

Declining seahorse populations linked to loss of essential marine habitats

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ABSTRACT: Habitat loss is a key driver in the decline of terrestrial and marine species worldwide. In the marine environment, habitat loss is considered to be a major threat to seahorses *Hippocampus* spp. This study assessed changes in the population abundance of White's seahorse *Hippocampus whitei* from mark-resight surveys undertaken between 2006 and 2015 in Port Stephens (New South Wales, Australia). The mark-resight closed population estimates found that the population at the Seahorse Gardens site had declined by 97 % between 2009 and 2015, and similarly, the population at the Pipeline site was found to decrease by 83 % over the same time period. Comparison of habitat composition data found a significant decline in available marine habitats between 2009 and 2015 at both sites. SIMPER analysis indicated that 5 habitat types (soft coral *Dendronephthya australis*, sand, seagrass *Halophila ovalis*, sponges and algae) contributed 76 % of the dissimilarity between the sampling periods. The preferred habitats of *H. whitei* had significantly declined at both sites, with *D. australis* declining at the Seahorse Gardens site by ~96 % from 2009 to 2015 and by ~73 % at the Pipeline site, whilst sponge habitat was also found to decline by ~49 % at the Seahorse Gardens site and ~25 % at the Pipeline site. The significant decline of *H. whitei* abundance is concerning, as there is no evidence to suggest that the populations are recovering, and given the large decline of habitat availability at both sites, it would be difficult for seahorses to recolonise these areas without some form of habitat remediation.

KEY WORDS: Habitat loss · *Hippocampus whitei* · Syngnathidae · Population abundance · Mark-resight · NOREMARK

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INTRODUCTION

Loss of habitats that species rely on is a recognised problem for both terrestrial and marine species (Brooks et al. 2002). Habitat loss is one of the greatest contributors to species declines worldwide (Fahrig 1997). In the marine environment, habitat loss is considered to have contributed to significant declines in marine diversity (Airoldi et al. 2008, McCauley et al. 2015), with habitat loss shown to impact on abundance of marine invertebrates (Carlton 1993, Hubbard et al. 2014), marine turtles (Gibbons et al. 2000, Christianen et al. 2014) and coral reef fishes (Munday 2004, Pratchett et al. 2012). One family of fishes considered to be under threat from habitat loss is the Syngnathidae; the

group containing seahorses, pipefish, pipehorses and seadragons. Seahorse populations are adversely affected by overfishing, collection for aquaria and curios, the degradation of important habitats (Vincent et al. 2011) and potentially environmental warming (Aurélio et al. 2013). The biology and ecology of seahorses—small home ranges, low population densities, monogamous mating and limited distribution—make them susceptible to localised anthropogenic impacts (Foster & Vincent 2004). As a result of concerns regarding worldwide declines in seahorse populations, all seahorses were listed in the Convention on International Trade in Endangered Species of Wild Fauna and Flora in 2004 and several species are classified as Threatened on the IUCN Redlist (IUCN 2015).

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There have been few studies that have assessed changes in seahorse populations in relation to habitat loss. In Florida, the population size of *Hippocampus zosterae* was observed to decline as a result of damage to seagrass caused by the demolition and construction phases of 2 adjacent marinas (Masonjones et al. 2010). The use of dynamite in the Philippines damaged coral reefs, contributing to a decline in *H. comes* populations (Marcus et al. 2007). In Malaysia, *H. kuda* abundance was reduced after extensive port development decimated large areas of seagrass (Vincent et al. 2011), which is considered to be a very important habitat for this species. The availability of holdfasts, which seahorses use to curl their tail around for anchoring, positively affects the densities of *H. guttulatus* when the percentage of holdfast coverage increases (Correia et al. 2015). Habitat degradation has negatively impacted seahorse species that utilise inshore habitats and there are concerns for the long-term conservation of several *Hippocampus* species (Vincent et al. 2011).

