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Ecological impact of single and semi-contiguous artificial rockpool installations on the assemblages and species richness of vertical seawalls

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ABSTRACT

Local improvements to species diversity through the creation of microhabitat features have been adopted as an approach for "Greening Grey Infrastructure" (GGI) in urbanised coastal ecosystems. To confidently implement these enhancements asset managers and engineers need quantitative information on the value of different feature types, densities, and configurations. We compared the biodiversity benefits of horizontal arrays of semi-contiguous 3 and 5 artificial rockpools with single isolated rockpool units and unenhanced sections of seawall. Rockpools were fixed within seawall sections 2 m wide at Mean High Water Neap Tide Level. At low tide, biota was monitored inside the pools, on the side of the pool units, the sea wall adjacent to the rockpools and in sea wall zones above and below the pools. After 36 months, species richness (all zones combined) of seawall sections with five rockpools was up to four times greater than controls and included protected and non-indigenous species. Increased richness was attributable to a higher density of rockpools and not rockpool contiguity. Grazers attracted to areas between and above rockpools modified assemblages that may limit persistence of algae. At one site, recovery of brown algae following disturbance during rockpool installation remained incomplete after 36 months. Benefits of arrays of semi-contiguous pools remain unclear, and deployment of individual rockpools (or similar enhancements) over a larger habitat area, that experience a wider range of conditions, may be at least as valuable. Quantifying species richness per unit size/ area of structure should assist managers and the development of metrics designed to measure ecological benefits in GGI.

1. Introduction

Along urban coasts, the loss of marine habitats and degradation of ecological communities has been particularly acute due to the cumulative pressure of pollution, invasive species, resource exploitation and development due to rising human populations (Vitousek et al., 1997; Stewart et al., 2010; Airoldi and Beck, 2007; Todd et al., 2019; O'Shaughnessy et al., 2020; Airoldi et al., 2021). This has caused the displacement and modification of natural habitats and significant negative interactions with coastal processes and habitat connectivity (Airoldi and Beck, 2007; Bulleri and Chapman, 2010; Dugan and Hubbard, 2010; Duarte et al., 2012; Firth et al., 2016; Heery et al., 2017; Bishop et al., 2017; Aguilera et al., 2020; Bugnot et al., 2021). These issues are compounded in regions experiencing rising sea levels due to thermal expansion and the melting of ice sheets and glaciers (Frederikse et al., 2020). Along many urban coasts the prevailing management approach to flood risk is to increase the number and size of hard defences such as breakwaters and seawalls (Dong et al., 2020; Hosseinzadeh et al., 2022). Yet these structures typically offer poor intertidal habitat for marine organisms as their surfaces are often smooth and homogenous and lack spatial and topographic complexity and geodiversity of natural rocky habitats such as holes, crevices, overhangs and rockpools of various shapes and sizes (Moschella et al., 2005; Vaselli et al., 2008; Bulleri and Chapman, 2010; Firth et al., 2013; Aguilera et al., 2014; Evans et al., 2015; Hall et al., 2018; MacArthur et al., 2019; Evans et al., 2021). Therefore, there is a need to create multifunctional structures that are constructed to minimise coastal flood risk and yet also provide quality habitats for marine organisms (Dafforn et al., 2015; Naylor et al., 2023).

Over the past decade, there have been many small-scale trials and

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experiments to try and improve the ecological quality of coastal infrastructure through a range of enhancements that aim to increase local habitat heterogeneity and mimic the complexity of natural rocky reefs (Strain et al., 2018a). The performance and potential application of these and other interventions has been usefully summarised (see O'Shaughnessy et al., 2020; Firth et al., 2024 for reviews) and although mostly evaluated over relatively short timescales (∼12 months) the majority have demonstrated a positive increase in local species richness at the patch-scale (few centimetres -metre).

The retrospective incorporation of features that remain moist or retain water at low tide, such as holes, crevices and pools, have been found to be particularly beneficial at increasing local species richness and abundance (Chapman and Blockley, 2009; Browne and Chapman, 2011, 2014; Chapman and Underwood, 2011; Firth et al., 2014; Hall et al., 2018; Farrugia Drakard et al., 2021,Drakard et al., 2024). In addition to the pool of water, retrofitting concrete artificial rockpool units on to vertical seawalls can create multiple microhabitats/niches that are attractive to different species (Hall et al., 2019; Bone et al., 2022; Drakard et al., 2021; Drakard et al., 2024). For example, spaces created behind the feature can provide a refuge for larger invertebrates and fish (Bishop et al., 2022; Farrugia Drakard et al., 2024). Upper and mid-shore habitats are often impacted the most from the development of coastal infrastructure (Dugan and Hubbard, 2010) and therefore in need of restoration and enhancement. Species assemblages on natural rocky shores and seawalls have been found to differ most at upper and mid-tidal levels (Chapman and Bulleri, 2003) and the addition of rockpool features at upper and mid-tidal heights can have the greatest increases in species richness and abundance of specific species compared to adjacent wall habitats (Browne and Chapman, 2011, 2014; Firth et al., 2013).

Most artificial rockpool deployments and other interventions have consisted of a few units within experimental projects, so their small habitat area and isolation from other microhabitat habitat features (e.g. on natural shores or other ecological enhancement features) may limit species richness and population size, particularly of more specialist and migratory species (Bender et al., 1998; Strain et al., 2017). To address the current biodiversity crisis, scaling-up ecological enhancements within coastal infrastructure is now becoming more important. For example, in England, coastal engineers and planners must now evaluate the ecological benefits, or biodiversity 'net gain', for Integrated Green Grey Infrastructure at the scale of the whole structure (UK Government, 2024; CIEEM, 2024), with similar legislation set to come into force in many other countries. Scaling-up of enhancement may involve (i) increasing the density of enhancement/ intervention/ rockpool units e.g. units per length of seawall or breakwater, (ii) increasing the variety of microhabitats, or (iii) increasing the overall surface area of enhanced infrastructure. Where larger schemes have been introduced (Sawyer et al., 2020; Bishop et al., 2022) the aim has been to increase spatial variation and contiguity of habitat to replicate that found on rocky shores (Strain et al., 2017).

In terms of rockpool deployments, either on their own or as part of a mix of enhancements, the landscape-scale impacts on assemblages occupying the surrounding seawall and on whole structures are relatively unknown (Strain et al., 2017). For example, it is possible that rockpool edges can provide crevice-type habitats that may attract gastropods and mobile arthropods with the potential to create grazing or predation 'halos' around above and below features (Fairweather, 1988; Johnson et al., 1998, 2008; Williams et al., 2000; Stafford and Davies, 2005; Hall et al., 2018). Variation in microclimate around the rockpools may also result in unique biotic assemblages not found beyond the rockpool itself (Ostale-Valriberas et al., 2018). Therefore, to better understand the value of larger scale interventions, sampling needs to be carried out on a broader-scale across the tidal range, and above and below enhancement units and features.

Intuitively 'more' units may be considered better than 'fewer', as increasing habitat area and spatial heterogeneity should yield higher species richness (Preston, 1960; Connor and McCoy, 1979; Johnson et al., 2003). Grazer and predator foraging activity may also be enhanced if the number or density of units is higher as this may increase the potential number of refuges for grazers and therefore the abundance of these species (Menge, 1978; Johnson et al., 1997; Stafford and Davies, 2005; Skov et al., 2011). However, where site resources are limited, optimising the density and configuration of retrofitted enhancement units (pools, tiles, artificial reefs) (Loke et al., 2015; Loke and Chisholm, 2022) may yield gains that are greater than the sum of the individual component parts. Additional habitat and interactions arising from different placement configurations have not been commonly investigated, especially in intertidal systems, (though see Loke et al., 2019). There may be significant ecological and/or societal benefits should edible or rare and/or protected species colonise the site through the provision of specialist microhabitats (Martins et al., 2010). An aspiration of the ecological enhancement of infrastructure is to mitigate the colonisation of non-native species (NNS) (Dafforn, 2017) as opportunistic generalists often settle on artificial structures (Glasby et al., 2007; Mineur et al., 2012; Airoldi et al., 2015). Where units are placed at higher density more species and greater numbers may be attracted due to greater availability of specific habitat features. Such attraction may provide benefits to rare species, but also has the potential to attract greater numbers of NNS (Zabin, 2015).

Here we evaluate a larger scale trial over 36 months that compared species richness and ecological assemblages on sections of seawall with single rockpools (R1) and sections of wall with different horizontal configurations of three (R3) and five (R5) semi-contiguous rockpools. Arrays of multiple semi-contiguous pools have the potential to increase local habitat complexity through the creation of gaps that mimic natural crevices between adjacent pool units. Configurations of multiple pools also have potential to create a larger area of shade on the seawall that may provide a refuge at low tide for mobile and sessile fauna and shade tolerant algae. On the seawall immediately above the pools, evaporative cooling effects at low tide on warm days may also be sufficient to provide a refuge and enhanced survival. So, there is potential for species richness to be greater in these R3 and R5 configurations.

Species richness (Alpha (α) diversity), defined as the number of species within a specific plot or area, was employed in this study as this has been widely used for comparing different ecosystems (Gotelli and Colwell, 2001) and in the comparison of natural and artificial habitats (Firth et al., 2013, 2014; Herbert et al., 2017; O'Shaughnessy et al., 2023), and measures of abundance of each species also taken to allow assemblage structure to be calculated and compared.

