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Unravelling Complex Interaction among Coastal Management and Marine Biodiversity: A Case Study in Southern Spain

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Abstract: In this work, we use an integrated modelling approach to explore the complex relationships and interconnections in marine ecosystems among anthropogenic pressures, biodiversity loss, delivery of ecosystem services (ES), and implemented conservation and management strategies. We selected 60 indicators at regional (34), national (12) and international (14) scales that provided long-term information during the 1985–2019 time frame. The results show a decline in marine biodiversity and its associated provisioning services despite the increasing number of responses delivered by a society which are not enough and/or need more time to exert their effects and highlight the pressure on exploited species of unknown conservation status. The decline in Provisioning ES is explained by the decrease in the overall biomass of the captures, mostly large and carnivorous commercially-targeted species and the increase in the number of small-bodied fish species included in the IUCN Red List. The degradation of ecosystem integrity and the continuing loss of biodiversity affect the ability of the ecosystem to provide Regulating ES. The Cultural ES delivery, related to artisanal fisheries, is better preserved in the Gulf of Cádiz. We conclude how the implementation of new management regulations is needed and should be developed through participatory processes to protect and improve marine ecosystem status.

Keywords: fisheries; marine ecosystem services; policies; PLS-PM; conservation; biodiversity; marine protected areas; indicators

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1. Introduction

Marine ecosystems have been a source of multiple services (i.e., food, climate regulation) and contribute to human wellbeing. There is evidence of pelagic fisheries in the Pacific Ocean as old as 42,000 years [1], and in the Mediterranean Sea, Neanderthals exploited coastal resources, including molluscs, fish, and marine mammals in the upper Palaeolithic [2].

The development of human societies and technologies has allowed the exploitation of a wide variety of coastal and marine ecosystems around the globe. The emergence of other ecosystem services (ESs) associated with marine and coastal ecosystems goes beyond the provisioning function, especially regulating and cultural ESs that intimately connect natural systems to human societies, such as global climate regulation, recreational activities, or artisanal fisheries [3,4]. The importance of these ESs to human populations has exponentially increased during the last century [5]. In fact, the world fisheries sector reached an all-time production record in 2018, when 96.4 million tons of marine fish, molluscs, crustaceans, and other invertebrates were captured [6], with at least 34.2% of the assessed world stocks overexploited [6–8]. This situation dramatically increases in the Mediterranean Sea, and according to the Scientific Technical Economic Committee for Fisheries of the European Commission (STECF), 83% of the assessed fish stocks are overexploited [9]. This level of exploitation is both a cause and consequence of a "single species" managing strategy, in which the main goal is to keep all exploited stocks at their maximum sustainable yield simultaneously. Two main strategies have been set in motion during recent decades to address this situation: Marine Protected Areas (MPAs) with different levels of allowed fishing activities and focusing on spaces and conservation efforts and the Ecosystem-Based Fisheries Management (EBFM) approach for exploited stocks focusing on fishing stocks and resource management.

MPAs with different degrees of use and exploitation have been an important conservation tool for marine ecosystems during the last 25 years [10–13] and are part of conservation programs set by the Convention on Biological Diversity (Decision VII/5 of the 2004 COP [14]), the United Nations Environment Program [15], the European Union [16], and national governments (i.e., Spanish marine protection law, [17]). The scientific and social awareness of the distressed state of the oceans and marine biodiversity and positive experiences regarding the implementation of "no-take" areas have given impulse to the creation of new MPAs in every world ocean.

The EBFM strategy arises as a response to the traditional "single-stock" managing procedure that focuses exclusively on the provisioning service of the fishing activity. This ecosystem approach includes multiple species assessments and evaluations of the fishing stocks and considers possible relationships within the considered species assembly and with other species. It also takes into account the effects of the fishing industry on nonexploited biodiversity, ecosystem functioning and other ecosystem services [18–22].

Managing fisheries in a sustainable way that allows the production of high-quality food, income, and livelihood for human societies depending upon the fisheries, understanding the way fisheries create other ESs beyond provisioning, and minimising the negative effects on biodiversity are the challenges for the 21st century. The ES framework merges human needs and impacts with biodiversity state and conservation issues, allowing for an integrated decision-making process when managing coastal and marine ecosystems.

The objective of this work is to identify and understand the main interactions between marine ES and biodiversity and to assess the impact of different marine and coastal management strategies. We hypothesise that anthropogenic pressures increase marine biodiversity loss and decrease their associated ecosystem services, thus, decreasing the delivery of ecosystem services. Indeed, biodiversity loss impacts the conservation policy responses positively through the launching of regulations and MPAs.

2. Materials and Methods

2.1. Study Area

The study area is located in two biodiversity hot spot areas [23–25]: (i) the Alborán Sea and (ii) the Gulf of Cádiz (Figure 1). They were chosen because they are included in the same administrative region, the Autonomous Community of Andalusia, but they belong to two different marine Demarcations and present different physiographic attributes. Due to biogeographic differences, the Alborán Sea and Gulf of Cádiz were studied separately, and every indicator was obtained separately for both areas, except those related to the conservation and management normative delivered at a regional scale.

The Alborán Sea is located in the westernmost part of the Mediterranean. It is a region of extraordinary oceanographic conditions due to its position in the transition zone between two seas: the Atlantic Ocean and the Mediterranean Sea, and its complex orography. It is in contact with the Atlantic Ocean through the Strait of Gibraltar, which, with a depth of approximately 350 m and approximately 14 km wide, is the only passage of water between the Mediterranean Sea and the Atlantic. The continental shelf is narrow, with an average width of 5 km and an average depth of 100 m. The continental slope is very complex and has numerous canyons and collapses [26]. The Alborán Sea basin is divided into two "subbasins" by a submarine ridge that extends in a southwest-northeast direction and rises to the surface at its northeastern end, giving rise to the island of Alborán. The average depth of the western subbasin is 500 m, reaching a maximum depth of 1300 m. The eastern subbasin reaches depths between 1800 and 2000 m and is separated from the Algerian-Balearic basin by a 500 m slope [27]. This set of oceanographic, physiographic, and climatic conditions forms bodies of deep waters rich in nutrients that are more or less deep depending on the meteorological conditions but never disappear and give rise to areas of great wealth, making the Alborán Sea one of the most productive areas of the entire Mediterranean with primary productivity of up to 150 mg/m^2 [28–30].

