



## Boat anchoring impacts coastal populations of the pen shell, the largest bivalve in the Mediterranean



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### ABSTRACT

The decline of important coastal habitats, like seagrass meadows, is likely to influence populations of associated species, like the noble pen shell, *Pinna nobilis*. Here we used a Bayesian formulation of individual covariate models to derive a reliable estimate of populations of *P. nobilis* in shallow, and thus usually most impacted, areas around the island of Majorca, Balearic Islands, Spain. At six evaluated sites we find quite distinct densities ranging from 1.4 to 10.0 individuals/100 m<sup>2</sup>. These differences in density could not be explained by habitat factors like shoot density and meadow cover, nor did dislodgement by storms (evaluated by maximum wind speeds at the sites) seem to play an important role. However, noble pen shell density was related to anchoring as at sites where anchoring was not permitted the average density was 7.9 individuals/100 m<sup>2</sup> while in sites where ships anchored the density was on average 1.7 individuals/100 m<sup>2</sup>. As for the conservation of *Posidonia oceanica* meadows, for the associated population of *P. nobilis* it would be of utmost importance to reduce anchoring pressure as a conservation measure for these endangered and protected bivalves.

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### 1. Introduction

Coastal marine biodiversity is expected to decrease as a consequence of the biotic and abiotic changes resulting from anthropogenic activities (Hendriks et al., 2006; Jordà et al., 2012; Vaquer-Sunyer and Duarte, 2008; Waycott et al., 2009). Yet, despite a general consensus on this scenario, the current state of many benthic populations and the factors threatening their persistence are still poorly understood (Irish and Norse, 1996; Kochin and Levin, 2003; Lawler et al., 2006). Consequences of this knowledge gap are major uncertainties concerning adequate managerial strategies to address the emerging conservation problems (Norse and Crowder, 2005) for benthic populations in littoral areas. Marine communities in coastal areas are characterized by the presence of 'ecosystem engineers', species able to modify the physical and geochemical conditions in their environment, facilitating the life of

other organisms in the community (Bouma et al., 2009; Jones et al., 1994). The reduction of ecosystem engineers is likely to create an extinction cascade difficult to evaluate (Coleman and Williams, 2002; Gutiérrez and Jones, 2008; Gutiérrez et al., 2012; Ormerod, 2003). The most important engineer species in the Mediterranean Sea are corals, bivalves and seagrasses. Seagrass, particularly *Posidonia oceanica*, meadows directly modify the nature and complexity of sediment composition and contribute to increase water clarity (Duarte, 2000; Gutiérrez et al., 2012; Hendriks et al., 2010). However, *Posidonia* meadows are declining (Marbà and Duarte, 2010; Marba et al., 2005) in parallel to mounting impacts of human activities in Mediterranean coastal ecosystems (e.g. Vaquer-Sunyer and Duarte, 2008). The decline in *Posidonia* meadows, resulting from compounded local and global effects, is so dramatic as to severely impact these ecosystems or even possibly drive them to functional extinction before the end of this century (Jordà et al., 2012). This will impact the populations of species associated with *Posidonia*, some of them of particular conservation importance.

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The species associated with *P. oceanica* which status is most compromised is arguably the noble pen shell, *Pinna nobilis*, the largest bivalve of the Mediterranean Sea. The noble pen shell is threatened by ocean acidification, habitat loss and/or direct human disturbance like recreational or commercial fishing, their use for decorative purposes, and incidental mortality by trawling and anchoring (Richardson et al., 1999; Katsanevakis, 2007a; Katsanevakis, 2009; Rabaoui et al., 2007). Typical reported values of noble pen shell densities within the Mediterranean basin are in the range of a few (1–10) individuals per 100 m<sup>2</sup> (Moreteau and Vicente, 1982; Vicente et al., 1980; Zavodnik et al., 1991). Its populations are in decline, and the species is listed as endangered and protected under the European Council Directive 92/43/EEC (EEC, 1992). The noble pen shell is particularly vulnerable to anchoring impacts, associated with the increasing use of Mediterranean coastal areas (Milazzo et al., 2004). The noble pen shell is typically associated with meadows of the seagrass *P. oceanica*. Even though there are populations of noble pen shells that are not associated to seagrass meadows (Addis et al., 2009; Katsanevakis, 2005, 2007b), this is an exception and normally populations are closely linked to seagrass habitats. Seagrass provides shelter for small animals from storms that can dislodge them (Garcia-March et al., 2007; Hendriks et al., 2011), increases food supply for filter feeders by reducing current flow and trapping particles (Hendriks et al., 2008; Peterson et al., 1984) and provides shelter from predators. Additionally, since it is illegal (EC 1626/94) to use bottom trawls, seines or similar nets above seagrass habitats in the Mediterranean, association with this habitat effectively protects against commercial fishery. In littoral areas used for recreational tourism, many meadows have been impacted by anchoring and pollution from recreational boating, which is insufficiently regulated around the islands (Procaccini et al., 2003; Sánchez-Camacho, 2003). Seagrass habitats are sensitive to damage from dragging anchors (Backhurst and Cole, 2000; Ceccherelli et al., 2007; Duarte, 2002; Walker et al., 1989). The noble pen shell has relatively fragile shells, stands upright in the seagrass meadows, and protrudes up to 70 cm above the sediments, and can be damaged directly by the anchor track. The persistence of noble pen shell populations is dependent on anthropogenic impacts, but also on habitats properties. However, the latter are poorly understood.

