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TITLE OF THE Ph.D. THESIS

Coastal wetlands food provisioning ecosystem services under a changing climate

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“Tra lo stagno di Mistras, che può dirsi una continuazione di quello di Pontis, e la torre grande, vennero praticati molti canali, che comunicano da una parte col mare, e dall’altra collo stagno di Cabras. Su questi canali trovasi appunto la così detta Peschiera di Pontis*, in cui pur sono le abitazioni dei pescatori, i quali stanno nell’acqua quanto le folaghe. I molti canali, che attraversano in ogni senso quelle spiagge, sono tratto tratto attraversati da ponticelli e le acque vengono qua e là divise da palizzate e da fitte barriere di canne; le quali, mentre lasciano libero corso al liquido elemento, tengono prigionieri i pesci.”*

Enrico Costa, 1887. La bella di Cabras.

The so-called “Peschiera di Pontis”. Cabras lagoon. Photo by Elisa Serra

Summary

Climate change (CC) is projected to affect coastal wetlands in multiple ways, altering their physical structure, community composition, ecological processes, and functions. However, society and ecosystems are deeply interconnected, since humans depend on ecosystems to live, while at the same time shaping and transforming ecosystems and their dynamics. The concept of socio-ecological systems (SESs), captures this complex interdependence, emphasizing that social and ecological systems cannot be fully understood separately. CC-induced modification of ecological processes and social system could indeed generate cascading and non-linear effects on both components, posing severe threats to SESs ability in providing Ecosystem Services (ESs).

In this challenging procedural context, the main goal of this my PhD project was to offer an entry point for the exploration of CC impacts on socio-ecological systems, by integrating and systematizing existing data, and generating new empirical evidence through mixed methodologies, both quantitative and qualitative.

Starting from a narrative literature review, I focused on what is known (and what is missing) about CC and coastal wetlands ESs. While this chapter confirms the growing interest of the scientific community towards this topic, I also highlight past and current methodological and knowledge gaps that still hamper our ability to fully understand and anticipate future CC impacts on SESs.

Then, taking Sardinia coastal lagoons and food provisioning ESs as case study I conducted two complementary assessments to evaluate CC vulnerability of SESs.

Notably, I explored the vulnerability of the ecological component, with a focus on those characteristics involved in the provisioning of the ESs that determine ecosystem propensity or predisposition to be adversely affected. Whit this study I contributed to fill a gap in the European context by scaling down existing vulnerability assessments to the specificity of the Sardinian context, emphasising that ecological vulnerability can be unevenly distributed in space, and highlighting the importance of conducting fine-scale regional scale assessment to identify where (and why) management intervention should be targeted.

Thus, by adopting a deductive-inductive mixed methodology, I explored the characteristics of the social elements of the SESs involved in the provisioning of the ESs that determine their propensity or predisposition to be adversely affected by CC, encompassing related concepts

such as social sensitivity and lack of capacity. I also identified barriers and enablers toward adaptation, ultimately contributing to the broader debate on socio-ecological vulnerability in coastal lagoon small-scale fisheries with a SES lens, providing a first case study of this kind in Sardinian lagoons.

Finally, to evaluate how and if existing knowledge can be incorporated in emerging ecosystem accounting frameworks, I explored the feasibility of compiling ecosystem accounts for Sardinia according to the System of Environmental-Economic Accounting - Ecosystem Accounting (SEEA EA), a recognized international statistical standard, using available data sources. While describing the potential of ecosystem accounting to inform cross-sectoral management and policy decisions, also considering the ongoing threats posed by CC, the availability, the choice of indicators and the data uncertainty handling played a central role in my study, potentially affecting the full operationalization of the accounts at subnational scale in the near future, especially in effectively tracking ecosystem condition and their services over time.

Though aware that the case studies included here represent only a few pieces of the complex mosaic of SESs, my thesis has put light on aspects which, while relevant at first instances only at a regional scale, can nonetheless contribute to support advancing in future implementations of local-based CC adaptation strategies.

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Thesis Outline

This thesis, aimed at exploring the effects of Climate Change (CC) on food provisioning ecosystem services (ESs) of transitional socio-ecological systems, is divided in 6 chapters.

Chapter 1 introduces the definitions of ecosystem services, socio-ecological systems and climatic change, as well as the main gaps about and challenges for their study. Contents of this chapter represent the knowledge-building basis that allowed the choice of methods and tools adopted in this thesis. It also zooms on transitional socio-ecological system in Sardinia, to provide an overview of the selected case area.

Chapters 2, **3**, **4** and **5** are structured as self-standing scientific papers according to the IMRaD format: Introduction, Methods, Results, and Discussion.

Chapter 2, entitled "Climate change impacts on coastal wetland ecosystem services: a narrative review" reviews the available scientific literature dealing with the assessment of CC impacts on coastal wetlands ESs with the aim of identifying the most recurrent CC-related drivers of impact, the most often applied methodologies and the knowledge gaps.

Assessing the vulnerability of a socio-ecological system to CC requires the comprehension and assessment of both social and ecological attributes of the system under scrutiny.

To operationalise this need, **Chapter 3** (Climate change ecological vulnerability assessment of food provisioning ecosystem services) and **Chapter 4** (Socio-economic perspectives on climate vulnerability of food provisioning) report the results of complementary assessments at the regional scale (Sardinia) of:

- Sardinian ecological vulnerability, with a focus on ecosystem stocks/assets and conditions involved in their ability to provide ecosystem services, and
- their social vulnerability, with a focus on human capital/assets involved in the interaction with the ecological component to provide ESs.

For the ecological component (**Chapter 3**), considering the regional scale of this study, the analysis was somehow constrained by data availability. Secondary data collected at the local scale at regional level were used to derive composite indicators of ecological vulnerability while, at the same time, identifying critical data gaps that could only be addressed through new monitoring efforts.

For the social component of the system (**Chapter 4**, jointly developed under the supervision of Dr. Tiziana Luisetti and Prof. Irene Lorenzoni), where secondary information at the local scale was still missing, data collection through questionnaires was undertaken to explore vulnerability of food provisioning systems through a combined deductive-inductive approach.

Chapter 5 (jointly developed under the supervision of Dr. Tiziana Luisetti and Prof. Irene Lorenzoni), entitled "Monitoring ecosystems and their services in a changing climate context: Experimenting SEEA EA at local and regional scales in Sardinia" explores how the existing knowledge can be incorporated into ecosystem accounting frameworks, using the System of Environmental-Economic Accounting-Ecosystem Accounting (SEEA EA).

Finally, **Chapter 6** considers the relevance and innovation of the thesis work in relation to the context of coastal socio-ecological systems and climate change impact research, providing conclusions and future research perspectives.

Chapter 1. Introduction

1.1. Ecosystems services and socio-ecological systems

Society and ecosystems are deeply interconnected: humans depend on ecosystems to live, while at the same time shaping and transforming ecosystems and their dynamics (Haines-Young & Potschin, 2010). The concept of socio-ecological systems (SESs; Ostrom, 2009; McGinnis & Ostrom, 2014), captures these complex interdependences, emphasizing that social and ecological systems cannot be fully understood separately, as they are interdependent and, thus, co-evolve (Berkes & Folke, 2000; Ostrom, 2009).

During the last decades, the concept of Ecosystem Services (ES) has emerged as a central paradigm for linking ecosystem structure, processes and functioning with human well-being (Costanza et al., 2014). The ESs concept, rooted in the convergence of ecology and economics in the late 1970s, evolved till the widely known definition proposed by Costanza (1997, p.253) as “the benefits human populations derive, directly or indirectly, from ecosystem functions”.

The meaning of the ESs term and concept has evolved over time, and several classification systems have been developed since the beginning of the 21st century. The Millennium Ecosystem Assessment (MEA; 2005) represented an important milestone in ESs research (La Notte et al., 2017) and recognition, providing a comprehensive assessment of the consequences of ecosystem change on human well-being (MEA, 2005).

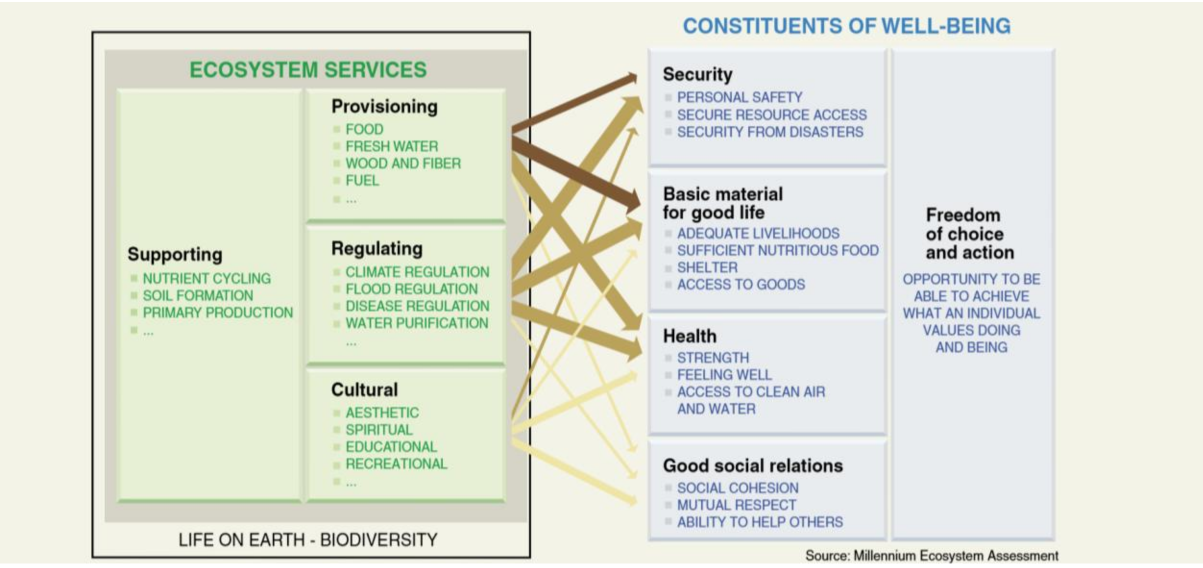

Figure 1.1 Linkages between Ecosystem Services and Human Well-being, From MEA, 2005.

The MEA defines ES as ""benefits people obtain from ecosystems" (MEA, 2005, p.40), and introduces a categorisation consisting in four main groups: provisioning services, regulating services, supporting services, and cultural services, where:

- Supporting services form the foundation for all other ESs
- Provisioning services supply tangible products such as food, fuel and fiber
- Regulating services delivery benefits through the regulation of ecosystem processes, including climate and water regulation, erosion control among others
- Cultural services offer intangible benefits, contributing to the maintenance of human well-being through spiritual enrichment, reflection, cognitive development and aesthetic and recreational experiences.

Since then, there has been a significant increase in the number of studies and initiatives, such as The Economics of Ecosystems and Biodiversity initiative (UNEP, 2012), the Mapping and Assessment of Ecosystems and their Services initiative (MAES; Maes et al., 2013), the Integrated system for Natural Capital and ecosystem services Accounting (INCA) project (La Notte et al., 2022), the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES; Díaz et al, 2015), among many others. Not all initiatives use the same categories as those proposed by MEA, for instance, The Economics of Ecosystems and Biodiversity (2012) disaggregated ESs into provisioning, regulating, habitat and cultural.

The connection between ecosystems and human well-being is frequently represented through the "ecosystem service cascade," conceived as a production chain where ecosystem structures and processes are progressively translated into benefits and values for the society through a series of intermediate stages (Haines-Young & Potschin, 2010).

Building on the ES cascade model, the Common International Classification of Ecosystem Services (CICES, see Haines-Young, 2023 and Table 1.1 for version 5.2), developed in the context of work on the System of Environmental and Economic Accounting (SEEA; Haines-Young, 2023), introduced a more standardised and hierarchical classification of ESs to improve cross-study comparability, by reducing the number of categories to three (*i.e.* CICES sections; Haines-Young, 2023, Haines-Young & Potschin, 2023; Table1). According to CICES, ESs are defined as "the contributions that ecosystems make to human well-being" (*i.e.* "*what ecosystem do*"; Haines-Young & Potschin, 2018, p.7) and are distinct from the goods and benefits that people subsequently derive from them (Haines-Young & Potschin-Young, 2018; Haines-Young, 2023).

Although ecosystem outputs derived from living structures and processes remain the focus of this classification, CICES also provides a way to classify the non-living (abiotic) contributions (i.e. geophysical ecosystem outputs; Haines-Young, 2023).

The concept of ESs is thus clearly related to human needs (Krohs & Zimmer, 2023). ESs does not manifest as such within the ecosystem *per se*, but only in the perspective of their utility for humans (Krohs & Zimmer, 2023).

Notwithstanding the significant progress made towards achieving a standard classification of ESs, a globally agreed and used classification is still lacking. While existing conceptual frameworks provide a basis for categorising and understanding ESs, their practical assessment requires methodologies capable of capturing the complexity and the contribution of both ecological processes and the socio-economic context in which ESs and their benefits are generated, extracted and used (Burkhard et al., 2012; McDonough et al., 2017).

Table 1.1 Overview of CICES classification (V5.2) for biotic/biophysical ecosystem services (upper three levels in the classification only. From Haines-Young, 2023)

Section	Division	Group
Provisioning	Biomass	- Cultivated terrestrial plants for nutrition, materials or energy - Cultivated aquatic plants for nutrition, materials or energy - Reared animals for nutrition, materials or energy - Reared aquatic animals for nutrition, materials or energy - Wild plants (terrestrial and aquatic) for nutrition, materials or energy - Wild animals (terrestrial and aquatic) for nutrition, materials or energy
	Genetic material from all biotas	- Genetic material from plants, algae or fungi - Genetic material from animals - Genetic material from organisms
Regulating & Maintenance	Transformation of biochemical or physical inputs to ecosystems	- Reduction of nutrient loads and mediation of wastes or toxic substances of anthropogenic origin by living processes - Mediation of nuisances of anthropogenic origin
	Regulation of baseline flows and extreme events	- Erosion control - Hydrological cycle and water flow regulation - Hazard mitigation
	Regulation of physical, chemical, biological conditions	- Lifecycle maintenance, habitat and gene pool protection - Pest and disease control- Regulation of soil quality - Water conditions - Atmospheric composition and conditions
Cultural	Intellectual and representative interactions with natural environment	- Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting - Indirect interactions with living systems
	Physical and experiential interactions with natural environment	- Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting (i.e. broadly recreational activities)
	Spiritual, symbolic and other cultural interactions with natural environment	- Elements of living systems that are indirectly appreciated and have significance for people without their presence in the environmental setting - Other biophysical characteristics of species or ecosystems that are appreciated in their own right by people

1.2. The Mediterranean Sea: hotspot of biodiversity, hotspot of changes

The Mediterranean basin, a micro-tidal sea known for its cultural and exceptional biological diversity, has been recognized as a hotspot for climate change (CC) (MedECC, 2020; Chiggiato et al., 2023; Lionello et al., 2023; Diffenbaugh & Giorgi, 2012).

For thousands of years, the Mediterranean region has been strongly shaped by human activities and today is home to over 500 million inhabitants, with dense urban areas and major industrial facilities located close to sea level, representing the world's leading tourist destination and one of the busiest maritime corridors globally (Ali et al., 2022).

Since 1990, its population has grown by nearly one third, with coastal zones showing an increase of more than 42% around coastal wetlands, such as coastal lagoons areas, population densities are higher of those of other coastal areas (Ali et al., 2022).

Global warming has already reached 1.5 °C above pre-industrial levels, exceeding global average rates since 1980s, with more frequent high-temperature extreme events, increasing in frequency and intensity of droughts events (Ali et al., 2022).

Warming of the sea surface temperature follows the same trend, with regional variations between 0.29-0.44 °C per decade since the early 1980s (Darmaraki et al., 2019). Sea level has risen at a rate of nearly 1.4 ± 0.2 mm/year during the 20th century (Wöppelmann & Marcos, 2012) and is projected to rise further, being likely to reach 0.52 m (0.32-0.81 m) for scenarios SSP1-1.9, up to 1.22 m (0.91-1.78 m) for SSP5-8.5 relative to 1996-2014 climatology by 2100 (IPCC WGI AR6 Chapter 9, Fox-Kemper et al., 2021).

Mediterranean coastal wetlands, at the interface between land and sea, are ecosystems considered particularly vulnerable to CC (de Vicente, 2021; Schuerch et al., 2018).

Modelling projections for the Mediterranean indicate that coastal wetlands are highly threatened, with potential losses exceeding 70-90% by 2100 under medium sea level rise (SLR) scenario and if no adaptation and mitigation measures will be taken (Schuerch et al., 2025; Figure 1.2).

Among coastal wetlands, coastal lagoons, of which saltmarshes are integral features (Corbau et al., 2022), are particularly exposed to the impacts of climate change (Brito et al., 2012), mainly due to their confinement, shallow depth, and location at the interface between land and sea (Derolez et al., 2025; Pérez-Ruzafa et al., 2019).

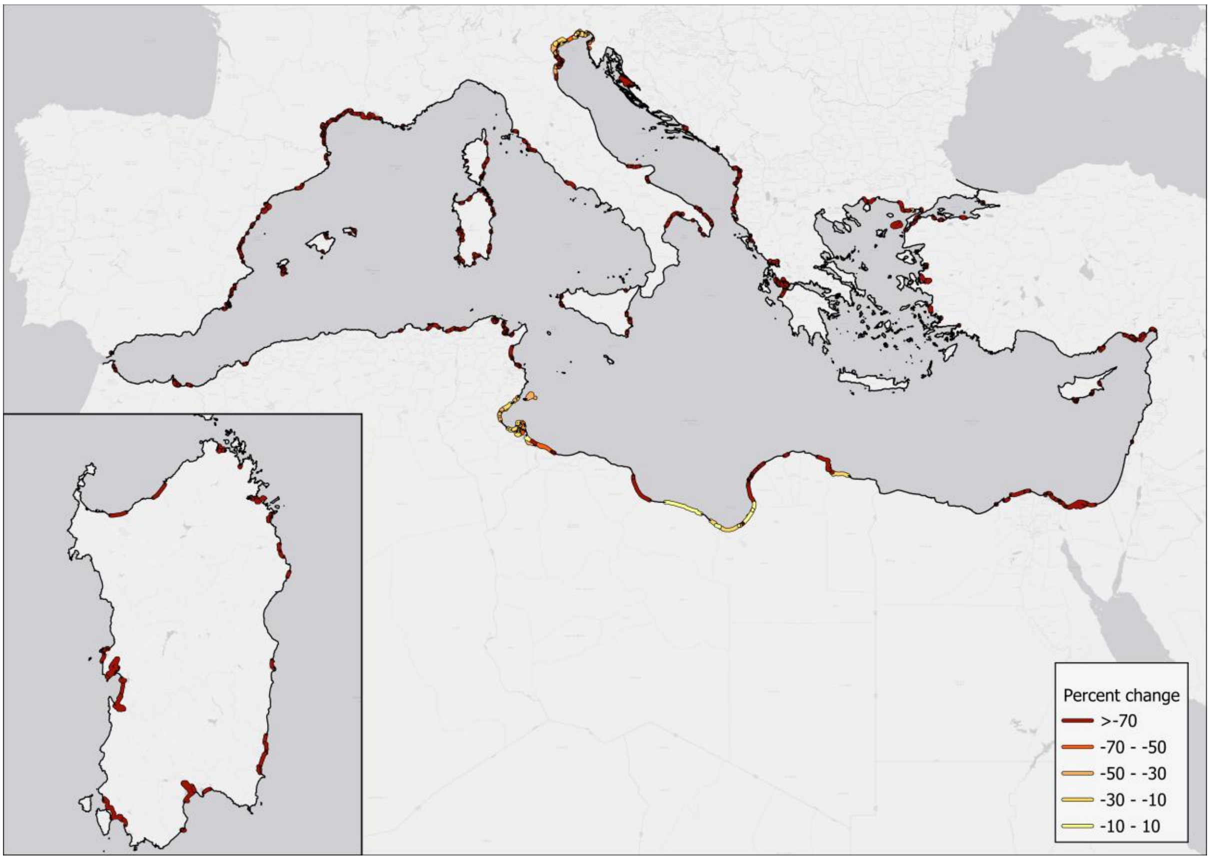

Figure 1.2 Spatial distribution of modelled changes of Mediterranean coastal marsh areas between 2020 and 2100 for a medium climate scenario SSP2-4.5 under coastal management scenario where wetland inland migration is highly constrained. (From Schuerch et al., 2025). In the lower panel the predictions for Sardinia are shown. All of these changes are also likely to affect the other adjacent ecosystems such as lagoon ecosystems, which are deeply interconnected to saltmarshes dynamics.

Coastal lagoons, encompassing a wide hydro-morphological variability ranging from leaky to chocked systems (Umgiesser et al., 2014; Kjerfve, 1994), are transitional water ecosystems naturally characterized by strong natural physico-chemical disturbance and large environmental fluctuations (Lacoste et al., 2023; McLusky & Elliott, 2007; Kjerfve, 1994).

Their hydrological regime is primarily driven by the balance between freshwater inflows and seawater intrusion, which, in turn, depend on the width and morphology of the sea inlets, riverine inputs, wind and wave forcing, and sea-level conditions (Ennas et al., 2025).

Riverine inputs, driven by precipitation patterns, directly shape physical-chemical and trophic characteristics of the lagoons, affecting salinity, nutrient concentration and availability, sedimentary organic matter quantity, composition and degradation rates, microbial assemblages, and thus the overall biogeochemical cycles (Pusceddu et al., 1996; Bianchelli et al., 2020; Magri et al., 2020; Bartoli et al., 2021; Palmas et al., 2025).

Sedimentary trophic conditions in coastal lagoons, while partly controlled by hydrodynamics and sedimentological processes, are also modulated by biological mechanisms such as *in situ* organic carbon production, heterotrophic consumption, and microbial-mediated degradation (Ennas et al., 2025).

Being shallow water ecosystems, the sediment trophic status of coastal lagoons generally shows greater temporal stability compared to the water column above, where rapid hydrological and biological dynamics make water column descriptors of trophic state more variable and less conservative (Dell'Anno et al., 2002).

Under CC these dynamics are expected to intensify (Snoussi et al., 2008). Sea level rise and warming will likely lead to changes in the physical properties of coastal lagoons, which are likely to experience an increase of their vulnerability to eutrophication (Brito et al., 2012) and a progressive hydrological homogenization, a process that has been described as a loss of "hydrodiversity" (Ferrarin et al., 2014).

Past model projections suggested that anomalies in temperature and salinity will be amplified in these confined systems compared to the adjacent open sea (Ferrarin et al., 2014; Figure 1.3), with temperature anomalies on average 15% higher, and of an extent similar to that expected for the atmospheric temperature, being particularly pronounced in shallow lagoons with low flushing rates (Ferrarin et al., 2014).

Such physical-chemical alterations caused by CC are expected to propagate through the entire ecosystem, affecting local biogeochemical cycles (Mora et al., 2013) and thus, influencing organic matter decomposition, food web length (Hansson et al., 2013), species behaviour, ultimately reducing site-specific species richness and fostering a homogenization of biota (Ferrarin et al., 2014).

Furthermore, recent experimental evidence from a Mediterranean lagoon indicates that marine heatwaves (i.e. "discrete prolonged anomalously warm water events"; Hobday et al., 2016, p.229) can induce marked shifts in planktonic communities, altering trophic interactions and potentially affecting the whole ecosystem functioning (Pulina et al., 2025).

Similarly, heatwaves have been shown to alter the biogeochemistry of lagoon sediments, reducing the quantity and nutritional quality of organic matter, thereby altering benthic food availability for the benthos and potentially constraining the overall ecosystem functioning (Palmas et al., 2025).

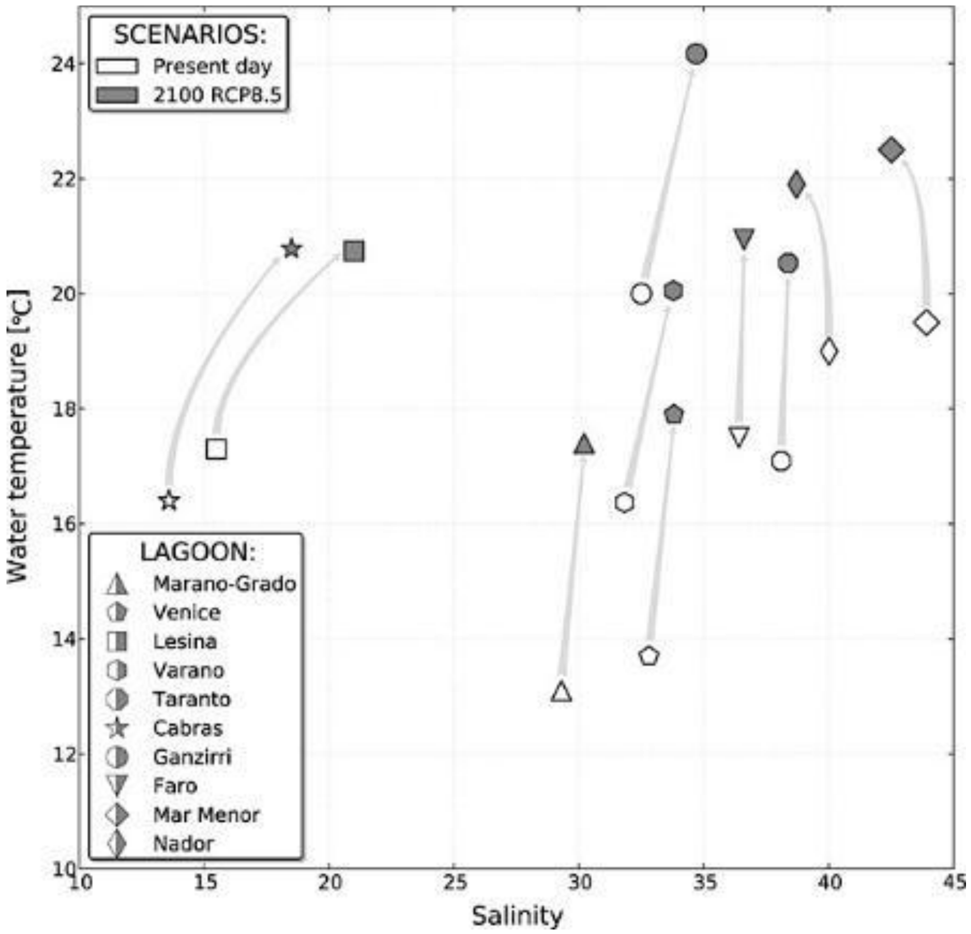

Figure 1.3 Present-day (empty markers) and predicted for the end of the century under the RCP8.5 scenario (filled markers), basin-wide average salinity, and water temperature in 10 Mediterranean lagoons. From Ferrarin et al., 2014

However, while important changes are generally expected in the physical and ecological characteristics of coastal lagoons due to CC (de Vicente, 2021; Palmas et al., 2025; Anthony et al., 2009), assessments of these changes and their cascading impacts for the Mediterranean Sea from at basin down to the local scale remain scarce, yet (Ferrarin et al., 2014).

1.3. Transitional socio-ecological systems under threat: Sardinia as a case study

Being the second largest island in the Mediterranean Sea, Sardinia, located in the western part of the basin, has a surface of 24,090 km^2. A recent inventory identified 2501 wetlands, both inland and coastal, covering a total surface area of 494.2 km^2 (Fois et al., 2021).

Among these, Sardinia hosts one of the richest and extensive networks of coastal lagoons in Italy which, together with other transitional environments, represent important socio-ecological systems (Giampaoletti et al., 2025; Fois et al., 2021). These ecosystems reflect the geographic and climatic characteristics of the region, while also displaying specific features shaped by the historical management of watersheds, adjacent marine areas, and the lagoons themselves (Padedda et al., 2019).

The current trophic conditions of Sardinian coastal lagoons are not substantially different from the past, with a moderate number of areas classified as eutrophic or hypertrophic (Padedda et al., 2019; Pusceddu et al., 2007), and some of theme experiencing the occurrence of dystrophic events (Magni et al., 2008; Padedda et al., 2019; Palmas et al., 2025).

Key drivers influencing the trophic status of coastal lagoons include nutrient loads, that often exceeded lagoon' tolerance thresholds, and the limited water turnover, a critical factor in regulating the dilution and removal of excess nutrients, oxygen supply, and salinity fluctuations (Padedda et al., 2019).

Due to the Mediterranean climate regime, Sardinian coastal lagoons are subject to high seasonal variability in the quantity and quality of water inputs, with freshwater inputs historically higher in winter and nearly absent in summer, when the water balance mainly depends on seawater inputs (Padedda et al., 2019).

As a result, water turnover time varies greatly, from an average of around 10-20 days observed in winter to more than 100-200 days in the summer season (Padedda et al., 2019). Moreover, in many Sardinian lagoons, these conditions can be exacerbated where fixed fishing structures (i.e. *lavorieri*) located in the sea channels are present (Padedda et al., 2019).

Local climatic conditions can also exacerbate lagoon eutrophication processes, particularly during summer, when water temperatures may exceed 30 °C along with concurrent periods of calm winds (Padedda et al., 2019).

Considering their characteristics, as well as their strong link with coastal communities, Sardinian coastal lagoons constitute an emblematic and complex example of socio-ecological systems, thus representing an ideal case study for exploring the challenges related to ESs provisioning and flows as affected by climatic change.

However, a recent study reviewing the conservation of Sardinian wetlands has shown that scientific efforts carried out so far on these ecosystems have been disproportionately focused on the largest sites (such as the Santa Gilla lagoon in Southern Sardinia; Fois et al., 2021). Moreover, while identifying several other important knowledge gaps, Fois and co-authors (2021) highlights also the lack of updated and site-specific ecological information, insufficient integration of ecological and social dimensions (e.g. the study of human attitudes), and limited multidisciplinary approaches (e.g., one health approach). These severe knowledge limitations, all of which exacerbated by poorly effective monitoring activities, pose important constraints for the conservation and management of these fragile ecosystems (Fois et al., 2021).

Indeed, while grey literature (see the Regional Strategy for Climatic Changes, RAS, 2024) and several peer-review studies have addressed the trophic status and ecological functioning of Sardinia coastal lagoons also in relation with climate change (e.g., Palmas et al. 2025; Ennas et al. 2025; Pulina et al., 2025; Ferrarin et al., 2014), knowledge on the combined ecological and social impacts of climate change remain, if any, still fragmented.

1.4. Thesis' foundational concepts and definitions

As explained in the above sections in this chapter, and noted in several studies, as CC impacts cascade through the socio-ecological systems, ultimately impairing ESs and related benefits for human well-being, there is a pressing need for structured approaches to assess how CC affects socio-ecological systems (Berrouet et al., 2018), especially in transitional areas such as coastal lagoons.

Adopting a socio-ecological systems perspective can help emphasize the linkages and the interdependence of both social and ecological systems, moving beyond the analysis of the isolated components and recognizing their dynamic often non-linear relationships and the complex feedback that shape them across time and scale (Raufflet, 2000; Biggs et al., 2012; IPCC, 2014; 2022; Hossain et al., 2020).

In this context, vulnerability and associated risk frameworks can provide a lens to analyse these threats. Vulnerability assessment, representing a critical foundation for understanding and anticipating CC impacts, have so far played a key role in assessing the climate vulnerability of species, ecosystems and human communities at different scales and levels of organization (IPCC, 2014; 2022).

Vulnerability-based prioritization management has been also recognized to be critical in disaster management and strategic planning (Mahmutoğulları et al., 2025), since it can inform the implementation of targeted and relevant prevention, adaptation, and mitigation options (Fakhruddin et al., 2020).

The concept and definition of vulnerability originated in the late 1960s, and since then have gradually evolved (Estoque et al., 2023). Already 50 years ago, White stated that:

"Vulnerability *is the degree to which a system, sub-system, or component is likely to experience harm due to exposure to a hazard, either a perturbation or stress"* (White, 1974).

Shortly afterwards, Timmermann (1981, p.17), in his exploration of vulnerability and resilience concepts in socioeconomic systems, stated that:

"[…] *'vulnerability' is useful rhetorically: the introduction to this paper charted the rise in concern which has forced us to consider the idea, if not the term. And yet, 'vulnerability' is a term of such broad use as to be almost useless for careful description at present, except as a rhetorical indicator of areas of greatest concern. For a start, vulnerability has only one true opposite - invulnerability - which probably does not obtain in the real world*".

Later, in its Third (TAR; IPCC 2001) and Fourth (AR4; IPCC 2007) Assessment Reports (Nguyen et al. 2016; Aslam et al. 2018) the IPCC defined vulnerability to CC as:

"*The degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate variation to which a system is exposed, its sensitivity, and its adaptive capacity*" (IPCC, 2001, p.225).

The later Special Report on Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation (SREX; IPCC, 2012) and the Fifth IPCC Assessment Report (AR5) in 2014 introduced a significant conceptual shift toward a risk-based approach (Estoque et al., 2023). These reports, including the last one published in 2022, defined risk as a function of hazard, exposure, and vulnerability (IPCC, 2014, 2022).

In this revised framework, vulnerability is now conceived as an intrinsic property of a system, regardless of its exposure (which now refers to exposed elements) to climate hazards, (IPCC 2012, 2014), and is defined as:

"*The propensity or predisposition to be adversely affected. Vulnerability encompasses a variety of concepts and elements including sensitivity or susceptibility to harm and lack of capacity to cope and adapt*" (IPCC, 2014b, p.128).

Nonetheless, this conceptual shift has generated ongoing debate in the scientific community, as many studies have not yet fully adopted the revised concepts, and continue to rely on the earlier TAR/AR4 framework, due to its clearer application, particularly regarding how exposure should be treated in standalone vulnerability analyses, and since differential vulnerabilities can relies on differential exposures (Estoque et al., 2023; Ishtiaque et al., 2022).

Thus, despite their widespread adoption, vulnerability and risk assessments face several challenges in their operationalization (Estoque et al., 2023). They have been widely applied across different socio-ecological contexts to identify drivers and metrics, yet they are though often limited by indicator selection, conceptual inconsistencies, and lack of site-specific data (Birkmann et al., 2022; Eriksen et al., 2021; Kasthala et al., 2024).

As a result, to date, there is no one standard approach nor clarity how to operationalize such assessments (Fakhruddin et al., 2020).

These issues are particularly evident when applying vulnerability assessments to socio-ecological systems (Berrouet et al., 2018). Despite the growing recognition of SESs as interconnected entities where social and ecological components deeply link with one each other, vulnerability analysis of these systems is still in its infancy (Berrouet et al., 2018). Studies have explored the assessment of vulnerability of SESs with different conceptual frameworks and analytical approaches, noting that most of them have still not adequately integrate the link between the ecological and social systems (Berrouet et al., 2018).

This thesis draws upon the IPCC's (2022; 2014) revised vulnerability concept and the insights and challenges from past vulnerability application on SESs and includes the vulnerability of both ecological and social components of the SESs related to the ecosystem services flow. Thereby, in this thesis, from this point forward, the following definitions will be adopted:

Ecological vulnerability includes those characteristics of the ecological component of the SES involved in the provisioning of the ESs that determine their propensity or predisposition to be adversely affected: it encompasses the sensitivity and capacity (IPCC 2022) of habitat and species contributing to the ESs flow.

Social vulnerability reflects those characteristics of the social elements of the SES involved in the provisioning of the ESs that determine their propensity or predisposition to be adversely affected (Berrouet et al., 2018), encompassing related concepts such as social sensitivity and lack of capacity (IPCC 2022).

Social sensitivity represents the degree to which the social system is affected and is defined by factors that influence the impacts of a hazard (IPCC, 2022), such as the level of dependence of the social system on the capacity of the ecological system to provide the food provisioning ES (Berrouet et al., 2018).

Social capacity represents the ability of a social system to anticipate, prepare for, and respond to current and future CC impacts. Capacity can be further distinguished into two key aspects, coping and adaptive capacity (IPCC, 2022).

Social capacity to cope is the ability of the social component of a system to address, manage, and overcome adverse conditions in the short to medium term, by using available skills, resources, and opportunities (IPCC, 2022).

Social adaptive capacity is the ability of social systems to adjust to potential harm, seize opportunities (IPCC 2022; Berrouet et al., 2018).

While vulnerability assessments can provide a lens for understanding SESs and CC threats and planning intervention and adaptation strategies, their application remains limited to point-in-time analyses. This can be especially true when such assessments are embedded within adaptation strategies and planning documents, such as in the case of the Sardinian Regional Adaptation Strategy to Climate Change (SRACC; RAS, 2024).

Although SRACC recognized the importance of monitoring and reporting activities, their implementation may follow rather long and vague cycles, often relying on periodically updated reports without clear temporal targets, as also observed for the Italian National Strategy (European Commission, 2018). Moreover, their effective application strongly relies on the availability of robust time-series data, which often represent a strong limitation.

Thus, in the context of CC, there is growing demand for tools that can systematically integrate ecological and socio-economic information to support decision-making and be mainstreamed across different policy strategies. To design CC adaptation measures, it is not sufficient to capture one-time impacts or sporadic analysis, since decision-making needs consistent, comparable, and repeatable tools that track how ecosystem conditions and service delivery evolve over time.

In this sense, vulnerability and risk assessments need to emerge in a decision-making context where changes in SESs and ESs are monitored in a robust and systematic manner.

The need for integrated monitoring approach linking ecosystems and human wellbeing has led to increasing interest in ecosystem accounting frameworks, such as the UN's System of Environmental-Economic Accounting - Ecosystem Accounting (SEEA EA; Edens et al., 2022). Such approach, based on an official international statistical standard which integrate ecological and economic data, provides systematic information on ecosystem extent, condition, and ecosystem services flows (United Nations, 2021).

The SEEA EA framework is developed for a yearly-basis reporting on ecosystem services and on a triennial cycle for tracking changes in ecosystem condition and extent, thus providing a more consistent basis for tracking changes over time (Regulation EU 2024/3024).

While a promising monitoring framework, the application of the SEEA EA framework is still limited, and mostly focused on terrestrial habitats (Gaglio et al., 2024). As mentioned above for the vulnerability assessments, data gaps remain the major obstacle, especially in coastal areas (Buonocore et al., 2021; Buonocore et al, 2020; Chen et al., 2020; Gaglio et al., 2024; Townsend et al., 2018).

1.4.1. Thesis main aim(s)

In this challenging procedural context, the main goal of my PhD project was to offer an entry point for the exploration of CC impacts on socio-ecological systems, by integrating and systematizing existing data, and generating new empirical evidence through mixed methodologies, both quantitative and qualitative.

After setting the general foundation with a systematic literature review, this thesis narrows down to a peculiar case study: Sardinia's transitional ecosystems, and notably coastal lagoons, which represent emblematic examples of SES in which local communities and ecosystems have co-evolved, continuously influencing and shaping one each other.

Throughout this thesis, socio-ecological systems and the actual flow (i.e. the historical and actual ESs extraction; the *de facto* used ESs, Burkhard et al., 2014) of food provisioning ESs were analysed as a concrete example of how CC impacts on socio-ecological systems can be explored even in a context of data limitation.

The PhD project adopted an interdisciplinary approach, integrating applied ecology, environmental economics, and social sciences. The rationale behind this integration lies in the inherently complex nature of coastal socio-ecological systems, which cannot be adequately understood or managed through the lens of a single discipline.

The ecological dynamics of transitional ecosystems are indeed strongly shaped by local communities and the demand for the ecosystem services they provide, the value they attribute to them, as well as their perceptions and behaviours.

Addressing CC threats in these systems therefore requires analytical tools and conceptual frameworks that bridge disciplinary boundaries. Rather than simply juxtaposing different methods from different disciplines, this study actively seeks integration by designing research questions that require inputs from all three domains.

This integration was achieved by aligning scales of analysis, ensuring conceptual consistency as outlined in previous sections, and adopting a SES perspective throughout the research design.

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Chapter 2. Climate change impacts on coastal wetland ecosystem services: a narrative review

2.1. Introduction

Coastal wetlands, encompassing a wide range of eco-geomorphic settings and habitats such as mangroves, salt marshes, estuaries, and lagoons, are widely acknowledged for their high productivity (Alongi, 2014; Marchand et al., 2022). These ecosystems can be found along the coasts of all continents (Hopkinson et al., 2019), occupying a substantial portion of low-elevation coastal zones (Törnqvist et al., 2021). They are characterised by a unique combination of biogeochemical and hydrological features, as well as by peculiar and well-adapted communities with the ability to thrive in these stressful environments (Marchand et al., 2022; Windham-Myers et al., 2022). Coastal wetlands, being located at a crucial interface between terrestrial and aquatic ecosystems, represent peculiar transitional ecosystems that provide a plethora of ecosystem goods and societal benefits essential to humanity (Costanza et al., 1997; Franco et al., 2006; Costanza et al., 2014; Barbier et al., 2019).Despite huge conservation and management efforts to preserve wetlands, habitat loss and ecosystem degradation still affect them, with more than 50% of the world's coastal wetlands having disappeared since the beginning of the last century (Gardner et al., 2015; Li et al., 2018). Land reclamation, pollution, resources overexploitation and invasive allochthonous species are among the main causes of coastal wetlands' loss and degradation (Millennium Ecosystem Assessment, 2005; Blankespoor et al., 2014). Insofar, their conversion into agricultural and urbanized areas, and the continuously increasing human population inhabiting on their coasts, heavily impacted these ecosystems (MEA, 2005).

Climate Change (CC) is expected to impact coastal wetlands in different ways, impairing their structure, communities' composition, ecosystem processes, and functions (Lorrain-Soligon et al., 2023; Schuerch et al., 2018; 2025). These impacts could be reversible until critical points (*i.e.*: tipping points) are reached and crossed, otherwise leading to possibly irreversible changes (Lenton et al., 2019). The magnitude of CC impacts is predicted to be enormous in coastal wetlands characterised by a large dependence from a small amount of foundation species (Osland et al., 2013). The expected CC hazards for coastal wetlands include Sea Level Rise (SLR), water and air temperature increase, changes in frequency and intensity of precipitation patterns, but also extreme episodic climatic events such as heatwaves, intense storms, larger wave height, floodings, and droughts (Zhang et al., 2019; Chu et al., 2019; IPCC, 2021; Salimi et al., 2021; Ennas et al., 2024). These CC manifestations are expected to have profound effects on coastal wetland ecosystems (IPCC, 2014).For instance, albeit with regional and local differences in the magnitude and rate, global mean sea level is projected to rise by the end of the 21st century at faster rates than in the last three millennia (Vermeer & Rahmstorf, 2009), reaching values up to 0.77 (0.63-1.01) m under the SSP5-8.5 scenario (Shared Socio-economic Pathway "Fossil-fueled Development"; very high greenhouse gases emissions, CO_2 emissions triple by 2075 and up to 4.4 warmer temperatures; AR6; Fox-Kemper et al., 2021). SLR is predicted to lead to salt-water intrusion in coastal wetlands (Eidam et al., 2020; Robins et al., 2014), determining changes in nutrient dynamics and biogeochemical cycles (Herbert et al., 2015; Tully et al., 2019), by affecting the composition and activity of microbial communities, possibly impacting nitrification and coupled nitrification-denitrification processes (Santoro, 2010) and altering the release of ammonium and phosphate from the sediment (Cai et al., 2023). The increasing salinity caused by enhanced saltwater intrusion due to SLR will transform coastal wetland habitats, especially when intrusion rate exceeds their accretion ability (Rayne et al., 2021), changing the composition and structure of communities and promoting the shift towards communities dominated by more salt-tolerant species and the upstream shift of the existing vegetation communities. This, in turn, will trigger cascading effects on hydrodynamic conditions and water circulation (Saintilan et al., 2019). Similarly, salinity intrusion will also have effects on coastal wetlands fauna at all trophic levels, leading to changes in the spatial distribution of some taxa towards innermost wetland areas and the local extinction of others (Fujii and Raffaelli, 2008). Although these impacts will drive a reorganization of coastal wetland processes and functions, there is a consensus in the scientific literature that the consequences will be particularly negative where migration of coastal wetlands vegetation in inner areas is restricted due to physical barriers, urban infrastructures, or other land-uses (Morris et al., 2002; Raabe & Stumpf, 2016).

Global warming is also particularly concerning, as temperature represents one of the most important factors influencing organisms, their metabolism (Wendering & Nikoloski, 2023), oxygen demand, immunity, growth and mortality rates, reproduction and spawning time (Mugwanya et al., 2022). Rising water temperature caused by global warming could also enhance primary productivity. This, in turn, could promote changes in community structure and assemblages (Martinez-Megias and Rico, 2022), affecting life at all levels of hierarchical organization, from the molecular level to the metabolic and phenological ones (Wendering & Nikoloski, 2023). These changes will ultimately affect morphological and functional traits of organisms and will potentially lead to shifts in their growth patterns, resources' uptake and exploitation (Gray and Brady, 2016). Rising temperatures will also alter animals' distribution and abundance, as well as species' migration patterns, thus influencing species mortality, social and feeding behaviour, developmental and reproductive timing (Nagelkerken et al., 2023; Predragovic et al., 2023). These changes will also be most likely exacerbated by the additional effects of certain opportunistic non-indigenous and invasive species, which might benefit from warmer temperatures and newly available ecological niches also created by local extinctions, thereby potentially worsening the direct impacts of CC on local species (Pyke et al., 2008; Walther et al., 2009). Overall, with the expected temperature increase by the end of the century, some coastal wetland species are likely to undergo a poleward redistribution (Cavanaugh et al., 2013). This could be the case of the well-known phenomena of mangrove encroachment into areas historically dominated by saltmarshes in Australia, New Zealand, South Africa, Peru, China, Mexico and the USA (Cavanaugh et al., 2013; Saintilan et al., 2014). However, it remains still unclear whether and where mangrove displacement will produce "winners" or "losers". Thus, the effects of wetland shifts caused by global warming are far from clear-cut. On the one hand, spreading of mangrove encroachment could enhance the provision of ecosystem services such as carbon storage (Kellewey et al., 2016), as well as strengthen storm surge protection (Adgie and Chapman, 2021). On the other one, the loss of saltmarsh habitats may trigger cascading effects on faunal species that depend on them, weakening the strength of established trade-offs (Cavanaugh et al., 2013).

While incremental change in atmospheric and water temperatures may drive distributional shifts in coastal wetland species, the increase in the intensity, duration, and frequency of "*discrete prolonged anomalously warm water events*", commonly referred to as marine heatwaves (Hobday et al., 2016, p.229) and of heatwaves (i.e. atmospheric heatwaves as at least three consecutive days exceed a calendar day threshold defined as the 90th percentile value for temperature; Perkins and Alexander 2013), could have far more pronounced impacts. Some coastal wetlands plant species and communities often exist close to their physiological limits (Domínguez et al., 2021). Thus, these extreme events can temporarily expose them to temperatures above their critical physiological thresholds, resulting in reduced biomass assimilation and growth rates (Slot et al., 2016), increased leaf death, and, ultimately, enhanced individual mortality rates (Doughty et al., 2023). Marine heatwaves can also cause a profound alteration in the biogeochemistry of coastal wetlands' sediments, leading, for example, to a decrease of food availability and quality for the benthic fauna, but also possibly affecting wetlands' sediment's ability to store carbon (Palmas et al., 2025).

Warming temperature and heat waves can also exacerbate the alteration in precipitation patterns, leading to increased evaporation and resulting in more frequent and intense storms and flooding phenomena in some areas and to anomalous drought in others (Allen et al., 2018). The effects of these extreme weather events, with changes in their timing, frequency, intensity, and distribution, are not easy to predict. Although these events are expected to exacerbate the long-term effects of CC on coastal wetlands (Day et al., 2008), the final impacts will also largely depend on local and regional diversity in ecosystems conditions, as well as in external pressure and adaptive responses (Davies-Vollum et al., 2021). Moreover, the projected changes in precipitation patterns will undoubtedly affect coastal wetlands conditions, and consequently the species inhabiting these environments, altering the freshwater inputs, thus causing modification, for example, in wetlands water salinity and dissolved oxygen concentrations (Milly et al., 2005). Other effects of precipitation changes, as well as of flooding events, could include alterations in water residence time, in the sedimentary load delivery and nutrient inputs to the adjacent sea (Anthony et al., 2009), as well as in altered sedimentary C degradation and sequestration (Ennas et al., 2024; Ennas et al., 2025). These latter, in turn, based they are interrelated with microbial diversity (Danovaro and Pusceddu, 2007), will further provoke substantial changes in patterns and dynamics of benthic biodiversity and associated ecosystem functioning (Danovaro and Pusceddu, 2007).

2.1.1. From ecological impacts to effects on human well-being

Given the complex and high-valued socio-ecological features that characterize coastal wetlands, the consequences of CC go well beyond their ecological impacts, and can be expected to be more severe where their role in supporting national economies and human well-being is more relevant, as well as where there is a limited adaptation capacity to the new conditions (Lawrence et al., 2020; Srinivasan, 2011). CC-induced modification of ecosystem processes and functions could indeed generate cascading effects (Sarà et al., 2021), posing severe threats to their ability and contribution in providing Ecosystem Services (ESs) (Fairchild et al., 2021; Mehvar et al., 2019).

ESs can be defined as the benefits that humans derive from nature (Costanza et al., 1997), a concept that bridges biodiversity, ecosystems functioning, natural capital and human wellbeing (Paul et al., 2020). Coastal wetlands ESs include primary production, water flow regulation, sediment stabilization and soil maintenance (Sathirai and Barbier, 2001), water quality improvement, and nutrient cycling (Costanza et al., 2008). Coastal wetlands also fulfil the important service of biodiversity support, providing habitats and refugia for many species of aquatic invertebrates, fish, and insects (Calder et al., 2019), as well as breeding, resting, and feeding areas for birds (Saintilan et al., 2019). Furthermore, coastal wetlands' regulating and maintenance services contribute to human well-being in several ways, playing a fundamental role in climate regulation, due to their capacity to modulate atmospheric concentrations of greenhouse gases, particularly for those that mostly contribute to the rising global warming (IPCC, 2007). They indeed, by behaving as detritus traps (Pusceddu et al., 2003; Palmas et al., 2025), contribute to mitigating CCs effects through blue carbon sequestration and long-term carbon storage (Navarro et al., 2023). The regulating and maintenance services of wetlands also encompass hazard mitigation dampening the impact of storms by attenuating their effects and providing erosion mitigation and shoreline protection (Barbier et al., 2019; Marois & Mitsch 2015; Sun and Carson, 2020). Provisioning ESs include processes and functions mediated by human activities (*i.e.* human capital) deputed to produce "tangible" goods or benefits (Rova et al., 2019). Provisioning services can be directly extracted, consumed and, sometimes, traded (Dhyani & Dhyani 2016). In coastal wetlands these benefits include provisioning of raw materials, as well as energy from biomass and food (Newton et al., 2014; 2018). Within provisioning ESs, food production from fisheries and aquaculture activities in coastal wetlands are important contributors to human well-being considering either people's employability or well-being economic value (Naskar et al., 2017). Additionally, coastal wetlands support a wide range of human activities, for instance by supplying shelter and nursery areas for recreational species (Barbier et al., 2019), but also entailing cultural values, representing identity places with traditional recipes, local customs and traditions, as well as cultural heritage, spiritual benefits, leisure and tourism, and landscape enjoyment (Newton et al., 2014). Cultural ecosystem services provided by coastal wetlands, thus, encompass a wide range of nonmaterial benefits arising from natural ecosystems, including cultural heritage, spiritual values, and educational and knowledge systems (Elwell et al., 2020). The understanding of the propagation of the CC effects on processes, functions, and organisms to the services provided by coastal wetlands represents a critical objective to reach for the evaluation of adaptive management policies.

Generally, natural and human capital involved in the delivery of provisioning services in coastal wetlands are considered particularly vulnerable to temperature increase, SLR and changes in precipitation patterns (Melaku Canu et al., 2011; Newton et al., 2018). CC could potentially have adverse effects on provisioning ESs at different levels. Considering, for example, food provision from fisheries, CC may have different impacts at different levels of biological organization (Harley et al., 2006; Lehodey et al., 2006; Tasker 2008), affecting species stocks productivity, and altering body size and primary productivity (Rijnsdorp et al., 2009). In the case of food provisioning services from aquaculture, some imported species, often more resistant than native ones, could instead take advantage of CC driven increasing temperatures (Marti et al., 2009). Coasts erosion associated with SLR, and intensified storms can also affect the stability of coastal wetlands fishing-related infrastructures, thus negatively, though indirectly, impacting the human access to food services (Davies-Vollum et al., 2021). Additionally, the newly suitable conditions for non-indigenous species (NIS) favoured by sea warming could impact fishing activities in different ways, considering - for example - the potential negative effects of invasive NIS on fishing gears, nets, and facilities (Katsanevakis et al., 2014). Such negative effects have already been observed for several NIS, like the polychaete *Ficopomatus enigmaticus* (Fauvel, 1923) (by fouling fishing gears and equipment or reared clams), the cladoceran *Cercopagis pengoi* (by attaching to fishing gears and clogging nets and trawls) (Katsanevakis et al., 2014), the ctenophore *Mnemiopsis leiydi* (A. Agassiz, 1865; by clogging nets) (Marchessaux et al., 2020), and, most recently, by the blue crab *Callinectes sapidus* (by plundering fishery resources, Marchessaux et al., 2023; Cabiddu et al., 2025). NIS can also have direct (*e.g.* by predation) and indirect negative effects on commercially exploited autochthonous species (Pyke et al., 2008), by causing the degradation of habitats that serve as refuge or nursery grounds (Katsanevakis et al., 2014). Nevertheless, expanding NIS could become a commercially exploitable resource, thus with a potential positive economic effect on food provisioning benefits (Cavraro et al., 2022).

