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An ecological assessment of Australia's first community oyster gardens

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Summary **Oyster gardening is a community-driven activity where oysters are grown in cages hanging off docks or other coastal infrastructure. Besides the provision of adult oysters for restoration programmes, oyster gardening may also support other ecosystem services such as providing habitat for fishes and invertebrates as well as encouraging community involvement and citizen science. Australia's first oyster gardening programme was undertaken in a canal estate on Bribie Island in Moreton Bay, Queensland between October 2016 and November 2017. Oyster gardens consisting of plastic mesh cages were deployed with either three species of bivalves (polyculture), or exclusively Sydney Rock Oysters (monoculture) to investigate whether the habitat value differed between the two garden types. After one year of growth, polyculture cages supported higher abundances and species richness of both invertebrates and fish compared to the monoculture gardens. Our study showed that oyster gardening can provide habitat for a range of invertebrate and fish species in the highly modified coastal environment of a canal estate. Further studies are needed to discern whether these oyster gardens would also support larger and mobile fauna, such as species with commercial and recreational importance.**

Key words: *citizen science, Isognomon ephippium, oyster gardening, reef restoration, Saccostrea glomerata, Trichomya hirsuta.*

Implications for managers

- Oyster gardening can provide habitat for a range of invertebrate and fish species in the highly modified coastal environment of a canal estate
- Submerged cages may provide some indication of the habitat value of extinct subtidal oyster reefs
- Oysters grown in cages suspended form pontoons grow rapidly, and can be used to supplement oyster restoration initiatives
- Oyster gardening is ideally suited to citizen science, and generates interest and engagement for local conservation issues

Introduction

Oysters are ecosystem engineers in shallow coastal waters that provide many ecosystem services, including coastal protection, sediment stabilisation and water quality improvements through filtration (Grabowski & Peterson 2007; Zu Ermgassen *et al.* 2020). Reef-forming oyster species often provide a hard substrate in a predominantly soft environment and introduce structural complexity that provides habitat, nursery and feeding grounds for a diverse assemblage of marine organisms (McLeod *et al.* 2019). However, oyster reefs are severely threatened, with over 85% of reefs lost globally (Beck *et al.* 2011). Australian oyster reefs mirror global trends, with a loss of 90% of the two primary reef building species (Sydney Rock Oyster, *Saccostrea glomerata* and the Australian Flat Oyster, *Ostrea angasi*) compared to historical levels (Gillies *et al.* 2018). Further, while historical *Sydney Rock Oyster* reefs in Moreton Bay formed at depths greater than 3.6 meters below the low tide mark (Lergessner 2008), current day Sydney Rock Oysters appear to be functionally extinct in the subtidal environment (Diggles 2013). Growing recognition of the important ecosystem services provided by oyster reefs, combined with an awareness of their substantial historical reductions have led to an increased interest in restoring shellfish reefs in Australia (Gillies & Crawford 2017; McAfee *et al.* 2020).

While oyster reef restoration is relatively new in Australia (Gillies *et al.* 2018), reestablishment of lost oyster reefs and their ecosystem services have been successfully implemented elsewhere over the last four decades (Fitzsimons *et al.* 2019). Oyster reef restoration on the east coast of the United States has often been implemented at large scales using industrial techniques, however, there have also been many small-scale citizen science initiatives. Among these, 'oyster gardening' has been a popular initiative and is increasingly being

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used in other areas of the world (Brumbaugh & Coen 2009; Oesterling & Petrone 2012). Enlisting the help of citizen scientists not only broadens the scale and scope of restoration activities with limited budgets, but can also have altruistic benefits to participants by allowing them to contribute to community needs (Toomey *et al*. 2020; Agnello *et al*. 2022). Oyster gardening is non-commercial oyster aquaculture, typically conducted by citizen scientists or volunteers where individuals grow oysters off docks or other coastal infrastructure in floats or cages (Westby *et al*. 2019). Oysters and other shellfish grown in submerged cages can be harvested for human consumption or can be used as a source of adult broodstock for restoration projects. However, they can also provide further benefits such as improving local water clarity and nutrient cycling, providing habitat for invertebrates and fishes, and supporting research and education (Marenghi & Ozbay 2010; Oesterling & Petrone 2012; Saurel *et al*. 2019).

