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Are intertidal habitats keeping up with nutrient export? Insights from modelling climate and management scenarios

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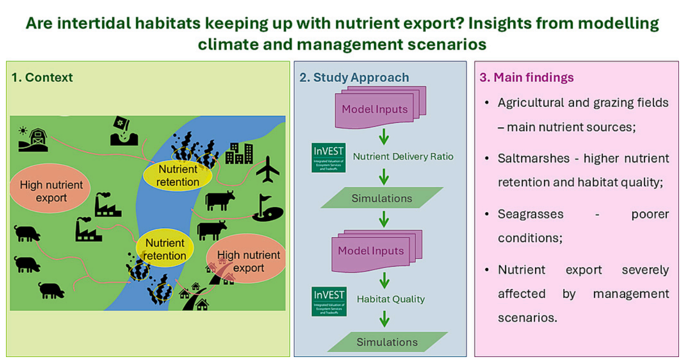
HIGHLIGHTS

- Habitat loss and eutrophication are major threats to estuaries;
- Two InVEST models to assess importance and vulnerability of intertidal habitats;
- Largest sources of nutrients are agricultural and grazing fields;
- Saltmarshes exhibit higher nutrient retention and better habitat quality;
- Management scenarios affected most nutrient export and habitat quality.

GRAPHICAL ABSTRACT



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ABSTRACT

Habitat loss and eutrophication are major threats to estuarine systems leading to biodiversity decline and loss of ecosystem services. These stressors are particularly severe in estuaries situated in highly urbanised and agriculture-dominated landscapes. While several studies have assessed nutrient dynamics in estuaries, few have focused specifically on the role of intertidal habitats as buffer areas, highlighting the novelty of this study by assessing if nutrient retention abilities of buffer habitats are able to keep up with the high nutrient export from nearby land-uses. This study employs two InVEST models (Nutrient Delivery Ratio and Habitat Quality). Scenarios also employed different management strategies and climate-change simulations to assess alterations in nutrient export and habitat quality. Agricultural and grazing fields were the largest sources of nutrients into the system, while intertidal habitats, particularly saltmarshes, exhibited high nutrient retention rates (> 80 %). Nutrient export was most severely affected by management scenarios, particularly Business-as-usual and Ecological Protection, while no significant changes were observed in Climate-change scenarios. Contrarily, habitat quality declined under the Economic Development scenario. For example, filamentous algae lost 22.02 %

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of area under excellent conditions. Nutrient export remained unaffected by the Economic Development Scenario. Saltmarshes were consistently of high quality, while seagrasses were in poorer condition (less than 1 % of the seagrass area was under excellent conditions). This study also revealed that intertidal habitats are keeping up with nutrient export, however, the upraising impacts of climate and land-use changes require update management strategies that actively change the practices in the region. Insights from such modelling approaches can aid decision-makers, offering guidance for developing policies regarding conservation of natural habitats and sustainable agriculture practices.

1. Introduction

Estuaries are functionally diverse systems, that endure strong physical and chemical gradients, including high nutrient levels from natural processes (*e.g.*, freshwater discharges) and human activities (*e.g.*, industrial discharges; Elliott and Whitfield, 2011; Medeiros et al., 2021). Nutrient sources can be categorised into point or diffuse, the first include localised discharges from facilities (*e.g.*, wastewater treatment plants), whereas the latter include dispersed inputs (*e.g.*, terrestrial runoff; Pinckney et al., 2001).

Vegetated intertidal habitats serve as natural buffer zones, effectively controlling nutrient levels and reducing the risk of estuarine eutrophication (Pinckney et al., 2001). Natural estuarine habitats, such as saltmarshes, slow water movement, enabling particle settlement and nutrient retention by plants into sediment (Cahoon et al., 2021). Nutrient retention involves removing nutrients from the water column, reducing nutrient pressure further downstream, while nutrient export refers to nutrients not retained, which reach the stream. Monitoring nutrient export is important for maintaining water quality and preventing further ecological consequences, such as biodiversity loss, which can severely affect estuarine biodiversity and ecosystem functions (OSPAR, 2006). While nutrients are crucial to support biological processes, excessive nutrient export can cause eutrophication, promoting increased primary production and leading to biofilm formation in the water's surface. This process limits gaseous exchanges and reduces light penetration, ultimately leading to hypoxia and biodiversity decline (Hautier et al., 2009). Biogeochemical modelling helps quantify nutrient export and project impacts of poor water quality on ecological structures and functioning. One commonly used modelling tool for this purpose is the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) software, developed by the Natural Capital Project (https://naturalcapitalproject.stanford.edu/). It presents a suite of open-source and deterministic models designed to map ecosystem services across different ecosystems, such as terrestrial (*e.g.,* Chen et al., 2023). In estuaries, several studies have employed one (*e.g.,* Wang et al., 2025) or more than one InVEST models (*e.g.,* Wu et al., 2025). This is also a user-friendly interface with relatively low data requirements, which makes it especially useful in data-scarce regions. As eutrophication is controlled by nutrient retention, the InVEST *Nutrient Delivery Ratio* (NDR) model supports the assessment of potential eutrophication. This model simulates yearly nutrient transport to streams and serves as a proxy for the ecosystem service water purification, according with the Common International Classification of Ecosystem Services (CICES v5.1; Natural Capital Project, 2023). The NDR model has faced criticism for relying exclusive on diffuse sources of nutrients and accounting only for yearly nutrient export (Redhead et al., 2018). However, their limitations are counterbalanced by its strengths, particularly in watersheds dominated by diffuse agricultural sources of nutrients. Alternative modelling options include *Soil and Water Assessment Tool* (SWAT) which could provide results in shorter time frames (less than a year) and integrate point nutrient sources, but demands high-resolution input data, which is often not available (Upadhyay et al., 2022). Therefore, the NDR model was selected based on several factors: being free, intuitive, capable of generating spatial outputs, and easily reproducible for monitoring purposes by management entities.

Water purification can be affected by habitat loss, a major threat in catchments dominated by agricultural activities and depends greatly on factors such as ecosystem condition and integrity (Culhane et al., 2020). Incorporating habitat degradation enables a more comprehensive assessment of intertidal habitats' capacity to support water purification, which can be done by using the InVEST model *Habitat Quality.* This model combines land use patterns and impact of habitat pressures, contributing to the identification of areas under increased stress (where pressures' weight is higher; Moreira et al., 2018).

This study aims to address the current knowledge gap regarding the intertidal habitats to withstand increasing environmental pressures, especially the ones derived from intensive agricultural activity. Understanding upstream nutrient sources is crucial to address downstream eutrophication and habitat loss. It is also important to explore different scenarios, namely anticipating future environmental conditions and evaluating how different policy choices could affect ecosystems and ecosystem services (IPBES, 2016). This information will contribute to designing appropriate management measures. The three main objectives of this research are to: 1) evaluate the impact of intertidal habitats on nitrogen and phosphorus export in a temperate estuary; 2) assess how climate-change and management strategy scenarios influence land use and, consequently, nutrient export; 3) understand how management scenarios can influence intertidal habitat quality and degradation due to identified anthropogenic pressures. The combined results can inform management decisions on the main risks of estuaries integrated in agricultural landscapes, including eutrophication and habitat loss. The novelty of this study lies on the use of two InVEST models applied specifically to a set of habitats (intertidal) by combining quality of the habitat with ability to provide the service water purification, particularly in agricultural watersheds, highlighting the importance of assessing if buffer habitats can keep up with the threats posed by the landscape.

2. Methods

2.1. Study approach

In this study, two InVEST models were used to understand the importance of intertidal habitats for water purification, and their vulnerability in the case study (Sado estuary, Fig. 1). **Section 2.2.** provides a brief description of the study area, including local conservation status and the main pressures affecting water quality, to contextualise the modelling approach. The InVEST *Nutrient Delivery Ratio* (NDR) model (**Section 2.3.**) was used to measure the impact of intertidal habitats on the export of nutrients, specifically nitrogen (N) and phosphorus (P). The *Habitat Quality* (HQ) model (**Section 2.4.**) was used to assess overall health of intertidal habitats, by combining information about pressure and habitat vulnerability (Fig. 1). The NDR model was also used as an input for the HQ model since excessive nutrient export can develop into a pressure under certain conditions (*i.e.,* shallow zones with low water flow). Data required to populate the model was collated *via* a comprehensive literature review (including grey literature). ArcGIS pro (v3.4.2.) was used to manipulate the data collected and analyse the outputs of InVEST. A baseline simulation was performed for each model (*i.e.,* current situation for land-use and nutrient input). A model validation was performed by using data collected from other projects. On top of the baseline simulation, each model produced a number of different outputs for each scenario: Business-as-Usual scenario (*i.e.*, prediction scenario for 2033 assuming the same trend as the last ten years), two Climate-change scenarios (*i.e.*, scenarios for Precipitation and Submersion for 2050 and 2100) and four Management Strategy scenarios (*i.e.*, Economic Development and Ecological Protection scenarios). Each scenario is described in detail in **Section 2.5.**. Additionally, a Sensitivity Assessment (see **Section 2.6**) was performed to both models to assess the influence of the different parameters into the outputs.

2.2. Description of study area

The Sado river basin (Portugal) was selected as the model domain, representing an ecosystem characterised by an agricultural landscape and multiple uses (*e.g.,* trading hub), that put the system at risk of eutrophication and habitat loss. The study area is located in a temperate climate with an average annual precipitation of approximately 600 mm (Alves et al., 2024). The source of Sado river is in Serra da Vigia (230 m elevation) and after 180 km it flows into the Atlantic Ocean *via* the Sado estuary (Caeiro et al., 2005). The basin spans 7692 km^2 while the Sado estuary covers 212.4 km^2 (Fig. 2). Sado is considered a mesotidal estuary with an average depth of eight meters, maximum depth of 50 m (Brito et al., 2023). Four municipalities envelope the estuary - Setúbal, Palmela, Alcácer do Sal and Grândola (Alves et al., 2024). Hydrologically, the estuary is well mixed with minimal stratification, water flow is mainly forced by tides (maximum tidal height is 3.9 m; Biguino et al., 2024).

The Natural Reserve of Sado Estuary, established in 1980, covers 329.71 km^2 to protect the ecological integrity of the estuary (ICNF, 2024). Additionally, Sado is recognised under the RAMSAR Convention, as well as a Special Protection Area and a Site of Community Importance from Natura 2000 (ICNF, 2024). Apart from environmental importance, the estuary supports key economic activities, including industry, fisheries, aquaculture, tourism. It is influenced by a population of 217,282 inhabitants across the four municipalities (INE, 2021). Significant transformations over the last century, mainly associated with the urbanisation process in the Setúbal region and industrial development around the port, have shaped the area's current landscape (Alves et al., 2024).