At the end of each full year of deployment, we investigated the impact of these different rockpool treatments at two spatial scales:

(i) Zone - the effect of the installations on assemblages and species richness within the tidal zone above the rockpool deployment (Zone 1), the zone of rockpool deployment (Zone 2- equivalent to Mean High Water Neap Tide level) and on the seawall zone below the rockpools (Zone 3). Each sampled zone was approximately 2.0 m wide x 0.25 m high.
(ii) Seawall Section - the effect of the installations on assemblages and species richness within a whole seawall section, combining species found within all tidal Zones 1–3. Each wall section sampled was approximately 2.0 m wide by 1.0 m high.

We tested three main hypotheses:

1. In Zone 2, treatments with 3 and 5 rockpools will have a higher species richness than single rockpool and control treatments;
2. Whole seawall sections (Zones 1–3 combined) will have a higher species richness in treatments with 3 and 5 rockpools than with single rockpools and control treatments.
3. Differences in assemblages above (Zone 1) and below (Zone 3) the zone of rockpool deployment will be greater in seawall treatments

with 3 and 5 rockpools compared to treatments with single rockpools and seawall controls.

2. Materials and Methods

2.1. Study sites

Artificial rockpools were fixed to vertical concrete seawalls at two sheltered sites with northerly-westerly aspect on the south coast of England: Bouldnor, a sheltered, rural, open coast on the Isle of Wight, and Sandbanks, a sheltered, urban location within Poole Harbour (Fig. 1).

At Bouldnor, the tidal range is 3 m and mean inshore surface salinity is 34 PSU. The seawall extends 1 km west of Bouldnor to Yarmouth which has a small but popular harbour. The seawall is colonised by dense, luxuriant zones of *Fucus spiralis* and *Ascophyllum nodosum.* Grazing gastropods, including limpets, are rare and barnacles are scattered where there have been fucoid disturbances. The foreshore below the seawall consists of fucoid-covered boulders and mobile mixed sediments. Permanent natural rockpools are not common on the shore but are present at other sites along the coast 4 km west of Bouldnor. Busy ports at Southampton and Portsmouth are situated 25 km to the east.

The north side of the Sandbanks peninsula is a sheltered urban location within Poole Harbour where the tidal range is 2 m, and the salinity varies between 26.3 and 34.5 PSU. Hard vertical seawalls protect dense residential and tourist infrastructure along much of the shoreline. The seawall is colonised by green algae, barnacles (dominated by *Austrominius modestus*), limpets (*Patella vulgata*) and patches of *F. spiralis*, above small clumps of mussels (*Mytilus edulis*) and red algal turf. The foreshore below the seawall is medium sand with an assemblage dominated by lugworms *Arenicola* spp. and amphipods. There are no natural rockpools within Poole Harbour, however they are present at rocky shores within marine protected areas 10 km to the west. Sandbanks has a significantly high resident and visiting human population and dense tourist infrastructure, nearby marinas, and water sports facilities. There is significant marine infrastructure around the port of Poole on the north side of the harbour and within the harbour there are fisheries for cockle (*Cerastoderma edule*) and Manila clam (*Ruditapes philipinarum*) and aquaculture for Pacific oyster (*Magallana gigas*).

2.2. Artificial rockpools

The concrete artificial rockpools were made of quick-setting low-carbon cement (Vicat Prompt), and manifactured by *Artecology* (Isle of Wight). Each pool is 32 cm wide, has a water depth of ~10 cm and overall capacity of 1.5 L of sea water. Small (9 mm) hemispherical depressions were created on the outside of the rockpool unit to simulate the surface roughness and habitat complexity of nearby limestone rocky shores. The manufacturing process can be viewed here (https://www.youtube.com/watch?v=mX0YXjWotWE). Details of dimensions and installation process may be found within the Supporting Information.

2.3. Experimental design

At each study site we established different configurations of rockpools and associated control treatments along an 80 m section of vertical seawall, herewith termed the ‘experimental wall’. Adjacent, but separated from the experimental wall by approximately 50 m, we established a ‘control wall’ of the same length where we did not fix any rockpools and was otherwise devoid of enhancements. All rockpool and control wall treatments were established in July 2020. Rockpool treatments were single rockpools (R1) and horizontal semi-contiguous rows of three (R3) and five rockpools (R5) fixed at Mean High Water Neap Tide level (Fig. 2).

At each study site, each treatment was 2 m wide and replicated (n = 5) randomly over the 80 m section of experimental seawall (sections were allocated treatments based on randomly generated numbers from 1

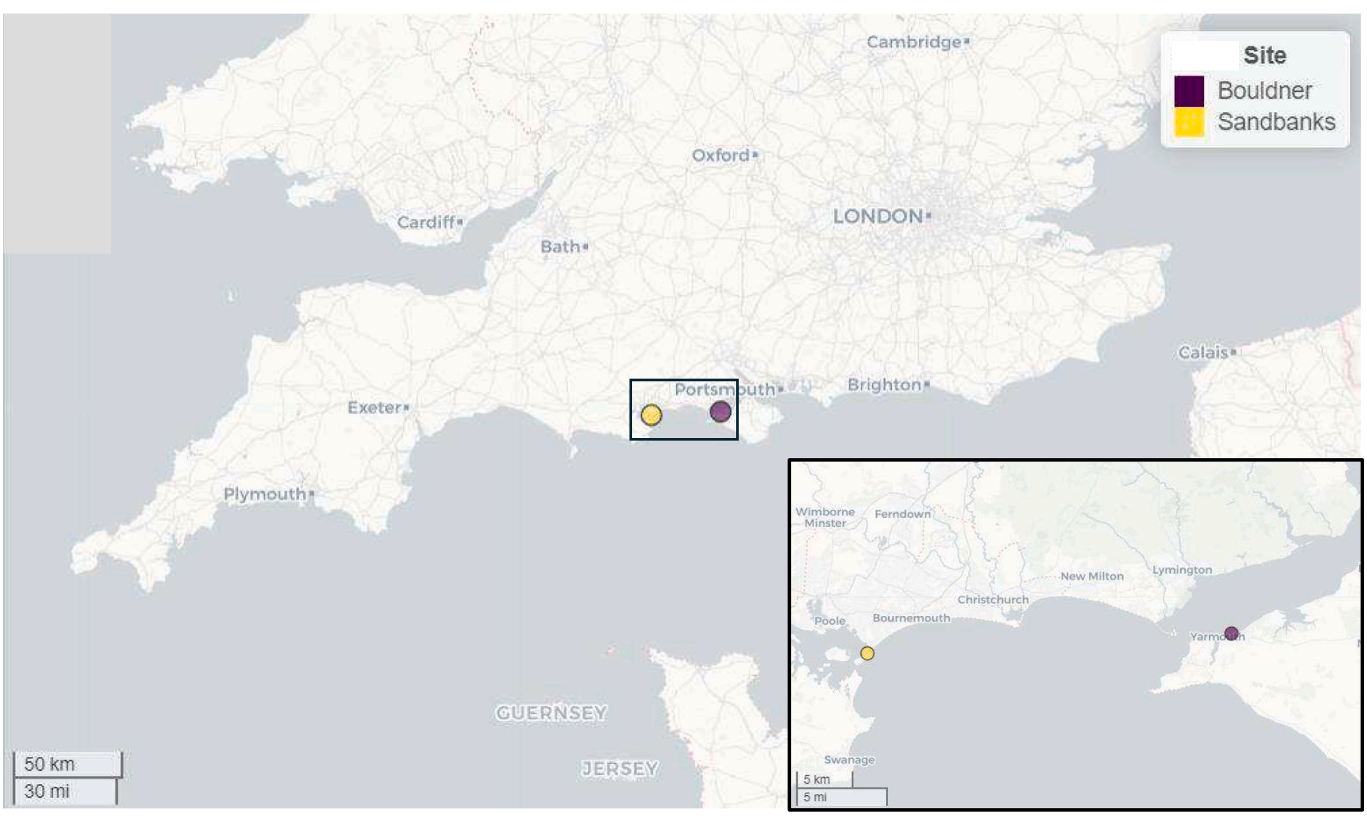

Fig. 1. Location of study sites on the South Coast of the UK. Map copyright of OpenStreetMap contributors (openstreetmap.org/copyright) and Carto (https://carto.com/attribution).

