Science of the Total Environment 872 (2023) 162244

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Mapping and assessing seagrass meadows changes and blue carbon under past, current, and future scenarios

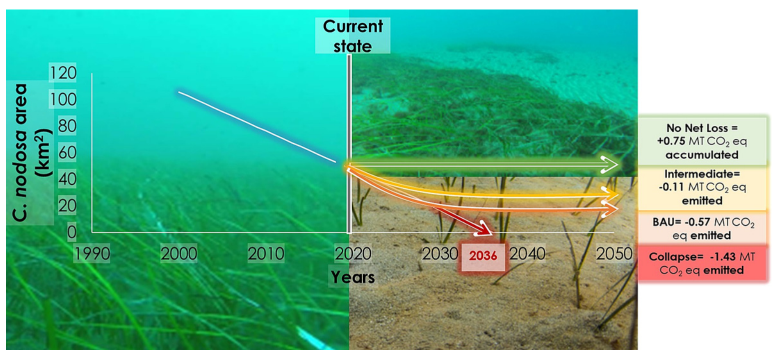
Miriam Montero-Hidalgo [a,*], Fernando Tuya [b], Francisco Otero-Ferrer [b], Ricardo Haroun [b], Fernando Santos-Martín [a]

[a] Rey Juan Carlos University, Chemical and Environmental Technology Department, Madrid, Spain
[b] Biodiversity and Conservation Research Group, IU-ECOAQUA, Universidad de Las Palmas de Gran Canaria, Telde, Spain

HIGHLIGHTS

- 50 % of seagrass meadows have disappeared in the last 2 decades.
- Mapping and assessing seagrass blue carbon storage and sequestration
- InVEST Blue Carbon model for past, current, and future scenarios
- C stock in the Canary Islands corresponds to 1.5 % nationally and 0.015 % globally.
- A proposal through blue carbon assessment to support decision making.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Fernando A.L. Pacheco

Keywords:
Cymodocea nodosa
Marine spatial planning
Ecosystem services
InVEST Blue Carbon model
Canary Islands
Atlantic

ABSTRACT

Seagrasses store large amounts of blue carbon and mitigate climate change, but they have suffered strong regressions worldwide in recent decades. Blue carbon assessments may support their conservation. However, existing blue carbon maps are still scarce and focused on certain seagrass species, such as the iconic genus *Posidonia*, and intertidal and very shallow seagrasses (<10 m depth), while deep-water and opportunistic seagrasses have remained understudied. This study filled this gap by mapping and assessing blue carbon storage and sequestration by the seagrass *Cymodocea nodosa* in the Canarian archipelago using the local carbon storage capacity and high spatial resolution (20 m/pixel) seagrass distribution maps for the years 2000 and 2018. Particularly, we mapped and assessed the past, current and future capacity of *C. nodosa* to store blue carbon, according to four plausible future scenarios, and valued the economic implications of these scenarios. Our results showed that *C. nodosa* has suffered ca. 50 % area loss in the last two decades, and, if the current degradation rate continues, our estimations demonstrate that it could completely disappear in 2036 ("Collapse scenario"). The impact of these losses in 2050 would reach 1.43 MT of CO_2 equivalent emitted with a cost of 126.3 million € (0.32 % of the current Canary GDP). If, however, this degradation is slow down, between 0.11 and 0.57 MT of CO_2 equivalent would be emitted until 2050 ("Intermediate" and "Business-as-usual" scenarios, respectively), which corresponds to a social cost of 3.63 and 44.81 million €, respectively. If the current seagrass extension is maintained ("No Net Loss"), 0.75 MT of CO_2 equivalent would be sequestered from now to 2050, which corresponds to a social cost saving of 73.59 million €. The reproducibility of our methodology across coastal ecosystems underpinned by marine vegetation provides a key tool for decision-making and conservation of these habitats.

1. Introduction

Atmospheric concentrations of CO_2 and other greenhouse gases are at levels unprecedented in the last few years, surpassing 40 % since pre-

* Corresponding author at: Rey Juan Carlos University, c. Tulipán, s/n, 28933 Móstoles, Madrid, Spain.
E-mail address: miriam.montero@urjc.es (M. Montero-Hidalgo).

http://dx.doi.org/10.1016/j.scitotenv.2023.162244
Received 11 November 2022; Received in revised form 10 February 2023; Accepted 10 February 2023
Available online 14 February 2023

industrial times; the main carbon sources are fossil fuel emissions and the net land use change emissions (Buis, 2019; IPCC, 2013). The Paris agreement seeks to achieve a balance between anthropogenic emissions and carbon reservoirs conservation and this is under the spotlight of climate change mitigation efforts (Wedding et al., 2021). In this sense, carbon accumulation has been extensively studied in terrestrial ecosystems, contrary to what occurs in coastal and marine ecosystems, where the data availability is still scarce (Macreadie et al., 2019, 2021; Townsend et al., 2014). It is widely known that blue carbon sequestered by coastal ecosystems, such as salt marshes, mangroves, and seagrasses, contribute to remove large amounts of CO_2 from the atmosphere and sea, helping to regulate the climate (Duarte et al., 2005; Fourqurean et al., 2012; Lavery et al., 2013).

Seagrass meadows store more than twice carbon relative to terrestrial forests (Fourqurean et al., 2019). This occurs because soils are saturated with water, maintaining an anaerobic state, and not allowing aerobic microbial carbon oxidation and release, which drives carbon accumulation over time and, therefore, creates long term carbon reservoirs through centuries to millennia (Chmura et al., 2003; Houghton, 2003; Tanaya et al., 2018). Despite seagrass meadows occupy <0.2 % of the world's oceans, they are responsible for approximately 10 % of the yearly estimated organic carbon burial, with ca. 27.4 T C annually (Duarte et al., 2005). Fourqurean et al. (2012) estimated that the global seagrass carbon stock was between 15,372 and 30,744 MT CO_2 eq and in Spain, a recent study estimated that carbon storage by *C. nodosa* is 29.5 MT CO_2 eq and by all seagrass species is 227 Mt. CO_2 eq (González-García et al., 2022).

Beyond the blue carbon storage and sequestration function, and so the climate regulation service, a multitude of ecosystem services provided by marine phanerogams have been identified so far, and the ecosystems formed by them have been classified therefore as one of the most valued habitats around the world (Costanza et al., 1997). Seagrasses benefit humans both directly and indirectly throughout services, such as coastal protection, water purification (Ascioti et al., 2022; Duarte et al., 2013), fish nursery (Espino et al., 2011; Lau, 2013), biodiversity conservation (Barbier, 2013), nutrient recycling, sediment trapping, habitat provision for numerous commercially important and endangered marine species, and food security for many coastal communities around the world (Barbier et al., 2011; Sousa et al., 2012).

Despite the ecological role of seagrasses, such as *Posidonia oceanica*, *Cymodocea nodosa*, or *Zostera noltii*, and the services they provide to human well-being at local and global levels, they have suffered strong regressions in recent decades (Short et al., 2011; Waycott et al., 2009). Global seagrass losses since the eighties is estimated, at least, at 29 % (Waycott et al., 2009) and the annual rate of seagrasses decline reaches 1.4 % (Short et al., 2011). The main anthropogenic pressures identified are: water contamination (Waycott et al., 2009), increased turbidity and eutrophication (Burkholder et al., 2007), and mechanical damages on the seabed (Ceccherelli et al., 2007), including boat anchoring (Montefalcone et al., 2008), and alterations of the habitat due to coastal works (Perez-Ruzafa et al., 1991).

Some areas of the planet are more vulnerable than others to anthropogenic pressures. This is the case of Overseas Countries and Territories (OCTs) and Outermost Regions (ORs), typically small oceanic islands and archipelagos, which depend on their natural heritage to sustain their economies, in particular their coastal areas (Sieber et al., 2018, 2022). Among these ORs, the Canary Islands are one of the richest biodiversity hotspots in Europe (Benzaken and Renard, 2010). Here, seagrass meadows are mainly distributed along the leeward coasts of the islands, under the direct effect of anthropogenic pressures. In this sense, *Cymodocea nodosa*, the dominant seagrass species, has suffered a strong regression at the archipelago-scale during the last decades (Fabbri et al., 2015; Tuya et al., 2013a, 2013b, 2014b, 2014c).

Despite a lot of seagrass distribution maps in the literature (Traganos et al., 2022; Ward et al., 2022), only a few have mapped the blue carbon associated with this habitat (Simpson et al., 2022). Importantly, blue carbon has been only mapped for certain seagrass species from some areas of the world, such as the iconic genus *Posidonia* from the Mediterranean (Monnier et al., 2021, 2022). However, assessments of opportunistic seagrasses (such as *Cymodocea* sp.) and their associated blue carbon are limited to intertidal and very shallow waters (Esteban et al., 2018; Kilminster et al., 2015). In this sense, seagrasses are often classified into three groups, according to their life story traits (Kilminster et al., 2015), including: persistent, opportunistic, and colonising species. While most cartographic and associated blue carbon assessments have been implemented for persistent species (e.g., *Posidonia* spp.), those focusing on opportunistic seagrasses (such as *Cymodocea* sp.) are limited to <10 m depth (Esteban et al., 2018; Kilminster et al., 2015). However, certain opportunistic seagrass species can reach large extension in deeper waters. For example, the most frequent bathymetric distribution of *C. nodosa*, in the Canary Islands, is typically between 10 and 20 m depth (Brito, 1984; Espino, 2004; Pavón-Salas et al., 2000).

Due to the low water transparency in the Canary Islands and the depth of *C. nodosa* in this archipelago, the use of satellite imagery is not convenient. Therefore, our study used reliable *C. nodosa* maps obtained from Side Scar Sonar (SSS) and underwater video techniques. A combination of both techniques is usually applied on small areas but, in this case study, covered the whole archipelago, as an input to obtain accurate blue carbon maps at a relevant resolution for managers (20 m/pixel). In summary, our study is the first contribution assessing blue carbon of an opportunistic seagrass species in deep waters based on validated distribution maps (10–30 m depth).

Carbon storage in seagrass meadows is strongly location-dependent, even within the same species, and large variability between seagrass species is often found (Duffy et al., 2022; Lavery et al., 2013; Simpson et al., 2022; Tuya et al., 2019). This study avoided biases by extrapolating average carbon storage of species that tend to dominate the literature, i.e., *Posidonia oceanica*, and used blue carbon data measured locally and recently in Gran Canaria (Bañolas et al., 2020).

The objectives of the study were: (i) to quantify changes in *C. nodosa* extension and carbon stocks overtime (2000 − 2020); (ii) to define plausible future scenarios for carbon stocks based on the management of anthropogenic pressures; and (iii) to value the socio-economic impact of these scenarios in monetary terms. Likewise, this research provides useful information related to the location where degradation of seagrass meadows should be avoided, and conservation measures should be prioritized to preserve *C. nodosa* and its carbon storage function.

2. Methodology

2.1. Study area and seagrass distribution

The Canarian archipelago (Spain) is composed by eight main volcanic islands in the North-east Atlantic Ocean, 95 km offshore the western African continent and about 1400 km from the Iberian Peninsula (Fig. 1. A). The seagrass *C. nodosa* has records in all the islands of the archipelago (Ruiz de la Rosa et al., 2015), mainly distributed along the leeward coasts of the islands, forming extensive meadows in the eastern islands (Lanzarote and Fuerteventura), as well as in the south of Gran Canaria and Tenerife (Brito, 1984; Haroun et al., 2003). In the western islands, however, sea bottoms are rockier and more abrupt, with reduced insular shelves. For this reason, the number and extension of meadows are reduced on the island of La Gomera, with an occasional presence in El Hierro (Moreira-Reyes et al., 2013; Ruiz de la Rosa et al., 2015), and the existence of meadows in La Palma to be confirmed (Ruiz de la Rosa et al., 2015). The seagrass *C. nodosa* majorly appears in shallow waters (<10 m). However, in this archipelagic context, this seagrass species can reach large extension in deeper waters, with the most frequent bathymetric distribution between 10 and 20 m depth (Brito, 1984; Espino, 2004; Pavón-Salas et al., 2000).