2.3. Nutrient delivery ratio model

The InVEST NDR model employs a mass balance approach to represent the long-term flow of nutrients (Nitrogen and Phosphorus) from catchments to streams. While the model cannot directly quantify the ecosystem service water purification, it can assess the effect of different habitats on nutrients exportation. The model computes nutrient export per pixel (Exp_i) relying on nutrient loading ($load_i$) and nutrient delivery (NDR_i; **Eq. 1**). For a catchment, the sum of each pixel nutrient export results in the total export (Exp_T; **Eq. 2**).

$$Exp_i = load_i \bullet NDR_i \tag{1}$$

$$Exp_T = \sum Exp_i \tag{2}$$

Nutrient loading ($Mload_i$) is based on the land use and land cover (LULC) map and associated loading rates, taking into consideration the pixel potential runoff (RPI_i; **Eq. 3**). The last is a ratio between pixel runoff (RP_i) and the average of runoff for all the catchment (RP_{av}; **Eq. 4**).

$$Mload_i = load_i \times RPI_i \tag{3}$$

$$RPI_i = RP_i / RP_{av} \tag{4}$$

Nutrient delivery (NDR_i) is based on the flow path, which considers the slope, provided by a Digital Elevation Model (DEM), and retention efficiency (*i.e.*, expected maximum nutrient retention for a given LULC). It depends on the proportion of nutrients that are not retained by downslope pixels (NDR_{dn}) and the connectivity index (IC_i; **Eq. 5**). NDR_{dn} is based on the retention efficiency (eff_i; **Eq. 6**). IC_i represents the probability of a nutrient to reach the stream. It is dependent on the average slope gradient ($\overline{S}$), upslope contributing area (A), length of the flow path (d_i) and slope gradient (S_i; **Eq. 7**). IC_0 and k_b are calibration parameters.

$$NDR_i = NDR_{dn} \bullet \left(1 + \exp\left(\frac{IC_0 - IC_i}{k_b}\right)\right)^{-1} \tag{5}$$

$$NDR_{dn} = 1 - eff_i \tag{6}$$

$$IC_i = log_{10}\left(\frac{\overline{S}\sqrt{A}}{\sum_i \frac{d_i}{S_i}}\right) \tag{7}$$

2.3.1. Baseline simulation: Information required and data preparation

To apply the NDR model, the first step was to define the catchment area. Considering the extensive size of the Sado river basin, the model was applied to two catchments: the river basin and the estuary basin (Table 1). The two catchments are connected, therefore comparing the

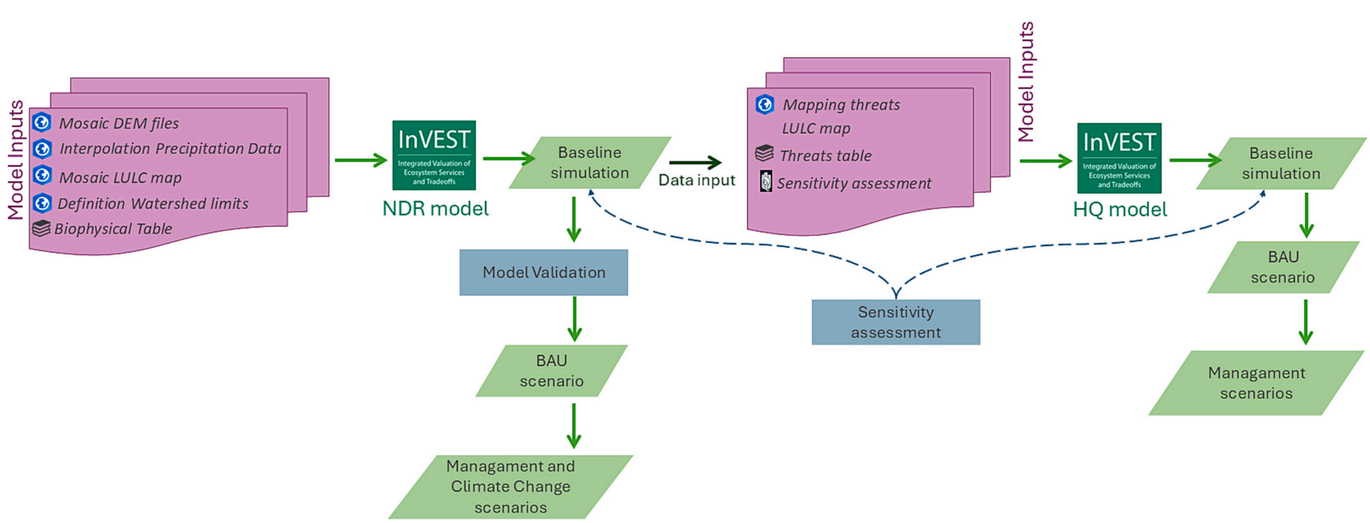

Fig. 1. Conceptual flowchart of the study design. Two InVEST models are employed, Nutrient Delivery Ratio (NDR) and Habitat Quality (HQ), whereas the Baseline simulation (Current situation) are used as inputs for the HQ model. Business-as-Usual (BAU) scenarios support additional scenarios.

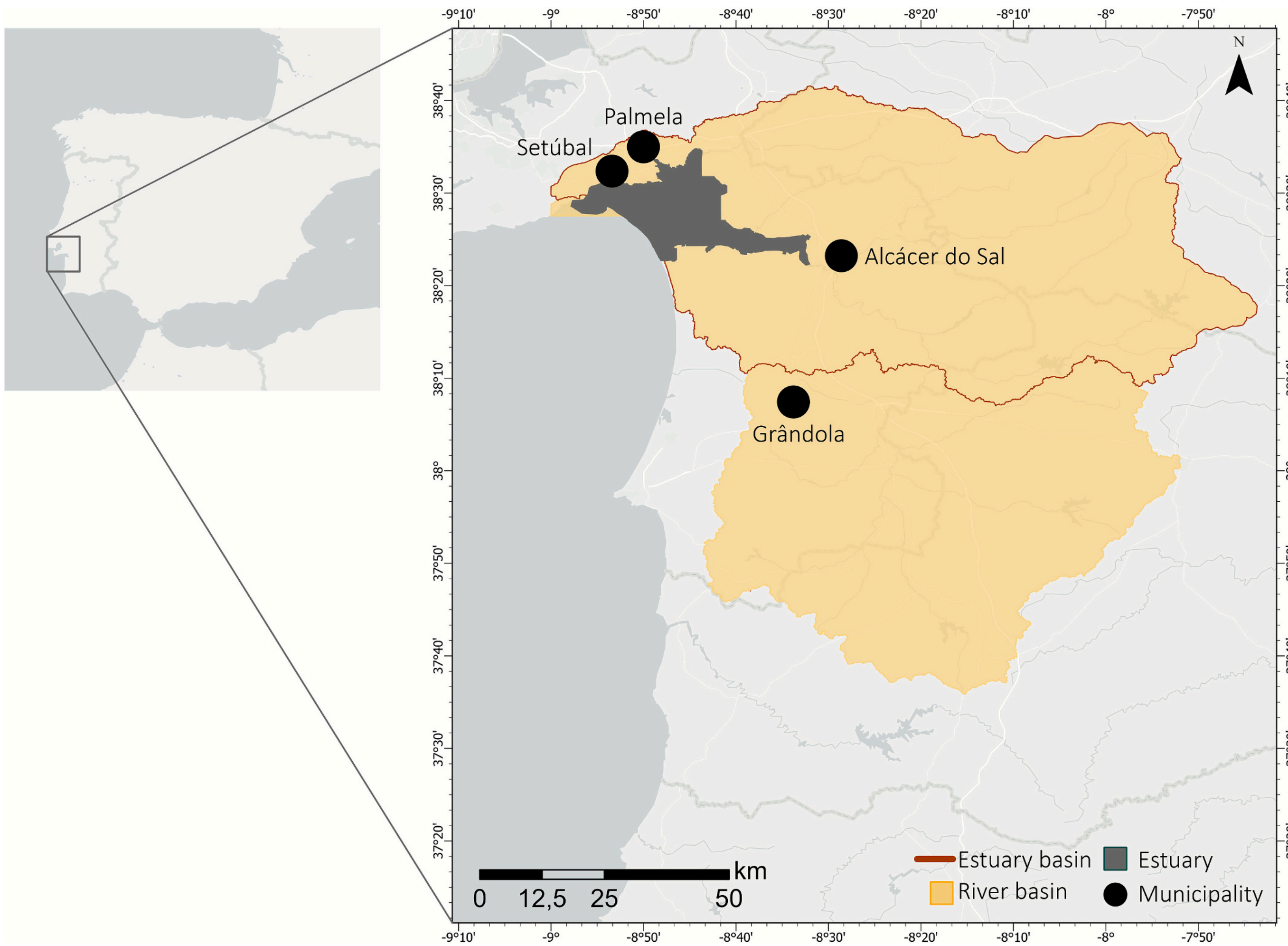

Fig. 2. Map of study area. Left inset: Location of Sado river basin in Iberian Peninsula. Right inset: Map of Sado river Basin and Sado estuary basin showing main municipalities and estuary location.

Table 1
Data sources of Nutrient Delivery Ratio model. APA – Portuguese Environment Agency; COS – Portuguese Land Use and Occupation Map; DEM – Digital Elevation Model; DGT – General-Directorate of Territory; LULC – Land Use Land Cover; SNIAmb – National System of Environment Information; SNIRH – National System of Water Resources Information.

Dataset	Source	
Catchments	Sado River Basin raster from SNIAmb	https://dados.gov.pt/pt/datasets/bacias-hidrograficas-das-massas-de-agua-de-portugal-continental-cdg-sniamb/
LULC	COS, 2018 (DGT, 2018)	https://www.dgterritorio.gov.pt/dados-abertos
Intertidal Map	Sado intertidal Map from Afonso et al. (2024)	doi:https://doi.org/10.1016/j.rsase.2024.101306
DEM	STRM-DGT (30-m) from NASA	https://dwtkns.com/srtm30m/
	DEM (2-m) from APA and DGT	https://www.dgterritorio.gov.pt/dados-abertos
Bathymetry	General Bathymetric Chart of Oceans	https://www.gebco.net/data_and_products/gridded_bathymetry_data/
Precipitation	SNIRH from APA	https://snirh.apambiente.pt/

nutrient export from both allows the determination of nutrient sources. The estuary basin was delineated using the *Watershed* function from ArcGIS pro, using as input the flow accumulation map based on the Sado river basin raster (generated by the *Flow accumulation* function from ArcGIS pro). The main pour points (*i.e,* points where water flows out of a catchment) were identified by locating the main tributaries of the Sado river.

For the LULC map, the Portuguese land use and occupation map of 2018 (DGT, 2018) was used. COS has a resolution of 1 ha per pixel, compatible with the nomenclature of the CORINE Land Cover map, which constitutes a reference product for LULC in Europe and Portugal (Table 1). The 83 LULC land classes from COS 2018 were reclassified into 17 classes (**Table S1**), depending on the vegetation and impermeabilization capacity. Due to the low spatial variability of intertidal habitats obtained from COS 2018, this LULC map was combined with an intertidal map of Sado estuary, previously published by Afonso et al. (2024). The combination of datasets with different resolutions (LULC - 1 ha; DEM – 2 m) can introduce some uncertainties, however, no additional detailed LULC maps were available. Despite this, InVEST can incorporate data with different resolutions.

The NDR model relies on a biophysical table that maps each LULC class to its biophysical properties related to nutrient loadings and retention. For Nitrogen (N) and Phosphorus (P), the table contains information on nutrient loading, efficiency of retention and critical length. Details can be found in **Appendix II**.

The Digital Elevation Model (DEM) was used to classify streams and to calculate slope (Eq. 7). A combined DEM, with two sources of data (Table 1), increased the precision in the area of interest. Bathymetry data was used to artificially replace the DEM mean sea-level by the hydrographic zero, so the intertidal habitats would mostly be out of water and be recognised as land by the model. Streams were classified considering a Threshold Flow Accumulation (TFA) of 100. TFA represents the number of upslope pixels that flow into a pixel before being classified as a stream. The threshold value (100) was selected based on the similarly with the hydrologic network previously published by Portuguese Environmental Agency.

To get the runoff potential index (RPI_i; **Eq. 4**) the annual averaged precipitation for a 20-year period (2004–2023) was used (Table 1). The authors are aware that a 20-year period may not be representative of current trends given the potential changes in the precipitation patterns in recent years. However, it was necessary to use a longer period to reduce spatial distortions in the face of data scarcity and to reduce the need to interpolate from point data. For interpolation purposes, the *Empirical Bayesian Kriging* from ArcGIS Pro was used. To avoid biased maps, data from meteorological stations from outside the study area, within a buffer zone of 30 m, was included in the interpolation process.

The Borselli k parameter (k_b; **Eq. 5**) is the calibration parameter that shapes the degree of connection from land to the stream, as well as the nutrient delivery ratio (*i.e.,* nutrient that reaches the stream). The InVEST default value (2) was used (Natural Capital Project, 2023).

2.3.2. *Validation of baseline simulation*

Water samples collected from seven estuary sites (Data not published, Fig. 3A) were used to validate the outputs from the NDR model. A buffer zone of three km was defined for each site. Comparison of water samples (validation data) with the NDR baseline simulation for the buffer zones (model data) showed a strong relationship ($R^2 = 0.767$ for P export and $R^2 = 0.625$ for N export) for both nutrients (Fig. 3B).