CC can also impact the recreational benefits of coastal wetlands, affecting timing and route of migratory birds (Ramirez et al., 2018), thus posing the risk of losing the cultural value and the revenue due to the decline of the wildlife observation market and hunting industry, as observed for inland wetlands (Ansley et al., 2023). In the same way, the expected CC impact on fish availability or distribution will consequently be reflected also in the business linked to sport-recreational fisheries (Townhill et al., 2019). CC can also impact tourism with indirect societal change impacts (Day et al., 2021). For instance, tourism demand could decrease because of the political instability caused by CC (Reinhardt and Toffel 2017) or because travel may become less socially acceptable with the increasing poverty generated by CC (Gössling et al., 2020). Notably, since tourist activities are linked to certain peculiar environmental characteristics of wetlands that, in turn, influence the users' preferences, CC induced modifications of wetlands will most likely have cascading consequences on their recreational use as well (Salpage et al., 2019). Indeed, tourism-related cultural ES could be affected in different ways by CC, since it could potentially benefit some touristic destinations but negatively impact consumers' demand for other ones (Simpson et al., 2008). In contrast, CC could also lead to positive changes in cultural ESs in some areas, including changes in tourist routes, causing significant economic consequences, for example by lengthening the tourist season and, thus, letting touristic flows to grow (Marcinkevičiūtė et al., 2021; Patrolia et al., 2017).

Nonetheless, despite their importance has been an increasingly attractive key research focus in the last 10-15 years, CC impacts on coastal wetland ESs remain poorly understood, and, nowadays, it is still not clear how these socio-ecological systems will respond to CC (Schuerch et al., 2018). With the Millennium Ecosystem Assessment (MEA) in 2005 (MEA, 2005), ESs became a well-established topic in scientific research (Seppelt et al., 2011) and significant efforts have been devoted to the study of ESs provided by coastal wetlands. Since then, several approaches for ESs assessment and evaluation have blossomed, including expert-based look-up table at land use/land cover scale (e.g.: Burkhard et al., 2009; Kienast et al., 2009), or modelling techniques ranging from qualitative models (Haines-Young et al., 2012), to spatially explicit methods such as InVEST (Sharp et al., 2020), Bayesian Belief Networks models (Landuyt et al., 2013), and Petri nets model (Fongwa et al., 2010; Rova et al., 2019).

The increasing scientific effort in the study of coastal wetlands ESs is noticeable considering the multitude of reviews published on this topic and dealing with ESs in different coastal wetland typologies (Barbier et al., 2011; Xu et al., 2020), ESs economic evaluation methodologies (Delle Grazie & Gill, 2022), ESs and disservices of mangroves and saltmarshes (Friess et al., 2020). Some work focused on specific ESs categories (Shepard et al., 2011;

Barbier, 2016) as well as on the linkages between ESs and Ecosystem-based approach in policy decision-making (White et al., 2010), on the economic value of ESs (Mehvar et al., 2018), or on specific geographical areas (Anthony et al., 2009; Engle 2011; Sierzen et al., 2012).

Despite ongoing efforts, since the ecological dynamics and processes in wetlands have not yet been fully clarified and, given the complexity and spatial-temporal heterogeneity of the drivers of change, no consensus has been reached on the future effects of CC on the benefits provided by coastal wetlands. In addition, to the best of my knowledge, no comprehensive review on the CC impacts on ESs provided by coastal wetlands exists, yet.

To provide insights on this gap of knowledge, the available scientific literature dealing with the assessment of CC impacts on coastal wetlands ESs was examined. The aims of this analysis are identifying the most recurrent variables used to assess CC impacts on coastal wetlands ESs, the main methodologies applied to study the impacts of CC on ESs, and the most important impacts of CC on ESs.

2.2. Overview of the scientific literature

A literature review was performed following the PRISMA protocol (Preferred Reporting Items for Systematic Reviews and meta-Analyses; Page et al., 2021). The Scopus and Web of Knowledge databases in December 2024 were searched for "ecosystem service*" AND (lagoon* OR saltmarsh* OR "salt marsh*" OR marsh* OR mangrove* OR “coastal wetland*” OR estuar*) AND "climat* chang*" on title, abstract and keywords sections. Only articles in English were included in the following analyses. No date restrictions were applied in the selection of studies for this review. After duplicates removal, the articles were screened for abstract and were retained when the following three requisites were all included: i) the study was carried out in the selected ecosystem types; ii) the study assessed/mapped/quantified/valued ESs; iii) the study evaluated/assessed the effects of climate change on ESs.

Book chapters, and conference materials were excluded from the analyses. Review articles were retained for discussion purposes, but no data were extracted to avoid duplicates in the results. References that did not incorporate climate change impacts but only referred to ESs importance in climate change mitigation and/or adaptation were excluded.

After the eligibility screening, the remaining articles were then examined to select only papers that reported explicitly, qualitatively, or quantitatively, the effects of CC on the ability of coastal wetlands to provide ESs.

For the included studies bibliometric information, location of the case studies, the ESs and climate changes variables considered, as well as the methodologies and data sources type used to assess the relationship between CC and ESs were extracted. The response variables were then extracted according to the following classification criteria. First, the ESs assessed in the reviewed literature were integrated and harmonised, to avoid ambiguity due to the different ESs' classification schemes, often not declared. In particular, the classification scheme for Marine and Coastal Ecosystem Services (MCES; Liquete et al., 2013) was applied using the following categories and subcategories:

- provisioning ESs: food, energy, materials, water provisioning services
- regulating and maintenance ESs: atmospheric composition and climate regulation, coastal protection and flow regulation, water purification and regulation, biological regulation, lifecycle maintenance, soil formation and composition; primary production
- cultural ESs: recreation and tourism, cognitive - intellectual and representative interactions, symbolic and aesthetic values, others.

The methodologies and tools applied to evaluate the CC impacts on ESs were categorized into qualitative, quantitative, and mixed approaches. The assessment methodologies in monetary, biophysical, socio-cultural, or mixed terms were also evaluated. The data collection methods used to assess the ESs were distinguished in primary (direct observations), secondary (official databases, expert knowledge), and mixed primary/secondary data sources.

The Sankey diagrams were produced using ggplot2 and ggalluvial R 4.3.0 packages (Wickham 2016) linking both ESs categories to the assessment methodologies and the ES subcategories to the CC related causative variables.

2.3. Results

2.3.1. Overview of the reviewed literature

A total of 1853 studies were initially collected, of which 822 were removed as duplicates or non-peer reviews. The remaining studies were screened for title and abstract following the criteria delineated above. After this screening, the full text of 202 articles was read, and the not pertinent papers were removed. After this final screening step, a total of 49 references (4 reviews and 45 original articles) were retained for the review (See Supplementary Material for a reference list).

The first study was published in 2006, followed by a steady increase in publications starting from 2016. Considering the study area of the original articles, most of the screened papers were produced in North America (USA N=17, Mexico N=2, Panama=1), followed by 7 papers reporting case studies in European countries, Asia (N=7), Oceania (4), South America (3) and Africa (N=1) (Figure 2.1). Among them, most reported case studies were presented with a local scale focus (N=31), while the remaining 18 adopted a regional or national scale. Only one paper analysed several case studies located across all the continents and covering 21 countries (Newton et al., 2018), while 2 references adopted a global perspective (van der Belt et al., 2006; Ouyang et al., 2024). Finally, one paper only used a simulated idealized system representative of shallow temperate estuaries to explore CC impact on ESs (Guyondet et al., 2022).

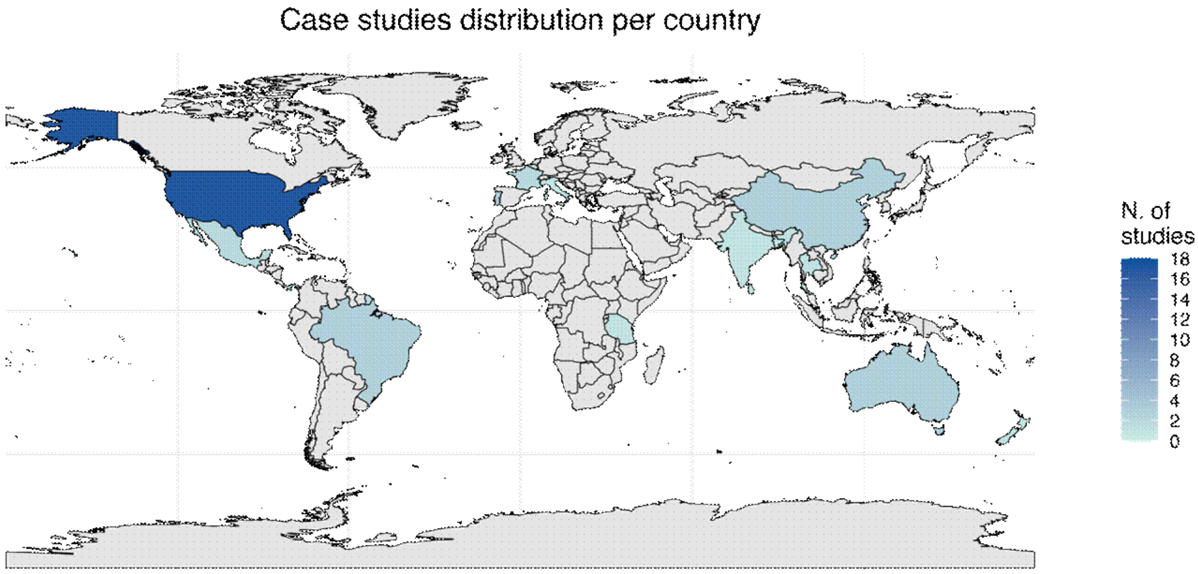

Figure 2.1 Global coverage of the reviewed literature considering single-country case studies

2.3.2. Ecosystem types; ESs and main drivers of CC

Overall, across the case studies, many studies (N=28) focused on single habitats, with the predominance of salt marshes (N=19) and mangroves (N=7), while others (N=15) examined multiple habitats simultaneously or coastal wetland habitat in general (N=1). Comparatively, coastal lagoons and estuary were the habitat receiving less attention. While most of the papers (N=42) address CC effects on ES provided by the vegetational ecosystem components, 3 references explored the impacts on ES provided by estuarine-associated bivalve fauna.

Regarding the ecosystem services addressed, three articles gave a unique value for all the considered ESs (*i.e.* Total economic value -TEV- and unique biophysical value). Among the others, most of the selected papers (N=24) assessed only one ES category at a time, while 10 focused on the assessment of two ESs categories and 8 considered all the three ESs categories. Regulating and maintenance ESs was the most assessed category (N=22 single ES category studies; N=17 multiple ES category studies), while fewer explored CC impacts on provisioning (N= 1 single ES category studies; N= 13 multiple ES category studies) and cultural ESs (N=1 single ES category studies; N=13 multiple ES category studies).

As far as the CC-related causative agents on ESs delivery are concerned, 30 out of 45 the reviewed original articles considered only one variable, while 3 papers addressed generally CC impacts and 3 the CC-driven effect of mangrove encroachment into saltmarshes. As shown in Figure 2.2, Sea Level Ris e (SLR) was the most common CC long term-related factor (N=24 one-driver studies; N=3 multiple drivers studies), followed by increasing temperatures (N=7 multiple drivers studies), change in precipitation patterns (N= 5 multiple drivers studies) and salinity (N=3 multiple drivers studies). Six references explored CC-related extreme events. Among the studies assessing future impact of CC, the majority utilized projections based on IPCC scenarios (N=19), while fewer adopted custom scenarios that were not based on official projections.

2.3.3. Models and methodologies applied to study the impacts of CC on ES

Across the articles included in the review, different quantitative and mixed qualitative-quantitative methods were applied to assess the impacts of CC on ESs. In more details, 10 studies carried out the analysis using qualitative or mixed quantitative-qualitative methodologies, while the majority (N=35) adopted quantitative methods. Among the quantitative studies, 8 original papers adopted a monetary-based assessment approach, while 16 papers assessed the ESs from a biophysical point of view, and 1 used a socio-cultural metric. The remaining 11 articles applied a mixed-based assessment approach. The articles adopting qualitative approaches use ranking/ relative importance, or descriptive/narrative approaches for ESs and CC assessment (Figure 2.3).

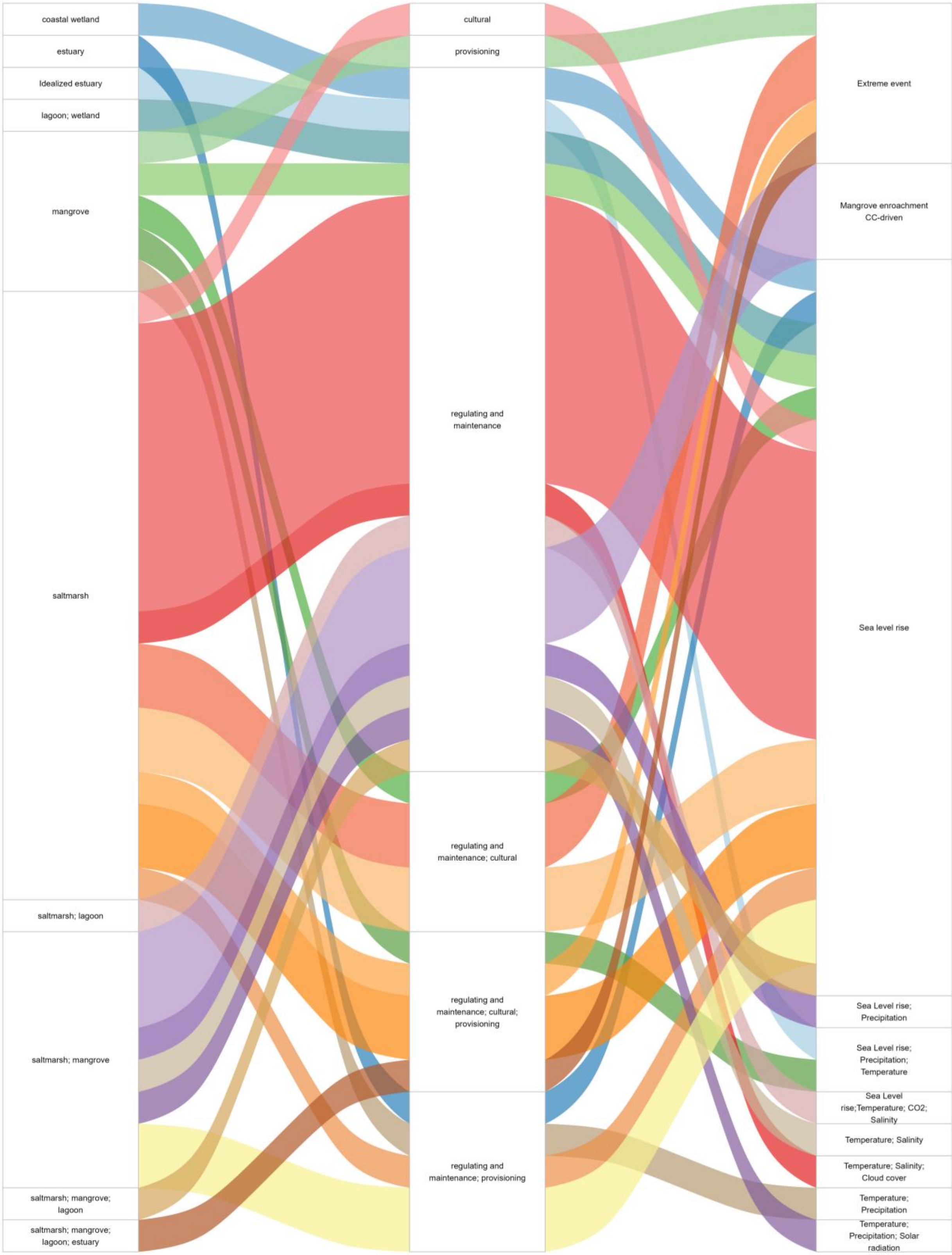

Figure 2.2 Sankey diagram showing the linkages among habitat, ES categories and CC-related drivers of change. Nodes are represented by rectangles; arcs represent flows between nodes. The width of the arcs is proportional to the flow quantity

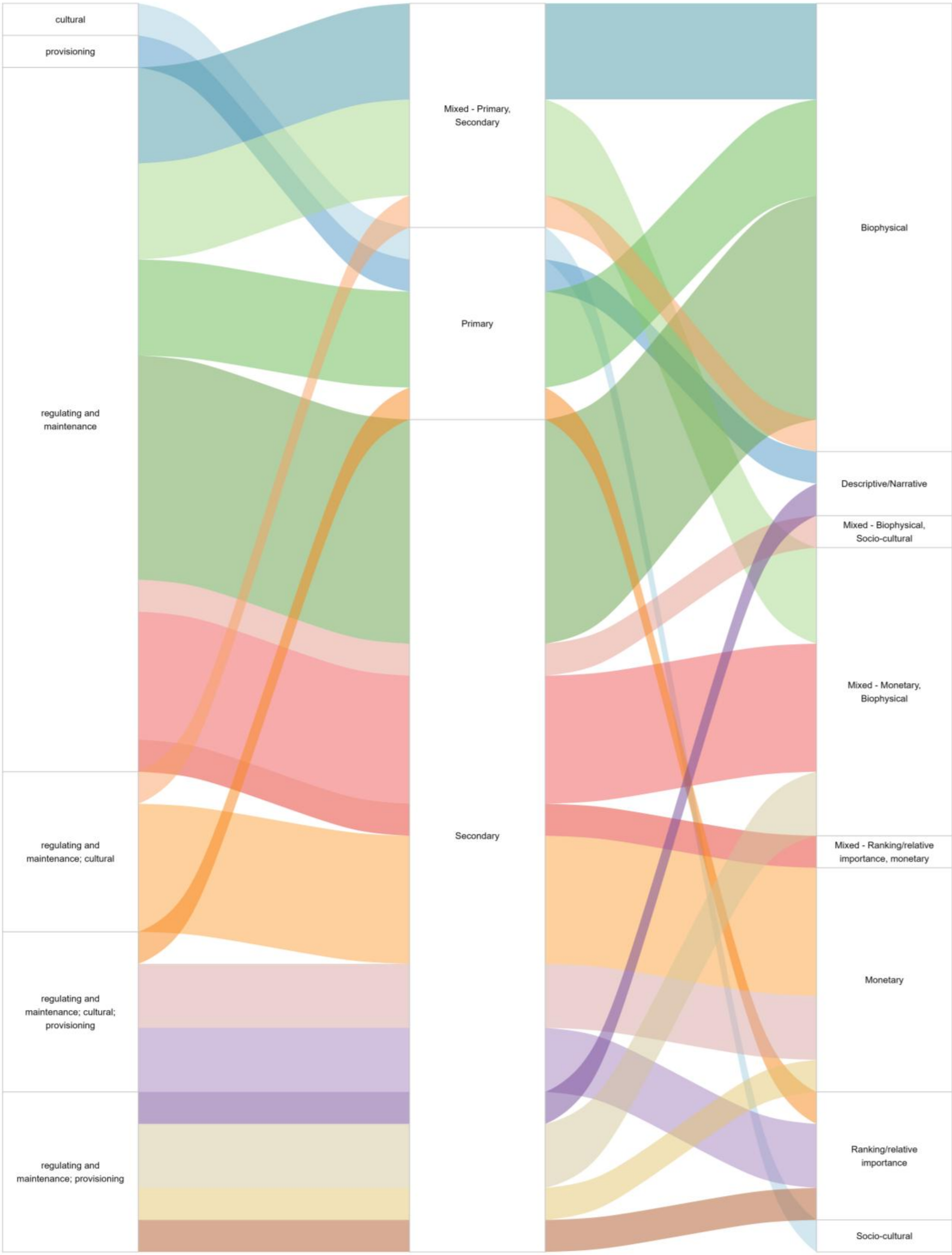

Figure 2.3 Sankey diagram showing the linkages among ES categories, data sources and methodology. Nodes are represented by rectangles; arcs represent flows between nodes. The width of the arcs is proportional to the flow quantity

Most of the reviewed papers used secondary data sources (N=28), while fewer used primary data or a combination of primary and secondary data sources. Primary data sources comprised field sampling (Macy et al., 2021), biophysical manipulative experiments (Nelson et al., 2012), interviews and questionnaires (Nyangoko et al., 2022; Debnath et al., 2024), and outcomes from focus groups or participatory workshops (van der Belt et al., 2006; Cervantes-Escobar et al., 2023). The use of secondary sources included data extracted from peer-review articles and public databases or by means of expert knowledge (Asmus et al., 2019).

Among the reviewed papers, the land-cover change assessment emerged as the predominant methodology to evaluate the impacts of SLR on coastal wetlands ESs. This approach primarily focuses on determining changes in ESs in terms of area loss or gain, which is assumed to directly, and usually linearly, influence the capacity of coastal wetlands to deliver ESs. The capacity of a specific land-use/land-cover type or habitat to provide a certain amount of an ES was often assessed using literature-based data sources, or by means of expert-based knowledge.

The land-use/land-cover (LULC) change was evaluated through several methodologies and tools, including marsh migration models (Sea Level Rise Affecting Marshes Model, SLAMM - Runting et al., 2017; Yoskowitz et al., 2017; Zhi et al., 2022; Li et al., 2023; Alemu et al., 2024, static models like the Bathub or "bucketfill" method (Fernandez-Diaz et al., 2022; Meixler et al., 2023; Cuhna et al., 2024), the Coastal Wetland Equilibrium Model (CWEM), and the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) suite, or a mixture of them (Tanner et al., 2023). Some authors adopted already developed scenarios of LULC change retrieved from available literature (Kuhfuss et al., 2016). Among these, InVEST was used to explicitly assess both the ESs delivered and CC-induced changes. Among the different InVEST models suite, the "Carbon" model (Kotagama et al., 2022; Zhi et al., 2022) and the "Habitat quality" model (Zhi et al., 2022) were used. Moritsch et al. (2022) adopted a combination of the Coastal Wetland Equilibrium Model (CWEM), a point- and cohort-based numerical model, and the MOSAICS habitat classification model, to estimate the impacts of SLR, suspended sediment, and inland habitat migration on total carbon accumulation in a macrotidal estuary in the northwest USA. Their approach, enhancing the traditional "bathub-style" approach, considered the feedback of tidal elevation, soil vertical accretion rate, plant biomass and carbon accumulation. In contrast, the "bathtub" approach used for example by Fernández-Díaz and co-authors (2022), and Meixler and co-authors (2023) for modelling the impacts of SLR on coastal wetlands habitat, assumed that elevation of sediment surfaces remained static, a constraint possibly leading to overestimated submergence of coastal ecosystems, due to the exclusion of sediment accretion (Kirwan et al., 2016).

A mechanistic growth and decomposition model was used by Anastácio et al. (2013) to evaluate the growth and mercury sequestration by *Bolboschoenus maritimus* (a flowering plant from the family Cyperaceae) under different scenarios of temperature and salinity. Similarly, Isdell et al. (2020) applied a spatial-correlative modelling approach to study the effects of SLR comparing the current and projected water filtration and nutrient removal on ESs provided by ribbed mussels. Considering the concept of environmental risk assessment, Asmus et al. (2019) evaluated the CC impacts in terms of risk to lose ESs in the Patos Lagoon (Brazil) through a predictive model that integrated stakeholders' perceptions and different IPCC scenarios. ES-based integrated risk evaluations are uncommon. Indeed, in the context of risk assessment human-related risk and ecological risk have historically been evaluated separately (Munns et al., 2016).

Considering the indirect effects of CC, such as the mangrove encroachment into saltmarshes, Doughty and coauthors (2016) explored changes in carbon stocks over a 7-year period in a representative salt marsh-mangrove ecotone to give insights on effects of its ability to store carbon. The CC-driven impacts of mangrove encroachment were also explored with reference to the saltmarsh coastal protection service, by adopting a mixture of field data and modelling techniques (Doughty et al., 2017). For the same purposes, Macy and co-authors (2021) evaluated with field sampling and manipulation experiments, the future implications for carbon and nitrogen stocks of black mangrove encroachment into *Spartina alterniflora*-dominated salt marsh along a mangrove expansion front at Port Fourchon, Louisiana (USA).

Among the papers that assessed more than one ES, the one by Rova et al. (2019) was unique in explicitly incorporating trade-offs, synergies, and other interactions among ESs. They proposed the use of Petri nets, a model built upon Socio-Ecological Systems (SES) framework (Ostrom, 2009), that incorporates CC impacts and management scenarios with socio-ecological elements involved in the provision of multiple ESs. Most of the reviewed articles indeed mainly focused on exploring ESs in isolations, and their changes in terms of area loss or gain, while ignoring ESs bundle and thus the trade-offs or synergies among them. The use of this innovative approach instead, allowed the modelling of temporal changes in

ESs interactions, thus representing a promising tool for the exploration of possible management and climate scenarios (Rova et al., 2019).

The papers assessing CC effects in a qualitative or mixed quantitative-qualitative point of view adopted different approaches, using both questionnaires, surveys as well as participatory techniques.

By integrating fishers' community perceptions and indigenous knowledge, including interviews with residents, fishers, fish sellers, and middlemen involved in the fish trade, with climate date across twelve wetlands in West Bengal (India), Debnath and co-authors (2024). developed a region-specific framework for estimating an indicator of vulnerability, considering the likelihood of negative effects on ESs as a vulnerability component (Debnath et al., 2024).

The application of the Press-Pulse Dynamics (PPD) framework to Bahía Almirante, on the Caribbean coast of Panama, reflects the growing use of social-ecological system approaches to study in deep systems interactions, resilience, and vulnerability (Collin et a., 2024) and highlighting ESs as a key linkage between the two components of these systems. Through the PPD lens, they examined Bahía Almirante's socio-ecological system, exploring its different components, interdisciplinary linkages, feedback, and causal hypotheses derived from traditional single-discipline studies (Collin et al., 2024). While these kinds of approaches have already advanced the understanding of the dynamics of socio-ecological systems, their implementation remains challenging due to multiple of disciplines required.

Similarly, recognizing the importance of active participation of communities in decision-making and conservation efforts in light of CC threat, Cervantes-Escobar and co-authors (2023) engaged in participatory workshops four communities in northwestern Mexico to understand similarities and differences in their perception regarding CC effects on ES, adopting the Metaplan methodology, a qualitative participatory technique based on the full visualization of the brainstorming process (Metaplan, 2009).

2.4. Climate change impacts on ecosystem services

In the reviewed literature, a decline in monetary value of food provision ecosystem services by coastal wetlands has been reported by some authors. For instance, Rova et al. (2019) predicted a negative variation for provisioning ESs under a Business-As-Usual scenario in the

Venice Lagoon. In contrast, assuming proportionality between the variation in habitat areas and ESs delivery, Kuhfuss et al. (2016) estimated for the Languedoc-Roussillon region (France), a net positive change in the supply of provisioning services in response to 1 meter of SLR. This was especially true under the adaptive management scenario of "strategic retreat of infrastructure and buildings", in which the retreat of wetlands is unconstrained (Kuhfuss et al., 2016).

The projected impacts identified from my literature review, highlight that regulating and maintenance ESs are likely to be altered in different ways from CC.

On a national scale, Young et al. (2021) highlighted that 38% of mangrove/tidal marsh Australian ecosystems will experience a decrease in soil carbon stocks suitability, used as an indicator for mangrove/tidal marsh carbon stock. The predicted declines are attributed to declining precipitations, due to the positive relationship between rainfall and SOC stocks, but also to warming temperatures, and decline in elevation, used as a proxy for SLR (Young et al., 2021). Their findings highlight the importance of ecosystems landward migration, as the key for the maintenance of blue carbon stocks in the future, especially in areas where sea level rise is the main driver of their loss.

These results are corroborated at nested spatial scales from the analysis presented from Tanner and co-authors (2023). By comparing CC impacts in two different locations, they demonstrate that, depending on geographical location, 1-meter SLR can lead to increases or decreases in marsh extent in the future and can result in either increases or decreases in net carbon sequestration in a given area, largely influenced by the availability of upland areas for marsh migration. Even more importantly, they also show that, even when carbon sequestration increases locally, the net present value of this ecosystem service will further decline under worse SLR scenarios. This discrepancy among biophysical and monetary values can be explained by the temporal distribution of ES benefits, since the net present value discounts future carbon benefits. In the "winner" areas, much of the new marsh area -and thus the derived carbon sequestration- will emerge only during the last decades of the current century, following earlier erosion-driven marsh losses. Although these newly created marshes will sequester significant amounts of carbon, their benefits are realized far into the future and thus discounted more heavily in net present value terms (Tanner et al., 2023).

When shifting at the landscape level, Meixler et al. (2023), estimating total present-day carbon sequestration in Jamaica Bay urban-coastal ecosystem, predicted a SLR-induced wetland area loss from 17% up to 63%, based on the 10^{th} to 90^{th} percentile projection, respectively. These findings are coupled with the predictions of a consequent decrease of above- and below-ground carbon storage from 15% to 51% (Meixler et al. 2023).

Kotagama et al., 2022, by means of the InVEST tool to model blue carbon dynamics, predicted that, with the rising of the sea level, carbon stocks could decrease with increasing extents of inundation in the Chilaw Lagoon (Sri Lanka), ultimately fostering by 2150 declines of the total carbon sequestration capacity, the monetary value of carbon sequestration, and carbon accumulation. At the same time, carbon emissions were also predicted to rise as the level of disturbance caused by SLR will increase (Kotagama et al., 2022).

The future impacts of SLR are also expected to increase lagoon surface in France, and along the increase the volume of water, it was estimated an increased capacity to dilute N and P, prospectically providing positive effects (Herivaux et al., 2018).

Zooming on the species level, Isdell and coauthors (2020), by investigating the consequences of different SLR scenarios in the Chesapeake Bay, estimated a 50% reduction of filtration and nitrogen regulating services provided by the Atlantic ribbed mussel (*Geukensia demissa*). Similarly, a reduction of mercury sequestration service related to increased salinity conditions was found for the salt marsh halophyte *Bolboschoenus maritimus* in Portugal (Anastacio et al., 2013).

When focusing on different spatial scales, Duarte and coauthors (2021) found that most coastal systems of Portugal are expected to experience declines in regulating services such as blue carbon, nitrogen, and phosphorus storage, mainly due to sea level rise and osmotic stress. An exception was observed in one of the ecosystems studied, which showed relative resilience due to its dominance of halophyte species adapted to submersion and submerged photosynthesis. Moreover, the study highlights how rising atmospheric CO_2 could enhance primary productivity in C3 halophytes, though this does not always lead to proportional increases in ES value. Interestingly, while total ES values may increase at broader spatial scales due to an overall vegetation expansion, value per unit area often decreases. Notably, while vegetation expansion might increase total ecosystem service values at broader spatial scales, the value per unit area often tends to decrease due to reduced local productivity or spatial dilution of benefits.

Considering studies at national scale but across different ecosystem types, uncertainties regarding future impacts on carbon dynamics arise. Although Young (2021) projected a decline in carbon storage within mangroves/tidal marshes, the overall results of the study are overturned when also considering another ecosystem they considered - seagrass meadows - which falls outside the scope of this review. The majority of seagrass meadows areas are indeed predicted to have increased soil carbon stocks suitability in the future (Young et al., 2021). Moreover, while considering salt marshes and mangroves together can revealed overall negative impacts on the carbon sequestration ES, distinguishing the impacts between the two habitats unveiled two different trajectories, with the potential area suitable for mangrove carbon transferred to the greatest extent, showing a trend of northward expansion (Chen et al., 2025).

Mangrove expansion was also explored as an indirect effect of CC since it is driven by the declining frequency of severe freeze events and by the increasing warming climate (Osland et al., 2013). Mangrove encroachment into saltmarshes in a Louisiana coastal wetland can in future lead to increased aboveground nitrogen and carbon sequestration, resulting in larger stocks in aboveground compartments (Macy et al., 2021). Soil stocks indeed, which represent the most stable and long-term pools due to soil burial, were found to not differ between marsh and mangrove habitats (Macy et al., 2021). Similarly, marsh conversion to mangrove over a 7-year period has been estimated to increase overall carbon storage by 22–26% also in Florida, a result attributed to aboveground components and not influenced by belowground (soil) stocks. (Doughty et al., 2016).

When considering the coastal protection service provided by saltmarshes, Doughty and co-authors (2017) showed potentially positive impacts - with 470% of erosion prevention more than salt marshes habitat - deriving from their conversion into mangroves due to CC.

Overall, the studies included in this review, agree on the fact that mangroves conversion may not be entirely negative with respect to the potential ESs delivery, and could make coastal wetlands more resilient to the effects of future CC. However, it is important to highlight that the full range of consequences of mangrove range expansion on socio-ecological ecosystems has yet to be fully understood (Doughty et al., 2017).

Among the reviewed literature, little emerged about the possible effects of CC on cultural ESs on coastal wetlands. The scarce number of outputs related to the impact of CC on cultural ESs in coastal wetlands can be attributed to several factors such as data availability, weak or missing interdisciplinary cooperation or community engagement, and several methodological challenges (Chan et al., 2012). However, some key aspects emerged from the case study presented by Rova et al., (2019) in which, in all the Business-As-Usual scenarios, CC are expected to produce a net positive effect on tourism-mediated cultural ecosystem services, as the number of visitors to the lagoon under scrutiny could rise due to more favourable conditions for tourists. Moreover, a general trade-off has been observed between tourism-mediated ecosystem services and all other ESs, which are instead characterized by a general declining trend (Rova et al., 2019).

To date, challenges around defining and measuring cultural Ess remains (Bennet et al., 2009) and, thus, is not surprising that understanding the consequences of CC on cultural ESs is a current key challenge, made difficult by the multitude of disciplines required to understand the underlying mechanisms. Indeed, when dealing with cultural ESs, problems related with their assessment, such as problems with value judgments, continue to persist, ultimately leading these services to be considered among the most difficult to be characterized and valued (Chan et al., 2012; Bennet et al., 2009).

2.5. Limits, gaps of knowledge and future perspectives

In this chapter I report the results of a literature review aimed at highlighting the progress made in studying the impacts of CC on ESs provided by coastal wetlands. As CC continues to pose challenges to biodiversity and ecosystem functioning, thus impacting the delivery of benefits that coastal wetlands provide, the comprehension of the complex relationships between climate and ESs remains critical to sustain and foster the humans' well-being (Bakure et al., 2022; Wedding et al., 2022). The fundamental role of ESs has been widely recognized at different policy levels, and the ESs concept has progressively entered political agendas worldwide (Bouwma et al., 2018). This is well documented in several EU documents, such as the EU Biodiversity Strategy 2030 (European Commission, 2021), but also in the EU Marine Spatial Planning Directive (MSPD; 2014/89/EU), and in the Integrated Coastal Zone Management (ICZM). The importance of ESs integration in decision-making processes is also highlighted by the linkage between the traditional System of National Accounts (SNA) with the central framework of ecosystem accounts (SEEA CF and SEEA EA; United Nations, 2009). Indeed, ecosystem accounting is now emerging as a powerful tool for integrating ecological and economic information, providing a systematic framework to assess the contributions of ecosystems to human well-being and sustainable development (United Nations et al., 2021).

The findings of this chapter confirm the growing interest of the scientific community towards this topic, reflecting a broader trend that can be observed for other ecosystem types (e.g. see ESs of high mountain areas; Palomo et al., 2017 and Buonocore et al., 2021 exploring trend and evolution in marine ESs concept). The analysis revealed both the progress made and the limitations and gaps in the current scientific knowledge. Indeed, despite a growing research attention, the response of coastal wetlands ESs to climate change threats, has yet to be fully understood (Ostrowski et al., 2021). Overall, from the piece of literature analysed, little emerges about the drivers that determine the vulnerability of these ecosystems, partly because most studies rely on the baseline assumption that ESs supply is directly proportional to area loss or gain and used merely static approaches, without fully considering the complexity of the underlying ecological processes.

Understanding cascading effects remains particularly challenging, as it requires accounting for the structural and functional complexity of coastal wetlands (Li, et al., 2018). This complexity is further compounded by the practical difficulties of conducting field studies or experiments in these peculiar environments, which are often time-consuming, logistically demanding, and expensive. As a result, empirical data are frequently limited, leading researchers to increasingly rely on predictive models to anticipate ecological responses (Sequeira et al., 2018) and the effects of CC on ESs provision. However, such models are often constrained by both data deficiencies and limited transferability across different spatial and temporal contexts (Yates et al., 2018), further hindering our ability to capture the full range of cascading impacts. Such complexities may explain the paucity of references that privilege the use of primary data and is also reflected in the relatively low number of studies reviewed that relied on value transfer approaches to assess both biophysical and monetary values of ESs.

Although some progress has been achieved, there is still a significant lack in the understanding of ESs demands and trade-offs, as well as the beneficiaries who rely on and benefit from them, and the feed-backs between ecological and socio-economic subsystems (Schoubroeck et al., 2024). Most of the reviewed studies gave the assessed ESs values without explicitly linking them to human-wellbeing, thereby neglecting the social benefits that derive from coastal wetlands ecosystems. This in turn, causes a lack in the assessment, quantification and understanding of the impacts at broader spatial scales - regardless of whether they are negative or positive.

The results of this review also show that there is an uneven distribution of studies throughout the world, with most of the reviewed literature focused on case studies from the USA and European countries, suggesting that more efforts are necessary in other areas, such as in developing countries, which are likely to experience the most severe and disproportionate social consequences caused by CC impacts on ESs (Kanan & Giupponi 2024; Srinivanas, 2011). The limited amount of research that analysed multiple CC drivers also underlines the urgent need to explore the effects of multi-variable interactions. Indeed, the evaluation of single impacts can be informative but does not provide a complete picture, especially if not included in the socio-economic context, which, together with CC, drives ESs supply and demand (Runting et al., 2016). Moreover, the predominance of SLR-focused studies, most often carried out adopting a static "habitat loss" perspective, spotlights the lack of integration with other macro climatic drivers (e.g. changes in precipitation patterns, warming) which could have important effects on coastal wetlands ESs (see Osland et al., 2015). Considering ESs typologies, the regulating and maintenance category was the most represented, while provisioning and cultural ESs received, comparatively, less attention. Such imbalance may not necessarily reflect less interest by the scientific community, but it could be attributed either to the literature search string used here (e.g. for the provisioning ESs) or the lack of transdisciplinary research for the characterization of some services, notably the cultural ones (Chan et al., 2012). However, the tendency to evaluate regulating and maintenance services at the expenses of the other categories was also highlighted in previous reviews on ESs (Egoh et al., 2012; Crossman et al., 2012; Haase et al., 2014; Englund et al., 2017; Campagne et al., 2020).

It is also noteworthy that there are still no consistent ESs classification and assessment in the reviewed case studies, thus potentially leading to a limited comparability of the results (Newton et al., 2018). Indeed, the different existing approaches for ESs classification, conceptualization (e.g. service; benefit; good concepts) and assessment are nowadays debated, because of their own strengths and weaknesses, such as ESs double counting (Newton et al., 2018). In this regard, I highlighted the need to harmonize ESs assessment methodologies among and inside the different research fields, along with ESs indicators based on clear socio-ecological assumptions, but also capable of being informative, measurable, and relevant for both researchers and society. Since an assessment time-quick and based on key information strata is more acceptable (Knight et al., 2006), these evaluation methods, would be particularly useful in land-use decision making for planners and practitioners, usually with limited resources in both funding and time (Egoh et al., 2007; Knight et al., 2006). Another important aspect, therefore, concerns making assessments of ESs on a local scale rather than employing ESs indicators from secondary sources or case studies conducted in other contexts or at different spatial scales (Runting et al., 2016). Indeed, when assessing the biophysical or monetary value of ESs with a benefit transfer approach, the outputs can be prone to enormous bias and inaccuracy due to the lack of transferability between study areas (Plummer 2009).

Finally, it is advisable to consider a trans-disciplinary perspective, to address the challenge of deeply understanding the linkages between ecosystem function, processes and ecosystem benefits that derive, to effectively manage the socio-ecological implications and consequences of CC on coastal wetlands. In fact, given that for a sustainable ESs management it is necessary to be able to distinguish the capacity, the flow, and the demand (Rova et al., 2019), a trans-disciplinary approach will be crucial in disentangling the different facets of coastal wetlands ESs. This, in turn, will better allow the integration of ESs knowledge and CC vulnerability in coastal wetlands management, thus moving beyond a scientific research focus toward an effective practice in decision-making.

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Chapter 3. Climate change ecological vulnerability assessment of food provisioning ecosystem services

3.1. Introduction

Coastal wetlands, including coastal lagoons, estuarine systems, mangroves, saltmarshes, represents peculiar transitional ecosystems that contributes to provide a plethora of Ecosystem Services (ES), goods and benefits to humanity (Costanza et al., 1997; MEA, 2005; Franco et al., 2006; Costanza et al., 2014; Barbier et al., 2019). However, the capacity of coastal wetlands to continue providing ESs under changing environmental conditions is increasingly under threat (Mehvar et al., 2019; Sarà et al., 2021). Climate change (CC) is indeed emerging as one of the major drivers of ecological and socio-economic transformation in these peculiar socio-ecological systems (Schuerch et al., 2018).

Climate change is affecting coastal ecosystems worldwide through a wide range of impacts, including rising air and water temperatures, increasing frequency and intensity of extreme weather events, shifts in salinity regimes due to altered precipitation patterns (IPCC, 2022). All these stressors can have significant impacts on the structure and functioning of coastal wetland ecosystems and the services they provide, such as nutrient cycling, carbon storage and sequestration, habitat provisioning, coastal protection (Palmas et al., 2025; Guild et al., 2024). Among the goods and benefits that humans derive from these ecosystems, food from wild aquatic animals holds particular importance for the subsistence and economies of many local communities (Benè et al., 2016; FAO, 2015; 2018; Kolding et al., 2014).

This is especially true in the Mediterranean region, where traditional small-scale fisheries located in coastal lagoons have so long played a fundamental role in ensuring nutrition, food security and sustainable coastal livelihoods (Malorgio et al., 2017; Pérez-Ruzafa et al., 2012). These fisheries target both resident and migratory fish species and are often organized through cooperatives and or consortia that maintain traditional knowledge and practices, while at the same time lower some of the costs related to fishing activities (Madau et al., 2018).

Traditional management activities have certainly contributed over time to preserve these peculiar ecosystems, although much of the coastal lagoons have progressively degraded and declined due to land reclamation, conflict of uses and the increasing effects of CC (Cataudella et al., 2015; Katselis et al. 2003). The potential impacts of CC on these socio-ecological systems are indeed of particularly interest, and their vulnerability has become increasingly evident especially after the COVID-19 pandemic (Islam et al., 2022).

In coastal marine CC-related research, the IPCC vulnerability assessment frameworks have played a key role in assessing the climate vulnerability of fish species, ecosystems, fishing and local communities at different scales. The initial frameworks, introduced in the IPCC's Third (TAR) and Fourth Assessment Reports (AR4), defined vulnerability as "*the degree to which a system is susceptible to, and unable to cope with, adverse effects of climate change, including climate variability and extremes*". According to this conceptualization, vulnerability depends on system's exposure, sensitivity, and adaptive capacity (IPCC, 2001; 2007).

The Special Report on Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation (SREX; IPCC, 2012) and the Fifth Assessment Report (AR5) in 2014 introduced a shift toward a risk-based approach (Estoque et al., 2023), defining risk as a function of hazard, exposure, and vulnerability (IPCC, 2014). Hereinafter, vulnerability is thus considered as an intrinsic property of a system, independent of its exposure to climate hazards, which now refers to exposed elements (e.g., people, assets, species or ecosystems) (IPCC 2012, 2014).

As a result, current assessments tend to focus on indicators of sensitivity and adaptive capacity, although distinguishing between them remains challenging, since many indicators contribute to both (Morrison et al., 2015; Pecl et al., 2014). Nevertheless, this conceptual shift has generated debate, as many studies have not yet adopted the revised concepts, and continue to rely on the earlier TAR/AR4 frameworks, due to its clearer operationalization, particularly regarding how exposure should be treated in standalone vulnerability analyses (Estoque et al., 2023; Ishtiaque et al., 2022).

However, despite growing interest toward risk and vulnerability assessments, significant challenges remain in evaluating the climate-related vulnerability of coastal wetland socio-ecological systems. These assessments are of paramount importance to generate the knowledge required for effective management and decision-making interventions (Gitay et al., 2011).

To date, one of the main limitations include the lack of spatially and temporally resolved data at appropriate scales, as long-term time series of high-resolution environmental data are often missing but essential to assess current ecosystem conditions, detect trends, and understand natural annual and inter-annual variability in ecosystem processes and structure (Brown et al., 2021; Gitay et al., 2011). Such data are also critical for identifying site- and species-specific ecological thresholds and plasticity, which serve as a basis for understanding potential changes in the functional characteristics of the system (Brown et al., 2019).

In addition, the development of appropriate metrics to assess vulnerability to multiple stressors remains limited, as does the availability of data on wetland sensitivity and adaptive capacity (Brown et al., 2021). Coastal wetlands are transitional ecosystems that span both terrestrial and marine habitats, but most indicators are not designed to cross this realm. As a result, significant gaps of knowledge exist at the land-sea interface, where integrated metrics are lacking to adequately capture the dynamics of this transitional zone (Brown et al., 2021). These gaps certainly still hinder the ability to anticipate how these fragile ecosystems and the benefits we derive will respond to ongoing and future environmental change.

At the European level, the EU Strategy on Climate Change Adaptation (2021) emphasizes the urgency of accelerating adaptation across all sectors and governance levels, with particular attention to local actions as the foundation of climate resilience. In this context, several member states, including Italy, have developed a National Strategy (SNAC; MASE, 2015) and adopted its National Adaptation Plan (PNACC, 2024), which provides a legal and strategic framework for identifying vulnerabilities and implementing adaptation measures at multiple scales.

In line with this national and EU visions, the Sardinia Region adopted its Regional Strategy for Climate Change Adaptation (SRACC) in 2019 (enforced in 2024), formally recognizing the need to integrate climate resilience into regional and local policies. Among the priority sectors implemented in the last release of the SRACC coastal and transitional environments, which include coastal lagoons of high ecological, cultural, and economic value, have received a specific attention. The SRACC provides the policy framework substrate for developing targeted adaptation actions to safeguard the ecological integrity and socio-economic benefits of these ecosystems.

In Sardinia, where numerous coastal lagoons are designated as protected areas under national, EU and/or international legislation, understanding the vulnerability of food provisioning services from small-scale fisheries is crucial for guiding sustainable management and climate adaptation strategies. However, despite growing political commitment, integrated vulnerability assessments for coastal wetlands remain scarce, and spatially explicit analyses are still lacking.

This study addresses these gaps of knowledge by assessing the climate change vulnerability of food provisioning by Sardinian coastal lagoons. It provides an update and comprehensive analysis based on the IPCC climate risk assessment framework, with the aims to: (i) adapt existing risk framework to coastal lagoons, focusing on climate risk for food provisioning ESs; (ii) identify key ecological drivers of vulnerability; (iii) compare relative vulnerability and risk among coastal lagoons; and (iv) provide evidence-based recommendations to support adaptive management of food provisioning services in Mediterranean coastal wetlands.

3.2. Methods

3.2.1. Study Area

Sardinia, the second-largest island in the Mediterranean Basin, has a surface of 24,090 km^2. A recent inventory identified 2501 wetlands, both inland and coastal, covering a total surface area of 494.2 km^2 (Fois et al., 2021). Among these, 16 wetlands (covering approximately 92 km^2), are designated Ramsar sites, and 116 are included in the Natura 2000 network. Additionally, coastal lagoons are of particular importance for small-scale fisheries. These systems are generally managed by local cooperatives or consortia employing traditional practices. Considering data availability, such as the level of spatial and temporal detail of catch data and environmental data, this study encompassed a total of 15 coastal lagoons (Table 3.1) along the Sardinian coastline.

Figure 3.1 Maps and location of the studied lagoons. a=Calich; b=Is Benas, Cabras, Santa Giusta and S'Ena Arrubia; c=Pauli Biancu; d=Porto Pino; e=Nora; f= Santa Gilla; g=Colostrai; Feraxi; Peschiera San Giovanni and Sa Praia; h=Tortolì; i=Avalè Su Petrosu.

Table 3.1 Main characteristics of studied sites.

Site	Ramsar	Natura 2000	Area
Avalè Su Petrosu	-	ITB020013	60
Cabras	IT021	ITB030036 - ITB034008	2230
Calich	-	ITB013044	92
Colostrai	-	ITB040019 - ITB043025	137
Feraxi	-	ITB040019 - ITB043025	60
Is Benas	IT035	ITB030035 - ITB034007	100
Su Stangioni Sant'Efisio (Nora)	-	ITB042216 - ITB034004	30
Pauli Biancu Turri	IT022	ITB030032 - ITB034004	12
Peschiera San Giovanni	-	ITB040018	110
Porto Pino	-	ITB040025	500
Sa Praia	-	ITB040018	86
Santa Gilla	IT018	ITB040023 - ITB044003	1800
Santa Giusta	-	ITB030037/ITB030033 - ITB34005	870
S'Ena Arrubia	IT016	ITB030016 - ITB034001	187
Tortolì	-	-	289

3.2.2. Climate vulnerability assessment framework

Based on the last IPCC climate risk assessment framework, I explored the vulnerability component tailored to the specificities of Sardinian coastal lagoons and to the availability of relevant data. The analysis focused on small-scale fisheries operating in these wetlands, which are generally underrepresented in climate risk studies. The assessment was informed by a literature review, data from official and online sources, and expert elicitation.

Climate vulnerability was assessed using specific indicators designed to measure the ecological vulnerability of the food provisioning ecosystem services provided by coastal lagoons. Two main domains were evaluated: (i) species vulnerability, representing the sensitivity and adaptive capacity of species to climate change, and (ii) habitat vulnerability, reflecting the susceptibility of the habitat compartment supporting the services. The assessment focused on climate-induced air temperature changes hazard.

Species vulnerability was evaluated using a combination of species-specific biological traits and site-level metrics, emphasising thermal preferences, habitat specificity, lifespan, and stock status. The approach followed those by Engelhard et al., (2024), Payne et al., (2021), Pinnegar et al. (2019), and Hare et al. (2016), and was adapted to the characteristics of the selected case studies and the available data. Data were compiled for 4 key fish and shellfish species caught in Sardinian lagoons, representing more than 80% of the total landed biomass. The following metrics were assessed:

Temperature specificity: based on the species' temperature range, espressed as the difference between TP10 and TP90; species with narrow ranges were considered more vulnerable (Pinnegar et al., 2019; Rijnsdorp et al. 2009); data were retrieved from FishBase (Froese & Pauly; 2023) SeaLifeBase (Palomares and Pauly, 2019; last access 05/2025).

Habitat specificity and mobility: derived from information on vertical and horizontal habitat preferences and mobility; species with high habitat specificity and low mobility were considered most vulnerable (Rijnsdorp et al., 2009); data were obtained from FishBase (Froese & Pauly; 2023) and SeaLifeBase (Palomares and Pauly, 2019; last access 05/2025).

Lifespan: used as a proxy for resilience, with shorter lifespans indicating higher adaptability; lifespan data were analysed alongside growth and trophic traits (Cheung et al., 2005; Hare et al., 2016; Jones & Cheung, 2018; Cheung et al., 2018; Payne et al., 2021); data were obtained from FishBase (Froese & Pauly; 2023)and SeaLifeBase (Palomares and Pauly, 2019; last access 05/2025).

Thermal Safety Margin (TSM): defined as the difference between a species' maximum preferred temperature (90th percentile of the full temperature range; TP90; Pinnegar et al. 2019) and the mean temperature recorded over the past 20 years for each lagoon; species thermal preference data were obtained from AquaMaps (last access 05/2025; Kaschner et al. 2016); a higher TSM indicates greater thermal tolerance; species site-specific TSM vulnerability scores were expressed as percentile ranks. The 20-years (2002-2021) mean temperature in each lagoon was calculated from environmental data systematized in the framework of the AZA (Increase of aquaculture site potential and Preparation of the regional plan for areas allocated for marine aquaculture and aquaculture in inland waters) project, funded by the European Maritime and Fisheries Fund (EMFF).

Stock Status (CMSY++): overfishing has been indicated as the first non-CC related driver increasing vulnerability for fishery (Peck and Pinnegar, 2018; Das et al., 2020); stock assessments were carried out using the CMSY++ model using catch -only data (Froese et al., 2023). Fishery-dependent catch data were obtained from the Fishery Service, Department of Agriculture of the Autonomous Region of Sardinia (for more details on CMSY++ methodology see Appendix 2). Stock status was defined according to the B/BMSY and F/FMSY of the last year of a time series (last F/Fmsy and last B/Bmsy) according to Table A2.2; Zhai et al., 2020). None of the scrutinized stocks felled within the category "Healthy stock size about to be depleted". Stock status vulnerability was then classified on a 0-1 scale, ranging from healthy to severely depleted, according to the following rule:

Table 3.2 Stock status vulnerability scores based on stock status category assigned from CMSY++ model outputs.

