Habitat restoration: Early signs and extent of faunal recovery relative to seagrass recovery

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ABSTRACT

The overall intent of restoration is often not only to restore the habitat *per se*, but to restore the ecosystem services it supplies, and particularly to encourage the return of fauna. Seagrass meadows act as habitat for some of the most diverse and abundant animal life, and as the global loss of seagrass continues, managers have sought to restore lost meadows. We tested how quickly the epifaunal richness, abundances and community composition of experimental restoration plots recovered to that in an adjacent natural seagrass meadow relative to the recovery of seagrass *per se*. Seagrass structure in the restoration plots took three years to become similar to a nearby natural meadow. The recovery of epifaunal richness and total abundance, however, occurred within one year. These results suggest that although recovering habitats may not be structurally similar to undisturbed habitats, they can support similar richness and abundances of epifauna, and thus have greater economic and social value than otherwise might have been expected. Nevertheless, whilst epifaunal richness and total abundance recovered prior to the recovery of seagrass structure, full recovery of seagrass was required before the composition and relative abundances of the epifaunal community matched that of the natural seagrass meadow.

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1. Introduction

Habitat restoration can help to alleviate habitat loss or re-establish ecosystem structure and function (Elliott et al. 2007; Reynolds et al. 2013). Often, a primary motivation for habitat restoration is to restore the richness and abundance of fauna associated with the lost habitats (e.g. Muotka et al. 2002; Ruiz-Jaén and Aide, 2005). However, restoration success varies, due to the inherent difficulties involved in restoring complex environments (Elliott et al. 2007; Irving et al. 2010). Further, ecosystems are not stable through time, meaning the "baseline" that should be used for restoration targets is often uncertain. Therefore, setting goals for restoration success based solely on compositional or structural attributes that were characteristic of the system prior to disturbance can be problematic (Hobbs and Harris, 2001).

Restoration success is often most reliably assessed by comparing structural and functional attributes of the restoration site to those of a neighbouring undegraded habitat or reference site (Hobbs and Harris, 2001; Ruiz-Jaén and Aide, 2005; Benayas et al. 2009). A general element of structural restoration is the replenishment of plant species which provide the physical structure of an ecosystem (McCay et al. 2003). Recovery of structure, however, does not necessarily lead to the return of ecosystem function (Zedler and Lindig-Cisneros, 2000). For example, arthropod diversity in restored coastal sage scrub was lower than in undisturbed habitat after 15 years, even though vegetation was structurally similar (Longcore, 2003).

In marine systems, seagrass meadows form ecologically and economically important coastal habitats (Short and Wyllie-Echeverria, 1996; Beck et al. 2001; Duarte, 2002; Orth et al. 2006). Due to their coastal location, seagrass meadows are highly susceptible to disturbance from natural and anthropogenic sources (Short and Wyllie-Echeverria, 1996; Ralph et al. 2006), and approximately 29% of the world's seagrass habitat has been lost (Waycott et al. 2009). As a consequence, seagrass restoration has become an element of coastal management, with early research

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primarily focused on establishing the most effective techniques of transplantation (Van Keulen et al. 2003; Bell et al. 2008; Cunha et al. 2012).

The success of seagrass restoration projects has, however, been limited, with only 30% of studies reporting success (Fonseca et al. 1998), which is thought to be primarily due to poor site selection (Fonseca, 2011). Restoration success can be defined by a lack of detectable differences in structure (e.g. shoot density) between recovering treatments and undisturbed treatments. Studies that do report 'success', generally do so based on short-term monitoring (<1 year), and hence long-term success is often not known (Cunha et al. 2012). Increasing the length of restoration monitoring may increase the ability to identify successful restoration. For example, long-term monitoring of seagrass restoration near Tampa Bay, Florida, showed the recovery of seagrass to be slow during the first 3 years, followed by rapid recovery 4–7 years after restoration was implemented (Bell et al. 2014). Further, the recovery of ecosystem function rather than structure, is only infrequently used to assess restoration success (e.g. Bell et al. 1993; Fonseca et al. 1996; Sheridan et al. 2003). As functional diversity, being the varying functional characteristics of the organisms residing in an ecosystem, is thought to have the greatest influence on ecosystem function (Tilman et al. 1997; Dıaz & Cabido, 2001), it may provide a measurable index of the restoration of ecosystem function. However, in systems where little is known about the functional characteristics of the organisms present, other measures such as species richness may be used as an indicator (Tilman, 2001). For restoration to be successful, restored seagrass patches should persist and recover similar ecosystem function, such as the recovery of fauna due to the provision of habitat, to that of a natural undisturbed seagrass meadow (Fonseca et al. 1998).