Cymodocea nodosa presents a warm-temperate distribution across the Mediterranean Sea, spreading into the Atlantic Ocean along the southern Iberian Peninsula and the NW coast of Africa, which includes Madeira and the Canary Islands (Alberto et al., 2006; Hartog and Kuo, 2006; Mascaró et al., 2009; Tuya et al., 2014c) and reaching Senegal, the southern range edge of the species (Cunha and Araújo, 2009). Contrary to persistent

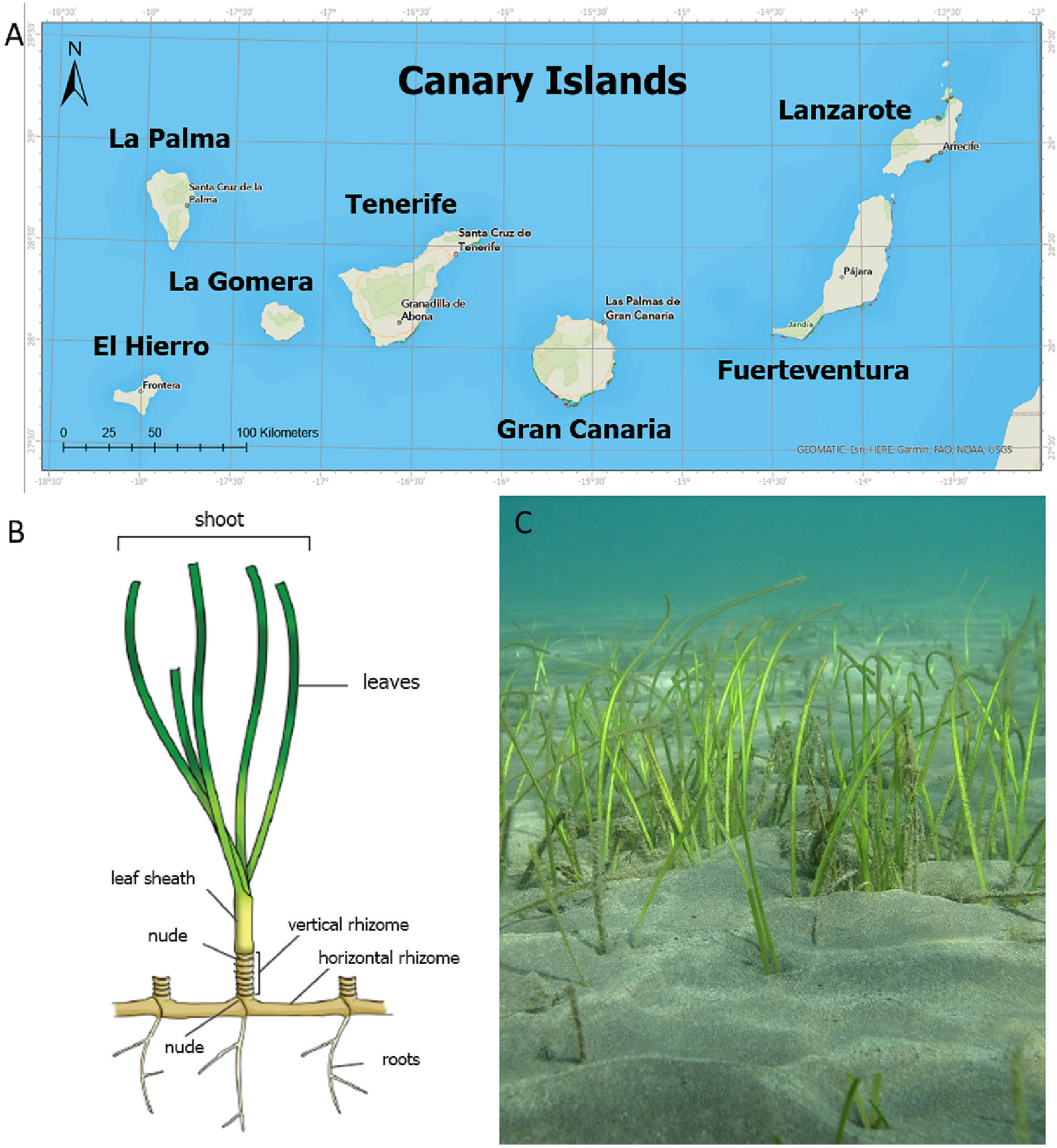


Fig. 1. A) Map of the Canary Islands, B) illustrative diagram of a seagrass shoot, and C) *C. nodosa* meadow, which leaves length often seasonally change between 12 and 15 cm in winter and 30–35 in summer.

seagrass species, such as those within the genus *Posidonia* (Kilminster et al., 2015), this is a fast-growing (i.e., opportunistic) species (sensu Orth et al., 2006) with an annual reproduction pattern (Caye and Meinesz, 1985) and a clear annual (seasonal) pattern in vitality. Maximums in leaf growth and canopy structure are typically observed in spring-summer, and minimums in winter (Cancemi et al., 2002; Máñez-Crespo et al., 2020; Marbà et al., 1996; Tuya et al., 2006). In the Canary Islands, the evolution of *C. nodosa* has followed a "founder effect", so all genotypes derived from a few initial colonizers (Alberto et al., 2006). Therefore, populations from the Canary Islands are a genetic unit that differs from both the Atlantic Iberian and Mediterranean populations (Alberto et al., 2008). This has consequences on the resilience of the plant to human disturbances, with populations from the Canaries more sensitive to stressors that their Mediterranean counterparts (Tuya et al., 2019, 2021). This has been linked to large regressions of *C. nodosa* across the Canary Islands (Fabbri et al., 2015; Tuya et al., 2014c), which has been connected with human pressures, such as nutrient enrichment of the water column (Tuya et al., 2013b), release of brine (Portillo, 2014) and construction of infrastructures (Manent et al., 2020). The species is included in eight 'Special Area of Conservation', under the EU 'Natura 2000' network (EU Habitat Directive, EU 92/43, 21 May), and included since 2016 in the Spanish National Catalogue of Endangered Species (Order AAA/1351/2016).

Table 1
Sources of information to derive metrics for the InVEST model on carbon pool dynamics in *C. nodosa* seagrass meadows.

InVEST inputs	Description	Elemental carbon units	CO_2 eq	Source
Biomass-initial (tonnes/ha)	Carbon stocks in the biomass pool (above- and below-ground) for *C. nodosa* in 2000 (the year of the baseline map)	0.83	3.05	Tuya et al. (2013b); Máñez-Crespo et al. (2020)
Soil-initial (tonnes/ha)	Carbon stocks in the soil pool in 2000	85.70	314.52	Bañolas et al. (2020)
Litter-initial (tonnes/ha)	Carbon stocks in the litter pool in 2000	0.11	0.40	Local experts
Biomass-yearly-accumulation (tonnes/ha year)	Annual rate of CO_2 eq accumulation in the biomass pool (above- and below-ground)	1.68	6.17	Tuya et al. (2019); Bañolas et al. (2020); Tuya et al. (2013b); Reyes et al. (1995)
Soil-yearly-accumulation (tonnes/ha year)	Annual rate of CO_2 eq accumulation in the soil pool	0.05	0.18	Bañolas et al. (2020); Serrano et al. (2016)

InVEST inputs	Description	Value	Source
Biomass-half-life (year)	The half-life of carbon in the biomass pool	0.27	Murray et al. (2011)
Biomass-high-impact-disturb (ratio)	Proportion of carbon stock in the biomass pool that is disturbed when a cell transitions away from this LULC class in a high-impact disturbance	1.00	
Soil-half-life (year)	The half-life of carbon in the soil pool	1.00	
Soil-high-impact-disturb (ratio)	Proportion of carbon stock in the soil pool that is disturbed when a cell transitions away from this LULC class in a high-impact disturbance.	0.50	

2.2. InVEST coastal blue carbon model

This study applied the InVEST Blue Carbon V3.9.2 model[1] to map and assess coastal blue carbon changes in the Canary Islands between 2000 and 2020. It was used to value avoided emissions, as well as to identify locations where there are net carbon gains or losses over time. It was chosen because is a free open-source software model to map coastal blue carbon changes and is supported by a well-recognized institution, such as Stanford University and the Natural Capital Project. These features allow the reproducibility of this study around the world. Despite the relevance of seagrasses as blue carbon habitats, most studies that used InVEST to assess blue carbon have focused on mangroves and wetlands, ca. 82 vs 18 % of studies (Cai et al., 2021; El-Hamid et al., 2022; Hernández-Blanco et al., 2022; Kacem et al., 2022; Rosa et al., 2022). To our knowledge, this is the first study estimating blue carbon of a deep-water and opportunistic seagrasses using the InVEST strategy based on validated distribution maps (Esteban et al., 2018).

The InVEST Blue Carbon model considers three carbon pools: living biomass such as leaves, rhizomes and roots (above- and below-ground compartments, respectively), sediment carbon into soil, and standing dead carbon as detritus (litter). This model requires: (1) maps of coastal ecosystems that store carbon (in a raster format), (2) the amount of carbon stored in each carbon pool, (3) the rate of annual carbon accumulation in the living biomass and sediments, (4) the impacts of human actions on carbon storage, as well as (5) their level of disturbance, (6) magnitude, and (7) timing of loss.

The total carbon stock was calculated by adding up all carbon stocks in the 3 pools. Meanwhile, carbon stocks for a given year and pool were calculated by adding the net carbon sequestration for that year to the stocks available in the prior year. The model assumes that the carbon accumulation and emission rates for *C. nodosa* is linear through time points. Based on spatial layers of *C. nodosa* changes between years, the model estimates the carbon lost to the atmosphere over time when *C. nodosa* is disturbed. In this model, net sequestration refers to the amount of carbon lost (emitted) and gained (accumulated). Carbon yearly accumulation and initial stock values were included in a biophysical table for each pool (Table 1). It assumes that a disturbance event happens in the first moment of the year in which the transition occurs and, after it, the disturbed carbon is emitted over time at an exponential decay rate. The exponential decay function needs some inputs as the proportion of carbon disturbed from *C. nodosa* presence to its absence, and the carbon half-life in each pool (Table 1). Finally, the model outputs were spatially displayed in several blue carbon maps for the years 2000 and 2020.

[1] https://naturalcapitalproject.stanford.edu/software/invest-models/coastal-blue-carbon

2.3. Data collection

2.3.1. Spatial information

To map the habitat of *C. nodosa* meadows over time, all *C. nodosa* official spatial information available were requested to the Biodiversity Service of the Canary Islands Autonomous Government. Six datasets were delivered by the technical authorities based on field monitoring campaigns with in-situ techniques carried out between 2000 and 2018 (Barquín and Martín, 2011; Monterroso et al., 2016, 2018). The first dataset refers to the whole Canary Islands archipelago and gathers the initial eco-cartographic studies based on the acoustic SSS and ground truthing via underwater video techniques, mostly conducted between 2002 and 2005, and included in the Bionomic Atlas of the Canary Islands (Barquín and Martín, 2011). The cartography of subsequent years (2005–2011) confirmed the presence of *C. nodosa* in these areas and overlapped previous registers. Hence, the polygons generated during 2000–2011 were dissolved with ArcGIS Pro and considered as the *C. nodosa* maps for the entire archipelago for the initial year (2000), as the initial time reference. We took 2000 as the "historical" baseline, because there are no notable changes on the seagrass distribution in the period in which the most significant cartographies were implemented, i.e., the cartography in this period was very similar between 2005, 2002 and 2000.

The remaining five datasets corresponded to different islands and were carried out between 2015 and 2018 in those locations where *C. nodosa* presence had been verified in previous studies; the Bionomic Atlas of the Canary Islands (2011) was used as a reference (Monterroso et al., 2015, 2016, 2018). The five layers are based on the acoustic tool SSS and ground truthing via underwater video techniques and all of them provide information on the level of substrate coverage (high, medium or low), unlike the previous cartography, which did not include this local distinction attribute. High coverage means that >50 % of the seabed is covered by seagrasses, medium coverage refers to a range between 25 and 50 %, and low coverage refers to <25 %. Because there is no unified cartography for the whole archipelago, at the same year, those maps produced during the period 2015–2018 were considered as the *C. nodosa* distribution in 2018.

As a result, the years "2000" and "2018" were selected as the two time periods to compare for this study. Maps from 2000 and 2018 were compared using tools from ArcGIS Pro and regression rates were calculated between these years. Both maps were converted from shapefiles to raster format, and a spatial resolution of 20 m of pixel size was used to maintain the maximum resolution admissible for the InVEST Blue Carbon model.

2.3.2. Carbon estimates

The InVEST Blue Carbon V3.9.2 model calculated carbon stocks and sequestration based on the changes in the distribution of *C. nodosa* over time, and the carbon pools information referred to the entire Canary Islands. All the carbon pools information was input via a biophysical table, which contains the estimates of carbon stored in each pool, the rate of annual carbon accumulation in the biomass and sediments, the impacts from human activities on carbon storage, as well as their level of disturbance, magnitude, and timing of loss (Table 1).

Most metrics included in the model were obtained from local field data collections, which increases the accuracy of results, instead of using national or globally collected data elsewhere. The first parameter required was the "biomass initial" and refers to the carbon stocks in the biomass pools of *C. nodosa* in 2000, incorporating both above- and below-ground compartments. It was calculated by multiplying the mean value of carbon content in leaves and rhizomes-roots tissues (Tuya et al., 2013b) by the mean dry weight per m^2 of leaves and rhizomes-roots, respectively (Máñez-Crespo et al., 2020). The "biomass-yearly-accumulation" was calculated by adding the above- and below-ground biomass yearly accumulation rates. The above-ground component was obtained by multiplying the mean value of the annual leaf growth (per shoot) of *C. nodosa* in Gran Canaria (Tuya et al., 2019) by the mean value of shoot density (Bañolas et al., 2020) and by the mean value of carbon content in above-ground tissues (Tuya et al., 2013b). The below-ground component was obtained by multiplying the mean value of the elongation rate of rhizomes of *C. nodosa* (Reyes et al., 1995) by the unit weight of rhizomes, and by the mean value of carbon content in below-ground compartments (Tuya et al., 2013b).

