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Reconstruction of *Cymodocea nodosa*'s dynamics as a tool to examine the conservation status of a Mediterranean declared marine protected area

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Abstract

Seagrass habitats rank amongst the most valuable ecosystems in the biosphere. They support fisheries production, climate change mitigation, water quality improvement, and coastal protection. Faced with the current global crisis of accelerating losses of this key component of coastal environments, strong efforts have been expended within the conservation movement to flatten their decline curve. Although understanding the functioning of seagrass ecosystems and how they respond to stressors has improved over the last years, major gaps exist for West Africa including basic ecological and distributional knowledge. This study aimed to investigate, for the first time, the structural development and dynamics of two *Cymodocea nodosa* meadows in Al Hoceima National Park (the unique official marine protected area of the Mediterranean Moroccan coast) using the reconstruction techniques as an indirect measurement of seagrass growth. *C. nodosa's* leaves were remarkably invaded by epiphytes while leaf production, shoot density, above and belowground biomass, and vertical and horizontal rhizome elongation were in the lowest recorded values. Shoot age structure showed that since the second-year class, the survival rate of shoots has decreased. Regarding the severely slow recruitment rates, the populations' net growths declined. Besides this negative growth trend, CymoSkew^m biotic index assigned a slightly polluted ecological status to both meadows. The seagrass faces numerous pressures, mainly fishing by trawling and alien species invasion. The primary conservation challenge for *C. nodosa* of Al Hoceima National Park is ensuring that this marine protected area becomes operationally implemented and actively managed. Until then, the seagrass habitat-forming species is not safeguarded.

Keywords: Seagrass; Al Hoceima National Park; reconstruction techniques; population dynamics; effective management and conservation.

Introduction

Seagrasses are flowering marine plants that inhabit coastal ecosystems worldwide (Short *et al.*, 2018). Although they are valuable to humankind, seagrasses are underappreciated compared to colourful coral reefs and mighty mangroves in terms of research and conservation measures (Unsworth *et al.*, 2019). However, their ecosystem service value has been estimated to be three times higher than that of coral reefs and ten times higher than that of tropical forests (Short *et al.*, 2018).

Seagrass meadows are of fundamental importance to global food security, supporting 20% of the world's most significant fisheries (Unsworth *et al.*, 2018) with a total value of at least €200 million per year in the Mediterranean alone (Jackson *et al.*, 2015). Seagrasses' leaves can reduce water velocity and thereby attenuate the impact of wave action on coastlines (Christianen *et al.*, 2013; Paul, 2018). Their extensive rhizome structure also plays a vital role in sediment accretion and stabilization (Barbier *et al.*, 2014). Their quick response to many biotic and abiotic changes and their ability to improve water quality by filtering, cycling, and storing nutrients and pollutants make them a bioindicator of ecosystem health in coastal waters (Orth *et al.*, 2006; Orfanidis *et al.*, 2010; Sandoval-Gil *et al.*, 2016; Boutahar *et al.*, 2019, 2021). Seagrass meadows are also efficient carbon sinks (Mazarrasa *et al.*, 2018). Although they represent only 0.2% of the ocean's surface, it is estimated that they store 20% of oceanic blue carbon (Fourqurean *et al.*, 2012) with an economic value up to $13.7 billion per annum (Pendleton *et al.*, 2012). The large quantities of dissolved inorganic carbon taken up during photosynthesis can also buffer ocean acidification, thus contributing to the resilience of calcifying organisms, such as corals (Manzello *et al*., 2012) and shellfish (Wahl *et al.*, 2017), living within or adjacent to seagrasses.

Despite their extensive ecosystem services, it has been estimated that 30% of the known seagrass areal extent has disappeared since seagrass areas were initially recorded in 1879 (Waycott *et al*., 2009), and more than half of the meadows are predicted to be lost in the next 100 years (Pendleton *et al*., 2012). Human disturbances, such as coastal development, eutrophication, pollution, and physical destruction by dredging and trawling, play a key role in the loss of seagrasses (Ralph *et al*., 2006; Salinas *et al.*, 2020). In addition, climate change is a growing concern as rising sea levels and increasing ocean temperature may cause future seagrass losses (Fortes *et al.*, 2018).

