Pre-proof verion of Hsiung, A.R., Tan, W.T., Loke, L.H., Firth, L.B., Heery, E.C., Ducker, J., Clark, V., Pek, Y.S., Birch, W.R., Ang, A.C. and Hartanto, R.S., 2020. Little evidence that lowering the pH of concrete supports greater biodiversity on tropical and temperate seawalls. Marine Ecology Progress Series, 656, 193-205.

Little evidence that lowering the pH of concrete supports greater biodiversity on tropical and temperate seawalls

Amanda R. Hsiung[1¶], Wen Ting Tan[1¶], Lynette H.L. Loke[1*], Louise B. Firth[2], Eliza C. Heery[1], James Ducker[2], Victoria Clark[2], Y. Shona Pek[3], William R. Birch[3], Ambert C.F. Ang[1], Rania S. Hartanto[1], Tiffany M.F. Chai[2], Peter A. Todd[1]

[1] Experimental Marine Ecology Laboratory, Department of Biological Sciences, National University of Singapore, 14 Science Drive 4, Block S3, Singapore 117557
[2] School of Biological and Marine Sciences, University of Plymouth, Plymouth PL4 8AA, United Kingdom
[3] Institute of Materials Research & Engineering, 2 Fusionopolis Way, Innovis, Singapore 138634

¶*These authors contributed equally to this work*
*Corresponding author, lynetteloke@gmail.com

Concrete is one of the most commonly used materials in the construction of coastal and marine infrastructure despite well-known environmental impacts, including a high carbon footprint and high alkalinity (~pH 13). There is an ongoing discussion regarding the potential positive effects of lowered concrete pH on benthic biodiversity, but this has not been investigated rigorously. Here, we designed a manipulative field experiment to test whether carbonated (lowered pH) concrete substrates support greater species richness and abundance, and/or alter community composition, in both temperate and tropical intertidal habitats. We constructed 192 experimental concrete tiles, half of which were carbonated to a lower surface pH of 7–8 (vs control pH of >9), and affixed them to seawalls in the United Kingdom and Singapore. There were two sites per country and six replicate tiles of each treatment were collected at four time-points over a year. Overall, we found no significant effect of lowered pH on the abundance, richness, or community assemblage in both countries. Separate site- and month-specific generalized linear models (GLMs) showed only sporadic effects: i.e, lowered pH tiles had a small positive effect on early benthic colonisation in the tropics but this was later succeeded by similar species assemblages regardless of treatment. Thus, while it is worth considering the modification of concrete from an environmental/emissions standpoint, lowered pH may not be a factor for enhancing biodiversity in the marine built environment.

Key words: pH, eco-engineering, biodiversity, concrete

1. INTRODUCTION

Coastal marine ecosystems have experienced dramatic changes during the last century, often driven by urbanisation and exemplified by the proliferation of man-made structures such as seawalls, breakwaters, and groynes (Heery et al. 2017, Todd et al. 2019). In major coastal cities, including Sydney, Hong Kong, and Singapore, these artificial structures can comprise over 50% of shorelines (Chapman & Bulleri 2003, Lam et al. 2009, Lai et al. 2015). Designed to prevent erosion and provide flood protection (Chapman 2003, Todd et al. 2019), sea defences are likely to become more prevalent with growing coastal populations, rising sea levels and increasing storm frequencies (Nicholls et al. 2007, Temmerman et al. 2013). Concomitantly, there has been growing research interest in the ecological functioning of these man-made structures (Bulleri & Chapman 2010, Dafforn et al. 2015, Firth et al. 2016b). However, compared to natural rocky shores, artificial structures tend to support lower species diversity and/or abundances (e.g., Moschella et al. 2005, Lai et al. 2018), different ecological communities (e.g., Chapman & Bulleri 2003, Lam et al. 2009), and higher numbers of non-native species and/or homogenised species assemblages (e.g., Bulleri & Airoldi 2005, Glasby et al. 2007).

Concrete, a composite material comprising Portland cement, water, and a mixture of coarse and fine aggregates, is one of the most commonly used building materials in coastal and marine infrastructure (Dugan et al. 2011). While the physical characteristics of concrete (e.g. durability, strength, and workability) have made it a ubiquitous component of the modern built environment (Dyer 2014), the production process of concrete has a high carbon footprint (Waters & Zalasiewicz 2018). It has also been suggested that concrete has a negative effect on the recruitment of marine biota due to its high surface alkalinity (pH ~13) (Lukens & Selberg 2004, Perkol-Finkel & Sella 2014), reducing initial rates of species colonization (Nandakumar et al. 2003) and favouring alkotolerant taxa such as barnacles and serpulids over algae (Hatcher 1998, Dooley et al. 1999). This high surface alkalinity potentially compounds the known negative effects of hard coastal defences such as the loss of habitat area (Lai et al 2015), compression of the intertidal zone due to its steep gradient (Firth et al. 2014, Loke et al. 2019a), low structural complexity (Chapman & Bulleri 2003, Moreira et al. 2007), and higher desiccation (Tan et al. 2018, Zhao et al. 2019) and temperature risk (Aguilera et al. 2019). With such changes in material and physical structure, seawalls have been considered sub-optimal intertidal habitats and there is a general consensus that the expansion of hard coastal defences at a global scale presents a huge threat to coastal and marine biodiversity (Bishop et al. 2017, Heery et al. 2017).

In response to these threats, ecological engineering—the integration between engineering principles and maximised ecological value—has been increasingly adopted in the marine environment (Strain et al. 2018, Chapman et al. 2018). The aim is to alleviate the negative impacts associated with artificial structures and to increase their ecological functioning (Morris et al. 2019). In particular, "hard" engineering, the physical modification of existing seawalls or use of habitat enhancement units (Chapman & Underwood 2011), has been experimented in several countries, both temperate and tropical (Dafforn et al. 2015, Firth et al. 2016a, Loke et al. 2019b). However, ecological engineering techniques applied to seawalls have generally targeted the physical (topographical) differences between natural rocky shores and artificial structures. Therefore, habitat enhancement units tend to focus on manipulating the surface complexity of substrates to incorporate water-retaining features and/or increase structural complexity, via the creation of cavities and the retrofitting of tiles with varying surface topography (Firth et al. 2013, 2014, Loke et al. 2017, Strain et al. 2018). Nevertheless, even with ecological engineering efforts, concrete is often used, as it fulfils industry building and construction safety standards and is easily moulded into various shapes and designs (Waltham & Dafforn 2018).

Some studies have suggested that the material of habitat enhancement units should also be manipulated to increase their ecological benefits (Dennis et al. 2018). Partial replacement of cement or coarse aggregates with more environmentally-friendly materials such as granulated blast-furnace slag and pulverised fly ash has been shown to improve the live cover of benthic organisms on concrete substrates (Dennis et al. 2018, McManus et al. 2018). Altering the concrete matrices also resulted in higher live cover and primary productivity of pre-fabricated habitat units (Perkol-Finkel & Sella 2014, Sella & Perkol-Finkel 2015). On top of increasing species diversity, using natural materials in concrete can reduce its environmental footprint (Dennis et al. 2018). Many of these studies postulated that the reduced pH from these modifications may be beneficial for biotic recruitment (Perkol-Finkel & Sella 2014, Sella & Perkol-Finkel 2015, McManus et al. 2018). pH can influence the colonisation of algae and barnacles at early stages (Guilbeau et al. 2003), which can, in turn, result in different succession patterns (Almeida & Vasconcelos 2015). With contrasting effects of pH on different taxa (Guilbeau et al. 2003), sites with different benthic community assemblages could also be influenced to varying degrees.

