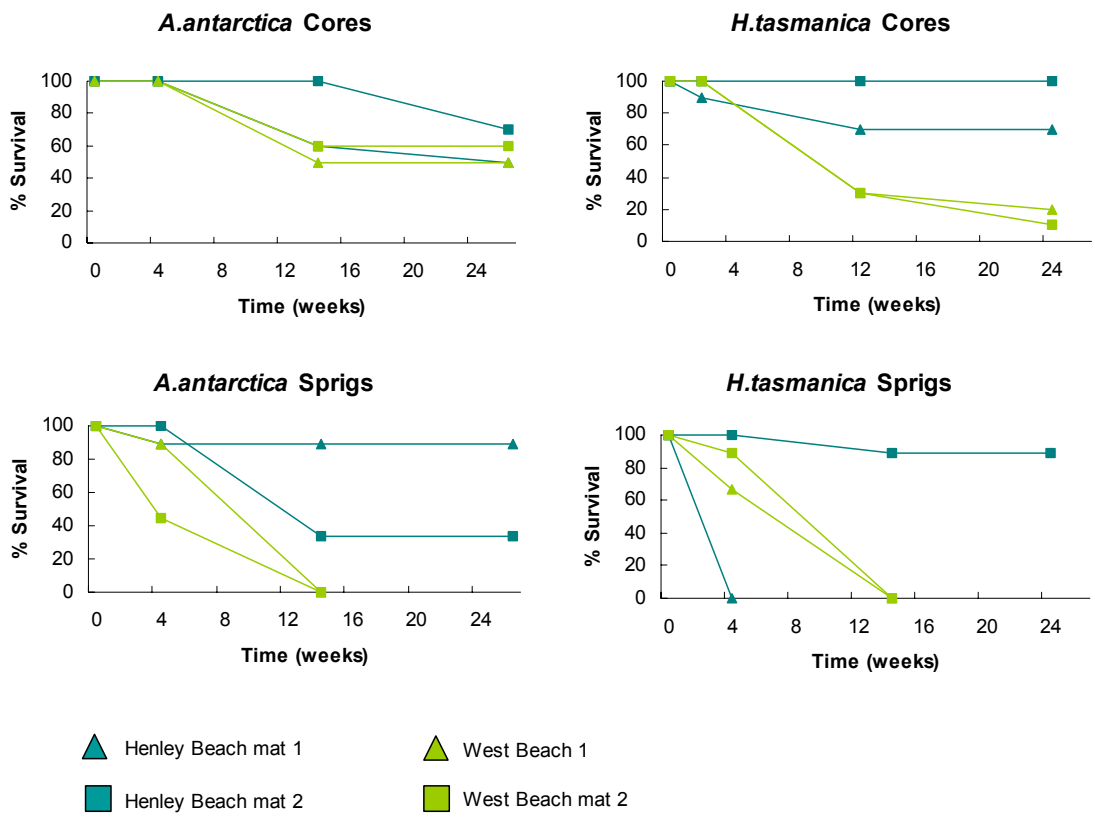


**Figure 3.2 Survival rates for *A. antarctica* and *H. tasmanica* cores and sprigs from March until September 2003.**

Cores tended to have higher survival rates than sprigs (**Figure 3.2**). This is at least partly explained by the fact that the sprigs were attached directly to the mats, which in 5 out of 8 cases (all 4 at West Beach) were dislodged, presumably during storms, destroying all attached sprigs. The loss of mats did not have such a large effect on cores. However, looking only at sprigs on mats that remained intact, they still had a lower survival rate than cores.

As a result of the loss of the hessian mats at West Beach, and poor visibility and weather conditions at this site during the June monitoring trip, shoot counts could not be obtained past April 2003. As a result, only data obtained from the Henley Beach planting site are reported. Of the transplants that survived, the number of shoots per sprig did not change significantly over time for either species, suggesting there was no major decline in the health of the plants (**Figure 3.3** and **Table 3.1**). While there was also no significant decline in the mean number of shoots per core for *A. antarctica*, there was a 42% decline in the number of shots per core for *H. tasmanica* between April and June 2003, indicating a decline in the condition of the plants (**Figure 3.3** and **Table 3.1**). It is worth noting that the shoot densities can vary seasonally in healthy seagrass beds. As a result, decreases in shoot density do not necessarily mean that the transplants are performing poorly relative to plants in intact meadows. However, differences between methods still provide a good indication of relative perfomance of plants subjected to the transplanting procedures. Thus, *H. tasmanica* cores obviously performed poorly relative to sprigs and to *A. antarctica*.