Our first objective was to derive an estimate for the population density of *P. nobilis* around the Balearic Islands and investigate whether habitat characteristics or physical forcing are determining the spatial differences in population density and structure. Capture-mark-recapture (CMR) models, based on multiple observations of marked individuals, can be used to estimate animal abundance (Seber, 1982; Williams et al., 2002). CMR models include a set of parameters to account for the observational process, such as detection failures (Schwarz and Anderson, 2001; Williams et al., 2002), expected to be a problem for organisms living in seagrass meadows. Traditional models rely on the hypotheses that all individuals are equally likely to be captured. If not corrected, unequal catchability leads to biased estimates of the animal abundance (Pollock et al., 1990). Hendriks et al. (2012) found that the probability of detection of noble pen shells in *Posidonia* meadows is positively associated with shell size, but that this association is similar across sites. In contrast, we expect population density and structure to vary spatially. Royle (2009) have showed how size-dependent recapture can be incorporated into models for population abundance using data augmentation techniques (Kéry and Schaub, 2012; Royle and Dorazio, 2008). Here we extended the model to stratified data and simultaneously analyzed the CMR data from five different sites. We then test whether site dependent differences were influenced by site-specific anchoring pressures or by the physical characteristic of the habitat. This information can be used to focus conservation efforts for the population of endangered bivalves in coastal areas.

## 2. Methods

### 2.1. Study area

We conducted a survey along the coastline of Majorca, Balears, Spain at six sites, Magalluf (39°30.1'N, 2°32.36'E), Cala d'Or (39°22.164'N, 3°13.887'E), Pollença (39°53.792'N, 3°05.523'E), Es Cargol (39°16.394'N, 3°2.476'E), Sta. Maria (39°9.00'N, 2°56.96') and Es Castell (39°9.12'N, 2°55.48'); Fig. 1). The six sites had a uniform depth between 5 and 6 m but contrasting anchoring pressure and physical characteristics. Magalluf is an area with important tourist development with associated pollution (Medina, 2004)

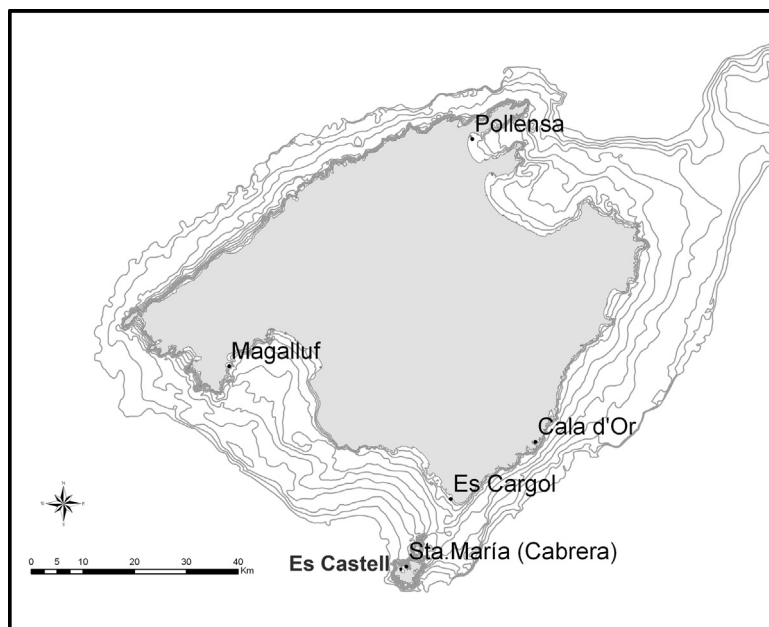


Fig. 1. Map with the locations of the sites where surveys were conducted around the island of Majorca, Balearic Islands, Spain.

**Table 1**

Site characteristics. Max wind speed is the directional component (taking into account exposure of the sites) of the wind velocity p95 (95th percentile) averaged over 2008, 2009 and 2010. Fragmentation was measured in 2009, while shoot density and coverage are measured between 2001 and 2008.

	Es Castell	St. Maria	Cala d'Or	Pollença	Magalluf	Es caragol
Anchoring	Absent	Absent	Absent	Medium	High	High
Physical forcing						
Wind (m/s)	10.5 ± 0.6	10.5 ± 0.6	10.9 ± 0.2	11.3 ± 0.3	10.5 ± 0.4	10.5 ± 0.6
Habitat						
Shoot density	569.8	865.9	384.0	612.9	494.7	1012.5
Meadow coverage	93.9	93.5	95.7	97.7	87.7	52.1

and physical disturbance of the habitat by anchoring of boats. Cala d'Or is also an important tourist destination, but at the time of the surveys, the site was closed for anchoring in summer to prevent spreading of the invasive alga *Caulerpa taxifolia*. Pollença is an extensive bay in the North of the island, which gently slopes so the surveyed area at 5 m depth is far from the typical recreational areas. Es Cargoll is located at an exposed area on a south-eastern tip of the island.

Sta. Maria is a bay in the archipelago of Cabrera where access by visitors is prohibited and anthropogenic pressures are negligible, while Es Castell is located in the same archipelago at the entrance of an enclosed bay (Es Port) harboring many boats in summer, but mooring is only allowed at supplied permanent mooring buoys so no physical damage is expected in this area (see Table 1).

