REGULAR PAPER

Life history and population dynamics of an endangered seahorse (*Hippocampus capensis*) within an artificial habitat

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Abstract

Species-specific life-history information is critical for successful conservation, particularly in establishing an accurate baseline status. Obtaining such information is challenging for most species, but in particular for rare and threatened marine species. To facilitate future conservation of the endangered Knysna seahorse (*Hippocampus capensis*) this study aimed to determine important life-history information for this species. Visible implant fluorescent elastomer (VIFE) tags were used to mark 78 seahorses within a residential marina estate in the Knysna estuary, South Africa, in February 2018. Using a mark-resight approach, the size and movement patterns of the population and growth rate of seahorses were determined over a 14-month period. The closed population estimate for *H. capensis*, within Thesen Islands Marina, was estimated to be 134 (118–152 95% C.I.) in February 2018 compared to only 72 (48–108 95% C.I.) in February 2019. The species showed rapid initial growth with males and females having similar rates of growth based on the specialised von Bertalanffy growth function model. The importance of Reno mattresses as a habitat for *H. capensis* was confirmed based on the high abundance and site fidelity of the population, which emphasises the conservation potential of heavily modified environments for threatened seahorse species. The use of VIFE tags was deemed effective in studying this endangered seahorse and allowed the collection of important information for this species which can be used in future Red List assessments and conservation actions.

KEYWORDS
mark-recapture, South Africa, syngnathidae, VIFE

1 | INTRODUCTION

Successful conservation and management of threatened species depend on accurate and current information regarding population dynamics and life-history parameters. In the 2019 report from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) an estimated one million species are at risk of extinction (Díaz *et al*., 2020), which emphasises the need for evidence-based conservation actions. Monitoring plays a critical role in successful conservation, and according to Legg and Nagy (2006) monitoring provides information to (a) indicate when a system departs from the desired state, (b) measure the success of conservation actions and (c) detect the effects of disturbances. It is, therefore, important to establish a baseline state using specifics such as population parameters with which future changes can be compared, without which shifting baselines become the status quo and no clear conservation conclusions can be made (Pauly, 1995).

Basic life-history information such as population size, habitat use and movement patterns are integral to establish a baseline state for a species. The IUCN Red List Assessment and a species' extinction risk can be used as a baseline state with which future changes can be

compared (Rodrigues *et al.*, 2006). Once a baseline state for a species is set, the effectiveness of conservation and management actions can be assessed over time using ecological information such as increases or decreases in population size.

Rare and cryptic species, in particular rare fish species, are notoriously difficult to study which has resulted in poor monitoring and lack of information on basic life history and population parameters (Jerde *et al.*, 2011; Thompson, 2004). One group of small cryptic marine fishes that are considered to be threatened worldwide are the seahorses (*Hippocampus* spp.) which includes 45 recognised species (IUCN, 2020; Short *et al.*, 2018, 2020). Seahorses are under threat from over-fishing and loss of marine habitats (*i.e.*, seagrass and sponge habitats) (Vincent *et al.*, 2011) with several species listed as "Threatened" (*n* = 14) and "Data Deficient" (*n* = 17) on the IUCN Red List (IUCN, 2020).

Two seahorse species are listed as "Endangered" on the IUCN Red List and face a significant risk of extinction in the wild. The first is *Hippocampus whitei* Bleeker, 1855, previously classified as Vulnerable, whose status was updated in 2018 based on new ecological information on its range, population size and habitat vulnerability (Harasti & Pollom, 2017). The second is *Hippocampus capensis* Boulenger, 1900, first listed as Endangered in 2000 and considered to be the most threatened seahorse in the world known to occur in only three estuaries along the south coast of South Africa (Knysna, Swartvlei and Keurbooms estuaries) (Pollom, 2017).

Previous research on *H. capensis* has focused on population abundances (Bell *et al.*, 2003; Lockyear *et al.*, 2006), range (Lockyear *et al.*, 2006) and habitat use (Claassens, 2016; Teske *et al.*, 2007). The Knysna seahorse is found within submerged vegetative habitats (De Villiers *et al.*, 2019; Teske *et al.*, 2007) and has adapted to withstand extreme and rapid salinity fluctuations (Riley, 1986), making it a true estuarine species. The Knysna estuary is considered the most important estuary for the long-term conservation of this species (Teske *et al.*, 2003), as seahorse populations within the other two smaller estuaries are vulnerable to floods (Lockyear *et al.*, 2006) and mouth breaching (Russell, 1994).

Although the majority of seahorse species around the world are found to occur in natural habitats, several species utilise artificial habitats (Correia *et al.*, 2013; Harasti *et al.*, 2010; Simpson *et al.*, 2020), including *H. capensis* (Claassens, 2016). Recent studies on *H. capensis* have found that it has adapted to living in artificial habitats such as Reno mattresses (Claassens, 2016; Claassens *et al.*, 2018; Claassens & Hodgson, 2018a). It was found that consistently high densities of *H. capensis* occurred within artificial Reno mattress habitat located in a residential marina estate (Thesen Islands Marina) in the Knysna estuary (Claassens, 2016; Claassens & Hodgson, 2018a), and an *in-situ* choice experiment found that *H. capensis* was significantly more likely to remain on or move onto artificial Reno mattress habitat compared to Cape Eelgrass *Zostera capensis* Setchell, 1933 (Claassens *et al.*, 2018).

Although recent research on *H. capensis* has focused on its use of artificial habitats, no research has been undertaken to assess its life history, site fidelity and population parameters within this habitat. The lack of these data makes it difficult to assess if this species is still in decline or increasing in abundance which is essential for the IUCN Red List assessment of this species. This study aimed to assess population abundance, site fidelity, movement and growth rates of the Endangered Knysna seahorse within an artificial habitat. The life-history information generated by this research will be the first for *H. capensis* and will provide important baseline data that will be used to inform future IUCN Red List assessments of this species and local conservation strategies.

2 | MATERIALS AND METHODS

This research was conducted under authorisation received from the South African National Parks (SANParks) Board (Permit No: CLAA-L/2017-004) and the Department of Environmental Affairs (Permit No: RES2017/103). Ethical authorisation for the research was received from the SANParks Animal Use and Care Committee (Permit No: 012/17).

2.1 | Study site

This study was conducted within the Knysna estuary (Figure 1), considered the most important estuary for the future survival of *H. capensis* (Teske *et al.*, 2003). The Knysna estuary is an estuarine bay (Whitfield, 1992) and South Africa's most important estuary in terms of conservation significance (Turpie *et al.*, 2002). This estuary contains the largest beds of *Z. capensis* in southern Africa (Adams, 2016), and in 2018 an estimated 70% (632 ha) of the estuary was covered by subtidal vegetation (Wasserman *et al.*, 2020). The submerged vegetative habitats of the Knysna estuary are, however, threatened by ongoing nutrient enrichment caused by non-compliant wastewater discharge (Claassens *et al.*, 2020; Human *et al.*, 2016).

