



Research article

Potential feeding sites for seabirds and marine mammals reveal large overlap with offshore wind energy development worldwide

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ABSTRACT

Offshore wind energy is experiencing accelerated growth worldwide to support global net zero ambitions. To ensure responsible development and to protect the natural environment, it is essential to understand and mitigate the potential impacts on wildlife, particularly on seabirds and marine mammals. However, fully understanding the effects of offshore wind energy production requires characterising its global geographic occurrence and its potential overlap with marine species. This study aims to generate risk maps of interaction between offshore and seabirds and marine mammals based on the distribution of their potential foraging areas. These maps will allow visualisation of the spatial occurrence of risk and its severity for both groups. To achieve it, we built a structural equation model of three levels (plankton, fish, and top predators) to predict small-ranged seabirds and marine mammal spatial richness as a proxy of potential feeding sites. Later, we overlapped these maps with global wind density (as a proxy of potential offshore development areas) to identify risk areas. Our results pointed to simplified trophic chain models that effectively explained the richness of small-ranged seabirds and marine mammals. Our risk maps reveal a high overlap with potential offshore wind development. Low-risk areas were located mainly in so-called Global North countries, suggesting vast knowledge gaps and potential hidden risks in these areas. Importantly, the highest risk values were found outside the Marine Protected Areas for both groups, underscoring the necessity for strategic planning and the expansion of renewable energy sources to avert potential conservation challenges in the future.

1. Introduction

The imperative need to reduce CO₂ emissions has propelled the strategic rise of renewable energies onto the global economic and political agenda (European Commission, 2021; Congress.gov, 2020; SCIJ, 2021; IEA, 2021). As concern grows over the challenges posed by climate change to energy supply and production costs (Díaz and Guedes Soares, 2020), renewable energies sources — such as solar, wind, tidal, hydraulic, and geothermal — have been steadily growing as the offer more sustainable and lower emissions production (Ellabban et al., 2014). Among these, wind energy emerges as one of the most cost-effective options due to its versatile locations and vast global wind potential (Veers et al., 2019). In fact, offshore wind energy development

projects have increased 9% worldwide (Lee and Zhao, 2022) and are projected to generate 140.8 GW in Europe, 136.3 GW in Asia, and 49.5 GW in North America by 2030 (Williams and Lee, 2022).

Due to the rise of offshore wind energy over the last decade, there is a growing interest in its impact on marine ecosystems and the wildlife they harbor (Bergström et al., 2014). Various research studies support the notion that these structures can have positive effects. For instance, they can create new habitats for numerous benthic species (Kramer et al., 2015), leading to the designation of the term 'reef effect' (Wilhelmsson et al., 2006; Degraer et al., 2020). Additionally, they can potentially act as 'marine reserves' (Shields and Payne, 2014) since many countries exempt fishing activities from these areas (Hammar et al., 2016). However, offshore wind energy is not without negative

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environmental impacts (Sayed et al., 2021; Bergström et al., 2014), which span from exploration (Bailey et al., 2014) to construction (Langhamer, 2012), operation (Croll et al., 2022; Fernández-Bellón et al., 2019), and decommissioning (Maxwell et al., 2022). These impacts include potential collisions with marine fauna, habitat disturbance, and noise pollution affecting marine mammals and fish during the construction phase. Operational phase impacts involve constant low-frequency underwater noise that can disrupt marine life (Croll et al., 2022). Recent studies suggest that wind wakes from offshore wind farms can cause up to a 20% change in phyto- and zooplankton distributions in the North Sea, potentially affecting the entire marine food web (Daewel et al., 2022). Additionally, research indicates that these changes can lead to significant alterations in nutrient distribution and primary production within and beyond wind farm areas (Daewel et al., 2022), thus potentially affecting the entire marine food web. Consequently, the potential impact of these structures on ecosystems could result not only in behavioural changes due to habitat alteration but also, in the worst-case scenario, affect species populations through direct mortality (Martin et al., 2023; Rezaei et al., 2023). To anticipate and mitigate the potential impacts of offshore wind energy on marine wildlife it is necessary to identify risk areas that contribute to designating favourable areas for infrastructure deployment (e.g. see Galparsoro et al., 2022). One way of achieving this is by considering foraging ecology information in the designation of protected areas (Pérez-García et al., 2022). Such information is particularly relevant in the case of top predators (i.e. seabirds and marine mammals). Despite offshore wind energy could have an impact on marine species at each trophic level (Wang et al., 2024), seabirds and marine mammals are particularly vulnerable to the impacts of offshore wind energy (Phillips et al., 2023). Worryingly this new emerging threat must be added to other well-known primary threats, such as climate change, bycatch, invasive species, pollution, hunting, or maritime traffic (Marchowski, 2021; Nelms et al., 2021).

These two groups usually show social patterns that lead to a large aggregation of multiple species during foraging (Veit and Harrison, 2017). However, up-to-date and fine-resolution data on species trophic ecology is seldom available (but see, Peschko et al., 2021; Garthe et al., 2023), which would explain their limited use on a global scale despite the rapidly growing development of offshore wind energy (Calado and Castro, 2021). Even when fine-scale information is not available, generating risk maps with coarser information at the global level could help identify areas where those aggregations are likely to occur and could target or delimit areas that ensure coexistence with marine fauna conservation efforts. We argue that assessing potential risk areas for sensitive species such as seabirds and marine mammals can benefit from generating spatial information based on crucial ecological aspects of species, such as feeding behaviour and knowledge of ecosystem functioning with coarser scale data which is more largely available. This information would be critical to evaluate the impact of wind energy development prior to the construction of infrastructures and to avoid possible future impacts at the species and community level. Moreover, risk maps may help assess the effectiveness of Marine Protected Areas in safeguarding these regions.

The present study aims to determine potential risk areas at a global scale for offshore wind energy on seabirds and marine mammals—two taxonomic groups which are particularly vulnerable to offshore wind energy development—based on the delimitation of potential feeding sites (Nelms et al., 2021; Galparsoro et al., 2022; Phillips et al., 2023). To achieve this, we will first identify potential feeding sites of seabirds and marine mammals by building a three-level “bottom-up” trophic chain (i.e. primary producers, secondary consumers, and apex predators) in a structural equation model framework. In particular, we will assess the influence of the biomass of each trophic level on the observed spatial patterns in seabird and marine mammal richness. We would then use this information to identify potential feeding sites (hereafter PFSs) to define overlapping areas with potential wind energy areas and identify the risk areas for seabirds and marine mammals. Finally, we evaluated

the role of Marine Protected Areas (hereafter MPAs) located in coastal zones in protecting seabirds and marine mammals from offshore wind development risk.