The rapid expansion of coastal cities has increased the demand for space suitable for development. As a result, natural coastal habitats in highly urbanised areas are being replaced by a proliferation of grey infrastructure (Airoldi *et al*. 2021; Bugnot *et al*. 2021). Canal estates, artificial residential waterways, are increasingly common globally and are constructed to maximise waterfront real estate potential (Waltham & Connolly 2011). Australia has the greatest expanse of residential canal estates in the world, with approximately 440 km of linear urban waterways (Waltham & Connolly 2007). These constructed waterways provide a lower habitat value for estuarine species assemblages compared to natural undisturbed waterways (Brook *et al*. 2018; Waltham *et al*. 2020). Canal estates are dominated by flat vertical concrete walls and soft sediment, and the construction of infrastructure like pontoons can have both negative and positive impacts on invertebrate communities. For example, the introduction of artificial structures may negatively impact invertebrate communities through direct displacement and/or shading (Dafforn *et al*. 2015; Heery *et al*. 2017). Artificial structures have also been found to provide substrates for colonisation by invasive species (Dafforn *et al*. 2009; Obaza & Williams 2018), facilitating their introduction, persistence and spread in developed coastal ecosystems. In contrast, infrastructure like pontoons and seawalls may provide stable substrate for the recruitment of native invertebrates and plants in these artificially bare habitats, thus increasing invertebrate biodiversity (Forrest *et al*. 2009). Due to severe modification, lack of water circulation and excessive nutrient inputs, these systems are often fundamentally different from natural estuarine systems (Waltham & Connolly 2007), with little chance of successful recovery post-development.

While natural habitat restoration is often no longer possible in these developed locations, oyster gardening has proven a popular activity among the citizens of many canal estates overseas (Toomey *et al*. 2020). Participating in these activities can increase citizen scientist engagement with their local environment, improve scientific literacy and environmental awareness (Lewandowski & Oberhauser 2017). In this context, oyster gardening activities could provide an opportunity for both improving the ecological value of highly modified coastal ecosystems like pontoon estates, while engaging the local population with the ecosystem at their doorstep. Further, in substrate-limited areas where restoration of natural oyster reefs is no longer a feasible option, oyster gardens could function as floating oyster reef analogues, potentially providing some of the services of their natural counterparts. However, it remains unknown to what extent oyster gardens mimic their natural counterparts, and contribute to the ecosystem functioning of canal estates.

In Australia, the first community-based oyster gardening programme was undertaken on Bribie Island in Moreton Bay, Queensland between October 2016 and November 2017. This programme was established to assist a shellfish reef restoration initiative in nearby Pumicestone Passage, Moreton Bay. Residents from a canal estate on Bribie Island volunteered to grow out juvenile oysters alongside their floating pontoons. Once oysters reached sufficient size, they were used in experimental trials to restore subtidal shellfish reefs in Moreton Bay (http://restorepumicestonepassage.org). While the trials were focussed on restoring the historically abundant Sydney Rock Oyster (*S. glomerata*) and measuring the effects of reef restoration on invertebrates (Diggles 2018) and fisheries production (Gilby *et al*. 2021), the temporary oyster garden cages provide an opportunity for exploring their ecological value in canal estates prior to deployment. We incorporated a multi-species approach by including two bivalve species that often occur in remnant Sydney rock oyster reefs in Moreton Bay; Hairy Mussels (*Trichomya hirsuta*) and Leaf Oysters (*Isognomon ephippium*) (Diggles 2018). Polyculture cages containing these three species were included to test if this led to improved oyster growth and/or survival as well as greater habitat value for fishes and invertebrates. Further, while intertidal Sydney Rock Oyster reefs provide important habitat for a wide range of marine invertebrates (McLeod *et al*. 2019), relatively little is known about the now functionally extinct subtidal reefs (Gillies *et al*. 2018). The oyster gardens are far from a perfect analogue of subtidal reefs, however, these gardens may provide an indication of the value of historic subtidal Sydney Rock Oyster reefs in Australia. Here, we (i) explore the invertebrate and fish communities supported by temporary oyster gardens in a canal estate, and (ii) the effect of single versus multispecies composition of the shellfish community in cages. These data will support managers and practitioners in designing oyster restoration projects, and the management of highly altered coastal ecosystems like canal estates.

