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Marine macroinvertebrate ecosystem services under changing conditions of seagrasses and mangroves

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ABSTRACT

This study aimed to investigate the impact of changing environmental conditions on MMI ES in seagrasses and mangroves. We used data from satellite and biodiversity platforms combined with field data to explore the links between ecosystem pressures (habitat conversion, overexploitation, climate change), conditions (environmental quality, ecosystem attributes), and MMI ES (provisioning, regulation, cultural). Both seagrass and mangrove extents increased significantly since 2016. While sea surface temperature showed no significant annual variation, sea surface partial pressure CO2, height above sea level and pH presented significant changes. Among the environmental quality variables only silicate, PO4 and phytoplankton showed significant annual varying trends. The MMI food provisioning increased significantly, indicating overexploitation that needs urgent attention. MMI regulation and cultural ES did not show significant trends overtime. Our results show that MMI ES are affected by multiple factors and their interactions can be complex and non-linear. We identified key research gaps and suggested future directions for research. We also provided relevant data that can support future ES assessments.

1. Introduction

The ecosystem services (ES) concept plays a crucial role in drawing society's attention to the connection between ecosystems and human well-being (Villa et al., 2014; Broszeit et al., 2019). It also provides evidence for the establishment of a holistic and sustainable ecosystem management approach (Cavanagh et al., 2016). The management of ecosystems also underscores that every ecosystem is modulated by its biophysical context (such as temperature, water flow and species interactions) and anthropogenic pressures (Barbier, 2017; Cruz-Garcia et al., 2017). Thus, to achieve sustainable management of ES we need improved understanding of how the services are provided to humans by the biota; and how the ES providers (the biota) will respond to changes in their environment (Cruz-Garcia et al., 2017; Prather et al., 2013). This raises the need for ES assessments (ESA) which encompass a well-organized process that reveals the importance of ecosystems in human wellbeing (Cruz-Garcia et al., 2017). ESA help to generate the knowledge base for improved policy and decision-making and build capacity for analysing and delivering information (MA, 2005). In fact, ESA are instrumental in informing the UN Convention on Biological Diversity (CBD), which aim to safeguard biodiversity (Geneletti et al., 2020; Maes et al., 2013). In addition, ESA are an essential component of the Sustainable Development Goals (SDGs), which seek to promote global sustainability and emphasize the multifaceted sustainability of nature (Johnson et al., 2019). By tracing the impact of ecosystem change on human well-being, ESA help to link and align the SDGs with the long-term goals of the UN CBD, making them an important part of the international political agenda.

The fact that ESA forms the support base for global policies and decision-making makes it more important to every environment, including macroinvertebrate communities in aquatic environments (Rife, 2018; Prather et al., 2013). Macroinvertebrates are important in

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ecosystems because they show great diversity and abundance, constituting about 80% of all the over 1.6 million described eukaryotic species (Brusca and Brusca, 2004; Kellert, 1993). Macroinvertebrates, including marine macroinvertebrates (MMI), shape ecological processes such as food webs (Hauer and Resh, 2017; Sarker et al., 2021), nutrient and water cycling within their habitats and the adjacent systems (Aghajari Khazaei et al., 2021; Mashar et al., 2021) contributing to essential regulation ES (e.g., Adite et al., 2013; Mashar et al., 2021; Mermillod-Blondin and Rosenberg, 2006). Other significant regulation ES include hydrological flux and climate regulation (Kristensen and Kostka, 2005; Mermillod-Blondin and Rosenberg, 2006; Gentry et al., 2020), habitat modification (Nakamura and Kerciku, 2000; Michio et al., 2003; McLeod et al., 2019; Chakraborty et al., 2022), among others. In addition, MMI provide relevant cultural ES including education through scientific studies (as in Caro-Borrero and Carmona-Jiménez, 2019; Roveri et al., 2020), aesthetics and tourism (Gentry et al., 2020), heritage through archaeological studies (Franceschini et al., 2020), inspiration for arts (van der Schatte Olivier et al., 2020) to mention a few. Furthermore, MMI provide provisioning services - for example serving as a source of protein and materials for humans (Prather et al., 2013; Rife, 2018; Willer and Aldridge, 2020). By occupying multiple trophic levels, macroinvertebrates interact with various trophic groups, thereby (in)directly influencing ES at both global and local scales (Prather et al., 2013; Traill et al., 2010). Thus, MMI sustain high biodiversity, ecosystem functioning and ES in healthy ecosystems, which are significant at both ecological and socio-economic scales (Prather et al., 2013; Rife, 2014, 2018).

MMI and their habitats, such as seagrasses and mangroves, are sensitive to changes in the environment which can alter species compositions and their functions (Amone-Mabuto et al., 2017; Bandeira et al., 2021; Mashar et al., 2021), and thus their capacity to provide ES. Aside from their high susceptibility to changes in their environment, in most macroinvertebrate ESA the focus is on a small selection of ES such as water clarity, for example through filter-feeding (Cavanagh et al., 2016; Damanik-Ambarita et al., 2016; Gentry et al., 2020; van der Schatte Olivier et al., 2020) and food provisioning (Chatterji et al., 2002). This makes MMI ESA scarce (Baaloudj et al., 2022), resulting in a poor understanding of their ES. The scarcity of information mostly lie within the regulation and cultural services, which is a global trend (Barbier, 2017; Small et al., 2017). This is because data and indicators for measuring provisioning services are often readily accessible and tangible through resource users (Mengist et al., 2020; Barbier, 2017), unlike the other services which benefits are not as easily perceived by society (Oleson et al., 2015; Small et al., 2017). Thus, there are complexities associated with measuring MMI regulation and cultural ES, which result in data limitation. This has substantial effects on the awareness of policymakers and the scientific community (Mengist et al., 2020), thereby posing unanticipated dangers to human well-being (Mengist et al., 2020). Unearthing the importance of the different MMI ES to humans requires data on their occurrence, distribution and functional traits, as well as on their ecosystem's pressures and condition. Digital/online databases have been used in many instances for global, regional, and local biodiversity and ecosystem research (Chapman et al., 2019; Nelson and Ellis, 2019; Nelson and Paul, 2019) because they offer a historical, and spatial view of biodiversity and ecosystem processes. The expensive nature of collecting new data in the field, inaccessibility to some habitats, and lack of temporal advantage make digital and online data valuable (Ball-Damerow et al., 2019; Nelson and Ellis, 2019; Parmesan and Hanley, 2015). Thus, online databases which combine datasets from multiple field visits and studies can be powerful tools to evaluate MMI ES and to trigger the conservation of their habitats (Häyhä and Franzese, 2014).

Seagrasses and mangroves are among the most productive ecosystems playing a crucial role in human well-being (Kruitwagen et al., 2010; Mendoza et al., 2019). These ecosystems are ecologically linked, allowing fish and MMI to migrate from one to another in search of shelter, food, spawning and nursery habitats (Kruitwagen et al., 2010; Nagelkerken, 2009; Amone-Mabuto et al., 2017). By so doing, they help to maintain biodiversity upholding an array of ES, namely fisheries including MMI populations, of economic significance. Therefore, as seagrasses and mangroves face increasing degradation due to anthropogenic pressures and global change (Aghajari Khazaei et al., 2021; Amone-Mabuto et al., 2017), MMI biodiversity, their ES and human populations that depend on them for their livelihoods become increasingly uncertain (Rife, 2018). This is mostly prominent in developing countries where there is low budget for ecosystem research, ecosystem management, and low per capita income and a high reliance of coastal communities on coastal resources (Bandeira et al., 2021; Cruz-Garcia et al., 2017; Adite et al., 2013).

Mozambique presents extensive areas of seagrasses and mangroves along the coast which have also been declining (Pereira et al., 2014; Bandeira et al., 2021). In Mozambique, like in many developing countries, coastal communities rely on marine resources for livelihoods and food security (Vicente and Bandeira, 2014). As the country is committed to meeting its UN SDGs including "conserve at least 10% of coastal and marine areas" (i.e., SDG 14.5), "to minimise and address the impacts of ocean acidification through enhanced scientific cooperation at all levels" (SDG 14.3), and improve food security (SDG 2: "No hunger"), coastal resources, particularly MMI remain important food source for local communities, especially the rural poor (Paula & Silva, 1998; Amone-Mabuto et al., 2017). By upholding many ecological processes, MMI assemblages provide several ES and strengthen food safety in the country (Duque et al., 2022). Therefore, food security becomes a problem if MMI decline (Chitará-Nhandimo et al., 2022). The Inhambane bay (INB) in Mozambique, accommodates both seagrass meadows and mangrove forests which are prone to several pressures (Amone-Mabuto et al., 2017; Bandeira et al., 2021), namely temperature variation, sea level fluctuations, pollution and other human disturbances (Aghajari Khazaei et al., 2021; Mashar et al., 2021). Amone-Mabuto et al. (2017) and Bandeira et al. (2021) have documented the impact of natural disasters such as cyclones, while Solana et al. (2020) studied the effects of environmental variability such as temperature and sea level fluctuations on these habitats and associated species in the INB. Despite the importance of MMI ES, their quantification has been limited by the complex nature of these services and data scarcity. Furthermore, there is a lack of understanding regarding the links between MMI ES and changes in seagrasses and mangroves. The wide variations in physical and biological structure in INB (Euliss and Mushet, 2004), make it a suitable model for evaluating multiple pressures and their corresponding influence on MMI ES.

This study aimed to assess the ES provided by MMI in seagrasses and mangroves in connection with pressures and ecosystem conditions using available digital data applied to the INB. We tested the hypothesis that changes in the conditions of both ecosystems affect MMI ES. This research integrated both online and field data allowing a time series analysis. Specifically, we addressed the following questions: i) are the pressures related to habitat conversion, fishing effort, and climatic parameters affecting seagrasses and mangroves? ii) Are there any changes in ecosystem conditions, namely environmental quality and ecosystem attributes? iii) What are the changes in the ES provided by MMI, namely, provisioning, regulation and maintenance, and cultural? The idea behind this study was to provide in-depth baseline information on the relevance of MMI ES from seagrasses and mangroves, primarily to support well-being of local communities and to provide data for sustainable decision-making processes. The study also provides a typology that simplify the quantification of MMI ES which have been challenging to many in the past. This is foundation for developing future methods.

2. Materials and methods

2.1. Study area

The INB (Fig. 1) is a mesotidal coastal environment on the southeast

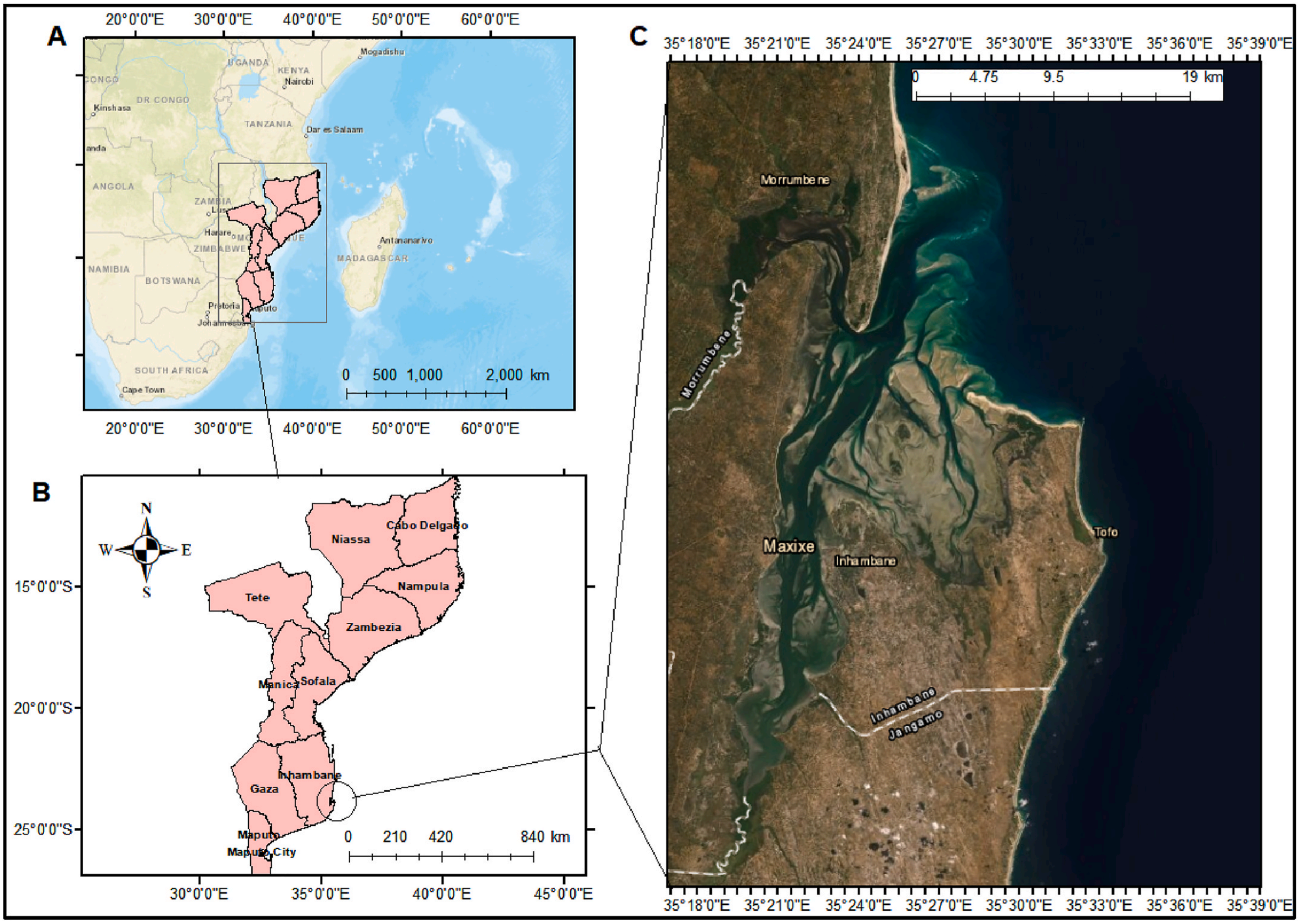

Fig. 1. Map of study area. A) Map of Southern Africa depicting Mozambique, B) Mozambique Provincial map C) the Inhambane bay in southern Mozambique. (Source: ArcGIS 10.6 base map, and digital platform OCHA Services, accessed on May 08, 2022).

