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Detecting trends in seagrass cover through aerial imagery interpretation: Historical dynamics of a *Posidonia oceanica* meadow subjected to anthropogenic disturbance

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A R T I C L E I N F O

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A B S T R A C T

Over the last century, *P. oceanica* meadows have undergone significant regression along Mediterranean coasts due to anthropogenic disturbances. Using an integrated approach based on historical aerial photography interpretation in GIS environment and Bayesian modelling, we mapped the upper limits of a *P. oceanica* meadow and identified the most influential disturbance factors acting along the coast of Giglio Island, where several local human-mediated impacts have historically co-occurred from 1968 to 2013. Model selection based on the DIC criterion suggested that the presence of the impacts is suitable for describing the seagrass coverage variation. Similarities in *P. oceanica* cover within 13 investigated zones were highlighted when considering the most relevant impacts, such as harbour expansion, mining, and anchoring. The detected adverse effects indicate the need for implementing management actions focusing on the present and past sources of impact to reduce their effect on *P. oceanica* beds actively.

1. Introduction

Coastal zones constitute transitional ecosystems between the land and sea, representing one of the most dynamic natural environments and essential contexts in which human activity, economy, ecology and geomorphology interact (Blanco-Murillo et al., 2022; Fabbri, 1998). Coastal areas offer a variety of habitats and provide important and valuable goods and services to humankind (Barbier et al., 2011; Costanza et al., 1997; Cullen-Unsworth and Unsworth, 2013; Reid, 2005). Since time immemorial, the coastal zone has been a centre of human activity because of its high biological productivity and easy accessibility, resulting in the world's most populated area (Primavera, 2006). For this reason, coastal zones face some significant environmental issues such as i) urban sprawl, ii) pollution of estuarine and coastal waters, iii) marine resources exploitation, iv) coastal hazards and risks (Fabbri, 1998).

Seagrasses are a mixed group of flowering plants living in shallow coastal marine and estuarine environments worldwide, thriving both on soft and rocky bottoms and capable of forming extensive meadows (Green et al., 2003). Seagrass meadows are among the most productive and threatened habitats on Earth (Orth et al., 2006; Waycott et al., 2009). ***Posidonia oceanica*** (L.) Delile, 1813 is the most important and widespread endemic seagrass species in the Mediterranean Sea, capable of developing large meadows from the sea surface level up to 40–45 m depth (Dennison, 1987; Duarte, 1991). It forms one of the most valuable coastal ecosystems on Earth in terms of goods and services for its ecological, physical, economic and bio-indicator role (Boudouresque et al., 2012; Salomidi et al., 2012; Vassallo et al., 2013).

Due to its wide distribution and unique features, *P. oceanica* is protected by EU legislation and local measures both at species and habitat levels. Despite the legal framework, *P. oceanica* meadows are rapidly declining during the last century mainly due to human activities, climate changes and alien species invasion (Casoli et al., 2021; Jordà et al., 2012; Mancini et al., 2019; Marba et al., 2014; Montefalcone et al., 2007; Ruiz and Romero, 2003; Telesca et al., 2015). Although a

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widespread regression of *P. oceanica* meadows is reported at the basin level, the magnitude of this decline derived from anthropogenic pressures may vary depending on the considered geographic location and mapping technique (Bonacorsi et al., 2013). To counter and reverse this habitat loss, identifying local disturbances affecting the seagrass is a fundamental step for developing specific protection measures aimed at mitigating and regulating these pressures (Boudouresque et al., 2009). Effective coastal zone management plans and conservation efforts on *P. oceanica* could benefit from improved knowledge about both seagrass spatial distribution and human-mediated sources of impact.

Marine spatial planning and integrated coastal zone management supported by accurate mapping are pivotal in promoting sustainable growth of maritime and coastal activities and using coastal and marine resources sustainably, as also recently highlighted by the European Commission (Schaefer and Barale, 2011). Mapping seagrass spatial distribution can be carried out using a range of approaches mainly divided into direct and indirect methods. The technique is chosen according to the activity's aim, the region's size or locality to be mapped and site-specific features (McKenzie et al., 2001). The former approaches are represented by in situ observations, i.e., visual methods accomplished by scuba operators, remotely operated vehicles (ROV), and real-time towed video cameras. The indirect methods are represented by satellite imagery, such as Landsat and ESA, aerial photography, and unmanned aerial vehicles (UAV) imagery.

Aerial photography can be conducted at various scales and in various formats (e.g., colour, black and white, infra-red) and has become the most common source for seagrass mapping studies (McKenzie et al., 2001). Historical aerial photographs have been used as a helpful tool for studies on the dynamics of *P. oceanica* beds (Bonacorsi et al., 2013; Leriche et al., 2006; Maccarrone, 2010; Meehan et al., 2005; Pasqualini et al., 2014; Pasqualini et al., 2001; Pasqualini et al., 1999; Pasqualini et al., 1998). Traditionally, *P. oceanica* data are derived from aerial photographs by manually delineating the seagrass boundaries, including the upper and, according to the plant depth range, the lower edge. It has been demonstrated that coastal geomorphologic features, hydrodynamism and chemical-physical features of the column water, together with human activities, determine the *P. oceanica* occurrence and spatial extension of the meadow edges (Infantes et al., 2009; Maccarrone, 2010; Montefalcone et al., 2018; Vacchi et al., 2010).

The actual *P. oceanica* meadows distribution and health status can be interpreted only by investigating the seagrass historical dynamic. For this reason, historical *P. oceanica* distribution maps represent an important tool to provide a reliable basis for inferring changes over time in the surface area occupied by the seagrass; furthermore, some regression cases are questionable without a reliable baseline (Boudouresque et al., 2009). Nevertheless, historical maps are largely lacking or have low accuracy on most coastlines of the Mediterranean Sea (less than 42% of the coastlines present historical maps); therefore, there is an urgent need to adopt a set of standardised and efficient indicators and a robust comparative baseline to accurately assess the losses and possible gains at the Mediterranean scale (Telesca et al., 2015). To this end, with the actual distribution charts, historical maps need to be used to map and quantify the surface area covered by the seagrass and determine its dynamic over time.

