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Integrating Ecosystem Services into Systematic Conservation Planning on an Island's Coastal Protected Area Network

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Abstract

Integrating ecosystem services (ESs) into systematic conservation planning (SCP) has been gaining attention in recent years. For Mediterranean islands, this integration can serve as a path towards effective implementation of conservation strategies and management. In this study on the island of Cyprus, we map and assess three coastal-related ESs using Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) models. The model outputs are used in the MARXAN decision-support tool for the identification of trade-offs and congruences between ESs across the terrestrial and marine realms. We compared low and high equal targets alongside service-specific target combinations to examine trade-offs and congruences both between ESs and across realms, using two Marxan cost configurations. We found spatial congruence between biodiversity and the targeted coastal protection and recreation services. However, the extent of this congruence partly reflects differences in the spatial distribution of the services and the area selected under each scenario. Trade-offs were limited and occurred between recreation and biodiversity. The spatial allocation of priority multiservice sites spans across the terrestrial and marine realms; however, the proportion leans towards the marine realm. Our results show that highly popular tourist sites do not act as multiservice sites because of intensive human presence that affects the quality of the protected area and the congruence of the regulating services they may provide. The study contributes to the growing field of cross-realm ecosystem services prioritization, offering insights for more integrated and effective conservation planning in island contexts.

Keywords: cross-realm; integrated coastal zone management; Natura 2000 network; prioritization; Mediterranean

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1. Introduction

Natural ecosystems and human societies are at a critical point where decisions and actions to sustain them in a healthy state need to be transparent, effective, and well-targeted. In response to the unprecedented environmental challenges of habitat loss and anthropogenic climate change, the need to link nature conservation with human well-being led to the development of the concept of ecosystem services (ESs)—the benefits that human societies receive from ecosystems [1]. The term ES entered ecological science in the late 70s–early 80s, and since then many definitions and classification systems have been proposed [1–4]. The diversity in frameworks and definitions stems from variations in how ESs are generated, delivered, and contributed to human well-being [5–7], as well as from gaps in understanding ES flows at local and regional scales [8]. Furthermore, the communication

of the importance of ESs to policymakers, for the promotion of their protection to safeguard human well-being, led to the development of different methods aiming to value ESs economically [9–11]. However, the monetary valuation of ESs faced criticism for treating nature as a commodity, and for being insufficient because it is inherently impossible to assign monetary value to all ESs, particularly cultural ones that reflect the deep relationships and cultural roots between people and their natural environment [12–14].

Despite the increasing interest and effort to define, classify and assess ESs, their protection remains a low priority for stakeholders [15,16]. Although ESs are explicitly mentioned in policies such as the EU Green Deal and the UN Sustainable Development Goals, and strategies such as the EU Biodiversity Strategy and the Marine Strategy Framework Directive, clear and legally binding legislation for their protection is still lacking. In recent years, the integration of ESs into biodiversity conservation goals, particularly in the context of systematic conservation planning (SCP), has been gaining attention to align conservation objectives with human well-being [8,17–19]. SCP is based on spatially explicit and target-driven information for the identification of cost-effective priority areas [20,21]. However, identifying cost-effective spatial priorities for ESs is inherently more complex, as it requires not only a deep understanding of ES supply, demand, and flow but also a deep understanding of the relationships among them [22]. Positive relationships (synergies) and negative relationships (trade-offs) occur among multiple ESs, and understanding the drivers that form these relationships is essential for their effective management and protection [23].

The relationship between biodiversity and ESs inherently involves both trade-offs and synergies. However, the two often do not spatially (congruence) or functionally (synergy) align [17], which can undermine biodiversity conservation efforts if not carefully mapped, valued, and prioritized [24]. Setting explicit quantified targets for ESs is challenging, as guidelines or criteria for defining appropriate amounts of ESs to conserve are still lacking [25]. In the absence of standardized approaches, most studies rely on area-based proportion methods; for example, protecting a fixed percentage of high-value service areas [24]. While this approach aligns with biodiversity planning approaches, it may be inappropriate for many ESs, particularly those with diffuse flows or spatially mismatched supply and demand [26]. Villareal-Rosas et al. (2020) [27] proposed a decision-support framework to guide ES planning and minimize the risk of ineffective decisions. They emphasized that targeting multiple ESs in the planning process rather than a single ES can optimize accounting for trade-offs and lead to more cost-effective conservation outcomes. In addition to the targeting of multiple ESs, integrating priorities across terrestrial, freshwater, and marine ecosystems further enhances the cost-effectiveness of conservation networks [28–31]. To date, existing cross-realm applications predominantly focus on biodiversity [32–34], whilst fewer focus on ESs [35,36].

A cross-realm approach is particularly relevant for islands, where spatial limitations and the strong land–sea interactions amplify ecological dynamics [37]. Islands are often biodiversity hotspots but face heightened risks due to climate change, land use change, and—in the case of Mediterranean islands—touristic development that can jeopardize not only biodiversity but also their cultural heritage. Furthermore, the dependency of island communities on ES provision is greater than that of their counterparts on mainland [38]. Although many ES assessments have been conducted for islands, very few studies concentrate on Mediterranean islands, and even fewer assess ecosystem services provision by integrating more than one realm [39].

To address these gaps, our study applies two established tools, InVEST [40] and Marxan [41], for ES assessment and spatial prioritization, respectively, within a common terrestrial-marine framework. In contrast to realm-specific assessments, our approach incorporates habitat characteristics and relevant human pressures from both terrestrial

and marine realms into the assessment of ESs [42]. This allows the contributions of both realms to be considered jointly in the assessment, instead of being evaluated independently. The application of a common spatial prioritization framework allows the investigation of conservation priorities to be explored across the terrestrial-marine framework, thereby contributing to the advancement of cross-realm SCP.

Cyprus island represents a useful example to demonstrate this approach, given the complex interplay between economic development and environmental conservation. The island is heavily reliant on ESs, particularly those provided by coastal and marine ecosystems. This dependency is reflected in its economy, with over 50% of the national GDP generated by coastal and marine activities [43]—a figure expected to increase with the continued growth of sectors such as aquaculture, shipping, energy, and tourism. In addition, more than half of the population lives in coastal areas, intensifying the pressure on these ecosystems. Cyprus also illustrates significant shortcomings in environmental governance, particularly reflected in the management of PA [44]. For example, many coastal and marine PA function as little more than 'paper parks', lacking both effective management plans and enforcement mechanisms.

The aim of the paper is to address the spatial trade-offs and congruence between three coastal-related ESs, using Cyprus' coastal protected areas network as a case study, using spatially explicit indices. Specifically, we address the following research questions:

1. Where are the supply areas of the three ESs located within the coastal protected areas network, and to what extent do they spatially overlap?
2. How do different representation targets alter the spatial configuration of priority areas and their allocation between the terrestrial and marine realms?
3. How does the integration of socioeconomic proxy costs reflecting human activities and pressures influence the prioritization of areas and their allocation between terrestrial and marine realms?

2. Materials and Methods

2.1. Study Area

Cyprus is located at the easternmost part of the Mediterranean Sea (Levantine Sea), covering 772 km of coastline, of which only 296 km fall under the effective control of the Republic of Cyprus [43]. Cyprus has a rich and diverse biodiversity due to its geological structure, climatic conditions, geographic location (isolated island) and topography [45], making it a biodiversity hotspot. According to the report on the definition of the landward limit of Cyprus's Integrated Coastal Zone Management (ICZM), as mandated by the Barcelona Convention, the coastal zone extends up to 5 km landward [46] and up to 12 nautical miles seawards (territorial waters). The protection of species and habitats on the island is guided by the application of national and European laws (EU Birds and Habitats Directives). In total, the PA coverage (Natura 2000 sites) of Cyprus (area effectively controlled by the Republic of Cyprus) amounts to 1694 km^2 of land area and 8464 km^2 of marine area (Natura 2000 Barometer, accessed September 2026). Out of the total 69 Natura 2000 sites, six are coastal MPAs, and one is an offshore MPA. In addition to the MPAs, there are 11 artificial reefs that are protected under national protection law (Fisheries Law (106 (I) 2004)). The political situation on the island affects the PA network because of the suspension of *acquis communautaire* in the area not effectively controlled by the Republic of Cyprus, and the exclusion of the UK Sovereign Base Areas (SBAs) from the Accession Treaty of Cyprus [47].

In this analysis, we aim to map and assess three ES-related indices within the PA network and the zone defined by the ICZM Protocol. The landward limit and the 12-nautical-mile marine territorial waters delineate the study area, aligning with a cross-realm

approach that integrates both terrestrial and marine realms. This integration allows each ES to be assessed across the combined terrestrial-marine area, simultaneously accounting for the contributions of habitats from both realms to ES supply, thus capturing spatial patterns in ES distribution across the coastal zone. The analysis focuses on the PA network, allowing for an explicit evaluation of how ES priorities can be integrated into existing biodiversity conservation planning. To achieve this, detailed spatial information on habitat distribution is required. This is available within the PA network but is not consistently available outside of it. Within the PA network, only Sites of Community Importance (SCIs) (Figure 1) were included, as these are designated under the Habitats Directive (92/43/EEC), for which spatially explicit information on habitats is available. Conversely, Special Protection Areas (SPAs) designated under the Birds Directive (2009/147/EC) were excluded because habitat information is not a requirement for their designation, and a corresponding spatially explicit habitat dataset was therefore not available. Moreover, areas not under the effective control of the Republic of Cyprus were omitted from the analysis due to data unavailability regarding habitat types and distribution, and because EU nature directives do not apply there.

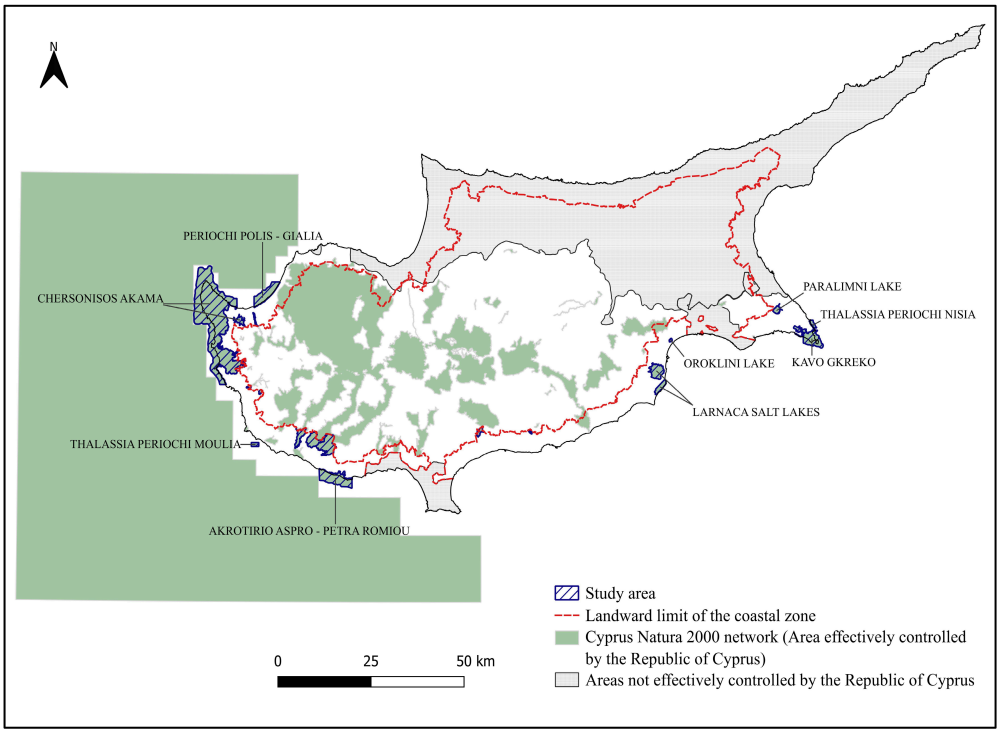

Figure 1. Map of the Natura 2000 network in Cyprus and delineation of the study area.