2.4. *Habitat quality model*

The *Habitat Quality* (HQ) model uses biodiversity as a proxy for habitat quality. In this study, the model was used to assess overall quality and degradation of each intertidal habitat and potential impacts that could lead to the loss of habitats and associated ecosystem services. According with the model guidelines, habitat quality is defined as the

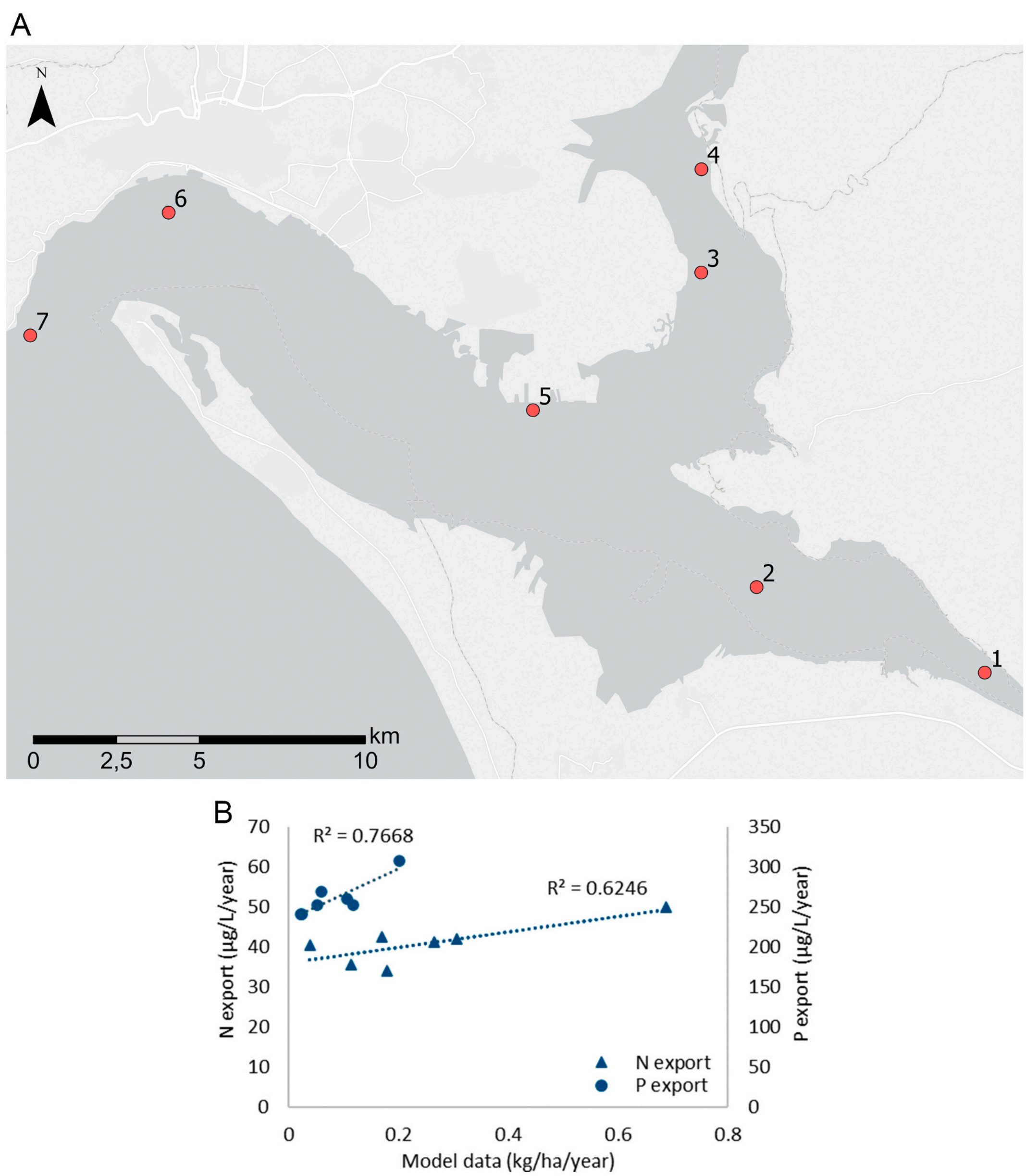

Fig. 3. Inset (A): sampling points in Sado estuary used to validate the modelled data. Inset (B): Linear regression comparing validation data and Nutrient Delivery Ratio model outputs, and associated determination coefficient (R^2).

ecosystem ability to provide conditions appropriate for persistent communities, whereas habitat degradation is defined as the cumulative impact of all identified threats on each habitat (Natural Capital Project, 2023). The model treats all habitats equally, assessing only the pressures in the area and the sensitivity of each habitat to those pressures. It is commonly used for conservation assessments, allowing for the evaluation of the extent and degradation of different habitat types within the study area.

The impact (i) of each pressure (r) in a grid cell (y) of a habitat (x) is quantified differently depending on whether the decay is linear (*i.e.*, impact is equal within the influence area) or exponential (*i.e.*, impact changes with the distance from the source). For linear decay (**Eq. 8**), the impact is defined based on the linear distance between grid cells x and y (d_{xy}) and the maximum effective distance of pressure across space (d_{max}). For exponential decay, the constant 2.99 (Natural Capital Project, 2023) reduces the impact of the pressure by 95 % at the d_{max} (**Eq. 9**).

$$i_{rxy} = 1 - \left(\frac{d_{xy}}{d_{r\ max}}\right) \text{ if linear} \tag{8}$$

$$i_{rxy} = exp\left(\left(-\frac{2.99}{d_{r\ max}}\right)d_{xy}\right) \text{ if exponential} \tag{9}$$

The total pressure impact (D_{yx}; Eq. 10) is measured based on the degradation weight of each pressure (w_r; *i.e.,* the relative damage of a pressure to all habitats), the impact of the pressure (i_{rxy}), the level of accessibility (β_x) and the relative sensitivity of habitat types to the pressure (S_{jr}). Each pressure must be mapped and can have a unique number of grid cells due to variations in raster resolution, with Z_r indicating the set of grid cells per pressure map.

$$D_{yx} = \sum_{r=1}^{R}\sum_{z=1}^{Z_r}\left(\frac{w_r}{\sum_{r=1}^{R} w_r}\right) r_y \bullet i_{rxy} \bullet \beta_x \bullet S_{jr} \tag{10}$$

The quality of the habitat (Q_{xj}; Eq. 11) comprehends the habitat suitability (H_j) and total pressure impact (D_{yx}). Two scaling parameters are used: s and k. The s is set to 2.5, and the k is the half-saturation constant that is set by the user.

$$Q_{yx} = H_j\left(1 - \left(\frac{D_{yx}^s}{D_{yx}^s + k^s}\right)\right) \tag{11}$$

Degradation and quality scores are opposites, when one increases the other decreases.

2.4.1. Baseline simulation: Information required and data preparation

Similarly to the NDR model, the LULC map resulted from a combination of a previous LULC map (DGT, 2018) and an intertidal map for habitats, previously developed in Afonso et al. (2024).

To run the HQ model, two types of information are essential: pressures and sensitivity of each habitat to each pressure. MarESA (Marine Evidence and Sensitivity Assessment https://www.marlin.ac.uk/) were used as the main guidelines to assess both. Due to the similarities of intertidal habitats between Portugal and the UK, the MarESA marine evidence and sensitivity assessment was considered acceptable to fill data needs. Regarding the pressures list, the main activities occurring in the estuary were first diagnosed and identified based on MarESA. The pressures associated with each activity and their influence on the intertidal habitats considered herein were identified (Fig. 4). The weight of each pressure was defined on a scale from 0 (very low impact) to 1 (very high impact), depending on the impact of each activity in terms of frequency and extension. Five categories were defined (See **Appendix III** for more detail). Furthermore, pressure extension was defined as the maximal distance of impact, with the minimal distance defined as the raster cell size of LULC map – 50 m (Afonso et al., 2024). The attenuation types can be continuous and similar (linear) or change with distance from the source point (exponential). For each identified pressures a map was created, considering the abundance or density of the pressure. Details can be found in **Appendix III**.

Habitat sensitivity includes information about habitat vulnerability to the listed pressures. The value is normalised to range between 0 (no sensitivity) and 1 (high sensitivity). The habitat sensitivity values were defined based on the MarESA sensitivity assessment for most habitats,

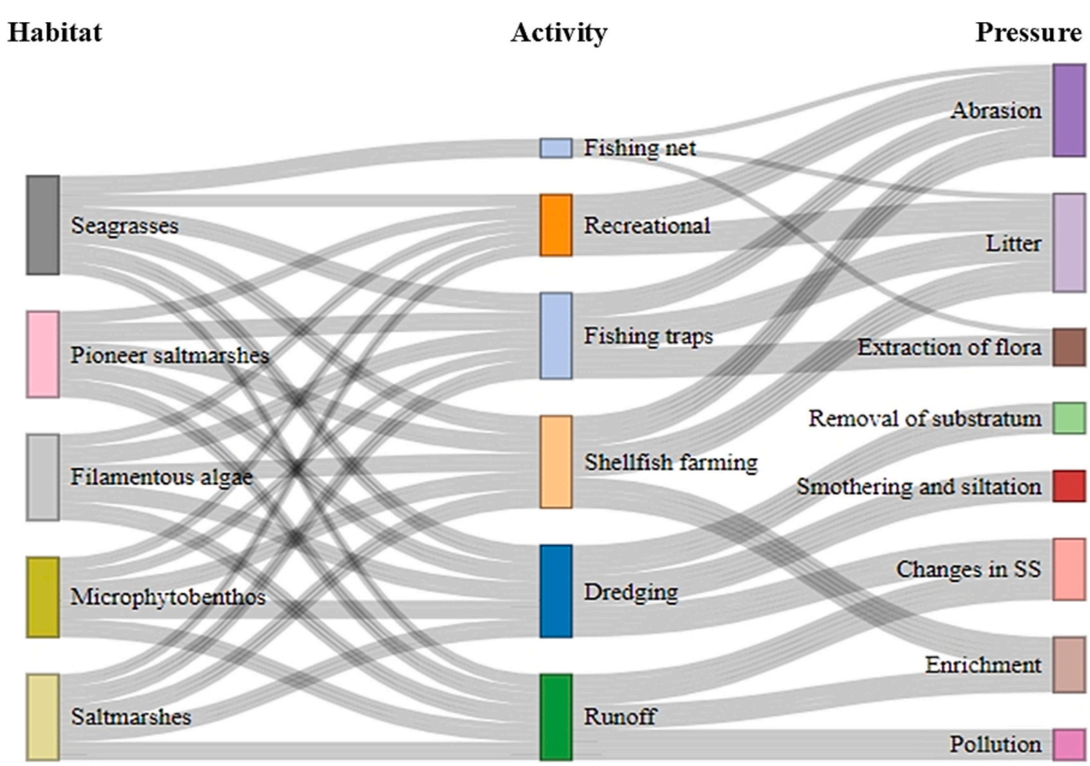

Fig. 4. Diagram connecting Habitat to activity and pressure, adapted from MarESA project (https://www.marlin.ac.uk/). SS - Suspended Solids.

except for saltmarshes, which were not included in the MarESA assessment. However, this sensitivity assessment does not consider the characteristics of the Sado estuary. Therefore, an advisory board of experts (four senior researchers) on ecology and estuarine processes in the Sado Estuary was consulted to assign values based on their perspectives. This information was used to validate the sensitivity assessment performed. Confidence in the assigned values was defined as 0.5 (*i.e.,* peer-review publications outside of study area), 0.8 (*i.e.,* MarESA https://www.marlin.ac.uk/) and 1.0 (*i.e.,* advisory board values agree with MarESA). The final sensitivity number results from the multiplication of both values. The values can be found in **Appendix III.**

2.5. Scenarios

Seven scenarios were developed: i) Business-as-Usual (BAU) scenario; ii) two Management Strategy scenarios focused on Ecological Protection *vs* Economic Development; iii) four Climate-change scenarios, two of each focused on changes in Precipitation patterns and Submersion due to sea-level rise. Seven were used to assess nutrient export, and three to assess habitat quality (BAU and two management strategy scenarios). Under the BAU scenario (Table 2), the 2023 River Basin Plan (RBP) predicted an increase in agricultural land and nutrient inputs from agriculture and livestock production in 2033. The LULC map and nutrient input (Biophysical Table) were adapted to this scenario. Regarding the LULC map, this scenario included a 9 % increase in agricultural areas and a 14 % increase in livestock production. Agricultural expansion was simulated by using the InVEST *Scenario Generator*, with these expansions occurring near existing agricultural areas and with the areas being replaced by either grazing land or forests, depending on proximity and availability. Grazing areas were considered more likely to transition to agriculture, while forests would require more preparation (*e.g.*, cutting trees, licenses). Consequently, the farming fields expansion was divided into 5 % for grazing areas and 4 % for forests. Nutrient inputs from RBP were incorporated into the Biophysical Table.