2. Materials and methods

2.1. Seabirds and marine mammal data

We selected only small-ranged seabirds and marine mammals exhibiting restricted geographical distribution (hereafter seabird and marine mammal richness) This selection was based on the premise that species with smaller ranges are more threatened than those exhibiting larger distribution ranges (Ripple et al., 2017). Furthermore, these species usually show higher extinction rates and are more sensitive to anthropogenic-driven alterations within their range (Chichorro et al., 2019). Hence, they become a suitable subject to investigate the potential effects of offshore wind energy on them. Small-ranged species were determined as those with a geographic range size smaller than the global median (i.e., the 50% of species with the smallest ranges; median = 16.8 and 16.5 million km² for seabirds and marine mammals, respectively; Jenkins and van Houtan, 2017; Supp Mat Table 1). Distribution of these species was obtained at a resolution of 100 km from <https://biodiversitymapping.org/>. According to Jenkins and van Houtan (2016), richness maps were derived from digital distribution maps for all the world's birds in Birdlife International (2018) and IUCN (2018) for birds (n = 82) and marine mammals (n = 24), respectively. Finally, seabird and marine mammal richness raster were summed for further analysis.

Based on existing literature, we obtained raster data for each marine ecosystem food web component (see Lynam et al., 2017; for an overview), including phytoplankton biomass, zooplankton biomass, fish biomass and fish richness. We extracted phytoplankton, zooplankton, and fish biomass spatial raster data from Hatton et al. (2021) at a 70 km resolution. We calculated phytoplankton biomass (g/m²) from sea surface temperature and satellite chlorophyll *a* measurement obtained from MODIS-Aqua monthly climatology (Moderate Resolution Imaging Spectroradiometer aboard the Aqua spacecraft, 4-km resolution) from 2002 to 2016 and aggregated to a 100 km spatial resolution (Hatton et al., 2021). We used zooplankton biomass data from >200 k water samples and interpolated over the whole ocean based on environmental correlates (more details on Hatton et al., 2021). We summed the biomass of micro, meso and macro-zooplankton to get a unique measure of total zooplankton biomass (g/m²). We estimated fish biomass (g/m²) from two data-constrained global ecosystem models (see Petrick et al., 2019; Carozza et al., 2017; Hatton et al., 2021). Finally, we obtained fish richness from Jenkins and Van Houtan (2016) generated spatial dataset at a resolution of 100 km (freely available at <https://biodiversitymapping.org/>). All the raster were resampled to 100 km resolution to make them comparable.

The above variables (phytoplankton biomass, zooplankton biomass, fish biomass, fish richness, and small-ranged seabird and marine mammal richness) were extracted using a random sample of 500 points within the summed small-ranged seabird and marine mammal distribution ranges. We took a conservative approach by selecting 500 random points, as structural equation models are sensitive to large sample sizes, which can produce inconclusive results. By limiting our

Table 1

Results of the model for risky areas within Marine Protected Areas. Abbreviations: SE=Standard Error, P=P-value. Reference levels for comparisons in the model were “seabirds + mammals” and “outside”, respectively.

| Variables | Estimate | SE | t-value | P |
|-----------|----------|-------|---------|--------|
| Intercept | 0.435 | 0.004 | 98.81 | <0.001 |
| Mammals | 0.229 | 0.006 | 36.20 | <0.001 |
| Seabirds | 0.230 | 0.005 | 36.22 | <0.001 |
| Inside | -0.384 | 0.008 | -43.55 | <0.001 |

selection to 500 points, we ensured adequate spatial coverage of the sea surface without inflating the sample size, thus avoiding these issues.

For each sampling point, a 50 km buffer was generated, and the mean values of each variable were extracted. The buffer size was chosen to encompass a sufficiently large area to capture all the spatial variability on a large scale.

2.2. Structural equation model and identification of potential feeding sites

We considered phytoplankton biomass could influence higher trophic levels (e.g., zooplankton, fish and apex predators such as seabirds; Lynam et al., 2017). At the same time, zooplankton biomass could shape marine mammal distribution (Pendleton et al., 2020). Likewise, zooplankton biomass influences seabird density and associations between species (Sydeman et al., 2010) and could even determine fish biomass (i.e., via effects on annual recruitment; Lomartire et al., 2021). Fish abundance, which is positively related to fish biomass (Pauly and Pitcher, 1988), is related to seabird abundance at a local scale (De La Cruz et al., 2022b). Similarly, fish richness may play an important role in modulating predator richness in marine ecosystems (Emmerson and Raffaelli, 2004). Finally, top predators such as seabirds and marine mammals primarily feed on fish or at least include them in their diet in most cases (Jelicich et al., 2022).

Based on this reasoning, we proposed a trophic chain consisting of four hierarchical levels present in marine food webs: 1) phytoplankton, 2) zooplankton, 3) fish, and 4) seabirds and marine mammals (Jelicich et al., 2022). Each level influences the next, ultimately impacting top predators through bottom-up control (Heath et al., 2014). Based on this trophic chain structure and using data on phytoplankton biomass, zooplankton biomass, fish biomass (Hatton et al., 2021), fish richness, and seabirds and marine mammal richness we built a structural equation model. Despite other mechanisms (e.g., top-down regulation; Lynam et al., 2017) and other environmental variables (e.g., climatic anomalies; Chust et al., 2014; Ratnarajah et al., 2023) could also modulate spatial distribution or biomass of critical components of marine ecosystems we only consider bottom-up interactions between the selected variables, as our final purpose was to identify whether lower trophic levels could be potentially used as proxies for potential feeding sites for marine mammals and seabirds and not to describe exhaustively the links between each one of the trophic levels considered.