coast of Mozambique in the centre of the Inhambane Province (Solana et al., 2020), enclosed within the coordinates −23.44 S and −24.00 S, and 35.3 E and 35.99 E. The surface area of the bay is about 25,000 ha, with seagrass beds and mangroves forests as the prominent vegetation (Amone-Mabuto et al., 2017; Halare, 2012), situated along a coastline of about 700 km long (Halare, 2012). The mean depth of the inner bay is less than 5 m, increasing to 10 m (Chitará-Nhandimo et al., 2022) or more in the outer bay. The Inhambane province administratively has 14 districts which population has grown from 1,123,079 in 1997 to 1,454, 804 in 2017 (according to the City population website, Mozambique). Inhambane (the capital), Morrumbene, Jagamo, and Maxixe are the central districts bordering the bay (Halare, 2012). The Morrumbene river joins the sea from the NW of the bay, forming the Morrumbene estuary (Day, 1974). Two seasons are recorded in the bay: a rainy season from October–March and a dry season from April–September (Meisfjord, 1998). The bay's management is both community-based and government-based through the Inhambane Maritime Administration (Pereira et al., 2014). There is also an NGO, Ocean Revolution of Mozambique, co-supported by the Foundation Ensemble and the Marine Conservation Institute leading the management of the INB through a participatory approach. Through this joint management, the bay has, over the years, received some protection by establishing the Inhambane Bay Community Conservation Network (IBCCN), locally known as Sidika. This resulted in the establishment of 12 protected zones in the bay, known as Locally Marine Managed Areas for seagrass protection against degradation (Chitará-Nhandimo et al., 2022).

2.2. *Data acquisition*

2.2.1. *Ecosystem pressures and condition*

Pressures, the condition of the bay, and their indicators were identified through a bibliographical search, including the guidelines of the Millennium Ecosystem Assessment (MA, 2005), the EU Mapping and Assessment of Ecosystem Services (MAES, 2014, 2018) and local studies like Solana et al. (2020), Grifoll et al. (2020), Amone-Mabuto et al. (2017), and Bandeira et al. (2021). The identified pressures included habitat conversion, overexploitation (fisheries), and climate change, while the ecosystem condition included environmental quality and ecosystem attributes (MAES, 2014, 2018) (Table 1).

2.2.1.1. Habitat conversion. To assess the habitat conversion regarding a change in the areas of seagrasses and mangroves in the INB, cloud-free (≤3.0) Sentinel 2A satellite images for 2015, 2016, 2017, 2019, and 2021 were obtained from the European Space Agency (ESA) Copernicus Science Hub. Landsat 05 (Thematic Mapper, TM) and Landsat 08 (Operational Land Imager, OLI) images for 2009 and 2010, and 2013 and 2014 were also obtained from the USGS earth explorer, respectively. It is worth noting that two Landsat images were downloaded for each year (row/path: 166/076 and 166/077) to ensure the images cover the entire bay. Seagrass data (area) for 1992, 1998, 2001, and 2004 were extracted from Amone-Mabuto et al. (2017). High-resolution images for accuracy assessment were obtained from the Google Earth Engine (GEE) plugin (alpha Version: 0.0.5) in QGIS (Donchyts, 2022).

Table 1
Ecosystem pressures and condition (environmental quality and ecosystem attributes), and macroinvertebrate (MMI) ecosystem services (ES), data acquisition, sources and analysis approach used for the study.

Data	Data Source	Data analysis approach
PRESSURES **Habitat Conversion** Landsat LT05_166076_2009/05/30 LT05_166077_2009/05/30 LT05_166076_2010/11/09 LT05_166077_2010/11/09 LC08_166077_2013/09/14 LC08_166077_2013/09/14 LC08_166076_2014/11/20 LC08_166077_2014/11/20 Sentinel images S2A_L1C_2015/11/23 S2A_ L1C_2016/07/30 S2A_ L1C_2017/11/22 S2A_ L1C_2019/12/27 S2A_ L1C_2021/05/05 Seagrass area data for 1992, 1998, 2001, and 2004	USGS Earth explorer Copernicus scihub Amone-Mabuto et al. (2017) Google Earth Engine	i) Mosaic of Landsat images ii) Pre-processing: Orthorectification; radiometric correction; land masking and clipping with bathymetric data; and pan-sharpening iii) Random Forest classification randomForest function (Model 1) tuneRF function (Model 2) iv) Accuracy assessment: High resolution download; generation of stratified random points; assignment of IDs; accuracy assessment v) Area covered by landcover class $Class\ Area = \frac{Model1 + Model2}{2}$ vi) Estimation of seagrass and mangrove cover (in ha)
Overexploitation: fisheries National Fisheries	Afonso and Mafuca (2001); Ministry of the Sea Inland Waters and Fisheries (Mozambique)	Fishing effort (proxy): Number of artisanal fishing licenses issued per year
Climate change Sea surface temperature, sea surface pH, surface partial pressure of carbon dioxide, and sea level **CONDITION** **Environmental Quality** Dissolved oxygen, silicate, nitrate concentration, phosphate concentration, chlorophyll concentration, phytoplankton concentration, net primary productivity	Copernicus Marine Service	i) Extraction of datasets into CSV. ii) Monthly averaging of data for each year iii) Conversion of $mmolm^{-3}$ to mgl^{-1} iv) Time series analysis Trend line computation Annual statistical differences Monthly statistical differences Seasonal statistical difference computation v) Establishment of relationship between climate and environmental quality variables
Ecosystem attributes MMI online dataset MMI field data Area of Locally Marine Managed Areas (LMMAs, represent Marine Protected Areas)	Bento et al. (2023) Chitará-Nhandimo et al. (2022)	i) Filtration of a MMI online dataset; Combination with field data (MarInSeaMan dataset); Fixing missing functional attributes ii) Mapping of MMI occurrences and distribution iii) Species occurrence matrix formation iv) Calculation of Simpson diversity and function diversity (FD) v) Determination of differences in FD per decade vi) SDG 14.5 commitment (10%) $\%\ protected\ area = \frac{area\ protected}{Total\ area\ of\ the\ bay} * 100.$
ECOSYSTEM SERVICES **Provisioning** MMI fishery production data **Regulation** Species occurrences Functional traits **Cultural** Sport fisheries Species occurrences Protected area	Ministry of the Sea Inland Waters and Fisheries (Mozambique) MarInSeaMan dataset Ministry of the Sea Inland Waters and Fisheries (Mozambique) MarInSeaMan dataset Chitará-Nhandimo et al. (2022)	i) Combination of data 2005–2020 ii) Total annual fishery production iii) Annual differences in fishery production computation iv) Link between fish production and fishing effort (licences) v) MMI Trophic guild and function formation vi) Matrix construction vii) Functional group contribution (C_f) viii) Calculation of ES score through summation of C_f and species richness ix) Total number of sports and recreational fishing licences issued per year 2009–2020 x) Species Richness xi) total area and distribution of protected areas

2.2.1.2. Overexploitation: fisheries. Datasets for evaluating the MMI exploitation rates (fishing effort) were obtained from literature such as Afonso and Mafuca (2001) and the Statistical Bulletin of Fisheries and Aquaculture 2005–2012, 2006–2017, and 2009–2020 (Ministry of the Sea Inland Waters and Fisheries, 2015; 2019, 2022). The data extracted from these reports and studies comprised the number of artisanal fishing licenses issued per year (as a proxy for fishing effort).

2.2.1.3. Climate change. The evaluation of pressures linked to climate change was based on Mozambique's commitment to the United Nations' Sustainable Development Goal (SDG) 14: "Life below water". Specifically, Mozambique addresses the SDG 14.3: "to minimise and address the impacts of ocean acidification through enhanced scientific cooperation at all levels". Therefore, climate change gap-free (Level 4) historical data were collected from the Copernicus Marine Service (CMS) website. These datasets included sea surface temperature (SST) – foundation SST (i.e., water column temperature down to 10 m depth) and skin SST (i.e., SST from the surface up to the first 1 m depth), sea surface pH (down to 5 m), sea level anomalies (SLA), and surface partial pressure of carbon dioxide ($spCO_2$). Except for SST, where daily records from 1990 to 2021 (up to September 2021 for skin SST and May 2021 for foundation SST) were available, all other datasets were daily records from 1993 to 2020, and SLA data consisted of monthly records (1993–2020). The above climate datasets were collected within latitudes −23.40 S, −24.2 S and longitudes 35.50 E and 36.0 E, which include physicochemical properties of the rivers, streams and seas that exchange water with the bay.

2.2.1.4. Environmental quality. Environmental quality (ecosystem condition) data were collected from the CMS website. These datasets included dissolved oxygen (DO), silicate (Si), nitrate (NO_3), phosphate (PO_4), chlorophyll *a* (Chl *a)*, phytoplankton concentrations, and net primary productivity (NPPV).

2.2.1.5. Ecosystem attributes. MMI data were compiled from a dataset constructed for the COBIONET project by Bento et al. (2023). This dataset comprises validated and georeferenced historical data of Mozambique and São Tomé and Príncipe from museum collections, scientific studies, and online databases such as the Global Biodiversity Information Facility (GBIF). In this study, the original dataset was filtered to obtain data relative to 5 MMI phyla from 1950 to 2020 within the INB at latitudes −23.6 S, −23.90 S and longitudes 35.30 E, 35.55 E. The filtered data combined with field data from Chitará-Nhandimo et al. (2022) were organised into a final MMI dataset entitled "MarInSeaMan - Marine invertebrates from seagrasses and mangroves of the Inhambane Bay, Mozambique", which can be consulted in Zenodo database (https://doi.org/10.5281/zenodo.7453891). The MarInSeaMan dataset includes taxonomic level to the species or genera, collection date, collection location, coordinates (WGS84), ecosystem type (seagrasses and mangroves) and functional traits (habitat occupation, trophic guild, and trophic level). The MMI species functional traits also include functional groups (attributes): habitat occupation – zooplankton, benthos, nekton, and neuston; trophic guild – predator, filter feeder, deposit feeder, suspension feeder, scavenger, microphage, generalist, symbiont, parasite, and grazer; trophic level – carnivore, detritivore, omnivore, and herbivore. However, some species in the original dataset had no functional attributes (i.e., the characteristics for a functional trait were missing). This was corrected using literature and the World Register of Marine Species database. For the trophic level, the attributes for species in which the trophic level was not identified were denoted as NoData. The final MarInSeaMan dataset provided data to evaluate ecosystem attributes (condition) related to species occurrence, distribution, and diversity (Simpson diversity index and functional diversity) of MMI in the INB through mapping.

Data on ecosystem attributes (condition) related to the amount of protected area was assessed under Mozambique's commitment to the United Nations SDG 14.5: "to conserve at least 10% of coastal and marine areas". For this study, data on all protected areas (known as the locally managed marine areas – LMMAs) in the INB was collected from Chitará-Nhandimo et al. (2022).

2.2.2. Ecosystem services

An extensive literature review was used to collect data on the ES provided by species included in the MarInSeaMan dataset and external and local reports and scientific literature on the INB. The provisioning services of MMI were evaluated under the scope of Mozambique's commitment to the Sustainable Development Goal (SDG) 2. "Zero hunger", by addressing the production of edible MMI in the INB. The MMI fishery production data was extracted from the Statistical Bulletins of Fisheries and Aquaculture 2005–2012, 2006–2017, and 2009–2020 (Ministry of the Sea Inland Waters and Fisheries, 2015; 2019, 2022) to quantify provisioning ES. Data to assess the MMI regulation and maintenance ES were obtained from the MarInSeaMan dataset. Data for assessing the cultural ES regarding recreation, sense of stewardship, and biodiversity for research and education were obtained from the Statistical Bulletin of Fisheries and Aquaculture 2005–2012, 2006–2017 and 2009–2020 (Ministry of the Sea Inland Waters and Fisheries, 2015; 2019, 2022), the scientific literature such as Chitará-Nhandimo et al. (2022) and the MarInSeaMan dataset, respectively. See Table 1 for all the data used for this study and their sources.

2.3. Data analysis approach

A summary of the data analysis approach is presented in Table 1.