The focus of this study is *Hippocampus whitei*, the White's or Sydney seahorse, that is endemic to estuaries of the New South Wales (NSW) central coast region (Harasti et al. 2012), one of the most densely populated region of Australia (DECCW 2009). In NSW, seahorses are totally protected under the *Fisheries Management Act 1999* making it illegal for them to be taken from the wild. The main locations where *H. whitei* are considered to be most abundant are in Port Jackson (Sydney Harbour) and in Port Stephens (Harasti et al. 2012). Extensive studies have been undertaken on *H. whitei* within Port Stephens assessing life history and population parameters (Harasti et al. 2012), flash photography impacts (Harasti & Gladstone 2013), habitat preferences, site fidelity and movements (Harasti et al. 2014b). Populations of seahorses in Port Stephens can also be affected by predation, as increased numbers of predators in a protected area cause a decrease in seahorse abundance (Harasti et al. 2014a). The data collection for these studies concluded in 2010; however, since 2010, there has been an apparent decline in the numbers of seahorses at various sites across Port Stephens (Harasti 2014). The rate and extent of this decline is unknown, and given that *H. whitei* is listed as data deficient on the IUCN Redlist (IUCN 2015), information on changes in population abundance would assist in the future conservation and management of this species. In addition to the apparent decline of seahorses, apparent habitat loss has also been observed at the 2 sites where seahorses are known to occur in high

abundance in Port Stephens; however, the level of habitat loss has not been quantified.

The specific objective of this study was to assess the level of decline in populations of *H. whitei* within Port Stephens, if any, at 4 sites where abundance of the species has previously been recorded. Specifically, the aims of this study were to assess whether (1) the population abundance of *H. whitei* has changed from 2007–09 to 2013–15 using relative abundance estimates from timed swim counts, (2) the population abundance of *H. whitei* has changed from 2006–09 to 2014–15 using mark-resight closed population estimates; (3) the relative abundance of predators had increased at the sites, and (4) habitat composition at sites within Port Stephens had changed over a period of 6 yr.

MATERIALS AND METHODS

This study was undertaken at 4 sites in Nelson Bay, Port Stephens, NSW, Australia with the sites located within the Port Stephens–Great Lakes Marine Park (PSGLMP) (Fig. 1). The 2 main study sites were dive sites known as the 'Pipeline' and the 'Seahorse Gardens'; both are approximately 6000 m² in extent and have the largest known populations of *H. whitei* in Port Stephens (Harasti et al. 2014a). Both these sites are located within the Habitat Protection Zone (HPZ) of the PSGLMP. The HPZ allows recreational fishing; however, there are restrictions on commercial fishing methods, with habitat-damaging fishing methods such as trawling prohibited. The other 2 sites surveyed were Fly Point and Little Beach, both approximately 6000 m², located within a no-take sanctuary zone where all forms of fishing and collecting have been prohibited since 1983, and where there is minimal anchoring as a result of the no-fishing regulations. The marine habitats at these sites have previously been mapped at various levels (Davis et al. 2015, Poulos et al. 2016); the sites contain a range of marine habitat-forming species (seagrasses, soft corals and sponges) at depths of 3–11 m, and a diverse range of mobile marine species have been recorded (Poulos et al. 2013), including rare and protected species (Harasti 2015, Scott et al. 2015).

Annual population abundance estimates

Population abundance surveys were undertaken annually at 2 sites, the Seahorse Gardens and Pipeline, between June–August 2006–2009 and between

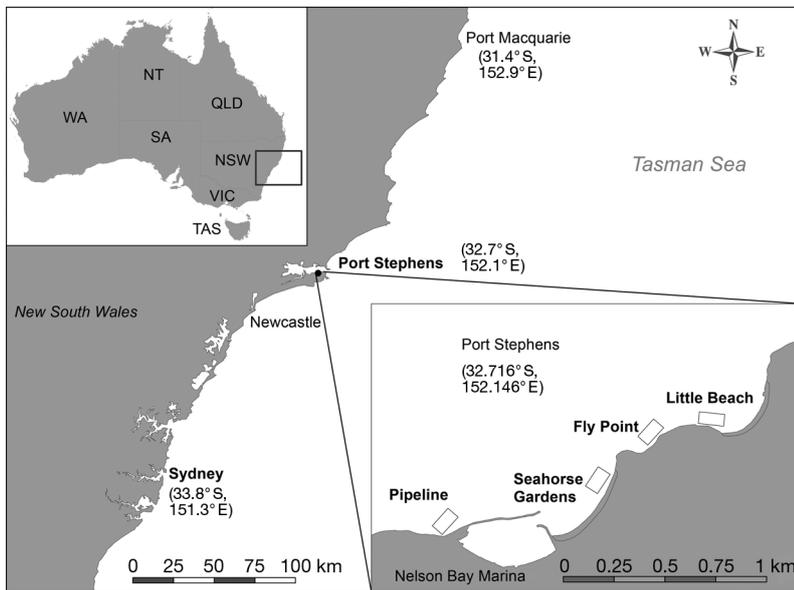


Fig. 1. Location of *Hippocampus whitei* survey sites, Port Stephens, New South Wales, Australia

