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Research Article

Recovery of Donor Meadows of *Posidonia sinuosa* and *Posidonia australis* Contributes to Sustainable Seagrass Transplantation

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Donor meadow recovery is important in deciding whether removal of material from natural seagrass meadows is a sustainable activity. Thus an investigation into meadow regrowth was undertaken as part of a large-scale seagrass rehabilitation effort in Cockburn Sound, Western Australia. Several plug extraction configurations were examined in *Posidonia sinuosa* and *Posidonia australis* meadows to monitor shoot growth into plug scars. No significant differences in shoot growth between extraction configurations were observed, and both species increased their shoot numbers over two years, with *P. sinuosa* showing a significantly better recovery rate than *P. australis*. *P. sinuosa* shoot recovery into extracted areas was 2.2 ± 0.1 shoots over 24 months, similar to shoot changes in controls (2.3 shoots over the same period). *P. australis* shoot recovery for each configuration was 0.8 ± 0.3 shoots in 24 months compared with 1.5 shoots in the controls. Based on the number of regrowing shoots, the predicted recovery time of a meadow is estimated at 4 years for *P. sinuosa* and three years for *P. australis*. Different plug extraction configurations do not appear to affect meadow recovery, and it can be concluded that established meadows of both species are sustainable providers of planting units for rehabilitation measures.

1. Introduction

The rate of worldwide seagrass decline has been estimated at 110 km^2 y^{-1} since 1980 and 29% of the total area has disappeared since measurements were recorded in 1879 [1, 2]. Most seagrass losses have been correlated with increases in human activities [3], including coastal development, building of groynes, seawalls and associated dredging of channels and harbours. For example, 21,000 ha of seagrass have been lost globally to dredging alone [4]. In an effort to mitigate for these losses, a number of attempts to restore seagrasses have been undertaken [5–9] using natural beds as donor sites for planting material [5, 10–12]. Mitigation is a compensation measure addressing removal of existing habitat when the agent of loss and responsible party are known [5] and often involves using material from a donor site without damaging it beyond recovery. This is important in deciding whether removal of material from natural vegetation is a sustainable activity [4, 13, 14]. In Cockburn Sound, Western Australia, donor material was obtained from meadows to be dredged, that is, no other meadows were affected. However, this may not always be the case and recovery of the donor meadows is paramount to ensuring it remains a sustainable activity. Despite this, the implications of damaged donor beds resulting from restoration activities have not been studied extensively. Unless there is an alternative source of planting units (PU's) (e.g., seeds or seedlings), the overall success of restoration through transplantation is dependent on the recovery of the donor beds.

Few studies have documented postdisturbance recovery rates of seagrasses and data on the extent of seagrass, meadow recovery is scarce due to seagrass meadows having either failed to recover or recovery was very slow following stress on the meadows [15]. Many transplant projects have been carried out worldwide but have generally included only a few seagrass species [11]. *Zostera marina*, or eelgrass, has been

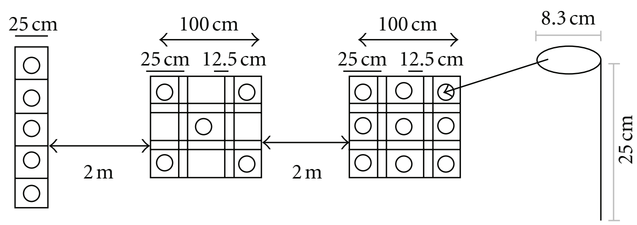


Figure 1: Experimental layout of plug removal to test donor meadow recovery for three treatments: "line 5", "block 5", and "block 9", with schematic view of marker rings (8.3 diameter equals the diameter of the corer) which were placed in the core extraction holes to monitor shoot regrowth into them.

used successfully in the Northern Hemisphere as it tends to recover well [5]. *Zostera* species, however, tend to grow in different hydrodynamic conditions to those of the *Posidonia* species described in this study for Western Australian coastal conditions and, therefore, the success of *Zostera* species cannot be directly transposed to *Posidonia* species. In the Mediterranean Sea, deep meadows (15–30 m) of *Posidonia oceanica* recovered after being disturbed by repeated trawling. Although recovery was slow at a rate of 1–7 cm year^{-1} [16, 17], it demonstrates the capacity of *Posidonia* species to naturally rehabilitate after large-scale and repeated disturbances once the disturbance is taken away [18].