Figure 1. Bathymetric map of the marine areas included in the study elaborated from the EMODnet Bathymetry Viewing and Download portal service available at https://portal.emodnet-bathymetry.eu, accessed on 30 October 2022.

The Gulf of Cádiz is located on the Atlantic side of Andalusia, has a continental shelf of extremely variable width, is wide in the central zone (where the 100 m isobath is approximately 50 km from the coast) and it narrows when approaching Portugal and the Strait of Gibraltar. The continental slope conforms to the 200 m bathymetric curve. It is a stretched seabed with a gentle slope and mostly sandy sediments. The continental slope is more complex in relief and includes canyons and ravines corresponding to the mouth of the Guadiana River. In the Gulf of Cádiz, there are also three abyssal plains more than 4300 m deep, separated from each other by seamounts. There are also two particularities of interest: mud volcanoes located in the central sector of the middle slope at depths between 700–1100 m and carbonate deposits that form chimneys through which methane gas is expelled and that give rise to unique ecosystems due to their metabolism based mainly on this gas. For the purposes of marine water circulation, the Gulf of Cádiz is connected to the water inlet system from the Atlantic to the Mediterranean through the Strait of Gibraltar. Studies carried out in recent years [31–35] show that the biodiversity of the Gulf of Cádiz goes far beyond the simple presence of species of commercial interest and that there is an important information gap regarding species without commercial interest, especially when the focus moves away from the coast.

2.2. Indicators Selection and Data Sources

The time series of the indicators in this study covers three decades, spanning the period 1985–2019. Indicators were evaluated separately for each of the two study areas that are characterised by distinct environmental features, species composition and fishing pressure. The collected indicators were chosen following three main criteria: (1) indicators containing rigorous and contrasting information proceeding from official statistical databases and repositories; (2) indicators are temporally explicit, meaning that trends could be measured through a time frame (1985–2019); (3) indicators are quantifiable, and they could be easily compared.

Eleven of the indicators were related to biodiversity loss representing the "exploited marine biodiversity" (i.e., number of captured bony fish, cartilaginous fish, mollusc and crustacean species) and the "jeopardised marine biodiversity" (i.e., changes in Conservation IUCN threatened categories (NT, VU, EN, and CR, [36]) for captured species, seabirds, cetaceans and sea turtles, as well as the species included in the Spanish and Andalusian Catalogues of threatened species). Seven indicators were related to ESs: two of provisioning, extracted from biomass of all landed species data of the Andalusian Regional Government; and the number of fish producer organisations, fishermen associations, and business associations representative of the fisheries and aquaculture sector; three of cultural (i.e., variables summarising the development of artisanal fishing gear (number, Gross tonnage and Total power of artisanal fleet vessels), and two of Regulating ESs (i.e., proxies of ecosystem integrity: Trophic Marine Index and (MTI) and Primary Productivity Required (PPR)). In addition, five anthropogenic pressures, the direct drivers of change, were subdivided into three main groups: (1) overexploitation (i.e., variables summarising the development of industrial fishing gear, number, gross tonnage, and power of industrial fisheries fleet vessels), (2) demographic (i.e., human population density in the coastal area), and (3) land use change (i.e., the transformation of coastal ecosystems). Conservation responses derived from biodiversity loss were measured using 19 indicators (e.g., MPAs' number and size and regulations). Finally, 14 indicators regarding management policies were also gathered at regional, national, and international scales. All compiled indicators gathered and/or estimated on a yearly basis, their assignment to the different blocks of the path analysis dimensions, and the data source used for obtaining the database are summarised in Table S1.

Additionally, we found a limited amount of information regarding the presence in the marine areas of cetaceans, seabirds, and marine turtles [37–44], and we included them in the analysis, when possible, i.e., when the amount of data was large enough to be comparable with the other indicators. It is important to state that all fish and shellfish (molluscs and crustaceans) data used in this work are landings of commercially exploited species and do not necessarily reflect the actual biomass removed from the ocean that can be higher due to illegal fishing and discards [33].

2.3. Temporal Trends of Indicators

For a better understanding of the overall models, the general trends of the indicators were examined (Figures S1 and S2). The main concern was the extreme decline of the state of biodiversity in the Gulf of Cádiz and in the Alborán Sea over the period of study. In both marine regions, the indicators related to exploited and jeopardised biodiversity loss have steadily increased since 1985. The delivery of ESs followed different trends in each studied area. In the Gulf of Cádiz, all the ESs had a decreasing trend with a breakpoint in 2000. In the Alborán Sea, the Provisioning ES showed a decreasing trend along the time series, but the rest of the ESs followed a less clear pattern. In addition, in both areas, indicators related to Conservation and Management policies reached another tipping point in 2009–2010, when the implementation of the new marine protected areas belonging to the Natura 2000 Network began, including their associated Sites of Community Importance (hereafter SCIs), Special Protection Areas (SPAs) and Special Areas of Conservation (SACs) as explained in WWF (2014) [45]. The overexploitation indicator declined over time, whereas the spatial transformation of the coast and population increased over time in both areas.

2.4. Conceptual Model and Statistical Analysis

We used Partial Least Square Path Modelling (PLS-PM) to explore the relationships and interconnections among biodiversity, marine ESs delivery, pressures, conservation responses and management strategies of the two marine areas described above (Figure 2). The PLS-PM framework allows us to analyse multiple relationships between blocks of manifest (observed) variables, hereafter MVs, in which each block plays the role of a theoretical concept (e.g., biodiversity loss or ecosystem services delivery) that appears in the form of a latent (unobserved) variable, hereafter LVs [46–48]. It has been previously used to investigate the causal relationship between drivers of change (climate and fishing effort) on marine ecosystem services [49] or among fisheries exploitation, environmental conditions, and ecological descriptors of communities [50].