One straightforward technique for lowering concrete pH for experimental work is through concrete carbonation. Carbonating concrete ex-situ, also known as accelerated carbonation, has traditionally been used to simulate the carbonation process that occurs naturally when concrete is exposed to air (de Ceukelaire & van Nieuwenburg 1993, Neves et al. 2013). This is often performed to test for the effects of long-term carbonation on concrete's metal leaching abilities (Sazzad bin Shafique et al. 1998), compressive strength (de Ceukelaire & van Nieuwenburg 1993, Chi et al. 2002), and durability (Roy et al. 1999) as carbonation can alter the physical properties of concrete by densifying the concrete surface (Chi et al. 2002, Fernandez Bertoz et al. 2004). However, to our knowledge, no previous studies have tested the effects of this approach on benthic diversity and composition.

Whether changes in concrete pH alone (i.e. while keeping structure texture and composition constant) affects the overall species recruitment on habitat enhancement units is unknown. To determine this, we fabricated topographically-complex concrete tiles and carbonated half of them to obtain lower surface alkalinity, from here on referred to as "carbonated tiles". To test for generality, the experiment was conducted in a temperate country (United Kingdom) and a tropical country (Singapore). Specifically, we tested the following hypotheses: (1) carbonated tiles would support higher macrofaunal abundance and species richness than standard non-carbonated tiles, and (2) carbonated tiles would support different biological communities from standard non-carbonated tiles, and these differences would be consistent across time and sites with different community assemblages.

2. MATERIALS & METHODS

2.1. Tile design and fabrication

A total of 192 experimental tiles were constructed for this study using a single tile design. The face of each tile measured 14 cm × 10 cm (Fig. 1) and had a smooth and pitted façade (on left and right hand side, respectively). The smooth surface was designed for photographic analysis of epibenthic percentage cover while the pitted side was designed to create water-retaining features that would act as refugia for colonising macrofauna (Loke and Todd, 2016); this was achieved using the software *CASU* (Loke et al. 2014). After measuring the angle of seawalls at the chosen study sites, we then adapted all tiles so that the resultant slope of the front facing façade after installation was standardised at 60° (Fig. 1C–D).

Masters of the tiles were created following Loke and Todd's (2016) protocol, using silicone rubber moulds (Freeman Bluesil™ V-340). Tiles were then cast from the moulds using cement/aggregate ratio = 1/3 and water/cement ratio = 3/5. Pre-drilled holes were set in the centre of the concrete tiles for installation on seawalls.

2.2. Tile carbonation

Carbonation is often performed by diffusing high concentrations of carbon dioxide into a sealed chamber containing the concrete (Sanjuán et al. 2003, Chang & Chen 2006). Carbon dioxide reacts with calcium hydroxide and calcium–silicate–hydrate in concrete to form calcium carbonate and water, reducing the alkaline content in the tiles and lowering its pH (Fernández Bertos et al. 2004). In this experiment, a CO_2 chamber was created using a large cooler box and dry ice (Short et al. 2001, Venhuis & Reardon 2003).

Trials were conducted using concrete coupons (5 cm × 5 cm × 2 cm) to determine the best carbonation conditions (wet or dry), and the duration of curing (2, 6, 12, 20 days) and carbonation (7, 22, 29 days) required to reduce the pH of the tiles. Concrete coupons were split in half using a tile saw and the surface and cross section of the split tiles were stained with two pH indicator dyes: (1) Phenolphthalein and (2) Bromothymol blue to test the effectiveness of carbonation. Phenolphthalein, a pH indicator which transitions from colourless to light pink around pH 8, becoming a dark pink when pH value exceeds 9, is typically used to assess the extent of carbonation in concrete (Fig. 2B; Chang & Chen 2006, Thiery et al. 2007). Bromothymol blue, which is less commonly used to test concrete pH, transitions from yellow to light blue from pH 6 to 7, becoming dark blue for pH values above 8 (Guilbeau et al. 2003). When the stained carbonated tiles were colourless (phenolphthalein) and light blue (bromothymol blue), it indicated that the external front-facing surface of the carbonated tiles had a pH estimated to be between 7 and 8 (Fig. 2A).

After several trials were conducted, it was found that the tiles were more rapidly carbonated when dry as opposed to wet, and when they were left to cure for longer before being exposed to CO_2. Carbonation duration (>28 days), however, was the most important variable to achieve a pH of less than 8 (Fig. 2A). A sub-sample of the final batch of tiles were assessed using the indicator dyes, which showed that the surface of the carbonated concrete tiles was no more than than pH 8.

Attempts were also made to quantify the pH of the concrete tiles using a pH meter, but there has been a longstanding lack of a standardised protocol for measuring the pH of pore fluid in concrete (Alonso et al. 2012). Additionally, while the method is often used to test for internal concrete pH, it does not give an accurate measurement of surface pH. Therefore, this method was only used to confirm the differences in internal pH between treatments at the 6-month time point (Fig. S1, Table S1). All tiles were prepared in Singapore before half were sent to the UK.

2.3. Study sites

Tiles were deployed in two locations, one temperate and one tropical climate, with two seawall sites at each location. Plymouth (United Kingdom) was chosen as the temperate location and Singapore was chosen as the tropical location.

2.3.1. Plymouth, United Kingdom

Plymouth is a port city located on the south-west coast of England, United Kingdom, where the English Channel broadens into the Atlantic Ocean. 33% of the coastline within Plymouth Sound is artificial (mostly seawalls) (Knights et al. 2016). In Plymouth, the tiles were installed in February 2018 onto two vertical seawalls at: (i) Turnchapel (50.359, 4.1178) and (ii) Cremyll (50.3648, 4.1633).

2.3.2. Singapore

Singapore is a tropical city-state located just over one degree north of the equator, separated from Peninsular Malaysia by the Straits of Johor in the north and from Indonesia by the Straits of Singapore in the south. Over 63% of Singapore's coastline is made up of seawalls (Lai et al. 2015). In Singapore, tiles were carbonated from January to February 2018 and were installed in late February and early March 2018 at two southern islands: (i) grouted granite rip-rap seawall at Pulau Hantu (1.22611, 103.75222) and (ii) vertical seawall at Pulau Seringat (1.23, 103.85056).

2.4. Field experimental design, sampling and laboratory procedures

At each site, 24 of each tile treatment (carbonated and non-carbonated) were installed along seawalls at mid-shore height, approximately 1.5 m above chart datum, and spaced at least 0.5 m apart. Six replicates of carbonated and non-carbonated tiles were removed randomly at 3, 6, 9 and 12 months. However, due to unforeseen temporary restricted access to Pulau Hantu, collection for the 9-month time point could not be carried out, hence we included a 15-month time point instead for that site.

Prior to removal of the tiles, fast-moving organisms were picked and placed into self-sealing plastic bags. The tiles were then photographed (for subsequent algal cover analysis) before being removed from the seawall and placed into larger self-sealing plastic bags. Algal cover was quantified using CPCe image analysis software (Kohler & Gill 2006), with percentage cover tabulated from 40 random point intercepts on the smooth surface of the tile. Four common functional groups were used to categorise the algae composition in both countries following Loke et al. (2016) (Table 1).

After algal removal from the smooth surface, the tiles were placed into the freezer (–20 °C) for subsequent sorting, counting and identification using a dissecting microscope. All specimens were identified to species or morphospecies level except for polychaetes, which were identified to family level (Loke & Todd 2016, Loke et al. 2017, 2019a).

2.5. Statistical analysis

As tiles were lost due to wave action, there was an unequal number of replicates for some sites and treatments (Table S2), but there were at least four replicates per treatment per site per time point. Data were first examined for the presence of outliers, heterogeneity, non-normality and overdispersion (Zuur et al. 2010). We then tested for differences in total abundance and species richness using generalised linear models (GLMs). Models with Poisson error were first constructed separately for the two countries with treatment, site, and month (categorical) as fixed effects, but models with negative binomial error were subsequently used to analyse abundance due to over-dispersed data.