To assess life history, movement and population parameters for *H. capensis* it was necessary to mark animals so that they could be individually identified. The marking of animals was undertaken by using visible implant fluorescent elastomer (VIFE) (Northwest Marine Technologies, Anacortes, WA, USA; www.nmt.us) tags that are commonly used in seahorse studies where individual identification is required (Curtis, 2006; Harasti & Gladstone, 2013). This method is considered to be non-intrusive, long-lasting and has not been found to have any adverse effects on those seahorse species studied using this approach (Harasti *et al.*, 2012; Woods & Martin-Smith, 2004).

Owing to the endangered status of *H. capensis* and the fact that VIFE tags have been used on this species in only one *ex situ* study (Le Cheminant, 2000), stringent restrictions were placed on the research permit for this study by the authorising authority (SANParks). The permit for the marking of individual *H. capensis* was limited to a maximum of 80 adults. In addition to this, to determine seahorse growth rate, a maximum of 15 juveniles could be marked. Owing to these limitations, the focus was on one site known to have consistently high seahorse densities (Claassens, 2016; Claassens *et al.*, 2018) to test the efficacy of using VIFE tags as a tool to determine population size, site fidelity and
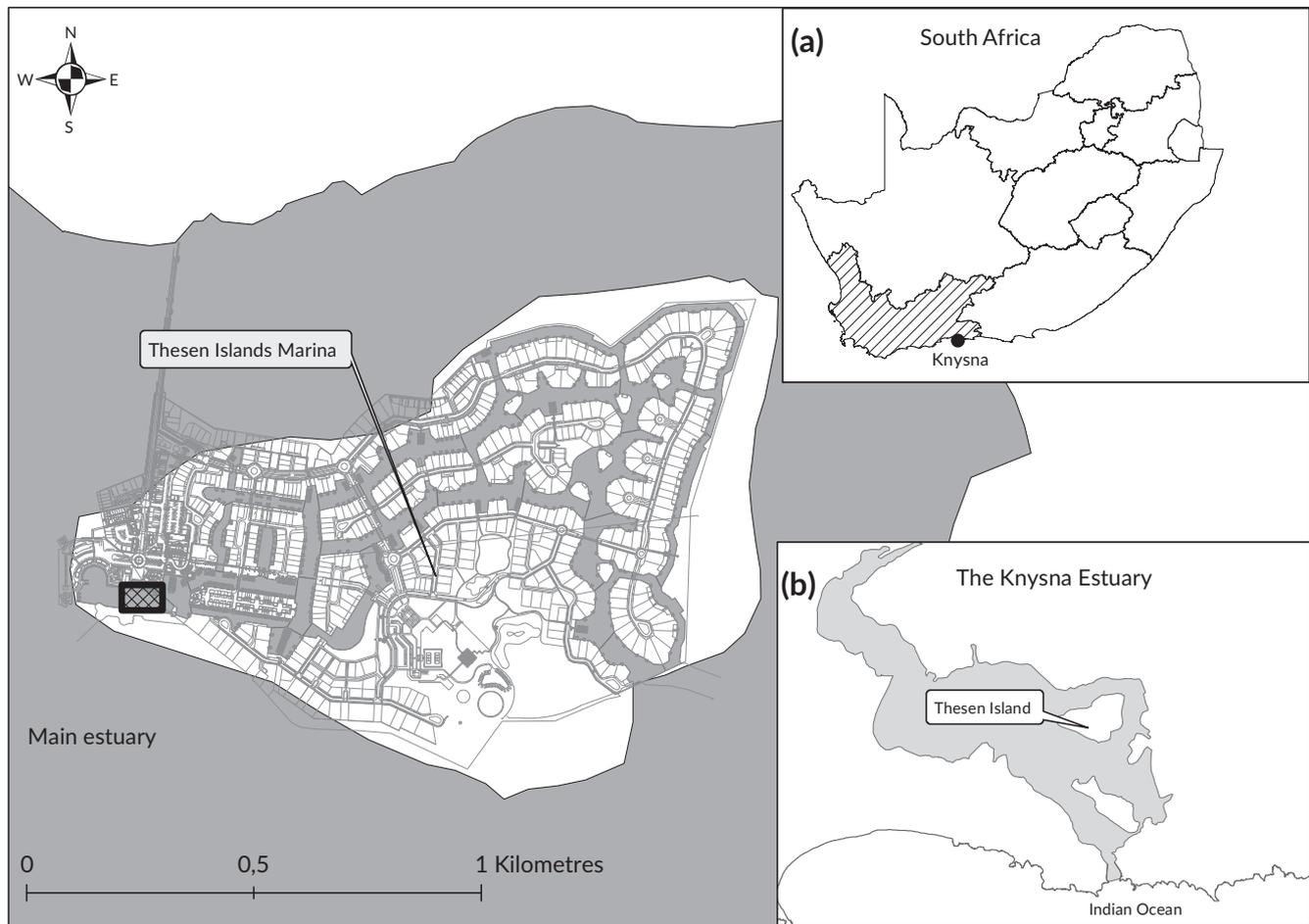
FIGURE 1 Locality of Knysna (Insert A) and the Knysna estuary (Insert B). Main map shows the layout of Thesen Islands Marina with the hatched area showing the study site

growth rates of *H. capensis*. The site, located in Thesen Islands Marina, consisted of a 50 m² (25 × 2 m) area of Reno mattress habitat which has been described in detail in Claassens (2016), Claassens *et al.* (2018) and Claassens and Hodgson (2018a). *Hippocampus capensis* has consistently been found in this site, specifically within the Reno mattress habitat which includes a complex environment with a high number of crevices and holdfasts. This habitat has also been colonised by various invertebrate and fish species. In the absence of macroalgal growth on the Reno mattress, such as *Asparagopsis taxiformis* (Delile) Trevisan de Saint-Léon, 1845, (which was the case during this study), the majority of seahorses use the Reno mattress wire as a holdfast (Claassens, 2016; Claassens & Hodgson, 2018a). Owing to the limited extent of the site and no variation in depth or temperature across the site surveyed, depth and temperature were not deemed relevant explanatory variables for consideration.

In February 2018, 78 adult seahorses (as per the permit conditions) were tagged using VIFE tags following the method described by Woods and Martin-Smith (2004) and monitored for a 14-month period. Every seahorse was given a unique code of three 1–2 mm red elastomer tags. The sex of seahorses was determined by the presence (male) or absence (female) of a brood pouch. Seahorses smaller than

4 cm with no distinct sexual characteristics were considered juvenile (Lockyear *et al.*, 2006). During initial marking, each seahorse was measured as per the protocols developed by Lourie (2003). All animals were marked *in situ* with the capturing, marking and release taking approximately 1 min. After marking, the seahorse was placed back on the holdfast it was found on. During the 14-month study period a number of monthly surveys focusing on specific aspects – which included surveys to (a) determine the population size through mark-resight surveys, (b) assess the relative population abundance, (c) conduct seahorse measurements and (d) to monitor seahorse movement – were conducted.