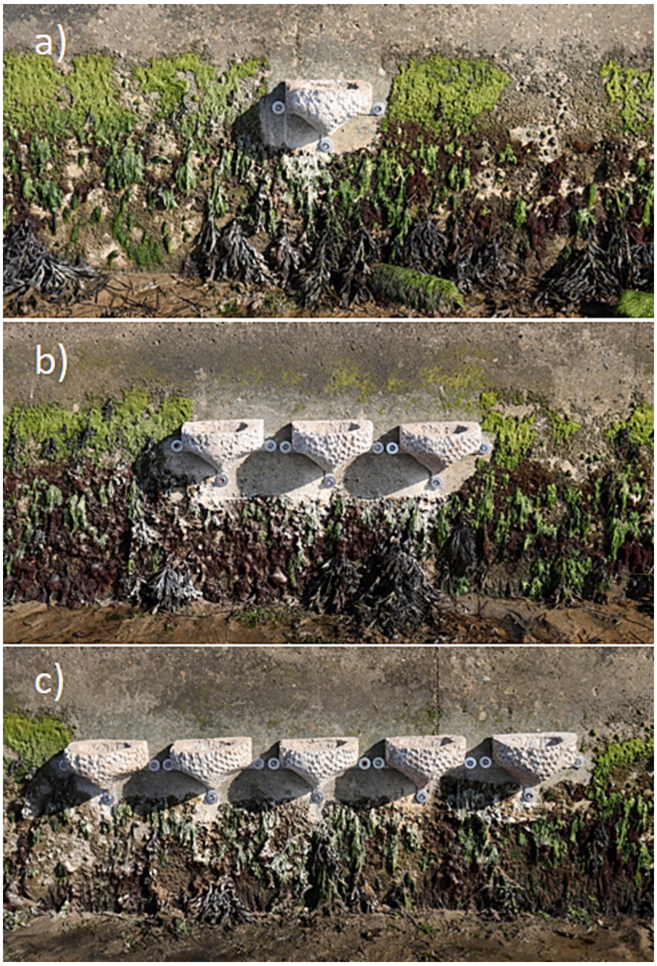

Fig. 2. Two-metre-wide sections of seawall at Sandbanks with (a) single (RP1), (b) three (RP3) and (c) five (RP5) rockpools treatments fixed at Mean High Water Neap Tide level. Photo July 2020 shortly after installation.

to 5 corresponding to the five different treatments on the main wall). For the R5 (five rockpool) treatment, a 15 cm gap was necessary between each rockpool to minimise damage to the seawall and to accommodate the rockpool brackets, therefore, these treatments were considered 'semi-contiguous'. While it would have been possible to extend this distance in the 2 m wide zone for the three rockpool (R3) treatment, it was decided to fix the configuration similarly, i.e. semi-contiguous (15 cm apart) as opposed to equally spaced across the 2 m wall section. This was considered beneficial as the creation of more closely connected surfaces and habitats on artificial structures, mimicking habitat natural habitats, has been argued as a major challenge for the design of eco-engineering solutions on marine infrastructure (Bishop et al., 2017; Strain et al., 2017).

Among the rockpool treatments and on the control wall, we established 2 m width 'control sections' of seawall (n = 5) where there were no rockpools. As some of the existing seawall biota was disturbed when we mounted the rockpools, we also established 'procedural control' seawall sections of 2 m width (n = 5) by scraping off a similar amount of algae and removing attached fauna but did not mount any rockpools. These were randomly interspersed with the other treatments (see above) to establish if the installation process had an impact on the species richness in rockpools and in zones above and below the pools during the monitoring period. A 2 m space separated each of the rockpool treatment sections, experimental controls and procedural controls.

All experimental and control wall sections encompassed the tidal range between Mean High Water Spring Tide level and the base of the seawall, which equated to approximately Mean Tide Level at both study sites.

2.4. Data collection

At each site, monitoring of rockpools and all control sections was carried out at low tide at intervals of 1, 3, 6, 12, 18, 24 and 36-months post-installation.

Within each treatment section, five 25 × 25 cm quadrats were placed on the seawall immediately above and below each rockpool, referred to as Zone 1 and Zone 3 respectively. Within the Zone of the rockpools (Zone 2), quadrats were placed over the rockpool basin, on the underside of each rockpool and on the immediately adjacent seawall, (except for R5 treatments as pools occupied the whole section width). Quadrats on the seawall were separated by 15 cm.

Percentage cover (canopy and understorey) of all algae and encrusting faunal taxa for each quadrat was identified to species level where possible and measured in situ. If a species had less than 4 % cover it was recorded as present and assigned a value of 1. Within the quadrats, counts were made of individual mobile species (e.g., gastropod molluscs and decapod crustaceans) and individual sessile benthic species identified to species level where possible. A small aquarium net was passed through each pool twice to capture swimming animals including fish and amphipods. The sampling design enabled approximately 80 % of the 80 m length of each seawall to be sampled.

After 36 months the density of observed limpet aggregations on the seawall around rockpools at Sandbanks was investigated by placing five rectangular 10 × 25 cm quadrats in each seawall section at MHWN immediately above the rockpools, and in the 15 cm gaps between rockpools. In treatments with single rockpools this was done either side of the rockpool and at three other locations of equivalent orientation and height. This method was also replicated in control sections of the experimental and control seawall.

2.5. Abiotic measurements

At each monitoring interval we measured water temperature and salinity in each of the rockpools using a hand-held multimeter (YSI 30). Depth of water and any sediment in the rockpools was also recorded.

2.6. Statistical analysis

Analysis of species richness at the two sites were conducted separately. A single value of species richness for each zone was obtained by pooling results of the five replicate quadrats. At each site, this gives $n = 5$ replicate values above the rockpools (Zone 1) the rockpool zone (Zone 2) and below the rockpools (Zone 3) for each treatment. Initially, a three-way mixed model ANOVA was performed on the annual species richness data. Data were square-root transformed to help meet assumptions of count data and used as the dependent variable. Fixed factors were 'Year' (1,2,3), 'Zone' (1, 2, 3), and 'Treatment' (rockpool treatments R1, R3, R5, procedural control, experimental control, control wall) and 'Quadrat Location ID' was included as a random factor due to repeated measures (i.e. recording of the same section of shore over successive time periods). Assumptions of the models were checked using fitted model vs. residual plots (Zuur et al., 2009). Analysis of two-way interaction plots involving 'Year' demonstrated relatively minor differences over time, and as such the model was altered to a two-way mixed ANOVA model with Quadrat Location ID nested within Year as a random effect, with just Zone and Treatment (and their interaction) as fixed effects. The percentage contribution of the random component to explaining the variance in the model was very small (<2 % of the variance explained in all cases). The analysis presented in the results and the multiple comparisons are based on the two-way ANOVA model with random effects, using the emmeans package in R (version 4.3.2) for post-hoc analysis.

To compare differences in species richness between treatment whole wall sections treatment (i.e. Zones 1–3 combined) a single value of species richness was obtained by pooling values from all zones (n = 5 per treatment). The same random error structure as above was used, but here Treatment was the only fixed effect.

At Sandbanks, observed limpet densities were compared using two-way ANOVA with Treatment (rockpool treatments R1, R3, R5, experimental control, procedural control, control wall) and Position (immediately above and between pools) as fixed factors.

After 36 months, when there had been a large measure of assemblage convergence between experimental controls and procedural controls, variation between Treatments in each Zone 1–3 were compared separately at both study sites with one-way PERMANOVA using the Vegan package in R (version 4.3.2). Sample data from each quadrat were square root transformed before analysis and the Bray–Curtis resemblance matrix was used with 9999 permutations for both analyses. SIMPER analysis was used to establish species similarity between assemblages Plymouth Routines in Multivariate Ecological Research (Primer-e v.7, Clarke and Gorely, 2015).

3. Results

Following installation in July 2020, assemblages both inside and outside all the rockpools in all treatments, and within the disturbed procedural control sections of the seawall, changed rapidly over the first 18 months. During this period, green algae (*Ulva* spp.) dominated the exterior of the pools and disturbed areas on the walls at both study sites, while the brown algae *Pylaiella littoralis* was dominant within rockpool interiors. Opportunists including tunicates, hydrozoa and bryozoans (*Bugula* spp.) colonised fresh mortar surfaces, especially on the inside of the rockpool interiors. Over the first 18 months water siphoned out of the rockpools through luxuriant growths of filamentous *Ulva* spp. during low tide, reducing water levels in the rockpools by up to half. In September 2020, two months after installation, a moderate settlement of non-native barnacle *Austrominius modestus* had occurred on the exterior of the rockpools and the seawall at Sandbanks, but not at Bouldnor, where densities of barnacles on the seawall are generally very low (<100 per m^2). Mobile species including decapods *Carcinus maenas*, *Palaemon elegans* and the intertidal benthic fish *Liphophrys pholis* sought refuge in the rockpools at each site within the first months, and throughout the monitoring period. In spring 2021, between 9 and 12 months post-installation, approximately 50 % of the *Ulva* on the exterior of the pools at Sandbanks had reduced due to desiccation and grazing by gastropods *Littorina littorea* and *L. saxatilis,* that had moved on to the rockpools from the seawall, but not limpets (*Patella* spp.). This did not happen at Bouldnor as grazers on the seawall were at very low density, though some grazed patches were created by a few topshells (*Steromphala umbilicalis*) that moved up from lower levels of the wall and took refuge beneath the rockpool or within the cavities on the rockpool exterior. However, grazing pressure on the *Ulva* remained very low. By January 2021, 18 months post-installation, much of the *Ulva* on the pool exteriors and disturbed areas at both sites was being replaced by the brown algae *Fucus spiralis* that had settled the previous autumn. *F. spiralis* continued to grow and develop during 2021 and 2022 yet became much more luxuriant at Bouldnor due to the near absence of grazers. By July 2023, three years post-installation, assemblages within disturbed areas around rockpool at Sandbanks and within procedural controls had converged with undisturbed areas and experimental control sections. This had not fully occurred at Bouldnor as parts of the canopy of *A. nodosum* removed during installation had not re-established, although plants were visible below a dense coverage of *F. vesiculosus* and *F. spiralis* that had occupied that space.