**Table 3.1. Results for a repeated measures ANOVA for differences in numbers of shoots per transplant for *Amphibolis antarctica* (A.a.) and *Heterozostera tasmanica* (H.z.) over the three monitoring times.**

| Treatment | F value | Hypothesis df | Error df | Significance |
|---|---|---|---|---|
| **A.a. Cores** | 2.021 | 2 | 13 | 0.172 |
| **A.a. Sprigs** | 0.922 | 2 | 13 | 0.432 |
| **H.z. Cores** | 10.262 | 2 | 13 | 0.002 |
| **H.z. Sprigs** | 3.109 | 2 | 13 | 0.118 |

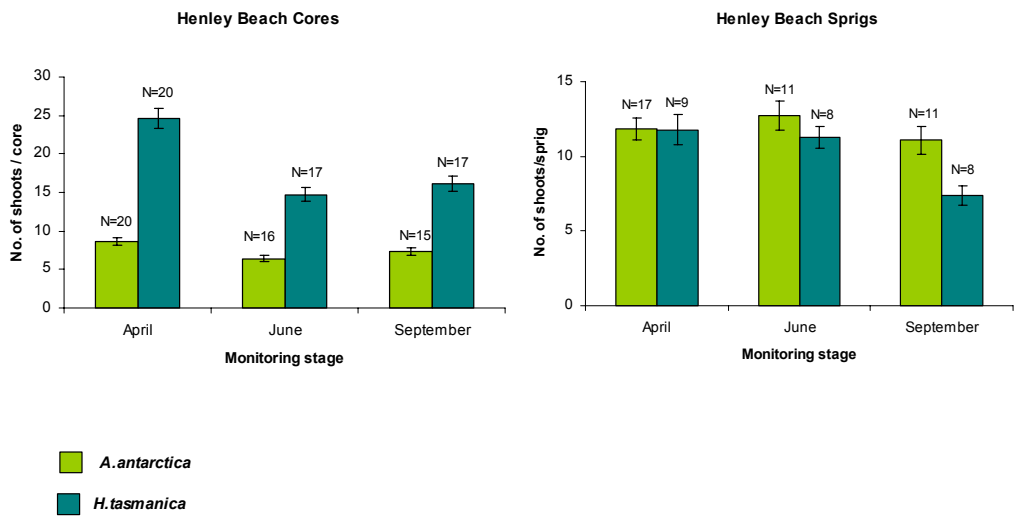


**Figure 3.3 Mean number of shoots from the surviving cores and sprigs for the April (first), June (second) and September (third) monitoring trips at the Henley Beach site. Error bars show the standard error of the mean.**

### 3.1.3 Physiological Condition

In general, measurements of effective quantum yield in April, approximately four weeks after transplantation, point to a decrease in the photosynthetic efficiency of the transplants, with the exception of the *Heterozostera tasmanica* sprigs (**Figure 3.4** and **Appendix Table 2.1a & b**). This suggests most of the transplants are experiencing a decline in health relative to control plants at the donor and planting sites. Significant differences in effective quantum yield between treatments were evident for *Amphibolis antarctica* (**Appendix Table 2.1a**). Both cores and sprigs returned lower average yield measurements than controls, suggesting poorer health, however, this difference was only significant compared to the donor meadow control (**Figure 3.4**). Effective quantum yield measurements also differed significantly between treatments for *H. tasmanica* (**Appendix Table 2.1b**). The average yield measurement for cores was lower and more variable than for other treatments, including sprigs (**Figure 3.4**), suggesting that the transplant process impacted the health of cores but not sprigs.

Signs of physiological stress were still evident for *A. antarctica* cores and sprigs in June 2003, approximately 4 months after transplanting. Effective quantum yields for both cores and sprigs were significantly lower than for control plants in both the donor and planting beds (**Figure 3.5** and **Appendix Table 2.2a**). Unfortunately it was not possible to monitor the *H. tasmanica* transplants any further. During the June monitoring trip, large amounts of drift vegetation covering the site, combined with storm damage and poor visibility, prevented divers from finding many transplants. By October 2003, the drift vegetation had decreased and divers confirmed that most of the *H. tasmanica* plants at West Beach were missing.

Effective quantum yields were not significantly different between treatments by the time of the last monitoring trip, suggesting that plants had largely recovered from the physiological stress associated with transplantation (**Figure 3.5** and **Appendix Table 2.2 b & c**).