2.2. Selection Criteria for the ESs

ESs were selected based on the following criteria:

(i) Relevance to the coastal zone and islandscapes: The selected ESs needed to be ecologically and socially significant for island environments. For instance, island coastlines are vulnerable to erosion due to exposure to storm surges and sea-level rise, and recreation is a key economic driver for many islands, particularly Mediterranean islands such as Cyprus.

(ii) Availability of spatially explicit data: The mapping and assessment of ESs in this study relied on the availability of high-resolution, spatially explicit datasets.

(iii) Data representativeness: Data needed to be representative of the study area, ensuring appropriate resolution and complete coverage.

2.3. Integrated Valuation of Ecosystem Services and Trade-Offs (InVEST) Models

To map and assess the selected ES, we used the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) models (InVEST® v3.19.0: https://naturalcapitalproject.stanford.edu/software/invest, accessed on 18 September 2026). InVEST is a suite of spatially explicit models designed to map and assess individual ESs, providing spatially explicit outputs that can be used to assess spatial trade-offs and congruences. The models use spatial input data and produce biophysical or economic outputs that reflect the flow and magnitude of services across terrestrial and marine environments, considering ecosystem structure and function alongside human presence and activities [40]. Following the application of the aforementioned criteria, three InVEST models were selected and implemented: Coastal Vulnerability (CV), Habitat Quality (HQ) and Visitation: Recreation and Tourism (RT). These models generated indices representing coastal exposure, habitat quality and visitation, respectively, which were used as proxies for coastal protection, biodiversity maintenance and recreation. Although the coastal exposure index derived from the CV model does not directly measure coastal protection, it was interpreted at the biophysical level as an indication of the capacity of coastal segments to provide natural protection against erosion and inundation. Accordingly, less exposed coastal segments were assumed to have greater capacity to provide natural protection. The selected ESs were classified according to the Common International Classification of Ecosystem Services (CICESs) framework [4] (Table S1 of the Supplementary Materials S1). All InVEST models were applied according to their respective guidelines. The contribution of this study lies in its operationalization within a cross-realm framework, whereby information from both realms was incorporated into the construction of the three indices. The approach to this integration differed among the models and is explained in the description of each model below.

The CV model uses spatially explicit information on a set of biophysical characteristics that act upon and influence the coastline dynamics. The model integrates geomorphology, wind and wave characteristics, surge potential, relief, and the mitigating role of coastal habitats to estimate an exposure index (EI) for each coastline segment [48,49]. Each input variable is assigned a Relative Exposure Index (REI), expressed as a ranked score from 1 to 5 (1 = very low exposure; 5 = very high exposure), using a combination of user-defined and model-derived values. The overall EI is then calculated as the geometric mean of the individual REIs [40]:

$$EI = \left(\prod_{i=1}^{n} R_i \right)^{1/n} \quad (1)$$

In this study, six input variables were used (Table 1) to calculate the exposure index (EI). Equation (1) becomes as follows [40]:

$$EI = \left(R_{Geomorhpology} R_{Relief} R_{Habitats} R_{Wind\ exposure} R_{Wave\ exposure} R_{Surge} \right)^{1/6} \quad (2)$$

Table 1. Datasets and their sources used in the InVEST Coastal Vulnerability model.

Biophysical Data	Source
Geomorphology	European Marine Observation and Data Network (EMODnet) [50]
Wind	Cyprus Meteorological Service
Wave	Copernicus Marine Service [51]
Habitats	Article 17 reporting of the EU Habitats Directive (92/43/EEC) [52]
Bathymetry Elevation	Eratosthenes Database (2014) [53]

In this study, the integration of terrestrial and marine realms was reflected in the mitigating role of the habitats. Habitat types from both realms were included in the model and assigned exposure ranks according to their capacity to reduce coastal exposure. A

maximum protection distance was specified for each habitat, beyond which the habitat exerts no protective effect (Table S2 of the Supplementary Materials S1). The assigned scores ranged from 1 to 5 and were based on the guidelines provided in the InVEST Coastal Vulnerability documentation [40], while the specific ranking scheme was informed by approaches developed by Gornitz (1990) and Hammar-Klose & Thieler (2001) [54,55]. Habitat ranking followed earlier research by Liquete et al. (2013) and Drius et al. (2019) [56,57], and protection distances were calculated using QGIS Desktop (version 3.44.10) [58] as the distance between each habitat's centroid and the coastline [40]. The exposure rank assigned to habitats was then calculated by applying the following equation [40]:

$$R_{Hab} = 4.8 - 0.5 \sqrt{\left(\begin{matrix} N \\ 1.5 \; max \\ k \end{matrix} \left((5 - R_k)^2 \right) + \sum_{k=1}^{N} (5 - R_k)^2 - \begin{matrix} N \\ max \\ k \end{matrix} \left((5 - R_k)^2 \right) \right)} \quad (3)$$

where R_k is the exposure rank of habitat k, and N is the total number of habitats. According to the model, the habitat with the lowest exposure rank is weighted 1.5 times higher than other habitats present near a coastline segment [40]. Consequently, coastal segments containing more than one habitat type may receive a lower exposure rank, reflecting a greater capacity to mitigate coastal exposure compared with segments containing only one habitat type [40].

The wind, wave, surge, and relief REIs were derived following the methodology proposed in the InVEST Coastal Vulnerability documentation [40]. Local bathymetry and wind data were used to calculate the surge exposure index (R_{Surge}), and elevation data to calculate the relief index (R_{Relief}). For the computation of the wind exposure index (R_{Wind}), wind data were obtained from the four coastal stations of the Cyprus Meteorological Service for the period 2016–2021. The index was computed using a sector-based formulation that multiplies wind speed in each of the 16 equiangular sectors (0°, 22.5°, etc.) by the proportion of high-percentile occurrences and fetch distance [40]:

$$R_{wind} = U_n \times P_n \times F_n \quad (4)$$

where U_n is the average wind speed of the highest 10% wind speeds in sector n, P_n is the proportion of winds in the top 10% that fall within sector n, and F_n is the fetch distance along that sector. Because empirical wave-buoy data were not available for Cyprus, the wave exposure index (R_{wave}) could not be derived directly by the CV model. Instead, we used satellite data from the Mediterranean Sea Waves Reanalysis dataset provided by the Copernicus Monitoring Environment Marine Service [51]. Specifically, the primary swell significant wave height was extracted as a proxy for wave energy, under the assumption that higher swell wave heights correspond to greater wave energy acting upon the coastline. The data were sampled at the coastline, assigning each coastline point the corresponding wave height value. Where a grid cell did not intersect a coastline segment, nearest-neighbor resampling was applied to assign the closest wave height value. Finally, these values were reclassified into ranked wave exposure index values (1 = very low exposure to 5 = very high exposure) to harmonize them with the model's ranking scheme. The final exposure index (EI) was subsequently computed by incorporating the customized wave index into the geometric mean equation (Equation (2)). The resulting index represents the relative exposure of each coastline segment to erosion and inundation, considering both physical coastal characteristics and the mitigating role of natural habitats.

The HQ model was used to map and assess the habitat quality in the study area [40]. It assumes that areas with high habitat quality support higher biodiversity. The model

combines information on habitat extent and threats to produce habitat quality maps. The habitat quality is calculated from the following equation [40]:

$$Q_{xj} = H_j\left(1 - \left(\frac{D_{xj}^z}{D_{xj}^z + k^z}\right)\right) \quad (5)$$

where H_j (0~1) is the habitat suitability of habitat type; 1 indicates habitat with the highest suitability and 0 with the lowest suitability. Accordingly, natural habitats were assigned a value of 1, whereas human uses were assigned a value of 0. The parameter z is a scaling parameter, and k is the half-saturation constant (default value is 0.5) that is used to spread and distribute habitat quality values for a better visualization of the index changes in the study area. D_{xj} is the total threat level, and is calculated by the following formula [40]:

$$D_{xj} = \sum_{r=1}^{R}\sum_{y=1}^{y_r}\left(\frac{w_r}{\sum_{r=1}^{R}\omega_r}\right) r_y i_{rxy} S_{jr} \quad (6)$$

D_{xj} depends on four factors: the relative impact of each threat w_r, the distance between habitat and threat source, and the impact of the threat across space i_{rxy}. The latter term is determined by a linear or exponential distance-decay function [40]:

$$i_{rxy} = 1 - \left(\frac{d_{xy}}{d_{r\ max}}\right) \text{ if linear} \quad (7)$$

$$i_{rxy} = exp\left(-\left(\frac{2.99}{d_{r\ max}}\right) d_{xy}\right) \text{ if exponential} \quad (8)$$

where d_{xy} is the linear distance between cells x and y, and d_{rmax} is the maximum impact distance of threat r. In the exponential decay, the constant 2.99 reduces the impact of the threat by 95% at the specified maximum distance of the threat's effect. The term S_{jr} represents the relative sensitivity of the habitat to each threat in the study area.

Given the lack of information for these numerical parameters, we provided a structured questionnaire to seven local experts from academia and national environmental management authorities to assign values to each model parameter. During the expert assessment, habitat sensitivity was evaluated with respect to threats originating in both realms, rather than being restricted to threats occurring within the same realm as the habitat. Consequently, the HQ estimates for each habitat were influenced by the spatial distribution of threats, with their impact determined by the habitat's sensitivity to each threat, in combination with the corresponding maximum impact distance and decay function, irrespective of the realm in which the threat originated or the habitat occurred. The final parameter values represent the average value of all expert scores. The decay function (linear or exponential) for each threat was selected based on the majority consensus among the experts, considering their level of confidence. Of the habitat types included in the assessment, expert input was obtained directly for 13. The remaining habitats were either assigned the same scores as other habitats belonging to the same aggregated habitat classification, or parameterized using baseline InVEST sample data [40], supplemented by values from similar case studies in the Mediterranean region [59]. The habitat types included in the assessment are listed in Table 2. The final parameter values are detailed in Table S3 of Supplementary Materials S1, and the detailed expert elicitation methodology is detailed in Supplementary Materials S2.

No independent field verification or external statistical validation of the HQ index was performed in the study area. Furthermore, no sensitivity analysis was conducted to assess how variations in threat weights, maximum impact distances, decay functions and habitat sensitivities might influence the resulting index values. To examine the consistency of the modelled HQ estimates with available independent information, a semi-quantitative comparison was undertaken using two sources: (i) the conservation status of Annex I habitats

reported under Article 17 of the Habitats Directive (92/43/EEC) for the 2019–2024 reporting period [60], and (ii) the ecological status assessments of Annex I coastal marine habitats reported by Kletou et al. (2024) [61]. These assessments represent relevant independent sources of information identified as available for the study area and were therefore used as complementary reference points to examine the consistency of the modelled HQ estimates with existing evaluations of the habitats.

Table 2. Coastal and Marine habitat data according to Annex I of the Habitats Directive (* priority habitats) used in the Habitat Quality model. Source: Article 17 reporting of the EU Habitats Directive (92/43/EEC) (https://data.gov.cy/) [52].

Habitat Group Classification	Habitat Code
Schlerophyllous scrubs (mattoral)	5212, 5220 *, 5330, 5420
Coastal and Halophytic habitats	1150 *, 1210, 1240, 1310, 1410, 1420
Coastal Sand dunes and inland dunes	2110, 2230, 2240, 2250 *
Coastal and Marine Habitats	CY04, CY05, 1110, 1120 *, 1170, 8330
Freshwater Habitats	3140, 3150, 3170 *, 3290
Natural and Semi-natural grassland formations	6220 *, 62B0 *
Rocky habitats and caves	8210, 8310
Forests	9290, 92C0, 92D0, 9320, 9540

The two sources differ from the HQ model in their objectives, assessment indicators and spatial scales. Article 17 assessments provide an overall evaluation of the conservation status of Annex I habitat types at the Member State level over a six-year reporting period. This assessment considers habitat's range, area, specific structure and functions, and future prospects. The assessment by Kletou et al. (2024), on the other hand, focuses specifically on Annex I coastal marine habitats and evaluates their ecological status at the level of individual Natura 2000 (N2K) sites, using specific indicators and assessment protocols. For example, *P. oceanica* (1120*) was assessed using the Posidonia oceanica Rapid Easy Index (PREI), while sandbanks (1110) associated with the established *Cymodocea nodosa* meadows were assessed using the CymoSkew Ecological Quality Ratio (EQR). These comparisons are descriptive and are intended to examine consistency between the modelled HQ estimates and available independent information, and do not constitute a validation of the index.