Faced with this situation, and in order to guarantee the seagrass meadows' future sustainability, the setting up of monitoring surveys based on exhaustive characterisation of natural populations in response to local and global changes is becoming more crucial than ever before (York *et al.*, 2017; Unsworth *et al.*, 2018; De los Santos *et al.*, 2019).

In order to update all the management plans of the marine protected area Al Hoceima National Park (Mediterranean coast of Morocco), the mapping inventory of key marine habitats of conservation interest identified the presence of *Cymodocea nodosa* (Ucria) Ascherson meadows in the park. This seagrass species is common in the Mediterranean Sea, the North Atlantic coasts of Africa, the South Atlantic coast of Europe and around Madeira and the Canary Islands (Cunha & Duarte, 2007). Even though it is considered to be resilient to natural and anthropogenic stresses and shows a high environmental plasticity that allows it to colonise lagoons, bays, estuaries and open coastal waters (Canals & Ballesteros, 1997; Papathanasiou *et al*., 2015), it is classified as an endangered species under Annex II of the Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean of the Barcelona Convention (RAC/SPA-UNEP/MAP, 2013). Along the Mediterranean coast, the extension decline of *C. nodosa* meadows was recorded in many areas: Mar Menor lagoon, Spain (Perez-Ruzafa *et al.*, 2012), Urbinu lagoon, France (Fernandez *et al.*, 2006), Gulf of Tigullio, Italy (Barsanti *et al*., 2007), and Ghar El Melh lagoon, Tunisia (Shili *et al*., 2002).

Given the underrepresented knowledge of seagrasses in West Africa (McKenzie *et al.*, 2020) and the marked regression of this species throughout its range distribution, this study aimed to closely monitor the development of *C. nodosa* meadows in Al Hoceima National Park and provide the first baseline quantitative and qualitative database to fill the knowledge gaps on these ecosystems and promote their conservation. We investigated *C. nodosa* structural development using the reconstruction techniques, an indirect measurement of seagrass growth, which has been proposed to overcome the lack of long-term data. This dating method allows researchers to quickly evaluate leaf production and rhizome growth rates and their changes over time and, from this, derive shoot demography and population dynamics (Duarte *et al.*, 1994). The balance between shoot recruitment and mortality rates is thus used to forecast the meadow's expansion, steady-state or regression (Duarte & Sand-Jensen, 1990). In addition, the angiosperm biotic index CymoSkew^{m}, based on *C. nodosa* leaf length (Orfanidis *et al.*, 2020), was applied for biomonitoring the quality status of Al Hoceima coastal waters.

The management status of the park is also discussed and measures to promote the conservation of seagrass habitats are proposed.

Materials and Methods

Study site

Al Hoceima National Park (AHNP) is located on the Mediterranean coast of Morocco (35°10′N, 4°07′O) at approximately 150 km east of the Strait of Gibraltar, west of the city of Al Hoceima. The surface area of its land part is 28860 ha, while the marine part covers 19600 ha covering a coastline of 40 km (RAC/SPA-UNEP/MAP, 2020). AHNP is the unique official marine protected area of the Mediterranean Moroccan coast under the Specially Protected Areas and Biological Diversity Protocol of the Barcelona Convention. Moreover, it was classified in 2009 by the United Nations as a specially protected area of Mediterranean importance (SPAMI) (RAC/SPA-UNEP/MAP, 2009).