With differences in sample numbers between sites at some time points (described above) and significant differences in abundance and species richness between months and sites, we removed interaction terms (Table S3) and evaluated whether treatment effects differed by subsequently modelling the abundance and richness data separately for each site and month with treatment as the sole predictor. Site- and month-specific models of richness tended to be under-dispersed, and were therefore fit with Conway-Maxwell-Poisson (COM-Poisson) regressions (Sellers & Shmueli 2010). Negative binomial error structure was maintained for site- and month-specific models of abundance. Univariate tests were performed in R v3.6.0 (R Core Team 2019). COM-Poisson models were constructed and evaluated using the 'COMPoissonReg' package (Sellers et al. 2017) while negative binomial regression was performed using the 'glm.nb' function in the 'MASS' package (Venables & Ripley 2002).

We used permutational distance-based multivariate analysis of variance (PERMANOVA; Anderson 2001) to test for differences in community composition between treatments (we removed 15th month data as they were un-replicated in time; please see the Methods section for more information). As both countries hosted no overlapping species, analyses were conducted separately for temperate and tropical systems. The abundances were log(X+1)-transformed and the full resemblance matrix was calculated on Bray-Curtis similarities and *p* values were generated using 9999 unrestricted random permutations of residuals. PERMANOVA revealed significant differences in community composition among months, but did not reveal significant differences among treatments; canonical analysis of principal coordinates (CAP) plots were then used to examine these temporal differences. All multivariate analyses were performed using the PRIMER v7 with the PERMANOVA add-on (Anderson et al. 2008).

3. RESULTS

3.1. Abundance and species richness

A total of 78,114 individuals of 68 species/morphospecies were collected and identified from experimental tiles across both countries. Of these, 13 were temperate species from Plymouth, and 55 were tropical species from Singapore. Although there were more unique species found on carbonated tiles than non-carbonated tiles at both sites in Plymouth, this was not observed in Singapore (Table 2; further details in Table S5, S6). Additionally, all species found from both countries were native, with the exception of the non-native *Austrominius modestus* in Plymouth and *Siphonaria guamensis* in Singapore (Gallagher et al. 2015, Tan et al. 2018), both of which were found on both treatments at both sites in their respective countries.

GLMs showed a significant effect of month on abundance and species richness in both Plymouth and Singapore. There was also a significant effect of site on abundance and species richness in Singapore (Table 3), with lower rates of colonisation at Pulau Hantu (Fig. 3). There was, however, no significant effect of treatment in either country (Table 3, S7).

Site- and month-specific GLMs revealed that there were significant effects of carbonation at some months and sites, but they were not ubiquitous and none occurred in the final 12-month, time point (Table 4; further details in Table S8, S9). Carbonated tiles had greater total abundance than non-carbonated tiles at Cremyll at the 9-month time point, and at Pulau Hantu at the 6-month time point (Table 4). In Singapore, species richness was greater on carbonated tiles than non-carbonated tiles at the 3-month time point at Pulau Seringat, and at the 6-month time point at Pulau Hantu. There were no other significant effects of carbonation detected from site- and month-specific GLMs.

3.2. Community composition

PERMANOVA revealed significant differences in colonising assemblages among months (SS = 124360; Pseudo-$F_{3,70}$ = 39.06; p = <0.001, SS = 38734; Pseudo-$F_{3,67}$ = 8.6198; p = <0.001, for Plymouth and Singapore respectively; Table 5) and sites (SS = 3309.5; Pseudo-$F_{1,70}$ = 3.1183; p = <0.05, SS = 60739; Pseudo-$F_{1,67}$ = 40.55; p = <0.001, for Plymouth and Singapore respectively; Table 5), but none between treatments regardless of country or month (Table 5). Despite significant results for the interaction term (site × treatment × month) in Singapore, no significant differences were detected when pair-wise comparisons were conducted between treatments within sites and months.

In Plymouth, barnacle *A. modestus*, dominated the surfaces of all tiles (Fig. 4). Despite having higher percentage cover on carbonated tiles than non-carbonated tiles at the 3-month time point, there was no observed difference at the final 12-month time point. In Singapore, biofilm which dominated at 3-month and 6-month time points was succeeded by barnacles and encrusting algae by the 9-month time point (Fig. 4). However, mean barnacle cover fell from 31% to 18% between 9-month and 12-month time points (Fig. 4). Although there appears to be marginal differences between treatments at the 9-month time point, with higher barnacle percentage cover than algae on non-carbonated tiles, this was not observed at the final 12-month time point (Fig. 4).

4. DISCUSSION

Findings from our bilateral one-year study indicate that lowering the pH of concrete did not significantly increase the abundance and species richness of intertidal benthic organisms on retro-fitted enhancement tiles, and did not significantly alter the community composition they support. Concrete is generally considered damaging to the environment, yet it remains one of the most utilised materials in the world and is prevalent in the construction of marine and coastal infrastructure (Bulleri & Chapman 2010, Waters & Zalasiewicz 2018), including marine biodiversity enhancement units. Some researchers have proposed that lowering the pH of concrete would further increase species richness on enhancement units (Perkol-Finkel & Sella 2014, Huang et al. 2016, Reef Ball Foundation 2017). However, previous studies that showed positive effects of lowered concrete pH on benthic diversity were only conducted over short time periods (3–4 weeks; Guilbeau et al. 2003, Nandakumar et al. 2003), in subtidal areas with little/no emersion (Perkol-Finkel & Sella 2014, Sella & Perkol-Finkel 2015), or had also made additional adjustments to the concrete composition and surface texture (Perkol-Finkel & Sella 2014, Sella & Perkol-Finkel 2015, Dennis et al. 2018) which made it difficult to discern if pH was indeed responsible for the positive effect. Given that the current experiment, which tested the effects of pH alone, found no overall significant differences in species recruitment on the tiles, lowering pH might not be an efficacious ecological engineering technique for increasing intertidal biodiversity on artificial structures.

While the effects of soil pH on plants have been thoroughly studied in the terrestrial environment (Bååth & Arnebrant 1994, Robson 2012), the influence of substrate pH on benthic marine life remains poorly understood (Nandakumar et al. 2003, Sekar et al. 2004). Higher species richness on carbonated concrete at earlier time-points in Singapore (3-month at Pulau Seringat and 6-month at Pulau Hantu; Fig. 3) could be related to greater biofilm (e.g., cyanobacteria, diatoms) and microalgal development. pH has been regarded an important factor in the colonisation of natural biofilms (Sekar et al. 2004); further, carbonating concrete can create smaller pore diameters when calcium is precipitated into carbonate form (Roy et al. 1999) that can also encourage microalgal attachment (Guilbeau et al. 2003). These layers of biofilm and microalgae are food resources which could have provided greater foraging opportunities for grazers (Irigoyen et al. 2011), such as limpets (e.g., *Siphonaria guamensis*, *Patelloida saccharina*) and snails (e.g., *Nerita undata*). For example, higher abundance of individuals found on carbonated concrete tiles from Pulau Hantu at the 6-month time point was also mainly due to a single snail species, *N. undata*, a microalgal feeder (Underwood 1984). Concrete carbonation, however, had little or no effect at sites which had low algal growth generally, such as at Cremyll and Turnchapel in the UK (Fig. 4).

Even though there might be some early differences in abundance and species richness between tile treatments in Singapore, the effects of carbonation did not persist. Biofilm formation can strongly influence the settlement of macrofouling taxa such as barnacles, serpulids and mussels (reviewed by Almeida & Vasconcelos 2015), but the lack of significant differences between treatments beyond six months suggests that, even if there were differences in initial microalgal attachment, it was not enough to influence subsequent successional species. Additionally, the surface pH of non-carbonated tiles in Singapore appeared to have reduced to <8 by month 6 (Figure S1). This is in line with findings by Dooley et al. (1999) who suggested that the pH of concrete surface will approach seawater pH after three to six months in marine environments. As such, colonisers may not experience major differences in concrete pH between tiles of different treatments after a few months of seawater exposure.