2.2 | Population abundance

Population abundance estimates for *H. capensis* were undertaken using both open and closed population models. A once-off marking period took place over 7 days which consisted of daily dives where all adult seahorses found were marked until 78 animals in total were marked by mid-February 2018. The fact that all marking took place during a single, initial event, with no subsequent marking taking place,

necessitated that a mark-resight analysis approach was followed (McClintock *et al*., 2009, 2019).

Directly after the marking session, an open population abundance analysis was undertaken for the Thesen Islands Marina population between February 2018 and March 2019. Monthly surveys (*n* = 14) were undertaken during this period where one diver on SCUBA (LC) swam around the study site where the 78 seahorses had been marked, for an hour, and all marked and unmarked seahorses were recorded. The site was searched by starting on one end of the study site and methodically searching through the Reno mattress habitat moving towards the opposite side. All animals marked during the initial marking session were considered to be present within the survey site and available for recapture during the first open population survey that took place in February 2018.

Seeing that the subsequent surveys occurred over a period of 13 months (March 2018–March 2019), the population was considered to be open as immigration, emigration, births and mortality would have occurred during the survey period (Seber, 1982). A Cormack-Jolly Seber (CJS) model for live recapture was applied, and seahorse sex was set as the attribute group across 14 encounter occasions (Cooch & White, 2017). Both sex and time (month) and the interaction between these two attributes were included as model parameters to determine the effect on survivability (φ) and probability of resighting individuals (*p*). The most parsimonious model was chosen based on Akaike's Information Criteria (AIC).

Population abundance was estimated from the CJS model by calculating the ratios of the number of individuals detected during each survey over the corresponding estimate of recapture probability. Recapture probability can be defined as the probability that an animal alive at time *t* is also captured at time *t* and is estimated by comparing the ratios of frequencies of different encounter ratios (Cooch & White, 2017). Seeing that recapture probability is used in the calculation, abundance was not estimated for the initial survey (February 2018). When a CJS model is used in MARK, the final survival and resightability estimates are confounded and cannot be used (Cooch & White, 2017). The last survey was thus excluded from the analyses (March 2019) which provided abundance estimates for a 12-month period (March 2018–February 2019) (Gimenez, 2016). This analysis was conducted in both Program Mark (White & Burnham, 1999) and by using the package "RMark" in R (Laake, 2013).

In addition to the 14-month open population abundance surveys, three closed population abundance surveys were conducted in February 2018, August 2018 and February 2019. Each primary survey consisted of four consecutive daily secondary surveys, where all marked and unmarked seahorses within the study site were recorded for each 1 h dive. During each primary survey, the population was considered to be geographically and demographically closed. Across the three primary surveys (February 2018–February 2019), the population was considered open, assuming that emigration, immigration, births and deaths could occur (Cooch & White, 2017; McClintock *et al*., 2019). A Poisson-log normal mark-resight model was used to determine population size seeing that animals were individually

identifiable, the number of marked individuals in the population (for primary surveys 2 and 3) was unknown and sampling was with replacement within secondary surveys (McClintock, 2017; McClintock *et al*., 2019). Considering the limitations set by the small sample size and resighting rates, estimates of population abundance for each of the three closed surveys were derived using a zero-inflated Poisson-log normal estimator (McClintock *et al*., 2019) in Mark.

In addition to the above, relative seahorse abundances were estimated from February 2018 to March 2019 through additional monthly 1 h visual surveys where all seahorses, both marked and unmarked, observed were recorded.

2.3 | Recapture probability and apparent survival rate

Recapture probability (*p*) and apparent survival rate (φ) were estimated from the monthly mark-resighting data using a CJS model analysed in Program MARK (Cooch & White, 2017). Differences between sex for *p* and φ were tested using AIC using models that assessed time dependence and sex.

2.4 | Length and length-at-age

Each seahorse was individually measured using straight line measurements (to the nearest 1 mm) with two measurements recorded: (a) tail length: tail to base of abdomen (bottom of pouch in males) and (b) trunk length: bottom of abdomen/pouch to top of coronet. All measurements were done *in situ* using a ruler. An estimate for total height (H_T) was obtained by combining both measurements. After checking for normalcy using a Shapiro–Wilk test (*P* > 0.5), the mean H_T of males and females across seasons was compared using a two-factor ANOVA.

To obtain estimates of growth rate for *H. capensis,* length-at-age was estimated using length increments derived from resighting data of marked individuals from February 2018 to March 2019. It was assumed that the growth of *H. capensis* corresponded to the specialised von Bertalanffy growth function (SVBGF) (Von Bertalanffy, 1938). The SVBGF equation is $L_t = L_\infty (1 - e^{-K(t-t_0)})$, where L_t is the length at time *t*, *K* is the growth rate constant (measurement of how quickly a fish approaches its asymptotic length), L_∞ is the maximum asymptotic length and t_0 is the theoretical age at length zero. To estimate *K* and L_∞, a "forced" Gulland–Holt plot was used where the mean growth rate ($\bar{y}$) and the mean length ($\bar{v}$) were used to estimate *K* in the equation: $K = \bar{y}/(L_\infty - \bar{v})$ with the parameter L_∞ estimated using the mean of the length of 10 largest male and female *H. capensis* (King, 1995). Both *K* and L_∞ parameters were used in the SVBGF to develop a growth model for male and female *H. capensis* within Thesen Islands Marina. To ensure accuracy of the growth model, remeasurements of juveniles were needed, and a maximum of 15 juveniles were allowed to be tagged throughout the project period. Marked juveniles were, however, not included in any population abundance estimates. Seahorses marked as juveniles were sexed, where possible, during

subsequent resightings. Only five juveniles were resighted during the entire project and included in the growth curve analysis. To obtain an estimate of size at birth for *H. capensis*, newborn seahorses (*n* = 25) from a local aquarium display were measured to the nearest millimetre.

