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**Swimming nets have positive effects on populations of the endangered White's seahorse
*Hippocampus whitei***

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Abstract

1. Habitat degradation and destruction arising from rapidly increasing urbanization represents one of the most significant threats to biodiversity. Human populations are continuing to increase around coastal regions, and as marine habitats are displaced by artificial structures it is important to understand how marine species may be impacted by these changes in habitat availability. The endangered seahorse *Hippocampus whitei* has been observed inhabiting protective swimming nets in Sydney Harbour, Sydney, Australia, even in the presence of natural habitats.
2. This study tested whether the presence of a swimming net results in increased seahorse numbers at sites around Sydney Harbour, or whether seahorses are attracted away from natural habitats. Density surveys and mark-recapture population estimates were done at sites with pre-existing swimming nets and compared to control sites where only natural habitat was present. A manipulative experiment was conducted in which panels of swimming net material were installed at two sites in Sydney Harbour, with comparisons to control sites over a period of 14 months (April 2018 to June 2019) to test whether the installation of swimming nets would affect seahorses on surrounding natural habitat or increase site abundance.

3. The pre-existing and installed swimming nets were found to support greater densities of *H. whitei* as well as some increases in site-scale abundance, with no effects on seahorse density on natural habitats. It is likely that increased seahorse production is occurring on the nets, with no evidence that seahorses are being attracted away from natural habitat, however, effects may vary across survey occasions and sites. Furthermore, swimming nets may serve as a useful replacement habitat in locations where natural habitat has become sparse or absent.

Keywords

Coastal; Seagrass Meadow; Endangered species; Habitat management; Urban Development; Fish

Introduction

Rapid coastal urbanization and anthropogenic disturbance has resulted in significant declines in the availability of natural habitats, which in combination with climate change, has been associated with concurrent declines in abundance for many marine taxa (Airoldi & Beck, 2007). Anthropogenic activity around coastlines has resulted in an overwhelming abundance of artificial structures in a phenomenon referred to as “ocean sprawl” (Bishop et al., 2017; Firth et al., 2016). The ecological impact of artificial structures as well as their suitability as a replacement for natural habitat is a topic of debate and may vary depending on the affected taxa, as well as characteristics of the structure itself (Dafforn et al., 2015; Strain et al., 2018). Replacement of natural habitat with artificial structures is often associated with declines in

habitat heterogeneity and complexity resulting in decreased biodiversity (Bulleri & Chapman, 2010; Gittman, Scyphers, Smith, Neylan, & Grabowski, 2016). Conversely, there is some evidence that structures such as artificial reefs provide habitat for taxa in the absence of natural habitat, with positive effects on the diversity and abundance of marine fish (Burt, Feary, Cavalcante, Bauman, & Usseglio, 2013; Wen, Pratchett, Shao, Kan, & Chan, 2010). When planned and created with an ecological function in mind, artificial structures may be able to act as a powerful conservation tool in a heavily urbanized coastal environment (Dafforn et al., 2015). For example, artificial structures such as seawalls can be modified to increase complexity and emulate natural habitat, with inhabiting communities resembling those found on natural habitats to some degree (Chapman & Blockley, 2009; Morris et al., 2018; Strain et al., 2018).

Due to the varied ecological effects that artificial structures can have on marine taxa, careful consideration must be given before promoting their use as a potential conservation tool.

While artificial structures in marine ecosystems such as artificial reefs have been associated with rapid colonization and relatively high abundance of some (often economically important) fish taxa (Bohnsack, Johnson, & Ambrose, 1991; Reubens et al., 2013), it is important to establish whether such structures are resulting in production (i.e. increased fish abundance) or simply aggregating fish. This attraction-production dilemma has been well discussed in the artificial reef literature, with the capability of production varying from reef to reef depending largely on design and complexity (Baine, 2001; Pickering & Whitmarsh, 1997). Well designed, complex artificial reefs have been shown to be highly productive fish

habitats (Becker, Taylor, Lowry, & Grabowski, 2016; Smith, Lowry, Champion, & Suthers, 2016).

There are a number of marine taxa that inhabit shallow, heavily urbanized coastal waters and are consequentially at high risk of disturbance (Johnston et al., 2015; Mayer-Pinto et al., 2015). Such taxa are frequently affected by replacement of natural habitats with artificial structures; therefore, novel approaches to conservation are necessary in many cases.

Seahorses (*Hippocampus* spp.) are one such taxa, with 12 species currently listed as Vulnerable and two as Endangered on the IUCN Red List (2019). Seahorses can be especially susceptible to significant declines in abundance following anthropogenic habitat disturbance due to their small home ranges, small brood size (relative to other teleost species) and poor swimming ability (Foster & Vincent, 2004; Vincent, Foster, & Koldewey, 2011).

Seahorses' natural habitat largely consists of erect biogenic structures such as seagrass, algae, soft coral and sponge, though many species have been found to inhabit artificial structures in absence of these habitats. Seahorses require holdfasts onto which they may attach with their prehensile tail while feeding and resting (Foster & Vincent, 2004). In the Knysna estuary, South Africa, the Knysna seahorse (*Hippocampus capensis*) has been found to inhabit rock filled Reno mattresses in higher densities than natural habitat (Claassens, 2016), and was found to selectively inhabit these structures even in the presence of natural habitat (Claassens, Booth, & Hodgson, 2018). Artificial habitats have been considered a potential conservation tool for seahorses in the face of dwindling natural habitat, with some increases in the density of the seahorses *Hippocampus guttulatus* and *Hippocampus hippocampus* found after trialling

artificial holdfast units in the Ria Formosa Lagoon (Correia, Koldewey, Andrade, & Palma, 2015; Correia, Palma, Koldewey, & Andrade, 2013).

Hippocampus whitei is a species of seahorse endemic to the eastern coast of Australia, and their preferred natural habitats in Port Stephens (soft coral *D. australis* and sponges) (Harasti, Martin-Smith, & Gladstone, 2014) have been in significant decline from 2009 to 2015 (Harasti, 2016). As these habitats are largely absent in Sydney Harbour, *H. whitei* is instead found inhabiting the seagrass *Posidonia australis*, however, this seagrass has been in decline by greater than 10% each year from 2009 to 2014 (Evans, Griffin, Blick, Poore, & Verges, 2018). These declines in natural habitat availability have contributed to significant declines in *H. whitei* abundance (Harasti, 2016), and the species was updated to Endangered on the IUCN Redlist (Harasti & Pollom, 2017) and listed as Endangered under local legislation in November 2019. Informed conservation strategies will be necessary to prevent this species from further population declines, such as the consideration of artificial habitats in areas where natural habitats are in decline. For example, purpose-built artificial habitats referred to as “Seahorse Hotels” have been shown to successfully support populations of *H. whitei* (Simpson, Coleman, Morris, & Harasti, 2020).

Hippocampus whitei has been observed inhabiting protective swimming nets around Sydney Harbour, likely as a result of declines in natural habitat availability. These structures encircle swimming enclosures with the purpose of preventing shark interactions and can be found in several locations on both the north and south sides of the harbour. *Hippocampus whitei* has been found to inhabit swimming nets in greater densities and abundances than on natural habitat, even when both are in close proximity (Clynick, 2008; Harasti, Glasby, & Martin-

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Smith, 2010), and experiments in aquaria have shown that *H. whitei* display a significant choice for swimming net material over natural habitats (Simpson, Morris, Harasti, & Coleman, 2019). However, no previous studies have made comparisons of seahorse density and population abundance between natural and artificial habitats across multiple sites, nor have they tested whether the deployment of swimming nets has a result on site-scale seahorse density and population abundance. Swimming nets therefore have the potential to be employed as a conservation tool for seahorses when natural habitats decline; however, there are a number of important factors to be considered. Local management agencies often clean swimming nets of fouling epibiota or remove nets during winter months which can have significant impacts for seahorses that inhabit the net. It is also unknown whether swimming nets are resulting in localized seahorse population increases, and no research has explored whether the presence of a swimming net is having a net positive effect or simply attracting seahorses away from natural habitats. Swimming nets in Sydney Harbour are regularly braided with copper to inhibit fouling and strengthen the material, which may reduce the habitat value of the nets and result in leaching of copper into the marine environment (Abou El-Naga, El-Moselhy, & Hamed, 2005; Moraitou-Apostolopoulou & Verriopoulos, 1979). The use of copper free, “seahorse friendly” swimming net material is currently being trialled at Parsley Bay, NSW.