July–August 2014–2015 using mark-resight estimates of tagged individuals. Population estimates for 2006 are reported in Harasti et al. (2014a) and are included in this analysis. Population estimates were not feasible for the other 2 sites, Little Beach and Fly Point, because of low abundance at these localities (Harasti et al. 2014a). For consistency, and to allow comparisons over time periods, the same methods used in 2006 for obtaining *H. whitei* population estimates through mark-resight were used in the subsequent surveys. Seahorses at both sites were tagged annually with visible implant fluorescent elastomer (VIFE; Northwest Marine Technologies, www.nmt.us) using the tagging techniques described for *H. whitei* in Harasti et al. (2012) with each seahorse receiving 3 individual coloured tags (green, orange, red or yellow) in a unique pattern to allow for individual identification. Animals were tagged *in situ*, with *H. whitei* placed back on their holdfast following the 1–2 min tagging process. To obtain estimates of population abundance, a population survey consisted of 5 surveys over consecutive days where all VIFE tagged and untagged *H. whitei* were recorded, with dive time approximately 70 min per survey and surveys conducted during the daytime. On each day, all resighted tagged animals were individually recorded and any unmarked animals were subsequently tagged. The aim of tagging animals continuously over the 5 d survey period was to attempt to have all seahorses on the site tagged, as the more animals tagged would improve population abundance esti-

mates and associated confidence limits (Seber 1982). Diving was only conducted on the high tide as this was the time of clearest water and least current.

Relative seahorse and predator abundance

Estimates of monthly relative abundance of *H. whitei* and known predators were undertaken across the 4 sites from August 2007 to July 2009 and August 2013 to July 2015 using 60 min timed swims, using the same method as previously conducted in Harasti et al. (2014a). The timed swims were conducted monthly within each of the 4 study sites, where the author undertook a haphazard swim around the site searching through all

available habitats for the presence of any seahorses. To minimise problems associated with non-independence, the start and end point varied from survey to survey and the dive pattern varied from month to month. In addition to the recording of the seahorses, predator species known to attack or feed on *H. whitei*, as reported in Harasti et al. (2014a), were also recorded during the 60 min timed swim (fishes: dusky flathead *Platycephalus fuscus*, eastern red scorpionfish *Scorpaena jacksoniensis*, striped anglerfish *Antennarius striatus*; octopus: Sydney octopus *Octopus tetricus*, blue-lined octopus *Hapalochlaena fasciata*). Water temperature was recorded monthly at each site from 2 Mares dive computers (www.mares.com).

Habitat composition

Data on habitat composition for both the Pipeline and Seahorse Gardens sites were collected in January 2009 as part of a study on habitat preferences for adult and juvenile *H. whitei* (Harasti et al. 2014b). Data in 2009 were collected by using a point-transect method to provide an estimate of each habitat type (Choat & Bellwood 1985) where 20 transects measuring 30 m in length were haphazardly placed at each site. To avoid overlapping of transects, the initial transect was placed at one end of the site with all subsequent transects laid approximately parallel to the first, and separated from the previous transect by

at least 5 m, towards the other end of the study site. To obtain an estimate of habitat coverage, at 1 m intervals the habitat directly below the transect was recorded providing a total of 600 habitat points for each site. For direct comparison, this point-count habitat collection method was replicated in February 2015 at both the Pipeline and Seahorse Garden sites using the same habitat categories (i.e. sponge, soft coral, kelp and macroalgae habitats) as described in Harasti et al. (2014b) with all data collected by myself for consistency.