The RT model was used as a proxy to assess the cultural ecosystem services in the study area. The model predicts the spatial distribution of recreation and tourism person-days based on the location of natural habitats, accessibility, cultural sites and other landscape or seascape features that may influence people's decision about where to recreate [62]. The model output is a map depicting visitation rates across the gridded study area. To estimate visitation potential, the RT model builds a linear regression model to estimate the contribution of landscape or seascape features to the overall visitation [40]:

$$y_i = \left(\beta_{0+}\, \beta_1 x_{i1} + \cdots + \beta_p x_{ip}\right), \text{ for } i = 1 \ldots n \quad (9)$$

where x_{ip} represents the spatial coverage or presence of predictor feature within each grid cell; i, n is the total number of cells in the study area, and the dependent variable y_i is quantified as Photo-User-Days (PUD) derived from geotagged images uploaded to the Flickr database. The regression estimates β_p coefficients, which essentially represent the direction and magnitude of the relationship between each predictor and the visitation rate [40]. In this study, we compiled a list of predictors that are likely to influence recreational choices within the coastal and marine environments [63–65]. Because the analysis focuses on a coastal PA network, the predictors incorporate ecotourism elements such as natural trails, dive sites, artificial reefs, archaeological monuments, and important areas for birds. Predictors representing recreational activities and opportunities in both terrestrial and

marine environments were included simultaneously in the regression analysis, allowing the estimated visitation patterns to reflect the contributions of both realms within a common spatial framework (Table S4 of the Supplementary Materials S1).

To capture distinct spatial patterns of visitation, the study area was divided into discrete grid cells. To identify the optimal grid cell size, we performed a sensitivity analysis testing cell sizes of 0.25 km^2, 1km^2 and 4km^2. The 1km^2 grid cell size was selected because it best captured the most coherent spatial visitation patterns across the study area.

2.4. Spatial Prioritization

The second part of the methodology involved the spatial prioritization of the selected services to examine potential spatial trade-offs and congruences within a common land–sea framework. We used Marxan software [41], a widely used decision-support tool for the identification of priority areas for conservation. It uses a heuristic optimization algorithm (simulated annealing) to minimize an objective function that accounts for the costs of selected planning units (PUs) that comprise the study area (usually the study area is divided into squares or hexagons), and penalties for not achieving the conservation target set or for missing connections. The output of the simulated annealing algorithm is a near-optimal solution. The optimization problem is as follows [41]:

$$\sum_{i=1}^{m} c_i x_i + BLM \sum_{i1=1}^{m} \sum_{i2=1}^{m} x_{i1}(1 - x_{i2}) cv_{i1,i2}\ ,\ \ for\ i \in \{1, 2,\ \ldots,\ m\}, \tag{10}$$

$$subject\ to\ \sum_{i=1}^{m} a_i x_i \geq t_j \forall j,\ \ for\ j \in \{1, 2,\ \ldots, m\} \tag{11}$$

where x_i is a control variable for the group of m planning units that takes a value of 1 when the planning unit i is selected and 0 when not; c_i is the cost of the planning unit i; a_i is the presence of a conservation feature in the planning unit i; $cv_{i1,i2}$ is the penalty for missing the connection between a given pair of planning units ($i1$ and $i2$) in the solution, weighted by the Boundary Length Modifier (BLM), which determines the importance given to the boundary length relative to the cost of the reserve system; t_j is the target for each conservation feature. Under these premises, the objective function that we tried to minimize is as follows [41]:

$$Obj.function = \sum_{i=1}^{m} c_i x_i + BLM \sum_{i1=1}^{m} \sum_{i2=1}^{m} x_{i1}(1 - x_{i2}) cv_{i1,i2} + \sum_{j=1}^{n} SPF_j H\left(s_j\right)\left(\frac{s_j}{t_j}\right) \tag{12}$$

where there are n conservation features under consideration; SPF_j is the Species Penalty Factor applied when not adequately representing a conservation feature j; $H(s_j)$ is a heaviside step function that takes a value of 0 when s $\geq$ 0 and 1 otherwise; s is the shortfall in targets not achieved and is measured as t_j representation achieved; the ratio $\frac{s_j}{t_j}$ equals 1 when the conservation feature j is not represented within the solution and approaches 0 as the level of representation approaches the target amounts t_j.

To apply Marxan, we divided the study area into square PUs sized 1km^2, to be consistent with the InVEST model outputs. The resulting planning unit layer comprises 483 PUs. The Marxan cost parameter assigns a numeric value to each PU to minimize overall costs while meeting conservation targets. Costs may represent any type of socioeconomic factors (e.g., costs of buying or leasing land, loss of revenue from putting an area into conservation, or proxy costs derived from human land or sea uses), or they may represent the area of the planning unit, thus promoting equal representation potential in the optimal solution. In this study, we applied two cost configurations to investigate their influence on spatial prioritization. Under the equal cost configuration, each PU was assigned a cost equal to its area (1 km^2), whereas under the variable cost configuration, each PU was assigned a

surrogate cost representing human activities and associated pressures. Consequently, the equal cost configuration allowed us to examine prioritization based on the spatial distribution and quantity of the index value of each ES, thus equal representation potential without differentiating among PUs. In contrast, the variable cost configuration incorporated spatial differences arising from the intensity of human pressures into the prioritization process (Figure 2). Although the analysis is applied within a PA network where management actions such as Natura 2000 designations or fishing prohibitions are active, some areas are influenced by human activities disproportionately despite their protection status. For example, wetlands are impacted by agricultural activity, runoff, and extensive infrastructure development; sandy beaches bear a heavy burden from human presence; and certain marine sites are influenced by shipping lanes and port activities. Cost types and scores were adapted from Leontiou et al. (2022) and Christodoulou et al. (2021) [34,47] and are indicated in Table S5 of the Supplementary Materials S1.

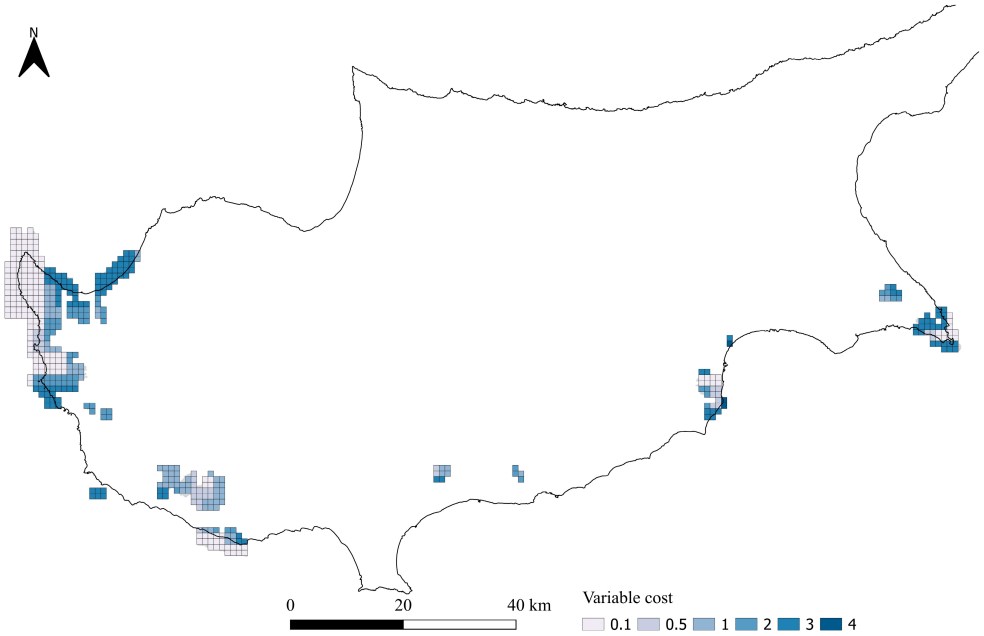

Figure 2. Spatial distribution of variable cost (for more details, see Leontiou et al., 2022 [34]).

The output index values of the InVEST CV, HQ and RT models were used as conservation features in the Marxan decision-support tool. In contrast to conventional Marxan applications where conservation features typically represent spatial distributions of species or habitats, the present analysis uses these index values to represent the spatial distribution and magnitude of ES supply [8,36]. The coastal exposure index has an opposite direction of interpretation to the other two indices. Therefore, to align the direction of interpretation across all three indices, the coastal exposure index was transformed prior to its incorporation into Marxan. Specifically, its reciprocal was calculated so that higher values represent greater relative biophysical coastal protection, whereas lower values represent lower relative biophysical coastal protection. Because all indices had different numerical ranges, they were normalized to a 0–1 scale prior to their integration into Marxan. This ensured that all features were represented on a common numerical scale.

Hereafter, the indices that were used as conservation features in Marxan are referred to as CP (coastal protection), HQ (habitat quality) and RT (recreation). For each feature, the Marxan representation target was expressed as a proportion of its total amount across all PUs. Thus, a representation target of 30%, for example, required the selected PUs to collectively represent at least 30% of the total amount of the respective index in the final solution.

In contrast to species and habitats, representation targets for ESs are inherently difficult to define and be meaningful [12] due to their complexity in the generation and distribution, as well as the absence of policy-driven targets on 'how much' to conserve. Therefore, different proportions of low and high representation targets were explored to identify values that yield spatially informative solutions. Very low targets (10–25%) prevented Marxan from exploring alternative sites to achieve an optimal solution, whereas very high targets (80–100%) resulted in the selection of almost all available areas. Low and high representation targets of 30% and 60%, respectively, were ultimately selected, as they demonstrated spatially sound patterns of priority areas for targeted ESs. Based on the above, five Marxan scenarios were developed with varying representation targets for each feature:

(a) Equal Low Targets scenario, where all ES indices were assigned an equal representation target of 30% of their total amount in the study area;
(b) Equal High Targets scenario, where all indices were assigned an equal representation target of 60%;
(c) Prioritizing biodiversity through habitat quality (HQ) scenario, where HQ was assigned a high representation target of 60% versus a target of 30% assigned to CP and RT;
(d) Prioritizing coastal protection (CP) scenario, where CP was assigned a high representation target of 60% versus 30% assigned to HQ and RT; and
(e) Prioritizing recreation (RT) scenario, where RT was assigned a high representation target of 60% versus 30% assigned to the HQ and CP.

All scenarios were run under both equal and variable costs. The Species Penalty Factor (SPF) in the Marxan equation is used to penalize Marxan solutions when targets are not met. In our analysis, SPF was set to 2 to ensure all targets were met. The Boundary Length Modifier (BLM) controls the compactness of the selected PUs, and it is an essential parameter for minimizing network fragmentation while balancing total area and costs. To identify the appropriate BLM value, we used the BLM calibration tool (QMarxan Toolbox), testing a range of values (0.001, 0.01, 0.1, 1, 10, and 100) to determine the optimal trade-off between total boundary length and solution cost. Based on calibration, the final BLM value of 0.1 was selected. The Marxan scenarios and parameter values are summarized in Table 3.

Table 3. Marxan scenarios and parameter values.

Scenarios	Representation Targets for ES			SPF	Runs	BLM
	HQ	CP	RT			
Equal Low Targets	30%	30%	30%	2	100	0.1
Equal High Targets	60%	60%	60%	2	100	0.1
Prioritizing HQ	60%	30%	30%	2	100	0.1
Prioritizing CP	30%	60%	30%	2	100	0.1
Prioritizing RT	30%	30%	60%	2	100	0.1

2.5. Comparison Between Scenarios

We statistically and descriptively compared the best solutions generated by Marxan across the different scenarios to examine potential spatial trade-offs and congruences, investigate spatial similarities among solutions obtained under alternative cost configurations and representation targets and investigate the allocation of priority areas across the terrestrial and marine realms. The analyses followed methodologies suggested by Harris et al. (2014) [66].