We constructed our structural equation model to validate our proposed trophic relationships and estimate the direct and indirect effects of lower trophic levels on the observed small-ranged seabird and marine mammal richness at a global scale (Fig. 1A). For this end, we used the 500 random points (see section 2.1). Larger samples increase the likelihood of obtaining a statistically significant p value, potentially reducing the chances of achieving a well-fitting model (Lefcheck, 2016). Structural equation models were constructed using the R package 'piecewiseSEM' version 2.1 (Lefcheck, 2016). We developed three full models including all the covariates for seabirds and marine mammals separately and for the combined richness of both groups. We did not perform backward or forward stepwise selection and decided to retain all the terms within each full model set for two main reasons: 1) we uphold that our food chain should retain all the elements and have enough support for not eliminating crucial components. Otherwise, the latter may compromise the rationale in the relationship between all trophic levels of our chain. 2) We are aware that some subset models have higher support by Akaike Information Criterion, nonetheless dropping variable biases remaining coefficients and p values (Whittingham et al., 2006). The goodness-of-fit for each model was measured using Fisher's C statistic and coefficients of determination (R^2) values (Lefcheck, 2016).

We used seabird and marine mammal richness separately and the combined richness of the two groups, along with significant terms included in the full structural equation models affecting their richness (i.e. fish biomass), as an indicator of immediate food availability for these

species to identify potential feeding sites (hereafter PFS). Potential feeding sites were spatially represented using bivariate raster maps, one for each covariate, and lately summed and rescaled to 0 and 1, indicating the likelihood of PFS on each raster cell ranging from low (0) to high (1). Consequently, we generated three distinct global PFS maps: 1) small-ranged seabirds, 2) small-ranged marine mammals, and 3) combining both of them.

2.3. Offshore wind energy data extraction

We selected wind power density data (W/m^2) as a proxy for potential wind energy development zones (e.g., Dunnett et al., 2022; Patidar et al., 2022). We used raster data from the Global Wind Atlas at 250 m resolution (Badger et al., 2023), providing information up to 200 km offshore and at a global height of 200 m. Turbine maximum heights typically range from 50 to 300 m (Mathern et al., 2021; Wang et al., 2018); hence, 200 m was considered the average height. Regarding the distance from the coast, 200 km was chosen to encompass the maximum possible surface, provided that the furthest offshore wind farm is located 120 km away, considering the advancement of technologies and floating structures (Díaz and Guedes Soares, 2020).

2.4. Assessment of risk areas

We performed a Spearman correlation test to determine whether power density was higher in areas exhibiting higher PFS likelihood and thus having a higher risk for the target species group. To do so, we first generated a 50 km buffer around each raster point of the PFS likelihood raster previously generated and extracted the mean values from the wind power density raster. We also computed confidence intervals for the correlation values to assess the uncertainty associated with correlation estimates.

In order to identify risk areas through spatial overlap between PFS and wind power density, again, we constructed a bivariate raster using PFS likelihood and power density data (W/m^2). Before this, we adjusted the resolution of our PFS and wind power images to a suitable 25 km scale. We deemed this resolution appropriate for delineating risk areas for the joint utilization of offshore wind energy by governmental bodies and wind companies while avoiding areas with high feeding potential. We conducted this procedure independently for the PFS of seabirds and marine mammals and the combined PFS of both groups.

2.5. Estimation of risk within Marine Protected Areas

We evaluated the mean risk inside and outside MPAs for seabirds, marine mammals and both groups together. We did so by calculating the mean risk within MPAs polygons obtained from <https://www.protectedplanet.net/en>. Then, in order to determine the effectiveness of MPAs in accounting for risky areas, we performed Generalized Linear Model with quasibinomial distribution in which we included mean risk as response variables and whether risk cells felt outside or inside existent MPAs. We also added groups (seabirds, marine mammals or both) as a factor to untangle whether there were differences in the risk between groups outside and inside MPAs.

All spatial analyses and calculations were conducted using the open-source R software (R Core Team, 2023). Significance was set at 0.05.

3. Results

3.1. Structural equation model

Our results revealed that models fitted well to the proposed structure of simplified trophic chain for small-ranged seabirds, marine mammals and the combined richness of both groups (Fisher's C ; $C_2 = 4.93$, $p = 0.09$; $C_2 = 4.90$, $p = 0.083$ and $C_2 = 4.91$, $p = 0.085$, respectively; Fig. 1A–C). We found that all lower trophic levels, except for fish