Hence, the following paper aims to determine the dynamic of the *P. oceanica* upper limit surrounding Giglio Island over 45 years through aerial photography interpretation and assess the effects of local disturbances on the seagrass cover.

2. Materials and methods

2.1. Study area

The study area is represented by the Island of Giglio (central Tyrrhenian Sea, Italy), one of the seven main islands composing the Tuscany Archipelago National Park (TANP). The TANP was designated in July 1996 as a coastal marine protected area by the National Government to preserve its marine biodiversity and shelter the coastal areas from increasing human pressures. Giglio Island is the second island of the archipelago in terms of size (21.2 km^2) and represents a natural, historical, and cultural heritage. The underwater environment of Giglio Island is characterised by the presence of a vast and almost continuous *P. oceanica* meadow growing on matte, sand, and rock from a few centimetres below the sea surface up to 37 m depth. The meadow runs all over the island except for the west-south quadrant, characterised by vertical cliffs and steep bottoms, a harsh environment for *P. oceanica* thriving. The upper and lower edges, i.e., respectively, the landward and seaward boundaries defining the meadow, are localised at different depths and distances from the coastline due to both natural (seabed slope, hydrodynamic forces, photosynthetically active radiation) and anthropogenic factors (Infantes et al., 2009; Montefalcone et al., 2010; Pace et al., 2017; Vacchi et al., 2012, 2010).

2.2. Historical aerial photographs

2.2.1. Zones identification and historical aerial photographs retrieval

The study area was divided into 13 different zones, identified along the coast of Giglio Island (Fig. 1) extending from approximately −1 m up to −18 m. Two main factors were adopted to determine zones' selection: i) the presence of *P. oceanica* with an upper limit extending up to the coastline derived by recent satellite imagery assessment, and ii) the availability of historical images to enable comparisons of common areas between 1968, 1976, 1987, 1994, 2005 and 2013. Therefore, almost every ten years (according to aerial orthophotos availability), historical aerial photographs of Giglio Island were sourced from archives of the GEOscopio web portal (GEOscopio), maintained by the Tuscany Region (Territorial and Environmental Information System, SITA). The database included high-quality digital scans (300 DPI) of orthorectified (Datum: WGS-84) images deriving from low-altitude flights (comprised between 1000 and 5000 m) carried out by the Italian Body of Aerial Photogrammetric Surveys (EIRA) and the Military Geographic Institute (IGM) for several regional and national mapping projects (Orbetello, Volo Alto, Isole RT, AGEA). Photogrammetric colour cameras acquired the aerial photographs (e.g., Zeiss RMKA 23, Zeiss RMK TOP 15) after 1994 and black and white cameras (e.g., Wild AG 9, Zeiss 119013) before that period, with a ground resolution comprised between 1.5 m/pixel and 40 cm/pixel, respectively. Aerial photographs affected mainly by waves, turbidity, sun glare and/or at scales greater than 1:5000 were rejected as unsuitable for the fine-scale identification of seagrass meadow limits.

2.2.2. Historical aerial images alignment and georeferencing

Subsequently, the downloaded historical aerial images were aligned and rectified in a GIS environment (ESRI ArcMap 10.6) to a recent (2014) high spatial resolution (50 cm/pixel) RGB WordView-2 satellite image. At least ten control points identified over natural (rocks) and man-made features (road intersections, buildings) were selected to perform a 2nd order polynomial transformation and a cubic convolution interpolation procedure. Since the older photographs had fewer features in common with the more recent aerial photographs, the aerial photographs were georeferenced using control points from the most recent to the oldest. Finally, the control points were used to check transformation accuracy. The total error was computed using the root mean square (RMS) of all the residuals to compute the RMS error. This value described how consistent the transformation was between the different control points. This error is a good assessment of the reliability of image transformations (Vericat et al., 2009). The forward residual that expressed error in the same units as the data frame spatial reference showed a mean rectification error that varied from 1.3 ± 1.95 m for 1968 to 0.59 ± 0.75 m for the 1965 aerial images. The maximum rectification error reported for older images varied from 3.8 m for the 1968 images to 1.7 m for the 2013 images. When considering the inverse