To calculate the initial soil carbon pool metric, the carbon stored in the roots and rhizomes (initial below-ground biomass) was subtracted from the value of carbon in the soil, obtained in situ by Bañolas et al. (2020). This subtraction was performed so as not to double count the carbon in the below-ground compartment. Additionally, the "soil-yearly-accumulation" was calculated by multiplying the average carbon concentration obtained in Bañolas et al. (2020) by the minimum sediment accumulation rates included in Serrano et al. (2016) and by the dry sand density. The initial litter carbon pool was estimated as 20 % of the above-ground biomass and 10 % of the below-ground compartment (Tuya & Espino, unpublished data).

The "Biomass-half-life" and "soil-half-life" metric values were taken from the user guide recommendations for seagrasses (Murray et al., 2011). Likewise, the "biomass-high-impact-disturb" and "soil-high-impact-disturb" values followed user guide recommendations, which considers that a high disturbance over the biomass pool achieve a 100 % of carbon loss from biomass, while a high disturbance over the soil pool just remove the top carbon (50 % carbon loss from soil).

Finally, all metrics were converted to CO_2 equivalent (CO_2 eq) units, by multiplying by 3.67 due to the difference between the CO_2 atomic mass and elemental carbon.

2.4. Future scenarios

Future scenarios were conceptualized according to the plausible trends of seagrass meadows over time in the Canary Islands (Fig. 2). Four scenarios were considered with a 2050 fixed time horizon: (i) "No Net Loss" scenario (NNL); (ii) "Business-As-Usual" scenario (BAU); (iii) "Collapse scenario" (COL); and (iv) "Intermediate" scenario (INT). In the NNL scenario, the current seagrass area distribution was maintained from 2018 to 2050. In the BAU scenario, *C. nodosa* decreases following the current degradation rate (3 km^2 y^{-1}) until reaching 20 km^2 in 2029 and then, the seagrasses distribution is maintained from 2030 to 2050. In the COL scenario, the degradation rates of *C. nodosa* is constantly maintained until its completely disappearance in 2036 (assuming a linear decline function). In the INT scenario, an intermediate loss between the NNL and BAU scenarios is considered, which means that seagrass meadows decreases until 30 km^2 approximately, and then the seagrass distribution is steady maintained from 2030 to 2050.

To spatially represent the effect of each scenario in a map, we followed González-García et al. (2022) and also considered different pressures that alter seagrass meadows, based on those provided by Spanish marine authorities (MITECO, 2019a, 2019b). Among the pressures that affect the Canarian marine area, we selected those with a proven impact over *C. nodosa,* discarding those with no demonstrated disturbance (e.g., underwater

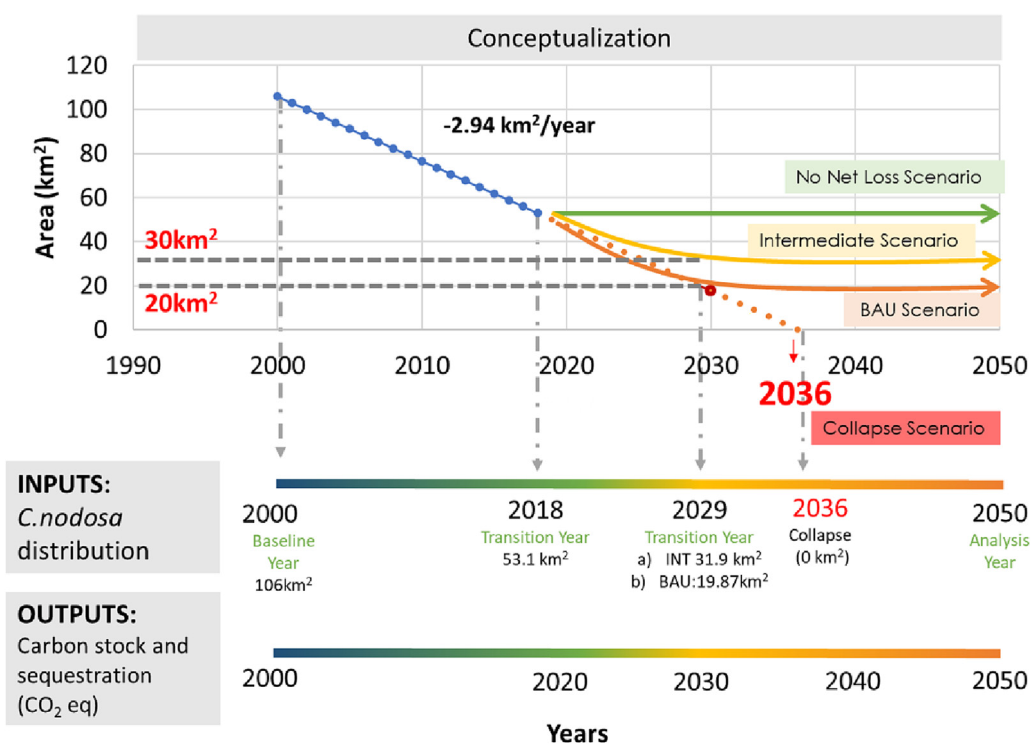


Fig. 2. Conceptualization of plausible trends of *C. nodosa* distribution over time with timelines displaying the snapshot maps considered in the model and the outputs obtained.

Table 2
Pressures selected in this study, including description, thresholds, and associated sources.

Pressures	Descriptions	Threshold (range)	Threshold Source
Hydrographic alterations	Construction of ports, marinas, and defence infrastructures (dikes) inducing alteration in sediment dynamics, including resuspension and retention, discharges of dredged port material, regeneration of beaches and creation of artificial beaches.	1.75 (0–2.25)	Manent et al. (2020)
Nutrients	Areas under large inflow of nutrients (nitrogen and/or phosphorus) due to dumping of dragged material, aquaculture facilities, and urban wastewater discharges from treatment plants.	1.61 (0–2.5) Concentration units: 0.1 mg/L (0.01–0.15 mg/L)	Tuya et al. (2013b)
Salinity	Areas under salinity alterations due to the release of brine from desalination plants, urban wastes and industrial wastes.	1.63 (0–3.25) Concentration units: 38 (36–40 UPS)	Portillo (2014); Ruiz de la Rosa et al. (2015)

noise). Three final pressures were considered: (i) hydrographic alterations, (ii) nutrients, and (iii) salinity (Table 2). Each pressure is displayed in a shapefile mesh of 9 × 9 km of cell size, which includes an index calculated by adding the drivers that generate the pressure. Then, a set of different threshold values were selected based on previous local studies and local expert consultation. Nutrients and salinity thresholds were determined through a change of scale between the range of values of the index in the shapefile and the range of concentration values found in the local scientific literature. Those cells with *C. nodosa* presence that exceed the threshold value were removed from the *C. nodosa* distribution map in 2018 and the new maps with areas of *C. nodosa* subtracted were those that describe future scenarios (Fig. 2). To follow the tendencies conceptualized for each scenario (Fig. 2), we increased the value of pressures in a 50 % for the INT scenario and in an 80 % for the BAU scenario to obtain a total *C. nodosa* distribution area of about 30 and 20 km^2, respectively. Finally, to generate the map for the COL scenario, we considered that 2036 was the year in which *C. nodosa* would completely disappear if the decline tendency was linear. Each *C. nodosa* distribution map created according to the four future scenarios was entered into InVEST, and carbon stocks and sequestration maps were subsequently obtained.

2.5. Economic valuation

The economic valuation of coastal blue carbon was estimated using the “Social Cost of Carbon” (SCC) of net carbon sequestration between 2020 and 2050. This approach monetizes the damage that an increase in CO_2 emissions would cause in the future, and includes the costs to avoid natural catastrophes, such as floods due to rising sea levels, rising temperatures, etc. It is also an indicator of what society would be willing to pay now to avoid the future damage caused by incremental carbon emissions (Liu et al., 2022; Rennert et al., 2022; U.S. IAWG SCGG, 2021). The time lag between the causes and effects of climate change complicate efforts to tackle the problem. The SCC fills that gap by bringing those future damages costs to the present, in order to increase awareness about the impact of our current actions and make informed decisions.

Due to the protection of *C. nodosa* in the Canary Islands and that changes in carbon emissions are public policy responsibilities, the benefits of zoning coastal areas for development against the social losses from carbon emissions (SCC) should be considered by decision makers. This valuation tries to shed light on this question by providing data to avoid costs if current *C. nodosa* is conserved and the social costs of emitting carbon if other non-sustainable scenarios occur. To estimate the SCC, emissions corresponding to three periods were considered: from 2020 to 2030, from 2030 to 2040, and from 2040 to 2050. The SCC of the last year of each period was multiplied by the amount of CO_2 emitted in that period. As local carbon emissions affect the atmosphere on a global scale, the SCC taken in this study was considered at a global scale. The data source used is the last Technical Support Document carried out by the US Interagency Working Group on the Social Cost of Greenhouse Gases[2] (U.S. IAWG SCGG, 2021). Among the discount rate available, a 2.5 % was selected because it is into the 1–3 % range advised by experts who participated in 2015 in the survey carried out by The Centre for Climate Change Economics and Policy (CCCEP) and The Grantham Research Institute on Climate Change and the Environment (Drupp et al., 2015). Finally, the selected estimation of SCC was changed from dollars to euros (1 US $ = 0.91 €) resulting in 81 €/tonnes CO_2 eq in 2030, 93€/tonnes CO_2eq in 2040, 105€/tonnes CO_2 eq in 2050.

[2] The US IWG SCGG has the commitment to ensuring that the SCC values estimates reflect the best available science and methodologies

3. Results

3.1. Changes in the distribution and extent of seagrass meadows

Overall, seagrass meadows area in 2000 covered 106 km^2 for the whole archipelago; around 50 % has been lost since then, with a regression rate of 3 km^2 y^{-1} (Figs. 2 and 3.A). For the remaining 53 km^2 that still exist nowadays, just 7 km^2 has a high seagrass cover (Fig. 3.A).

In 2000, the islands with the largest area were Gran Canaria (34 km^2), followed by Fuerteventura (32 km^2) and Lanzarote (23 km^2) (Fig. 3B). Tenerife, La Gomera and El Hierro had already reduced areas (13.1, 2.8 and 0.04 km^2, respectively, Fig. 3B). El Hierro had a small seagrass meadow (0.04 km^2) in the east of the island. There is no record from La Palma Island.

In 2018, Gran Canaria was still the island with the largest area, followed by Tenerife (instead of Fuerteventura) (Fig. 3B). Therefore, Fuerteventura is the island that has lost the largest area from 2000 to 2018 (23.2 km^2, ca. 71.6 %, Fig. 3B). Lanzarote experienced ca. 56.6 % reduction in area (13.2 km^2) and Gran Canaria lost ca. 37.5 % and 12.9 km^2 (Fig. 3B). Tenerife has lost ca. 19 %, corresponding to 2.5 km^2. Despite La Gomera just lost about 1.2 km^2, this was a considerable reduction (42.8 % loss), when compared to 2000 (2.8 km^2, Fig. 3B). There is no record from El Hierro in recent times. Fig. 3.C shows a local example in the decay of *C. nodosa* seagrass meadows through 2000 to 2018, as a consequence of an industrial port construction.

3.2. Plausible future changes in the distribution and extent of seagrass meadows

Fig. 4 shows the spatial distribution of pressures under each plausible future scenario, including a NNL scenario, where no meadows are lost (Fig. 4.A). The zones affected by the INT and BAU scenarios are shown in the Figs. 4.B and 4.C, with a remaining seagrass area distribution of 31.9 and 19.9 km^2, respectively. Some of the orange areas where *C. nodosa* is not present nowadays contained *C. nodosa* in the past, such as those located in the north-east of Gran Canaria.

Fig. 5.A provides information on the islands in which the pressures would cause the greatest losses in km^2. Fig. 5.B-E show which cells are affected by the current pressures, either in combination (5.B) or individually (5.C, 5.D, 5.E). Nutrients only affects two cells that otherwise do not contain *C. nodosa* (5.C), and coincide with the metropolitan areas of the two capitals in Tenerife and Gran Canaria. On the contrary, hydrographic alterations (5.D) and salinity (5.E) are the pressures that most cells containing *C. nodosa* are disturbed. These cells are mostly concentrated around the two most populated islands (Tenerife and Gran Canaria).