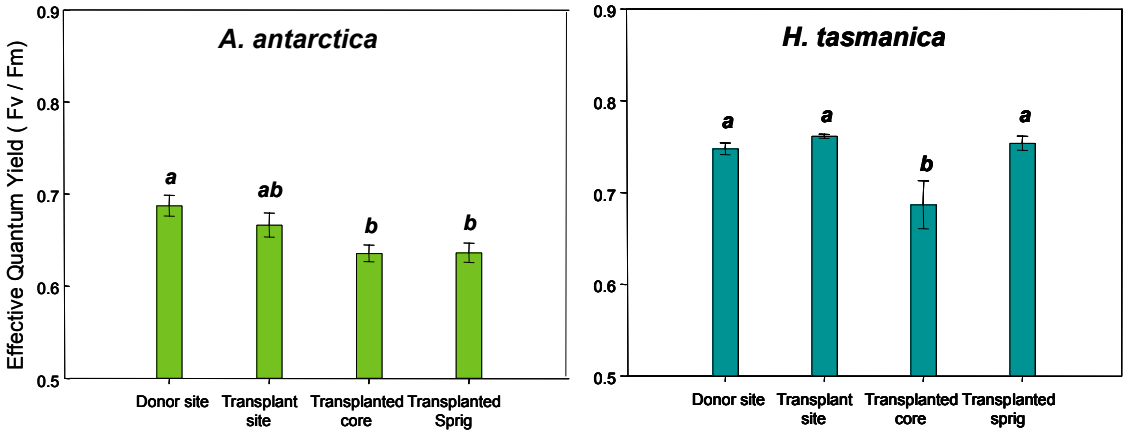


**Figure 3.4 Maximal effective quantum yield (Fv/Fm) for control and transplant plants of *H. tasmanica* at the donor and planting sites four weeks after transplanting. Letters above each bar indicate Tukey test results i.e. different letters indicate that the means are significantly different from one another. Error bars show the standard error of the mean.**

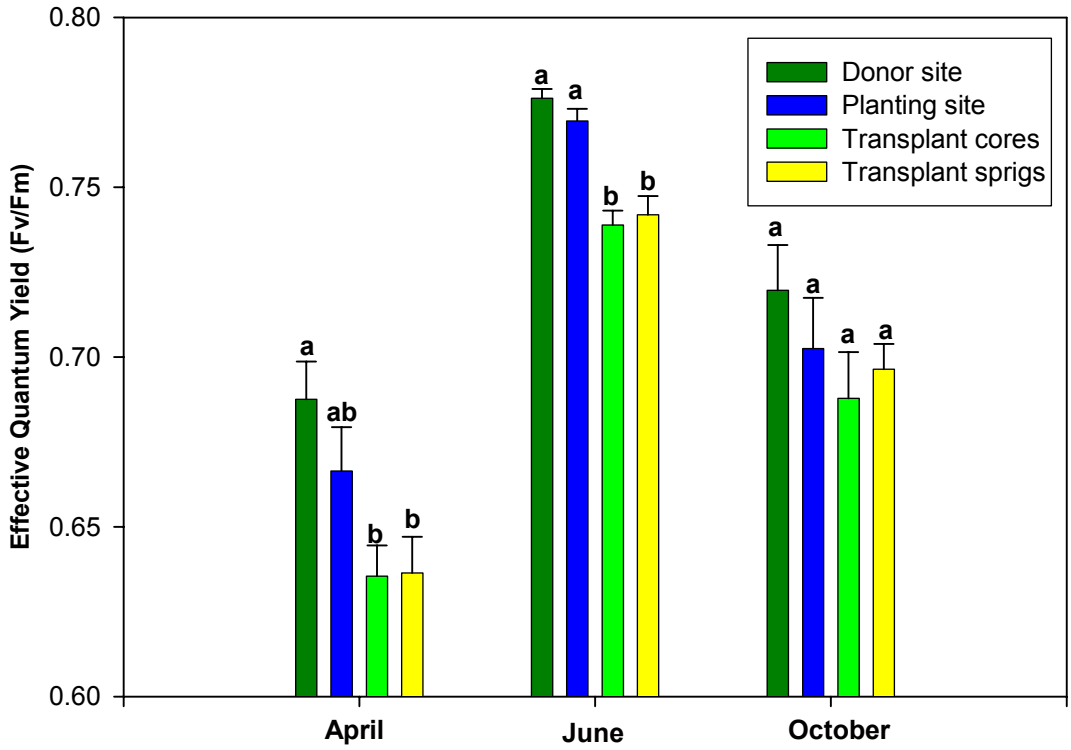


**Figure 3.5 Maximal effective quantum yield (Fv/Fm) for control and transplant plants of *A. antarctica* over the three monitoring times. Letters above each bar are results from *post hoc* Tukey tests i.e. different letters indicate that the means are significantly different from one another. Error bars show the standard error of the mean.**

## 3.2 Environmental Conditions

### 3.2.1 Planting and Donor Sites

Variation was evident in the nature of the sediments and dominant vegetation between the four sites. The donor site for *A. antarctica* is relatively close to the Henley Beach planting site (<1 km) and conditions and sediment structure appear similar. Both are presumably subject to similar tidal movement and water quality; although the *A. antarctica* donor site is approximately 1 m shallower. As mentioned earlier, the donor site for *H. tasmanica* was located in relatively shallow water and therefore subject to different environmental conditions (such as greater wave energy during storms). In particular, sediments from the *H. tasmanica* donor site appeared much coarser than those from the planting sites.