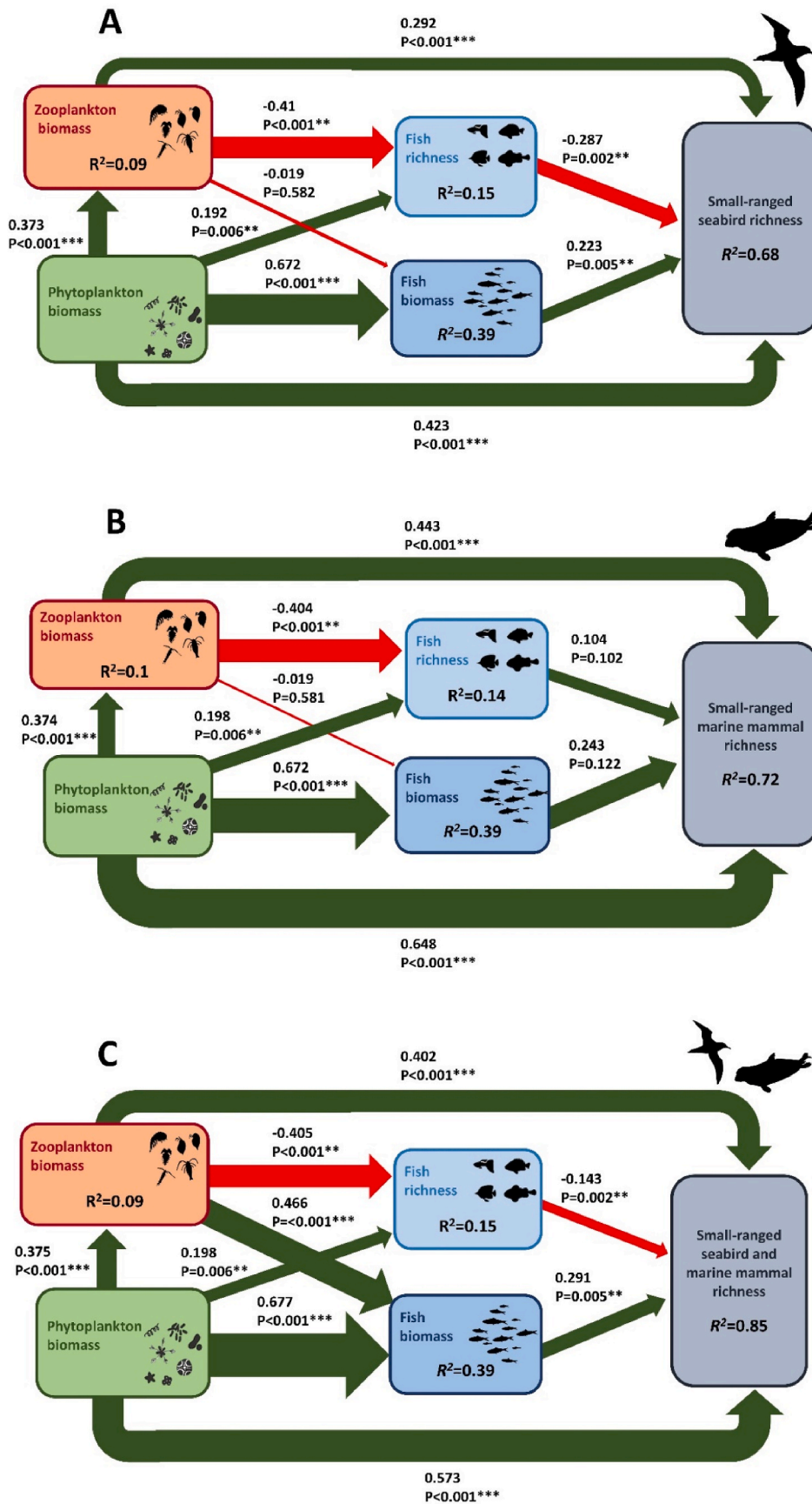


Fig. 1. Structural equation model structure and results for small-ranged seabird and marine mammal richness and the combined richness of both groups (A, B, and C). Red and green arrows denote significant negative and positive effects, respectively. The width of the arrows reflects standardized path coefficients and indicates each predictor's relative effect sizes. We provided p-value and significant ones are represented with ***.

richness, positively influenced the richness of small-ranged seabirds (Fig. 1A). Similarly, all lower trophic levels positively impacted the richness of marine mammals (Fig. 1B). Finally, all trophic levels, excluding fish richness, positively influenced the combined richness of small-ranged seabirds and marine mammals (Fig. 1C).

3.2. Identification of potential feeding sites

Small-ranged seabirds, marine mammals and the combination of the two groups exhibited distinct potential feeding site distributions (Fig. 2A–C). Specifically, the percentage of area with high feeding potential was larger in seabirds (17.3%) than that of marine mammals and the combined feeding potential of both groups (8.2% and 11.8%, respectively). In North America, a high potential was observed along the northern coast, along the coast of Greenland, and in South America, in the western and southeastern parts. In Europe, greater potential was identified along the coasts of the Nordic countries, the United Kingdom, Ireland, France, and the western region of the Iberian Peninsula. The southern coastal zone stood out in Africa, spanning from Namibia to approximately Port Elizabeth in South Africa. In Asia, areas of higher potential were found in Japan, the Sea of Okhotsk, and the Bering Sea. Lastly, in Oceania, potential zones were identified along the coastal areas of Melbourne, Tasmania, and New Zealand.

3.3. Assessment of risk areas

Model results showed a strong positive correlation between PFS

likelihood and wind power density at the global scale for seabirds, marine mammals and both groups together (Spearman's rank correlation; $\rho = 0.61$ [Lower 95% CI = 0.60, Upper 95% CI = 0.62], $p < 0.001$, $\rho = 0.5$ [Lower 95% CI = 0.49, Upper 95% CI = 0.51], $p < 0.001$, $\rho = 0.64$ [Lower 95% CI = 0.63, Upper 95% CI = 0.65], $p < 0.001$, respectively; Fig. 3). Overall high-risk areas for seabirds, and marine mammals and both groups concentrated in the western belt of North and South American coast and northern Europe, Oceania and eastern Asia, specifically the Bering Sea region. Significantly, low-risk areas expanded across the global southern coast, irrespective of the targeted taxonomic group (Fig. 4A–C).

3.4. Estimation of risk within Marine Protected Areas

Our model showed that areas of higher risk were located outside MPAs whilst, risk was generally lower within them for seabirds, marine mammals and both groups (0.59 ± 0.26 , 0.59 ± 0.26 and 0.45 ± 0.34 , respectively) (Table 1; Fig. 5). Moreover, we observed that risk was slightly higher for seabirds and marine mammals separately compared to both groups taken together, particularly within MPAs.

4. Discussion

4.1. Use of trophic relationships as predictors of potential feeding sites

Our findings suggest a global-scale bottom-up control mechanism, whereby biomass of lower trophic levels exerts an influence on the