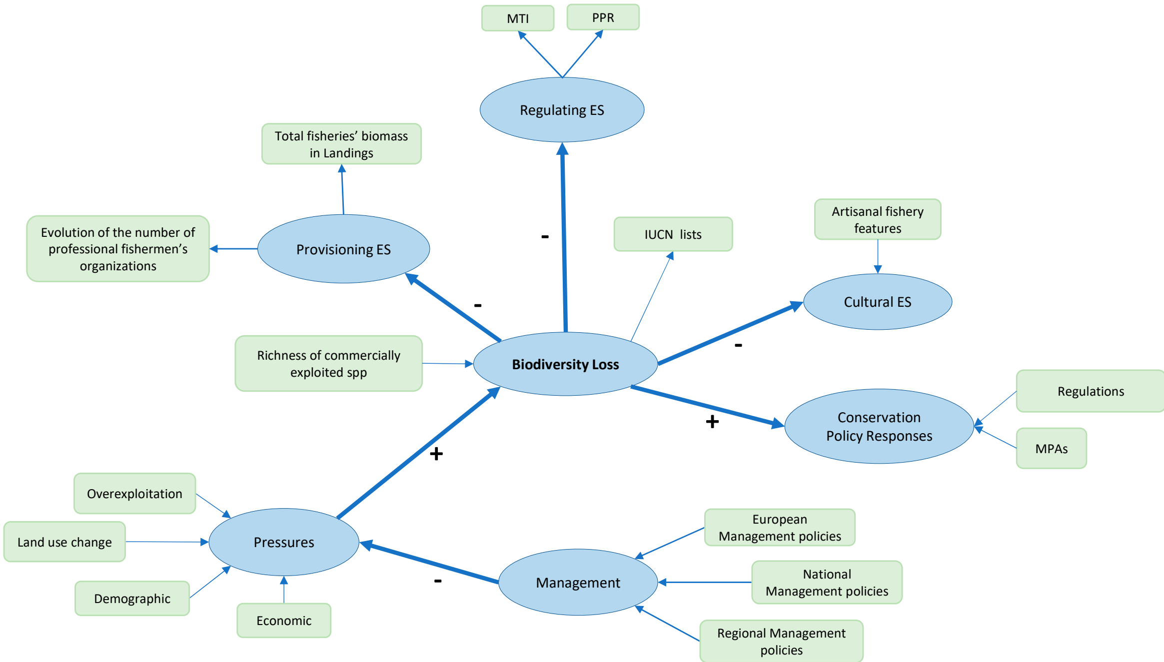

Figure 2. Conceptual path model linking anthropogenic pressures, biodiversity loss, marine ESs, and conservation and management strategies of the two marine areas described above. The inner model consists of seven latent variables (LVs, blue ovals) that affect the endogenous variables through different pathways. Each LV is measured by its own block of manifest variables, which form the outer model (MVs, green boxes). We hypothesise that anthropogenic pressures increase marine biodiversity loss and decrease their associated ecosystem services, thus, decreasing the delivery of ecosystem services. Indeed, biodiversity loss positively impacts conservation policy responses through the launching of regulations and MPAs.

Every PLS-PM model is composed of two submodels: the structural model and the measurement model. The structural model, also called the inner model, takes into account the relationships among the LVs. The measurement model, the outer model, considers the relationships between each LV and its corresponding MVs [48].

In our case, the inner model consists of seven latent variables (e.g., Biodiversity Loss, Provisioning, Regulating and Cultural ESs, Anthropogenic Pressures, Management Policies and Conservation Responses) that affect the endogenous variables through different pathways. Each LV is measured by its own block of manifest variables (indicators), which form the outer model (Figure 2, Table S1).

We hypothesise that biodiversity loss (LV) will be increased by anthropogenic pressures (LV) affecting marine ecosystems and, consequently, will have an impact on the delivery of ESs (LVs) and human wellbeing. Thus, conservation responses (LV) and management policies (LV) are the strategies and actions implemented to preserve the integrity of marine ecosystems and their biodiversity and to prevent or mitigate the effect of the pressures that trigger biodiversity loss (Figure 2).

Moreover, the formulation of the outer model depends on the direction of the relationships between LVs and MVs. There are three types of outer models: the reflective, the formative, and the MIMIC. In the reflective model, each MV is the effect of the corresponding LV and plays the role of an endogenous variable in the specific outer model of the block. In the formative model, each MV is the cause of the corresponding LV and each MV or each sub-block of MVs represents a different dimension of the underlying concept. The MIMIC model is a mixture of the reflective and formative models within the same block of MVs [47,48]. In our path model, indicators related to the LVs of ESs were constructed in a reflective mode (outwards directed model), whereas the ones related to the LVs of Pressures, and Conservation and Management responses, were built in a formative way (inwards directed model). For the LV Biodiversity Loss, we use the MIMIC approach to build the interactions among variables of the outer model (Table S1). In a reflective model, any change in the LV will produce the same directional change in the reflective indicators. Alternatively, in a formative model, each MV is an exogenous variable that could have a different effect on the underlying LV. Once the whole model is conceptualised (i.e., inner and outer models), the PLS-PM is implemented through an iterative algorithm that separately estimates the various measurement models and then, in a second step, estimates the path coefficients in the structural model. Before running the algorithm, all indicators were standardised by subtracting the mean of each value and dividing it by the standard deviation. Then, the general increasing or decreasing trends of the indicators were examined to ensure that they explained the ecological meaning of the assessed component.

2.5. Evaluation of the Path Regression Model

The validation of PLS-PM must consider the three parts of the model: (1) the assessment of the measurement or outer model, (2) the assessment of the structural or inner model, and (3) the validation of the whole model. The quality of the measurement model depends on whether the indicators' nature is reflective or formative. The reflective indicators need to have a strong mutual association with its LV and not any other one; if an indicator is not loyal to its LV, it must be deleted [46]. In other words, the reflective indicators must follow unidimensionality, and there are two indices to check it: Cronbach's alpha and the Dillon–Goldstein's rho. Cronbach's alpha is a coefficient that evaluates how well a block of indicators (MVs) measures their corresponding LV, whilst Dillon–Goldstein's rho focuses on the variance of the sum of variables in the block of interest. As a rule of thumb, for both indices, a value alpha greater than 0.7 is considered acceptable [47,48]. The next things to examine are the loadings and the communalities that are contained in the outer model. The loadings are correlations between an LV and its MVs. In turn, communalities are squared correlations. Loadings greater than 0.7 are acceptable. Communality is calculated with the purpose of checking how much of the variability in MVs is explained by its LV scores and is calculated as the average of all squared correlations between each MV and its underlying LVs. MVs with low communality scores (i.e., <0.7) are those for which the model is not working, and the researcher may use this information to drop such variables from the analysis [48]. Then it is necessary to check the cross-loadings, which are the loadings of an indicator with the rest of the LVs, to verify that the shared variance between LV and its MVs is larger than the variance shared with other LVs by looking at the diagonal of the cross-loading matrix block by block.