Several differences were observed between the two planting sites. Although both sites supported substantial *Posidonia* meadows, the Henley Beach site also had a number of smaller *A. antarctica* beds. While *A. antarctica* was not naturally present at the West Beach site, there was a small amount of *Halophila australis* and *Heterozostera tasmanica* (**Figure 2.5** and **Figure 2.6**). Another major difference between the two planting sites was the amount

of drifting vegetation. West Beach consistently had large amounts of drifting vegetation (mostly *Posidonia* leaves), which was particularly thick during June when it prevented divers from locating many of the transplant mats. By comparison, much smaller and variable amounts of drift vegetation were observed at Henley Beach.

It is likely that the West Beach planting site is also subject to higher water movement than the Henley Beach site, particularly during times of increased wave energy. Divers reported a more noticeable swell at West Beach on a number of occasions when both sites were visited on the same day. Both sites have similar orientation and so it is unlikely that there are large differences in tidal currents experienced at each site.

### 3.2.2 Physical conditions

Diver observations suggested water quality varied between sites. Underwater visibility was generally poorer at West Beach than at Henley Beach. This was particularly evident in June, when monitoring occurred soon after a storm (**Table 3.1**). The West Beach planting site is in close proximity to both the Torrens river and Barcoo outlets and as such, is likely to be more severely impacted by the effects of stormwater runoff. Divers noted extremely turbid water on 20/02/03, during the first major rainfall event following the 2002/3 summer.

Limited sampling of a range of physical water quality parameters was undertaken in June and the only substantial difference between sites was a 1.5$^{o}$ C lower temperature at the *H. tasmanica* donor site. This site was much shallower and probably subject to less mixing and a more variable temperature range is not unexpected. Although expensive, in a future monitoring program, it would be worthwhile sampling the sediment nutrient and organic content at the donor and planting sites on a seasonal basis, as these are likely to have a significant effect on seagrass growth and survival (Ralph *et. al.* in press).

**Table 3.1 Water quality data collected from all sites on the second monitoring trip in June.**

| Location | Depth (m) | Temp $^{o}$C | pH | Salinity (‰) | Turbidity (NTU) | $DO_2$ (mg $L^{-1}$) |
|---|---|---|---|---|---|---|
| *A. antarctica* donor site | Surface (0) | 14.76 | 8.11 | 38.33 | 0 | 9.69 |
| | Bottom (6.0) | 14.66 | 8.11 | 38.28 | 0 | 8.90 |
| *H. tasmanica* donor site | Surface (0) | 13.01 | 8.06 | 38.90 | 1.15 | 9.79 |
| | Bottom (2.8) | 12.93 | 8.09 | 38.83 | 4.50 | 9.60 |
| Henley Beach planting site | Surface (0) | 14.63 | 8.10 | 38.23 | 4.65 | 8.90 |
| | Bottom (6.0) | 14.59 | 8.11 | 38.30 | 0 | 8.73 |
| West Beach planting site | Surface (0) | 15.32 | 8.09 | 38.03 | 0 | 9.53 |
| | Bottom (6.8) | 14.58 | 8.10 | 38.30 | 2.95 | 9.02 |

Light availability, as determined through sporadic instantaneous measurements, was highly variable between the sites on a given day (Table 3.2). Light intensity for each site was measured just below the surface of the water (~5 cm) and on the bottom. On the bottom, light intensity ranged from a low 12 to 381 $\mu$mol $m^2$ $s^{-1}$. The *H. tasmanica* donor site received comparatively high levels of light compared with the planting sites, which is not surprising as this site is the shallowest. While it is possible that a difference in light availability may exist, the highly variable nature of light data collected for the two planting sites shows that more light data measurements (preferably simultaneous continuous measurements i.e. light loggers) are required before any definitive conclusions can be made.

Of more relevance for seagrass survival and growth is the percentage of light reaching the canopy compared with that just below the surface, which is known as percent surface irradiance (Table 3.2). This is a relative measure of water clarity, which unlike light intensity is dependent upon time of day and weather conditions. It has been widely reported that seagrass meadows require a minimum of 8 to 20% surface irradiance for continued survival, depending on species (Dennison *et al*., 1993; Cabello-Pasini *et al*., 2002). However, for seagrass transplants the recommended minimum light requirements are at least 25% surface irradiance, due to the higher energy requirements for growth and repair of the transplants (Fonseca *et al*., 1998). The limited light data from this study indicate that the minimum light requirement for transplants was not met at either planting site during the June monitoring trip. If this is typical of the winter season, then the transplants may not be receiving enough light to grow, and possibly to survive if they don't have sufficient energy reserves stored. Hence good water clarity in the following spring/summer season will be crucial not only to allow for transplant growth and repair, but to store carbohydrate reserves for the next winter. It may thus be beneficial to compare the success of transplants established early in the growing season with transplants established later in the year.