## 2.2. Data collection, habitat descriptors and anchoring

Noble pen shell surveys were carried out in 2 years (2007 and 2010). In each site, we randomly positioned underwater tape of 30 m length as transect line. Each transect was randomly assigned a team of two divers, with at least one experienced diver on each team. Capture–recapture data were collected along a strip with 1.5 m width at each side of this line. Each diver marked all noble pen shells found within this strip using a metal peg inserted until level with the sediment with a discrete tag displaying a unique alphanumeric code. Once at the end of the transect line, divers switched side and searched for already marked noble pen shells marked by the previous diver ('re-capture'). The width of each marked individual was noted on a PVC bar and measured in the laboratory. On subsequent surveys, diver teams changed randomly to minimize a possible 'diver' effect on recapture probability. A total of 13 different SCUBA divers participated to the project for a total of 15 different two-diver teams. Team composition did not affect the counts (more information on the methodology in Hendriks et al. (2012)). Capture–mark sessions were organized on different days throughout the year but mostly concentrated either in summer (water temperature allows for a longer or more comfortable immersion time) or winter (shorter leaf length of the seagrass facilitates searching effort). On average 14 transects per site were surveyed in 2007 and 8 in 2010.

At each site we evaluated shoot density and seagrass coverage (as% area) of *P. oceanica* meadows as possible predictors for population structure of *P. nobilis*. Coverage was estimated from scoring habitat along a 30 m transect. Shoot density was obtained from a permanent monitoring program established on the same locations (Marbà et al., 2005; Marbà, N., unpublished data). Strong wind events may lead to dislodgement of individuals through the enhancement of waves or wind-induced currents. We used wind speed as a proxy to characterize the physical stress present on each site. Hourly wind data were obtained from the outputs of the HIRLAM model run by the Spanish meteorological agency (AEMET) at 0.05° (~5 km) resolution. In order to identify areas subject to stronger physical stresses we compute the 95th percentile of wind intensity over the years 2008, 2009, 2010. In particular we use the

wind velocity from the direction to which each particular site was exposed to get an idea of the maximal stresses.

Each site has a different anchoring regime and regulation and as consequence, data on the anchoring pressure from recreational tourism were difficult to obtain. In 2008, the number of registered recreational boaters only on the island of Mallorca was 324,522 (CITTIB, 2009). There is no legislation on the total number of boats that can access the anchoring areas around the island (Balaguer et al., 2011), but in most sites there is a delimitation of bathing areas and a no navigation area is established 200 m from the coast when a beach is present, or 50 m for other types of coast. However, many boats disregard or are unaware of this legislation and anchor in shallower restricted areas close to shore (Balaguer et al., 2011). Balaguer et al. (2011) divided the coast of Mallorca in three areas, Eastern, Central and Northern, and estimated the number of boats that navigate these waters. We used their estimates as an index of anchoring pressures for the four sites that fall into the areas considered (Cala d'Or, Es Cargoll fall into the East area, Magalluf in the Middle and Pollença in the North). The estimated number of boats would come down to 258, 200 and 584 boats per km<sup>2</sup> of seabed area (seagrass beds included) available and commonly used for anchoring in these areas for the Eastern, Central and Northern area, respectively (Table 1 in Balaguer et al., 2011). However, Cala d'Or was closed off during our survey period so now anchoring was allowed in this area. The remaining two sites are inside the National Park of Cabrera Islands where anchoring is off-limits (St. Maria) or strictly regulated (Es Castell, Table 1).

## 2.3. Model formulation and parameter estimation

Each site was considered as a closed population in which  $N$  individuals are sampled on  $J$  occasions with  $J = 2$ , leading to a sample of  $n$  unique individuals. We considered the detection constant over the occasions, and converted the encounter histories to capture frequencies of the sample of  $n$  unique individuals (Royle and Dorazio, 2008), where each individual  $i$  were captured  $y$  times (with  $y = 1$  or  $2$ ). Data were analyzed using a Bayesian formulation of individual covariate models based on parameter-expanded data augmentation technique (DA, hereafter; Liu and Wu, 1999; Royle and Dorazio, 2007). The general concept is to physically augment the observed data set with a fixed, known number, say  $M - n$ , of "all zero" encounter histories, and to analyze the augmented dataset (of size  $M$ ) with a new model. This new model is a zero-inflated version of the conventional known- $N$  model, and could be easily fitted using Markov chain Monte Carlo (MCMC) sampling (see e.g. Royle and Dorazio (2010) for further details on DA). Given the augmented dataset, we introduced a set of latent variables  $z_i$  for  $i = 1, 2, \dots, M$  which are Bernoulli trials with the parameter  $\psi$ . This parameter is the probability that an individual from the augmented data list is an element of the population. Conceptually, the population of  $N$  individuals represents a subset of some hypothetical super-population of individuals  $M$ . With the DA technique, the problem of estimating population size ( $N$ ) is converted into that of estimating inclusion probability ( $\psi$ ), since the expected value for

$N$  is equal to  $M\psi$  (Kéry and Schaub, 2012). Population size  $N$  could potentially be any integer between 0 and  $M$ , and DA just induces for  $N$  a discrete uniform prior on the interval  $(0, M)$ . By estimating the value for each  $z_i$  we can estimate noble pen shell abundance in a way that naturally excludes the structural zeros in the augmented data. If  $z_i = 0$ , then individual  $i$  from the super-population of size  $M$  does not correspond to an individual in the population exposed to sampling, whereas if  $z_i = 1$  the individual is a member of the population of size  $N$ . An estimator of the total population size is then simply derived as  $N = \sum_{i=1}^M z_i$  (Royle, 2009). We can estimate the total number of individuals in each specific site or for groups of sites sharing similar characteristics, the only difference being the indexes we use in the summation. Because the total sampled area was different from site to site we derived the density (individuals/100 m<sup>2</sup>) of *P. nobilis* by dividing  $N$  for the specific sampled area at each model iteration to obtain a credible interval of  $N$ . To model the stratified population size we augmented each group-specific dataset, and then fitted a model with group-specific inclusion probability ( $\psi_{group}$ ) to the ensemble dataset (see below). In this way we estimated the latent variable for the unobserved individuals in each specific group of interest, as well as the related shell width.