Wear et al. (2010) developed a novel seagrass restoration technique, using biodegradable hessian (burlap) bags to stabilize the sediment and facilitate the natural recruitment of *Amphibolis antarctica* seedlings, with the overall intention of re-establishing an extensive continuous seagrass meadow, which was present in the area prior to substantial seagrass loss (>5200 ha) (Neverauskas, 1987; Nayar et al. 2012). This technique has allowed *A. antarctica* seedlings to become established and create new patches (Irving et al. 2013), which have persisted for >5 years (Tanner, 2014). *Amphibolis* is a large perennial structure-forming seagrass that grows in similar environments to *Posidonia* (Shepherd and Womersley, 1981; Bryars and Rowling, 2009). Unlike many other large seagrasses, most of the biomass is above-ground (Paling and McComb, 2000), and it has long, wiry vertical stems that support clusters of small leaves (rather similar to a bottlebrush in appearance), rather than long strap-like leaves. It is also unusual in being viviparous, with seedlings released from the parent plant in winter and drifting until they encounter a suitable attachment point (Cambridge, 1975; Ducker et al. 1977). Attachment is via a comb-like rosette at the base of the seedling that entangles in features such as *Posidonia* root mat (Kirkman, 1999; Rivers et al., 2011), and it is this feature that allows it to attach readily to hessian bags (Wear et al. 2010).

Here, we explicitly seek to estimate the early signs and extent of motile epifaunal recovery relative to seagrass recovery of the series of small-scale experimental seagrass restoration patches described by Tanner (2014). We define motile epifauna as non-sedentary small invertebrates which are directly associated with above-ground seagrass structure. To estimate the early signs and extent of recovery, we compare initially small and expanding patches of restored seagrass to an adjacent continuous natural seagrass meadow. We consider this to be the gold standard for recovery in this situation, as this meadow is well established (hundreds if not thousands of years), large, and not subject to fragmentation, and

therefore best reflects the natural situation. Additionally, we tested whether the time scale of epifaunal recovery in these restoration patches matched the time scale of seagrass recovery. If epifaunal recovery occurs before seagrass recovery, then demonstrating this may assist managers by showing early signs of achievement, thus justifying continued investment in restoration.

2. Materials and methods

2.1. Restoration site and sampling design

Structural recovery and epifaunal use were examined in an experimental seagrass restoration site located just inshore of a large, naturally occurring *A. antarctica* meadow, in approximately 8 m water depth, along the Adelaide metropolitan coast, South Australia (35° 1′ S, 138° 18′ E). The natural *A. antarctica* meadow consists of a dense continuous canopy, with the edge of the meadow being an abrupt change from dense seagrass to bare sand. The current edge of the natural seagrass meadow marks the margin of seaward retreat of inshore seagrass at this site due to eutrophication (Westphalen et al. 2005). In recent years, extensive effort has been invested in improving water quality, allowing a small amount of natural seagrass recovery in deeper waters (Bryars and Neverauskas, 2004), and prompted initial studies on restoration.

Restoration trials at this site began in 2007 by deploying hessian bags to promote the recruitment of *A. antarctica* seedlings, which are released from the adjacent natural meadow. Hessian bags (area 0.35 m² per bag) were deployed approximately bimonthly, from September 2007 to October 2009 and again from January 2011 to March 2013. On each deployment, ten replicate bags, which represent a restoration plot, were filled with ~25 kg of clean play pit sand to anchor them and deployed on sandy substrate, shoreward of and parallel to the natural meadow. Bags were placed end-to-end in a double row by divers ~0.5–1 m apart, making restoration plots rectangular in shape. Each bimonthly deployment was separated by ~2–3 m and there was a minimum distance of 10 m between restoration plots deployed in different years. All bags were deployed within 50 m of the natural meadow, and extended over a distance of ~100 m (Fig. S1). The variation in recruitment of *A. antarctica* seedlings with distance from the natural meadow has previously been tested at this site, and bags located within ~80 m of the natural meadow effectively recruit *A. antarctica* seedlings (Irving et al. 2013). While not formally measured due to the small size of the entire site (~2 km between the edge of the seagrass and shoreline, Wear et al. 2010) there were no obvious environmental gradients present. Importantly, there was no measurable difference in water depth between the offshore and inshore margin of the restoration site (~8 m water depth). In addition, previous measurements showed that seafloor light intensities at this site averaged 15–18% of surface irradiance (86.83 $\pm$ 22.71 μmol m² s^{-1}) (Irving et al. 2010).

We used a space-for-time substitution approach (also known as a chronosequence) to establish the time scale for the recovery of the restoration site. Space-for-time substitution (SFT) has long been used in ecology, particularly as a standard method for looking at successional theory, where time-scales are generally sufficiently long that standard replicated experimental designs are not feasible (Pickett, 1989). This technique has allowed us to assess the time scale and extent of epifaunal recovery by taking a series of samples from restoration plots of known ages, representing a "single snapshot" of succession, instead of sampling the one site multiple times. *A. antarctica* samples with associated epifauna were collected from three restoration plots of known ages (based on year and month of bag deployment), 1 year (July 2011 deployment), 3 years (February 2009 deployment) and 5 years (September 2007