Stock status	Vulnerability
Healthy	0
Recovering	0.33
Fully overfished/outside of safe biological limits	0.67
Severely depleted	1

Species scores were calculated at two levels and adopting the following weights (w):

- The biological-traits score, computed as the weighted mean of lifespan (w=0.25), habitat specificity (w=0.50), and temperature specificity (w=0.25)

- The site-level score, computed for each species and site as the unweighted mean of TSM and stock status.

The species vulnerability metric was the weighted mean of species biological traits-level (w=0.33) and site-level (w=0.67) scores.

To accounting for the contribution of each species to the actual food provisioning ES flow in each site, fishery-dependent catch data were used to calculate the relative species importance. The final overall species vulnerability score per site was then calculated by weighting the species vulnerability scores by their relative contribution to total catch biomass.

Habitat vulnerability at the site level was assessed using indicators reflecting the ecosystem's capacity to maintaining its functioning and continuous ability to providing services under climate change. The following metrics were considered; (i) the Elements of Biological Quality (EQB), as defined by the EU Water Framework Directive (2000/60/EC), (ii) the presence and impact of invasive alien species (IAS), which cause ecological disruption and affect fisheries; and (iii) a water exchange capacity index, used as a proxy of hydrological connectivity.

Ecological status data were taken from the Review and Update of the River Basin Management Plan for the Sardinia River Basin District (Third Planning Cycle, 2021–2027). The ecological status classification is based on the assessment of Biological Quality Elements (BQEs): macrophytes (macroalgae and angiosperms - MAQI), and benthic macroinvertebrates (M-AMBI). Quality status as summarized into five quality classes (high, good, sufficient, poor, and bad) were scored in vulnerability terms. However, since no site were classified as having bad status, the following scoring was considered: 0 for high status, 0.33 for good status, 0.67 for sufficient and 1 for poor status.

Hydrological connectivity describes water movement and the transport of matter, energy, and organisms (Olds et al., 2016; Podda and Porporato 2023; Pringle, 2003), encompassing both structural and functional aspects (Bracken et al., 2013). Reduced connectivity can limit water renewal, increasing the system's susceptibility to climate-induced stressors such as warming or heatwaves. Due to the high time and financial costs associated with detailed hydrological assessments, a water exchange capacity indicator was used as a proxy, as available data did not allow a more comprehensive evaluation (Brown et al., 2019; Podda and Porporato, 2023). This indicator was calculated as the average of three sub-indicators reflecting lagoon isolation due to sedimentation or obstruction: (i) silting of the sea inlet (bocca a mare), (ii) silting of the main channel, and (iii) silting within the lagoon. Each sub-indicator was scored on a semi-quantitative scale: 0 for low vulnerability (no sedimentation or obstruction), 0.5 for medium vulnerability (partial sedimentation), and 1 for high vulnerability (severe sedimentation/full obstruction).

To incorporate the contribution of biological invasions to lagoon vulnerability, the presence of three invasive allochthonous species (IAS) was assessed: the Atlantic blue crab *Callinectes sapidus* Rathbun, 1896 (Brachyura: Portunidae), the polychaete *Ficopomatus enigmaticus* (Fauvel 1923), and the ctenophore *Mnemiopsis leidyi* A. Agassiz, 1865. For each species, presence was scored using a semi-quantitative scale: 1 for stable presence and reported impacts, 0.5 for low or occasional presence and low impacts, and 0 for absence. The overall score of vulnerability to IAS (IAS score) was calculated as the mean value of vulnerabilities to the four scrutinized species.

The food provisioning vulnerability score for each site was computed by averaging the overall species vulnerability and habitat vulnerability scores. In addition to this score, the food provisioning vulnerability "central" scenario", vulnerability was also calculated under "optimistic" and "pessimistic" stock assessment scenarios, defined by using the upper and lower confidence levels of *F/FMSY* and *B/BMSY* (ucl and lcl). Notably the optimistic scenario was defined by combining the upper B/BMSY and lower F/FMSY, while the pessimistic by combining lower B/BMSY and the upper F/FMSY.

The use of stock status as a site-specific metric has some operational and interpretational limitations, as data on stock status are often not available, particularly in data poor contexts such as coastal lagoons. In this study, indeed, several sites had to be excluded from the analysis because stock assessments could not be performed, either due to limited or missing time series or to data series available only at fishing concession level and not at the site level. To evaluate the effect of including or excluding this metric on ecosystem service vulnerability rankings, a leave-one-out analysis (Engelhard et al., 2024; Ramos et al., 2022) was then conducted, comparing food provisioning vulnerability for each coastal lagoon calculated with (central, optimistic and pessimistic scenarios) and without the stock status metric ("no stock" scenario).

3.3. Results

3.3.1. Species vulnerability

3.3.1.1. Biological traits-level

The temperature specificity, defined as the range between TP10 and TP90, was low (>12°C) for *M. cephalus*, indicating a broad thermal tolerance and suggesting that this species is relatively robust to temperature variations. The other three species showed medium temperature specificity (range between 8 and 12 °C; Table 3.3), implying a moderate sensitivity to temperature changes.

As far as the habitat specificity is concerned, *M. cephalus* and *S. aurata* were classified in the medium-high vulnerability category (score = 0.67), reflecting their partial dependence on coastal lagoon habitats during specific life stages. The bivalve *R. decussatus* exhibited the highest habitat specificity score (1.0), consistent with its sedentary and benthic adult life and its strong reliance on lagoon habitats. Conversely, *D. labrax* showed the lowest habitat-related vulnerability, as its broader horizontal mobility and capacity to exploit different habitats reduce its dependence on coastal lagoons (Table 3.3).

Table 3.3 Summary of the metrics used for vulnerability at biological-trait level

Species	TP10	TP90	T specificity	Habitat score	Lifespan
Mugil cephalus	15.81	27.83	12.02	0.67	16
Dicentrarchus labrax	10.19	20	9.81	0.33	14.3
Sparus aurata	12.72	21.44	8.72	0.67	11
Ruditapes decussatus	10.72	20.46	9.74	1	8

3.3.1.2. Site level

Thermal safety margin values (i.e. the raw difference between a species' upper thermal tolerance limit and the medium temperatures) ranged from a minimum of 0.01 °C for European seabass in the Nora lagoon, up to 9.85 °C for the flathead grey mullet in the Feraxi lagoon (Table A4.1).

The Thermal Safety Margin (TSM) score, representing the normalized difference between a species' upper thermal tolerance limit and the medium temperatures recorded in each site, showed marked interspecific and spatial variability. Among the analysed species, *M. cephalus* consistently exhibited the lowest TSM scores (often <0.10). Conversely, *D. labrax* (0.80-1), *S. aurata* (065-0.85), and *R. decussatus* (0.75-0.87) displayed higher TSM scores. From a spatial perspective, the Nora and Tortolì lagoons exhibited the lowest TSM values and, therefore, the highest TSM vulnerability scores. In contrast, the Feraxi and Peschiera San Giovanni lagoons exhibited relatively higher TSM values and, thus, lower TSM vulnerability scores (Figure 3.2; Table A4.1).

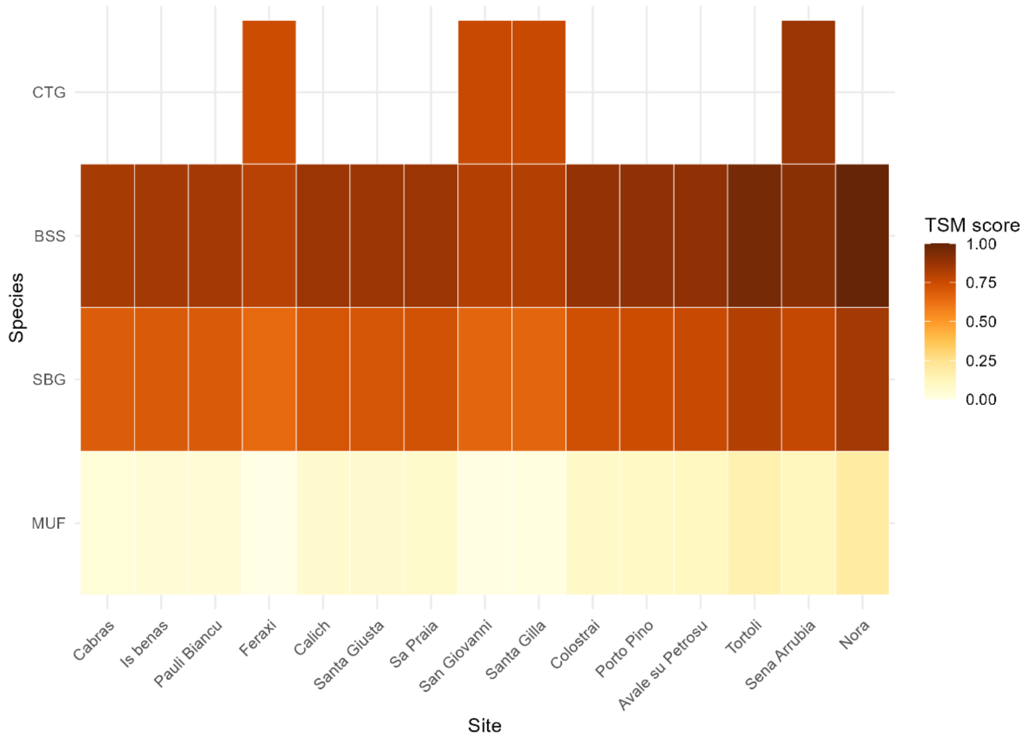

Figure 3.2 Heatmap of the thermal safety margin TSM score (normalized value of species TSM) for each species in each site. Dicentrarchus labrax *- BSS,* Mugil cephalus *- MUF,* Sparus aurata *- SBG, and* Ruditapes decussatus *- CTG*

The results from the stock assessment models (Table A4.2; Appendix 6) and the three scenarios built from the results are illustrated in Figure 3.4, Figure 3.3, and Figure 3.5.

In the "central" scenario, the flathead grey mullet exhibited five populations in healthy condition, two in recovery, and eight fully overfished/outside safe biological limits. The european seabass showed a similar distribution of stock categories, with several stocks under pressure. The gilthead seabream displayed a slightly more balanced distribution of the stock condition categories, with six healthy populations, eight overfished, and one in recovery. In contrast, the grooved carpet shell resulted the most vulnerable: no healthy stocks were identified, and overexploitation was the prevailing condition, with biomass levels consistently below those required to produce maximum sustainable yields (Figure 3.5A).

Considering the optimistic (i.e. lower *F/FMSY* and upper *B/BMSY*) and pessimistic (i.e. upper *F/FMSY* and lower *B/BMSY*) scenarios, some site and species exhibited greater variability in stock assessment results than others. Notably, in the optimistic scenario (Figure 3.5B) only three stocks (*M. cephalus* in Cabras and Pauli Biancu, *S. aurata* in Avalè su Petrosu) exhibited a fully overfished/outside safe biological limits status. In contrast, in the pessimistic scenario (Figure3.5C), almost all the stocks showed fully overfished/outside safe biological limits status, with 8 stocks being severely depleted, and only few stocks in healthy (*M. cephalus* in Feraxi and *S. aurata* in Tortolì lagoon) or recovery conditions (*S. aurata* in Santa Giusta lagoon and *D. labrax* in Peschiera San Giovanni). The plots illustrated in Figure 3.3, Figure 3.4 show the uncertainty associated with B/Bmsy and F/Fmsy central values estimates, expressed as deviation from the central value (last.B/Bmsy and last.F/Fmsy, set at zero for visualization purposes).

The variability in B/Bmsy values showed most intervals spanning approximately ±0.4 units around the central values. In contrast, F/Fmsy lower and upper values exhibited higher uncertainty, with intervals wider than those for B/Bmsy, and exceeding, in some cases, +9 units from the central values. Notably, while for B/Bmsy the uncertainty spanned similarly both for the lower and upper values, for F/Fmsy the uncertainty was most associated with upper values, resulting in graphs highly skewed toward positive deviations.

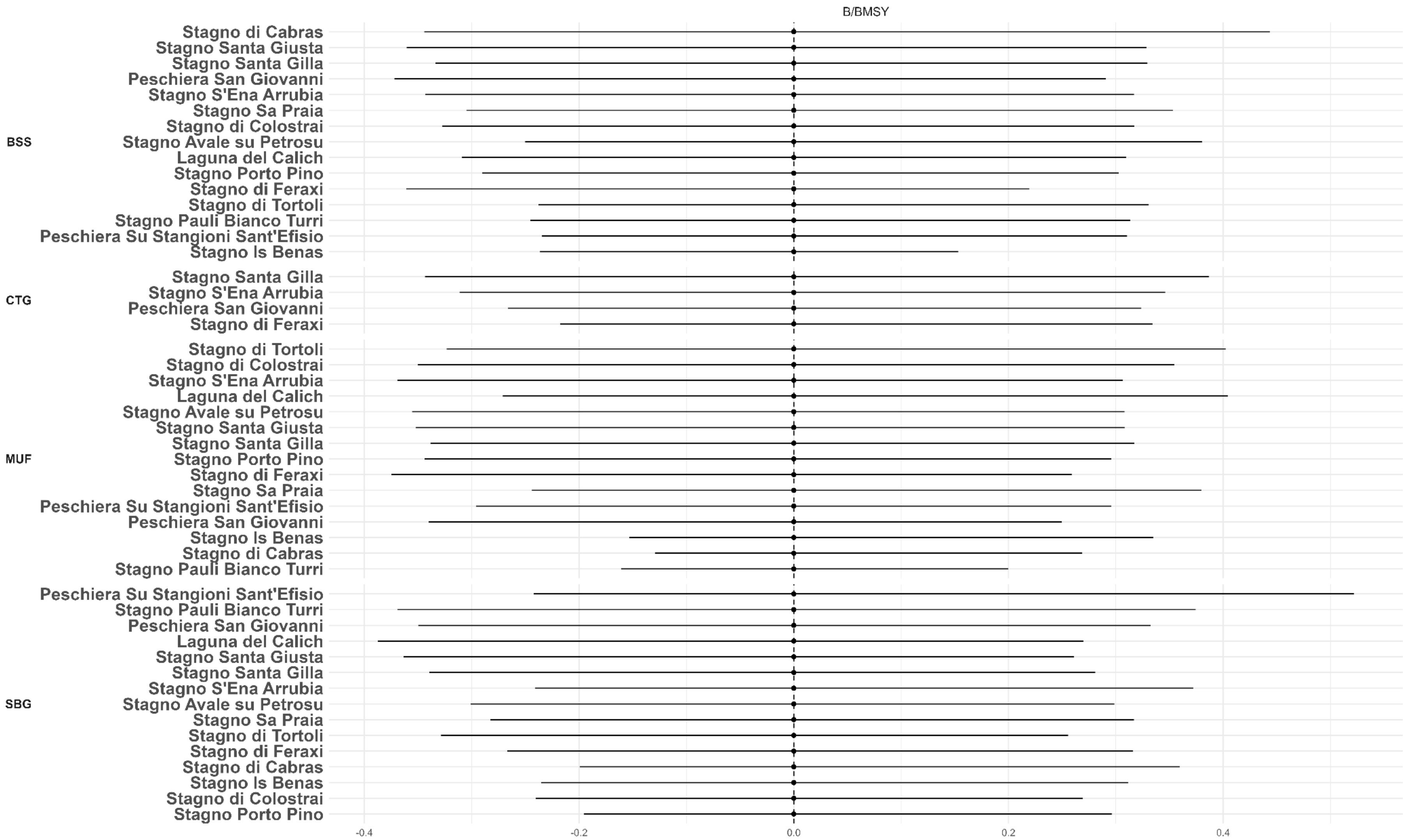

Figure 3.3 Variability in B/Bmsy upper and lower values (from ucl and lcl) around "central" values.

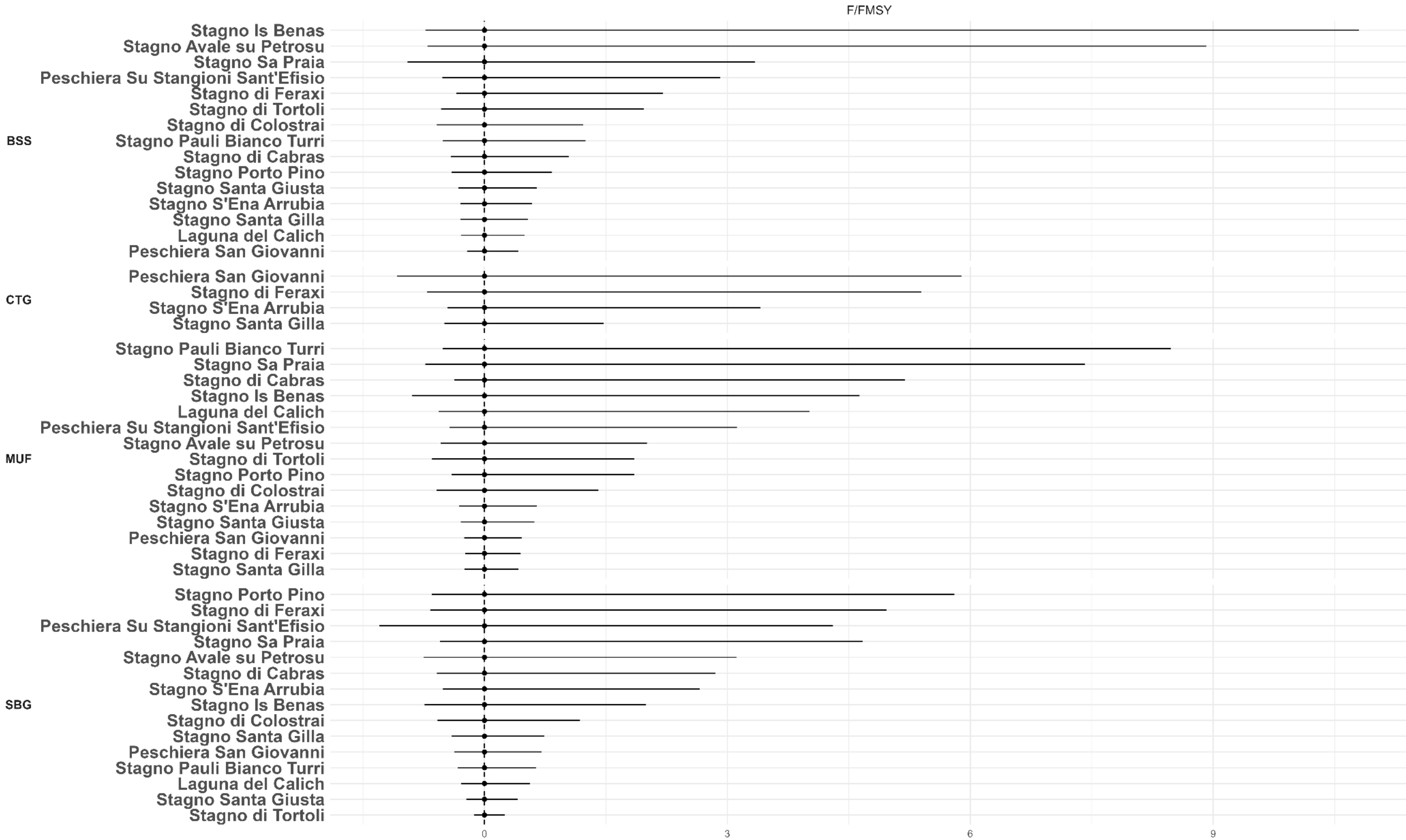

Figure 3.4 Variability in F/Fmsy upper and lower values (ucl and lcl) around "central" values

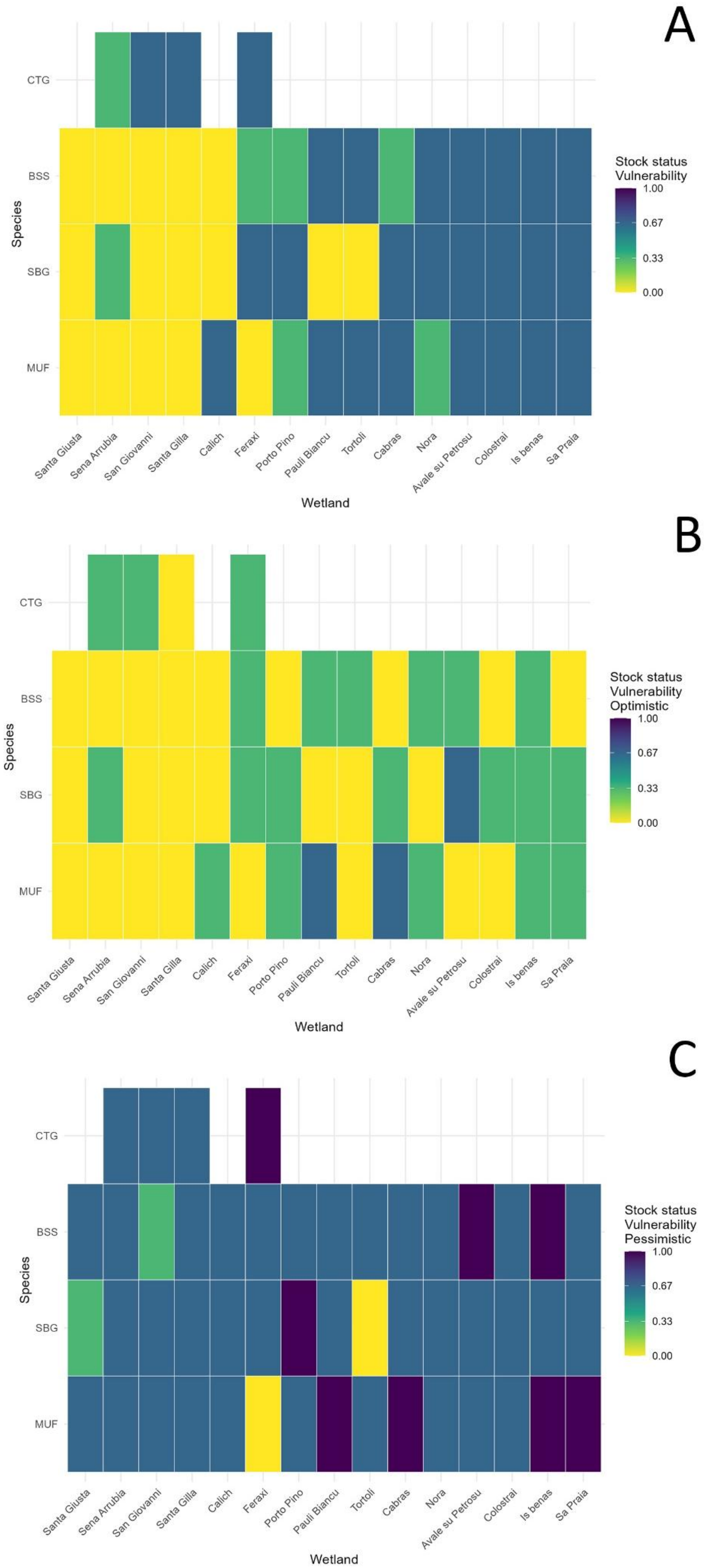

Figure 3.5 Heatmaps of the stock status vulnerability for each species in each site for the 3 stock assessment status scenarios. Dicentrarchus labrax - *BSS,* Mugil cephalus - *MUF*, Sparus aurata - *SBG, and* Ruditapes decussatus – *CTG.*

3.3.2. Habitat vulnerability

The ecological status classification based on the assessment of Biological Quality Elements (BQEs) revealed differences among the lagoons under scrutiny, with 5 lagoons classified as poor, 3 as sufficient, 4 good and 3 lagoons found in high ecological status. The water exchange capacity greatly varied among sites, with S'Ena Arrubia lagoon showing the highest water exchange capacity vulnerability (0.83) and the Santa Gilla one the lowest (0). Similarly, the S'Ena Arrubia shows the worst vulnerability score for the IAS metric (1). The remaining lagoons scored moderate-low (0.33) or moderate-high (0.83) values. The final Habitat vulnerability (Hb) scores across the analysed lagoons ranged from 0.25 to 0.92 (Figure 3.6), reflecting the marked differences in ecosystem functioning, hydrological connectivity, and susceptibility to ecological disturbances driven by alien species. The highest habitat vulnerability was recorded in the S'Ena Arrubia lagoon (Hb=0.92), followed by the Cabras one (Hb=0.71), while San Giovanni and Avalè su Petrosu lagoons exhibited the lowest vulnerability values (both Hb=0.25).

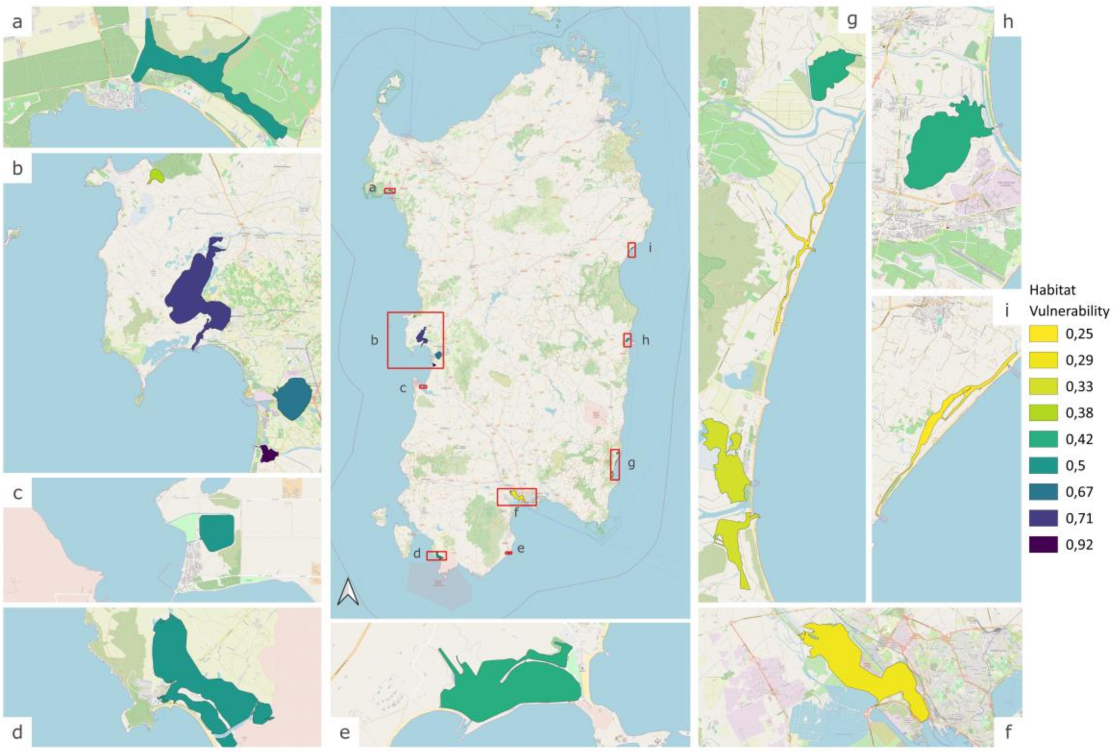

Figure 3.6 Habitat vulnerability score across each lagoon. a=Calich; b=Is Benas, Cabras, Santa Giusta and S'Ena Arrubia; c=Pauli Biancu; d=Porto Pino; e=Nora; f= Santa Gilla; g=Colostrai; Feraxi; Peschiera San Giovanni and Sa Praia; h=Tortolì; i=Avalè Su Petrosu.

3.3.3. Food provisioning vulnerability

The overall vulnerability of the food provisioning ecosystem service across the Sardinian lagoons showed moderate variability, with scores ranging from moderately low (0.32) to moderately high vulnerability (0.65; Figure 3.7) in the central scenario.

Considering the final food provisioning ES vulnerability scores across the four scenarios (without stock status, central stock status, optimistic and pessimistic scenarios), some differences emerged. The resulting maps of all the scenarios evaluated are illustrated in Figure 3.7, Figure 3.8, Figure 3.9, and Figure 3.10.

As shown in Table 3.4, the leave-one out analysis shows that the inclusion of the stock status metric modifies the relative ranking of some sites, while others ranking remained unvaried.

Notably, Tortolì lagoon is ranked among the less vulnerable in the relative ranking when considering the stock status under all scenarios, while it appears to be within the top 5 in the relative rankings under the "no-stock" scenario.

Similarly, but to a lesser extent, the Cabras lagoon is ranked as the second most vulnerable lagoon in the relative rankings under all stock scenarios, while when the stock status is not considered, it appears, comparatively, less vulnerable.

Table 3.4 Leave-one-out analysis to assess the effect of including stock status metric. The first part of the table shows the vulnerability scores while the second half shows the relative vulnerability rankings by site.

Scenario	**Vulnerability scores**				**Vulnerability rankings**			
Site	**No stock**	**central**	**optimistic**	**pessimistic**	**No stock**	**central**	**optimistic**	**pessimistic**
Avale su Petrosu	0.34	0.39	0.31	0.40	15	14	15	15
Cabras	0.52	0.58	0.57	0.64	6	2	2	2
Calich	0.44	0.47	0.43	0.50	11	9	8	8
Colostrai	0.45	0.46	0.37	0.46	10	10	12	10
Feraxi	0.43	0.42	0.39	0.43	13	12	11	13
Is benas	0.46	0.48	0.42	0.50	9	7	9	9
Nora	0.51	0.50	0.44	0.52	7	5	6	7
Pauli Biancu	0.56	0.47	0.46	0.56	3	8	5	5
Porto Pino	0.56	0.53	0.48	0.59	4	3	4	4
Sa Praia	0.48	0.50	0.43	0.52	8	4	7	6
San Giovanni	0.40	0.33	0.31	0.41	14	15	14	14
Santa Gilla	0.44	0.41	0.34	0.45	12	13	13	12
Santa Giusta	0.57	0.50	0.50	0.59	2	6	3	3
Sena Arrubia	0.69	0.64	0.64	0.73	1	1	1	1
Tortoli	0.52	0.45	0.42	0.45	5	11	10	11

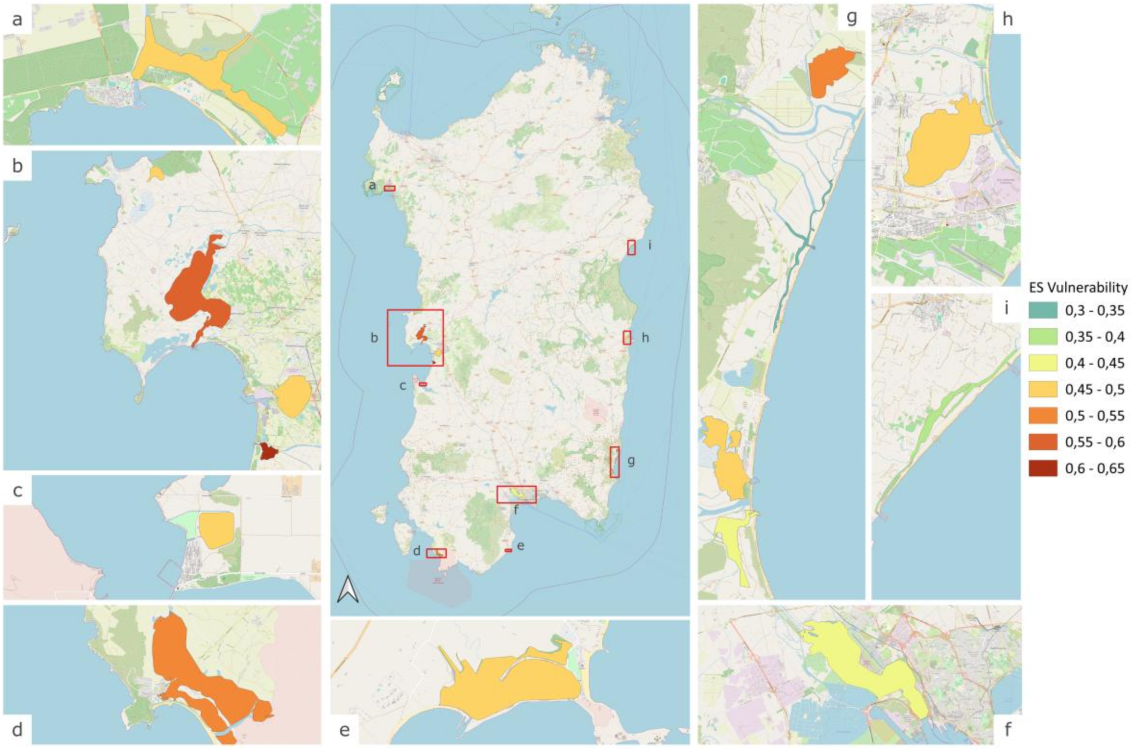

Figure 3.7 ES Vulnerability across each lagoon central scenario. a=Calich; b=Is Benas, Cabras, Santa Giusta and S'Ena Arrubia; c=Pauli Biancu; d=Porto Pino; e=Nora; f= Santa Gilla; g=Colostrai; Feraxi; Peschiera San Giovanni and Sa Praia; h=Tortolì; i=Avalè Su Petrosu.

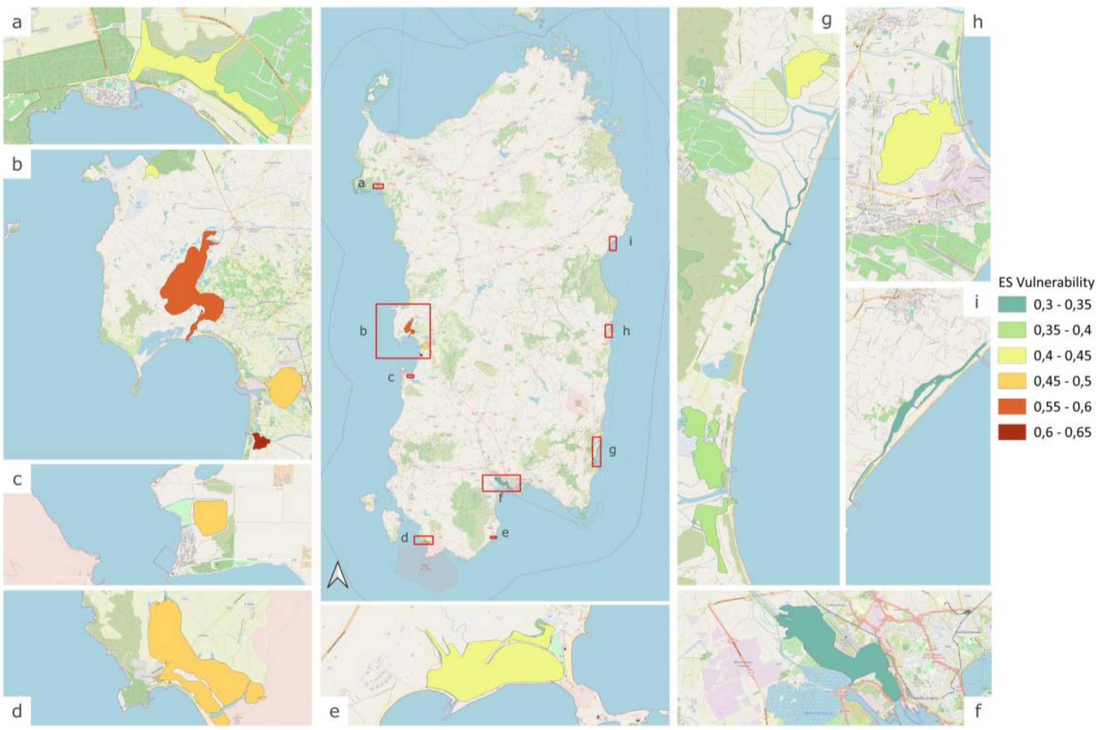

Figure 3.8 ES Vulnerability Map - Optimistic scenario across each lagoon. a=Calich; b=Is Benas, Cabras, Santa Giusta and S'Ena Arrubia; c=Pauli Biancu; d=Porto Pino; e=Nora; f= Santa Gilla; g=Colostrai; Feraxi; Peschiera San Giovanni and Sa Praia; h=Tortolì; i=Avalè Su Petrosu.

Figure 3.9 ES Vulnerability Map - Pessimistic scenario across each lagoon. a=Calich; b=Is Benas, Cabras, Santa Giusta and S'Ena Arrubia; c=Pauli Biancu; d=Porto Pino; e=Nora; f= Santa Gilla; g=Colostrai; Feraxi; Peschiera San Giovanni and Sa Praia; h=Tortolì; i=Avalè Su Petrosu.

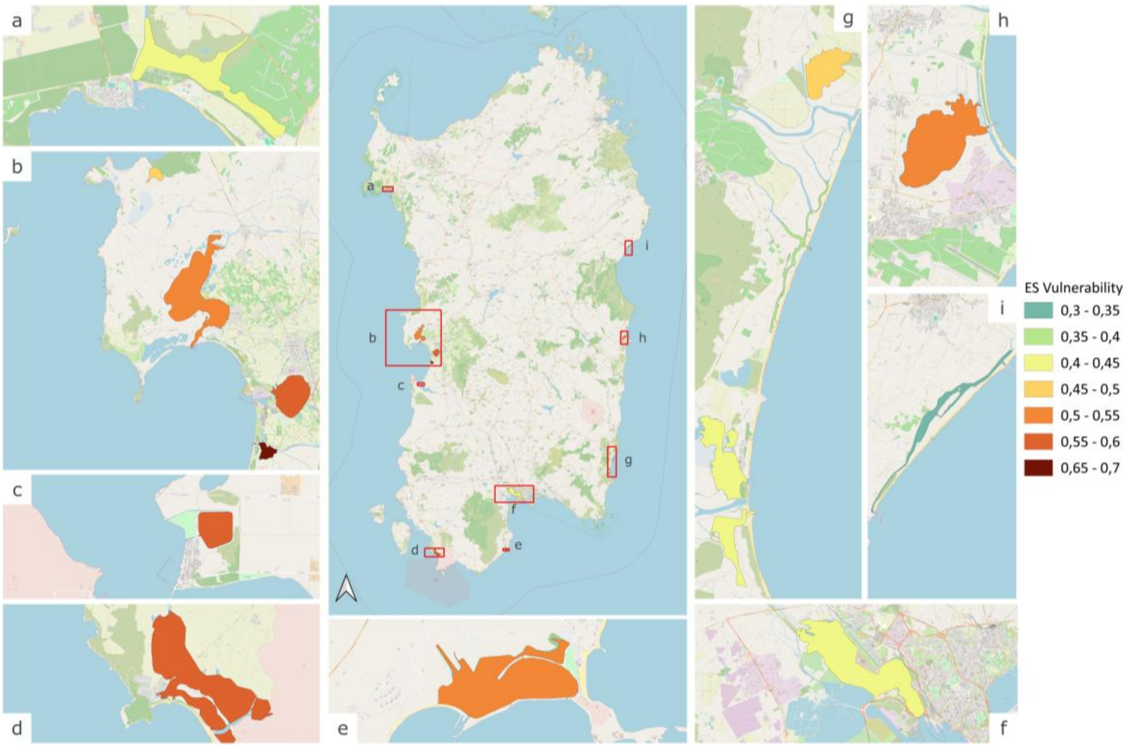

Figure 3.10 Vulnerability Map - No stock scenario across each lagoon. a=Calich; b=Is Benas, Cabras, Santa Giusta and S'Ena Arrubia; c=Pauli Biancu; d=Porto Pino; e=Nora; f= Santa Gilla; g=Colostrai; Feraxi; Peschiera San Giovanni and Sa Praia; h=Tortolì; i=Avalè Su Petrosu.

3.4. Discussion and limitations

As previously noted, despite the growing attention to vulnerability and risk assessments, there is still a general lack of studies explicitly addressing regional specificities, particularly within Europe (Aragão et al., 2022). This gap of knowledge may partly be explained from the outcomes of previous national-scale assessments, which suggested generally low vulnerability (and high adaptive capacity) for the European continent compared to less economically developed regions (Allison et al., 2009; Ding et al., 2017).

However, more recent research emphasizes the need to downscale such assessments to regional and local contexts (Pinnegar et al., 2019), as large-scale analyses may overlook the spatial heterogeneity of vulnerability, exposure, and hazard. Downscaling such assessments is, therefore, crucial for informing local management and decision-making, since the design of effective adaptation and mitigation strategies ultimately depends on capturing the ecological and socio-economic specificities of individual regions (Holsman et al., 2019, 2020). The regional scale of this study enabled me to show that vulnerability is not evenly distributed across the different socio-ecological systems analysed.

Findings revealed marked interspecific differences in species vulnerability, driven by biological traits and site-specific metrics. For instance, *M. cephalus* exhibited a broad temperature specificity, suggesting a relative robustness to thermal variability, while the bivalve *R. decussatus* was classified as highly vulnerable due to its benthic adult life. This is consistent with other vulnerability studies, were species highly associated to the sediment compartment, due to their limited movement capacity, are considered at greater risk than those in the water column (Fulton, 2011; Ramos et al., 2022).

Thermal Safety Margin (TSM) results identified spatial heterogeneity across the different sites considered, with some sites in the south-east and east Sardinian coast emerging as thermal stress hotspots, while sites with lower TSM vulnerability scores showing comparatively safer thermal conditions. These findings can help to identify climate vulnerability and potential refugia (Pörtner & Farrell, 2008; Sunday et al., 2014). Sites with higher mean annual temperatures may therefore represent priority areas for monitoring, especially if the magnitude of hazard will be high, while thermally safer sites could function as refugia, supporting species persistence under CC.

Habitat vulnerability scores highlighted the importance of ecosystem condition and the heterogeneity in Sardinian sites. Some site displayed particularly high scores, reflecting a combination of poor ecological status, reduced hydrological exchange, and the massive presence of invasive alien species, while others exhibited lower scores and vulnerability.

Moreover, by comparing the final food provisioning ES vulnerability scores across the four scenarios (without stock status, centrale stock status results, optimistic and pessimistic scenarios) this study shows that the inclusion of the stock status metric can affect the relative ranking of some sites, while for others the effect was negligible. Differences between the "no stock" and the "central" stock scenarios can be explained by the combined effect of habitat vulnerability and the relative contribution of individual species to food provisioning. Notably, where a species with high relative importance for the ecosystem service exhibited extreme stock status values, its contribution tended to disproportionately drive the site-level vulnerability, either inflating or reducing it, compared to the "no stock" scenario, where vulnerability was determined solely by Thermal Safety Margin (TSM) and habitat scores.

Variability between the central and the optimistic and pessimistic stock scenarios mainly reflected the uncertainty range derived from the stock assessment outputs. When uncertainty was high and involved species of major importance for the lagoon, the site-level vulnerability score shifted accordingly, influencing at large the overall ranking.

For example, in the Cabras lagoon, differences in site-level vulnerability between the "no stock" and "stock" scenarios were primarily driven by the stock status of *M. cephalus*, which accounts for more than 75% of the total food provisioning contribution. While its stock vulnerability score remained moderate (0.67) under both central and optimistic scenarios, it reached the maximum value (1.0) under the pessimistic scenario. Given the high weighting of *M. cephalus* in the final indicator, such inflated score markedly amplified the overall site vulnerability, explaining why the Cabras lagoon is consistently ranked among the most vulnerable Sardinian lagoons, when stock status is considered.

Conversely, in Tortolì lagoon, where *S. aurata* dominates the catch composition (and thus the species importance for the ESs) and displayed a healthy stock status across all scenarios, site-level vulnerability remained comparatively low. In the "no stock" scenario, however, the Tortolì lagoon ranked among the most vulnerable lagoons, due to its high TSM scores (second only to the scores of the Nora lagoon) combined with intermediate habitat vulnerability, conditions that inflated its vulnerability when stock information was omitted.

Another key finding of my study, thus, concerns the influence of including stock status information on site-level vulnerability rankings. In particular, three main insights emerge from this analysis.

First, omitting stock status can lead to misleading conclusions, either over- or underestimating vulnerability.

Second, the uncertainty in stock status (here expressed as variability among stock assessment results scenarios) cascade on the uncertainty of the final vulnerability indicator, highlighting the importance of model validation through monitoring efforts. Indeed, even if a commonly employed (e.g. within ICES, STECF and GFCM) stock assessment model was applied, CMSY is not exempt from problems. In a recent study, Bouch and coauthors (1021) found that CMSY tends to produce negative bias in relation to the ICES analytical assessments, underestimating relative stock biomass, while overestimating the relative fishing mortality (Bouch et al., 2021), thus posing challenges for fish managers.

Third, stock status information may not only help refining vulnerability assessment but also providing insights about whether rebuilding sustainable stock levels could serve as an effective measure to reduce overall food provisioning ES vulnerability. In this sense, and considering the limitation explained above, there is the need to more effort in monitoring stock status with data-rich assessment methodologies, in order to improve estimates and enable their use for management purposes (Bouch et al., 2021).

As in all indicator-based assessments, the selection of indicators, data availability and quality, and mismatches in temporal and spatial scales of datasets affected my results. One limitation of this study relates to the estimation of the TSM using TP90 values obtained from the literature, a conservative approach that may introduce bias. AquaMaps TP90 values are typically derived from global species distribution and may not fully capture local population-level thermal tolerances. For widely distributed species, thermal limits can vary substantially among populations (Dressler et al., 2023), challenging the use of global parameters to predict local vulnerability to warming.

Populations chronically exposed to high temperatures over generations may either exhibit greater resilience through adaptation or acclimatization (Fangue et al., 2006; Gracet, 2022; Shulte, 2007), or conversely, show higher vulnerability due to their proximity to upper thermal limits (Rijnsdorp et al., 2009). Evidence of inter-population specific thermal traits has been widely documented, for instance, for *Oncorhynchus mykiss* where individuals from warmer sites exhibited higher critical thermal maxima compared to populations from cooler environments (Dressler et al., 2023).

These complexities underscore the importance of experimentally site-specific derived critical thermal limits or TP90 values to reduce indicators uncertainty and provide more realistic vulnerability assessments.

However, the current availability of such fine-scale data is very limited, with information being fragmented across both species and regions. This lack of consistent and spatially resolved data limits an effective application of vulnerability assessment for regional-scale studies such as the one presented here, stressing the need for future research efforts focusing on population-specific thermal tolerance.

In addition, this study did not include species of high ecological and socio-economic relevance, such as *Anguilla Anguilla* (L., 1758), a species considered highly vulnerable to both climate and anthropogenic stressors (Podda et al., 2022) and classified as Critically Endangered (CR; IUCN, 2014), but also important for food provisioning in several lagoons in Sardinia (Podda et al., 2022). Similarly, certain sites, characterized by multiple vulnerability drivers, including invasive species and high susceptibility linked to limited water exchange capacity, were not included due to the lack of site-level data that hampered the possibility to perform stock assessment models. Future investigations should prioritize filling these gaps by expanding both the species and site coverage, as their inclusion could substantially refine regional vulnerability assessments.

3.5. Conclusion

This study presented the first regional-scale assessment of the climate vulnerability of the food provisioning ES in a selection of Sardinian socio-ecological lagoon systems, using available secondary data and vulnerability category pre-imposed to build tailored vulnerability indicators.

By integrating habitat and species level indicators, the ES vulnerability was explored, and insights into the fine-scale heterogeneity of vulnerability patterns across different socio-ecological systems were provided.

Despite the challenges encountered, especially those related to data availability, the approach outlined here could be the starting point to be adapted in future vulnerability assessments by adding greater detail for the indicators and species that now lack in availability, but also by integrating the vulnerability factors of species related to their life phase in adjacent environments, especially for those that are considered migratory between freshwater and marine ecosystems.

Beyond its regional relevance, the adoption of this approach can inform future management strategies, highlighting where adaptation measures could be targeted to reduce vulnerability, and, by identifying which component of vulnerability contributes most to each site ESs vulnerability, it allows to determine the most appropriate adaptation or response measures. However, to improve the reliability of this assessment, future efforts should focus on improving data coverage and granularity, and refining and extending indicator selection.

3.6. References

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Chapter 4. Socio-economic perspectives on climate vulnerability of food provisioning ecosystem services

This chapter was developed in collaboration with the School of Environmental Sciences at University of East Anglia (UEA) and the Centre for Environment Fisheries & Aquaculture Science (Cefas) under the supervision of Dr. Tiziana Luisetti and Prof. Irene Lorenzoni during the period spent abroad hosted by UEA and Cefas at the Collaborative Centre for Sustainable Use of the Seas (CCSUS).

4.1. Introduction

Coastal and marine ecosystems provide a wide array of goods and services to human societies, including climate regulation, extreme events buffering, biodiversity support, art inspiration and food resources for nutrition (Barbier et al., 2014, 2017; Costanza et al., 2014). Among these, food provisioning represents an important ecosystem service (ES), contributing to food security, livelihoods, as well as cultural identity for coastal communities worldwide (Barbier, 2019). Globally, in 2022, approximately 62 million people were employed in the primary fisheries and aquaculture sector, with a significant proportion engaged in artisanal and small-scale operations (FAO, 2024).

In recent years, many studies have drawn attention to how climate change (CC) is impacting marine and coastal systems, and with them, the fishing sector (Cochrane et al., 2009; Jones et al., 2015). Warming water temperatures, sea level rise, altered precipitation patterns, and extreme weather events have been linked to shifts in ecosystem structure, processes and functioning (Guild et al., 2024; IPCC, 2022), including changes in species distribution, composition, reproduction timing, growth and survival rates (Poloczanska et al. 2013), ultimately impacting human communities that rely on them. The past and expected impacts of CC are not uniform across regions; rather, they depend on local geography, ecological characteristics -such as species composition and status, habitat type, condition, and connectivity- and socio-economic factors, including governance, resource dependence, infrastructure, market rules (IPCC, 2022; Payne et al., 2021; Zahnow et al., 2025).

In the future, multiple synergistic stressors, including pollution, habitat degradation, invasive species, unsustainable extraction, and CC, are likely to push coastal fisheries into increasingly precarious conditions (Hare et al., 2016; Murciano et al., 2021). Indeed, fishing incomes in many countries already fall below national living wage thresholds (Giron-Nava et al., 2018), that, in many cases, further amplifies the impacts. Understanding the future impacts of CC on these socio-ecological systems is thus crucial to inform sustainable policies and actions that support local adaptation and longer-term sustainability of fisheries and the livelihoods of those who currently depend on them (Berrouet et al., 2018).

To this aim, climate vulnerability assessments (CVAs) have offered a structured, even if not standardized, framework to understand, quantify, and synthesize the impacts of CC on socio-ecological systems (Allison et al., 2009; Cinner et al., 2012; Colburn et al., 2016; Gaichas et al., 2014; Hare et al., 2016; Payne et al., 2021; Pinnegar et al., 2019). Within such assessments, vulnerability have been evaluated through the adoption and the implementation of a plethora of concepts and models (see Estoque et al., 2023 for a review on IPCC vulnerability concept implementation in CVAs). In these assessments, vulnerability encompasses a variety of concepts and elements, but its conceptualization evolved over time (Begum et al., 2022; Estoque et al., 2023). In fact (see chapter 1 in this thesis) with the Special Report on Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation (SREX; IPCC, 2012) and the Fifth Assessment Report (AR5) in 2014, IPCC introduced the risk-based approach (Estoque et al., 2023), defining "risk" as a function of hazard, exposure, and vulnerability (IPCC, 2014). Contrary to previous assessments, vulnerability is now considered as an intrinsic property of a system, independent of its exposure to climate hazards, which now specifically refers to the exposed elements (e.g., people, assets, species or ecosystems) (Estoque et al., 2023; IPCC 2012, 2014, 2022).

In this context, CVAs have been applied at multiple spatial scales and are particularly useful in identifying which components of a system are most vulnerable to stressors and why (Aragão et al., 2022; Cinner et al., 2012; Colburn et al., 2016; Payne et al., 2021).

Indicators are central to quantitative CVA assessments, as they provide the means to represent the dimensions of vulnerability in measurable terms (Kasthala et al., 2024) and to evaluate change. Notably, indicator choice can shape perceptions and understandings of vulnerability, and influence policy priorities (de Sherbinin et al., 2019; Papathoma-Köhle et al., 2019). Studies of existing assessments have highlighted their strengths, as well as conceptual inconsistencies, methodological gaps, and a lack of local-level data (Eriksen and Kelly, 2007; FAO, 2015; Kasthala et al., 2024;)

While the traditional quantitative CVA approach provides a valuable and often cost-effective starting point for assessing vulnerability, sources of concern are the lack of clarity associated with the concept of vulnerability still persisting in the scientific literature (Ford et al., 2018), in other words, what is being measured and how, as well as data availability at different levels and scales of analysis (Kasthala et al., 2024). The quantitative CVAs also risk oversimplification and have a limited ability to capture complexities of local realities and inter-relationships between specific system characteristics. To date, many CVAs adopt indeed a predominantly deductive approach, applying pre-defined categories of vulnerability and assuming *a priori* the directionality of effects through measurable quantitative elements

(Fletcher et al., 2021), while devoting comparatively less attention to the underlying “remote root causes” (Wisner et al. 2004, p.7) that shape vulnerability in specific contexts (Fletcher et al., 2021; Huggel et al., 2016; Ribot 2014).

To overcome these challenges, it has been argued that CVAs must reflect the dynamics and specific characteristics of local contexts, including species- and site-specific metrics (Albo-Puigserver et al., 2022; Bueno-Pardo et al., 2021), the sensitivity of ecological and social communities , as well social, cultural, and institutional factors that affect systems’ capacity to respond and adapt to change (Murciano et al., 2021). Several studies have stressed the importance of combining quantitative indicators with qualitative approaches, which provide valuable contextual information (Bryman, 2007) and allow a more nuanced understanding of social inequalities, perceptions, and structural drivers of vulnerability (Huggel et al., 2016; Jepson & Jacob, 2007; Jacob et al., 2010; Jacob et al., 2001; Marshall & Marshall, 2007; Tuler et al., 2008).