Individual covariate for unobserved individuals was estimated by assuming shell width as normally distributed, with a mean and a variance to be estimated. Then, by introducing in the observation process the relationship between shell width and detection probability, the population mean and variance of the individual covariate was corrected for the size-biased sampling. We therefore specified this dependence as  $\text{logit}(p_i) = \alpha + \beta x_i$ , with  $x_i \sim \text{Normal}(\mu_x, \sigma_x^2)$ , where  $\alpha$  parameterizes the detection probability (on the logit scale) for an average size individual while  $\beta$  parameterizes the variation in detectability in relation to shell width ( $x$ , previously centered by subtracting the mean). Given this basic model formulation, we aggregated data in different ways to address specific questions by building the corresponding models. First, we tested for a seasonal effect on detection probability, aggregating data from all sites in relation to sampling season (winter or summer) considering  $\text{logit}(p_{i,season}) = \alpha_{season} + \beta x_i$ , with  $x_i \sim \text{Normal}(\mu_x, \sigma_x^2)$ . A second model assumed a site-specific mean and variance for shell width and the basic constraint for detection probability,  $\text{logit}(p_{i,site}) = \alpha + \beta x_{i,site}$ , with  $x_{i,site} \sim \text{Normal}(\mu_{x,site}, \sigma_{x,site}^2)$ . To test the effect of environmental covariates (shoot density, fragmentation, meadow coverage and maximum wind speed) and an anthropogenic factor (presence/absence of anchoring) on noble pen shell density we assumed the probability that the  $i$ th individual is a member of the population exposed to sampling to depend on meadow parameters, wind speed, or anchoring. These effects were modeled separately, with the same model formulation. In this case, given the site-specific standardized values (by subtracting the mean and divided for the standard deviation) of each predictor, we let  $\psi_{cov}$  denote the probability that an individual from a site with a specific covariate ( $cov$ ) value is a member of the population of noble pen shells exposed to sampling (i.e. group-specific inclusion probability). Thus, the model of  $z_i$ , for the  $i$ th individual detected in a site with a covariate value  $cov_{site}$ , can be written as  $z_i|cov \sim \text{Bernoulli}(\psi_{cov})$ , with  $\text{logit}(\psi_{cov}) = \alpha_{\psi} + \beta_{\psi} cov_{site}$ . Parameter  $\beta_{\psi}$  represents the slope for the relationship between the covariate value of a specific site and the number (and/or density) of noble pen shells present in that site. Similarly, we let  $\psi_{anch}$  denote the probability that an individual, from a site with or without anchoring, is a member of the population of noble pen shells exposed to sampling. As before the model for the latent state of the  $i$ th individual in a certain site with anchoring present or not was  $z_i|anch \sim \text{Bernoulli}(\psi_{anch})$ , which provides an explicit connection between the presence of anchoring in a site and the number (and thus density) of noble pen shells pres-

ent in that site. In the model we then derived the difference in the estimated densities of individuals between sites with and without anchoring, together with its related uncertainty, as a direct measure of the effect of anchoring on noble pen shell density. In the models with the continuous predictors we assumed a site-specific population mean and standard deviation for shell width,  $x_{i,site} \sim \text{Normal}(\mu_{x,site}, \sigma_{x,site}^2)$ , while in relation to anchoring we were interested in modeling and evaluating the shell width population structure in relation to this human-related factor, as  $x_{i,anch} \sim \text{Normal}(\mu_{x,anch}, \sigma_{x,anch}^2)$ .

Posterior masses for the estimates of population size  $N$  were located well away from the upper bounds, indicating that sufficient data augmentation was used. For shell width mean parameter ( $\mu_x$ ), a normal priors with mean 0 and variance 1000 was used (replicating the analyses with a uniform prior between  $-10$  and  $10$ ), whereas for precision ( $\tau = 1/\sigma^2$ ) a gamma prior with shape and scale both equal to 0.001 was used. For the  $\alpha$  and  $\beta$  parameters we repeated the analyses using both a normal prior with mean 0 and variance 1000, and a uniform prior between  $-10$  and  $10$ . For the inclusion parameters  $\psi$  a uniform prior between 0 and 1 was used.

Summaries of the posterior distribution were calculated from three independent Markov chains initialized with random starting values, run 100,000 times after a 50,000 burn-in and re-sampling every 30 draws. For our analyses the Brooks–Gelman–Rubin convergence diagnostic (Brooks and Gelman, 1998) was less than 1.003 for all parameters, which indicate convergence. Model formulations were implemented in program WinBUGS (Lunn et al., 2000), executed from R (R Development Core Team, 2011) with the package R2WinBUGS (Sturtz et al., 2005). An R script with the WinBUGS model specification is provided as supporting information.

#### 2.4. Population structure in relation to shell width

From the marginal posterior distribution of parameter  $z_i$  we summarized noble pen shells' frequency in relation to shell width. Thus, from the super-population of latent variables  $z_i$  we extracted and tabulated data for individuals that are members of the population of  $N$  individuals exposed to sampling (those with  $z = 1$ ). We then summarized size dependent population structure for the different sampling sites and in relation to anchoring.