Initially, we calculated the representation achieved for all ESs across all best solution scenarios. The threshold used by Marxan to determine whether a feature successfully met its target was set to 95%. We then investigated the spatial similarity of the best solution networks identified by Marxan across all scenarios. A Jaccard resemblance matrix was constructed in

R, using the *vegdist* function from the vegan package. Subsequently, a non-metric multidimensional scaling (NMDS) ordination was generated to visually assess and interpret spatial resemblance among the alternative solutions. To further examine how prioritizing individual ESs influenced the selection of priority areas, we compared the best solution outputs from the service-targeted scenarios against the Equal Low Targets scenario, which served as the baseline (30% representation target for all ESs). Finally, we assessed how the spatial allocation of priority areas varied across realms by calculating the total number of selected PUs within each realm. PUs were classified as terrestrial, marine or interface (i.e., where a PU was intersected by the coastline). This classification allowed us to compare the proportional representation of each realm across the different optimized solutions. The overall workflow of the methodology is summarized in Figure 3, while definitions of key terms and descriptions of the models used throughout the Methods are provided in Appendix A.

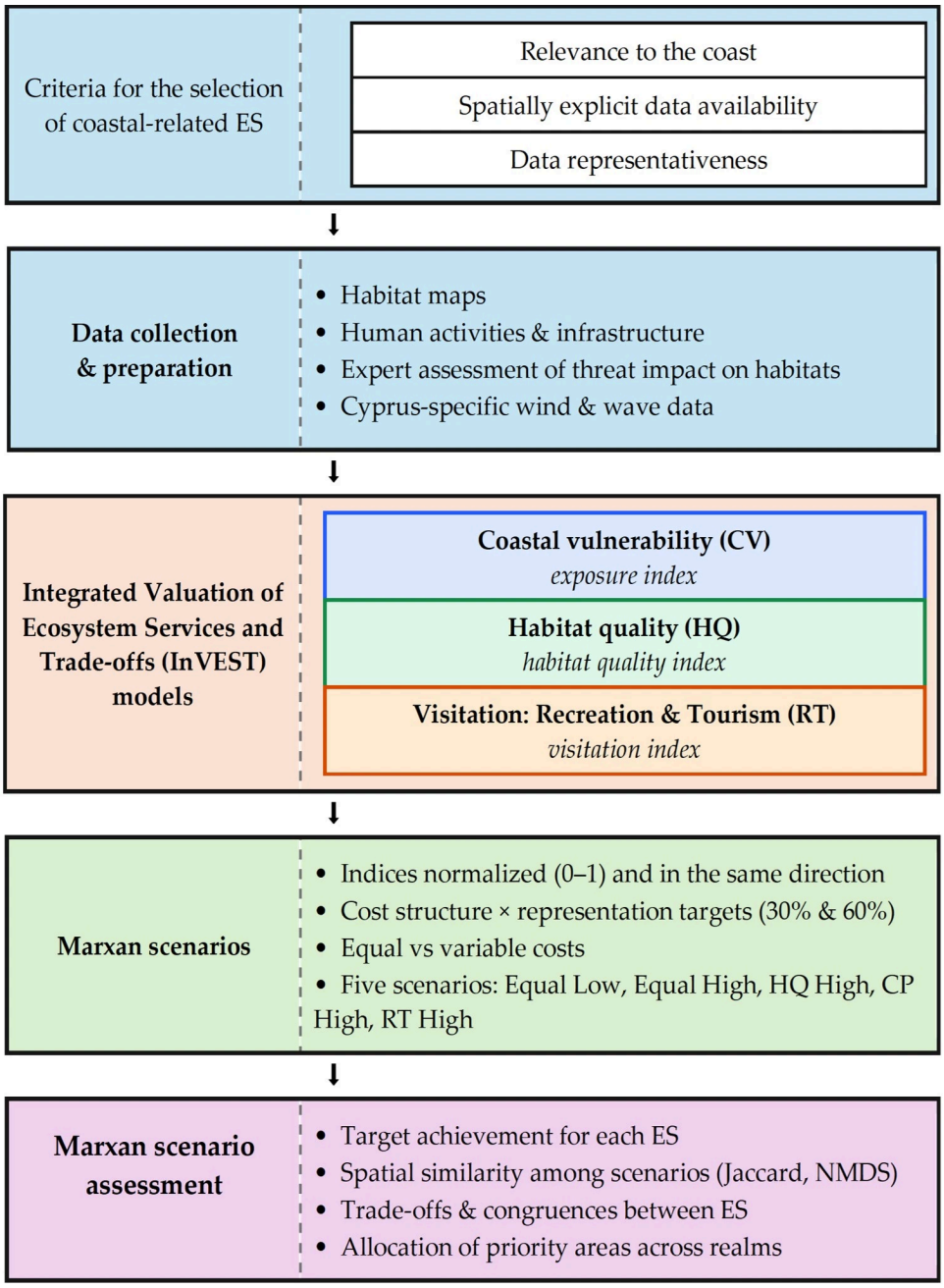

Figure 3. Overview of the methodological workflow for ES assessment and prioritization across realms.

3. Results

3.1. Coastal Vulnerability (CV) Model

The output of the CV model represents the raw coastal exposure index at each predefined coastline point (set to 1 km herein), calculated based on the Relative Exposure Index (REI) of specific bio-geophysical variables (see Equation (1)). Low index values indicate a low exposure of a coastal segment to storm-induced erosion and inundation, whereas high values indicate high exposure. Figure 4 shows the spatial distribution of the exposure index across the study area, with values ranging from 1.2 to 3.4. Index values greater than 3 occur predominantly within the Polis-Gialia PA (Figure 4a), located in the northwest part of the study area. The high exposure in this region is primarily driven by high REIs for surge, relief and geomorphology. Similarly, the Larnaca Salt Lakes complex (Figure 4d) exhibits high exposure index values, characterized by high geomorphology and relief REIs alongside an absence of protective marine habitats. In contrast, low to medium exposure values are observed in the Chersonisos Akamas PA to the west (Figure 4a), though certain segments display high exposure driven by high wind, wave and relief REIs. Low to medium exposure values are also found in the southwestern Akrotirio Aspro—Petra tou Romiou PA (Figure 4b). Here, low relief and favorable habitat REIs compensate for relatively high wind and wave REIs. Finally, the easternmost part of the study area, CapeCape Greco PA (Figure 4c), shows medium to low exposure index values, driven by low wind, wave and geomorphology REIs.

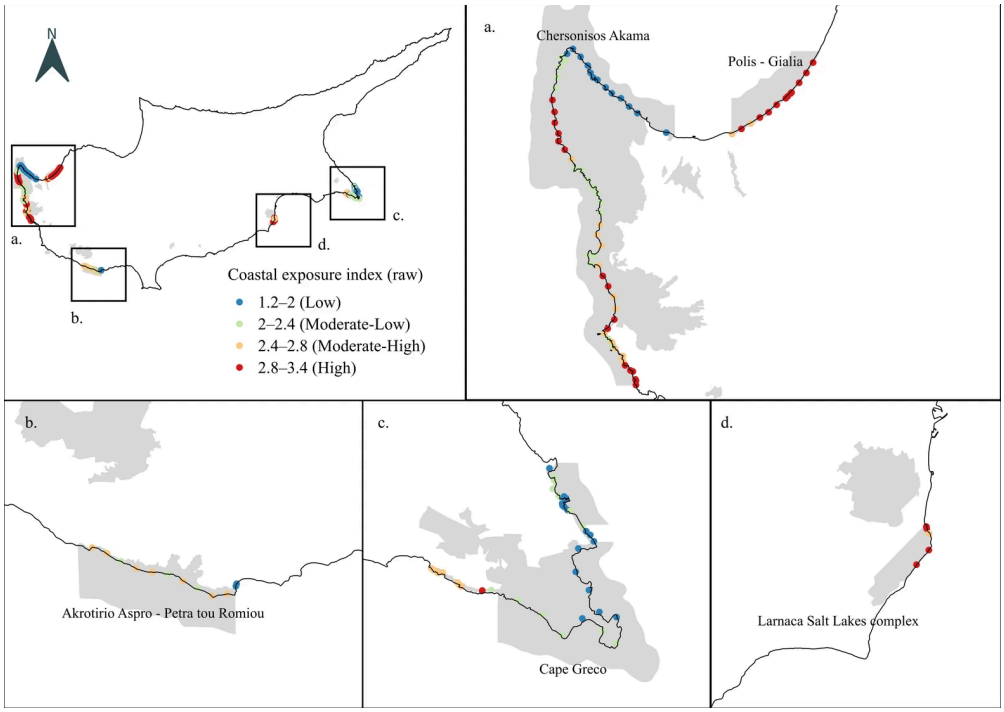

Figure 4. Spatial distribution of the raw coastal exposure index across the study area. Inset maps (**a**–**d**) highlight localized index values within the coastal PAs. Index values are grouped into four classes using quartile classification.

3.2. Habitat Quality (HQ) Model

The InVEST HQ model produces a continuous raster output representing habitat quality index values, with the spatial distribution across the study area illustrated in Figure 5. Low habitat quality values occur predominantly within terrestrial ecosystems, driven by land-based developments that are substantially more extensive than marine-based activities.

Notably, coastal halophytic habitats attained the lowest quality scores. For instance, three of the island's wetlands within the Natura 2000 network—Larnaca Salt Lakes complex, Oroklini Lake (Figure 5d), and Paralimni Lake (Figure 5c)—exhibit the lowest habitat quality values. Areas with sclerophyllous vegetation such as in Cha Potami, Koilada Diarizou and Xeros Potamos PAs (Figure 5b) demonstrate medium habitat quality (~0.5 index value) across nearly their entire distribution.

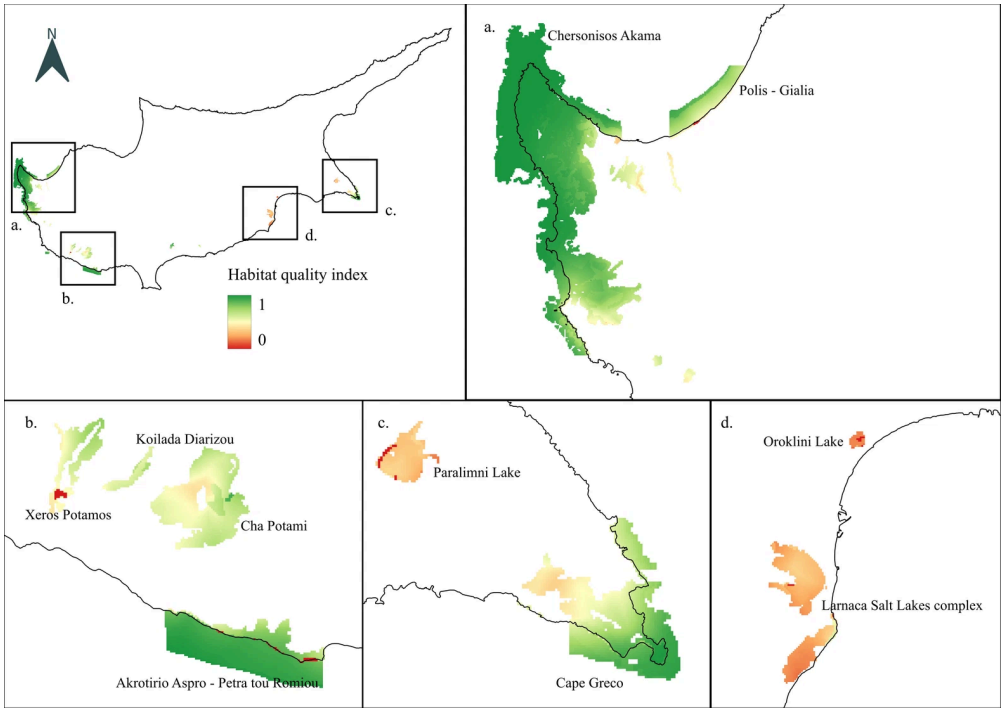

Figure 5. Spatial distribution of the habitat quality index across the study area. Inset maps (**a**–**d**) highlight localized index values within the coastal PA network. Index values are displayed along a continuous scale from 0 (lowest quality) to 1 (highest quality).