In dealing with formative indicators (the ones considered as forming an LV), it is necessary to compare the outer weights of each indicator in order to determine which MVs contribute most effectively to the LV. The elimination of an indicator is recommended when high multicollinearity occurs [47].

The quality of the structural model is evaluated by examining three indices or quality metrics: the predictive power of the model R^2, the redundancy index, and the Goodness-of-Fit (GoF). The R^2 values explain the amount of variance in the endogenous LV explained by its independent LVs, $R^2 > 0.6$ is considered "High" [48]. The redundancy measures the amount of variance of MVs in an endogenous LV that is predicted from the independent latent variables associated with the endogenous LV. High redundancy means a high ability to predict. Overall model predictive performance is measured by the goodness of fit, GoF, which is calculated as the geometric mean of the average communality and the average R^2 value. Since it takes into account communality, this index is more applicable to reflective indicators than to formative indicators. Acceptable values of GoF are those greater than

0.7 [48]. Additionally, path coefficients are used to estimate the strength and direction of the relationships between exogenous LV (e.g., Pressures and Management policies) and endogenous LV (e.g., Biodiversity Loss).

Finally, bootstrapping analysis was used as a final check of the quality of the model pathways and results, using the 95 % bootstrap confidence interval to evaluate whether the parameters were significantly different from zero. All statistical analyses were performed with XLSTAT 2020.4.1.

2.6. Final Indicators Selection

The construction of the PSL-PM models presented below has been designed with an iterative process of review and refinement of the indicators included with a posterior validation of the models obtained. In this way, we started from an initial set of 63 indicators (Table S1) submitted to a first path analysis iteration model. After checking the validation, the delivered models were not acceptable. Those indicators that showed low communalities and cross-loadings (<0.7) or high multicollinearity were eliminated from the model (i.e., they are marked with asterisks in Table S1). Indeed, the rest of the indicators were reassigned to different integrated indices after summing up the gross value of each indicator of the same category and then standardising them (e.g., "Overexploitation" summarised the effect of the pressure done by industrial fisheries effort and integers the time series data for n° industrial vessels, Gross tonnage of industrial vessels and Total Power of industrial vessels (kW), Table S1). Finally, we built our PLS-PM models with 21 indicators for the Gulf of Cádiz and 22 for Alborán.

3. Results

Model Evaluations

The overall PLS-PM for both studied areas had a high predictive power with an R^2 = 0.961 for the Gulf of Cádiz and an R^2 = 0.909 in Alborán (Table 1).

Table 1. Values of R^2 representing the predictive power of the overall model of each studied area.

Marine Area	R^2	F	R^2 (Bootstrap)	S.E.	Lower Confidence Limit (95%)	Upper Confidence Limit (95%)
Gulf of Cádiz	0.961	804.409	0.957	0.014	0.914	0.984
Alborán	0.909	330.068	0.922	0.019	0.879	0.959

Indeed, the GoF of the overall, outer, and inner models were >0.7 in all the cases in both areas (Table 2), thus implying a good selection of the indicators used in the full-time series used.

Table 2. Goodness of fit for the models of the two studied areas; values are acceptable when they are larger than 0.7.

Model	GoF	GoF (Bootstrap)	S.E.	Lower Confidence Limit (95%)	Upper Confidence Limit (95%)
			Gulf of Cádiz		
Overall model	0.729	0.719	0.051	0.602	0.836
Outer model	0.966	0.941	0.057	0.804	1.000
Inner model	0.876	0.872	0.014	0.829	0.898
			Alborán		
Overall model	0.804	0.797	0.069	0.654	0.921
Outer model	0.966	0.942	0.069	0.803	1.000
Inner model	0.956	0.952	0.013	0.915	0.972

The outer models were explained by the unidimensionality of the reflective MVs to their LVs (Biodiversity Loss, Regulating ES, and Provisioning ES) assessed by Cronbach's α and the Dillon–Goldstein ρ always >0.7 (Table 3).

Table 3. Check of the unidimensionality of the reflective indicators (MVs) of each block (LVs) in the outer model when Cronbach's a and Dillon–Goldstein are >0.7 for the Gulf of Cádiz (GC) and Alborán (A) marine areas.

	Cronbach's α		Dillon–Goldstein ρ	
Latent Variable	**GC**	**A**	**GC**	**A**
Biodiversity loss	0.971	0.945	0.977	0.956
Regulating ES		0.859		0.934
Provisioning ES	0.839	0.918	0.925	0.961

In our outer models, all the LVs were measured effectively by all their reflective MVs; they showed loadings and communalities bigger than 0.7, except for the IUCN (fisheries) in its LV Biodiversity Loss in Alborán (Table 4). Indeed, all these loadings and communalities scores were smaller in the Alborán area than in the Gulf of Cádiz.

Table 4. Summary of the Loadings and Communalities contained in the outer model of each studied area. Acceptable values for reflective blocks should be >0.7 (marked in bold).

Latent Variables	Manifest Variables	Loadings	Communalities	Loadings	Communalities
		Gulf of Cádiz		Alborán	
Management policies (formative)	European Management policies	0.635	0.403	0.655	0.428
	National Management policies	0.915	0.837	0.924	0.853
	Regional Management policies	0.946	0.896	0.977	0.954
Pressures (formative)	Population (inhabitants/yr)	−0.911	0.829	−0.901	0.813
	Overexploitation	0.997	0.995	0.994	0.988
	Land use change	−0.836	0.699	−0.858	0.736
Biodiversity Loss (Mixed)	nº bony fish spp. (formative)	0.962	0.925	0.913	0.834
	nº cartilaginous fish spp. (formative)	0.915	0.838	0.847	0.717
	nº molluscs spp. (formative)	0.642	0.412	0.436	0.190
	nº crustacean spp. (formative)	0.928	0.861	0.803	0.645
	UICN (fisheries) (reflective)	**0.963**	**0.928**	**0.785**	0.616
	IUCN (unexploited) (reflective)	**0.919**	**0.845**	**0.866**	**0.750**
	IUCN (Global) (reflective)	**0.980**	**0.960**	**0.956**	**0.913**
	IUCN (Mediterranean) (reflective)			**0.856**	**0.733**
Cultural ES (reflective)	Artisanal fishery features	**1.000**		1.000	
Regulating ES (reflective)	PPR	**1.000**		**0.936**	**0.876**
	ITM			**0.936**	**0.877**
Provisioning ES (reflective)	Total fisheries' biomass for all groups (kg)	**0.906**	**0.822**	**0.956**	**0.913**
	Fishermen	**0.947**	**0.896**	**0.966**	**0.934**
Conservation Policies (formative)	Number of regulations for protected areas	0.186	0.035	0.331	0.110
	Total Number of Marine Protected areas and reserves	0.995	0.990	0.998	0.996
	Changes in the Total Marine Protected Area Surface (ha)	0.971	0.943	0.981	0.963