### 3. Results

We marked a total of 356 individuals, with an average shell width of 14.57 cm  $\pm$  0.27 SE. Average detection probability did not differ between the two sampling seasons, with widely overlapped 95% credible intervals (hereafter 95%CRI) for the two estimates ( $p_{winter} = 0.578$ , 0.502–0.653, 95%CRI;  $p_{summer} = 0.587$ , 0.522–0.652, 95%CRI). Site-specific estimates of noble pen shell density varied, on average, from 1.4 (1.2–1.6, 95%CRI) to 10.0 (9.0–11.2, 95%CRI) individuals/100 m<sup>2</sup> (Table 2 and Fig. 2). As expected, detectability was positively affected by shell width, with a 95%CRI for the slope parameter that did not encompass zero ( $\beta = 0.126$ , 0.078–0.174, 95%CRI; Table 2 and Fig. 3). Structural parameters of seagrass meadows were not related to noble pen shell density as the effect of shoot density was not relevant and the 95%CRI for the related parameter did encompass zero ( $\beta_{\psi} = -0.028$ ,  $-0.183$ – $0.126$ , 95%CRI), and noble pen shell density was not significantly affected by meadow coverage ( $\beta_{\psi} = 0.044$ ,  $-0.128$ – $0.217$ , 95%CRI). Dislodgement by storms did not seem to be an issue in our populations as no significant effect of wind speed on noble pen shell density was found ( $\beta_{\psi} = 0.061$ ,  $-0.150$  –  $0.292$ ,

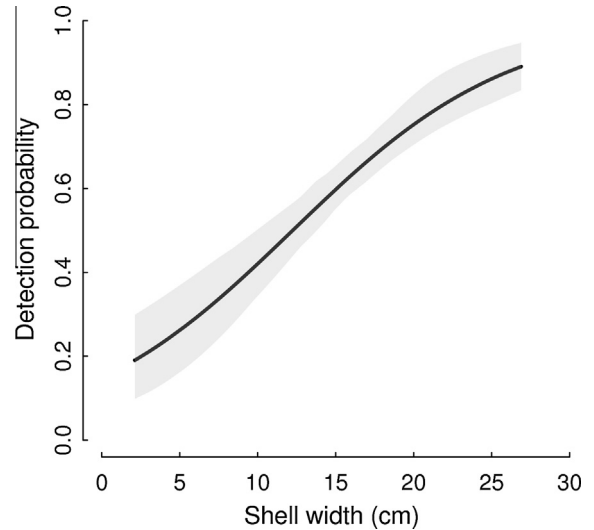


**Table 2**

Posterior summary of model parameters for data of the noble pen shell aggregated at site level. Densities were derived parameters expressed as individuals/100 m<sup>2</sup>,  $\alpha$  is the detection probability (on the logit scale) for an average size individual,  $\beta$  is the slope for the relationship between detectability and width shell,  $\psi$  is the “zero-inflation” parameter associated with data augmentation,  $\mu_{x,site}$  is the site-specific mean shell width,  $\sigma_{x,site}$  is the shell width standard deviation. Note that  $\alpha$ ,  $\beta$ , and  $\psi$  are site-independent. Posterior mean and related 95% credible interval are reported for each parameter.

Parameter	Mean	2.5%	97.5%
Density Cala d’Or	4.8	4.3	5.5
Density Es Cargol	1.4	1.2	1.6
Density Es Castell	10.0	9.0	11.2
Density Magalluf	1.8	1.4	2.2
Density Pollença	1.7	1.4	2.2
Density St. Maria	8.8	8.0	9.9
$\alpha$	0.350	0.123	0.574
$\beta$	0.126	0.078	0.174
$\psi$	0.315	0.279	0.356
$\mu_{x,Cala\ d'Or}$	12.653	11.618	13.605
$\mu_{x,Es\ Cargol}$	8.116	6.691	9.394
$\mu_{x,Es\ Castell}$	17.090	16.033	18.058
$\mu_{x,Es\ Magalluf}$	11.688	8.894	14.129
$\mu_{x,Pollença}$	9.891	4.595	14.172
$\mu_{x,St.\ Maria}$	14.237	13.239	15.126
$\sigma_{x,Cala\ d'Or}$	3.866	3.247	4.629
$\sigma_{x,Es\ Cargol}$	3.139	2.343	4.388
$\sigma_{x,Es\ Castell}$	4.307	3.686	5.083
$\sigma_{x,Es\ Magalluf}$	5.441	3.878	7.984
$\sigma_{x,Pollença}$	9.770	6.965	13.880
$\sigma_{x,St.\ Maria}$	4.668	4.090	5.376

95%CRI). In contrast, average noble pen shell density was different in relation to anchoring, with 7.9 (7.1–8.9, 95%CRI) individuals/100 m<sup>2</sup> in sites without anchoring pressure and 1.7 individuals/100 m<sup>2</sup> (1.4–2.1, 95%CRI) in sites where anchoring was permitted (Table 3 and Fig. 4). Site-specific population structure in relation to shell width showed certain variability in both mean and standard deviation (Table 2 and Fig. 5). Average shell width varies from 8.12 cm (6.69–9.39, 95%CRI) for Es Cargol to 17.09 cm (16.03–18.06, 95%CRI) for Es Castell. Average shell width standard deviation was smaller in Es Cargol (3.14 cm, 2.34–4.39, 95%CRI) and wider in Pollença (9.77 cm, 6.97–13.88, 95%CRI; Table 2). Size-dependent population structure showed differences also in relation to anchoring (Fig. 6). Mean estimated shell width was higher in sites without anchoring (14.44 cm, 13.61–15.19, 95%CRI) with no