Conversely, marine habitats, predominantly *Posidonia oceanica*, show relatively high index values. For example, within the Chersonisos Akama PA (Figure 5a), marine habitat quality represents the highest values. Conversely, in Cape Greco PA (Figure 5c), lower index values are displayed, mirroring a similar pattern observed in the surrounding landscape.

The comparison of the mean HQ index for each habitat across the study area with the latest report for Article 17 of the Habitats Directive for the period 2019–2024 (EEC) showed that habitats classified as FV represented mean HQ values ranging from 0.18 (habitat 1420) to 0.98 (habitat 2250*). Habitats classified as U1 also ex exhibited a wide range, from 0.22 (habitat 1310) to 0.98 (habitat 2240), while a single habitat classified as U2 (habitat 2230) has a mean HQ value of 0.99 (Table 4).

In the field assessment by [61], the indicator for habitat 1110 was available for two sites: Thalassia Periochi Nisia and Chersonisos Akama. Both were classified as having Good or High ecological status, respectively, while their corresponding mean HQ values were 0.65 and 0.92. For habitat 1120*, no PREI values were available; however, all five sites assessed were classified as having High ecological status, with the corresponding HQ mean index values ranging from 0.71 to 0.96. For habitat 1170, EEI-c index values were reported for six sites, ranging from 0.876 to 0.965, while the corresponding HQ values ranged from 0.52 to 0.94 (Table 5).

Table 4. Mean Habitat Quality (HQ) index values for the habitat types included in the analysis and their corresponding conservation status under Article 17 of the EU Habitats Directive (2019–2024 reporting period) for Cyprus. Habitat codes marked with an asterisk (*) denote priority habitat types under Annex I. Code 5212 represents the Cyprus-specific subdivision of Annex I habitat type 5210 (Arborescent matorral with *Juniperus* spp.); its conservation status is therefore reported under the corresponding Annex I habitat type. Categories such as CY04 and CY05 are not Annex I habitat types and were therefore excluded from the comparison with Article 17 conservation status. FV = Favorable; U1 = Unfavorable–Inadequate; U2 = Unfavorable–Bad.

Habitat Code	HQ Mean Index Value (0–1)	Article 17 Conservation Status (2019–2024)
1110	0.85	FV
1120 *	0.93	FV
1150 *	0.23	FV
1170	0.91	FV
1210	0.74	U1
1240	0.76	FV
1310	0.22	U1
1410	0.26	U1
1420	0.18	FV
2110	0.82	U1
2230	0.99	U2
2240	0.98	U1
2250 *	0.98	FV
3140	0.61	FV
3170 *	0.70	FV
5212	0.88	FV
5220 *	0.94	U1
5330	0.89	FV
5420	0.68	FV
6220 *	0.72	U1
8210	0.85	FV
8310	0.94	FV
8330	0.79	FV
9290	0.64	FV
9320	0.70	FV
9540	0.90	FV
62B0	0.89	FV
92C0	0.41	U1
92D0	0.23	U1

Table 5. Comparison of modelled Habitat Quality (HQ) index values for three coastal marine habitat types within Natura 2000 sites with field-based ecological indicators reported by [61]. The comparison includes CymoSkew EQR for habitat 1110 (Sandbanks associated with *Cymodocea nodosa* meadows), PREI for habitat 1120* (*Posidonia oceanica* beds), and EEI-c for habitat 1170 (Reefs). Field-based assessments were available for selected habitat–site combinations only. HQ = Habitat Quality; N2K = Natura 2000; EQR = Ecological Quality Ratio; PREI = Posidonia oceanica Rapid Easy Index; EEI-c = Ecological Evaluation Index continuous (For details about the indices, see [61]).

N2K Site	1110 HQ Mean Index Value	CymoSkew EQR Index	Ecological Status	1120* HQ Mean Index Value	PREI Status	1170 HQ Mean Index Value	EEI-c Index
Cape Greko (CY3000005)	0.77	–	–	0.85	High	0.81	0.94
Thalassia Periochi Nisia (CY3000006)	0.65	0.51	Good	0.71	High	0.65	0.97
Polis Gialia (CY4000001)	0.67	–	–	0.83	High	0.52	0.91
Thalassia Periochi Moulia (CY4000006)	0.85	–	–	0.91	High	0.91	0.91
Chersonisos Akama (CY4000010)	0.92	0.79	High	0.96	High	0.94	0.94
Akrotirio Aspro–Petra Romiou (CY5000005)	0.92	–	–	0.91	–	0.80	0.88

3.3. Visitation: Recreation and Tourism (RT) Model

The output of the RT model represents the spatial distribution of the visitation index across the study area, reflecting the influence of the predictors used in the regression analysis. Figure 6 represents the spatial patterns of the index. The highest concentrations of tourist visitation occur at the Larnaca Salt Lakes complex (Figure 6d), Cape Greco (Figure 6c) and Chersonisos Akama (Figure 6a).

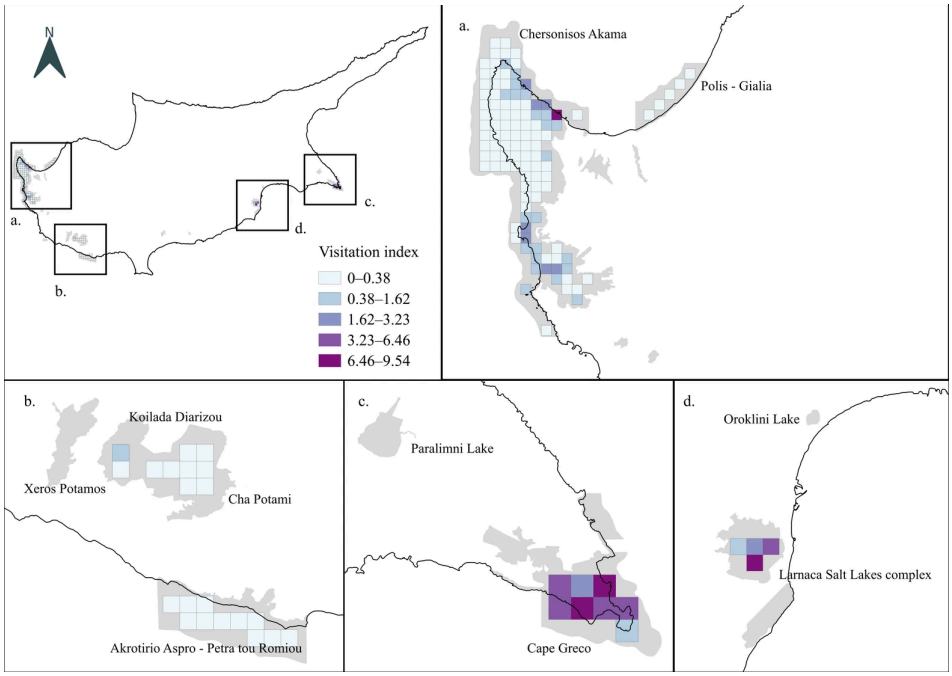

Figure 6. Spatial distribution of the visitation index across the study area. Inset maps (**a**–**d**) highlight localized index values within the coastal PA network. Index values are classified using Natural breaks (Jenks).

Regression results (Table S6 of the Supplementary Materials S1) indicate that the presence of natural trails, secondary roads, beaches, and monuments significantly and positively influence an individual's choice of recreation site. An increase in these variables is associated with a higher visitation index. In contrast, mean elevation and distance from tourist accommodation (e.g., houses or hotels) exert a significant negative influence. Non-significant predictors from the initial model were removed, and the analysis was re-run using only significant variables to improve model performance and statistical reliability.

3.4. Marxan Scenarios Assessment

Spatial prioritization results were evaluated using the Marxan best solution file, which shows the optimal priority network required to meet representation targets at minimum cost and boundary length. In addition, the Marxan selection frequency file was used to assess PU irreplaceability. Selection frequency represents the number of times a given PU is selected for inclusion in the optimal network across all model runs. For instance, PUs selected in more than 80 of the 100 model runs demonstrate high irreplaceability and were therefore prioritized for network inclusion.

Table 6 summarizes the main results obtained from the best solution files across all scenarios. The optimal priority network increased in size as representation targets increased (e.g., Equal Low Target vs. Equal High Target scenarios). When targeting specific ESs, particularly HQ, the size of the priority network increases substantially compared to the other scenarios. A similar trend was observed for boundary length, where the Equal High Target and Prioritizing HQ scenarios yielded the longest boundary lengths under both cost configurations.

Table 6. Main Marxan output results from the best solution and selection frequency files for each scenario.

Scenario	Equal Cost					Variable Cost				
	Equal Low	Equal High	HQ High	CP High	RT High	Equal Low	Equal High	HQ High	CP High	RT High
Priority area network (km^2)	85	180	176	86	92	91	195	186	93	103
Boundary length (km)	72	126	134	88	80	56	158	142	80	86
Percentage of irreplaceable PUs	32%	87%	87%	47%	41%	85%	99%	100%	85%	83%

The spatial variability of priority areas across the different scenarios is shown in Figure 7. Under the equal cost configuration, the selection of priority areas is distributed among three PAs: Chersonisos Akama and Cape Greco (included across scenarios 1–5), and Akrotirio Aspro-Petra tou Romiou (included in scenarios 2 and 3). The priority networks generated under the variable cost configuration represented a similar spatial configuration, though additional sites were included depending on the scenario.

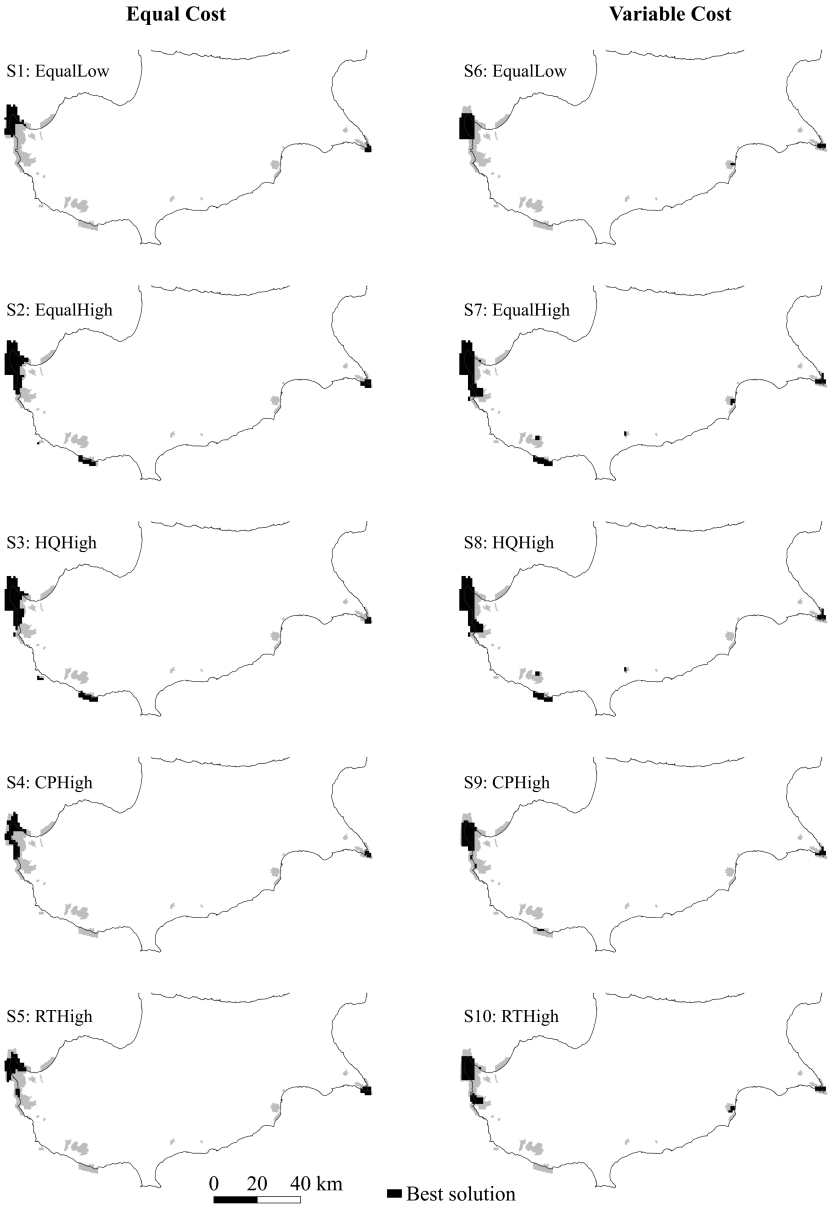

Figure 7. Best solution for each scenario under equal-cost and variable-cost configurations.