The dominant features of the park coast are its very high rocky cliffs reaching more than 500 m that fall into the sea with steep slopes, constituting a grandiose natural landscape. These cliffs are locally interrupted by creeks of sand and gravel that often correspond to the wadis' outlets. Some of these streams only function after abundant and continuous rainfall, resulting in the irregularity of the water regimes of these wadis (IUCN, 2012). The beaches are relatively rare, small in extent, and appear only in the protected zones of the bays. The marine area of the park, located in the Alboran Sea, is under the influence of the Western Anticyclonic Gyre (WAG) formed by the opposite circulatory movements of the Mediterranean Sea and the Atlantic Ocean waters entering through the Strait of Gibraltar (Garcia-Lafuente *et al*., 2017). The exchange of these two water masses with different properties results in high primary productivity in this region (Abdellaoui *et al*., 2017), and remarkably enhances the diversity of benthic habitats and species communities listed in many international conventions and protection agreements (RAC/SPA-UNEP/MAP, 2020).

Sample collection and laboratory analyses

During the marine survey of July 2019, two continuous *Cymodocea nodosa* meadows were identified near Boumehdi Beach (35.14 N-04.00 W) and east of the Cala Iris islet (35.9 N-04.20 W) with estimated areas of 184254 m² and 46319.63 m^{2}, respectively (Fig. 1).

At -15 m and -9 m depth from Boumehdi and Cala

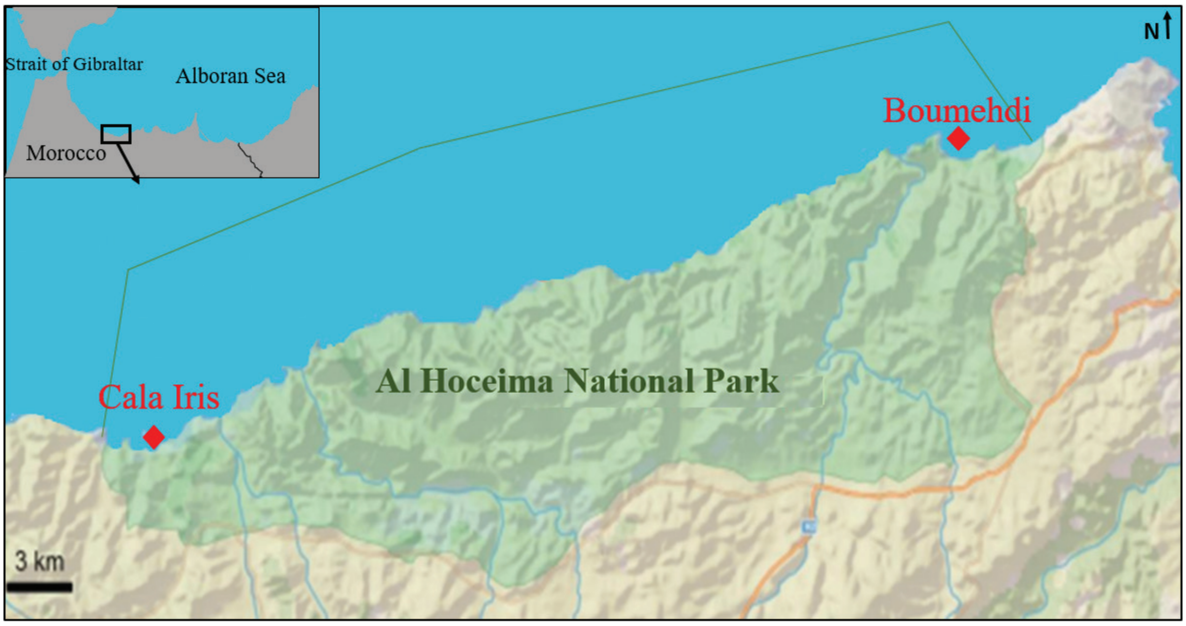

Fig. 1: Map showing the two studied *Cymodocea nodosa* meadows in Boumehdi beach and Cala Iris bay, in the marine part of the Al Hoceima National Park, along the Mediterranean coast of Morocco.

Iris, respectively, referring to the dense center part of each meadow, five randomly thrown quadrats of 1 x 1 m^2, separated by 1 m from each other, were photographed to estimate the meadows cover. Images were digitally analysed using Adobe Photoshop 6.0© (Adobe). Then, a digital network of 64 squares was superimposed onto the photographs and adjusted using the distortion tool RAC/SPA-UNEP/MAP,, 2015).