Some scholars have argued that qualitative approaches, potentially used with quantitative work through mixed methods studies, can help collect insights that may not otherwise be elicited through quantitative studies alone (Gordon et al., 2022; Lindkvist et al., 2022; Singh et al., 2025). Although methodologically diverse, such mixed methods approaches can contribute to also revealing how communities themselves conceptualize risks, adaptation strategies, and barriers to change (Chapagain et al., 2025; Fitchen et al., 1987). They can also expose context-specific factors such as power relations, institutional trust, and cultural values that are not easily quantifiable but are central to shaping vulnerability and adaptive capacity (Arxer et al. 2025; Cadag & Gaillard, 2012; Siders, 2019; Singh et al., 2025; Smit & Wandel, 2006).

The importance of assessing vulnerability using a variety of methodologies has been similarly recognized also in the Fifth IPCC Assessment Report (2014), highlighting that risk management does not entail only an accurate quantification of risks, and that approaches acknowledging diverse qualitative values, goals, and priorities can enhance adaptation effectiveness (Mach et al., 2016). Moreover, the most recent IPCC Sixth Assessment Report (2022) also recognized the importance of including local and indigenous knowledge in assessment processes, to overcome the exclusion of this form of knowledge in previous assessments (Begum et al., 2022). Similarly, other literature on adaptation and coping capacity highlights the role of barriers and enablers and their effect, thus emphasizing the role of more in depth and integrated approaches (Maltby et al., 2023). Although complex and resource intensive, integrating qualitative and quantitative insights into vulnerability assessments is therefore critical to avoid a purely quantitative and potentially technocratic perspective, ensuring community priorities are recognized, and that they ultimately develop more legitimate and locally grounded adaptation strategies (Van Aalst et al., 2008; Ribot, 2014).

4.1.1. Climate vulnerability in socio-ecological systems

The Mediterranean region has been recognized as a CC hotspot, due to changes that typically occur faster than the global average, such as temperature increases, altered precipitation regimes, and sea level rise (Giorgi, 2006; IPCC, 2014; Lionello et al., 2023; 2018). In the Mediterranean, coastal lagoons and estuarine areas can be considered complex socio-ecological systems (SES) that deliver multiple ecosystem services (ESs). The SES concept and framework (McGinnis and Ostrom, 2014; Ostrom, 2009, 2007) gained mainstream recognition in the1990s, together with the introduction of the ES concept (Daily 1997), emphasizing the deeply connection between ecological and human systems. The recognition of this connection provides thus a useful basis to start analysing the drivers of vulnerability in these complex systems. Lagoon-based small-scale fishery is characterized by artisanal practices, a high dependency on a limited number of species, as well as strong exposure to both environmental and economic stressors (Pérez-Ruzafa & Marcos, 2012). Typically, Mediterranean coastal lagoon fishery relies on the migratory movements of fish between the lagoon and the sea, and sometimes also to inland freshwater (Pérez-Ruzafa & Marcos, 2012), taking advantage from the entrance of juvenile fish, easily passing lagoon sea inlet through structures such as *lavorieri*, then growing in lagoon and riverine systems, and being later captured when attempting to migrate back seaward as adults for reproduction (Pérez-Ruzafa & Marcos, 2012).

By the late 20th century, this traditional method was carried out across approximately 29,000 hectares of Mediterranean coastal lagoons, producing an estimated 13,000 tonnes annually of species like sea bass, sea bream, mullet, and eel (FAO, 1979). Although relatively small in scale, these fisheries continue to be essential for the livelihood of many families and hold deep cultural significance (Basurto et al., 2025). However, fishers in coastal lagoons, similarly to small-scale fishery in the rest of the world, are often marginalized from decision-making processes, lacking institutional support (Basurto et al., 2025; Jentoft, 2019) and cooperation (Béné et al., 2006) and without adequate adaptation strategies in the face of growing climate pressures, as also observed about traditional fishing practices in many parts of the world (Allison et al., 2009: Allison and Ellis, 2001; Davies-Vollum et al., 2021; FAO, 2020).

Adopting an ES lens can provide a useful framework to assess climate change impacts on these complex SES; it allows an exploration of the contributions of both ecological and socio-economic components to the food provisioning ESs of coastal and lagoon fisheries. This also enables an understanding of their vulnerabilities. To do so, it is relevant to consider that on the one hand, the vulnerability of the ecological system varies according to species, their characteristics, life history and population status, as well as on the condition of habitats (as discussed in the previous chapter).

On the other hand, the socio-economic dimension is shaped by fishers and the wider social system (i.e. governance, economics and political setting) that interact with and within these ecosystems: through extraction (fishing activities), market demand, regulations, and management practices (Maltby et al., 2023), they transform the ecosystem service into an actual flow of services and benefits, while, at the same time, influencing its long-term sustainability.

Previous literature has explored the assessment of vulnerability of SES with different conceptual frameworks, but, to date, there is paucity of in-depth consideration of the linkages between ecological and social systems (Chapagain et al., 2025; Berrouet et al., 2018). Regarding the latter, Berrouet and coauthors (2018) defined, within a SES vulnerability framework, social vulnerability as a “function of the dependency that the beneficiaries have on the ecosystem service and the adaptation of beneficiaries to that change”. Regarding the adaptation component in SES, Maltby and co-authors (2023) explored the possible causes of adaptation success or failure, defining the barriers as “factors that make it harder to plan and implement adaptation actions” and the enablers as those factor that “make it easier to plan and implement adaptation actions, expand adaptation options, or provide ancillary co-benefits”.

Drawing on IPCC's (2022) revised vulnerability concept and building on insights and challenges highlights from past vulnerability application on SESs, in this chapter the Social vulnerability reflects those characteristics of the social elements of the SES involved in the provisioning of the ESs that determine their propensity or predisposition to be adversely affected (Berrouet et al., 2018), encompassing related concepts such as social sensitivity and lack of capacity (IPCC 2022).

Social sensitivity represents the degree to which the social system is affected and is defined by factors that influence the impacts of a hazard (IPCC, 2022), such as the level of dependence of the social system on the capacity of the ecological system to provide the food provisioning ES (Berrouet et al., 2018). Social capacity represents the ability of a social system to anticipate, prepare for, and respond to current and future CC impacts. Capacity can be further distinguished into two key aspects, coping and adaptive capacity (IPCC, 2022).

Capacity to cope is the ability of the social component of a system to address, manage, and overcome adverse conditions in the short to medium term, by using available skills, resources, and opportunities (IPCC 2022). Adaptive capacity is the ability of social systems to adjust to potential harm, seize opportunities (Berrouet et al., 2018; IPCC 2022). An example is acquiring the knowledge to implement new farming practices or change target species, also thanks to collaboration with R&D. A lack of capacity can greatly increase a system's vulnerability and, consequently, its overall risk.

Yet, there are many coastal-marine area in the Mediterranean where no such vulnerability assessments have been undertaken. For example, in Sardinia, no targeted climate vulnerability assessment has been conducted for the lagoon fisheries sector, neither at the regional nor local scale.

The study outlined in this chapter therefore aims to contribute to fill this gap of knowledge and explore, for the first time ever (to the best of my knowledge) the vulnerability to climate change of food provisioning services in Sardinian coastal lagoons. By adopting the ES lens and focusing on the socio-economic component, I investigate, mainly in a qualitative way, the past impacts of CC and explore the vulnerability (and its component of sensitivity, adaptive and coping capacities as outlined above) in two social groups: fisheries cooperatives and fishers. In this sense, using methods from the social sciences can help in framing the findings considering political, cultural and socio-economic contexts (de Ruiter & van Loon, 2022).

4.2. Materials and methods

4.2.1. Study area and context

Sardinia is the second largest island in the Mediterranean Sea. Lagoon-based fisheries are small-scale, characterized by artisanal practices, a high dependency on a limited set of species, and strong exposure to both environmental and economic stressors. Sardinian coastal lagoons host a variety of commercially and ecologically important fish species, such as *Mugilidae* (flathead grey mullets), *Dicentrarchus labrax* (European seabass), *Sparus aurata* (gilthead seabream), and *Anguilla anguilla* (European eel), (Madau et al., 2018) whose vulnerability to climate change varies due to differing ecological niches and life histories.

In Sardinia, fishing activities in transitional waters like coastal lagoons are commonly organized into fisheries cooperatives and/or consortia that fishers form part of. This form of organization enables fishers to reduce some costs, particularly transaction costs, while fostering more efficient resource management (Madau et al.,2018).

As a result, decision-making about fisheries operates at two levels: (1) individual fishers who conduct fishing activities and (2) collective management via cooperatives and consortia, which potentially enables strategic directions, oversees financial and administrative tasks (e.g., co-management of common fishing grounds, purchase of shared technical inputs, payment of taxes), and allocates investments to collective capital endowment (Madau et al., 2018).

This research is set within a context that represents an important turning point for lagoon fisheries in Sardinia. Notably, for months in 2024 and early 2025 there have been protests and mobilisations by fishery cooperatives and associations seeking to resolve various perceived problems and to call for a greater coordination, among which the most important was the request to initiation of extraordinary maintenance works within the lagoons, financed with 6 million euros under Article 13, paragraph 47 of Regional Law 27/10/2021, but never started, till summer 2025.

Figure 4.1 One of the banners displayed during the mobilizations of fishers' cooperatives. [It translates into: "If I don't have oxygen, I will only be a memory"] (Photo by Antonio Loi, Marceddì)

4.2.2. Methods: Questionnaires on ES vulnerability from a social perspective

The vulnerability assessment framework in IPCC AR6 (2022), for the reasons outlined above, was the starting point to explore the social vulnerability (social sensitivity, lack of coping and adaptive capacities) of lagoon-based fisheries in Sardinia, with a focus on the human component of food provisioning ESs flow.

The focus of this study are the "direct" beneficiaries of the lagoon food provisioning ES namely fishers and fishing cooperatives. Although I recognise the importance of other relevant actors in the lagoon socio-ecological system; however, these are not included in the empirical data collection phase of this exploratory study. Social sensitivity and social capacity were elicited as part of this study by means of structured questionnaires, developed and administered with Cooperative/consortia level, and fishers in Sardinian lagoons.

The questionnaire methodology was chosen as it represents a practical research tool relatively cost-effective and allows for extensive data collection, which have been increasingly applied to investigate complex issues such as environmental change, social identity, mobility, quality of life, community relations, and social networks (McGuirk and O'Neill 2016). While the questionnaire approach has limitations, such as the depth of qualitative insights, it also offers several strengths, helping to provide systematic evidence of social trends, processes, values, attitudes, and interpretations (McGuirk & O'Neill 2016).

To develop the questionnaires, an extensive literature search was done on IPPC AR6 and social vulnerability-related papers to identify existing factors and/or indicators involved in vulnerability of food provisioning ESs. Socio-demographic characteristics such as age, work experience, and education are often used in the literature as variables to assess vulnerability (Junio et al., 2015; Badjeck et al., 2013; Flanagan et al., 2011); however, their effects can be multi-directional, making their interpretation difficult, if not ambiguous. In this study, these data were collected primarily to characterize the study sample.

Regarding the sensitivity component, one "economic" factor has been identified and related to dependence on the natural resource (*i.e.*, the food provision from coastal lagoons); this was included in the questionnaires so it could be explored both at cooperative and fishers' level. Coping and adaptive capacities were explored through eliciting an understanding of how and whether social networks (drawing from work by Bodin and Crona 2008; D'agata et al., 2020; Richmond and Casali 2022; Stoeckl et al., 2017) and broader economic and organizational dynamics may influence these capacity factors (Table 4.1).

I also included items that elicited the perception of barriers and enablers to adaptation, the impacts of past events, and awareness of climate-related risks (Ankrah, 2018; Chen, 2020; Martins and Gasalla, 2018). Here, I recognise these cannot be unambiguously related to vulnerability for two main reasons: they can be influenced by other components such as past exposure (e.g., the degree of damages experienced in the past related also to exposure); they have not previously been explored in this case study context (the literature indicates that the relationship between perceived barriers and enablers with adaptive or coping capacity is highly site-specific).

To avoid forcing these questions into categories where their conceptual meaning might not strictly match, I referred to them as a separate category named "Risk Awareness and Past Impacts" (Table 4.2). Within this category, the question related to barriers and enablers towards adaptive and coping capacity are of particular importance, since the vulnerability literature identifies them as key factors influencing whether adaptive capacity can or cannot be translated into effective actions (Eisenack et al., 2018; Maltby et al., 2023). Moreover, within this category, recognizing that climate change-related warmer temperatures and heatwaves can affect not only the ecological component of the SES but also the working capacity of fishers engaged in outdoor activities, due to highest risk of increased injury and labour productivity losses (Bednar-Friedl et al., 2022; Dellink et al., 2019; Gosling et al., 2018; Szewczyk et al., 2018; Orlov et al., 2019), I included specific questions exploring both past impacts and future perceptions of heat waves on fishers' work conditions.

This study adopts a combined deductive-inductive approach. While I drew upon existing vulnerability categories already identified in the literature to guide the questionnaire's design, I also relied on predominantly open-ended questions to try capture unanticipated factors, dynamics and concepts emerging from the fieldwork and fishers' voices (Allen 2017; Mwita 2022). This dual strategy enabled me to remain anchored in established frameworks while open to context-specific insights revealed by the respondents.

The full questionnaires are reported in Appendix 3 – Questionnaires. Ethics approval for the study was granted by the University of East Anglia (UEA) General Research Ethics Committee (GREC).

Table 4.1 Components of social vulnerability, drivers and themes identified from the literature, and the resulting questions in questionnaires used with fishers and cooperatives in this study. ID columns represent the question ID number in questionnaires. Reference columns report a non-exhaustive list of literature (used in this thesis) on vulnerability and its components.

Vulnerability Component	Theme	Variable	Reference	ID	Respondent	Question in this study
Sensitivity	Economic	Dependence from coastal lagoon food provisioning ES	Berrouet et al., 2018; Ding et al., 2017; Morzaria-Luna et al., 2014	24	Fishers	Is fishing and/or aquaculture your only source of income?
				3	Cooperative	What activities does the cooperative carry out?
				4	Cooperative	Does the cooperative engage in other economic activities in/in the [lagoon name]? (e.g., recreational fishing, tourism)If yes, which activities?
Capacity to Cope and Adapt	Networks	Social capital (informal; fosters cooperation and mutual aid)	Stoeckl et al., 2017; D'agata et al., 2020; Bodin and Crona 2008; Richmond and Casali 2022; Cinner et al., 2018; Pelling et al., 2008; Adger et al., 2003	25	Fishers	Do you collaborate or share knowledge/resources with other cooperatives or fishers to address challenges in your work? If so, how?
		Formal networks & technical/scientific support	Nguyen et al., 2019; Lutz et al., 2014	39	Cooperative	Has your cooperative ever received technical/scientific support from research or business support organizations to address problems caused by extreme events or climate change?
	Economic / management characteristics	Adaptation strategies	Ara Begum et al., 2022; Salvadeo et al., 2021	32	Cooperative	Has the cooperative ever implemented strategies to cope with climate change?If not, why?
		Access to credit or financing	Cinner et al., 2018; Ding et al., 2017; Pomeroy et al., 2020	33	Cooperative	Has the cooperative ever requested financial support to cope with problems caused by extreme climate events?If yes, which ones?If no, why was support never requested?
				34	Cooperative	Did the cooperative receive the requested subsidies? Please explain.
				35	Cooperative	Has the cooperative requested financial support for interruptions of fishing activities caused by climate events?If yes, which ones?If no, why was support never requested?
				36	Cooperative	Did the cooperative receive the requested subsidies? Please explain.

Table 4.2 Risk awareness and past impacts component and the resulting questions of questionnaires for both fishers and cooperative levels. ID columns represent the question ID number in questionnaires.

Theme	Variable	ID	Respondent	Question in this study
Perception, risk awareness and Past Impacts	Perceived enablers (factors that could influence adapting and copying capacities)	40	Cooperative	Have there been factors that enabled your cooperative in implementing adaptation or mitigation strategies in the past? Which ones?
		41	Cooperative	Which factors do you think will enable your cooperative in implementing adaptation or mitigation strategies in the future?
		26	Fishers	Do you consider collaboration/sharing knowledge/resources important for tackling climate change challenges? If yes, how and why?
	Perceived barriers (factors that could influence adapting and copying capacities)	42	Cooperative	Have there been barriers that limited your cooperative in implementing adaptation or mitigation strategies? Which ones?
		43	Cooperative	Which barriers do you think will limit your cooperative in implementing adaptation or mitigation strategies in the future?
		43a	Cooperative	Do you think these barriers can be overcome? If yes, how? If not, why?
	Awareness (factors that could influence adapting and copying capacities, but influenced also by past exposure)	3	Fishers	Do you believe the climate is changing?
		7-9	Fishers	Do you think climate change will affect your work in the future? If yes, what type of effect? Why? Can you give an example?
		17-18	Fishers	Do you think the expected increase in frequency and intensity of extreme heat events will affect your work? Why?
		19	Fishers	Do you think extreme heat events will affect other aspects of your life, such as physical or psychological wellbeing? If yes, how?
	Degree to which activity is affected by past experience (related both to sensitivity and past exposure)	6	Cooperative	Has the [lagoon name] ever experienced environmental changes or events that affected the cooperative's activities?
		9	Cooperative	What type of effect did they have on the cooperative's activities?
		10	Cooperative	In which years did the following extreme events occur that caused damage to the cooperative's activities? What types of impacts/damages occurred?
		2	Fishers	Have you recently experienced environmental changes that affected your work? How did they affect your activity?
		4-6	Fishers	Do you believe that climate change has affected your work activity in/in the [lagoon name] so far? If yes, what type of effect? Why? Can you give an example?
		10	Fishers	Among the following extreme events caused by climate change, which have you witnessed in/in the [lagoon name] where you work?
		11-13	Fishers	If you selected "……", what effects did you experience on your activities in/in the [lagoon name]?
		16	Fishers	Has extreme heat ever affected the way you perform your work activities?

4.2.3. Data collection

For questionnaires at the cooperative level, an e-mail was sent in February to the relevant cooperatives in Sardinia to explain the purpose of the study and to request their willingness to participate. A total of 14 cooperatives/consortia were contacted. Out of these, 8 cooperatives (Fig. 4.2) agreed to participate, resulting in a response rate of 57% (indicated in this study as respondents C1-C8). The questionnaires were then administered face-to-face with cooperatives representatives between February and May 2025, written consent was requested, as well as the possibility of linking the responses to the specific lagoon. At the individual fisher level, questionnaires were administered face-to-face directly in situ in the Cabras lagoon between April and May 2025, approaching fishers during their working hours to ensure accessibility and minimize disruption. Approximately 40 fishers were approached directly in the Cabras lagoon, and 27 agreed to participate (indicated in this study as F1-F27), resulting in a response rate of ca. 68%. An information sheet was provided to all participants, detailing the purpose of the study and how research data would be stored and used, a written consent form and without providing background information on climate change impacts to avoid influencing their responses. All interviews were conducted in Italian and subsequently translated into English for data analysis. Respondents (all over 18 years of age) were informed about the aim of the survey, the voluntary nature of participation, and the confidentiality of their responses.

4.2.1. Data analysis

The quantitative questionnaire data were first entered into two Excel sheets (one for fishers and one for cooperative representatives' responses). The responses were numerically coded to allow basic descriptive quantitative analyses using R. The open-ended responses were inputted to a document (ascribing numerical identifiers to those responses from each participant, thus maintain anonymity).

Thematic analysis (following the process outlined in Braun & Clarke 2021) was then undertaken on this textual data to identify barriers and enablers among fishers and cooperative representatives. This analysis was complemented by additional themes that emerged from respondents' answers to other questions, when relevant.

Figure 4.2 Map of the case studies. In red the socio-ecological systems where questionnaire at cooperative-level were administered. a= Is Benas lagoon; b=Cabras lagoon; Santa Giusta lagoon; Pauli Biancu Turri lagoon; c=Marceddì; d=Porto Pino; e=Santa Gilla lagoon; f=Tortolì lagoon. In dark red, Cabras socio-ecological system, were both cooperative and fisher-level questionnaire were administered.

A semi-inductive approach was used, coding and re-coding data before grouping into broader themes (Braun & Clarke 2006) of barriers or enablers using the NViVo software (Lumivero, 2023). This allowed context-specific themes to emerge, while also being informed by insights from relevant literature on barriers and enablers to climate adaptation (e.g. Maltby et al., 2023; Klein et al., 2014; Islam et al., 2014; Biesbroke et al., 2013).

Barriers and enablers were reported according to the level at which they emerged (i.e. whether among individual fishers, cooperatives, or across both levels, see Table 4.5 and Table 4.6 Enablers identified in results section).

4.3. Results

4.3.1. Overview

According to 2024 internal fisheries statistics, the Cabras consortium comprises approximately 100 affiliated fishers. Therefore, the 27 fisher-level questionnaire interviews in this study covered about 25% of the total consortium members. Fishers interviewed were 49 years on average (median = 50 years), with ages ranging from 28 to 65 years. Regarding their experience, most fishers (10 out of 27, equivalent to 37% of the respondents) have been working in this lagoon for over 30 years, 26% (7 out of 27 less than 10 years). Similarly, in terms of professional experience as fishers, nearly half of respondents have been in this profession for over 30 years (13 out of 27), with only a small proportion having less than 10 years of experience as fishers (3 out of 27).

As for the interviews with cooperatives, the 8 organisations which took part in the study collectively account for most of the total catch from lagoons at the regional level. Moreover, they represent a broad range of cooperative types, differing in number of members, annual revenue, and internal organisational structure, thus providing a sample that is reasonably representative of the socio-economic heterogeneity of Sardinian coastal lagoons small-scale fishery (Figure 4.2).

4.3.2. Sensitivity - Dependence on food provisioning from the lagoons

The interviewed cooperative representatives explained about their varying degrees of dependence on wild fishery resources reflected in the diversity of their supplementary activities. The Tortolì cooperative is engaged in multi-species aquaculture, a restaurant, lagoon tours, and environmental education through a teaching farm. Cabras consortium combines lagoon fishing with product transformation and fishing restaurant, including guided visits to a local fishing museum. Porto Pino has recently added a seasonal food truck to its core fishing operations. S'Ena Arrubia integrates fishing and single-species aquaculture, operates a mobile food truck, and outsources product processing. Santa Gilla consortium combines fishing and single-species aquaculture with sporadic engagement in cultural events related to lagoon and fishery and a food truck. Finally, other cases show more limited diversification: one offers occasional guided tours; another is entirely reliant on fishing; and one complements fishing with recreational sport fishing services. These cases highlight how economic dependence on natural fishery resources remains strong across most cooperatives, although expressed through varying degrees of activity diversification.

Regarding fishers in the Cabras lagoon, different levels of natural resource dependence can be observed. Although most fishers (24 out of 27) are engaged in activities other than lagoon fishing, their livelihoods still largely depend on fishery resources - both in the lagoon and at sea. This dependency increases their sensitivity to climate-related impacts and may be further exacerbated if their adaptive or immediate response capacities are limited. Two fishers rely solely on lagoon fishing as their only source of income, highlighting a very high sensitivity to climate variability and change due to their strong dependence on lagoon wild resources.

4.3.3. Capacity to adapt and cope

When asked about collaborating or sharing knowledge and/or resources to discuss and address challenges or problems related to their sector, most respondents reported frequent communication and knowledge exchange within the cooperative, both during work and in informal settings. This dialogue often revolves around fishing practices, observed environmental changes, and potential solutions, reflecting a strong sense of shared identity and collaborative problem-solving, pointing to the strength and support of their social networks. Some fishers highlighted the importance of curiosity and mutual support:

"*Curiosity and sharing are very important to face problems and understand how fishing is changing*." (F3)

However, a few participants indicated a reduction in social interactions in recent years, partly attributed to the COVID19 pandemic and increased workload, suggesting that structural and contextual factors may affect social capital over time and in turn, diminish fishers' capacity to cope and adapt to recent changing circumstances.

Similarly, the elements related to capacity to adapt, and cope analysed at the cooperative level reveal differences among them but suggest a general low capacity. Regarding formal networks and technical/scientific support, the majority of cooperative representatives reported only general availability of assistance related to ecosystem management or business diversification, without a specific focus on climate change. However, two cooperatives reported participation and involvement in initiatives explicitly addressing climate-related challenges: one involved the pilot installation of real-time monitoring buoys for key physical-chemical parameters to made available to end-users (fishers) real-time data about unfavourable conditions and enabling prompt responses within the e.INS project (Spoke 9, University of Cagliari, Sardinia), while the other engaged in pilot TransformAr project implementing automated sluice gate systems to enhance lagoon coping capacity, since the infrastructure is designed to activate automatically in response to the detection of anomalies in lagoon conditions. Similarly, when considering access to subsidies or financial compensation for damages caused by extreme climatic events, only three cooperatives reported having applied for such support. However, all noted that the amount of subsidies received was significantly lower than their perceived losses.

4.3.4. Climate change awareness, past and expected impacts

When asking Cabras lagoon fishers about their views about climate change, most respondents (26 out of 27) declared that climate is changing. One fisher, who reported not believing in anthropogenic climate change, nonetheless acknowledged observing changes in the weather, which he interpreted as part of natural climatic cycles. Another fisher also affirmed that the climate is changing due to natural phenomena rather than anthropogenic stressors. Almost all fishers (24 out of 27) reported that climate change is already affecting their fishing activities. Among them, most described the impacts as negative, while one reported experiencing both positive and negative effects. The remaining three respondents reported not experiencing any impact of CC on their activity so far. Similarly, 89% of respondents believe that climate change will affect their work in the future. Nearly all of them anticipate negative impacts, except for one fisher who stated that he was unsure of what to expect. Most fishers reported having experienced at least one extreme event in the lagoon, while one fisher reported having experienced two, and only one stated that he had never witnessed any. Extreme heat was cited by all respondents who reported experiencing extreme events, with one fisher additionally mentioning a flooding event. One respondent reflected on how the duration of heatwaves has changed, noting that "heatwaves have always existed, but now they last much longer than in the past: what used to be a few days can now stretch for weeks." He also pointed out that "rainfall has become much less frequent", suggesting a broader perception of shifting climatic patterns affecting lagoon dynamics. Another fisher shared his perception that "there seems to be a shortage of water and less freshwater coming in, also because of agricultural water rationing," indicating how local agricultural practices might be influencing water availability in the lagoon from his point of view. When asked about observable changes in the climate in the 8 Sardinian lagoons, responses at cooperative level most frequently reported heatwaves events (N=7), followed by flooding events (N=4), and storm surges (N=3), all attributing these phenomena to climate change. Respondents who had observed extreme heat events in their lagoon described them as having changed over time in both frequency, intensity, and duration. While several participants acknowledged that heatwaves have always occurred to some extent, they consistently emphasised that recent events differ significantly from past experiences. The respondent for Cabras lagoon also reported more severe impacts exacerbated due to lagoon morphology highlighting how local environmental characteristics and conditions, such as shallow depth and limited water circulation, can amplify the ecological impacts of climate stressors:

"There have always been hot spells, but due to the lagoon's shallow, still waters, that heat now has more impact." (F7)

A cooperative respondent commenting on flooding events noted: "Before, [floods] it happened, but the difference is: Now the climate has changed, the whole system has changed, all the season [changed]. Water comes all at once. A flood today brings as much water as we used to get in an entire year." (C4)

Similarly, another respondent remarked on the increasing intensity and persistence of flooding:

"It used to rain regularly and moderately, and the lagoon could handle it. Now it stays like that for a month and a half, the water just doesn't drain anymore." He added, "The seasons don't exist anymore." (C3)

4.3.4.1. Impacts

Cooperative representatives reported that heatwaves led to hypoxia in the lagoon, causing near-mass mortality events. These extreme events were also associated with damage to essential equipment. Such events were sometimes averted by favourable winds or extraordinary management interventions. Similarly, Cabras fishers consistently reported that both temperature-related climatic changes are affecting fishing activities in multiple ways. Overall temperature increases were reported to impact the timing of key species' life cycles, including eel migrations and mullet spawning for *bottarga* production. Warmer conditions have also been associated with longer periods of cormorant presence, changes in species composition, and favoured increases in species such as the sea walnut (*Mnemiopsis leidyi* A. Agassiz, 1865; Figure 4.3), and the polychaete worm *Ficopomatus enigmaticus* (Fauvel, 1923) that clogs supply channels thus exacerbating the reduction of oxygen levels during heatwaves. Heatwaves were reported by the fishers to be repeatedly associated with near-mass mortality events, highlighting the ongoing risk of fish die-offs in the lagoon. Increased salinity due to reduced freshwater input has been reported to favour the proliferation of marine and alien species and higher abundances of cormorants. Storm surges and floods have repeatedly caused substantial damage to infrastructure such as working piers and fishing barriers.

The direct impacts of climate change, of invasive alien species (IAS, whose occurrence has been also associated with increasing temperatures), and other impacts and damages reported by the cooperative representatives are summarised in Table 4.3.

Figure 4.3 M. leidyi in S'ena arrubia lagoon (Photo provided by Cooperativa Sant'Andrea; Ph. Alessandro Porcu)

Figure 4.4 Flooding event in S'Ena Arrubia lagoon in 2020 (Photo provided by Cooperativa Sant'Andrea; Ph. Alessandro Porcu).

Table 4.3 The climate change (CC) and invasive alien species (IAS) Impacts reported by cooperatives representatives.

Stressor	Typology	Impact category	Description	N.
CC	Heatwaves	Near-mass mortality events	Near mass mortality events or mortality events, associated with hypoxic condition due to heatwaves	5
		Damage to equipment	Heatwaves damaged the cooperative's cold storage facilities	1
		Market	Fishers are forced to sell bivalve earlier in the season (by July), when market prices are lower, to avoid keeping bivalve in the lagoon during heatwaves that could cause mass mortality, resulting in reduced income.	1
IAS	*C. sapidus*	Damages to fishing gear	Broken/torn nets, fyke nets damaged (*bertivelli*), other gear damage	6
		Reduction of catches	Decline in fish and clam catches due to predation	5
		Non-marketable catch	Mechanical damages on fishes rendering the product unmarketable	2
		Fishers	Occasional minor wounds while fishing by hand	1
	M. leidyi	Fishing effort and catchability	Clogging or compromising eel traps, nets filled with sea walnut, changes in fishing efforts due to decline in catch and increasing in workload due to cleaning procedures	6
		Fish quality	Decrease in fish quality due to competition for food resources	1
		Reduction of catches	Decline in fish and bivalve due to predation on eggs and larvae	2
	F. enigmaticus	Damage to fishing gear and other equipment	Nets torn, fishing gear compromised, damage to boats and motors	3
		Navigation and access issues	Obstruction of canals and lagoon which causes difficulties in boat passage and slowing of fishing activities	3

When asking about past effects of intense heat on fishing activities, fishers reported a variety of impacts, both on work activities and on physical and psychological well-being (Table 4.4). Approximately 22% of respondents reported that their work activities are generally planned based on temperature, while 1 respondent declared that he has had to interrupt working activity due to high temperatures.

Physiological effects were the most reported, with 33% of fishers experiencing symptoms such as physical fatigue, loss of strength, heat-related tiredness, headache, blurred vision, or fainting. This highlights that intense heat can pose significant health risks to fishers, potentially compromising both safety and productivity. Psychological effects, although reported only from one respondent, were also noted, with stress being the only manifestation mentioned. General discomfort, which could encompass both physical and psychological unease, was reported by 2 respondents, indicating that heat can reduce comfort and well-being even if the heat does not force them to stop working. 11 respondents reported no impact of intense heat on their activities. It should be noted that two of the respondents reported never being affected by intense heat, as their working hours do not involve outdoor activities during the hottest periods of the day.

Table 4.4 Typology of impacts of intense heat during fishing activities by fishermen taking part in the research. Note: the sum is not 100% because respondents could report more than one impact

Category	Description of effect	Number of fishers reporting this
Working activity	Working activities planned according to temperature Working activity interrupted	6 1
Psychological effects	Mental effect referred as “stress”	1
Physiological effects	Physical fatigue/Loss of strength/Weakness/low blood pressure; Heat-related tiredness; Headache; Blurred vision; Fainting	9
General discomfort	General discomfort that could be both physical and psychological effect reported as “*fastidio*”	2
No impact reported	No effect reported related to intense heat	11

When asked whether an increase in the intensity and frequency of extreme heat events is expected to affect their fishing activities and why, responses varied across the 27 interviewed fishers. The majority of respondents (18 out of 27) reported that they anticipate impacts on their work. Six respondents stated that they do not expect impacts, although some expressed uncertainty about the future:

“*No, it’s subjective, but it depends on age, we will see in the future*” (F13)

Similarly, those who answered ‘I don’t know’ also expressed uncertainty:

“*I don’t know. Because beyond a certain point I think it can cause problems. I know they also introduced a law for workers on construction sector, to prevent them from working during the hot hours. Then the years go by, and one has to see whether with age one can withstand the heat less*” (F15)

Moreover, some fishers emphasized concerns about the potential effects of heat on the fish species they catch. Similarly, when asked whether extreme heat events are expected to affect their wellbeing, most of them (19 out of 27) expected that intense heat negatively affects their wellbeing. The responses reveal that intense heat can have multiple impacts on fishers' wellbeing. Many respondents reported general discomfort without specifying concrete effects, some also described possible impacts - already experienced - on daily habits and leisure activities. For example, several reported reactive responses, such as modifying their schedules, staying indoors during peak heat, or avoiding recreational activities like swimming or going to the beach. Some responses also expressed concerns for environmental consequences, such as the link between high temperatures and wildfire risk or altered environmental conditions.

4.3.5. Barriers and enablers toward adaptation and coping capacity

Drawing on information from both cooperative representatives and individual fishers, the questionnaire findings highlight several barriers that hinder respondents' capacity to adapt and cope with CC, as well as enabling factors that could support them toward adaptation. In this section, I explore these perceived barriers and enablers, as expressed by the participants and further interpreted through the lens of this study, to better understand the dynamics shaping adaptation within fishers and cooperatives.

The thematic analysis of questionnaire responses allowed me to classify the barriers into five main domains: institutional and governance factors; social capital; economic and financial assets; perception, awareness and agency; and knowledge and technical skills.

Within the institutional and governance theme, both groups highlighted limited institutional participation and support, unclear regulations, and complex bureaucratic and authorization processes. These factors were perceived as discouraging proactive adaptation, delaying access to resources, and constraining the flexibility of both individual fishers and cooperatives. Some respondents also felt that certain stakeholders or actors had greater influence in decisions, which led to decisions being made for their own benefit rather than collective benefit.

Environmental regulations and resource use conflicts were also seen as limiting factors for fisheries activities, while insufficiently evidence-based or outdated norms further reinforced a sense of misalignment between management decisions and local realities.

Barriers linked to social capital emerged in terms of distrust and collaboration challenges. Fishers and representatives reported a sometimes presence of lack of mutual confidence, and general skepticism towards collaboration, perceived as inherent to the condition of being a fisherman, associated with individualism. Institutional distrust was more commonly present, with formal authorities often perceived as unresponsive, unfair, or lacking legitimacy.

In the economic and financial domain, structural constraints were poorly highlighted, including limited financial capacity at cooperative level, reducing opportunities for investment in adaptation measures, but also for coping with climate related emergencies.

Finally, barriers related to perception, awareness, and knowledge were evident, and somewhat also related to institutional barriers. Fishers expressed feelings of powerlessness, marginalization, and abandonment, weakening their sense of agency. Some participants felt indeed that they had little or no voice in the decision-making processes related to coastal lagoon management.

Limited collaboration with research institutions and a perceived lack of technical-scientific support further reduced adaptive capacity, while low problem-solving capacity within communities was also noted.

Building on the identification of barriers, my analysis also highlighted several enablers of adaptation as perceived by both fishers and cooperative representatives. In general, the reported enablers have been somewhat presented by the interviewed as the positive counterpart to the perceived barriers (i.e. what was seen as a barrier when absent was conversely perceived as an enabler when present, as in the case of the institutional participation/support). These enablers were grouped into six broad domains: institutional and governance factors, social capital, perception and agency, knowledge and technical skills, environmental characteristics and management, and other adaptation strategies.

In the institutional and governance domain, stronger participation and support from public institutions, including improved engagement in decision-making, technical assistance, and financial support, were seen as crucial for enhancing adaptive and coping capacities. Simplifying bureaucratic and authorization procedures was also identified as an important enabler, reducing administrative burdens and facilitating access to new opportunities. Additionally, establishing evidence-based norms and improving internal governance within cooperatives were perceived as key measures to strengthen trust and effectiveness in fisheries management.

Regarding social capital, bridging networks and promoting collaboration within and among communities were highlighted as important enablers for bringing new ideas and solutions, and making a collective decision-making toward adaptation and the resolution of common problems. In fact, what emerged from the interviews, both representatives and the fishermen, was a sense of shared problems, a factor that connects and unites them beyond geographical or territorial divisions and possibly overcoming any individualism or potential internal tensions.

Knowledge and technical skills also emerged as critical enablers. Technical-scientific collaboration, access to relevant knowledge, and adoption of technological innovations were identified as factors to improve adaptive capacity, increase efficiency, and respond effectively to changing environmental conditions. This is the case, for example, of technologies such as environmental sensors, which enable real-time monitoring and prediction of anoxic conditions caused by rising temperatures, cited from one of the respondents.

Environmental characteristics and management also are perceived to play possible enabling roles: being in relatively undisturbed or healthy ecosystems was indeed perceived as a prerequisite for enhancing resilience, while management actions aimed at restoring lagoon conditions, such as improving water circulation, support both ecological and socio-economic adaptation.

Finally, adaptive strategies, such as diversification through aquaculture or species restocking, were cited among the possible enablers, to reduce dependency on vulnerable stocks and respond flexibly to environmental changes.

Overall, among the barriers and enablers identified, those related to the institutional and governance themes were mentioned more frequently and as more urgent, as also highlighted from the number of codes within this category compared to the other themes.

Table 4.5 Barriers to adaptation identified from the questionnaire analyses

Theme	Code	Description	Respondents
Institutional and Governance Factors	Limited institutional participation/support	Perceived limited involvement or support from public institutions, including a lack of engagement in decision-making processes, assistance, or financial support and subsidies for adaptation measures and copying capacity. *C: "If the funds arrive late, recovering also takes longer"*	Both
	Regulatory ambiguity	Confusing or unclear regulations that create uncertainty about responsibilities, compliance requirements, and permitted actions, discouraging proactive adaptation	Cooperatives
	Complex bureaucracy & authorization processes	Time-consuming and inefficient administrative procedures that delay or complicate the implementation of adaptation strategies, infrastructure improvements, or access to funding. *C: "The timing of procedures and bureaucracy for emergency issues is long"*	Cooperatives
	Environmental regulations and conflict of uses	Rules and restrictions aimed at environmental protection are perceived as limiting factors for fishing or multiple-uses activities in the lagoons, and as constraints to flexibility or adaptation options. *C: "For mercerella [F. enigmaticus], a limiting factor was not being able to move it for environmental restrictions, therefore not allowing us to solve problems in the lagoon"*	Cooperatives
	Perceived insufficient fact-based norms	Management decisions and regulation viewed as lagging in their responsiveness or being out of sync with fishers' experiences. Science and information used in management decisions are perceived to be inadequate, leading to decisions based on inconsistent criteria, subjective interpretations, or outdated practices.	Cooperatives
	Internal Governance	Related to internal governance factors perceived as barriers *F: "It is difficult to make decisions because there are so many of us."*	Fishers
Social Capital	General distrust	General distrust in collaboration	Both
	Institutional distrust	Lack of confidence in formal institutions, such as government bodies, regulatory agencies, and fisheries management authorities. It reflects the perception that these institutions are not acting in the best interests of the community, are unresponsive, unfair, or lack legitimacy.	Both
Economic and Financial assets	Financial	Economic and financial constraints related to limited economic internal capacity of the cooperative *C: "There are no economic conditions to be able to do it; we are a small cooperative"*	Cooperatives
Perception, Awareness, and Agency	Feeling powerless	Fishers' perceptions of lacking agency; that is, little or no control or influence over decisions and changes affecting their livelihoods *F: "The problem is that the decisions that truly affect fishing are not made by us."* *C: "It doesn't depend on us"*	Both

Theme	Code	Description	Respondents
	Perceived abandonment and marginalization	The belief that institutions or broader society have neglected or excluded the fishing sector or specific communities, reinforcing feelings of isolation, social injustice, and resistance to top-down adaptation strategies. *F: "No one helps us. We are left on our own."*	Both
Knowledge and technical skills	Limited technical-scientific collaboration and support	Low level of support and collaboration from research institution can lead to low adapting and copying capacities	Cooperatives
	Low problem-solving capacity	Perceived low knowledge capacity to solve problems *F: "We know what the problem is, but we don't know how to solve it."*	Fishers

Table 4.6 Enablers identified from the questionnaire responses using thematic analysis. Note that some of the themes reflect those also identified for barriers (see previous table).

Theme	Code	Description	Respondents
Institutional and Governance Factors	More institutional participation/support	More involvement and support from public institutions, including improved engagement of fishers in decision-making processes, technical assistance, or financial support and subsidies for adaptation measures and copying capacity.	Cooperatives
	Simplify bureaucracy & authorization processes	Simplifying bureaucratic procedures and authorization processes was identified as a potential enabler of adaptation and copying capacities, by reducing administrative burdens and making it easier for fishers to access new opportunities or implement changes.	Cooperatives
	More evidence-based norms	Establishing norms and regulations grounded in scientific evidence is perceived as crucial to improve management effectiveness and build trust between fishers and the government, facilitating better adaptation to environmental changes.	Cooperatives
	Internal governance	Improving internal governance within cooperative perceived as enablers	Cooperatives
Social Capital	Bridging social capital for solutions	Social networks are important to have a major role in decision-making *F: "Discussing with others always brings new ideas and solutions; ideas merge, creating innovative outcomes."* *C: "We created a committee with the fishers. We joined forces with the trade associations and with other cooperatives".*	Both
Perception, Awareness, and Agency	Recognition of their own role	Recognition of the role of fishers in managed in a good state the resources and ecosystems C: "We don't have the concession just to fish; we also have to know how to maintain it in good condition and take care of it"	Cooperatives
Knowledge and technical skills	Technical-scientific collaboration and support	Technical and scientific collaboration and support as key enablers for adaptation, providing fishers with access to relevant knowledge needed to adapt.	Cooperatives
	Technological innovation	Adopting new technologies as a key factor to enhance their adaptive capacity, improve efficiency, and better respond to changing environmental conditions. *C: "Technological innovation that helps us address the issue of hypoxia related to high temperatures"*	Cooperatives
Environmental characteristics and management	Characteristics of the ecosystem	Being located in a relatively undisturbed or healthy ecosystem was identified as an enabling factor, as it supports the resilience of fisheries resources and provides more stable conditions for adaptation *C: "We have an ecosystem that is still largely intact"*	Cooperatives
	Management of ecosystem	Environmental restoration of lagoon conditions involves actions aimed at recovering the ecological health, water quality, and habitat integrity of the lagoon system. This process enhances the resilience of the resource system by improving the natural environment that supports fisheries and associated livelihoods.	Cooperatives
Other	Diversification (cited as enabler but is an adaptation measure)	Diversifying activities with aquaculture or restocking of species seen as a key enabler, allowing fishers to respond to changing stock availability and reduce dependency on vulnerable resource	Cooperatives

4.4. Discussion

Vulnerability is increasingly recognized as a cross-scale issue in CC literature (Peterson, 2000), is “tied to local history, social relations, and place” (Adger et al., 2009), highlighting that is a concept nested across different levels and interconnected in complex, and often non-linear dynamics (Biggs et al., 2012; IPCC, 2014; IPCC, 2022). Indeed, local-level dynamics and phenomena may be shaped or constrained by broader governance and regulating settings (Adger et al., 2009), with decisions and processes at higher levels cascade down to individual or community adaptive capacity and vulnerability perception, creating feedback loops within the whole socio-ecological system (Adger et al., 2009).

Although my analysis identified several themes related to barriers and enablers of adaptation, it is important to acknowledge that some relevant aspects related to fishers may not have emerged. This is primarily because fishers were not asked direct questions on barriers and enablers, but these questions were explicitly included only in questionnaire for cooperative representatives. This decision was intentional, as interviews were conducted during working hours and I sought to minimize survey fatigue by limiting the length in time and complexity of the questionnaire. As a result, the findings on barriers and enablers mainly reflect perspectives captured indirectly through other questions, rather than a comprehensive exploration of these factors.

Overall, considering the economic level of dependence from food provisioning ES, the results of my study indicate that while some cooperatives exhibit a certain degree of diversification through complementary activities, such as aquaculture, tourism, food services, food processing, their primary economic income remains strongly rooted in coastal lagoon fishery resources. According with the small-scale fisheries literature, this high dependence can be related to higher degree of economic damages, especially if considering the unprofitability of some fisheries without subsidies, with broader impacts on employees and wages (Sala et al., 2018). Similarly, the findings at fisher’s level highlight high economic dependence from fishery resources, since most of them are only engaged with fishing activities. These findings suggest that focusing only on the cooperative/consortia level may overlook important intra-group variability - as exhibited by the fishers’ responses - and highlights the value of exploring individual livelihood strategies to better understand overall vulnerability of the socio-ecological system.

Furthermore, this economic dependence can also be exacerbated when some barriers constrain their capacity to adapt (Allison & Ellis, 2001; Cinner et al., 2018) and can have effects on fishers and cooperative copying capacities. Overall, the findings of my study therefore suggest the cooperatives might have limited internal capacity to buffer potential income fluctuations caused by environmental shocks.

My findings suggest also that, according to cooperative representatives, while formal networks and scientific collaborations exist, their contribution to the climate change adaptive capacity of cooperatives remains to date limited, as most support focuses on general environmental management rather than targeted responses or adaptation strategies to climate risks. The few cases involving CC-related specific pilot initiatives, such as real-time monitoring systems and automated sluice gate management, highlight the potential role of research-driven projects in enhancing preparedness and resilience, yet their implementation appears sporadic and geographically uneven. In coastal lagoons like those under scrutiny in my study, adaptation could be fostered by enhancing more applied research, for example by developing site-specific strategies and technologies, based on the specific needs of the social groups and tailored to the ecological characteristics of the lagoon. Similarly, by operationalizing monitoring efforts and pilot studies into early-warning systems available for fishers can help copying strategies and, at the same time, provide valuable time-series data to scientist and decision-makers to build a more evidence-based knowledge.

Consistently with previous literature, findings from my study reveal widespread awareness and concern about climate change among fishers and cooperative representatives. However, such awareness often could not translate into meaningful individual or cooperative-level engagement in adaptation, nor in concrete actions (Maltby et al., 2023).

My study identified elements which the fishers and cooperative representatives perceive to impede or facilitate their ability to adopt effective adaptation or coping measures and / or mechanisms. As outlined above, my study is the first of this kind undertaken in Sardinian lagoons. In the climate adaptation and vulnerability scientific literature, analysing the barriers and enablers to adaptation is considered a key aspect for understanding the factors that influence the success or failure of adaptation efforts and for guiding future policy decisions (Eisenack et al., 2014; Lee et al., 2022; Klein et al., 2014). Barriers are generally conceptualized as either the reason, for specific actors in a particular context, for capacity not being translated into action (Adger, 2009; O'Brien et al., 2006) or as the basis of low adaptive capacity (Einesack et al., 2012). Barriers are also perceived differently by different actors, and notably are well distinct from limits to adaptation (Eisenack et al. 2014)

Concurrently with the wider literature, here I provided findings on the key barriers and enablers of climate adaptation and coping capacity, included but not limited to social, economic, financial, governance and institutional elements (Adger et al., 2009; Galappaththi et al., 2021; Islam et al., 2014; Maltby et al., 2023; Morgan, 2011).

While this analysis only captures perceptions from fishers and their representatives, these perceptions reveal insights spanning multiple levels of governance, covering cascade aspects of the broader socio-ecological system. The findings suggest that institutional barriers are perceived to play a key role in constraining respondents' ability to adapt to environmental and climate stresses. Importantly, when institutions are perceived as responding inadequately to changing conditions and emerging risks, this can increase the overall vulnerability of the system (Adger, 2006). In some way, the lack of trust in institutional decisions, perceived as "distant" and not evidence-based in contrast to the experiences and knowledge of the participants in the studied socio-ecological systems, recall the concept of place attachment (i.e. an emotional connection to certain places or landscapes, involving both their physical characteristics and social dimension; Devine-Wright, 2013; Levicka, 2011), even if not explored in this chapter. Other studies found that this "sentiment" was strongest among those perceiving governance processes as weak (Clarke et al., 2018) and demonstrated a strong link among place attachment of individuals and the importance they attribute to participate in governance processes (Mesch & Talmud, 2010). These point to the suggestion for future research to explore how the sense of place attachment interacts with communities' self-perceived knowledge of their local environments, and the ways in which these affects their views about and perceived role in governance processes and decision-making.

Moreover, as other studies indicated, different barriers cannot be fully understood or captured in isolation (Eisenack et al., 2014), since barriers can evolve over time (Adger et al., 2009; Biesbroek et al., 2013) or could interact to further impede and constrain adaptation (Biesbroek et al., 2013; Islam et al., 2014); for example, the interlinkages between the perceived limited support from institutions and the institutional distrust that both fishers and cooperative representatives mentioned.

The erosion of trust often represents a long-term process, extending beyond fisheries institutions to include broader regional governance arrangements (see Dixon et al., 2024 about rebuilding trust in UK fishery). As previous authors suggest, actions that promote and strengthen more effective regional and adaptive governance can provide a foundation to building trust in fisheries governance, through more face-to-face interaction, enhanced democratic processes, and a stronger focus on resolving local issues (Dixon et al., 2024). However, when distrust has already emerged, as in this case, rebuilding trust can be substantially more complex (Dixon et al., 2024).

Capacity to adapt is also constrained by economic barriers (as found by Adger et al., 2007). Although the internal financial barrier of cooperatives was not frequently mentioned explicitly, it emerged as a potential important barrier needing future investigation, because many respondents referred to difficulties in obtaining subsidies. In other words, the need to request financial support from institutions can hide the underlying constraints in their own internal financial capacity, linking internal resource limitations to institutional-level barriers found. This typology of “resource-based” barriers (Moser & Ekstrom, 2010; Klein et al. 2014) can be closely linked to the economic viability of small-scale fisheries (Sala et al., 2018), as the cooperatives’ ability to maintain operations, diversify activities, or invest in adaptive measures depends on their net benefits over time (Schuhbauer & Sumaila, 2016). However, some argue that their presence does not necessarily mean there is a need for a greater (economic) capacity, but rather that the actual and available resources can be better utilized (Biesbroek et al., 2013; Burch 2010).

An example of how existing environmental resources and assets have been capitalised and used for the maintenance of the resource itself is evidenced, for instance, by projects such as the one in Valle Dogà, Venice (Italy; https://www.bluev.it/copia-di-le-nostre-attivit%C3%A0). There, a company engaged in extensive aquaculture of fish species assessed the ecosystem's carbon capture and storage capacity, producing certified carbon credits (Verified Carbon Unit) that can be sold on the voluntary market, thereby funding further sustainable actions on the environmental system they manage.

Moreover, the issue highlighted by the interviews is consistent with findings in the scientific literature, which acknowledge that while government relief may be provided, such economic support often takes a long time to materialize (Bellquist et al., 2021). This delay, combined with the fact that compensation rarely matches the full extent of the damage, can exacerbate vulnerability and limit the capacity to adapt of small-scale fisheries.

This research further shows that cooperatives often face difficulties in quantifying losses, particularly for damages that have cascading effects over subsequent years, for example, the loss of juvenile fish, which may lead to reduced income in the following years. Nonetheless, the results of this chapter should also be read considering the specific socio-political context in which this research took place, as clarified in section 4.2.1. The mobilisations in which fishing cooperatives were engaged (before the summer season, when concerns were particularly strong about the potential consequences of high temperatures in the absence of channel maintenance works) may have amplified negative perceptions and especially the narratives of barriers as see in results section, even if, in the questionnaire for fishers' representatives, this potential bias was partially limited by first asking about enablers.

The thematic analysis showed (within the theme of Perception, Awareness, and Agency) that individual forms of agency ("autonomous, purposive and creative actors, capable of a degree of choice"; Lister, 2004) are important and can act as both barriers (such as feelings of powerlessness and perceived marginalization) and enablers (notably the recognition of one's own role were highlighted). Consistently with other studies, my findings underline indeed the emergence of a "social concept of agency" (Adger 2003; Ostrom et al. 1999) within the theme Social Capital, where the enablers to capacity to cope and adapt materialize and express through collective action, cooperation, and networks of trust, which can transform individual constraints into shared solutions. In this sense, Fisheries Local Action Groups (FLAGs) instituted in Sardinia under the 2014-2020 EMFF programming period, as well as associations for the protection and representation of cooperatives, have and can continue to, for example, playing an important role in shaping and supporting fisheries' participation in decision-making processes within the Sardinian sector (Madau et al., 2018).

