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Vertical arrays of artificial rockpools on a seawall provide refugia across tidal levels for intertidal species in the UK

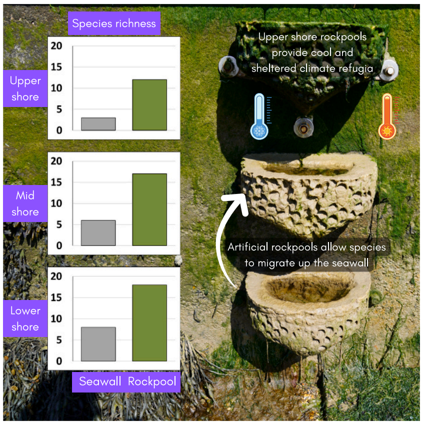
Jessica R. Bone [a,c], Rick Stafford [a,*], Alice E. Hall [b], Roger J.H. Herbert [a]

[a] Bournemouth University, Fern Barrow, Poole BH12 5BB, UK
[b] University of Plymouth, Drake Circus, Plymouth PL4 8AA, UK
[c] Natural England, London, UK

HIGHLIGHTS

- Artificial rockpools can retain both water and sediment, providing multiple habitats for intertidal fauna
- Although higher artificial rockpools have lower species richness, they have a greater impact on the seawall biodiversity
- Artificial rockpools may play a role in mitigating impacts of sea level rise on the intertidal zone of coastal structures

GRAPHICAL ABSTRACT



Upper shore rockpools provide cool and sheltered climate refugia

Artificial rockpools allow species to migrate up the seawall

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ABSTRACT

Eco-engineering of coastal infrastructure aims to address the insufficient intertidal habitat provided by coastal development and flood defence. There are numerous ways to enhance coastal infrastructure with habitat features, but a common method involves retrofitting artificial rockpools. Often these are 'bolt-on' units that are fixed to existing coastal infrastructure but there is a paucity of literature on how to optimise their arrangement for biodiversity.

In this study, 24 artificial rockpools were installed at three levels between High Water Neaps and Mean Tide Level on a vertical concrete seawall on the south coast of the UK. The species abundance of the rockpools and adjacent seawall were surveyed at low tide for 2 years following rockpool installation and compared. Over the course of the study, sediment had begun to accumulate in some of the rockpools. At the 2-year mark, the sediment was removed and assessed for macrofauna. Algal biomass of the seawall and rockpools was estimated using previously obtained dry weight values for the dominant algae taxa.

After 2 years, it was determined that artificial rockpools successfully increase species richness of seawalls, particularly at higher tidal levels where water-retaining refugia are crucial for many species. The rockpools

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* Corresponding author.
E-mail address: rstafford@bournemouth.ac.uk (R. Stafford).

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hosted 37 sessile taxa and 9 sessile taxa were recorded on the seawall. Rockpools increased the vertical elevation for brown canopy-forming seaweeds by providing better attachment surfaces. Although the retained sediment only hosted 3 infaunal species, it was observed to provide shelter for shore crabs during surveys. As sea levels and ocean and air temperatures continue to rise, vertical eco-engineering arrangements will play a crucial role in allowing species to migrate up the tidal zone, negating habitat loss and localised extinction.

1. Introduction

The biodiversity of rocky shores may largely be attributed its structural and topographical complexity, which includes surfaces of varying rugosity and rockpools (also known as tide pools) (Metaxas and Scheibling, 1993). The retreat of the tide on a rocky coastline exposes intertidal biota to the risk of desiccation stress and associated mortality, and predation (Little et al., 2009). However, rockpools provide refuge and resources throughout the tidal cycle (Martins et al., 2007; Firth et al., 2013; Legrand et al., 2018) and permit species that cannot live out

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Fig. 1. The rockpool location in Hamble Harbour on the south coast of the UK.

Fig. 2. Top: the seawall in 2017 prior to rockpool installation. Bottom: the seawall during rockpool installation in October 2020.

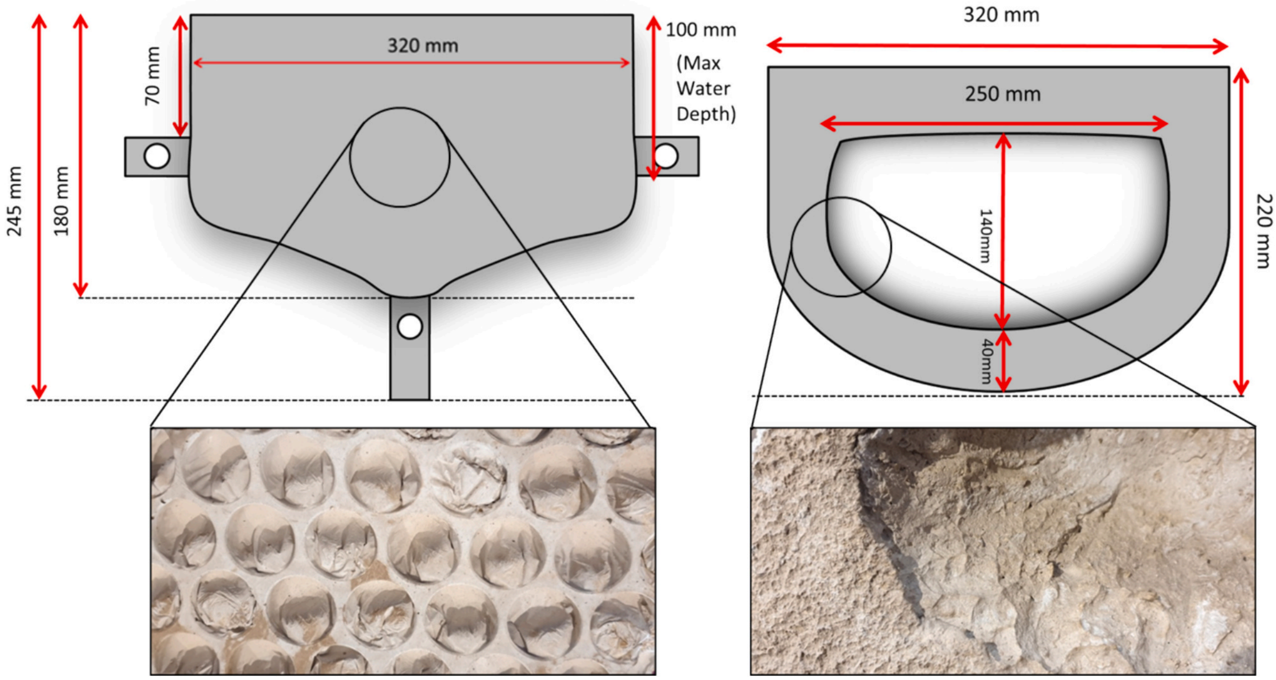

Fig. 3. The artificial rockpool dimensions.

Table 1
The quadrat size and number of quadrat replicates recorded in each habitat per rockpool.

Habitat	Quadrat size	Replicates
Artificial rockpool - interior	25 cm^2	1 – the whole pool basin
Artificial rockpool - exterior	25 cm^2	2 – one on each exterior half
Seawall	25 cm^2	2 – one either side of the rockpool

of water to extend their distribution into the intertidal zone. Rockpools are crucial to food web connectivity (Noel et al., 2009) on rocky shores, particularly at low tide (Mendonça et al., 2018; Vinagre and Mendonça, 2023). At high tide, rockpools serve as feeding grounds for juvenile transient fish, intertidal benthic fish and crabs (Mendonça et al., 2019; Bone et al., 2024).

Ecological enhancement (also referred to as eco-engineering) is used to add habitat to artificial coastal and marine structures (Bergen et al., 2001; Mitsch and Jorgensen, 2003; Odum and Odum, 2003; Airoldi et al., 2005) which are ecologically poor surrogates for natural hard substrate environments (Connell and Glasby, 1999; Chapman, 2003; Moschella et al., 2005; McKinney, 2006; Glasby et al., 2007; Vaselli et al., 2008; Pister, 2009; Firth et al., 2013; Earp et al., 2023). There are numerous ways this can be achieved which are well summarised (Naylor et al., 2017; Strain et al., 2017; O'Shaughnessy et al., 2020; Evans et al., 2021) but examples include drill-cored or hammered pits in rip-rap rock armour (Firth et al., 2014; Evans et al., 2015; Ostale-Valriberas et al., 2018; Chee et al., 2020), standalone pre-cast intertidal reef blocks and pools (Firth et al., 2014; Perkol-Finkel and Sella, 2015; Waltham and Sheaves, 2018), habitat features integrated or recessed within the structure through bricks or form liners (Chapman and Blockley, 2009; Chapman and Underwood, 2011; Firth et al., 2014), and retrofitted tiles, rockpools and panels (Browne and Chapman, 2011; Browne and Chapman, 2014; Morris et al., 2017; Hall et al., 2019; MacArthur et al., 2019; Bishop et al., 2022; Kosova et al., 2023).

Retrofitted (installed after the construction) 'bolt-on' artificial rockpools are commonly used to add ecological enhancement to existing artificial coastal structures (Naylor et al., 2017), particularly for research purposes (Browne and Chapman, 2011; Morris et al., 2017; Morris et al., 2018; Hall et al., 2019; Drakard et al., 2023; Bone et al., 2024). These precast, 'off-the-shelf' units are now available on a commercial basis (Evans et al., 2019). However, there is a paucity of literature on how the arrangement (i.e., number, pattern/configuration) of retrofitted interventions can impact species abundance outcomes (but see Loke et al., 2019).