Shellsand dredging activities on Success Bank and harbour development in Cockburn Sound have led to the requirement for development and implementation of rehabilitation measures for seagrasses [19]. Currently, this focuses on Cockburn Sound, where around 80% of meadow area were lost between 1967 and 1999 through a variety of industrial and coastal development activities [20–22]. As a habitat compensation measure for the most recent shellsand dredging and harbour development activities, environmental regulators stipulated that 2.1 ha of seagrass habitat should be replaced by 2012. As part of the development and implementation of procedures for this rehabilitation, a technique for mechanical removal of small cores has been developed. The longer-term goal is one of sustainably using this technique to source material from donor meadows on a large scale using automatic extractors. *Posidonia* species do not recover easily from disturbances partly due to their rhizomatous growth, that is, through spreading, and division, of horizontal shoots [23, 24]. In Western Australia, rehabilitation has also been carried out using planting material from nearby seagrass stands [11, 25–27]. However, the success and timescale of seagrass recovery remains to be investigated [22].

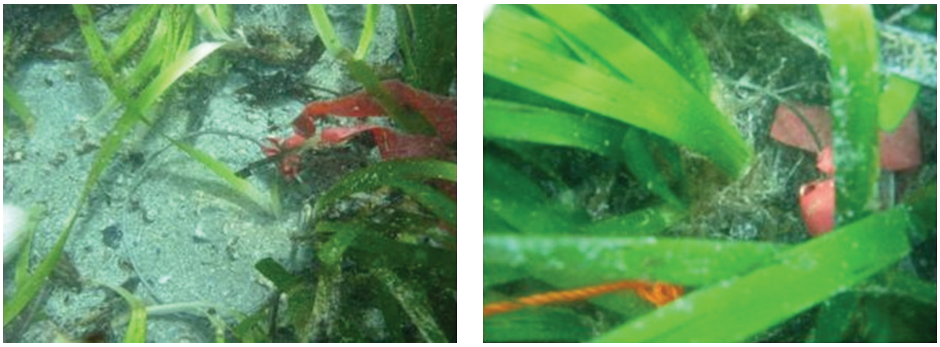
Therefore, this study examines the potential effects of mechanical extraction of donor material in existing *Posidonia sinuosa* and *Posidonia australis* meadows. Three extraction configurations of increasing intensity were used to test potential meadow recovery, which was assessed by monitoring growth of new shoots into extracted plug areas for a period of two years. We postulate that the configuration with increased intensity of coring will give an indication of recovery for larger size disturbances.

2. Methods

Suitable donor and transplant sites were identified near a rehabilitation site on Southern Flats, Cockburn Sound; *Posidonia sinuosa* at S 32°15′042′′, E 115°43′028′′ and *Posidonia australis* at S 32°14′918′′, E 115°42′900′′. The layout of the three density plug extraction configurations in the meadow recovery study were "line", where one plug was removed from each of the five 25 cm × 25 cm quadrats forming a row of a total of 1.25 m (Figure 1), "block five" (a total of five plugs m^{-2} removed, one plug from each 25 cm × 25 cm quadrat, Figure 1), and "block nine" (nine plugs m^{-2} removed, one plug from each 25 cm × 25 cm quadrat, Figure 1). Line, block five and block nine treatments were placed 2 m apart. For each species, four replicates of each configuration were used along with one control. Each replicates and control were placed five meters apart but were located within the same continuous seagrass meadow. Plugs of *Posidonia australis* and *Posidonia sinuosa* were removed using a steel corer, 8.3 cm in diameter and 25 cm depth (same size as mechanical removal device); metal marker rings were placed into the resulting bare area such that the ring was visible but was anchored with the ring attachment to 25 cm (Figure 1) within the sediment to monitor shoot growth into it at 3, 10, 13, and 24 months (Figure 2). As the sand is unconsolidated in this area, the resulting core holes collapsed in themselves; additional sand was not filled into the gaps. The metal marker rings were placed into the remaining hole directly after coring. Marker rings of 8.3 cm diameter were placed into adjacent undisturbed meadows within 5 m of the experimental plots, using the configurations described above, to act as reference plots with no plant material or sediment removed. The number of shoots within the rings was measured, and this was monitored over the same time intervals. Extraction configurations (line, block five, and block nine) were compared for each species. The recovery of shoots over 24 months into hole configurations of line, block five and block nine, was compared for *P. sinuosa* and *P. australis* using a two-way ANOVA (species × configuration). All assumptions for the ANOVA were met and if yielding a significant result ($P < 0.05$), a post hoc pairwise comparison of the means was performed using either Tukey's HSD test or the Student's *t*-test. Analyses were performed using JMP for Windows (Version 8.0, SAS Institute Inc.).