This study tested whether swimming net presence is having an effect on populations of *H. whitei* at sites where natural habitat for seahorses (seagrass *P. australis*) is present. To explore this, experiments were conducted to test the hypotheses that: A) there would be a difference in seahorse density on *P. australis* between sites where there was a pre-existing

swimming net in proximity to seagrass and control sites where no net was present, B) seahorse density would be greater on swimming nets than on *P. australis*, and C) site scale seahorse population abundance would be greater at sites where a swimming net is present. As mensurative experiments are usually confounded in space and time (Underwood, 1997), a manipulative experiment was also conducted that tested the hypotheses that: A) there would be a difference in seahorse density on *P. australis* seagrass between sites at which panels of swimming net were installed and control sites, B) that there would be greater seahorse density on swimming nets compared to in *P. australis*, and C) that there would be a difference in site-scale seahorse abundance between net installation sites and control sites (or that swimming net presence would result in increased site-scale seahorse abundance at installation sites).

Methods

Study sites

This study was done at six sites in Sydney Harbour, NSW, Australia. Four of these sites (Delwood Beach, Fairlight, Little Manly Cove and Quarantine Station; Figure 1) had *H. whitei* populations inhabiting patches of *P. australis* seagrass, and two sites (Clifton Gardens and Manly Cove) had *H. whitei* populations in *P. australis* patches as well as found on pre-existing swimming nets near the seagrass (within approximately 10 m).

The *P. australis* meadows at each site were approximately 500 m², and often comprised a mosaic of smaller patches of varying blade density, and located at a depth of between 2 and 6

m. The meadows of *P. australis* were surrounded or interspersed with bare sand, patchy *Sargassum* sp. or other seagrass species such as *Zostera muelleri* or *Halophila ovalis*.

Delwood Beach and Fairlight were used for net installation in the manipulative experiment (detailed below). The swimming nets located at Manly Cove and Clifton Gardens had a perimeter of approximately 150-200 m with an average depth of 3 m. Both nets were polypropylene with a square mesh of 10 cm, with a line thickness of 3 mm.

Assessing seahorse populations at sites with pre-existing swimming nets

Seahorse density was quantified using ten replicate 5 min timed diver surveys conducted at four sites in April 2018. Of these, two sites had a swimming net and the natural seahorse habitat *P. australis* in proximity to one another, and two contained *P. australis* alone. At sites where only *P. australis* was present, 5 min surveys (n = 10) were done on the seagrass, and at sites where both habitats were present, 5 min timed surveys (n = 10) were done on each habitat. On each survey, divers would continuously search through *P. australis* seagrass that had not already been searched on that dive, recording the number of seahorses observed. The total length (measured from the top of the seahorse coronet to the end of the tail) and sex (determined through presence of a brood pouch) of each seahorse was also recorded on each survey. Seahorses smaller than 70 mm were classified as juveniles, as the absence of a developed brood pouch at this size makes sexing these animals difficult (Harasti, Martin-Smith, & Gladstone, 2012).

The null hypothesis that swimming net presence has no effect on seahorse density within *P. australis* was tested with a two-way ANOVA, with net presence as a fixed factor (two levels: net present, net absent), and site as random factor (Four levels: Delwood Beach, Fairlight, Manly Cove and Clifton Gardens). The null hypothesis that there is no difference in seahorse density on natural and artificial habitats was tested with a two-way ANOVA, with habitat as a fixed factor (two levels; swimming net, seagrass) and site as a random factor (two levels: Manly Cove, Clifton Gardens). The assumption of homoscedasticity was assessed with a Levene's test. In the case of homoscedasticity violations, data were transformed ($\ln+1$). If homoscedasticity was still violated, tests were still run and interpreted with caution as ANOVA's are robust to heterogeneity of variance in cases where the denominator is greater than ten (Sokal & Rohlf, 1995; Zar, 1996). ANOVA is robust against non-normality regardless of transformation so this assumption was treated as inconsequential (Sokal & Rohlf, 1995; Underwood, 1997). These analyses were run in Primer with PERMANOVA add-on, on a Euclidean resemblance matrix. Non-significant interaction terms were eliminated from the model (Underwood 1997).

To assess any effects that swimming net presence had on site-scale seahorse abundance, population estimates were derived from mark-recapture. Every new seahorse encountered was given a unique tag for identification purposes using visual implant fluorescent elastomer (VIFE, Northwest Marine Technologies, USA; www.nmt.us), with each seahorse given three 1-2-mm coloured tags with a combination of colours *in situ*. This tagging method was chosen because of the long-lasting nature of the tags (up to 5 years) and the capacity to create many unique identifying codes whilst minimizing disturbance and harm to the animals (Harasti et

al., 2012). Individuals were placed back onto the habitats on which they were found after tagging. Five separate tagging occasions occurred within seven days at each site. Seahorse populations were not expected to experience any immigration, emigration, births or mortalities over the seven day period the population was considered “closed”. Population estimates were derived using program MARK based on a closed captures (full likelihood p and c) model.

Assessing the effects of installed experimental swimming net panels on seahorse populations

Panels of swimming net were installed adjacent to meadows of *P. australis* at two sites (Delwood Beach and Fairlight) in September/October 2018. To estimate effects of net presence on seahorse density in their natural habitat, six surveys consisting of ten replicate 5 min timed surveys for seahorses were conducted on *P. australis* at these two sites prior to net installation at monthly/bimonthly intervals. These surveys were continued at monthly intervals until the net panels were removed in June 2019, resulting in 18 surveys per site. To establish whether effects on seahorse populations were due to net presence, two control sites were used where populations of *H. whitei* seahorses existed in *P. australis* seagrass. Eighteen surveys were also conducted at these sites. The number of seahorses, length and sex was recorded. Once artificial nets were installed, a five-minute survey was also conducted on each panel of swimming net on the same day that a survey was conducted on the seagrass at that site.

Twelve replicate panels of swimming net were installed at each site, consisting of six replicates of each of the two different types of swimming net that can be found at swimming

enclosures around Sydney Harbour: (1) regular copper braided netting (mesh size of 10 cm, thickness of 3 mm, white colour); and (2) seahorse friendly copper free netting (mesh size of 10 cm, thickness of 4 mm, blue colour). Net material was sourced from Haverford PTY LTD, Sydney. Each replicate panel of netting was 1 m in height and 5 m in length, deployed in a triangle formation and pinned into the substratum with steel star pickets (Figure 2). A triangle formation was used here to increase structural integrity of the nets once they were deployed, as well as reduce the chances of destruction as a result of boat anchoring. This formation also allowed for replicate nets to be placed randomly throughout the site with adequate distance between replicates (1.5 times replicate diameter), while also allowing each replicate to be placed at roughly equal distance from natural habitats and removing any directional bias that may occur if non-triangle net panels were to be deployed at different angles to natural habitats. Net panels were placed at approximately 6 m depth, and approximately 10 m from natural *P. australis* habitats.

The null hypothesis that there would be no difference in seahorse density on *P. australis* in the first and final six surveys between “net installation” sites and “seagrass only” sites over time was tested using a nested repeated measures analysis of variance (ANOVA) with net presence as a fixed factor (two levels: net present, net absent), before/after as a fixed within-subjects factor (two levels: before, after), site as a random factor (four levels, nested within net presence), and time as the repeated measure. The assumption of sphericity was tested using Mauchly’s test. If this assumption was violated, the Greenhouse-Geisser correction was interpreted. The analysis was performed in SPSS.

The null hypotheses that there would be no difference in seahorse density between habitat types at sites where swimming nets were installed only was tested with a repeated measures analysis of variance (ANOVA). This analysis was conducted only on the final six surveys of data collection after seahorse numbers had established on the installed nets, with habitat as a fixed factor (two levels: net and seagrass), site as a random factor (two levels) and time as the repeated measure. To assess overall long-term trends at each site, regressions were used to test the hypothesis that there was a relationship between time and seahorse density.