Shoot density was estimated *in situ* as the number of shoots within 0.04 m^2 quadrats (10 replicates at each station). For biometry and biomass determination, five cylindrical cores of 0.15 m in diameter and 0.12 m long, separated by 1 m from each other, were extracted at each station. Around 200 shoots connected to their horizontal rhizomes were handily harvested for plant growth history and population dynamics. All samples were then rinsed free of sediment carefully to avoid shoot breakage, transferred into plastic bags, and frozen until further analysis.

In the laboratory, core material was sorted into aboveground (leaves) and belowground (rhizomes and roots) *C. nodosa* tissues. Foliar epiphytes were scraped off using a glass slide (Dauby & Poulicek, 1995). Twenty shoots were randomly selected from the five aboveground core samples and dissected for number of total leaves per shoot and for number, length and width of leaf by category (differentiated (with sheath), and not differentiated (without sheath)). Leaf area index (LAI) was determined by multiplying mean surface area of shoots (only one face) by meadow shoot density. Plant tissues and epiphytes were then oven dried until constant weight (minimum 48 h at 60 °C) to determine dry biomass per shoot (g_{DB} $shoot^{-1}$) and per meadow surface unit (g_{DB} m^{-2}).

Shoot age, i.e., the age of vertical rhizomes (144 at Boumehdi and 130 at Cala Iris), was estimated by counting the number of internodes and standing leaves and dividing the resultant number by the annual average number of leaves produced by the plant (Duarte *et al.*, 1994). This average was derived from the annual cycles in internode lengths and, in turn, allowed the calculation of the annual leaf plastochron interval (PI, the time elapsed between the formation of two consecutive leaves). Average leaf turnover was determined as the product of the inverse of the rate of production of new leaves and the average number of standing leaves per shoot. The annual vertical rhizome elongation rate was calculated as the slope of the linear regression between the length of vertical rhizomes and the shoot age. The production rate of horizontal rhizome internodes was estimated as the linear regression slope of the number of horizontal internodes between consecutive shoots against their age difference (Duarte *et al.*, 1994). The product of this slope and the average length of the rhizome internodes estimated the average annual horizontal rhizome elongation rate (Duarte *et al.,* 1994). The population dynamics of the two meadows were characterised by the shoot mortality and recruitment through clonal growth as described by Duarte *et al.* (1994). The annual gross shoot recruitment rate (R_{gross}) was calculated as the difference between the total number of shoots and the number of shoots older than one year. Shoot mortality rate (M), assumed constant over shoot age and years, was obtained from the exponential decay regression of the shoot age frequency distribution, where it is:

$N_t = N_0 e^{-M t}$,

where N_t is the number of shoots in age class t, N_0 is the number of shoots recruited into the population, and M is the mortality rate. The net rate of shoot population growth ($R_{net} = R_{gross} - M$) was then used to forecast the future development of the meadows if growth conditions were maintained. Net recruitment positive values indicate expanding seagrass meadows; negative values indicate the regression of meadows.

Concurrently, from all sampled shoots, length of 220 leaves was measured to classify the meadows ecosystems' ecological status according to CymoSkew^m index values using its web computational tool (http://index.cymoskew.gr/).

Statistical analyses

Differences in measured parameters among both stations were analysed using the Student's T test. Raw or log-transformed data were tested for normality and homogeneity of variance to meet the assumptions for parametric statistics. Statistical analyses were performed using the Statistica software (version 7.1; StatSoft and TIBCO Software Inc., Palo Alto, CA, USA). Statistical analyses were not performed for the horizontal rhizome elongation or mortality and recruitment rates, because just one value was obtained for each population.

Results

In Boumehdi and Cala Iris, *C. nodosa* grows in dense monospecific beds covering 18.4 ha and 4.63 ha, respectively (Fig. 1).