Throughout the monitoring period from August 2020 to July 2023, salinity and temperature of retained water in rockpools was similar to the seawater (Table 1). At Bouldnor, water temperature in the rockpools was typically slightly cooler than on the water's edge at low tide (Table 1), although salinity was comparable. Rockpool temperature and salinity at Sandbanks was more variable due to its more estuarine location, however, neither site, in any season, showed large increases in temperature or salinity as compared to some reported values for rockpools on natural shores. At both sites, only a trace of sandy sediment was found in the bottom of rockpools. Small rocks, occasionally thrown up from the beach below, were left in situ.

3.1. Species richness

At both study sites a significant interaction between zone and treatment was found at the (Sandbanks $F_{10,18} = 31.9$, $p < 0.001$; Bouldner $F_{10,250} = 21.3$, $p < 0.001$) with treatments with 1, 3 and 5 rockpools in Zone 2 showing increases in species richness compared to other zones (Figs. 3, 4). At both study sites there were no significant differences in species richness between 1, 3 and 5 rockpool treatments in Zone 1 (above pools) and Zone 3 (below pools) (Figs. 2, 3; post-hoc tests, $p > 0.05$ in all cases see supporting information).

All seawall sections with rockpool treatments (zones combined) showed a significant increase in mean species richness over the 36 month period, except for the single rockpool sections at Bouldnor that remained relatively stable after the first year survey (Fig. 5). Overall, at both sites, seawall sections with rockpools had significantly higher species richness than experimental controls, procedural controls and control wall sections (Sandbanks $F_{5,82} = 63.9$, $p < 0.001$; Bouldnor F $F_{5,82} = 98.1$, $p < 0.001$). Post-hoc comparison tests showed significant differences in mean species richness between most combinations of seawall sections at each study site (exceptions were between R3 and R5 treatments and between experimental and procedural controls on both shores; $p > 0.05$ in post-hoc tests – see supporting information for individual comparison p values).

Over the 36 month period, background changes in species richness on the control seawall and in control sections on the experimental seawall differed between study sites. At Bouldnor, significant differences between the control seawall and experimental seawall at the start of the experiment were maintained, yet although overall background species richness increased slightly, this was not significant. At Sandbanks, there was a significant decline in species richness over the 36 month period on the control seawall and both experimental and procedural control

Table 1
Mean low tide rockpool water temperature and salinity at Sandbanks and Bouldnor (mean of all 45 pools). Measurements made at each sampling interval. Values of sea water measured from the shore are shown in parentheses. nd = no data.

	Sandbanks		Bouldnor	
	Rockpool Water Temp °C (Sea)	Rockpool Salinity PSU (Sea)	Rockpool Water Temp °C (Sea)	Rockpool Salinity PSU (Sea)
Aug 2020	21.2 (21.0)	32.7 (33.3)	18.1 (20.1)	34.4 (34.2)
Oct 2020	14.4 (14.0)	30.3 (30.4)	11.4 (13.5)	32.5 (33.8)
Jan 2021	3.0 (nd)	24.9 (nd)	4.0 (6.5)	32.2 (32.5)
April 2021	10.4 (10.2)	31.4 (29.3)	10.0 (13.2)	32.6 (32.3)
July 2021	20.6 (23.6)	31.7 (30.7)	20.5 (20.6)	33.4 (34.4)
July 2022	nd	nd	19.1 (21.4)	34.5 (34.5)
July 2023	17.8 (18.1)	32.0 (32.2)	17.7 (20.6)	34.0 (34.0)

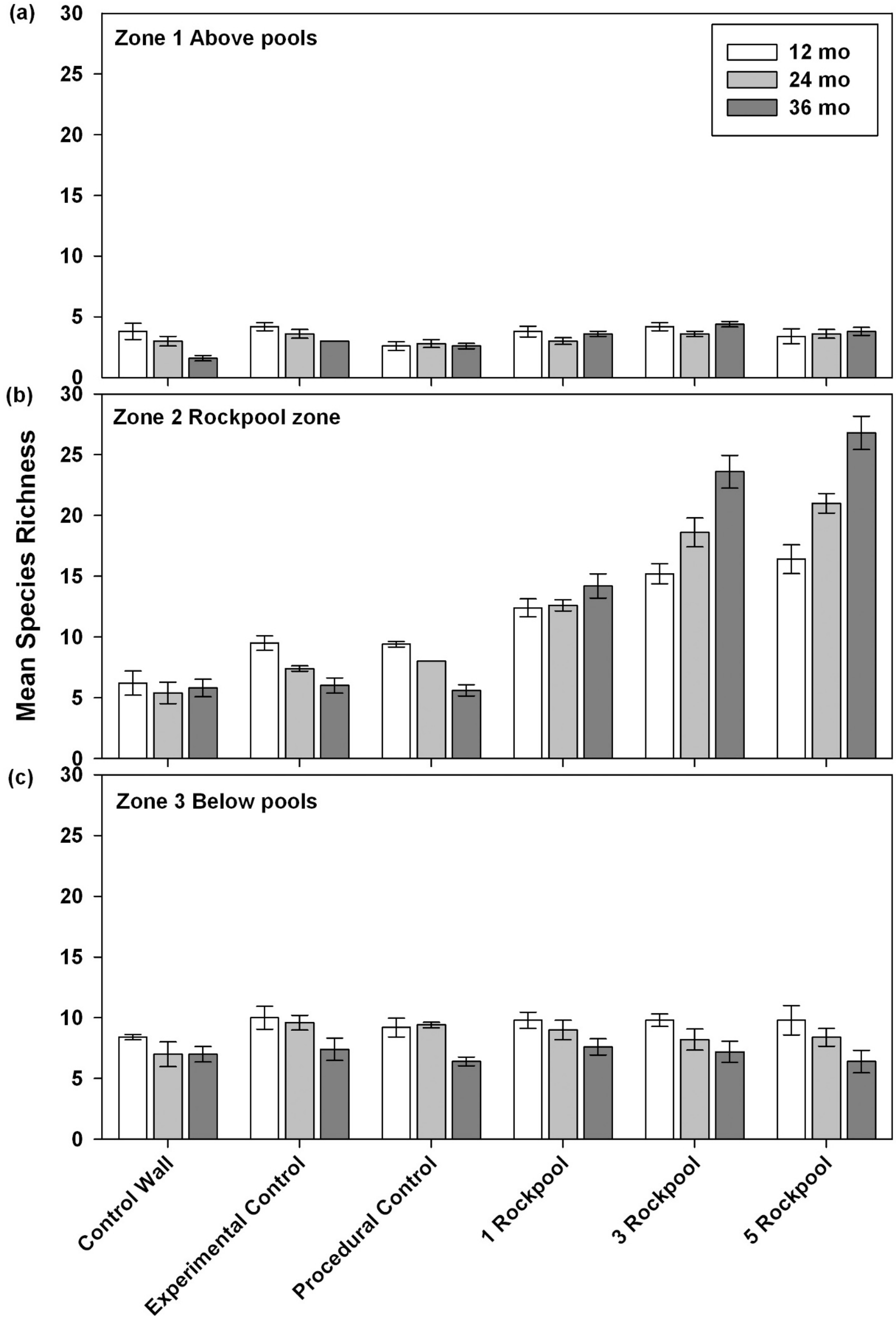

Fig. 3. Mean species richness in each zone for each treatment on the seawall at Sandbanks, 12, 24 and 36 months after deployment in July 2020. (a) Zone 1 above rockpools, (b) Zone 2 the rockpool zone, (c) Zone 3 below the rockpools. See Table 2 for results of treatment comparison tests. Error bars show ± SE.

seawall sections. This decline was mostly due to loss of *F. spiralis* patches, which reduced from a mean coverage of 42 % in 2020 to 7 % in 2023 on the control seawall. Fauna associated with *F. spiralis* patches including *Littorina obtusata* and *Actinia equina*, also declined. On the control wall *F. serratus* was lost in Zone 3 and red algal turf, dominated by non-native red alga *Caulacanthus okamurae,* that provided cover for other species including crabs *C. maenas,* also declined from a mean coverage of 18 % in 2020 to 4 % in 2023.

At Bouldnor, species new to seawall sections with rockpools were almost entirely algae not found in Zone 1 or 3 and all these were found within the rockpool interiors. The number of red algae (Rhodophyceae) recorded in these sections doubled that of controls and included *Ceramium* sp., *Griffithsia corralinoides* and *Halurus flosculosa*. Porifera, Tunicata and Bryozoa not found at Zone 3 below rockpools, also colonised the rockpools, but were infrequent. Opportunistic bryozoan species of *Bugula* and *Scrupocellaria* recorded in rockpools within the first three months did not persist. Over the entire monitoring period, 44 % of the 59 species recorded across the entire control seawall and experimental seawall at Bouldnor (total 160 m) were within rockpool interiors (Table 1 in Supporting Information provides these details, as well as breakdown in species by taxonomic grouping). At the 36 month census 45 % were found within rockpools interiors only.

