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1 **Can transplanting enhance mobile marine invertebrates in ecologically engineered rock**
2 **pools?**

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14 Running headline: Artificial rock pool colonisation

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24 **Abstract**

25 The field of eco-engineering has burgeoned in recent years in response to the proliferation of
26 artificial structures. Adding water-retaining features to seawalls has been successful in
27 increasing biodiversity relative to the surrounding structure. Artificial rock pools may not,
28 however, completely mimic natural rock pools. Here, we compared natural colonisation,
29 through dispersal and recruitment, of intertidal mobile species to water-retaining flowerpots
30 on seawalls with that into rock pools. This represents the more usual ‘passive’ approach to
31 eco-engineering where features are built to enhance biodiversity and are allowed to colonise
32 naturally, as opposed to seeding or transplanting organisms to features. While flowerpots
33 supported some mobile species not found on the seawall, other species common on natural
34 shores did not recruit to flowerpots. Thus, in a second experiment we tested the effectiveness
35 of an ‘active’ approach through transplanting mobile organisms to flowerpots to expedite the
36 colonisation process. For the species examined, however, most individuals did not stay in the
37 flowerpots for more than 24 hours after being transplanted. Further understanding of the
38 processes (e.g. dispersal distances, recruitment) influencing colonisation of eco-engineered
39 habitats is needed to effectively inform management of marine infrastructure, particularly for
40 projects targeted at restoration rather than enhancement.

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42 Key words: Benthic ecology; Biodiversity; Coastal structures; Ecosystem management;
43 Green engineering; Seeding; Sydney; Urbanisation

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48 **1. Introduction**

49 Artificial structures are proliferating in the coastal environment (Dugan et al., 2011; Firth et
50 al., 2016a). In some areas, more than 50% of the natural foreshore has been replaced by
51 artificial structures, such as seawalls, breakwaters and pilings (Dugan et al., 2011; Chee et al.,
52 2017). In addition to loss and fragmentation of natural habitats (Goodsell et al., 2007; Heery
53 et al., 2017), artificial structures provide novel habitats for colonisation, which often support
54 fewer (Chapman, 2003; Gittman et al., 2015) and different combinations of species (Connell
55 and Glasby, 1999; Davis et al., 2002; Bulleri and Chapman, 2004; Bulleri et al., 2005; Lam et
56 al., 2009) in comparison to those on natural hard substrata. In response to the impacts of
57 artificial structures on marine communities (Bulleri and Chapman, 2010; Bishop et al., 2017),
58 an increasing research effort has focused on ecological engineering of artificial structures to
59 facilitate greater colonisation of native species (see reviews: Chapman and Underwood, 2011;
60 Dafforn et al., 2015a; Firth et al., 2016a). A fundamental difference between artificial
61 structures and natural shores is the lack of physical complexity and microhabitat diversity
62 (e.g. rock pools, crevices, pits) on artificial structures for organisms to colonise (Chapman,
63 2003). To combat the homogeneous habitat provided by much built infrastructure, adding
64 complex surfaces and novel habitats either retrospectively or during construction has been
65 successful at increasing biodiversity relative to that which is found living on the surrounding
66 structure (Chapman and Underwood, 2011; Firth et al., 2014a).

67 In intertidal systems, mobile organisms are notably absent from artificial structures
68 (Chapman, 2003). The majority of work in the intertidal zone has been done on groynes and
69 seawalls (Strain et al., 2018), hence that is where most of our current knowledge comes from.

70 Although certain mobile species have been recorded in greater abundances on seawalls
71 compared to rocky shores (e.g. some limpets and chitons; Bulleri et al., 2005), for the
72 majority of mobile species, abundances tend to be lower than on natural rocky shores (Bulleri
73 and Chapman, 2004; Bulleri et al., 2005). Further, a subset of mobile species that are found
74 on rocky shores are not found on seawalls (Chapman, 2003), and similar patterns have been
75 observed for other structures (e.g. groynes and breakwaters, Firth et al., 2013; Aguilera et al.,
76 2014). Species that are unable to live on artificial structures are often those found in
77 microhabitats that tend to be absent on walls, such as rock pools and deep crevices
78 (Chapman, 2003; Firth et al., 2013). Seawalls can vary in geology and topography (Chapman,
79 2003), and walls that are characterised by more crevices can support greater numbers of
80 particular mobile species (Moreira et al., 2007). Similarly, some gastropods (e.g. *Morula*
81 *marginalba*) only inhabited seawalls when other habitat-forming organisms were present (e.g.
82 oysters) (Jackson et al., 2008). Thus, the introduction of eco-engineered microhabitats (e.g.
83 rock-pools, crevices, pits) might enable the range of certain species to extend onto seawalls
84 that would otherwise not be found there and therefore enhance native biodiversity on
85 seawalls.

86 A 'passive' approach has been taken for a majority of eco-engineering projects to
87 date. This is the "build it and they will come" scenario (Palmer et al., 2008), where eco-
88 engineered features are installed and an emergent assemblage as a product of recruitment
89 and/or dispersal and succession is measured (hereafter 'colonisation') after a period of time.
90 Colonisation of mobile species that are not found on nearby existing artificial structures to
91 eco-engineered habitats will depend on recruitment from the water column (Chapman and
92 Underwood, 2011). ~~In contrast,~~ Any mobile species found on the existing artificial structure

93 would be able to disperse into eco-engineered habitats by moving from the surrounding
94 structure (Underwood, 1977). Thus, the assemblage that colonises eco-engineered habitats
95 will depend on both the pre-existing surrounding benthic assemblage as a source for dispersal
96 and the larval pool as a source for recruitment.