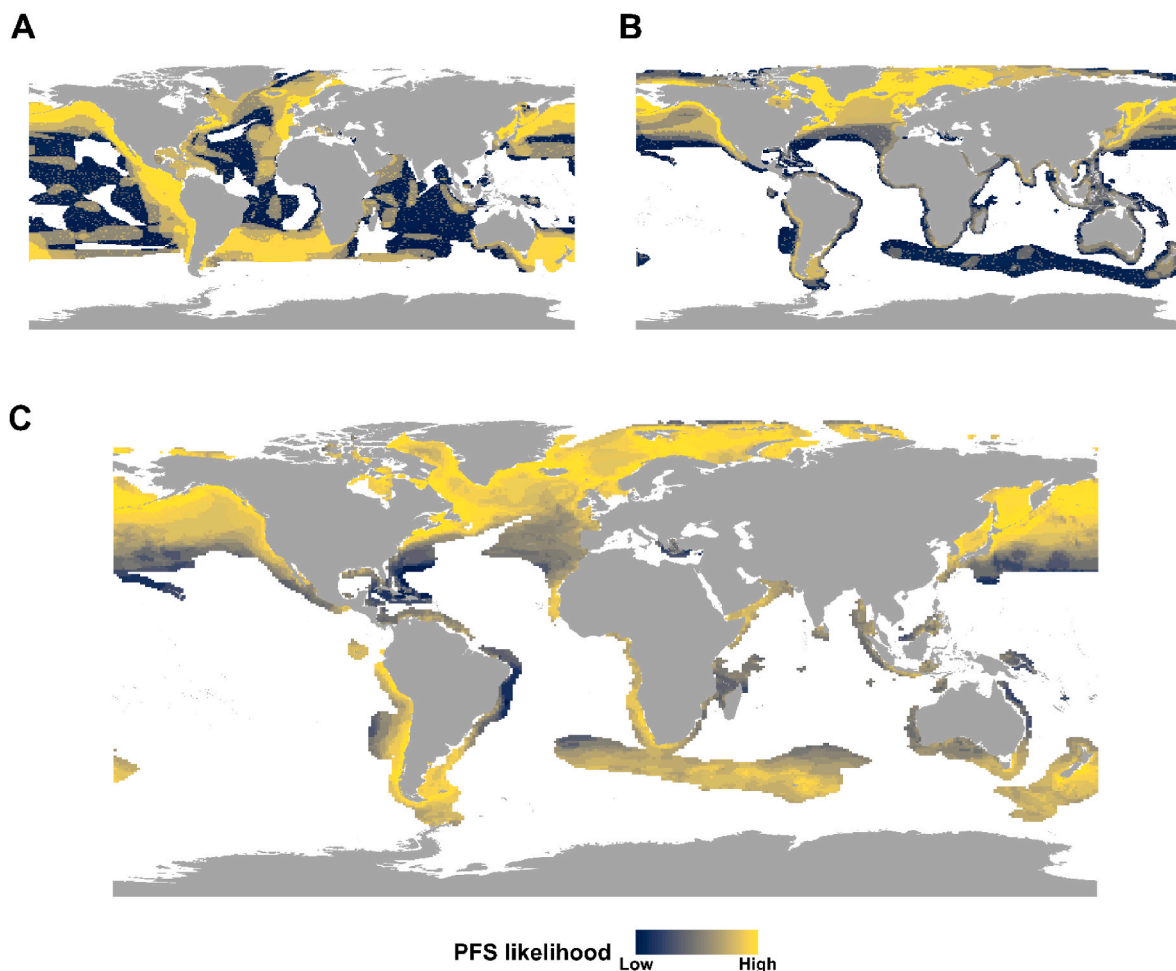


Fig. 2. Bivariate maps of potential feeding sites (PFS) likelihood ranging from 0 (low) to 1 (high) for small-ranged seabirds and marine mammals and for both groups combined (A, B, and C).

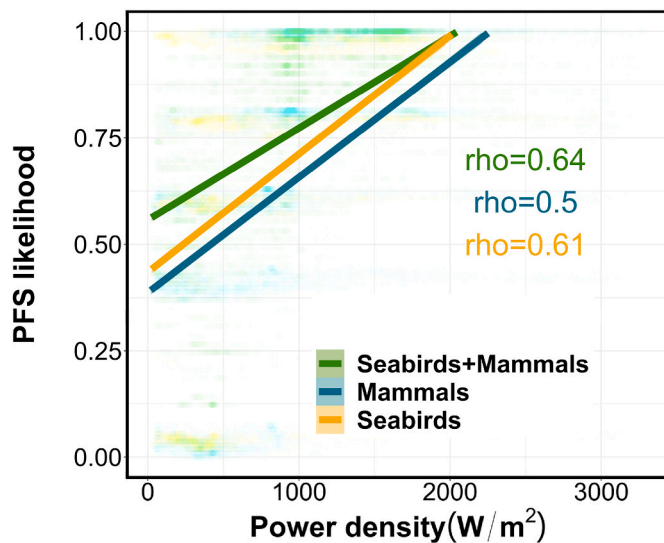


Fig. 3. Correlation between potential feeding sites (PFS) likelihood and wind power density, for seabirds, marine mammals and both groups combined. Number represent Spearman correlation values. Dots in the background showed each cell value.

population dynamics and distribution of higher trophic levels (see also Tucker et al., 2014; Lassalle et al., 2011). These findings are consistent with previous studies which have found evidence of this trophic structure (see Lynam et al., 2017). A strong positive relationship was identified between the richness of apex predators such as seabirds and marine mammals and the biomass of lower and immediate trophic levels. This positive correlation can be explained mainly by the generalist diet exhibited by these species (Jelicich et al., 2022). Some seabirds and marine mammals groups feed on phytoplankton and zooplankton (Evans et al., 2021; Hipfner et al., 2020; Pendleton et al., 2020), which has been demonstrated to facilitate their optimal development (Hipfner et al., 2020). Moreover, phytoplankton can also be utilized by seabirds to locate fish schools due to the substances they release when feeding on krill (Abolaffio et al., 2018) which could explain the observed results. About zooplankton, although it is not a direct component of the diet for certain species (e.g., seabirds), it is consumed by their prey, which may explain the indirect strong positive relation between zooplankton biomass and target groups. Indeed, as evidenced by our model for grouped seabird and marine mammal richness, zooplankton blooms have been observed to enhance trophic interactions and subsequently increase the biomass of higher trophic levels (i.e., fish) (Nishizawa et al., 2020).

It is noteworthy that, despite fish representing a crucial food source for the majority of species groups (Forero et al., 2004; Jelicich et al., 2022), our results showed that fish richness is not a relevant variable explaining marine seabird and combined richness of seabird and marine mammals richness. Seabirds often rely on specific prey species rather than overall fish diversity. The abundance of particular forage fish species is more important than fish richness (Mínguez et al., 2003). Lower levels, such as phytoplankton and zooplankton, showed a stronger effect on this particular group, indicating a more specialized diet based on these resources (e.g., Iverson et al., 2019). On the other hand, fish biomass greatly influenced seabird and marine mammal richness and the combined richness of seabird and marine mammals at large scale. This finding aligns with prior research that has identified fish biomass as a key driver of seabird distribution variations, along with other factors such as fish distribution (Ward and Lewis, 2016).