2.3.1. Ecosystem pressures and condition

2.3.1.1. Habitat conversion. All Landsat images for each year were mosaicked. Images were pre-processed through orthorectification, radiometric correction, atmospheric correction, land masking and clipping, and Gram-Schmidt pan-sharpening in QGIS 3.24 (Tesler version) and ArcGIS version 10.6, respectively. Historical landcover analyses were performed for five landcover classes (i.e., seagrasses, mangroves, land, water, and sand but more focus on mangrove and seagrass cover) in R version 4.2.1 using Random Forest classification models (RF). RF was chosen for its robustness in using multiple decision trees to make classification decisions, assuming that individual tree predictors make incorrect predictions (Breiman, 2001; Shapiro, 2018; Traganos and Reinartz, 2018; Collins et al., 2020). Two random forest models (models 1 and 2) were built for each year using the random Forest (model 1), and tuneRF (model 2) functions in R. Each model was built using 1000 trees, as trees ≥250 make the models more stable and reduce variance (Breiman, 2001; Probst and Boulesteix, 2017). Since the areas mapped for each landcover class from the two models were not significantly different, the average of the two regions was used as the area coverage of the landcover class considered. Accuracy assessment of the landcover analyses was conducted using high-resolution images from GEE and random point generation. Sentinel 2 and Landsat (05 and 08) high-resolution images for each year (i.e., the year in which an image was obtained for the classification) were downloaded using GEE and python plugins in QGIS. The QGIS Semi-Automatic Classification Plugin (SCP) was used to generate 250 (50 for each class) stratified random points (1 × 1 pixel size) on each classified raster. On the high-resolution image, the random points were assigned IDs that reflect IDs they represent in the classified raster to form reference points (Reference training sample). The reference points were then used to calculate the Kappa hat, users, and producers' accuracies for each landcover class and the Kappa hat and overall accuracy for the whole analysis for each year.

2.3.1.2. Overexploitation: fisheries. Artisanal fisheries licences issued were totalled for both Inhambane province and the whole of Mozambique for each year from 1994 to 2020.

2.3.1.3. Climate change and environmental quality. The CMS climatic and environmental quality datasets were extracted from their netCDF formats into readable texts and CSV using Panoply version 4.10.3 and R 4.2.1. As all L4 datasets from CMS are already pre-processed, data for each variable were summarised (averaged) on monthly bases for each year. For example, to address SDG 14.3 (ocean acidification), the data for each variable was summarised into yearly averages and compared with standard normal conditions of tropical oceans. To better compare the results of the climatic variables to standards of environmental quality, DO, Si, NO_3, PO_4, Chl *a* and phytoplankton concentrations units were converted from millimoles per cubic metre (mmolm^{-3}) to a milligram per litre (mg l^{-1}) (using 1 mmol/l to 46 mg/l) (everydaycalculation.com). First, mmol m^{-3} was converted to mmol/l and then to mg l^{-1} (i.e. 1 mmolm^{-3} = 0.001 mmol/l = 0.001*46 mg/l) (Convert measurement units). The final data were subjected to time series analyses in R version 4.2.1. The Theil–Sen nonparametric estimator computed a trend line for each variable. Annual statistical significances were computed for each variable of the available time series using the Mann-Kendall trend test (MAES, 2018; Maes et al., 2020; Tiwari and Pandey, 2019). Theil–Sen nonparametric and Mann-Kendall trend tests were used because they are able to deal with non-normal distributions which mostly arise in time-series data (Tiwari and Pandey, 2019). Further, one-way analysis of variance tests were performed to account for monthly differences in each variable. The chi-square test (χ^2) was applied to test the differences between the rainy season (October–March) and the dry season (April–September). Principal component analysis (PCA) was used to assess the relationships between pressure and condition variables. All variables used in the PCA were scaled to ensure no variable had an advantage over another due to the differences in units of measurement. Spearman correlation was further used to print the magnitude of the relationships between the variables showing the coefficients of determination (R^2).

2.3.1.4. Ecosystem attributes. The MMI data were mapped to evaluate species occurrences and distribution in the INB. However, due to the low number of occurrences of the phyla Cnidaria and Annelida, they were not included in further analysis. The Simpson biodiversity index (D) was used to assess MMI diversity of each ecosystem, according to Pélissier et al. (2003) and Leps et al. (2006). With all the attributes for species and functional traits identified, each species' occurrences for each year were quantified using the sampling dates, locations (coordinates and, to some extent, the locality) and the sampling organisations/observers (also considering the catalogue number if applicable). These data treatments yielded two matrices, species × functional group (the occurrence of each species across all samples as rows × each functional group within each trait as columns) and species × sample (occurrence of each species as rows × each sample, i.e., years as colums) for each ecosystem (seagrasses and mangroves). Samples were later grouped in decades starting from the 1950s resulting in a matrix species × samples (decades in this case 1950–1959, 1960–1969, 1970–1979, …., 2010–2020). Using the Macro excel by Leps et al. (2006), each matrix was relativized. Using the category function in the Macro excel, the species matrix was used to calculate dissimilarity among species (Botta-Dukát, 2005). Functional diversity (FD) was then computed using RAO's quadratic functional diversity index in the macro (Schleuter et al., 2010). RAOs quadratic FD assumes that if the proportion of i-th species in a community (in a year) is *pi* and dissimilarity of species, *i* and *j* is *dij,* the Rao coefficient has the form: (Leps et al., 2006; Schleuter et al., 2010).

$$FD = \sum_{i=1}^{s} \sum_{j=1}^{s} d_{ij} p_i p_j \qquad \text{Eqn. (1)}$$

Therefore, when the number of species *s* in a community (i.e., year or decade for this study) has *dij* varying from 0 to 1 (pair of species with entirely different traits), *dij* = 1 for any pair of species (so each pair of species is altogether different), shows that *FD* can be expressed as the Simpson diversity index in form 1 minus Simpson index of dominance *D*:

$$\text{FD} = 1 - \sum\nolimits_{i=1}^{s} p_i^2 \qquad \text{Eqn. (2)}$$

The RAO's quadratic was used because its flexibility in accepting data without abundances of species (Leps et al., 2006). The Fisher Chi-square was used to determine the difference of functional diversity for a particular functional group among decades. To determine the amount of protected area in line with the SDG 14.5 approach, the area for all 12 protected areas in the INB was summed up and expressed as a percentage of the total area of the bay.

2.3.2. *MMI ES identification and classification*

ES provided by MMI from the INB were organised into a table comprising provisioning, regulation and maintenance, cultural services, and the indicators used to quantify those services. The ES identified were classified according to the Common International Classification of Ecosystem Services (CICES) and their correspondence to the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) typology was imparted.

2.3.3. *MMI ES quantification*

2.3.3.1. Provisioning services. The provisioning ES of the INB MMI included the assessment of artisanal and semi-industrial (shrimps) fisheries production from 2005 to 2020. First, the statistical differences in production between years were estimated using Chi-square. Then, the yearly fishery production was analysed and matched against the artisanal fishery licenses issued per year using a linear regression with a log of the fisheries production.

2.3.3.2. Regulation and maintenance services. For the quantification of the regulation and maintenance ES, functions of MMI were connected to the services they provided according to Nakamura and Kerciku (2000) and Lam-Gordillo et al. (2020). It is important to note that only the trophic guild trait was used in the ES assessment because it has a wide diversity that splits species into multiple functional groups (attributes). Based on Lam-Gordillo et al. (2020), we did an a priori analysis to identify the MMI trophic guild functional groups and their functions that lead to a particular regulation and maintenance ES (see Appendix A for the associated functions that interlink the functional groups and ES). Subsequently, a sample × trophic guild functional groups matrix (sample as rows and functional groups as columns) was constructed. This matrix consisted of the number of species under each trophic guild functional group for each year (sample). To quantify each regulation and maintenance ES, the contribution (C_f) of each functional group (*f*) to a particular ES was first calculated using Eqn. (3).

$$C_{f_i} = 1/N_f \times S_{f_i} \qquad \text{Eqn. (3)}$$

where $C_{f\,i}$ = the contribution of the *i*-th functional group; N_f = the total number of functional groups contributing to a particular ES; and $S_{f\,i}$ = the number of species in the *i*th functional group considered. A score level (SL) for each regulation ES was then calculated by summating all the functional contributions linked to it (see Fig. A.1 and Table A.1 from the Appendix) using the equation below:

$$SL_i = \sum_{i=1}^{s} (C_f ij) \qquad \text{Eqn. (4)}$$

where SL_i = the score of *i*-th regulation and maintenance ES considered.

This quantification was done only for seagrasses each year. Due to the lack of data on mangrove MMI, a qualitative assessment of mangrove and seagrass MMI regulation and maintenance ES was conducted using the changes in the area of each ecosystem as a proxy, with the assumption that a change in mangrove/seagrass cover will have a linear change in the services provided by MMI.

2.3.3.3. Cultural services. The cultural services were quantified through three indicators: (1) the total number of recreational and sport fishing licenses issued annually by the Mozambican maritime authorities, both locally and nationally, to assess activities promoting health and recuperation or enjoyment through active or immersive interactions. The total number of licenses issued locally in the Inhambane province and nationally were compared; (2) the area of each locally managed marine areas (LMMAs) in the INB, which relates to the creation of traditional ecological knowledge and culture or heritage as a sense of stewardship or belonging to the surrounding communities, indirectly conserving MMI through the conservation of seagrass meadows. The area of each LMMA was expressed as a percentage of the total area of the INB and mapped to their locations; (3) the species richness of MMI over the years between 1950 and 2018, in both seagrass and mangrove ecosystems, were analysed to assess the diversity of MMI to learn from either by scientific investigation, the creation of traditional ecological knowledge, education and training.

3. Results

3.1. *Ecosystem pressures and condition*

3.1.1. *Habitat conversion*

The temporal changes in seagrass cover were not consistent (Fig. 2; see also Appendix B for classification accuracy of the individual land-cover classes and the overall accuracy for the analysis for each year), with the area changing significantly from 1992 to 2021 ($\chi^2 = 4133.9$, df = 12, $p < 0.001$). Seagrass increase between 2016 and 2021 was rather consistent. The lowest cover of seagrasses was recorded in 2001. The

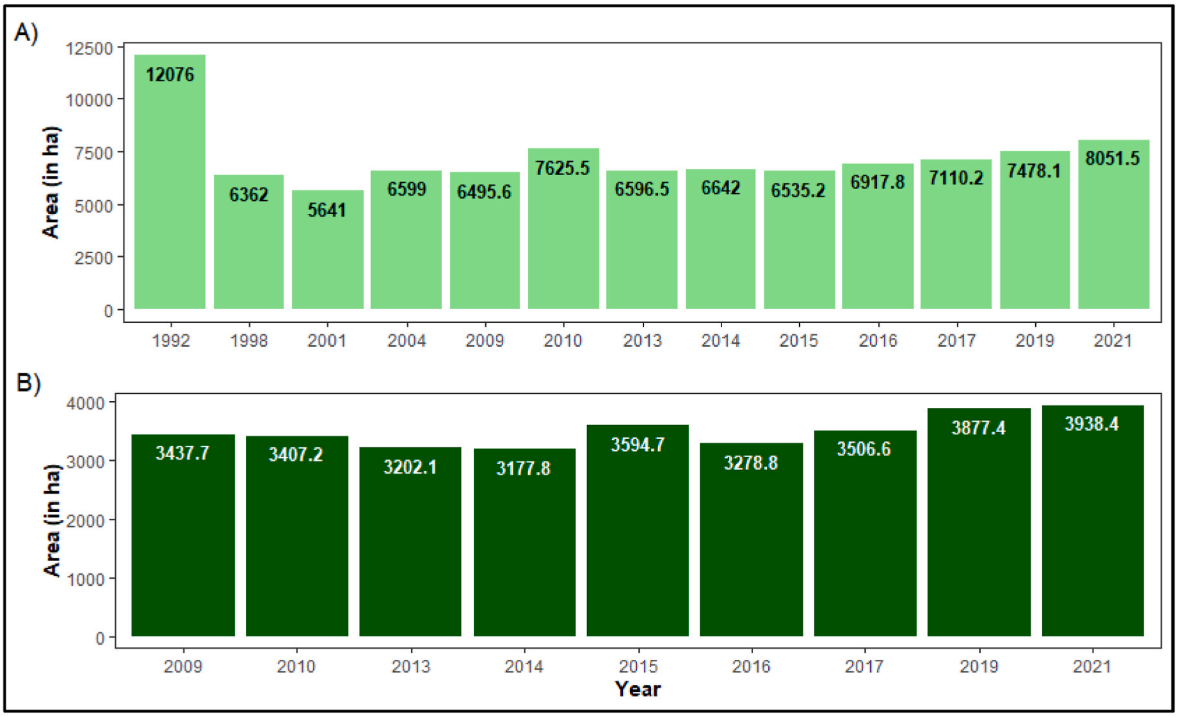

Fig. 2. Land cover changes in A) seagrass cover and B) Mangrove cover of the Inhambane Bay (See also Appendix B for classification accuracy of the individual landcover classes for the years 2009–2021).

changes in mangroves were not consistent across the years from 2009 to 2021, decreasing from 2009 to 2014. The mangrove area in 2019 and 2021 was significantly higher than in the previous years ($\chi^2 = 170.99$, df = 8, $p < 0.001$). However, there has been a consistent increase since 2016. The lowest cover was observed in 2014.