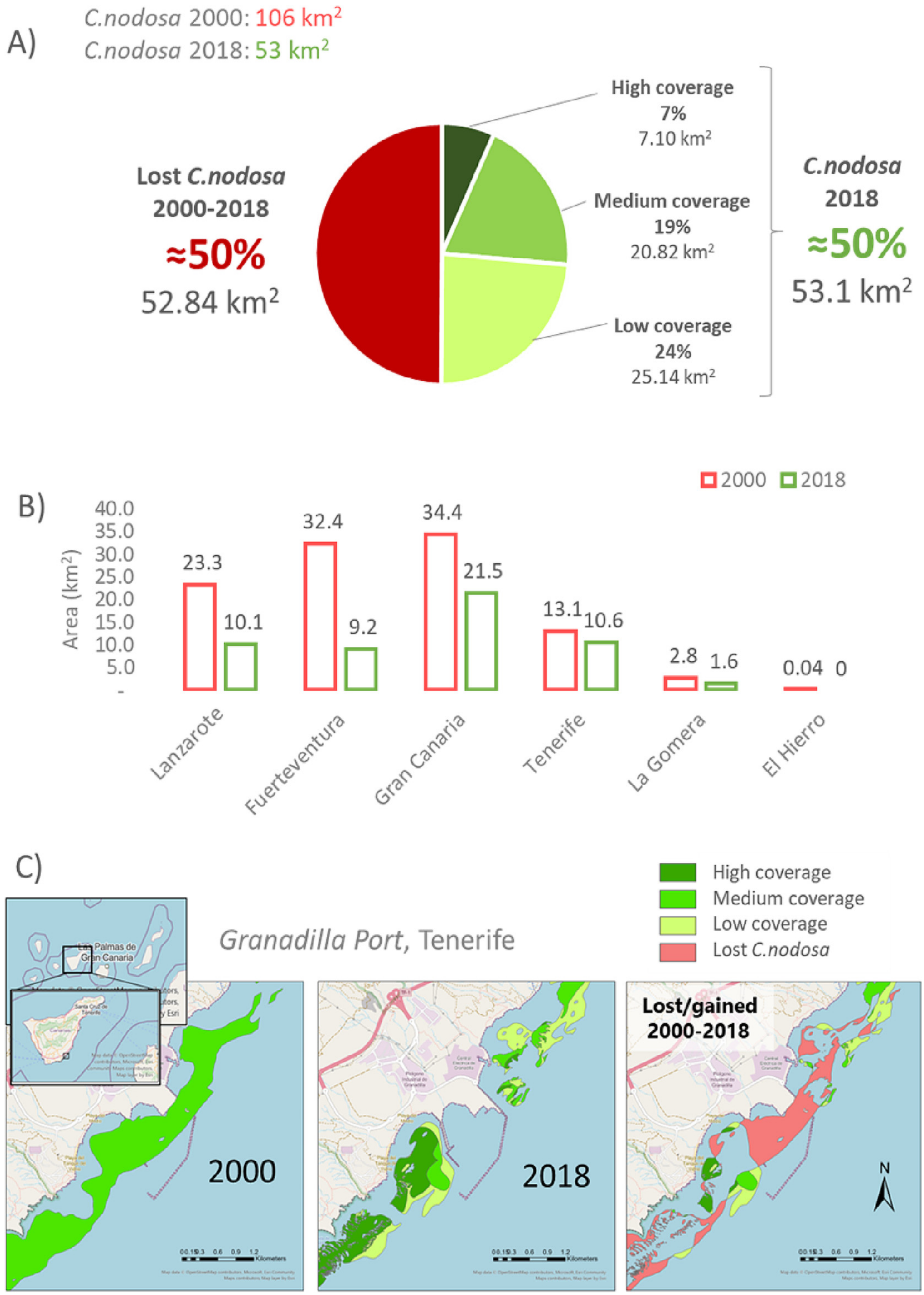


Fig. 3. A) Overall temporal changes in seagrass meadows area between 2000 and 2018, B) seagrasses meadow area by islands between 2000 and 2018, C) a local example of seagrass meadow degradation from Granadilla in Tenerife.

For the INT scenario, Gran Canaria and Tenerife would show the largest losses with ca. 38 and 69 % loss, corresponding to 8.1 km^2 and 7.3 km^2 in absolute terms, respectively. For the BAU scenario, these losses would add up to ca. 73 and 76 % loss (15.6 and 8.1 km^2), respectively. La Gomera would not suffer changes under the INT scenario, whereas for the BAU scenario, it would be the most disturbed island and would lose all *C. nodosa* meadows (ca. 1.6 km^2). Seagrass meadows from Fuerteventura would suffer the same impact for both the BAU and INT scenarios (3.7 km^2, a ca. 40 % loss). Meadows from Lanzarote would decrease ca. 20 % more in the BAU, in relation to the INT scenario, with a loss of ca. 2km^2 and 4.2 km^2, respectively.

3.3. Changes in coastal blue carbon overtime

Fig. 6 shows an example of spatial changes in carbon stocks over time, according to the different future scenarios and past records (in tonnes

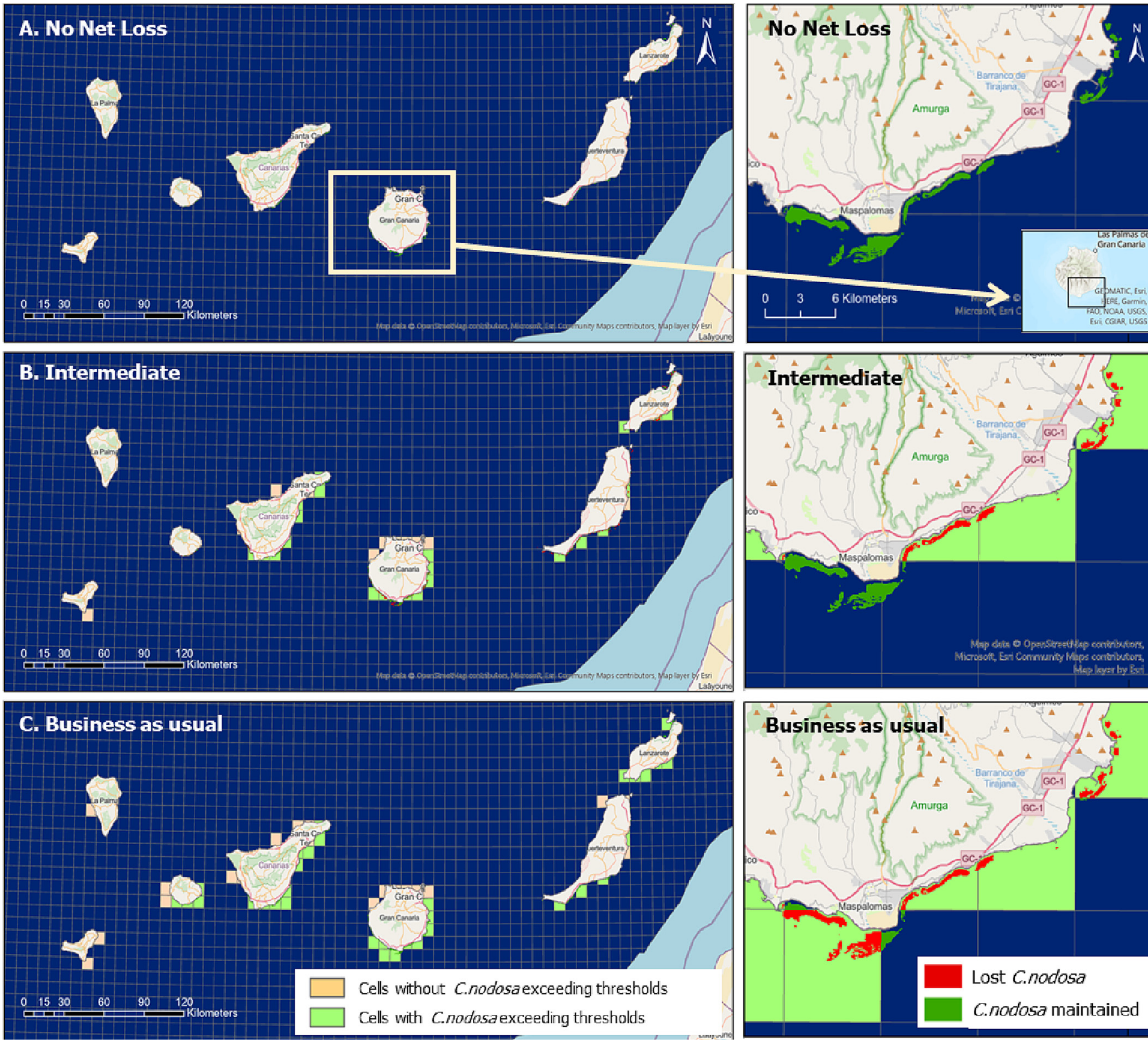


Fig. 4. Spatial distribution of pressures under each plausible future scenario: A) NNL- there are no pressures that overcome the thresholds and *C. nodosa* area is maintained until 2050; B) INT- some cells overcome the thresholds and *C. nodosa* is removed from them; C) BAU- *C. nodosa* was removed in those cells in which pressures exceeded the thresholds. Green cells are those in which thresholds are exceeded and contain *C. nodosa*. Orange cells are those in which thresholds are exceeded, but *C. nodosa* is not currently present. On the right, an example where coloured red surface shows areas where *C. nodosa* has been removed in each scenario.

CO_2 eq per ha). For this pilot case, a total of 318 t CO_2 eq per ha were observed in 2000, with continuous decreases onwards in those zones where *C. nodosa* has disappeared (light green), while carbon stocks have increased until 635 t CO_2 eq per ha (dark green) in those areas where *C. nodosa* has been maintained.

Fig. 7 accounts for the total carbon stocks, in Mega tonnes CO_2 eq, for the whole archipelago and each scenario, as well as the carbon fluxes or net sequestration between four time periods (2000–2020, 2020–2030, 2030–2040 and 2040–2050). The stocks for 2000 and 2020 were the same for all scenarios, because the difference between scenarios were introduced in 2029. Fig. 7 displays how, as the scenario became more unfavourable, carbon stocks concurrently decreased in the same year. For the NNL scenario, an increase was detected over time, becoming constant during the last 2 periods (2030–2040 and 2040–2050). The carbon stock reached 3.9 Mt. CO_2 eq in 2050 and the amount of carbon accumulated, and emissions avoided was 0.75 Mt. For the INT scenario, carbon stocks decreased substantially between 2020 and 2030, while slowly increasing from 2030 onwards; specifically, 0.05 Mt. between 2030 and 2040, and 0.20 Mt. during the following 10 years, reaching 3 Mt. CO_2 eq in 2050 and a net flux of -0.11 Mt. This means that 0.11 Mt. would be emitted from 2020 to 2050 for this scenario. In contrast, for the BAU scenario, an exponentially decreasing trend would be observed, with a considerable drop from 2020 to 2030 (-0.59 Mt. CO_2 eq) and a considerable drop the following ten years (-0.10 Mt). From 2040 to 2050, a slight accumulation would be registered again to finally reach 2.58 Mt. in 2050. The net emissions over time, for the BAU scenario, would be 0.57 Mt. Similarly, the COL scenario would imply a drastic decline from 2020 to 2040 and a minored decrease from 2040 to 2050, with a maintained value of 1.7 Mt. CO_2 eq in this period. Finally, the net amount emitted from 2020 to 2050, for this COL scenario, would be 1.43 Mt. CO_2 eq.