Substrate alkalinity is also unlikely to affect primary or secondary consumers during low tide, since leaching occurs when concrete is submerged in water (Li et al. 2005). Calcium oxide (CaO) in Portland cement reacts with water to form calcium hydroxide (CaOH), contributing to the high pH of the substrate. Lowering concrete pH via carbonation can also influence the solubility of metals, where copper, cadmium and cobalt are increasingly mobilised, and calcium and strontium become more tightly bound (Sazzad bin Shafique et al. 1998, Fernandez Bertoz et al. 2004), but this mostly occurs during submersion. Nevertheless, the water-retaining pits of the non-carbonated concrete tiles still accommodated a higher abundance and richness of benthic organisms than the flat surfaces of the tiles. Water-retaining features of habitat enhancement units, even non-carbonated concrete ones, provide organisms with shelter from desiccation and thermal stresses (Firth et al. 2016a, Loke et al. 2019b). This adds to the growing evidence that habitat structure may have a larger influence on community assemblages than substratum material (Anderson & Underwood 1994, Coombes et al. 2015).

At small scales, the presence of motile fauna (i.e., gastropods, non-encrusting polychaetes, decapods) is often highly influenced by the availability of refugia and foraging opportunities in habitats (Schmidt & Scheibling 2007, Irigoyen et al. 2011). The empty shells of dead barnacles provide additional complex micro-habitat (<5 mm) structures (Chalmer 1982, Dean & Connell 1987). In this study, many barnacles died in Singapore after initial colonisation, which then served as microhabitats for smaller organisms such as the crab *Nanosesarma minitum*, snails *Zafra* spp. and polynoids (Fig. 5). At a larger scale, seawall design and location can affect benthic colonisation (Jackson 2014). For instance, slope differences can affect the susceptibility of seawalls to extreme surface temperatures, with sloping seawalls absorbing more solar radiation compared to vertical ones (Zhao et al. 2019). Additionally, Pulau Hantu is a particularly sheltered site compared to Pulau Seringat (Loke et al. 2016). Both temperature and wave exposure can affect hard-shore communities (McQuaid & Branch 1984, 1985), and lower abundance and species richness at Pulau Hantu (sloping) compared to Pulau Seringat (vertical) at all time points is likely due to their very different gradients. These biotic and abiotic influences on the succession of the tiles may play a greater role in controlling community patterns compared to the pH of the concrete tiles.

Furthermore, barnacles and serpulids often settle on new intertidal substrate surfaces, both natural (Dean & Connell 1987, Tejada-Martinez et al. 2016) and artificial (Chalmer 1982, Coombes et al. 2017), during early successional phases. While carbonated concrete had previously reduced the settlement of “alkotolerant organisms” (Dooley et al. 1999, Huang et al. 2016) and promoted algal growth (Guilbeau et al. 2003), this effect was not evident in the current experiment. In fact, there were significantly more barnacles on carbonated tiles than non-carbonated tiles at Cremyll at the 9-month sampling point (Table 3, Fig. 3).

To gain a more comprehensive understanding on the effects of concrete pH, future studies can take regular measurements of the tile pH as well as the seawater pH in the water-retaining pits of the tiles. There is also a lack in standardised protocol for testing the pH of other hard substrates such as granite, limestone and other naturally occurring rocks (Aho &

Weaver 2006), which would be useful for investigating the role of substrate pH in influencing marine biodiversity. Nevertheless, this study provides some insight to the potential effects of pH on marine benthic colonisation from an ecological engineering perspective.

As the demand for urban coastal development rises in response to the threats of sea level rise and increasing coastal populations, it is important to consider engineering solutions that can maximise the ecological functioning of artificial structures. However, the influence of substrate pH on benthic colonisation is relatively understudied with little evidence to support the hypothesis that lowering concrete pH can increase species richness or abundance of organisms. Our experiment indicates that the effects of pH on benthic colonisation is non-significant and we suggest that manipulation of the physical structure of habitat enhancement units, such as increasing topographical complexity and adding water-retaining features, is a more effective eco-engineering approach to enhancing the ecological value and species diversity on seawalls.

Acknowledgements

We thank members of the Experimental Marine Ecology Laboratory and Tropical Marine Science Institute and Richard Ticehurst for their assistance in the field. We also thank Tan Siong Kiat from Lee Kong Chian Natural History Museum for his help with the identification of specimens. This research was funded by the National Research Foundation, Prime Minister’s Office, Singapore under its Marine Science Research and Development Programme (Award No. MSRDP-P05).

References

Aguilera MA, Arias RM, Manzur T (2019) Mapping microhabitat thermal patterns in artificial breakwaters: Alteration of intertidal biodiversity by higher rock temperature. Ecol Evol 9(22):12915–12927

Aho K & Weaver T (2006) Measuring water relations and pH of cryptogam rock-surface environments. Bryologist 109(3): 348–357

Almeida JR, Vasconcelos V (2015) Natural antifouling compounds: Effectiveness in preventing invertebrate settlement and adhesion. Biotechnol Adv 33(3–4):343–357

Alonso MC, Garcia CJL, Walker C, Naito M, Pettersson S, Puigdomenech I, Cuñado MA, Uorio M, Weber H, Ueda H and Fujisaki K (2012) Development of an accurate pH measurement methodology for the pore fluids of low pH cementitious materials. SKB Report R-12-02. Sweden: Swedish Nuclear Fuel and Waste Management Co.Anderson MJ (2001) A new method for non-parametric multivariate analysis of variance. Austral Ecol 26:32–46

Anderson MJ, Underwood AJ (1994) Effects of substratum on the recruitment and development of an intertidal estuarine fouling assemblage. J Exp Mar Biol Ecol 184(2):217–236

Anderson M, Gorley RN, Clarke RK (2008) Permanova+ for primer: Guide to software and statistical methods. Primer-E Limited.

Bååth E, Arnebrant K (1994) Growth rate and response of bacterial communities to pH in limed and ash treated forest soils. Soil Biol Biochem 26(8):995–1001

Bishop MJ, Mayer-Pinto M, Airoldi L, Firth LB, Morris RL (2017) Effects of ocean sprawl on ecological connectivity: impacts and solutions. J Exp Mar Biol Ecol 492:7–30

Bulleri F, Airoldi L (2005) Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the north Adriatic Sea. J Appl Ecol 42(6):1063–1072

Bulleri F, Chapman MG (2010) The introduction of coastal infrastructure as a driver of change in marine environments. J Appl Ecol 47:26–35

Chalmer PN (1982) Settlement patterns of species in a marine fouling community and some mechanisms of succession. J Exp Mar Biol Ecol 58(1):73–85

Chang CF, Chen JW (2006) The experimental investigation of concrete carbonation depth. Cement Concr Res 36(9):1760–1767

Chapman MG (2003) Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. Mar Ecol Prog Ser 264:21–29

Chapman MG, Bulleri F (2003) Intertidal seawalls—new features of landscape in intertidal environments. Landsc Urban Plan 62:159–172

Chapman MG, Underwood AJ (2011) Evaluation of ecological engineering of “armoured” shorelines to improve their value as habitat. J Exp Mar Biol Ecol 400:302–313

Chapman MG, Underwood AJ, Browne MA (2018) An assessment of the current usage of ecological engineering and reconciliation ecology in managing alterations to habitats in urban estuaries. Ecol Eng 120:560–573

Chi JM, Huang R, Yang CC (2002) Effects of carbonation on mechanical properties and durability of concrete using accelerated testing method. J Mar Sci Technol 10(1):14–20

Coombes MA, Viles HA, Naylor LA, La Marca EC (2017) Cool barnacles: Do common biogenic structures enhance or retard rates of deterioration of intertidal rocks and concrete?. Sci Total Environ 580:1034–1045

Dafforn KA, Glasby TM, Airoldi L, Rivero NK, Mayer-Pinto M, Johnston EL (2015) Marine urbanization: an ecological framework for designing multifunctional artificial structures. Front Ecol Environ 13(2):82–90

De Ceukelaire L, Van Nieuwenburg D (1993) Accelerated carbonation of a blast-furnace cement concrete. Cement Concr Res 23(2):442–452.