3. Results

The initial numbers of shoots in the control rings for *Posidonia sinuosa* and *Posidonia australis* were 5.5 ± 0.7 and 3.1 ± 0.4, respectively. *P. sinuosa* averaged one shoot after three months (Figure 3(a)). There were no significant differences, within each species, between the line, block five, and block nine treatments. In contrast, mean *P. australis* shoot growth into the extraction holes for each density configuration was less than 0.25 shoots in the first three months (Figure 3(b)), while the control showed a slight decline (−0.3 shoots) over the same time period. After 13 months, both species had increased shoot densities, with *P. sinuosa* showing significantly better growth than *P. australis* ($F_{2,18}$ = 0.8955,



(a) (b)

Figure 2: (a) *P. sinuosa* meadow recovery (Southern Flats), one new shoot after three months within the core marker ring; (b) *P. australis* meadow recovery (Southern Flats), control with five shoots.

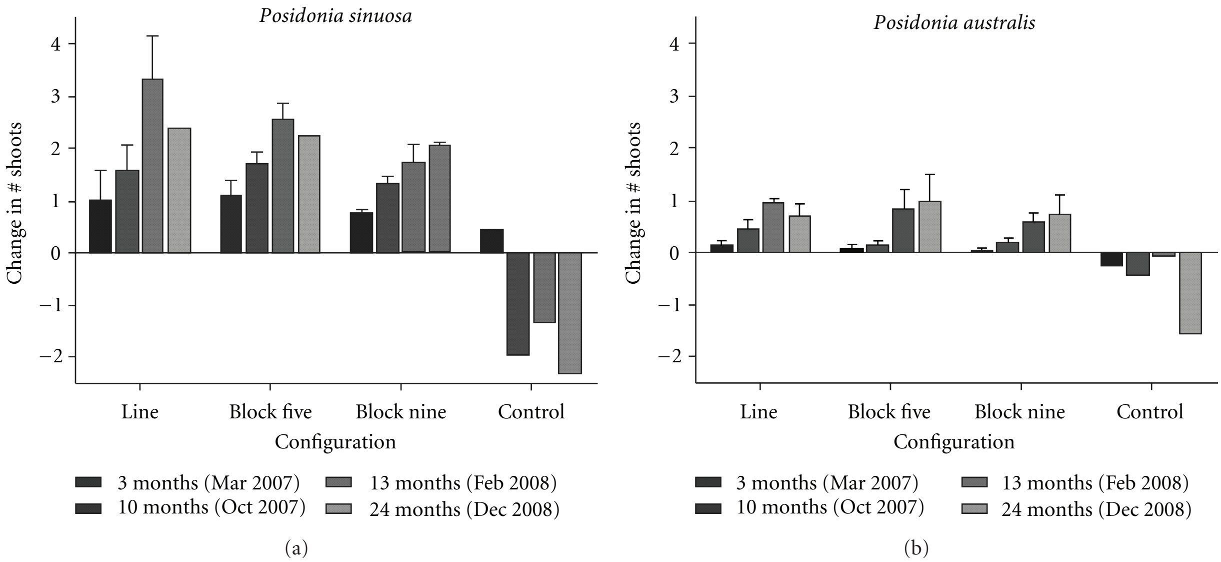


Figure 3: Change in mean number of (a) *Posidonia sinuosa* shoots and (b) *Posidonia australis* shoots over 3, 10, 13, and 24 months after plug extraction ($t = 0$) in configurations of line, block five, and block nine (mean ± SE, n (replicates) = 4, n (control) = 1).

$P < 0.05$). There were no statistical differences, however, between the line, block five, and block nine configurations within each species. After 24 months, *P. sinuosa* again showed significantly higher growth (2.2 ± 0.1 shoots present) than *P. australis* (0.8 ± 0.3 shoots present, $F_{2,14} = 20.1891$, $P < 0.05$), and there were also no significant differences between extraction configurations. After 24 months, the *P. sinuosa* controls showed a loss of 2.3 ± 0.3 shoots present, and *P. australis* controls decreased their number of shoots by 1.5 ± 0.1 shoots.