3.1.2. Overexploitation: fisheries

The issuance of artisanal fishing licenses (used as proxies of fishing efforts - overexploitation) in the Inhambane province since 1994 was on average 1203 permits per year⁻, rising from 270 in 1994 to 1782 in 2020 (see Appendix C). Nationwide issuance of artisanal fishing licenses also increased since 2009, on average 16,018 permits per year, ranging from 8791 in 2009 to 29,545 in 2020 (see Appendix C).

3.1.3. Climate change

The mean annual climatic pressures on the INB are summarised in Fig. 3A (see also Appendix D for monthly variation). The sea surface temperature, both at the skin (S_{SST}) and the foundation (F_{SST}) levels showed an increasing trend since the 1990s, but the annual variations were not significant (Mann-Kendall, t = 0.033, p = 0.336; and t = 0.054, p = 0.115, respectively). The lowest and the highest mean S_{SST} were registered in 1994 (25.348 ± 0.484 °C) and 1998 (26.337 ± 0.482 °C), respectively. The lowest F_{SST} mean temperature was recorded in 1994 (25.382 ± 0.485 °C) and the highest in 2021 (27.750 ± 0.524 °C). Significant statistical differences were observed among months both for S_{SST} and F_{SST} (One-way ANOVA, $F_{11,369} = 606.22$, $p < 0.001$; $F_{11,365} = 551.12$, $p < 0.001$, respectively), where both S_{SST} and F_{SST} lowest mean temperature was recorded in August (S_{SST}: 23.624 ± 0.076 °C and F_{SST}: 23.684 ± 0.075 °C). The highest mean temperatures for S_{SST} and F_{SST} occurred in February (28.142 ± 0.065 °C and 28.195 ± 0.069 °C, respectively). The F_{SST} mean temperature (27.023 ± 0.096 °C) of the rainy season was higher than the mean temperature (24.933 ± 0.096 °C) of the dry season, but no significant differences were found ($\chi^2 = 0.084$, df = 1, p = 0.770). For $S_{SST,}$ the mean temperature (26.966 ± 0.0966 °C) of the rainy season was higher than the mean temperature (24.889 ± 0.095 °C) of the dry season, but the differences were not significant ($\chi^2 = 0.083$, df = 1, p = 0.773). The overall mean temperature for the INB since 1990 was 25.919 ± 0.086 °C (range: 22.79 to 29.04 °C) for S_{SST} and 25.983 ± 0.056 °C (range: 22.84 to 29.04 °C) for $F_{SST.}$

The pH of the INB (Fig. 3A) showed both significant annual decreasing trend (t = −0.303, $p < 0.001$) and monthly differences (One-Way ANOVA, $F_{11, 324} = 85.70$, $p < 0.001$). In the opposing direction (Fig. 3A), surface partial pressure of carbon dioxide ($spCO_2$) experienced significant increasing annual trends (Mann-Kendall, t = 0.299, $p < 0.001$) and monthly differences (One-Way ANOVA, $F_{11, 324} = 82.12$, $p < 0.001$). The highest mean pH was registered in 1996 (8.089 ± 0.007), while the lowest was recorded in 2019 (8.051 ± 0.006). For $spCO_2$, the lowest mean value was observed in 1996 (33.830 ± 0.517 Pa) and the highest in 2019 (37.496 ± 0.602 Pa). Also, January presented the lowest mean pH (8.042 ± 0.002) but the highest $spCO_2$ (38.243 ± 0.232 Pa). Continuing with the opposite trends, the highest mean pH (8.101 ± 0.002) and the lowest mean $spCO_2$ (32.700 ± 0.207 Pa) were registered in August. Overall, the mean pH of the INB bay was 8.070 ± 0.001 (range: 8.02 to 8.13), while the mean $spCO_2$ stood at 35.577 ± 0.127 Pa (range: 30.64–40.87 Pa). The pH of the rainy season (8.054 ± 0.001) was not significantly different ($\chi^2 = 6.52e\text{-}5$, df = 1, p = 0.990) from the pH in the dry season (8.087 ± 0.001). Unlike the pH, the mean $spCO_2$ was higher in the rainy season (37.114 ± 0.133 Pa) than in the dry season (34.040 ± 0.135 Pa), but the difference between the means for the two seasons was not significant ($\chi^2 = 0.133$, df = 1, p = 0.715).

Surface height above sea level (Fig. 3A), also called sea level anomalies (SLA), increased significantly annually (Mann-Kendall, t = 0.372, $p < 0.001$). The lowest (−0.016 ± 0.006 m) and highest (0.098 ± 0.011 m) means of SLA were observed in 1997 and 2020, respectively. The month effect was not statistically significant (One-Way ANOVA, $F_{11, 324} = 1.14$, p = 0.331). The lowest mean SLA was recorded in March (0.026 ± 0.008 m) and the highest in September (0.057 ± 0.011 m). The mean overall SLA in the INB since 1993 was 0.041 ± 0.002 m (range: 0.13 to 0.17 m). The mean SLA during the rainy season (0.043 ± 0.004 m) was higher than in the dry seasons (i.e., 0.039 ± 0.004 m) but not statistically different ($\chi^2 = 5e\text{-}05$, df = 1, p = 0.994).

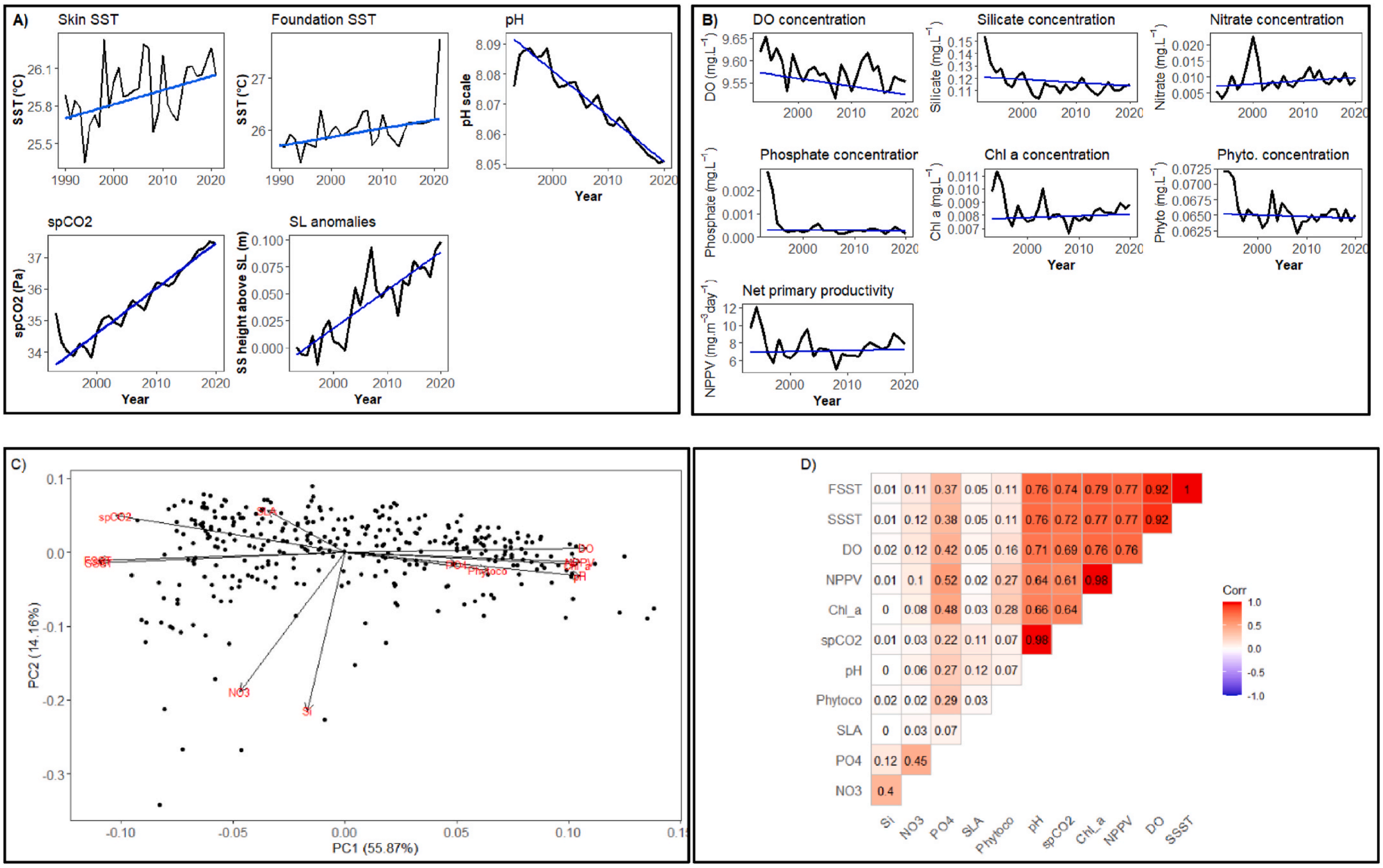

Fig. 3. Temporal variation in climate change and environmental quality variables and their relationships. A). Annual trends in climate change pressures (sea temperature – S_{SST} and F_{SST}, pH, surface partial pressure of carbon dioxide – $spCO_2$ and sea level anomalies – SLA). B) Annual trends in the Inhambane Bay condition (environmental quality - DO, Si, NO3, PO4, Chl a, phytoplankton concentrations and NPPV) between 1990 and 2021. C) Principal components analysis for the relationship between physicochemical variables. D) Correlation matrix indicating variations explained in the relationships between variables (Spearman correlation test, R squared). (See also Appendix D – monthly variations in climatic and environmental quality variables).

3.1.4. Environmental quality

The INB condition related to annual mean values of environmental quality variables is presented in Fig. 3B (see also Appendix D for monthly variation). The INB presented a non-significant (Mann-Kendall, $t = -0.03$, $p = 0.343$) annual decreasing trend in dissolved oxygen (DO) (Fig. 3B), with the highest DO average in 1994 (9.655 ± 0.062 mg/l) and the lowest in 2007 (9.516 ± 0.067 mg/l). There was a significant monthly variation (One-Way ANOVA, $F_{11,\ 324} = 493.94$, $p < 0.001$) in DO concentration, with the lowest mean in February (9.283 ± 0.010 mg/l) and the highest in August (9.893 ± 0.012 mg/l). No significant difference ($\chi^2 = 0.003$, df = 1, $p = 0.955$) was found between the DO concentrations in the rainy (9.455 ± 0.015 mg/l) and the dry (9.697 ± 0.015 ml/l) seasons. Overall, the mean DO concentration of the INB in 27 years (1993–2020) was 9.576 ± 0.012 mg/l (range: 9.18–10.02 mg/l).

The concentration of silicate (Si) (Fig. 3B) had a significant annual decreasing trend (Mann-Kendall, $t = -0.173$, $p < 0.001$). The mean Si concentration was highest in 1993 (i.e., 0.154 ± 0.008 mg/l) and lowest in 2003 (0.104 ± 0.004 mg/l). There was a significant variation among months (One-Way ANOVA, $F_{11,\ 324} = 8.64$, $p < 0.001$), where the lowest monthly mean Si concentration was recorded in November (0.102 ± 0.002 mg/l) and the highest in March (0.129 ± 0.006 mg/l). The mean Si concentration during the rainy season (0.113 ± 0.002 mg/l) was lower than that of the dry season (0.120 ± 0.002 mg/l), but the difference was not significant ($\chi^2 = 0.0002$, df = 1, $p = 0.990$). Overall, the Si concentration of the INB stood at 0.116 ± 0.001 mg/l (range: 0.08–0.22 mg/l).

Nitrate (NO_3) had an increasing annual trend, but phosphate (PO_4) had an opposite trend (Fig. 3B). However, for NO_3, there was no significant annual change in the trend (Mann-Kendall, $t = 0.071$, $p = 0.051$), ranging on average from 0.003 ± 0.001 mg/l in 1994 to 0.023 ± 0.010 mg/l in 2000. The annual changes in PO_4 were significant (Mann-Kendall, $t = -0.174$, $p < 0.001$), ranging from 0.003 ± 0.0003 mg/l in 1993 to 0.0001 ± 3.89e-5 mg/l in 2008. Both nutrients, presented significant monthly variations (One-Way ANOVA: NO_3, $F_{11\text{-}324} = 7.95$, $p < 0.001$; PO_4, $F_{11,324} = 3.72$, $p < 0.001$). The lowest mean NO_3 concentration was recorded in October (0.002 ± 0.001 mg/l) and the highest in March (0.022 ± 0.005 mg/l). As for PO_4, the lowest concentration was registered in April (0.0001 ± 5.24e-5 mg/l) and the highest in July (0.0007 ± 0.0001 mg/l). The mean concentrations of NO_3 in the rainy (0.012 ± 0.002 mg/l) and dry (0.007 ± 0.001 mg/l) seasons were not significantly different ($\chi^2 = 0.001$, df = 1, $p = 0.971$). The mean PO_4 concentrations in the rainy (0.0004 ± 5.61e-5 mg/l) and dry (0.0005 ± 4.96e-5 mg/l) seasons were not significantly different ($\chi^2 = 0.008$, df = 1, $p = 0.929$). Overall, the INB presented an average NO_3 concentration of 0.009 ± 0.001 mg/l (range: 9.92e-05 to 1.23e-01 mg/l), and a mean PO_4 concentration of 0.0004 ± 3.74e-5 mg/l (range: 2.58e-06 to 4.50e-03 mg/l).