Pressures and the indicators related to the exploited biodiversity were measured using their weights, and they showed different values at each marine area (Figures 3 and 4) and, in both cases, differed from our hypothesised conceptual model in the negative sign of the Pressure MVs over Biodiversity Loss. The weights of the Alborán's MVs of Conservation and Management Policies were always smaller than the ones found in the Gulf of Cádiz, while the Pressure's weights showed the opposite trend as well as the number of exploited bony fish, which had bigger weights in Alborán.

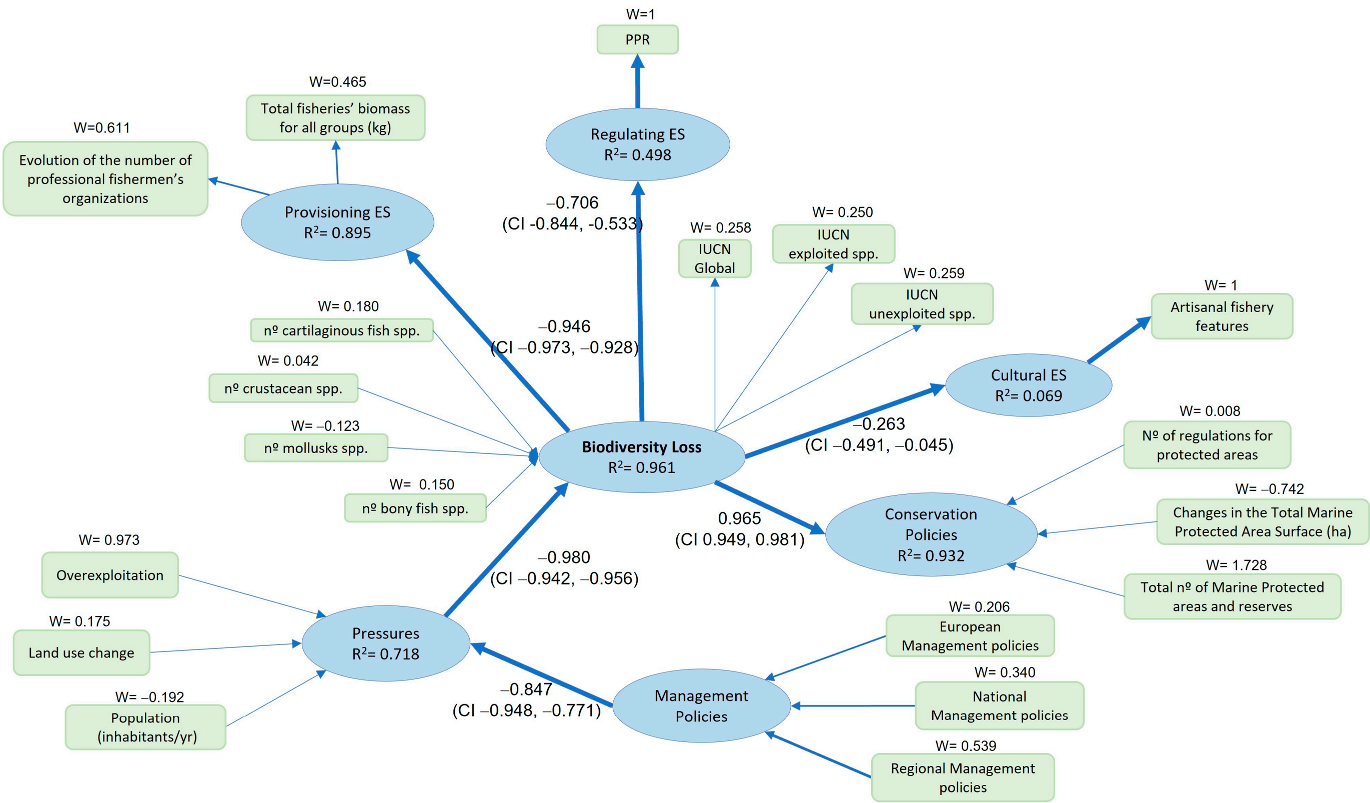

Figure 3. Representation of the PLS-PM of the full time series of the Gulf of Cádiz showing the weights of the MVs of the outer model; each value is positioned over the green boxes of each MV. The strength and direction of all the different pathways of the inner model were validated by bootstrapping, and the 95 % confidence interval (CI) is given below each path coefficient.

In the Gulf of Cádiz, the formative MV with a high weight in the LV Biodiversity Loss was the number of exploited cartilaginous fish species (0.180 ± 0.047, Regional Management policies in the Management Policies LV (0.589 ± 0.215), the number of Marine Protected areas and reserves in the LV Conservation Policies (1.728 ± 0.420), and Overexploitation in Pressures (0.973 ± 0.141). In the Alborán Sea, the formative MV with a high weight in the LV Biodiversity Loss was the number of exploited bony fish species (0.362 ± 0.068) instead of elasmobranchs, and the same indicators of the Gulf of Cádiz in the rest, Regional management policies in the Management Policies LV (0.636 ± 0.199), the number of Marine protected areas and reserves in the LV Conservation Policies (0.878 ± 0.191), and Overexploitation in Pressures (1.074 ± 0.112) (Tables S2 and S3).

Most of the cross-loadings of the reflective MVs showed that the shared variance between LV and its MVs was larger than those shared with the other LV looking at the diagonal of the cross-loading matrix block by block in both outer models except for the evolution of the number of professional fishermen's organisations that showed the highest value in Pressures instead of in Provisioning ES in the Gulf of Cádiz (Table S4) and IUCN (Mediterranean) which had a high cross-loading value in the Conservation Policies LV instead of Biodiversity Loss in Alborán (Table S5).