Finally, when analysing socio-ecological systems, it is essential to consider how adaptation strategies affect both ecosystems and human well-being (Coulthard, 2012). Adaptation strategies may simultaneously generate new forms of harm, maladaptation or unintended harms, which must be carefully taken into consideration in policies promoting adaptation as a pathway to risk reduction (Coulthard, 2012). For example, reliance on subsidies may increase external financial dependency, while diversification strategies still link on fishing resources could create only an illusion of adaptation. Indeed, in the fisheries sector already heavily dependent on subsidies, the emergence of mandatory insurance schemes introduces only a partial “cure” (for example in Italy), providing a sort of economic protection, but at the same time reinforcing structural dependence on external support and without removing the underlying vulnerability.

4.5. Conclusions

The research fueling this chapter was driven by the need to improve our understanding about the different dimensions of vulnerability within the socio-ecological system of coastal lagoons in Sardinia as no studies of these existed. This study provides an initial picture of the vulnerability of food provisioning services in Sardinia lagoon socio-ecological systems, highlighting the key role that the local context can play in shaping vulnerability and perceived vulnerability, also considering different social levels. While vulnerability at the cooperative level can reflect structural sensitivity components of the cooperatives as well as governance and social capital dynamics that interact with other structural factors to shape adaptive and reactive capacities, shifting the focus to the individual fishers helps to broaden our understanding at a level that is rarely captured when examining the organization as a whole.

Indeed, even when decisions are made at the cooperative level, it is important to assess vulnerability at a finer scale, as governance and social capital dynamics are inevitably shaped by the individual actors within the cooperative, who directly interact with the surrounding environment. However, further research is needed to broaden the scope, for instance by considering the vulnerability of indirect beneficiaries of ES such as the broader coastal lagoon community, the consumers and other economic activities linked to the fisheries sector.

Moreover, as my results are based on a limited set, even if very rich in contents, of interviews, I encourage further research to examine barriers and enablers within other fishing communities in Sardinia through participatory approaches and consider these in relation to the broader socio-ecological contexts.

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Chapter 5. Monitoring ecosystems and their services in a changing climate: Experimenting the SEEA EA at the local and regional scales in Sardinia

This chapter was developed in collaboration with the School of Environmental Sciences at University of East Anglia (UEA) and the Centre for Environment Fisheries & Aquaculture Science (Cefas) under the supervision of Dr. Tiziana Luisetti and Prof. Irene Lorenzoni during the period spent abroad hosted by UEA and Cefas at the Collaborative Centre for Sustainable Use of the Seas (CCSUS).

5.1. Introduction

Ecosystem accounting has emerged as a powerful tool for integrating ecological and economic information, providing a framework to systematically assess the contributions of ecosystems to human well-being and sustainable development.

The System of Environmental-Economic Accounting Central Framework (SEEA CF) was adopted by the UN Statistical Commission in 2012 as the international standard for environmental-economic accounting (United Nations, 2014b) and focused on the connections between the economy and the environment via material flows and asset accounts, and related economic transactions (Edens et al., 2022). The scope of the SEEA CF excludes ecosystem services (ES) and does not address various valuation aspects, such as the costs associated with ecosystem degradation. Consequently, ecosystem characteristics, functional interactions, and spatial dimension of natural capital fall outside the scope of the SEEA CF (Comitato Capitale Naturale, 2022).

As a response to the growing policy demands for integrated approaches to measure ecosystems and their relation to the economy and well-being, the SEEA - Experimental Ecosystem Accounting (SEEA EEA; United Nations, 2014a) was initially conceived to extends and complements the SEEA CF by prioritizing ecosystem services and the conditions of natural assets (Comitato Capitale Naturale, 2022; Edens et al., 2022). After a piloting phase and several revision processes, the SEEA - Ecosystem Accounting (SEEA EA; Edens et al., 2022) was finalised in 2024 and is now a globally recognized framework that integrates ecological data with economic information to provide key information to well-manage the ecosystems (United Nations, 2025). It must be noted that the SEEA EA chapters 8 to 11, which deal with monetary valuations, were excluded from the recognition of international statistical standards, and are recognized only as statistical principles (Edens et al., 2022), due to the considerable methodological challenges that the statisticians community still needs to resolve (Femia & Capriolo, 2022).

The SEEA EA methodology, initially adopted in its draft form in March 2021 by the UN Statistical Commission as the official international statistical standard covering the first seven chapters (i.e., accounts that define the physical aspects of ecosystem assets and the services they provide; Comitato Capitale Naturale, 2022), can indeed play a central role in organising and synthesising basic ecological and economic data to produce key indicators and evidence across all stages of the policy cycle (King et al., 2024; Vardon et al., 2016). Measuring and tracking changes in stocks and flows of ecosystem assets will enable policymakers to make informed decisions that reconcile economic growth with environmental conservation at different governance levels (Grilli et al., 2021).

While more than 90 countries have established or are developing SEEA (UNSC, 2025; United Nations, 2021), most of the attention in economic-environmental accounting has been directed toward the terrestrial domain (Gacutan et al. 2022). In Europe, publicly available accounts for marine ecosystems include those of the UK, the Netherlands, France and initial OSPAR accounts. The application of SEEA EA to marine-coastal waters remains limited (Chen et al., 2020; but see Silva et al., 2024, Bordoni et al., 2023, Rigo et al., 2021 for natural capital assessment in marine ecosystems), particularly regarding the evaluation of ecosystem condition. This holds true particularly for coastal waters and transitional aquatic ecosystems (Dvarskas et al., 2019; Gacutan et al., 2022) such as coastal lagoons (Gaglio et al., 2024). While the SEEA EA is generally meant to be adopted at the national level, most ecosystem management decisions are made at the regional or local scale. Recognising this, the need for further research on how to operationalise accounts for policy and management purposes emerges particularly at regional and local scales.

This study seeks to contribute to this growing body of scientific efforts, by exploring the methodological development of the SEEA EA framework at the sub-national scale (regional and local). More in details, here, I explore the methodological challenges related to the compilation of SEEA EA in Sardinia (Italy) using available data sources, with a specific emphasis on the (sea) food provisioning ecosystem service in coastal lagoons. With this chapter my aim was to focus on identifying challenges and the extent of data gaps, wherever present, and to offer insights on data needs and on the potential of SEEA-based accounting in supporting and informing identified relevant sectoral policies at regional scale.

5.2. Accounting principles in the SEEA EA framework

Ecosystem Accounts consist of multiple linked accounts that present ecosystem-related values in either bio-physical units or monetary terms (see Figure 5.1). In 2024, the Regulation (EU) 2024/3024 introduced ecosystem accounts among the mandatory accounts that member states must compile.

These accounts are tables in which either stocks or flows between ecosystems, society, and the economic actors are assessed. Each spatial area of a particular ecosystem type is described as an ecosystem asset, characterized by a unique set of abiotic and biotic components and their interactions (United Nations, 2025).

The ecosystem stocks and their changes over time are measured via ecosystem extent and condition accounts. The ecosystem services accounts organise information on the supply of ecosystem services by different ecosystem types as physical and monetary flows over a certain time of interval (i.e. accounting period), which can be minimum a year, but can span over few years depending on the ecosystem service considered (United Nations, 2025).

According to the SEEA-EA guidelines (United Nations, 2025), ecosystem accounts include tables for the ecosystem extent, ecosystem condition, physical ecosystem service flow, monetary ecosystem service flow, and monetary ecosystem assets (see Figure 5.1).

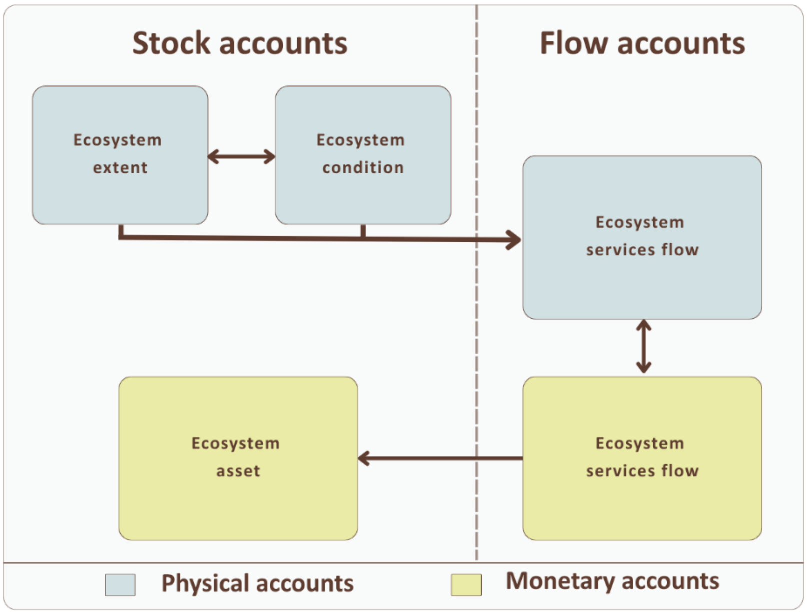

Figure 5.1 Core SEEA EA Accounts. Adapted from United Nations 2021)

5.2.1. Physical accounts

Ecosystem extent accounts provide important insights, allowing to analyse changes in between accounting periods when time series are available, as it organizes, over time, information on the extent of different ecosystem types in terms of geographical extent and any managed (changes in ecosystem area as a result of planned and implemented actions, usually with specific management, conservation, or restoration objectives) or unmanaged (either increase or decrease in the area of an ecosystem type without planned human management or restoration interventions) changes to the extent (see in Figure 5.2 the suggested template for an extent account elaborated by the SEEA EA).

Opening extent (km²; Ha)		Ecosystem type		
	MAES code	Code 1	Code 2	
	Habitat name	Habitat 1	Habitat 2	Total
		Area 1	Area 2	Total area
Additions to extent				Tot Area
Manage expansions		Area	Area	
Unmanaged expansions		Area	Area	
Reductions in extent				Tot Area
Managed reductions		Area	Area	
Unmanaged reductions		Area	Area	
Net change in extent		Area	Area	Tot area
Closing extent		Area	Area	Tot Area

Figure 5.2 Ecosystem extent account (adapted from United Nations 2025)

The EU ecosystem typology (ET; Table 5.1) has been endorsed and adopted as a common framework to harmonize ecosystem accounting and reporting across the EU. Its development builds on the main existing EU-wide classifications (MAES ecosystem typology and the EUNIS habitat classification) while drawing additional context from the IUCN Global Ecosystem Typology (IUCN GET; Eurostat, 2024).

The EU ET uses three hierarchical levels of increasing detail to classify European ecosystems typologies. Level 1 (L1 EU; Annex IX of Regulation) of the classification consists in 12 ET and represents the mandatory reporting level for ecosystem extent to Eurostat. In addition, countries may voluntarily report ecosystem extent at Level 2 (L2 EU). Level 3 is designed to address specific national needs while remaining comparable across the EU (Eurostat, 2024).

Table 5.1 Description of EU ecosystem typology, level 1. From Eurostat, 2024

N	Ecosystem type	Description
1	Settlements and other artificial areas	Human habitats strongly modified by people, with buildings and other man-made structures. Includes residential, industrial, commercial, transport areas, urban green areas, extraction sites, dumping and construction sites. Excludes peat extraction sites. Also referred to as *urban ecosystems*.
2	Cropland	Food production areas, intensively or extensively managed. Includes perennial/annual crops, agro-ecosystems with natural vegetation, agro-forestry (e.g. cork and holm oak stands), and agricultural mosaics. Semi-natural elements (hedges, ponds, margins) are included.
3	Grassland (pastures, semi-natural and natural grasslands)	Areas dominated by herbaceous vegetation (grasses, forbs, mosses, lichens). Includes modified (sown, grazed, hay/silage) and semi-natural grasslands. May include wooded pasture and agroforestry used for grazing. Associated small semi-natural elements included.
4	Forest and woodland	Tree-dominated ecosystems with canopy cover >30% (temperate) or >10% (boreal/Mediterranean). Thresholds adjusted for extreme climatic conditions.
5	Heathland and shrub	Vegetation dominated by shrubs or dwarf shrubs, with canopy cover <30%. Includes natural harsh-condition ecosystems and secondary ecosystems from extensive human use (e.g. heathlands, sclerophyllous vegetation).
6	Sparsely vegetated ecosystems	Ecosystems with low vegetation density (≥70% barren soil). Includes degraded or extreme-condition areas, bare rocks, glaciers, inland dunes, sand plains. Despite sparse cover, it may host high biodiversity of adapted species.
7	Inland wetlands	Areas affected by water year-round or seasonally (flooding, shallow groundwater). Includes marshes, mires, bogs, fens, and peat extraction sites. Excludes seasonally flooded grasslands/heathlands.
8	Rivers and canals	Permanent freshwater linear surface waters. Includes natural rivers, streams, and artificial canals/ditches for transport, drainage, or supply.
9	Lakes and reservoirs	Permanent freshwater non-linear water bodies. Includes natural lakes and artificial reservoirs mainly for supply or energy generation.
10	Marine inlets and transitional waters	Land–water interface ecosystems under tidal influence, salinity >0.5‰. Includes lagoons, estuaries, and other transitional waters.
11	Coastal beaches, dunes and wetlands	Coastal ecosystems influenced by marine conditions (salt spray, saline groundwater, tidal flooding). It includes beaches, dunes, coastal saltmarshes, salines, and small wetlands between dunes.
12	Marine ecosystems	Marine ecosystems below mean sea level, from nearshore to deep waters. It includes the entire water column, seabed, and pelagic zone.

The ecosystem condition account is a fundamental component of the SEEA-EA framework, and it is formed by two main tables (United Nations, 2025). It serves as a primary assessment of the quality of an ecosystem by measuring its abiotic and biotic characteristics (United Nations, 2021; 2025). The underlying idea is that the extent and condition of ecosystem assets influence the flows of ecosystem services, both in the present and in the future, and therefore they must be monitored. Additionally, in certain cases, the supply and use of ecosystem services can also affect the condition of ecosystems (Czucz et al., 2019; United Nations, 2025).

The relationship between stocks and flows is captured by the concept of ecosystem capacity. Unlike the experimental SEEA-EEA framework, where the capacity account was proposed as a separate component, in the SEEA-EA capacity is no longer treated in a standalone account but is instead embedded within the ecosystem condition account, which can also organize data relevant to the measurement of the capacity of an ecosystem to supply different ecosystem services (United Nations, 2025).

These accounts include ecosystem information and variables (Figure 5.3) organized and rescaled, through reference levels, into a set of indicators (Figure 5.4) that are relevant for assessing the overall quality of the ecosystems (Czucz et al., 2019; United Nations, 2021, 2025). Ideally, the variables should be selected according to their link to ecosystem processes, and hence into the ecosystem functioning (United Nations, 2025), thus reflecting “stocks” rather than the associated “flows” and should be sensitive to change. Moreover, they should be policy-relevant, being capable of conveying clear normative messages (Czucz et al., 2019; 2021a, 2021b).

According to the SEEA-EA framework, ecosystem condition accounts ideally compare the current condition to a reference condition to provide insights into the ecological state of the ecosystem. Also, the structure of these accounts is specific to the ecosystem type and depends on the selected variables, with data availability being a primary factor. It is also recommended that ecosystem characteristics which seem to be relevant, but which are not covered adequately by available data sources, should be highlighted as data gaps (Czucz et al., 2021a, 2021b). For the calculation of each indicator, with respect to upper and lower reference limits for the specific condition variables, the following general formula can be applied:

I = (V-VL)/(VH-VL)

where I is the value of the indicator, V is the value of the variable at the opening or closing date, VH is the high reference level value (upper reference limit value) and VL is the low reference level value (United Nations, 2025).

Regulation (EU) 2024/3024 stated that statistics for ecosystem extent and ecosystem condition accounts shall be compiled and transmitted from the member states every 3 years, and within 24 months of the end of the reference year.

SEEA Ecosystem Condition Typology Class		VARIABLES		ECOSYSTEM TYPE		
		Descriptor	Unit	Opening value	Closing value	Change
Abiotic	Physical state	Variable 1				
	Chemical state	Variable 2				
Biotic	Compositional state	Variable 3				
		Variable 4				
	Structural state	Variable 5				
		Variable 6				
	Functional state	Variable 7				
		Variable 8				
	Waterscape	Variable 9				

Figure 5.3 Ecosystem condition variables table (adapted from United Nations 2025)

SEEA Ecosystem Condition Typology Class		INDICATORS		ECOSYSTEM TYPE						
				Variable value (observed)		Reference values		Indicator value (rescaled)		
		Descriptor	Unit	Opening value	Closing value	Upper level	Lower level	Opening value	Closing value	Change
Abiotic	Physical state	Variable 1								
	Chemical state	Variable 2								
Biotic	Compositional state	Variable 3								
		Variable 4								
	Structural state	Variable 5								
		Variable 6								
	Functional state	Variable 7								
		Variable 8								
	Waterscape	Variable 9								

Figure 5.4 Ecosystem condition with variables and indicators rescaled with reference values table (adapted from United Nations 2025)

The reporting of ecosystem services flow in physical terms follows the structure of the so-called Supply and Use Tables (SUTs), also described in the SNA and SEEA CF. SUTs, from an ecosystem accounting point of view, are described as “accounting tables structured to record flows of final ecosystem services between economic units and ecosystems and flows of intermediate services among ecosystems" (United Nations, 2025).

Economic units are identified with the primary users (e.g. fishing industry, households, government) and can be assessed by referring to the Statistical Classification of Economic Activities of the European Community (NACE, Nomenclature statistique des Activités économiques dans la Communauté Européenne).

A fundamental principle of the SUTs is that, within any accounting period, the amount of ecosystem services supplied must match the amount used (i.e. the quantity in the supply table should be equal to the quantity recorded in the use table) (United Nations, 2025), reflecting the supply–use identity already outlined in the SEEA CF (para. 3.35).

While in the supply tables the ESs are recorded in reference to the ecosystem type that supply them, the use tables record the flow of ESs in terms of their utilization by the economic units in case of final ESs ("those ESs in which the user of the service is an economic unit"; United Nations, 2025), and by ecosystem type in case of intermediate ESs ("those ESs in which the user […] is an ecosystem asset and where there is a connection to the supply of final ecosystem services"; United Nations, 2025). Regulation (EU) 2024/3024 states that statistics for ecosystem services accounts shall be compiled on a yearly basis, "provided that modelling tools are made available by the Commission (Eurostat) for calculating ecosystem services for ecosystem services accounts" or every 3 years in absence of such tools.

5.2.2. Monetary accounts

In the SEEA EA framework, monetary valuation of ecosystem services is based on the principle of exchange values (e.g. market prices). This approach ensures consistency with the System of National Accounts (SNA) - the international statistical framework used to measure national economies - and allows direct comparison with economic aggregate indicators such as output, added value, and Gross Domestic Product (GDP). According to SEEA EA 6.9, "ESs are the contributions of ecosystems to the benefits that are used in economic and other human activity" and can be classified as SNA (e.g. fish catch) or non-SNA benefits (e.g. climate regulation) (6.1 SEEA EA).

Ecosystem services are recorded in the SUTs in monetary terms, following the same structure as in physical terms. Thus, the value of a service is generally obtained by multiplying its physical flow (e.g. tonnes of fish) by an appropriate unit price. (United Nations, 2021; 2025).

Since prices for ESs are generally not observed, SEEA EA provides a hierarchy of valuation methods and related unit prices that can be used, depending on the context. These include:

- directly observed prices (*e.g.* stumpage fees for timber, carbon credit prices, water purification charges)
- prices from similar markets (using proxies from comparable goods/services)
- residual value or resource rent methods, which infer the contribution of ecosystems to market goods by subtracting the costs of labour, capital, and intermediate inputs from total revenues
- cost-based methods (replacement cost, avoided damage, averting behaviour)
- simulated exchange values (estimating hypothetical market outcomes using demand and supply functions)

When ESs contribute to the production of goods and services that fall within the production boundary of the SNA (i.e. are "inputs into an existing […] joint production process"; SEEA EA 6.17) their values are already embedded in produced goods and services values. This is the case for provisioning services such as food from fishery: the contribution of the ecosystem is embodied in the market value of the harvested fish, which is recorded in the national accounts as part of the economy's output. At the national level, the monetary valuation of these ESs can thus be obtained by separating the value of the final products to isolate the share that is directly linked to the ecosystem's contribution (United Nations, 2021; 2025).

This approach aligns with national accounting principles by distinguishing ecosystem services from the final benefits they contribute to, and by isolating the contribution of the ecosystem asset net of human inputs such as labour, produced capital and intermediate goods. In this sense, the monetary valuation reflects the ecosystem's role in production minus the value generated by economic actors.

5.2.3. Progress and Positioning of Italy in Ecosystem Accounting

In Italy, the implementation of national-scale ecosystem mapping, the "Italian ecosystems map, (Capotorti et al., 2023 v.2.0; first version from Blasi et al., 2017), aligned with the "Mapping and Assessment of Ecosystems and their Services" (MAES) initiative under Target 2 of the EU Biodiversity Strategy to 2020, has provided an important achievement for conservation planning and a starting point for developing a comprehensive ecosystem accounting system.

Such ecosystem map, now available in its second version and produced at a 1:100.000 scale, organizes Italian ecosystems into 98 legend categories and is structured according to nested hierarchical ecoregional divisions, as proposed in the "Italian ecoregion map" (Blasi et al., 2018). These range from broad divisions to more localized provinces, sections, and subsections, with the province level identified as the most appropriate spatial unit for national ecosystem accounting. This hierarchical approach facilitates the integration of socio-economic statistics with ecological characteristics, enabling tailored analyses according to purpose, scale, and relevant ecosystem services (Capotorti et al., 2023).

Building on this framework, which hierarchically classifies ecosystems across multiple scales, from broad macro types to detailed ecosystem types, Italy has supported important national initiatives including the Red List of Italian Ecosystems and the initial development of a national system of ecosystem accounts, as proposed by the Italian Committee for Natural Capital (Capotorti et al., 2023; Rendon et al., 2025).

Recently, Italy developed a new mapping of ecosystem types, produced by ISPRA and aimed at accounting for ecosystem extent for 2012 and 2018, obtained by integrating the main local and Pan-European Copernicus Land Monitoring Service data with ISPRA's National Land Consumption Map, according to the MAES (Mapping and Assessment of Ecosystems and their Services) classification system (De Fioravante et al., 2023).

Building upon this technical and spatial foundation, Italy is thus advancing the implementation of the SEEA EA framework through a dedicated project within the National Statistical Programme. This initiative seeks to expand and refine ecosystem extent and condition accounts, improve spatial detail, and strengthen the links between ecosystem condition and service flows in line with evolving EU specifications. A key priority of such an attemptsis the definition and communication of monetary values following the "Italian approach," which emphasizes technical rigor and policy relevance while incorporating non-monetary information to reflect broader social and ecological values (Femia & Capriolo 2022).

Importantly, Italy has expressed reservations about the global consensus to confer international statistical standard status to monetary aggregates related to ecosystems, as found in the SEEA EA framework (Femia & Capriolo 2022). This cautious stance stems from technical concerns regarding the application of exchange value concepts to ecosystems and their services, which do not inherently possess tradable exchange values, but owns only the rights of use do. Italy advocates for recognizing a plurality of monetary values "*connected to*" ecosystems rather than imposing a single uniform exchange value (Femia & Capriolo 2022).

5.3. Methods

5.3.1. Study area

Sardinia is the second largest island in the Mediterranean (Fig. 5). According to the Map of the Ecosystems of Italy (Capotorti et al., 2023; v. 2.0, 2021) and the Map Ecoregion in Italy (Blasi et al., 2018), Sardinia belongs to the Mediterranean Division, Tyrrhenian Province, "Sardinian Section-2B4", and comprises 4 subsections. Its coastal lagoons, many of which are designated as Ramsar sites and included in the Natura 2000 network, represent crucial socio-ecological systems that deliver multiple ecosystem services.

The provisioning services in Sardinian coastal lagoons are primarily represented by fishery and aquaculture activities. Hosting a variety of commercially and ecologically important fish and shellfish species such as Mugilidae (grey mullets), European seabass, gilthead seabream, *Ruditapes decussatus*, and *A. anguilla* (European eel), these transitional water bodies support small-scale fisheries that are strongly rooted in local tradition and often organized through cooperatives or *consortia (*Madau et al., 2018). Although relatively small in scale, these fisheries are essential for the livelihood of many families and hold deep cultural and economic significance (Fois et al., 2021). In Sardinia, the Regional Government grants long-term concessions to fishing cooperatives, while a small number of lagoons are privately owned. In this study, I aimed to analyse Sardinian transitional waters, and notably coastal lagoon ecosystems though with a different degree of investigation depending on data availability.

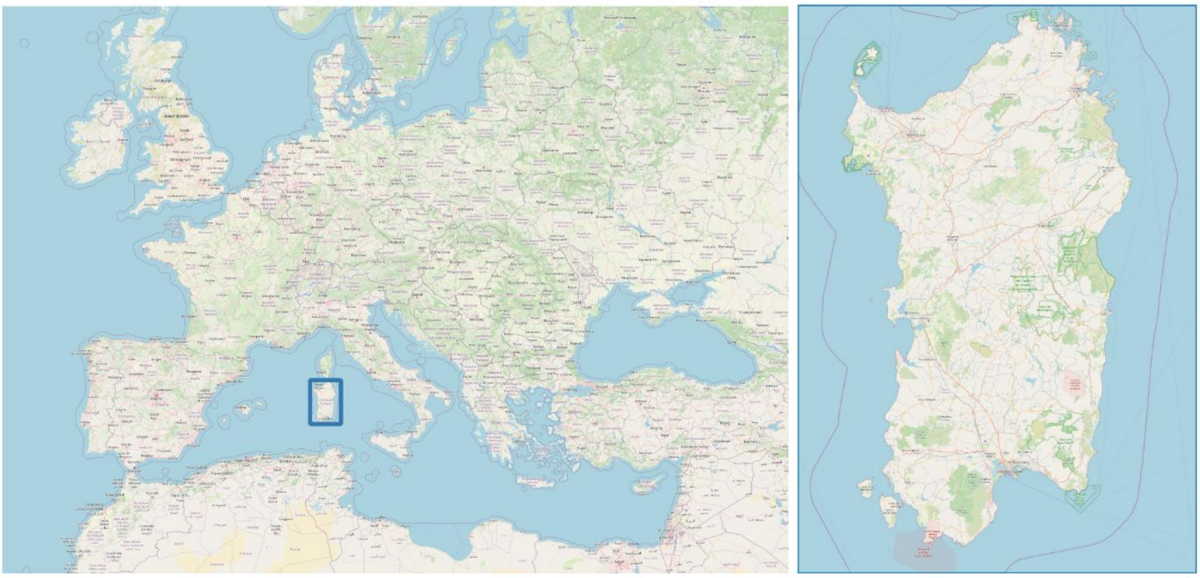

Figure 5.5 Localization of the case study

5.3.2. SEEA EA extent and condition - Sardinian transitional waters

According to the SEEA-EA framework, the ecosystem extent account aims to describe and monitor changes in the spatial distribution and area of ecosystem types over time. Regarding this case study, I refer to transitional aquatic ecosystems as classified under the EU reporting category: “Marine inlets and transitional waters” at the first level, which includes “coastal lagoons”, “estuaries and bays”, and “intertidal flats” at the second level (Eurostat, 2024).

The feasibility for compiling ecosystem extent account was explored by firstly searching for official data provided by the Sardinian Region and other official sources regarding the high resolution extent of transitional aquatichabitats and any changes in their area over time, including those resulting from the emergence of new habitats, such as those formed by the polychaete worm *Ficopomatus enigmaticus* (Figure 5.6), or through human intervention activities.

Figure 5.6 Ficopomatus enigmaticus in S'Ena Arrubia lagoon (Photo by Andrea Alvito - Teramare)

Unfortunately, systematically and updated collected SEEA-EA aligned regional-scale datasets on the extent of transitional inlets and their changes over time were not found; the only available information was limited to fragmented documents, mainly related to site-specific assessments, or scientific papers carried out for individual lagoons and specific scientific purposes.

Since updated regional mapping was neither available, two national-scale sources were considered: the SEEA EA-aligned map developed by ISPRA (De Fioravante et al., 2023) and the ecosystem map of Italy V 2.0 (Capotorti et al., 2023; 1:100.000 minimum mapping unit 25 ha).

The accuracy of these maps was assessed using ancillary datasets, including the Italian Corine Land Cover 2018 (1:100.000 minimum mapping unit 25 ha), the 2008 Sardinia Corine land cover (CLC) (1:25.000), and Carta della Natura (Camarda et al., 2011; 1:50.000). To account for different classification between different strata, the following crosswalk was adopted.

Table 5.2 Adopted crosswalk table for the different classification of ecosystem typologies

EU	CLC	Ecosystem Map of Italy	De Fioravante et al. (2023)
10 Marine inlets and transitional waters	5.2 Marine waters	G18 -narrow Saltmarsh ecosystems of Sicily and Sardinia coasts with *Cymodocea nodosa*, *Nanozostera noltei*, *Ruppia spiralis*, *Zannichellia obtusifolia*, *Althenia filiformis subsp. Filiformis* Corresponding to CLC 5.2.1, 5.2.2.	10 - wider Included under "10 - Marine inlets and transitional waters", but this classification also comprises saltmarshes and salines.
10.1 Coastal lagoons	5.2.1 Coastal lagoons	No lower level of detail	No lower level of detail
10.2 Estuaries and bays	5.2.2 Estuaries	No lower level of detail	No lower level of detail
10.3 Intertidal flats	4.2.3 Intertidal flats	No lower level of detail	No lower level of detail

Ecosystem condition tables provide a structured approach to collect and aggregate variables in indicators describing the characteristics of ecosystem assets and how they have changed over the accounting period (see 5.2.1).

To verify the feasibility of compiling condition accounts in the present case study, I first reviewed existing ecological datasets collected for reporting under the EU Water Framework Directive (WFD), as well as other official data sources. Ecosystem variables were selected based on five classes for abiotic and biotic characteristics and one indicator for seascape characteristics, following SEEA-EA guidelines and recent revisions (Czúcz et al., 2021; Maes et al., 2020; United Nations, 2021; 2025; Vallecillo et al., 2022). Variables and related reference values, where feasible, were chosen based on technical competence, feasibility of analysis, and relevance, adhering to SEEA-EA guidelines (United Nations, 2021) while considering data availability limitations. Variables from the WFD were chosen to represent the ecosystem condition, plus other indicators which are not actually systematically assessed and collected to date.

5.3.3. SEEA EA ecosystem service physical and monetary flow accounts

To evaluate the feasibility of reporting the biophysical flow of the wild fish and other natural aquatic products provisioning services (i.e. ecosystem contributions to the growth of fish and other aquatic biomass that are captured in uncultivated production contexts by economic units for various uses, primarily food production - SEEA EA code 1.6; sensu CICES v5.2 1.1.6.1 "wild aquatic animals used for nutritional purposes"), I used existing datasets on fish and shellfish catches. Monthly recorded fishery-dependent data for Sardinian coastal lagoons, collected annually by the Fishery Service of the Department of Agriculture within the Regional Authority, were requested and checked for their reliability and to verify presence or absence of the three more usual sources of bias that can be found in this type of datasets (Caddy et al., 1998; Watson and Pauly, 2001):

1) the possible presence of species from aquaculture within wild species catch data (for example, the presence of seabass biomasses from aquaculture activities within the lagoon)
2) the presence of landings concerning the marine ecosystem within coastal lagoon catch data (for example spiny lobsters) and
3) products that are traded illegally.

I also identify the sector that act as primary user in receiving ESs from the ecosystem (i.e. the sectors that receive ESs directly from the ecosystem type- ET), by referring to the Statistical Classification of Economic Activities in Europe and Italy.

For the monetary flow accounts, gross price data were obtained from the Italian Institute of Agri-Food Market Services (ISMEA) database (https://www.ismea.it/flex/FixedPages/IT/WizardPescaMercati.php/L/IT), which reports minimum and maximum prices as sold in the fish market of Cagliari (the most important fish market in Sardinia) for the years 2012 to 2018. Unfortunately, ISMEA discontinued publishing these records after 2018. Since the Cagliari fish market neither publishes price lists nor provides historical data post-2018, gross prices from 2018 onwards were retrieved from the Fishery Department of Sardinia Region up to 2020. It should be noted that the Sardinia Region only holds post-2018 prices for certain months and selected species, since these data are collected primarily for the purpose of distributing subsidies related to the management of piscivorous birds, particularly the great cormorant (*Phalacrocorax carbo*). The output value was then calculated up to 2018 from the quantity of fish and shellfish caught during the year multiplied by the maximum and minimum annual market gross price, differentiating between species, whenever possible.

To value the ecosystem service of food provisioning, for this study, I used the resource rent value method (see Appendix 5 for methodological notes). I used the resource rent factors in Figure 5.7 and made the calculations using the financial statements of two Sardinian lagoon cooperatives as no official statistical data regarding this was found in national or international fishery statistics such as the Scientific, Technical, and Economic Committee on Fisheries (STECF). Noticeably some cooperatives carry out secondary activities beyond fishing in lagoon (e.g. fishing at sea, food processing), so costs are mixed up and are difficult to disaggregate. This, together with financial statements that are often presented in a simplified form by the cooperatives, made difficult to isolating the costs specifically related to fish resource extraction, which are necessary to calculate the resource rent value. Due to confidentiality concerns and considering that the resource rent was calculated for only two lagoons (corresponding to two cooperatives only), the detailed data are not here disclosed, as it could indirectly reveal the identity of the cooperatives. However, they can be made available upon request for verification and reproducibility purposes. The methodological challenges and data gaps encountered in the ESs monetary valuation, such as the ones just described, are addressed in the following sections. All the monetary values, both for output and resource rent values, have been adjusted to the reference year 2020 using the appropriate GDP deflator.

Output

plus subsidies on production

less intermediate consumption

less taxes on production

less compensation of employees

Gross operating surplus

less consumption of fixed capital

Resource rent

Figure 5.7 Calculation steps to calculate the resource rent value as adopted in this study (adapted from UN, 2021)

Table 5.3 Cost typology from financial statement used to calculate the cost of human input for resource rent

Resource rent factor	**Production costs**	**Subcategory**	**Description**
Intermediate consumption	Raw, ancillary, consumable materials and goods	-	Costs of materials and goods used in production
Intermediate consumption	Services	-	Expenses for external services like transport, maintenance
Intermediate consumption	Use of third-party assets	-	Costs for renting or leasing assets owned by others
Compensation of employees	Personnel costs	Wages and salaries	Employee gross salaries
		Social security charges	Employer contributions to social security
		Termination, retirement, other personnel costs	Severance, retirement benefits, other personnel expenses
Consumption of fixed capital (depreciation)	Depreciation	Amortization of intangible assets	Cost allocation for intangible assets (e.g., fishing concessions, internal accounting or management software))
		Amortization of tangible assets	Cost allocation for physical assets (e.g., machinery)
Taxes on production	Taxes on production	-	Taxes

5.4. Results

5.4.1. SEEA EA extent and condition accounts for the Sardinia marine inlets and transitional waters

Table 5.5 reports an estimate of the overall extent of Sardinian “marine inlets and transitional waters” ET at 1 level considering different mapping datasets consulted (Figure 5.9). It should be highlighted that the noticeable discrepancy between the De Fioravante (2018) dataset and the others (Table 5.5) may be attributed to the different categorization of ecosystem type.

My analysis highlighted indeed some mismatches between the available maps and the Eurostat (2024) typology adopted for the ecosystem extent accounts. In particular, the National Map (De Fioravante et al., 2023; Tab. 4) is based on MAES ET, reporting differences with Eurostat (2024) classification. It classifies “marine inlets and transitional waters” including saltmarshes intertidal flats and saltworks, whereas according to EU category they should be reported under “11 - Coastal beaches, dunes and wetlands.”

Furthermore, verification of ecosystem type attribution revealed inconsistencies with some transitional aquatic ecosystems reported under the “freshwater ecosystems” category instead the coastal lagoon one (Figure 5.8).

These discrepancies indicate limitations in directly applying national maps to regional SEEA EA reporting and underline the need for sub-national validation procedures of ecosystem classifications.

Table 5.4 MAES ecosystem types adopted for Ecosystem Map of Italy (From De Fioravante et al., 2023).

I Level	II Level	Description
Terrestrial ecosystems	Settlements and other artificial areas	Urban areas where most of the human population live and which also include significant areas for synanthropic species associated with urban habitats
	Cropland	Areas mainly dedicated to agricultural production, even with the presence of important natural areas
	Grassland (Pastures, semi-natural and natural grasslands)	Areas with a prevalence of herbaceous vegetation, which can include managed pastures and natural and semi-natural pastures
	Forest and woodland	Areas dominated by woody vegetation
	Heathland and shrub	Dominated by moors, heathland, and sclerophyllous vegetation
	Sparsely vegetated ecosystems	Naturally unvegetated or sparsely vegetated habitats, usually with extreme climatic conditions, such as bare rocks, glaciers, dunes, beaches, and sand plains
	Inland wetlands	Natural or modified mires, bogs, and fens, as well as peat extraction sites
Freshwater ecosystems	Rivers, canals, lakes, and reservoirs	Permanent freshwater, inland water courses, and water bodies
Marine ecosystems	Marine inlets and transitional waters	Coastal wetlands, lagoons, estuaries, i.e., areas on the land–water interface under the influence of tides and with salinity levels greater than 0.5 ‰

Figure 5.8 Mismatch in ecosystem classification. Is Benas and Avalè Su Petrosu transitional water bodies reported under "freshwater" (light blue colour; From De Fioravante et al., 2023). In the lower panel the main channel and sea inlet of Is Benas lagoon (Photo: Elisa Serra).

Table 5.5 Ecosystem extent for all Sardinian marine inlets and transitional waters (L1, sensu Eurostat 2024).

Extent (ha)	Camarda et al., 2011 (2010)	Capotorti et al., 2023 (2018)	De Fioravante et al.,2023 (2018)	Sardinian Land Cover (2008)
Total	11921	11262.4	17440.4	10283.9

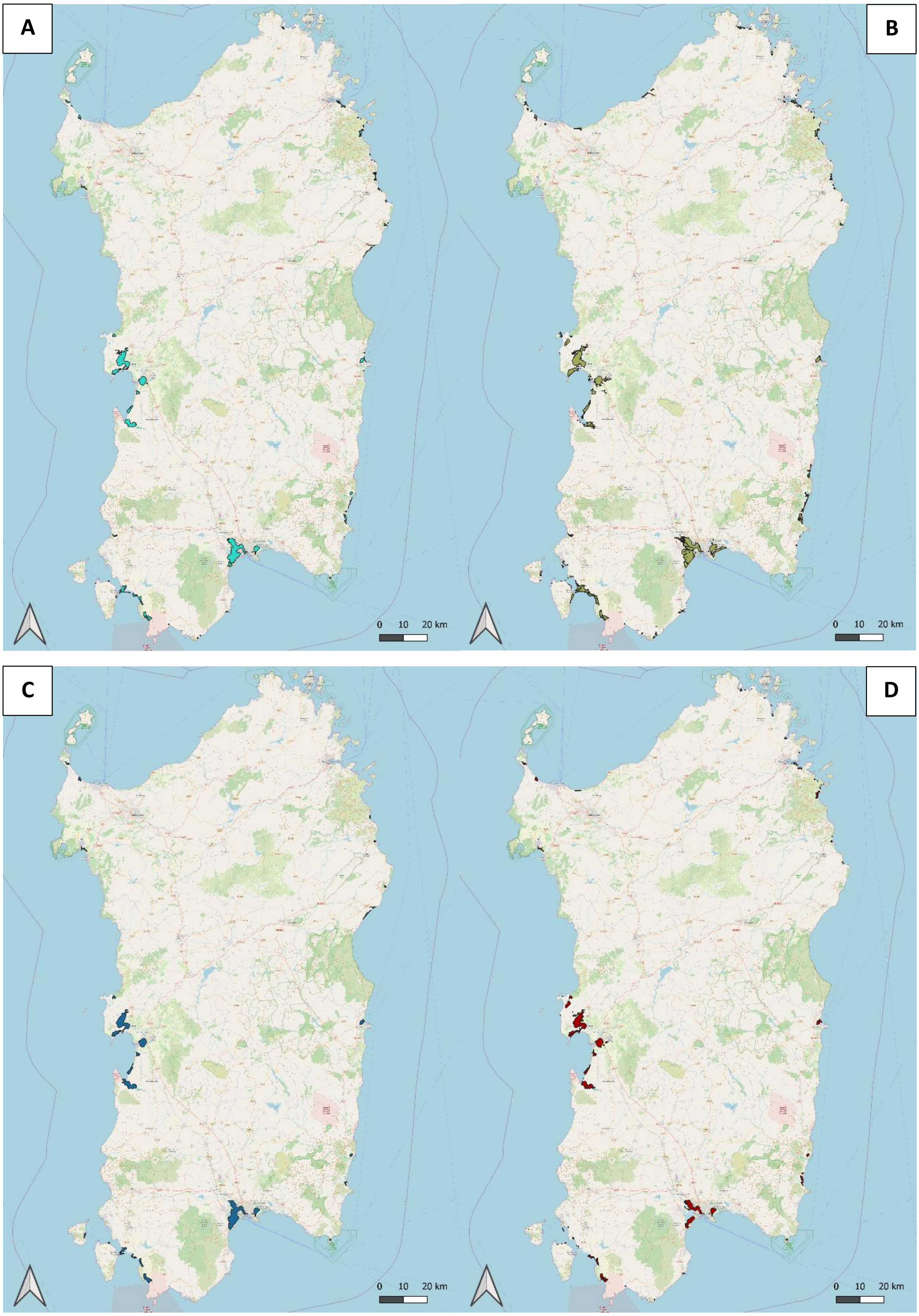

Figure 5.9 Marine inlets and transitional waters from: A – Camarda et al., 2011 ; B - De Fioravante et al., 2023; C - Capotorti et al., 2023; D - Sardegna land cover 2008.

For the ecosystem condition account, after reviewing existing environmental datasets, the selected set of condition variables to represent key abiotic, biotic, and landscape-level characteristics is presented in Table 5.6. Among them, I also retained variables for which data are not yet available but considered important for a comprehensive assessment of ecosystem condition, in line with SEEA-EA recommendations. At the local scale, descriptors from Table 5.6 should be translated into variables and indicators that capture the condition of that specific system. For instance, the descriptor water temperature becomes the variable "mean annual temperature". At the regional scale, the same descriptors are aggregated or transformed to provide indicators that allow for comparison across multiple ecosystems. However, the passage to indicators requires a consistent time series to set the reference values, not available for most of the variables selected (e.g. for temperature). This lack of historical data constrains the possibility of establishing robust reference values, both at the local scale, and at the regional scale - where comparisons among ecosystems would benefit from standardized baselines. In addition, when collected systematically, physical-chemical parameters are typically measured at discrete time points, often with quarterly monitoring campaigns but not always carried out in the same months each year. As a result, even comparisons of the same site across different years may be of limited reliability, further complicating the construction of robust indicators.

In Table 5.7, condition variables and indicators at regional level are presented only for those possible to be calculated. It is important to note that commercial species population status was included in Table 5.6 among the biotic ecosystem condition indicators, as it reflects the compositional condition of fish/shellfish populations and thus the system capacity to provide provisioning services. However, in Sardinia these data are not systematically collected. In Table 5.7 is then presented a proxy, by evaluating the biomass of commercially important fish and shellfish species relative to their biomass at Maximum Sustainable Yield (MSY) based on B/B MSY obtained from CMSY++ model outputs presented in Chapter 3 and Appendix 4 and 6. Similarly, for the hydrological connectivity, three water exchange capacity variables obtained in Chapter 3 were used as a proxy, as existing available data did not allow a more comprehensive evaluation.

Table 5.6 Condition variables for marine inlets and transitional waters

SEEA EA ETC		VARIABLES		Units	Systematically collected?
		DESCRIPTOR	Description		
Abiotic	Physical state	Water temperature	Influenced not only by air temperature, but also by water levels and inflows from the sea and/or rivers, depth etc. Is therefore a combined result of climate - and therefore also climatic changes - and the ecosystem management.	°C	Yes
		Marine heatwaves	The marine heatwaves can be a key condition variable reflecting, during the accounting period, punctual thermal stress events in lagoon ecosystems. Increased frequency and intensity of heatwaves influence the overall ecosystem condition, and can lead to habitat degradation, loss of temperature-sensitive species, and shifts in ecosystem structure and function, ultimately affecting ecosystem resilience and the provision of water ecosystem services.	N and duration	No
	Chemical state	Chemical state WFD	The assessment of the chemical status of transitional waters (good or not good) based on the evaluation of the presence of pollutants. From a One Health perspective, this can be considered a highly relevant condition variable: the presence of priority or priority hazardous substances affects not only ecosystem quality, but also food safety (through bioaccumulation in fishery products) and the health of coastal communities that consume them.	a-dimensional	Yes
		Dissolved oxygen	The amount of oxygen dissolved in water, which is available for biota and enters via diffusion from the atmosphere. Rapidly moving water will typically have more dissolved oxygen than stagnant water, as will water with lower amounts of biomass.	mg l−1	Yes
		Chl-a concentration	Photosynthetic pigment used as an indicate algal levels in water	µg l−1	Yes
		Salinity	Strongly influences species distribution and composition	psu	Yes
		pH	Reflects acidification, photosynthetic activity, and decomposition. Sensitive to organic inputs and climate-related changes.	unit	Yes
		Total N concentration	Essential for nutrient balance. Excess nitrogen can trigger eutrophication, harmful algal blooms, and oxygen depletion.	mg l−1	Yes
		Total P concentration	Together with TN, it is crucial to assess nutrient loading. Often the limiting nutrient in freshwater and coastal systems.	mg l−1	Yes
		Total Organic Carbon - sediment	The total amount of organic carbon in the sediment.	mg g−1	Yes
Biotic	Compositional state	EQB Macroinvertebrates	Macrozoobenthos is one of the Biological Quality Elements (BQEs) considered in the application of the Water Framework Directive (WFD; EU, 2000) as it can rapidly respond to natural and anthropic stress.	a-dimensional	Yes
		Fish and shellfish biomasses or population status	Reflects the compositional condition of fish/shellfish populations and thus the system capacity to provide provisioning services.	tons or a-dimensional	No

SEEA EA ETC		VARIABLES		Units	Systematically collected?
		DESCRIPTOR	Description		
		Fish and shellfish Shannon diversity index (H)	Measures the diversity of fish and shellfish communities by considering both the number of species (richness) and the relative abundance of each species (evenness). Higher values indicate communities with a greater variety of species and a more balanced distribution of individuals among those species.	(0 to ∞)	No
		NIS	Number of invasive non-indigenous species affecting ecosystems; or proportion of the species group or spatial extent of the broad habitat type which is adversely altered due to NIS or richness or abundance	n; ratio; extent	No
	Functional state	Dystrophic events	Dystrophic crises on the other hand represent one of the most dramatic consequences of eutrophication in Mediterranean coastal lagoons, whereby, nutrient enrichments enhance the dense growth of benthic and pelagic primary producers causing imbalances to production/respiration rates and abrupt variations in their physicochemical and ecological properties leading at times to anoxia and massive mortality events (Basset et al., 2013).	n	No
		Functional richness	Functional properties may be more robust than structural ones in assessing ecosystem condition and resilience. The calibration and application of such indices require the identification of biological reference conditions specific to the type of water body under study. However, to date, not all lagoons have sufficient data (e.g. for meiozoobenthos) to reliably establish these reference conditions.	n	No
	Structural state	Harmful algal blooms (HAB)	Overgrowth of algae species releases toxic substances into lagoon waters, depletes deep water oxygen, decreases water column transparency, decrease the overall aesthetic value in lagoons, and affecting organisms.	n	No
		EQB Macrophytes	The Macrophyte Quality Index (formally adopted in Italy for the ecological status classification of transitional waters under Directive 2000/60/EC, integrates two organic quality elements: macroalgae and aquatic phanerogams. MaQI is used to provide a synthetic ecological classification of the ecosystem through structural parameters (cover, composition, and abundance) of green and red algae, the number of macroalgal taxa distinguishing sensitive from opportunistic or indifferent taxa, and the cover of individual aquatic phanerogams.	a-dimensional	Yes
Landscape	Connectivity	Hydrological connectivity	Hydrological connectivity describes water movement and the transport of matter, energy, and organisms, encompassing both structural and functional aspects. Reduced connectivity, induced by natural factors but also from man-made modification, can also limit water renewal, increasing the system’s susceptibility to climate-induced stressors such as warming or heatwaves.	a-dimensional	Sporadic, qualitatively

Table 5.7 Condition table at regional level for Sardinian transitional waters. The range between upper and lower reference level indicates the number of ecological units or stock assessed.

SEEA CONDITION CLASS	ECOSYSTEM TYPOLOGY	VARIABLE	UNITS	Value observed	Year	Low RF	Upper RF	Indicator
Abiotic	Chemical state	Ecosystems with chemical state WFD good	N	21	2019-2021	0	41	0,51
Biotic	Compositional	Ecosystems with EQB Macroinvertebrates in good status	N	26	2019-2021	0	41	0,63
		Commercial interest Fish and shellfish with biomass status >1	N	16	2024	0	49	0,30
		Ecosystems impacted from NIS	N	13	2024	15	0	0,13
	Structural	Ecosystems with EQB Macrophytes >=good	N	11	2019-2021	0	41	0,27
Landscape	Hydrological connectivity - water exchange capacity	Ecosystems experiencing burying of sea inlets	a-dimensional	12	2024	15	0	0,2
		Ecosystems experiencing burying of main channel	a-dimensional	12	2024	15	0	0,2
		Ecosystems experiencing burying within lagoon	a-dimensional	10	2024	15	0	0,33

5.4.2. SEEA EA ecosystem service biophysical and monetary flow account for the Sardinian lagoons

Based on available fishery-dependent data on fish catch biomass collected by the Regional Authority, I collected data, that presented gaps in both spatial and temporal coverage, spanning from 1992 to 2024. I was able to compile the aggregated ecosystem service biophysical flow accounts for the years 2006 up to 2024 (Table 5.8) from 23 fishing concessions located in sites falling under the category “marine inlets and transitional waters”, and notably in coastal lagoons.

However, for some ecosystems, ecosystem-level data are not available, as fishery data are collected and reported at the level of cooperatives or consortia rather than by individual ecosystem units, thus precluding a disaggregated compilation at the site level.

Considering the reliability of the dataset, the source of bias related to the presence of aquaculture biomasses was partially eliminated by excluding species whose occurrence could be exclusively linked to aquaculture production, specifically pacific oysters (*Crassostrea gigas*) and mussels (*Mytilus galloprovincialis*). For European seabass (*Dicentrarchus labrax*) and gilthead seabream (*Sparus aurata*), data were assumed to refer to wild catches, given the limited number of aquaculture sites farming these species; nonetheless, this assumption requires further investigation (see Discussion, section 5.5.2). The second source of bias (i.e. landings from marine ecosystems reported within the ET analysed) was minimized by retaining only landings of species typically inhabiting lagoons and removing those most likely attributable to coastal or marine fishing outside lagoons. The third source of bias, concerning unreported or illegal catches, could not be addressed due to data limitations. Overall, my data cleaning improved the reliability of the time series, although some uncertainties remain.

Starting from 2020, the time series of the physical flow of this ecosystem service also includes data related to the catch (and sale) of the Atlantic blue crab (Callinectes sapidus; Figure 5.10), an invasive alien species that has recently spread in the Mediterranean Sea and ,particularly in Italian seas (Mancinelli et al., 2013, Marchessaux et al., 2023), with the first record in a Sardinian brackish lagoon dating back to 2017 (Culurgioni et al., 2018).