Dean RL, Connell JH (1987) Marine invertebrates in an algal succession. I. Variations in abundance and diversity with succession. J Exp Mar Biol Ecol 109(3):195–215

Dennis HD, Evans AJ, Banner AJ, Moore PJ (2018) Reefcrete: Reducing the environmental footprint of concretes for eco-engineering marine structures. Ecol Eng 120:668–678

Dooley KM, Knopf FC, Gambrell RP (1999) pH-Neutral Concrete for Attached Microalgae and Enhanced Carbon Dioxide Fixation-Phase I (No. AC26-98FT40411-01). Federal Energy Technology Center, Morgantown, WV (US); Federal Energy Technology Center, Pittsburgh, PA (US).

Dyer T (2014) Concrete Durability. CRC Press.

Dugan JE, Airoldi L, Chapman MG, Walker SJ, Schlacher T, Wolanski E, McLusky D (2011) Estuarine and coastal structures: environmental effects, a focus on shore and nearshore structures. Estuar Coast Sci 8:17–41

Fernández Bertos M, Li X, Simons SJR, Hills CD, Carey PJ (2004) Investigation of accelerated carbonation for the stabilization of MSW incinerator ashes and the sequestration of CO2. Green Chem 6:428–436

Firth LB, Thompson RC, White FJ, Schofield M, Skov MW, Hoggart SPG, Jackson JE, Knights AM, Hawkins SJ (2013) The importance of water retaining features for biodiversity on artificial intertidal coastal defence structures. Divers Distrib 19:1275–1283

Firth LB, Thompson RC, Bohn K, Abbiati M, Airoldi L, Bouma TJ, Bozzeda F, Ceccherelli VU, Colangelo MA, Evans A, Ferrario F, Hanley ME, Hinz H, Hoggart SPG, Jackson JE, Moore P, Morgan EH, Perkol-Finkel S, Skov MW, Strain EM, van Belzen J, Hawkins SJ

(2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. Coastal Eng 87:122–135

Firth LB, Browne KA, Knights AM, Hawkins SJ, Nash R (2016a) Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment. Environ Res Lett 11(9):094015

Firth LB, Knights AM, Bridger D, Evans AJ, Mieszkowska N, Moore PJ, O'Connor NE, Sheehan EV, Thompson RC, Hawkins SJ (2016b) Ocean sprawl: challenges and opportunities for biodiversity management in a changing world. In: Hughes RN, Hughes DJ, Smith IP, Dale AC (eds) Oceanography and Marine Biology. CRC Press, pp. 201–278

Gallagher MC, Davenport J, Gregory S, McAllen R, O'Riordan R (2015) The invasive barnacle species, *Austrominius modestus*: Its status and competition with indigenous barnacles on the Isle of Cumbrae, Scotland. Estuar Coast Sci 152:134–141

Glasby TM, Connell SD, Holloway MG, Hewitt CL (2007) Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions?. Mar Biol 151(3):887–895

Gosselin LA, Chia FS (1995) Distribution and dispersal of early juvenile snails: effectiveness of intertidal microhabitats as refuges and food sources. Mar Ecol Prog Ser 128:213–223

Guilbeau BP, Harry FP, Gambrell RP, Knopf FC, Dooley KM (2003) Algae attachment on carbonated cements in fresh and brackish waters—preliminary results. Ecol Eng 20(4):309–319

Hatcher AM (1998) Epibenthic colonisation patterns on slabs of stabilised coal-waste in Poole Bay, UK. Hydrobiologia 367(1–3):153–162

Heery EC, Bishop MJ, Critchley LP, Bugnot AB, Airoldi L, Mayer-Pinto M, Sheehan EV, Coleman RA, Loke LHL, Johnston EL, Komyakova V, Morris RL, Strain EMA, Naylor LA, Dafforn KA (2017) Identifying the consequences of ocean sprawl for sedimentary habitats. J Exp Mar Biol Ecol 492:31–48

Huang X, Wang Z, Liu Y, Hu W, Ni W (2016) On the use of blast furnace slag and steel slag in the preparation of green artificial reef concrete. Constr Build Mater 112:241–246

Irigoyen AJ, Trobbiani G, Sgarlatta MP, Raffo MP (2011) Effects of the alien algae *Undaria pinnatifida* (Phaeophyceae, Laminariales) on the diversity and abundance of benthic macrofauna in Golfo Nuevo (Patagonia, Argentina): potential implications for local food webs. Biol Invasions 13(7):1521–1532

Jackson JE (2014) The influence of engineering design considerations on species recruitment and succession on coastal defence structures. PhD dissertation, University of Plymouth, United Kingdom.

Knights AM, Firth LB, Thompson RC, Yunnie ALE, Hiscock K, Hawkins SJ (2016). Plymouth—a world harbour through the ages. Reg Stud Mar Sci 8(2) 297–307

Kohler KE, Gill SM (2006) Coral Point Count with Excel extensions (CPCe): a visual basic program for the determination of coral and substrate coverage using random point count methodology. Comput Geosci 32:1259–1269

Lai S, Loke LHL, Hilton MJ, Bouma TJ, Todd PA (2015) The effects of urbanisation on coastal habitats and the potential for ecological engineering: A Singapore case study. Ocean Coast Manage 103:78–85

Lai S, Loke LHL, Bouma TJ, Todd PA (2018) Biodiversity surveys and stable isotope analyses reveal key differences in intertidal communities between tropical seawalls and rocky shores. Mar Ecol Prog Ser 587:41–53

Lam NW, Huang R, Chan BK (2009) Variations in Intertidal assemblages and zonation patterns between vertical artificial seawalls and natural rocky shores: A case study from Victoria Harbour, Hong Kong. Zool Stud 48(2):184–195

Li L, Nam J, Hartt WH (2005) Ex situ leaching measurement of concrete alkalinity. Cement Conc Res 35(2):277–283

Loke LHL, Todd PA (2016) Structural complexity and component type increase intertidal biodiversity independently of area. Ecology 97:383–393

Loke LHL, Liao LM, Bouma TJ, Todd PA (2016) Succession of seawall algal communities on artificial substrates. Raffles Bull Zool 32:1–10

Loke LHL, Bouma TJ, Todd PA (2017) The effects of manipulating microhabitat size and variability on tropical seawall biodiversity: field and flume experiments. J Exp Mar Biol Ecol 492:113–120

Loke LHL, Chisholm RA, Todd PA (2019a) Effects of habitat area and spatial configuration on biodiversity in an experimental intertidal community. Ecology 100(8):e02757

Loke LHL, Heery EC, Lai S, Bouma TJ, Todd PA (2019b) Area-independent effects of water-retaining features on intertidal biodiversity on eco-engineered seawalls in the tropics. Front Mar Sci 6:16

Loke LHL, Jachowski NR, Bouma TJ, Ladle RJ and Todd PA (2014) Complexity for artificial substrates (CASU): software for creating and visualising habitat complexity. PLoS One 9(2):e87990

Lukens RR, Selberg C (2004) Guidelines for Marine Artificial Reef Materials. A Joint Publication of the Gulf and Atlantic States Marine Fisheries Commissions, p. 7

McManus RS, Archibald N, Comber S, Knights AM, Thompson RC, Firth LB (2018) Partial replacement of cement for waste aggregates in concrete coastal and marine infrastructure: a foundation for ecological enhancement?. Ecol Eng 120:655–667

McQuaid C, Branch GM (1984) Influence of sea temperature, substratum and wave exposure on rocky intertidal communities: An analysis of faunal and floral biomass. Mar Ecol Prog Ser 19(1):145–151

McQuaid C, Branch GM (1985). Trophic structure of rocky intertidal communities: Response to wave action and implications for energy flow. Mar Ecol Prog Ser 22(2):153–161

Moreira J, Chapman MG, Underwood AJ (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. J Exp Mar Biol Ecol 347(1–2):134–143

Morris RL, Heery EC, Loke LHL, Lau E, Strain EMA, Airoldi L, Alexander KA, Bishop MJ, Coleman RA, Cordell JR, Dong YW, Firth LB, Hawkins SJ, Heath T, Kokora M, Lee SY, Miller JK, Perkol-Finkel S, Rella A, Steinberg PD, Takeuchi I, Thompson RC, Todd PA, Toft JD, Leung KMY (2019) Design options, implementation issues and evaluating success of ecologically engineered shorelines. Oceanogr Mar Biol Annu Rev 57:169-228

Moschella PS, Abbiati M, Åberg P, Airoldi L, Anderson JM, Bacchiocchi F, Bulleri F, Dinesen GE, Frost M, Gacia E, Granhag L, Jonsson PR, Satta MP, Sundelof A, Thompson RC, Hawkins SJ (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. Coast Eng 52(10–11):1053–1071

Nandakumar K, Matsunaga H, Takagi M (2003) Microfouling studies on experimental test blocks of steel-making slag and concrete exposed to seawater off Chiba, Japan. Biofouling 19(4):257–267

Neves R, Branco F, de Brito J (2013) Field assessment of the relationship between natural and accelerated concrete carbonation resistance. Cement Concr Compos 41:9–15.