Chlorophyll *a* (Chl *a*) concentration (Fig. 3B) had a non-significant annual decreasing trend (Mann-Kendall, $t = -0.048$, $p = 0.186$). The highest and the lowest Chl *a* concentrations were recorded in 1994 (0.011 ± 0.001 mg/l) and 2008 (0.007 ± 0.001 mg/l), respectively. There were significant monthly differences in Chl *a* concentration (One-Way ANOVA, $F_{(11,\ 324)} = 114.01$, $p < 0.001$) where the lowest mean monthly Chl *a* concentration occurred in March (0.005 ± 0.0001 mg/l) and the highest in July (0.015 ± 0.001 mg/l). The concentration of Chl *a* in the rainy season (0.006 ± 0.0001 mg/l) was comparatively lower than Chl *a* in the dry season (0.011 ± 0.0003 mg/l), but this difference was not significant ($\chi^2 = 0.001$, df = 1, p = 0.970). The overall mean of Chl *a* concentration in the INB was 0.008 ± 0.0002 mg/l (range: 0.004–0.02 mg/l).

Phytoplankton concentration (Fig. 3B) had a significant annual decreasing trend (Mann-Kendall, t = −0.115, p = 0.002). The highest mean phytoplankton concentration was registered in 1993 (0.072 ± 0.002 mg/l), and the lowest in 2008 (0.062 ± 0.001 mg/l). There were significant monthly differences (One-Way ANOVA, $F_{11, 324} = 20.72$, $p < 0.001$) in phytoplankton concentrations among months. The average month phytoplankton concentration was lowest in May (0.059 ± 0.0005 mg/l) and highest in August (0.074 ± 0.0.0011 mg/l). The mean phytoplankton concentration in the rainy season (0.065 ± 0.0004 mg/l) was lower than that of the dry season (0.066 ± 0.0006 mg/l), but the difference was not significant ($\chi^2 = 0.000$, df = 1, p = 0.990). The overall mean phytoplankton concentration in the INB bay was 0.066 ± 0.0003 mg/l (range: 0.06–0.09 mg/l).

Net primary productivity (NPPV) presented decreasing trends (Fig. 3B) over the years, but the annual differences were not significant (Mann-Kendall, t = −0.040, p = 0.278). The highest mean concentration of NPPV was registered in 1994 (12.068 ± 2.287 $mgm^{-3}day^{-1}$) and the lowest in 2008 (5.012 ± 0.873 $mgm^{-3}day^{-1}$). There were significant variations of NPPV among months (One-Way ANOVA, $F_{11, 324} = 123.81$, $p < 0.001$) where the lowest mean NPPV (2.631 ± 0.182 $mgm^{-3}day^{-1}$) was recorded in March and the highest (17.027 ± 0.707 $mgm^{-3}day^{-1}$) in August – a similar trend to the Chl *a* concentration. The rainy season presented a mean NPPV (4.201 ± 0.168 $mgm^{-3}day^{-1}$) that was less than half of the mean NPPV recorded during the dry season (11.087 ± 0.461 $mgm^{-3}day^{-1}$); however, the difference was not significant ($\chi^2 = 3.102$, df = 1, p = 0.078). Overall, the average NPPV of the INB was 7.644 ± 0.309 $mgm^{-3}day^{-1}$ (range: 1.67 to 26.42 $mgm^{-3}day^{-1}$).

The variations in the relationships between pressures and condition of the INB, were explained by the first two axes (ca. 70%) of the PCA (Fig. 3C). Sea surface temperature (SST) was negatively correlated with all variables indicating variations ranging from 37% in PO_4 to 92% in DO (Fig. 3D). However, NO3, Si, spCO2 and SLA had positive relationships with SST, where the lowest variation was visible between SST and SLA (5%) and the highest between SST and spCO2 (72–74%). On the other hand, the pH was positively associated with all variables except spCO2, NO3, and SLA**.** There was above 99% of positive variations explained between NPPV and Chl *a* concentration (Regression coefficient = 1.641, F = 7946, $r^2 = 0.960$, $p < 0.001$).

3.1.5. Ecosystem attributes

Regarding the INB condition related to ecosystem attributes, species occurrences of MMI identified in the INB as well as their distribution are summarised in Fig. 4A. Overall, from 1953 to 2020, there were 4963 MMI occurrences (within 227 species) in the INB shared among five phyla: Arthropoda (76.85%), Mollusca (22.24%), Cnidaria (0.75%), Annelida (0.04%), and Echinodermata (0.12%), in 12 classes, 113 families, and 134 genera. Out of the 227 species identified in the INB, about 54.19%, 36.56%, 6.61%, 1.76%, and 0.88% were Mollusks, Arthropods, Echinoderms, Annelids, and Cnidarians, respectively. MMI species richness (142 species) and occurrences (4814) in seagrasses were higher than MMI richness (97 species) and occurrences (149) in mangroves. The MMI Simpson diversity index in seagrasses (Fig. 4B) for each decade was higher than 0.5 ranging from 0.80 (2002–2009) to 0.96 (in the 1950s). Functional diversity in seagrass habitat occupation ranged from 0.0003 (in the 1950s) to 0.28 (2010–2018), but the differences were not significant ($\chi^2 = 0.65816$, df = 4, p = 0.956). Trophic guild functional diversity varied between 0.0133 (in the 1970s) to 0.087 (in the 1950s), but not statistically ($\chi^2 = 0.066$, df = 4, $p > 0.05$). For the trophic level, functional diversity was not statistically different ($\chi^2 =$

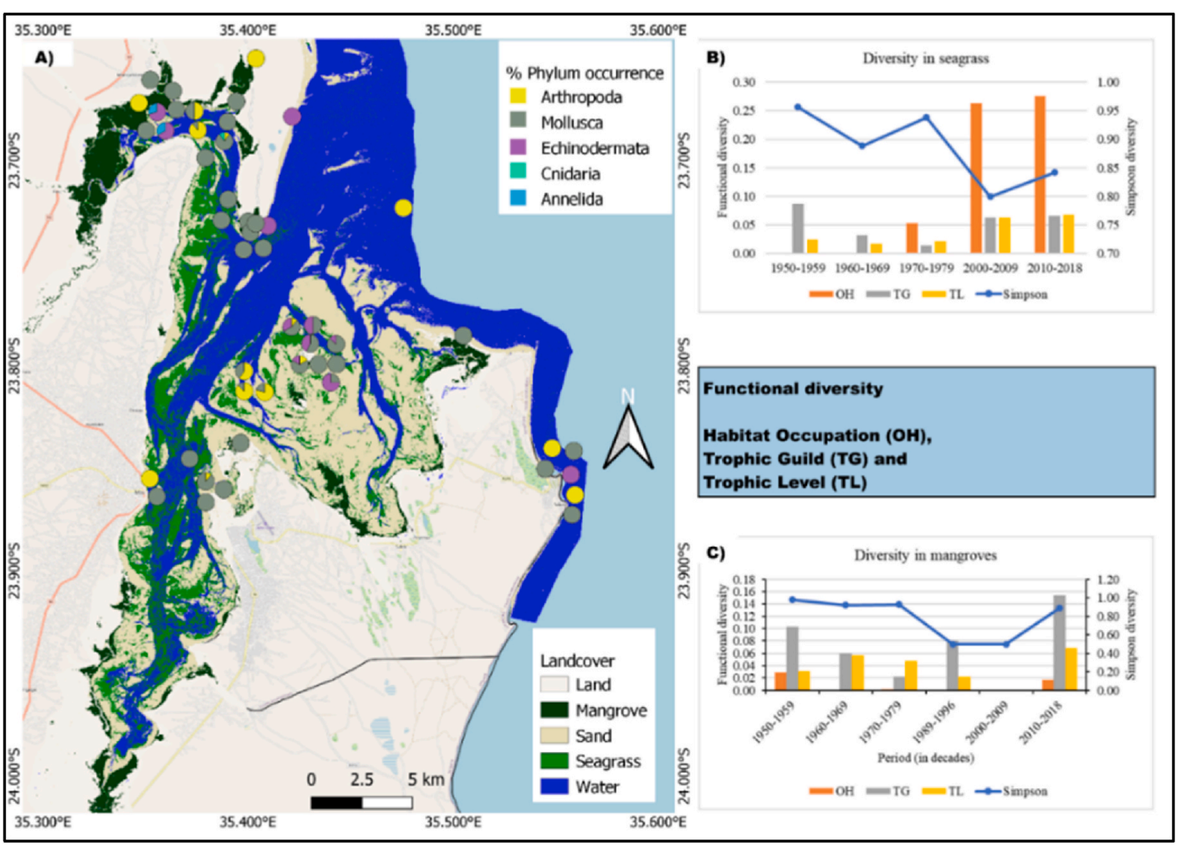

Fig. 4. Distribution and functional diversity indexes of the Inhambane bay A) Marine macroinvertebrate occurrences and distribution where pie charts represent each phylum's occurrence percentage. Simpson and Functional diversity indexes for B) Seagrass and C) Mangrove ecosystems.

0.062, df = 4, p > 0.05), varying from 0.017 (in the 1960s) to 0.068 (2010–2018). The Simpson diversity index in mangroves (Fig. 4C) ranged from 0.98 (1950–1959) to 0.5 (1989–1996 and 2000–2009). There was no significant difference in functional diversity for habitat occupation across all decades in mangroves (χ^2 = 0.093, df = 5, p = 0.999), varying from 0.0 (1989–2009) to 0.029 (in the 1950s). Similarly, the trophic guild of MMI in mangroves did not significantly vary between decades (χ^2 = 0.223, df = 5, p = 0.999), ranging between 0.0 (2000–2009) and 0.154 (2010–2018). The functional diversity of mangroves in the trophic level was not significantly different (χ^2 = 0.083, df = 5, p = 0.999), varying from 0.0 (2000–2009) to 0.069 (2010–2018).

Regarding the amount of protected area, out of the total area (25000 ha) covered by the INB, the seagrass LMMAs covered approximately 4.7% (1173.9 ha) of the bay (see Appendix E for detail in the area of each LMMA).

3.2. MMI ES identification and classification

The MMI ES identified in the INB are summarised in Table 2 (see also Appendix F for definition, examples, and MMI ES sources of information), including the CICES ES classification (section, class and code), the correspondence to the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) code, and the indicators used to quantify the ES.

3.3. MMI ES quantification

3.3.1. Provisioning services

The MMI (lobsters, crabs, cephalopods, and shrimps) artisanal fisheries production in the Inhambane Province (Fig. 5A), between 2012 and 2020, was 33669 tons representing 13.46% of the province's total fishery production. The MMI production increased steadily from 2012 (1030 tons) to 2015 (2140 tons) and then dropped by 68.04% from the previous year in 2016 (684 tons). However, since 2017, the MMI production in the Inhambane province achieved a steep rise (average 6640.25 tons per year; range: 1908–10680 tons). The MMI production was significantly different among years (χ^2 = 134737, df = 9, p < 0.001). The MMI production in 2020 was higher than the other years and lowest in 2016. The production across all years (2012–2020) regarding the percentage of each MMI to the total MMI production showed that crabs contributed about 61.8%, followed by cephalopods (29.9%), shrimps (7.9%) and lobsters (~0.5%) with significant differences (χ^2 = 90.797, df = 3, p < 0.001). However, MMI production in the INB (considering the four bordering districts: Jagamo, Maxixe, Inhambane Municipal, and the Morrumbene) contributed only 35% to the total MMI production (1028 tons) in the Inhambane province in 2012 (see Appendix G for the MMI artisanal fisheries production in the Inhambane province by districts – tons and percentage). The total (i.e., all fisheries stocks) artisanal fisheries production per year in the Inhambane province was positively correlated with the number of licenses issued per year (Regression coefficient = 6.41e-04, F = 7.54, p = 0.029, R^2 = 52.0%, intercept = 9.00). However, the total MMI production per year in the Inhambane province was not significantly correlated with the number of fishing licenses issued per year (Regression coefficient = 4.29e-04, F = 0.36, p = 0.568, R^2 = 5%, intercept = 7.05). The shrimp production from semi-industrial fisheries (Fig. 5B) increased from 2016 to 2018, and since then it has declined.

3.3.2. Regulation and maintenance services

Climate regulation and habitat modification were the two principal regulation and maintenance services performed by marine macroinvertebrates in INB, whereas hydrological flux was the least performed (Table 3). Across all years, 2018 had the highest regulation and maintenance ES score level, presenting the lowest score for hydrological flux and the highest for climate regulation. On the other hand, 1964 presented the lowest number of regulation services having no representative for three services, and low scores for the available climate regulation and habitat modification services. The qualitative assessment (see Appendix H) of regulation ES provided by MMIs in both ecosystems (using changes in their area as proxy) indicated that overall seagrass MMI ES increased since 2017, and mangrove MMI ES increased since 2019.