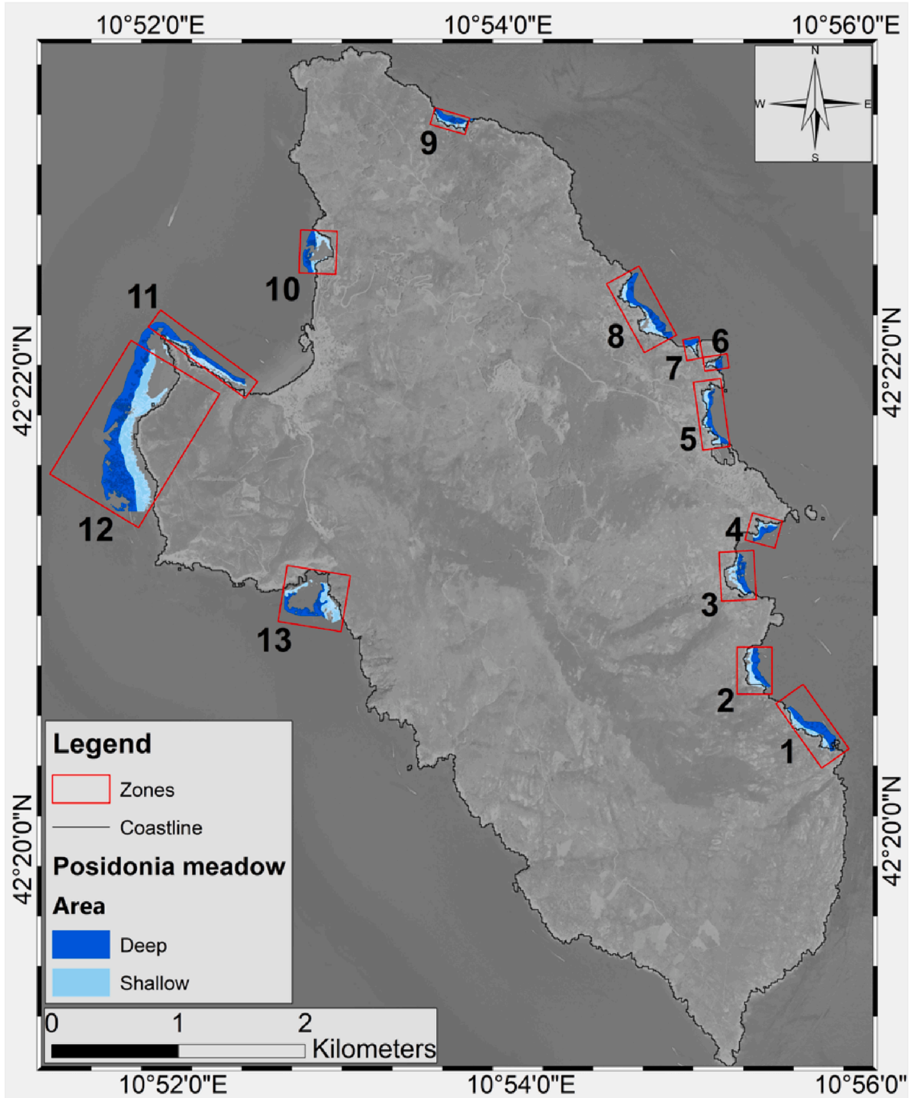

Fig. 1. Study area representing Giglio Island. The 13*P. oceanica* upper limits investigated are identified by the red rectangles and the black numbers. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

residual that showed the error in the pixels units a mean accuracy of rectification of 0.74 ± 0.03 pixel was achived, which is well under the conventional requirements of less than 1 pixel (Rozenstein and Karnieli, 2011), making these spatial errors acceptable for the present study of historical seagrass change. After rectification, the pixel size of each of the images was resampled using a bilinear technique to produce a consistent pixel size of 1 m × 1 m (Kendrick et al., 2000). The successful geo-registration allows comparison between historical aerial images.

2.3. Imagery interpretation and Posidonia meadow polygons delineation

Seagrass cover was manually outlined (Fig. 1 Supp. Mat.) with an accuracy of approximately 1 m (scale 1:100) using the "freehand tool" available in the advanced editing toolbar of ArcMap 10.6 (Esri, 2011). The same trained operator carried out photo interpretation and manual digitisation of polygons to reduce sampling biases. 1 m-bathymetric contours data was derived from a geophysical survey performed in 2012 during Costa Concordia removal project, using a hull-mounted RESON Seabat 7125 SV2 Multi Beam Echo Sounder (MBES). Using the depth contour of 5 m, the bathymetric range of the identified meadows in the 13 zones was divided into two main classes: "Shallow" and "Deep". The "Shallow" depth class was extended from 0 to 5 m depth, whereas the "Deep" one from 5 to 10 m. Even though the study area was characterised by clear waters, implying that the meadows' lower limits are extended more than 30 m depth, we limited our analysis to the upper limits up to 10 m because the historical imagery's low spectral and spatial resolution did not allow an accurate detection of lower limits. Due to historical imagery quality, only two cover classes were identified: "seagrass" and "other habitats" encompassing *P. oceanica* meadow and sandy/rocky/dead matte areas, respectively. Finally, the meadow surface area was automatically calculated using the Calculate area tool, included in the Spatial statistic toolbox.

2.4. Anthropogenic disturbances

Despite the normative framework protecting *P. oceanica* and including the island within the TANP, no active protection is undergone on the meadow all over the island. For this reason, *P. oceanica* has been directly and indirectly threatened by several anthropogenic disturbances factors mainly represented by: i) anchoring, ii) land-use change (LUC), iii) mining and iv) harbour (Table 1, Fig. 2). Once any sources of impact were detected, these areas were manually outlined by enclosed polygons to estimate their extension (Table 1) and their centroids were used to measure the distance from the nearest edge of the meadows with

Table 1
Main disturbance factors impacting *Posidonia oceanica* cover detected by aerial photo interpretation from 1968 to 2013.

Impact type	Description	Photointerpretation	Effects on seagrass	Year	Zones	Area interested by impact (m^2)
Anchoring	Anchoring (effects of anchor and chain of leisure and professional boats)	Presence of leisure and professional boats, sheltered bays by winds	Mechanic damage by anchor and chain, clods detachment, holes and canal formation increasing erosion by currents (Collins et al., 2010; Francour et al., 1999; Pergent-Martini et al., 2022)	1968	8	14,850
				1976	4	2571
				1987	1–4	30,600
				1994	6, 7, 9, 13	17,174
				2005	1–4; 6, 7, 9, 10, 11, 13	25,145
				2013	1–4; 6, 7, 9, 10, 11, 13	34,132
				1968	8	10,589
Land-use change (LUC)	Land-use change (roads and buildings construction, agrarian activities)	Detection of new anthropogenic structures (roads and buildings), agrarian activities (terracing), transplanted pinewood	Change in sedimentary fluxes from land to sea, increase in water turbidity, and increase in rainfall-runoff events (Arnáez et al., 2017; López-Merino et al., 2017; Ruiz and Romero, 2003; Saunders et al., 2017)	1976	1, 3, 8, 10	50,797
				1987	4, 5, 9	12,920
				2005	4	156
Mining	Mining (extraction activities along coastal mines)	Detection of mines and associated structures for mineral extraction	Change in sedimentary fluxes from land to sea, metal accumulation (Marín-Guirao et al., 2005)	1968	11, 13	7290
Harbour	Harbour (Enlargement of the harbour)	Harbour enlargement and detection of changes in breakwaters' length	Increase in water turbidity, change in sedimentary fluxes (Holon et al., 2015; Roca et al., 2014)	1976	5	3183
				1994	5	1053
				2013	5	1888

the "proximity toolset" available in ArcMap 10.6 (Esri, 2011).

2.5. Statistical analysis:

The database undergoing the statistical analysis consists of 348 records given by 6 temporal observations for each of 58 areal units. Areal units were obtained by first dividing the 13 zones into 29 subzones of comparable extension, then splitting each subzone in two, according to the bathymetric class (deep/shallow). For each areal unit the relative *P. oceanica* surface cover was obtained by dividing the meadow surface coverage by the extension of the unit. Additional variables include: deep/shallow classification, mean depth and slope, presence of 4 anthropogenic disturbance categories (anchoring, harbour, land use change, mining), presence of sea-bottom morphology (derived from MBES data) and 6 indicators of overall image quality: flight height (4 levels), ground resolution (5 levels), sea conditions (categorical), presence of reflections (categorical), georeferencing error, image quality.