2.5 | Movement

During the initial marking period, the location of each marked seahorse was recorded by a diver towing a tethered Garmin GPS (www.garmin.com) on a surface float that was time-synchronised to a SUUNTO dive computer. The location of all subsequent recaptures during the 14-month period, during all subsequent surveys, was recorded following the same process. When an animal was resighted, the location was logged as well as the sex of the animal, the time of day as classified in Claassens and Hodgson (2018b) and the season. In addition to the monthly population surveys, 2–3 additional dives were conducted per month to record movement within the study site. Wider searches (100 m diameter search area) outside of the study site were conducted on an *ad hoc* basis. After checking for normalcy using a Shapiro–Wilk test (*P* > 0.05), the data were found to be non-normally distributed, and a generalised linear model was used to assess differences in movement between sex, season and time of day (morning, noon or afternoon).The most parsimonious
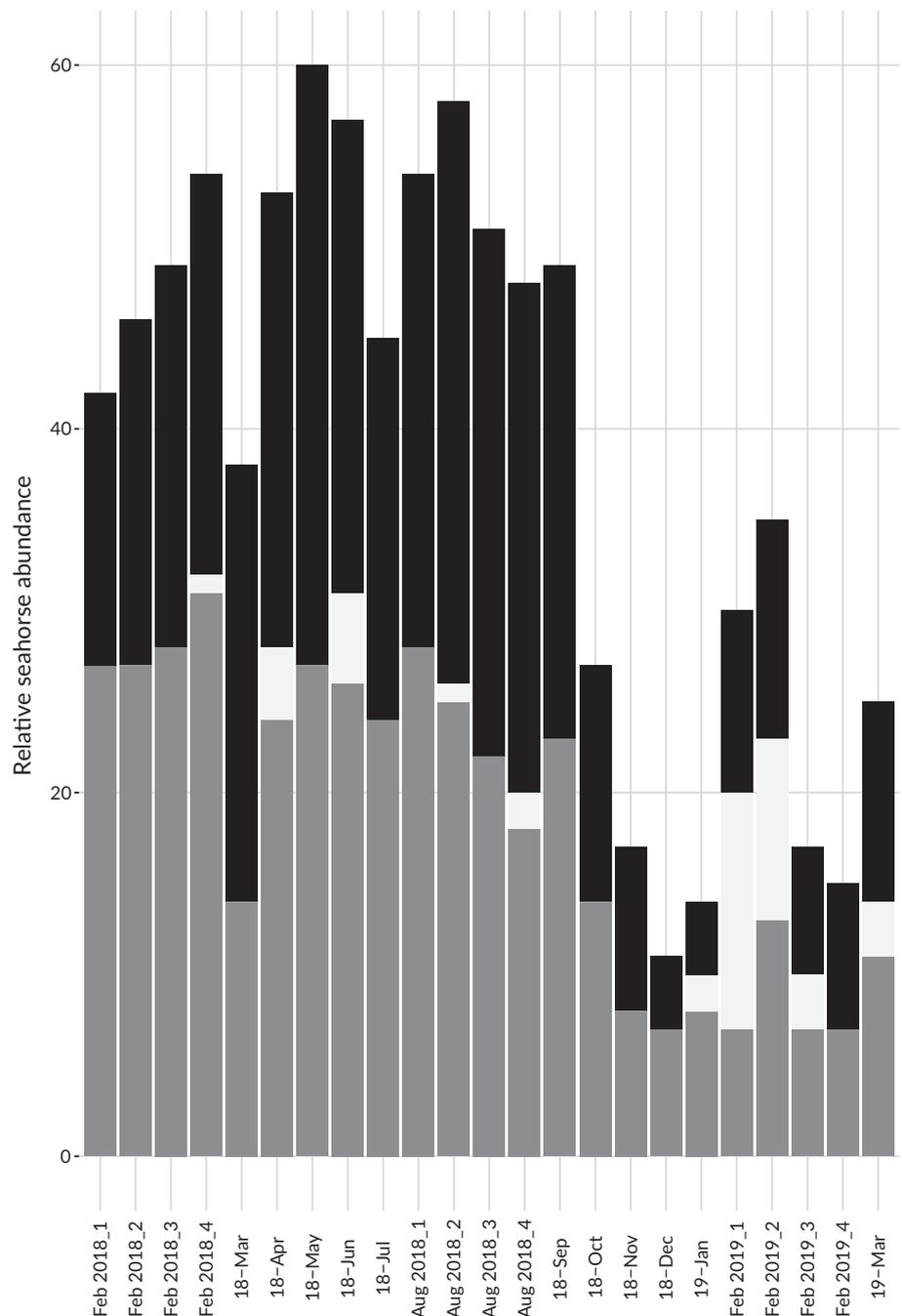
FIGURE 2 Relative abundance for *Hippocampus capensis* over a 14-month period (February 2018 to March2019) on Reno mattress habitat in Thesen Islands Marina (■) Female, (□) Juvenile, and (■) Male

model was chosen using AIC, and model results were summarised using an ANOVA.

3 | RESULTS

3.1 | Population abundance

Of the 78 animals marked on 22 February 2018, 28% of them were never seen again. The number of resightings was slightly less for females, and the maximum number of times an animal was resighted was 17 times over 388 days. During subsequent surveys after March 2019, marked animals were still noted on the site, and the longest period an animal was recorded since the initial marking

was 21 months. A maximum seahorse density of 1.2 individuals per square metre ($n = 60$) were recorded in May 2018 (Figure 2) with an overall mean (±S.D.) density of 0.26 ± 0.22 individuals per square metre recorded for the entire study period (Table 1).

From all the candidate models developed for the open population assessment, the model which included a sex effect for survivability and a time effect for resighting probability was chosen based on the AIC. Open population estimates from March 2018 to February 2019 (12-month period) showed a constant decreasing trend from 71.07 animals to 28.41 animals (Figure 3).

The estimated closed population size for the Thesen Islands Marina *H. capensis* population decreased from 134 in February 2018 to 72 in February 2019 (Table 2).

	Mean ± S.D. density	Maximum density	Minimum density
Male	0.18 ± 0.09	0.36	0.02
Female	0.18 ± 0.10	0.46	0.00
Juvenile	0.02 ± 0.04	0.20	0.02