Nicholls RJ, Tol RS, Hall JW (2007) Assessing impacts and responses to global-mean sea-level rise. In: Schlesinger ME, Kheshgi HS, Smith J, de la Chesnaye FC, Reilly JM, Wilson T, Kolstad (eds) Human-induced climate change. Cambridge University Press, Cambridge, pp. 119–134

Perkol-Finkel S, Sella I (2014) Ecologically active concrete for coastal and marine infrastructure: innovative matrices and designs. In: Allsop W, Burgess K (eds) From Sea to Shore – Meeting the Challenges of the Sea. ICE Publishing, pp.1139–1149

R Core Team (2019). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Reef Ball Foundation (2017) Reef Ball Foundation. http://reefball.org (accessed 9 Dec 2019)

Robson A (2012) Soil acidity and plant growth. Elsevier.

Roy SK, Poh KB, Northwood DO (1999) Durability of concrete—accelerated carbonation and weathering studies. Build Environ 34(5):597–606

Shafique MSB, Walton JC, Gutierrez N, Smith RW, Tarquin AJ (1998) Influence of carbonation on leaching of cementitious wasteforms. J Environ Eng 124(5):463–467

Schmidt AL, Scheibling RE (2007) Effects of native and invasive macroalgal canopies on composition and abundance of mobile benthic macrofauna and turf-forming algae. J Exp Mar Biol Ecol 341(1):110–130

Sekar R, Venugopalan VP, Satpathy KK, Nair KVK, Rao VNR (2004) Laboratory studies on adhesion of microalgae to hard substrates. In: Ang PO Jr. (ed) Asian Pacific Phycology in the 21st Century: Prospects and Challenges, Springer, Dordrecht, pp. 109–116

Sella I, Perkol-Finkel S (2015) Blue is the new green-ecological enhancement of concrete based coastal and marine infrastructure. Ecol Eng 84:260–272

Sellers KF, Shmueli G (2010) A Flexible Regression Model for Count Data. Ann Appl Stat 4(2):943–961

Sellers KF, Lotze T, Raim A (2017) COMPoissonReg: Conway-Maxwell Poisson (COM-Poisson) Regression, p. 380

Short NR, Purnell P, Page CL (2001) Preliminary investigations into the supercritical carbonation of cement pastes. J Mater Sci 36(1):35–41

Strain EMA, Olabarria C, Mayer-Pinto M, Cumbo V, Morris RL, Bugnot AB, Dafforn KA, Heery E, Firth LB, Brooks PR, Bishop MJ (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit?. J Appl Ecol 55(1):426–441

Tan WT, Loke LHL, Yeo DCJ, Tan SK, Todd PA (2018) Do Singapore's seawalls host non-native marine molluscs?. Aquat Invasions 13(3):365–378

Tejada-Martinez D, López DN, Bonta CC, Sepúlveda RD, Valdivia N (2016) Positive and negative effects of mesograzers on early-colonizing species in an intertidal rocky-shore community. Ecol Evol 6(16):5761–5770

Temmerman S, Meire P, Bouma TJ, Herman PMJ, Ysebaert T, de Vriend HJ (2013) Ecosystem-based coastal defence in the face of global change. Nature 504:79–83

Thiery M, Villian G, Dangla P, Platret G (2007) Investigation of the carbonation front shape on cementitious materials: Effects of the chemical kinetics. Cement Conc Res 37:1047–1058

Pre-proof verion of Hsiung, A.R., Tan, W.T., Loke, L.H., Firth, L.B., Heery, E.C., Ducker, J., Clark, V., Pek, Y.S., Birch, W.R., Ang, A.C. and Hartanto, R.S., 2020. Little evidence that lowering the pH of concrete supports greater biodiversity on tropical and temperate seawalls. Marine Ecology Progress Series, 656, 193-205.

Todd PA, Heery EC, Loke LHL, Thurstan RH, Kotze DJ, Swan C (2019) Towards an urban marine ecology: characterizing the drivers, patterns and processes of marine ecosystems in coastal cities. OIKOS 128(9):1215–1242

Underwood AJ (1984) Microalgal food and the growth of the intertidal gastropods *Nerita atramentosa* Reeve and *Bembicium nanum* (Lamarck) at four heights on a shore. J Exp Mar Biol Ecol 79(3):277–291

Venables WN, Ripley BD (2002) Modern Applied Statistics with S. Fourth edition. Springer.

Venhuis MA, Reardon EJ (2003) Carbonation of cementitious wasteforms under supercritical and high pressure subcritical conditions. Environ Technol 24(7):877–887

Waltham NJ, Dafforn KA (2018) Ecological engineering in the coastal seascape. Ecol Eng 120:554–559

Waters CN, Zalasiewicz J (2018) Concrete: the most abundant novel rock type of the Anthropocene. In: DellaSala DA, Goldstein MI (eds) Encyclopedia of the Anthropocene 1:75–85

Zhao K, Yuan J, Loke LHL, Chan SHM, Todd PA, Liu PLF (2019) Modelling surface temperature of granite seawalls in Singapore. Case Stud Therm Eng 13(9):100395

Zuur AF, Ieno EN, Elphick CS (2010) A protocol for data exploration to avoid common statistical problems. Methods Ecol Evol 1(1):3–14

Tables

Table 1. Functional categories used for classifying algae in this study, adapted from Loke et al. (2016).

Functional group	Dominant Component Taxa (examples from Singapore)
Microalgae/biofilm	Unidentified cyanobacteria and diatoms, bare surfaces were also classified in this group due to difficulty in differentiating visually.
Encrusting algae	Ralfsiaceae and/or Neoralfsiaceae
Ephemeral green turfs	*Ulva* spp.
Red/brown turfs	*Parviphycus antipae*, *Gelidiopsis variabilis*, *Dictyota* spp. and Ceramiales

Table 2. Total number of species and unique species found on each tile treatment at each site across all time points.

	Total number of species		Total number of unique species	
Sites	Carbonated	Non-carbonated	Carbonated	Non-carbonated
Cremyll	11	8	4	1
Turnchapel	8	7	2	1
Pulau Hantu	19	21	4	6
Pulau Seringat	41	41	5	5

Table 3. Analysis of deviance results for negative binomial and poisson GLMs for total abundance and species richness in Plymouth (left) and Singapore (right). Significant p-values, as determined by likelihood ratio tests, are shown in bold.

