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A cost-effective approach to enhance scleractinian diversity on artificial shorelines

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ABSTRACT

Seawalls, which have replaced many natural shorelines in coastal cities, are increasingly built to alleviate the impacts of rising sea levels. To mitigate the consequential loss of biodiversity, novel approaches such as ecological engineering have been adopted to enhance the biodiversity on these artificial structures. However, the majority of research to date has focused on physical modifications of intertidal seawalls, and such habitat enhancement efforts are labour- and cost-intensive. We examined the feasibility of transplanting nursery-reared scleractinian corals on subtidal seawalls in Singapore with the help of volunteers. Fragments of six hard coral species (*Pocillopora damicornis, Hydnophora rigida, Merulina ampliata, Podabacia crustacea, Echinopora lamellosa, Platygyra sinensis*) were tested. Fragments of all species fared well in the nurseries with a mean survival rate of 98.5% and 1.1 to 4-fold increase in live tissue area. Six months after transplantation to a seawall, only 50% of *P. damicornis* transplants survived, while those of *M. ampliata* decreased in size. Transplants of the other four species exhibited sustained growth and high survival rates (>90%), suggesting that they were more suitable than the former two species as candidates for transplantation onto subtidal seawalls. Scleractinian cover at the transplant site increased from 3% to 20% and generic richness increased from two to eight. The estimated project costs were almost US$ 23,000 if only researchers were involved in the effort, but the inclusion of volunteers in fieldwork and data analyses could help to bring the expenditure down by up to 23%. The study demonstrated the feasibility of transplanting corals onto subtidal seawalls to mitigate the impacts of coastal development, and highlighted its potential for application on other artificial structures. The findings also show that synergy between the community and scientists helps to reduce overall costs and is beneficial for biodiversity enhancement initiatives.

1. Introduction

Natural shorelines and coastal habitats have been extensively modified (Bulleri and Chapman, 2010) to meet the demands of more than 275 million coastal inhabitants (Airoldi et al., 2005; Burke et al., 2011). Artificial structures such as seawalls and breakwaters have replaced many natural shorelines for the purposes of wave attenuation and erosion reduction (Airoldi et al., 2005). Seawalls are increasingly built to mitigate the impacts of sea level rise and frequent flooding that are predicted to intensify with global climate change (Airoldi et al., 2005; Hallegatte, 2009; Lai et al., 2015). However, the construction of these structures replaces existing natural habitats, and the consequential loss of biodiversity and ecological function due to coastal development is a grave concern (Chapman, 2003; Bulleri and Chapman, 2010; Browne and Chapman, 2011; Firth et al., 2014). The physical attributes of seawalls, such as inclination, surface area and water retention rates, are significantly different from natural rocky shores and limit the level of biodiversity that can be supported (Chapman, 2003). Furthermore, these coastal infrastructures are not built to support the recruitment and establishment of biological communities. Due to the lack of microhabitats on seawalls relative to the natural habitats such as rocky

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shores, mobile species such as gastropods and echinoderms are among the most impacted (Chapman, 2003). Nevertheless, despite their lack of a complex structure subtidal artificial substrates can also unintentionally enhance recruitment and density of rare coral species (Gilbert et al., 2015) or more common ones (Ho et al., 2016).

In recent years, ecological engineering has been suggested as a way to reconcile the loss of marine biodiversity (Rosenzweig, 2003). This involves integrating ecological principles into the construction or modification of coastal defence structures to enhance ecosystem functions (Rosenzweig, 2003; Loke et al., 2015). One approach, which incorporates habitat complexity into the design of the structures (Airoldi et al., 2005), aims to increase species diversity without compromising the integrity of seawalls (Browne and Chapman, 2011; Chapman and Underwood, 2011; Loke et al., 2016). For instance, Browne and Chapman (2011) demonstrated that the attachment of artificial structures, which mimicked intertidal rock-pools, helped increase the number of species inhabiting the seawalls by 110%. Similarly, Loke and Todd (2016) retrofitted concrete substrates of various geometric designs onto intertidal seawalls and reported enhanced species richness and community composition. The increase in physical complexity of these coastal defence structures generates additional microhabitats, which in turn enhances marine biodiversity (Loke et al., 2016).

Where opportunities for construction or modification of the infrastructure are limited, transplantation of marine biota can be a viable way to increase the ecological value of seawalls (Kinoshita, 2009; Ng et al., 2015). Scleractinian corals are prime candidates for transplantation, as they form the main framework of coral reefs, which support 25% of the world's marine species (Roberts et al., 2002; Burke et al., 2011) and confer shoreline protection by reducing wave energy and erosion rates (Moberg and Folke, 1999; Kunkel et al., 2006). Coral transplantation is a strategy that has been widely employed to assist the recovery of degraded reefs (Toh et al., 2012, 2014; Rinkevich, 2014; Gomez et al., 2014) or to relocate corals away from reefs that will become lost or degraded due to dredging, pollution and construction works (Plucer-Rosario and Randall, 1987; Kilbane et al., 2008; Seguin et al., 2008; Kenny et al., 2012). Since seawalls are large and stable substrates that support natural coral recruitment and growth (Chou et al., 2010; Ng et al., 2012; Tan et al., 2012), the transplantation of corals on these structures may be considered as an option to assist scleractinian colonization (Ng et al., 2015). The increased habitat complexity generated from the established transplants would subsequently help attract and support more marine organisms such as fish and molluscs (Cabaitan et al., 2008; Villanueva et al., 2012; Ng et al., 2016).