We want to understand the similarities between the 13 zones of the Giglio island coastal area in terms of relative Posidonia coverage and the relationships with the presence of anthropogenic impacts and environmental features. To this aim, we use a Beta regression model (Ferrari and Cribari-Neto, 2004) for clustered proportions, which is a generalised linear model based on the Beta likelihood with random effects for Z clusters or groups of zones on the response mean:

$$Y_{it}\ \mathrm{Beta}(\mu_{it}, \tau_{it}) \tag{1}$$

where Y_{it} is the relative coverage of *P. oceanica* for the i-th areal unit ($i = 1,\ldots,58$) at time t ($t = 1,\ldots,6$), μ_{it} is the mean of the Beta distribution and τ_{it} its precision. Further, μ_{it} and τ_{it} are respectively related to the set of available environmental and impact data $\{x_{it}\}$ by the following expressions:

$$\mathrm{logit}(\mu_{it}) = \beta_{t\mu} + \beta_{0\mu}^{z_{k_i}} + \sum\nolimits_{h=1}^{H} x_{ith}\beta_{h\mu} \tag{2}$$

$$\log(\tau_{it}) = \beta_{0\tau} + \sum\nolimits_{l=1}^{L} x_{itl}\beta_{l\tau} \tag{3}$$

In the mean predictor $\beta_{t\mu}$ is an unstructured random effect of time. The number of available time points (only 6) was considered too small to properly estimate a time dependence correlation structure, such as Autoregressive or Random Walk (Krone et al., 2017). The term $z_{k_i} = 1, \cdots, Z$ denotes the cluster membership of the k_i-th zone where the i-th areal unit occurs ($k_i = 1,\ldots,13$) and $\beta_{0\mu}^{z_{k_i}}$ is the effect of cluster z_{k_i}. Similarities among the 13 zones are evaluated as cluster memberships once the effects of time, anthropic impacts and environmental explanatory variables are simultaneously removed. Here we consider linear predictors for mostly categorical or discrete explanatory variables, though in the case of continuous covariates more flexible dependence structures (e.g., based on splines or local regression) could be envisaged (Di Brisco et al., 2022). Hence, by the same model, we investigate the influence of anthropic impacts on the *P. oceanica* coverage and the presence of Z residual homogeneous clusters of zones. We perform the model estimation in the Bayesian setting, implementing our code in the JAGS programming environment (Plummer, 2003). Weakly informative prior distributions are chosen for all parameters:

$$\beta_{\mu}, \beta_{\tau}\ N(0, 1000) \tag{4}$$

$$z_j\ \mathrm{Multinomial}(\pi), j = 1, \cdots, 13 \tag{5}$$

$$\pi\ \mathrm{Dirichlet}_Z(\alpha = 1) \tag{6}$$

Several alternative specifications of the Beta regression model in (1)-(6) were estimated with and without the random effect of time, changing the explanatory variables and the number of clusters Z. For all models we ran the MCMC sampler for 160,000 iterations with 2 chains and a burn-in of 80000, keeping 5000 samples for inference after thinning. Chain convergence was assessed using Gelman type diagnostic including potential scale reduction factor and effective sample size (provided

Fig. 2. Main local human disturbances affecting the studied meadow in Giglio Island from 1968 to 2013. From the top left to the bottom right panel: a-b) mining activities in Giglio Campese (NW); c-d) past (1956) and actual (2013) view of the Giglio harbour; e-f) land-use change (LUC) relative to the construction of the desalination plant (e) and terracing (f); g) works for the harbour enlargement; h) intensive boat anchoring on the coastal meadow during the summer.

automatically by JAGS). The presence of label switching in the multinomial clustering mechanism was checked using the approach of Papastamoulis (2014), which is implemented in the R package label.switching (Papastamoulis, 2016); the results shown are those obtained after the use of the package. The uncertainty of the effects β_μ and β_τ was evaluated by 95% posterior credible intervals. Credible intervals containing the value 0 were meant to correspond to non-influential effects. Beta regression models were compared in terms of DIC (Spiegelhalter et al., 2014), WAIC and WAIC2 (Gelman et al., 2014).

3. Results

3.1. Seabed and zones' features

The shallow and deep limits of the investigated zones were localised between 0.2 and 18.7 m, respectively. A gentle slope characterised the seabed colonised by the seagrass with an inclination between 0.0 and 56.6 degrees. Due to the island's geomorphologic feature, the eastern coast's meadow, except for zone 2, was characterised by lower seabed slope values compared to the western side's prairies (Table 1 Supp. Mat.).

3.2. The impacts

Over the whole study period, the four anthropic impacts affected the meadows of the 13 investigated zones differently in terms of i) frequency of occurrence and ii) distance from the meadow. The impact frequencies over time are reported in Table 2. Anchoring and land-use change (LUC) were the most frequent and detected over the whole study period (1968–2013), compared to mining and harbour, which were less observed and detected only in specific years: 1968–1944 and 1994–2013, respectively. Anchoring showed a strong increasing trend in occurrence, whereas LUC was mainly focused from 1976 to 1994 (Table 2 and Fig. 3**a**). It appeared that only a few zones were affected by

Table 2
Frequency of impacts affecting the considered meadow of Giglio Island from 1968 to 2013.