Optimising the arrangement of intertidal interventions is crucial for incorporating ecological enhancement at scale, as it informs engineers where interventions should be focussed and where they may be less necessary. Tidal height of rockpools determines how long they remain emersed at low tide, with those higher in elevation remaining emersed for longer (Martins et al., 2007; Firth et al., 2013). Consequently, the physico-chemical parameters of the retained water are impacted by exogenous abiotic factors (solar radiation, evaporation, pool area/ depth/ volume) (Daniel and Boyden, 1975; Truchot and Duhamel-Jouve, 1980; Huggett and Griffiths, 1986; Metaxas and Scheibling, 1993; White et al., 2014), biological processes (photosynthesis, respiration), and species interactions (Underwood and Jernakoff, 1984; Benedetti-Cecchi et al., 2000). Rockpools experience greater variations of these physico-chemical parameters than the sea (Morris and Taylor, 1983) but a lower magnitude than that of the emergent substrate (Metaxas and Scheibling, 1993). A rockpool at a higher tidal elevation is likely to experience more extreme temperatures, pH, and salinity than rockpools at a lower tidal elevation that are emersed for less time (Little et al., 2009; Legrand et al., 2018). As a result, the biota they support is generally limited to species that are especially tolerant of more hostile conditions, such as *Ulva* sp. green algae (Legrand et al., 2018; Hall et al., 2019). Tidal height is an important factor in deciding where interventions should be placed within an artificial coastal structure or a flood defence scheme (Firth et al., 2016; Naylor et al., 2017). Therefore, maximising biota on a vertical seawall may be more effective at moderate tidal elevations (Firth et al., 2013). Installing retrofitted interventions may alter the distribution of species on the existing structure, for example through the provision of microhabitats favoured by grazers (Fairweather, 1988; Aguilera et al., 2014; Aguilera et al., 2022) or by ameliorating temperature and desiccation stressors by creating shaded, damp areas (Meager et al., 2011; Baxter et al., 2023). Some studies have been conducted on artificial rockpools along a vertical gradient (Browne and Chapman, 2014) in the northern hemisphere (Hall et al., 2019; Bone et al., 2022) but independence between levels and replication of rockpools was low. Otherwise, studies have favoured a horizontal array (Browne and Chapman, 2011; Morris et al., 2017; Morris et al., 2018; Drakard et al., 2023). In this context, an experiment was undertaken to determine the impact of vertical distribution and

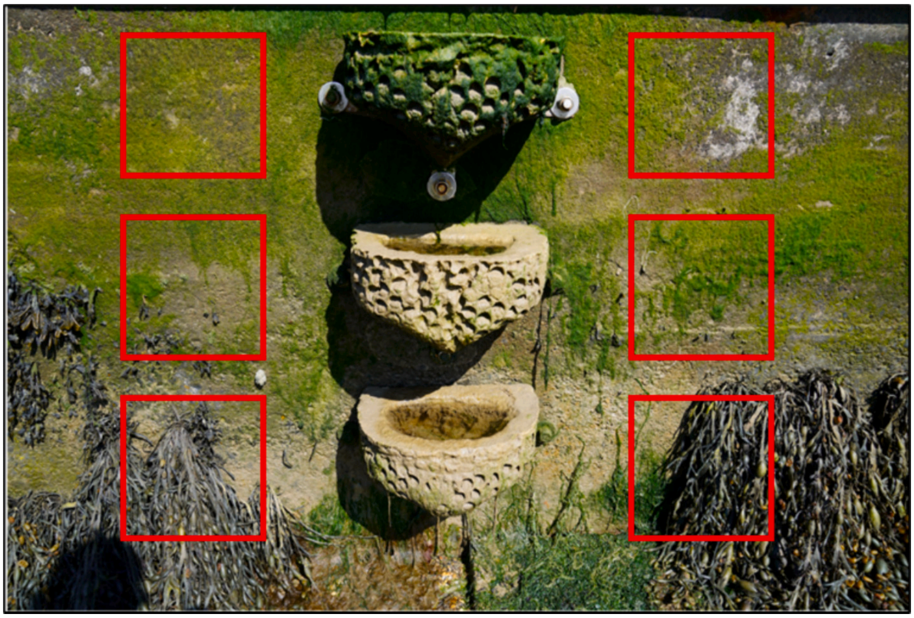

Fig. 4. The placement of photo-quadrats on the adjacent seawall.

Fig. 5. The colonisation of column 4 at 6 months (a.), 12 months (b.), 18 months (c.) and 24 months (d.)

tidal height on the colonisation of 'bolt on' rockpools on a vertical seawall. As part of the Marineff Project 24 artificial rockpools (Vertipools™) were installed in Hamble Harbour. We aimed to determine how tidal height impacts species abundance on the seawall and artificial rockpools and hypothesised that species abundance will be greater on the artificial rockpools compared to the adjacent seawall.

2. Method and materials

The site was based in Hamble Harbour on the River Hamble, Southampton Water on the south coast of England (Fig. 1), a highly modified, muddy estuary very popular for sailing with over 3000 moorings. It is sheltered and protected from prevailing south-westerly winds by the Isle of Wight and its location within Southampton Water. Nearby intertidal habitats are primarily soft-sediments and those provided by artificial structures, such as seawalls. The rockpools were affixed to a > 100-year-old intertidal vertical concrete seawall which in some areas, due to its age, had a slight positive camber. The seawall is western-facing and can be accessed at low tide on foot (Fig. 2). Visual assessment of the seawall biotic community prior to rockpool installation indicated it was species poor, with some old *Ascophyllum nodosum* growth dominating the bottom quarter of the wall. At the seawall toe was thick mud and occasional rubble which provided attachment substrate for fucoids and *A. nodosum*. Permission to install the rockpools was provided by the asset owners Warsash Sailing Club and Hamble Harbour Authority.

The artificial rockpools were handmade by *Artecology Ltd*. using Vicat Prompt cement, sharp sand ballast (≤10 mm) and potable water. They were pre-cast in standardised moulds to the dimensions shown in Fig. 3. The rim and rockpool interior were made with a stippled texture, and the exterior was finished with concave hemispherical pockets achieved using a bubble wrap form liner. A short video of the rockpool manufacture may be viewed here (https://youtu.be/mX0YXjWotWE?si=ubOEqQcCeMfpMh6p).

Prior to installation the seawall substrate was scrubbed clear of fouling organisms to allow the rockpools to sit flush against the seawall surface. A stainless-steel bracket and hardware were used to fix the rockpools to the seawall in eight columns of three, with the bottom rockpool at mean tide level, the top rockpool at high water neap tide level, and the middle rockpool installed equidistant between both top and bottom rockpools. A short video of the rockpool installation method may be viewed here. Each column was installed within a section of

Table 2

Main test results for abiotic factors salinity, temperature, water depth and sediment depth. Significant values show in bold.

Factor	numDF	denDF	F-value	*p*-Value
Salinity				
Tidal level	2	21	1.55	0.2366
Survey interval	3	63	172.73	<**0.0001**
Level:interval	6	63	0.89	0.5095
Temperature				
Tidal level	2	21	22.84	<**0.0001**
Survey interval	3	63	263.15	<**0.0001**
Level:interval	6	63	4.05	**0.0017**
Water depth				
Tidal level	2	21	3.051	0.0687
Survey interval	3	63	9.611	<**0.0001**
Level:interval	6	63	3.031	**0.0114**
Sediment depth				
Tidal level	2	21	46.39332	<**0.0001**
Survey interval	3	63	58.74117	<**0.0001**
Level:interval	6	63	8.20152	<**0.0001**

Table 3

Species recorded in the rockpool interiors, on the rockpool exteriors and on the seawall. Mean abundance given with standard deviation (±) given in brackets. Non-native taxa indicated with an asterisk (*). Taxa indicated with a dagger (†) denote those recorded as numeric counts, all other taxa were recorded as percentage cover.

	Rockpools interior	Rockpools exterior	Seawall
Brown seaweeds			
Ascophyllum nodosum	0	0	15.1 (±28.4)
Fucus spiralis	5.7 (±16.6)	1.9 (±7.5)	1.5 (±5.7)
Fucus vesiculosus	6 (±20.6)	2.7 (±9.8)	4.5 (±13.5)
Pylaiella littoralis	14.6 (±23)	0.1 (±1.5)	0
Red seaweeds			
Catanella sp.	0.01 (±0.07)	0	0.08 (±0.7)
Ceramium sp.	0.2 (±2.2)	0.01 (±0.09)	0
Dumontia contorta	0.02 (±0.1)	0	0
Porphyra sp.	0.01 (±0.1)	0.1 (±0.3)	0.003 (±0.05)
Polysiphonia sp.	0.1 (±0.7)	0	0
Green seaweeds			
Blidingia minima	0	28.2 (±41.5)	16.7 (±27.2)
Cladophora sp.	2 (±11.9)	0	0
Diatoms	0.01	7.3 (±26.0)	0
Ulva spp.	11.2 (±18.4)	14.5 (±31.8)	0.2 (±1)
Porifera			
Halichondrea panacea	0.01 (±0.07)	0	0
Sycon ciliatum†	0.01 (±0.1)	0	0
Hydrozoa			
Clava multicornis	0.01 (±0.07)	0	0
Annelida			
*Ficopomatus enigmaticus**	0.1 (±0.4)	0	0
Spirobis spirobis	0.04 (±0.2)	0	0
Spirobranchus triqueter	0.1 (±0.2)	0	0
Crustacea			
Amphipoda sp.†	0.1 (±0.5)	0	0
Anurida maritima†	0.01 (±0.1)	0	0
*Austrominius modestus**	0.02 (±0.1)	1.1 (±4.3)	0.6 (±1.8)
Carcinus maenas†	0.3 (±0.6)	0.04 (±0.3)	0
Corophium volutator†	0.01 (±0.07)	0	0
Gammaridae sp.†	0.01 (±0.1)	0	0
Ligia oceanica†	0.1 (±0.3)	0.07 (±0.4)	0
Palaemon sp.†	1.6 (±3.3)	0	0
Semibalanus balanoides	0.01 (±0.07)	0.02 (±0.1)	0
Mollusca			
Hydrobidae sp.†	0.02 (±0.1)	0	0
Littorina littorea†	0.1 (±0.5)	0.04 (±0.2)	0.02 (0.2)
Littorina obtusata†	0.03 (±0.2)	0.02 (±0.2)	0
Littorina saxatalis†	0	0.01 (±0.07)	0
Mytilus edulis†	0.1 (±0.4)	0	0
Steromphala umbilicalis†	0.01 (±0.07)	0.01 (±0.07)	0
Vertebrata			
Lipophrys pholis†	0.02 (±0.2)	0	0
Total taxa:	32	16	9

seawall separated by equally spaced buttresses to maintain independence. Installation was completed in October 2020.