TABLE 1 Mean relative seahorse density (individuals per square metre) recorded for *Hippocampus capensis* between February 2018 and March 2019 in Thesen Islands Marina
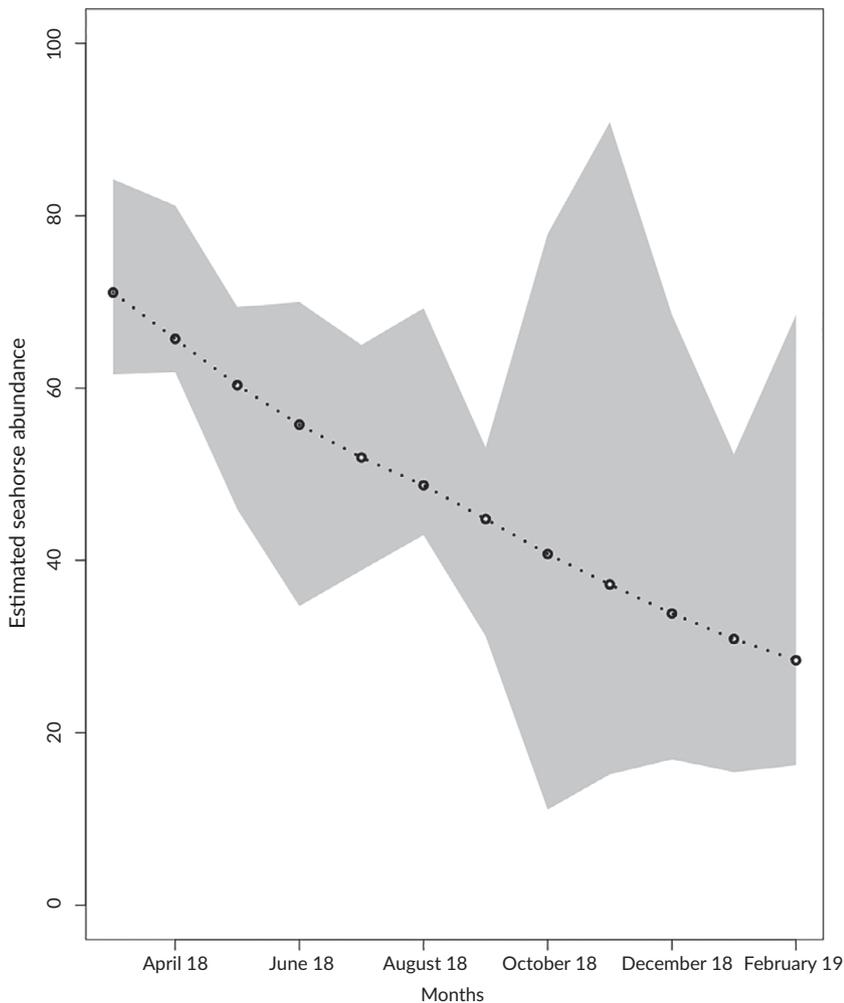
FIGURE 3 Open population estimates for *Hippocampus capensis* for a 12-month period (March 2018 to February 2019). Shaded area indicates the 95% C.I. for the estimates

3.2 | Recapture probability and apparent survival rate

For the open population assessment, monthly survival rate (ϕ) and resighting probability (p) varied across surveys from March 2018–February 2019 with the most suitable CJS model (AICC = 749.17, AICc Weights = 0.75, deviance = 290.26, parameters = 13) being constant for ϕ and time dependent for p. The survival rate estimated during the open population assessment was 0.92 ± 0.02 (± S.E.), whereas the average resighting probability varied across time as shown in Figure 4.

3.3 | Length at age

On average (S.D.), male seahorses (H_T = 8.15 ± 1.40 cm) were significantly (ANOVA, df = 1, F34.95, P = <0.01) larger than females (H_T = 7.30 ± 1.55 cm) (Figure 5), and no significant size variability was observed across seasons. The mean size of the captive newborn seahorses was 0.11 ± 0.01 cm (n = 25). The largest recorded male *H. capensis* was 10.30 cm, whereas the largest female was 11.40 cm.

L_∞ from the Gulland–Holt forced plot was 10.03 cm for males (n = 78 remeasurements) which was less than females (L_∞ = 10.23 cm; n = 103 remeasurements). Growth parameters from the VBGF model (Figure 6) indicated that the species grows rapidly in the initial 200 days with male growth rate K = 1.8 (year^{-1}) being less than the female growth rate of K = 2.2 (year^{-1}).

TABLE 2 Closed population estimates for *Hippocampus capensis* based on three primary closed population assessments in Thesen Islands Marina

Closed population assessment	Estimate ± S.E.	95% C.I. Lower	Upper
February 2018	134.38 ± 8.622	118.52	152.37
August 2018	133.21 ± 17.20	103.52	171.40
February 2019	72.17 ± 15.03	48.19	108.08

3.4 | Movement

From the 628 movement measurements collected over the 14-month period, only 6 measurements from three individuals were greater than 30 m. The furthest distance moved by a seahorse was 77.6 m. These measurements were considered to be outliers and removed from the analyses.

The average (S.D.) distance moved by the Thesen Islands Marina seahorse population was 4.45 ± 3.71 m and 4.54 ± 3.82 m by females and males, respectively. No significant differences in movement were found between sex (ANOVA, df = 1, F = 0.09, P = 0.75) or time of day (ANOVA, df = 2, F = 2.08, P = 0.12). Movement differed significantly (ANOVA, df = 3, F = 3.05, P = 0.03) across seasons with less movement found during spring (Figure 7). A significant interaction was, however, found between sex and time of day (ANOVA, df = 2, F = 3.59, P = 0.03) and season and time of day (ANOVA, df = 6, F = 3.63, P < 0.01), showing that movement not only varies across seasons, but also between sexes across the day within seasons (Figure 7).

4 | DISCUSSION

This is the first study to provide information on life history and population parameters for the endangered seahorse *H. capensis*. The scope of this research was, however, limited by the permit restrictions placed on it in terms of the total number of animals (n = 80) that were allowed to be marked as well as the number of sites (n = 1) that could be included in the study. A critical aspect that could not be investigated owing to these limitations was the recruitment of *H. capensis* over a longer time period which would have provided insights into the status of the Thesen Islands Marina population and its role as a source population for the larger estuary. In addition, the precision of the growth curve would have benefited with a higher number of re-measurements, particularly juveniles. Being limited to only marking 80 adults and 15 juveniles limited the data that could be collected as new recruits and small juveniles were unable to be marked once the permit quota was reached. Although these permit restrictions limited