Year	None	Anchoring	LUC	Harbour	Mining	Anc/LUC	Anc/Har	LUC/Har	Anc/LUC/Har
1968	51	1	1	0	4	1	0	0	0
1976	36	6	10	0	4	1	0	1	0
1987	34	5	11	0	1	6	0	0	1
1994	27	12	8	2	1	7	0	0	1
2005	30	16	5	2	0	4	1	0	0
2013	34	17	3	2	0	2	0	0	0
TOT	**212**	**57**	**38**	**6**	**10**	**21**	**1**	**1**	**2**

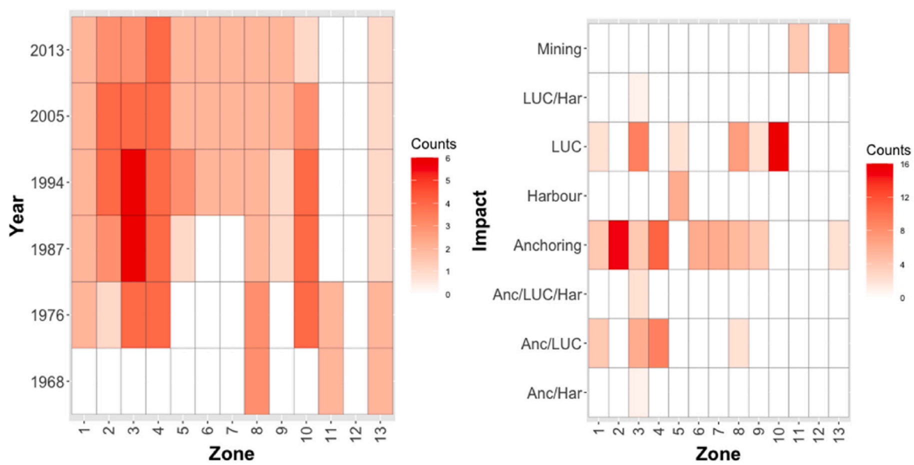

Fig. 3. Frequencies of impacts affecting the 13 investigated zones. a) impacts counts by year and b) impacts counts by zone. Note that the lack of impacts ('none' category) was removed from the plots to facilitate the visualisation.

multiple disturbances; however, zone 12 only was free from impacts of some kind (Fig. 3**b**).

The four impact categories were differently (Table 3) distributed when considering the distance from the meadows (Fig. 4). The activities capable of more directly impacting the meadow were the boat anchoring and the harbour enlargement (with mean values ± SD of 42.0 ± 23.1 m and 66.1 ± 24.6 m, respectively) followed by LUC and mining (with mean values ± SD of 118.8 ± 41.6 m, 169.6 ± 53.1 m, respectively).

3.3. Effects of disturbances and seabed features on meadow dynamics

The Beta regression model in (1)-(6) was estimated considering 21 alternative combinations of the 14 explanatory variables defined in Sec. 2.5 for the two predictors in (2) and (3), with and without the unstructured random effects of time $\beta_{t\mu}$, for $Z = 1$ to 13 clusters of the 13 zones. Model selection and assessment was based on a blend of inspection of influential effects by 95% credible intervals and Bayesian information criteria (DIC, WAIC, WAIC2). For the sake of brevity we don't report results for all the $21 \times 2 \times 13 = 546$ tested models. Overall, including the time effects we obtained higher values of information criteria and non influential $\beta_{t\mu}$ estimates. Notice that, as the impact presence varies with time, the effects of time and impact presence interact. As a results, model selection based on the information criteria suggest that the relevance of the impacts in the model causes the time effect to be not influential. It was also quite clear that $Z = 3$ was the optimal number of zone clusters according to DIC, WAIC and WAIC2. A smaller number of cluster was never supported by the information criteria, but whenever a larger one was, only 3 distinguishable cluster effects could be obtained. Therefore Tab. 3 contains the values of the three information criteria for 21 models with alternative specifications of the predictors in (2) and (3), for 3 clusters of zones and without the time random effect.

The best performing model in terms of the three information criteria contains the effects of the bathymetric class, of the mean depth and slope, of the presence of anchoring, harbour and mining impacts on the mean and only the effect of the bathymetric class on the precision. Then, according to the three information criteria, the presence of the impacts is suitable to describe the variation of the *P. oceanica* mean coverage. Land-use change (LUC) was not included in the final model, being not influential according to information criteria and 95% credible intervals. Conversely, impacts of boat anchoring, mining activities and harbour enlargement were influential and detrimental factors for *P. oceanica* mean cover (Fig. 5**, left**). Comparing the effects of the impacts, the harbour enlargement had the major negative effects on the mean seagrass cover, followed by mining activities and boat anchoring. In addition, slope and bathymetry were influential factors in determining the mean seagrass cover (Fig. 5**, left**). The seabed slope was directly correlated to *P. oceanica* mean extension: higher slope values increased the seagrass cover. The two bathymetric classes (shallow and deep) differently influenced the mean seagrass level: the shallow depth class was a relevant detrimental factor in determining the *P. oceanica* mean cover if compared to the deep one. Specifically, within the shallow depth class, the higher the depth, the significantly greater the mean seagrass surface; conversely, within the deep depth class, the relation between the depth and the seagrass cover is not relevant. Indicators of image quality were not influential on the precision of the P. oceanica coverage that proved to be higher for the shallow bathymetric class (Fig. 5**, right**).

The scatterplot of the posterior mean $\widehat{\mu}_{it}$ against the observed data y_{it} (Fig. 4 Supp. Mat.) allows to validate the ability of the model to fit the data. Moreover, the central 50% and 95% credible intervals of the absolute differences $|y_{it} - \widehat{\mu}_{it}|$ are respectively [0.134, 0.110] and [0.00, 0.300], thus showing acceptable absolute residual values for relative frequencies.

3.4. Similarities among zones according to meadow dynamic

Considering the meadow within the 13 investigated zones over the study period (from 1968 to 2013), we reported that, given the time

Table 3
DIC, WAIC and WAIC2 for 21 alternative specifications of the Beta regression model for 3 clusters without the time random effect. M and P respectively mean that the corresponding variable belongs to the mean and precision predictors, * indicates the presence of an interaction term.