Targeted mapping of the soft coral *Dendronephthya australis* habitat was previously undertaken at both sites in 2011 as part of a broader mapping program for the PSGLMP (Poulos et al. 2016). Mapping used a diver-towed GPS system (Garmin Map60; www.garmin.com) attached to a float at the surface, whilst filming underwater with a video camera time-synchronised to the GPS. When a patch of *D. australis* was located, its perimeter would be followed while maintaining a tight line to the GPS on the surface; the time-point and location information were then combined to identify the area of *D. australis* coverage as described by Poulos et al. (2016). The GPS data were downloaded onto a computer and soft coral polygon files were produced in ARCGIS 9.1 (www.arcgis.com). This towed GPS mapping method was replicated in June 2015 to assess actual changes in the area of occurrence of *D. australis* at both sites.

Data analysis

Annual population abundance

Annual population abundance estimates were derived from the joint hypergeometric maximum likelihood estimator (JHE) using Lincoln-Petersen estimates in the program NOREMARK (White 1996). An extension in the program NOREMARK allows additional marked animals to be added to the population between daily sighting occasions. The *H. whitei* populations were considered to be closed in that no immigration, emigration or mortality was expected during the consecutive 5 d survey periods.

Relative seahorse and predator abundance

The hypothesis that mean seahorse abundance would differ between August 2007–July 2009 and August 2013–July 2015 survey periods was tested by a 2-factor permutational multivariate analysis of

variance (PERMANOVA) using PERMANOVA+1.0.5 within PRIMER-E 7 (www.primer-e.com/) (Anderson et al. 2008). The factor survey period was analysed as fixed with 2 levels (2007–09, 2013–15) and the factor site was analysed as fixed with 4 levels (Pipeline, Seahorse Gardens, Fly Point, Little Beach). Each monthly survey was treated as a replicate ($n = 24$) for each survey period. Analysis was done on the Euclidean distance similarity matrix with significance determined from $n = 9999$ permutations. The same 2-factor PERMANOVA design was used to test the hypothesis that predator abundance would differ between the 2 survey periods (2007–09 and 2013–15). Post-hoc evaluations of significant results were done using pairwise *t*-tests. Water temperature was analysed using a 2-factor ANOVA, with factor survey period analysed as fixed with 2 levels (2007–09, 2013–15) and the factor site analysed as fixed with 4 levels, with analysis conducted in SPSS 20 (www.ibm.com/spss).

Habitat composition

The hypothesis that habitat composition would differ between August 2007–July 2009 and August 2013–July 2015 for the 2 locations (Pipeline and Seahorse Gardens), was tested using a 2-factor PERMANOVA. The number of occurrences of each habitat on each transect was summed, and data were square-root transformed. The analysis was done on the Bray-Curtis similarity matrix, with site and year both analysed as fixed factors, and significance was determined from $n = 9999$ permutations. To assess which habitats were responsible for the dissimilarity in the habitat composition, a similarity percentage routine (SIMPER) using Bray-Curtis similarity was performed (Clarke & Gorley 2006).

RESULTS

Population abundance estimates

The mark-resight closed-population estimates for *Hippocampus whitei* at both sites in 2014–15 indicates that the populations are greatly reduced when compared to population abundance estimates from 2006–2009 (Fig. 2). The population at the Seahorse Gardens in 2009 was estimated to be 370 (95%CL: 239–612) compared to only 10 animals (95%CL: 10–12) in 2015; a reduction of 97% over 6 yr. Similarly, the Pipeline population estimate in 2009 was 260 (95%CL: 225–309) compared to 45 (95%CL:

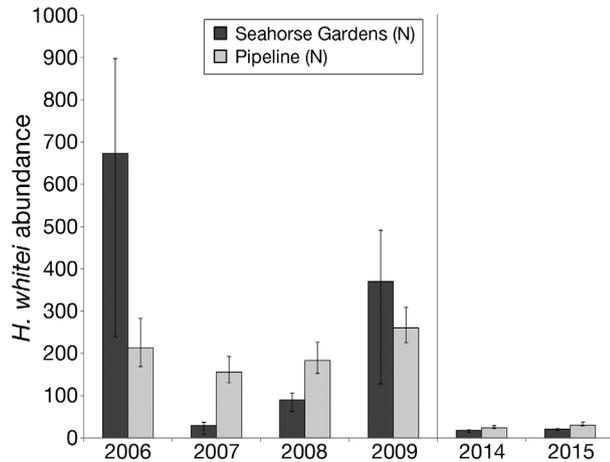


Fig. 2. Changes in *Hippocampus whitei* population abundance estimates (with 95% confidence limits) at 2 sites in Port Stephens (2006–09 and 2014–15) using closed population model with JH Estimator in the program NOREMARK

42–52%) in 2015; a reduction in population abundance of 83%. There were no records of tagged seahorses moving between any of the sites.