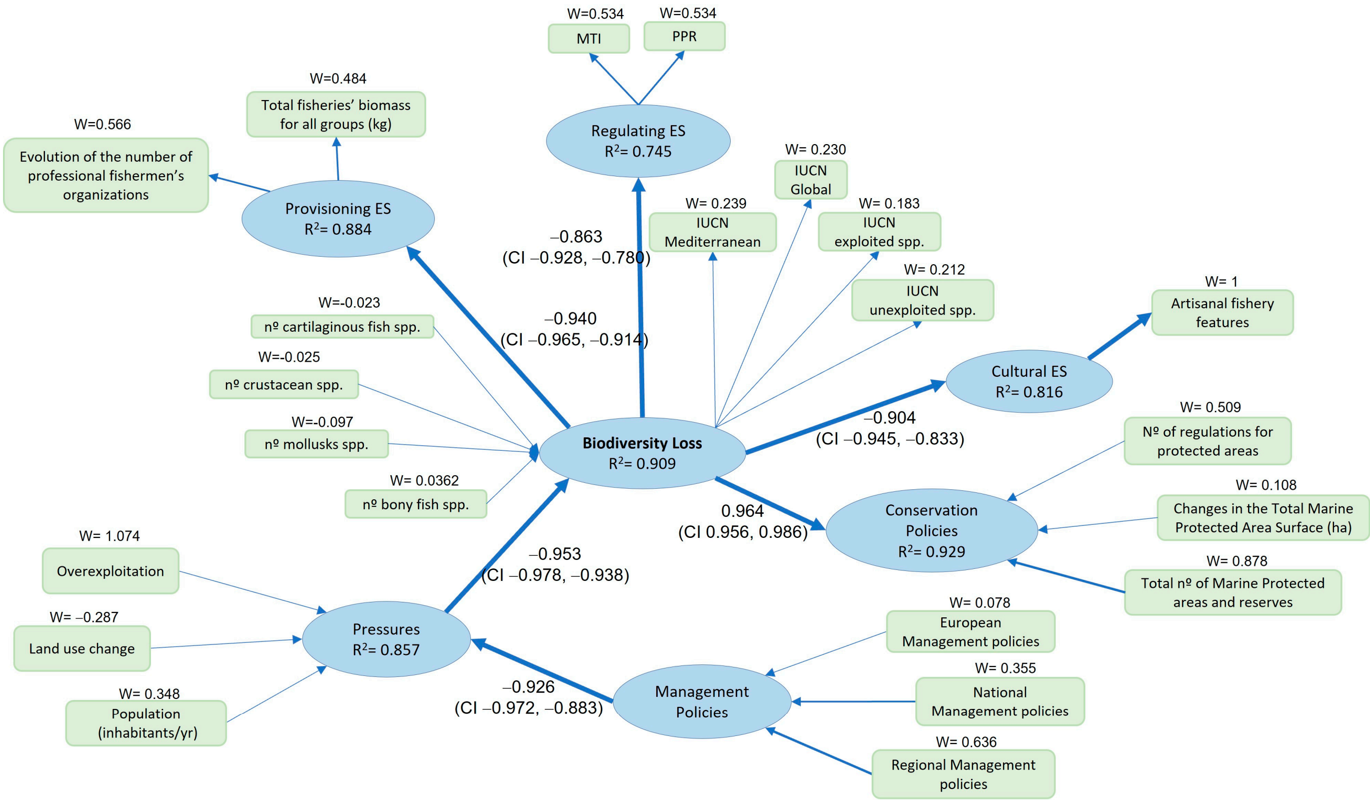

Figure 4. Representation of the PLS-PM of the full time series of the Alborán Sea showing the weights of the MVs of the outer model; each value is positioned over the green boxes of each MV. The strength and direction of all the different pathways of the inner model were validated by bootstrapping, and the 95 % confidence interval (CI) is given below each path coefficient.

Evaluation quality indices of the inner models for both areas are summarised in Table 5. In the Gulf of Cádiz, the LV that had the highest predictive power is Biodiversity Loss (R^2 = 0.961), and in Alborán, it is the block of Conservation Policies (R^2 = 0.929). The observed interrelations between pairs of LVs (Figures 3 and 4) matched the proposed conceptual model (Figure 2) except for the path coefficient direction between anthropogenic Pressures and Biodiversity Loss. In both areas, the sign was negative (Gulf of Cádiz = −0.980 (CL −0.992, −0.956); Alborán = −0.953 (CL −0.978, −0.938), meaning that the observed decrease in the intensity of anthropogenic Pressures is not translated in a lower rate of Biodiversity Loss as opposed to our initial hypothesis that stated that an increase in Pressures intensity would result in a higher rate of Biodiversity Loss.

Table 5. Summary of the evolution of marine environmental education centres in Andalucía.

Marine Area	Centre	Opened	Closed
Gulf Cádiz	Marina El Terrón	1996	2008
	Museo Marítimo Matalascañas	2002	2011
	Centro de Interpretación del Atún de Almadraba	2008	2011
Alborán Sea	Aula del mar de Benalmádena	1989	2011
	Aula del Mar de Málaga	1989	-
	Aula del Mar "El Corralete" Cabo de Gata	2000	2007

Path coefficients between Biodiversity Loss and Regulating ESs are lower than [0.9] in both areas (Gulf of Cádiz = −0.706 (CL −0.844, −0.533) and Alborán = −0.863 (CL −0.928, −0.780)), in the Gulf of Cádiz the path coefficient between Biodiversity Loss and Cultural

ESs is also lower than [0.9] (-0.263 (CL -0.491, -0.045)). The rest of the path coefficients in both models are higher than [0.9] (Figures 3 and 4).