	Deep/ shallow	Mean depth	Mean slope	Anchoring	Harbour	Land use change	Mining	Bottom morphology	Flight height	Ground resolution	Sea conditions	Presence of reflections	Georeferencing error	Image quality	DIC	WAIC	WAIC2
1	M*/P	M*	M	M	M		M								−557.125	−561.461	−558.1923
2	M*/P	M*	M	M	M	M	M								−556.737	−560.870	−557.6174
3	M*/P	M*/P	M	M	M		M						P		−554.429	−559.169	−555.229
4	M/P		M	M	M		M								−537.888	−541.301	−538.3416
5	M/P	M	M	M	M		M	M					P		−536.995	−540.033	−536.5341
6	M*	M*	M	M	M		M						P		−523.963	−527.241	−524.1499
7	M	M	M	M	M		M						P		−510.173	−512.426	−509.5578
8	M		M	M	M	M	M						P		−508.832	−511.613	−508.5964
9	M		M	M	M								P		−506.697	−509.156	−506.7586
10	M	M	M	M	M								P		−504.313	−507.557	−504.9084
11	M	M	M	M	M	M	M	M	P	P	P	P	P	P	−497.428	−504.938	−498.0019
12		M*	M*	M	M	M	M	P					P		−422.286	−424.635	−421.1316
13		M	M	M	M	M	M	P					P		−414.857	−417.977	−414.5806
14		M	M	M	M	M	M				P		P		−408.002	−413.406	−409.5575
15		M	M	M	M								P		−407.354	−409.581	−407.2831
16		M	M	M	M		M						P		−406.786	−409.431	−407.0453
17		M	M	M	M	M	M				P		P		−406.584	−411.007	−406.6712
18		M	M	M	M	M	M				P		P	P	−405.02	−409.521	−404.8785
19		M	M	M	M		M						P		−403.576	−407.187	−404.6916
20		M	M	M	M	M	M	M					P	P	−402.095	−407.845	−403.3239
21		M	M	M	M	M	M	M	P				P	P	−402.41	−401.278	−406.4893

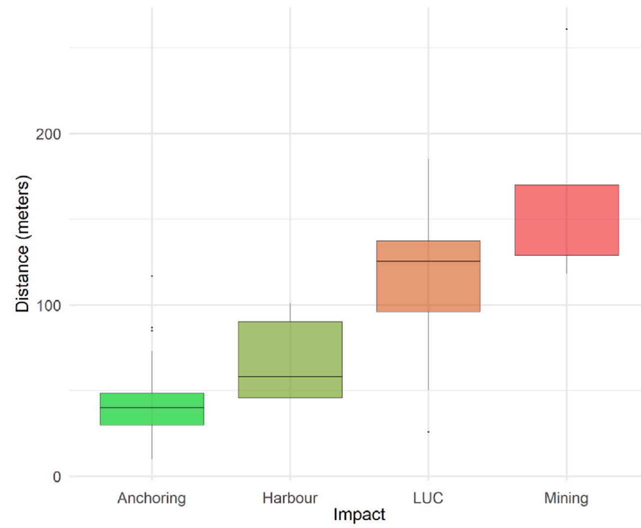

Fig. 4. Distance of impacts from the closest meadows.

variability of the impacts, the effect of time is not influential on the *Posidonia* coverage, passing from 45.81 ha to 43.28 ha (-5%). However, over time we noticed reduction and expansion phenomena within specific zones according to the two bathymetric classes (Fig. 2 Supp. Mat..). For instance, in zones 6, 10 and 13, the contraction of the meadow was confirmed by a decreasing trend in cover (up to − 47% in the shallow depth of zone 10), whereas the remaining zones showed stability or an increasing trend in cover, such as zone 2 and 11 (up to + 215% in the shallow depth of zone 11). As also shown by the model, reduction patterns were mostly reported in the shallower part of the mapped meadows.

Once the effects of the four impacts and bathymetry were removed, three clusters could be defined based on the values of $\beta_{0\mu}^{z_{k_i}}$ for the 13 zones (Figs. 6-7). The first cluster, characterised by higher seagrass cover, was represented by zones 1, 2, 5, 7, 8, and 9. The second one, with intermediate *P. oceanica* cover values, was composed of zones 3, 4, 11 and 12. The third cluster, characterised by a lower mean seagrass cover, encompassed zones 6, 10 and 13.

4. Discussion

The present study gives a detailed picture of the temporal dynamic of the *P. oceanica* meadow surrounding Giglio Island and assesses the effects of major local disturbances affecting the seagrass since 1968. Mapping the investigated meadow's upper edge and its comparison with historical maps provides a reliable method for inferring changes over time in the surface area colonised by the seagrass. Together with the actual distribution charts, historical maps have been used to map and quantify the surface area covered by seagrasses and determine their dynamic over time (Bonacorsi et al., 2013; Leriche et al., 2006; Maccarrone, 2010; Meehan et al., 2005; Pasqualini et al., 2014; Pasqualini et al., 2001; Pasqualini et al., 1999; Pasqualini et al., 1998).

The current *P. oceanica* meadow distribution and conservation status result from the seagrass historical dynamic. Therefore, analysis of the historical maps represents an essential tool for developing a robust comparative baseline to accurately assess losses and possible gains at the Mediterranean scale (Telesca et al., 2015). Using data on current seagrass distribution only is poorly explanatory in evaluating both the trajectories of change and the dynamic of the meadows and represents a substantial limitation in providing a baseline of past ecosystem conditions (Boudouresque et al., 2009).

Our findings are informative to define the meadow dynamics from 1968 to 2013 and highlight that no significant loss in *Posidonia* coverage

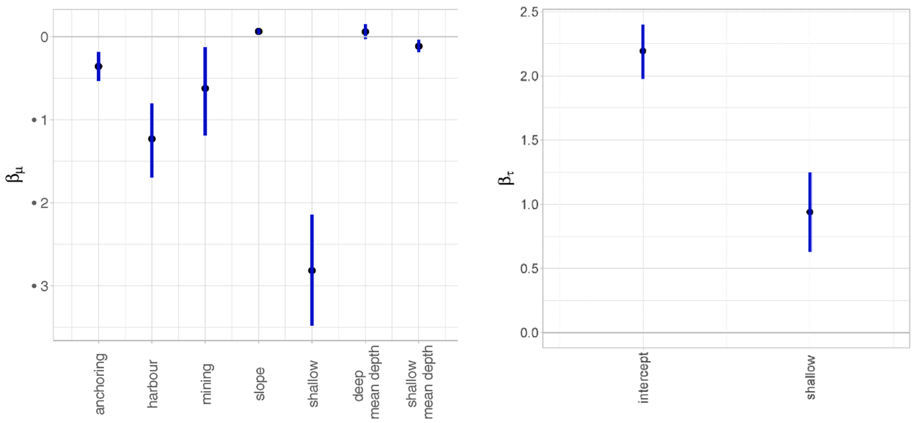

Fig. 5. Estimated effects of the presence of anthropic impacts, seabed and meadow features on the mean seagrass level (left) and precision (right) according to the selected model. Segments represent 95% credible intervals (if the line crosses the zero reference the effect is not influential).

