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Eco-engineering of modified shorelines recovers wrack subsidies

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ABSTRACT

Wrack (stranded phyto-detritus) from terrestrial and marine sources is an important source of carbon and nutrients for many intertidal habitats. In urbanised estuaries, seawalls may act as a barrier to the transport of wrack between terrestrial and marine habitats and, where they reduce the width and habitat complexity of the intertidal zone, negatively impact on wrack accumulation and retention on estuarine shorelines. We assessed differences in the accumulation of wrack between natural (sandy beach, rocky reef and rockpools) and armoured stretches of shoreline, and the efficacy of eco-engineering interventions (man-made rockpools and planted saltmarshes) in enhancing the accumulation and retention of wrack. Surveys conducted at 4 sites in Kogarah Bay, New South Wales, Australia, on 3 dates during autumn-winter 2016 revealed that on natural shorelines most wrack accumulated at the high intertidal mark. Placement of seawalls restricted wrack accumulation to the low intertidal zone, in front of the seawall. The cover of wrack was significantly less in the low intertidal zone in front of seawalls than either the high or low tidal elevations of the natural sandy beaches and rockpools. The man-made rockpools in the low tidal zone had higher cover and biomass of marine and terrestrial wrack than the low intertidal zone of the seawalls, and both these and the planted saltmarshes also supported the accumulation of wrack at the mid intertidal elevation. Experimental deployments of marked seagrass wrack revealed that the eco-engineered habitats enhanced wrack retention over a 7 day period compared with the seawalls. However, the artificial rockpools also retaining more wrack than the equivalent natural habitat. Our results suggest that the structural complexity of eco-engineered habitats can be effective at trapping and retaining wrack, with potential flow-on effects to benthic assemblages through increased food and shelter resources. When designing eco-engineering interventions for modified shorelines, scientists and managers should consider not only their impacts on biodiversity but also on key ecosystem functions.

1. Introduction

Many intertidal communities, including those of tidal flats, mangroves, saltmarshes, sandy beaches and rocky shores, are dependent on accumulations of wrack (stranded phyto-detritus), (Dugan et al., 2003; Orr et al., 2005; Olabarria et al., 2010). Wrack provides food and structural habitat for invertebrates which, in turn, fuel predatory fish and shorebirds, and is an important component of carbon and nutrient cycles (Thompson et al., 2002; Spiller et al., 2010; Dugan et al., 2011b). Wrack can be transported across habitat boundaries by wind and water and may contain detritus from both terrestrial and marine primary producers (Polis et al., 1997; Bartels et al., 2012). The accumulation of wrack on shorelines is dependent on rates of detrital production by donor habitats, transport processes that deliver wrack, as well as structural (e.g. vegetation, cobbles) and physical (e.g. wave energy, tidal regime, shoreline slope) features of the recipient habitat that affect wrack retention (Orr et al., 2005). Hence, human activities that modify the identity and productivity of primary producers, that change coastal circulation and drainage patterns, and/or that alter structural features of shorelines have the potential to have large impacts on wrack pathways and the food webs they support (Kirkman and Kendrick, 1997; Defeo et al., 2009; Dugan et al., 2011a).

Shoreline hardening or coastal armouring (i.e. the construction of artificial structures for coastal defence) is increasing in response to burgeoning human population growth along the coast (Gittman et al., 2015; Firth et al., 2016). It is estimated that in some coastal areas > 50% of the natural nearshore habitats have been replaced by coastal defence structures that protect reclaimed land, coastal property and infrastructure from erosion and inundation (Lam et al., 2009; Dafforn et al., 2015; Gittman et al., 2015; Firth et al., 2016). These artificial

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structures are often placed in transition zones, such as the intertidal, serving as physical barriers to the movement of organisms, energy and materials between terrestrial and aquatic environments (Bishop et al., 2017). Coastal defence structures typically have less habitat complexity as compared to the natural habitats they replace (Bulleri and Chapman, 2010). When placed parallel to the shoreline below the high water mark, their predominately vertical orientation can reduce the width of the intertidal zone (Sobocinski et al., 2010; Heatherington and Bishop 2012; Harris et al., 2014; Heerhartz et al., 2014) and may also impact adjacent marine habitats by increasing current strength and wave reflection (Dugan et al., 2011a). The net effect can be reduced intertidal accumulations of wrack on armoured shorelines (Sobocinski et al., 2010; Heatherington and Bishop, 2012; Harris et al., 2014; Heerhartz et al., 2014) which in turn support reduced abundances and altered assemblages of invertebrates (Dethier et al., 2016), with cascading impacts through higher trophic levels (Sobocinski et al., 2010; Heatherington and Bishop, 2012; Harris et al., 2014; Heerhartz et al., 2014).

Marine eco-engineering seeks to mitigate the impacts of artificial structures to biodiversity (Chapman and Underwood, 2011; Firth et al., 2016; Strain et al., 2017), ecological connectivity (Airoldi et al., 2005; Bishop et al., 2017) and ecosystem functioning (Morris et al., 2017). At smaller scales of millimetres to meters, eco-engineering may include modification of the design of new structures (Chapman and Underwood, 2011; Perkol-Finkel et al., 2017), or increasing the roughness, texture and/or microhabitats of existing structures, through additive (i.e. retrofit with artificial or natural habitats) or subtractive (i.e. grout removal, drilling) processes (Strain et al., 2017). Larger-scale interventions (meters to 100s of meters) typically involve partially or wholly removing the need for artificial structures by planting coastal habitats, including mangroves, saltmarshes, wetlands (Currin et al., 2008; Currin et al., 2010; Gedan et al., 2011) or biogenic reefs (Piazza et al., 2005; Scyphers et al., 2011) that stabilize sediments and/or dampen wave energy. Studies have addressed the efficacy of these interventions in enhancing biodiversity (Chapman et al., 2017; Strain et al., 2017; Toft et al., 2017), but their effects on ecological functions such as connectivity between land and water and the resulting accumulation of wrack remains untested. Where these interventions add structural elements to the intertidal zone, or restore its width, they may enhance retention and accumulation of wrack on intertidal shorelines relative to traditional coastal armouring.