Monitoring surveys were undertaken in the early to mid-afternoon on an ebbing tide at 1-, 6-, 9-, 12-, 15-, 18-, 21- and 24-month intervals following completion of installation. During surveys, species abundance data was obtained for each rockpool through visual assessment of percentage cover and numeric counts of organisms to species level wherever possible, with the rockpool interior (pool of water) and exterior (rockpool underside) surveyed separately (Table 1). The size of the quadrat used (25 cm^2) was based on the total surface area of the rockpool interior basin which was approximately 25 cm^2. All macrofaunal taxa within the rockpool interior basin were recorded, including sessile taxa and mobile taxa present in the water column such as prawns. The quadrat for the rockpool exterior covered most of each half of side/underside surface area, and so the same area was surveyed for each interval.

High resolution photos were taken using a Sony A7R3 CDC camera (Sony FE 24-70 mm zoom lens) permitting high quality data collection of photo-quadrats. For each rockpool a photo-quadrat of the seawall was taken either side of the rockpool at the same tidal level approximately 15 cm from the rockpools (Fig. 4).

During surveys, the seawall was characterised with high abundances of *Ascophyllum nodosum* at lower tidal levels and *Ulva* spp., *Blidingia minima* at higher tidal levels, but otherwise consisted of bare concrete and no understorey algal turf was observed. Due to a high abundance of canopy-forming algae obscuring the underlying seawall at the lower tidal level, it was not possible to ascertain the presence of mobile fauna from seawall photo-quadrats and therefore mobile fauna, such as snails and limpets, have been excluded from data analysis. All results use data derived from sessile species abundance only. A list of mobile fauna species identified in the artificial rockpools was collated separately.

After it became evident that sediment accretion was occurring in some of the rockpools, extra care was taken not to disrupt the sediment during surveys to avoid disturbance to infauna. At the final 24-month interval survey all sediment was removed from each rockpool ($n = 24$) for macrofaunal analysis and stored in formalin in labelled watertight zip-lock bags for analysis. Sediment samples were later rinsed of formalin under a fume hood and macrofauna were picked under a Leica stereomicroscope and identified to highest taxonomic resolution. To determine how much biomass grows on the rockpools compared to the seawall, percentage cover was used to convert existing dry weight values for algae species at the 24-month survey interval only. Dry weight values and algal species used can be found in Supplementary material.

Abiotic data were collected at 6-, 12-, 18- and 24-month intervals. A YSI multimeter was used for each rockpool to determine salinity (ppt) and temperature (Celsius). A ruler was used to measure water and sediment depth.

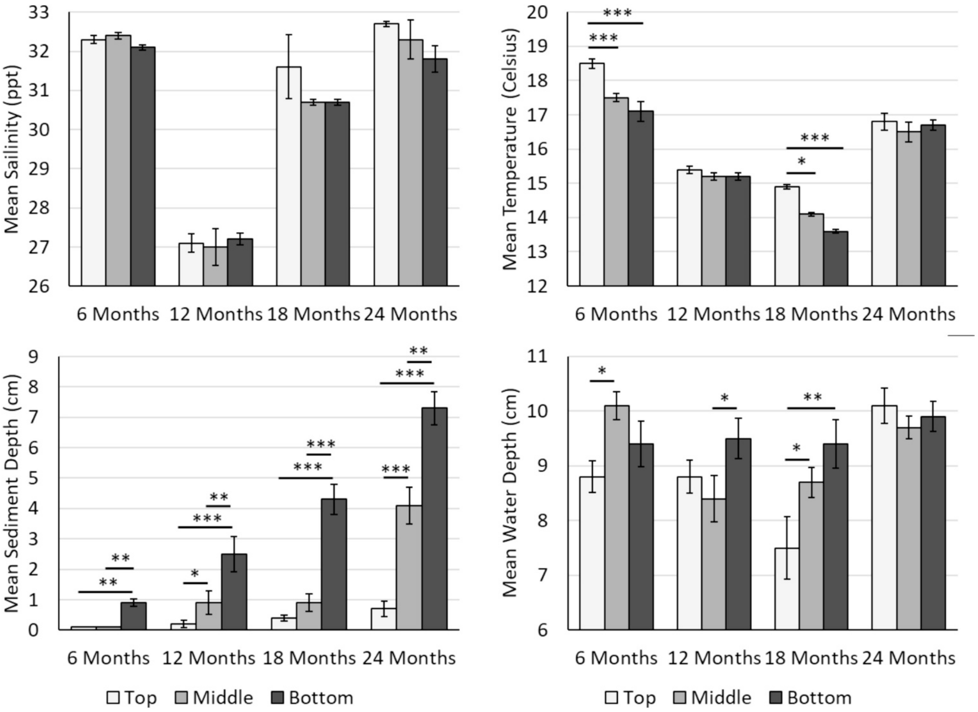

Fig. 6. Abiotic factors (top left, clockwise) salinity, temperature, water depth and sediment depth. Error bars show standard error. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Six, 12, 18 and 24 months correspond to April 2021, October 2021, April 2022 and October 2022 respectively.

Table 4
Main test results for species richness for rockpool interiors and exteriors and seawall. Significant values show in bold.

Factor	numDF	denDF	F-value	p-Value
Rockpool exterior and seawall				
Habitat	1	66	6.3426	**0.0142**
Tidal level	2	66	44.0404	<**0.0001**
Survey interval	7	654	94.0442	<**0.0001**
Habitat:level	2	66	7.2603	**0.0014**
Habitat:interval	7	654	90.6078	<**0.0001**
Level:interval	14	654	5.2773	<**0.0001**
Habitat:level:interval	14	654	2.0513	**0.0127**
Rockpool interior and seawall				
Habitat	1	66	24.4711	<**0.0001**
Tidal level	2	66	35.0013	<**0.0001**
Survey interval	7	462	7.9857	<**0.0001**
Habitat:level	2	66	10.9943	**0.0001**
Habitat:interval	7	462	18.4841	<**0.0001**
Level:interval	14	462	4.2027	<**0.0001**
Habitat:level:interval	14	462	1.6539	0.0622

2.1. *Data analysis*

To test for statistically significant differences in percent cover, numeric counts, species richness, and between the rockpools and seawall over time, linear mixed effect models were run using the “nlme” package (Pinheiro et al., 2020) in R Studio (Version 1.2.1335). Only sessile species were used in statistical analysis to limit underestimation of species that photo-quadrats may have missed, such as mobile fauna. Assumptions of statistical tests were verified by examination of residuals against fitted model plots, as per Zuur et al. (2009). Where clear lack of normality or heteroskedasticity were identified, transformations were made to the data (log+1 transformations) before further statistical analysis. Habitat (rockpool interior, rockpool exterior and seawall), Level (top, middle, bottom) and Intervals (1-, 6-, 9-, 12-, 15-, 18-, 21- and 24-months) were fixed factors. To account for repeated measures within the rockpool and seawall, as the same rockpool and seawall photo-quadrats were surveyed each time, rockpool and photo-quadrat ‘ID’ was included as a random factor. To test for statistically significant differences in biomass between the habitats over different tidal levels using only the 24-month data, generalised linear models (GLM) were run with Habitat (rockpool interior, rockpool exterior and seawall) and Level (top, middle, bottom) as fixed factors. Quasi-Poisson distribution was used as data were over dispersed (Crawley, 2012). Pairwise tests were run using the “emmeans” package (Lenth, 2021).

Plymouth Routines in Multivariate Ecological Research (Primer-e v.7) was used to perform individual PERMANOVAs to test for differences in assemblage structure between Habitat, Level and Habitat * Level using sessile species abundance data from 24-month interval only (Anderson, 2005). Data were square root transformed to avoid the weighting of common species over rare. A Bray–Curtis resemblance matrix was used with 9999 permutations and PERMANOVA run with unrestricted permutation of raw data. Significant results were followed by post hoc tests to determine if factors were significantly different. Multidimensional scaling (MDS) plots were used to visually demonstrate assemblage similarity between Habitat * Level.

3. Results

After two years in the intertidal environment, all rockpools were present and in good repair, despite significant storms in the intervening years since installation (Fig. 5). On the middle and top seawall sections, it was possible to ascertain the presence of mobile fauna due to the very low abundance of canopy-forming seaweed. Only 1 *Littorina littorea* was recorded on the seawall. In and on the rockpools, 13 mobile species were recorded (Table 3).