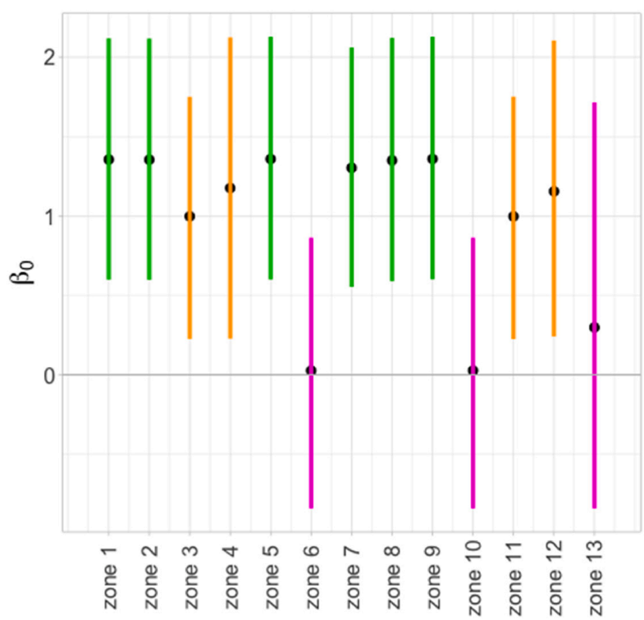

Fig. 6. Estimated effects of zones $\beta_{0\mu}^{Z_{k_i}}$ on *P. oceanica* cover. Colours refer to three clusters: green for high seagrass levels, orange for intermediate values and purple for low ones. Segments represent the 95% confidence interval (if the line crosses the zero reference the effect is not significant). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

occurred during the study period. Despite we reported a slight reduction (-5%) in the total *P. oceanica* cover, a clear trend of diminution (up to −47%) in the *P. oceanica* levels was highlighted in specific zones, due to the increase of impacts in recent years. This evidence confirms that local disturbances, even if acting at a small scale, could severely affect the meadow if any mitigation measures are adopted. Hence, being the Mediterranean and Black Sea ecosystems have been threatened by historical and current pressures, the cumulative impact assessment, also at a local scale, could be considered a valuable tool for achieving the objectives of the EU maritime policy and the UNEP's Mediterranean Action Plan (MAP) to move Mediterranean marine management towards an ecosystem approach (Micheli et al., 2012).

Currently, *P. oceanica* regression is a phenomenon observed almost all over the Mediterranean coastlines (Telesca et al., 2015), especially along the meadows' upper limits (Montefalcone et al., 2018), which major causes are attributable to human-mediated disturbances (Marba et al., 2014). To face this habitat loss, the European Union has increased efforts to reduce the local impacts on *P. oceanica* meadows by improving coastal seawater quality management (e.g., Water Framework Directive) and enhancing seagrass conservation strategies (e.g., Marine Strategy Framework Directive, Habitat Directive). To this end, identifying local disturbances affecting the seagrass is a fundamental step for developing specific protection measures to mitigate and regulate the pressures (Boudouresque et al., 2009).

Our study identifies four specific human-mediated disturbances affecting *P. oceanica* since 1968. The outcomes show that the most detrimental and significant stressors affecting the seagrass are harbour enlargement, mining and boat anchoring. This evidence confirms that coastal development, including construction seawalls, harbours, mining (Leriche et al., 2006; Montefalcone et al., 2018, 2010, 2007; Ruiz and Romero, 2003), and boat anchoring (Ceccherelli et al., 2007; Ganteaume et al., 2005; Montefalcone et al., 2008, 2006; Pergent-Martini et al., 2022; Seytre and Francour, 2008) have a negative effect of *P. oceanica*. Coastal development induces the destruction and deterioration of *P. oceanica* meadows through direct sediment burial, increase in turbidity, upstream hyper sedimentation, and downstream erosion, with modifying effects of coastal drift and pollution (Boudouresque et al., 2009, 2012). The Mediterranean countries have a high population growth rate and strongly attract tourists, especially during summer (Houngnandan et al., 2020). Therefore, many Mediterranean shores have suffered since the first or the second half of the 20th century from rapid urban development, the construction of new seaside resorts and marinas, and the extension of the existing harbours (Boudouresque et al., 2012).

It should be noted that, in this study, impacts derived by land-use change (LUC), such as land reclamation, construction of buildings and roads, sewage, desalination plants, and agricultural practices, that are well-known for the detrimental effect on seagrasses, are not significant in determining diminution in *P. oceanica* cover. This could be explained by i) the small temporal scale at which the LUC impacts act in Giglio Island and ii) the geological nature of the substrata where LUC is localized. In fact, LUC activities are focused in 1976, 1987 and 2005 with a strong decreasing frequency of occurrence over recent times. In addition, all the coastal areas interested by LUC are characterised by monzogranitic substrata, a less erodible rock than the NW sedimentary Triassic calcareous formation (sandstone and clay schist, Fig. 3 Supp. Mat.) where mining activities occurred which erosion may lead to the production of fine sediments that can be easier transported to the sea (Probst and Suchet, 1992).