The most notorious differences between scenarios, in relation to carbon stocks, took place in 2040 and 2050 (Fig. 7.A), whereas the most relevant

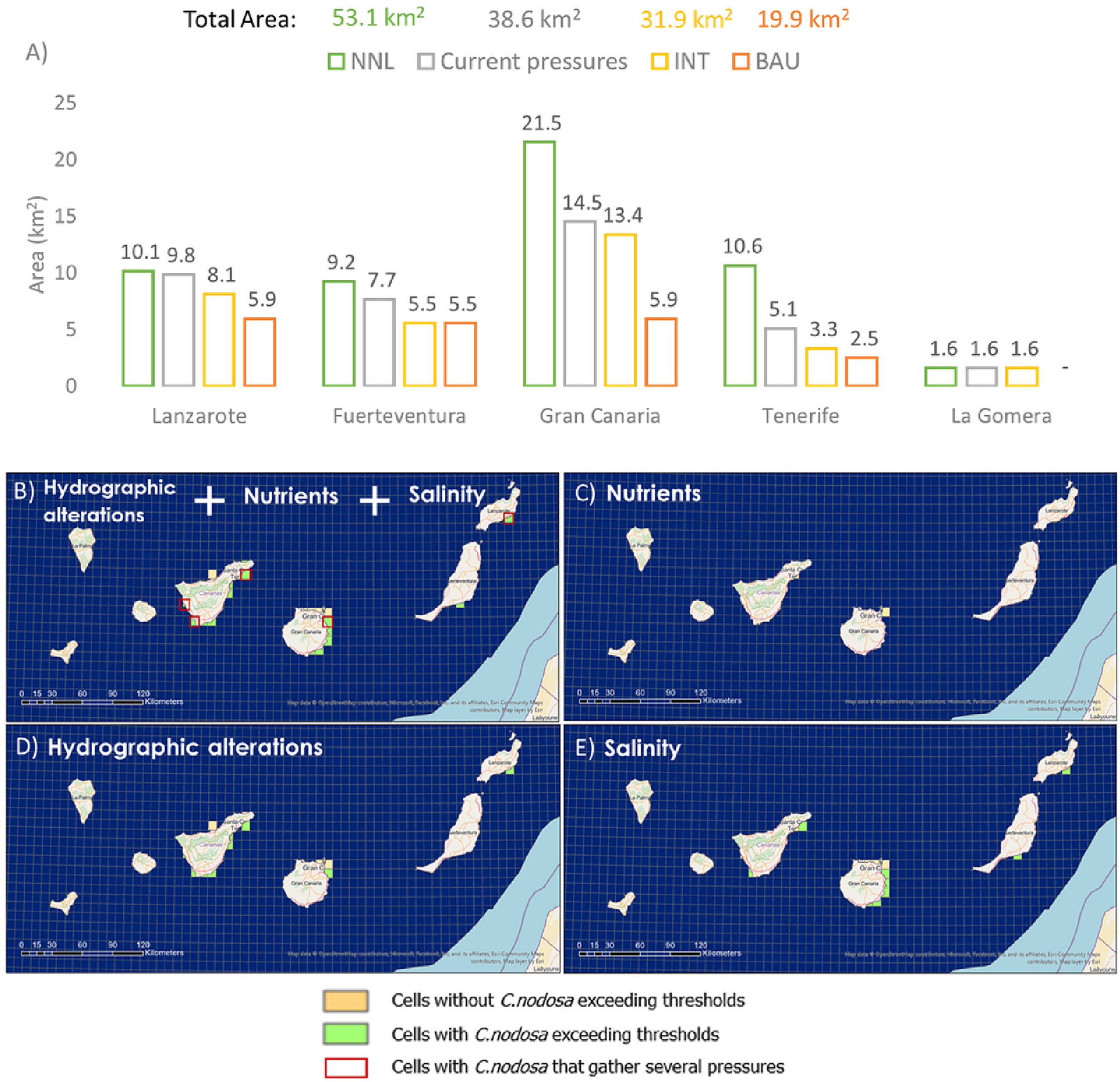


Fig. 5. A) Total area covered by *C. nodosa* in each island according to each scenario; B-E) Cells where pressures currently overcome the thresholds (in orange and green) and cells where the current *C. nodosa* distribution is being affected by current pressures (in green). For future prioritization, zones with *C. nodosa* where current pressures overlap are highlighted in red.

differences between net sequestration values were during the 2030–2040 period (Fig. 7.B).

3.4. Economic valuation of coastal blue carbon

The monetary value of the blue carbon stock in the Canary Islands, in 2020, is 218.4 million €, corresponding to 0.56 % of the Canary GDP. Fig. 7.C shows the social cost of carbon based on net carbon sequestration for each scenario in the 2020–2050 period. Positive values of the net sequestration refer to the accumulation of CO_2, while negative values refer to emissions. Otherwise, negative values of SCC obtained refer to avoided costs, while positive values of SCC refer to social costs caused by the damage that an increase in CO_2 emissions would cause in that future scenario. The NNL scenario is the only one that reached a net accumulation of CO_2 eq and the consequent avoided social cost, estimated at 73.59 M€ (0.19 % of the Canary GDP). On the contrary, emissions caused by the INT, BAU and COL scenarios would suppose costs of 3.6 M€ (0.01 % of the Canary GDP), 44.8 M€ (0.11 % of the Canary GDP) and 126.3 M€ (0.32 % of the Canary GDP), respectively.

4. Discussion

4.1. Decline of C. nodosa and carbon storage

This study has quantified a decline of ca. 50 % of the seagrass meadows area from 2000 to 2018 in the Canary Islands, at an annual rate of ca. 3 km^2. By contrast, global seagrass losses (29 % according to Waycott et al., 2009) and the global rate of seagrasses declines (1.4 % according to Short et al., 2011) is much lower than that registered in the archipelago. These considerable losses in the Canary Islands result from several factors: (i) the fast increase in population and tourist pressures on the coasts since the 70s (Espino et al., 2008; Tuya et al., 2014b), (ii) the large vulnerability of *C. nodosa* in the Canary Islands, due to its low genetic variability (Alberto et al., 2006) and associated low resilience to human impacts (Tuya et al.,

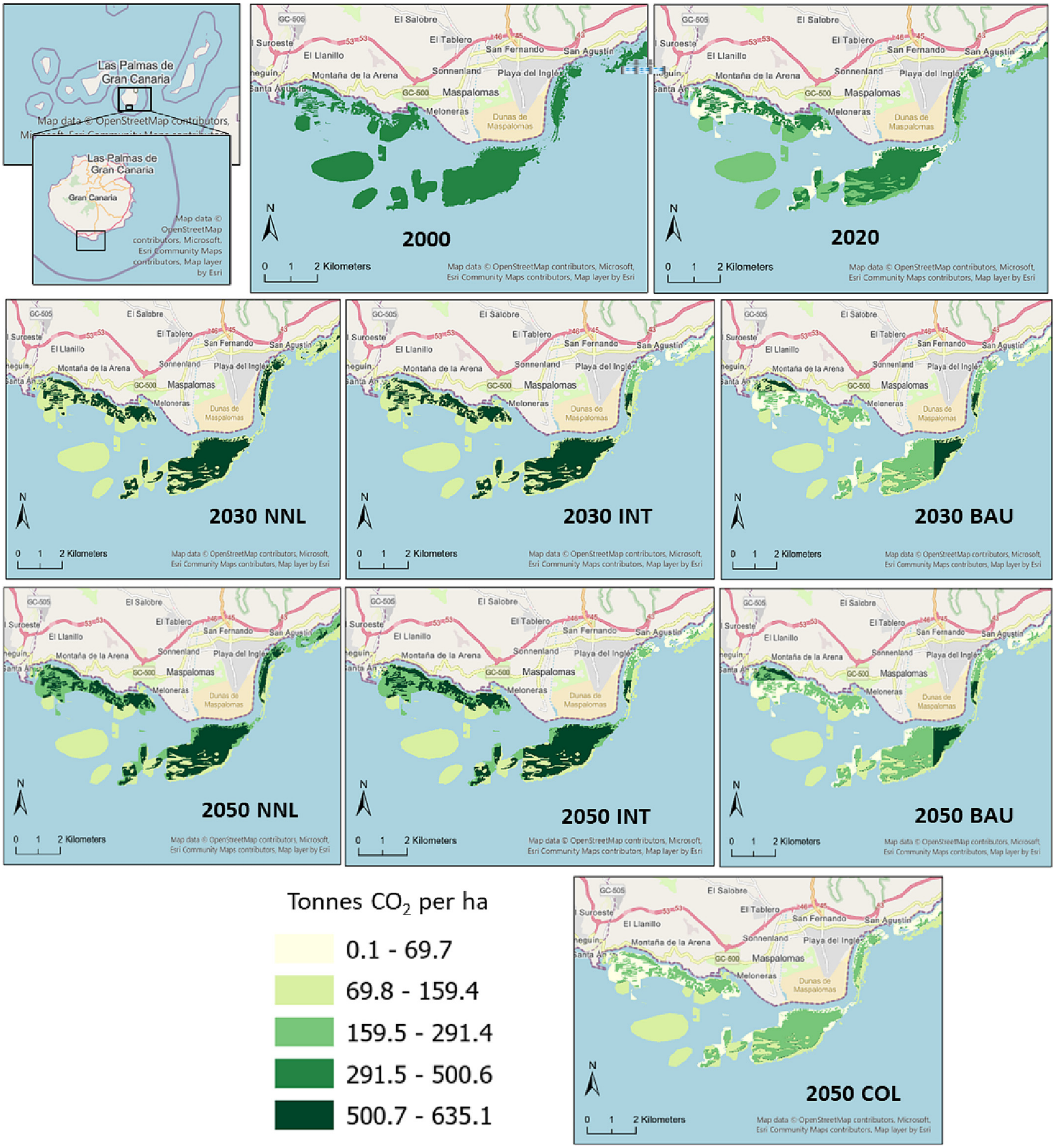


Fig. 6. Spatial representation of coastal blue carbon model outputs over time in Maspalomas (Gran Canaria), as a pilot case. Maps from 2000 and 2020 are carbon stocks based on *C. nodosa* field records. Maps from 2030 and 2050 denote changes in carbon stocks over time through the different scenarios considered (NNL-No Net Loss, INT-Intermedium, BAU-Business as usual, and COL-collapse).

2019, 2021), (iii) the exposure of meadows to an additional natural pressure: large, highly energetic, oceanic swells of the Atlantic Ocean (Portillo, 2014), and (iv) the effect of global warming, which is causing the proliferation of opportunistic species with greater tropical affinity, such as the cyanobacteria *Lyngbya majuscula* that has periodically covered and asphyxiate a large part of seagrass meadows in the easter islands of the archipelago (Martín-García et al., 2014).

If we translate the loss of seagrass meadows into coastal blue carbon estimations, a total amount of 0.21 Megatonnes CO_2 eq emitted was estimated. Comparing the outcomes of this research with those obtained by González-García et al. (2022), at the national level for Spain, the carbon stock in 2020 in the Canary Islands is 3.16 Megatonnes CO_2 eq, which is similar to the 3.17 Mt. CO_2 eq provided by this study. This occurs because the average carbon stock considered by González-García et al. (2022) in

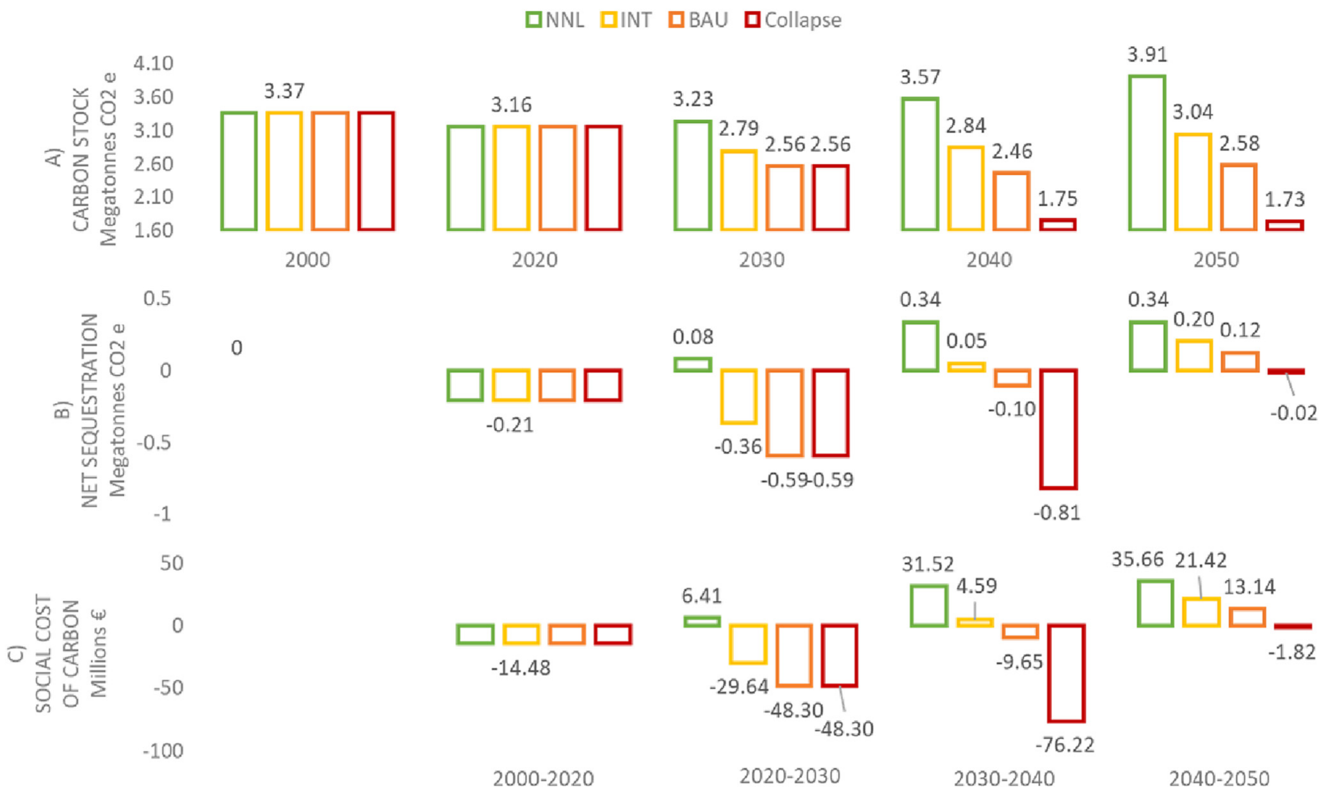


Fig. 7. Coastal blue carbon stocks and net sequestration over time under the four future scenarios. The "Social Cost of Carbon" is based on net carbon sequestration. NNL: No Net Loss, INT: Intermediate scenario, BAU: Business-as-usual.