Methods

This research was undertaken on land and sea country of the Kabi Kabi people, Traditional Custodians of Yarun, and the authors pay our respects to their Elders past and present.

Figure 1. Oyster garden after seven months of deployment (left panel). The cage in the image contains a mix of Sydney Rock Oysters (*Saccostrea glomerata*), Leaf Oysters (*Isognomon ephippium*) and Hairy Mussels (*Trichomya hirsuta*). Top right: Marble Fortescue (*Centropogon marmoratus*) one of the most common fish found in association with the oyster gardens. Middle right: The Oyster Goby (*Omobranchus anolius*), made up 80% of the total abundance of fish in oyster gardens. Bottom right: Oyster gardens also supported a diverse range of invertebrates, like this sea urchin.

Citizen science

The deployment of oyster gardens on Bribie Island was part of a locally led citizen science initiative with canal estate residents, with the joint aim of informing and educating the local community through active participation while generating a source of live shellfish for future reef restoration research. Participants were enrolled following a local information session and committed to maintain cages monthly and provide annual counts of cage contents.

Thirty households on Bribie Island, Moreton Bay, Australia, were given oyster gardens to deploy off their floating pontoons in October 2016. The cages were made of 1 cm^2 plastic mesh, 60 × 30 × 30 cm in dimension (approximately 54 litres volume, Fig. 1). Each household suspended two cages one metre below the pontoon, one monoculture cage stocked with 200 individuals of juvenile Sydney Rock Oysters (*Saccostrea glomerata*), and a polyculture cage with a mixed community of 100 Sydney rock oysters, 30 Leaf Oysters (*Isognomon ephippium*) and 70 Hairy Mussels (*Trichomya hirsuta*). The Leaf Oysters and Hairy Mussels were collected by hand from the Pacific Harbour canal system on Bribie Island in October 2016. The juvenile Sydney Rock Oysters were collected from a nearby oyster lease run by Sebastiani Oyster Farms (Fisheries Permit 186854). Cages were submerged throughout the experiment, except for during (~monthly) maintenance where pontoon owners cleaned the cages to remove fouling organisms and known oyster predators such as flatworms.

Two levels of experimental controls were added to the experiment: (i) garden controls, and (ii) cage controls. First, we added empty cages to two sites, to test the main effect of bivalve presence. Then, we included a second level of control for the cage itself, by performing a sweep with a net collecting any organisms present in the water around the pontoons equivalent to the area sampled when collecting the cages.

Associated organisms

To investigate the habitat value of submerged gardens, we surveyed their associated organisms approximately seven months after their initial deployment. Oyster garden cages from 10 pontoons were scooped up using a 2 mm mesh net to prevent the loss of any organisms. The cages were then emptied onto a plastic tarpaulin, separating shellfish from other organisms. All invertebrates and fishes present within the cages were then recorded and sorted into broad taxonomic categories (family or order). The 10 surveyed pontoons were selected based on pontoon owner availability on the day of sampling.

Survival and growth of oysters

To obtain an estimate of oyster growth rate and survival during the duration of their deployment, the size and number of dead oyster shells were recorded from a random subsample of 50 Sydney Rock Oysters in both monoculture and polyculture gardens (S1). Further, the dorsoventral measurement (in millimetres) and wet weight (in grams) of a subsample of 80 Leaf Oysters were recorded from multiple polyculture cages suspended from a single pontoon (S2). All measurements were performed over three time points: prior to deployment (November 2016), after six months (April 2017), and after 12 months (November 2017).