3.1. *Abiotic*

Salinity (ppt) ranged from 23.8 ‰ to 34.9 ‰ and was generally higher in the top rockpools. Main tests showed salinity did not significantly differ between rockpool levels, or level*interval, but it did

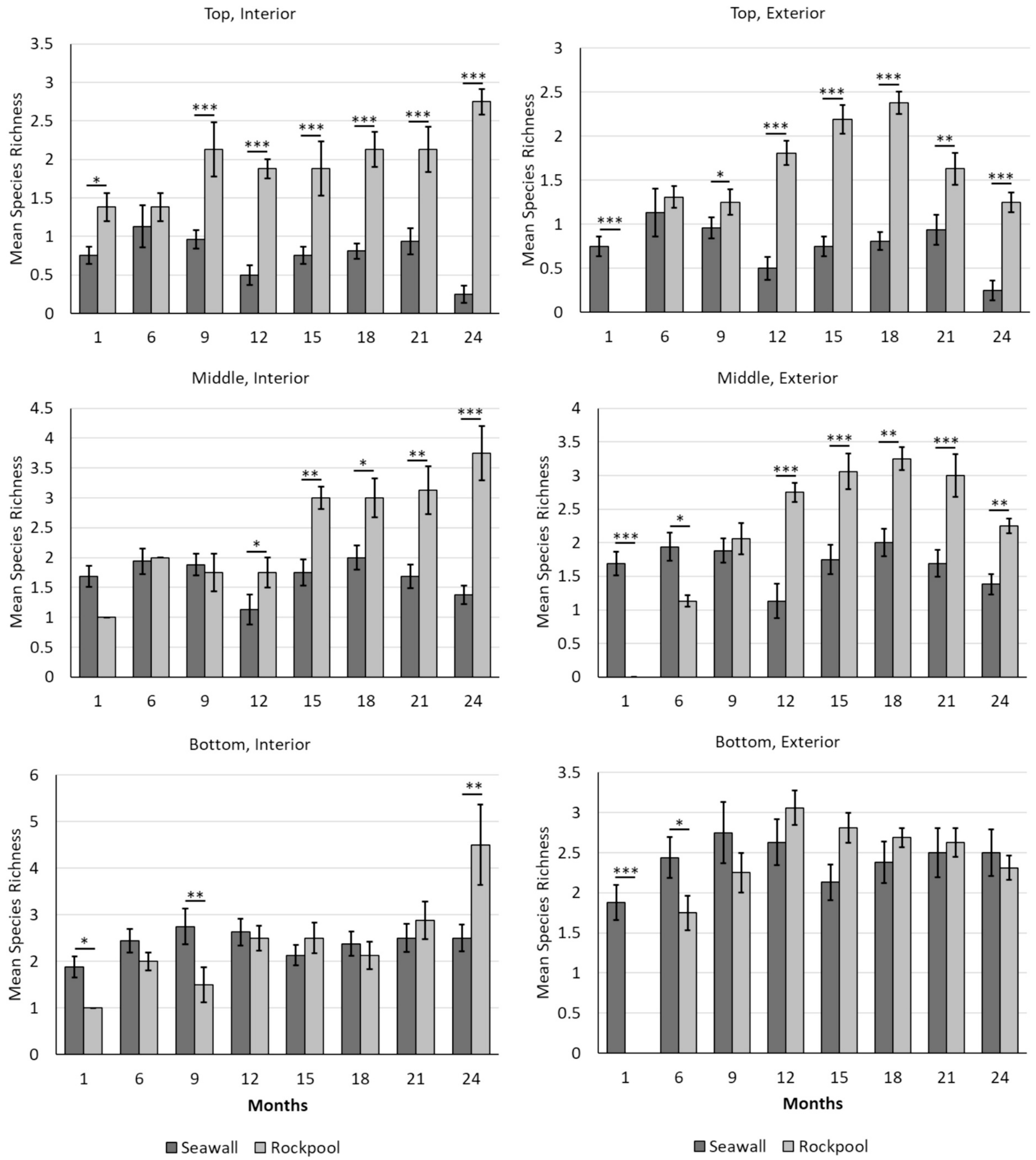

Fig. 7. Mean species richness for rockpool interiors and seawall (left column) and rockpool exteriors and seawall (right column). Mean species richness is per 25 × 25 cm quadrat. Error bars show standard error. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). One, 6, 9, 12, 15, 18, 21 and 24 months correspond to November 2020, April 2021, July 2021, October 2021, January 2022, April 2022, July 2022 and October 2022 respectively.

between intervals (Table 2). Temperature ranged from 13.4 °C to 19.0 °C and was again generally slightly higher in the top rockpools (Fig. 6). At the 6- and 18-month interval (both April), the temperature was significantly greater in the top rockpools compared to the middle and lower rockpools, but not at the 12- and 24-month intervals (both October).

Water depth ranged from 4.6 to 11.5 cm and varied between levels and between intervals, with no clear trend. Water depth in the middle rockpools was significantly greater than the top rockpools on two occasions, and significantly greater in the bottom rockpools than the top and the middle rockpools on one occasion. Water depth varied between rockpool levels the least at the final survey interval of 24 months. The maximum sediment depth recorded was 8.5 cm, and there is a positive temporal trend with sediment accretion increasing every interval for the bottom rockpools. Sediment accretion does not increase beyond 1 cm in the middle rockpools until 24 months, when the mean sediment height increases to 4 cm. Mean sediment height in the top rockpools does not exceed 1 cm throughout the study period. The bottom rockpool sediment depth is significantly greater than middle and top rockpool sediment depth for all intervals, with middle rockpool significantly greater than the top rockpool at 12- and 24-months.

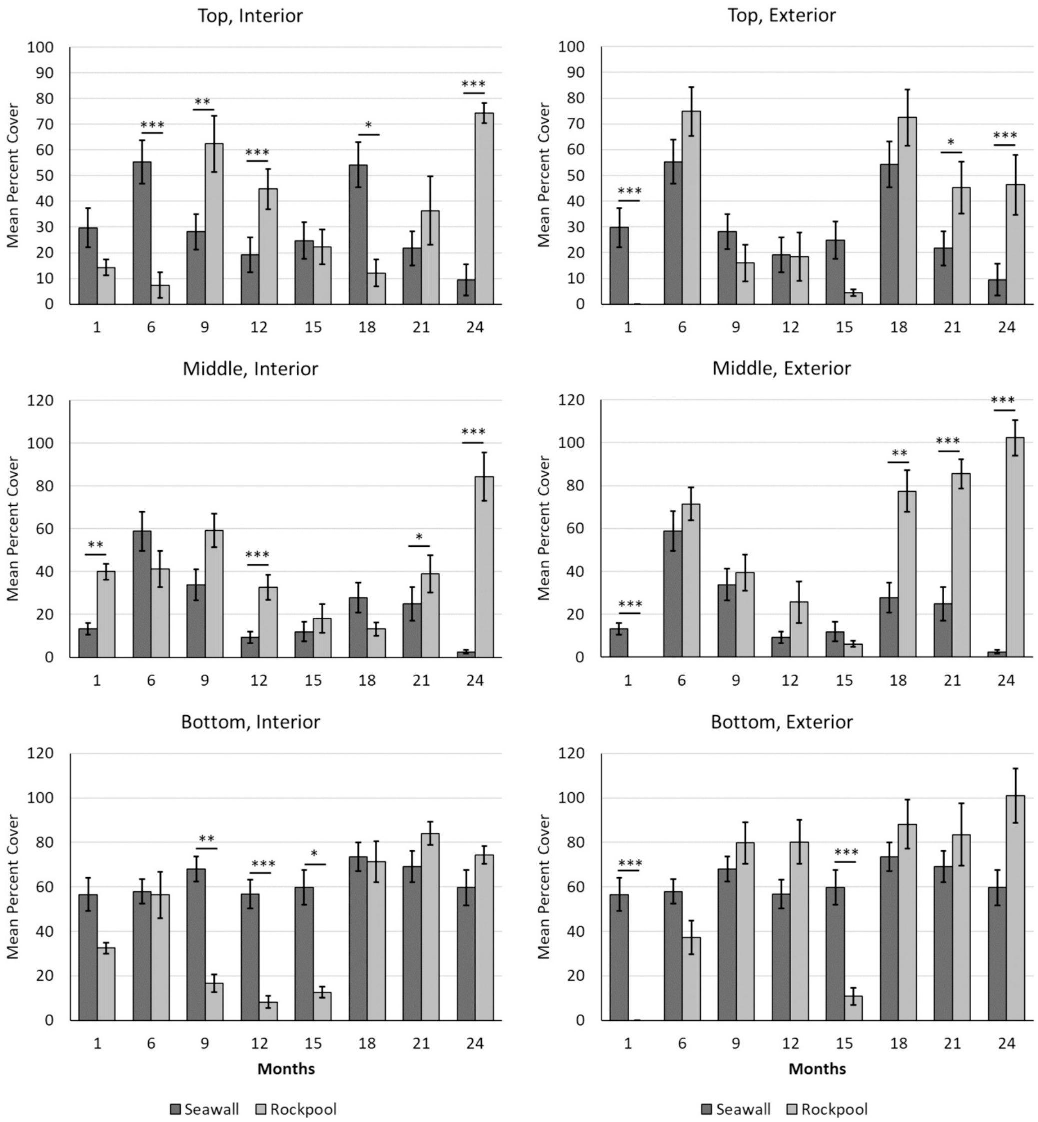

Fig. 8. Percent cover for rockpool interiors and seawall (left column) and rockpool exteriors and seawall (right column). Error bars show standard error. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001).

Table 5
Main test results for percentage cover for rockpool interiors and exteriors and seawall. Significant values show in bold.

Factor	numDF	denDF	F-value	p-Value
Rockpool exterior and seawall				
Habitat	1	66	0.0018	0.9662
Tidal level	2	66	20.3074	<**0.0001**
Survey interval	7	654	65.448	<**0.0001**
Habitat:level	2	66	4.6499	**0.0129**
Habitat:interval	7	654	55.2333	<**0.0001**
Level:interval	14	654	9.5191	<**0.0001**
Habitat:level:interval	14	654	2.2834	**0.0047**
Rockpool interior and seawall				
Habitat	1	66	4.4793	**0.0381**
Tidal level	2	66	19.4243	<**0.0001**
Survey interval	7	462	9.6703	<**0.0001**
Habitat:level	2	66	6.6515	**0.0023**
Habitat:interval	7	462	18.6644	<**0.0001**
Level:interval	14	462	3.0848	**0.0001**
Habitat:level:interval	14	462	7.8868	<**0.0001**

Table 6
PERMANOVA main test results after 24 months for assemblages for rockpool interiors, exteriors and seawall. Significant values show in bold.