All new seahorses found on each survey were given a unique identification VIFE tag as per above. To assess the population abundance of *H. whitei* at each site, an open population Jolly-Seber model was applied using the POPAN model in program MARK (White & Burnham, 1999). As surveys were conducted over a 14 month period, the population of *H. whitei* at each site was considered “open” as it was likely that births, mortalities, immigration and emigration of seahorses would have occurred. This model provides an estimate of the population size (N) at each tagging occasion as well as estimates of the total number of seahorses to have inhabited the site over the course of the experiment. Estimates for survivability (ϕ) and the probability of resighting individuals (p) are also given.

The null hypotheses that a) there would be no significant difference in site persistence of *H. whitei* on *P. australis* meadows at sites that have presence or absence of a swimming net; and b) that there would be no difference in site persistence of seahorses on *P. australis* and swimming net were tested. To do this, a Kaplan-Meier log rank test was used in SPSS (Kaplan & Meier, 1958; Pollock, Winterstein, Bunck, & Curtis, 1989). This analysis observed the presence or absence of the first 10 seahorses to inhabit *P. australis* at all sites as

per previous studies that explores *H. whitei* site fidelity (Harasti & Gladstone, 2013), and swimming nets at Delwood Beach and Fairlight, from first appearance until the end of the experiment.

Results

Assessing seahorse populations at sites with pre-existing swimming nets

No significant difference in seahorse density in *P. australis* was found between sites where a swimming net was present or absent (Table S1; Figure 3). At sites where both a swimming net and *P. australis* was present, seahorse density was significantly greater on the swimming nets than the seagrass (Table S2), with the mean seahorse density on swimming nets (1.65 ± 0.27 seahorse per timed survey) 3.66 times the mean density in seagrass (0.45 ± 0.17 seahorses per timed survey).

The populations of *H. whitei* at both sites where swimming nets were present (Clifton Gardens and Manly Cove) were estimated from program MARK to be greater than those at control sites (Table 1; Figure 4). The estimated population of *H. whitei* at Clifton gardens was 561% and 672% greater than the estimated populations at control sites (Fairlight and Delwood respectively) with the confidence intervals greater than that of the controls, while the estimated population at Manly Cove was 85% and 116% greater with confidence intervals within that of the controls (Figure 4).

At Clifton Gardens, 47 out of the total 101 tagged seahorses were resighted on another occasion after the initial tagging, and the greatest number of resightings for a seahorse was 5.

At Manly Bay, 16 out of the total 28 tagged seahorses were resighted after tagging, and the greatest number of resightings for a seahorse was 4. At Fairlight, 13 out of the total 18 tagged seahorses were resighted after tagging, and the greatest number of resightings for a seahorse was 3. At Delwood Beach, 8 out of the total 16 tagged seahorses were resighted after tagging, and the greatest number of resightings for a seahorse was 3.

Assessing the effects of installed experimental swimming net panels on seahorse populations

For the seahorse density within seagrass, a significant time*before/after*net presence interaction was found indicating that there were differences in seahorse density between the sites where nets were installed and the control sites at various survey times both before and after net installation (Figure 5; Table S3). However, regression analyses showed that there were no significant long-term positive or negative trends in seahorse density in *P. australis* over time at any site, as time did not significantly predict variation in seahorse density on this habitat at either of the net presence sites (Fairlight: $R^2 = 0.002$, $F_{(1,17)} = 0.040$, $P = 0.844$; Delwood Beach: $R^2 = 0.036$, $F_{(1,17)} = 0.602$, $P = 0.449$) or the control sites (Quarantine station: $R^2 = 0.021$, $F_{(1,17)} = 0.351$, $P = 0.562$; Little Manly $R^2 = 0.030$, $F_{(1,17)} = 0.494$, $P = 0.492$; Figure 7).

For the sites where experimental nets were installed, a significant site*habitat interaction was found (Table S4), showing that the difference in seahorse density across habitats varied across the two sites. At Delwood Beach, seahorse density tended to be greater on the nets than the seagrass, however, the opposite was true at Fairlight, where seahorse density was lower on the net than the seagrass (Figure 6). Regression analysis shows that time

significantly predicted seahorse density on the net panels at both Fairlight ($R^2 = 0.532$, $F_{(1,12)} = 12.493$, $P = 0.005$) and Delwood Beach ($R^2 = 0.802$, $F_{(1,12)} = 44.519$, $P = <0.001$; Figure 7), indicating that populations of seahorses on these structures were increasing over time.

Seahorse population analysis

The number of seahorses at Delwood Beach was estimated to have peaked in October/November which is the start of the breeding season for *H. whitei*, followed by a decline through the autumn months and another small rise in the winter (Figure 8). This pattern was also observed at both Quarantine and Little Manly, despite smaller (on average) populations. The population at Fairlight was consistent with an estimated 25 seahorses, with a peak in June 2018 and minor temporal fluctuations throughout the 12 surveys. The only site to display a net positive trend over the course of the experiment was Delwood Beach.

A greater population of *H. whitei* was estimated to have inhabited the two sites where swimming net panels were installed over the course of the experiment, with few overlapping confidence intervals (Table 2). At the site-scale, average survivability was approximately the same at all sites except for Quarantine where it was substantially less (Table 2). Survivability of seahorses found on the installed nets was generally slightly lower than those found in *P. australis*. Additionally, the average resight probability of seahorses was generally greater on installed panels of swimming net than on *P. australis*.

The longest length of time that a seahorse was observed to be inhabiting *P. australis* was 14 months, with two seahorses (found at Delwood Beach and Little Manly, respectively) tagged in April 2018 and later resighted in June 2019. Kaplan-Meier comparisons showed that there

was no significant difference in the site-persistence of *H. whitei* on seagrass across three sites, Delwood Beach, Fairlight and Little Manly ($\chi^2 = 1.492$, d.f. = 2, $p = 0.474$). Quarantine was excluded from this analysis due to insufficient seahorse numbers. Kaplan-Meier comparisons showed that there was no significant difference in the habitat-persistence of *H. whitei* across habitat types at Delwood Beach ($\chi^2 = 1.601$, d.f. = 1, $p = 0.206$). Fairlight was excluded from this analysis due to insufficient seahorse numbers inhabiting the panels of swimming net.

Four seahorses were observed to have moved between habitats at Delwood Beach, with two seahorses having moved from the nets onto seagrass, one seahorse moving from the seagrass onto the nets, and one seahorse moving from the seagrass onto the nets, then later back onto the seagrass. All other seahorses observed inhabiting the nets were new to the site, or not previously observed in the seagrass. No seahorses were observed to have moved between habitats at Fairlight, with seahorses observed inhabiting the nets being new to the site or not observed previously on natural habitat.

Discussion

This study provided strong evidence that swimming nets result in an overall increase in seahorse populations in Sydney Harbour. Both pre-existing and installed swimming nets were found to support greater densities of the threatened *H. whitei* seahorse, while there was little difference in the seahorse density on natural habitats with and without net presence. As coastal urbanization continues to expand rapidly, resulting in the replacement of natural habitats with artificial structures (Bishop et al., 2017; Firth et al., 2016), it is important to assess not only how to conserve natural habitats but also how these structures may act as a

potential habitat for threatened marine species. This is essential when the presence of artificial structures is unavoidable, such as cases where artificial structures are pre-existing as they fulfil a required purpose (e.g. swimming nets).

When dealing with rare and threatened marine species, issues of high variability and small sample sizes may arise, which may complicate data analysis and interpretation, especially those including the tagging and tracking of marine animals (Sequeira et al., 2019). Our results show that *H. whitei* numbers varied significantly across survey occasions as well as between sites, having the potential to obscure trends in the data caused by important factors such as net presence. Variability in seahorse abundance between sites also made comparisons to control sites difficult, as abundance varied strongly between control sites as well as fluctuating between survey occasions. Large fluctuations in population abundance has been reported for seahorse species (Correia, Caldwell, Koldewey, Andrade, & Palma, 2015), and such variability underlines the importance of long-term population monitoring and multiple control sites in experiments dealing with rare and threatened species. In some cases, seahorse numbers were too small to perform accurate population analyses in program MARK resulting in greater confidence intervals and overestimates at some sites (Quarantine).