At Sandbanks, species new to seawall sections with rockpools over the monitoring period were mostly red algae, which tripled in number, and these were all found entirely within the rockpool interiors. These included *Ceramium* sp., *Chylocladia verticillata,* and *Halurus flosculosa.* The brown algae *Pylaiella littoralis* and *Dumontia contorta* did not persist beyond the 1 month and 6 month census respectively, yet a variety of

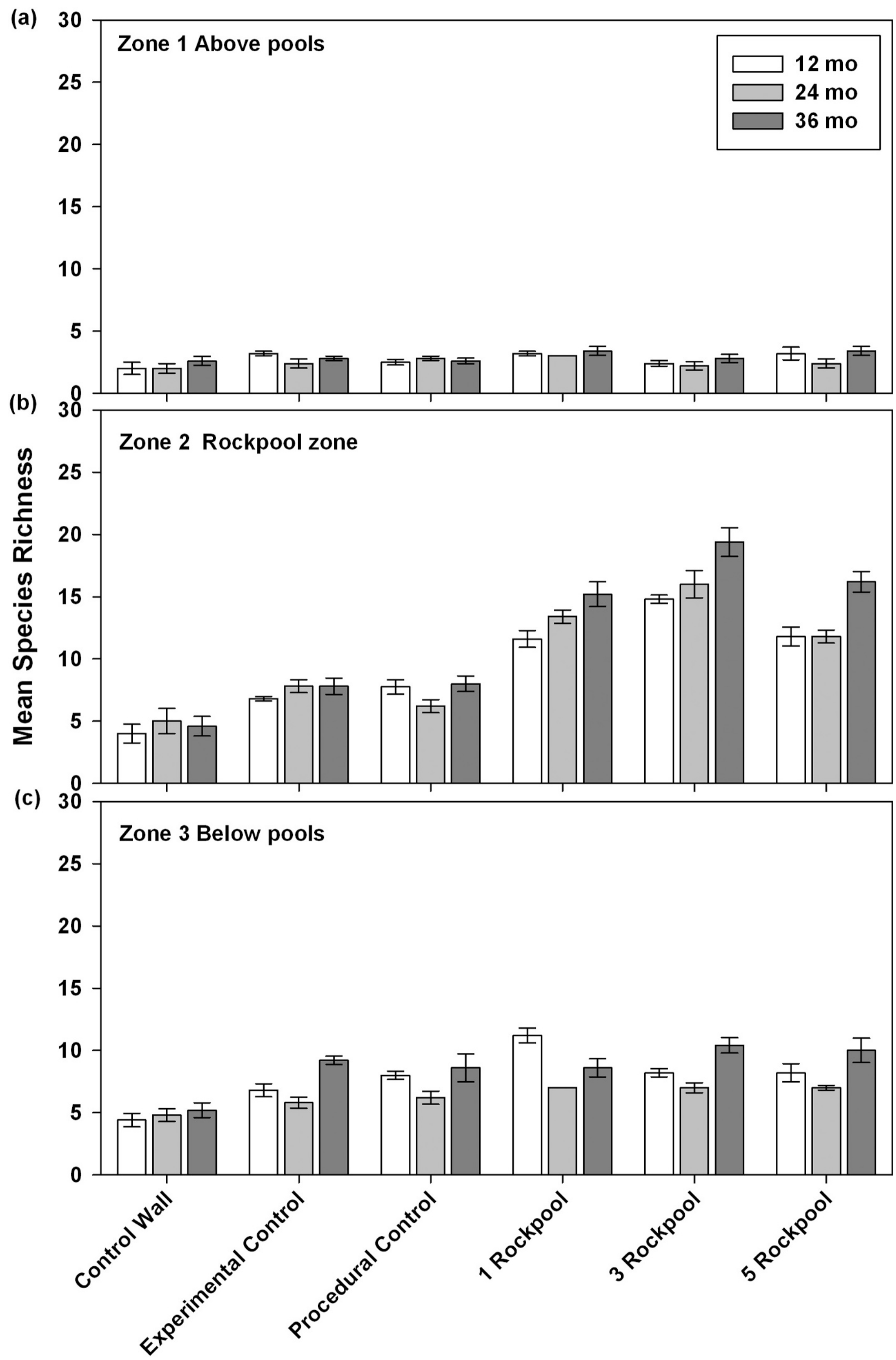

Fig. 4. Mean species richness in each zone for each treatment on the seawall at Bouldnor, 12, 24 and 36 months after deployment in July 2020. (a) Zone 1 above rockpools, (b) Zone 2 the rockpool zone, (c) Zone 3 below the rockpools. Error bars show ± SE.

faunal species also colonised the pools, including cnidaria *Kirchenpauria pinnata*, and *Anemonia viridis,* and tunicates *Aplidium glabrum*, *Ascidiella aspersa*, and *Ciona intestinalis*.

In 2021 there was a settlement of native oysters (*Ostrea edulis*), with spat first observed in each rockpool treatment in January 2022. In July 2022, fourteen native oysters were recorded all of which grew and persisted to the end of the monitoring period. By July 2023, one individual of shell diameter 77 mm manually opened on site had healthy-looking gonads ready to spawn. Over the monitoring period, 64 % of the 67 species recorded across both the control seawall and experimental seawall at Sandbanks (160 m) were within rockpool interiors (Supporting Information). At the 36 month census, 67 % were found within rockpools only.

At both sites, the mean number of NNS on seawall sections increased with the installation of the rockpools. Nearly all were species that require full submergence, including algae and tunicates. At Bouldnor, the only additional species was the algae *Sargassum muticum*. At Sandbanks, of the twelve NNS recorded in seawall sections with rockpools, four did not persist beyond the first year after installation and at 36 months the proportion of NNS in seawall sections with rockpools was 12 %. At Sandbanks, more NNS were found in multiple rockpool treatments than in single rockpool and control sections, however there was no association between species presence and rockpool contiguity.

At 36 months, although treatment replication is low, the initial

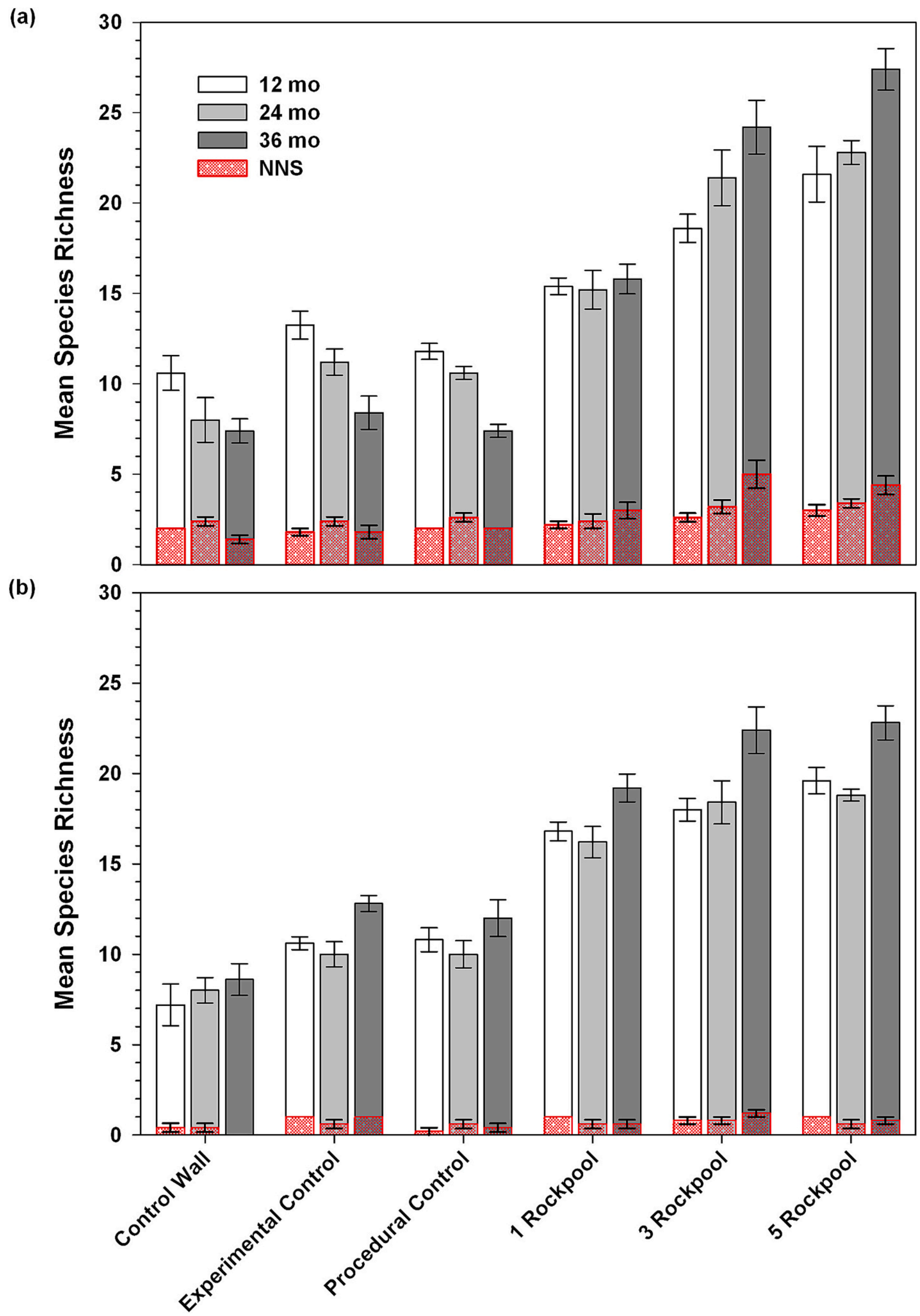

Fig. 5. Mean number of species recorded across whole wall sections (Zones 1–3 combined) of each treatment at each annual census (a) Sandbanks and (b) Bouldnor. Mean number of non-native species (NNS) shown in red. Error bars are ± SE. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

slopes of species accumulation (SAC) curves for each rockpool treatment (Fig. 6a) are at least as high for single rockpool deployment as for multiple rockpools. The stepped profiles of the continuous R5 SAC curves (Fig. 6b) indicate that more new species are encountered at different locations along the sea wall as the habitat changes.