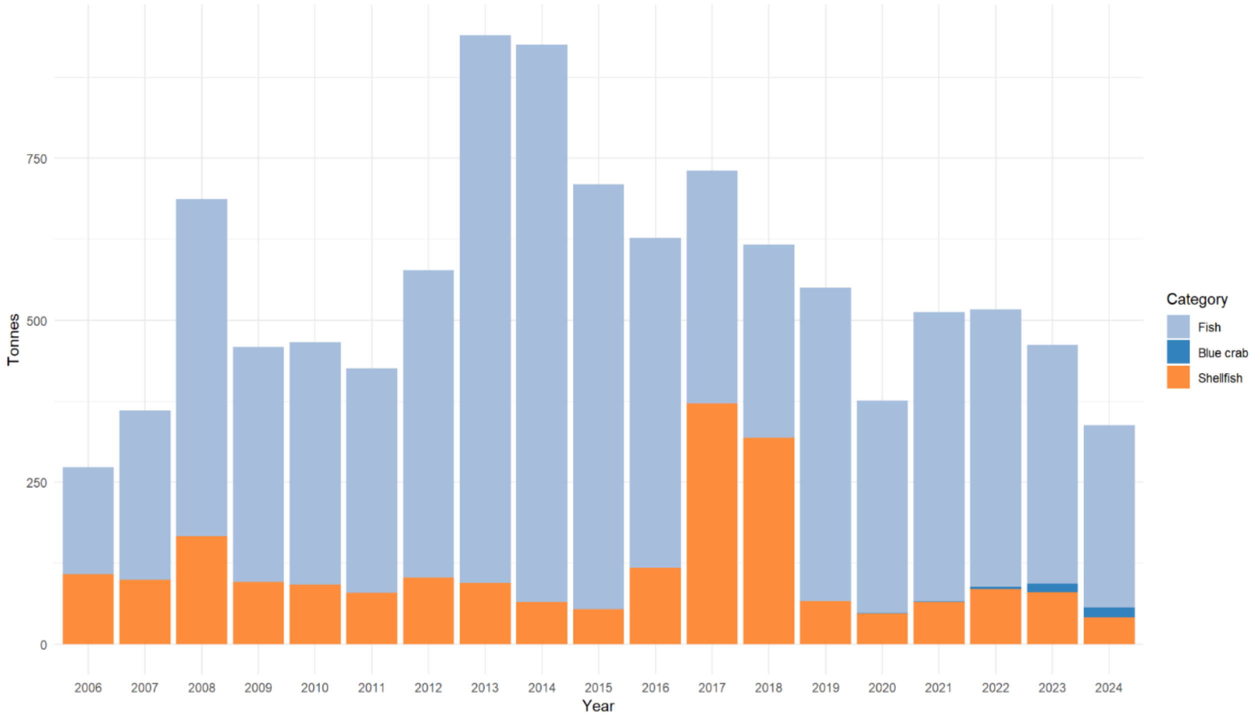

Figure 5.10 ES physical flows at regional level from 2006 to 2024

Table 5.8 Physical flow accounts

ES physical flows (Kg)	2006	2007	2008	2009	2010	2011	2012	2013	2014
Fish	164.913	261.585	520.482	36.2993	374.356,7	346.717,8	474.996,4	846.002,6	860.861,6
Shellfish	107.538,6	98.965,5	166.306	95.654,4	91.688,4	78.893,4	102.163,6	93.803,8	64.268,9
Total	272.451,6	360.550,5	686.788	458.647,4	46.6045,1	425.611,2	577.160	939.806,4	925.130,5

ES physical flows (Kg)	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Fish	655.429	509.861,9	358.861,2	298.237,7	483.517,7	327.618,2	447.239,31	428.622,62	368.165,61	281.289,99
Shellfish	54.011,6	117.382,7	371.475	318.419,3	66.320,8	47.943	65.084,27	88.077,48	93.571,11	56.453,17
Total	709.440,6	627.244,6	730.336,2	616.656,9	549.838,5	375.561,2	512.323,58	516.700,1	461.736,72	337.743,16

In Table 5.9 an example of compilation of ESs physical flows presented in the form of SUT for the year 2024 is reported.

Table 5.9 Example of SUT table for year 2024

SUT	Units	EU	ET
2024		Fishery (03.11)	Marine inlets and transitional waters
Supply			
Food provisioning	**Kg**		**337.743,16**
Use			
Food provisioning	**Kg**	**337.743,16**	

In Table 5.10, the output values calculated from the exchange values and the physical flows are presented.

Table 5.10 Monetary flow accounts (output values)

ES Output values	Unit	2012	2013	2014	2015	2016	2017	2018
Min prices	**2020 Euro**	2.854.861	4.399.198	3.711.143	3.163.453	3.226.775	5.453.550	4.556.729
Max prices		5.240.394	6.900.842	5.604.573	4.648.793	4.606.859	7.241.303	5.919.635

The resource rent was calculated for two cooperatives only and with reference to the years 2017 and 2018, based on subsidies and costs reported in their financial statements. The resulting resource rents were negative in both cases. Therefore, I also investigated what happened when I directly subtracted the costs from the production value reported in the financial statements, and also in this case, the resource rent remains negative. As mentioned in Section 5.3.3, due to confidentiality concerns, the detailed cost data used and related RR results are not here disclosed but can be made available upon request for verification and reproducibility purposes.

5.5. Discussion - Empirical challenges

Overall, as reported in other studies, the collection and reporting of ecosystem-related data for accounting purposes is limited by several factors, such as technical and procedural constraints (Bagstad et al., 2021; Dvarskas, 2019). Key data-related challenges encountered were mostly related to a lack of temporal and spatial coherence in the datasets for the ecological condition variables, as well as for biophysical and monetary flows. Here, I discuss the main challenges encountered in this study for managing data for the SEEA EA compilation at the local level and reflect on how to build datasets in the future that match ecosystem accounting data needs.

5.5.1. Extent and condition data availability and gap

Similarly to studies in other ecosystem types (e.g. see Bartolini et al., 2024 for marine ecosystems, Farrell et al., 2021 for multiple ecosystem types at catchment scale and Mengo et al., 2022 for marine and coastal accounts for Small Island Developing States), despite the importance of coastal lagoons ecosystems, I was unable to reconstruct a comprehensive time series for the extent and conditions for my study area.

This limitation arises from the lack of systematically collected data on these habitats over time. Although some information exists for sites designated under Natura 2000, key data remain missing; for example, the presence of *F. enigmaticus* is not currently documented for many ecosystems. However, the Sardinia Region has recently started habitat mapping activities (for year 2024), which will allow for improved spatial and temporal resolution in future ecosystem extent accounts at local and regional level.

My findings also revealed mismatches in the spatial data currently available from national maps produced for the compilation of ecosystem extent accounts, due to its resolution. These misalignments pose challenges for the direct use of national datasets in regional accounting exercises, particularly when aiming to ensure consistency with EU-level reporting obligations under the amended Regulation (EU) 691/2011. These limitations go beyond the regional scale exercise of this chapter, affecting all geographical scales of reporting, posing potential challenges in correctly reporting ecosystem extent at national level, as already shown for some EU countries (EEA 2016, Grêt-Regamey 2017, Grunewald 2020, Hein 2020, La Notte 2017). Especially, it is difficult to determine if any changes in extent were attributable either to managed or unmanaged changes or to different ways of collecting the spatial data.

Developing high-resolution spatial maps aligned with the Eurostat typology to enable the production of ecosystem extent accounts at the regional scale is therefore recommended. This recommendation will be fulfilled by the publication of the updated ecosystem maps of Italy for 2021, with 10 m of spatial resolution and detail up to the third level of EU ecosystem typology classification (see Cimini et al., 2025 for a presentation of the methodological flow and the preliminary outcomes of the map).

As discussed for extent accounts, the compilation of condition accounts poses several challenges (Grilli et al., 2021) as the methodology is to date not standardised or consistently implemented, being the least developed accounts within the EU and at national levels (Cummins et a., 2023; Farrell et al., 2021). Data availability is one of the most important enabling criteria which at the same time remains a limiting factor in ecosystem condition full implementation (Czucz et al., 2021; Lange et al., 2022). This is particularly true for coastal lagoon ecosystems (Gaglio et al., 2024), where there is often no operational objective to systematically collect data on environmental characteristics beyond those already included in international regulatory frameworks such as the Water Framework Directive (WFD). However, WFD indicators, even if highly informative and used in other studies (Rendon et al., 2019) and SEEA EA reporting (ONS, 2025), can be not sufficient alone to fully assess the condition of transitional ecosystems, and notably coastal lagoons, especially if the aim is also to understand how condition indicators are link with the capacity and flows of ESs human-mediated (Rova et al., 2022).

Moreover, the current lack of understanding of the interlinkages between ecosystem extent, condition and ES capacity and flows, the absence of targeted and reliable time-series data, as well as the need for agreed reference levels (Maes et al., 2020), affect my ability to build complete condition accounts. For some variables I used proxy, taking advantage from the results of Chapter 3, such as the modelled biomass status expressed as B/Bmsy and the water exchange capacity index.

Moreover, although *F. enigmaticus* reefs can be classified as a separate habitat type for extent accounting purposes, it is important to recognize the three-dimensional and interconnected nature of these reef structures within coastal lagoons. These reefs, being capable of massive growth (Costa et al., 2019; Goulletquer, 2016) physically develop inside the lagoon environment and strongly interact with adjacent habitats, influencing sediment composition, water flow, and benthic communities (Brundu & Magni 2021). As an invasive habitat engineer species, *F. enigmaticus* not only alters the spatial extent of habitats (Katsanevakis et al. 2014) but also significantly impacts the ecological condition of the whole lagoon (Costa et al., 2019). This means that while extent can be mapped as discrete classes, the condition assessment must consider the cumulative and interactive effects these reefs have on the whole ecosystem. Therefore, although habitat classification for extent is a useful simplification, condition indicators should reflect these ecological interdependencies and the functional impacts of invasive reef-building species.

Finally, as also observed for urban ecosystem accounts, the socio-ecological nature of most of the coastal lagoon systems risk to be only partially represented when considering the condition variables of the "ecological side" as suggested in SEEA EA. In line with the suggestion from Babì Almenar and co-authors (2025), a possible solution that would avoid affecting the interoperability of SEEA EA accounts could be the introduction of an additional condition table specifically addressing the "social" dimension of the system to provide a more comprehensive assessment of lagoons socio-ecological condition. In this regard, the social accounting framework currently being developed under the guidance of the Global Ocean Accounts Partnership (GOAP; Shellock & James, 2024) for the Ocean accounts represents an attempt to integrate social indicators within ecosystem accounting. Collecting and analysing data on variables such as income levels, access to lagoon resources, dependency on lagoon assets, food security, and other socio-economic dimensions of coastal lagoons, will enable the regular and comprehensive monitoring in the ecosystem accounts, and the identification of the most vulnerable socio-ecological systems.

In advancing the implementation of robust condition accounts, and allowing the construction of indicators to go beyond GDP to track and monitor ecosystem degradation as well as benefits from restoration and conservation actions, future efforts then should prioritise:

1) the identification of a European common and accepted set of condition variables for marine inlets and transitional waters, particularly for coastal lagoons, applying the enabling and quality criteria for variable selection, and which are sensitive enough to both evaluate the overall condition and detect climate change related shifts.

(2) the identification of reference value that take into account the specificity of these ecosystems, that is widely recognised by the scientific community and harmonised at national or European scales.

(3) the integration of coordinated sampling strategies between scientific institutions and governmental agencies, thereby enhancing the temporal and spatial coverage of long-term, standardised datasets, reducing data and sampling fragmentation, and preventing disproportionate representation of certain sites over others.

5.5.2. Physical and monetary flow data availability and gap

My findings, in line with other studies, highlight significant challenges related to data availability in compiling ecosystem services physical flow accounts, particularly at the disaggregated level.

The aggregation of fishery data at the cooperative or consortium level, rather than at specific ecosystem units, limits the spatial resolution and ecological specificity of my accounts. This is likely due to the focus of institutional data collection efforts, which are oriented towards monitoring economic entities such as cooperatives or consortia, in line with administrative and economic monitoring objectives, rather than ecological or spatially explicit ecosystem accounting. This aggregation, reflecting institutional priorities and data collection frameworks that predominantly emphasize economic and administrative units over ecological boundaries, is in some way in contrast with the definition of ecosystem assets, their services and their links to the economy from a perspective that considers ecosystems as distinct ecological entities, regardless of the legal and or economic ownership of ecosystem assets *sensu* SNA. Consequently, this can hinder the ability to link ESs flows directly to discrete ecosystem areas, reducing the accuracy and usefulness of ecosystem accounting for site-level management and conservation decisions. While aggregated data at the regional scale can still provide valuable insights, this lack of spatial disaggregation poses a potential significant limitation for assessments aiming to inform management and policy at local scales.

Moreover, I also highlight the potential limitations in the preliminary data cleaning step I performed, aimed to isolate physical flows attributable to wild fisheries within the lagoon. Indeed, it cannot be excluded that among the retained data there remain residual physical flows linked to marine-coastal fishing or sporadic aquaculture activities occurring inside the coastal lagoon. From the perspective of valuing food provisioning services produced solely by wild species in the lagoon, these potential ambiguities in the data may hinder an accurate compilation and origin double counting problems. This challenge could be addressed through enhanced data collection efforts aimed at clearly distinguishing between species farmed under semi-intensive aquaculture, those caught from wild populations at sea and those from wild populations in lagoons. In this sense, SEEA EA data needs can serve as a catalyst for refining monitoring systems and aligning data collection practices more closely with ecological boundaries and service flows (Usubiaga-Liaño et al., 2025).

Moreover, since I could not compile complete condition indicators over time due to data availability constraints, it was not possible to properly describe trends in the biophysical flow of the ecosystem service in relation to ecosystem conditions. However, it is important to note that, since the proxy used for the ecosystem service was biomass, this is influenced not only by ecosystem conditions but also by the social component responsible for resource extraction. In this context, a meaningful interpretation of these biophysical flow data can be made considering the results presented in Chapter 3, where B/Bmsy (used as a condition indicator in this chapter) and F/Fmsy (not included among conditions, as it represents a pressure; in line with the SEEA EA guidelines) were calculated. Indeed, even if not necessary and mandatory in the development of SUTs, the assessment of ecosystem capacity ("the ability of an ecosystem to generate an ecosystem service under current ecosystem condition, management and uses, at the highest yield or use level that does not negatively affect the future supply of the same or other ecosystem services from that ecosystem" SEEA EA, 2025) is recognized as useful in interpreting accounting entries (SEEA EA 2025; La Notte et al., 2025). The combination of these two indicators, the stock status referred to the maximum sustainable yield can help to assess the sustainability of the actual flow of the ecosystem service, providing important insights useful for the evaluation of the asset degradation and the future development of ecosystem capacity accounting (SEEA EA, 2025; La Notte et al., 2025).

Furthermore, together with the habitat vulnerability indicators developed in Chapter 3 and applied as condition measures in this chapter (hydrological connectivity, EQB, invasive alien species), if monitored over time, SEEA EA reporting can help in integrating data to periodically evaluate the ecological vulnerability of the ecosystem service to climate change.

Given the regional scale of this study and its ecosystem accounting perspective, it is important to note that national statistics did not distinguish between fishing activities carried out in marine versus lagoon environments (classified under ATECO code 03.11.0 for marine and lagoon fishing and related services, 2022 version). For this reason, unlike other studies (see Bogaart et al., 2023), it was not possible to rely on these to calculate the output value of ecosystem service monetary flows. While the use of official data usually provides advantages in terms of reliability, my use of financial statements offered the benefit of a higher level of disaggregation (Bogaart et al., 2023).

In line with similar findings (see Bogaart et al. 2023 for food provisioning or Zabel et al., 2024 for water supply), my analysis revealed cases of negative resource rent values, highlighting challenges in ES monetary valuation. Negative resource rents can happen, and it is often difficult to understand the cause behind as it could be caused by a range of factors, including low demand, oversupply, exploitation costs exceeding the revenues generated, or external drivers such as regulatory changes, distortion in market prices (Bogaart et al., 2023; Zabel et al., 2024). In this study, this outcome may have been influenced using wholesale price data rather than retail prices to calculate the ES output values, which may not fully reflect revenues from direct or retail sales, with the effect of underestimating the resource rent values.

The production value reported in the financial statements instead reflects the actual prices at which the catch was sold, which do not necessarily coincide with the wholesale prices used, since my estimates are based on annual minimum and maximum values and do not account for interannual price fluctuations. Moreover, cooperatives may also engage in direct retail sales, thus adopting different prices. Finally, due to the level of aggregation of the financial statements, the costs used in the calculation of the resource rent probably included costs related to sea fishing that could not be excluded, and which therefore further reduced the final resource rent value.

Generally, when zero or negative resource rents occur, they can have the implication that the ESs considered does not have a value, and the ecosystem's contribution is null (Obst et al., 2016). Similarly, in the case study analysed in this chapter, overestimating the confidence and the underestimating the limits of the economic method applied and its resulting negative resource rents, weakly related to underlying ecological processes and thus to nature contribution to ES, can lead to misrepresents the contribution of marine inlets and transitional waters to human well-being, and leading to the defined "paradoxical results of irrelevance for ecosystem services" (Femia et al., 2023).

In fact, a key but commonly violated assumption, in ES valuation is that the use of the resources is considered sustainable (Bateman et al., 2011; United Nation, 2014a). It has been observed that, when resource extraction is carried out sustainably, prevailing market structures can drive unit resource rents down to zero, as also in the case of public goods (e.g. water; Zabel et al., 2024; Obst et al., 2016).

Overall, the occurrence of low or negative resource rents values is more likely to reflect the influence of existing market arrangements (such as subsidies) governing the exploitation of the resources (Obst et al., 2016). Another study carried out in Italy, emphasized that these approaches risk to misrepresenting the contribution of ecosystems when prevailing institutional arrangements distort exchange values, as they provide “a poor measure of what is at stake, as they are related neither to the ecological value of ESs, nor to their social value, but represent only the income appropriated by ESs’ economic owners, i.e. by those who use them in production or benefit in asset property.” (Femia et al., 2023; Femia & Capriolo, 2022).

Such studies have therefore concluded that resource rent approaches are reductive (Femia & Capriolo, 2022) and unsuitable in situations where market structures prevent observed prices from reflecting a reasonable exchange value for the ecosystem service in question (Obst et al., 2016), thus requiring future efforts to align and standardize these valuations.

Recognising the abovementioned limitations related to the monetary valuation of ESs, some scholars propose alternative methodologies to evaluate ESs, developing, for example, biophysical evaluation methodologies such as the emergy method (Odum, 1996). Emergy, adopting the so called “donor-side” perspective (Paoli et al., 2020), assesses the ESs “cost of production” in terms of biophysical flows used to support their generation (Ulgiati et al., 2011), a value which could also be converted into monetary estimates representing biosphere investment in monetary equivalents (Vassallo et al., 2017). The strengths recognized to these methodologies are related to their capacity to value ESs independently from the value humans (i.e. the users) attribute to a service (Puselli et el., 2011), potentially overcoming some of the challenges related to services whose value is poorly or not at all reflected due to market mechanisms (Vassallo et al., 2013).

While the application of such approaches falls beyond the scope of this chapter, and is outside the current operational boundaries of SEEA EA, their conceptual contribution as complementary and mutually supportive can be recognised. Together with other alternative valuation methodologies, they can help to reflect about the interpretative limits resulting from the application of an exchange value-based monetary valuation.

The challenge related to monetary accounts is particularly urgent considering the current EU policy context. Regulation (EU) 2024/3024 requires indeed that the Commission (Eurostat), together with Member States, shall assess by June 2026 the methodological feasibility of monetary valuation for ecosystem services, including alternatives where values are missing, in line with SEEA EA standards.

The outcomes of this process may lead to a legislative proposal amending the Regulation to formally integrate monetary accounts, stressing the need to investigate what valuation methods may be capable to capture the ecological nexus between ecosystems and ESs provision.

5.5.3. Beyond the challenges: linking SEEA EA to policy

Vardon et al. (2016) highlighted the central role of the SEEA EA in systematically organising and summarising basic data to generate indicators and aggregates that can inform all stages of the policy cycle. To deliver truly policy-ready evidence, however, ecosystem accounts must align with the analytical requirements, processes, and procedures that shape decision-making.

According to the SEEA EA (United Nations, 2025), the ability to cross-classify ecosystem extent, condition, and service flow data with information on legal and economic ownership is highly policy-relevant, as it supports the assessment of management responsibilities, benefit distribution, and the integration of ecological assets into income, capital, and degradation-adjusted measures.

In the case of lagoons, this partitioning of ownership across service types provides a more accurate reflection of governance arrangements and enables ecosystem accounting outputs to inform both regional resource management and broader national policy objectives. In the case of Sardinia, legal ownership is generally vested in the Regional Government, which grants long-term concessions to fishing cooperatives for their use; a small number of lagoons are privately owned.

From the SEEA EA perspective, this distinction between legal ownership (the entity recognised by law as the owner) and economic ownership (the unit entitled to the benefits from the use of the asset and bearing the associated risks) is essential for linking ecosystem accounts to the national account's framework (United Nations, 2025). In this context, the economic ownership of provisioning services (e.g., wild fisheries, aquaculture) lies primarily with the concessionary cooperatives, while the economic ownership of non-SNA services, such as regulation services (e.g., coastal protection, water quality) and some cultural services is attributed to an *ecosystem trustee*, typically a government sub-sector.

More broadly, linking ecosystem accounting outputs to regional instruments can provide an evidence base that is both consistent with EU-level reporting requirements and tailored to regional governance structures. In this way, the SEEA EA not only supports compliance with statistical obligations but also serves as a strategic tool for integrating ecological assets into decision-making processes across multiple policy domains, from climate adaptation to sustainable resource management.

In Sardinia indeed, several coastal lagoons fall within the boundaries of Natura 2000 and Ramsar sites. Therefore, the Regional Government plays a central role in planning and coordinating conservation and restoration initiatives, as it is responsible for the management of Natura 2000 areas, although some functions are now delegated to lower levels of governance, such as MPAs or municipalities.

At the same time, the ecological classification of lagoon ecosystems is carried out under the Water Framework Directive by regional environmental agencies. SEEA EA data needs and its output align with existing evidence needs since monitoring for ecosystem protection and management can be supported and feed from SEEA EA time series data on ecosystem extent and condition accounts.

Similarly, thanks to the data reported in physical and monetary accounting tables and in a spatially explicit manner, SEEA EA reporting could be particularly useful in highlight possible trade-offs, synergies, as well as general trends in resource exploitation for spatial planning, such as allocated zones for aquaculture or maritime spatial planning plans, in the case of water resources.

Table 5.11 provides some high-level detailed examples of how and to what SEEA EA reporting can meet the evidence needs of some policy instrument, with reference to the Regional Sustainable Development Strategy of Sardinia (RSDS; RAS, 2021 Del. n. 39/56 2021) and the non-exhaustive example of the accounts presented in this chapter. The RSSD integrates climate change considerations, and the conservation and enhancement of ecosystem services into regional planning and decision-making.

It recognizes that human well-being is intrinsically linked to the health of the natural ecosystems to which it belongs. It serves as a strategic framework to guide sustainable development pathways that balance environmental protection, socio-economic growth, and adaptation to climate-related impacts, promoting the long-term well-being of communities and natural systems (RAS, 2021).

Finally, the relevance of the SEEA EA therefore lies not only in its capacity to meet the evidence needs of existing policy instruments, as detailed in table 5.11, but also in the fact that these instruments, in turn, can generate data that feed into and enhance the effectiveness of SEEA EA reporting.

In this sense, SEEA EA not only aligns with policy evidence need thanks to its outputs, but also SEEA EA needs are in line with the needs of the policy evidence too.

Table 5.11 Connection between SEEA EA and Regional Strategy for Sustainable development of Sardinia according to its strategic theme, objectives intervention and proposed actions.

Strategic theme	Strategic objective	Lines of interventions	Proposed actions	How the SEEA EA aligns with evidence needs
2 "Greener Sardinia, for the conservation, management, and enhancement of resources and land for an ecological and resilient transition and climate change"	2.1 "Conserve biodiversity, restore and enhance ecosystem services."	2.1.1 Conservation and monitoring of biodiversity and enhancement of ecosystem services	-Restoration of ecological connectivity and reduction of habitat fragmentation, including through re-naturalisation interventions -Integration of the value of natural capital (ecosystems and biodiversity) into plans, policies and accounting systems -Expansion of the areas of natural heritage under protection -Implementation of appropriate monitoring systems aimed at ensuring the conservation and improvement of the status of biodiversity, endemic species and species at risk, to prevent impacts arising from climate change and anthropogenic pressures	By monitoring, evaluation of changes due to restoration, emerging threats or management interventions supported by time series data on Ecosystem extent and condition accounts. The proposed action "integration of natural capital into plan, policies and accounting systems" is supported by the whole SEEA EA framework
		2.1.2 Restoration of natural heritage through targeted actions for habitats and endangered species	-Monitoring campaigns for endangered species and habitats -Reforestation plan with endemic species and environmental restoration of degraded areas related to soils, also aimed at reducing landslides, capturing CO_2, and mitigating the microclimate	By systematically organizing ecological data for monitoring and evaluation of changes supported by time series data on Ecosystem extent and condition, as well as on ES physical flows
		2.1.3 Raising awareness of the importance of biodiversity for human well-being and health	-Launch communication and information campaigns aimed at raising awareness among agricultural, livestock, and fisheries operators on the importance of safeguarding habitats and species -Communication and information campaigns on the importance of biodiversity for human well-being and health, addressed to local administrations, economic operators, citizens, and schools -Promote education for sustainability, starting from the value of biodiversity conservation and the preservation of ecosystem services, as a basis for achieving social equity, health, economic development, and quality of life	By generating structured evidence on the state of ecosystems and nature contribution through the services they deliver to the economy and society, SEEA EA can provide easily available information for raising awareness on the link between nature and well-being through all the accounts

5.6. Conclusions

This study represents a first attempt to explore the feasibility to compile disaggregated ecosystem accounts at the regional and local scale for coastal lagoons in Sardinia, providing new insights into the complexity of assessing ecosystem stocks and flows within the SEEA EA framework, but also their usefulness for policy and management purposes.

Despite highlighting several gaps, flaws and limitations to the effectiveness of the disaggregated ecosystem accounts operationalization, my results contribute to the growing body of conceptual studies and empirical applications of SEEA EA accounts at sub-national scale. In this study, I have highlighted the methodological and data-related challenges that remain for the full accounting of coastal lagoon ecosystems and related services provided, particularly with regard to ecosystem extent mapping, condition indicators, and the physical quantification of service flows. Data limitation particularly affected the biophysical assessment, for both ecosystem condition and ES flows. Furthermore, a reflection on the conceptual and operational challenges posed using the widely accepted valuation methodologies for monetary accounts, which could lead to misrepresent nature contribution to human wellbeing, is due. Although the debate on this topic is still open, the need for the acceptance of a multifaceted, pluralistic perspective in ES study (Costanza, 2024) that consider the different methodologies that could be adopted to value ESs as complementary and offering different points of view, rather than being considered as "rivals", is being recognised. While data limited full population of the proposed SEEA EA tables, this first exploration can be instructive and offer valuable lessons to regional authorities, resource managers, as well as researchers. For researchers, it identifies methodological priorities, such as the integration of high-resolution spatial data and the refinement of indicators to better capture ecosystem conditions and service flows, while advancing the ecosystem services flow monetary valuation methodologies to be more in line with the aims of the SEEA EA. Being informed with ecosystem accounting data, decision-makers can indeed plan and evaluate actions, strategies and investments oriented towards monitoring, conservation, or restoration of coastal lagoons, considering the ongoing threats posed by climatic changes and the trade-offs among different ecosystem services and relying on a broader, more comprehensive set of information.

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Chapter 6. Summary and future perspectives

Climate change (CC) is projected to affect transitional water ecosystems in multiple ways, altering their physical structure, community composition, ecological processes, and function, ultimately impairing their capacity to provide good and services for human's wellbeing.

Despite initiatives at global, national, and local levels, CC impacts are expected to intensify over this century. As a result, mainstreaming adaptation is increasingly recognized in science and policy as essential for fostering sustainable development. Because ecosystems rarely are isolated, the consequences of CC will extend beyond the ecology. Climate-driven shifts in processes can propagate through cascading effects, undermining the delivery of the ecosystem services (ESs) they provide.

Since ecosystem services flows are often co-produced through the interaction between ecological components and human inputs and are related to human needs (and thus wellbeing), the impacts of climate change also cascade on the human capacity to interact/manage ecosystems, generating cumulative and often non-linear effects on services flows. This perspective highlights the need for a socio-ecological systems (SESs) based approach to effectively integrate the ecological and social dimensions while explicitly accounting for the continuously changing dynamics of SESs, thereby supporting more robust and context-specific adaptation strategies.

Whitin this context, my thesis started from the assumption that in socio-ecological systems, ecological processes, human activities, economic drivers, institutional rules and governance processes co-produce and strongly influence the provision of ecosystem services.

Consequently, CC vulnerability and its components (sensitivity, copying and adaptive capacity) are not properties of a single "half" of the system (as well as the broader concept of risk) but rather they emerge from both as well as from their interactions.

Starting from a narrative literature review (**Chapter 2**), I focused on what is known (and what is missing) about climate change and ES in coastal wetlands. While this chapter confirms the growing interest of the scientific community towards this topic, I also highlight past and current methodological and knowledge gaps that still hamper our ability to fully understand and anticipate future CC impacts on SESs.

Then, I focused on an applicative case studies that link **Chapter 3** (Climate change ecological vulnerability assessment of food provisioning ecosystem services) and **Chapter 4**

(Socio-economic perspectives on climate vulnerability of food provisioning) in which I addressed a comprehensive analysis of two complementary studies carried out in Sardinia (Italy) at the regional level, considering the two sides of the socio-ecological system:

- Ecological vulnerability (Chapter 3), with a focus on ecosystem stocks/assets and conditions involved in ecosystem ability to provide ecosystem services, and
- Social vulnerability (social sensitivity, lack of coping and adaptive capacities) (Chapter 4), with a focus on human capital/assets involved in the interaction with the ecological component to provide ES.

The need to begin the exploration by separating the two components was mainly dictated by the inequity in data availability between them and the subsequent necessity to primarily delve qualitatively into the social component, with the ambition of establishing a baseline for future integrated evaluations.

The results of the study presented in **Chapter 3** revealed interspecific differences in organisms' vulnerability, with species such as *M. cephalus* showing greater robustness to thermal variability, while benthic organisms such as *R. decussatus* emerged as more vulnerable one. Thermal Safety Margin (TSM) indicators identified species near their thermal limit and allowed to highlight species thermal stress hotspots in south-eastern sites of Sardinia, as well as potential species thermal refugia, underlining the spatial unevenness of vulnerability. Habitat vulnerability scores further reflected the spatial differences of ecological status, hydrological exchange, and invasive alien species across sites, thus influencing the vulnerability of the different SES. Importantly, the inclusion of stock status information was seen to influence site-level final ES vulnerability rankings, especially where dominant species accounted for most of the ES flow exhibited uncertainty in stock status or extreme values. These findings demonstrate that omitting stock status can bias such assessments, while its inclusion, despite inherent uncertainties in the data used to build stock assessment models (that can be solved with new monitoring efforts), can provide more realistic insights into site-level vulnerability and, thus, more informative outputs for decision-makers. The findings illustrated in **Chapter 3** then contributed to fill a gap in the European context by scaling down existing vulnerability assessments to the specificity of the Sardinian context, emphasising that vulnerability is unevenly distributed in space, and highlighting the importance of conducting fine-scale regional scale assessment to identify where (and why) management intervention should be targeted. Finally, since this study relied on secondary data only, ecological vulnerability was assessed through a deductive approach based on predefined categories. This obviously represents a potentially severe limitation, which could be addressed in future research by adopting a coupled deductive-inductive approach, combining manipulative experiments and other methods to better characterize site-specific vulnerability factors (sensitivity, capacity to adapt and to cope with) that were not captured here, as well as the interaction among and within them and different hazards.

In **Chapter 4** I explored ES vulnerability of the social component of SES, focusing on both cooperative- and fisher-level scales. My results showed widespread awareness of climate change among both cooperative representatives and individual fishers, with most perceiving current and expected impacts as predominantly negative. Respondents consistently reported experiencing extreme events such as heatwaves, floods, and storm surges, linked to near-mass mortality events in the lagoons, damages to infrastructure, and shifts in species dynamics, including altered fish life cycles and the proliferation of invasive species. Importantly, climate extremes, and notably heatwaves, were also reported to affect fishers directly, not only by disrupting fishing activities, (but also aquaculture, changing farming periods and thus affecting prices) but also by potentially having effects on their physical health, safety, and overall well-being. Furthermore, I showed that while cooperatives represent the formal organizational level, a high dependence on wild fish resources for their income is presented both at cooperative- and fisher-level, ultimately shaping their vulnerability. Such dependence, while increasing their vulnerability to CC, can amplify climate-related impacts in terms of economic damages, occupational insecurity, particularly when barriers such as institutional distrust, delayed or inadequate compensation, and limited internal financial capacity interact. Importantly, my findings revealed intra-group variability, with individual fishers often facing different constraints than the cooperatives, highlighting the value of disaggregating analyses across multiple social scales. Moreover, the study identified barriers and enablers that span multiple dimensions, with perceptions of limited institutional support emerging as a critical potential driver of vulnerability. This highlights the need to consider the issues of trust, local agency, and multi-level governance into adaptation strategies, as they can either constrain or enhance the capacity to adapt and respond to CC. By capturing and exploring these dynamics, **Chapter 4** contributes also to the broader debate on socio-ecological vulnerability in small-scale fisheries with a socio-ecological lens, providing a first case study of this kind in Sardinian lagoons.

In **Chapter 5**, Monitoring ecosystems and their services in a changing climate: Experimenting the SEEA EA at the local and regional scales in Sardinia, I explored the feasibility of compiling accounts according to the SEEA EA framework using available data sources for Sardinian transitional waters, and notably coastal lagoons, including the identification of knowledge and data gaps, and critically reviewing how SEEA-based accounting can support and inform identified relevant sectoral policies at regional scale. While in the previous chapter I explored where and possibly why the socio-ecological systems are vulnerable, the SEEA EA, a recognized international statistical standard, can potentially offer a structured assessment framework through the periodic monitoring of changes in ecosystems and their services over time.

In **Chapter 5** I described the potential of ecosystem accounting to inform cross-sectoral management and policy decisions, also considering the ongoing threats posed by climatic change, providing a bridge between ecological assessments and governance tools. Likewise, as seen in the previous chapters, the availability, the choice of indicators and the data uncertainty handling played a central role, as these limitations affected the reliability and operationalization of the accounts, especially in effectively tracking ecosystem condition and their services over time. In this sense, this chapter emphasized the importance of environmental monitoring, as well as the possibility that SEEA-EA data needs offer to collect data and build indicators from the different accounts capable to capture changes, not only in ecosystem condition, but also in the vulnerability on the ecological compartment of the socio-ecological system.

6.1. Strengthen local evidence and monitoring efforts: a road ahead for effective CC research in SESs

Considering the findings of my thesis, I contend that the management of the socio-ecological systems will increasingly benefit from approaches that integrate ecological evidence with social and economic perspectives. In conclusion, looking ahead, the following future directions to improve our capacity to understand these complex relationships can be drawn. First, new empirical data collection should be prioritized on:

- gather species-specific tolerance thresholds to key climate related environmental variables, such as temperatures, derived from combined manipulative studies and in situ observations
- strengthen the evaluation of stock assessment status in transitional ecosystems, and particularly in coastal lagoons, validating model outputs with recruitment and population dynamics monitoring, to better capture changes in fish populations and disentangle the different effects of anthropogenic (fishing) and climate drivers
- collect ecological data on currently underrepresented sites and variables, such as benthic community dynamics, invasive species interactions, and habitat connectivity, which strongly influence climate vulnerability but remain to date poorly quantified
- collect relevant socio-economic data with a particular focus on information on livelihoods, dependence on ESs, and other vulnerability metrics to better characterize the of the social dimension of SES

Second, existing monitoring programmes and initiatives should strengthen their applicability by:

- Increasing their temporal resolution and spatial coverage for key environmental parameters (e.g. temperature, salinity, dissolved oxygen, nutrient concentrations) also through the operationalization of real-time monitoring systems, which are currently only at the pilot stage, so that they can effectively support and complement existing programmes.
- Updating and validating high resolution maps for habitats in coastal and transitional ecosystems at sub-national scale.
- Enhance data collection efforts on ESs flows to avoid ambiguity and bias in the existing dataset.

More broadly, these efforts will need to be embedded within governance systems that promote the integration of multi- and interdisciplinary research, in order to better capture the multifaceted impacts of CC on socio-ecological systems.

Strengthening the dialogue between disciplines, and between qualitative, quantitative and mixed methods approaches, while taking advantage of the new opportunities provided by emerging integrated frameworks, such as ecosystem accounting, will be essential. At the same time, fostering collaborations and participatory processes between authorities, research institutions, and local communities can ensure that the produced evidence will not be only scientifically sound but also operationally relevant and rooted in the reality.

Such an approach can help rebuild trust between local communities and institutions or academia and ensure that CC research is developed within contexts where the complexity of socio-ecological systems is fully recognized. In this way, the generation of new integrated knowledge will not only enhance our capacity to anticipate and mitigate climate-driven risks but will also support proactive governance strategies that are not just researching *"about"* climate change and SESs but are developed from the bottom *"for"* addressing climate change and a transition toward a transformative adaptation.

6.2. Concluding remarks

Significant progress has been made in the field of socio-ecological systems and climate change. This is true in academic research but also policy and decision making, as an increasing number of peer-review studies, policy strategies and actors address a range of aspects within this theme. However, considering the current limitations, the applicative research using socio-ecological integrated approaches can, will and must continue to evolve. Finally, though aware that the case studies included here represent only a few pieces of the complex mosaic of socio-ecological systems, this thesis has put light on aspects which, while relevant at first instances only at a regional scale, can nonetheless contribute to support advancing in future implementations of local-based CC adaptation strategies.

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Appendix 1 - Articles included in the literature review

Table A1. 1 Articles included in the literature review.

Authors	Year	Title	Source title	Reference
Chen X.; Li M.; Zhang Z.	2025	Climate change challenges coastal blue carbon restoration in China	Journal of Environmental Management	Chen et al., 2025
Alemu I J.B.; Ofsthun C.; Medley G.; Bowden A.; Cammett A.; Gildesgame E.; Munoz S.E.; Stubbins A.; Randall Hughes A.	2024	Evaluating ecosystem services in urban salt marshes: Assessing vulnerability to sea-level rise and implications for coastal management	Journal of Environmental Management	Alemu et al., 2024
Collin R.; Adelson A.E.; Altieri A.H.; Clark K.E.; Davis K.; Giddings S.N.; Kastner S.; Mach L.; Pawlak G.; Sjögersten S.; Torres M.; Scott C.P.	2024	Using forty years of research to view Bahía Almirante on the caribbean coast of Panama as an integrated social-ecological system	Estuarine, Coastal and Shelf Science	Collin et al., 2024
Cunha J.; Cabecinha E.; Villasante S.; Gonçalves J.A.; Balbi S.; Elliott M.; Ramos S.	2024	Quantifying the role of saltmarsh as a vulnerable carbon sink: A case study from Northern Portugal	Science of the Total Environment	Cuhna et al., 2024
Costa M.; Wartman M.; Macreadie P.I.; Ferns L.W.; Holden R.L.; Ierodiaconou D.; MacDonald K.J.; Mazor T.K.; Morris R.; Nicholson E.; Pomeroy A.; Zavadil E.A.; Young M.; Snartt R.; Carnell P.	2024	Spatially explicit ecosystem accounts for coastal wetland restoration	Ecosystem Services	Costa et al., 2024
Debnath S.; Sarkar U.K.; Kumari S.; Karnatak G.; Puthiyottil M.; Das B.K.; Das A.; Ghosh B.D.; Roy A.	2024	Exploring the vulnerability of the coastal wetlands of India to the changing climate and their adaptation strategies	International Journal of Biometeorology	Debnath et al., 2024
Harrison, E; Stephenson, F; Rullens, V; Pilditch, C; Ellis, J	2024	Predicting the cumulative effects of multiple stressors on shellfish ecosystem service potential	Ocean & Coastal Management	Harrison et al., 2024
Ouyang X.; Maher D.T.; Santos I.R.	2024	Climate change decreases groundwater carbon discharges in global tidal wetlands	One Earth	Ouyang et al., 2024
Roy M.S.; Byrnes J.E.K.; Mavrommati G.	2024	Mitigation policies buffer multiple climate stressors in a socio-ecological salt marsh habitat	Sustainability Science	Roy et al., 2024
Cassalho F.; de S. de Lima A.; Ferreira C.M.; Henke M.; de A. Coelho G.; Miesse T.W.; Johnston J.; Coleman D.J.	2023	Quantifying the effects of sea level rise driven marsh migration on wave attenuation	Environmental Monitoring and Assessment	Cassalho et al., 2023

Authors	Year	Title	Source title	Reference
Cervantes-Escobar A.; Ruiz-Luna A.; Berlanga-Robles C.A.	2023	Social perceptions of ecosystem services delivered by coastal wetlands: their value and the threats they face in northwestern Mexico	Ethnobiology and Conservation	Cervantes-Escobar et al., 2023
Kotagama O.W.; Pathirage S.; Perera K.A.R.S.; Dahanayaka D.D.G.L.; Miththapala S.; Somarathne S.	2023	Modelling predictive changes of blue carbon due to sea-level rise using InVEST model in Chilaw Lagoon, Sri Lanka	Modeling Earth Systems and Environment	Kotagama et al., 2023
Li C.; Fang S.; Geng X.; Yuan Y.; Zheng X.; Zhang D.; Li R.; Sun W.; Wang X.	2023	Coastal ecosystem service in response to past and future land use and land cover change dynamics in the Yangtze river estuary	Journal of Cleaner Production	Li et al., 2023
Meixler, MS; Piana, MR; Henry, A	2023	Modeling present and future ecosystem services and environmental justice within an urban-coastal watershed	LANDSCAPE AND URBAN PLANNING	Meixler M.S. et al., 2023
Tanner K.; Strong A.L.	2023	Assessing the Impact of Future Sea Level Rise on Blue Carbon Ecosystem Services on Long Island, New York	Sustainability (Switzerland)	Tanner et al., 2023
Fernández-Díaz V.Z.; Canul Turriza R.A.; Kuc Castilla A.; Hinojosa-Huerta O.	2022	Loss of coastal ecosystem services in Mexico: An approach to economic valuation in the face of sea level rise	Frontiers in Marine Science	Fernández-Díaz V.Z. et al., 2022
Guyondet, T; Filgueira, R; Pearce, CM; Tremblay, R; Comeau, LA	2022	Nutrient-Loading Mitigation by Shellfish Aquaculture in Semi-Enclosed Estuaries	Frontiers in Marine Science	Guyondet et al., 2022
Moritsch M.M.; Byrd K.B.; Davis M.; Good A.; Drexler J.Z.; Morris J.T.; Woo I.; Windham-Myers L.; Grossman E.; Nakai G.; Poppe K.L.; Rybczyk J.M.	2022	Can Coastal Habitats Rise to the Challenge? Resilience of Estuarine Habitats, Carbon Accumulation, and Economic Value to Sea-Level Rise in a Puget Sound Estuary	Estuaries and Coasts	Moritsch et al., 2022
Nyangoko B.P.; Berg H.; Mangora M.M.; Shalli M.S.; Gullström M.	2022	Community perceptions of climate change and ecosystem-based adaptation in the mangrove ecosystem of the Rufiji Delta, Tanzania	Climate and Development	Nyangoko et al., 2022
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Appendix 2 - Stock assessment methodology

CMSY (Martell and Froese 2013; Froese et al., 2017) is a widely used method for assessing fish stocks in data-limited situation. By estimating Maximum Sustainable Yield (MSY) and related reference points using only catch records and basic information that can be generated by default rules (Bouch et al., 2021), CMSY simulate population trajectories with surplus production models, retaining those consistent with catches and species resilience (Froese et al., 2017). CMSY assume logistic population growth, reliable catch data, and plausible values for intrinsic growth rate (r) and carrying capacity (K). The required estimation of resilience of the stock, ranging from high, medium, low or very low, can be determined by mean of expert knowledge or from resources such as FISHBASE (Froese et Pauly, 2018). The prior about biomass, for start, intermediate and final biomass ranges can be defined by expert opinion or by default rules as described in Froese et al., (2017). Its newer version, CMSY++ (Froese et al., 2023) and a Bayesian implementation of a modified Schaefer model are nested within a single JAGS model (Froese et al., 2023). CMSY ++ now offer advantages in considering the degree of technological creep, as well as handling better catch uncertainty, resilience priors, as well as diagnostics, providing more robust and reliable stock assessments.

If on one side its main strengths are represented by minimal data needs, simple implementation, and transparent assumptions, limitations include sensitivity to biomass priors, lack of age/size structure, reduced reliability with short or biased data (Martell & Froese, 2013; Rosenberg et al., 2014), as well as the tendency on overestimating fishing mortality and underestimating biomass (Bouch et al., 2021). However, CMSY is recognized in international frameworks for precautionary management (GFCM 2022a, 2022b, 2023, 2024). Catch-fishery dependent data for the period 2006–2024 for four commercial species caught in Sardinian coastal lagoons were used for this study. Catch data were obtained from the official statistical source, which is the Fisheries Service of the Autonomous Region of Sardinia (RAS). Below, the characteristics of each raw stock dataset are presented. The stock assessment was performed using the R software (R version 4.3.1) environment and the CMSY++ (CMSY_2019_9f.R) model code developed and freely accessible by Froese and co-authors. (2023).

Table A2. 1 Summary of the raw stock data and time-series.

Scientific name	English name	Lagoon	Time series
Mugil cephalus	Flathead grey mullet	Calich	2006-2023
Mugil cephalus	Flathead grey mullet	Nora	2007-2024
Mugil cephalus	Flathead grey mullet	Avalè su Petrosu	2008-2017
Mugil cephalus	Flathead grey mullet	Cabras	2007-2024
Mugil cephalus	Flathead grey mullet	Colostrai	2010-2024
Mugil cephalus	Flathead grey mullet	Feraxi	2005-2024
Mugil cephalus	Flathead grey mullet	Peschiera San Giovanni	2010-2024
Mugil cephalus	Flathead grey mullet	Tortolì	2007-2024
Mugil cephalus	Flathead grey mullet	Is Benas	2010-2024
Mugil cephalus	Flathead grey mullet	Pauli Bianco Turri	2006-2023
Mugil cephalus	Flathead grey mullet	Porto Pino	2006-2024
Mugil cephalus	Flathead grey mullet	S'Ena Arrubia	2006-2024
Mugil cephalus	Flathead grey mullet	Sa Praia	2008-2024
Mugil cephalus	Flathead grey mullet	Santa Gilla	2006-2024
Mugil cephalus	Flathead grey mullet	Santa Giusta	2010-2024
Sparus aurata	Gilthead seabream	Calich	2006-2023
Sparus aurata	Gilthead seabream	Nora	2006-2024
Sparus aurata	Gilthead seabream	Avalè su Petrosu	2004-2024
Sparus aurata	Gilthead seabream	Cabras	2007-2024
Sparus aurata	Gilthead seabream	Colostrai	2006-2024
Sparus aurata	Gilthead seabream	Feraxi	2005-2024
Sparus aurata	Gilthead seabream	Peschiera San Giovanni	2008-2024
Sparus aurata	Gilthead seabream	Tortolì	2007-2024
Sparus aurata	Gilthead seabream	Is Benas	2010-2024
Sparus aurata	Gilthead seabream	Pauli Bianco Turri	2006-2023
Sparus aurata	Gilthead seabream	Porto Pino	2006-2024
Sparus aurata	Gilthead seabream	S'Ena Arrubia	2006-2024
Sparus aurata	Gilthead seabream	Sa Praia	2008-2024

Scientific name	English name	Lagoon	Time series
Sparus aurata	Gilthead seabream	Santa Gilla	2006-2024
Sparus aurata	Gilthead seabream	Santa Giusta	2011-2024
Dicentrarchus labrax	European seabass	Calich	2006-2023
Dicentrarchus labrax	European seabass	Nora	2006-2024
Dicentrarchus labrax	European seabass	Avalè su Petrosu	2004-2024
Dicentrarchus labrax	European seabass	Cabras	2007-2024
Dicentrarchus labrax	European seabass	Colostrai	2006-2024
Dicentrarchus labrax	European seabass	Feraxi	2005-2024
Dicentrarchus labrax	European seabass	Peschiera San Giovanni	2008-2024
Dicentrarchus labrax	European seabass	Tortolì	2007-2024
Dicentrarchus labrax	European seabass	Is Benas	2010-2024
Dicentrarchus labrax	European seabass	Pauli Bianco Turri	2006-2023
Dicentrarchus labrax	European seabass	Porto Pino	2009-2024
Dicentrarchus labrax	European seabass	S'Ena Arrubia	2006-2024
Dicentrarchus labrax	European seabass	Sa Praia	2009-2023
Dicentrarchus labrax	European seabass	Santa Gilla	2006-2024
Dicentrarchus labrax	European seabass	Santa Giusta	2010-2024
Ruditapes decussatus	Grooved carpet shell	Feraxi	2005-2019
Ruditapes decussatus	Grooved carpet shell	Peschiera San Giovanni	2008-2024
Ruditapes decussatus	Grooved carpet shell	S'Ena Arrubia	2012-2021
Ruditapes decussatus	Grooved carpet shell	Santa Gilla	2011-2024

Species stock status was thus be defined according to the B/BMSY and F/FMSY following Zhai et al., 2020:

Table A2. 2 Definition of fish stock status for fisheries management adopted in this work (Zhai et al., 2020).

***B/B*MSY**	***F/F*MSY**	**Stock Status**
>1	<1	Healthy stocks
>1	>1	Healthy stock size about to be depleted by overfishing
>0.5	<1	Recovering stocks
<0.5	<1	Fully overfished/outside of safe biological limits
>0.2&<1	>1	Fully overfished/outside of safe biological limits
<0.2	>1	Severely depleted

Input Parameters and Data

The following parameters are used as model inputs: resilience of species and prior relative biomass (B/k) ranges corresponding to depletion levels at the start, intermediate, and end time series. Resilience is a preliminary (or 'prior') estimate of the resilience of the species, corresponding to the intrinsic growth rates. The possible classes are 'high', 'medium', 'low', 'very low'. The r ranges assigned for CMSY++ were obtained from www.FishBase.org (accessed on March 2024) and represents the 'atypical' range of intrinsic growth rate of a species population specified as r.low–r.hi.

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Appendix 3 – Questionnaires

Fishers' questionnaire

Questionnaire Introduction
Dear participant,
With this survey we are interested in collecting information on the past and future impacts of climate change on fishing and aquaculture activities in the coastal wetlands of Sardinia. The aim of this research is to evaluate how these changes have influenced in the past and will influence the work activity of fishermen and aquaculture farmers. The data we will collect will be useful to have a first overview of the vulnerability of workers in this sector, and to begin to think about how to improve the response and adaptation capabilities of fishermen and aquaculture farmers in the future. The collection of these data is carried out within the PhD Project of Elisa Serra, in collaboration between the University of Cagliari, the IMC Foundation - International Marine Centre (Torregrande) and the University of East Anglia (UK).

1 General Perception and Impacts Section

In this section of the questionnaire, we are focusing on your experience and perceptions of environmental changes regarding your activities in [name of the coastal wetland].

1)What is your current activity?

☐ Fishing

☐ Aquaculture

☐ Both

2) Have you experienced (in the past / currently) any environmental changes that have affected your work in [name of the coastal wetland] in any way? What and how did they affect you? Please tell me in your own words. [open ended]

In recent years, there has been an increasing awareness about climate change and extreme weather events, as well as their possible effects on economic activity, natural resources and human well-being. Below we will ask you some questions on this topic.

3)Do you think the climate is changing?

☐ Yes

☐ No

☐ I don't know

4) To date, do you think that climate change is having any effect on your work in [name of the coastal wetland]?

☐ Yes

☐ No

☐ I don't know

5) What kind of effect?

☐ Positive

☐ Negative

☐ Both

☐ I don't know

6) Why? Can you give an example? [open ended]

7) Do you think that climate change will have any effect on your work in [name of the coastal wetland] in the future?

☐ Yes

☐ No

☐ I don’t know

8)What kind of effect?

☐ Positive

☐ Negative

☐ Both

☐ I don’t know

9) Why? Can you give an example? [open ended]

10) Of the following extreme events caused by climatic changes, which have you witnessed in [name of the coastal wetland] where you work?

☐ Heatwaves

☐ Storm surges

☐ Floods

☐ Other (specify)____

☐ None

11) If you selected “extreme events such as storm surges,” what effects have you experienced on your activities in [name of the coastal wetland]?

☐ Change in catch quantities (☐ increase / ☐ decrease)

☐ Deaths

☐ Change in fishing seasons

☐ Change in the composition of fished species

☐ Damage to equipment

☐ Damage to fishing and/or aquaculture infrastructure

☐ Interruption of activities due to impossibility of carrying out work during adverse weather conditions

☐ Other (specify): __________

☐ I don't know

☐ None

12) If you selected “extreme events such as flooding,” what effects have you experienced on your activities in [name of the coastal wetland]?

☐ Change in catch quantities (☐ increase / ☐ decrease)

☐ Deaths

☐ Change in fishing seasons

☐ Change in the composition of fished species

☐ Damage to equipment

☐ Damage to fishing and/or aquaculture infrastructure

☐ Interruption of activities due to impossibility of carrying out work during adverse weather conditions

☐ Other (specify): __________

☐ I don't know

☐ None

13) If you selected “extreme events such as heat waves,” what effects have you experienced on your activities in [name of the coastal wetland]?

☐ Change in catch quantities (☐ increase / ☐ decrease)

☐ Deaths

☐ Change in fishing seasons

☐ Change in the composition of fished species

☐ Other (specify): __________

☐ I don't know

☐ None

Section 2 - Focus on heatwaves

Previously we talked about impacts on the ecosystem or on the facilities related to your work. We would like to ask a few more questions about heat waves and how they affect your work activity

14) During the summer (june-september), how many hours do you work outdoors on average per day? [open ended]

15) During the summer, what times do you work outdoors? [open ended]

16) Has intense heat ever affected your work performance?

☐ No

☐ Yes, it slowed down my work. Specify why ____

☐ Yes, I had to stop working. Specify why ____

☐ Other (specify): __________

17) The intensity and frequency of intense heat events are expected to increase in the future. Has intense heat ever affected your work performance?

☐ No

☐ Yes

18) Why? [open ended]

19) Do you think heatwaves will also influence your wellbeing? If so, why? [open ended]

Vulnerability Section - Adaptive and cope capacities

In this section we will ask you for some additional information, useful for better understanding factors that may determine vulnerability to the impacts of climate change on your job.

20) How old are you? [open ended]

☐ I prefer not to answer

21) What is your level of education?