97 Differential recruitment is one of the simplest models that could explain any
98 differences between assemblages found on natural shores and those in eco-engineered
99 habitats. Indeed, differential recruitment of organisms has been shown to help explain the
100 difference in intertidal assemblages found on seawalls and rocky shores (Bulleri, 2005), as
101 well as between shaded and non-shaded habitat on seawalls (Blockley and Chapman, 2006),
102 on seawalls which are sheltered or exposed to wave action (Blockley and Chapman, 2008)
103 and on seawalls of different materials (Iveša et al., 2010). There is considerable research over
104 many decades showing that small-scale habitat features, such as free space (Bracewell et al.,
105 2013), surface texture and chemistry (Coombes et al., 2015; McManus et al., in press), holes,
106 cracks and crevices (Kohler et al., 1999), patterns of water flow (Mullineaux and Butman,
107 1991) and interactions between various factors (Knights et al., 2012) influence recruitment of
108 intertidal species. Recruitment plates or pads are often used experimentally to provide a new
109 and standardised substratum to test for differences in recruitment among particular treatments
110 (e.g. McGrath et al., 1994; Blockley and Chapman, 2006). Many intertidal mobile species,
111 however, naturally recruit to softer biogenic substratum, such as algal turfs (Chapman et al.,
112 2005; Matias, 2013). For these species, artificial units of habitat (AUHs) made of synthetic
113 turf have been used as algal mimics to study colonisation to different habitats (Kelaher, 2005;
114 Matias, 2013). Artificial units are recruited to quickly by a large diversity and number of

115 intertidal mobile invertebrates, including juveniles of those found as adults in rock pools and
116 on open rock (Matias, 2013).

117 With respect to eco-engineering of artificial shorelines, the focus has been on
118 comparing assemblages among natural and/or surrounding substrata at various stages of
119 succession (e.g. Browne and Chapman, 2014; Firth et al., 2014b; Evans et al., 2015), but few
120 studies have focused specifically on early life history stages (but see Coombes et al., 2015).
121 An understanding of how eco-engineered habitats are colonised can be useful to management
122 if a specific species or assemblage is targeted, for example the chance of colonisation of these
123 species may be enhanced by deploying novel habitats at their peak time of recruitment
124 (Chapman and Underwood, 2011). An alternative method of manipulating the assemblage
125 that develops on artificial structures (e.g. seawalls, breakwaters) or eco-engineered habitats
126 (i.e. the added habitat) is to take a more ‘active’ approach by ‘seeding’ the structures with
127 cultured species (eg. oysters ~~or kelp~~, Strain et al., 2017; ~~Walls et al., In review.~~), or through
128 transplanting individuals from natural populations (eg. algae, Perkol-Finkel et al., 2012). An
129 active approach may be taken by a way of ‘nudging nature’ to target desirable species or
130 expedite the colonisation process and does not depend on the vagaries of natural recruitment.
131 Such active management as a tool has not been tested for mobile invertebrates.

132 Here, water-retaining ‘flowerpots’ (Browne and Chapman, 2014; Morris et al., 2017)
133 were attached to vertical seawalls in Sydney Harbour, Australia, with the objective to provide
134 habitat for at least some of the mobile species that were lacking on seawalls (Chapman,
135 2003). Indeed, both previous (Chapman and Blockley, 2011; Browne and Chapman, 2011,
136 2014) and subsequent (Morris et al. in press) studies showed that the addition of water
137 retaining features (cavities and flowerpots) onto seawalls in Sydney Harbour extended the

138 habitat for some species that were otherwise rare, or not found on that seawall. However,
139 other species that were common on natural shores in the harbour, were not present on the
140 seawall and did not recruit to artificial rock pools. On natural shores, some of these species
141 are confined to pools at low tide (e.g. starfish). Some gastropods, however, may show
142 preference for rock pools (e.g. *Austrocochlea porcata*, Underwood, 1976) but can be found
143 on emergent rock, and some use rock pools as refugia under harsh conditions (Underwood,
144 1977).

145 In two sequential experiments, we tested passive and active approaches to facilitate
146 the colonisation of mobile invertebrates into flowerpots. For the passive approach, we
147 compared the emergent assemblage (as a product of recruitment and dispersal from the
148 surrounding habitat) of mobile species between natural rock pools and flowerpots. Further,
149 we measured recruitment to rock pools and flowerpots using AUHs to determine whether
150 differences in mobile species was due to a lack of recruitment into flowerpots, or the
151 dispersal of species into rock pools from surrounding emergent substrata. Where species did
152 not recruit into flowerpots, an ‘active’ experiment tested whether it was possible to introduce
153 individuals by transplanting target species into flowerpots. Specifically, we predicted: (1) the
154 assemblage (i.e. composition and abundance) of mobile species will differ between
155 flowerpots and natural rock pools; (2) any differences in mobile species between natural rock
156 pools and flowerpots was because the mobile species not found in pots only recruit to natural
157 rock pools; (3) alternatively, species only found on natural shores can disperse between
158 natural pools and the surrounding rock, but cannot disperse into flowerpots, as these species
159 are not found on seawalls and; (4) when these species are transplanted into flowerpots, the

160 proportion that remains in flowerpots will be equal to those translocated into natural rock
161 pools ~~because flowerpots do serve as suitable habitat for these species.~~

162

163 2. Methods

164 2.1 Experiment 1: Passive approach

165 Concrete flowerpots (7L, 315 mm diameter, modified from those developed by
166 Browne and Chapman (2011)) were attached to seawalls at the mid-shore tidal level at
167 Blackwattle Bay (33.8720°S 151.1991°E, Figure 2), Sydney Harbour (Morris et al., 2017;
168 Figure 1b). Twenty flowerpots were deployed onto concrete and sandstone seawalls in
169 December 2013 (10 flowerpots) and February 2014 (10 flowerpots) at four sites (5 at each
170 site, ~4m apart) that were separated by at least 100 m. Natural rock pools were sampled at
171 Mrs Macquarie's Point (33.8597°S 151.2225°E), which was the closest natural rocky shore
172 (Figure 2). The rocky shore at Mrs Macquarie's Point was divided into two sites (40 and 50
173 m long), separated by at least 100 m. Natural rock pools had established communities and
174 were therefore cleared of all biota at the start of the experiment to make the substrata
175 comparable to new flowerpots with respect to any existing biota. To clear the pools, sponges
176 were used to drain the water and sessile invertebrates were removed using a metal paint
177 scraper and algae were removed with a metal brush. The pools were then washed with ~~dilute~~
178 hydrochloric acid (Bondall, 320 g/L) to remove any remaining biofilms (Underwood and
179 Skilleter, 1996).