4. Discussion

Manual rhizome-shoot planting commonly used for rehabilitation in many coastal areas worldwide would seem to be a better solution than removing plugs, in terms of causing less damage to the donor meadows. It is, however a very resource-demanding process [8], and in some cases it will take a long time to establish a rehabilitated meadow. Plugs or larger sods are, therefore, used because ultimately they may reduce meadow establishment time. While either choice can be made when sourcing a donor (or salvage) area that will ultimately be removed by human activities (e.g., a marina development), in those cases where seagrasses are to be harvested from natural populations, the impact on the donor bed becomes imperative to the overall success of the rehabilitation process.

Posidonia species have been considered to display slow plagiotropic rhizome extension rates and it has usually been assumed that recovery from disturbance would be slow [23, 24, 28]. Within a meadow, it might also be expected that the predominant growth pattern is orthotropic and the ability of rhizomes to switch to plagiotropic growth forms to fill in holes, such as those left by plug extraction, would be fundamental to hole recovery. However, our study shows that both *Posidonia sinuosa* and *Posidonia australis* seem to regrow well. After two years of monitoring, the number of shoots regrowth into plug scars amounted to 2.2 ± 0.1 shoots per ring for *Posidonia sinuosa*. Assuming the number of shoots in the control configurations represents a recovery target, it can be concluded that donor meadow regeneration from this extraction method takes three years in this species. Applying similar logic to *P. australis*, which had 1.6 ± 0.2 shoots per ring at the end of the monitoring period in the control configuration, it would take donor meadows of this species four years to recover from plug extraction.

Plugs in our study were removed from the middle of established "steady-state" meadows for both species. However, growth into extracted holes from the surrounding meadow was more rapid in *P. sinuosa*, reflecting more rapid lateral growth. Paling and McComb [29] reported mean rhizome growth rates over 30 days of 4.7 ± 0.03 cm in *P. sinuosa* and 6.5 ± 0.06 cm in *P. australis*. Bastyan and Cambridge [25] found differences in growth rates in *P. australis* from edges and centres of meadows with mean rates of 15–21 cm yr^{-1} and with maxima up to 33 cm yr^{-1}, respectively. *P. sinuosa* tended to have faster linear rhizome growth rates [30], even though there is proportionately less rhizome material than *P. australis* [29]. This may indicate the ability by *P. sinuosa* to switch more rapidly to plagiotropic growth than *P. australis*. It should also be noted that within-species variation in growth can be high; *P. australis* rhizome growth can vary from 9.1 ± 1.0 cm yr^{-1} to 22.3 ± 1.4 cm yr^{-1} depending on location [31].

The current study provides evidence that it is reasonable to assume that meadows of *P. sinuosa* and *P. australis* in Cockburn Sound can be sustainable providers of small sized planting units for rehabilitation. We make a case that the higher-intensity cored configurations give an indication of meadow recovery whereby the rhizosphere has been disturbed repeatedly over an area of 1 m^2 $replicate^{-1}$ despite the small core size. We have shown that with taking small core size from *P. australis* and *P. sinuosa* donor beds showed signs of recovery within three months with an estimated total recovery between three and four years based on the number of shoots regrowing into the core scars. This compares favourably to cited natural recovery periods of 20–30 years for many seagrass species or even 100 years as predicted for *Posidonia australis* [31]. These recovery predictions are in accordance with observations of natural recovery rates of seagrass meadows. Studies on *Posidonia* species impacted by flooding in Albany, Western Australia, indicated a near complete return to natural density after 1.5 years [25]. Seagrass losses of 13 m^2 day^{-1} caused by urchin grazing were observed in *Posidonia* species in Western Australia, but were seen to recover naturally by 15% over a period of four years once the source of disturbance was taken away [32]. Average rates of new shoots into seagrass populations cover rates varying from 0.26 shoots $year^{-1}$ in the large seagrass species *Enhalus acoroides* to 4.81 shoots $year^{-1}$ in the small species *Halodule wrightii* [33]. Variability exists within species, between years and site, in the rate of recruitment of new shoots into populations [34, 35]. However, comparisons of shoot growth and recovery are complicated considering the dynamic nature of the reference meadows.

While it is probable that rhizome growth and subsequent shoot production would be increased in bare areas compared to dense meadows, as there is little competition between shoots [25, 36], growth forms in the middle of the meadow are likely to be of the slower growing, orthotropic form [25]. In this form, more resources are allocated to increasing shoot density than lateral spread; it can reasonably be expected that some time is required for plants to switch to plagiotropic growth.

With the increasing efforts to restore seagrass populations in coastal zones worldwide, this study shows that meadow recovery can be achieved using an appropriate configuration based on dynamic growth specifics of the seagrass concerned.

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