☐ None

☐ Primary education

☐ Lower secondary education [middle school]

☐ Upper secondary education [high school]

☐ bachelor's degree

☐ master's degree

☐ PhD

☐ Prefer not to answer

22) How many years have you been doing this job? [open ended]

23) How many years have you been working in [name of the coastal wetland]? [open ended]

24) Is fishing and/or aquaculture your only source of income?

☐ Yes

☐ No

☐ I prefer not to answer

☐ Other (specify)

25) Do you collaborate or share knowledge with other cooperatives or other fishers to address challenges related to climate change? How? [open ended]

26) Do you consider collaboration or sharing of knowledge and/or resources with other cooperatives or other fishermen important to address the challenges posed by climate change in your work sector? If so, how/why? [open ended]

Final section

27) Thank you for the information you provided us while filling out the questionnaire. After addressing the topic of climate change in relation to your activities, is there anything else you would like to add?

____

Cooperatives' Questionnaire

Questionnaire Introduction
Dear participant,
With this survey we are interested in collecting information on the past and future impacts of climate change on fishing and aquaculture activities in the wetlands of Sardinia. The aim of this research is to evaluate how these changes have influenced the activities of the cooperative and what are the factors that determine and could determine in future the vulnerability of the SME to climate change. The data we will collect will be useful to have an initial overview of this work sector at a regional level, and to begin to think about how to improve the response and adaptation capabilities of cooperatives in the future. The collection of these data is carried out within the PhD Project of Elisa Serra, in collaboration between the University of Cagliari, the IMC Foundation - International Marine Centre (Torregrande) and the University of East Anglia (UK).

General section with socio-demographic information

1) Name of the cooperative [open ended]

2) How many years have you been working in this cooperative? [open ended]

3) What economic activities does the cooperative carry out?

☐ Fishery
☐ Aquaculture
☐Both

4) Does the cooperative carry out other economic activities in [name of the coastal wetland]? (e.g. fish tourism, excursions)

☐ Yes
☐No

4a) If yes, which ones? [open ended]

Section 1 - General impacts and perception on fisheries/aquaculture activity based on past experience

5) What is the biggest threat, problem, or challenge your sector is facing? [open ended]

6) Has the [name of coastal wetland] ever been affected by environmental changes that have affected the SME's activities?

☐ No, we have never had environmental changes that have affected the cooperative's activities
☐ Yes, Extreme events such as heat waves
☐ Yes, Extremes such as storm surges
☐ Yes, Extreme events such as floods
☐ Yes, Presence of new species
☐ Other (specify): __________

7) With reference to the changes or events you stated in the previous question, what kind of change have you observed? (example: more or less frequent events, more or less intense events, new events that were not observed before)

(To fill in only those previously declared)	Enter what perception they have of the change/event	I don't know / I'm not sure
Extreme events such as heat waves		
Extreme events such as storm surges		
Extreme events such as floods		
Presence of new species		
Other change		

8) In recent years, there has been an increasing discussion about climate change and extreme weather events, as well as their possible effects on economic activities, natural resources and human well-being. Do you think that the changes or events we discussed in the previous two questions are caused by climate change?

(To fill in only those previously declared)	Yes, totally	Yes, but only partly	No (why?)	I don't know / I'm not sure
Extreme events such as heat waves				
Extreme events such as storm surges				
Extreme events such as floods				
Presence of new species				
Other change				

9) What kind of influence did they have on the cooperative's activities?

(To fill in only those previously declared)	Positive	Negative	Both	I don't know / I'm not sure
Extreme events such as heat waves				
Extreme events such as storm surges				
Extreme events such as floods				
Presence of new species				
Other change				

Section 2 – Extreme events, magnitude of impacts on fisheries/aquaculture activity based on past experience

Now we would like to know more about the problems caused by extreme events that have negatively affected the SME's fishing/aquaculture activities. If you have never encountered this problem, go to the next section.

10) In which years did you observe the following extreme events that caused damage to the cooperative's activities? What type of impacts/damage did they cause? (One row for each year reported)

Extreme event name	Mass mortality event	Decrease in catch	Equipment damage	Damage to infrastructure	Other
Year					

Subsection 2 - Monetary estimates of damages from extreme events

11) Can you provide us an economic estimate of the damage suffered for each category? Please report the estimate in terms of “k of euros”. (One row for each year reported)

Extreme event name	Mass mortality event	Decrease in catch	Equipment damage	Damage to infrastructure	Other
Year					

12) Did the damage we just talk about cause a stop in fishing and/or aquaculture activities? If so, for how much and what kind of lost earnings did it cause? [open ended]

Section 3 - Presence of new species, magnitude of impacts on fishing/aquaculture activities based on past experience

In this section we would like to learn more about the presence of new species in the lagoon. If you have not observed this phenomenon, go to the next section.

13) Is the blue crab present in [name of the coastal wetland]?

☐ Currently Yes

☐ Currently no

☐ Currently no, but it has been present in the past

14) Do you consider this species as a threat or an opportunity for the cooperative's activity? Why? [open ended]

15) Does your market this species?

☐ Yes, as a fresh product

☐ Yes, as a processed product

☐ Yes, both

☐ No

16) If so, can you tell us how much blue crab (unprocessed) has sold in recent years? (One row for each year reported)

Year	Kg
Year	

17) If it is not marketed, what is the reason and what is the alternative usage or end-life? (e.g. self-consumed, waste disposal)

____

18) Has this species caused damage to the cooperative's fishing and/or aquaculture activities?

☐ No

☐ Yes, damage to equipment

☐ Yes, decrease in catch

☐ Other (specify) _____

19) If so, can you provide a monetary estimate? [open ended]

20) Is this species present in [name of the coastal wetland]?]

☐ Currently Yes

☐ Currently no

☐ Currently no, but it has been present in the past

21) Do you consider this species as a threat or an opportunity for the cooperative's activity? Why? [open ended]

22) Has this species caused damage to the cooperative's fishing and/or aquaculture activities?

☐ No

☐ Yes, damage to equipment

☐ Yes, decrease in catch

☐ Other (specify) _____

23) If yes, can you provide a monetary estimate? [open ended]

24) Is this species present in [name of the coastal wetland]?

☐ Currently Yes

☐ Currently no

☐ Currently no, but it has been present in the past

25) Do you consider this species as a threat or an opportunity for the cooperative's activity? Why?] [open ended]

26) Has this species caused damage to the cooperative's fishing and/or aquaculture activities?

☐ No

☐ Yes, damage to equipment

☐ Yes, decrease in catch

☐ Other (specify) ______

27) If yes, can you provide a monetary estimate?]

28) Is this species present in [name of the coastal wetland]?

☐ Currently Yes

☐ Currently no

☐ Currently no, but it has been present in the past

29) Do you consider this species as a threat or an opportunity for the cooperative's activity? Why? [open ended]

30) Has this species caused damage to the cooperative's fishing and/or aquaculture activities?

☐ No

☐ Yes, damage to equipment

☐ Yes, decrease in catch

☐ Other (specify) ______

31) If yes, can you provide a monetary estimate? [open ended]

Section 4 – Cope and adaptive capacities

32) Has the SME ever implemented strategies to address climate change?

☐ No

☐ Fishing season changed

☐ Modernization of the cooperative's infrastructure

☐ Diversified economic activity

☐ Other (specify) ______

32a) If not, why [open ended]

33) Has the SME ever requested financial support to address problems caused by extreme weather events?

☐ Yes

☐ No

33a) If yes, which ones? [open ended]

33b) If not, why did you never ask for? [open ended]

34) Given the requests made, did the SME receive the requested subsidies? Give reasons for your answer [open ended]

35) Has the SME ever requested financial support to address interruptions in fishing activities caused by climate events?

☐ Yes

☐ No

☐ There have never been interruptions in fishing activities caused by climate events

35a) If yes, which ones? [open ended]

35b) If not, why did you never ask for? [open ended]

36) Given the requests made, did the SME receive the requested subsidies? Give reasons for your answer [open ended]

37) Has the SME ever requested economic support to address problems caused by invasive species?

☐ Yes

☐ No

37a) If yes, which ones? [open ended]

37b) If not, why did you never ask for? [open ended]

38) Given the requests made, did the SME receive the requested subsidies? Give reasons for your answer [open ended]

39) Has your SME ever received technical-scientific support from research institutes or business support organizations to address problems caused by extreme weather events or climate change?

☐ Yes

☐ No, I have never requested technical-scientific support

☐ No, I am interested in technical-scientific support, but I have never received it

☐ Other (specify) _____

39a) If yes, what kind of support did you receive? [open ended]

40) Are there factors that facilitate or have facilitated your cooperative in implementing climate change adaptation or mitigation strategies? Which ones? [open ended]

41) What factors could facilitate your SME in the future in implementing climate change adaptation or mitigation strategies? [open ended]

42) Have there been any barriers or factors that actually limit or have limited your SME in implementing climate change adaptation or mitigation strategies? Which ones? [open ended]

43) What do you think are the barriers or factors that will limit your SME in implementing climate change adaptation or mitigation strategies? [open ended]

43a) Do you think you can overcome these barriers? If so, how? If not, why? [open ended]

Final section

44) Thank you for the information you provided us while filling out the questionnaire. After addressing the topic of climate change in relation to SME's activities, is there anything else you would like to add?

____

Appendix 4 – Ecological vulnerability

Table A4. 1 Thermal safety margin (TSM) and TSM scores for each species in each site.

Species	Site	TSM	TSM score
Mugil cephalus	Avale su Petrosu	8.78	0.11
	Cabras	9.43	0.04
	Calich	9.14	0.07
	Colostrai	8.94	0.09
	Feraxi	9.85	0
	Is benas	9.32	0.05
	Nora	7.84	0.2
	Pauli Biancu	9.34	0.05
	Porto Pino	8.84	0.1
	Sa Praia	9.09	0.08
	Peschiera San Giovanni	9.72	0.01
	Santa Gilla	9.7	0.02
	Santa Giusta	9.14	0.07
	S'Ena Arrubia	8.68	0.12
	Tortoli	8.27	0.16
Dicentrarchus labrax	Avale su Petrosu	0.95	0.9
	Cabras	1.6	0.84
	Calich	1.31	0.87
	Colostrai	1.11	0.89
	Feraxi	2.02	0.8
	Is benas	1.49	0.85
	Nora	0.01	1
	Pauli Biancu	1.51	0.85
	Porto Pino	1.01	0.9
	Sa Praia	1.26	0.87
	Peschiera San Giovanni	1.89	0.81
	Santa Gilla	1.87	0.81
	Santa Giusta	1.31	0.87
	S'Ena Arrubia	0.85	0.91
	Tortoli	0.44	0.96
Sparus aurata	Avale su Petrosu	2.39	0.76
	Cabras	3.04	0.69
	Calich	2.75	0.72
	Colostrai	2.55	0.74
	Feraxi	3.46	0.65
	Is benas	2.93	0.7
	Nora	1.45	0.85
	Pauli Biancu	2.95	0.7
	Porto Pino	2.45	0.75
	Sa Praia	2.7	0.73
	Peschiera San Giovanni	3.33	0.66
	Santa Gilla	3.31	0.66
	Santa Giusta	2.75	0.72
	S'Ena Arrubia	2.29	0.77
	Tortoli	1.88	0.81
Ruditapes decussatus	Feraxi	2.48	0.75
	Peschiera San Giovanni	2.35	0.76
	Santa Gilla	2.33	0.76
	S'Ena Arrubia	1.31	0.87

Table A4. 2 Stock assessement results used to calculate the "central", "optimistic" and "pessimistic" scenarios.

Species	Stock code	B/Bmsy	F/Fmsy	lcl B/Bmsy	ucl B/Bmsy	lcl F/Fmsy	ucl F/Fmsy
European seabass	BSS-Stagno Avale su Petrosu	0.37	1.05	0.12	0.75	0.35	9.97
	BSS-Stagno di Cabras	0.83	0.81	0.49	1.27	0.39	1.85
	BSS-Laguna del Calich	1.11	0.71	0.80	1.42	0.42	1.21
	BSS-Stagno di Colostrai	0.88	1.34	0.55	1.20	0.75	2.56
	BSS-Stagno di Feraxi	0.67	0.84	0.31	0.89	0.49	3.05
	BSS-Stagno Is Benas	0.35	1.22	0.11	0.50	0.49	12.02
	BSS-Nora	0.49	0.99	0.26	0.80	0.47	3.90
	BSS-Stagno Pauli Bianco Turri	0.67	1.12	0.42	0.98	0.61	2.37
	BSS-Stagno Porto Pino	0.80	0.94	0.51	1.10	0.53	1.77
	BSS-Stagno Sa Praia	0.70	1.79	0.40	1.05	0.84	5.13
	BSS- Peschiera San Giovanni	1.30	0.50	0.93	1.59	0.28	0.92
	BSS-Stagno Santa Gilla	1.13	0.71	0.80	1.46	0.41	1.24
	BSS-Stagno Santa Giusta	1.13	0.68	0.77	1.46	0.36	1.33
	BSS-Stagno S'Ena Arrubia	1.26	0.64	0.92	1.58	0.34	1.23
	BSS-Stagno di Tortoli	0.58	1.10	0.34	0.91	0.56	3.07
Flathead grey mullet	MUF-Stagno Avale su Petrosu	0.73	1.09	0.37	1.04	0.55	3.10
	MUF-Stagno di Cabras	0.18	0.44	0.05	0.45	0.07	5.63
	MUF-Laguna del Calich	0.50	1.07	0.23	0.90	0.50	5.08
	MUF-Stagno di Colostrai	0.85	1.29	0.50	1.20	0.70	2.70
	MUF-Stagno di Feraxi	1.46	0.53	1.09	1.72	0.29	0.98
	MUF-Stagno Is Benas	0.29	1.18	0.14	0.63	0.29	5.81
	MUF-Nora	0.53	0.84	0.23	0.83	0.41	3.96
	MUF-Stagno Pauli Bianco Turri	0.22	0.69	0.06	0.42	0.18	9.16
	MUF-Stagno Porto Pino	0.69	0.92	0.35	0.99	0.51	2.77
	MUF-Stagno Sa Praia	0.37	1.08	0.13	0.75	0.35	8.50
	MUF- Peschiera San Giovanni	1.18	0.65	0.84	1.43	0.40	1.11
	MUF-Stagno Santa Gilla	1.27	0.63	0.93	1.59	0.38	1.05
	MUF-Stagno Santa Giusta	1.22	0.65	0.87	1.53	0.36	1.27
	MUF-Stagno S'Ena Arrubia	1.25	0.72	0.88	1.56	0.40	1.36
	MUF-Stagno di Tortoli	0.76	1.31	0.44	1.16	0.66	3.16
Gilthead seabream	SBG-Stagno Avale su Petrosu	0.67	1.78	0.37	0.97	1.03	4.89
	SBG-Stagno di Cabras	0.40	0.90	0.20	0.76	0.31	3.75
	SBG-Laguna del Calich	1.26	0.71	0.87	1.53	0.42	1.27
	SBG-Stagno di Colostrai	0.72	1.43	0.48	0.99	0.85	2.61

Species	Stock code	B/Bmsy	F/Fmsy	lcl B/Bmsy	ucl B/Bmsy	lcl F/Fmsy	ucl F/Fmsy
	SBG-Stagno di Feraxi	0.49	1.26	0.22	0.81	0.59	6.23
	SBG-Stagno Is Benas	0.65	1.69	0.41	0.96	0.95	3.69
	SBG-Nora	0.58	2.09	0.34	1.10	0.79	6.39
	SBG-Stagno Pauli Bianco Turri	1.17	0.74	0.80	1.54	0.41	1.37
	SBG-Stagno Porto Pino	0.31	0.92	0.11	0.61	0.27	6.72
	SBG-Stagno Sa Praia	0.50	1.08	0.22	0.82	0.53	5.75
	SBG-Peschiera San Giovanni	1.12	0.92	0.77	1.45	0.55	1.62
	SBG-Stagno Santa Gilla	1.06	0.96	0.72	1.34	0.55	1.70
	SBG-Stagno Santa Giusta	1.35	0.56	0.99	1.61	0.34	0.97
	SBG-Stagno S'Ena Arrubia	0.51	0.91	0.27	0.88	0.40	3.57
	SBG-Stagno di Tortoli	1.43	0.29	1.10	1.69	0.16	0.54
Ruditapes decussatus	CTG-Stagno di Feraxi	0.37	1.11	0.15	0.70	0.40	6.50
	CTG-Stagno di S. Giovanni	0.53	1.95	0.26	0.85	0.87	7.84
	CTG-Stagno Santa Gilla	0.75	1.04	0.41	1.14	0.55	2.51
	CTG-Stagno S’Ena Arrubia	0.56	0.93	0.25	0.91	0.47	4.34

Appendix 5 – Methodological notes on monetary valuation

Fishing cooperatives that operate in Sardinian coastal lagoons under concession (where they hold economic ownership but not legal ownership) are required to pay an annual fee. This fee is defined by Ministerial Decree No. 595/1995, which set the baseline rules and the amount to be paid for domain concessions. The amount has been updated through the more recent Regional Law of Sardinia No. 9/2018, which called for a new regional decree to revise these fees.

However, the concession fee currently paid by fishing cooperatives operating in Sardinian coastal lagoons does not reflect the ecological characteristics that underpin the ES capacity of these ecosystems, unlike agricultural land rents that may vary according to acreage, soil fertility, hydrological conditions, or the presence of pollinators, thus indirectly capturing the value of ecosystem services (SEEA EEA 5.64). Pursuant to Ministerial Decree No. 595/1995, subsequently recalled by Regional Law of Sardinia No. 9/2018, the fee is determined based on the surface area of the water body under concession, rather than on ecological parameters (e.g. habitat quality, hydrological regime, biodiversity) that directly influence the provision of ecosystem services. As a result, for Sardinian coastal lagoon the fee represents a purely administrative rent, disconnected from the actual biophysical and ecological value of the resource. Thus, rental prince cannot be used to evaluate the ES in our case study. This is why, even violating some assumptions[1], I calculated the ES value using the resource rent methods, similarly to other studies.

For the Resource rent calculation, I considered the output values derived from biomasses and wholesale prices provided by ISMEA, using both minimum and maximum price

[1] SEEA EEA 5.81: However, a number of market conditions must be in place for estimates of unit resource rent to constitute accurately a price for the ecosystem services that takes into account the potential for degradation of the resource. These conditions, among others, require that the resource be extracted or harvested in a sustainable way and that the owner of the resource seek to maximize his or her resource rent. 5.82 Often, these conditions are not met. In particular, if there is open access to the resources and no charge by the owner for access, then the marginal unit resource rent will approach zero, thereby implying that the price of the ecosystem service is zero. Thus, whether the resource rent approach to valuing ecosystem services is appropriate will depend on the access conditions in place. (p.119)

ranges. These values were complemented with subsidies directly related to lagoon fisheries, while costs were subtracted.

Cost items explicitly attributable to marine resource extraction, as reported in the financial statements, were excluded from the analysis, as well as subsidies connected to such activities. Since the financial statements were presented in a simplified form, it was not possible to disentangle other potential cost components in detail. However, it can be reasonably assumed that the incidence of additional costs linked to marine resource extraction is negligible compared to the costs sustained for the exploitation of lagoon resources, except for salaries, which can contribute to bias in my analysis, requiring further investigation on how disaggregate these costs.

Example of the nature of individual revenue/cost items of exceptional significance or incidence that were eliminated for the calculation of resource rent based on my output values and for the calculation of resource rent based on their financial statements includes the compensation payments for sea clearances due to military exercises., which can be attributed solely to marine fishing.

Appendix 6 – Kobe plot

In the following figures the Kobe plot from CMSY++ outputs are presented, representing the time series of F/Fmsy on the y-axis and of B/Bmsy on the x-axis. The plot is divided into four quadrants, defined for the stock biomass and fishing mortality relative to BMSY and FMSY, respectively and the 4 traditional categories used for Kobe phase plot. The orange area indicates healthy stock about to be depleted by overfishing; the red quadrant indicates that the stock is overfished and is undergoing overfishing. The yellow area indicates reduced fishing pressure on stocks recovering from still too low biomass levels. The green area indicates sustainable fishing pressure and healthy stock size capable of producing high yields close to maximum sustainable yield. The shape around the assessment of the final year triangle indicates uncertainty for 50%, 80% and 95% confidence levels. The legend in the upper right graph also indicates the probability of the last year falling into each of the coloured areas.

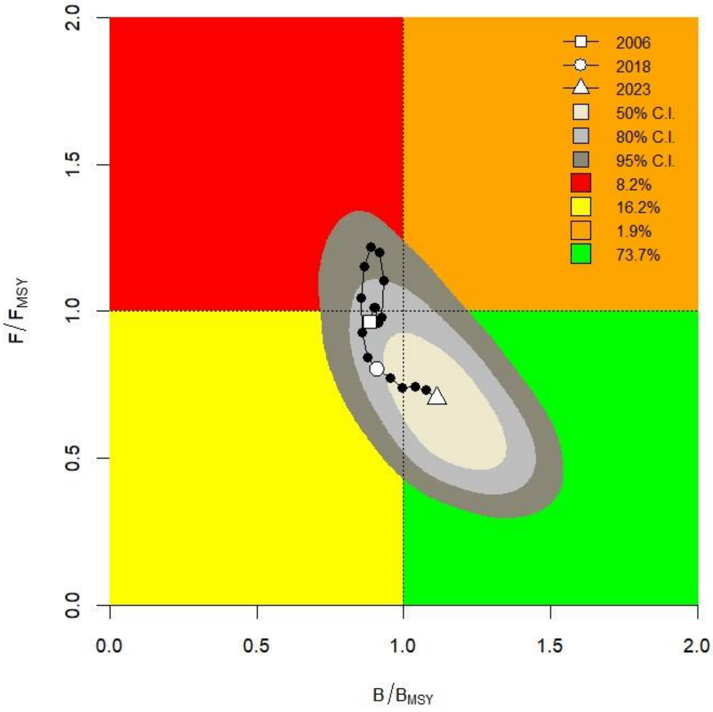

Figure A6. 1 Kobe plot for Dicentrarchus labrax *in Calich lagoon.*

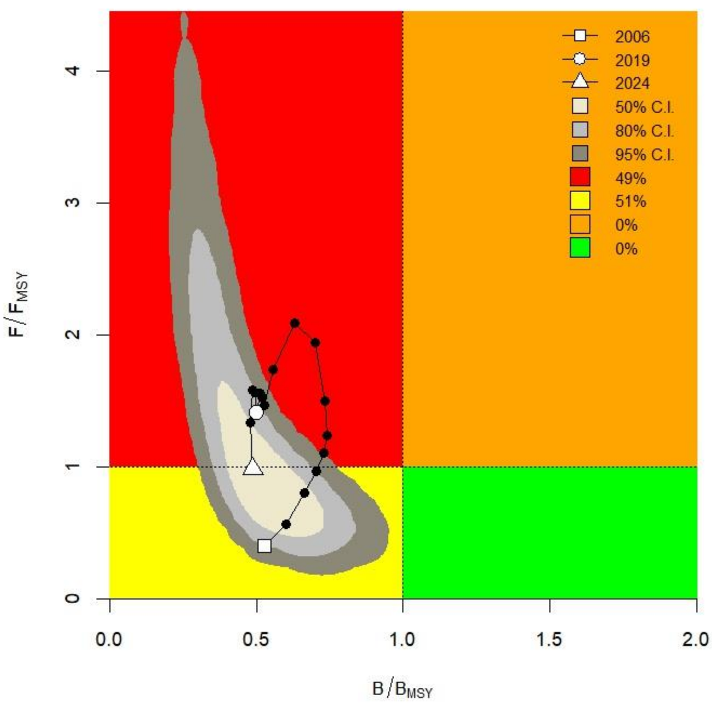

Figure A6. 2 Kobe plot for Dicentrarchus labrax *in Nora lagoon.*

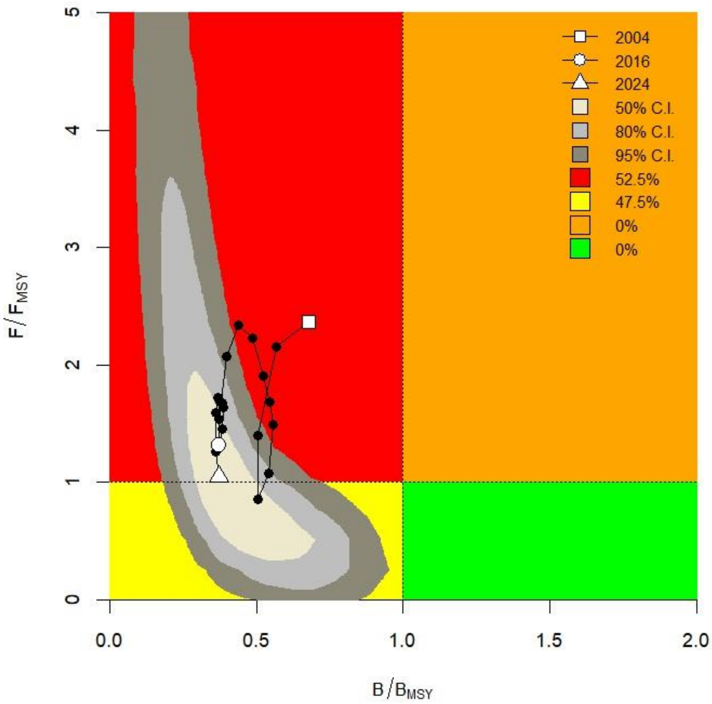

Figure A6. 3 Kobe plot for Dicentrarchus labrax *in Avalè su petrosu lagoon.*

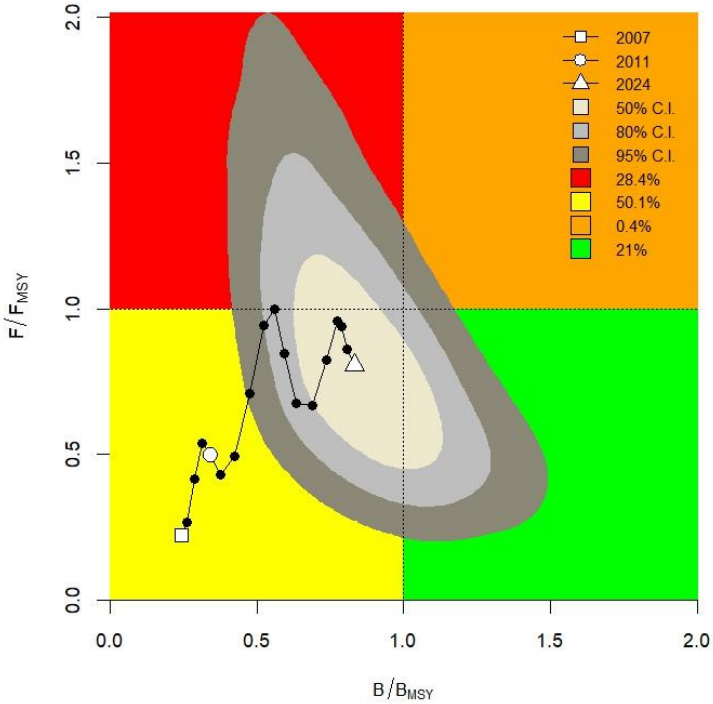

Figure A6. 4 Kobe plot for Dicentrarchus labrax *in Cabras lagoon.*

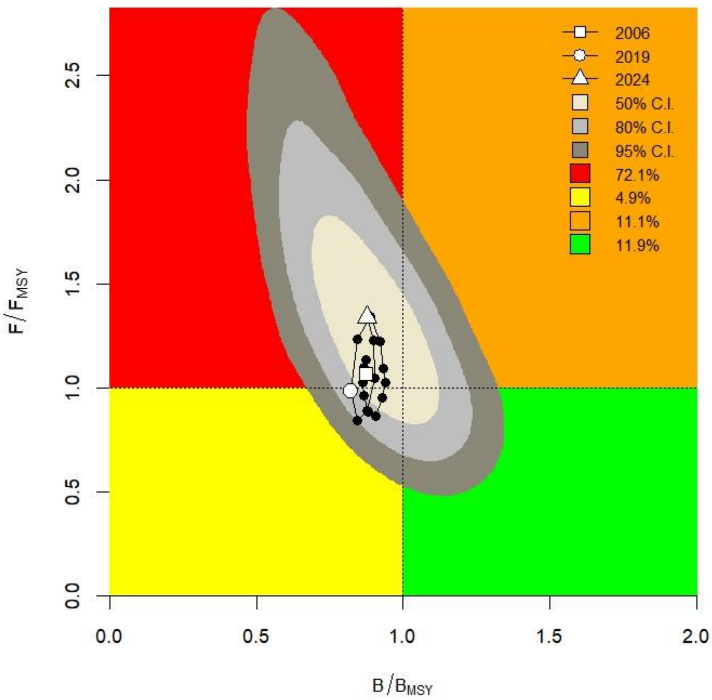

Figure A6. 5 Kobe plot for Dicentrarchus labrax *in Colostrai lagoon.*

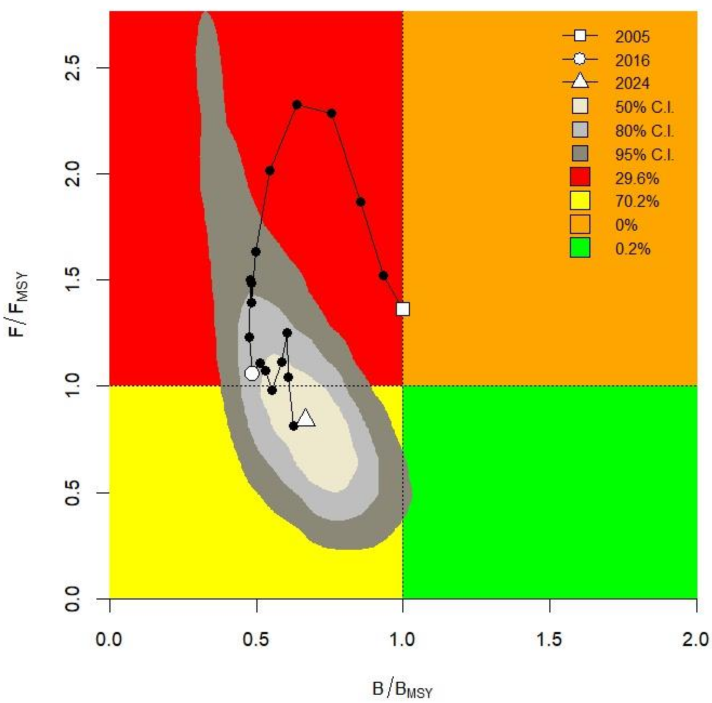

Figure A6. 6 Kobe plot for Dicentrarchus labrax *in Feraxi lagoon.*

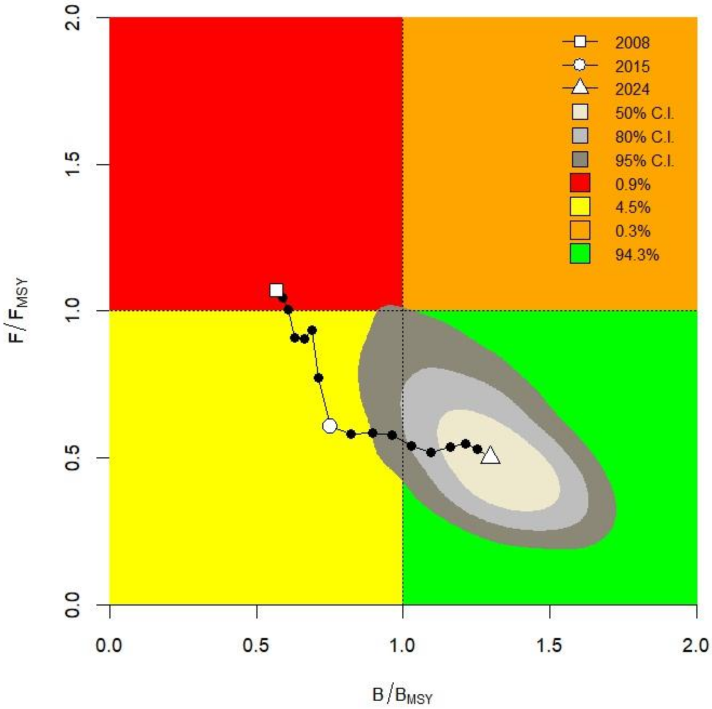

Figure A6. 7 Kobe plot for Dicentrarchus labrax *in Peschiera San Giovanni lagoon.*