Figure 8 represents the selection frequencies and subsequent PU irreplaceability across all scenarios. The proportion of irreplaceable PUs in each solution is notably high, especially under variable cost. For instance, targeting HQ under variable cost yields 100% of irreplaceable PUs included in the solution.

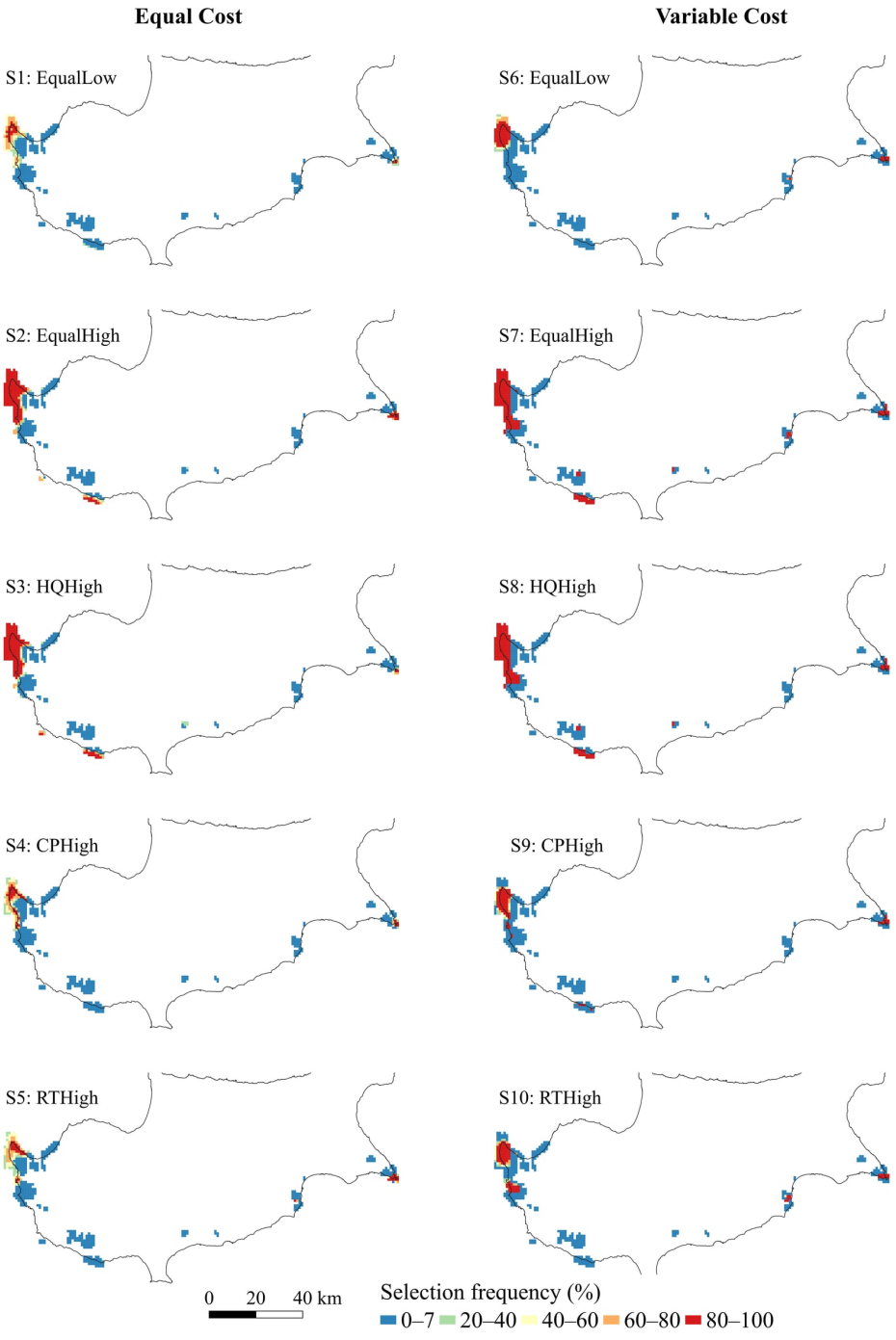

Figure 8. Selection frequency for each scenario under equal-cost and variable-cost configurations.

Notable spatial differences between the two cost configurations were observed in the RT High scenario (Figure 9). Under equal cost, priority sites are concentrated in two PAs, Chersonisos Akama and Cape Greco. Under variable cost, priority sites are still concentrated in the same PAs with the addition of Larnaca Salt Lakes.

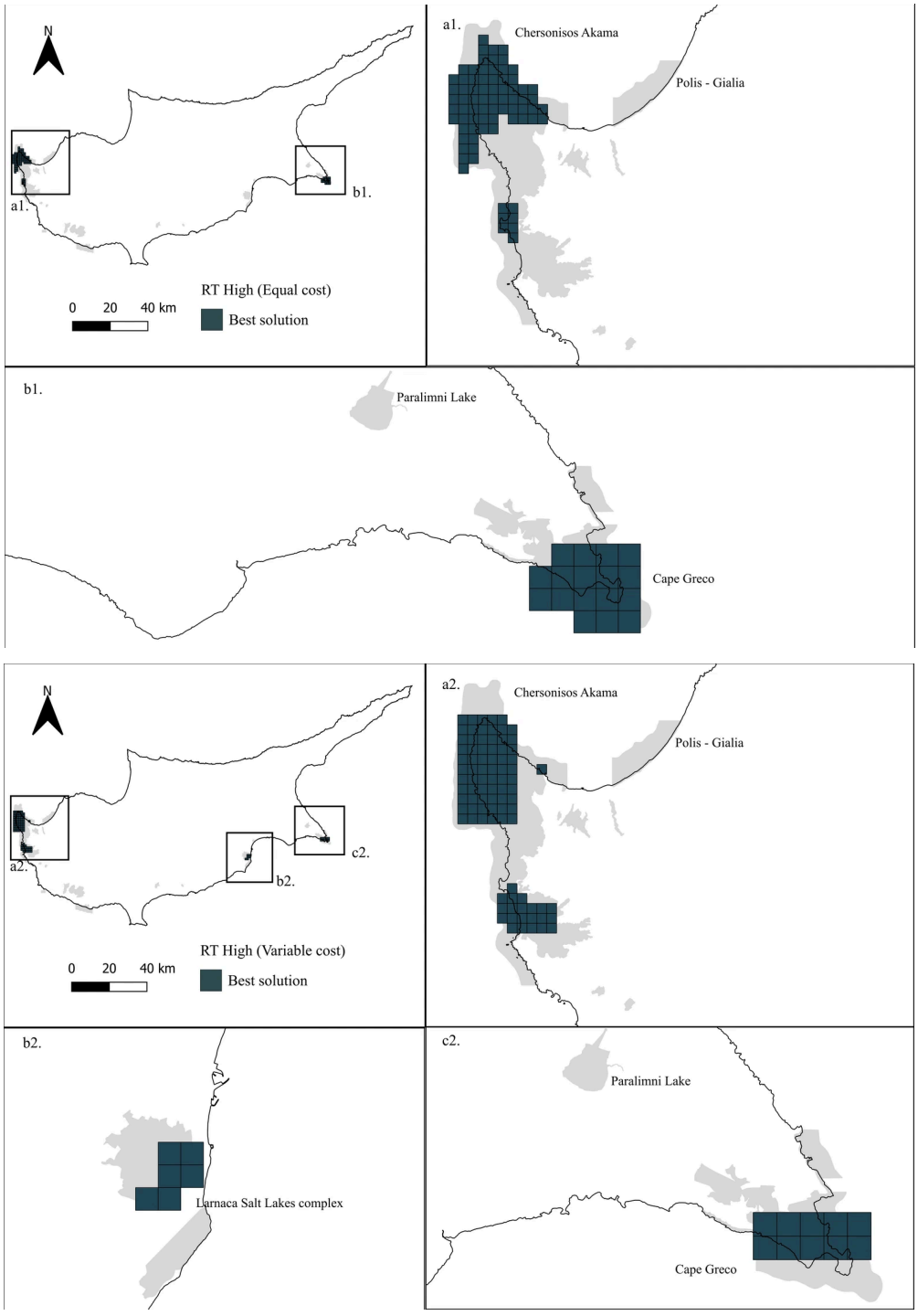

Figure 9. Best solutions for the RT High scenario under equal cost (**a1**,**b1**) and variable cost (**a2**–**c2**) configurations.

Selection frequency under equal cost represents greater variability compared to variable cost, especially in the PA of Chersonisos Akama (Figure 10). Highly selected PUs (>80%) are mainly located along the coastline towards the sea, under equal cost, whereas under variable cost the highly selected PUs dominate the terrestrial realm.

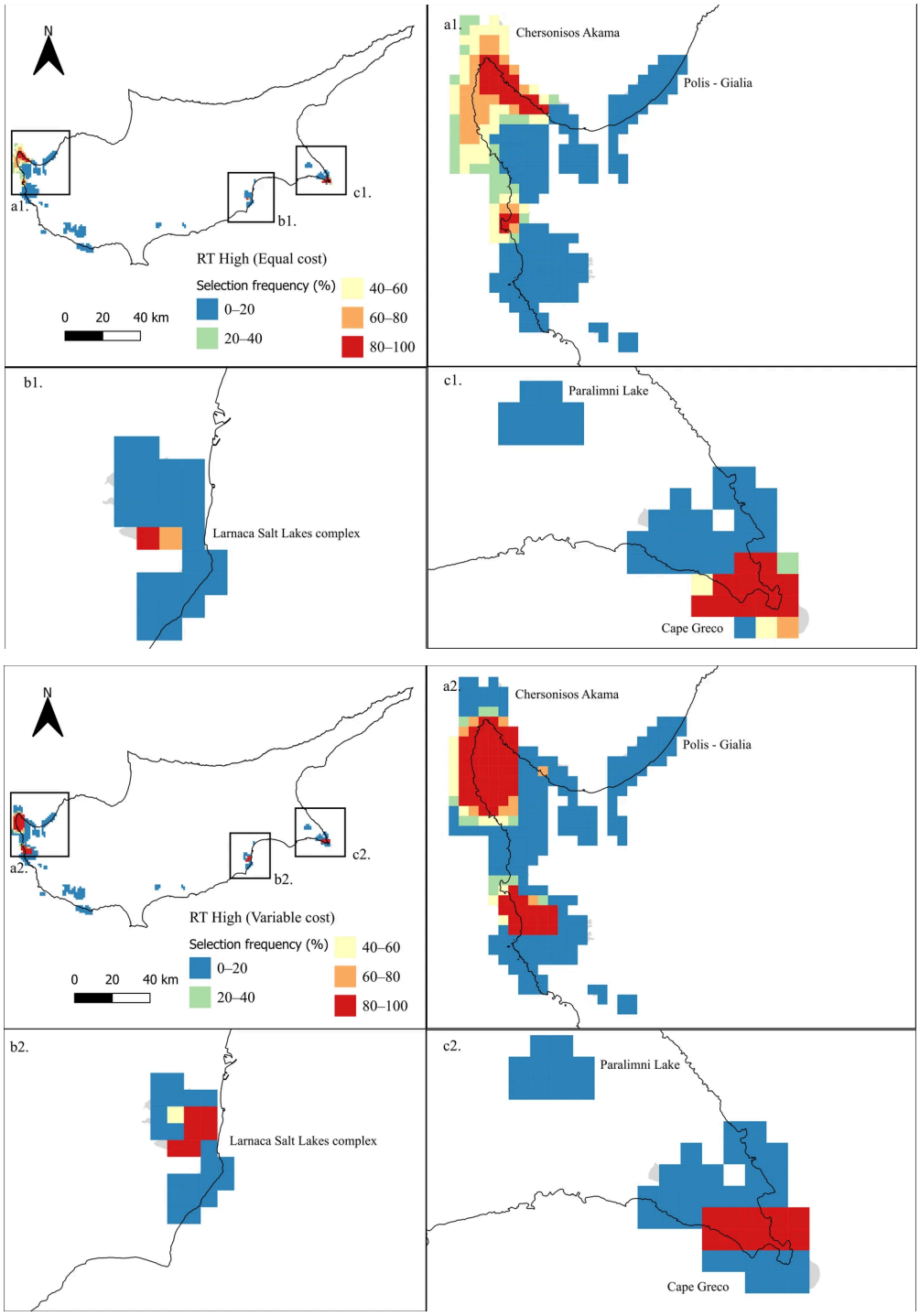

Figure 10. Selection frequency for the RT High scenario under equal cost (**a1–c1**) and variable cost (**a2–c2**) configurations.