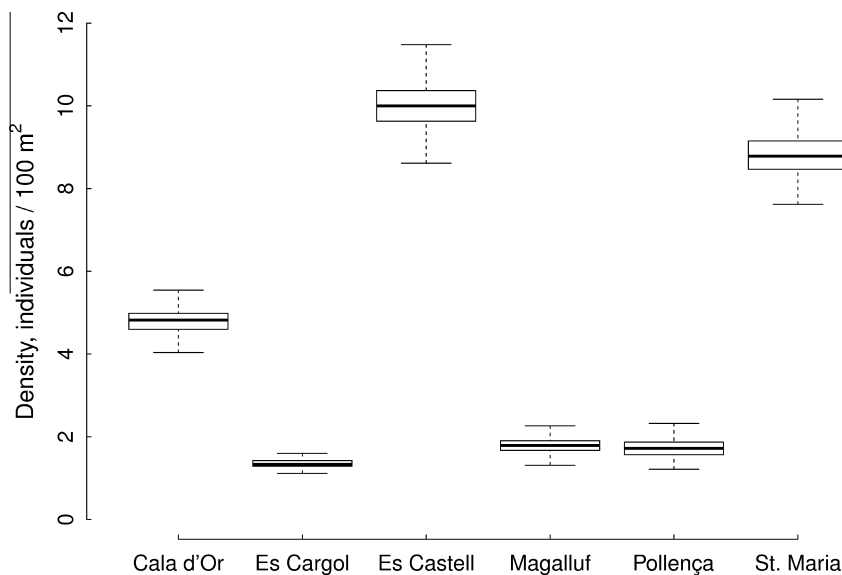


**Fig. 3.** Relationship between the detection probability ( $p$ ) of noble pen shells and the individual covariate, shell width. The shaded area represents 95%CRI.

overlapping credible intervals between the two estimates as mean size in presence of anchoring was 9.362 cm, 6.667–11.468 95%CRI; Table 3). The estimate for shell width standard deviation was distinctly lower in locations without anchoring (4.76 cm, 4.33–5.26, 95%CRI) than in those with this physical stressor present (6.91 cm, 5.59–8.78, 95%CRI; Fig. 6).

**4. Discussion**

We used a Bayesian formulation of individual covariate models to investigate the difference in structure and abundance of the noble pen shell populations in six coastal sites of the archipelago of Balearic Islands (Spain). The technique of data augmentation allowed us to derive reliable estimates for number of individuals as well as for the population structure, even if there was an effect of shell width on the detectability of the individuals. Processing the data according normal capture-mark protocols, i.e. without individual heterogeneity in detection probability, would have resulted in an under estimation of small individuals while over estimating the percentage of large animals in the population (Hendriks



**Fig. 2.** Site-specific densities (individuals/100 m<sup>2</sup>) of noble pen shell populations around the island of Majorca, Balearic Islands, Spain.

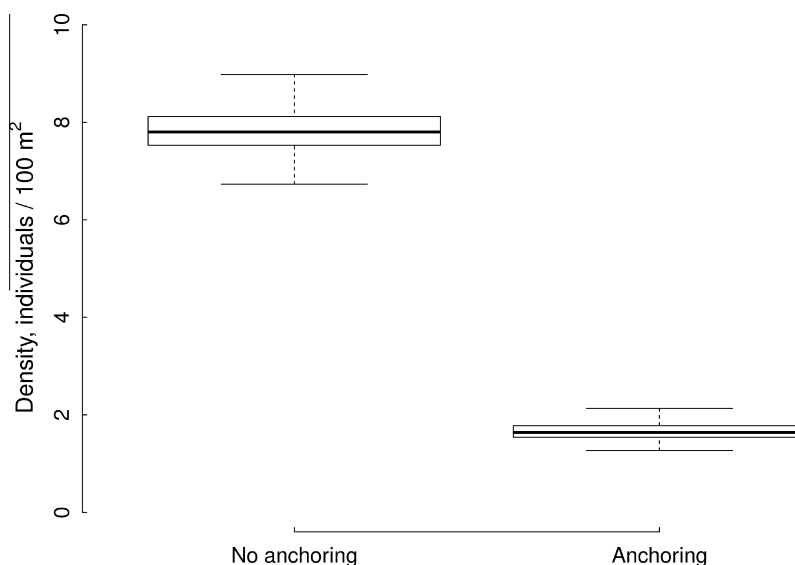
**Table 3**

Posterior summary of model parameters; noble pen shell in relation to anchoring. Densities were derived parameters expressed as individuals/100 m<sup>2</sup>,  $\alpha$  is the detection probability (on the logit scale) for an average size individual,  $\beta$  is the slope for the relationship between detectability and shell width,  $\psi_{no\ anchoring/anchoring}$  are the "zero-inflation" parameters associated with data augmentation and specific for locations without or with anchoring,  $\mu_{x,no\ anchoring/anchoring}$  are the site-specific means for shell width,  $\sigma_{x,no\ anchoring/anchoring}$  are the shell width standard deviations. Note that  $\alpha$  and  $\beta$  are anchoring-independent. Posterior mean and related 95% credible interval are reported for each parameter.