Descriptive morphological, structural, and growth parameters of both meadows are reported in Tables 1 and 2. Leaves of Cala Iris meadow were remarkably more invaded by epiphytes that exceeded leaf biomass per shoot and per m^2. Shoot density, cover, and leaf biomass were higher in Boumehdi without statistical differences ($p > 0.05$). Despite the higher leaf length of Cala Iris meadow ($p < 0.05$), the leaf surface exposed to light (m^2 of leaves per m^2 of sediment) was lower, as a direct consequence of the lower density value. Leaf formation rate was lower at Cala Iris meadow with 11 leaves shoot^{-1} year^{-1} compared to 15 leaves shoot^{-1} year^{-1} in Boumehdi with a turnover rate estimated to be 103 days and 116 days, respectively.

The relative contribution of leaf canopy to the total biomass, accounting only for 35% in Boumehdi, testified to the well-developed layer of rhizomes and roots in this meadow. The average length of vertical rhizomes was 2.19 ± 1.80 cm, where the longest length was 14.5 cm. Their growth rate was estimated to be 1.51 cm year^{-1}. Vertical rhizomes of Cala Iris were much shorter and ranged from 0.10 cm to 5.40 cm with an annual elongation rate of 0.94 cm year^{-1}. Horizontal rhizome growth was very low in both stations, as reflected by the slow elongation rates (4.8 - 7.2 cm year^{-1}).

Examination of age structure showed that since the second-year class, the survival of shoots decreased, reflecting the high mortality rates especially in Cala Iris (Fig. 2). The recruitment rates showed a similar pattern (0.37 year^{-1}, in both meadows). The calculated net shoot growth of both meadows was negative, with a slightly higher value in Boumehdi.

CymoSkew^{m} index values for each studied meadow are presented in Table 1. Value at Boumehdi was just slightly higher than at Cala Iris (1.183 vs. 1.161). According to the index ecological status classes boundaries, meadows of both stations were classified as slightly polluted.

Discussion

Epiphyte load is a result of the balance between light penetration, nutrient supply, and grazing pressure (Castejon-Silvo & Terrados, 2012). Lower depth development of Cala Iris meadow (9 m vs. 15 m at Boumehdi) in combination with human-induced stressors (fishing, tourism, and nearby port activities) may have promoted their epiphytic overgrowth. In addition, it is known that grazers have an important role in controlling the epiphytic load on the seagrass canopy. Since no data on grazers were collected in this study, it cannot be excluded that differences in grazing pressure between the two sites may have contributed to the observed differences of epiphytic biomass between Cala Iris and Boumehdi.

High epiphyte concentrations can severely reduce the availability of light to seagrass leaves (Tuya *et al.*, 2013; Tsioli *et al.*, 2021), one of the most limiting factors for their vegetative development (Leoni *et al.*, 2008). The low number and longer leaves in Cala Iris meadow is probably a co-adaptation mechanism to maximize light capture and avoid self-shading within the canopy (Ralph *et al.*, 2007). This growth trend has also been observed in *C. nodosa* meadows from impacted sites with low light levels (Marin-Guirao *et al.*, 2005; Orfanidis *et al.*, 2010; Papathanasiou *et al.*, 2015). Moreover, the lower leaf formation rate associated to the higher turnover rate at Cala Iris meadow reflects the fact that leaves disappeared