	Source	df	SS	MS	Pseudo-F	P (perm)
Rockpool exterior and seawall	Level	2	55,950	27,975	21.886	**0.0001**
	Habitat	1	47,356	47,356	37.048	**0.0001**
	Level * habitat	2	24,563	12,282	9.6082	**0.0001**
Rockpool interior and seawall	Level	2	45,464	22,732	25.388	**0.0001**
	Habitat	1	57,031	57,031	63.693	**0.0001**
	Level * habitat	2	27,602	13,801	15.413	**0.0001**

3.2. *Species richness*

The total number of taxa recorded in the rockpools overall was 37, including mobile and sessile fauna and 3 species that were found in the sediment only. Thirty-one taxa were recorded in the rockpool interiors, and 16 on the rockpool exteriors. Two non-native species were identified; the barnacle *Austrominius modestus*, which occurred on both the

Table 7
PERMANOVA post-hoc test results after 24 months for assemblages for rockpool interiors, exteriors and seawall. Significant values show in bold.

Level	Groups	t	P (perm)	Unique perms	Average similarity
Top	Exterior * seawall	1.7809	0.0547	9901	61.3 %
Middle	Exterior * seawall	2.443	**0.0018**	9945	40.1 %
Bottom	Exterior * seawall	5.566	**0.0001**	9945	12.2 %
Top	Interior * seawall	9.7818	**0.0001**	9951	21.4 %
Middle	Interior * seawall	8.5799	**0.0001**	9941	19.3 %
Bottom	Interior * seawall	9.3761	**0.0001**	9943	17.7 %

seawall and the rockpools, and *Ficopomatus enigmaticus*, a calcareous tubeworm that occurred in the rockpools only (Table 3). Both species have been recorded elsewhere in the harbour (unpublished data). The climate migrant *Steromphala umbilicalis* was also recorded, which has only colonised this region of the UK coast within the past 20 years (Herbert 2023, personal communication). The top rockpool interiors were generally the most species poor, with species richness increasing inversely with the tidal level of rockpools. This trend was not reflected on the rockpool exteriors. Twenty-one sessile taxa were recorded in and on the rockpools, compared to 9 sessile taxa on the seawall.

Main tests showed significant results for all factors and interactions for both rockpool interior and seawall and rockpool exterior and seawall, except for habitat:level:interval for rockpool exterior and seawall (Table 4). Mean species richness peaked at 24 months for the rockpool interiors at all levels and at 18 months for the top and middle rockpool exteriors and 12 months for the bottom rockpool exteriors (Fig. 7). From about 12 months onwards, the top and middle rockpool interiors and exteriors are significantly more species rich than the adjacent seawall. For the bottom rockpools, species richness is only significantly greater than the seawall in the rockpool interiors at 21 months. Seawall species richness is only significantly greater than rockpool species richness on seven occasions, and this predominantly occurs when comparing the seawall to the rockpool exterior during the first two survey intervals. Seawall species richness does not significantly exceed that of the rockpools after 9 months.

3.3. Abundance

Sessile percentage cover was highly variable throughout the study period with no linear trend observable (Fig. 8). However, the pattern of abundance for both rockpool interior and exterior broadly follows a seasonal succession pattern, with initial colonisation of the rockpools dipping around the 12-month interval (October 2021) as boreal autumn and winter occurs. Percentage cover then increases from approximately 18 months (April 2022) onwards as new settlement and growth occurs, particularly perennial and slower growing species such as fucoid algae.

Main tests showed significant results for all factors and interactions for both rockpool interior and seawall and rockpool exterior and seawall, except for habitat for rockpool exterior and seawall (Table 5). For the bottom rockpools, abundance on the seawall is significantly higher at 9-, 12-, and 15-months than for the rockpool interior and at 1- and 15-months on the rockpool exterior. At no point is abundance within the rockpool interiors and exteriors higher than the seawall. However, abundance in the rockpools is significantly greater than the seawall for the middle and top rockpools, particularly from 21-months onwards.

3.4. Assemblages

The seawall was dominated by the green algae *B. minima* and *A. nodosum*, and this remained stable over the course of the study. The exterior of the middle and lower rockpools were dominated by *Fucus spiralis* and *Fucus vesiculosis* on the rockpool rim and the barnacle *A. modestus* on the shaded underside. The upper rockpool exteriors were relatively devoid of fucoid algae and dominated by *B. minima*. All rockpool interiors were dominated by the brown filamentous algae *Pylaiella littoralis*, though the proportion of its dominance decreased with rockpool tidal level.

After 24-months, PERMANOVA main tests (Table 6) indicated that assemblage structures between both rockpool interiors and exteriors was significantly different to the seawall at tidal level, habitat and level*habitat. Post hoc tests (Table 7) show only the assemblages on the top rockpool exterior were not significantly different to the seawall and had an average similarity of 61.3 %, which can be attributed to the low species richness and shared dominance of the green algae *B. minima*. Otherwise, all other results were significant (Table 7). Rockpool interiors and seawall overall shared less average similarity than rockpool exteriors and the seawall. Average similarity between rockpool habitat and the adjacent seawall assemblages decreased linearly with tidal height. Grazing halos around the rockpools were observed (Fig. 5) but the organism responsible was not identified.

The MDS plots (Fig. 9) for the rockpool interiors and seawall indicate that the assemblages of each habitat remain consistently divergent throughout the study period. The similarity between the rockpool interiors appears greater than the similarity between the seawall quadrats, even across tidal levels. By 24-months, the bottom rockpool interior assemblages appear most similar to the seawall, whereas the rockpool interiors of other tidal levels remain more distant from their seawall counterparts. However, the MDS plots for the rockpool exteriors show greater similarity to the seawall throughout.

3.5. Sediment infauna

Ten taxa were recorded in the rockpool sediment (Table 8) at 24 months. The species identified are fairly typical of an intertidal estuary, such as shore crabs *Carcinus maenas* and prawns *Palaemon* sp., but infaunal species, such as bivalves and polychaete worms, were rare. Mean species richness increased inversely with tidal height, with the most species occurring in the bottom rockpools (Fig. 10). The sediment was mostly comprised of fine mud, with the occasional empty mussel shell or crab moult.

3.6. Biomass

After 24 months, main tests show significant differences for all factors for mean dry weight of sessile biota (Table 9), with tidal level contributing to variance the most (48.5 %). The biomass on the rockpools significantly exceeds that on the seawall for middle level, but on the bottom level, biomass is significantly greater on the seawall (Fig. 10). The mean dry weight of the bottom rockpools is the same as a single 25 × 25 cm quadrat on the adjacent seawall but the biomass of the seawall is more variable than that found on the rockpools. Mean biomass for the rockpool interior and exterior is relatively equal across tidal heights, each approximating half of the biomass for the whole rockpool (Fig. 11).

4. Discussion

This study demonstrates that from mean tide level to high water neaps, artificial rockpools can enhance the species abundance of concrete seawalls and the magnitude of this impact is greater at higher tidal levels where refugia are more fundamental to survival. Chapman and Blockley (2009) found similar results, with the addition of pools on intertidal seawalls at lower shore levels less different to the seawall in terms of species richness than those at mid and upper shore. The addition of a pool created a new assemblage not found on the seawall in

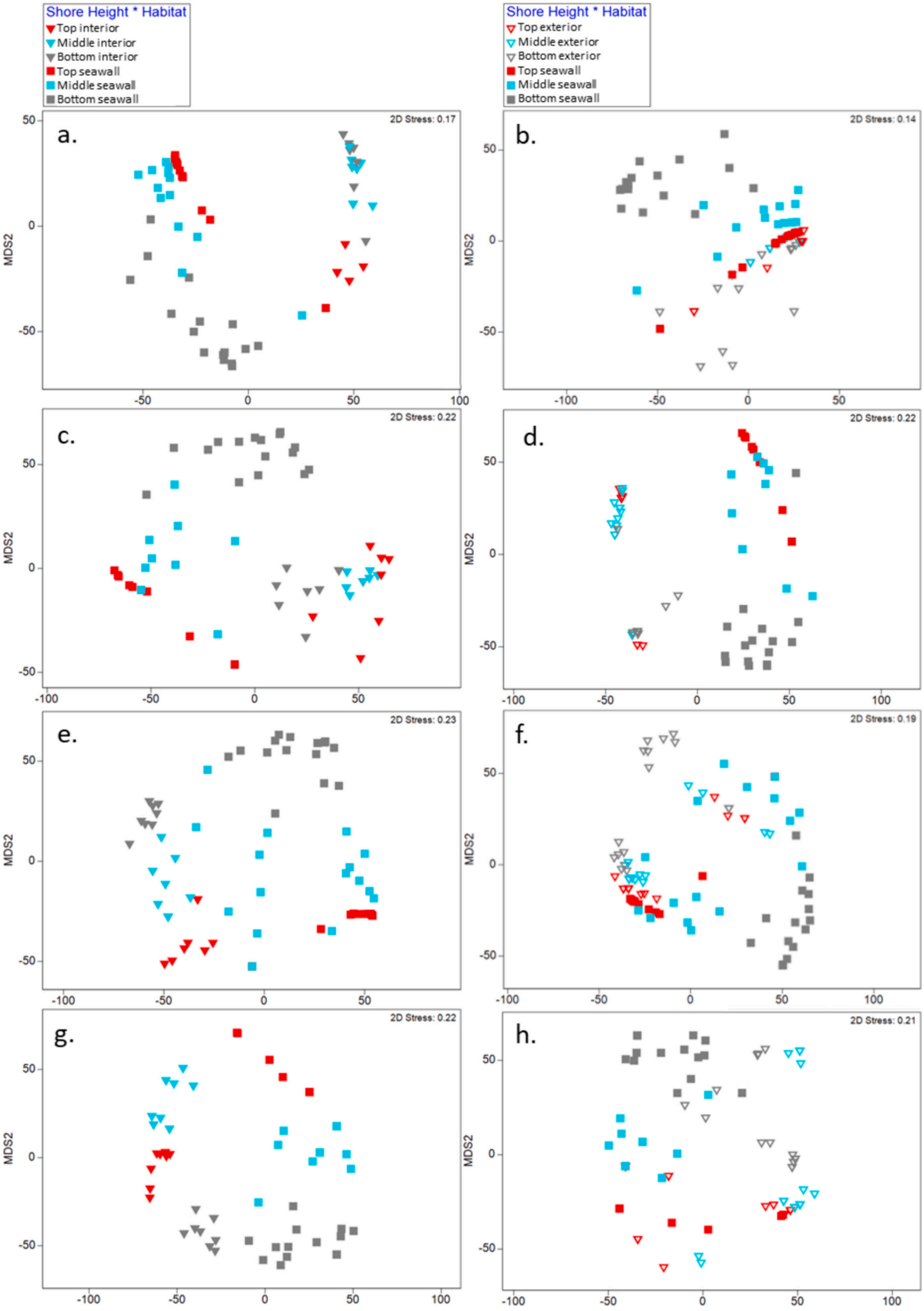

Fig. 9. Multidimensional scaling plots indicating rockpool interiors and seawall (left column) and rockpool exteriors and seawall (right column), with 6 months (a., b.), 12 months (c., d.), 18 months (e., f.), and 24 months (g., h.) using species abundance data. Triangles correspond to rockpools and squares correspond to the seawall. Red indicates top level, blue indicates middle level and grey indicates bottom level.