2020 (470.93 t CO_2 eq/ha) was comparable to the regional average of CO_2 obtained by our study at the same year (440.6 t CO_2 eq/ha). The average carbon stock provided by González-García et al. (2022) was between 104 and 1224 t of CO_2 eq h^{-1}, depending on *C. nodosa* locations, which indicates the importance of carrying out studies with local data, as we here have implemented.

Given that carbon storage by *C. nodosa* in Spain is 29.5 Mt. CO_2 eq and by all seagrass species is 227 Mt. CO_2 eq (González-García et al., 2022), the Canary Islands carbon stock accounts for ca. 11 % of the total carbon storage by *C. nodosa* across the entire coasts of Spain and ca. 1.5 % of the total carbon storage by all seagrasses. This shows the relevance contribution of Canarian seagrass to the national climate regulation service.

On a worldwide scale, Fourqurean et al. (2012) estimated that the global seagrass carbon stock was between 15,372 and 30,744 Mt. CO_2 eq; therefore, the carbon stock in the Canary Islands in 2020 (3.2 Mt. CO_2 eq) is ca. 0.015 % of the global carbon storage by seagrasses.

4.2. Which are the viable future scenarios for coastal blue carbon?

The Paris Agreement states that it is essential to reduce emissions by a 45 % in 2030, and achieve zero net emissions by 2050. In this sense, countries are urged to accelerate their climate action and to review and increase their 2030 goals before the end of 2022. In this context, our study contributes to the approach of a NNL scenario and provides useful information to act on the pressures and areas that favour carbon sequestration by the seagrass *C. nodosa.* The NNL scenario was the only one where an increase of carbon was detected over time (the amount of carbon accumulated, and emissions avoided was 0.75 MT) and, therefore, it is the only scenario that would comply with the Paris Agreement.

According to our outcomes, to reach a NNL scenario would imply to directly act on several human pressures in areas where such pressures currently overcome thresholds considered to guarantee a healthy state of the seagrass. According to the thresholds selected for each pressure, the current distribution of *C. nodosa* would be seriously affected if management actions are not implemented. Specifically, Tenerife and Gran Canaria would be the islands suffering the greatest impacts, as both islands would lose ca. 50 % and 30 % in the area covered by the seagrass, respectively. Most of the cells that exceeded the thresholds belong to salinity and hydrographic pressures. Human activities, such as water treatment and coastal works, are responsible of these impacts. Nutrients from waste discharges mainly affect the two capital cities (Santa Cruz de Tenerife and Las Palmas de Gran Canaria), but there are currently no seagrass meadows in any of the cells that exceeded these thresholds. In the past, however, there were sparse seagrass records in Las Palmas de Gran Canaria, which disappeared in the last cartography.

4.3. Economic valuation and trade-offs

The NNL scenario is the only one that would avoid the social costs associated to emissions in the future and would save 73.59 M€, which is 0.19 % of the Canary Islands Gross domestic product (GDP) and 2.7 times the budget dedicated, in 2022, to this item by the "General Directory to Fight Climate Change and Ecological Transition" of the Canary Islands (Canarian government budgets 2022[3]). On the contrary, the emissions caused by the INT, BAU and COL scenarios would suppose costs of 3.6 M€ (0.01 % of the Canary GDP), 44.8 M€ (0.11 % of the Canary GDP) and 126.3 M€ (0.32 % of the Canary GDP), respectively.

Given the protection of *C. nodosa* in the Canary Islands and that minoring carbon emissions are public policy responsibilities, the benefits of zoning coastal areas against the social losses from carbon emissions (SCC) should be considered by decision makers. This valuation tries to

[3] http://www.gobiernodecanarias.org/cmsgobcan/export/sites/hacienda/dgplani/galeria/Presupuestos/2022/proyecto_de_ley/TOMO-3-Resumenes.pdf

shed light on this question, by providing data as the avoided costs if current *C. nodosa* is conserved, and the social cost of emitting carbon if other non-sustainable scenarios occur. It is important to highlight that the economic value estimated by this study does not include the total value of *C. nodosa* seagrass meadows, for which the rest of the services that *C. nodosa* provides should have been evaluated, such as the fisheries support value (Tuya et al., 2014a). Rather, this study provides an economic approximation to just one service. Still, this is a useful way to insert the value of *C. nodosa* into decision-making.

4.4. Study limitations and future research

Some study limitations should be considered in future research. For example, sediment rates or trapping rates due to seagrass canopies should be further studied. We took information from other regions and these rates are dependent on species and location. For instance, bays and lagoons have higher rates than islands exposed to energetic seas, and the capacity of seagrasses to trap sediment might be lost under extreme flow conditions in wave-exposed environments. In addition, detritus generated annually by *C. nodosa* should be quantified, because there is a lack of current local information.

This study does not consider the carbon sequestration by species living in close association with *C. nodosa*, such as epiphytic algae or other benthic species (e.g.. the green algae *Caulerpa prolifera*, (Tuya et al., 2014b)), due to a lack of data on their distribution and accumulation rate. Therefore, their effect on carbon storage should be further studied.

Future research is needed on the relation between seagrass condition and blue carbon storage and sequestration. The ecosystem condition[4] inclusion in blue carbon assessment would be positive to ease the implementation of marine nature-based solutions (Watson et al., 2022).

The seagrass distribution maps used in this study were subjected to ground-truthing via underwater video techniques. Extensive ground-truthing for seagrass distribution has been recently demonstrated by even camera-equipped tiger sharks (Gallagher et al., 2022).

Besides, only direct anthropogenic pressures were considered when future scenarios were created. Indirect anthropogenic pressures on seagrass meadows, such as rising sea level and increasing temperature were not investigated, and further studies are required in this sense (Dahl et al., 2023). Additionally, we didn't explored how the results of this study can be used to promote the potential of the Canary archipelago seagrass meadows to national and international agencies as a practical scheme for nature-based solutions for climate change mitigation (Stankovic et al., 2021). Finally, a restoration scenario has not been fully explored (Tuya et al., 2017) due to, among other reasons, the exposure of local meadows to large, highly energetic, oceanic swells travelling the Atlantic Ocean (Portillo, 2014), but it may be considered in future research.

5. Conclusion

This study has demonstrated a severe decline (ca. 50 %) in the area covered by seagrass meadows, from 2000 to 2018, in the Canary Islands. This is the first study that has quantified and spatially represented the past, actual, and plausible future scenarios of *C. nodosa* and its associated blue carbon with a high spatial resolution (20 × 20 m). We have identified the areas and pressures that need to be acted upon, and created plausible future scenarios, to communicate the impacts on *C. nodosa* according to management decisions. A monetary valuation according to different scenarios was carried out to make the outcomes understandable for decision-makers and the general public. Importantly, the reproducibility of our methodology elsewhere guarantees its applicability across coastal ecosystems for conservation planning, particularly in promoting further research on seagrass meadows in other regions.

[4] According to Maes et al. (2018), ecosystem condition refers to the physical, chemical and biological condition or quality of an ecosystem at a particular point in time.

CRediT authorship contribution statement

Miriam Montero-Hidalgo: Conceptualization, Investigation, Data curation, Formal analysis, Methodology, Software, Visualization, Writing – original draft. **Fernando Tuya:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Supervision, Writing – review & editing. **Francisco Otero-Ferrer:** Conceptualization, Investigation, Formal analysis, Writing – review & editing. **Ricardo Haroun:** Investigation, Formal analysis, Writing – review & editing. **Fernando Santos-Martín:** Conceptualization, Investigation, Methodology, Project administration, Funding acquisition, Supervision, Writing – review & editing.

Data availability

Data will be made available on request.

Declaration of competing interest

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

Acknowledgements

This work was supported by the European Union under the MOVE-ON project “From case studies to anchor projects - setting the ground to advance MAES in Europe's overseas”, grant agreement Nº07.027735/2019/808239/SUB/ENV.D2, https://moveon-project.eu/.

We also thank the Biodiversity Service of the Canarian Government for providing cartographic databases, as well as to Alberto Gonzalez-García and Adrián García Bruzón for their generous support with the InVEST and ArcGIS software.

References

Alberto, F., Arnaud-Haond, S., Duarte, C.M., Serrão, E.A., 2006. Genetic diversity of a clonal angiosperm near its range limit: the case of Cymodocea nodosa at the Canary Islands. Mar. Ecol. Prog. Ser. 309, 117–129. https://doi.org/10.3354/meps309117.

Alberto, F., Massa, S., Manent, P., Diaz-Almela, E., Arnaud-Haond, S., Duarte, C.M., Serrão, E.A., 2008. Genetic differentiation and secondary contact zone in the seagrass Cymodocea nodosa across the Mediterranean-Atlantic transition region. J. Biogeogr. 35 (7), 1279–1294. https://doi.org/10.1111/j.1365-2699.2007.01876.x.

Ascioti, F.A., Mangano, M.C., Marcianò, C., Sarà, G., 2022. The sanitation service of seagrasses – dependencies and implications for the estimation of avoided costs. Ecosyst. Serv. 54, 101418. https://doi.org/10.1016/J.ECOSER.2022.101418.

Bañolas, G., Fernández, S., Espino, F., Haroun, R., Tuya, F., 2020. Evaluation of carbon sinks by the seagrass Cymodocea nodosa at an oceanic island: spatial variation and economic valuation. Ocean Coast. Manag. 187. https://doi.org/10.1016/j.ocecoaman.2020.105112.

Barbier, E.B., 2013. Valuing ecosystem services for coastal wetland protection and restoration: progress and challenges. Resources 2 (3), 213–230. https://doi.org/10.3390/resources2030213.

Barbier, E.B., Hacker, S., Kennedy, C., Koch, E., Stier, A., Silliman, B., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81. https://doi.org/10.1890/10-1510.1.

Barquín, J., Martín, L., 2011. Atlas Bionómico de las Islas Canarias. SIGMACAN Project-"CREACIÓN DE UN SISTEMA DE INFORMACIÓN GEOGRÁFICA (SIG) DE LOS FONDOS MARINOS SOMEROS DEL ARCHIPIÉLAGO CANARIO“.

Benzaken, D., Renard, Y., 2010. Future Directions for Biodiversity Action in Europe Overseas Convention on Biological Diversity, December 2010. December. IUCN.

Brito, A., 1984. El medio marino. In: Edirca, S.L. (Ed.), Fauna (marina y terrestre) del Archipielago Canario, pp. 27–86.

Buis, A., 2019. The Atmosphere: Getting a Handle on Carbon Dioxide. NASA’s Jet Propulsion Laboratory https://climate.nasa.gov/news/2915/the-atmosphere-getting-a-handle-on-carbon-dioxide/#:~:text=The concentration of carbon dioxide, it was near 370 ppm.

Burkholder, J.A.M., Tomasko, D.A., Touchette, B.W., 2007. Seagrasses and eutrophication. J. Exp. Mar. Biol. Ecol. 350 (1–2), 46–72. https://doi.org/10.1016/J.JEMBE.2007.06.024.

Cai, W., Zhu, Q., Chen, M., Cai, Y., 2021. Spatiotemporal change and the natural–human driving processes of a megacity’s coastal blue carbon storage. Int. J. Environ. Res. Public Health 18 (16). https://doi.org/10.3390/ijerph18168879.

Cancemi, G., Buia, M.C., Mazzella, L., 2002. Structure and growth dynamics of Cymodocea nodosa meadows. Sci. Mar. 66 (4), 365–373. https://doi.org/10.3989/scimar.2002.66n4365.