Relative abundance estimates for *H. whitei* and predators

A grand total of 1100 *H. whitei* were recorded in the monthly surveys across all sites from August 2007–July 2009 compared with only 427 recorded from August 2013–July 2015. Mean (\pm SE) monthly *H. whitei* abundance in 2007–09 was 11.56 ± 0.9 and was significantly higher compared to 4.45 ± 0.4 in 2013–15. There was a significant difference in the abundance of *H. whitei* between the 2 sampling periods ($F_{1,184} = 96.49$, $p < 0.001$); therefore, the hypothesis that abundance of *H. whitei* differs between the 2 survey periods is supported. There were significant differences in *H. whitei* abundance between the 4 sites ($F_{3,184} = 59.26$, $p < 0.001$) and there was a significant survey period \times site interaction ($F_{3,184} = 59.26$, $p < 0.001$). Pairwise tests indicate that the *H. whitei* abundance was significantly different between the 2 survey periods for both the Seahorse Gardens and Pipeline sites, whilst there was no significant difference in abundance between survey periods for Little Beach or Fly Point (Fig. 3).

The numbers of observed predators detected in between the 2 survey periods were similar across all 4 sites combined with a grand total of 690 in August 2007–July 2009 compared to 685 in August 2013–July 2015. There was no significant difference in the mean monthly abundance of predators recorded between

the 2 sampling periods ($F_{1,184} = 0.15$, $p > 0.05$); there was a significant difference in the numbers of predators between the 4 sites ($F_{3,184} = 413.22$, $p < 0.001$) and there was no significant sampling period \times site interaction ($F_{3,184} = 0.48$, $p > 0.05$). Pairwise tests indicate that predator abundance was significantly higher at the Fly Point and Little Beach sites (both sites located in no-take sanctuary zone) compared to lower predator abundance at the Pipeline and Seahorse Gardens sites where fishing occurred.

There was no significant difference in water temperatures between the 2 survey periods ($F_{1,184} = 0.02$, $p > 0.05$), no significant differences in water temperature across sites ($F_{3,184} = 0.25$, $p > 0.05$), and no significant sampling period \times site interaction ($F_{3,184} = 0.34$, $p > 0.05$).

Habitat composition

Habitat composition between 2009 and 2015 was found to be significantly different for the Seahorse Gardens and Pipeline sites ($F_{1,76} = 7.801$, $p < 0.001$), but there was no significant difference in habitat composition between sites ($F_{1,76} = 2.2$, $p > 0.05$) and no significant year \times site interaction ($F_{1,76} = 0.6$, $p > 0.05$). The SIMPER analysis determined that 5 habitat types contributed 76% of the dissimilarity amongst habitat composition between 2009 and 2015 (soft coral *Dendronephthya australis* 19.5%, sand 18.5%, *Halophila ovalis* 13.6%, sponges 13.1% and algae 11.5%). Comparison of habitat composition for 2 of

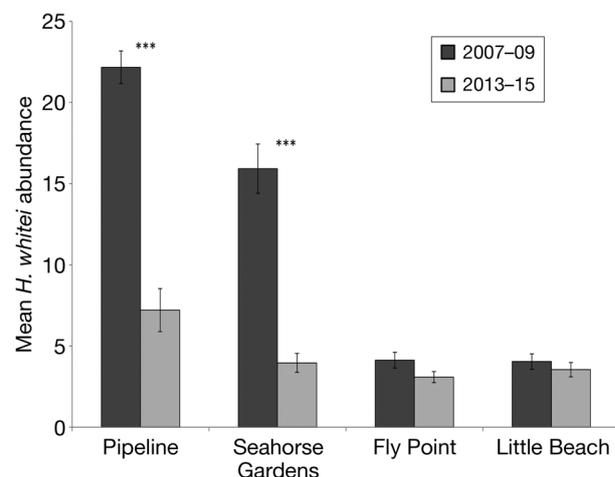


Fig. 3. Mean monthly relative abundance of *H. whitei* (\pm SE) recorded during 60 min dive surveys from Aug 2007–July 2009 and August 2013–July 2015 at the 4 sites in Port Stephens. ***Significant difference between survey periods ($p < 0.001$)