**Table 3.2 Light availability at the various sites measured each monitoring trip using a Li-Cor light meter.**

| Location | Date | Depth (m) | Light Intensity ($\mu$mol m$^2$ s$^{-1}$) | Percent Surface Irradiance (%) |
|---|---|---|---|---|
| *A. antarctica* donor site | 08-Apr-03 | Surface | 545 | |
| | | 6 | 94 | 17 |
| | 10-Jun-03 | Surface | 661 | |
| | | 6.5 | 12 | 2 |
| | 10-Sept-03 | Surface | 550 | |
| | | 7 | 381 | 69 |
| *H. tasmanica* donor site | 08-Apr-03 | Surface | 504 | |
| | | 2.6 | 221 | 44 |
| | 10-Jun-03 | Surface | 840 | |
| | | 3.8 | 176 | 21 |
| | 10-Sept-03 | Surface | 783 | |
| | | 3 | 225 | 29 |
| West Beach planting site | 08-Apr-03 | Surface | 339 | |
| | | 6.7 | 53 | 16 |
| | 10-Jun-03 | Surface | 493 | |
| | | 7.1 | 61 | 12 |
| | 10-Sept-03 | Surface | 294 | |
| | | 7.8 | 79 | 27 |
| Henley Beach planting site | 08-Apr-03 | Surface | 740 | |
| | | 6.2 | 128 | 17 |
| | 10-Jun-03 | Surface | 322 | |
| | | 6.8 | 24 | 7 |
| | 10-Sept-03 | Surface | 819 | |
| | | 7.5 | 176 | 22 |

### 3.2.3 Storms and Weather Observations

During the period between the establishment of the experimental planting sites and the June 2003 monitoring, several severe storms occurred along the Adelaide coast (**Table 3.3**). The first was in early autumn when wind speeds exceeded 60 km h$^{-1}$ and the second was in early winter, when wind speeds reached almost 70 km h$^{-1}$. This was a good test as to whether the transplants could survive the high wave energy and associated turbulence generated by these storms.

The planting sites are relatively shallow (5 - 7 m) and are likely to be affected by storm waves. Along the metropolitan coast it has been estimated that the significant wave height ($H_s$) is around 2 m (with a period of 5 s) at wind speeds of 60 km h$^{-1}$ (Adelaide Coast Protection Strategy Review, 1984). At a depth of 6.5 m, the representative sea bed water velocity that the seagrass transplants may experience is approximately 0.8 m s$^{-1}$; however, if we consider a 1/100 wave height ($H_{max}$) of 3.3 m, then the seabed peak orbital velocity could be as high as

1.25 m $s^{-1}$ (pers. com. M. Townsend[1]). Under these sea conditions, depth has a major influence on the seabed water velocity and if the transplants were planted closer inshore, for example at a depth of 3 m, this wave would have already broken and reformed into a 1.5 m wave with a peak seabed orbital velocity of 1.25 m $s^{-1}$. Seagrass meadows further offshore at 8 m depth would experience peak orbital velocities of around 1 m $s^{-1}$ (pers. com., M. Townsend). These preliminary calculations of seabed water velocities during storm events indicate the difficulty for seagrass to re-establish inshore and possibly may explain why seagrass beds are relatively intact offshore in terms.

**Table 3.3 Maximal wind speed and direction during storms as measured by the Adelaide airport station between the period of the 15 January to 17 June 2003. Note, winds from between south and north west (180o – 315o) have greatest fetch and generate the roughest sea states. Data from Bureau of Meteorology.**

| Date | Time | Wind speed (km $h^{-1}$) | Wind Direction (degrees) |
|---|---|---|---|
| 19 March | 7:39 | 63 | 240 |
| 4 June | 23:19 | 68 | 280 |
| 5 June | 18:17 | 63 | 320 |
| 6 June | 3:28 | 61 | 340 |

## 3.3 Cost Analysis

Sprigs were more cost effective than cores overall (**Table 3.4)**. The consumables purchased for installing cores were slightly more expensive than sprigs per planting day, although the planting equipment required for installing sprigs (per mat and per area) was marginally more expensive than cores (**Table 3.4**). The underwater time required to transplant a sprig was less than for cores, however, *A. antarctica* sprigs took longer to plant than *H. tasmanica* due to the more complex rhizome structure. Sprigs also required some surface preparation time prior to planting (weaving and securing onto the hessian mats) and consequently most of the diver time saved for sprigs was during the collection phase (**Table 3.4**). This is an advantage because underwater time is more costly.