4. Discussion

Our results show an alarming biodiversity loss since 1985 in the two marine areas studied. These results are consistent with the global deterioration of the ecological condition of marine ecosystems and biodiversity loss due to multiple anthropogenic pressures documented in different studies performed on a global scale [8,51,52], European scale [53–57], Mediterranean scale [58–63] and local scale, similar to the study by Torres et al. (2013) [64] in the Gulf of Cádiz or Tudela et al. (2005) [65] in the Alborán Sea. Surprisingly, there was a continuous increase in the number of management regulations as well as in the development of conservation policies (marine protected areas on international, national, and regional scales). Many of them were launched to fulfil the requirements of the European Marine Framework Directive (MSFD), which requires EU member states to achieve good environmental status of their seas by 2020 [16,66]. Indeed, the creation of Marine Protected Areas (MPAs) responds to the requirements of the Convention on Biological Diversity (CBD) for protecting and effectively managing 10% of the sea in MPAs by 2020 [67] to restore and conserve species, fisheries, habitats, ecosystems and their ESs. Despite the 1062 MPAs existing in the Mediterranean Sea, one of the marine biodiversity hotspots currently only covers 6% of the basin, far from the 10% CBD Koichi target, as shown by Claudet et al. (2020) [63]. In the marine areas of this study, we might expect that the management policies implemented by the institutions would be able to mitigate the intensity of anthropogenic pressures that trigger biodiversity loss, but based on our results, they are not able to do so (Figures 3 and 4). Indeed, the number of fishing regulations and the management plans for some species have not yet had the desired protection effect of guaranteeing a long-term recovery of the populations and sustainable fisheries.

The results of the inner models suggest that the pressures decrease as a result of the implementation of management measures (Table 4); however, the drop in the intensity of the pressures is mainly explained by the declining trend of overexploitation (Tables S4 and S5) that begins several years before the implementation of most of the measures (beginning of the 1990s in the Gulf of Cádiz, and at the beginning of the 2000s in the Alborán Sea) (Figures S1 and S2), when the European Maritime and Fisheries Fund (EMFF) started to support different financial lines to help fishers adapt their industrial fishing vessels and gear to a more sustainable fishing method. Moreover, these management policies increments and the decline in the anthropogenic pressures intensity were not able to slow down the biodiversity loss either for exploited species or for jeopardised ones in both areas, which could instead be a consequence of the overfishing of the ecosystem in the last decades [9,33,64,68]. Probably this behaviour in the interaction between anthropogenic pressures, biodiversity loss, and responses lies in the lack of efficiency of the given responses and /or the lack of time to probe their effects on an improvement of biodiversity state and ES delivery.

Another question arising from our results is whether the given conservation responses in these areas are inefficient and/or insufficient in protecting ecosystem integrity and biodiversity (Figures 3 and 4, Tables S4 and S5) or if there are other explanatory factors involved in this decline. There have been recent field studies addressing the effectiveness of MPAs in the mitigation of biodiversity loss that have shown their failure because of their insufficient size and/or inefficient boundary design [69]. It should also be mentioned that policies are often inefficient due to a lack of knowledge, or this is too limited to sustain them and is often lost in political interests (resulting in the so-called "paper parks"). The protection and conservation of marine metapopulations should undertake the design of a system of MPAs networks sufficiently large to avoid the mortality of individuals crossing their borders and sufficiently close to each other to guarantee connectivity among populations and their propagule dispersal [70–75]. Additionally, other failures in regard to their governance way of management have been addressed in some western Mediterranean

MPAs [76,77]. Moreover, the time since the beginning of protection due to the creation of the MPAs has been proven to be an important factor driving the ecological effectiveness of MPAs. Even this period of time has different effects depending on the taxa, which could be five years for some species targeted by fishing or decades for some top predators [78]. In some areas, such as the Cabo de Palos-Islas Hormigas MPA, Rojo et al. (2021) [79] analysed the recovery patterns after 23 years of protection and found that the biomass of piscivorous and macroinvertivore fish increased with time as their density decreased, suggesting top-down or consumer control of the food web. Thus, a similar delay in the effectiveness of regulations, actions and management plans could have occurred in the marine areas of this study. In Spain, the Natura 2000 network for MPAs began its implementation in 2009 for the Atlantic region and in 2010 for the Mediterranean region [45,80]. The total number of protected marine areas and reserves is the indicator with higher weight in the conservation policies enhancing the role of the different MPAs (Tables S6 and S7) in the conservation, but a decade is probably too short a period of time to reach good environmental status (Figures S1 and S2). The role of regional regulations is stressed as the MV with higher weight in the LV management policies (Figures 3 and 4, Tables S4 and S5), and thus the coordination between regional and national governments is paramount to achieve better governance of the MPAs since some of them have shared authority, while others only have national (e.g., marine reserves of Isla de Alborán and Cabo de Gata Níjar, all in the Alborán Sea area) or regional governance (e.g., marine reserve of Reserva de Pesca de la Desembocadura del Guadalquivir in the Gulf of Cádiz) (Tables S6 and S7). Moreover, the support of long-term management plans in MPAs, the improvement of the level of enforcement and the need to design a global network of MPAs have been suggested as key features to achieve global conservation of marine biodiversity [12,13,73,81,82]. In Europe, Katsanevakis et al. (2020) [57] proposed twelve measures for improving MPA management, including implementing adaptive management plans at all sites in the Natura 2000 Network, improving mechanisms for public participation in MPA planning and management, and prioritising conservation goals in full collaboration with stakeholders. In fact, the social effectiveness of large marine protected areas was demonstrated in the study by Ban et al. (2017) [83] of 16 MPAs located around the world, and they found that low levels of participation by resource users and limited external recognition were related to declines in wellbeing, whereas high participation in zoning, social monitoring, siting, rulemaking, and environmental monitoring were associated with improvements in wellbeing.