Table 8
Species recorded in the rockpool sediment at 24 months with mean abundance given.

	Top	Middle	Bottom
Annelida			
Hediste diversicolour	0.0	0.0	0.1
Polychaeta sp.	0.0	0.1	0.3
Crustacea			
Carcinus maenas	0.3	0.3	0.4
Palaemon sp.	0.5	2.1	5.3
Mollusca			
Bivalve	0.0	0.3	0.0
Hydrobidae sp.	4.6	1.4	9.4
Littorina littorea	0.0	0.3	0.3
Littorina obtusata	0.3	0.9	3.1
Mytilus edulis	0.0	0.0	0.1
Vertebrata			
Lipophrys pholis	0.0	0.0	0.1
Total taxa:	4	7	9

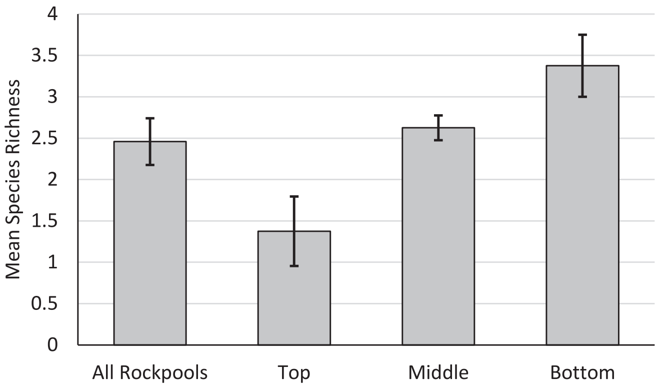

Fig. 10. Mean species richness of rockpool sediment at 24 months ($n = 24$). Error bars show standard error.

Table 9
Main test results for biomass on the rockpools and seawall after 24 months. Bold values indicate significant result.

Factor	Df	Deviance	Resid. Df	Resid. Dev	*P*	% Explained
Biomass						
Habitat	2.0	118.8	93.0	1331.9	<**0.0001**	8.2
Tidal level	2.0	703.0	91.0	628.8	<**0.0001**	48.5
Habitat * level	4.0	211.3	87.0	417.5	<**0.0001**	14.6

Hamble Harbour, in addition to sediment accretion that further provided habitat for a small number of infaunal taxa.

In the final surveys, species abundance for rockpool exterior and the seawall followed a similar pattern which was likely due to the high coverage of fucoid algae, predominantly *Fucus spiralis* and to a lesser extent *Fucus vesiculosus*, that developed on the rockpools. Species richness was greatest in the middle and bottom rockpools, which may be influenced by the less extreme temperature and salinity values recorded during the study. This reflects trends observed in natural (Raffaelli and Hawkins, 1996; Little et al., 2009; Legrand et al., 2018) and artificial rockpools (Firth et al., 2013). Further, species richness was significantly greater in the top rockpool interiors compared to the seawall just one month post instalment, which highlights the immediate benefits of refugia in the environmentally challenging artificial upper shore. These data suggest that artificial rockpools can supply ecological benefits across tidal levels and optimal placement of ecological enhancements is dependent on the desired outcome.

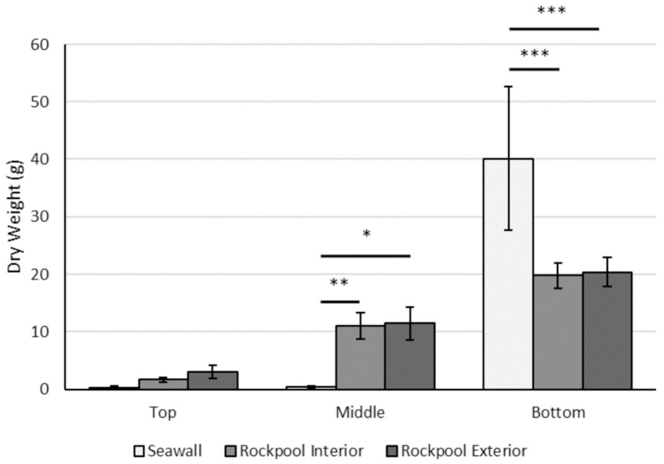

Fig. 11. Mean dry weight (g) of organisms for rockpool exterior and interior at 24 months. Error bars show standard error. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001).

It is clear from these results that artificial rockpools can facilitate the vertical elevation of sessile organisms that are typically found in the lower shore or recorded only at the base of the seawall, which is reflective of the existing literature (Hall et al., 2019). The brown canopy-forming seaweed *A. nodosum* was restricted to the lower seawall, but was recorded on rockpools of all levels, though rarely on the top rockpools. The succession of opportunistic green algae to dense fucoid canopy on the middle and lower rockpools was also reported in Hall et al. (2019). The holdfasts of seaweeds *F. spiralis* and *F. vesiculosus* on the rockpools were observed almost exclusively on the rockpool rim. The horizontal, rough-textured rim surface likely provided an ideal attachment surface (Fletcher and Callow, 1992) with good sunlight exposure, and this topographical pattern of fucoid colonisation has been observed on similar artificial rockpools elsewhere in the UK (Drakard et al., 2023). The incorporation of these features has facilitated the vertical migration of fucoid seaweed further up the tidal zone where it was otherwise absent. The vertical arrangement of the rockpools may have also impacted the assemblages that formed on them, due to varying exposure to sunlight. However, the aspect of the seawall meant that the shadows created by overhanging rockpools were usually cast on the adjacent seawall as opposed to rockpools below (Fig. 5).

The rockpools were also able to support more faunal species than the seawall, with a wide range of morphologies and traits, such as the delicate hydroid *Clava multicornis*. Most sessile faunal species were found exclusively in the rockpools, with only the non-native barnacle *Austrominius modestus* recorded on the seawall. Browne and Chapman (2014) found similar results, with only 2 sessile species on a vertical seawall in Sydney, but 7 in their bolt-on rockpools. Although mobile fauna was excluded from data analysis, they were abundant in the rockpools, particularly crabs *Carcinus maenas* and prawns *Palaemon* sp. Mobile organisms found in the rockpools, such as prawns and shanny *Lipophrys pholis*, would not survive on the seawall without water retaining features. The rockpools may also facilitate a halo effect (Fairweather, 1988; Johnson et al., 1998), where grazers that would otherwise struggle to survive on a vertical seawall lacking microhabitats (Chapman, 2006; Moreira, 2006; Jackson et al., 2008) can find refuge at low tide and graze the immediate surrounding seawall, impacting the assemblage composition of the structure they're fixed to. In a warming climate, fucoid cover may be of particular value to indigenous Northern species, such as *Patella vulgata* (Hawkins et al., 2008).

Sediment had accumulated in most of the middle and bottom rockpools by the end of the study, which has been recorded in other artificial rockpools (Firth et al., 2016; Waltham and Sheaves, 2018; Bone et al., 2022). Previous work on a similar model artificial rockpool on the south UK coast has indicated that retained sediment can successfully host an infaunal assemblage comparable to that of a disturbed estuary (Bone et al., 2022), but that was not realised in this study. Although 3 infaunal taxa (*Hediste diversicolor*, unidentified polychaete, unidentified bivalve) were recorded, their abundance was rare, and the assemblage was instead dominated by crabs and prawns. The deposited mud was not particularly deep ($\leq$ 7 cm) or compact, and so the low volume may have reduced its ability to provide a habitat analogous to a mudflat. The presence of crabs and fish in the sediment suggest it may get bioturbated by their movements and predation of infauna by crabs may limit their capacity to proliferate. However, the unintended retention of mud still plays a role as shelter, as crabs would bury themselves within the retained mud when disturbed during surveys.

4.1. *Application*

Sea levels are rising and will continue to rise (IPCC, 2022), combining with coastal development and land reclamation (Dugan et al., 2011; Duarte et al., 2013; Duarte, 2014) which results in coastal squeeze and intertidal habitat loss (Bugnot et al., 2021). Sea level rise scenarios (IPCC, 2022) indicate that the top rockpools installed at high water neaps may in future be at lower tidal levels and will continue to deliver sufficient habitat for intertidal fauna where there previously was none. Spreading eco-engineering interventions across the vertical tidal zone will ensure that intertidal habitat remains available in the future, providing vital steppingstones to species' survival and ecological resilience. This also emphasises the requirement for interventions to be integrated into coastal development and engineering at the design and planning phase, as bolt-on interventions may not possess the required multidecadal longevity due to their often-protruding design and elevated risk of dislodgement. For example, Browne and Chapman (2014) lost several bolt-on 'flowerpot' artificial rockpools to wave action in Sydney Harbour, Australia.