Data limitations

Given the nature of citizen science programmes and volunteer-based placement of oyster cages, there are a series of limitations with the data that must be acknowledged. The amount of maintenance and cleaning varied between pontoons, with some cages cleaned weekly, while others may not have been cleaned at all throughout the seven month period, and this was not recorded. Further, bivalves in some cages grew so fast that further growth may have been limited by the size of the cage. These examples were split into two cages and deployed on the same pontoon. To reduce ambiguity and subjectivity in describing the history of each split cage, we simply pooled all cages, categorised by the bivalve species present in each cage.

Analyses

Data on invertebrate communities were analysed using linear mixed-effects models, with garden type as fixed effect (three levels: garden control, polyculture and monoculture. Note that the empty scoop cage control was not included as a treatment as all values were zero. However, this control is included in Figures 2 and 3 (for visual comparison) as is site as a random effect, to account for any differences in maintenance and treatment of cages at the pontoons. Post-hoc tests for pairwise comparisons using least-squares means were performed using the emmeans package (Lenth 2021). Statistical analyses were conducted in R (R Core Team 2022), using the lme4 package (Bates *et al*. 2015). Assumptions were evaluated visually, and data were transformed to meet model assumptions (invertebrate abundance and species richness data required a log+1 transform, while species diversity (Simpsons D) required a cube transform). Test statistics from transformed data are reported on the transformed scale, not the response scale. An initial data inspection revealed a large quantity of isopods that had recruited to a control cage on a single site. To avoid the abundance data to be overwhelmed by this single datapoint, we removed isopods from abundance estimates, but retained them for analyses of species diversity and richness (see S1 for complete dataset). Fish data were highly variable, with extreme outliers, skewed distributions and not suitable for formal statistical analyses. These data are therefore described qualitatively in the text.

Results

Invertebrate assemblages

A total of 56 invertebrate taxa were found in the polyculture cages, 36 in monoculture cages, 17 in the empty cages (garden controls) and none in surrounding water (cage controls). Amphipods, decapod crustaceans and polychaetes made up almost 90% of the overall abundance in each cage (Fig. 2a). The pooled abundance differed among cages (Fig. 2b, linear mixed-effects model: F-value $10.87_{(2,12.4)}$, $P = 0.002$), with polyculture cages having greater abundance than the rest of the treatments (estimated means pairwise comparisons: garden control - polyculture: $t = -2.03$, df = 12.5, $P = 0.004$; monoculture – polyculture: $t = -0.99$, df = 11.4, $P = 0.018$), and with the cages containing only Sydney Rock Oysters and control cages having a similar abundance (garden control – monoculture: $t = -1.04$, df = 12.3, $P = 0.13$).

Oyster garden cages supported a rich assemblage of invertebrates, with polyculture cages supporting a greater species richness and diversity of species (Fig. 2c, d). Species richness differed among treatments (linear mixed-effects model: F-value $7.740_{(2,12.6)}$, $P = 0.006$). Polyculture cages supported a greater richness than monoculture and control cages, with no differences between the latter two types (estimated means pairwise comparisons: monoculture - polyculture: $t = -0.78$, df = 11.4, $P = 0.012$, garden control - polyculture: $t = -0.93$, df = 12.3, $P = 0.049$, garden control - monoculture: $t = -0.19$, df = 12.2, $P = 0.85$). The mean Simpson's diversity index differed between shellfish cages (Fig. 2d, linear mixed-effects model, F-value $15.5_{(2,13.2)}$, $P < 0.001$), with polyculture cages supporting a significantly larger diversity of species than both monocultures and the garden controls (linear mixed-effects model, estimated means pairwise comparisons: monoculture - polyculture: $t = -0.25$, df = 11.7, $P = 0.004$; garden control – polyculture: $t = -0.48$, df = 14.3, $P = 0.001$) while monoculture cages were not significantly different from garden controls (garden control – monoculture: $t = -0.22$, df = 14.1, $P = 0.11$).