the preferred seahorse habitat types for adults, *D. australis* and sponges (Harasti et al. 2014b), showed that the *D. australis* habitat had declined at the Seahorse Gardens by ~96% from 2009 to 2015 and by ~73% at the Pipeline, whilst sponge habitat was also found to decline by ~49% at the Seahorse Gardens and ~25% at the Pipeline. Sand was found to substantially increase at both locations with a ~40% increase in sand coverage at the Seahorse Gardens and ~41% at the Pipeline. The preferred habitat of juvenile *H. whitei*, the gorgonian fan *Euplexaura* sp. (Harasti et al. 2014b), was also absent from the 2015 habitat surveys even though it was recorded in 2009.

A comparison of the *D. australis* coverage from targeted mapping surveys also confirmed a reduction in available *D. australis* habitat for both locations. In 2011, the mapped area of *D. australis* habitat was approximately 3864 m² at the Pipeline site and 2681 m² at Seahorse Gardens, respectively, compared to only 690 m² (83% decrease) and 65 m² (98% decrease) in 2015 (Fig. 4).

DISCUSSION

Population abundance from mark-resight closed population surveys and the relative abundance estimates both indicate a significant decline in *H. whitei*

abundance at the 2 main study sites (Pipeline and Seahorse Gardens) in Port Stephens over a period of approximately 10 yr. Seahorse abundance was not found to decline at the 2 sites (Fly Point and Little Beach) located within the no-take sanctuary zone where anthropogenic damage is minimised because of its protected status. From 2009 to 2015, there was a significant decline in the coverage of marine habitats at both the Pipeline and Seahorse Gardens sites. Particularly, the preferred habitats of adult and juvenile *H. whitei*, the soft coral *D. australis*, sponge habitat and the gorgonian fan *Euplexaura* sp. (Harasti et al. 2014b), had all significantly declined in the 2015 habitat surveys at the Pipeline and Seahorse Gardens, or in the case of *Euplexaura* sp., disappeared altogether from the sites. Whilst there were other marine habitats still available at both these sites, particularly *Posidonia australis*, *Halophila ovalis* and macroalgae, these are not preferred habitats of *H. whitei* in Port Stephens (Harasti et al. 2014b), and *H. whitei* were subsequently not found to move into these habitats during surveys in 2013–15. The most dominant habitat at both these sites in 2015 was sand; a habitat in which *H. whitei* has not been recorded (Harasti et al. 2014b).

Given that the numbers of predators were not found to significantly differ across the 4 sites between the 2 sampling periods (2006–09 to 2013–15),