	Plymouth, UK					Singapore				
Source	df	Dev	Res df	Res Dev	P	df	Dev	Res df	Res Dev	P
Abundance - Neg. Bin. GLM										
Model			94	164.5				88	448.1	
Site	1	0.9	93	163.6	0.3388	1	253.9	87	194.2	**<0.0001**
Treatment	1	0.9	92	162.7	0.3411	1	2.5	86	191.6	0.1107
Month	1	48.7	91	114.0	**<0.0001**	1	83.0	85	108.6	**<0.0001**
Richness - Poisson GLM										
Model			94	50.8				88	350.9	
Site	1	0.1	93	50.8	0.8150	1	211.3	87	169.7	**<0.0001**
Treatment	1	0.4	92	50.3	0.5478	1	2.7	86	167.0	0.1008
Month	1	19.1	91	31.2	**<0.0001**	1	117.6	85	49.7	**<0.0001**

Table 4. Results from site- and month- specific GLMs for total abundance and species richness. Models for N used a negative binomial error distribution, while Conway-Maxwell-Poisson error was used in models for S. All contained treatment as the sole predictor. The table shows "--" when there was no difference between pH treatments, "C > NC" where carbonated tile treatments had higher abundance or species richness than the non-carbonated pH treatment, and "na" where no data were available. Complete coefficient summaries from each model are provided in Appendix A.

Country	Site	3-month	6-month	9-month	12-month	15-month
Abundance - Neg. Bin. GLM						
Plymouth, UK	Cremyll	--	--	C > NC	--	na

	Turnchapel	--	--	--	--	na
Singapore	P. Hantu	--	C > NC	na	--	--
	P. Seringat	--	--	--	--	na
Richness - Conway-Maxwell-Poisson GLM						
Plymouth, UK	Cremyll	--	--	--	--	na
	Turnchapel	--	--	--	--	na
Singapore	P. Hantu	--	C > NC	na	--	--
	P. Seringat	C > NC	--	--	--	na

Table 5. Permutational distance-based multivariate analysis of variance (PERMANOVA) results based on Bray-Curtis dissimilarities of the relative abundances (log-transformed) of 13 and 57 (Plymouth and Singapore, respectively) taxa in response to site, pH treatment and duration since deployment as fixed factors and their interactions.

Source	**df**	**SS**	**Pseudo-F**	**P(perm)**	**Unique perms**
Plymouth, UK					
Site	1	3309.5	3.12	**0.0379**	9942
Treatment	1	1343.4	1.27	0.2568	9940
Month	3	124360.0	39.06	**<0.0001**	9914
Site x Treatment	1	1944.1	1.83	0.1297	9941
Site x Month	3	3828.2	1.20	0.2841	9938
Treatment x Month	3	4979.9	1.56	0.1303	9936
Site x Treatment x Month	3	1703.4	0.54	0.8541	9951
Residual	70	83844			
Singapore					
Site	1	60739.0	40.55	**<0.0001**	9949
Treatment	1	1748.7	1.17	0.2903	9930
Month	3	38734.0	8.62	**<0.0001**	9910
Site x Treatment	1	519.0	0.35	0.9628	9936
Site x Month	2	27654.0	9.23	**<0.0001**	9927
Treatment x Month	3	5119.7	1.14	0.2892	9904
Site x Treatment x Month	2	5327.8	1.78	**0.0454**	9904
Residual	67	100360			

Figures

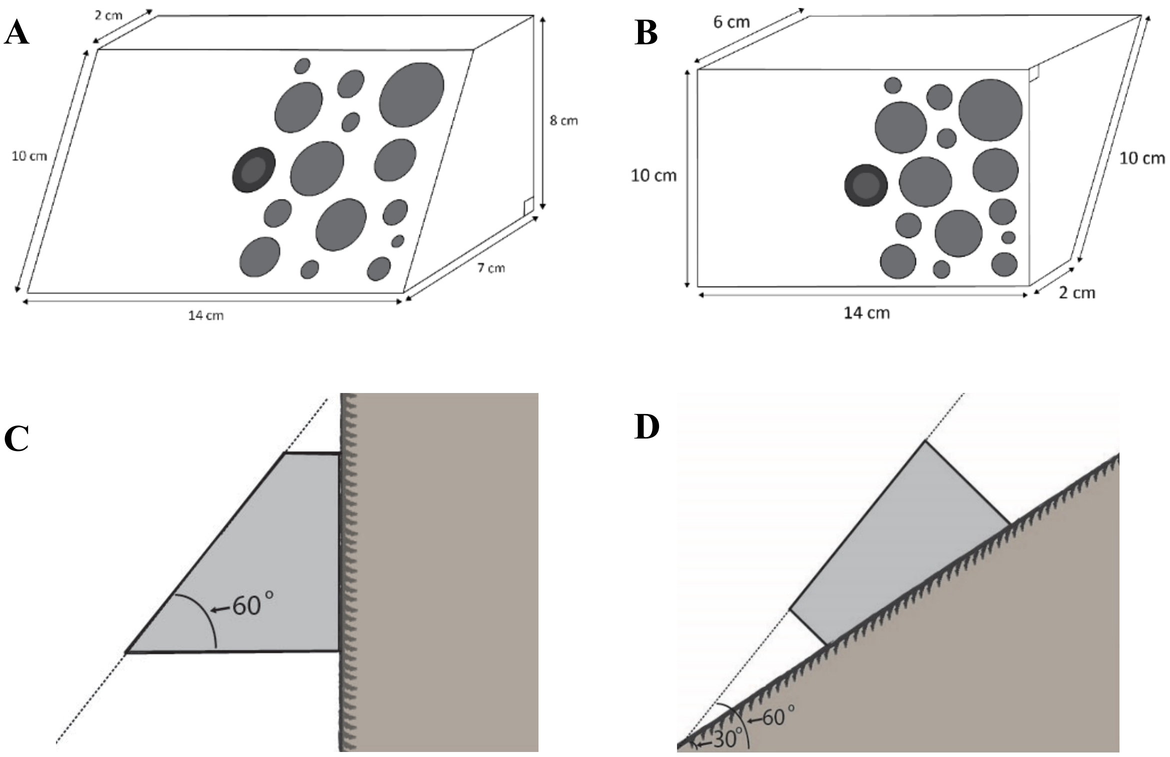

Figure 1. Dimensions of tiles for (A) vertical and (B) sloping seawalls, with schematics of the tiles when installed on the seawalls (C and D, respectively).

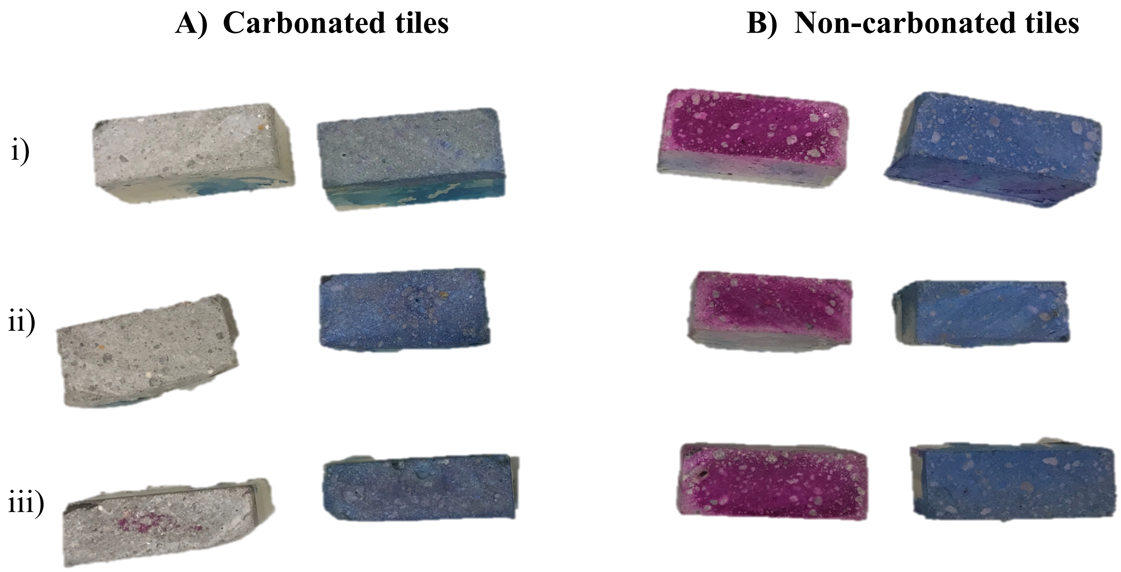

Figure 2. Images of (A) carbonated tiles stained with phenolphthalein (left) and bromothymol blue (right) after undergoing: i) 29 days of carbonation and 12 days of drying, ii) 22 days of carbonation and 20 days of drying, and iii) 22 days of carbonation and 6 days of drying, with (B) non-carbonated

tiles that dried for the same amount of time (control) stained with phenolphthalein (left) and bromothymol blue (right).