Table 2
Classification of the marine macroinvertebrate (MMI) ecosystem services (ES) in the Inhambane Bay (INB) according to CICES (v5.1) with correspondence to the IPBES typology codes and the indicators used for the ES quantitative assessment. (See also appendix Table F.1 for definitions, examples, and sources of the MMI ES).

Section	Class	Code	IPBES code	Identified MMI ES	Indicators for quantification
Provisioning services (Biotic)	Wild animals (aquatic) used for nutritional purposes	1.1.6.1	12	Food - Edible MMI in INB, e.g., lobster, shrimps, cephalopods, and crabs	Artisanal fishery (invertebrate) production (tons/year) Semi-industrial production (Shrimps) (tons/year)
Regulation & maintenance (Biotic)	Filtration and accumulation by animals	2.1.1.2	7 & 5	Water quality regulation & Climate regulation through carbon sequestration	All regulation and maintenance services were quantified through a score level (based on the total number of different functional groups and the number of species in each functional group) contributing to a particular ES
	Hydrological cycle and water flow regulation (Including flood control and coastal protection)	2.2.1.3	6	Hydrological flux, water quantity regulation	
	Bioremediation by microorganisms, algae, plants, and animals & Decomposition and fixing processes and their effect on soil quality	2.1.1.1 & 2.2.4.2	8	Decomposition precursors and nutrient fixation	
	Maintaining nursery populations and habitats (Including gene pool protection)	2.2.2.3	1	Habitat modification bioerosion, bioturbation, and food web stability	
Cultural (Biotic)	Characteristics of living systems that enable activities promoting health, recuperation, or enjoyment through active or immersive interactions	3.1.1.1	15	Recreation through sport fishing	Total number of recreational and sport fishing licenses issued per year
	Characteristics of living systems that enable the creation of traditional ecological knowledge and are resonant in terms of culture or heritage	3.1.2.1 & 3.1.2.3	15	Natural system for stewardship, sense of belonging and community togetherness	The number and total cover area of protected sites (locally managed marine areas)
	Characteristics of living systems that enable scientific investigation and the creation of traditional ecological knowledge, and education and training	3.1.2.1 & 3.1.2.2	15 & 16	Scientific and traditional knowledge, formal and informal education and training about MMI	Species richness per year in seagrasses and mangroves

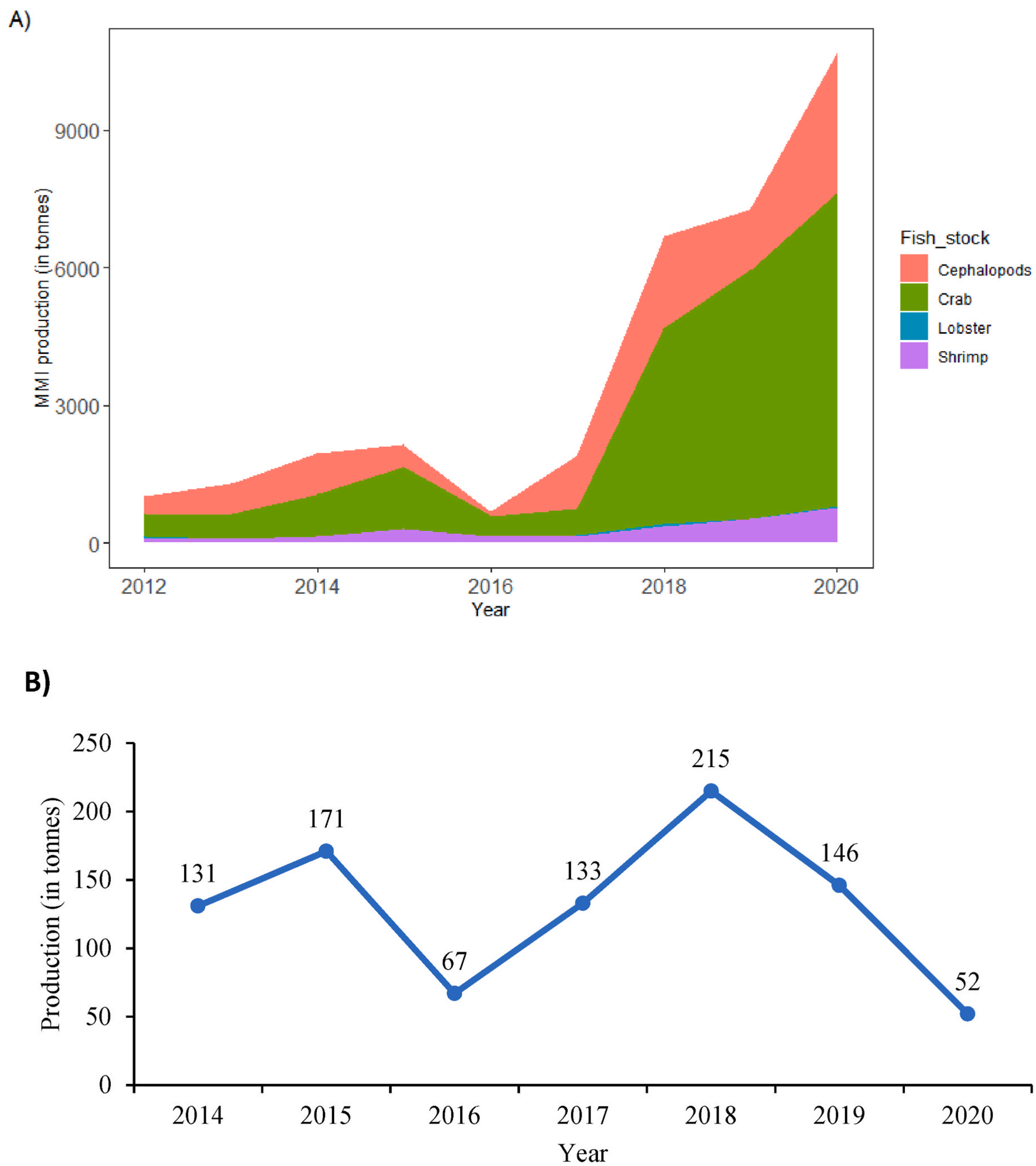

Fig. 5. Fisheries in the Inhambane province. A) Artisanal marine macroinvertebrate fisheries production in the Inhambane province between 2012 and 2017; and B) Semi-industrial production of shrimps between 2014 and 2020.

3.3.3. *Cultural services*

The number of licenses issued by the maritime authorities for recreational and sport fishing activities at both local (Inhambane province) and national levels, since 2009, was 3999 and 13181, respectively (Fig. 6). The number of licenses per year averaged 1465 at Inhambane and 4444 at the national level. The number of licenses issued in Inhambane province increased from 2009 to 2018, with punctual decreases in 2013 2015 and 2017. On the national scale, the number of licenses steadily increased from 2009 to 2018. However, the number of licenses has declined since 2018 at local and national levels.

The distribution of protected areas (the LMMAs) in the INB held by the local communities as natural systems for stewardship, sense of belonging and community togetherness is summarised in Fig. 7. Eight LMMAs were distributed across the eastern fringe, while four were at the western belt of the bay. The Inhambane city bordering the bay on the eastmost side holds about six of the LMMAs, totalling 888.34 ha representing approximately 75% of all the LMMAs. Only one, the Jogó LMMA, was found at the northwest portion of the bay. On the whole, the Marragane LMMA was the largest, covering an area of about 340 ha, situated in the southwest of the INB.

The MMI species richness in the INB, that can be linked to scientific and traditional knowledge, and formal and informal education and training, was higher in seagrasses (142 species) than in mangroves (97 species). Overall, 227 species of MMI were identified in the INB from 1953 to 2020. Most of the species identified in seagrasses were observed in 2018 (i.e., 46 species) against three species in mangroves (Fig. 8). The highest species richness, 46 species, in mangroves was recorded in 1954 against 22 species in seagrasses. The highest species richness of the bay

Table 3
Score level of marine invertebrate regulation and maintenance ecosystem services, weighted by the total number of different functional groups and the number of species within a trophic functional group connected to a particular regulation service, in the Inhambane Bay Seagrasses. (See also the qualitative assessment of regulation and maintenance ES of both Seagrasses and Mangroves in Appendix H).

Year	Water quality regulation	Hydrological flux, water quantity regulation	Decomposition precursors and nutrient fixation	Habitat modification, bioerosion, bioturbation, and food web stability	Climate regulation through carbon sequestration	Total Score Level
1954	5.02	1.60	4.02	5.10	6.02	**22**
1964	0.00	0.00	0.00	0.50	0.50	**1**
1968	3.00	1.00	2.00	1.50	2.50	**10**
1969	0.20	0.20	0.20	0.20	0.20	**1**
1971	0.60	0.60	0.60	7.60	7.60	**17**
2009	0.00	0.00	0.00	12.00	12.00	**24**
2010	0.00	0.00	0.00	12.00	12.00	**24**
2011	0.00	0.00	0.00	10.00	10.00	**20**
2012	0.00	0.00	0.00	9.50	9.50	**19**
2013	0.45	0.20	0.45	9.70	9.45	**20**
2014	0.00	0.00	0.00	9.50	9.50	**19**
2015	0.20	0.20	0.20	11.70	11.70	**24**
2016	0.40	0.40	0.40	13.90	13.90	**29**
2017	0.58	0.00	0.58	8.00	8.08	**17**
2018	7.12	2.20	6.20	15.62	16.95	**48**
2020	3.98	1.40	3.07	2.15	3.82	**14**

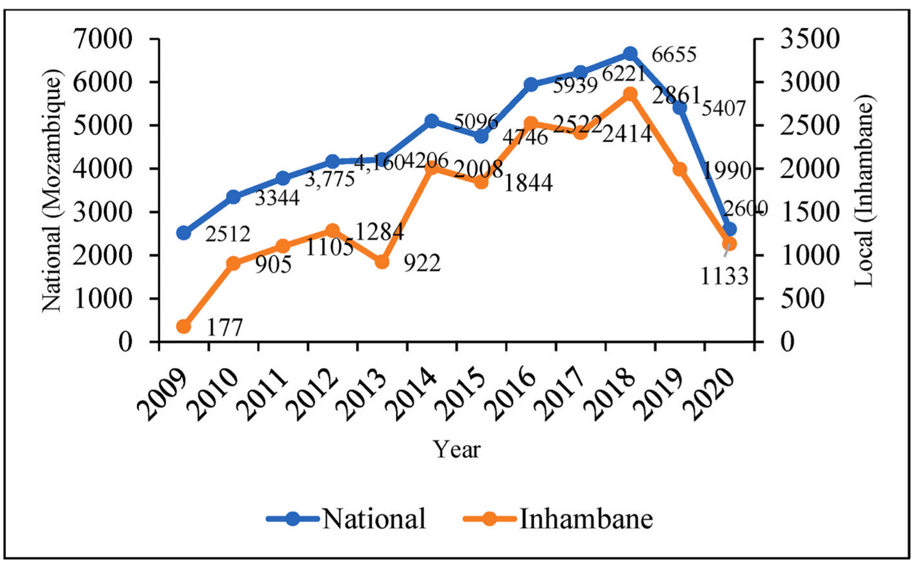

Fig. 6. Recreational and sport fishing licenses issued annually between 2009 and 2017 at the Inhambane province and at the Mozambique national level.

was observed in 1954 (i.e., a total of 66 species independent of habitats). Species of Annelida and Echinoderms were found only in mangroves and seagrasses, respectively. The number of species has increased since 2009, but sporadically. Of the 25 species found in the entire bay in 2009, 24 were in seagrasses against only one in mangroves. In 2020, 15 species were found in the bay, all observed in seagrass habitats. The highest species richness in seagrasses (i.e., 46) was found in 2018, while the highest species richness in mangroves (i.e., 42) was observed in 1954.