180

181 *2.2.1 Colonisation of flowerpots and natural rock pools through dispersal and recruitment*

182 Natural rock pools and flowerpots (the same over time, n=3) were sampled fortnightly
183 for the first four months following installation of the flowerpots in December 2013, monthly
184 thereafter until 1 year and quarterly until 18 months. The results presented here are part of a
185 larger dataset comparing different types of artificial rock pools, which determined the
186 replicate number (Morris, 2016). Mobile animals were counted for the whole available
187 substratum and abundance standardised to number per unit area (density). To account for
188 potential non-independence over time, analyses were done for the last time point only,
189 however, presence/absence of all species over the entire 18 months has been presented in
190 Table 1. A multivariate difference in the assemblage between rock pools and flowerpots was
191 determined using permutational multivariate analyses of variance (PERMANOVA, Anderson
192 2001) from 9999 permutations of the data. If there were not enough permutations to get a
193 reasonable test (less than 100), the Monte-Carlo P value was used (Anderson *et al.* 2008). A
194 SIMPER analysis (Clarke 1993) was used to identify the taxa contributing most to significant
195 differences between habitats. A threshold of $\geq 10\%$ contribution was chosen to select the
196 most ‘important’ taxa (Bulleri 2005).

197

198 *2.2.2 Patterns of recruitment to AUHs between flowerpots and natural rock pools*

199 In February 2014, rock pools (n=5) were cleared at Mrs Macquarie’s Point at the
200 same time as the flowerpots were installed, as described above. In each replicate rock pool or
201 flowerpot, one AUH was deployed to measure recruitment ($10 \times 10 \text{ cm}^2$, polyethylene, 10
202 mm frond length, 66.2 fronds per cm^2 ; Grass Manufacturers Pty Ltd, Sydney, NSW,

203 Australia). AUHs were deployed vertically into pools and pots (Figure 1 c,d). They were
204 attached using cable ties to saddle brackets in rock pools, which were attached to the
205 substratum with screws before the start of the experiment. Drilling may have damaged
206 flowerpots, so instead a steel rod was placed across the diameter of the pot to which the cable
207 ties were attached. Attachment methods meant that there was a gap between the substratum
208 and AUHs in both treatments. This was kept as consistent as possible as the deployment
209 method of AUHs can have an effect on the colonising assemblage (Chapman *et al.*, 2008).
210 The use of turfs to measure recruitment avoided the confounding factor of the existing
211 assemblage on recruitment, as the AUHs could be removed and replaced with new ones,
212 which created a “bare” surface for recruits.

213 Retrieval and replacement of AUHs was done every two months for a year. ~~Although~~
214 ~~we did not have a hypothesis about temporal patterns,~~ Recruitment was measured for one
215 year to maximise sampling all species that recruited at different times over the year
216 (Underwood and Fairweather, 1989). To retrieve the AUHs a plastic bag was placed over
217 each AUH, which prevented individuals escaping and the cable ties were cut to release the
218 turf from the brackets. The entire unit was preserved in 7% formalin-seawater before being
219 rinsed with freshwater over a 200 µm sieve. The animals were then sorted under a dissecting
220 microscope. All decapods, echinoderms and molluscs were identified to the greatest possible
221 taxonomic resolution (e.g. Chapman *et al.*, 2008). Decapods, echinoderms and molluscs were
222 the focus for this study as these were the three main groups forming the emergent mobile
223 assemblage of rock pools and flowerpots (Table 1). Data were summed for each species over
224 the year and the hypothesis that the total number of recruits of certain species differed
225 between habitats was tested using using χ^2 tests. These tested any significant difference from

226 the expected frequencies of equal counts of individuals in each habitat. Where the expected
227 value was less than 5 in the χ^2 calculations, an exact binomial test was used, which is robust
228 to a small number of observations (Sokal and Rohlf, 2012).

229

230 *2.2 Experiment 2: Active approach*

231 A transplant experiment of those mobile species that were found in rock pools but not
232 (or rarely) flowerpots, from pools to flowerpots, was done to investigate whether ~~seeded~~
233 individuals could persist in these artificial habitats. At Blackwattle Bay, a further installation
234 of flowerpots in January 2016 was used for this experiment (n=6). The flowerpots were left
235 for a few weeks prior to the start of the experiment, which allowed a biofilm to form
236 (MacLulich 1986, Murphy & Underwood 2006). An additional natural rocky shore at
237 Fairlight (33.7960°S 151.2770°E) was used in this experiment (Figure 2). At each location, 2
238 sites (15 m) were chosen, ~30 m apart. Four treatments (Chapman, 1986) were randomly
239 allocated to six rock pools each (or flowerpots; 3 per site): control (untouched animals at
240 Fairlight); disturbance control (animals picked up and placed back in the same rock pool at
241 Fairlight); translocation control (animals were disturbed and moved from Fairlight to natural
242 rock pools at Mrs Macquarie's Point) and transplanted animals (those that were disturbed,
243 moved from Fairlight to Blackwattle Bay and placed in flowerpots). Fairlight was used as the
244 control location in this experiment as Mrs Macquarie's Point was the closest rocky shore to
245 Blackwattle Bay and this allowed the translocate and transplant treatments to be similarly
246 distant from the control location (ca. 8 and 11 km, respectively), which avoided potential
247 confounding effects of geographic proximity on the interpretation of the behaviour of the
248 animals. The design of this experiment allowed determination of whether the outcome was

249 caused by the disturbance of the animals, movement to a new location or movement to a new
250 environment (Chapman, 1986).

251 Three of the most abundant species found in natural rock pools in Sydney Harbour
252 were used in the experiment: the snails *Bembicium nanum* and *Austrocochlea porcata* and the
253 starfish *Parvulastra exigua*. These species were either much less abundant (*A. porcata*) or
254 absent from flowerpots (*B. nanum* and *P. exigua*) or the seawall (Table 1). Ten individuals of
255 each species were transplanted in individual replicate groups (30 individuals = 3 replicates in
256 total), and the proportion (%) of individuals remaining per group was used as the response
257 variable. Starfish and snails at Fairlight were initially removed from replicate rock pools,
258 tagged and placed back in their original pools for the control treatment. Snails were marked
259 with shellfish tags and starfish were tagged using visible implant elastomer (Martinez et al.,
260 2013). The control animals had to be disturbed to be tagged as it was not possible to mark
261 individuals whilst underwater in rock pools. Therefore, tagging of control animals was done
262 24 hours prior to the start of the experiment to allow recovery from the disturbance, so at the
263 start of the experiment they were considered undisturbed (Chapman, 1986; Underwood,
264 1977). Individuals in the other treatments were tagged the next day with different coloured
265 tags. Animals from the disturbed treatment were tagged at Fairlight and placed back in the
266 rock pool from which they were taken. Other marked animals were translocated or
267 transplanted to natural or artificial rock pools at Mrs Macquarie's Point and Blackwattle Bay,
268 respectively.