Fish assemblages

Twelve fish taxa were identified across all oyster gardens, with 10 species present in polyculture cages, and five species present in monoculture and control cages. No fish were caught on the cage control scoops. Overall, the fish assemblage found in polyculture and monoculture cages was dominated by the Oyster Blenny (*Omobranchus anolius*) which represented up to 80% of all fish identified in the polyculture cages (median per cage, interquartile range, 3, 1–10) and 84% of all fish identified in monoculture cages (5.5, 4–8.5) but was absent from control cages. Other common fishes included the Marbled Fortescue (*Centropogon marmoratus*) and Krefft's Frillgoby (*Bathygobius krefftii*). In contrast, the fish assemblage in control cages was dominated by

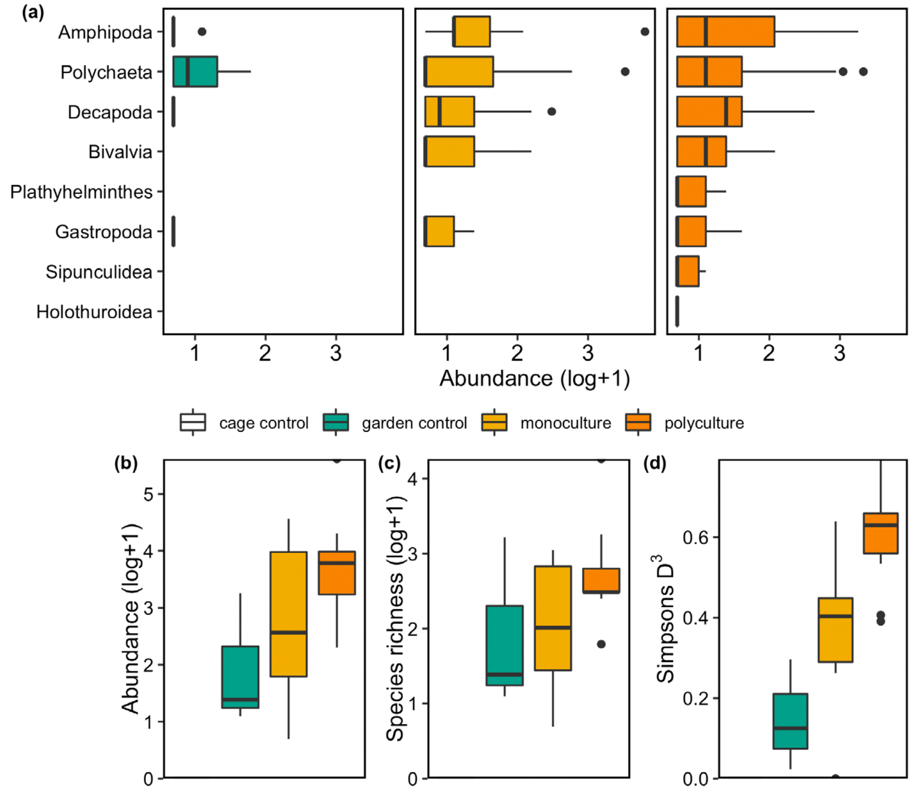

Figure 2. (a) Abundance of invertebrate taxa present in shellfish cages (boxplots: median; lower and upper hinges correspond to the first and third quartiles; whiskers extend from the hinge to the value no further than 1.5 × Inter Quartile Range from the hinge), (b) median pooled invertebrate abundance, (c) species richness and (d) Simpson's diversity index of invertebrate assemblages associated with oyster gardening cages. Monoculture cages contained exclusively Sydney Rock Oysters (*Saccostrea glomerata*), while polyculture cages contained a mix of *S. glomerata*, Leaf Oysters (*Isognomon ephippium*) and Hairy Mussels (*Trichomya hirsuta*). Controls were either empty cages (garden controls), or scoops of the surrounding water (cage controls). Note that all cage control values were zero.