3.2. Assemblages

3.2.1. Zone 1

In Zone 1 above the rockpools, PERMANOVA revealed significant differences in assemblages between treatments at both study sites (Sandbanks $F_{5,144} = 21.2$, $p < 0.001$; Bouldnor $F_{5,144} = 3.9$, $p < 0.001$). At Bouldnor, variation between treatments in Zone 1 was driven principally by differences in coverage of the algae *Fucus spiralis*, *Catenella caespitosa* and *Blidingia minima.* Pairwise differences between treatments and subsequent SIMPER analysis (see Supporting Information) showed that the red algae *Catenella caespitosa,* that had been significantly reduced in coverage during the establishment of the procedural controls and the R3 and R5 treatment sections had not recovered and had been replaced by *B.minima*. However, there were no significant differences in assemblage between procedural controls and R3 and R5 treatments (although a significant difference between procedural control and R1 treatment does occur, likely related to less disturbance in placing a single rockpool), or between R3 and R5 treatments. At Sandbanks, variation in treatments in Zone 1 was driven by differences in coverage of the barnacle *Austrominius modestus* and abundance of gastropods *Patella vulgata* and *Littorina saxatilis*. Significant pairwise differences between treatments and subsequent SIMPER analysis were mostly due to

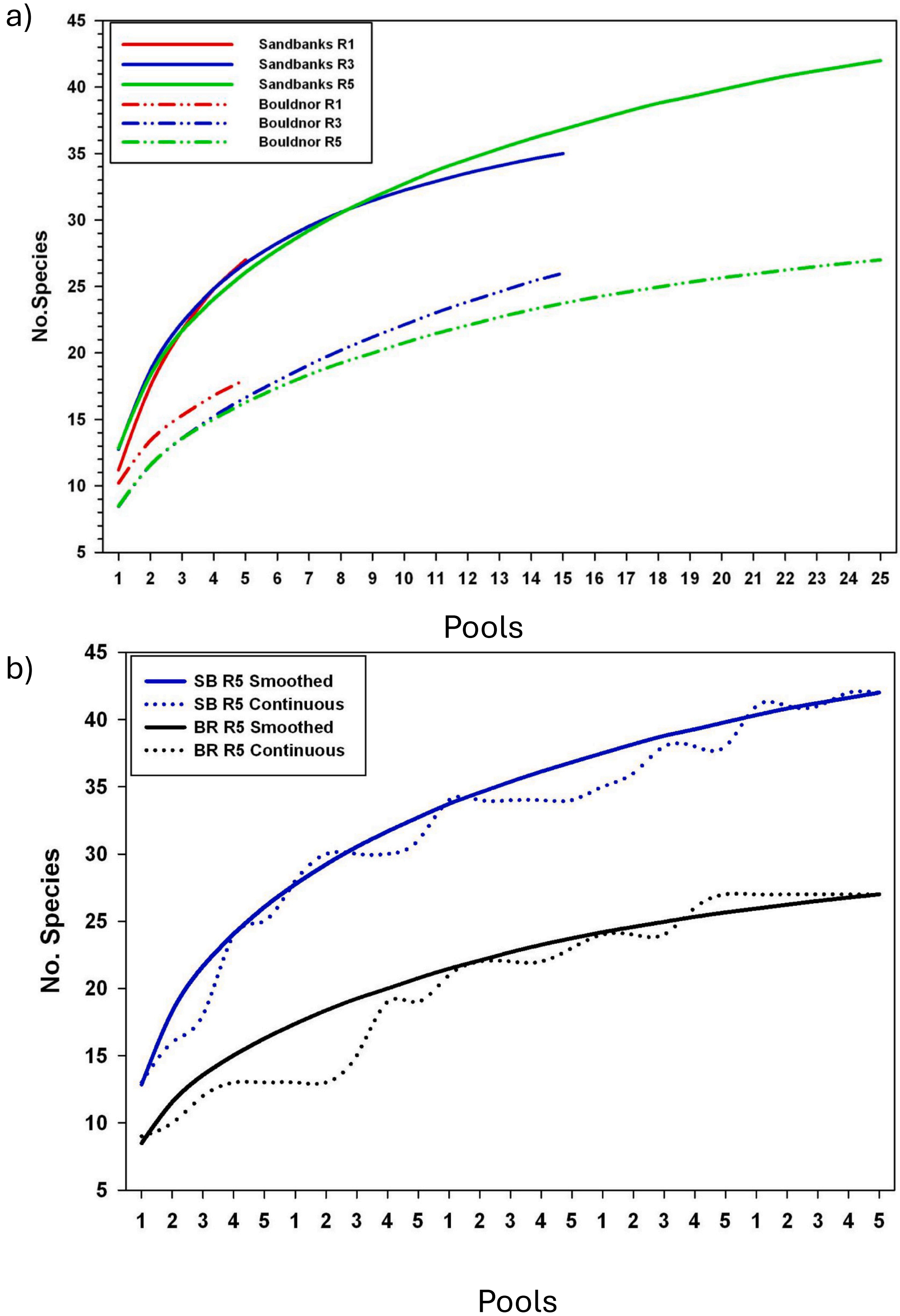

Fig. 6. (a) Mean species accumulation (*S*) curves for rockpool interior habitat at 36 month census for each treatment at Sandbanks and Bouldnor (R1 (single rockpool) $n = 5$ pools; R3 (three rockpools) $n = 15$ pools; R5 (5 rockpools) $n = 25$ pools) produced in Plymouth Routines in Multivariate Ecological Research (Primer-e v.7, Clarke and Gorely, 2015). Curves smoothed from 120 random permutations of sample order. **(b)** Continuous and mean species (*S*) accumulation curves for R5 rockpool interior habitat at 36 month census at Sandbanks (SB) and Bouldnor (BR). The continuous 'stepped' curve is produced from values of species richness of each pool as encountered in each replicate treatment 1–5 (shown on x-axis) along the seawall. Mean *S* curve produced from 999 permutations of original/continuous pool order.

variable abundance of the barnacle *A. modestus*, that may have been disturbed during the creation of the procedural controls, but the response to installation of R3 and R5 sections was inconsistent (see Supporting Infoirmation). Limpets (*Patella vulgata*) had increased in R3 and R5 sections contributing to assemblage dissimilarity between these and other treatments.

3.2.2. Zone 2

At 36 months the assemblages in Zone 2 of rockpool treatments had been transformed at both sites, and this was greatest for the R5 sections, where the pools occupied almost the entire zone (Fig. 7). PERMANOVA revealed significant differences in assemblages between treatments at both study sites (Sandbanks $F_{5234} = 8.23$, $p < 0.001$; Bouldnor $F_{5232} = 20.32$, $p < 0.001$). At Sandbanks, post-hoc comparison between treatments were all significant except for between experimental and procedural controls ($p = 0.07$), experimental control and single rockpool (R1) ($p = 0.19$) and three (R3) and five (R5) rockpool treatments. At Bouldnor, pairwise comparison between treatments were all significant ($p < 0.05$) (Supporting Information for full details). Pool interiors had been colonised by species characterised by the interior of rockpools on natural shores and those often found in zones lower than Zone 2, as described previously. Assemblages on the exterior of the pool units were different from all other Zone 2 treatments. At Sandbanks, these consisted mostly of a dense coverage of the barnacle *A. modestus* with *Littorina saxatilis* and resembled control areas, but with fewer limpets. At Bouldnor, the pool exterior was colonised by patches of *F. spiralis*, the red encrusting alga *Hildenbrandia rubra* and occasional littorinid molluscs. In sections where one (R1) and three (R3) rockpools had been installed, assemblages in quadrats placed beside the pools were not different to control sections of the seawall, although some initial disturbance during installation was still visible around the edges of pools at Bouldnor.