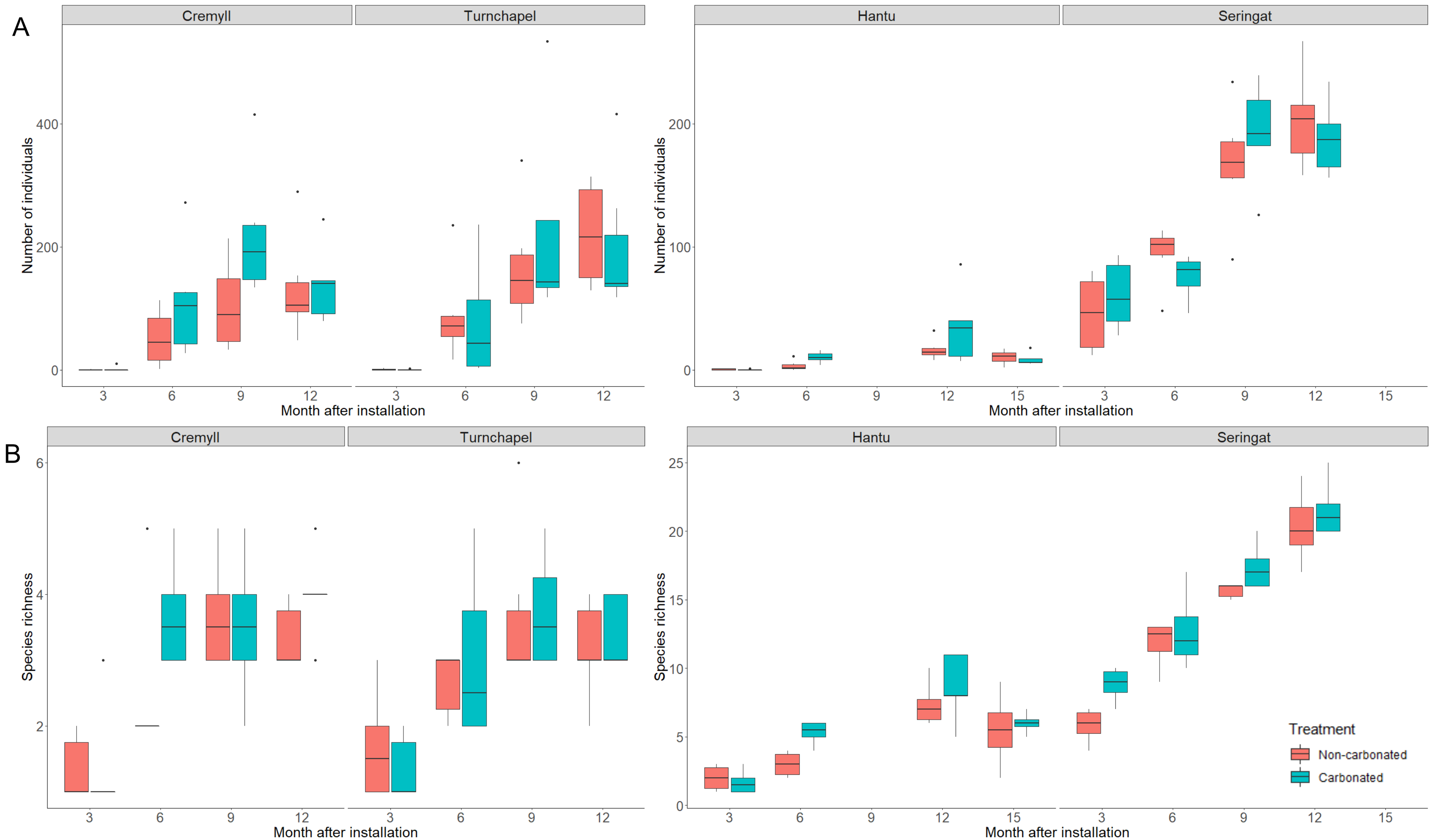

Figure 3. (A) Abundance (number of individuals) and (B) species richness on tile treatments (non-carbonated and carbonated) across four time points (3-month, 6-, 9-, 12- for Cremyll, Turnchapel and Pulau Seringat, 3-month, 6-, 12-, 15- for Pulau Hantu). Boxplot middle lines indicate the median; hinges indicate 75% and 25% quantiles (top and bottom, respectively); whiskers indicate highest and lowest values within 1.5 times the interquartile range from top and bottom hinges, respectively; dots indicate outliers.

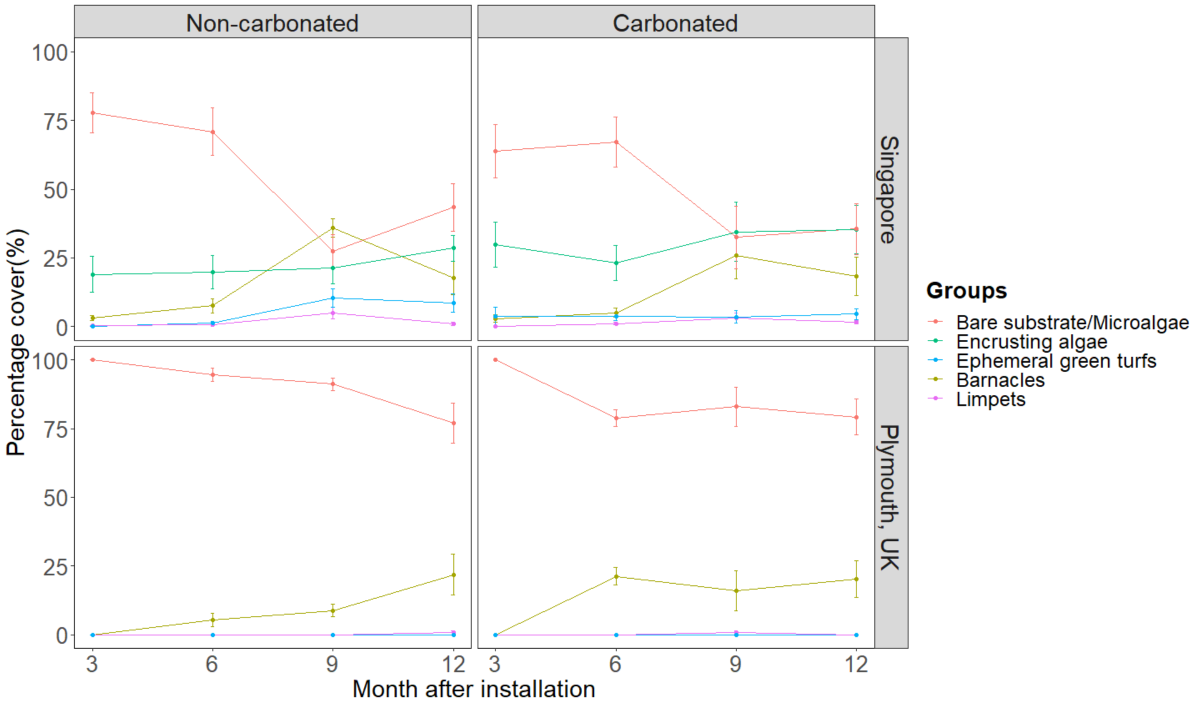

Figure 4. Changes in mean percentage cover (±1SE) of dominant taxa (mean > 1%) on the front surface of the tiles in Plymouth and Singapore over time.

Figure 5. Example of a non-carbonated tile at 12-month from Pulau Seringat, Singapore, with several empty barnacle shells that contributed to microhabitats for smaller organisms.