The two Management Strategy scenarios and four Climate Change-related scenarios were built onto the BAU 2033 scenario. The Management Strategy scenarios were designed to simulate the implications of nutrient export in the year 2050. The Ecological Protection (EP) scenario was developed in response to stakeholders' concerns about the input of toxic compounds from industry and farming industries in the estuary, and a general interest in reducing the nutrient and effluent pollution in the estuary (Afonso et al., n.d.). Since the European Green Deal goal will not be achieved in Portugal by 2030, considering the predictions in the RBP scenario, it could be achieved by 2050 through changes in agricultural and livestock production practices.

Table 2
Description of scenarios developed and employed for each model – Nutrient Delivery Ratio (NDR) and Habitat Quality (HQ). Climate-change scenarios were not applied to HQ model since does not affect habitat sensitivity.

Scenario, Year	Description	Model
Business-As-Usual (BAU), 2033	Simulation based on the principle that the trends of the last 10 years (2013–2023) will be repeated in the next 10 years (2023–2033), according to the 2023 River Basin Plan (APA, 2023).	NDR, HQ
Ecological Protection (EP), 2050	Simulation of nutrient export according with the Green Deal, nutrient export and pesticide reduced by 50 % (European Commission, 2019).	NDR, HQ
Economic Development (ED), 2050	Simulation of land-use changes by extending oyster production areas (17.58 km^2) into intertidal habitats.	NDR, HQ
Precipitation 2050, 2100	Simulation of changes in precipitation patterns, according to IPCC extreme scenario (AR5, RCP 8.5) (IPMA, Portal do Clima)	NDR
Submersion 2050, 2100	Simulation of changes in sea-level rise, according with IPCC extreme scenarios (AR6, RCP 8.5) (Antunes et al., 2019)	NDR

The Economic Development (ED) scenario was developed to predict possible consequences of investing solely in economic development in the region. In the Sado estuary, aquaculture activity has been growing over recent years and is expected to continue in the coming years (APA, 2023). This scenario incorporates the expansion of this activity (Table 2), according with the Project for Aquaculture Development in Portuguese Transitional Areas (DGRM, 2019).

Precipitation scenarios were considered to assess how rain pattern changes will affect nutrient export (Table 2). Two sets of data were used to predict how precipitation will change in 2050 (climatologic normal 2041–2070) and in 2100 (climatologic data 2071–2100). A climatologic normal is the averaged climate conditions over a standardised 30-year period. Unlike the Management Strategy scenarios, this one was run for two time periods: 2050, to compare with the Management Strategy scenarios, and 2100, to understand the potential implications of not addressing climate-change mitigation and adaptation actions adequately. This is crucial to inform local decision-makers.

Finally, for the Submersion scenario, models assessing how sea-level rise affects the Portuguese coast were used to identify areas that will be underwater in the coming years (Table 2). The scenario RCP 8.5 from IPCC is a conservative scenario, thus, it considers the worst-case scenario, which remains realistic under the current trend of emissions growth and limited efforts to mitigate impacts of climate-change. Therefore, the climate-change scenarios considered in this study were focused on the worst-case scenarios rather than in intermediate scenarios.

2.6. Sensitivity analysis

Sensitivity analysis was performed to assess the importance of the different parameters in both models. Commonly this type of analysis is used to assess how the variation of input parameters affects model outputs. For the NDR model, Redhead et al. (2018) was used as a reference for the sensitivity assessment approach (*i.e.*, variables to consider and variability in the input values). The criteria to choose the variables under analysis was to select numerical (*e.g.*, precipitation) or categorical variables (*e.g.*, critical length), since these are simpler to alter. Variability of the DEM and LULC was not considered due to lack of additional data to test the sensitivity.

Table 3 summarises all values tested for each parameter considered for the NDR model sensitivity analysis. Regarding the precipitation data, the climatologic normal for 2040–2070 (**Section 2.5.**) was used. The NDR model considers a ratio of precipitation (*i.e.,* precipitation per pixel divided by the average of all pixels). The high spatial homogeneity in precipitation data would signify an artificial lack of changes in precipitation. To accommodate this limitation, the current average annual precipitation (2004–2023) and climatologic normal (2040–2070) were used, which had a higher spatial heterogeneity. Lastly, precipitation increase and decrease of 50 % and 90 % were considered, following

Table 3
Sensitivity analysis variables and values associated. K_b – korselli value; N - Nitrogen; Normal – Climatological Normal (2040–2070); P - Phosphorus; TFA - Threshold Flow Accumulation.

Variables		Baseline value	Sensitivity Analysis
Precipitation		average 2004–2023	Normal ±50 %; ± 90 %
TFA		100	10; 1000; 10,000
K_b		2	0.5; 1; 4; 8
	N loading	Table S2	± 50 %; ± 90 %
	P loading	Table S3	± 50 %; ± 90 %
Biophysical Table	N efficiency rate	Table S2	± 50 %; ± 90 %
	P efficiency rate	Table S3	± 50 %; ± 90 %
	N critical length	1–60	± 20 m
	P critical length	1–60	± 20 m

Redhead et al. (2018); **Eq. 12**, Table 3).

$$[(Precip2023 - ClimNormal) \bullet (percentage\ of\ variation)] + ClimNormal \tag{12}$$

Regarding the TFA, three values (10; 1000; 10,000 – Table 3) were tested. These were selected following the work of Redhead et al. (2018) who showed that subtle variations in TFA made little difference, especially in larger catchments. Values below 100 were very likely to overestimate the stream network density and values higher than 10,000 created no watercourses (Redhead et al., 2018).

Regarding the k_b parameter, the value should be defined according to catchment characteristics (Redhead et al., 2018). However, due to a lack of field data, this study in concordance with Redhead et al. (2018) considered two categories of values below and higher than the default value (2): 0.5, 1, 4 and 8 (Table 3). Values higher than 8 made progressively less differences to the relationship between topography and nutrient delivery, and values below 0.5 collapse the function (Redhead et al., 2018).

Regarding the biophysical table variables, this study used the same methodology of Redhead et al. (2018) for nutrient loadings and efficiency rate (± 50 % and 90 %). The exception was the critical length variables, in this study a categorical variable within intervals of 20 m was used, thus, for the sensitivity analysis was used an addiction and reduction of one category (20 m; Table 3).

Sensitivity assessment was also performed for the HQ model. This is crucial to determine how the outputs are influenced by differences in habitat sensitivity to pressures. For each pressure individually, the weight was set to the value 0 (no weight) or 1 (maximum weight) for all intertidal habitats. To test for significant differences, the Kruskal-Wallis test (Kruskal and Wallis, 1952) was used due to the small number of samples. If significant differences (p-value$<$0.05) were found, Tukey's HSD was used to compare the multiple pairs and understand which pairs were causing the differences.

2.7. Statistical analysis

Outputs from the HQ model were analysed to identify significant differences. Initially, data was tested for normality (Shapiro-Wilks test) and homogeneity (Levene's test - Levene, 1960; Shapiro and Wilk, 1965). If the distribution was normal and data was distributed homogeneously, One-way ANOVA was used to assess significant differences. If the distribution was not normally distributed, the Kruskal-Wallis test was used. Finally, if significant differences (p-value$<$0.05) were found, Tukey's HSD was used to compare the multiple pairs and understand which ones were causing the differences.

3. Results

3.1. Impact of intertidal habitats on nutrient export

Export values were higher for nitrogen than for phosphorus (Fig. 5). Current situation maps presented higher export in the eastern and southern parts of the river basin, especially in agriculture and grazing fields (Fig. 5). The Sado river basin was dominated by grazing fields (26.97 %), forests (24.74 %) and waterbodies (19.31 %; **Table S1**). Nutrient export was higher in the river basin (10.66×10^5 N kg/year and 3.47×10^5 P kg/year), however, the Sado estuary was also a large nutrient contributor (6.04×10^5 N kg/year and 2.02×10^5 P kg/year; Table 4).

Assuming that the nutrients that reach a pixel are retained (if not exported), the amount of nutrients potentially retained by each land-use was calculated (Table 5). Two of the Baseline simulation outputs, modified load (See **Section 2.3.** for more details) and nutrient export, were used to calculate nutrient retention. All intertidal habitats present high nutrient retention rates (> 80 %), especially saltmarshes and pioneer saltmarshes (> 90 %; Table 5). In contrast, waterproof and extraction sites (Definition in **Table S1**) retain fewer nutrients (< 72 %; Table 5).

3.2. Scenario influence on nutrient export

Seven scenarios were simulated by the NDR model. Figs. 6 to 8 shows the spatial differences between scenarios. The minimum value defined as a change in nutrient export was 0.001 kg/pixel/year for Phosphorus (P) and 0.01 kg/pixel/year for Nitrogen (N). In the BAU scenario (Fig. 6) an increase of nutrient export was projected relative to the current situation (baseline), especially in the eastern and southern parts of the river basin. In the EP scenario (Fig. 7) a decrease in nutrient export was projected throughout the entire basin. All of the other scenarios were not significantly different from the BAU scenario, with small changes over the study area (Fig. 8**, S1**). In both Precipitation scenarios only differences in P export were projected, with an increase in the centre of the river basin and a decrease in the south (Fig. 8). In the Submersion scenarios was predicted that 1836.06 ha will be underwater in 2050, and 5325.13 ha in 2100, however, few changes in nutrient export were projected with an increase in submerged grazing fields and a decrease in submerged rice fields (**Fig. S1**).

3.3. Quality and vulnerability of intertidal habitats

Habitat quality was calculated as a percentage (1–100 %), and according to the attributed value four categories were defined: quality under 25 % - poor; quality between 25 and 50 % - moderate; quality between 50 and 75 % - good; quality over 75 % - excellent. Currently (baseline simulation), all areas occupied by saltmarshes were evaluated as being of excellent quality (Table 6). Large areas inhabited by pioneer saltmarshes and filamentous algae (>60 %) were also considered of excellent quality (Table 6). Conversely, more than 90 % of seagrasses were of low or moderate quality (Table 6).

Habitat degradation outputs are related to Habitat Quality Outputs (Table 7) with most habitats presenting as low degradation (>90 % of area; Table 7). In contrast, most (64.91 %) of the areas covered by seagrass were considered moderately degraded (Table 7).

3.4. Scenario influence on habitat quality and vulnerability

Only the BAU and Management Strategy scenarios were simulated by the HQ model. Table 8 compares differences (%) between scenarios in habitat area extent in the different categories of habitat quality. The EP scenario caused few or no changes in habitat quality, in comparison with the BAU scenario (Table 8). The extent of saltmarshes and filamentous algae habitat of excellent quality was expected to decrease in all scenarios (Table 8). However, while saltmarshes will maintain large areas of excellent quality, the extent of poor-quality filamentous algae habitat was expected to considerably increase, up to 40 % in the BAU and EP scenarios, and up to 63 % in the ED scenario (Table 8). The extent of pioneer saltmarshes in excellent quality was expected to increase (67 % -73 %), as was the extent of those in poor quality (13 % - 28 %; Table 8). Major changes for Microphytobenthos included a decrease in extent of good quality habitat with an increase in poor quality habitat (Table 8). The quality of seagrasses was not expected to change significantly, except for in the ED scenario where an increase of poor-quality habitat was expected (Table 8). No significant differences were found between scenarios (**Tables S7-S11**).

Habitat degradation was expected to increase in the BAU scenario, when compared to the current situation (baseline; Table 9). The EP scenario again presents almost no differences from the BAU scenario (Table 9). Areas with low degradation were expected to increase in the BAU scenario and decrease in the ED scenario (Table 9). No significant differences were found (**Tables S12-S14**).