269 The following day, the number of marked individuals found in or out of rock pools or
270 flowerpots were quantified. The experiment was repeated twice; the sites used at Fairlight
271 were different each time because a relatively large number of animals needed to be collected

272 to translocate and transplant. The null hypothesis that the proportions of starfish and snails
273 remaining in and out of pools were similar for all treatments were tested using a 3-way
274 ANOVA (Treatment: 4 levels and orthogonal; Experimental run: 2 levels and random; Site: 2
275 levels, random and nested in experimental run) using GMAV5. When necessary, data were
276 arcsine transformed to stabilise variances (Underwood, 1997).

277

278 3. Results

279 3.1. Passive approach: colonisation of flowerpots and rock pools

280 Of the 35 taxa found in total over 18 months, 25 were unique to natural rock pools
281 and three were unique to flowerpots (Table 1). ~~Of the seven species that were common to~~
282 ~~both habitats, the molluscs: *Austrocochlea porcata*, *Patelloida mimula*, *Siphonaria*~~
283 ~~*denticulata* and an unidentified species of nudibranch had higher density in natural rock pools~~
284 ~~compared to flowerpots (Table 1). Conversely, *Bedevea hanleyi*, *Bembicium auratum* and fish~~
285 ~~were more abundant in flowerpots than natural rock pools, although the former two were~~
286 ~~already present on the seawall (Table 1). At the end of the end of the experiment, there was a~~
287 significant effect of treatment on mobile species assemblage (PERMANOVA, Pseudo- $F_{1,2} =$
288 4.87, $P = 0.03$). Three taxa contributed more than 10% to the differences in the mobile
289 assemblage between treatments: *S. denticulata*, *B. auratum* and *A. porcata*. *Siphonaria*
290 *denticulata* and *A. porcata* were more abundant in natural rock pools, whereas *B. auratum*
291 were more abundant in flowerpots (although this species was already present on the seawall,
292 Table 1). Juveniles of some of the common grazing molluscs (*Austrocochlea* sp., *Bembicium*

293 sp., *Cellana tramoserica* and *Siphonaria* sp.) were recorded in rock pools, but not flowerpots
294 (Table 1).

295

296 3.2 Passive approach: recruitment to AUHs in flowerpots and rock pools

297 Only four replicates for each time and treatment remained at the end of the study as
298 AUHs were lost. Additionally, only data for the first 8 months of the study were available
299 because AUHs were lost from rock pools due to storms during the last 4 months. A total of
300 149 individuals from 14 taxa were recorded from the three groups: molluscs (15 taxa),
301 echinoderms (2 taxa) and decapods (1 taxon) (Table 2). Only two species recruited to the turf
302 that were relevant to the hypotheses in this study (i.e. they were found in rock pools, but not
303 in flowerpots or on the seawall), *Austrolittorina unifasciata* and *Parvulastra exigua* (Tables 1
304 and 2). Further, few recruits of these species were recorded and there was no significant
305 difference between turf in rock pools compared to flowerpots (Table 2). *Austrocochlea*
306 *porcata* and *Parasesarma erythroductyla* were found as either adults or juveniles in rock
307 pools and/or flowerpots (Experiment 1, Table 1), and as recruits in the turf (Table 2).
308 *Parasesarma erythroductyla* was only found in flowerpots as adults, and only recruited to
309 flowerpots rather than rock pools (Tables 1 and 2). *Austrocochlea porcata* was found in
310 greater densities in rock pools, however, there were few recruits recorded in either habitat
311 (Tables 1 and 2). The rest of the taxa that recruited to the AUHs were not found as adults in
312 either rock pools or flowerpots (Tables 1 and 2).

313

314 3.3 Active approach: transplantation experiment

315 For *Austrocochlea porcata*, the proportion of snails found in flowerpots after
316 transplantation was significantly less than all other treatments (Figure 3a; Table 3).
317 Nevertheless, there was also a significant effect of handling and location to which individuals
318 were moved for *A. porcata* as the proportion of translocated individuals in pools was similar
319 to the disturbance treatment, which were lower than the control (Figure 3a; Table 3). The
320 proportion of animals found outside of flowerpots was greater than the proportions found
321 outside of rock pools in the translocation treatment, and the control and disturbance
322 treatments (Figure 3b; Table 3).

323 The proportion of *B. nanum* remaining in pools after one day varied across treatments
324 for each experimental run (Figure 3a; Table 3). A lower proportion was found in flowerpots
325 compared to the proportions of translocated and control snails at the first experimental run
326 (Figure 3a). When the experiment was repeated, the proportion of *B. nanum* in locations
327 where they were translocated (rock pools on natural shore) and transplanted (flowerpots on
328 seawall) were similar and smaller compared to the control and disturbed treatments (Figure
329 3a). The proportion of *B. nanum* found outside of rock pools and flowerpots in the
330 transplantation and translocation treatments was similar, and these were greater than the
331 proportions of snails found for the control and disturbed treatments (Figure 3b; Table 3).

332 Note that the proportion of animals found in the disturbed treatment was lower than the
333 control treatment, which suggests an effect of handling on *B. nanum* behaviour.

334 The behaviour of *P. exigua* was clearly affected by the transplantation; the proportion
335 of *P. exigua* that remained in flowerpots was smaller than all treatments in natural rock pools
336 (Figure 3a; Table 3). No *P. exigua* were found outside of rock pools or flowerpots.