4. Discussion

4.1. Ecosystem pressures and condition

4.1.1. Habitat conversion

Across the globe, habitat loss through the reduction in habitat extent is perhaps the greatest threat to biodiversity (Horváth et al., 2019). Applying landcover analysis allowed us to detect the distribution and the temporal changes in the area coverage of seagrass and mangrove ecosystems in the INB, Mozambique. Between these two vegetation types, seagrasses occupy the most significant portion of the bay, concentrated mainly at the central and south-eastern ends of the bay. Conversely, mangroves are primarily concentrated in the north-western part close to the Morrumbene district. This is consistent with Barbosa et al. (2001) and Gullström et al. (2021). They indicated that in Mozambique, mangroves are well established and extensive in areas that receive more freshwater and alluvium discharge from rivers. Furthermore, mangroves are established extensively in sheltered areas compared to seagrasses avoiding high wave pressure. Mangroves create a monospecific canopied environment when dominant, thereby obstructing light away from understorey vegetations (Mendoza et al., 2019). Seagrasses are photophilic, shallow, sandy, limestone and flat surface vegetation established in the open spaces far from mangroves (Mendoza et al., 2019; Gullström et al., 2021). In accordance, Amone-Mabuto et al. (2017) found a low abundance and composition of seagrasses in the INB areas where mangroves grow, which further confirms why seagrasses were more concentrated in areas without mangroves, as also observed in the Mindanao and Bantayan islands of the Philippines (Sharma et al., 2017; Mendoza et al., 2019). Regarding habitat loss from this study, the INB lost about 33% of its seagrass between 1992 and 2021, while mangroves gained approximately 14.5%

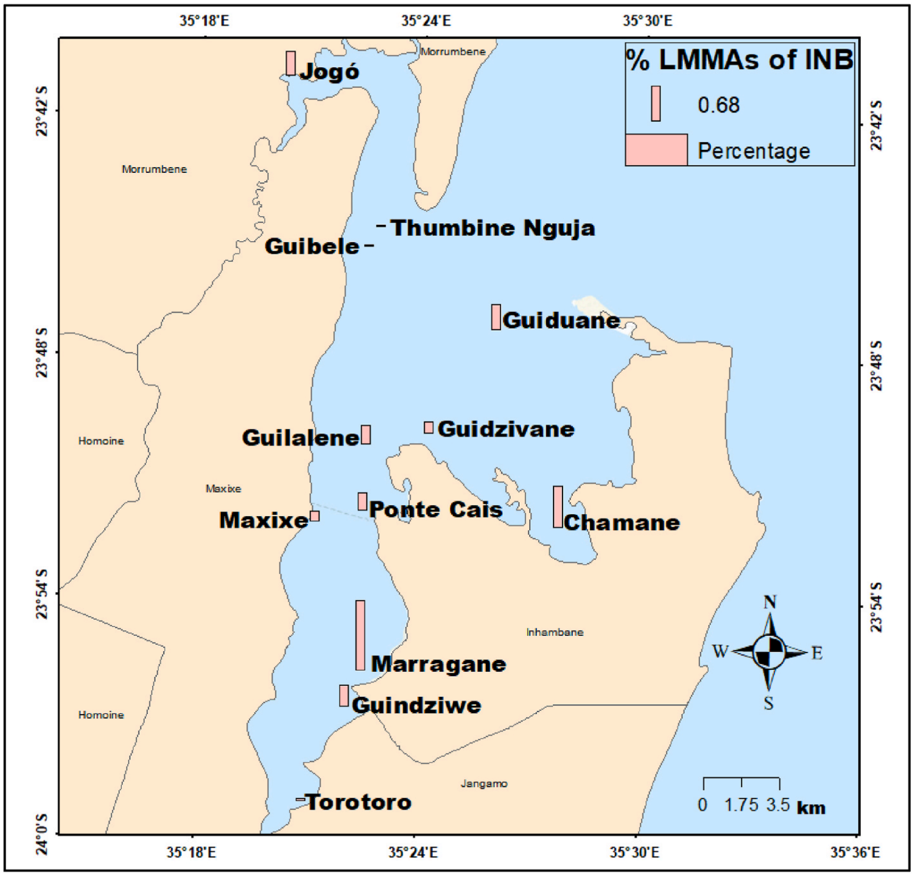

Fig. 7. Locally managed marine areas in the Inhambane bay. The length of the bar represents the percentage of area covered in the bay.

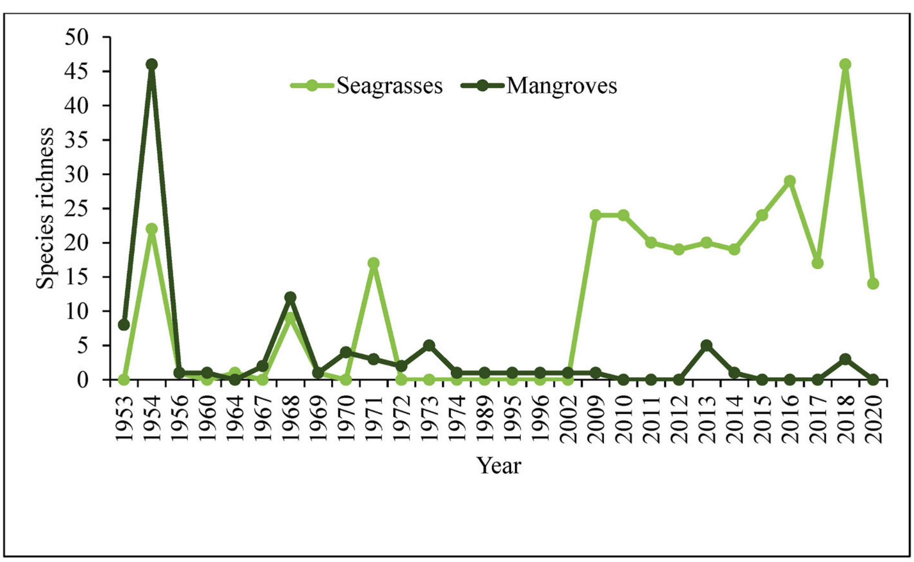

Fig. 8. Temporal evolution of Marine macroinvertebrate species richness of the Inhambane Bay in both Seagrasses and Mangroves.

increase from 2009 to 2021. Seagrass meadows are very fragile to extreme events like floods and cyclones (Amone-Mabuto et al., 2017; Bandeira et al., 2021). Since 1966, about 12 cyclone events have occurred in the Inhambane province, with some happening directly in the INB. Even though there are other anthropogenic factors such as the removal of seagrass for clam harvest (Amone-Mabuto et al., 2017), cyclones are the essential driver of seagrass cover changes in the INB (Amone-Mabuto et al., 2017; Bandeira et al., 2021). According to Macamo et al. (2016), mangroves in the bay have been resistant to cyclones based on a 14-year survey from 2000, supported by Campira et al. (2021). Therefore, the main factors that may affect mangroves negatively in Mozambique include mangrove wood cutting and urbanization allied to population growth (Gullström et al., 2021). The good news is that since 2016, in the INB, both seagrasses and mangroves have continuously increased in area, possibly due to the establishment of the 12 LMMAs (no-take zones). A similar account of the importance no-take zones in marine habitats has been reported by Rioja-Nieto and Sheppard (2008) who found higher area coverage of seagrass beds in no-take zones. Thus, despite the unrelenting anthropogenic stressors on marine and coastal habitats all across the world, well managed marine protected areas can be an insurance for coastal resilience against the disturbances (Alonso Aller et al., 2017). Shapiro (2018), who used the random forest classification method to survey mangroves in Mozambique, supports this study that even though there are human impacts on mangroves in the country, the net gains in mangrove cover in several provinces, override the losses. Goldberg et al. (2020), reported a decline of mangroves worldwide mainly due to anthropogenic pressures, especially land use changes. However, the authors found a decreasing trend in the decline of mangroves from 2000 to 2016, which they attributed to global conservation efforts.

4.1.2. Overexploitation: fisheries

Fishing efforts which comprise the amount of time spent, the number of gear, and all inputs used by fishers, including the crew number (according to the FAO), have been indicated as a potential overexploitation risk across the Mozambique channel (Zeller et al., 2021). The current study, using the number of artisanal fisheries licenses issued per year as an indicator of overexploitation, showed an increment of over 500% of licenses issued in the Inhambane province between 1994 and 2020. Uetimane (2018) pointed out the effect of the increase in effort on shrimp fishing in the INB, resulting in the annual closed seasons on the bay. The rise in fishing efforts in Inhambane parallels that of the national situation, even though while the national number of licenses keeps increasing, the number of licenses in Inhambane has seen a slight decline since 2018. Moreover, Blythe et al. (2014) associate the rise in fishing effort in the whole of Mozambique with higher investments in fishing gear and government supports. The support of governments and huge monetary investments are still a debate in many coastal nations. Therefore, the best way to reduce high fishing efforts is to incorporate the needs of the local communities in decision-making where alternative source of livelihoods can be considered. The consideration of provision of alternative livelihoods must compose of diversification where fishers will not only depend on fisheries for their livelihoods (Blythe et al., 2014).

4.1.3. Climate change

At least three independent factors, rising temperature, ocean acidification, and ocean deoxygenation, have been reported to pose severe stress to the ocean's biogeochemistry and its ecosystems in the subsequent centuries Gruber (2011). Our analysis of the trends in climate change using satellite data from CMS (since 1990) showed an increase in sea surface temperature (SST: S_{SST} and F_{SST}) and a significant rise in the concentration of hydrogen ions (decrease in pH), surface partial pressure of carbon dioxide ($spCO_2$), and sea surface height above sea level (sea level anomalies - SLA) over time. These changes support both the global predictions of Chemane et al. (1997) and Gruber (2011) in their assessment of the vulnerability of coastal resources and communities to climate change emphasising that the reality of global warming lays severe consequences on most low-lying areas of Mozambique and worldwide. This study shows that water temperature in the bay increases towards the bottom, considering that the foundation sea surface temperature (F_{SST}) was higher than the skin sea surface temperature (S_{SST}) at any instance. This is because depth is an essential factor in temperature stratification. Oczkowski et al. (2015) observed that in well-mixed mid-estuarine systems having a water depth of about 10 m, temperature increases on average by 0.2 °C per meter depth. This can harm the flora and fauna, especially highly sensitive species (Sumaila et al., 2014; Gruber, 2011). The INB can be likened to the systems described by Oczkowski et al. (2015), since it has a mean depth of 7.5 m (range: 9.3 m and 17.5 m) (Solana et al., 2020). The monthly variations in SST results support Solana et al. (2020) and Ortiz Porras (2019), where both studies observed the highest and lowest mean month SST in February and August, respectively. Also, Solana et al. (2020) support this study that between the rainy and dry seasons, the highest SST is recorded in the rainy seasons (October–March) at a point where the open sea SST is not significantly different from that of the bay. However, this is different during the dry cold season month (April–September) when the bay becomes warmer than the open sea.

In this study, the correlations between pH and SST, $spCO_2$, and dissolved oxygen DO confirm the earlier predictions by Sumaila et al. (2014). These authors predicted the likely increase in ocean acidification (decreasing pH) in the Western Indian Ocean (WIO) due to increasing temperature and deoxygenation. The INB is part of the Mozambique channel, which in the year 1800 had a pH of 8.18 ± 0.014 (Lo Monaco et al., 2021). Our study shows that the current mean surface pH of the INB is 0.11 less, a clear signal of anthropogenic carbon load. Our study recorded the lowest pH in the INB in 2019, which is consistent with Lo Monaco et al. (2021), who showed that the pH of the Mozambique channel in 2019 was 0.13 less than that in the 1800s. A significant decline in surface pH across the entire WIO has been reported by Sumaila et al. (2014), who assessed the impacts of ocean acidification on artisanal fisheries and subsistence livelihoods on countries in the WIO. The average surface pH found in the study (i.e., 8.070 ± 0.001) is consistent with the surface pH of the Indian Ocean (8.07 ± 0.02) (Jiang et al., 2019). Results of this study are supported by the work of Gruber (2011), Sumaila et al. (2014), and Lo Monaco et al. (2021) that showed the difficulty in disentangling $spCO_2$ from surface pH because these two variables significantly covaried. Therefore, we can advance that more intake of CO_2 will result in the decreasing pH trend observed here.

Regarding SLA, Solana et al. (2020) and Oritz Ortiz Porras (2019) support our observation of the increasing SLA with increasing tidal range in the bay. Oliveros et al. (2011) observations indicated that rising sea levels coupled with annual cyclones increase the flooding effect in the Inhambane province. However, while constant SLA increases is beneficial for mangrove ecosystem expansion (McKee and Vervaeke, 2018), seagrass ecosystems and coastal communities stand at risk. Hence, it is a matter of the ability of the mangroves to buffer seagrasses and coastal communities.

4.1.4. Environmental quality

This study showed that although DO and pH related to temperature and carbon dioxide increase presented a trend to decline, their estimates fall within the allowable limits. According to Bhuyan et al. (2020) the optimum ranges of DO and pH in estuaries are 5–10 mg/l (i.e., 60–120% at saturation) and 8–9, respectively, below which aquatic organisms become stressed. The $spCO_2$ and SST had strong negative correlations with DO in the INB. Still, in systems with limited nutrients, a non-significant variation in temperature over time and pH within the alkaline limits, change in DO is likely to remain negligible/constant. According to the Oczkowski et al. (2015), systems that receive more freshwater accompanied by well mixing are likely to have reduced salinity. This event reduces DO loss in water. The silicate concentration, although within the reference limits, decreased through time. This may be related to pH decrease, since a pH decrease drives the decrease in silicate availability (Taucher et al., 2022). In the south Indian Ocean, the DO is high but limited in nutrients with associated low productivity as illustrated by Gruber (2011) and Sumaila et al. (2014). This could explain the low nitrates, phosphates, Chl *a*, phytoplankton concentration, and net primary productivity of the INB, which is located in that zone. Phytoplankton concentration in water is measured using Chl *a* in the water, hence decline in Chl *a* corresponds to a decrease in phytoplankton with a consequent fall in NPPV. Miguel (2018) assessed the seasonal variations in the biogeochemistry of the Macuse estuary, central Mozambique. The author found that during the rainy seasons, silicate, nitrate, and phosphate concentrations increase due to high runoffs and the influx of allochthonous nutrients. However, here this process was only found for nitrates; the other nutrients were higher in the dry seasons; so perhaps the INB is not highly dependent on allochthonous inputs.