Overall *H. tasmanica* proved more time effective to transplant than *A. antarctica* (**Table 3.4**), which is likely the result of two main factors. First, the above ground shoots of *H. tasmanica* are less complex in structure and lower in biomass than *A. antarctica* (which has branching woody stems that are frequently covered in epiphytic algae), and were therefore easier to separate and put into cores. Secondly, *H. tasmanica* rhizomes are smaller in diameter, less

[1] Murray Townsend (CPB) calculated estimates of wave heights and seabed orbital velocities using Automated Coastal Engineering Systems software.

woody and grow closer to the surface of the sediments, all of which made *H. tasmanica* easier to collect.

**Table 3.4 Transplant costings calculated per hessian mat and per unit area ($m^2$) averaged over all sites. *H. tasmanica* was generally more cost effective than *A. antarctica*, and underwater, sprigs were quicker to collect and plant than cores. Data are in minutes unless indicated otherwise.**

| Task / Equipment | Species | Units | Cores | Sprigs |
|---|---|---|---|---|
| **Consumables** | | Per mat | $50 | $43 |
| **Planting Equipment** | | Per mat | $10 | $11.10 |
| | | Per area ( $m^2$) | $2.80 | $3.08 |
| **Surface Preparation** | | Per mat | - | 20 |
| | | Per area ( $m^2$) | - | 6 |
| **Diving Time** | ***A. antarctica*** | Per mat | 102 | 63 |
| | | Per area ( $m^2$) | 27 | 18 |
| | ***H. tasmanica*** | Per mat | 84 | 57 |
| | | Per area ( $m^2$) | 23 | 15 |
| **Total cost per established plant** | ***A. antarctica*** | **(mat value/no. of plants per mat)** | **$13.21** | **$11.70** |
| | ***H. tasmanica*** | **(mat value/no. of plants per mat)** | **$11.94** | **$11.28** |

Diver efficiency also was a factor influencing timing. The efficiency of the divers improved with experience, and given that planting of *A. antarctica* was carried out prior to *H. tasmanica*, gains in efficiency may have had some influence on the relative efficiency of the two species. In addition, the donor site for *H. tasmanica* was shallower than that for *A. antarctica*, which also made operations at that site easier in terms of locating the site and moving equipment and cores. Divers also noted that the visibility was consistently better at the *H. tasmanica* donor site, possibly due to coarser sediment and therefore faster settlement times, which also added to efficiency.

Conditions above and below the water also affected collection and planting efficiency in general, which no doubt added a degree of variability when attempting to compare the two techniques. For example, surface conditions affected the efficiency of locating sites and raising and lowering equipment, cores and sprig mats. Underwater, visibility affected the time required to locate sites and for divers to move equipment and plants.

# 4 Conclusions

Overall transplant success was notably lower for both cores and sprigs at the West Beach planting site compared with the Henley Beach planting site. In considering the suitability of donor dependent methods for seagrass rehabilitation along the Adelaide metropolitan coastline, several key issues need to be taken into consideration. Site selection is probably the most important step in the rehabilitation process (Seddon, 2002). On a broad scale, commitments by SA Water and the government, along with recent observations of regrowth in a number of areas (e.g. Bryars and Neverauskas, 2002) suggest that the coast as a whole is becoming more amenable to rehabilitation. However, on a small scale, this study suggests that careful consideration needs to be given when selecting sites.

Sites for this study were chosen based on past observations (by the Coast Protection Branch, DEH) that suggested these areas were not likely to suffer sediment loss. While this was true for the Henley Beach planting site, diver observations at the West Beach site suggest that there was in fact a loss from this area during the winter of 2003. This apparent loss in conjunction with poorer water quality (possibly due to proximity to the Barcoo and Torrens outlets) and large amounts of drifting seagrass wrack, no doubt contributed to the relatively low success of transplants at West Beach. To better understand inter-site differences it is suggested that future work (transplant or other methods) include the measurement of changes in sediment depth on a finer temporal scale (compared with the annual CPB rod profile monitoring). This could consist of simple measurements on site marker posts to assess relative change in sediment depth. It is also hoped that over time, our experience will provide a basis for identifying the characteristics of various sites that are best suited for specific rehabilitation methods. This is likely to be a balance between increasing success rates and reducing costs.

This was a relatively small-scale study (in terms of number of sites and replicate mats for both cores and sprigs within each site) limited by the resources available and as such, was not able to provide conclusive evidence as to wether the use of hessian mats was successful. A much larger study involving a number of sites located along a wave energy gradient, and increasing the area planted within each site (i.e. more replicate mates per species per site) would be advisable. It would also be worthwhile including several variations and refinements of methodologies tested. Equally important is a comprehensive monitoring program (including shoot densities at donor and control meadows) over several years to more fully investigate the potential of mats and other methodologies. Other factors that should be considered for a rehabilitation-monitoring program are continuous light measurements and seasonal sampling of the sediment nutrients.