4.2. Identification of potential feeding sites

Our approach contributes to an understanding of the trophic

relationships among these species (Lynam et al., 2017). Consequently, we addressed the primary considerations in marine spatial planning analysis (see Foley et al., 2010) by including variables providing highly relevant information (main trophic resources). These were used for conducting distribution zone analysis (Evans et al., 2021; Pendleton et al., 2020) as well as for delineating sensitive areas for target species (Püts et al., 2023).

The PFSs identified in our study were found to comprise a significant portion of the ocean's surface, including the coastal regions of North and South America, northern Europe, eastern Asia, and New Zealand, which were identified as areas with the highest potential. As might be expected, these regions encompassed the largest phytoplankton, zooplankton and fish biomass due to the ocean currents and physical characteristics that result in the formation of nutrient-rich waters (Bristow et al., 2017; see <https://www.ncei.noaa.gov/products/world-ocean-atlas>). The maps facilitate the prediction of an increased diversity of seabird and marine mammal congregations in these areas (Evans et al., 2021). However, it is essential to note that the spatiotemporal factors that could influence the PFS distribution are not feasible to consider. It is clear that the distributions of the groups under study are not static in terms of both abundance and occurrence. As an example, fish biomass shows predominant spatial variability while phytoplankton and zooplankton show greater temporal variability (Evans et al., 2021). Additionally, it is established in the literature that temporal fluctuations in temperature or salinity can affect fish diversity (Heino et al., 2013). In this regard, alterations in temperature, such as those resulting from global warming, have the potential to alter the distribution of species (Campana et al., 2020) and might reduce the predictive capacity of our models. It is therefore imperative to enhance our comprehension of the spatio-temporal dynamics of pivotal food resources (Frederiksen et al., 2006) to more accurately and dynamically assess the locations of PFS, thereby increasing their value in informing urgent management decisions.

4.3. Evaluation of risk areas for seabirds and marine mammals

Our analysis of potential risk revealed a substantial overlap between areas of high PFS and greater wind resource density. These results highlight the emergence of conflict zones between the studied species and future offshore wind energy infrastructures. As these areas serve as potential gathering points for numerous species of seabirds and marine mammals, establishing wind farms could pose a threat, impacting their abundance and potentially leading to displacement (Vallejo et al., 2017). It is important to note that seabirds are particularly vulnerable due to cumulative exposure during the operational phase (Goodale and Milman, 2020; Wing Goodale et al., 2019), while marine mammals are predominantly affected during the construction phase of such infrastructure (Amaral et al., 2020). It is therefore imperative that these areas are considered in spatial planning on a proactive basis.

The risk map revealed that the majority of potential feeding areas are situated in proximity to regions characterised by high wind intensity. Regions with a higher likelihood of harbouring potential risks were situated along the coastal areas of Canada, Chile, Argentina, the majority of European coasts, the Russian coasts of the Asian continent, the southern coast of Australia, and throughout the coastlines of New Zealand. The global advancement of wind energy has positioned Europe as the vanguard of this field, with many of its coastlines facing potential conflicts with seabirds and marine mammals, as evidenced by the findings of this study. Similarly, our analysis underscores the need for specific legislative measures to regulate the establishment of these developments (deCastro et al., 2019; European Union, 2014). However, even protected marine areas are affected by this issue, as installations in these zones are occasionally permitted under specific justifications (deCastro et al., 2019). China is the second most prominent nation in terms of wind energy development. However, caution must be exercised when interpreting these results, as they correspond to a lack of spatial