Caye, G., Meinesz, A., 1985. Observations on the vegetative development, flowering and seeding of cymodocea nodosa (Ucria) ascherson on the Mediterranean coasts of France. Aquat. Bot. 22, 277–289.
Ceccherelli, G., Campo, D., Milazzo, M., 2007. Short-term response of the slow growing seagrass Posidonia oceanica to simulated anchor impact. Mar. Environ. Res. 63 (4), 341–349. https://doi.org/10.1016/J.MARENVRES.2006.10.004.
Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. Glob. Biogeochem. Cycles 17 (4). https://doi.org/10.1029/2002GB001917.
Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387 (6630), 253–260. https://doi.org/10.1038/387253a0.
Cunha, A.H., Araújo, A., 2009. New distribution limits of seagrass beds in West Africa. J. Biogeogr. https://doi.org/10.1111/j.1365-2699.2009.02135.x.
Dahl, M., McMahon, K., Lavery, P.S., Hamilton, S.H., Lovelock, C.E., Serrano, O., 2023. Ranking the risk of CO2 emissions from seagrass soil carbon stocks under global change threats. Glob. Environ. Chang. 78 (December 2022), 102632. https://doi.org/10.1016/j.gloenvcha.2022.102632.
Drupp, M.A., Freeman, M., Groom, B., Nesje, F., 2015. Discounting disentangled: an expert survey on the determinants of the long-term social discount rate. SSRN Electron. J. 54. https://doi.org/10.2139/ssrn.2616220.
Duarte, C.M., Middelburg, J.J., Caraco, N., 2005. Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences 2 (1), 1–8. https://doi.org/10.5194/bg-2-1-2005.
Duarte, C.M., Losada, I.J., Hendriks, I.E., Mazarrasa, I., Marbà, N., 2013. The role of coastal plant communities for climate change mitigation and adaptation. Nat. Clim. Chang. 3 (11), 961–968. https://doi.org/10.1038/nclimate1970.
Duffy, J. Emmett, Stachowicz, J.J., Reynolds, P.L., Hovel, K.A., Jahnke, M., Sotka, E.E., Bostrom, C., Boyer, K.E., Cusson, M., Eklof, J., Engelen, A.H., Eriksson, B.K., Joel Fodrie, F., Griffin, J.N., Hereu, C.M., Hori, M., Randall Hughes, A., Ivanov, M.V., Jorgensen, P., Olsen, J.L., 2022. A Pleistocene legacy structures variation in modern seagrass ecosystems. Proceedings of the National Academy of Sciences of the United States of America 119 (32), 1–8. https://doi.org/10.1073/pnas.2121425119.
El-Hamid, H.T.A., Eid, E.M., El-Morsy, M.H.E., Osman, H.E.M., Keshta, A.E., 2022. Benefits of blue carbon stocks in a Coastal Jazan ecosystem undergoing land use change. Wetlands 42 (8). https://doi.org/10.1007/s13157-022-01597-9.
Espino, F., 2004. Una metodología Para el estudio de las fanerógamas marinas en Canarias. Revista de La academia Canarias deCiencias XV, 3–4 (XV), 237–256.. http://www.redmic.es/bibliografia/Docum_02939.pdf.
Espino, F., Tuya, F., Blanch, I., Haroun, R.J., 2008. Los sebadales de Canarias. BIOGES, Universidad de Las Palmas de Gran Canaria, p. 68.
Espino, F., Brito, A., Haroun, R., 2011. Ictiofauna asociada a las praderas de Cymodocea nodosa en las islas Canarias (Atlántico centro oriental): estructura de la comunidad y función de "guardería". Cienc. Mar. 37, 157–174.
Esteban, N., Unsworth, R.K.F., Gourlay, J.B.Q., Hays, G.C., 2018. The discovery of deep-water seagrass meadows in a pristine Indian Ocean wilderness revealed by tracking green turtles. Mar. Pollut. Bull. 134, 99–105. https://doi.org/10.1016/j.marpolbul.2018.03.018.
Fabbri, F., Espino, F., Herrera, R., Moro, L., Haroun, R., Riera, R., González-Henriquez, N., Bergasa, O., Monterroso, O., De La Rosa, M.R., De La Rosa, M.R., Tuya, F., 2015. Trends of the seagrass Cymodocea nodosa (Magnoliophyta) in the Canary Islands: population changes in the last two decades | tendencias de la fanerógama marina Cymodocea nodosa (Magnoliophyta) en las islas Canarias: cambios poblacionales en las dos última. Sci. Mar. 79 (1), 7–13. https://doi.org/10.3989/scimar.04165.19B.
Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M.A., Apostolaki, E.T., Kendrick, G.A., Krause-Jensen, D., McGlathery, K.J., Serrano, O., 2012. Seagrass ecosystems as a globally significant carbon stock. Nat. Geosci. 5 (7), 505–509. https://doi.org/10.1038/ngeo1477.
Fourqurean, J.W., Johnson, B., Kauffman, J.B., Kennedy, H., Lovelock, C.E., Megonigal, J.P., Rahman, A., Saintilan, N., Simard, M., 2019. Coastal blue carbon. Habitat Conservation. Ci , p. 860.. http://www.habitat.noaa.gov/coastalbluecarbon.html.
Gallagher, A.J., Brownscombe, J.W., Alsudairy, N.A., Casagrande, A.B., Fu, C., Harding, L., Harris, S.D., Hammerschlag, N., Howe, W., Huertas, A.D., Kattan, S., Kough, A.S., Musgrove, A., Payne, N.L., Phillips, A., Shea, B.D., Shipley, O.N., Sumaila, U.R., Hossain, M.S., Duarte, C.M., 2022. Tiger sharks support the characterization of the world's largest seagrass ecosystem. Nat. Commun. 13 (1), 1–10. https://doi.org/10.1038/s41467-022-33926-1.
González-García, A., Arias, M., García-Tiscar, S., Alcorlo, P., Santos-Martín, F., 2022. National blue carbon assessment in Spain using InVEST: current state and future perspectives. Ecosyst. Serv. 53, 101397. https://doi.org/10.1016/J.ECOSER.2021.101397.
Haroun, R., Wildpret de la Torre, W., Gil-Rodríguez, M.C., 2003. In: Canseco (Ed.), Plantas Marinas de las islas Canarias.
Hartog, C.Den, Kuo, J., 2006. Taxonomy and biogeography of seagrasses. Seagrasses: Biology, Ecology and Conservation, pp. 1–23 https://doi.org/10.1007/978-1-4020-2983-7_1.
Hernández-Blanco, M., Moritsch, M., Manrow, M., Raes, L., 2022. Coastal ecosystem services modeling in Latin America to guide conservation and restoration strategies: the case of mangroves in Guatemala and El Salvador. Front. Ecol. Evol. 10. https://doi.org/10.3389/fevo.2022.843145.
Houghton, R.A., 2003. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850–2000. Tellus Ser. B Chem. Phys. Meteorol. 55 (2), 378–390. https://doi.org/10.1034/j.1600-0889.2003.01450.x.
IPCC, 2013. Summary for policymakers. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y. (Eds.), Climate Change 2013: The Physical Science Basis. V.B. and P.M.M. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate43. Chemistry International, Cambridge. https://doi.org/10.1515/ci-2021-0407 Issue 4.
Kacem, H.A., Bouroubi, Y., Khomalli, Y., Elyaagoubi, S., Maanan, M., Rhinane, H., Maanan, M., 2022. The economic benefit of coastal blue carbon stocks in a moroccan lagoon ecosystem: a case study at Moulay Bousselham lagoon. Wetlands 42 (2). https://doi.org/10.1007/s13157-022-01533-x.
Kilminster, K., McMahon, K., Waycott, M., Kendrick, G.A., Scanes, P., McKenzie, L., O'Brien, K.R., Lyons, M., Ferguson, A., Maxwell, P., Glasby, T., Udy, J., 2015. Unravelling complexity in seagrass systems for management: Australia as a microcosm. Sci. Total Environ. 534, 97–109. https://doi.org/10.1016/j.scitotenv.2015.04.061.
Lau, W.W.Y., 2013. Beyond carbon: conceptualizing payments for ecosystem services in blue forests on carbon and other marine and coastal ecosystem services. Ocean Coast. Manag. 83, 5–14. https://doi.org/10.1016/J.OCECOAMAN.2012.03.011.
Lavery, P.S., Mateo, M.-A., Serrano, O., Rozaimi, M., 2013. Variability in the carbon storage of seagrass habitats and its implications for global estimates of blue carbon ecosystem service) variability in the carbon storage of seagrass habitats and its implications for global estimates of blue carbon ecosystem service. PLoS ONE 8 (9), 73748. https://doi.org/10.1371/journal.pone.0073748.
Liu, Y., Du, M., Cui, Q., Lin, J., Liu, Y., Liu, Q., Tong, D., Feng, K., Hubacek, K., 2022. Contrasting suitability and ambition in regional carbon mitigation. NatureCommunications 13 (1). https://doi.org/10.1038/s41467-022-31729-y.
Macreadie, P.I., Anton, A., Raven, J.A., Beaumont, N., Connolly, R.M., Friess, D.A., Kelleway, J.J., Kennedy, H., Kuwae, T., Lavery, P.S., Lovelock, C.E., Smale, D.A., Apostolaki, E.T., Atwood, T.B., Baldock, J., Bianchi, T.S., Chmura, G.L., Eyre, B.D., Fourqurean, J.W., Duarte, C.M., 2019. The future of blue carbon science. Nat. Commun. 10 (1), 3998. https://doi.org/10.1038/s41467-019-11693-w.
Macreadie, P.I., Costa, M.D.P., Atwood, T.B., Friess, D.A., Kelleway, J.J., Kennedy, H., Lovelock, C.E., Serrano, O., Duarte, C.M., 2021. Blue carbon as a natural climate solution. Nat. Rev. Earth Environ. 2 (12), 826–839. https://doi.org/10.1038/s43017-021-00224-1.
Maes, J., Teller, A., Erhard, M., Grizzetti, B., Barredo, J.I., Paracchini, M.L., Condé, S., Somma, F., Orgiazzi, A., Jones, A., Zulian, A., Petersen, J.E., Marquardt, D., Kovacevic, V., Malak, D.A., Marin, A.I., Czúcz, B., Mauri, A., Loffler, P., Werner, B., 2018. Mapping And Assessment of Ecosystems And Their Services: An Analytical Framework for Ecosystem Condition. L. Publications office of the European Union. https://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/pdf/5thMAESreport.pdf.
Manent, P., Bañolas, G., Alberto, F., Curbelo, L., Espino, F., Tuya, F., 2020. Long-term seagrass degradation: integrating landscape, demographic, and genetic responses. Aquat. Conserv. Mar. Freshwat. Ecosyst. 30 (6), 1111–1120. https://doi.org/10.1002/aqc.3325.
Máñez-Crespo, J., Tuya, F., Fernández-Torquemada, Y., Royo, L., Pilar-Ruso, Y.D., Espino, F., Manent, P., Antich, L., Castejón-Silvo, I., Curbelo, L., Terrados, J., Tomas, F., 2020. Seagrass Cymodocea nodosa across biogeographical regions and times: differences in abundance, meadow structure and sexual reproduction. Mar. Environ. Res. 162. https://doi.org/10.1016/j.marenvres.2020.105159.
Marbà, N., Cebrián, J., Enríquez, S., Duarte, C.M., 1996. Growth patterns of western Mediterranean seagrasses: species- specific responses to seasonal forcing. Mar. Ecol. Prog. Ser. 133 (1–3), 203–215. https://doi.org/10.3354/meps133203.
Martín-García, L., Herrera, R., Moro-Abad, L., Sangil, C., Barquín-Diez, J., 2014. Predicting the potential habitat of the harmful cyanobacteria Lyngbya majuscula in the Canary Islands (Spain). Harmful Algae 34, 76–86. https://doi.org/10.1016/J.HAL.2014.02.008.
Mascaró, O., Oliva, S., Pérez, M., Romero, J., 2009. Spatial variability in ecological attributes of the seagrass Cymodocea nodosa. Bot. Mar. 52 (5), 429–438. https://doi.org/10.1515/BOT.2009.055.
MITECO, 2019. Documento Marco General Evaluación Inicial y Buen estado ambiental Estrategias Marinas. Ministerio Para La Transición Ecológica (MITECO) , p. 247.. https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/estrategias-marinas/eemm_2dociclo_fases123.aspx.
MITECO, 2019b. Estrategias marinas de España. Anexo parte II: fichas del análisis de presiones e impactos. Ministerio Para La Transición Ecológica (MITECO).
Monnier, B., Pergent, G., Mateo, M.Á., Carbonell, R., Clabaut, P., Pergent-Martini, C., 2021. Sizing the carbon sink associated with Posidonia oceanica seagrass meadows using very high-resolution seismic reflection imaging. Mar. Environ. Res. 170. https://doi.org/10.1016/j.marenvres.2021.105415.
Monnier, B., Pergent, G., Mateo, M.Á., Clabaut, P., Pergent-Martini, C., 2022. Quantification of blue carbon stocks associated with Posidonia oceanica seagrass meadows in Corsica (NW Mediterranean). Sci. Total Environ. 838. https://doi.org/10.1016/j.scitotenv.2022.155864.
Montefalcone, M., Chiantore, M., Lanzone, A., Morri, C., Albertelli, G., Bianchi, C.N., 2008. BACI design reveals the decline of the seagrass Posidonia oceanica induced by anchoring. Mar. Pollut. Bull. 56 (9), 1637–1645. https://doi.org/10.1016/J.MARPOLBUL.2008.05.013.
Monterroso, O., Rodríguez, M., Riera, R., Pérez, O., Ramos, E., Álvarez, O., Domínguez, J., 2015. Memoria Final del "Seguimiento de sebadales en retroceso: Fuerteventura.". CIMA S.L. 55 pp.
Monterroso, O., Rodríguez, M., Riera, R., Pérez, O., Ramos, E., Álvarez, O., Domínguez, J., 2016. Memoria Final del "Seguimiento de sebadales en retroceso: Lanzarote y Gran Canaria.". CIMA S.L. 63 pp.
Monterroso, O., Rodríguez, M., Pérez, O., Ramos, E., Álvarez, O., Cruces, L., Ruiz, M., Miguel, A., González, M., 2018. Memoria final del estudio "Cartografía de Cymodocea nodosa en Tenerife y La Gomera.". CIMA S.L. 164 pp.
Moreira-Reyes, A., Acuna, D., Candelaria Gil-Rodriguez, M., 2013. Characterization of a seagrass ("sebadal") in La caleta, Valverde, El hierro, Canary Islands. Vieraea 41, 61–71.
Murray, B., Pendleton, L., Jenkins, W., Sifleet, S., 2011. Green Payments for Blue Carbon: Economic Incentives for Protecting Threatened Coastal Habitats. April, 52Nicholas Institute for Environmental. http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Green+Payments+for+Blue+Carbon+Economic+Incentives+for+Protecting+Threatened+Coastal+Habitats#0%5Cnhttp://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Green+payments+for+blue+carbon:+.
Orth, R.J., Carruthers, T.J.B., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., Short, F.T., Waycott, M., Williams, S.L., 2006. A global crisis for seagrass ecosystems. Bioscience 56 (12), 987–996. https://doi.org/10.1641/0006-3568(2006)56[987:AGCFSE]2.0.CO;2.