The use of swimming nets by *H. whitei*

Artificial structures and habitats have often been associated with increased abundance of some fish species (Charbonnel, Serre, Ruitton, Harmelin, & Jensen, 2002; Zalmon, Novelli, Gomes, & Faria, 2002), and increased densities of the seahorses *H. guttulatus* and *H. capensis* have been observed on artificial habitats in the Ria Formosa lagoon in Portugal

(Correia, Koldewey, et al., 2015) and in the Knysna estuary, South Africa (Claassens, 2016), respectively. In Sydney Harbour, greater densities of *H. whitei* have been reported on swimming nets than on surrounding natural habitats (Harasti et al., 2010). Our results corroborate these findings, as a greater density of *H. whitei* was found on pre-existing swimming nets than in natural habitats during the experiment. Similar results were found in our manipulative experiment, with a greater density of *H. whitei* on the installed nets than on *P. australis* at Delwood Beach.

Hippocampus whitei density was lower on the installed nets than on *P. australis* at Fairlight, though this is likely due to natural variation in seahorse recruitment to artificial habitats between sites. The time taken for seahorses to inhabit newly installed artificial structures has been found to vary temporally and geographically and can lead to variations in population density, with some studies showing that *H. whitei* can take up to four months to inhabit newly installed swimming nets (Clynick, 2008), and may take up to 15 months for a population to recover from disturbances such as removal or cleaning (Harasti et al., 2010). Seahorse recruitment onto artificial structures has also been found to vary significantly from site to site due to a number of factors such as the quality and complexity of natural habitats present (Correia, Koldewey, et al., 2015). Our results indicate that this study would have benefitted from deploying nets for longer, as the regression analyses showed that seahorse populations on the installed nets were increasing over the course of the experiment, and seahorse density on the installed nets at Fairlight likely would have become greater than on *P. australis* if the nets were deployed long enough for seahorse populations to fully establish. Unfortunately, the net panels had to be removed due to time constraints as well as permit conditions.

Epibiotic growth on artificial structures helps provide a source of mobile epifaunal prey for seahorses (Hellyer, Harasti, & Poore, 2011). A lack of epibiotic growth may explain the relatively large amount of time seahorses take to recruit onto newly installed nets (Clynick, 2008; Harasti et al., 2010), and the lack of established growth on the installed nets may explain the slow recruitment of seahorses onto the nets at Fairlight. The abundance of *H. whitei* on purpose built “Seahorse Hotels” increased over time after deployment, likely linked to increasing growth of epibiota on the habitats (Simpson et al., 2020).

There are a number of reasons that seahorses may inhabit artificial habitats in greater densities than on natural habitats such as *P. australis*, as was found here. Artificial habitats have often been found to be attractive to seahorse species, with a number of seahorse species choosing to inhabit artificial habitats over natural habitats (Claassens et al., 2018; Simpson et al., 2019). Additionally, seahorses may have increased reproductive success on the nets than on patchy natural habitat as they are more easily able to locate one another and breed. This is supported by the greater average resight probability of seahorses on swimming nets than on natural habitats as found here. Further, swimming nets may provide a diverse range of complex habitats due to the epibiotic growth that may accumulate on the nets.

The potential of swimming nets to increase *H. whitei* populations

Many artificial structures (e.g. seawalls) have been associated with biodiversity loss and decreases in species abundance, likely as a result of decreased habitat complexity (Chapman, 2003). Conversely, some artificial structures and reefs have been associated with

observations of increased abundance for important marine species (Charbonnel et al., 2002; Zalmon et al., 2002). Purpose-built Seahorse Hotels were found to support a relatively large population of *H. whitei* in an area where natural habitat had declined (Simpson et al., 2020), and the presence of artificial structures has been potentially associated with increases in site-scale seahorse density in other regions (Claassens, 2016; Correia, Caldwell, et al., 2015).

Previous studies have observed that swimming nets are inhabited by relatively large populations of *H. whitei* and may be useful as replacement habitat in areas where natural habitats are sparse (Clynick, 2008; Harasti et al., 2010). While swimming nets may provide a structure for seahorses to inhabit with increased density, there is no substantial evidence to suggest that similar increases in density are observed in surrounding natural habitat.

However, the mark-recapture population estimates presented here do provide some evidence that the presence of swimming nets may result in increases in site-scale seahorse abundance, likely through the addition of new habitat and increased holding capacity at sites. Increases in seahorse numbers have been associated with increased holdfast availability at sites where natural habitat is fragmented (Correia, Caldwell, et al., 2015; Correia, Koldewey, et al., 2015).

It is important, however, to understand whether artificial structures are resulting in increased species production at a site, or simply attracting fish away from natural habitats (Grossman, Jones, & Seaman Jr, 1997; Pickering & Whitmarsh, 1997). Where only attraction occurs at artificial reefs, this can increase the vulnerability of fish to human harvesting, as individuals are aggregated in an area making them easier to catch (Cabral, Alino, & Lim, 2014; Sadovy & Domeier, 2005). For seahorses in Sydney Harbour, an attraction to nets could increase

their vulnerability, as nets are maintained through removal and/or cleaning (Harasti et al., 2010). There remains a possibility of increased vulnerability to predation if natural predators of *H. whitei* were aggregated around swimming nets, though this has not been assessed in literature and future studies could do so.

There was no difference in seahorse density on *P. australis* seagrass between sites where a pre-existing swimming net was present and control sites, which is evidence that seahorses are not being attracted away from natural habitats onto established swimming nets. While comparing seahorse density on *P. australis* between sites where experimental panels of swimming net were installed and control sites, a significant interaction was found between survey times, before/after and net presence factors. This is likely a reflection of the high level of variability in seahorse density between sites as well as variation over survey occasions.

Hippocampus whitei abundance in *P. australis* has been shown to vary as a result of predator abundance, depth, habitat density and prey amounts (Manning, Foster, Harasti, & Vincent, 2018). It is likely that this variation does not indicate any long term differences in seahorse density on natural habitats between test and control sites due to swimming net presence, and this is supported by our regression analyses that showed that there were no long term increases or decreases in seahorse density over time at any sites over the course of the experiment. Additionally, there was relatively even movement of tagged seahorses between habitats at sites where net panels were installed, and similar site fidelity of seahorses on *P. australis* at sites with and without nets indicate a lack of attraction from one habitat to another.

Another concern of artificial habitats is that they may create an ecological trap; a phenomenon where animals prefer habitats where their fitness is lower than it would have been if they had selected another available option (Robertson & Hutto, 2006; Robertson, Rehage, & Sih, 2013). Assessing ecological traps, therefore, requires information about fitness and habitat preferences. *Hippocampus whitei* has been found to prefer swimming nets over *P. australis* in a laboratory experiment (Simpson et al., 2019). Here it was not possible to directly assess the fitness of individuals on nets and seagrass in the field, although modelled survivability estimates using MARK showed that survivability of individuals on the nets was ~10% lower on nets compared to seagrass, and the number of pregnant males and juveniles were also smaller on nets. While most examples of ecological traps are terrestrial, marine examples exist, such as the African penguin *Spheniscus demersus* becoming ecologically trapped in the degraded Benguela ecosystem (Sherley et al., 2017). Additionally, tuna have been shown to inhabit drifting fish aggregation devices (FADs) in high abundances despite decreases in health (Hallier & Gaertner, 2008) It is necessary to assess population survivability, reproduction and immigration/emigration of any taxa that may inhabit these structures to determine habitat quality (Runge, Runge, & Nichols, 2006). Though no evidence that seahorses are being attracted onto swimming nets away from natural habitats was found, caution should be used when considering swimming nets as a conservation tool. Future research should use field observations to further determine *H. whitei* survivability and fitness on natural and artificial habitats such as swimming nets, as well as continue to explore ways to make swimming nets a potentially higher quality habitat for seahorses (though further testing would be required in such cases to avoid the creation of a potential ecological trap).

Some recent suggestions are the addition of frayed material to increase net complexity (Hellyer et al., 2011) as well as seahorse friendly cleaning methods (Harasti et al., 2010).