337

338 4. Discussion

339 Here, we extend previous evaluation of water-retaining flowerpots on seawalls (Browne and
340 Chapman, 2011; Browne and Chapman, 2014) by comparing colonisation and recruitment of
341 mobile species to flowerpots with natural rock pools. Some species did recruit to the
342 flowerpots that were not found on the seawall (e.g. fish, nudibranch). Of the species that were
343 unique to natural rock pools (71%), few recruits were recorded in the AUHs in either natural
344 rock pools or in flowerpots. This suggests that the differences observed in the patterns of
345 mobile species between rock pools and flowerpots, were likely due to dispersal of adults into
346 rock pools from the surrounding emergent rock, rather than recruitment from the water
347 column. As recruitment of common rock pool species was limited, the effectiveness of
348 transplanting adults from natural shores to ~~seed~~ the flowerpots was tested. The results for *B.*
349 *nanum* were varied, with the proportion of individuals that stayed in flowerpots less than the
350 other treatments for the first experimental run, but similar to those moved to rock pools in the
351 second run. Similarly, the behaviour of *Bembicium auratum* varied among time and space in
352 previous translocation experiments (Crowe and Underwood, 1999). In contrast, fewer *A.*

353 *porcata* and *P. exigua* remained in flowerpots than the other treatments, which suggests that
354 flowerpots may not necessarily serve as a suitable habitat for these species.

355 Previous research has shown that intertidal species more frequently found in rock
356 pools and other microhabitats that provide shelter are absent from artificial structures
357 (Chapman, 2003; Moschella et al., 2005). Eco-engineered habitats can increase the number of
358 species found on artificial structures in comparison to the surrounding habitat (e.g. Chapman
359 and Underwood, 2011; Browne and Chapman, 2014; Firth et al., 2014a). Notably, some
360 mobile species were reported in the initial evaluation of the flowerpots (Browne and
361 Chapman, 2011) that were not recorded here. Obviously, this could be due to the many
362 differences between the experiments, including type of pots and timing and location of pot
363 deployment. Indeed, the mobile species assemblage found in flowerpots has been shown to
364 be variable between locations (Browne and Chapman, 2014). Further, here the flowerpot and
365 rock pool treatments were in different areas of the harbour and could have been influenced by
366 other localised processes (e.g. water flow or exposure to wave action, Mullineaux and
367 Butman, 1991; Firth et al., 2016b), separate from the effects of habitat. Unfortunately,
368 location and habitat could not be un-confounded in this study because there was no
369 permission to install the flowerpots onto seawalls where natural rocky shore was also present.
370 Rock pools in different locations of Sydney Harbour, however, have the same common
371 species present (Chapman and Blockley, 2009; Morris, 2016). Therefore, it could be expected
372 that these species would also be present where the flowerpots were installed, if there was
373 suitable habitat.

374 Few studies have made comparisons between eco-engineered and natural features (but
375 see Chapman and Blockley, 2009; Evans *et al.* 2015). Even if the goal of the flowerpots was

376 to simply increase the biodiversity of seawalls rather than to exactly mimic rock pools, a
377 comparison with the nearest natural equivalent is informative to understand what species can
378 do well in eco-engineered habitats, and what species may be missing. This could instruct
379 subsequent management objectives and/or future design of artificial rock pools. For instance,
380 natural rock pools have great variability in aspects such as topography, volume and depth
381 (Astles, 1993). Although, at least for mobile species, there has been little effect of rock pool
382 diameter and slope on the emergent assemblage recorded (Underwood and Skilleter, 1996;
383 Firth et al., 2014b), however, depth has been shown to influence the abundance of mobile
384 species in some cases (Astles, 1993), but not others (Firth et al., 2014b). In accordance with
385 one study that compared inbuilt cavities in seawalls to natural rock pools (Chapman and
386 Blockley, 2009), flowerpots supported some, but not all mobile organisms found in rock
387 pools. Conversely, the mobile species assemblage in drill-cored rock pools on a breakwater in
388 the UK was not significantly different from that in nearby natural rock pools (Evans et al.,
389 2015). The drill-cored rock pools were on a horizontal substratum, whereas flowerpots on
390 seawalls differ from pools on natural rocky shores in that they are surrounded by vertical,
391 rather than horizontal, substrata. Many species that use rock pools forage on the surrounding
392 substrata when conditions allow (Underwood, 1977; Noël et al., 2009; Martinez et al., 2017).
393 If these species cannot use the vertical habitat, then they may not be found in mimics of rock
394 pools deployed on seawalls (Chapman and Blockley, 2009). Alternatively, recruitment
395 processes may be responsible for the differences in assemblages, as colonisation of
396 flowerpots relies on recruitment from the water column for those species that are not present
397 on the seawall to disperse into flowerpots.

398 Here, there was little recruitment of mobile intertidal species to AUHs over 8 months
399 in either flowerpots or natural rock pools. Prior studies have shown that artificial turf units
400 are recruited to quickly by a large diversity and number of intertidal mobile species (Kelaher,
401 2005; Matias, 2013). In contrast to previous studies however, the AUHs in this study were
402 deployed vertically, rather than horizontally, to the substratum. This was done as the
403 predominant habitat in flowerpots was steeply sloping sides with a comparatively small
404 surface area of horizontal base. Orientation of the substratum has been shown to influence
405 patterns of biodiversity in some (Wendt et al., 1989; Glasby, 2000; Firth et al., 2014b), but
406 not all (Firth et al., 2015) studies. Goldberg and Foster (2002) showed reduced recruitment of
407 ~~geniculate~~ coralline algae to vertical seawalls in comparison to horizontal substrata. The
408 method of deployment of artificial units of habitat can have an effect on the colonising
409 gastropod assemblage (Chapman et al., 2008), although this has not been tested with regards
410 to orientation. Testing the effect of orientation on mobile species recruitment could be done
411 using further manipulative experiments. Furthermore, it is unknown whether different results
412 would have been achieved using a different recruitment substratum (i.e. a sandstone or
413 concrete tile). This research would give an insight into the factors affecting recruitment (e.g.
414 orientation, material), and could be useful to inform the design of eco-engineered habitat
415 features that may rely on recruitment for colonisation of species not found on the surrounding
416 seawall.