3.2.3. Zone 3

At Sandbanks, after 36 months, PERMANOVA showed that overall assemblage composition in Zone 3 beneath the rockpools did not differ between treatments ($F_{5, 144} = 1.75$, $p = 0.08$). There had been a reduction in the coverage of *F. spiralis* and *C. okamurae* turf since the beginning of the monitoring period, however this was common across all treatments and both control and experimental seawalls. At Bouldnor PERMANOVA showed significant differences between treatments in Zone 3 ($F_{5, 142} = 9.64$, $p < 0.001$). SIMPER analyses showed that variation between treatments was driven principally by differences in coverage of *Fucus spiralis*, *F. vesiculosus*, *Ascophyllum nodosum*,

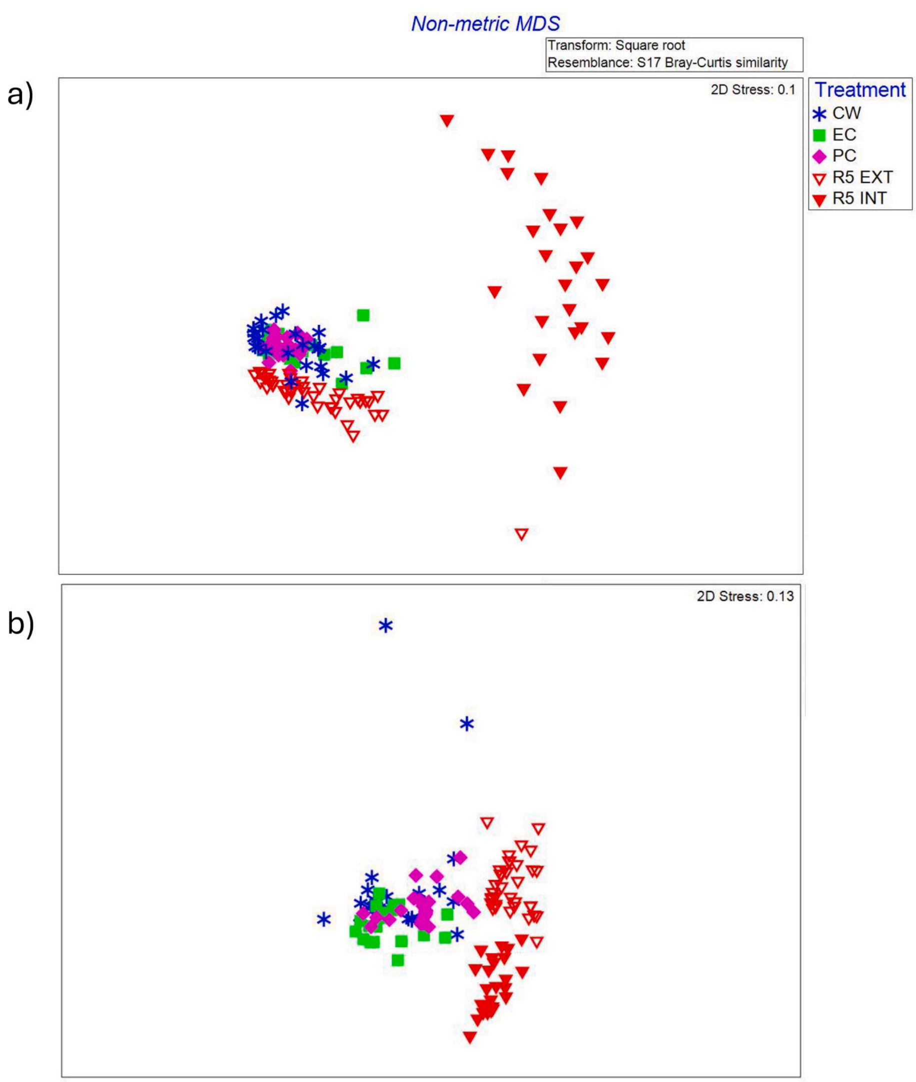

Fig. 7. nMDS plots of assemblages in R5 (5 rockpool) Zone 2 at 36 months (a) Sandbanks and (b) Bouldnor. CW = Control Wall, EC = Experimental Control, PC = Procedural Control, R5 INT = Interior of rockpools in R5 treatment, R5 EXT = Exterior of rockpools in R5 treatment. See text and Supporting Information for results of PERMANOVA analyses.

Cladophora rupestris, *Rhodothamniella floridula* and *Ulva* spp. (see Supporting Information). This variation of the canopy and understory can be attributed to the initial disturbance and removal of part of the *A. nodosum* canopy during rockpool installation, which had not completely recolonised. Convergence between procedural and experimental controls was low, with 22 % of dissimilarity between experimental and procedural controls attributed to lower coverage of *A. nodosum* (Supporting Information). Areas of *A. nodosum* canopy removed during installation had become colonised by a dense growth of *Fucus* spp. Pairwise comparisons showed no significant differences between procedural controls and R3 ($p = 0.27$) and R5 ($p = 0.27$) treatments, where there had been most disturbance (see Supporting Information for full analysis).

At both sites, at 36 months, PERMANOVA showed that assemblages within the interior and exterior of rockpools did not differ between treatments, suggesting that species colonisation of each rockpool had been random and was not determined by rockpool contiguity (Sandbanks $F_{2, 42} = 1.368$, $p = 0.143$; Bouldnor $F_{2, 42} = 1.066$, $p = 0.393$).

3.3. Limpet density and distribution

At Sandbanks, two-way ANOVA showed that the density of adult limpets in gaps between and immediately above pools did not differ. Yet density was significantly different across treatments (Fig. 8, Table 2). Tukey pairwise analysis showed that limpet density on wall sections with 3 and 5 rockpools was significantly greater than single rockpool sections and controls ($p < 0.05$, see supporting information).

4. Discussion

Over 36 months, we investigated the ecological impact of different configurations of rockpools on assemblages above and below the zone of deployment, in addition to changes within the rockpools themselves. As predicted, at both study sites. Seawall sections with multiple semi-contiguous rockpools had consistently higher species richness than sections with single rockpools. However, there was minimal ecological impact of the rockpools on surrounding zones, and all additional species recorded were within the pool interior. Additional species found occur mostly on natural rocky shores, infrastructure, and on subtidal reefs in the region (Herbert et al., 2022; Collins et al., 1990). Although natural rockpools are virtually absent from Poole Harbour, the artificial rockpools attracted species found in rockpool habitats beyond the harbour entrance. At Bouldnor, where there is some limited rockpool habitat, additional rockpool colonists were almost entirely species of algae. Density of grazers and filter feeders on the seawall at Bouldnor was initially very low, and these did not increase with installation of rockpools, although limpets (*P. vulgata*) and barnacles were present in moderate densities on other infrastructure within 1–2 km. It is possible that locally low larval supply and space competition with *A. nodosum* and other fucoids precluded recruitment.

Rockpool contiguity did not have any measurable impact on species richness and individual rockpool assemblages did not differ between treatments; each individual rockpool within the R3 and R5 treatments resembled a standalone habitat patch on the seawall with its own unique assemblage as if a single isolated pool. Therefore, the higher species richness in zones and seawall sections with three or five rockpools, compared to sections with single rockpools is assumed to be because of greater overall rockpool habitat area. As such, both hypothesis 1 and 2 have support from the data, but not for the reasons thought. Equally, single rockpool treatments also showed increased diversity from control sites in many cases, especially within zone 2 measurements. At 36 months, although treatment replication is low, the initial slopes on species accumulation (SAC) curves for each rockpool treatment (Fig. 5a) are at least as high for single rockpool deployment as for multiple rockpools. The stepped profiles of the continuous R5 SAC curves indicate that new species are encountered at different locations along the sea wall as the habitat changes. Therefore, with a limited availability of

Table 2
Two-way ANOVA of density of limpets (*Patella vulgata*) in seawall treatments at Sandbanks at two positions (above and between pools) at 36 mo. census.

	df	Sum. Sq	Mean Sq	F	*P*-value
Treatment	5	3.4754	0.6951	12.68	<0.0001
Position	1	0.0013	0.0013	0.024	0.878
Treatment*Position	5	0.2317	0.0463	0.845	0.519
Residuals	278	15.238	0.0548		

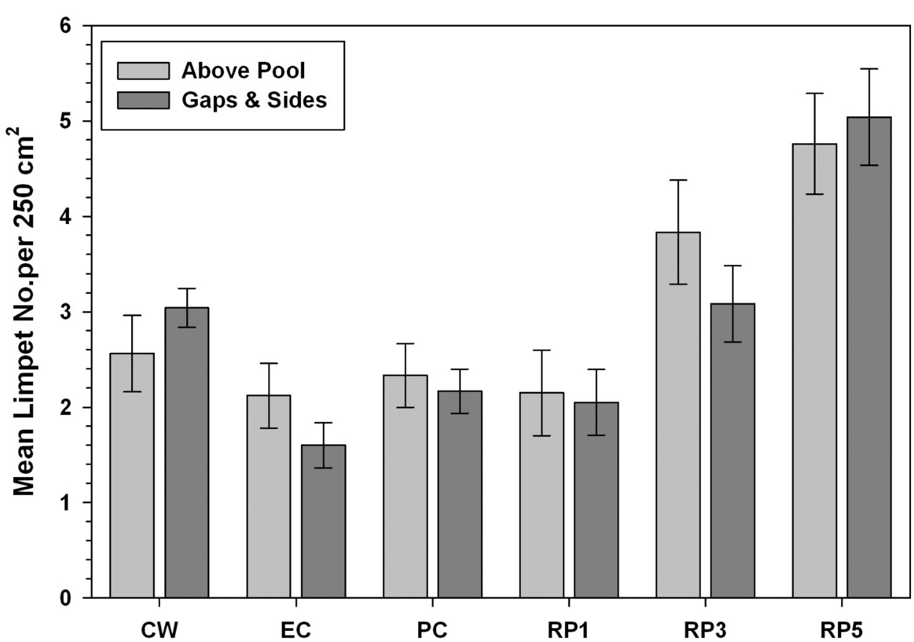

Fig. 8. Density of limpets (*P. vulgata*) on the seawall at positions immediately above rockpools and in gaps between/ beside rockpools at 36 months. Treatments CW, control wall, EC, experimental control, PC, procedural control, R1, single rockpool, R3 three rockpools, R5, five rockpools. See text for further details. Error bars show ± SE.

rockpools, overall species richness on the structure may be greater if installation of single rockpools was spread over a larger habitat area, which are exposed to greater range of environmental conditions, than a smaller area of semi-contiguous pools. At Bouldnor, where the benthos was almost exclusively algae, any physical contact between rockpool units might offer little benefit other than allowing vegetative growth to extend between rockpool exterior habitats. It has been argued that the creation of ecologically connected surfaces and habitats on artificial structures, thus mimicking habitat natural habitats, is a major challenge for the design of eco-engineering solutions on marine infrastructure (Bishop et al., 2017; Strain et al., 2017). It has been demonstrated that the artificial rockpools at Sandbanks also provide a refuge for invertebrates and fish at high tide (Bone et al., 2024). However, any differences in species behaviour between semi-contiguous and isolated pools requires further investigation.