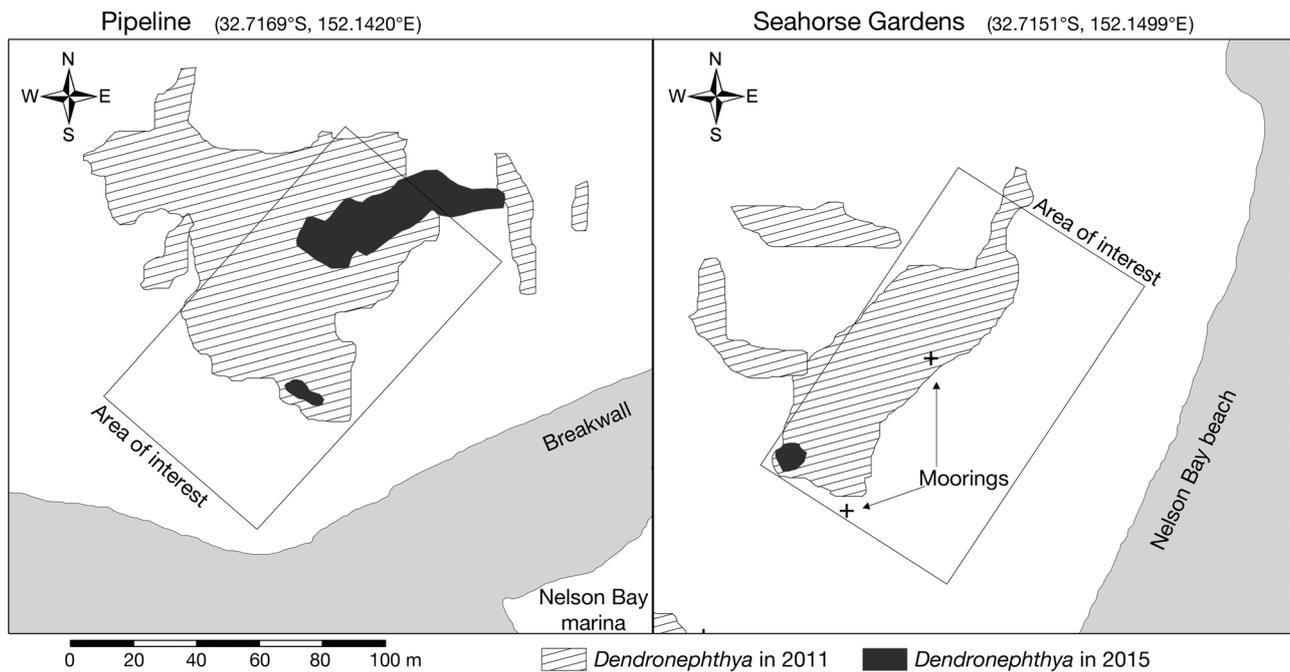


Fig. 4. Changes in the distribution of the soft coral *Dendronephthya australis* habitat from 2011 to 2015 at the Pipeline and Seahorse Gardens sites in Port Stephens from diver towed GPS mapping

it is unlikely that predation contributed to any decline in seahorse abundance as there was no recorded increase in numbers of predators. The significant difference in predator abundance between the 2 sites closed to fishing in the no-take sanctuary zone (Fly Point and Little Beach) and the 2 sites open to fishing (Pipeline and Seahorse Gardens) was not surprising given that Harasti et al. (2014a) had previously shown that predator abundance was significantly higher within this no-take fishing area.

There were no signs of obvious disease or illness, such as fungal infection, amongst the seahorses remaining on the sites, so it is improbable that disease would have been a contributing factor to seahorse decline. It is possible that *H. whitei* may have emigrated from the 2 sites affected by habitat loss; however, given that no tagged animals were recorded moving to other sites, and that the species displays strong site fidelity (Harasti et al. 2014b) and small home range (Vincent et al. 2005), it is unlikely that emigration contributed to the population declines. As the 4 sites are all located within 1.5 km of the coast, and water temperature did not significantly differ between survey periods or sites, water temperature is also unlikely to have affected the population abundance. Seahorse populations were found to remain stable at both sites within the protected area (Fly Point and Little Beach) and within these sites there is very little anchoring (if any) because of the no-fishing regulations, and no damage from moorings. Furthermore, there has been no apparent decline in marine habitats within the protected area, unlike the sites outside the protected area. Whilst predation, depth, water temperature, emigration and disease were unlikely to have contributed to *H. whitei* population declines, it is possible that other unknown factors may have contributed to the population declines at the 2 sites.

Declines in seahorse populations are not uncommon and have been recorded in other studies around the world (Martin-Smith & Vincent 2005, Caldwell & Vincent 2012). Populations of *H. guttulatus* varied significantly over a 13 yr period in the Ria Formosa Lagoon, Portugal, whilst *H. hippocampus* populations were not observed to significantly differ over the same time period (Correia et al. 2015). The observed fluctuations of *H. guttulatus* populations were positively correlated with the percentage availability of holdfasts (i.e. habitats that seahorses could curl their tails around), indicating that this species can increase in abundance if sufficient holdfasts are available (Correia et al. 2015). The findings of Correia et al. (2015) demonstrate the importance of hold-

fasts, whilst this study reinforces the importance of holdfasts to seahorses, as it demonstrates that seahorse populations can significantly decline when available habitats (holdfasts) decrease. Other studies have also documented the importance of holdfasts and different habitat types for various seahorse species (Curtis & Vincent 2005, Teske et al. 2007, Aylesworth et al. 2015, Gristina et al. 2015), whilst some species have been found to show ontogenetic changes in habitat use (Morgan & Vincent 2007, Harasti et al. 2014b).