It was hoped the mats would stabilise surrounding sediment and provide anchorage for expanding rhizome. While the mats were relatively successful at Henley Beach, many were lost from the West Beach site. It seems that once a section of the mat is exposed, wave energy in storm events acts upon that section to quickly dislodge the whole mat further and the loss of mats often (always in the case of sprigs) equates to plant loss. The contribution of the mats to stabilisation in the initial phase of transplantation could be tested in future studies through the comparison of transplants with and without the mats at several different sites.

For the sprig transplant method, the mats provided a convenient method of attaching and planting sections of rhizome and plant. However, when mats were lost, so were all the sprigs. Without attachment to a medium such as the mats, we presume that it very likely the sprigs would quickly be dislodged. While cores for both species provided a relatively tangled mass of root and rhizome material that could anchor plants, single rhizomes are likely to be more easily dislodged. The use of sprigs is therefore worthy of further investigation using other planting methods, including the use of smaller sections of mat or attachment to other media such as lengths of string or rope pegged beneath the substrate (with differing orientations to swell and current). Particularly considering the relatively lower cost of using sprig mats, they may be suitable in lower wave energy areas in the northern metropolitan region or sheltered waters elsewhere in South Australia.

The general condition of the transplants suggested that although survival seemed poor for the sprig method, this was largely due to the loss of mats during storms rather than a deterioration of the health of the sprigs that remained in place. Measurements of physiological condition using the PAM fluorometer, suggested that initially cores of both species and sprigs of *A. antarctica* were affected, while shoot counts suggested only *H. tasmanica* cores suffered a decline in health as a result of the transplanting process. Signs of physiological stress were gone within 6 - 9 months, although a longer period of monitoring (at least through the next summer growth season) would be required to see growth and expansion of transplants[2].

For large scale rehabilitation of seagrass meadows, costs are a fundamental consideration. This study found that overall *H. tasmanica* was more economical to transplant, particularly during collection. The sprig method was generally cheaper than the core method (e.g. dive time was reduced). Both methods are likely to be relatively expensive compared to non transplanting methods (i.e. donor bed independent methods) which will be discussed in the second report of this series.

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[2] This conclusion was confirmed as field observations on 3rd March 2003 at the West Beach planting site confirmed lateral expansion of the *H. tasmanica* cores and sprigs. This comes at the end of the summer growing season, almost one year after the transplants were installed.

It was noted during the course of this experiment that over winter, *A. antarctica* seedlings naturally recruited to exposed areas of hessian mat in high numbers. This observation was extremely interesting in that it may present a method of facilitating the recruitment of this species (and to a lesser extent others) relatively quickly and easily. As a result of this finding, two trials were opportunistically set-up (all though very late in the season) as a pilot study.

## 4.1 Recommendations

- Future monitoring should include assessment of sediment nutrients and organic content and include finer scale measurements of light availability.
- Future trials include the regular measurement of sediment movement into or away from trial planting sites.
- Future testing and development of donor bed dependent methods should include:
    1. Comparisons of cores and sprigs with and without hessian mats.
    2. Other methods of sprig attachment.
- Expand the current trials to include the use of hessian strips and possibly other methods for the facilitation of *A. antarctica* recruitment, beginning mid 2004 after the end of winter to maximise the use of the spring/summer growing season and avoid the winter storms.