Conclusion

In the face of rapid coastal urbanization and decreasing abundance of natural habitats for threatened marine taxa, it is important to both conserve these natural habitats as well as recognize where artificial structures may act as a potential conservation tool. The presence of swimming nets may provide seahorses with additional habitat in sites where natural habitat is becoming sparse, potentially resulting in increases in population numbers. Additionally, purpose built artificial structures have been found to support relatively abundant populations of seahorses in the absence of natural habitats (Simpson et al., 2020). It is likely that increased seahorse production is occurring on the nets with a net result of increased site-scale seahorse numbers, with no evidence that seahorses are being attracted away from natural habitat. The results show, however, that these effects may vary across survey occasions and sites. While swimming nets may serve as a useful replacement habitat while natural habitat becomes sparse, they are not a permanent solution and should be utilized in combination with other long-term plans to restore natural habitat as seahorse populations on nets may be vulnerable if nets are removed or cleaned. Strategies for the restoration of natural habitats should be prioritized, and seahorse friendly management strategies for swimming nets should be used. Swimming nets will hold most value as an artificial habitat for seahorses if left as permanent structures (Clynick, 2008) or are maintained using seahorse-friendly cleaning methods (Harasti et al., 2010). Future research should further explore fitness consequences for seahorses on artificial habitats such as swimming nets. Managers can implement the

findings of this research into conservation of the endangered *H. whitei* seahorses by promoting and creating public awareness of the importance of swimming nets as seahorse habitat, as well as seahorse friendly management of swimming nets. Managers should ideally plan for the deployment of new swimming nets in locations where they can fulfil their purpose as a safety net for swimmers whilst simultaneously providing new habitat where seahorse populations are in decline as a result of declining natural habitats.

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Authors contributions

Michael Simpson, Rebecca L. Morris, David Harasti and Ross A. Coleman conceived the ideas and designed the methodology; Michael Simpson collected the data; Michael Simpson analysed the data; Michael Simpson, Rebecca L. Morris, David Harasti and Ross A. Coleman interpreted the data and wrote the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

Compliance with ethical standards

All applicable international, national and/or institutional guidelines for sampling, care and experimental use of organisms for the study have been followed and all necessary approvals have been obtained. This research was conducted under DPI Scientific Collection Permit P17/0028-1.1, with ethics approval from the Sydney University Ethics Committee, Project Number: 2016/1066. No authors have any conflicts of interests to declare that may affect this research.

Data availability statement

The relevant raw data will be lodged in the University of Sydney library's Data Repository and a DOI can be provided upon acceptance.

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Tables

Table 1. Seahorse data collected during five diver surveys conducted at four sites in Sydney Harbour in April 2018, including Closed Jolly-Seber population estimates conducted in program MARK, with 95% confidence intervals or standard error (\pm) given in brackets.

Site	No. of seahorses observed	Site population estimate	Males observed	Estimated males	Females observed	Estimated females	Juveniles observed	Estimated juveniles	Smallest seahorse	Largest seahorse	Mean size
Fairlight	18	21 (16-26)	7	7 (6-8)	11	11 (10-12)	0	0	76 mm	130 mm	115.5 mm (\pm 2.66)
Delwood	16	18 (12-25)	5	5 (4-6)	9	9 (7-11)	2	2 (n/a)	62 mm	130 mm	104.5 mm (\pm 5.84)
Clifton Gardens	101	139 (87-190)	40	45 (37-53)	58	85 (64-106)	3	3 (0-8)	53 mm	140 mm	111.5 mm (\pm 1.78)
Manly Cove	28	39 (19-59)	10	11 (7-15)	14	17 (11-23)	3	3 (0-6)	60 mm	142 mm	111.5 mm (\pm 5.48)

Table 2. Seahorse data collected during monthly/bimonthly diver surveys conducted at four sites in Sydney Harbour from April 201 to June 2019, including open POPAN population estimates for seahorse populations conducted in program MARK with 95% confidence intervals and standard error (\pm) given in brackets.

Site	No. seahorses observed	Population estimate	Observed males	Estimated males	Observed females	Estimated females	Observed juveniles	Estimated juveniles	Observed Pregnant males	Resight probability	Average survivability	Smallest seahorse (mm)	Largest seahorse (mm)	Mean size (mm)	
Fairlight Site	57	87 (71-96)	28	41 (31-55)	25	42 (29-61)	4	4 (3-4)	11	0.24 (± 0.033)	0.88 (± 0.025)	45	137	109.8 (± 2.99)	
	Net	6	7 (6-8)	1	n/a	1	n/a	4	n/a	0	0.78 (± 0.129)	0.79 (± 0.125)	45	137	114.71 (± 8.35)
	Seagrass	51	79 (63-99)	27	39 (30-52)	24	40 (28-57)	0	0	11	0.21 (± 0.033)	0.88 (± 0.026)	49	106	69.00 (± 2.45)
Delwood Site	76	102 (89-118)	27	38 (29-49)	37	49 (40-59)	12	14 (12-15)	12	0.30 (± 0.018)	0.91 (± 0.018)	42	141	94.25 (± 2.88)	
	Net	24	26 (25-27)	3	3 (2-4)	10	11 (10-12)	11	12 (11-14)	2	0.88 (± 0.043)	0.84 (± 0.043)	42	129	70.5 (± 5.61)
	Seagrass	56	91 (71-116)	26	42 (29-62)	29	44 (33-60)	1	27 (19-39)	10	0.14 (± 0.023)	0.94 (± 0.021)	44	141	103.10 (± 2.77)
Little Manly	51	60 (51-69)	26	29 (24-34)	20	25 (19-33)	4	6 (2-15)	11	0.41 (± 0.034)	0.90 (± 0.019)	61	140	109.37 (± 3.00)	
Quarantine	21	86 (31-236)	9	15 (11-22)	12	23 (15-35)	0	0 (n/a)	2	0.23 (± 0.179)	0.41 (± 0.166)	70	143	102.90 (± 4.65)	

Figure legends

Figure 1. Location of study sites at which observational or manipulative experiments were held to assess the effect of swimming net presence on seahorse populations within Port Jackson, NSW, Australia.

Figure 2. Example of an experimental swimming net panel installed at Delwood Beach, NSW, Australia. Photograph taken in January, 2019.

Figure 3. Mean seahorse densities on *P. australis* at four sites in Sydney Harbour, with either swimming net present (Manly Bay and Clifton Gardens) or absent (Delwood Beach and Fairlight) at the site. Bars represent standard error.

Figure 4. Estimated population sizes of *H. whitei* at each of four sites in Sydney Harbour. Two of these sites contained seahorse habitat of *P. australis* and swimming nets (Manly Cove and Clifton Gardens) and at the other two sites, the seahorse habitat was *P. australis* alone. Bars represent 95% confidence intervals.