The seawalls in the two study sites differed with respect to their initial assemblages, which were regionally representative of harbour (Sandbanks) and sheltered coast (Bouldnor). Although artificial habitats may provide novel biodiversity on regional scales (O'Shaughnessy et al., 2023), species richness of coastal infrastructure is usually considered less than in natural habitats (Chapman and Bulleri, 2003; Moschella et al., 2005; Firth et al., 2016). Yet both these walls were not sparsely colonised. At Sandbanks, there was a moderate diversity of species present across the tidal range and at Bouldnor a very dense canopy of fucoid algae. It was therefore evident at the outset that the installations would create a measure of physical disturbance to these assemblages and that any full assessment of net gain or loss in species richness through ecological enhancement would require evidence of recovery. Monitoring of the treatments continued for 36 months, which is longer than most previous studies of enhancements on coastal infrastructure (Strain et al., 2018a; Firth et al., 2024). During the monitoring period the region experienced extreme and record-breaking air temperatures and the highest sustained sea temperature ever recorded on the south coast of England. In February 2022, three severe storms hit the south coast of England with the highest ever gust of 106 Kt (122 mph) recorded at the Needles, in the study region at the west end of the Isle of Wight. Although species richness of control treatments remained relatively stable throughout the 36 months at Bouldnor, there was a decreasing trend across controls at Sandbanks. This was also evident in Zones 1 and 3 on the experimental wall. Greater desiccation stress and/or dislodgement may have been responsible for the loss of canopy algae (*F. spiralis*), red turf and associated fauna on control sections and the control wall at Sandbanks. However, species richness in the rockpools bucked this trend. The maintenance of species richness in zones with pools and the presence of species elevated from lower zones demonstrates the potential refugia from climatic extremes the rockpool habitat offers (Hawkins et al., 2019).

There was no significant difference in species richness between R3 and R5 pool sections at both sites, so if this metric was the prime target objective, then the establishment of an array of three pools at this level could suffice. In R5 treatments, the pools occupy most of Zone 2, whereas in R3 the wall assemblage remains extant either side of the array, effectively increasing habitat diversity in that zone, which may be a preferred and cheaper option.

An aspiration for the design of ecologically enhanced infrastructure is to reduce the colonisation and coverage of non-native species (NNS) that are increasingly common on urban coasts and could become invasive (Dafforn, 2017; O'Shaughnessy et al., 2020). Yet relatively few studies have reported NNS (Strain et al., 2018a). Although the number of NNS at Sandbanks increased in sections with multiple pools, species presence did not appear to be associated with pool contiguity. Colonisation by NNS has been shown to be reduced if native species richness and coverage is higher and available space (resource) is lower (Stachowicz et al., 2002), although in marine algal assemblages the prevalence of functional types, such as canopy species, may also be important (Arenas et al., 2006). New colonists were mostly species that required continuous submergence, such as algae and tunicates and which were ubiquitous within rockpool and shallow water habitats in the region (Herbert et al., 2022; Collins et al., 1990). At Sandbanks, of the twelve NNS recorded in wall sections with rockpools, several early colonists were known opportunists of freshly exposed surfaces (Airoldi and Bulleri, 2011; Mineur et al., 2012; Johnstone et al., 2017). At 36 mo. the proportion of NNS of the total number of species at Sandbanks in wall sections with rockpools (12 %) was higher than in non-rockpool sections (6 %), nearby Boscombe Artificial Surf Reef (after five years) (6 %) (Herbert et al., 2017), and intertidal and lagoon habitats in Poole Harbour generally (9 %) (Herbert et al., 2022). The risk of NNS colonisation in rockpools was advised by O'Shaughnessy et al. (2020), yet on this busy urban coastline, all species are commonly found in natural habitats and artificial habitats.

The native oyster *Ostrea edulis*, a protected species in the UK, that settled in the rockpools at Sandbanks may occasionally be found on rocky and mixed sediment shores at extreme low water springs (ELWS), though normally in subtidal habitats. Settlement on concrete has been demonstrated experimentally (ter Hofstede et al., 2024) and the elevated position may have been a refuge from higher predation at lower tidal levels. In Sydney Harbour, higher habitat complexity has been found to reduce mortality of native oysters from fish predation (Strain et al., 2018b). Yet there appeared to be no association between survival and pool treatment in this work. Fitness and functionality of colonists within artificial habitats and ecologically engineered interventions is infrequently reported and there are concerns that some structures and interventions may create ecological traps (Hale and Swearer, 2016; Swearer et al., 2021). However, evidence of normal gonad and shell growth in the pools suggest this may not be the case here and is of interest considering current attempts to restore native oyster reefs.

4.1. Assemblages

Although convergence between disturbed and undisturbed areas was achieved after 36 months in terms of species richness, differences in species composition remained between disturbed (procedural) and experimental controls. At Sandbanks, reduced density, and abundance of *A.modestus* and alga *Blidingia minima* in Zone 1 could partially be due to initial disturbance, yet unusually high air temperatures experienced during the monitoring period may also have resulted in higher mortality and desiccation. Additionally, however there was a significant movement of limpets down the wall towards the pools from a level close to the upper limit of their vertical range, which could have resulted in higher grazing pressure and 'barnacle bulldozing' (Dayton, 1971). This resulted in significantly higher limpet densities in Zone 1 immediately above and in gaps between the pools in R3 and R5 sections, but not R1. Limpets are known to make short foraging and homing movements across the intertidal zone (Hawkins, 1981; Hawkins and Hartnoll, 1983) and may favour shady gaps between pools that offer thermal refugia, during hot weather (Lima et al., 2015; Virgin and Shiel, 2023) or protection from avian predators (Coleman, 2008). Although few *P. vulgata* were found within pools, a closer proximity to the pools may provide cooling and access to algae on the damp horizonal pool rim. This is the most tangible and convincing impact of semi-contiguous pools on assemblages observed during the monitoring period. For Sandbanks Zone 1, Hypothesis 3 is supported; dense pool arrays can have an impact on assemblages, however there was no significant difference in assemblages between treatments in Zone 3.

At Bouldnor the *A.nodosum* canopy, that was partially removed to facilitate rockpool installation, had not fully recolonised by 36 months and this space became occupied by dense *F.spiralis* and *F. vesiculosus*. It is known that physical removal of *A.nodosum* canopy on rocky shores can take over 12 years to recover fully (Jenkins et al., 1999, 2004) and that the understory can become bleached and damaged. Yet this was not observed at Bouldnor, possibly because of the north aspect and the presence of sufficient canopy cover remaining. It is unclear whether *A.*

nodosum cover will recover fully within Zone 2 in R3 and R5 sections where a significant proportion of available space is occupied by the pools. To some extent, this will depend on whether the species colonises the underside of pool units. Investigations by Hall et al. (2019) and authors surveys on larger, Vertipools™ installed nearby, suggest that approximately 7 % cover of *A.nodosum* is possible after five years and 40 % cover achievable after ten years, except for the more exposed out edges of the pools.

This study focussed on the differences in species richness (alpha (α) diversity) and assemblages between different configurations of artificial rockpools. Further investigations would benefit from use of other metrics including functional richness and composition biomass, and the contribution of enhancements to β-diversity, within and among sites, and regional gamma (γ) diversity (O'Shaughnessy et al., 2023).

5. Conclusions

An important objective of environmental enhancement is to replicate the species richness and variety of habitat and contiguity found on natural rocky shores. By casting more shade, evaporative cooling and creating crevices, overhangs and gaps between pools, arrays of multiple retrofitted rockpool units have the potential to create higher habitat diversity and species richness. However, we did not find this. In fact, with a limited availability of pools, rather than arrays of multiple pools fixed at the same zone in a small area, there may be at least as much, if not more, benefit from the installation of single pools spread over a larger area and tidal zone that may experience a greater range of environmental conditions. There may be benefits of contiguous arrays for mobile species at high tide, however this is yet untested. There are almost endless possible configurations of pools that can be tried such as vertical arrays spread over the tidal range and variation in pool size. Pools can be included within tiles of different shapes and sizes (Bishop et al., 2022). The merits of larger and deeper pools may also be worth investigating.

CRediT authorship contribution statement

Roger J.H. Herbert: Writing – original draft, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **Jessica R. Bone:** Writing – review & editing, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Alice E. Hall:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Conceptualization. **Stephen J. Hawkins:** Writing – review & editing, Conceptualization. **Rick Stafford:** Writing – review & editing, Funding acquisition, Formal analysis, 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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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2024.107432.

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