FIGURE 4 Monthly recapture probabilities (p) with standard errors for *Hippocampus capensis* across a 12-month period
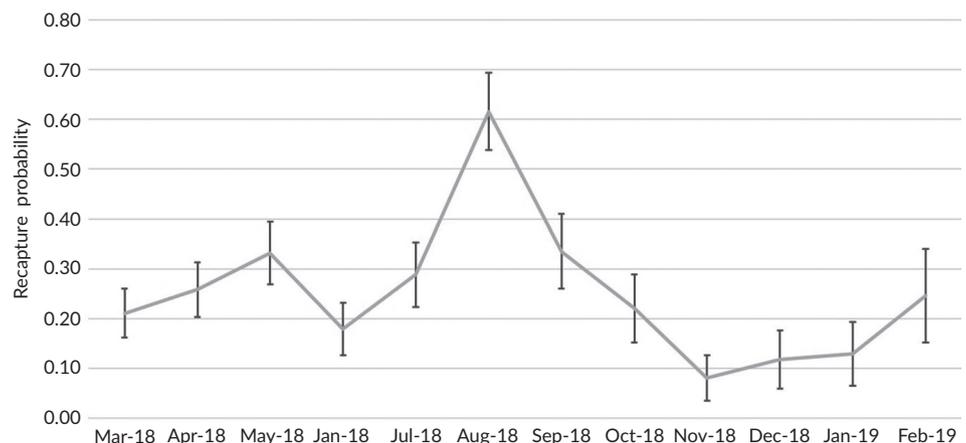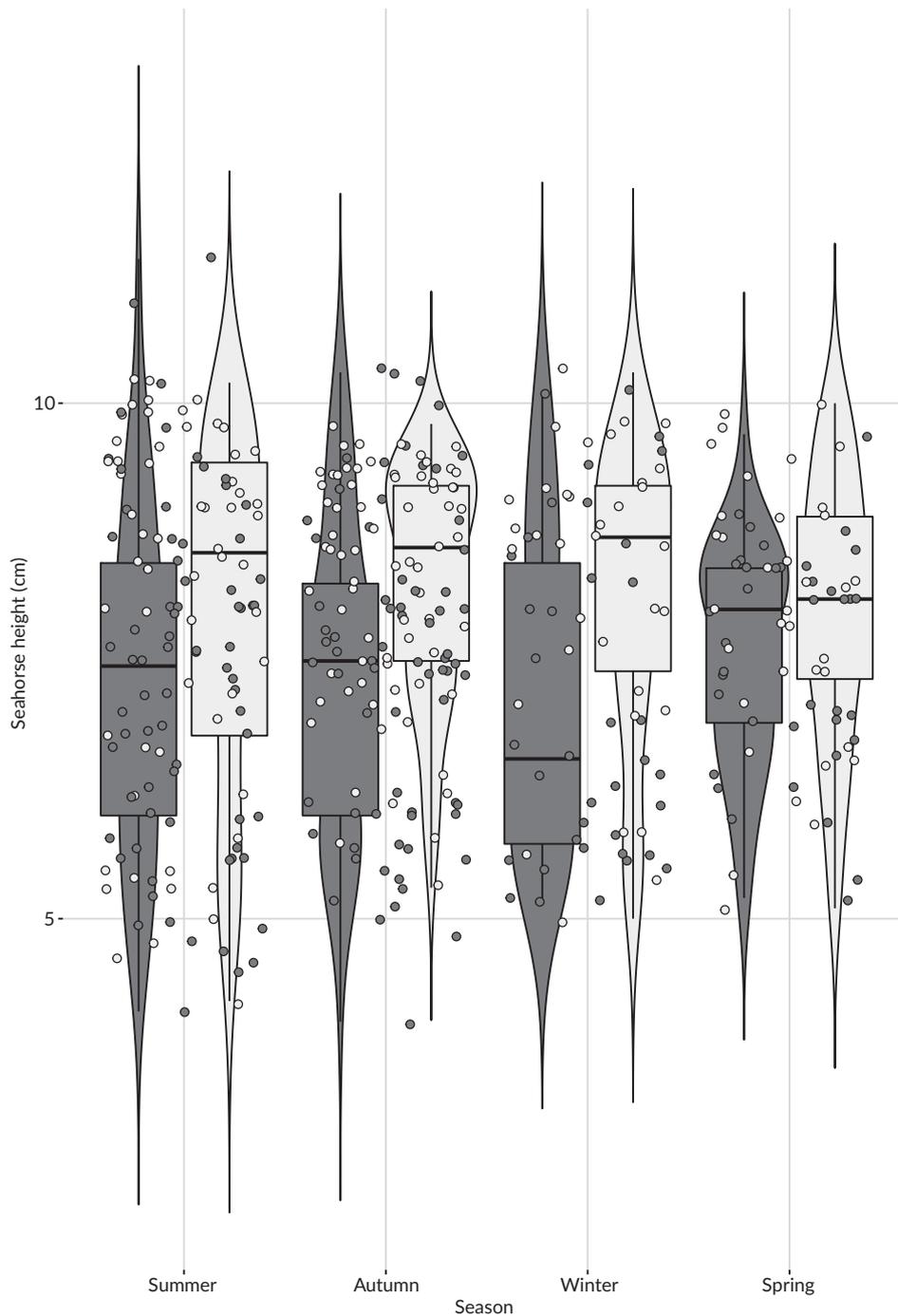
FIGURE 5 The mean (± S.D.) height for adult male and female *Hippocampus capensis* recorded in Thesen Islands Marina during 2018 and 2019 (▨) Female, and (▨) Male. (Summer = December–February; autumn = March–May; winter = June–August; spring = September–November)

what could be achieved in the current study, there were still sufficient data collected to provide life-history information that can be used for localised and site-specific conservation and management actions for this endangered species.

Several studies have shown that seahorse populations can fluctuate seasonally and temporally (Martin-Smith & Vincent, 2005; Harasti *et al*., 2014a; Correia *et al*., 2015a; Claassens & Hodgson, 2018a). In addition, different sampling approaches can significantly affect recapture rates; for example, recapture rates for *H. zosterae* increased from

0.47% to 33% by changing the sampling resolution (Masonjones *et al*., 2019). This study used closed population surveys to allow detailed population assessments through intensive surveys for one point in time and an open population model approach which allowed the assessment of changes through time on a monthly basis. The closed population estimate for *H. capensis* occurring within Thesen Islands Marina was 134 (118–152 95% C.I.) in February 2018 compared to only 72 (48–108 95% C.I.) in February 2019. The number of resighted marked animals and the estimated population abundance
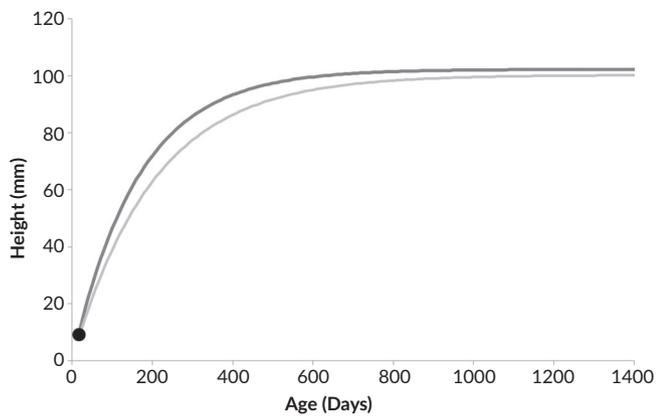
FIGURE 6 Estimated specialised von Bertalanffy growth curve for male and female *Hippocampus capensis* in the Knysna estuary. (●) = Size at birth from aquaria study (——) Female, and (——) Male

for *H. capensis* decreased across time with a marked decrease towards November 2018. In addition, the relative abundance of all seahorses on the site followed a decreasing pattern with a marked decrease from September to October 2018. It was also shown from the open population estimates that the probability of resighting marked individuals decreased over time and was particularly low for the months of November–December. Past research found a comparable decrease in *H. capensis* numbers and sightability during December 2015 (Claassens & Hodgson, 2018b) which was suggested to be owing to an increase in boat noise during the summer holiday period. As this study was focused on a site located within a residential marina estate, regular maintenance activities take place. At the study site, the wooden pylons of the adjacent jetty were replaced in September 2018 which could be a potential cause for the observed decrease in seahorse abundance in this study, as anthropogenic noise has been found to have an impact on seahorse behaviour (Anderson et al., 2011; Palma et al., 2019).