3.4.1. Comparison Between Target Representation Scenarios

Representation targets for all ESs were met in all scenarios (Figure 11). HQ and RT were close to their target in every scenario, except RT in the HQ High scenario under variable cost, where it exceeded its representation target by ~10%. In contrast, CP exceeded its target in most scenarios, under both cost configurations.

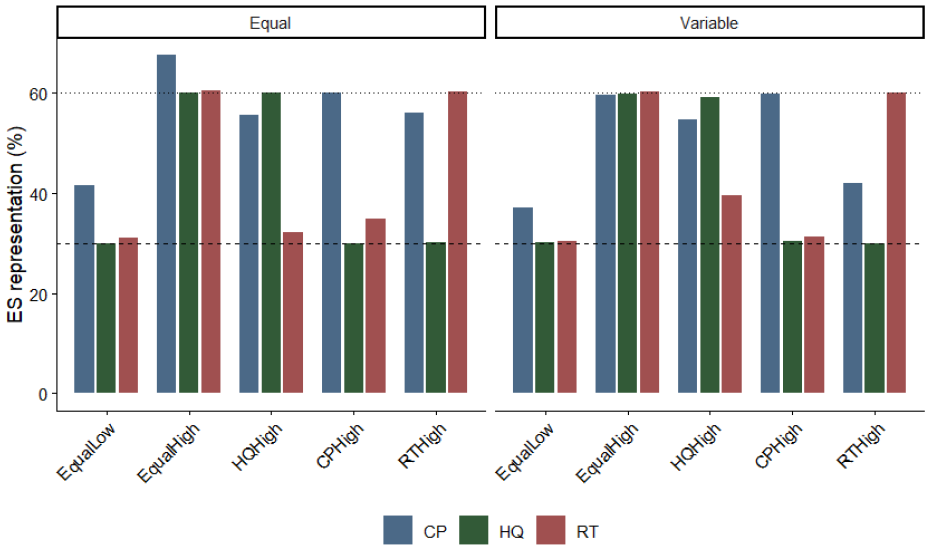

Figure 11. Percentage of the total amount of each feature (HQ, CP, and RT) represented in the best solution file under each target and cost scenario. Results are shown separately for equal and variable costs. Dashed and dotted horizontal lines indicate the 30% and 60% target levels, respectively.

3.4.2. Spatial Similarity Among Scenarios

The non-metric multidimensional scaling (NMDS) ordination of the 10 best solutions based on Jaccard dissimilarity of planning-unit selection is represented in Figure 12. Scenarios positioned close to each other represent similar spatial configurations, whereas greater distances between scenarios indicate greater differences in their spatial configuration. The stress value to assess the goodness of fit of the NMDS ordination was 0.086, indicating a good representation of the ordination of the Jaccard matrix in the 2-D space.

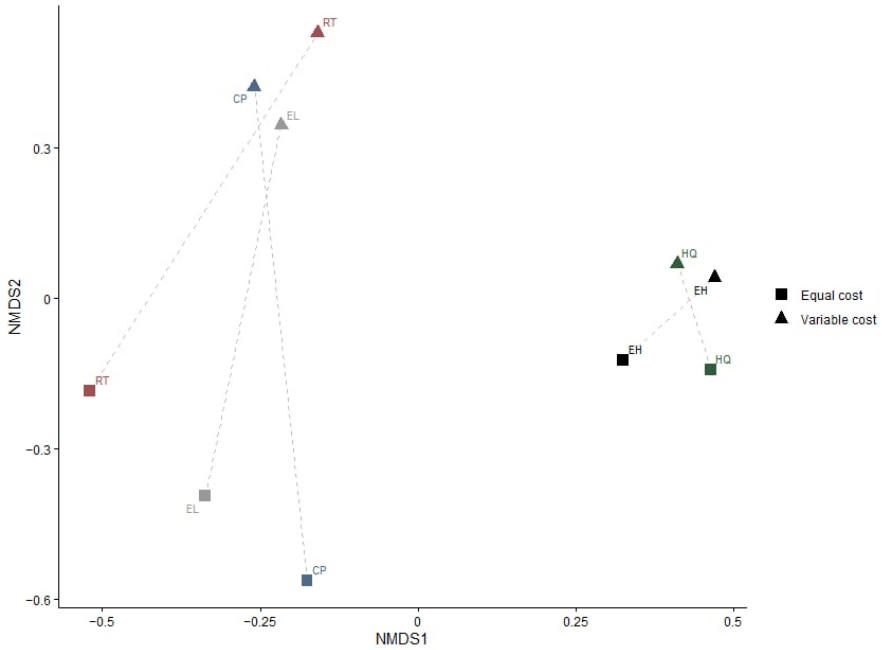

Figure 12. Non-metric multidimensional scaling (NMDS) ordination of the 10 best solutions based on Jaccard dissimilarity of planning-unit selection. Each point represents one target/cost scenario; squares indicate equal-cost scenarios and triangles indicate variable-cost scenarios. Grey dashed lines connect to the same target scenario under the two cost configurations. To save space, scenarios are abbreviated as RT = RT High; CP = CP High; HQ = HQ High; EL = Equal Low; and EH = Equal High.

The NMDS ordination showed that the spatial configuration of the best solution for each scenario was influenced by both the cost configuration and representation targets. The HQ High (green) and the Equal High (black) scenarios showed high spatial similarity and formed a distinct cluster separate from the other configurations, reflecting the difference in the number of selected areas required under those scenarios to achieve the set representation targets. They represented a high Jaccard similarity of 0.85 under equal cost and 0.94 under variable cost, indicating a small influence of the cost configurations. In contrast, the CP High (blue), RT High (red), and Equal Low (grey) scenarios showed a greater variation in spatial overlap between the cost configurations. Under equal cost, Jaccard similarity values ranged from 0.47 to 0.53, and under variable cost, Jaccard similarity increased, with values ranging from 0.47 up to 0.69.

3.4.3. Spatial Trade-Offs and Congruences Among ESs

Spatial trade-offs and congruences derived from prioritizing service-targeted scenarios were quantified relative to the Equal Low baseline scenario, where each feature was represented at 30% of its total amount (Figure 13). Prioritizing HQ increased the representation of CP across both cost configurations, and for RT, particularly under the variable cost configuration. Conversely, prioritizing CP exerted a small impact on HQ regardless of the cost configuration. Prioritizing RT resulted in a minor effect on HQ under equal costs, shifting to a minor trade-off under variable costs. Finally, prioritizing RT and CP resulted in mutual increases in the representation of CP and RT, respectively, with these increases being more pronounced under the equal cost configuration.

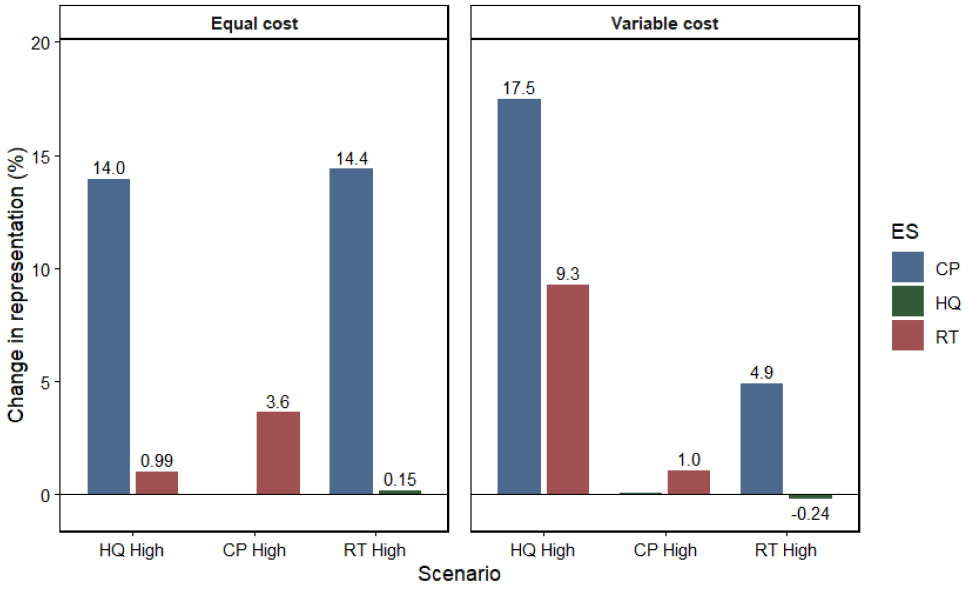

Figure 13. Spatial trade-offs and congruences among ESs. Change in the percentage representation of HQ, CP, and RT relative to the Equal Low target scenario under equal and variable cost. Positive values indicate an increase in representation, whereas negative values indicate a decrease.

3.4.4. Allocation of Priority Areas Across Realms

The investigation of spatial trade-offs across realms revealed a marine-dominated allocation under equal cost, except in CP High and RT High scenarios where the allocation of priority areas is higher along the interface. Under variable cost, priority areas are allocated more in the terrestrial realm, especially in the RT High scenario.

Under variable cost, marine realm representation of priority areas was higher only under Equal High and HQ High scenarios. The terrestrial realm was predominantly repre-

sented in the RT High scenario, while priority areas along the interface were more prevalent only in the CP High scenario (Figure 14).

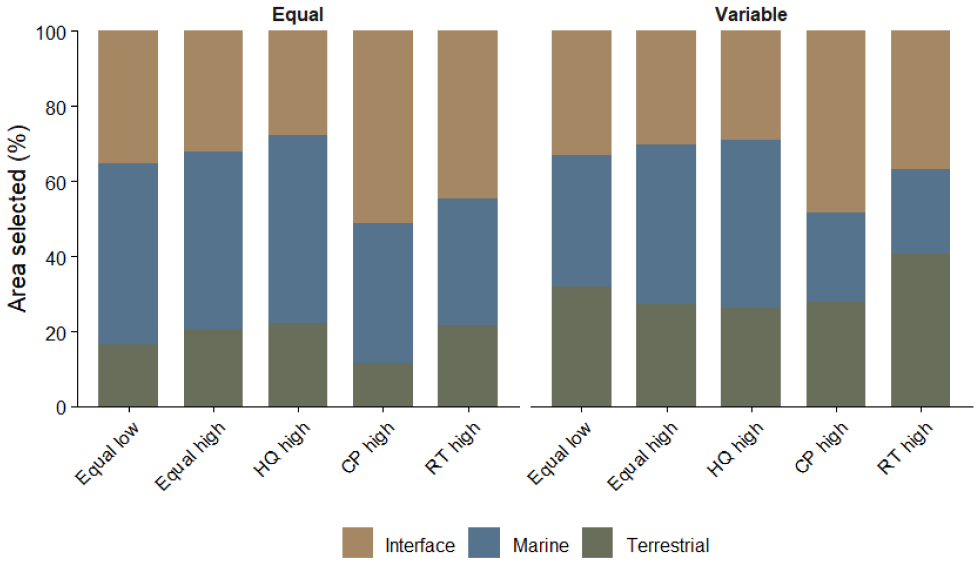

Figure 14. Allocation of priority sites across realms.