In addition, *P. oceanica* meadows are particularly sensitive to human activities causing direct impact by mechanical damage such as boat anchoring. Although this phenomenon is not recent, in the past few decades, due to the increase in leisure boating, the anchors' impact has

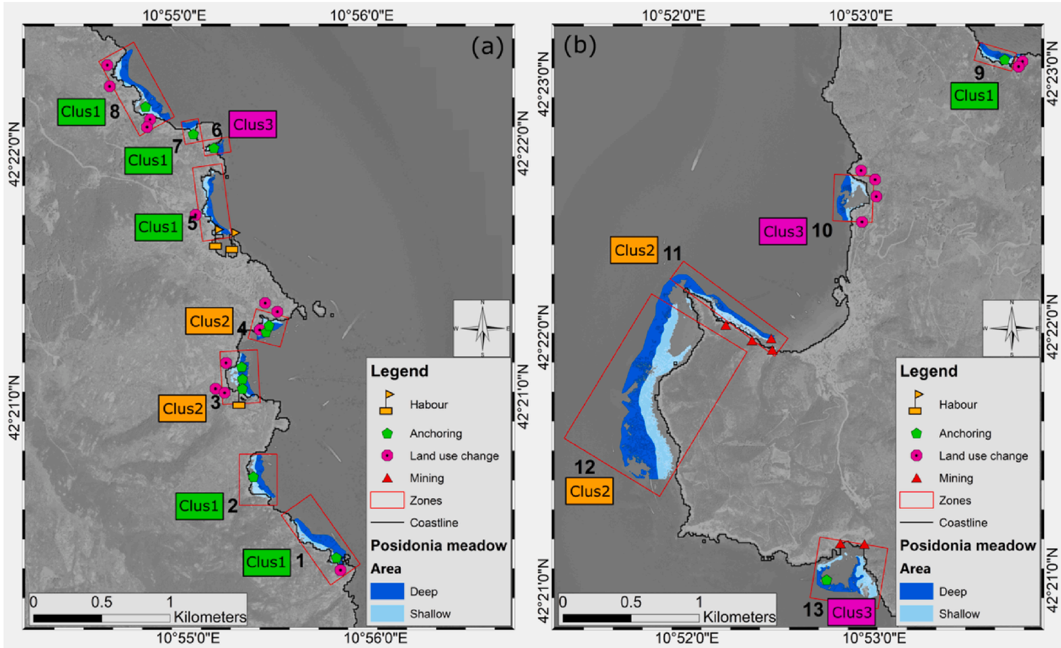

Fig. 7. Spatial distribution of zones according to the identified cluster from the model. Colours refer to three clusters: green for high seagrass levels, orange for intermediate values and purple for low ones. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

become worrying all over the whole Mediterranean Sea (Deter et al., 2017; Francour et al., 1999; Holon et al., 2015; Networking, 2019; Okudan et al., 2011). Being the *P. oceanica* meadows widespread over the shallow coastlines, they cover seabed areas coinciding with the ideal sites for pleasure boats anchoring. The mechanical damages from uncontrolled boats anchoring in shallow coastal waters appear responsible for localised regressions of *P. oceanica* meadows (Montefalcone et al., 2006; Pergent-Martini et al., 2022). The impacts of anchoring systems (e.g. anchors and chains) on *P. oceanica* can be recorded at two different levels: the individual level (the plant), where mechanical damage is the direct cause of pulling up leaves, rhizomes and clods of matte (Ceccherelli et al., 2007), and the population level (the meadow), where mechanical damage reduces shoot density and cover of the meadow (Ganteaume et al., 2005). The maximum sustainable mooring pressure has been demonstrated to be two anchorage events/ha/day (Boudouresque et al., 2012).

Furthermore, the direct action of anchors, by tearing out the plant shoots or sections of matte, reduces the cover of the meadow and enhances the forming of erosive inter-matte areas fragmenting the meadow (Boudouresque et al., 2012 and references therein). Nevertheless, until the boat anchoring is not regulated, clods of matte and cuttings derived from the uprooting of the chain system can be used as a vegetal material to be employed in *P. oceanica* restoration actions without using donor meadows (Mancini et al., 2022, 2021; Ventura et al., 2022).

Together with the aforementioned human activities, natural elements such as coastal geomorphology, hydrodynamism, and seabed characteristics are pivotal in shaping *P. oceanica*'s upper limits (Infantes et al., 2009; Maccarrone, 2010; Pace et al., 2017; Vacchi et al., 2017, 2012, 2010). We reported that zone 12, despite the lack of human impacts, shows a reduction of *P. oceanica* cover over time, especially in the shallower part of the meadow. This could be explained by the geographic exposure to rough seas events from SW and NW, the two quadrants where the strongest winds are registered (for further information, see National network for sea conditions monitoring at https://www.mareografico.it/). In addition, the shallow bottom makes this area one of the most affected by the waves.

In the present study, another influential effect on *P. oceanica* cover over time is determined by depth. Depth has been largely demonstrated to influence population structure, biomass partitioning and photosynthesis of *P. oceanica* (Dennison, 1987; Duarte, 1991; Olesen et al., 2002). The present study confirms the well-known correlation between seagrass occurrence and its spatial extent with bathymetry. In addition, we notice that the shallowest part of the *P. oceanica* upper limit shows a lower cover if compared with the deepest side of the upper limit due to the proximity of disturbances to the meadows.

Widespread and moderate disturbance factors, such as those observed along the coastlines of Giglio island over the years, cannot degrade or threaten the vitality of a *P. oceanica* meadow over vast areas but may trigger local reduction patterns as those highlighted in this study. Conversely, the synergy of severe disturbance events acting on small sectors of the coast, even if for a short period, can lead to the total disappearance of the meadow or decrease its vitality (Boudouresque et al., 2012; Mancini et al., 2019).

The present research also highlights the importance of long-time series in detecting possible changes in the population dynamics of this species. Lastly, this study underlines the importance of enforcing efforts to assess the identification of disturbances factors affecting meadows to prioritise and manage areas where cost-effective schemes for threats reduction, capable of reversing the patterns of change and ensuring *P. oceanica* persistence, could be implemented (Telesca et al., 2015).

CRediT authorship contribution statement

Gianluca Mancini: Data curation, Conceptualization, Methodology, Project administration. **Gianluca Mastrantonio:** . **Alessio Pollice:** Data curation. **Giovanna Jona Lasinio:** Data curation, Project administration. **Andrea Belluscio:** . **Edoardo Casoli:** . **Daniela Silvia Pace:** . **Giandomenico Ardizzone:** . **Daniele Ventura:** Methodology, Investigation, Data curation, Conceptualization, Methodology, Project administration.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2023.110209.

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