The Provisioning ES declined during the study period in both marine areas, showing a great decreasing trend in the Alborán Sea. In both areas, the threshold was found in 1999–2000, when the Agreement on Cooperation in the sea fisheries sector between the European Community and the Kingdom of Morocco [84], approved in 1995, was not renewed, and a significant decrease in the total captured biomass occurred and showed a contrasting tipping point in the Gulf of Cádiz because of this reason [85]. In both areas, this decline has had a negative correlation with the land-use change pressures that enhanced the deterioration of littoral ecosystem integrity due to the construction of harbours and coastal development. Known habitats of nurseries for fish and shellfish [86] and the destruction and alteration of these habitats could trigger a decline in the population density of fish and shellfish. Additionally, in both areas, the Provisioning ES has a negative coefficient path with the LV Biodiversity Loss, implying that the higher the loss of biodiversity, the lower the delivery of Provisioning ES (Figures 3 and 4). The Provisioning ES decline in the studied time series responds to two processes occurring at the same time; (1) the decrease in the overall biomass of the captures (mostly large-bodied commercially targeted species, predators with a high trophic level, and carnivorous fish), and (2) an increase in the number of small-bodied fish species (i.e., herbivores with a low trophic level) captured and/or included in the Red List IUCN catalogue; thus, the mean trophic levels of consumers would be lower in an overfished food web than in an undisturbed web [87]. Indeed, other stressors contributing to biodiversity loss that are not officially quantified are the effects of poaching and recreational fisheries that cause an underestimation of the real extracted biomass. There is an important illegal trade of fish that is very difficult to identify and quantify; however, Coll et al. (2012) [88] estimated that up to 43% of the real captures can be illegal in the area. Most nondeclared captures are discards and by-catch species on the Spanish Mediterranean coast, and depending on the fishing gear and habitat, they represent between 13% and 67% of the total biomass intake [89,90]. It is also important to consider extraction by sport-fishing boats, fish trade in the black market, subsistence fishing, nonregistered artisanal fishing, and poaching [91]. Unfortunately, since they provide data for a single time and not for the entire time series, the existing estimates are not applicable to our study. There is a lack of official fisheries reports related to them, and there is an urgent need to include them in our current regulations, as stated by Giménez-Casalduero (2021) [92]. This author reviewed the economic importance of recreational fisheries known for some areas, such as Málaga (Alborán Sea), where recreational fisheries contribute four more times to the markup by professional fisheries. In addition, this author also sounded the alarm about other environmental impacts of recreational fisheries, such as anchoring, which has a negative impact on the seagrass beds of the protected species *Posidonia oceanica* and has also been recorded in other Mediterranean marine areas [93].

Concerning Regulating ES, some differences were found between areas; although PPR (primary production required to sustain the fishery [94]) was selected as an important MV in both areas, MTI (Marine Trophic Index [95]) was only considered in the Alborán Sea (Figures 3 and 4, Figures S1 and S2). The MTI is designed to assess the mean trophic level of the catch, assuming that it will decline as a result of the overexploitation of the higher trophic levels corresponding to large and long-lived predators with high commercial value and is expected to follow the same trend as the PPR index. In this study, however, the MTI steadily increases in the Alborán Sea and shows a recovery pattern since the beginning of the 2000s. These patterns are consistent with those observed in the Gulf of Cádiz by Torres et al. in 2013 [64] and Baeta et al. in Portugal in 2009 [96] and, as they point out, probably related to the collapse of the small pelagic fish stocks and not to the recovery of species with higher trophic level. In both cases, the path coefficients connecting biodiversity loss and Regulating ES had a negative sign, meaning that the degradation of ecosystem integrity and biodiversity loss will negatively affect the ability of the system to provide this type of ESs.

Moreover, this loss of biodiversity is also connected with the delivery of Cultural ES in the Gulf of Cádiz as well as in the Alborán Sea (Figures 3 and 4). However, the coefficient path patterns (Figures 3 and 4) and general trends (Figures S1 and S2) are different in both areas. In the Alborán Sea, the delivery of Cultural ES declined slightly during the study period as a result of the development and protection of artisanal fishing fleets and the creation of fish producer organisations, fishermen associations, and business associations representative of the fisheries and aquaculture sectors; a move which led to a trend of improvement from the 1990s to 2004, reflecting the EMFF effects mentioned above. In the Gulf of Cádiz, however, the artisanal fishery was declining for the first half of the analysed time series and shows an upward trend since the beginning of the 2000s as a result of the general fleet management measures implemented at national and international levels, but especially as a result of regional regulations and initiatives that consider and impulse a stronger connexion between local fishermen and the ecosystems on which they fish [97–102] and a better knowledge of the interactions between industrial and artisanal fisheries in the area [103].

Overall, the complex interactions among the biodiversity, marine ESs delivery, pressures, conservation responses and management strategies of the studied marine areas are easily visualised through the PLS-PM models as addressed in the North Sea [49] and 12 marine areas of different regions of the world [50]. Both models had the same structure of the interactions but different coefficient paths in the inner model (i.e., relationships among LVs) and weights of the outer model (i.e., the contribution of the MVs to LVs) due to different physiographic seascapes and different social organisations. In both areas, the responses given by the society to protect marine biodiversity through management plans and conservation actions are not enough and/or need more time to exert their effects.

We can reinterpret these results using the socioecosystem scale of analysis. The relevance of the different integrated indicators of the social dimension is clear. It seems clear that there is a need to rethink and rebuild anthropogenic pressures and management and conservation policies to ensure the long-term protection and conservation of marine biodiversity through three main axes of action. First, actions have focused on changing and reducing the biodiversity that humans consume. This should be achieved by promoting the development of artisanal fisheries fleets and NGOs of fishers concerned about obtaining sustainable and healthy capture of fish and shellfish together with the reinforcement of environmental education promoting the creation and maintenance of museums and centres focused on these topics in the long term. The lack of interest of the regional governments in cultural and environmental education is regrettable; there were six centres in the late 1990s and early 2000s in Andalucía, and currently, only one remains open (Table 5).

Second, actions related to the improvement of MPA governance and regulations to improve current practices may compromise the effectiveness of conservation actions and plans. Third, we reinforce the need for the collection of high-quality, open-access data from the different institutions implied in the monitoring of the environmental status and the need to promote actions for better coordination among them.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/su15086544/s1, Figure S1: Time series of the indicator set used for the Gulf of Cádiz area; Figure S2: Time series of the indicator set used for the Alborán area; Table S1: Summary of the variables included in each block of indicators latent variables (inner model) and manifest variables (outer model) of the conceptual path model for the study area; Table S2: Weights of the MVs of the outer model of the Gulf of Cádiz. The weights of the formative indicators are used to determine which MV contribute most effectively to its forming LV (marked in bold); Table S3: Weights of the MVs of the outer model of Alborán. The weights of the formative indicators are used to determine which MV contribute most effectively to its forming LV (marked in bold); Table S4: Cross-loading matrix of the outer model of Gulf of Cádiz. Loadings of a MV with the rest of LVs. The shared variance between LV and its MVs must be larger than the shared with other LV looking at the diagonal of the cross-loading matrix block by block; Table S5: Cross-loading matrix of the outer model of Alborán. Loadings of a MV with the rest of LVs. The shared variance between LV and its MVs must be larger than the shared with other LV looking at the diagonal of the cross-loading matrix block by block; Table S6: Summary of the Marine Protected Areas (MPAs) of the marine areas of the Gulf of Cádiz; Table S7: Summary of the Marine Protected Areas (MPAs) of the marine areas of the Alborán Sea.

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