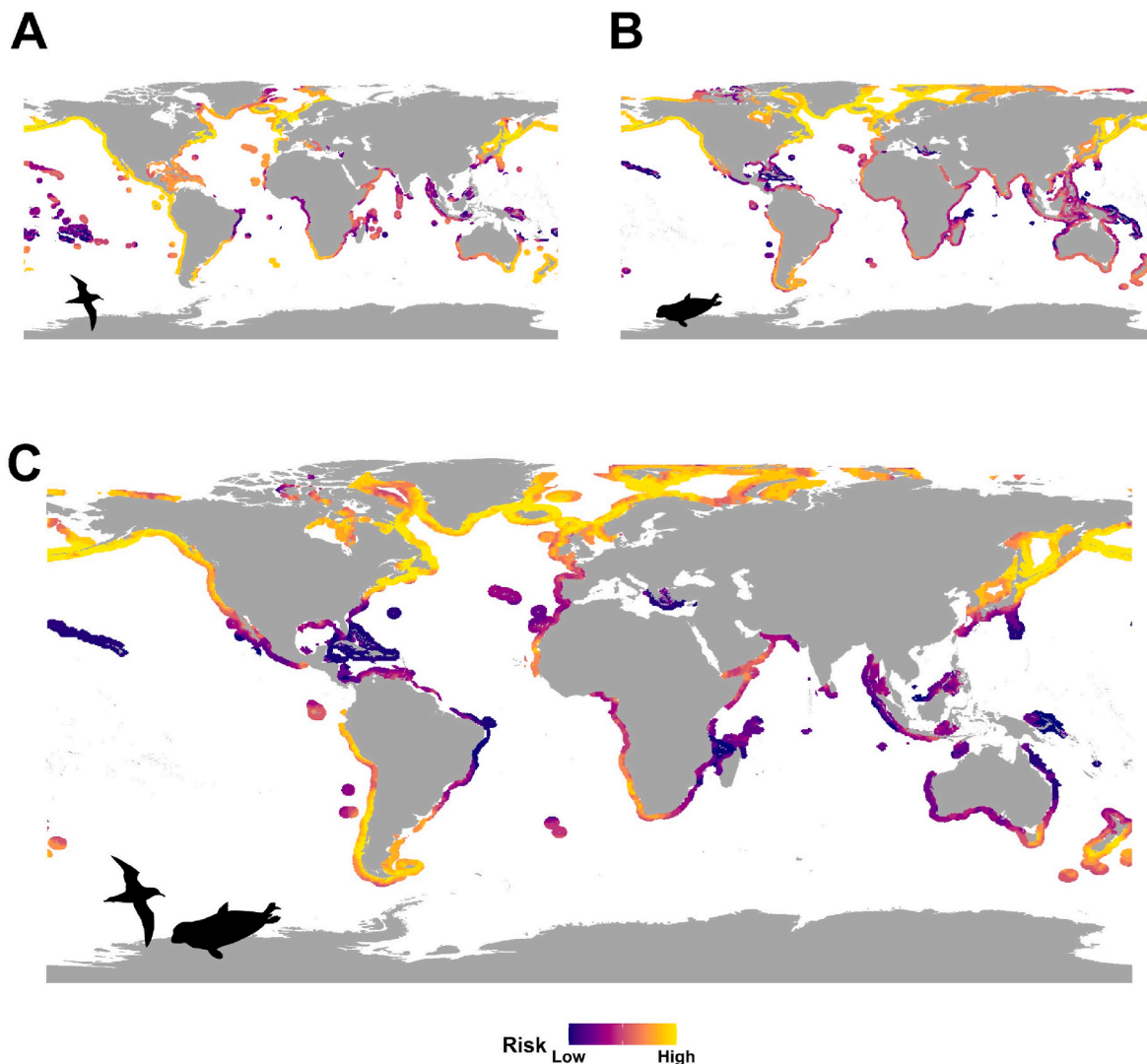


Fig. 4. Risk maps for small-ranged seabirds, mammals, and both groups combined (A, B, and C). Coloured areas showed sites where both potentialities as feeding site and wind power overlap and, therefore, assumed to have different risks ranging from low (0) to high (1).

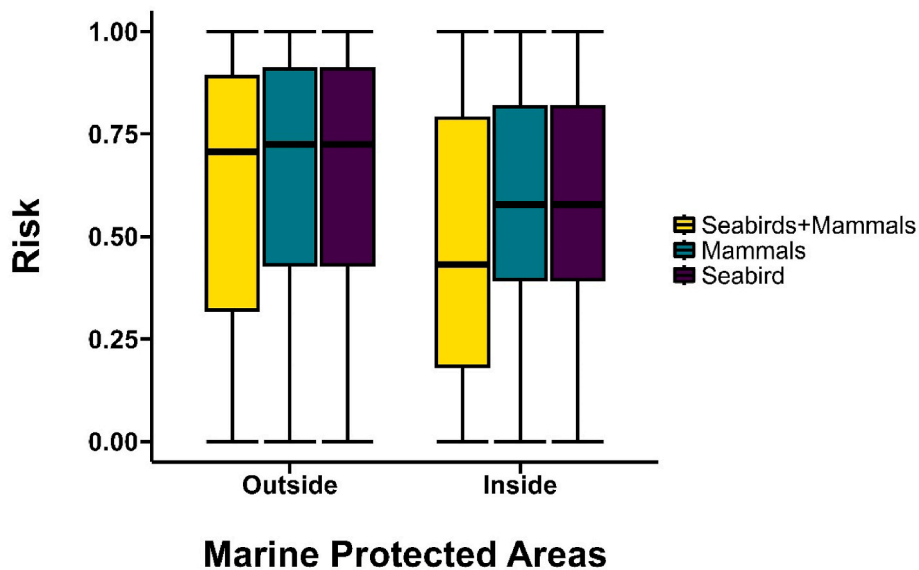


Fig. 5. Mean risk for seabirds, marine mammals and both groups together outside and inside Marine Protected Areas.

information rather than the actual absence of risk. In contrast, on the Asian continent, the eastern coast of Russia is marked by areas with potential conflict with these species, although its current offshore wind development is limited (Kudelin and Kutcherov, 2021).

In the Americas, zones with a high risk of conflict were identified in the northern and southern parts of the continent. Notwithstanding the considerable onshore wind development, Canada lacks offshore wind installations, largely due to the challenging geological conditions that prevail there (Eamer et al., 2021). Argentina and Chile exhibit high potential for offshore wind energy but lack active projects, reducing the risk of conflict with the studied species (Mattar and Villar-Poblete, 2014; 4COffshore, 2023). Conversely, Uruguay and Brazil have multiple projects currently in development that have yet to receive approval, suggesting a high risk of conflict with these species (Hernandez et al., 2021). When analysing the Oceania region, a notable level of risk is evident along the coast of Tasmania and the coastlines of New Zealand. Despite the current low levels of offshore wind energy development and the existence of pending project approvals, it is crucial to emphasise the considerable wind potential these areas possess (Salvador et al., 2022; 4COffshore, 2023).

A global-scale approach provides an initial perspective on conflict-prone areas, which require a more detailed evaluation due to the potential risks they may pose to wildlife. The findings derived from these analyses offer institutional entities, governmental agencies, and stakeholders the opportunity to establish priorities regarding project implementation zones, aiming to mitigate conservation challenges that might emerge for offshore wind energy development. The timely utilization of decision support tools proves most effective during the early stages of project planning, enabling the anticipation and prevention of potential future conflicts (Best and Halpin, 2019). Thus, despite its limitations, our study supports the implementation of “no go-to areas” based on species’ trophic ecology knowledge (e.g., areas where multiple species converge for feeding). This could alleviate conflicts between the limited distribution of seabirds and marine mammals and offshore wind energy.

Although our approach offers a straightforward solution (banning turbines in risky areas), it also has important implications at the socio-economic level. For instance, most low-risk areas are located in the Global South. These results may indicate that areas with nutrient-rich waters support a variety of organisms, including seabirds and marine mammals (de Souza and Morrison, 2024; Sun et al., 2024), but at the same time are regions with low-power density potential in which offshore wind energy development is still minimal compared to European countries. Therefore, if offshore wind energy is to be developed in such regions, it could generate an environmental and economic burden to countries where the poverty index is currently increasing. This issue is significant, considering that benefits and burdens from land (and sea) are unequally distributed worldwide (Meyfroidt et al., 2022). If measures are to be implemented to develop offshore wind energy in these areas, they must be accompanied by financial support, or else they will turn into so-called ‘green colonialism’ (see Normann, 2021). This would involve countries aiming to avoid the adverse effect of offshore wind energy development on their seascapes (e.g., by identifying “no go-to areas”). Finally, we are also aware that the applicability of these bans on developed countries may be considered unfeasible, given it may pose a radical policy change. This change seems unrealistic under environmental law regression concerning renewable energy development, particularly in Europe (Durá-Alemañ et al., 2023).