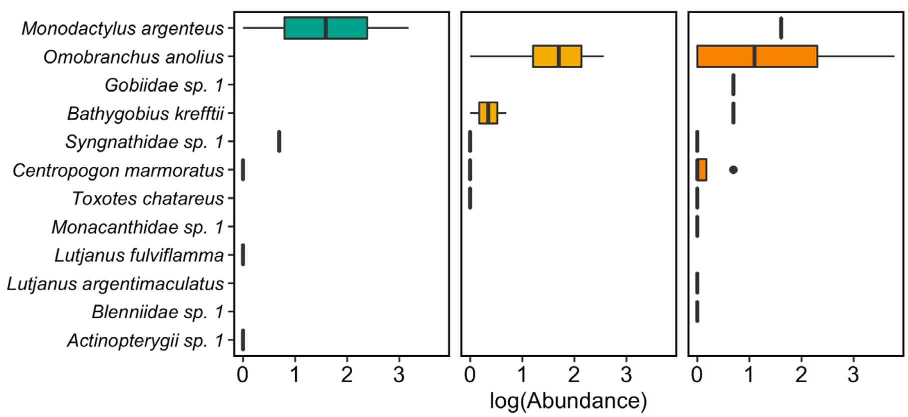

Figure 3. Abundance (boxplots: median; lower and upper hinges correspond to the first and third quartiles; whiskers extend from the hinge to the value no further than 1.5 × Inter Quartile Range from the hinge) of the fish assemblage associated with oyster gardening cages. Monoculture cages contained exclusively Sydney Rock Oysters (*Saccostrea glomerata*) while polyculture cages contained a mix of *S. glomerata*, Leaf Oysters (*Isognomon ephippium*) and Hairy Mussels (*Trichomya hirsuta*). Controls were either empty cages (garden controls), or scoops of the surrounding water (cage controls). Note that all cage control values were zero.

juvenile Butter Bream (*Monodactylus argenteus*) representing up to 83% of taxa in control cages (median 12.5, 6.75 - 18.25), while the species was present in a single polyculture cage (n = 5) and was absent in the monoculture cages. This result was driven by a very high number of this species associated with one control cage.

Discussion

Australia's first oyster gardening project provides insight into the value of small-scale restoration initiatives in highly urbanised environments. First, we found that submerged oyster cages in these environments supported a diverse assemblage of invertebrates and fishes. The invertebrate communities found living in association with the oysters were primarily composed of amphipods, polychaetes and crustaceans, common components of the natural assemblages of invertebrates found in remnant intertidal Sydney Rock Oyster reefs (McLeod *et al*. 2019). Similarly, the fish community associated with the oyster cages were composed of small cryptic species such as blennies, gobies and benthic-associated species such as the Marbled Fortescue (*Centropogon marmoratus*), all common residents of experimental subtidal reefs in Pumicestone Passage (Diggles 2018) and remnant intertidal Sydney Rock Oyster reefs (Cole *et al*. 2022). It is likely that the oyster gardens provided shelter from predation (from the live bivalves, disarticulated shells, and the mesh of the cages) and a food source supplied by associated invertebrate and fouling communities. Bivalve shells also provide nesting habitat for species such as *O. anolius* (Thomson & Bennett 1953). These results offer a glimpse into the potential biodiversity value of subtidal Sydney Rock Oyster Reefs, once common ecosystems that are now functionally extinct in most parts of Australia.

The invertebrate community composition found in oyster gardens was similar to that found in Queensland canal estates, with amphipods and copepods being the most abundant prey item in the stomach of fishes commonly associated with jetties and other artificial structures (Moreau *et al*. 2008). Oyster gardens in canal estates provide islands of structural complexity in a sea of soft sediment, and may provide critical habitat for invertebrate assemblages, similar to historically abundant subtidal oyster reefs (McLeod *et al*. 2019).

We did not detect any significant differences in the fish community between control cages and the two shellfish treatments, however we speculate that this may be due to the control cages, rather than an indication of the habitat provisioning of the oysters themselves. First, empty cages allowed the proliferation of macroalgae, which provide a structural habitat for many cryptic fish species in artificial environments (Feary *et al*. 2011; Dafforn *et al*. 2015). Second, the presence of a cage in our oyster gardens preclude the settlement of larger mobile fish species, disrupting the natural trophic linkage. While natural subtidal oyster reefs were likely to support higher abundances of fish than surrounding sediment areas, by hosting an abundance of invertebrate prey sources (McLeod *et al*. 2019), this service may not be detectable in our study due to the presence of cages. Indeed, the more accurate comparison for understanding the habitat provisioning of oyster gardens is the empty scoop cage controls in which no fish were detected. Indeed, oyster aquaculture infrastructure in Sydney Harbour, Australia, has been demonstrated to support a large abundance of fishes, with a similar community composition to natural biogenic habitats nearby (Martínez-Baena *et al*. 2022).