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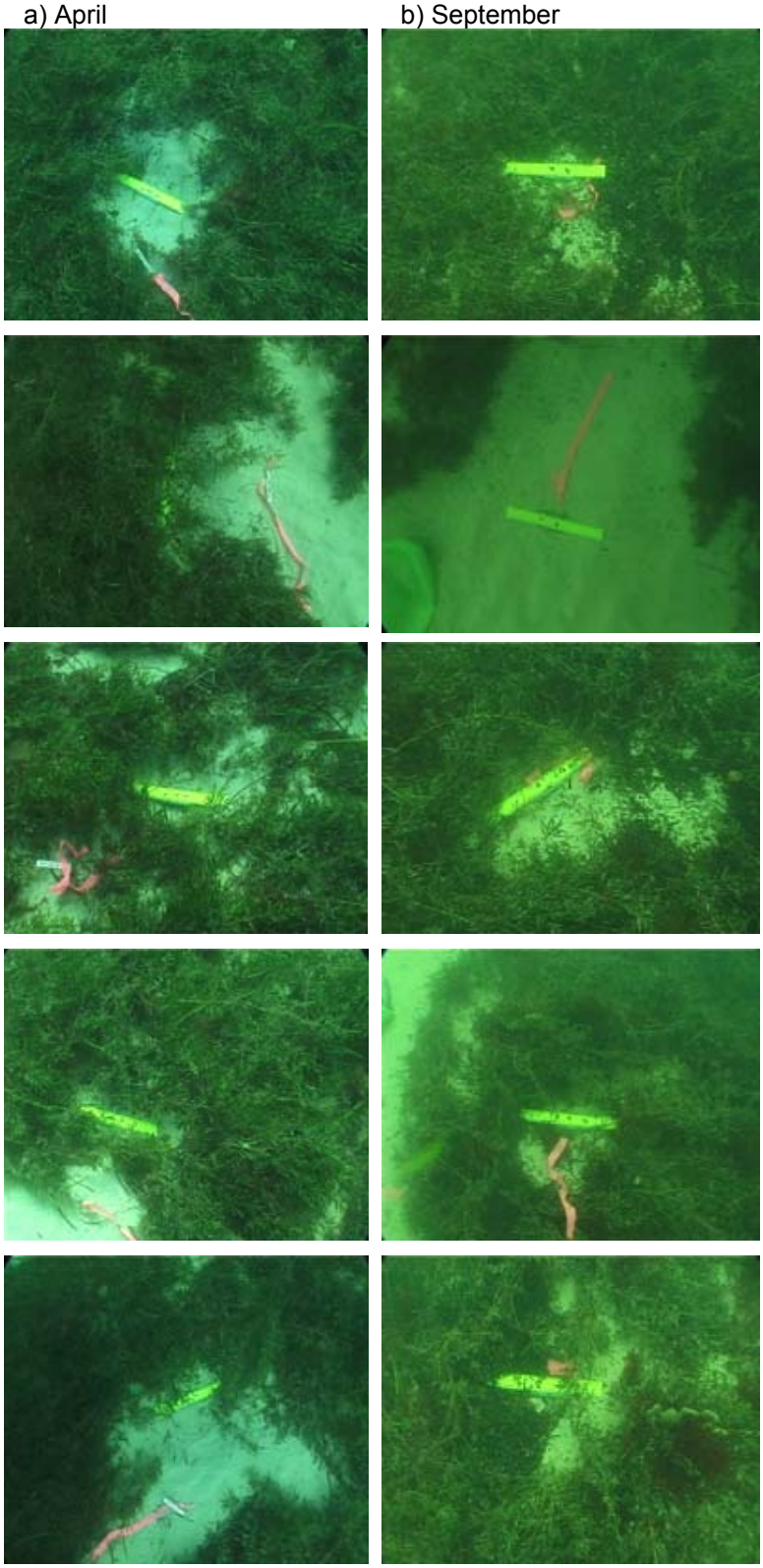
# Appendices

## Appendix 1. *Amphibolis* core hole recovery.

**Video stills of five *Amphibolis* cores holes at the time of the first monitoring trip (April) and the last (September). A substantial change in the size of holes is not obvious, however, it is suggested that future studies include standardized photographic monitoring of holes to make quantitative assessment of recovery possible.**

a) April b) September



## Appendix 2. Details of statistical analysis

**2.1. ANOVA tables for differences in effective quantum yield for *Amphibolis antarctica* (a) and *Heterozostera tasmanica* (b) transplants in April 2003, approximately 4 weeks after planting.**

| a) | **Sum of squares** | **df** | **Mean squares** | **F** | **Significance** |
|---|---|---|---|---|---|
| Between groups | 0.01885 | 3 | 0.006282 | 5.023 | .005 |
| Within groups | 0.04377 | 35 | 0.001251 | | |
| Total | 0.06262 | 38 | | | |

| b) | **Sum of squares** | **df** | **Mean squares** | **F** | **Significance** |
|---|---|---|---|---|---|
| Between groups | 0.03446 | 3 | 0.01149 | 5.403 | .004 |
| Within groups | 0.07440 | 35 | 0.002126 | | |
| Total | .109 | 38 | | | |

**2.2 ANOVA tables for differences in effective quantum yield for *Amphibolis antarctica* cores and sprigs in June (a), cores in October (b) and sprigs in October (c) of 2003. Note, October analyses had to be performed separately as data for cores and sprigs were collected on separate days and therefore controls differed.**

| a) | **Sum of squares** | **df** | **Mean squares** | **F** | **Significance** |
|---|---|---|---|---|---|
| Between groups | 0.01080 | 3 | 0.003600 | 21.011 | .000 |
| Within groups | 0.006168 | 36 | 0.0001713 | | |
| Total | 0.01697 | 39 | | | |

| b) | **Sum of squares** | **df** | **Mean squares** | **F** | **Significance** |
|---|---|---|---|---|---|
| Between groups | 0.0085780 | 2 | 0.004289 | 2.2094 | 0.1292 |
| Within groups | 0.052414 | 27 | 0.001941 | | |
| Total | 0.060992 | 29 | | | |

| c) | **Sum of squares** | **df** | **Mean squares** | **F** | **Significance** |
|---|---|---|---|---|---|
| Between groups | 0.001.8163 | 2 | 0.0009080 | 0.5876 | .5626 |
| Within groups | 0.0417264 | 27 | 0.001545 | | |
| Total | 0.0435427 | 29 | | | |