Abundance fluctuations of seahorse populations can be attributed to either natural or anthropogenic causes; for example, *Hippocampus abdominalis* Lesson, 1827, populations decreased by 79%–98% over a 4-year period in the Derwent estuary, Tasmania (Martin-Smith & Vincent, 2005) without any clear cause, and similarly *H. whitei* populations were found to show a significant decline at one location (Seahorse Gardens) in Port Stephens, NSW, Australia which then recovered within 2 years (Harasti et al., 2014a). A subsequent decline in natural habitats (soft corals and sponge habitats) as a result of anthropogenic (boat anchoring and moorings) and natural impacts (sand inundation) led to declines of over 90% of *H. whitei* populations in Port Stephens (Harasti, 2016). Past research on *H. capensis* noted population fluctuations which included a decrease in density from summer to winter observed within *Codium tenue* (Kützing) Kützing, 1856 beds (De Villiers et al., 2019) as well as within *Z. capensis* and Reno mattress habitats (Claassens & Hodgson, 2018a). Changes in population abundances could be attributed to seahorse movement owing to breeding behaviour as seahorse species have been found to congregate (Harasti et al., 2010) as well as disperse (Correia

et al., 2018; Freret-Meurer & Andreata, 2008; Masonjones et al., 2010) during the breeding season. It is unclear if these decreasing abundance patterns noted for *H. capensis* are an annual phenomenon with recruitment of new individuals taking place in late summer (January–February), or if it is owing to localised or environmental impacts. Longer-term research with continued marking of *H. capensis* and monitoring is required to determine the reasons for these observed patterns and to determine if *H. capensis* is following a decreasing population trend.

H. capensis was found to display rapid growth in its initial life stage which is similar to other seahorse species including *H. whitei* (Harasti et al., 2012) and *Hippocampus hippocampus* Linnaeus, 1758 (Curtis et al., 2017). Although this study was not able to determine time to sexual reproduction and sexual maturity for *H. capensis*, the estimated L_∞ indicates this species would be considered a medium-size seahorse and therefore may have similar times to maturity as recorded in other similar-sized species (*H. whitei*: Harasti et al., 2012, *Hippocampus guttulattus* Cuvier, 1829: Curtis & Vincent, 2006). The L_∞ estimates suggest that the females grow to a larger size than males; nonetheless, it was found from the actual measuring of individuals that the males grew significantly larger than the females. This is similar to other seahorse studies that have shown males can grow larger than females (Anderson, 2012; Rosa et al., 2007). A limitation of the developed *H. capensis* growth model was that only 78 adult seahorses were marked which made it very difficult to constantly find the same marked animals to remeasure growth over time and a lack of juveniles being observed on the site and resighted. Increasing the available pool of marked animals would increase the chances of being able to measure growth in individuals and therefore increasing the VBGF model.

The home range of most seahorse species is assumed to be small (Foster & Vincent, 2004), with estimates including ~50 m² for *Hippocampus reidi* Ginsburg, 1933 (Freret-Meurer & Andreata, 2008); ~5.5 m² for *Hippocampus comes* Cantor, 1849 (Perante et al., 2002) and between ~22 and 5 m² for *H. whitei* (Vincent et al., 2005). The estimation of home range is limited by the resighting rate of marked individuals, and of the few studies that have followed a mark-recapture approach *in situ*, resighting rates have been low; for example, only 30% of *H. guttulatus* were sighted again in the Ria Formosa (Correia et al., 2014); 24% of *H. reidi* were sighted again in Brazil (Mai & Velasco, 2012); and a mere 0.3% of *Hippocampus zosterae* Jordan & Gilbert, 1882, were sighted again in Florida (Masonjones et al., 2010). This study had a resight percentage of 72% which again emphasises the site fidelity of this species.

The distance moved by seahorses within a specific area, still following a mark-recapture approach, can, however, provide some insights into site fidelity. Studies have found that seahorses show high site fidelity (Caldwell & Vincent, 2013; Curtis & Vincent, 2006; De La Nuez Hernández et al., 2016; Harasti et al., 2014b; Moreau & Vincent, 2004; Siqueira et al., 2017). Similarly, this study found that *H. capensis* remained within a relatively small area (~70 m²) for an extended period of time, which emphasises the need to protect this specific habitat from any localised impacts. Although one individual
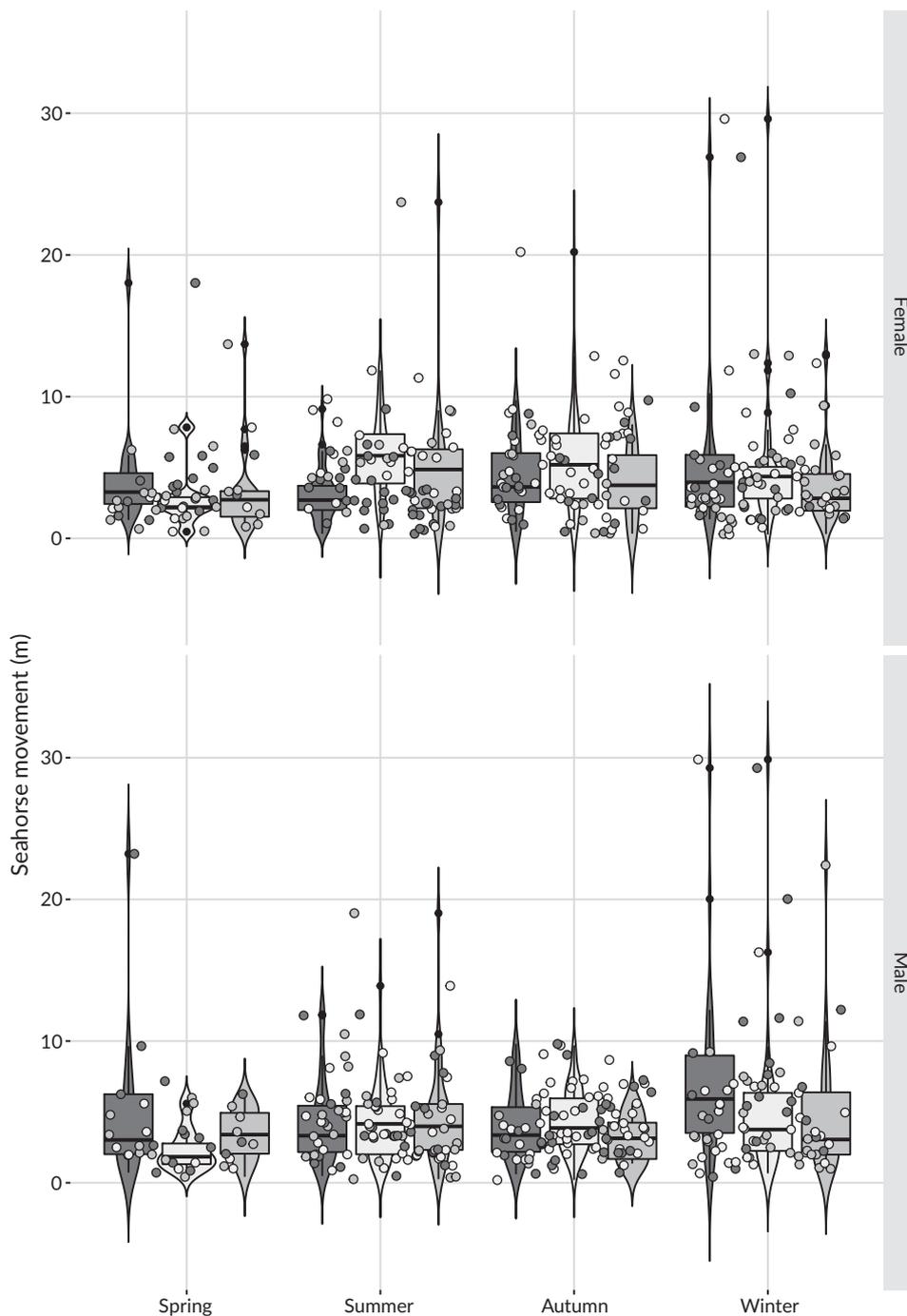
FIGURE 7 The mean (± S.D.) seasonal distances moved by adult male (M) and female (F) *Hippocampus capensis* across different daytime periods recorded in Thesen Islands Marina during 2018 and 2019 (■) Morning, (■) Noon, and (■) Afternoon. (Summer = December–February; autumn = March–May; winter = June–August; spring = September–November)