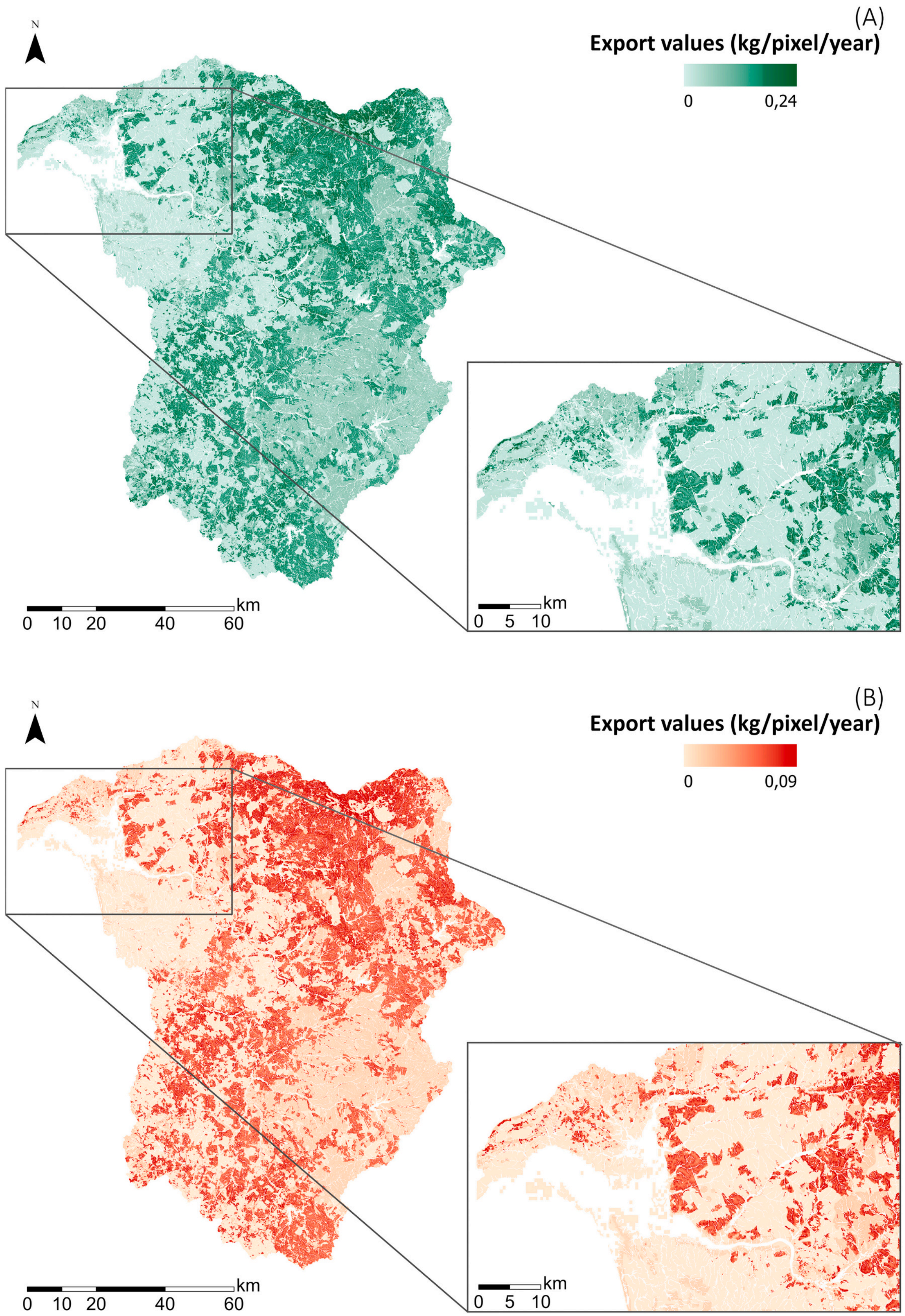

Fig. 5. Nitrogen (kg/pixel/year) (A) and Phosphorus (kg/pixel/year) (B) export in Sado river basin in the current situation (Baseline outputs). Right bottom insets: clip of estuary basin export of Nitrogen and Phosphorus.

Table 4
Total nutrient loading (*10^5 kg/year) and export (*10^5 kg/year) of Nutrient Delivery Ratio model. Values for the sub-watershed (Sado estuary) and Sado river watershed. N - Nitrogen; P - Phosphorus.

Nutrient	Sado Sub-watershed				River Watershed			
Scenarios	N		P		N		P	
	Load	Export	Load	Export	Load	Export	Load	Export
Baseline	29.35	6.04	9.80	2.02	52.02	10.66	17.02	3.47
Business-as-Usual	32.66	6.76	11.00	2.27	57.64	11.87	19.04	3.89
Ecological Protection	17.62	3.61	5.57	1.15	31.14	6.36	9.66	1.98
Economic Development	32.79	6.77	11.07	2.28	57.78	11.88	19.11	3.90
Precipitation 2050	32.63	6.75	10.99	2.27	57.64	11.87	19.03	3.89
Precipitation 2100	32.68	6.76	11.01	2.28	57.68	11.88	19.04	3.89
Submersion 2050	32.64	6.75	11.00	2.27	57.65	11.87	19.04	3.89
Submersion 2100	32.62	6.75	11.00	2.27	57.63	11.87	19.03	3.89

Table 5
Percentage of nutrient retention (Nitrogen and Phosphorus) calculated by land use class. Table S6 presents all the absolute values. Grey rows identify intertidal habitats.

	N retention	P retention
Agriculture	75.69	75.69
Agriculture trees	77.49	77.49
Eucalyptus forest	83.05	83.05
Extraction sites	67.52	67.44
Filamentous algae	83.04	83.03
Forest	83.18	83.18
Grazing field	79.92	79.92
Green space	83.57	83.57
Microphytobenthos	84.03	84.04
Pioneer saltmarshes	90.06	90.07
Saltmarshes	94.10	94.10
Seagrasses	83.57	83.57
Semi-waterproof surfaces	76.33	76.33
Shellfish farming	90.22	90.22
Shrubland	81.43	81.43
Waterbody	80.03	80.03
Waterproof	71.82	71.82

3.5. *Sensitivity assessment*

Sensitivity analyses were performed on the NDR and HQ model. Nutrient loading and efficiency rate had the highest impact in NDR outputs (Fig. 9), however, no parameter had a significant impact on the model outcomes (*p*-value >0.05; **Table S15**). The sensitivity analysis performed on the HQ model parameters (**Table S16**) showed significant differences for runoff in critical zones (p-value <0.05; **Table S17**). When runoff was set to a minimal value (0), the high-quality areas increase by 23.76 %, and when set to maximal value (1), the low-quality areas increase by 93.63 % (**Table S16**). Whereas analysis to degradation component (**Table S18)** presented significant differences for dredging and runoff in critical zones (**Table S19**).

4. Discussion

4.1. *Influence of intertidal habitats on nutrient export*

This study findings indicated a higher export of N, than P, consistent with previous research employing this approach (*e.g.,* Majumdar and Avishek, 2024). This can be considered an issue for estuarine and marine ecosystems where N is a limiting nutrient (*i.e,* the demand for a nutrient is higher than the stocks of it; Smith et al., 1999). The different concentrations of N and P export may be due to the form of each nutrient, while N is commonly dissolved and easily transported, on the other hand P is often bound to sediments and not readily available (National Council Research, 2000). Agriculture and grazing fields from Sado watershed are the primary sources of nutrients to estuarine waters, a pattern also reported in NDR simulations conducted by Majumdar and Avishek (2024). This tendency (higher N contamination from agricultural runoff) was also observed in previous studies performed in Sado estuary (Caeiro, 2004). Eutrophication is not currently a problem in Sado estuary and water quality is within standard levels (Biguino et al., 2024), understanding nutrient sources of nutrients and their potential impacts on local biodiversity and habitats is crucial.

Nutrient delivery maps showed that intertidal habitats, particularly saltmarshes and pioneer saltmarshes, exhibited low nutrient export and high retention (> 90 %). Commonly, saltmarshes can denitrify N delivered to the systems, transforming nitrate into other reduced forms (*e.g.,* NO_2 to N_2), and retain the P that reach the system (National Research Council, 2000). Thus, these habitats located near areas with high nutrient export, such as rice fields, might retain excessive nutrients, preventing excessive export to the water column. Rice fields require flooding during most of their growing period, in a single growing season 2 to 22 kg of N per ha of fertiliser are used (Zhao et al., 2012). Without saltmarsh buffers, excess nutrients could contaminate the water column.

The Sado estuary basin receives nutrients from the entire catchment (Biguino et al., 2024). Despite expectations of higher nutrient export from the river catchment, the estuary catchment encompasses more than half of the nutrient exported from river basin (56.66 % of N and 58.21 % of P). Even though the Sado estuary catchment is smaller, its land-uses have caused higher contamination of the streams.

Additionally, it is crucial to recognise potential uncertainties associated with the outputs of the NDR model. A potential source of uncertainty arises from using datasets with different spatial resolutions, which can lead to spatial misalignment. Here, DEM (2 m) and LULC (1 ha) were used, due to an absence of detailed and high-resolution LULC maps. In spatial modelling, the mismatch between datasets can impact the outputs due to generalisation of land-cover or introduction of artificial smoothing of slope. This is particularly aggravated in fragmented areas, where the outputs may not capture the fine scale resolution, especially with small-scale habitats, such as seagrass patches. However, InVEST models can accommodate different resolution mismatches by resampling and aligning the input data to a common grid (Natural Capital Project, 2023). Moreover, this NDR modelling assessment does not consider point sources of nutrients, such as effluents from local industries and Wastewater Treatment Plants (WWTP). According to the regional River Basin Management Plan, the local WWTPs and industries contribute with 5.74 % and 15.5 % of total N and P loading in the system, respectively (APA, 2023). A fraction of these nutrients may be found in Sado's streams, which will imply a higher export of nutrients than what was simulated by the model (Smith et al., 1999).

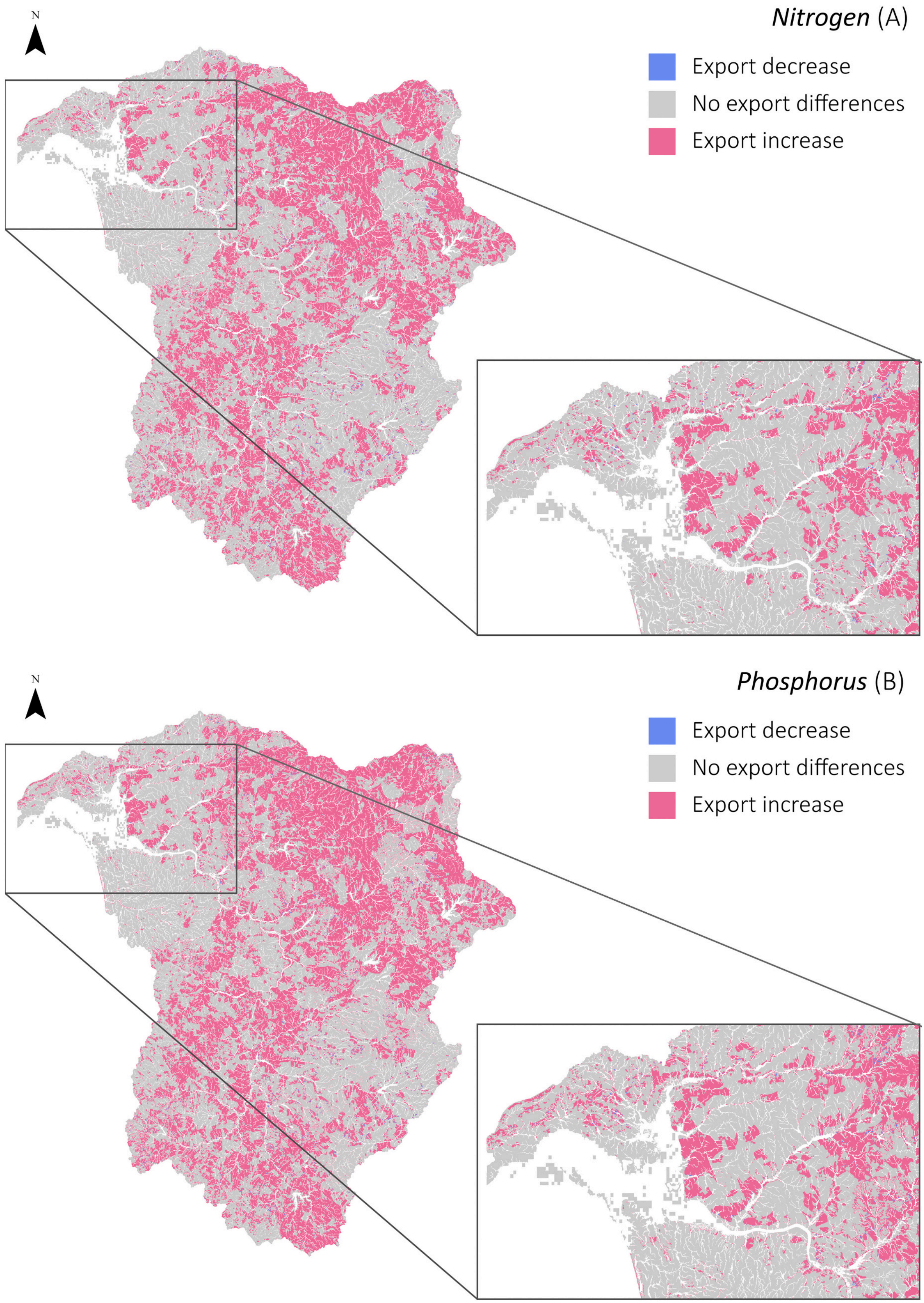

Fig. 6. Differences of Nitrogen (A) and Phosphorus (B) export between the Business-as-Usual scenario (2033) and the baseline output (current situation). Business-as-Usual scenario predicts changes in nutrient export if last 10 year's trend is maintained through the next 10 years. Right bottom insets: clip of estuary basin difference of Nitrogen and Phosphorus export. Export differences were only considered in values higher than 0.001 kg/pixel/year for Phosphorus and 0.01 kg/pixel/year for Nitrogen. Grey pixels indicate no changes in nutrient export, blue pixels indicate a decrease, and pink pixels indicate an increase.