Parameter	Mean	2.5%	97.5%
Density with anchoring	7.9	7.1	8.9
Density without anchoring	1.7	1.4	2.1
Density difference	6.2	5.5	7.1
$\alpha$	0.335	0.103	0.563
$\beta$	0.143	0.096	0.195
$\psi_{no\ anchoring}$	0.329	0.287	0.382
$\psi_{anchoring}$	0.356	0.271	0.469
$\mu_{x,no\ anchoring}$	14.440	13.608	15.187
$\mu_{x,anchoring}$	9.362	6.667	11.468
$\sigma_{x,no\ anchoring}$	4.761	4.327	5.259
$\sigma_{x,anchoring}$	6.905	5.587	8.779

et al., 2012). We found that population density and the number of large noble pen shells were less in sites with anchoring of recreational boats. Site-specific differences in population size and structure however can be attributed to many factors, including food availability.

The close association of the noble pen shell with its seagrass habitat and the facilitation seagrass provides in terms of mechanical shelter, food increase and shelter against predation leads to believe that meadow structure would affect the (sustainable) number of bivalves living within its boundaries. However, we did not find a significant effect of simple variables like shoot density or spatial cover on population density or size structure of *P. nobilis*. It is possible that the processes that structure the meadows of *P. oceanica* act on different time scales from those that structure the populations of the noble pen shell. *Posidonia* meadows can be very old and grow very slowly while recruitment and development of the population of noble pen shells would take place over time scales like 10 to maximum 30 years. On the other hand, despite the general decline in shoot density observed in the region (Marbà et al., 2005; Marbà and Duarte, 2010), current shoot densities may still be above a critical threshold to directly effect noble pen shell populations already established.



**Fig. 4.** Densities (individuals 100/m<sup>2</sup>) of the noble pen shell around the island of Majorca (Balearic Islands, Spain), in relation to presence/absence of anchoring.

We used wind speed as a proxy for storminess and wave action since we did not encounter suitable data on waves for all our sites. Climate models project less storms over the Mediterranean basin for the end of the century (Giorgi, 2008; Marcos et al., 2011). This would reflect on lower wave heights (Jordà et al. 2012; Medina, 2004), which will even decrease the pressure of dislodgement caused by wave action. We thus believe that natural physical forcing dislodging individuals during storms with high wave action is not a likely factor in structuring the populations of the noble pen shell around the Balearic archipelago, not now, nor in the near future.

Even though we were unable to test for other influences like pollution, we do not expect this to be the major determinant for noble pen shell population size as we found high densities in Cala d'Or (no anchoring), a site probably equally impacted as Magalluf or St. Elm (Anchoring) due to the proximity of a harbor and tourist accommodation. Incidental fishing is unlikely because of the association with the *Posidonia* meadow. The meadow would cover the animals and hide them from swimmers on the surface. *P. oceanica* canopy reach heights of over 1 m in summer, when recreational boating peaks and anchoring is likely. Diving clubs do not target the shallow *P. oceanica* meadows so illegal poaching by divers is very unlikely. Targeted poaching did occur historically but nowadays residents and tourists alike are relatively well informed about the protective status of the noble pen shell and the low population density also discourages this activity. Additionally, since it is illegal (EC 1626/94) to use bottom trawls, seines or similar nets above seagrass habitats in the Mediterranean, association with this habitat effectively protects against commercial fishery. Based on our results the major determinant of shallow populations of *P. nobilis* appears to be anchoring. Average noble pen shell density was different in relation to anchoring, with a difference of 6.2 (5.5–7.1, 95%CRI) individuals/100 m<sup>2</sup> between sites with anchoring compared to sites without this pressure. In 2008, the number of recreational boaters on the island of Mallorca was 324,522 (CITTIB, 2009). This cause of structural damage has far more effect than dislodgement by storms or habitat quality.

## 5. Conclusions and recommendations

Physical dislodgement by anchoring causes fast and unpredictable mortality on larger noble pen shells. Selective mortality of large individuals might have important consequences for the fu-