Here we examine the role of 2 eco-engineered intertidal habitats in enhancing the cover and biomass of wrack in a heavily urbanised estuary in southeast Australia. In this region, much of the natural intertidal shoreline has been replaced by seawalls. However at 3 sites the seawalls have been eco-engineered to include man-made rockpools in the low to mid intertidal zone and/or planted saltmarshes in the mid tidal zone (Heath and Moody, 2013). We tested the following hypotheses: 1) on armoured shorelines, wrack accumulations will be confined to the low intertidal zone; 2) armoured shorelines will support less wrack accumulations than the natural shorelines (rockpools, rocky shores, sandy beaches); shorelines that have been eco-engineered will 3) have greater accumulations and retention of wrack than armoured shorelines, and 4) similar accumulations and retention of wrack to analagous natural habitats. This study was conducted across 4 sites with 2 examples of each eco-engineered habitat to assess their importance for enhancing the cover and biomass of wrack in locations that are exposed to different levels of wind and waves.

2. Methods

2.1. Study site

The study was done in Kogarah Bay (Fig. 1), a heavily modified embayment of the Georges River estuary, New South Wales, Australia, 15 km south of Sydney city. The Bay is approximately 2300 m in length, 700 m in width and 3 m in depth (Alyazichi et al., 2015), with adjacent land-use a mixture of residential and urban parkland. Kogarah Bay is characterised by semi-diurnal tides, with a spring tidal range of 2 m.

Sampling and wrack deployments were done at 4 sites, across which there were 2 replicate sites of each of the following habitats: natural rockpools, natural sandy beaches, natural rocky shores, each spanning low to high intertidal elevations; eco-engineered habitat comprised of man-made rockpools in the lower and mid intertidal zone; and eco-engineered habitat comprised of saltmarsh planted in the mid intertidal zone (Fig. 2). At each of the 4 sites we also sampled stretches of sedimentary shoreline armoured by vertical seawalls. In Kogarah Bay there are no remanent natural saltmarshes, so it was not possible to sample this habitat.

At sites with man-made rockpools, there were 2 in the low intertidal and 2 in the mid-intertidal zone. These habitats were constructed of sandstone blocks, resulting in artificial rockpools that were much larger (average: 0.60 m depth and 20.20 m^2 area) than the natural rockpools (average: 0.20 m depth and 0.09 m^2 area) in the bay. The sites with planted saltmarsh had 2 mid intertidal patches (average: 60.02 m^2 area), containing a mixture of *Sarcocornia quinqueflora* and other native grasses *Suaeda australis* and *Sporobolus virginicus*. Seawalls were comprised of sandstone blocks that had been constructed at a tidal elevation of approximately 0.20 and extended vertically to 2.82 m above mean sea level, and were located within 200 m of either the natural or eco-engineered habitats (Figs. 1 and 2). In contrast, the natural rocky shores were characterised by predominantly horizontal sandstone substrate, with the rock pools interspersed. The sandy beaches were horizontal and contained a mixture of fine to medium size sand grains. Sampling and experiments were stratified by 3 intertidal elevations: low (0.35–0.45 m above mean sea level), mid (0.5–0.8 m above mean sea level) and high (0.95–2 m, above mean sea level).

2.2. Differences in wrack cover, composition and biomass among habitats

We conducted a survey of wrack accumulations across the intertidal zone of each of the habitats to test the hypotheses about differences in wrack accumulations between natural, armoured (seawall) and eco-engineered sites.

The survey was repeated on 3 dates during late autumn to early winter of 2016, each of which was 1–3 days after a 'storm event' (rainfall > 22 mm and primary wind direction S-WSW) which mobilises wrack from terrestrial and marine sources and deposits it on the shore (Heath pers. obs.). On each date, and for each habitat type, zone and site, we surveyed wrack in ten randomly placed 0.5×0.5 m quadrats. The cover of wrack was quantified within each quadrat by taking a digital photograph using an Olympus Tough Style camera, setting the measurement scale, tracing around the wrack, and converting this to percentage cover of wrack using image j (https://imagej.nih.gov/ij/). All wrack within each quadrat in the lower tidal zone of the seawalls, and the lower and mid tidal zone of the man-made rockpools, planted saltmarshes and natural rockpools was then collected by hand, bagged and returned to the laboratory for biomass assessment. We did not collect the wrack in the natural sandy beaches and rocky shores because the primary focus of the study was to compare eco-engineering habitats to seawalls or analogous natural habitats. Wrack samples were frozen at -80°C until they were processed (within 3 months of collection). Following thawing, each wrack sample was separately washed over a 1 mm sieve to remove all dirt, sorted into key marine (algae, seagrass) and terrestrial (leaves, bark) wrack components, which were then separately dried to constant weight (14 days) and weighed (to nearest 0.001 g). Throughout the sampling only small amounts of man-made materials were found in each quadrat, so we did not attempt to quantify the cover or biomass of rubbish in each of the different habitats.

At each location, the average water flow at the base of each seawall was estimated, using 3 clod cards (height 2 cm, width 1 cm, length 1 cm). The clod cards were made of plaster of paris (500 g of Dingo),