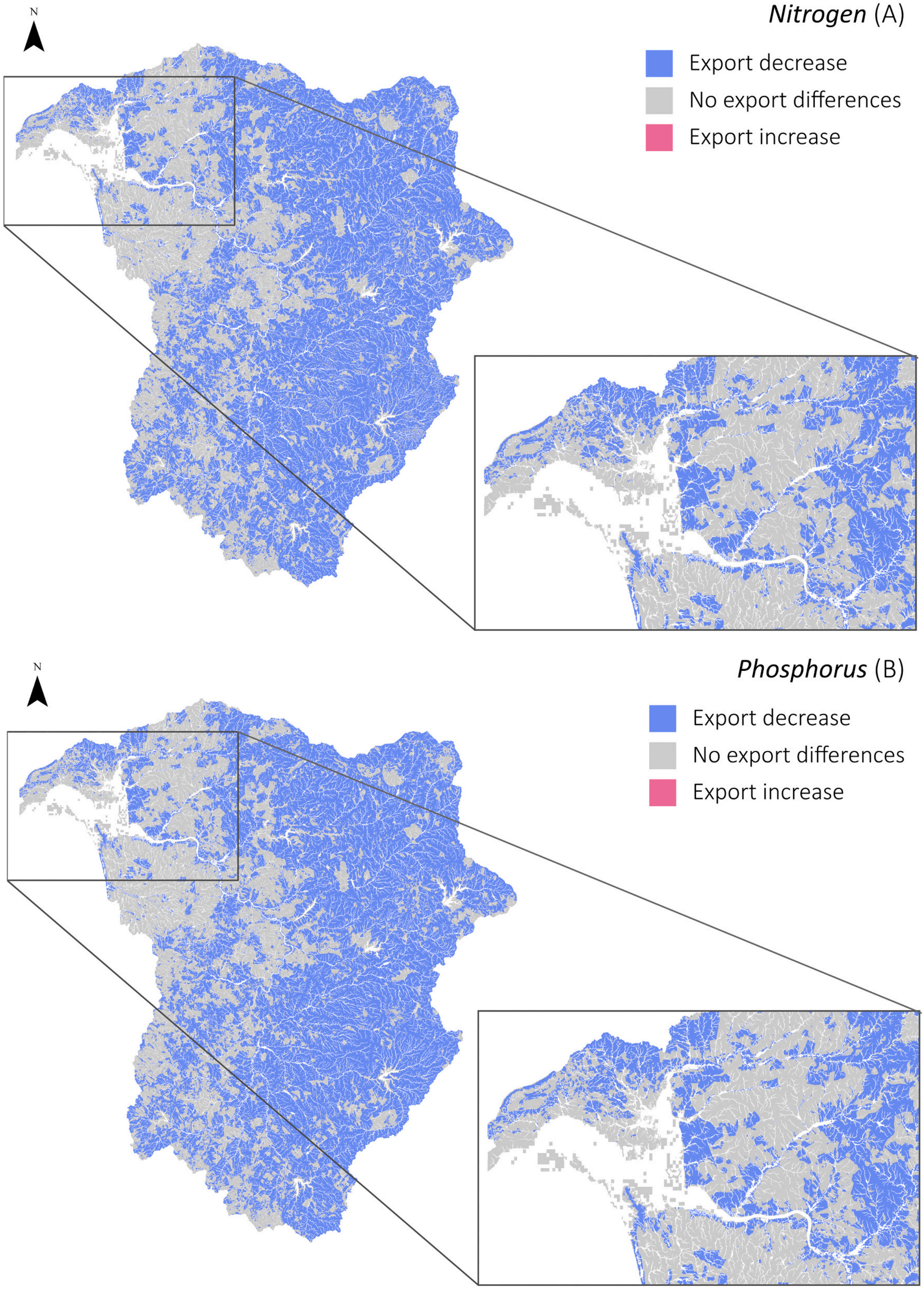

Fig. 7. Differences of nutrient export between Management Strategy (2050) and Business-as-Usual (2033) scenarios. Insets A and B: Ecological Protection scenarios, Nitrogen (A) and Phosphorus (B). Insets C and D: Economic Development scenarios, Nitrogen (C) and Phosphorus (D). Business-as-Usual scenario predicts changes in nutrient retention if last 10 year's trend is maintained through the next 10 years. Ecological Protection scenario predicts changes in nutrient retention until 2050 if Green Deal measures are put in practice in the study area. Economic Development scenario predicts changes in nutrient retention if shellfish farming is extended in the study area, one of the most developed economic activities in the study area. Right bottom insets: clip of estuary basin difference of nutrients export. Export differences were only considered in values higher than 0.001 kg/pixel/year for Phosphorus and 0.01 kg/pixel/year for Nitrogen. Grey pixels indicate no changes in nutrient export, blue pixels indicate a decrease, and pink pixels indicate an increase.

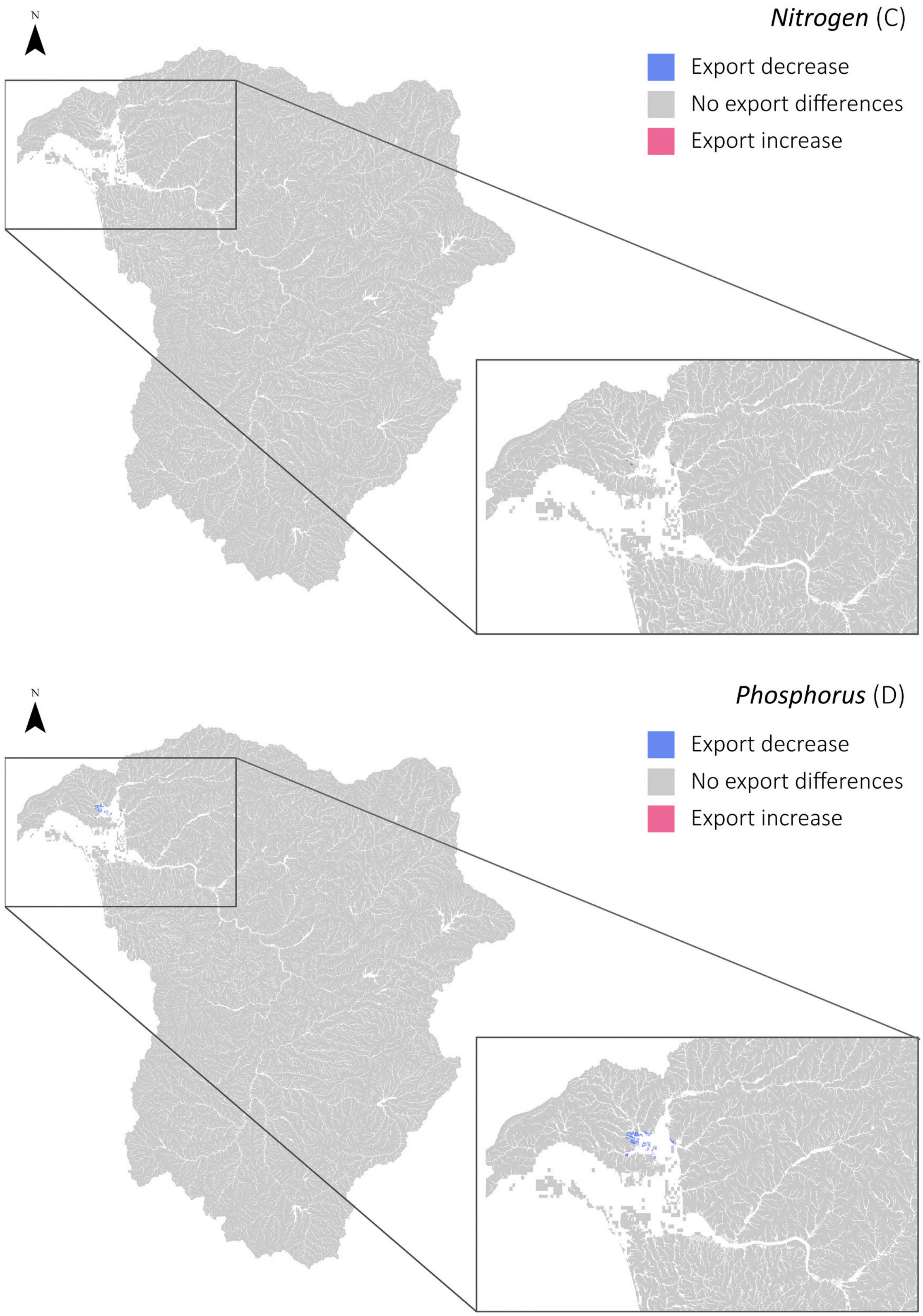

Fig. 7. (*continued*).

4.2. How are land use and nutrient export influenced by climate-change and management strategy scenarios?

The BAU scenario predicted a substantial increase in nutrient export, which is aligned with findings from other studies (*e.g.*, Han et al., 2021), where agricultural and grazing fields increased nutrient export due to higher nutrient loadings in these land-uses. Contrarily, the EP scenario had an expected decrease in nutrient export, similarly to results were found in Banerjee et al. (2024), where sustainable agricultural practices, with reduced nutrient input, resulted in reduced export, particularly in scenarios incorporating climate-smart agriculture. Combining changes in agricultural practices with active conversion measures (*e.g.,* restoring