4. Discussion

This study demonstrates the integration of ES priorities into systematic conservation planning within an established coastal PA network using a cross-realm approach. We mapped and assessed three coastal-related ESs by integrating information from terrestrial and marine realms and explored how different representation targets and cost assumptions affect the resulting spatial configuration of priority areas. We compared equal-cost (i.e., area-based) and variable-cost approaches to distinguish spatial patterns driven by ES representation targets from those influenced by the distribution of human uses and management-related costs [67], while alternative target scenarios were used to examine potential trade-offs and congruences among ESs.

4.1. Comparison of HQ Index with Available Independent Habitats Assessment

The semi-quantitative comparison of the modelled HQ values with the conservation status of Annex I habitats reported under Article 17 of the Habitats Directive showed no consistent correspondence between the two assessments. Habitats classified as Favorable (FV) exhibited mean HQ values ranging from low to high, and similarly, habitats with less favorable conservation status did not consistently exhibit lower HQ values. This may reflect differences in the objectives and spatial scales of the two assessments. Article 17 reporting integrates information on each habitat across its entire distribution within the Natura 2000 network, as well as information on conservation measures targeting that habitat. In contrast, our study considered only PAs falling within the ICZM boundary. Consequently, discrepancies may partly reflect the limited representation of certain habitats within our study area.

In contrast, the comparison with the ecological assessments reported by [61] provided a more spatially specific reference for selected coastal marine habitats within the N2K sites. The semi-quantitative comparison indicated some correspondence between the field-based assessments and the modelled HQ values, particularly for habitats 1110 (Sandbanks) and 1120* (*P. oceanica*), across the overlapping sites; however, this was not equally evident for habitat 1170 (Reefs). These comparisons nevertheless provide complementary information

for interpreting the spatial patterns represented by the modelled HQ index and highlight the value of site-level ecological assessments when examining habitat condition and identifying conservation priorities.

A comparable assessment against independent reference data was not possible for the other two model outputs, as equivalent spatially explicit assessments were not identified for the study area.

4.2. Comparison of Marxan Scenarios

Ten Marxan scenarios were run using two cost configurations and five combinations of ES representation targets. Across all scenarios, representation targets for all ESs were met. Results from the best solutions identified areas important for the provision of the targeted ESs within a network primarily designated for the protection of species and habitats. However, depending on which service was prioritized, spatial configurations differed, implying alternative management outcomes. For example, prioritizing recreation (RT), especially along the coastline, raises implications for the effective management and protection of certain species and habitats distributed solely therein (e.g., sand dunes/sea turtle nesting beaches).

Differences between the two cost configurations were particularly highlighted in scenarios with high HQ representation targets (Equal High and HQ High). While boundary length was consistently lower under variable cost than under equal cost in the other scenarios, this was the opposite in the high HQ scenarios. Marxan's ability to prioritize areas that are well-connected was constrained in these scenarios, resulting in a sparser solution [68]. Selection frequency was consistently higher under variable cost scenarios, with 83–100% of PUs across the scenarios exhibiting selection frequencies above 90%, indicating a persistent selection of the same PUs across Marxan runs. This consistency in site selection suggests that incorporating variable costs, which, in our case, represented human activities and associated pressures, influenced not only the spatial allocation of conservation priorities but also the consistency with which these areas were selected. Our finding aligns with previous comparative studies examining the use of socioeconomic proxy costs [69,70] and underscores the importance of selecting appropriate cost criteria when identifying conservation priorities [71].

The comparison of Marxan scenarios showed the dependency of the spatial configuration of priority sites on both the representation targets and the cost configurations. This dependency is consistent with previous studies showing that the integration of ESs into SCP can modify conservation priorities and associated costs considerably, as a reflection of the nature of a service and of the use of proxy costs [72]. The prioritization of CP and RT services resulted in smaller networks of priority areas, compared to the HQ priority scenario, as a reflection of the spatial differences in the distribution of the services in the study area. Coastal protection is spatially distributed along the coastline, whereas biodiversity and recreation are represented across a broader part of the study area. Consequently, the selection of a larger area to achieve a given representation target increases the extent of the priority network but also partly reflects the increase in the representation of other ESs without necessarily indicating a strong spatial congruence. This pattern mirrors the findings of Chan et al. (2006) [8] and Arkema et al. (2015) [73], who distinguished spatially constrained services versus widely distributed ones, and human-demanded services versus supportive ones, respectively. Nevertheless, our combined approach with the use of proxy costs and single-targeted prioritization scenarios allowed the identification of priority sites that are relevant for a particular ES despite making a comparatively limited contribution to other services [74].

Importantly, our results further enhance and highlight the value of cross-realm approaches when prioritizing ESs in the coastal zone [35,36,42]. Despite cost configuration and target representation for each ES, the distribution of high-selection frequency priority areas (>80% selection frequency) repeatedly concentrated within PAs encompassing substantial terrestrial and marine components, particularly Chersonisos Akama and Cape Greco. Their repetitive representation across all ten Marxan scenarios suggests that these locations extend beyond individual service supply, thus supporting combined priorities [8,75], and reinforcing the relevance of a cross-realm conservation planning framework [34,76,77].

5. Conclusions

This study contributes to the growing field of cross-realm ecosystem services prioritization, offering insights for more integrated and effective conservation planning and management in island contexts. Spatially explicit, multi-objective and cross-realm SCP can support the integration of ESs in PA management. The ability of SCP to spatially identify congruences and trade-offs, as well as to evaluate management plans and actions, can contribute to ecosystem-based management and provide a foundation for advancing Integrated Coastal Zone Management (ICZM). Such an approach is particularly relevant to Mediterranean islands where physical limitations and heavy reliance on ESs threaten both the biodiversity and ES provision.

Several limitations occur within this study and should be considered when interpreting the results. First, expert assessments were not available for all habitat types included in the HQ model. In particular, gaps in expert knowledge concerning natural grasslands and forests required the use of information from published literature, baseline InVEST data and, in some instances, standardized scores shared among habitats within the same aggregated classification. These approaches enabled the inclusion of a broader range of habitats but introduced uncertainty into their parameterization. Second, spatially explicit data were unavailable for certain recreational activities and associated pressures within coastal PAs. These included anchoring by privately owned recreational boats, which can affect marine habitats, especially *P. oceanica* meadows (habitat 1120*) and off-road recreational vehicle use (ATVs, quad bikes and buggies), particularly in Chersonisos Akama and Cape Greco. Consequently, these activities could not be incorporated into the spatial assessment of habitat quality. Third, no independent field verification or external statistical validation of the HQ index was undertaken in the study area, and no sensitivity analysis was conducted to examine the influence of variations in threat weights, maximum impact distances, decay functions and habitat sensitivities on the resulting HQ estimates. Previous studies have highlighted the sensitivity of InVEST HQ estimates to input data and parameter specifications [78,79], reinforcing the need to examine this limitation. The comparisons with Article 17 conservation status and the field-based assessments by [61] provided complementary information but were semi-quantitative and did not constitute validation of the index. Finally, the analysis was restricted to the PA network because standardized, high-resolution spatial information on habitat distribution was not consistently available outside it, preventing an equivalent island-wide assessment.

These limitations identify several priorities for future research. More comprehensive expert assessments and improved habitat-specific parameterisation would strengthen the representation of habitat sensitivities in the HQ model, particularly for natural grasslands and forests. The collection and spatial mapping of recreational boat anchoring and off-road vehicle activities would allow these pressures to be incorporated into future habitat quality assessments. Further work should also include sensitivity analyses of HQ model parameters and independent field-based assessments to examine the robustness and reliability of the modelled estimates. In particular, additional site-level habitat assessments using consistent

ecological indicators could provide a stronger basis for comparison with modelled outputs. Finally, the development of standardised, high-resolution habitat maps beyond the existing PA network would enable the extension of this approach to the wider coastal zone and support island-wide assessments of ES priorities.

PAs are currently lagging behind in adopting the ES concept [80]. Our findings highlight the importance of considering how different ESs and biodiversity priorities can be accommodated within existing PAs. Multi-zone planning approaches, such as Marxan with Zones, could be used in future applications to explore alternative spatial allocations of conservation objectives and differentiated management strategies [36,81], particularly in sites where competing priorities overlap. In addition, the difficulty of defining meaningful ES representation targets, in the absence of established policy-driven targets, has been recognised in previous research [12]. Our results reinforce the need for clearer guidelines and criteria for defining appropriate representation targets for ESs. Developing such criteria would provide a more transparent basis for incorporating ESs into conservation planning and supporting integrated management, particularly in Mediterranean island contexts.

With regard to the application of the analysis in a PA network, while an island-wide assessment would offer a broader geographical perspective, the lack of standardized, high-resolution habitat mapping outside the PA network prevents the application of such analyses at that scale. The study focuses intentionally on the PA network, which provides representative data necessary for ES assessment. Additionally, the evaluation of ESs within a designated PA network is vital for conservation and systematic planning. Our results critically demonstrate that legal designation alone does not guarantee ecological integrity or balanced management. Furthermore, this analysis provides an essential tool for PA managers to prioritize conservation efforts that focus on ecological integrity rather than solely species-targeted actions.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/land15112237/s1, Supplementary Materials S1 Tables S1–S6; Supplementary Materials S2 Elicitation Methodology and Assessment Scores for the HQ model (Excel file).

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Abbreviations

The following abbreviations are used in this manuscript:

ESs	Ecosystem services
SCP	Systematic Conservation Planning
CICESs	Common International Classification of Ecosystem Services
InVEST	Integrated Valuation of Ecosystem Services and Trade-offs
HQ	Habitat Quality
CV	Coastal Vulnerability
RT	Recreation and Tourism
CP	Coastal Protection
PA	Protected Area
PU	Planning unit
BLM	Boundary Length Modifier
SPF	Species Penalty Factor
ICZM	Integrated Coastal Zone Management
EU	European Union

Appendix A. Glossary

Ecosystem Services (ES): The benefits (and occasionally disbenefits or losses) that people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as flood and disease control; and cultural services such as recreation, ethical and spiritual, educational, and sense of place.

Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) is a suite of open-source software models for mapping and valuing ecosystem services.

Habitat Quality model (HQ): The InVEST Habitat Quality model uses habitat quality as a proxy to represent the biodiversity of a landscape, estimating the extent of habitat and vegetation types across a landscape, and their state of degradation.

Coastal Vulnerability model (CV): The InVEST Coastal Vulnerability model uses geophysical and natural habitat characteristics of coastal landscapes to compare their exposure to erosion and flooding in severe weather.

Visitation: Recreation and Tourism model (RT): The InVEST recreation model predicts the spread of person-days of recreation, based on the locations of natural habitats and other features that factor into people's decisions about where to recreate.

Marxan: Marxan is a decision support tool, designed to provide support to decision-makers in finding conservation solutions and other spatial planning problems. It produces a range of potential reserve configurations that meet the conservation objectives and consider all possible constraints such as cost, existing reserves, or other land uses. The main outputs of a Marxan analysis are the best solution and summed (i.e., selection frequency) files.

Planning unit: Spatial units within the study area, which can be defined as regular shapes (squares or hexagons) or irregular landscape-based features (e.g., river sub-catchments).

Objective function: the mathematical formulation of the minimum set problem, according to which Marxan analysis allocates a score (value) for every reserve configuration.

Marxan Solution: a binary output of Marxan indicating whether a planning unit is selected (1) or not selected (0) as part of the reserve system.

Best solution: one of the main output files of Marxan. Best solution is the run that comprises the selection of PUs with the smallest objective function score.

Summed solution (or selection frequency): the selection frequency of each planning unit, used as a surrogate of irreplaceability.

Cost (uniform, variable): a function Marxan acts to minimize in its search for achieving the representation targets. It reflects the costs of including a PU in the reserve. Costs are flexible and often relate to socioeconomic implications of establishing a new reserve (e.g., management costs; land acquisition costs).

Boundary Length Modifier (BLM): an optional parameter used in Marxan analysis to control the level of fragmentation in the reserve system.

Species Penalty Factor (SPF): the penalty given to a reserve system for not adequately representing conservation features.

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