To facilitate straightforward and accurate surveying, the rockpools in this study were devoid of macroscale features. In future, their design could be optimised by adding overhangs or ledges on the rim to create shaded areas, and a deeper pool to ameliorate extreme temperature values. This is particularly important in tropical regions, where rockpools have a higher risk of become ecological traps due to exceeding species' thermal limits during heatwave events (Vinagre et al., 2018). Browne and Chapman (2014) found intertidal assemblages in rockpools with two different depths were not significantly different on the upper shore, and so the impact of rockpool depth may not be as great on upper shore levels. However, design choices can be made to meet a variety of ecological needs and engineering standards. Consultation and collaboration with the appropriate experts are fundamental to implementing successful and climate resilient eco-engineering interventions.

5. Conclusion

This study has demonstrated that artificial rockpools can provide valuable hard and soft substrate habitats between high water neaps and mean tide level on a concrete seawall by retaining water and sediment. The rockpools provided crucial refugia to a wide range of species that were not otherwise present on the seawall and extended the vertical elevation of habitat-forming species such as canopy-forming brown seaweed. Installation of rockpool interventions should be incorporated across the tidal zone, including the upper shore where their presence will be of greater importance in the coming decades due to climate-change induced sea level rise and warming. Retrofitted, bolt-on rockpools have adequately demonstrated proof of concept and so future interventions should be integrated into the initial design phase of coastal infrastructure, negating the requirement for retrofitting.

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CRediT authorship contribution statement

Jessica R. Bone: Writing – review & editing, Writing – original draft, Visualization, Validation, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **Rick Stafford:** Writing – review & editing, Validation, Supervision, Methodology, Formal analysis. **Alice E. Hall:** Writing – review & editing, Supervision, Formal analysis. **Roger J.H. Herbert:** Writing – review & editing, Validation, Supervision, Methodology, Investigation, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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References

Aguilera, M.A., Broitman, B.R., Thiel, M., 2014. Spatial variability in community composition on a granite breakwater versus natural rocky shores: lack of microhabitats suppresses intertidal biodiversity. Mar. Pollut. Bull. 87 (1–2), 257–268.

Aguilera, M.A., Bulleri, F., Thiel, M., 2022. Weak effects of age but important role of microhabitats in community differences between breakwaters and natural rocky shores across a latitudinal gradient. Glob. Ecol. Biogeogr. 0, 1–13.

Airoldi, L., Abbiati, M., Beck, M.W., Hawkins, S.J., Jonsson, P.R., Martin, D., Moschella, P.S., Sundelof, A., Thompson, R.C., Aberg, P., 2005. An ecological perspective on the deployment and design of low-crested and other hard coastal defence structures. Coast. Eng. 52 (10–11), 1073–1087.

Anderson, M.J., 2005. Permanova Permutational Multivariate Analysis of Variance, a Computer Program. University of Auckland, New Zealand.

Baxter, T., Coombes, M., Viles, H., 2023. Intertidal biodiversity and physical habitat complexity on historic masonry walls: a comparison with modern concrete infrastructure and natural rocky cliffs. Mar. Pollut. Bull. 188, 114617.

Benedetti-Cecchi, L., Bulleri, F., Cinelli, F., 2000. The interplay of physical and biological factors in maintaining mid-shore and low-shore assemblages on rocky coasts in the north-west Mediterranean. Oecologia 123, 406–417.

Bergen, S.D., Bolton, S., Fridley, J., 2001. Design principles for ecological engineering. Ecol. Eng. 18 (2), 201–210.

Bishop, M.J., Vozzo, M.L., Mayer-Pinto, M., Dafforn, K.A., 2022. Complexity-biodiversity relationships on marine urban structures: reintroducing habitat heterogeneity through eco-engineering. Philos. Trans. R. Soc. B 377, 20210393.

Bone, J.R., Stafford, R., Hall, A.E., Boyd, I., George, N., Herbert, R.J.H., 2022. Estuarine infauna within incidentally retained sediment in artificial rockpools. Front. Mar. Sci. 8, 780720.

Bone, J.R., Hall, A.E., Stafford, R., Mir, N.F., Benny, J., Herbert, R.J.H., 2024. Artificial rockpools create habitat refugia on seawalls at high tide. Ecol. Eng. 206, 107318.

Browne, M.A., Chapman, M.G., 2011. Ecologically informed engineering reduces loss of intertidal biodiversity on artificial shorelines. Environ. Sci. Technol. 45, 8204–8207.

Browne, M.A., Chapman, M.G., 2014. Mitigating against the loss of species by adding artificial intertidal pools to existing seawalls. Mar. Ecol. Prog. Ser. 497, 119–129.

Bugnot, A.B., Mayer-Pinto, M., Airoldi, L., Heery, E.C., Johnston, E.L., Critchley, L.P., Strain, E.M.A., Morris, R.L., Loke, L.H.L., Bishop, M.J., Sheehan, E.V., Coleman, R.A., Dafforn, K.A., 2021. Current and projected global extent of marine built structures. Nat. Sustain. 4 (1), 33–41.

Chapman, M.G., 2003. Paucity of mobile species on constructed seawalls: effects of urbanisation on biodiversity. Mar. Ecol. Prog. Ser. 264, 21–29.

Chapman, M.G., 2006. Intertidal seawalls as habitats for molluscs. J. Molluscan Stud. 72 (3), 247–257.

Chapman, M.G., Blockley, D.J., 2009. Engineering novel habitats on urban infrastructure to increase intertidal biodiversity. Oecologia 161, 625–635.

Chapman, M.G., Underwood, A.J., 2011. Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. J. Exp. Mar. Biol. Ecol. 400, 302–313.

Chee, S.Y., Wee, J.L.S., Wong, C., Yee, J.C., Yusup, Y., Mujahid, A., 2020. Drill-cored artificial rock pools can promote biodiversity and enhance community structure on coastal rock revetments at reclaimed coastlines of Penang, Malaysia. Trop. Conserv. Sci. 13.

Connell, S.D., Glasby, T.M., 1999. Do urban structures influence local abundance and diversity of subtidal epibiota? A case study from Sydney Harbour, Australia. Mar. Environ. Res. 47 (4), 373–387.

Crawley, M., 2012. The R Book, 2nd edition. Wiley, Sussex, UK.

Daniel, M., Boyden, C., 1975. Diurnal variations in physio-chemical conditions within intertidal rockpools. Field Stud. 4, 161–176.

Drakard, V., Evans, A.J., Crowe, T.P., Moore, P.J., Coughlan, J., Brooks, P.R., 2023. Artificial rockpools: seaweed colonisation and productivity vary between sites but are consistent across environmental contexts. Mar. Environ. Res. 188, 106022.

Duarte, C.M., 2014. Global change and the future ocean: a grand challenge for marine sciences. Front. Mar. Sci. 1, 63.

Duarte, C.M., Pitt, K.A., Lucas, C.H., Purcell, J.E., Uye, S., Robinson, K., Brotz, L., Decker, M.B., Sutherland, K.R., Malej, A., Madin, L., Mianzan, H., Gili, J.M., Fuentes, V., Atienza, D., Pages, F., Breitburg, D., Malek, J., Graham, W.M., Condon, R.H., 2013. Is global ocean sprawl a cause of jellyfish blooms? Front. Ecol. Environ. 11 (2), 91–97.

Dugan, J.E., Airoldi, L., Chapman, M.G., Walker, S.J., Schlacher, T., 2011. Estuarine and coastal structures: environmental effects, a focus on shore and nearshore structures. In: Wolanksi, E., McLusky, D.S. (Eds.), Treatise on Estuarine and Coastal Science, vol. 8. Academic Press, Waltham.

Earp, H.S., George, R., Brooke, P.R., Drakard, V.F., Thompson, B.J., Fisher, B., Hayden, R., Crowe, T.P., Moore, P.J., 2023. The population structure, sex ratio and reproductive potential of limpets (*Patella* spp.) on natural shores and artificial structures in the Irish Sea. Mar. Environ. Res. 184, 105853.

Evans, A.J., Firth, L.B., Hawkins, S.J., Morris, E.S., Goudge, H., Moore, P.J., 2015. Drill-cored rock pools: an effective method of ecological enhancement of artificial structures. Mar. Freshw. Res. 67 (1), 123–130.

Evans, A.J., Firth, L.B., Hawkins, S.J., Hall, A.E., Ironside, J.E., Thompson, R.C., Moore, P.J., 2019. From ocean sprawl to blue-green infrastructure – a UK perspective on an issue of global significance. Environ Sci Policy 91, 60–69.

Evans, A.J., Moore, P.J., Firth, L.B., Smith, R.K., Sutherland, W.J., 2021. Enhancing the Biodiversity of Marine Artificial Structures: Global Evidence for the Effects of Interventions. Conservation Evidence Series Synopses. University of Cambridge, Cambridge, UK.

Fairweather, P.G., 1988. Predation creates haloes of bare space among prey on rocky seashores in New South Wales. Aust. J. Ecol. 13, 401–409.

Firth, L.B., Thompson, R.C., White, F.J., Schofield, M., Skov, M.W., Hoggart, S.P.G., Jackson, J., Knights, A.M., Hawkins, S.J., 2013. The importance of water-retaining features for biodiversity on artificial intertidal coastal defence structures. Divers. Distrib. 19, 1275–1283.

Firth, L.B., Thompson, R.C., Bohn, K., Abbiati, M., Airoldi, L., Bouma, T.J., Bozzeda, F., Ceccherelli, V.U., Colangelo, M.A., Evans, A., Ferrario, F., Hanley, M.E., Hinz, H., Hoggart, S.P.G., Jackson, J.E., Moore, P., Morgan, E.H., Perkol-Finkel, S., Skov, M.W., Strain, E.M., van Belzen, J., Hawkins, S.J., 2014. Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. Coast. Eng. 87, 122–135.