These findings highlight the importance of oysters and other bivalve species in providing habitat for invertebrates and fishes. Indeed, studies of the fish assemblages associated with the shellfish reef restoration trial in Pumicestone Passage found that restoration can significantly enhance both the diversity, abundance and density of harvestable fish (Gilby *et al*. 2021). Further, our findings indicate that oyster gardens with mixed bivalve species provide habitat to a more abundant and diverse invertebrate and fish community compared to gardens with only Sydney Rock Oysters. Given our study design, we are unable to tease apart whether this is due to the mixed species combination, or any of the individual species themselves. However, while the oyster gardens add habitat to the water column, the inclusion of several shellfish species likely provides further habitat complexity due to the combination of the different shapes and forms of the species. Habitat complexity and heterogeneity provided by aggregations of different foundational species is positively correlated with invertebrate composition and abundances, with greater species richness and abundances in more structurally complex habitats (Sueiro *et al*. 2011; Hughes *et al*. 2014). The structural heterogeneity provided by multiple shellfish species likely enhances the number of microhabitats and interstitial spaces through increased space-size heterogeneity, thereby supporting a greater invertebrate and fish species richness (St. Pierre & Kovalenko 2014). While our methods captured the cryptic invertebrates and fish communities living within the oyster garden structure it is likely that the mesh size of the cages, and our collection method, selected against mobile and larger species.

The three-dimensional structure of oyster aquaculture infrastructure is providing comparable foraging, resting and nursery grounds to many natural habitats like remnant oyster reefs, rocky reefs, seagrasses and mangroves (Mercaldo-Allen *et al*. 2019; Martínez-Baena *et al*. 2022; Theuerkauf *et al*. 2022). Further studies on oyster gardens initiatives are needed to determine the habitat provided to the larger and more mobile fish species that inhabit these canal estates.

Working with citizen scientists in this project provided multiple benefits. First, the project has engaged with 30 households within the canal estate, generating interest and engagement for a local conservation issue. Further, the oyster gardeners provided critical maintenance and infrastructure for the developing oyster gardens, which would have come at a substantial cost in a commercial setting. Further research into the human dimension of citizen science engagement is needed to understand the motivations and benefits experienced by the participants.

Conclusion

Australia has the greatest expanse of residential canal estates in the world, with approximately 440 km of linear urban waterways (Waltham & Connolly 2007). Given the lower habitat value provided for estuarine species compared to natural undisturbed waterways (Brook *et al*. 2018; Waltham *et al*. 2020), oyster gardening initiatives could be a great opportunity to increase the habitat value of these artificial ecosystems, while contributing similar provisioning and regulating services that mariculture provides (van der Schatte Olivier *et al*. 2020; Theuerkauf *et al*. 2022). While oyster gardens are limited structural analogues for natural oyster reefs, they provide habitat to a subset of species resident on natural reefs, and may provide clues to the habitat value of extinct subtidal oyster reefs in these modified environments. Upscaling oyster gardening initiatives in areas where large-scale oyster reef restoration programmes are planned could be an effective way to grow and store oyster broodstock, while boosting local fish and invertebrate biodiversity and providing opportunities for community involvement and citizen science.

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Supporting Information

Additional supporting information can be found in the following online files.

Figure S1. The mean abundance of invertebrate taxa, with isopods included. The isopod data are larger driven by a single control cage at a single site receiving orders of magnitude higher numbers of isopods. Figure 1a in the main manuscript depicts the same data with isopods removed.

Figure S2. The mean shell length (a), and estimated mortality of Sydney Rock Oysters (b), recorded in the first 12 months of oyster gardening cage deployment. Data were calculated from a random sample of 50 shells in each cage.

Figure S3. Mean shell height (dorsoventral measurement in mm, orange) and wet mean shell weight (in grams, green) of Leaf Oysters (*Isognomon ephippium*, n = 80) randomly sampled from polyculture oyster gardens from one pontoon after zero, six and 12 months deployment.