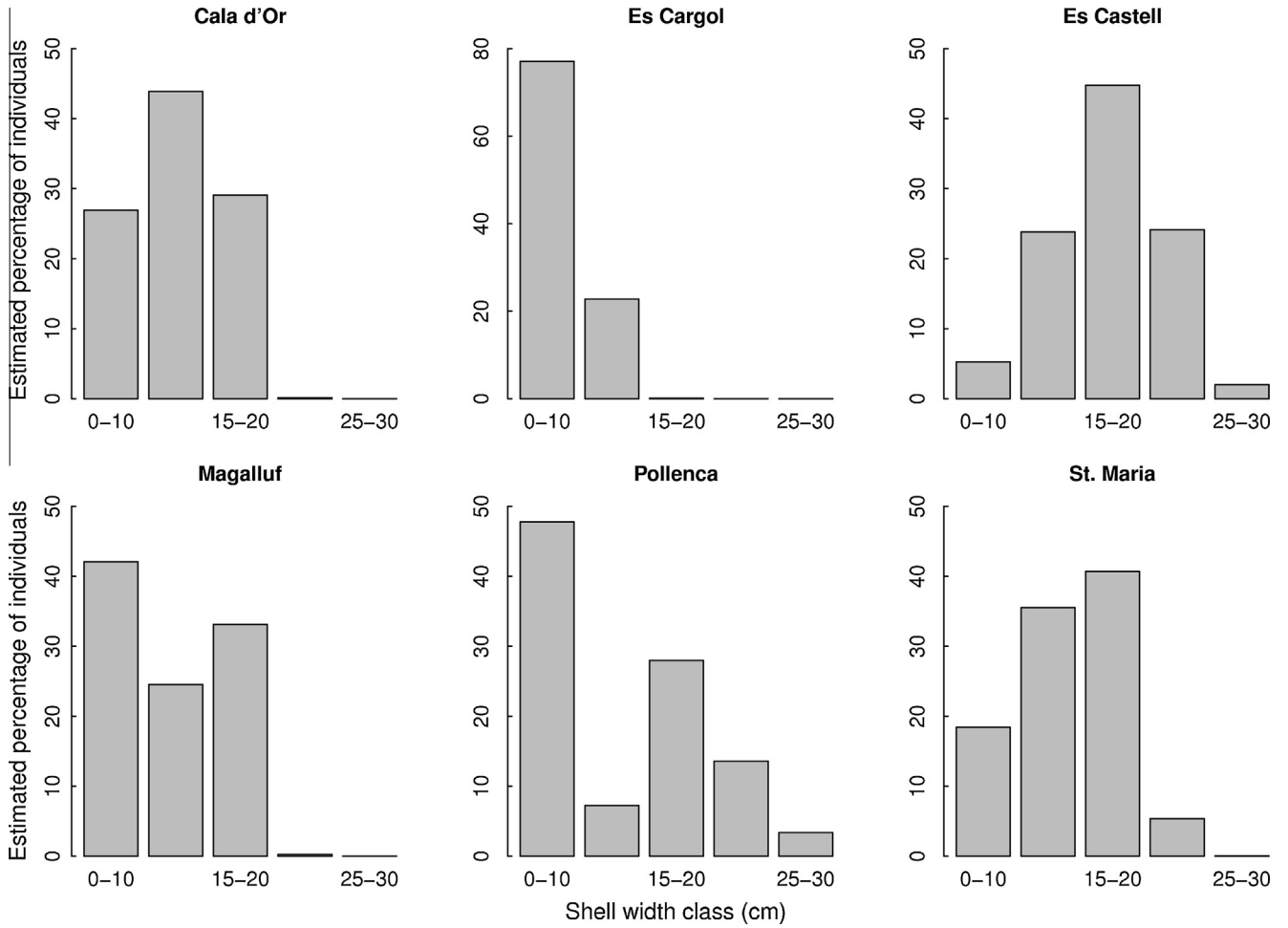


Fig. 5. Size (shell width) dependent population structure of the noble pen shell for each sampling site, around the island of Majorca (Balearic Islands, Spain). Estimated proportions of individuals for different dimensional classes are reported. Note the different y-axis scale for Es Cargol.

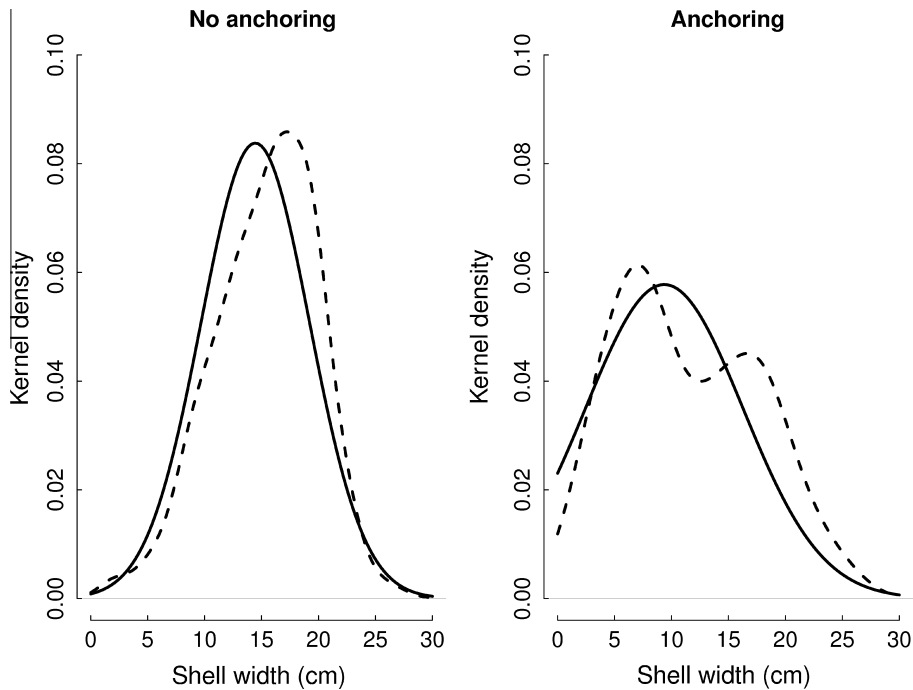


Fig. 6. Kernel density estimates for the size of noble pen shells (shell width) in relation to presence/absence of anchoring, dashed lines represent estimates for sampled individuals and solid lines for population estimates.

ture of the population (Coltman et al., 2003). Since larger individuals probably produce more eggs this could have direct implications for the amount of potential recruits and restocking of depleted areas. Also, a lower density of broadcast spawners renders fertilization in the water column less effective. Conservation efforts for the noble pen shell, *P. nobilis*, should prioritize the installation of permanent mooring buoys to decrease physical damage to seagrass meadows and the associated noble pen shell population.

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## Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.biocon.2013.01.012>. These data include Google maps of the most important areas described in this article.

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