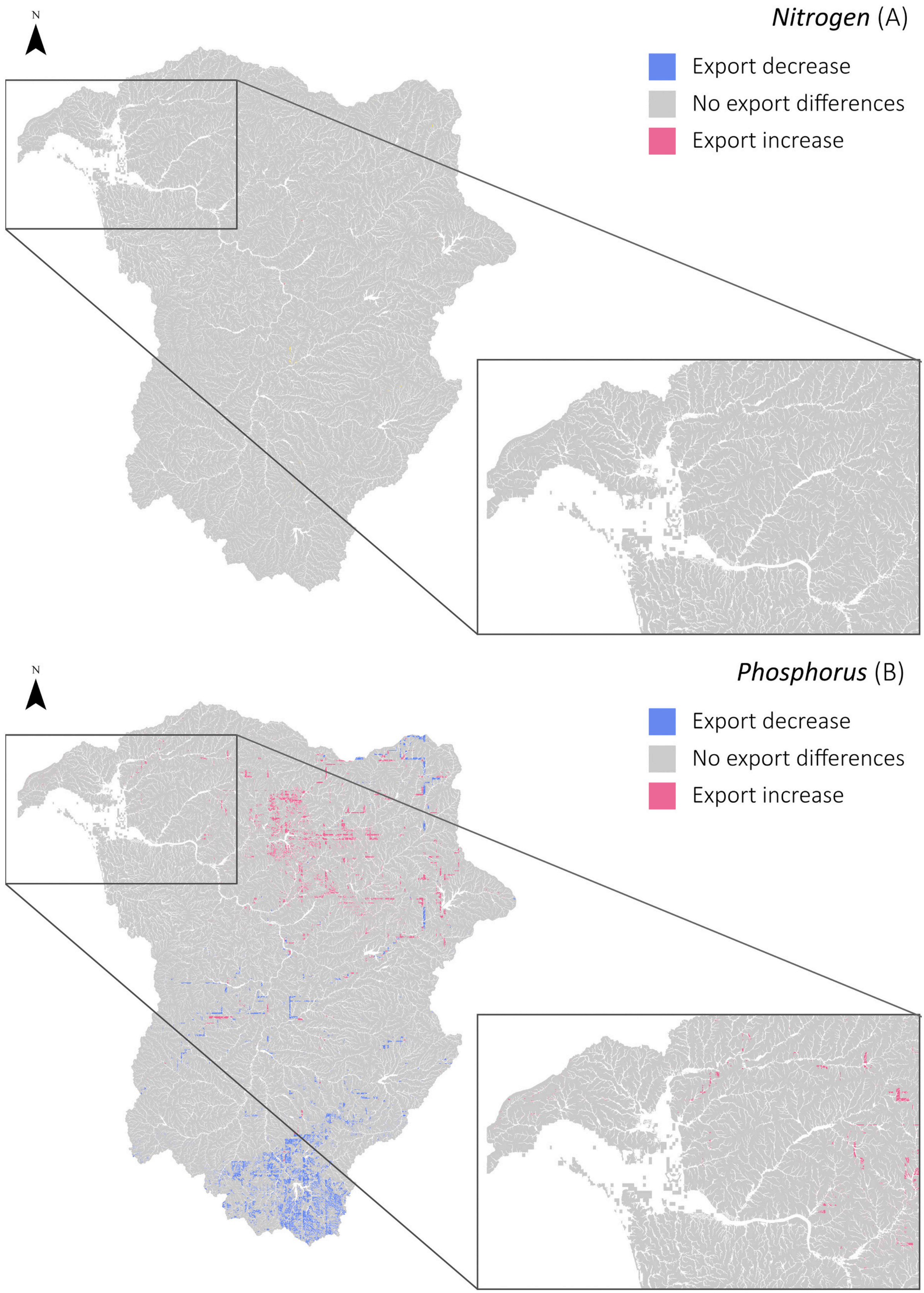

Fig. 8. Differences of Nitrogen (Insets A, C) and Phosphorus (Insets B, D) export between the Precipitation scenario for 2050 (A, B) and 2100 (C, D) and Business-as-Usual (2033) scenarios. Business-as-Usual scenario predicts changes in nutrient retention if last year's trend is maintained until 2033. Precipitation scenarios are designed according with the IPCC report worst-case scenario, which predicts changes in precipitation patterns for next years. Right bottom insets: clip of estuary basin difference of Nitrogen and Phosphorus export. Export differences were only considered in values higher than 0.001 kg/pixel/year for Phosphorus and 0.01 kg/pixel/year for Nitrogen. Grey pixels indicate no changes in nutrient export, blue pixels indicate a decrease, and pink pixels indicate an increase.

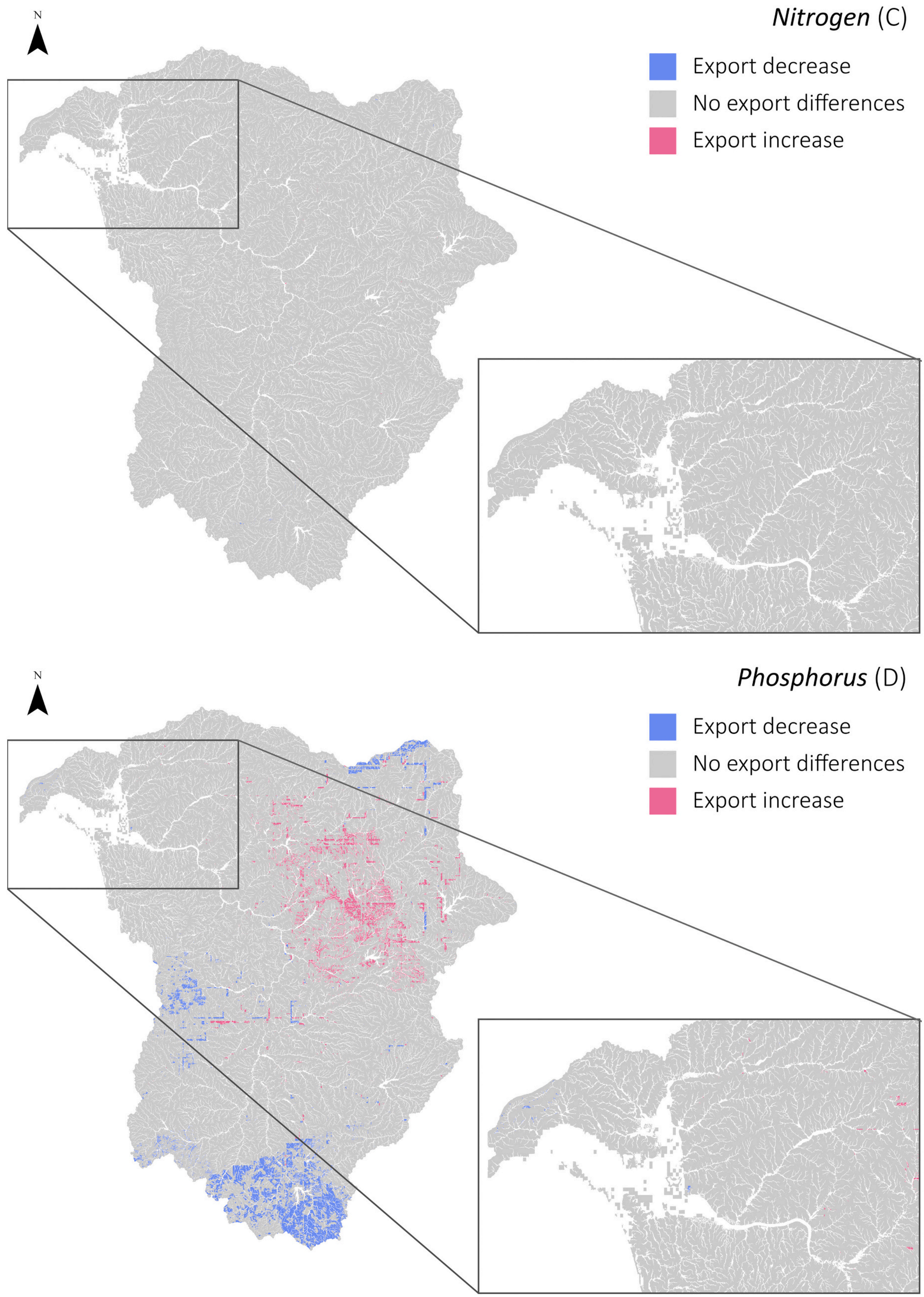

Fig. 8. (*continued*).

wetland vegetation) can bring substantial improvements into habitat conservation and health. The reduction of intensive land use combined with the reintroduction or enhancement of natural vegetation in agricultural dominated watersheds can support the recovery of biodiversity, increase resilience against climate-change impacts, and improve soil and water quality. In fact, Saraiva et al. (2007) conducted a

Table 6
Percentage of habitat area classified by habitat quality. Four categories were defined: poor (<25 %), moderate (25–50 %), good (50–75 %), and excellent (>75 %).

	Poor	Moderate	Good	Excellent
Filamentous algae	17.53	0.00	0.00	82.47
Microphytobenthos	13.96	2.38	83.09	0.56
Pioneer saltmarshes	6.42	0.00	29.90	63.68
Saltmarshes	0.00	0.00	0.00	100.00
Seagrasses	33.04	66.23	0.65	0.07

Table 7
Percentage of habitat area classified by habitat degradation. Two categories were defined: low degradation (0–25 %) and moderate degradation (25–50 %).

	Low	Moderate
Filamentous algae	99.83	0.17
Microphytobenthos	99.21	0.79
Pioneer saltmarshes	99.68	0.32
Saltmarshes	99.01	0.09
Seagrasses	35.09	64.91

modelling study focused on nutrient export in Portuguese estuaries and showed that in Sado estuary a reduction in N export (50 %) would cause a decrease in phytoplankton production with strong impacts through the food web.

Shellfish production has shown to improve water quality by removing particulates and assimilating dissolved nutrients from the water column (Brito et al., 2023). Previous studies have demonstrated that each oyster individual is able to remove $0.7*10^{-5}$ kg of N and $0.4*10^{-5}$ kg of P over the course of one year (Mao et al., 2006). However, the ED scenario did not reflect any changes in nutrient content. Under this scenario, intertidal habitats were replaced, especially seagrasses and filamentous algae (assigned with low nutrient input – 0.08 kg/ha/year - and lower nutrient retention efficiency – 83 %), with shellfish farms (assigned with higher nutrient input – 7.226 kg/ha/year – and higher nutrient retention efficiency – 90 %). The increase in nutrient retention may have been counterbalanced by the simultaneous rise in nutrient input leading to negligible changes in nutrient export. Rioux and Strong (2023) have highlighted that the NDR model responds better to severe changes in LULC map, as seen in simulations where converting every natural land-use into a developed area resulted in a significant increase in nutrient export.

Sensitivity assessment revealed that the NDR model was not responsive to changes in precipitation, explaining the rather low changes detected in the Precipitation scenarios. It would not be reasonable to consider these scenarios for decision-making, since the model could not develop realistic outputs. The Submersion scenario for 2050 predicted sea-level rise impacts on the Tróia and Marateca channels, submerging small dune habitats and part of saltmarshes in Tróia,

Table 8
Differences in habitat area (%) between scenarios and for each habitat quality category. Each column represents the changes between two outputs: Baseline outputs and Business-as-Usual (BAU); BAU and Ecological Protection (EP); BAU and Economic Development (ED). Red cells indicate an increase in area, and blue cells indicate a decrease in area. BAU scenario predicts changes in nutrient retention if last 10 year's trend is maintained through the next 10 years. EP scenario predicts changes in nutrient retention until 2050 if Green Deal measures are put in practice in the study area. ED scenario predicts changes in nutrient retention if shellfish farming is extended in the study area, one of the most developed economic activities in the study area.

Poor Habitat Quality (<25%)			
	BAU	**EP**	**ED**
Saltmarshes	↑3.09	0	↑9.81
Pioneer saltmarshes	↑6.29	0	↑14.99
Filamentous algae	↑22.51	0	↑20.28
Microphytobenthos	↑17.46	0	↑31.44
Seagrasses	↓8.35	↓0.16	↑42.1
Moderate Habitat Quality (25-50%)			
	BAU	**EP**	**ED**
Saltmarshes	↑0.03	0	↑0.01
Pioneer saltmarshes	↑0.26	0	↓0.04
Filamentous algae	↑0.40	0	↑0.02
Microphytobenthos	↓0.33	0	↑0.20
Seagrasses	↑8.06	↓0.49	↓43.94
Good Habitat Quality (50-75%)			
	BAU	**EP**	**ED**
Saltmarshes	↑1.45	0	↑0.45
Pioneer saltmarshes	↓16.33	0	↓8.62
Filamentous algae	↑1.78	0	↑1.72
Microphytobenthos	↓17.58	0	↓31.62
Seagrasses	↑0.37	0	↑1.45
Excellent Habitat Quality (75-100%)			
	BAU	**EP**	**ED**
Saltmarshes	↓4.57	0	↓16.26
Pioneer saltmarshes	↑9.78	0	↓6.33
Filamentous algae	↓24.69	0	↓22.02
Microphytobenthos	↑0.45	0	↓0.02
Seagrasses	↑0.59	↓0.66	↑0.21

Table 9
Differences in habitat area (%) between scenarios and for each habitat degradation category. Each column represents the changes between two outputs: Baseline outputs and Business-as-Usual (BAU); BAU and Ecological Protection (EP); BAU and Economic Development (ED). Red cells indicate an increase in area, and blue cells indicate a decrease in area. BAU scenario predicts changes in nutrient retention if last 10 year's trend is maintained through the next 10 years. EP scenario predicts changes in nutrient retention until 2050 if Green Deal measures are put in practice in the study area. ED scenario predicts changes in nutrient retention if shellfish farming is extended in the study area, one of the most developed economic activities in the study area.