4.1.5. Ecosystem attributes

Based on our results on the IMM biodiversity, the total number of species found in the INB was 227. Molluscs, followed by arthropods, were the dominant phyla. However, Day (1974), in his evaluation of the MMI biodiversity in the Morrumbene estuary, found 404 species with Arthropods (totalling 142 species) followed by Molluscs (117 species) as the dominant phyla. In our study, seagrasses presented more species than mangroves, but this needs confirmation from thorough field studies. A recent study by Chitará-Nhandimo et al. (2022), who combined direct field and digital data in 2017, 2018, and 2020 in seagrasses, found a decline in seagrass MMI richness in the INB. Our study on the overview of MMI diversity in the INB in terms of decades, revealed that MMI biodiversity (Simpson diversity index) was the highest in 1950–1959, and the lowest in 2000–2009 in seagrasses. Similar to the seagrass ecosystem, the Simpson diversity index in mangroves was highest in 1950–1959, but the lowest in 1989–1999. The functional diversity (FD) of MMI for the two functional groups (habitat occupation and trophic guild) assessed here was low (<0.5 for each) across both ecosystems. This represents reduced MMI functions with a consequential drop in MMI ES (Laureto et al., 2015). However, this low MMI FD may be due to sampling biases (e.g. sampling directed to certain taxa or ecosystem, randomness, limitation due to political turbulence from the late 60s–2000s) since our data was based on MMI species occurrences collected from different sources. Further studies designed to understand the MMI diversity (species richness and abundance) would help to overcome biases related to biodiversity assessments such as functional diversity, which can be directly linked to ES.

This study revealed that the management authority of the INB is halfway through achieving the Mozambique commitment toward the SDG 14.5, and the CBD and its Achi Biodiversity target 11, which stipulated that 10% of coastal and marine areas should be protected by 2020 (Grip & Blomqvis, 2018). This is because the ratio between the total area of all the 12 LMMAs and the total area of the bay was 4.7%. These measures can be improved in Mozambique and worldwide as surrounding communities are included in setting up local Marine Protected Areas (MPAs).

4.2. Ecosystem services of MMI

This study identified, classified, and quantified three primary ecosystem service types: provisioning, regulation and maintenance, and cultural services.

Food from crabs, cephalopods, shrimps, and lobsters was the only documented provisioning MMI ES extracted from the bay. The artisanal total MMI production contributes about 13% to the total fishery production in the province. In the INB, like other coastal communities in developing countries, the rural poor populations live on less than $1, so MMI are crucial as an alternative animal protein source (Chitará-Nhandimo et al., 2022; Uetimane, 2018; Bloecker, 2016). Many artisanal fishers capture mainly MMI through gleaning and beach seining activities (Chitará-Nhandimo et al., 2022), strongly contributing to food security and the achievement of the SDG2. In addition, the shrimp's contribution to semi-industrial fishing has also been recognised. Its recent decline has called for rapid measures to be taken to conserve shrimp stocks in the INB (Uetimane, 2018). Small-scale fisheries made up of subsistence artisanal and semi-industrial fishing contribute about 93% (91% and 2%, respectively) to Mozambique's total marine catch (Sumaila et al., 2014). In 2012, an official report indicated that approximately 400,000 Mozambique's population are employed in artisanal fisheries (Menon et al., 2021). Even though other areas, districts and bays in the Inhambane province contribute to the MMI production, the INB has received higher management attention since 2016, leading to the creation of 12 seagrass protected areas (LMMAs). According to Vandeperre et al. (2011), the establishment of no-take zones like MPAs (LMMAs in the INB) increases fish production. However, at this moment, associating the significant increase in MMI production in Inhambane since 2016 with the establishment of LMMAs in INB is difficult due to insufficient data. On the other hand, our literature search showed that many other provisioning services of MMI have been overlooked in the INB. For example, Prather et al. (2013) identified not just food but also other materials like silk from a worm; biochemical and pharmaceutical products: from sea urchins (e.g., containing holothurins, toxins for treating coronary disorders and cancer), cephalopods (e.g., octopus for hypertension), sponges (antiviral products for inhibiting common cold), chitinous crustaceans (for fighting malignant or dead cells and fungal infections) and adhesive chemicals from barnacles for tooth fillings (Rocha et al., 2019); and genetic materials. The inherent strength in the shells of bivalves (e.g., oysters) has been used in road construction (Etim et al., 2020), wall plastering and painting (Lee et al., 2022). This means that the knowledge of MMI ES in INB is either not profoundly understood, or these ES are not being explored. Thus, further studies are needed to fully understand the role of other MMI provisioning services.

The quantitative evaluation of regulation and maintenance ES provided by MMI in the seagrasses of INB included water quality regulation, habitat modification, water quantity regulation, MMI as decomposition precursors, and climate change regulations. The estimation of these ES was based on the number of trophic guild (TG) attributes and species richness of MMI in the INB, which relate to functional diversity (FD). Thus, high species richness increases FD of TG, which consequently ensures the contribution of ES evenly. For example, 1954, 1968, 1969, 2013, 2018, and 2020 (see Table 3) were years in which FD of TG were relatively high. Therefore, in these years seagrasses presented even ecosystem service contribution. On the other hand, years with low FD (due to low species richness) had few services, primarily favouring habitat modification and climate regulations. In our study MMI regulation and maintenance ES were probably underestimated due to data constraints related to occurrence data gaps in some decades, and the lack of reliable data on abundance. The availability of more consistent time series data on species occurrence and abundance would have allowed a better temporal estimation of these services. Nevertheless, our study provides relevant information that can support future regulation and maintenance ES assessments.

Concerning cultural services, namely recreation and sport fishing; presence of LMMAs which promote a sense of ownership, stewardship, and spiritual heritage; and MMI species richness for culture, education, scientific studies, or training, are crucial components of MMI services in the INB. All these services stem from the presence and increase in MMI biodiversity. These are services that are more intrinsic in nature and therefore are subjective. Maintaining the biodiversity, aesthetic and heritage values of MMI by conserving them and enhancing their key 'traditional' uses is the best way to strengthen their cultural services.

4.3. Data limitations

The results of this study were mostly based on data from satellite and digital biodiversity platforms, Natural History Collections (NHC) datasets, scientific literature and national reports on fisheries. Aside from the satellite data, the biodiversity digital platforms and NHC datasets had some limitations along with reduced literature studies in the study area. The MMI dataset contains mostly data from NHC, which was collected from the field dating from 1953 to 2020 but do not result from a systematic sampling methodology. There were some deficiencies in the dataset due to temporal and spatial biases. Large proportions of data were collected in certain years including 1954, 1968, and from the 2000s (between 2009 and 2020). The spatial disparities were as well within but more evident between ecosystems. For instance, more data were assembled in seagrass than in mangrove ecosystems. On the other hand, this study did not have access to data from other empirical studies including Day (1974) that recorded the highest number of marine macroinvertebrate species (i.e., 404) in the INB because it lacked a list and the number of the individual species associated with each phylum. We tried to reduce these biases during the calculation of functional diversity by combining data by decades. However, it is well to acknowledge that the temporal variations could not be completely solved in decades including 1960s, 1970s, 1989–90s, and 2000s. This was most likely related with the precolonial and post-civil wars that were going on during these periods in Mozambique (Coelho, 2002; Morier-Genoud et al., 2018), hence a higher possibility of hindering data collection in our study area as it may be for other areas of the country. Even though our dataset on INB reveals information about the diversity, trace historical composition of MMI, as well as knowledge of their ES, further analyses and interpretation will improve with the acquisition of more accurate proportional temporal and spatial data in the future.

4.4. Future perspectives

Increasing the area of seagrasses and mangroves, as observed in the INB, will have a positive effect on ecosystem environmental quality and biological attributes, namely those related to MMI distribution and biodiversity. With more available area, decision-makers can implement more protected areas to meet the SDG14.5, and the CBD. Increasing seagrass and mangrove cover will promote ecosystem health, contributing to a sustainable food production and thus food security concurring to achieve the SDG2. It will also uphold more effective MMI regulation and maintenance services, and the preservation of cultural services, contributing to the SDG14 in general. However, the increasing pressure from MMI fishing, as observed in the INB and in most coastal ecosystems, is expected to negatively affect MMI biodiversity, and it may also prevent the establishment of new protected areas in seagrasses and mangroves due to the scarcity of fishing areas. This, in turn, may hamper food sustainability and consequently food security, as well as MMI regulation and maintenance, and cultural services of those ecosystems. Therefore, it is extremely important to gather more data on MMI fisheries (from industrial to artisanal) and defining maximum sustainable yields to accurately implement sustainable fishery production and practices in collaboration with local communities.

The increasing pressure from climate variables, as observed in this study, namely the raise of sea surface temperature, sea level anomalies and spCO2, and the decrease of surface pH, which are expected to become more severe due to global changes, will negatively affect seagrass and mangrove condition and thus the MMI services. The studied pressures on the INB are not yet reflected on environmental quality and ecosystem attributes, which lay within normal limits, and they don't seem to be affecting the MMI ES. However, further studies are needed to obtain powerful data (replicable data obtained through well-defined sampling methods), especially on MMI biodiversity and abundance, for more accurate assessments.

The compilation of online data from digital platforms, scientific literature and reports, although with limitations, is a valuable tool to retrieve holistic temporal and spatial information about anthropogenic pressures, environmental variables, ecosystems and MMI biodiversity and ES. The data gathered during our study provides baseline information and suggests that the sustainable use of MMI ES needs a holistic approach considering, e.g., the trade-offs between the benefits of increasing both seagrass and mangrove cover, as well as the number of protected areas, and the negative pressures from fishing and climate change.

4.5. Policy Implications

Framing ecosystem pressures and condition of seagrasses and mangroves in the scope of MMI ES gives us a general perspective on how to address a sustainable use, conservation and management of these ecosystems to preserve the delivery of MMI ES in the long-term. Seagrasses and mangroves are highly productive and deliver important ES, but they are vulnerable to global changes, especially in developing countries where local communities are directly dependent on them and their resources. Thus, an integrated approach based on ES, using all the best available information and considering anthropogenic pressures and ecosystem condition, will foster effective interdisciplinary conservation and management plans across ecological and socio-economic systems.

4.6. Conclusion

This study assessed the links between changing conditions of seagrasses and mangroves and MMI ES through the aggregation of historical digital data and field data. The individual pressures analysed showed a negative trend, except for habitat conversion. The negative changes pose a severe threat and require urgent attention. This work established that increases in climate change variables, including, SST, spCO2 and SLA, show potential negative effects on environmental quality, namely Chl *a*, net primary productivity, phytoplankton, DO and PO_4 concentrations. Although the surface pH of the INB is within the allowable limits, it is significantly decreasing. The significant increases in $spCO_2$ and SST represent about 98.0% and 76.0% variation, respectively, which may hasten the decline of pH and pose severe threats in the future. This is especially concerning because pH had significant positive correlations with the environmental quality variables and, consequently, MMI ES. Therefore, periodic monitoring of pH, SST, and $spCO_2$ changes, along other environmental variables, is necessary to safeguard the health of seagrasses and mangroves and the human populations who depend on their MMI ES. The study examined the importance of MMI fisheries as an essential provisioning ES and highlights the need for greater attention to the increasing MMI and all fish stock production, alongside the annual increase in artisanal fishing licences. Overexploitation of MMI will result in declining MMI provisioning ES, which is detrimental to food security. Also, an in-depth survey of MMI provisioning ES is necessary to bring to light other potential services that can support the social-economic structure of local communities. The regulation and maintenance services, namely climate change regulation, water quality regulation, habitat modification, decomposition, and hydrological flux, varied through time showing inconsistencies primarily due to data gaps. This raises the need for more studies and data acquisition on MMI to improve our understanding of MMI's functions and services regarding regulation and maintenance ES. Nevertheless, this research established a methodology to quantify MMI regulation and maintenance ES through the combination of species richness and functional groups data, which can be used in the future. Conserving the biodiversity, aesthetic and heritage values of MMI and enhancing their 'traditional' uses is pivotal to strengthening their cultural services. The expansion of seagrasses and mangroves amid the recent cyclones in 2017 and 2019, and the establishment of LMMAs (since 2017) reveal a positive trend for cultural services, but this trend has simultaneously been paralleled by an increase in fisheries production. However, it is unclear whether the management strategy (LMMA establishment) significantly affects fisheries production other than fishing efforts. More empirical data is required to evaluate where fishing efforts can be controlled while preserving the area of MPAs and habitats for MMI cultural services. Future studies to look into the importance of habitat expansions and the establishment of LMMAs will also be necessary. Overall, this study provides a dataset and baseline information that can support future research, as well as decision-makers, to protect food security and human-wellbeing.

Authors contributions

Frederick Asante: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Resources, Data Curation, Writing - Original Draft, Writing - Review & Editing **Marta Bento:** Data Curation, Writing - Review & Editing, Visualization. **Stefanie Broszeit:** Conceptualization, Methodology, Validation, Investigation, Writing -

Review & Editing, Visualization. **Salomão Bandeira:** Methodology, Validation, Investigation, Writing - Review & Editing, Visualizatio. **Sadia Chitará-Nhandimo:** Data Curation. **Manuela Amoné-Mabuto:** Validation, Formal analysis. **Alexandra Marçal Correia:** Conceptualization, Methodology, Formal analysis, Validation, Investigation, Resources, Writing - Review & Editing, Visualization, Supervision, Project administration.

Declaration of competing interest

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

Data availability

Data have been deposited in Zenodo and the link has been provided in the materials and methods section 2.2.1.5.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marenvres.2023.106026.

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