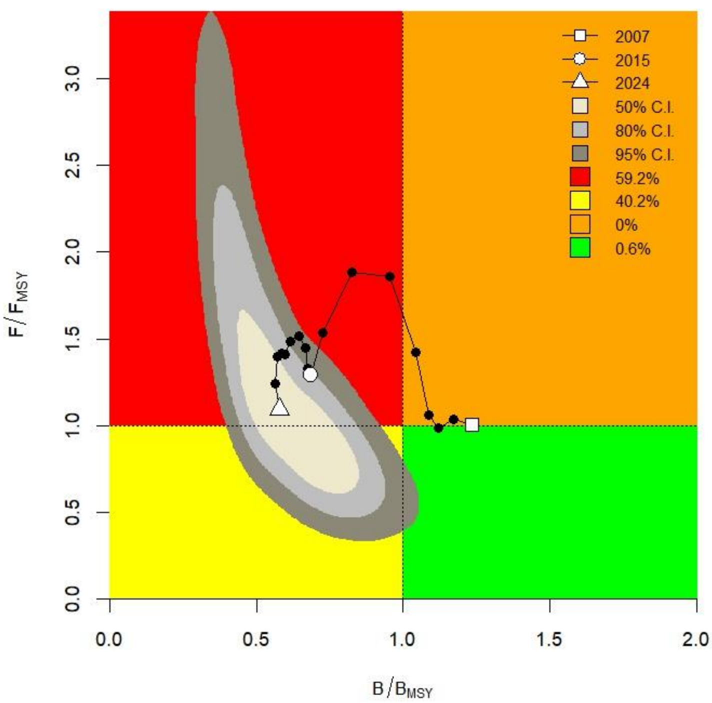

Figure A6. 8 Kobe plot for Dicentrarchus labrax *in Tortolì lagoon.*

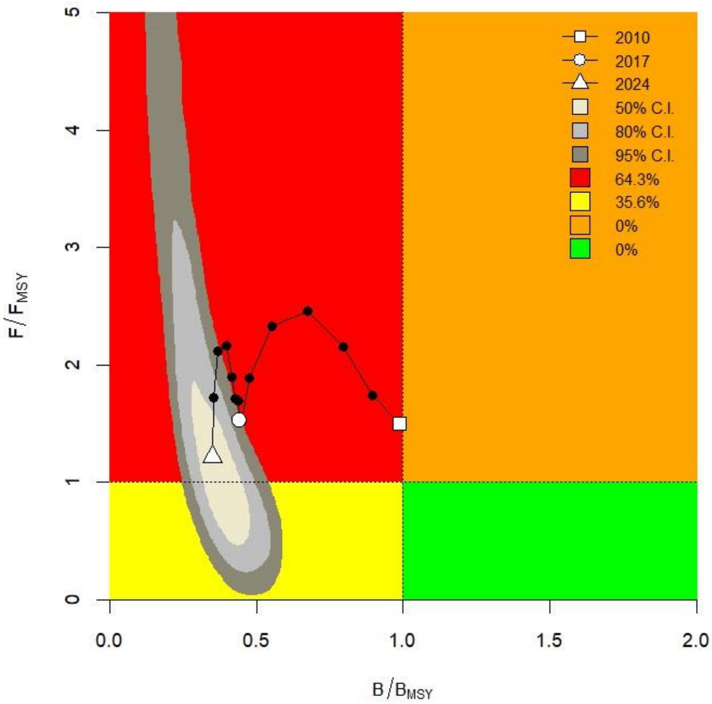

Figure A6. 9 Kobe plot for Dicentrarchus labrax *in Is Benas lagoon.*

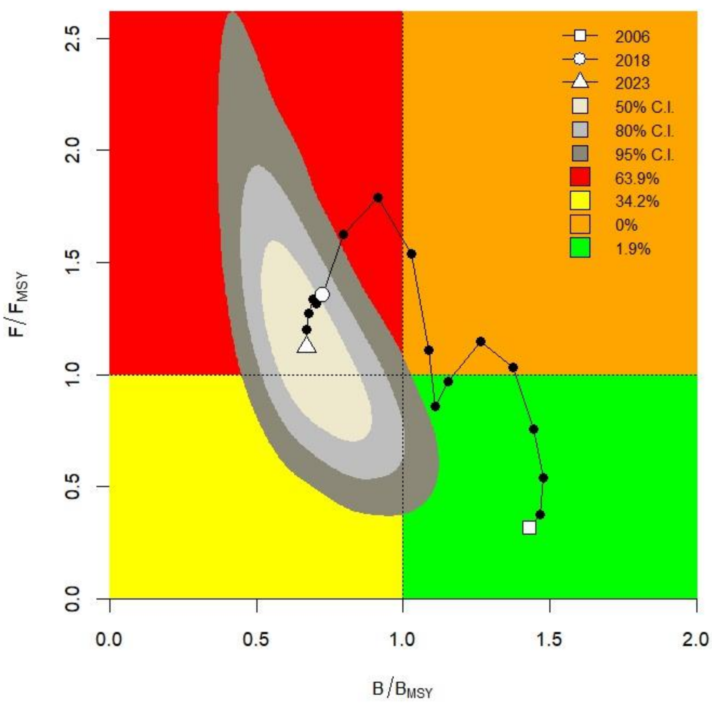

Figure A6. 10 Kobe plot for Dicentrarchus labrax *in Pauli biancu Turri lagoon.*

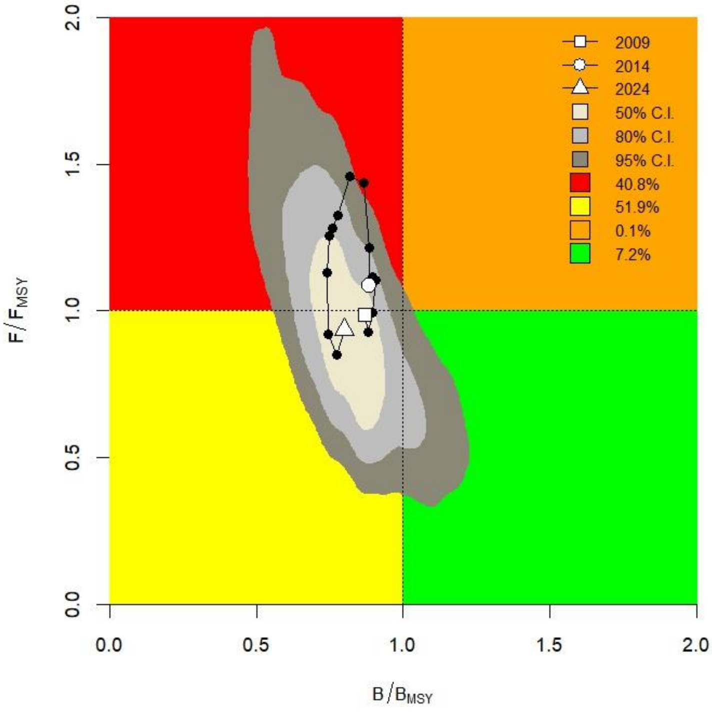

Figure A6. 11 Kobe plot for Dicentrarchus labrax *in Porto Pino lagoon.*

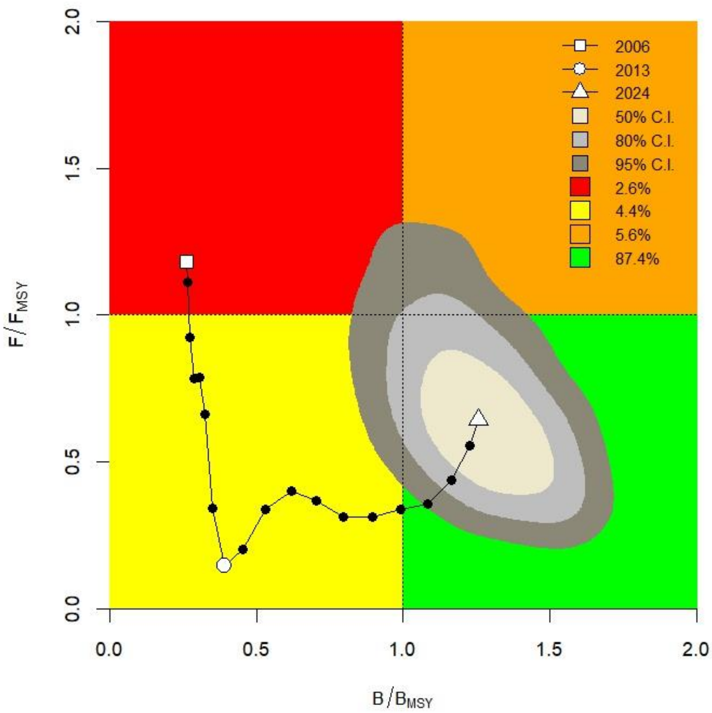

Figure A6. 12 Kobe plot for Dicentrarchus labrax *in S'Ena Arrubia lagoon.*

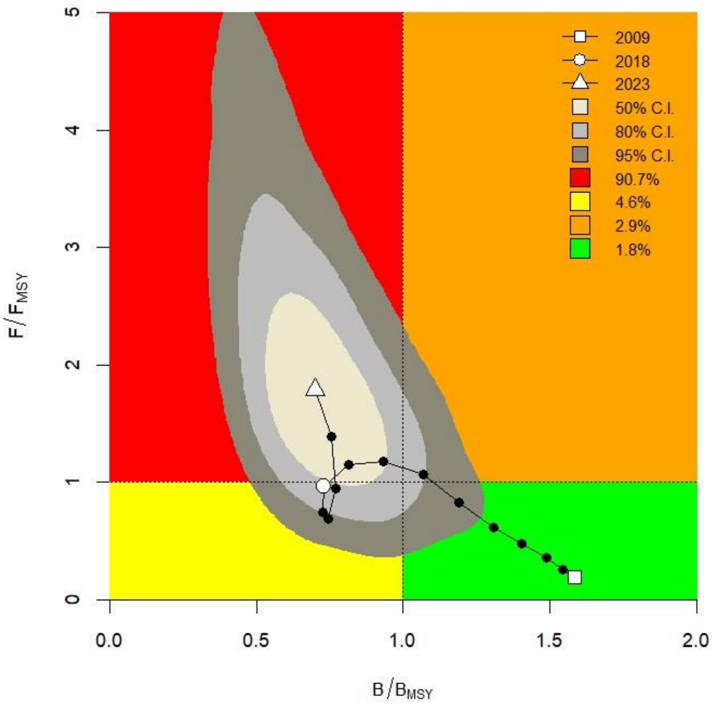

Figure A6. 13 Kobe plot for Dicentrarchus labrax *in Sa Praia lagoon*

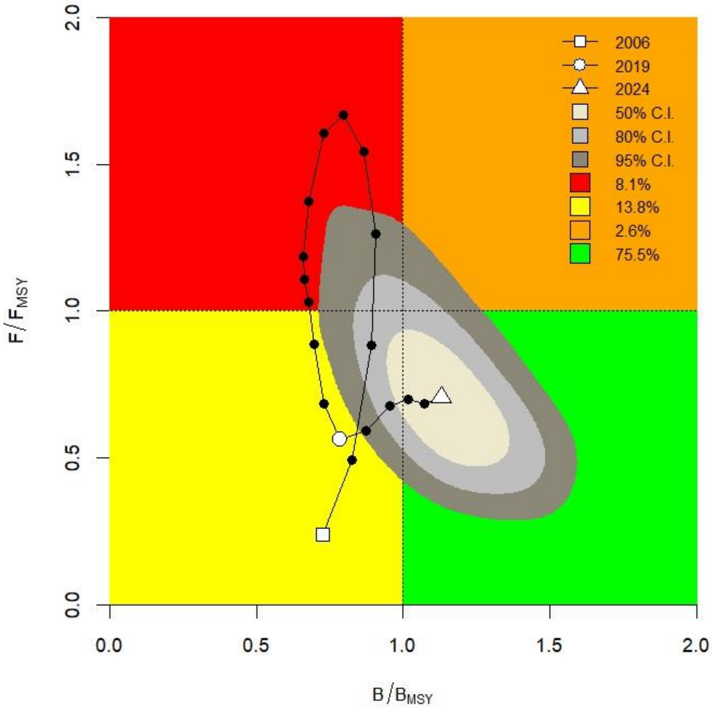

Figure A6. 14 Kobe plot for Dicentrarchus labrax *in Santa Gilla lagoon.*

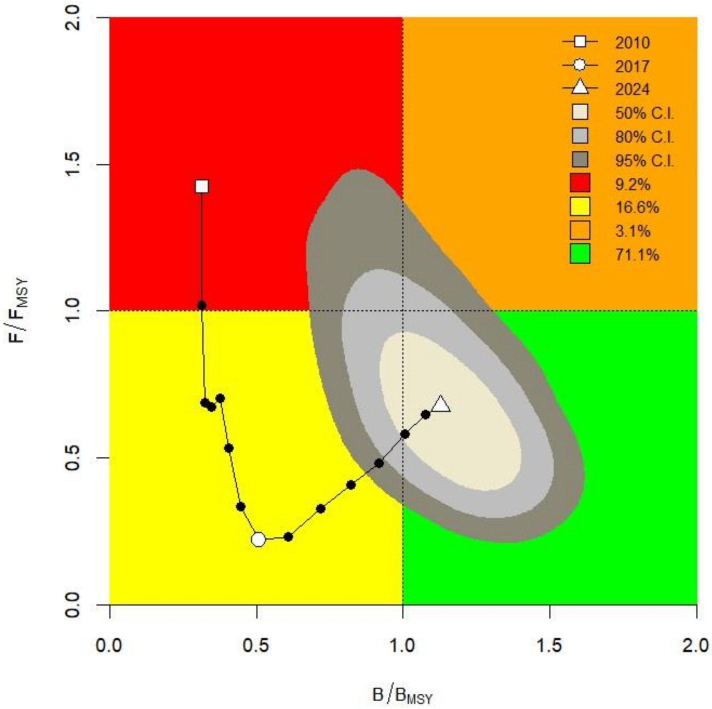

Figure A6. 15 Kobe plot for Dicentrarchus labrax *in Santa Giusta lagoon.*

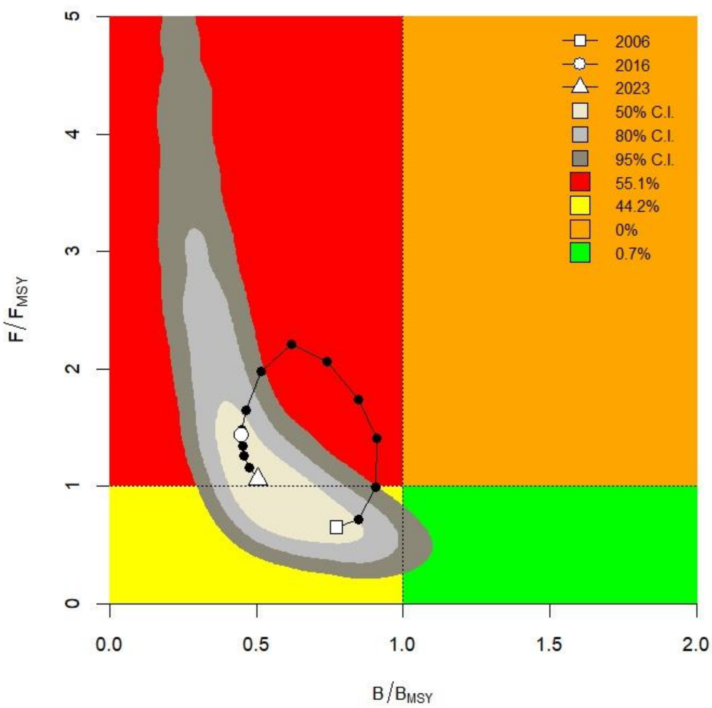

Figure A6. 16 Kobe plot for Mugil cephalus *in Calich lagoon.*

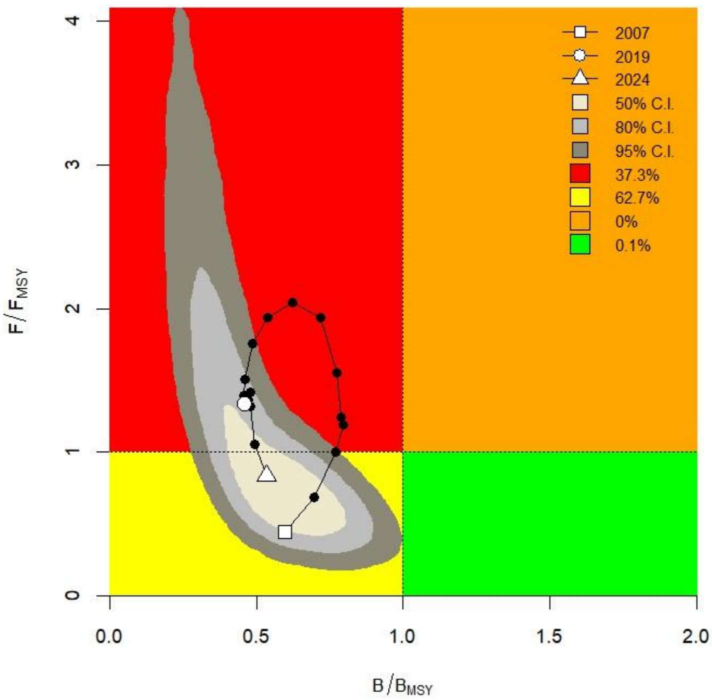

Figure A6. 17 Kobe plot for Mugil cephalus *in Nora lagoon.*

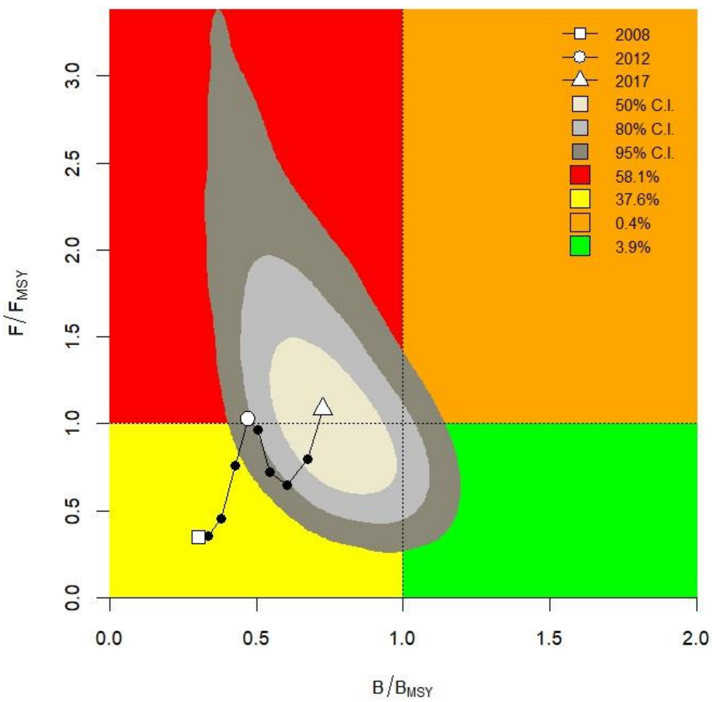

Figure A6. 18 Kobe plot for Mugil cephalus *in Avalè su petrosu lagoon*

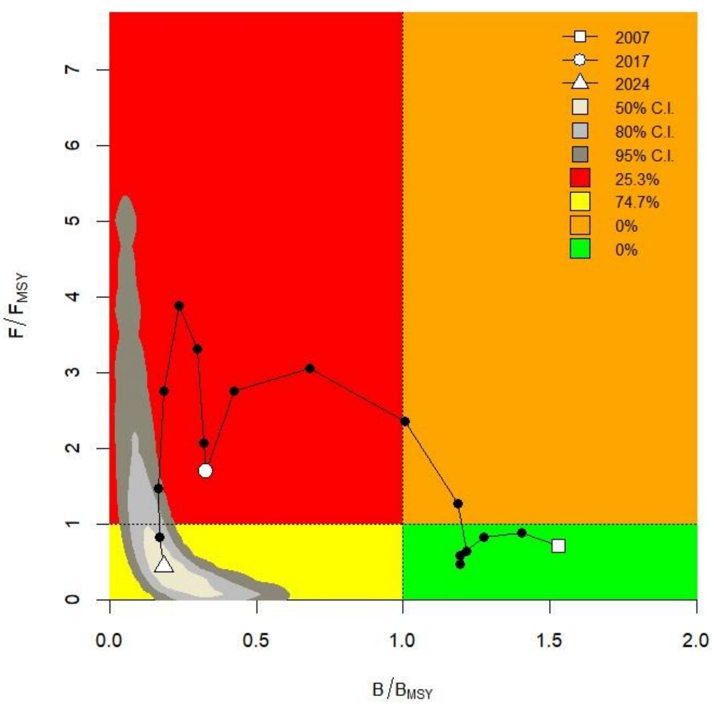

Figure A6. 19 Kobe plot for Mugil cephalus *in Cabras lagoon*

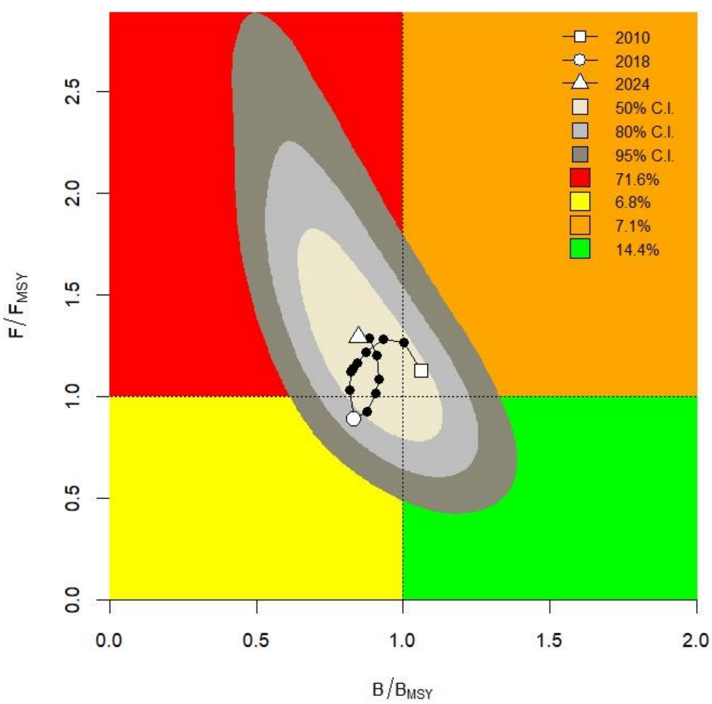

Figure A6. 20 Kobe plot for Mugil cephalus *in Colostrai lagoon*

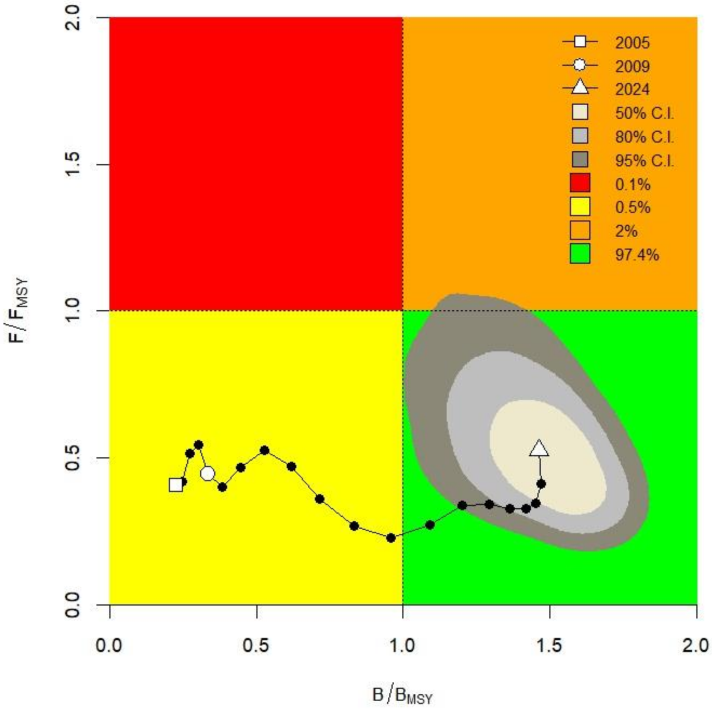

Figure A6. 21 Kobe plot for Mugil cephalus *in Feraxi lagoon.*

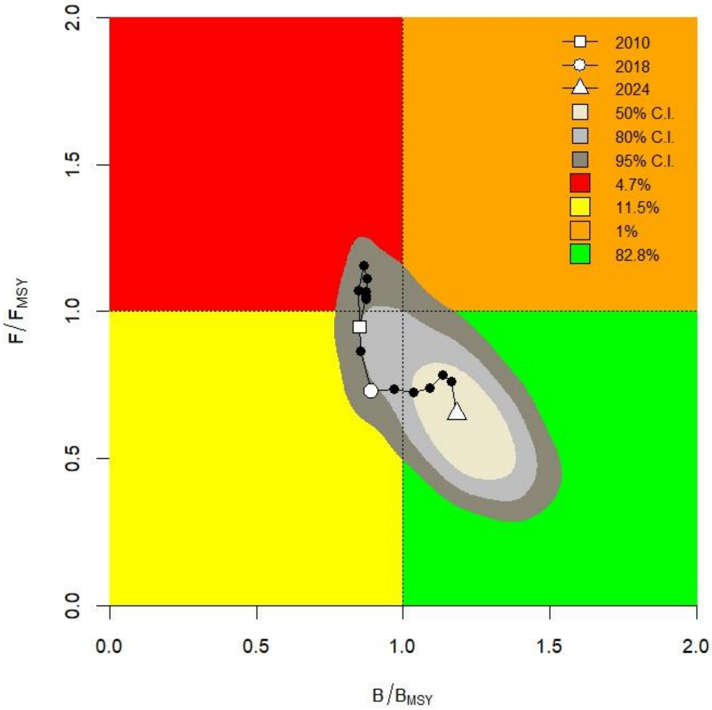

Figure A6. 22 Kobe plot for Mugil cephalus *in Peschiera San Giovanni lagoon.*

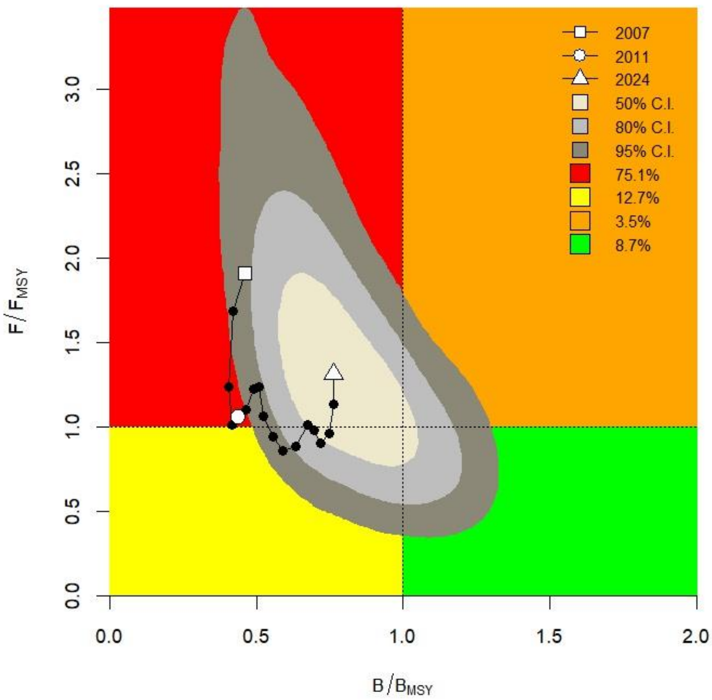

Figure A6. 23 Kobe plot for Mugil cephalus *in Tortolì lagoon.*

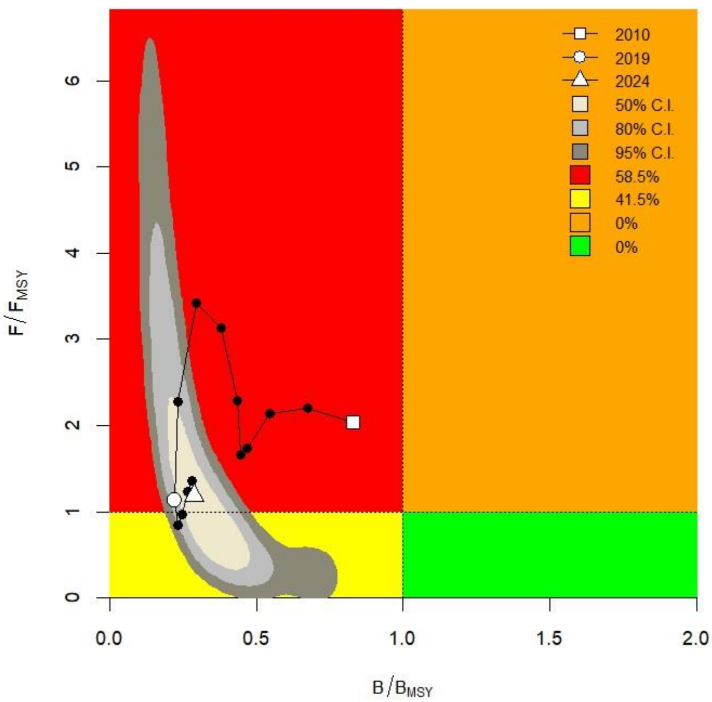

Figure A6. 24 Kobe plot for Mugil cephalus *in Is Benas lagoon.*

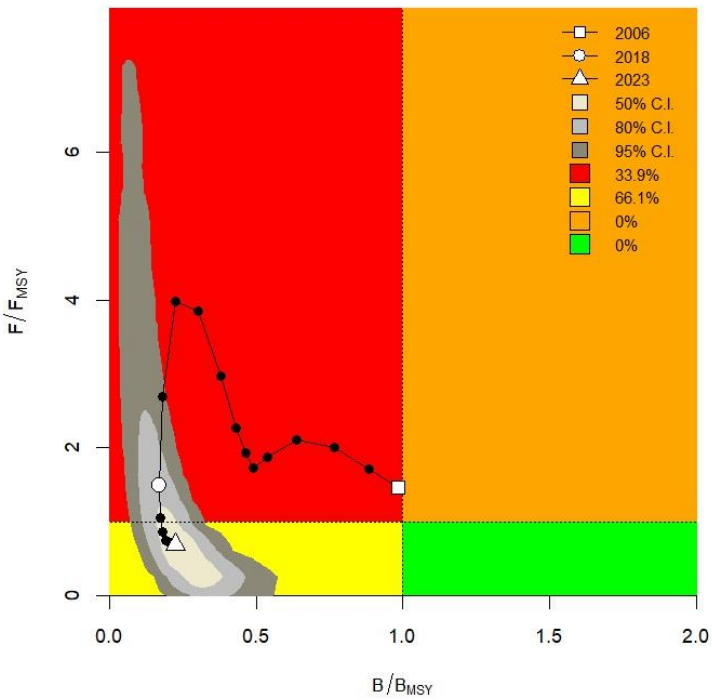

Figure A6. 25 Kobe plot for Mugil cephalus *in Pauli biancu Turri lagoon.*

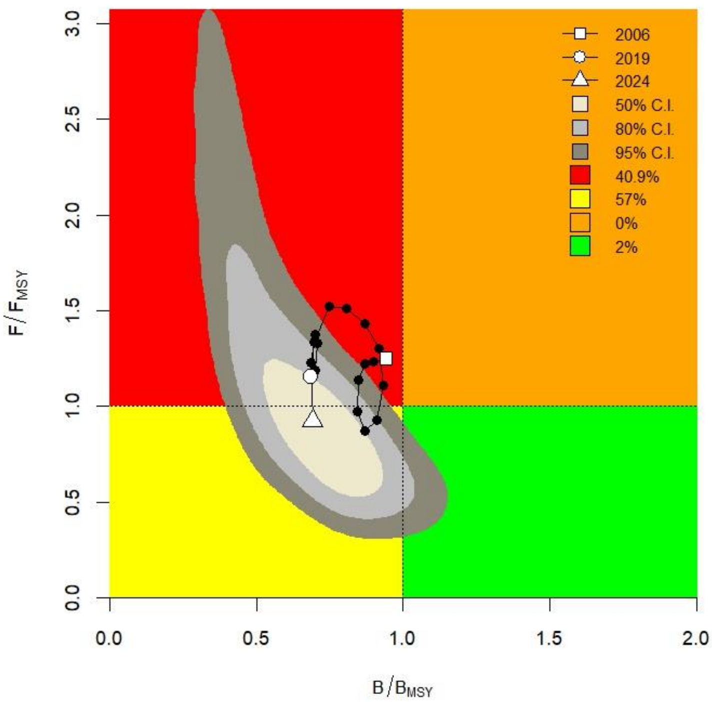

Figure A6. 26 Kobe plot for Mugil cephalus *in Porto Pino lagoon*

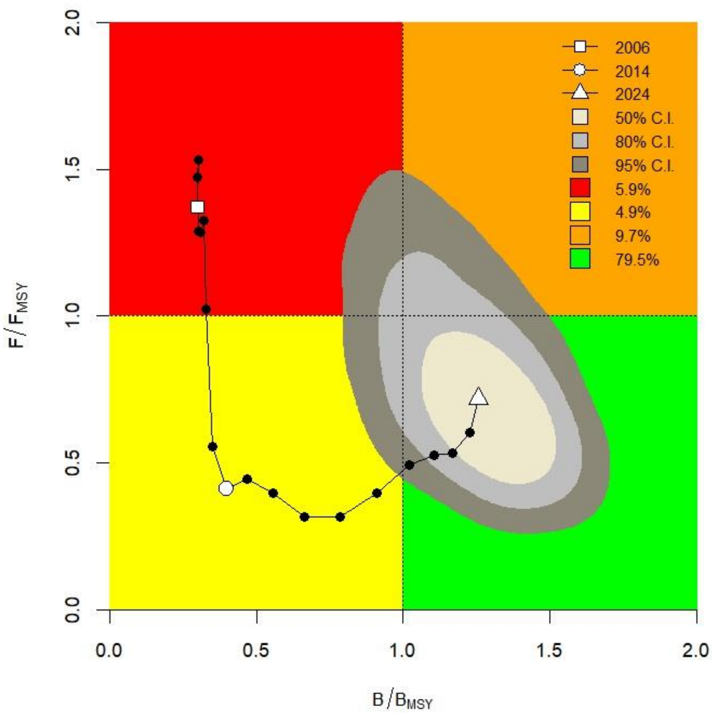

Figure A6. 27 Kobe plot for Mugil cephalus *in S'Ena Arrubia lagoon.*

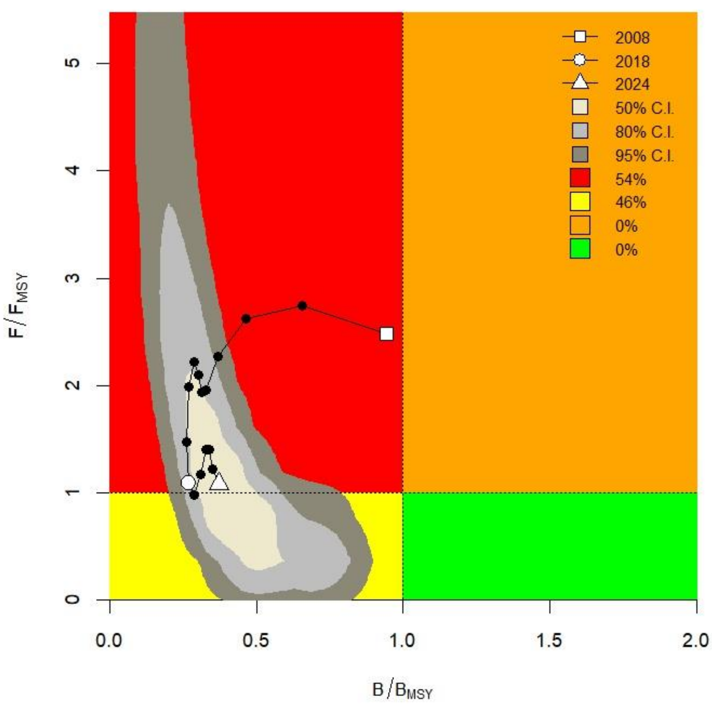

Figure A6. 28 Kobe plot for Mugil cephalus *in Sa Praia lagoon.*

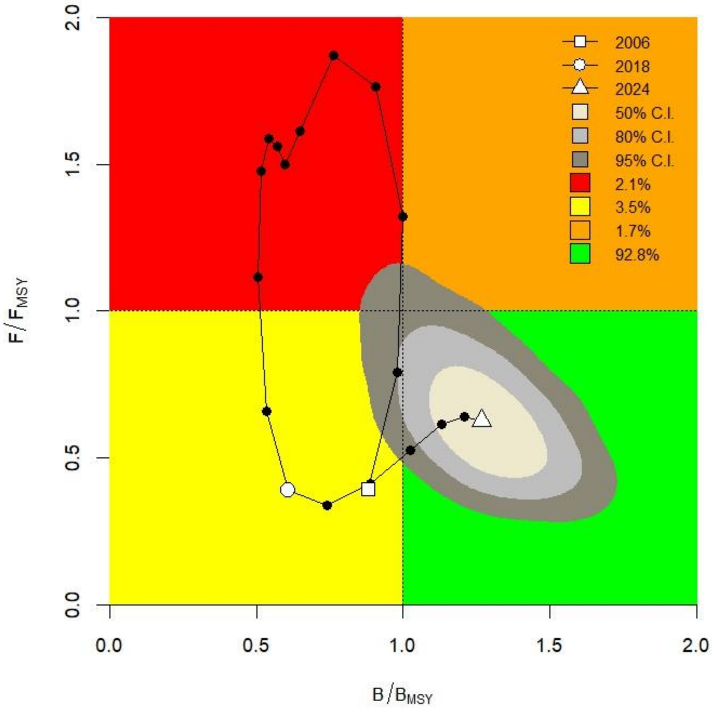

Figure A6. 29 Kobe plot for Mugil cephalus *in Santa Gilla lagoon.*

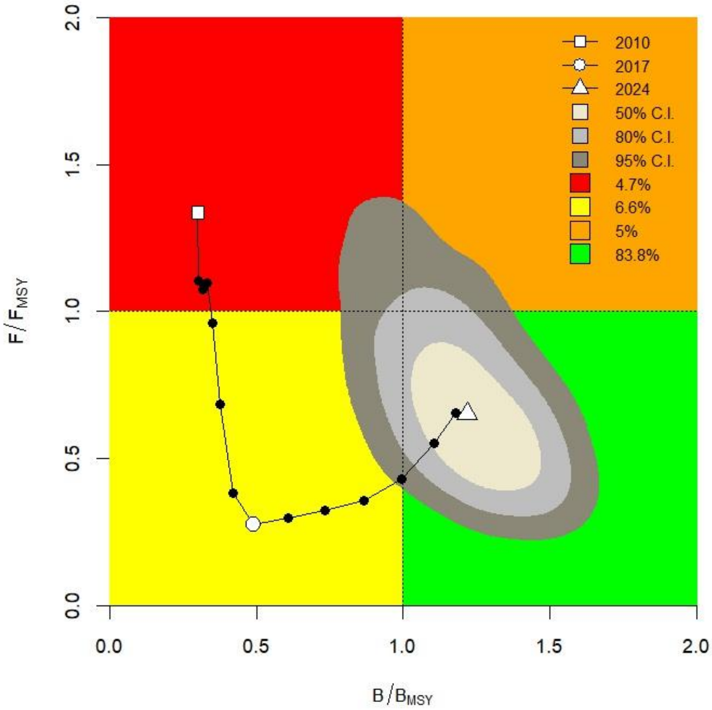

Figure A6. 30 Kobe plot for Mugil cephalus *in Santa Giusta lagoon.*

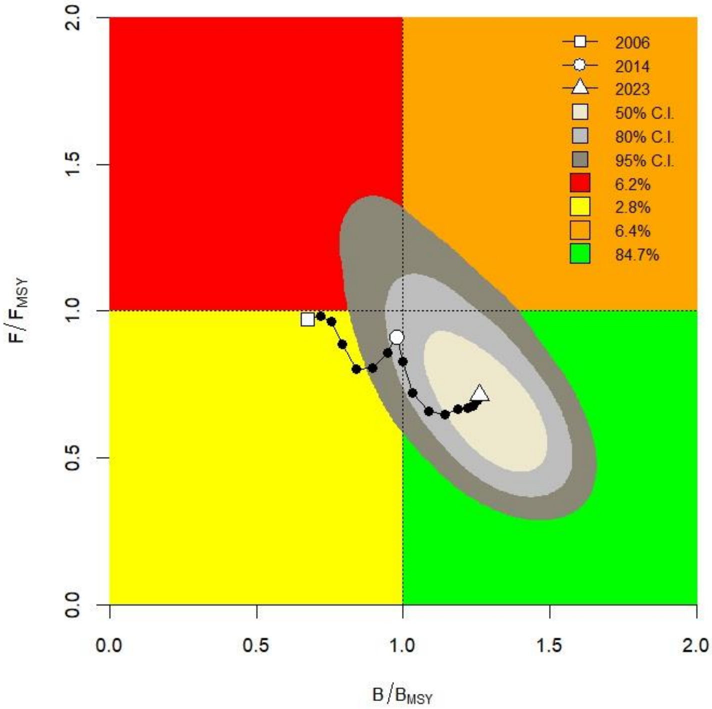

Figure A6. 31 Kobe plot for Sparus aurata *in Calich lagoon.*

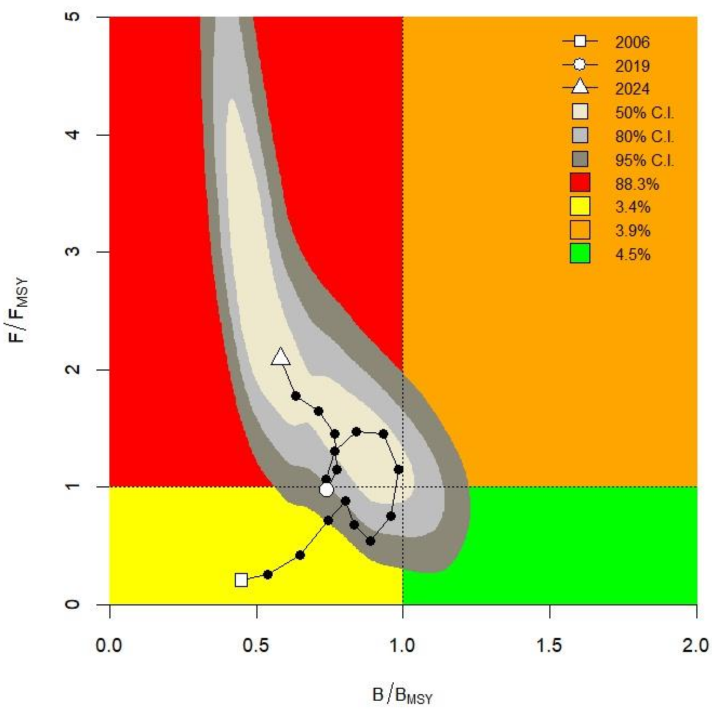

Figure A6. 32 Kobe plot for Sparus aurata *in Nora lagoon.*

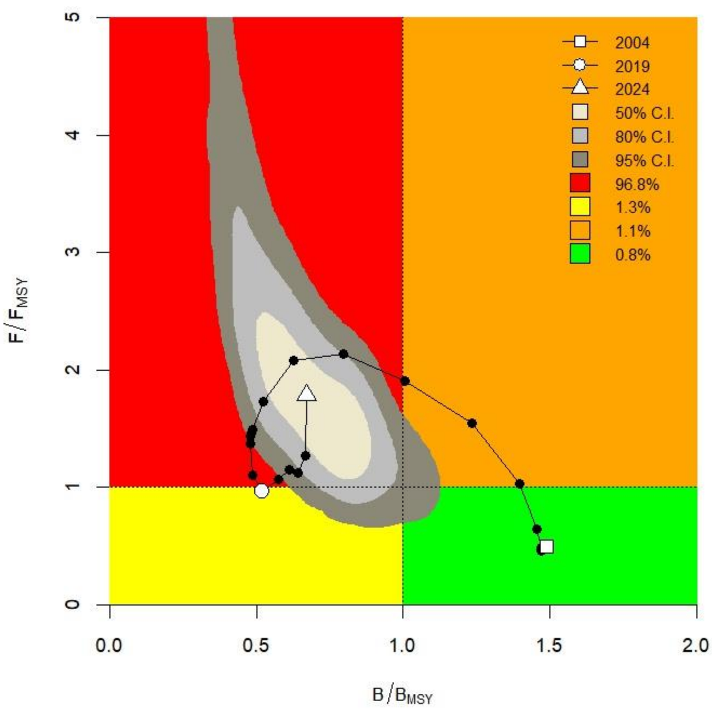

Figure A6. 33 Kobe plot for Sparus aurata *in Avalè su petrosu lagoon.*

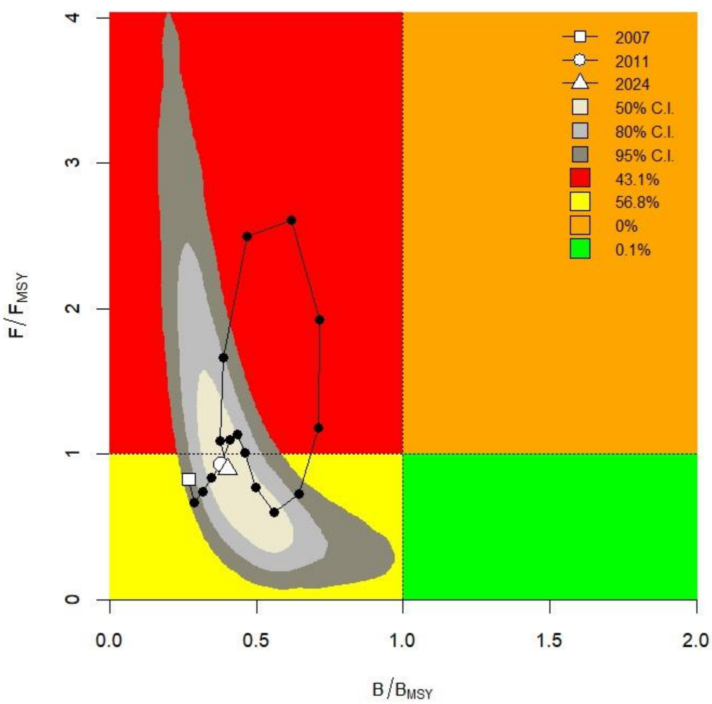

Figure A6. 34 Kobe plot for Sparus aurata *in Cabras lagoon.*

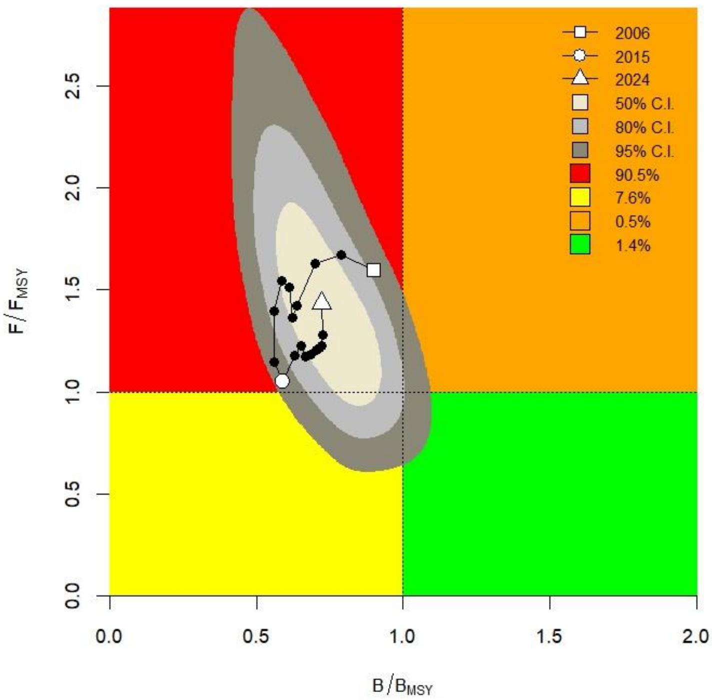

Figure A6. 35 Kobe plot for Sparus aurata *in Colostrai lagoon.*

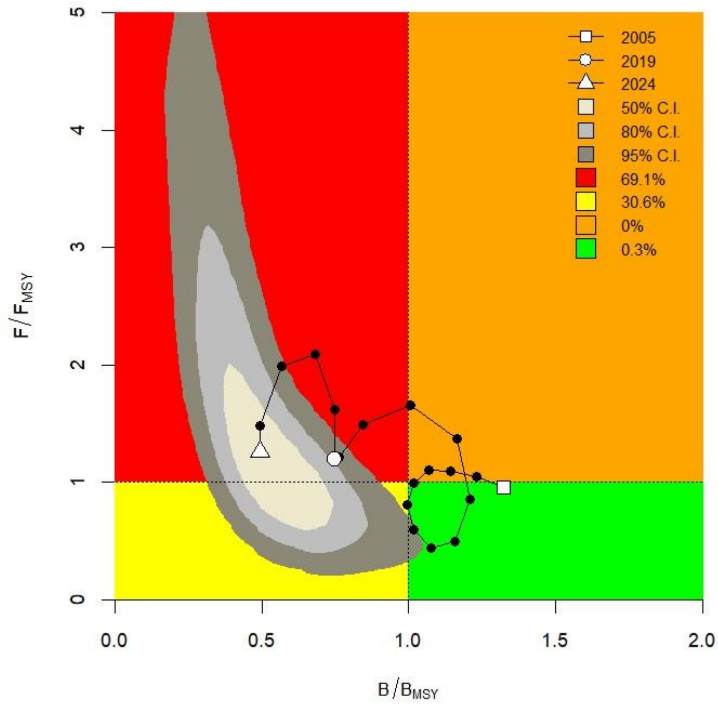

Figure A6. 36 Kobe plot for Sparus aurata *in Feraxi lagoon.*

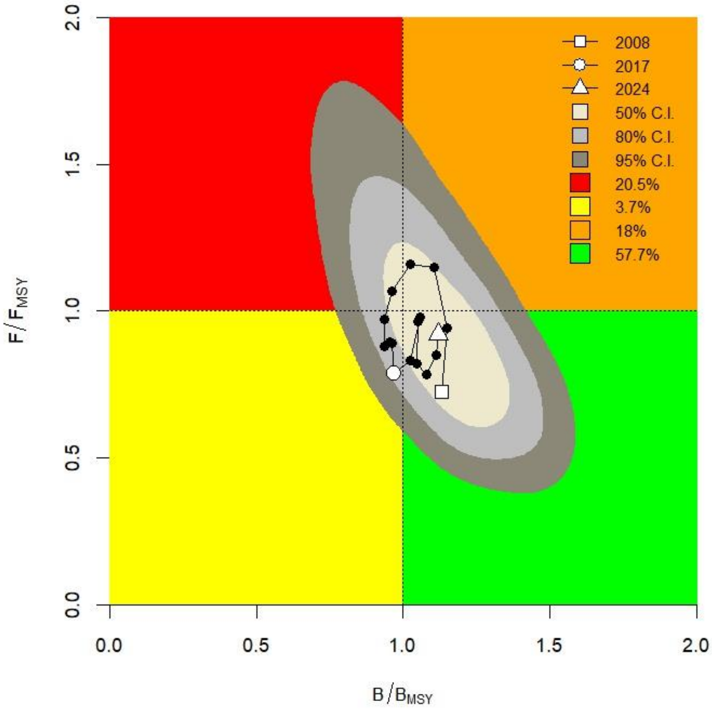

Figure A6. 37 Kobe plot for Sparus aurata *in Peschiera San Giovanni lagoon.*

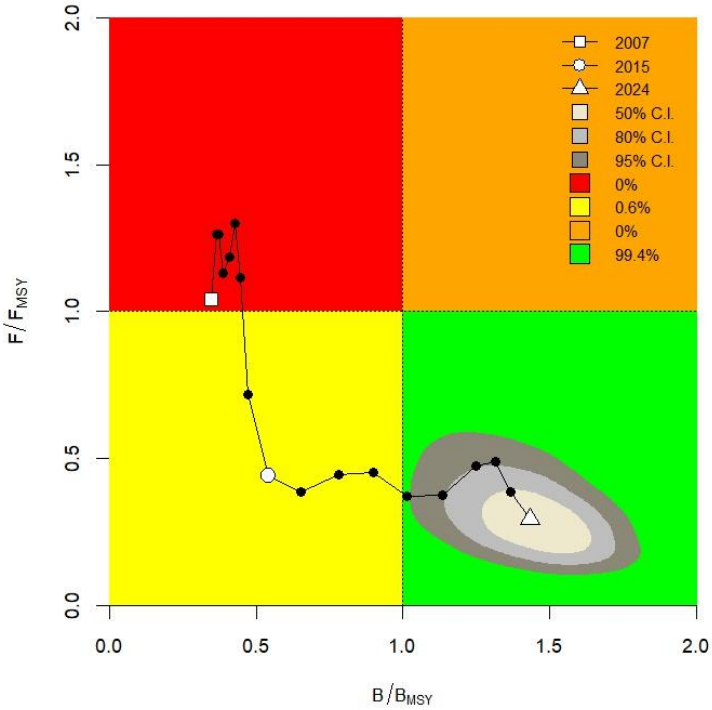

Figure A6. 38 Kobe plot for Sparus aurata *in Tortolì lagoon.*

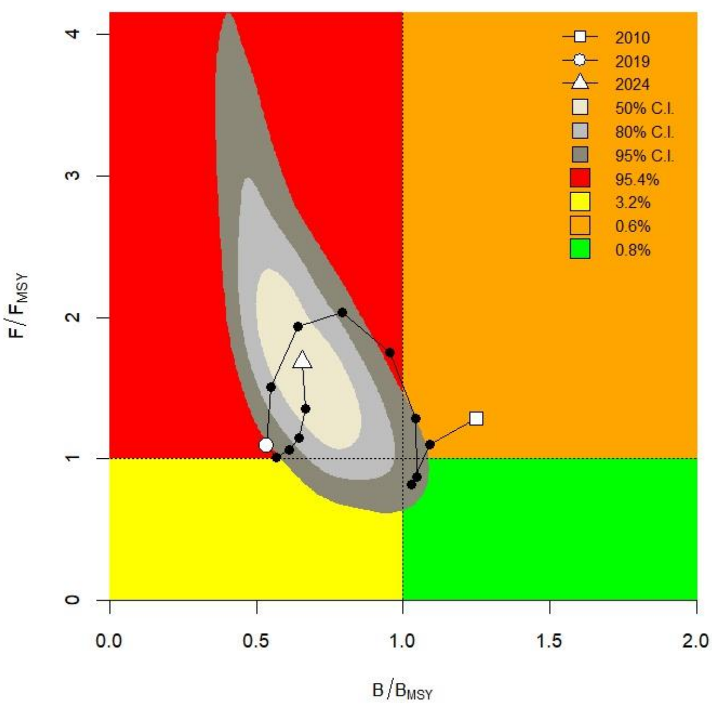

Figure A6. 39 Kobe plot for Sparus aurata *in Is Benas lagoon.*

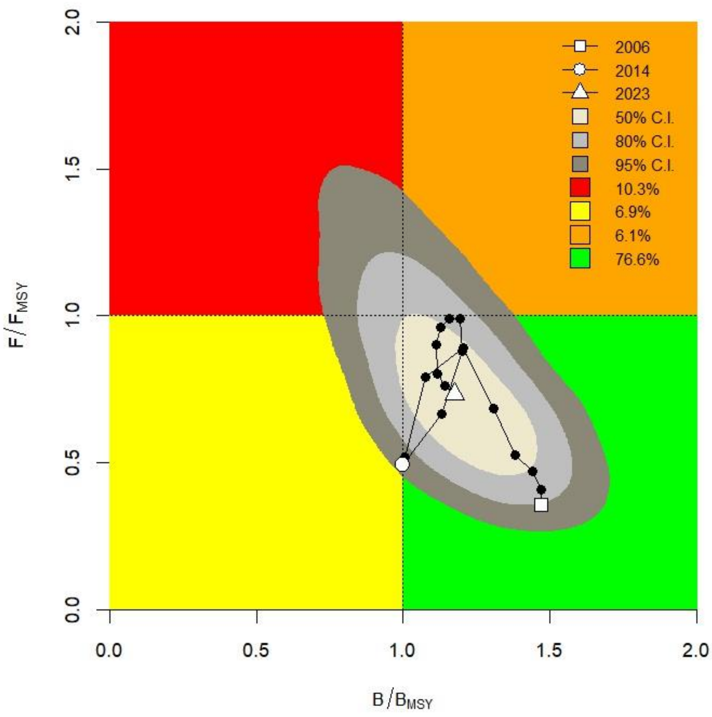

Figure A6. 40 Kobe plot for Sparus aurata *in Pauli biancu Turri lagoon.*

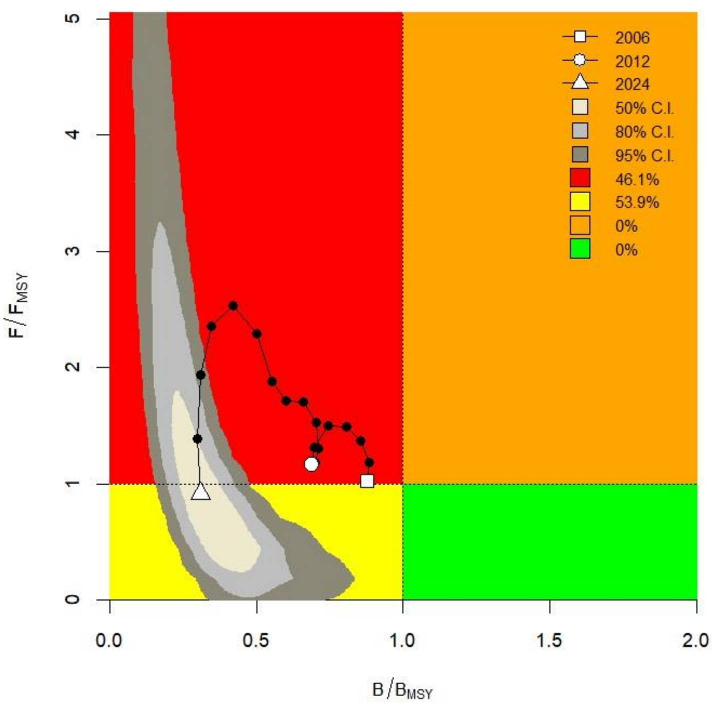

Figure A6. 41 Kobe plot for Sparus aurata *in Porto Pino lagoon.*

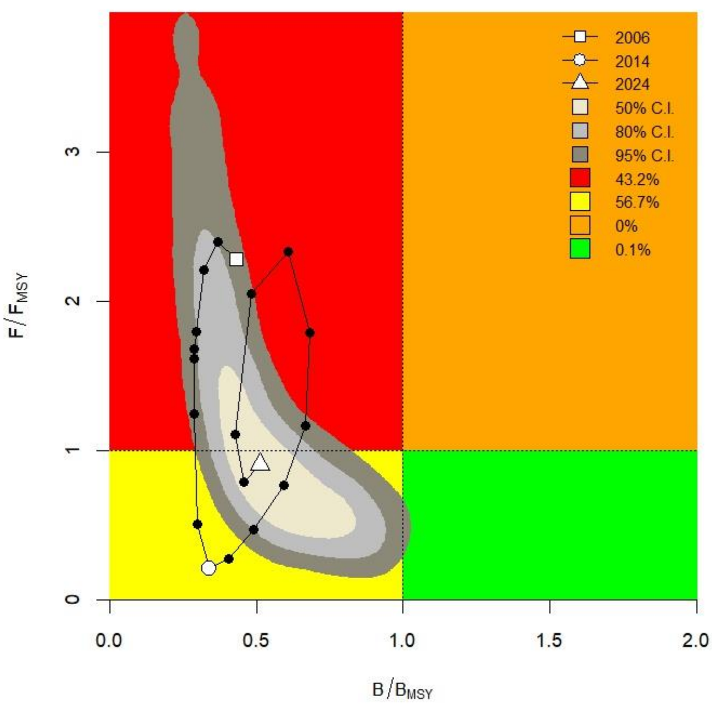

Figure A6. 42 Kobe plot for Sparus aurata *in S'Ena Arrubia lagoon.*

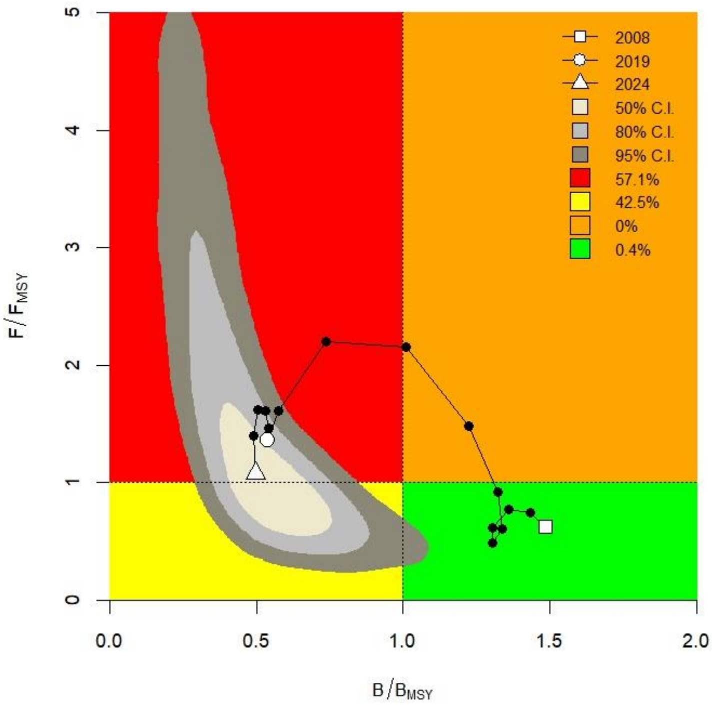

Figure A6. 43 Kobe plot for Sparus aurata *in Sa Praia lagoon.*

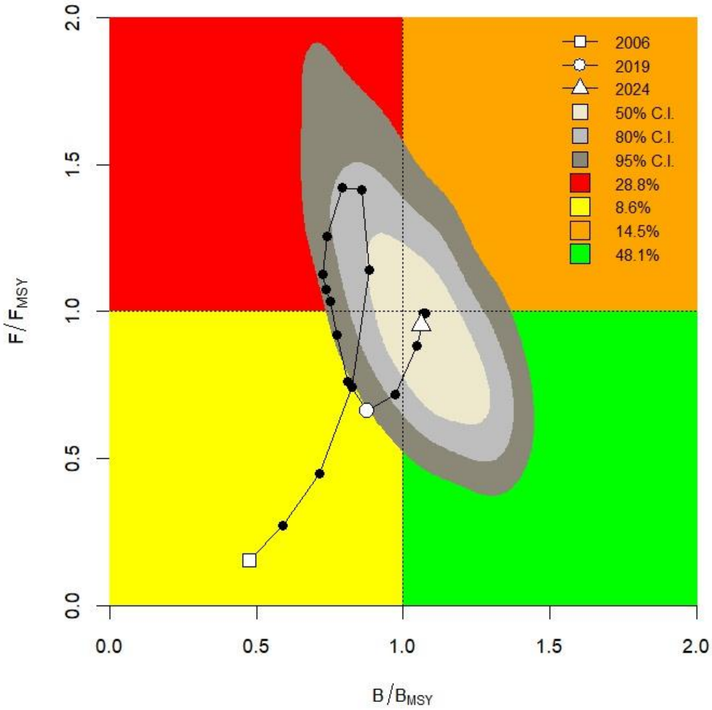

Figure A6. 44 Kobe plot for Sparus aurata *in Santa Gilla lagoon.*

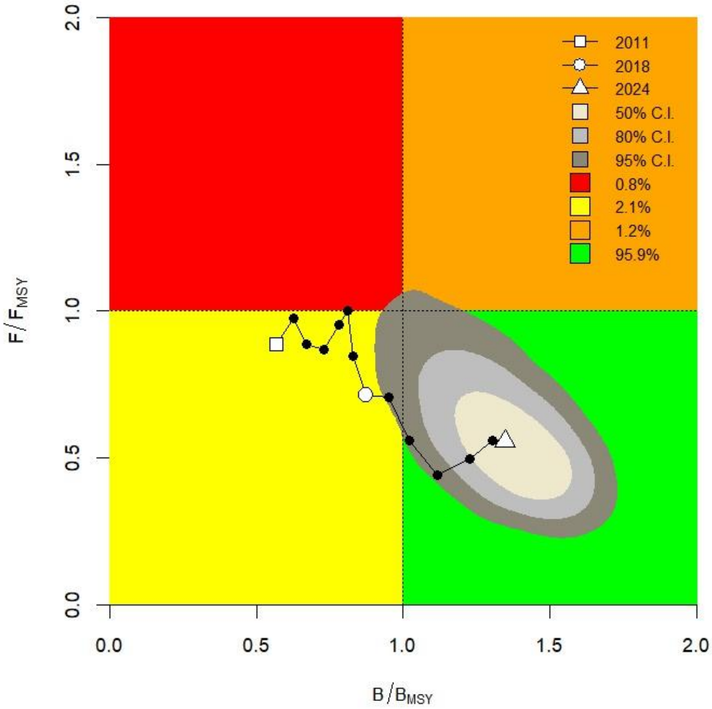

Figure A6. 45 Kobe plot for Sparus aurata *in Santa Giusta lagoon.*

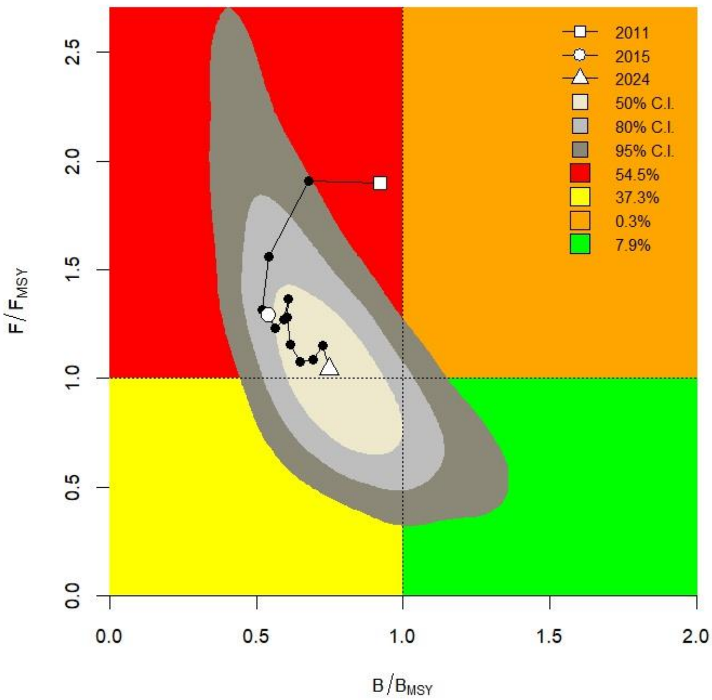

Figure A6. 46 Kobe plot for Ruditapes decussatus *in Santa Gilla lagoon.*

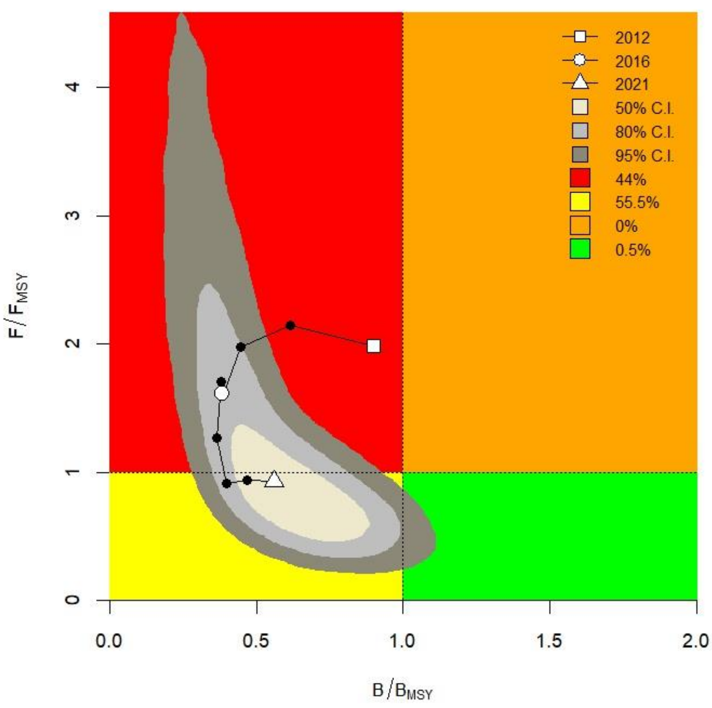

Figure A6. 47 Kobe plot for Ruditapes decussatus *in S'Ena Arrubia lagoon.*

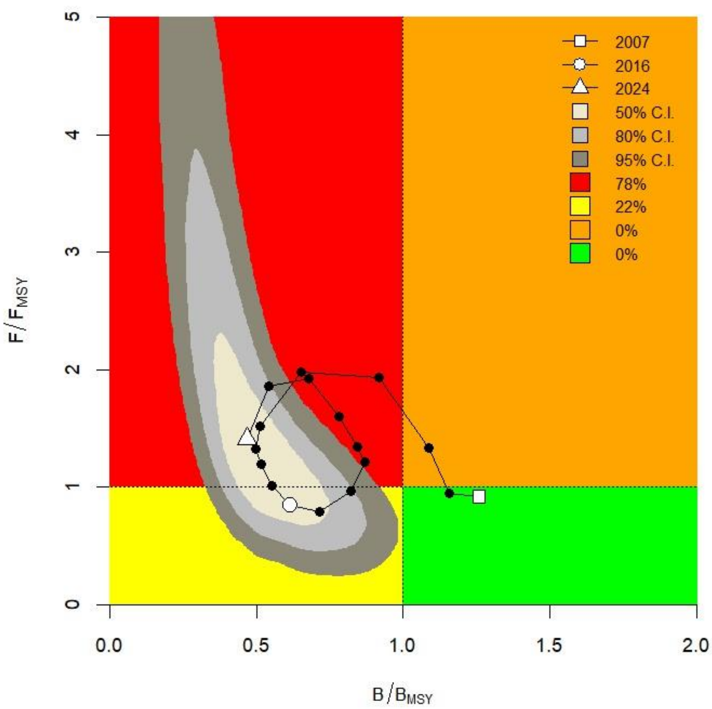

Figure A6. 48 Kobe plot for Ruditapes decussatus *in Tortolì lagoon.*

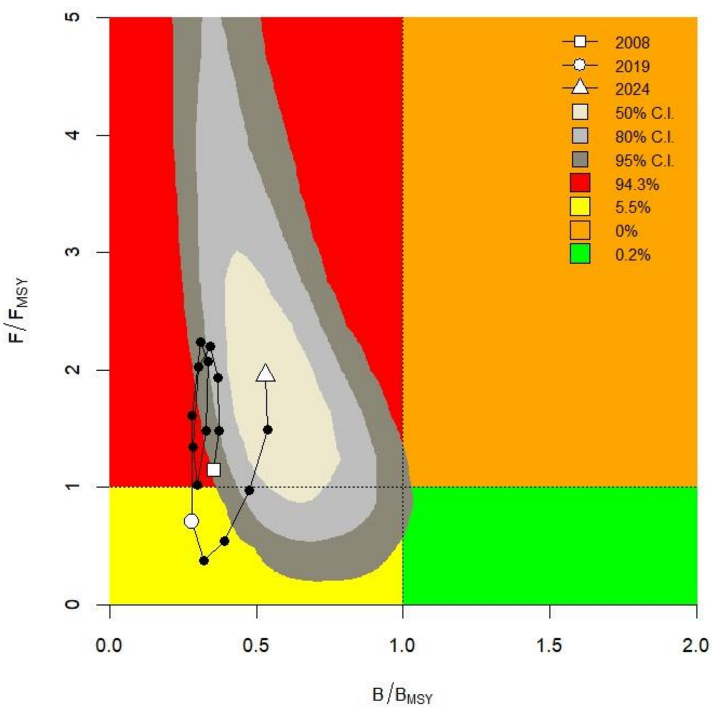

Figure A6. 49 Kobe plot for Ruditapes decussatus *in Peschiera San Giovanni lagoon.*

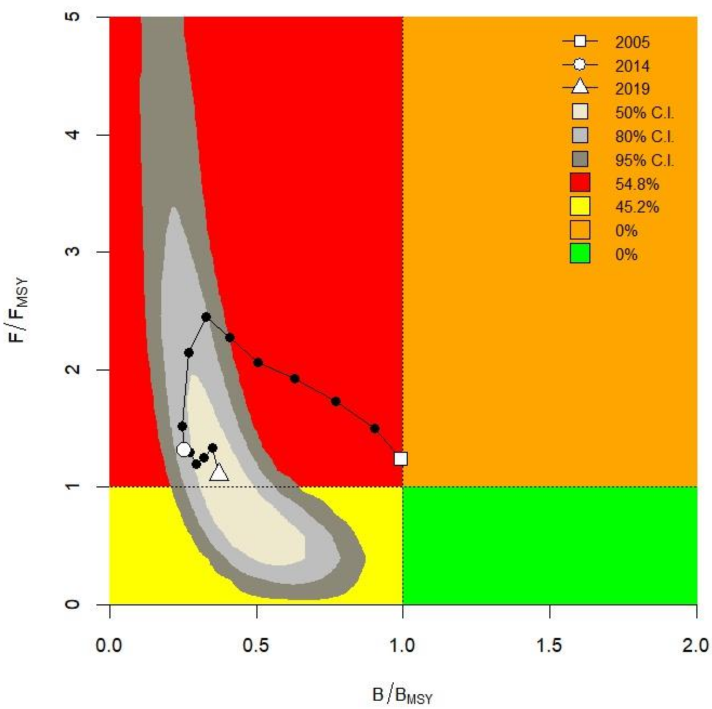

Figure A6. 50 Kobe plot for Ruditapes decussatus *in Feraxi lagoon.*

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