	BAU	EP	ED
Low degradation (0 - 25%)			
Saltmarshes	↑0.03	0	↑0.02
Pioneer saltmarshes	↓0.75	0	↓0.18
Filamentous algae	↓0.13	↓0.04	↑0.13
Microphytobenthos	↑0.15	0	↓0.12
Seagrasses	↓10.29	0	↑33.64
Moderate Degradation (25-50%)			
	BAU	**EP**	**ED**
Saltmarshes	↓0.04	0	↑0.02
Pioneer saltmarshes	↓0.06	0	↑0.17
Filamentous algae	↓0.13	0	↓0.13
Microphytobenthos	↓0.15	0	↑0.12
Seagrasses	↑10.27	0	↓33.67

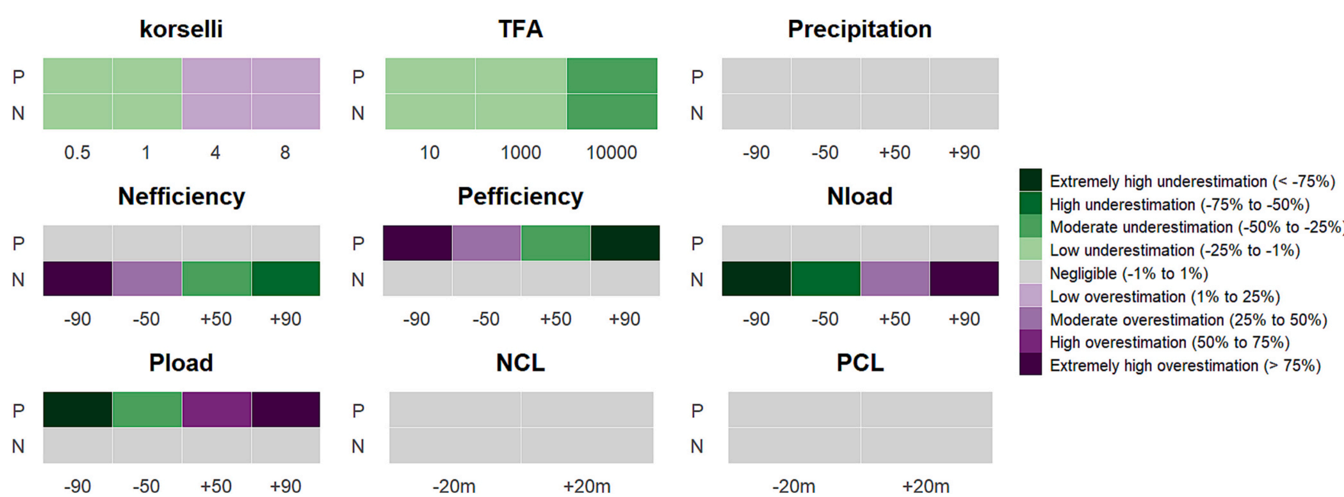

Fig. 9. Sensitivity of Nutrient Delivery Ratio (NDR) model to changes in parameters. Values attributed based on changes in total nutrient export (for nitrogen and phosphorus). Parameters included: Korselli (K_b), Threshold flow accumulation (TFA), Precipitation, retention efficiency of Nitrogen (N) and Phosphorus (P), N and P loading, and critical length of N and P. Green cells indicates an increase of nutrient export with the tested values, and purple cells indicates a decrease of nutrient export with the tested values.

and fish farming tanks, rice fields and grazing areas in Marateca. By 2100, these areas will be further submerged. However, in terms of nutrient export no significant changes were predicted. These findings suggest that sea-level rise may not immediately alter nutrient dynamics despite the changes caused in the LULC map. However, these results must be interpreted with caution. While the impact of sea-level rise on wetlands is poorly understood, it is known that wetland erosion can accelerate and alter sediment dynamics under longer periods of submersion, reducing habitat stability and increase their deterioration (Kirwan and Megonigal, 2013). This degradation compromises ecological functions, such as nutrient retention. The NDR model cannot simulate progressive degradation or loss of habitats. Therefore, in the long-term sea-level rise may impact negatively intertidal habitats, and consequently, impact nutrient export.

4.3. *How is habitat quality and vulnerability influenced by management scenarios?*

Currently, most intertidal habitats presented little degradation. As with other studies (*e.g.,* Huang et al., 2024), saltmarshes are generally in best condition. Contrarily, a large proportion of seagrass habitat was moderately degraded, suggesting they are at potential ecological risk and may already be under significant pressure. In agreement with previous studies, seagrass meadows in the Sado estuary were more degraded than other meadows in Portuguese estuaries, due to a deficiency in P (Vieira et al., 2022). Perez et al. (1991) showed that P plays an important role in seagrass growth. Although seagrasses have root systems that enable them to uptake P from sediment (Alexandre and Santos, 2020), it is possible that the nutrient requirements are not fulfilled in Sado estuary. Rice production may be the main cause of low *P* values, since this plant requires high concentrations of P to grow (Jiang et al., 2021). Local efforts are in place to protect and monitor seagrasses, such as the project Ocean Alive (https://www.ocean-alive.org/). However, there are still some additional measures that could be implemented, such as restoration projects.

The quality of almost all intertidal habitats decreased in the BAU scenario, except for seagrasses, which appear to thrive under increased nutrient export. These results align partially with Deng et al. (2024), who observed that habitat quality tends to decrease with agriculture expansion. Regarding seagrasses, the potential P insufficiency appears to have been overcome in the baseline simulation, leading to an improvement in the quality of seagrasses. Contrary to what was expected, the EP simulation did not improve the habitat quality, suggesting that reduction in agriculture pressure may be insufficient to reverse existing habitat degradation. Deng et al. (2024) noted that scenarios of conversion of unused or paddy lands to forests and wetlands substantially improved habitat quality. This suggests that more active land-use interventions may be more effective to yield ecological benefits.

The ED scenario presented extensive changes in habitat quality and degradation. Surprisingly, all habitats presented an increase in areas of both poor quality and low degradation. These contradictory outputs might be due to replacement of intertidal habitats by shellfish farming structures. In this scenario, degraded intertidal habitats were replaced by shellfish structures, and non-degraded habitats were preserved. Thus, the changes observed are caused by a model limitation which does not allow combined habitats (*i.e.,* overlap of two or more predefined land uses, such as seagrasses with shellfish farming). In summary, the changes observed do not reflect a change in quality or degradation of these habitats, only a change of land-use which resulted in a change of ratio between classes. Moreover, other studies indicated that high economic development, where land uses are overdeveloped, leads to poor habitat quality and low biodiversity (Sun et al., 2023), which is in according to our findings. Due to limitations in data availability, the HQ model outputs could not be calibrated or validated, thus, may not fully represent the actual conditions of the habitats. Without field-data for comparison, it is not possible to assess if the model is over- or underestimating the habitat quality. Therefore, the results must be interpreted carefully. While other studies corroborate the low quality of seagrasses in Sado estuary, supporting some of the model outputs, future simulations should still be interpreted with caution, given the uncertainties arising from unaccounted variables. In future studies, Habitat Suitability models based on Machine learning tools could be used to test this scenario, and implications for habitat quality.

4.4. Implications for decision-making

The Submersion simulations predict that by 2100 the area underwater will triple the submersion areas of the 2050 scenario with potentially wide repercussions for estuarine ecology and local stakeholders. To mitigate sea-level rise impacts, climate-change adaptation measures are essential, such as the recently implemented Municipal plan for Climatic Action for Setúbal.

The BAU scenario predicted an increase of nutrient export to the streams which may be counterbalanced with protecting and monitoring existing buffer habitats. Land-use intensification in the Sado watershed is likely to happen in the future, since rice production is a long tradition in the region and organic production is still unfeasible. Thus, increasing or establishing new buffer zones can be an important measure to further prevent nutrient export. Considering the predicted increase of damaging activities and alterations caused by climate-change over the coming decades, spatial planning should be conducted over long periods (>20 years) by employing a proactive and adaptative approach, which anticipates and prevents future issues that are not detectable in short-term decisions that may worsen environmental degradation. For example, action plans that combine the promotion of traditional agriculture practices by employing new practices that require the use of less fertilisers to avoid nutrient enrichment and deleterious impacts on the water quality. This may also be applicable to mitigate sea-level rise, which must be performed in an informed, adaptative, and long-term approach. Therefore, aligning with EU policy and particular the Water Framework Directive (Directive 2000/60/EC) goals to integrate early action measures and long-term catchment management.

The EP and ED scenarios yielded opposing results. The former positively impacted the nutrient export but did not change habitat quality, whilst the latter did not significantly affect nutrient export but had negative consequences for habitat quality. Future research could explore a combined strategy for management that adopts new methods to minimise the use of fertilisers and pesticides, fostering economic growth while safeguarding environmental health. However, to further improve habitat quality in Sado estuary, more active land-use changes are needed, which may include converting rice fields, which may be submerged by 2050, into natural habitats (*e.g.*, saltmarshes). Additionally, future studies should be performed to understand the impact of shellfish farming structures in the intertidal habitats, especially in seagrasses which are under poorer conditions.

4.5. Model limitations

InVEST provides a set of models, which are cost-effective and user friendly. However, they also have several limitations, including: exclusive mapping of one ecosystem service per model, inability to reflect seasonal variations or extreme events, and challenges in interpreting outputs, particularly for non-technical stakeholders. Specifically, the NDR model overlooks water components that influence nutrient transport, such as currents, assuming that nutrients are evenly distributed once they reach a stream. In addition, both phosphorus and nitrogen are modelled identically, despite differing cycles. Therefore, results should be considered with caution since N is a highly mobile nutrient, being easily transported, whereas P is usually bound to sediment, thus, it is not easily accessibly. Regarding the model outputs, this limitation can potentially imply a more accurate measure of N export than P. Moreover, this a so called "black box model", which limits the use to edit the input data and assess the outcomes, however, the user cannot change the internal logic and processes. This can limit their use for future scenarios, since it does not comprehend the complexity of these scenarios. For example, in a submersion scenario the model only considerers the changes in land use, however, does not consider the changes in habitat quality. Future studies aiming to validate InVEST outputs could consider simulating relevant variables using other models, such as SWAT.

Moreover, the HQ model relies on expert-defined values, which is inherently subjective. All habitats are treated equally, not incorporating their unique ecological characteristics. In addition, pressures originating from outside the study area that impact the study area cannot be included (Moreira et al., 2018; Natural Capital Project, 2023). Furthermore, the model does not include combined habitats, which can be limiting especially in wetlands due to the three-dimensionality of the system (*e.g.,* algae on the surface and sediment with seagrasses).

The debate over complex and simple models is longstanding and not unique to InVEST use. Some authors argue that simplification limits progress while complex models can emphasise their overfitting and impracticality (Oberpriller et al., 2021). It is important to continue to use models whilst understanding and acknowledging their limitations. Efforts should be done to promote the use of *in-situ* data for validation purposes, as well as to enhance model reliability.

5. Conclusions

Habitat loss and eutrophication are among the major threats to estuarine systems worldwide, particularly in regions dominated by urban development and intensive agriculture. This application of the InVEST models attempted to understand if intertidal habitats of Sado estuary are keeping up with nutrient export and anthropogenic pressures. It demonstrates that intertidal habitats, even those in poor condition, such as seagrasses, impact on nitrogen and phosphorus export in estuaries. By increasing nutrient retention, intertidal habitats act as buffers for areas of high nutrient export. Scenario analysis indicated that climate-change and management interventions can influence land use and consequently nutrient export. Scenarios focused on conservation presented a considerable impact in nutrient export, but not enough to improve the habitat quality. Further active land-use changes are necessary to successfully improve habitat quality. Contrarily, economic development caused a decrease in habitat quality without affecting nutrient export. Future research could explore the implications of combining Management Strategy scenarios, Economic Development and Ecological Protection, to balance conservation with stakeholder use.

CRediT authorship contribution statement

F. Afonso: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Z. Teixeira:** Writing – review & editing, Visualization, Validation, Methodology, Conceptualization. **M.C. Austen:** Writing – review & editing, Conceptualization. **S. Broszeit:** Writing – review & editing, Conceptualization. **C. Antunes:** Writing – review & editing, Methodology. **C. Rocha:** Writing – review & editing, Methodology. **A.C. Brito:** Writing – review & editing, Conceptualization.

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Declaration of competing interest

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2025.180953.

Data availability

No data was used for the research described in the article.

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