To date, most biodiversity enhancement efforts have focused on the intertidal zone of seawalls (Browne and Chapman, 2011; Ng et al., 2015; Loke et al., 2016). Less research has been carried out for subtidal seawalls, with only one study on the transplantation of macroalgae on submerged breakwaters reported to date (Kinoshita, 2009). The use of coral transplantation to enhance biodiversity on seawalls is rare, and none of the reported studies has examined the feasibility of assisted coral colonisation on subtidal seawalls. While there are growing efforts to assist the recovery of degraded reefs worldwide (Rinkevich, 2014; Ng et al., 2016), it is unknown if corals transplanted on seawalls would respond similarly to those transplanted on reefs, as the former are man-made habitats which support different biotic assemblages from the latter (Kinoshita, 2009; Ng et al., 2015). Seawalls are also exposed to a wider range of environmental fluctuations (*e.g.* thermally and hydrodynamically) that are potentially more stressful for the transplants (Kamphuis et al., 1992; Ng et al., 2015). Although some generalizations on these responses can be derived from the life history traits of individual coral species, reliable estimates are difficult since coral transplants can vary in biological responses, even between congenerics (Edwards, 2010; Ng et al., 2016). Extrapolations of transplant responses from published literature could thus be inaccurate (Browne and Chapman, 2011; Madin et al., 2016). The slow uptake of this approach is hindered by the paucity of empirical data (*e.g.* transplant survival rates and growth; Ng et al., 2016; Madin et al., 2016), therefore studies need to be systematically carried out to adequately assess the suitability of different coral species for transplantation onto these novel habitats.

Apart from the variability in biological responses elicited among species, the implementation and success of coral transplantation initiatives are also limited by the monetary costs involved. Recent studies indicate that coral transplantation incurs high labour and transportation costs (Toh et al., 2014), and that restoring coral reefs in developed economies could be 30 times more expensive than in developing countries (Bayraktarov et al., 2016). There is a clear need to improve the economic viability of transplantation efforts. A citizen science approach involving trained volunteers could help to lower expenditure (dela Cruz et al., 2014), improve the cost-effectiveness of transplantation projects (Spurgeon, 1999), and promote awareness of environmental issues (Marshall et al., 2012; Parkinson et al., 2016). However, detailed financial reports are scarce in the literature, and obtaining reliable cost estimates for coral transplantation is challenging as few studies have quantified the extent to which this approach could reduce the cost of transplantation (Bayaraktarov et al., 2016).

In this study, fragments of six coral species were propagated in an *in situ* coral nursery and thereafter transplanted to the subtidal zone of a seawall in Singapore, a tropical developed economy with an urbanised marine environment (Chou, 2008; Tan et al., 2016). Volunteers were actively involved in fieldwork and data analyses. We examined the biological responses (*i.e.* growth and survival) of the corals to determine their suitability for nursery-rearing and subsequent transplantation onto seawalls. In addition, we calculated the cost estimates for the project to approximate the savings that could be achieved with community involvement.

2. Methods

2.1. Coral collection, propagation and nursery phase

Forty-two coral colonies (each approximately 60 cm in diameter) were collected from Sultan Shoal, Singapore (1°14.21′N, 103°38.52′E) from August to December 2014. This comprised seven colonies from each of the six species commonly found at this site: *Podabacia crustacea, Merulina ampliata, Echinopora lamellosa, Hydnophora rigida, Pocillopora damicornis* and *Platygyra sinensis.* Each colony was broken up into fragments (7–10 cm diameter). The initial size of all the fragments did not differ significantly among species ($F_{5,18} = 2.098$, $p > 0.05$).

Thirty-six fragments of each species were randomly secured on four *in situ* nursery tables (0.5 m × 0.5 m) located at Lazarus Island, Singapore (1°13'41.76"N, 103°51'19.82"E). These tables were elevated 0.5 m above the sand-silt bottom (4 m deep). The corals were tagged and reared in the nursery for up to nine months before transplantation. The nurseries were maintained regularly to reduce the impact of fouling organisms on the corals (Toh et al., 2013a).

Photographs were taken directly above each coral fragment with a calibrated scale to monitor growth and survivorship, and then analysed using the software ImageJ (NIH). The following parameters were measured: live coral tissue area, length (L) and width (W). The areal growth rate of each fragment was calculated as the difference between the final and the initial live tissue area divided by the number of months spent in the nursery. Geometric mean diameter for each fragment (GMD) was calculated using the formula GMD = $\sqrt{(L \times W)}$. Linear growth rates were calculated as the