4.4. Estimation of risk within Marine Protected Areas

MPAs may seem ineffective in reducing exposure to wind energy development on both seabirds and mammals. These results are not surprising as most of the species found their feeding sites out of these areas due to the spatiotemporal dynamics of food sources (De La Cruz et al., 2022b). Hence, despite the risk within MPAs was presumably lower, the coverage of MPAs is still very limited. Currently of world

national waters (39% of total sea surface) 19.17% are designated as protected areas which seem insufficient to meet global conservation goals (Edgar et al., 2014). Increasing coverage of such areas may help encompass riskier sites for both groups, which, accompanied by the designation of “no go-to areas” may help reduce conflicts with offshore wind energy development. Moreover, implementing dynamic MPAs (Maxwell et al., 2020) by combining multiple species groups’ GPS tracking data would help in also solving conservation conflicts of highly mobile species (i.e., seabirds and marine fauna) where the static or traditional MPAs may fail, thus enabling compatibilization of species conservation and wind energy development. Nonetheless, we recommend being cautious in this instance, as the non-overlap between wind energy and MPAs may not always be possible. Particularly, in MPAs located in coastline areas. In these cases, there is a need for a rigorous assessment of their potential effect on sensitive groups, especially at a local scale where this overlap is unavoidable (e.g.; Lloret et al., 2023).

4.5. Potential limitations of the current study

It is important to acknowledge that our study is not without limitations which could potentially influence the outcome, particularly if they are to be interpreted on a smaller scale. Our model showed how fish and zooplankton biomass are variables providing key information for identifying potential feeding sites of marine predators, but we recognize that PFS may be influenced by multiple other factors that ultimately condition the current and future areas of potential interaction with offshore wind farms.

The predictive model we developed relies on various variables, yet we recognize that several important factors have been excluded, which could enhance its accuracy. These include anthropogenic activities, commuting patterns, migration corridors, and species with large distribution ranges. For instance, the distribution of seabirds and marine mammals is significantly affected by fishing activities, as these species often feed on discards or use vessels to locate their primary prey (De La Cruz et al., 2022; Karpouzi et al., 2007). Research has shown that global analyses of feeding zones for these species align closely with the findings of our study, particularly when utilizing fisheries data (Karpouzi et al., 2007). It is also worth mentioning that the resolution of covariates employed to construct risk maps, despite it being fine enough to give a general overview of risk areas, could be much improved by generating 1) more reliable species distribution maps at different temporal scales yield finer estimates of species richness (and abundance) and 2) a more detailed record of biomass estimates for considered covariates. Together, these two points may help build risk maps at higher resolution. Consequently, it would be intriguing to consider other variables of this nature for future analyses and studies to enhance the understanding and delve deeper into researching the effects of offshore wind energy on this and other species and its utilization for spatial planning and sensitive zone determination at more local scales.

It should be noted that there are other sensitive areas, such as commuting and migratory corridors or breeding areas for birds and marine mammals, that have not been modelled here (e.g., Mikami et al., 2022). On the other hand, it is important to highlight that while we assume the PFSs are feeding zones, we cannot definitively rule out that these species feed in those areas given that the data of tracked individuals were not used. Using detailed spatial tracking data has the potential to address this limitation, reduce uncertainty, and provide a more comprehensive understanding of the spatial and temporal patterns of usage in these areas, as well as in other potential transition zones and areas with significant overlap with existing or future wind turbine developments (Nelms et al., 2021; Peschko et al., 2020, 2021; Vanermen et al., 2020). Linked to the latter, generating both global and local long-term datasets of species abundance and richness at a finer resolution would assist in the identification of risk areas with greater precision and accuracy.

Additionally, wind power density offers a reasonable approximation

when considering future areas for offshore wind farm installations. However, more precise data from relevant authorities concerning these areas is required. This approach would enable a more comprehensive assessment of the impact of such installations on potential feeding sites. Moreover, the risk during different phases is subject to variation. For instance, accounting for different phases of turbine settlement, such as offshore prospection, preconstruction, construction, operation, and decommissioning activities, might offer insights into which of these phases had a more significant impact. This information could be valuable in reducing potential impacts, particularly in the context of achieving a balance between offshore wind energy and wildlife conservation.

5. Conclusions

This study highlights that an understanding of trophic relationships could be employed to identify the key factors modulating seabirds and marine mammal richness, such as biomass of primary food resources. The combination of the richness of target species and fish and zooplankton biomass revealed extensive areas of potential feeding sites at a global level. It is of particular significance that our analyses demonstrated a considerable degree of overlap between potential feeding sites and regions characterised by elevated power density. This could help to identify areas that should be avoided (i.e. “no go-to areas”) thereby guiding future offshore wind energy development in a manner that mitigates the potential harmful effects on seabirds and marine mammals. In this sense, it would be the opposite approach to what the European Union is doing with the Renewable Energy Acceleration Directive (EU Directive, 2023/2413), which identifies areas to encourage development.

Our findings showed that combining knowledge of species distribution, feeding behaviour, and ecosystem functioning is a highly valuable approach in spatial planning, and has the potential to facilitate the coexistence of wind energy development and wildlife conservation in marine ecosystems. Notably, the risk within MPAs was consistently lower than outside MPAs across all scenarios. This result highlights the inadequacy of current protection zones against renewable energy deployment and underscores the necessity for the development of specific measures to enable the conservation of marine biodiversity. To sum up, despite the inherent limitations, our approach represents a step forward in assessing the impact of offshore wind energy on biodiversity, particularly in light of the recent advent of this energy source.

CRedit authorship contribution statement

Jon Morant: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Ana Payo-Payo:** Writing – review & editing, Writing – original draft, Validation, Supervision, Methodology, Investigation. **Ana María-Valera:** Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Formal analysis, Data curation. **Juan Manuel Pérez-García:** Writing – review & editing, Writing – original draft, Validation, Supervision, Methodology, Investigation, Conceptualization.

Data availability statement

All the data used to conduct the present study and the raster map derived from analyses are freely available at Zenodo data repository: <https://zenodo.org/record/8383917>. Data for the small-ranged seabird and marine mammal richness and fish diversity is freely available at <https://biodiversitymapping.org/index.php/seabirds/>. Fish and zooplankton biomass data were obtained from Hatton et al. (2021) freely available in Zenodo <https://zenodo.org/record/5520055>.

Declaration of competing interest

There is no conflict of interest between the authors.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jenvman.2024.123808>.

Data availability

<https://zenodo.org/records/8383917>.

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