Pavón-Salas, N., Herrera, R., Hernández-Guerra, A., Haroun, R., 2000. Distributional pattern of seagrasses in the Canary Islands (Central-East Atlantic Ocean). J. Coast. Res. 16 (2), 329–335.

Perez-Ruzafa, A., Marcos-Diego, C., Ros, J.D., 1991. Environmental and biological changes related to recent human activities in the mar menor (SE of Spain). Mar. Pollut. Bull. 23 (C), 747–751. https://doi.org/10.1016/0025-326X(91)90774-M.

Portillo, E., 2014. Relation between the type of wave exposure and seagrass losses (Cymodocea nodosa) in the south of gran canaria (Canary Islands - Spain). Oceanol. Hydrobiol. Stud. 43 (1), 29–40. https://doi.org/10.2478/s13545-014-0114-2.

Rennert, K., Errickson, F., Prest, B.C., Rennels, L., Newell, R.G., Pizer, W., Kingdon, C., Wingenroth, J., Cooke, R., Parthum, B., Smith, D., Cromar, K., Diaz, D., Moore, F.C., Müller, U.K., Plevin, R.J., Raftery, A.E., Ševčíková, H., Sheets, H., Anthoff, D., 2022. Comprehensive evidence implies a higher social cost of CO2. Nature https://doi.org/10.1038/s41586-022-05224-9.

Reyes, J., Sanson, M., Afonso-Carrillo, J., 1995. Leaf phenology, growth and production of the seagrass Cymodocea nodosa at El medano (South of Tenerife, Canary Islands). Bot. Mar. 38 (1–6), 457–466. https://doi.org/10.1515/botm.1995.38.1-6.457.

Rosa, L.N., de Paula, Duarte, Costa, M., de Freitas, D.M., 2022. Modelling spatial-temporal changes in carbon sequestration by mangroves in an urban coastal landscape. Estuar. Coast. Shelf Sci. 276. https://doi.org/10.1016/j.ecss.2022.108031.

Ruiz de la Rosa, M., Tuya, F., Herrera, R., Moro Abad, L., Espino, F., Haroun, R., Manent, P., 2015. Praderas de angiospermas marinas de las Islas Canarias. Atlas de las praderas marinas de España, pp. 423–488.

Serrano, O., Lavery, P.S., López-Merino, L., Ballesteros, E., Mateo, M.A., 2016. Location and associated carbon storage of erosional escarpments of seagrass posidonia mats. Frontiers in Marine Science 3. https://doi.org/10.3389/fmars.2016.00042.

Short, F.T., Polidoro, B., Livingstone, S.R., Carpenter, K.E., Bandeira, S., Bujang, J.S., Calumpong, H.P., Carruthers, T.J.B., Coles, R.G., Dennison, W.C., Erftemeijer, P.L.A., Fortes, M.D., Freeman, A.S., Jagtap, T.G., Kamal, A.H.M., Kendrick, G.A., Judson Kenworthy, W., La Nafie, Y.A., Nasution, I.M., Zieman, J.C., 2011. Extinction risk assessment of the world's seagrass species. Biol. Conserv. 144 (7), 1961–1971. https://doi.org/10.1016/J.BIOCON.2011.04.010.

Sieber, I.M., Av Borges, P., Burkhard, B., 2018. Hotspots of biodiversity and ecosystem services: the outermost regions and overseas countries and territories of the European Union. One Ecosyst. 3, e24719. https://doi.org/10.3897/oneeco.3.e24719.

Sieber, I.M., Montero-Hidalgo, M., Kato-Huerta, J., Rendon, P., Santos-Martín, F., Geneletti, D., Gil, A., Trégarot, E., Lagabrielle, E., Parelho, C., Arbelo, M., van Beukering, P., Bayley, D., Casas, E., Duijndam, S., Cillaurren, E., David, G., Dourdain, A., Haroun, R., Burkhard, B., 2022. Mapping and assessing ecosystem services in Europe's overseas: a comparative analysis of MOVE case studies. One Ecosystem 7. https://doi.org/10.3897/oneeco.7.e87179.

Simpson, J., Bruce, E., Davies, K.P., Barber, P., 2022. A blueprint for the estimation of seagrass carbon stock using remote sensing-enabled proxies. Remote Sens. 14 (15). https://doi.org/10.3390/rs14153572.

Sousa, A.I., Lillebø, A.I., Risgaard-Petersen, N., Pardal, M.A., Caçador, I., 2012. Denitrification: an ecosystem service provided by salt marshes. Mar. Ecol. Prog. Ser. 448, 79–92. https://doi.org/10.3354/meps09526.

Stankovic, M., Ambo-Rappe, R., Carly, F., Dangan-Galon, F., Fortes, M.D., Hossain, M.S., Kiswara, W., Van Luong, C., Minh-Thu, P., Mishra, A.K., Noiraksar, T., Nurdin, N., Panyawai, J., Rattanachot, E., Rozaimi, M., Soe Htun, U., Prathep, A., 2021. Quantification of blue carbon in seagrass ecosystems of Southeast Asia and their potential for climate change mitigation. Sci. Total Environ. 783, 146858. https://doi.org/10.1016/j.scitotenv.2021.146858.

Tanaya, T., Watanabe, K., Yamamoto, S., Hongo, C., Kayanne, H., Kuwae, T., 2018. Contributions of the direct supply of belowground seagrass detritus and trapping of suspended organic matter to the sedimentary organic carbon stock in seagrass meadows. Biogeosciences 15 (13), 4033–4045. https://doi.org/10.5194/bg-15-4033-2018.

Townsend, M., Thrush, S.F., Lohrer, A.M., Hewitt, J.E., Lundquist, C.J., Carbines, M., Felsing, M., 2014. Overcoming the challenges of data scarcity in mapping marine ecosystem service potential. Ecosyst. Serv. 8, 44–55. https://doi.org/10.1016/J.ECOSER.2014.02.002.

Traganos, D., Lee, C.B., Blume, A., Poursanidis, D., Čižmek, H., Deter, J., Mačić, V., Montefalcone, M., Pergent, G., Pergent-Martini, C., Ricart, A.M., Reinartz, P., 2022. Spatially explicit seagrass extent mapping across the entire Mediterranean. Front. Mar. Sci. 9. https://doi.org/10.3389/fmars.2022.871799.

Tuya, F., Martín, J.A., Luque, A., 2006. Seasonal cycle of a Cymodocea nodosa seagrass meadow and of the associated ichthyofauna at Playa Dorada (Lanzarote, Canary Islands, eastern Atlantic) | Ciclo estacional de una pradera marina de Cymodocea nodosa y la ictiofauna asociada en Playa Dorada (La. Ciencias Marinas 32 (4), 695–704. https://doi.org/10.7773/cm.v32i4.1158.

Tuya, F., Hernandez-Zerpa, H., Espino, F., Haroun, R., 2013a. Drastic decadal decline of the seagrass Cymodocea nodosa at gran canaria (eastern Atlantic): interactions with the green algae Caulerpa prolifera. Aquat. Bot. 105, 1–6. https://doi.org/10.1016/j.aquabot.2012.10.006.

Tuya, F., Viera-Rodríguez, M.A., Guedes, R., Espino, F., Haroun, R., Terrados, J., 2013b. Seagrass responses to nutrient enrichment depend on clonal integration, but not flow-on effects on associated biota. Mar. Ecol. Prog. Ser. 490, 23–35. https://doi.org/10.3354/meps10448.

Tuya, F., Haroun, R., Espino, F., 2014a. Economic assessment of ecosystem services: monetary value of seagrass meadows for coastal fisheries. Ocean Coast. Manag. 96, 181–187. https://doi.org/10.1016/j.ocecoaman.2014.04.032.

Tuya, F., Png-Gonzalez, L., Riera, R., Haroun, R., Espino, F., 2014b. Ecological structure and function differs between habitats dominated by seagrasses and green seaweeds. Mar. Environ. Res. 98, 1–13. https://doi.org/10.1016/j.marenvres.2014.03.015.

Tuya, F., Ribeiro-Leite, L., Arto-Cuesta, N., Coca, J., Haroun, R., Espino, F., 2014c. Decadal changes in the structure of Cymodocea nodosa seagrass meadows: natural vs. Human influences. Estuar. Coast. Shelf Sci. 137 (1), 41–49. https://doi.org/10.1016/j.ecss.2013.11.026.

Tuya, F., Vila, F., Bergasa, O., Zarranz, M., Espino, F., Robaina, R.R., 2017. Artificial seagrass leaves shield transplanted seagrass seedlings and increase their survivorship. Aquat. Bot. 136, 31–34. https://doi.org/10.1016/J.AQUABOT.2016.09.001.

Tuya, F., Fernández-Torquemada, Y., Zarcero, J., del Pilar-Ruso, Y., Csenteri, I., Espino, F., Manent, P., Curbelo, L., Antich, A., de la Ossa, J.A., Royo, L., Castejón, I., Procaccini, G., Terrados, J., Tomas, F., 2019. Biogeographical scenarios modulate seagrass resistance to small-scale perturbations. J. Ecol. 107 (3), 1263–1275. https://doi.org/10.1111/1365-2745.13114.

Tuya, F., Fernández-Torquemada, Y., del Pilar-Ruso, Y., Espino, F., Manent, P., Curbelo, L., Otero-Ferrer, F., de la Ossa, J.A., Royo, L., Antich, L., Castejón, I., Máñez-Crespo, J., Mateo-Ramírez, Á., Procaccini, G., Marco-Méndez, C., Terrados, J., Tomas, F., 2021. Partitioning resilience of a marine foundation species into resistance and recovery trajectories. Oecologia 196 (2), 515–527. https://doi.org/10.1007/s00442-021-04945-4.

U.S. IAWG SCGG, 2021. Technical support document: social cost of carbon, methane, and nitrous oxide. Interim Estimates Under Executive Order 13990. February. Interagency Working Group on Social Cost of Greenhouse Gases, United States Government , pp. 1–44.. https://www.whitehouse.gov/wp-content/uploads/2021/02/TechnicalSupportDocument_SocialCostofCarbonMethaneNitrousOxide.pdf.

Ward, E.A., Aldis, C., Wade, T., Miliou, A., Tsimpidis, T., Cameron, T.C., 2022. Is all seagrass habitat Equal? Seasonal, spatial, and interspecific variation in productivity dynamics within Mediterranean seagrass habitat. Front. Mar. Sci. 9. https://doi.org/10.3389/fmars.2022.891467.

Watson, S.C.L., Watson, G.J., Beaumont, N.J., Preston, J., 2022. Inclusion of condition in natural capital assessments is critical to the implementation of marine nature-based solutions. Sci. Total Environ. 838, 156026. https://doi.org/10.1016/J.SCITOTENV.2022.156026.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. 106 (30), 12377–12381. https://doi.org/10.1073/pnas.0905620106.

Wedding, L.M., Moritsch, M., Verutes, G., Arkema, K., Hartge, E., Reiblich, J., Douglass, J., Taylor, S., Strong, A.L., 2021. Incorporating blue carbon sequestration benefits into sub-national climate policies. Glob. Environ. Chang. 69, 102206. https://doi.org/10.1016/J.GLOENVCHA.2020.102206.