Firth, L.B., Browne, K.A., Knights, A.M., Hawkins, S.J., Nash, R., 2016. Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment. Environ. Res. Lett. 11 (9).

Fletcher, R.L., Callow, M.E., 1992. The settlement, attachment and establishment of marine algal spores. Br. Phycol. J. 27 (3), 303–329.

Glasby, T.M., Connell, S.D., Holloway, M.G., Hewitt, C.L., 2007. Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions? Mar. Biol. 151, 887–895.

Hall, A.E., Herbert, R.J.H., Britton, J.R., Boyd, I., George, N.C., 2019. Shelving the coast with Vertipools: retrofitting artificial rock pools on coastal structures as mitigation for coastal squeeze. Front. Mar. Sci. 6, 456.

Hawkins, S.J., Moore, P.J., Burrows, M.T., Poloczanska, E., Mieszkowska, N., Herbert, R.J.H., Jenkins, S.R., Thompson, R.C., Genner, M.J., Southward, A.J., 2008. Complex interactions in a rapidly changing world: responses of rocky shore communities to recent climate change. Clim. Res. 37, 123–133.

Huggett, J., Griffiths, C.L., 1986. Some relationships between elevation, physicochemical variables and biota of intertidal rock pools. Mar. Ecol. Prog. Ser. 29, 189–197.

IPCC, 2022. In: Pörtner, H.-O., Roberts, D.C., Tignor, M., Poloczanska, E.S., Mintenbeck, K., Alegría, A., Craig, M., Langsdorf, S., Löschke, S., Möller, V., Okem, A., Rama, B. (Eds.), Climate Change 2022: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK and New York, NY, USA.

Jackson, A.C., Chapman, M.G., Underwood, A.J., 2008. Ecological interactions in the provision of habitat by urban development: whelks and engineering by oysters on artificial seawalls. Austral Ecol. 33 (3), 307–316.

Johnson, M.P., Hughes, R.N., Burrows, M.T., Hawkins, S.J., 1998. Beyond the predation halo: small scale gradients in barnacle populations affected by the relative refuge value of crevices. J. Exp. Mar. Biol. Ecol. 231 (2), 163–170.

Kosova, E., James, K., MacArthur, M., Vovides, A., Peters, J., Metcalfe, D., Roberts, K., Naylor, L.A., 2023. The BioGeo Ecotile: improving biodiversity on coastal defences using a multiscale, multispecies eco-engineering design. Ecol. Eng. 188, 106881.

Legrand, E., Riera, P., Pouliquen, L., Bohner, O., Cariou, T., Martin, S., 2018. Ecological characterization of intertidal rockpools: seasonal and diurnal monitoring of physico-chemical parameters. Reg. Stud. Mar. Sci. 17, 1–10.

Lenth, R.V., 2021. emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.6.3. https://CRAN.R-project.org/package=emmeans.

Little, C., Williams, G.A., Trowbridge, C.D., 2009. The Biology of Rocky Shores. Oxford University Press, New York, USA.

Loke, L.H.L., Chisholm, R.A., Todd, P.A., 2019. Effects of habitat area and spatial configuration on biodiversity in an experimental intertidal community. Ecology 100 (8), e02757.

MacArthur, M., Naylor, L., Hansom, J.D., Burrows, M.T., Loke, L.H.L., Boyd, I., 2019. Maximising the ecological value of hard coastal structures using textured formliners. Ecol. Eng. X 1, 100002.

Martins, G.M., Hawkins, S.J., Thompson, R.C., Jenkins, S.R., 2007. Community structure and functioning in intertidal rock pools: effects of pool size and shore height at different successional stages. Mar. Ecol. Prog. Ser. 329, 43–55.

McKinney, M.L., 2006. Urbanisation as a major cause of biotic homogenization. Biol. Conserv. 127 (3), 247–260.

Meager, J.J., Schlacher, T.A., Green, M., 2011. Topographic complexity and landscape temperature patterns create a dynamic habitat structure on a rocky intertidal shore. Mar. Ecol. Prog. Ser. 428 (1–12).

Mendonça, V., Madeira, C., Dias, M., Vermandele, F., Archambault, P., Dissanayake, A., Canning-Clode, J., Flores, A.A.V., Silva, A., Vinagre, C., 2018. What's in a tide pool? Just as much food web network complexity as in large open ecosystems. PLoS One 13 (7), e0200066.

Mendonça, V., Flores, A.A.V., Silva, A.C.F., Vinagre, C., 2019. Do marine fish juveniles use intertidal tide pools as feeding grounds? Estuar. Coast. Shelf Sci. 225, 106255.

Metaxas, A., Scheibling, R.E., 1993. Community structure and organisation of tidepools. Mar. Ecol. Prog. Ser. 98, 187–198.

Mitsch, W.J., Jorgensen, S.E., 2003. Ecological engineering: a field whose time has come. Ecol. Eng. 20 (5), 363–377.

Moreira, J., 2006. Patterns of occurrence of grazing molluscs on sandstone and concrete seawalls in Sydney Harbour (Australia). Molluscan Res. 26 (1), 51–60.

Morris, S., Taylor, A.C., 1983. Diurnal and seasonal variation in physicochemical conditions within intertidal rock pools. Estuar. Coast. Shelf Sci. 17, 339–355.

Morris, R.L., Chapman, M.G., Firth, L.B., Coleman, R.A., 2017. Increasing habitat complexity on seawalls: investigating large- and small-scale effects on fish assemblages. Ecol. Evol. 7, 9567–9579.

Morris, R.L., Golding, S., Dafforn, K.A., Coleman, R.A., 2018. Can coir increase native biodiversity and reduce colonisation of non-indigenous species in eco-engineering rock pools? Ecol. Eng. 120, 622–630.

Moschella, P.S., Abbiati, M., Aberg, P., Airoldi, L., Anderson, J.M., Bacchiocchi, F., Bulleri, F., Dinesen, G.E., Frost, M., Gacia, E., Granhag, L., Jonsson, P.R., Satta, M.P., Sundelof, A., Thompson, A.C., Hawkins, S.J., 2005. Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. Coast. Eng. 52 (10–11), 1053–1071.

Naylor, L.A., Kippen, H., Coombes, M.A., Horton, B., MacArthur, M., Jackson, N., 2017. Greening the Grey: A Framework for Integrated Green Grey Infrastructure (IGGI). University of Glasgow, Glasgow.

Noel, L.M.-L.J., Hawkins, S.J., Jenkins, S.R., Thompson, R.C., 2009. Grazing dynamics in intertidal rockpools: connectivity of microhabitats. J. Exp. Mar. Biol. Ecol. 370 (1–2), 9–17.

Odum, H.T., Odum, B., 2003. Concepts and method of ecological engineering. Ecol. Eng. 20, 339–361.

O'Shaughnessy, K.A., Hawkins, S.J., Evans, A.J., Hanley, M.E., Lunt, P., Thompson, R.C., Francis, R.A., Hoggart, S.P.G., Moore, P.J., Iglesias, G., Simmonds, D., Ducker, J., Firth, L.B., 2020. Design catalogue for eco-engineering of coastal artificial structures: a multifunctional approach for stakeholders and end-users. Urban 23, 431–443.

Ostale-Valriberas, E., Sempere-Valverde, J., Coppa, S., Garcia-Gomez, J.C., Espinosa, F., 2018. Creation of microhabitats (tidepools) in ripraps with climax communities as a way to mitigate negative effects of artificial substrate on marine biodiversity. Ecol. Eng. 120, 522–531.

Perkol-Finkel, S., Sella, I., 2015. Harnessing urban coastal infrastructure for ecological enhancement. Marit. Eng. 168 (MA3), 102–110.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., the R Core Team, 2020. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-144. URL: https://CRAN.R-project.org/package=nlme.

Pister, B., 2009. Urban marine ecology in southern California: the ability of riprap structures to serve as intertidal rocky habitat. Mar. Biol. 156 (5), 861–873.

Raffaelli, D., Hawkins, S.J., 1996. Intertidal Ecology. Kluwer Academic Publishers, Dordrecht, the Netherlands.

Strain, E.M.A., Olabarria, C., Mayer-Pinto, M., Cumbo, V., Morris, R.L., Bugnot, A.B., Dafforn, K.A., Heery, E., Firth, L.B., Brooks, P.R., Bishop, M.J., 2017. Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? J. Appl. Ecol. 55, 426–441.

Truchot, J.P., Duhamel-Jouve, A., 1980. Oxygen and carbon-dioxide in the marine intertidal environment – diurnal and tidal changes in rockpools. Respir. Physiol. 39, 241–254.

Underwood, A.J., Jernakoff, P., 1984. The effects of tidal height, wave-exposure, seasonality and rock-pools on grazing and the distribution of intertidal macroalgae in New South Wales. J. Exp. Mar. Biol. Ecol. 75, 71–96.

Vaselli, S., Bulleri, F., Benedetti-Cecchi, L., 2008. Hard coastal-defence structures as habitats for native and exotic rocky-bottom species. Mar. Environ. Res. 66 (4), 395–403.

Vinagre, C., Mendonça, V., 2023. Changing webs – variation of complex networks over a tidal cycle in an intertidal rocky reef. Ecol. Complex. 56, 101060.

Vinagre, C., Mendonça, V., Cereja, R., Abreu-Afonso, F., Dias, M., Mizrahi, D., Flores, A. A.V., 2018. Ecological traps in shallow coastal waters – potential effect of heat-waves in tropical and temperate organisms. PLoS One 13 (2), e0192700.

Waltham, N.J., Sheaves, M., 2018. Eco-engineering rock pools to a seawall in a tropical estuary: microhabitat features and fine sediment accumulation. Ecol. Eng. 120, 631–636.

White, G.E., Hose, G.C., Brown, C., 2014. Influence of rock-pool characteristics on the distribution and abundance of inter-tidal fishes. Mar. Ecol. 36, 1332–1344.

Zuur, A.F., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed Effects Models and Extensions in Ecology with R. Springer.