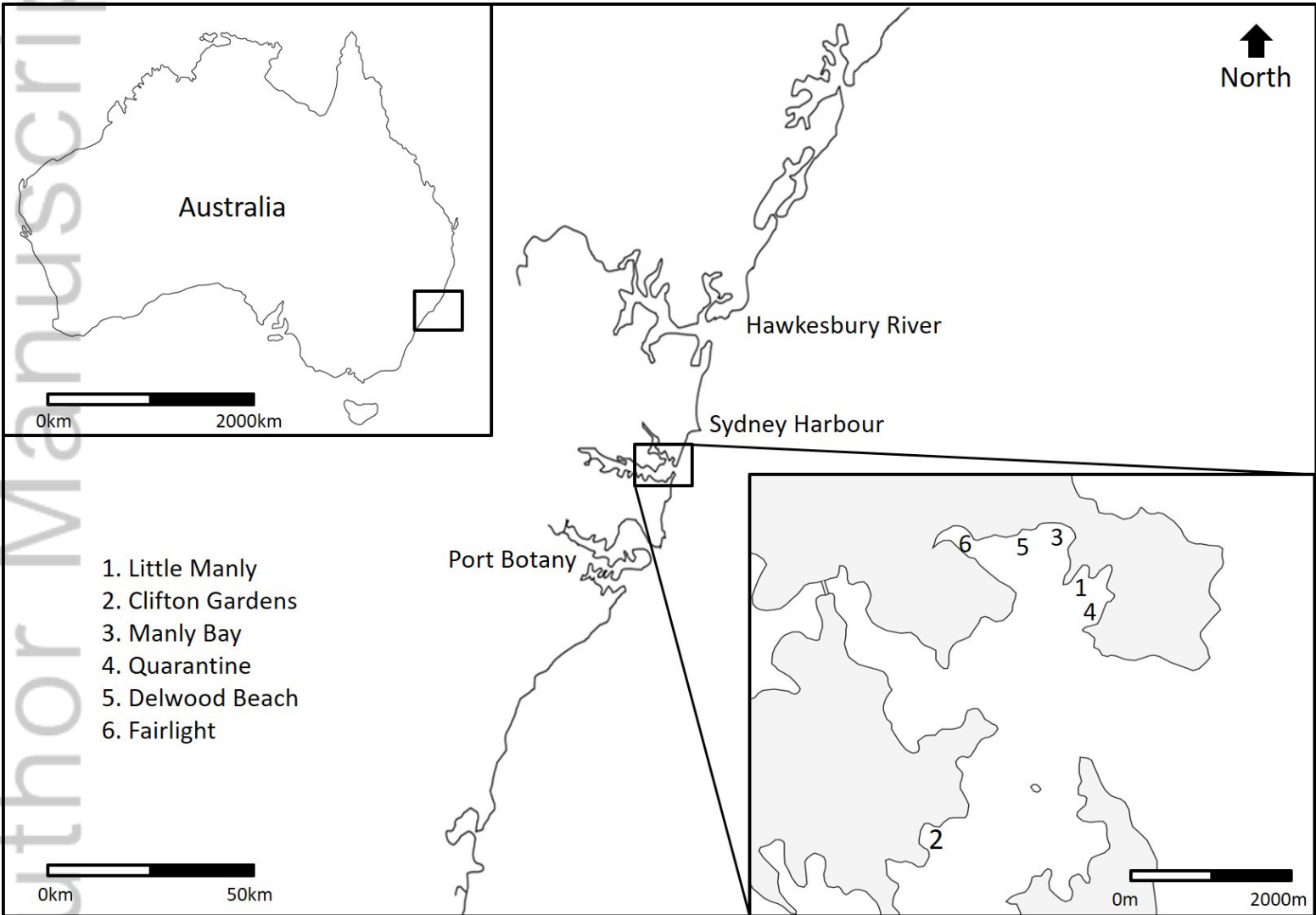
Figure 5. Seahorse density (seahorses per five minute timed search) on *P. australis* seagrass for four sites are shown, consisting of two sites where experimental panels of swimming net were installed (Delwood Beach and Fairlight) and two control sites (Little Manly and

Quarantine Station). Six surveys before net installation are shown, as well as the final six surveys prior to net removal. Bars represent standard error.

Figure 6. Mean *H. whitei* density on both *P. australis* seagrass and installed panels of swimming net material at two sites in Sydney Harbour, Australia. Bars represent standard error.

Figure 7. Mean *H. whitei* density In *P. australis* seagrass at four sites in Sydney Harbour, NSW, Australia; A) Fairlight, B) Delwood Beach, C) Quarantine station and D) Little Manly. At sites A and B, seahorse density on installed panels of swimming net material is also given. Trendlines are fitted to the data.

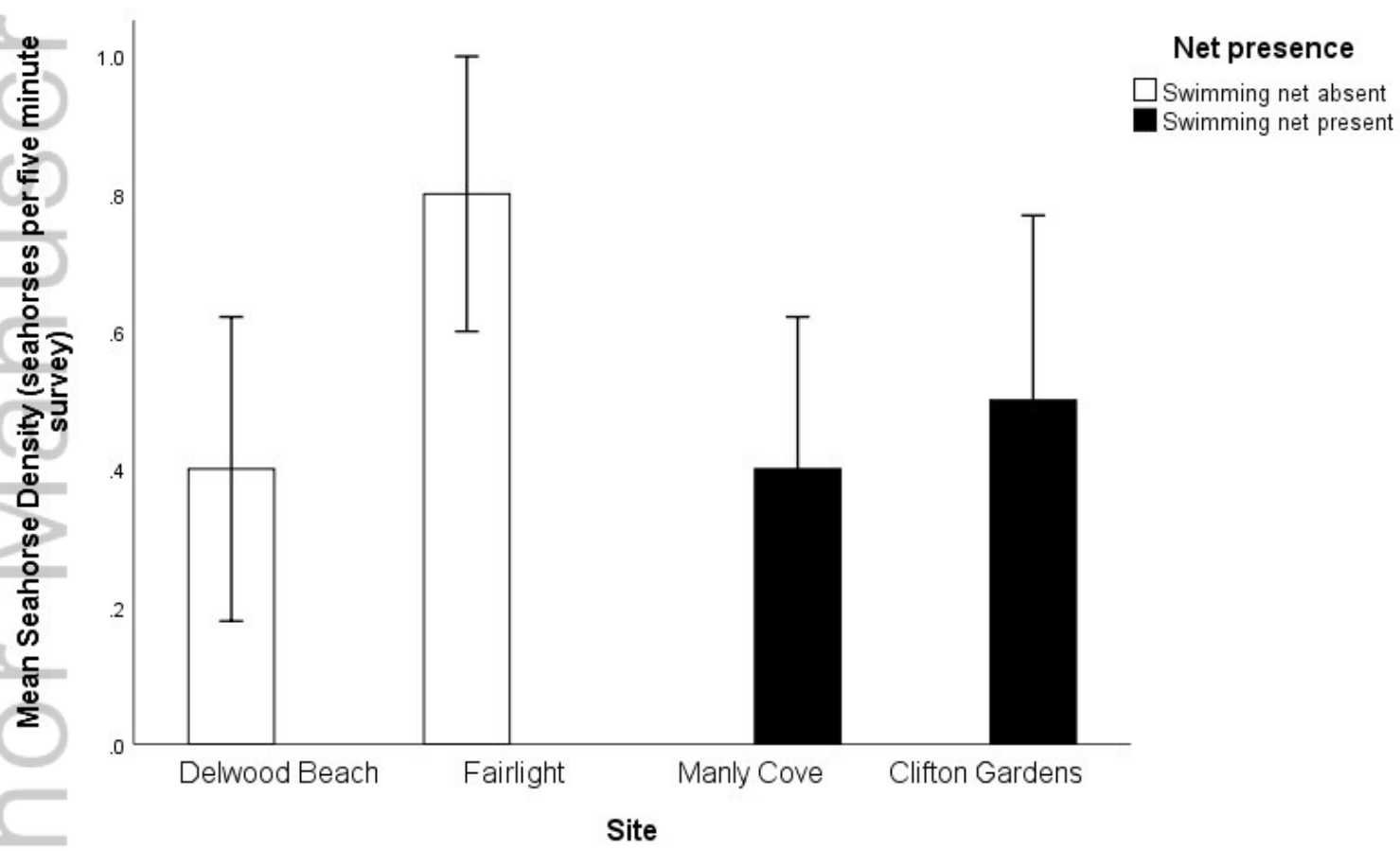
Figure 8. POPAN estimates of seahorse populations at four sites in Sydney Harbour, including A) Fairlight, B) Delwood Beach, C) Quarantine and D) Little Manly. POPAN estimates are displayed for both A) seagrass and B) site scale (including installed panels of swimming nets) populations of seahorses. Bars represent 95% confidence intervals.