417 It is likely that any species that are found in artificial rock pools, but not on the
418 seawall have recruited there. Little or no recruitment may be enhanced through transplanting
419 species from cultured or natural populations to artificial habitats prior or following
420 installation (Perkol-Finkel et al., 2012; Strain et al., 2017). Previous studies (Perkol-Finkel et

421 al., 2012; Browne and Chapman, 2014) have not used all of the transplant controls (i.e.
422 disturbance control and translocation) in the experiment, as done here. If the transplant fails,
423 however, this is useful to determine what aspect of the transplantation affected the organisms
424 i.e. disturbance, movement and/or artificial habitat, so methods may be revised in future
425 experiments. For instance, for *B. nanum* movement to another location (i.e. translocation or
426 transplantation), regardless of whether natural or artificial, may have an effect on the
427 behaviour of this species. Conversely, the proportion of *P. exigua* and *A. porcata* that
428 remained in the flowerpots was significantly smaller than the other treatments. This suggests
429 that some aspect of the flowerpot habitat is not suitable for these organisms. Observations at
430 high tide revealed that many of the starfish and snails left the flowerpots immediately when
431 the water lapped the edge of the pots (Morris and Martinez, pers. obs.), and few or none
432 returned the following day at low tide. Perhaps the vertical orientation of the seawall, and low
433 percentage of flowerpot in comparison to seawall habitat makes it difficult for organisms to
434 navigate back to the pots as the tide comes in. This seawall scale effect may particularly be
435 the case for *A. porcata* as they were found in flowerpots on rare occasions naturally.
436 Therefore perhaps flowerpots would be suitable habitats if the percentage of flowerpot habitat
437 relative to seawall was increased. For *P. exigua*, no individuals were recorded outside rock
438 pools. This could be the result of mortality outside of natural or artificial rock pools, or due
439 the difficulty of finding this species once it has left a rock pool, particularly amongst the
440 oyster matrix on the seawall. Notably, *B. nanum* and *A. porcata* transplanted to flowerpots
441 were seen inhabiting the seawall the following day. Whether these individuals can persist and
442 reproduce over time on the seawall needs further study.

443

444 **5. Conclusion**

445 Our study has raised some important considerations for the management of marine
446 infrastructure employing eco-engineering practices, in particular if the goal of eco-
447 engineering is to mitigate the loss of natural habitats (Perkins et al., 2015). Future work
448 should aim to further clarify the various processes contributing to colonisation of eco-
449 engineered habitats using larger-scale, more robustly replicated experiments.

450 Transplantation is increasingly being used to promote the presence of native species
451 on artificial structures, however the primary focus has been sessile habitat-forming
452 organisms, such as corals, macroalgae, shellfish and sponges (Perkol-Finkel et al., 2012; Ng
453 et al., 2015; Strain et al., 2017). Here, there was little recruitment of mobile intertidal species
454 over the year in any habitat. Thus, differences between rock pools and flowerpots were likely
455 due to the movement of adults between pools and the surrounding substratum, which cannot
456 occur in flowerpots as these species are not found on the seawall. Whether there was a supply
457 of larvae at the location where the flowerpots were installed and species were not recruiting,
458 or there was no supply of larvae remains unknown. Larval traps could be used to determine
459 whether larvae are in the water column, but not settling/recruiting or if they are absent around
460 the flowerpots (Chen et al., 2013). Nevertheless, this study does highlight the importance of
461 considering recruitment if the goal of eco-engineering is to achieve an assemblage similar to
462 natural habitats, which may be relatively slow, or might never occur. This may be particularly
463 the case for direct developers, which have lower dispersal than species with larval stages
464 (Hoskin, 1997; Johnson et al., 2001; Barbosa et al., 2013). Here, only two invertebrates with
465 direct development were seen, *P. exigua* and *Bedevea hanleyi*. *B. hanleyi* were already present
466 on the seawall (Table 1), and *P. exigua* has been noted occasionally within the oyster matrix

467 on the wall (Morris, pers. obs.). Although, adults with direct development can disperse by
468 floating or whilst attached to mobile substrata (e.g. wrack), known as rafting (Highsmith,
469 1985; Grantham et al., 2003). Further, it is not known whether the recruitment potential of a
470 particular species is reduced if they are absent from a habitat for too long (Perkins et al.,
471 2015). For example, recruitment of large numbers of sessile species can use all available
472 space in a habitat, making it unavailable for later arriving grazing species (Underwood et al.,
473 1983; Hawkins et al., 1992). Alternatively, some species will only settle in the presence of
474 conspecifics (Sweatman, 1983; Gebauer et al., 2011). The transplant experiment showed,
475 however, that even if species can recruit or if they are artificially seeded, eco-engineered
476 habitats may have design limitations that makes them unsuitable for some species to persist.
477 This research highlights the importance of manipulative experiments to disentangle the
478 processes (recruitment and/or post-recruitment) in artificial systems. It is these processes that
479 will determine the distribution of species colonising engineered features, and a good
480 understanding is needed if we are to move forward in informing management of marine
481 infrastructure.

482

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494

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741 **Table 1.** Summary of the taxa found over 18 months (18 surveys) in rock pools (RP) and
 742 flowerpots (FP). Taxa are indicated as not present (-) or present (+). Taxa indicated as not
 743 present or present on the seawall (SW) are listed from surveys before and after the
 744 installation of the pots (3 sampling times before and after, n = 12; Morris, 2016).

		RP	FP	SW
Amphipods	Caprellid spp.	+	-	-
Decapods	Crab sp. 1	+	-	-
	Crab sp. 2	+	-	-
	Crab sp. 3	+	-	-
	<i>Parasesarma erythroductyla</i>	-	+	+
	Shrimp sp. 1	+	-	-
Echinoderms	* <i>Parvulastra exigua</i>	+	-	-
Fish	Fish spp.	+	+	-
Molluscs	<i>Acanthochitona</i> sp.	+	-	-
	<i>Afrolittorina acutispira</i>	+	-	+
	<i>Austrocochlea porcata</i>	+	+	-
	<i>Austrocochlea</i> juvenile	+	-	-
	<i>Austrolittorina unifasciata</i>	+	-	-
	* <i>Bedevea hanleyi</i>	-	+	+
	<i>Bembicium auratum</i>	+	+	+
	<i>Bembicium nanum</i>	+	-	+†
	<i>Bembicium</i> juvenile	+	-	-
	<i>Cellana tramoserica</i>	+	-	+
	<i>Cellana tramoserica</i> juvenile	+	-	-
	<i>Montfortula rugosa</i>	+	-	-
	<i>Morula marginalba</i>	+	-	+
	Nudibranch sp. 1	+	+	-
	Nudibranch sp. 2	+	-	-
	<i>Onchidella patelloides</i>	+	-	-
	<i>Patelloida alticostata</i>	+	-	-
	<i>Patelloida mimula</i>	+	+	+
	Sea slug sp. 1	+	-	-
	<i>Siphonaria denticulata</i>	+	+	+
	<i>Siphonaria virgulata</i>	+	-	-
	Siphonariid juvenile	+	-	-