There has previously been a recorded decline of *H. whitei* abundance at the Seahorse Gardens site (decline of 96% between 2006 and 2009); however, this population recovered within a period of 2 yr (Harasti et al. 2014a). It is of concern that neither the Pipeline nor Seahorse Gardens populations recorded in 2013–15 are showing any signs of recovering to the levels recorded in 2006–09; given that there has been such a significant decline in available habitat, such recovery appears unlikely. With the lack of available natural habitats at the sites, there may be a need to develop artificial habitats or possibly transplant natural habitats, to encourage recolonisation at the sites, particularly the Seahorse Gardens site that is now predominantly sand habitat. *Hippocampus whitei* will colonise artificial structures, particularly structures that have considerable epibiotic growth (Harasti et al. 2010, Hellyer et al. 2011). *Hippocampus guttulatus* will also utilise different types of artificially created holdfasts, and placement of such structures could contribute to the settlement of seahorse populations (Correia et al. 2013). The potential use of artificial holdfasts to help *H. whitei* recolonise these sites therefore warrants further investigation; however, it would be of greater benefit to minimise any future loss of habitats important to *H. whitei*.

One of the obvious questions arising from the study is: What caused the decline in habitats at both sites? There are 3 factors that have been identified as contributing to the loss of soft coral *D. australis* and sponge habitat: (1) damage from newly installed moorings (Seahorse Gardens site only), (2) anchor damage and (3) sand inundation.

Firstly, in November 2013, the existing environmentally friendly screw moorings at the Seahorse Gardens site were replaced with traditional block and chain moorings in an asset exchange between the then NSW Marine Parks Authority and the NSW Roads and Maritime Services. Following replacement, the most northern mooring was placed in the wrong location, approximately 30 m to the north in amongst the soft coral and sponge habitats, and was

incorrectly configured: the subsurface floats did not keep the chain off the bottom, resulting in the chain scouring an area of approximately 1300 m² over a period of 6 wk. As the soft corals are only shallowly connected to the substrate (Davis et al. 2016), all soft coral colonies in the vicinity of the mooring were dislodged from the sediment with only sand habitat remaining. The mooring was subsequently moved back to its original proposed location on bare sand; however, irreparable damage had been done to the soft coral and sponge habitats. Boat moorings can cause significant damage to seagrass habitats (Demers et al. 2013, Glasby & West 2015); however, the damage caused to other sensitive marine habitats such as *D. australis* has not previously been quantified.

The second factor, anthropogenic damage caused by boat anchors, results from the 2 sites' attractions to boaters. The Seahorse Gardens site is protected from north-east winds over summer, making it a desirable anchorage, and the Pipeline is also popular for recreational fishing. Whilst anchoring has never been quantified at either site, video footage of anchor damage to both the *D. australis* and sponge habitats has been recorded at both sites (author's unpubl. data).

The third factor contributing to significant habitat loss, sand inundation, predominantly affected the Pipeline site. In 2010, a large volume of sand moved westwards within the Port, smothering marine habitats at various locations (Wainwright 2011). The sponge and soft coral habitat in the deeper northern section of the Pipeline site were completely inundated with sand causing considerable habitat loss; however, this sand movement and subsequent habitat inundation was considered to be a natural process (Wainwright 2011). Whilst the first 2 impacting factors can be addressed through management intervention and marine park regulations, the impact of sand movement smothering marine habitats is much more complex and difficult to address, if at all possible.

This study clearly demonstrates the importance of marine habitats to *H. whitei* and that loss of their preferred habitats can lead to localised declines in population abundance. It is concerning that the populations at both sites are showing no signs of recovery, and given that both sites have lost significant habitat, particularly the Seahorse Gardens site, it is difficult to see how populations can return to 2006–09 levels without some form of habitat restoration or the introduction of artificial habitats. Whilst marine protected areas (MPAs) could provide protection for marine habitats essential to seahorses, the use of MPAs would invariably lead to increased numbers of pred-

ators potentially causing seahorse populations to decline (Harasti et al. 2014a). Better management of essential seahorse habitats could be achieved through implementation of no-anchoring areas and installation of environmentally friendly moorings to minimise disturbance and degradation to benthic habitats. Whilst the Port Stephens estuary was previously considered a 'stronghold' for populations of *H. whitei* (Harasti et al. 2012, 2014a), the present study indicates that its long-term conservation within the Port Stephens waterway is at risk if essential marine habitats continue to be lost, and that continued population declines could ultimately lead to this species also becoming listed as Threatened on the IUCN Redlist.

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