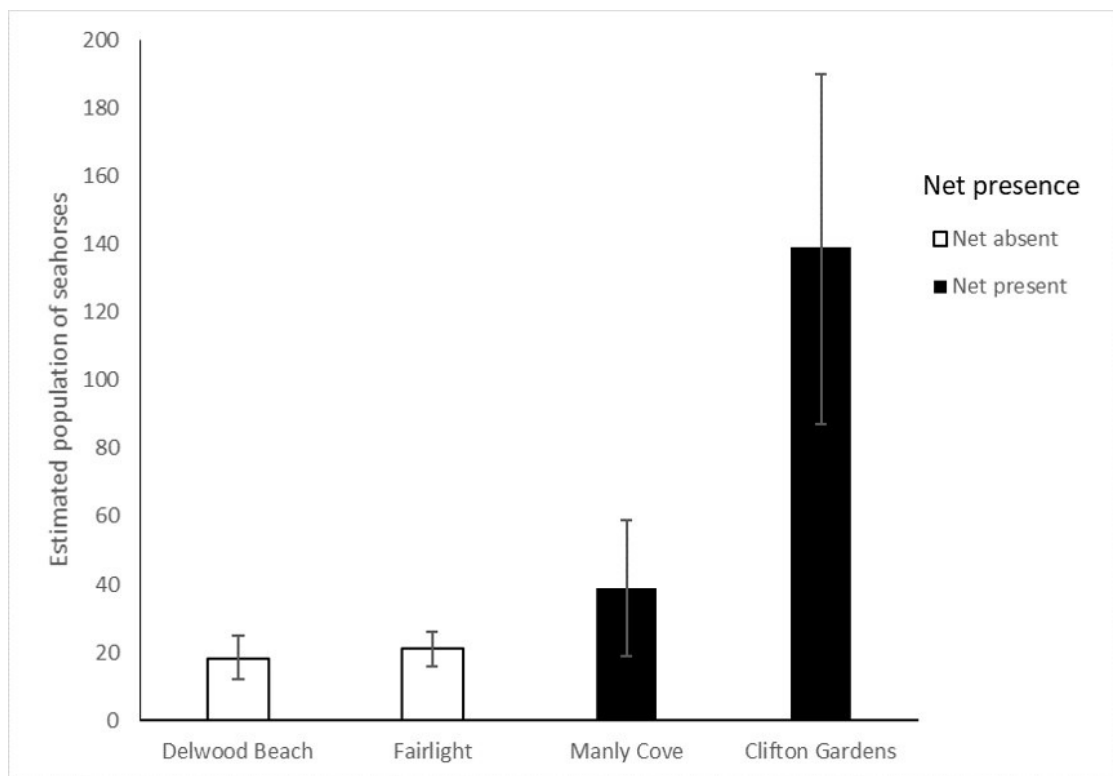
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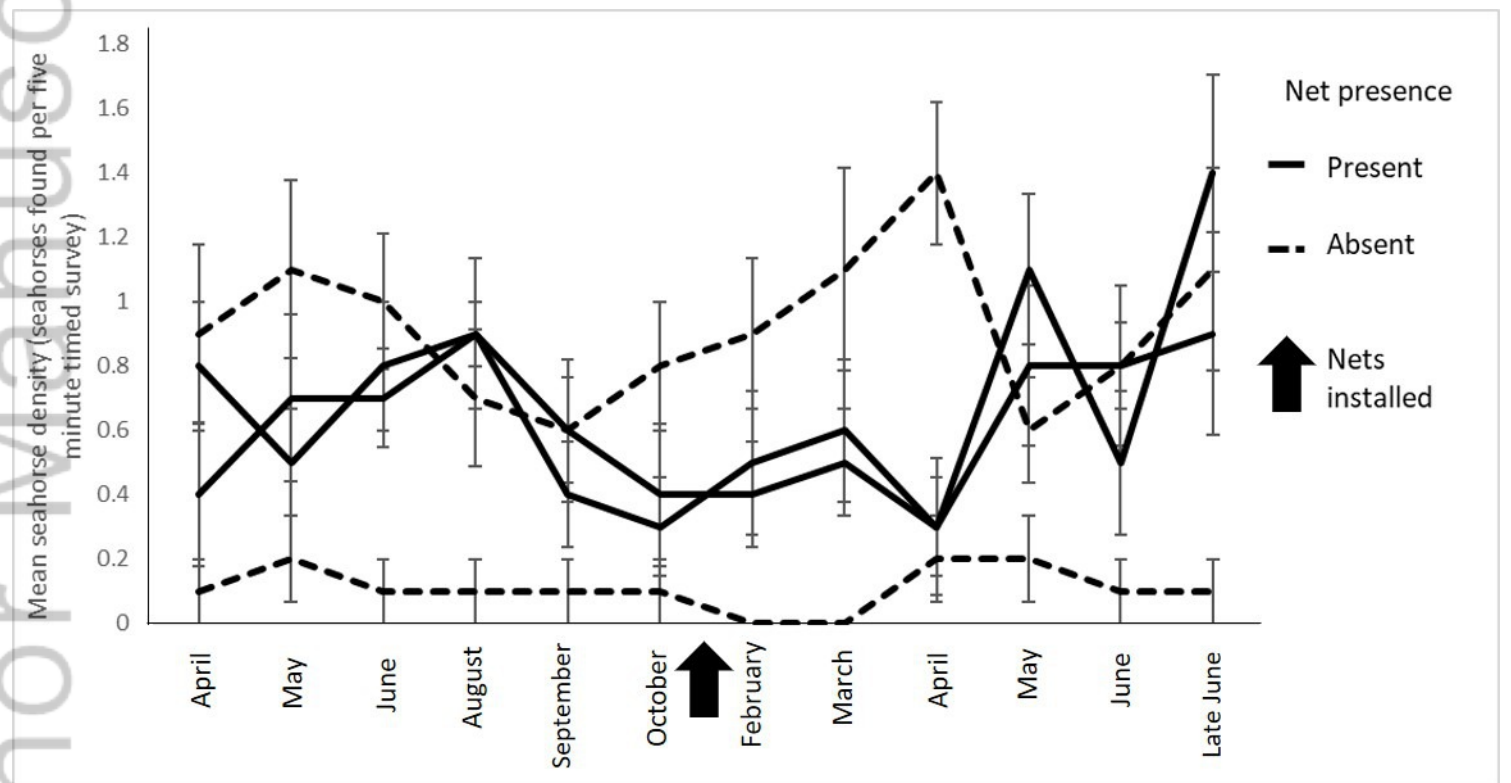
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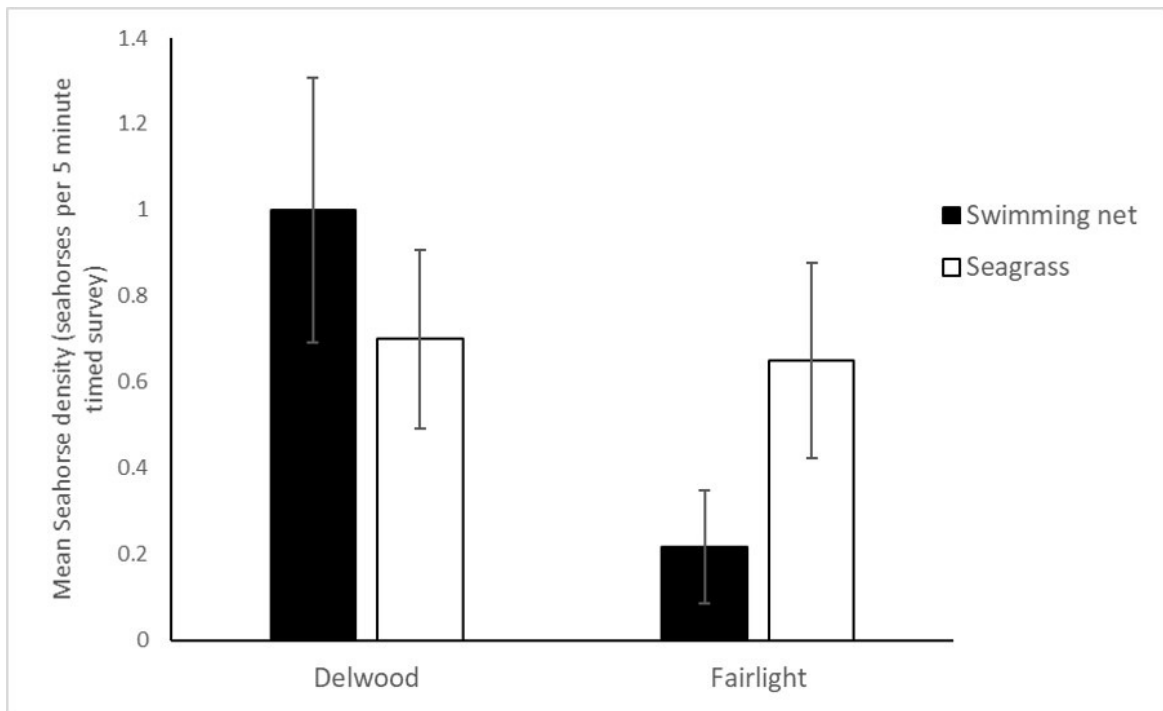