was recorded moving a distance of 70 m, the large majority of seahorses did not move far from the location of initial marking (5 m), which suggests this species displays strong site attachment. Even though movement patterns were found to be similar between males and females, females did move farther during the noon time period. In addition, seahorses were found to move shorter distances during autumn and spring, specifically in the morning. Differences in site fidelity between male and female seahorses vary across species, with *H. whitei* females found to move much greater distances than males (Harasti *et al*., 2014) and *H. comes* showing no difference in movement patterns between males and females (Perante *et al*., 2002).

Nonetheless, given the population fluctuations observed in this study there is potential that this species may migrate into other areas of the estuary. It would be beneficial to ensure that any potential harmful impacts on known seahorse aggregation sites (such as Thesen Islands Marina) are minimised and further research is warranted to assess if there are other locations outside the marina that the species may migrate to that could be subjected to anthropogenic impacts.

Artificial structures in the marine environment are known to aggregate species, resulting in higher abundances (Connell & Glasby, 1999) compared to natural habitats. Numerous studies have shown that seahorses will use artificial habitats (Correia *et al*., 2015b;

Hellyer *et al*., 2011; Otero-Ferrer *et al*., 2015), and they have also been found to selectively choose artificial habitats over natural habitats (Claassens *et al*., 2018; Simpson *et al*., 2019).The artificial Reno mattress habitat within Thesen Islands Marina has been found to be a suitable habitat for *H. capensis*, with significantly higher densities consistently found on this habitat compared to natural habitats (Claassens & Hodgson, 2018a). The maximum seahorse density recorded on the Reno mattress during this study was 1.2 individuals per square metre which is comparable to densities of *H. whitei* found on artificial swimming nets in Sydney Harbour (Harasti *et al*., 2012). Extremely high densities of *H. guttulatus* and *H. hippocampus* (maximum density of 13.1 individuals per square metre) were found on artificial structures deployed in the Ria Formosa, Portugal (Correia *et al*., 2015b).

The utilisation of this artificial habitat by an endangered species provides a novel conservation opportunity within this heavily modified environment and the habitat created by Thesen Islands Marina provides an opportunity for the application of reconciliation ecology as defined by Rosenzweig (2003). The possibility that these types of structures only act as aggregation devices (*e.g*., Baine, 2001; Grossman *et al*., 1997) with limited value in increasing biomass should, however, be considered when determining the effectiveness of such an approach. It is thus important to determine if the use of artificial structures as a conservation tool does in fact act as a source of individuals for the larger system. In areas where natural habitats of *H. capensis* have declined and there is no available Reno mattress habitats, purpose-built artificial habitats (Correia *et al*., 2015b; Simpson *et al*., 2020) could be trialled to help promote recovery of this endangered species.

Nonetheless, the priority would be to ensure that important natural habitats for *H. capensis* within the estuary, such as *Z. capensis* and *Codium tenue* (Bell *et al*., 2003; Claassens & Hodgson, 2018a; De Villiers *et al*., 2019; Lockyear *et al*., 2006; Teske *et al*., 2007), are protected and that anthropogenic disturbances are minimised.

The use of VIFE tags for marking *H. capensis* for the identification of individuals allowed the collection of important data on the growth, movement, home range and life history of *H. capensis*. These data can be used in future IUCN Red List assessments for this species and help local conservation managers in the management of this threatened species. Although the use of VIFE tags in seahorse research globally has been found to be the best approach for longer-term monitoring, using invasive techniques on an endangered species for the first time can present some risks (Bennett *et al*., 2016), and limitations on the research approach taken can be required by permitting authorities as in the case of this study.

Invasive methods do, however, provide information not always obtainable by non-invasive methods, and permit limitations can thus affect the value and application of the information generated (Heupel & Simpfendorfer, 2010). Biological species information is critical for effective conservation, and invasive research on a population has the potential to benefit a species as a whole, particularly where limited information is available (Heupel & Simpfendorfer, 2010). Owing to the dark colour of *H. capensis* and limited distinct facial markings, alternative approaches, such as the use of facial or spot markings to allow for individual identification as done in other *Hippocampus* spp. studies (Correia *et al*., 2014; Freret-Meurer *et al*., 2013), could be unsuitable. This approach has, however, not been tested in *H. capensis*, and future research to test this would be beneficial. Given that there were no observed visual effects found on *H. capensis* from the use of VIFE tags and that it allows individual seahorses to be monitored for several years (Harasti *et al*., 2012), this method of marking is considered to be suitable for any future research on *H. capensis*.

This study provided suitable evidence that a mark-resight approach using VIFE tags can be used to effectively monitor *H. capensis* populations, providing the first insights into its movement patterns and growth rates. Future research should use this approach to determine the overall population trends for this species across its range to determine the current status of this species.

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