	<i>Sypharochiton pelliserpentis</i>	+	-	+
	Unidentified juvenile limpet	+	-	-
Nematodes	Nematode sp. 1	-	+	-
Platyhelminthes	Cream platyhelminth	+	+	-
Polychaetes	Polychaete sp. 1	+	-	-

*Invertebrates with direct development

†One individual found

745 **Table 2.** χ^2 tests for the number of taxa summed over 8 months comparing taxa that had
 746 recruited to artificial units of habitat in rock pools (RP) and flowerpots (FP). Taxa in bold
 747 indicate those taxa there were specific hypotheses about. Statistical tests underlined indicate a
 748 binomial test was used, df = 1. *P < 0.05, **P < 0.01, ***P < 0.001

		RP	FP	χ^2
Decapod	<i>Parasesarma erythroductyla</i>	0	71	71.0***
Echinoderm	<i>Ophiuroid</i>	1	3	<u>ns</u>
	<i>Parvulastra exigua</i>	3	1	<u>ns</u>
Mollusc	<i>Austrocochlea porcata</i>	2	0	<u>ns</u>
	<i>Austrolittorina unifasciata</i>	2	0	<u>ns</u>
	Bivalve 8	0	6	6.0**
	<i>Cantharidella</i> sp.	1	0	<u>ns</u>
	Columbellidae	0	1	<u>ns</u>
	<i>Hiatella australis</i>	8	28	11.1***
	Muricidae	0	2	<u>ns</u>
	Pectinidae	1	3	<u>ns</u>
	<i>Tawera lagopus</i>	0	13	13.0***
	Turridae	0	1	<u>ns</u>
	Veneridae	1	1	<u>ns</u>

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762 **Table 3:** ANOVA results of the proportion of snails and starfish found in or out of pools from
763 the transplant experiment. Data were arcsin transformed to reach the assumption of
764 homogeneity of variances when necessary. Interaction terms were eliminated when the *P*
765 values were greater than 0.25 (Underwood 1997). *Post hoc* SNK test are shown for significant
766 factors.

		Proportion in pools			Proportion outside pools		
<i>Austrocochlea porcata</i>		(Cochran's <i>C</i> = 0.200 ^{ns})			(Cochran's <i>C</i> = 0.336)*		
Source	<i>df</i>	MS	<i>F</i>	<i>P</i>	MS	<i>F</i>	<i>P</i>
Treatment Tr	3	0.8203	37.46	< 0.001	1341.4681	11.52	< 0.05
Experimental run Er	1	0.0133	0.10	> 0.05	38.7768	0.10	> 0.05
Site Si (Er)	2	0.1321	6.04	< 0.01	373.0947	2.67	> 0.05
Tr x Er	3	<i>eliminated</i>			116.4674	0.60	> 0.05
Tr x Si (Er)	6	<i>eliminated</i>			195.2007	1.40	> 0.05
RES	32				139.5776		
<u>SNK results</u>		Tr	TP < TL = DC < C		Tr	C = DC = TL < TP	
		Si (Er)	ER 1: S1 = S2 ER 2: S2 < S1				
<i>Bembicium nanum</i>		(Cochran's <i>C</i> = 0.115 ^{ns})			(Cochran's <i>C</i> = 0.175 ^{ns})		
Source	<i>df</i>	MS	<i>F</i>	<i>P</i>	MS	<i>F</i>	<i>P</i>
Treatment Tr	3	0.5311	4.56	> 0.05	0.3485	13.94	< 0.001
Experimental run Er	1	0.0675	32.4	< 0.05	0.1102	12.9	> 0.05
Site Si (Er)	2	0.0021	0.09	> 0.05	0.0085	0.34	> 0.05
Tr x Er	3	0.1164	4.95	< 0.01	<i>eliminated</i>		
Tr x Si (Er)	6	<i>eliminated</i>			<i>eliminated</i>		
RES	32	0.0235			0.0250		
<u>SNK results</u>		Er	1 < 2		Tr	DC < C < TL = TP	
		Tr x Er	ER1: TP < TL = DC = C ER2: TP = TL < DC = C				

<i>Parvulastra exigua</i>		(Cochran's $C = 0.130^{ns}$ *)		
Source	<i>df</i>	MS	<i>F</i>	<i>P</i>
Treatment Tr	3	0.5194	7.55	< 0.05
Experimental run Er	1	0.2408	7.12	< 0.05
Site Si (Er)	2	<i>eliminated</i>		
Tr x Er	3	<i>eliminated</i>		
Tr x Si (Er)	6	0.0688	2.04	> 0.05
RES	32	0.0338		
<u>SNK results</u>		Tr	TP < C = D = TL	
	Er	1 > 2		

767 ^{ns} not significant; * arcsin transformed; C = control, D = disturbance control, TL = translocation, TP =
 768 transplantation; ER = Experimental run

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771 Figure 1. (a) Cavity rock pools and (b) flowerpot rock pools on seawalls in Sydney Harbour,

772 Australia. Artificial units of habitat in (c) natural rock pools and (d) flowerpots.

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774 Figure 2. Map of artificial (circle) and natural (triangles) shores in Sydney Harbour,

775 Australia.

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777 Figure 3: Proportion of individuals (mean \pm SE) of the snails *Austrocochlea porcata* and

778 *Bembicium nanum*, and the starfish *Parvulastra exigua* found (a) inside or (b) outside of

779 pools within the control (C), disturbance control (D), translocation (TL), and transplantation

780 (TP) treatments. White, grey and black bars are data pooled by sites at experimental run 1 (n

781 = 6) and 2 (n = 6), and pooled by sites and experimental run (n = 12), respectively.

782 *Parvulastra exigua* were not observed outside of rock pools or flowerpots.