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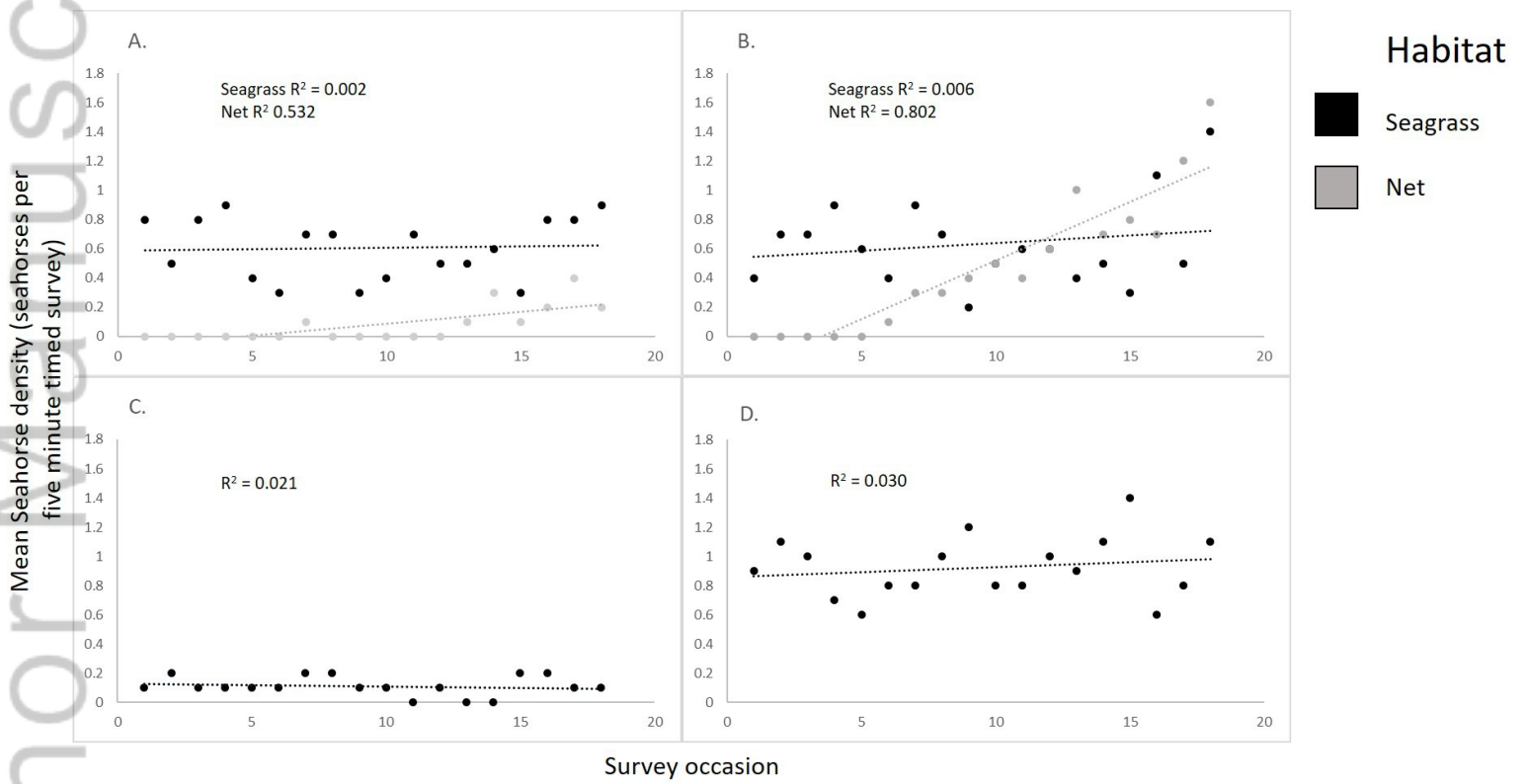
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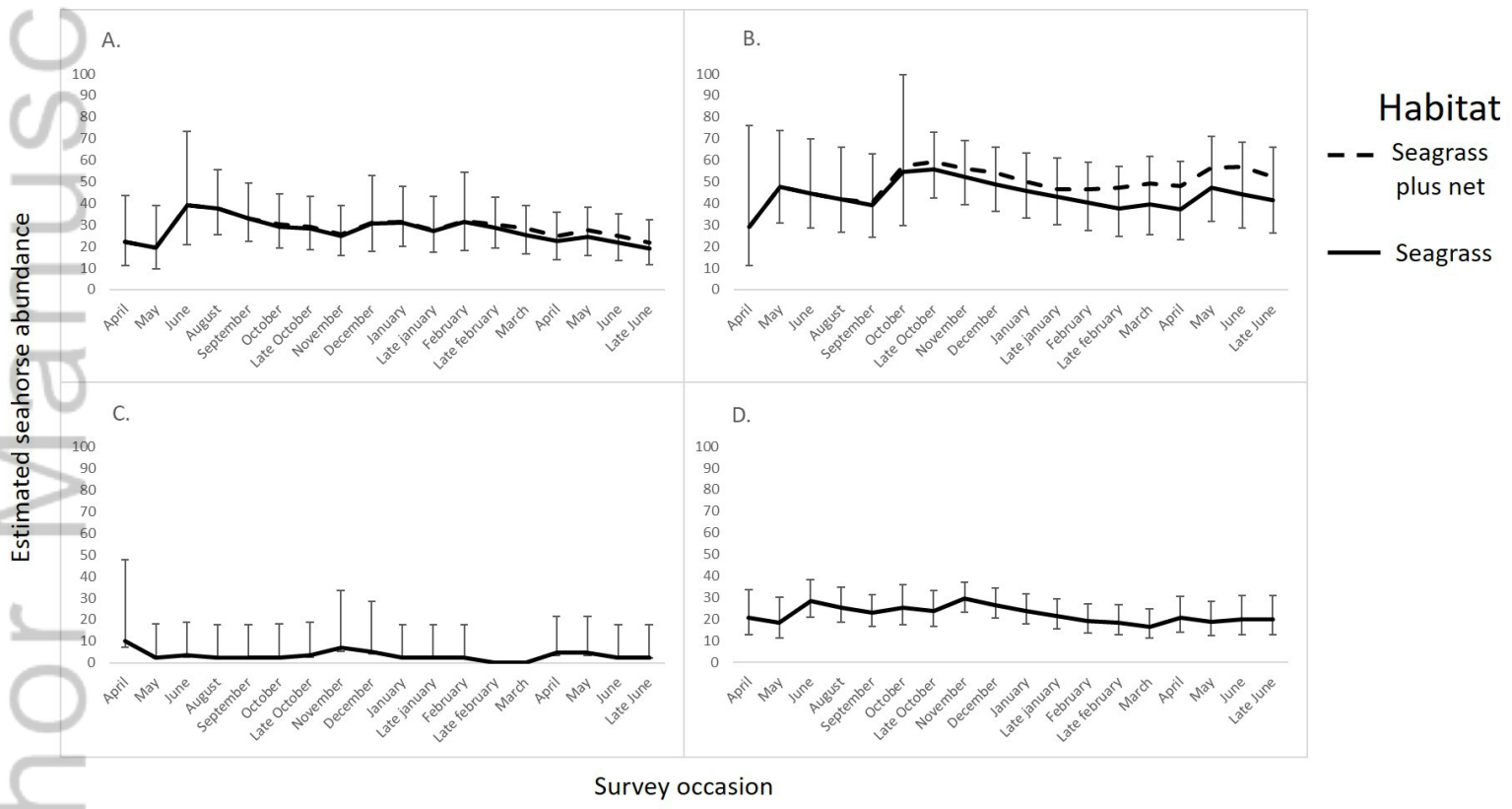
AQC_3451_Figure 5.jpg



AQC_3451_Figure 6.jpg



AQC_3451_Figure 7.jpg



Survey occasion

AQC_3451_Figure 8.jpg