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792 Figure 1:



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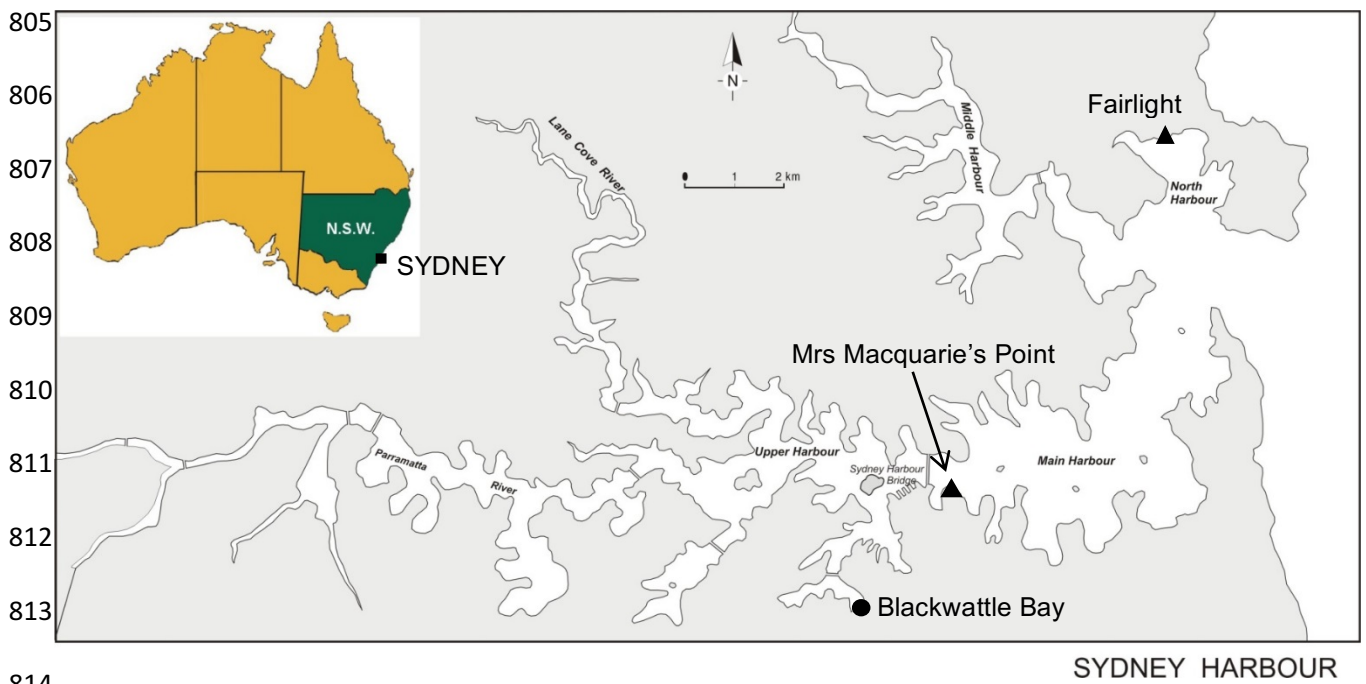
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804 Figure 2:



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Pre-proof: Please see Morris, R.L., Martinez, A.S., Firth, L.B. and Coleman, R.A., 2018. Can transplanting enhance mobile marine invertebrates in ecologically engineered rock pools?. Marine Environmental Research 141: 119-127. <https://www.sciencedirect.com/science/article/pii/S0141113618303404>

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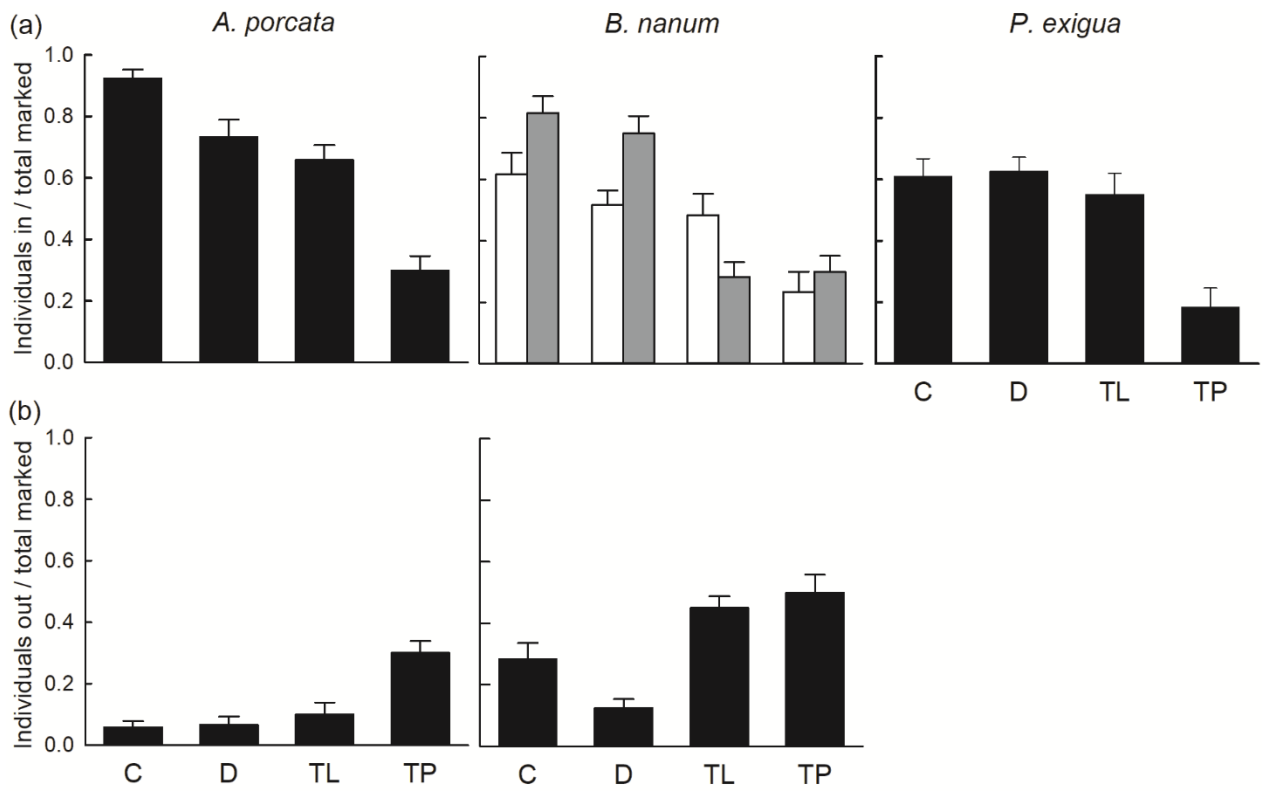
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829 Figure 3:



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