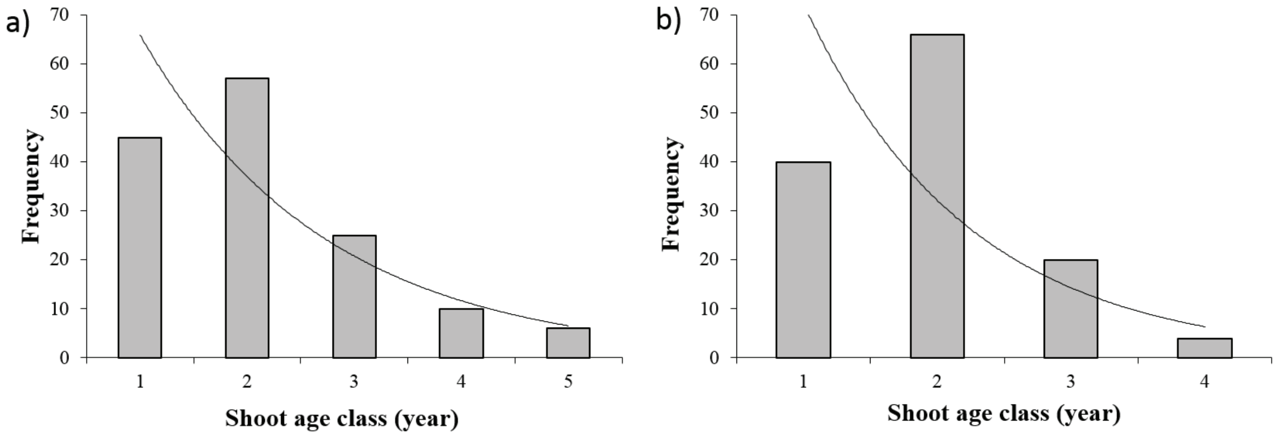

Fig. 2: Age-frequency distribution of *Cymodocea nodosa* populations at Boumehdi (a) and Cala Iris (b) meadows of Al Hoceima National Park. Solid lines show the fitted exponential decay line used to obtain the mortality rate (M).

Table 1. *Cymodocea nodosa* shoot density, cover, biomass, leaf morphometric measurements, rhizome growth, leaf formation, and CymoSkew^{m} index at Boumehdi and Cala Iris of Al Hoceima National Park. Significant differences between meadows ($p < 0.05$) are presented in bold. The standard deviation of the mean (within parentheses) is provided when possible. DB: dry biomass.

Stations	**Boumehdi**	**Cala Iris**
Parameters		
Shoot density (shoots m^{-2}, n = 10)	493 (118)	450 (184)
Shoot cover (%, n = 5)	98 (4)	94 (5.1)
Total number of leaves per shoot (leaves shoot^{-1}, n = 20)	**4.7** (1.2)	**3.1** (0.7)
Number of differentiated leaves per shoot (leaves shoot^{-1}, n = 20)	**2.7** (0.7)	**1.8** (0.4)
Number of undifferentiated leaves per shoot (leaves shoot^{-1}, n = 20)	**2.0** (0.8)	**1.4** (0.5)
Leaf length (mm, n = 20)	**148** (82)	**185** (89)
Leaf width (mm, n = 20)	2.7 (0.8)	2.5 (0.6)
Leaf Area Index (m^{2} m^{-2}, n = 20)	**0.97** (0.5)	**0.7** (0.2)
Shoot biomass (g$_{DB}$, n = 20)	0.05 (0.02)	0.04 (0.02)
Epiphyte load (g$_{DB}$ g$^{-1}_{DB}$ shoot^{-1}, n = 20)	**0.02** (0.01)	**0.09** (0.07)
Leaf biomass (g$_{DB}$ m^{-2}, n = 5)	41 (5)	30 (12)
Epiphyte biomass (g$_{DB}$ m^{-2}, n = 5)	**16** (3)	**51** (28)
Rhizome biomass (g$_{DB}$ m^{-2}, n = 5)	**50** (12)	**17** (2)
Root biomass (g$_{DB}$ m^{-2}, n = 5)	**26** (8.2)	**8.4** (3.6)
Belowground biomass (g$_{DB}$ m^{-2}, n = 5)	**76** (15)	**25** (4)
Total seagrass biomass (g$_{DB}$ m^{-2}, n = 5)	**117** (10)	**55** (13)
Vertical rhizome elongation (cm year^{-1})	1.51	0.94
Horizontal rhizome elongation (cm year^{-1})	7.2	4.8
Leaf production (leaves shoot^{-1} year^{-1})	15	11
Leaf appearance (leaves shoot^{-1} day^{-1})	0.04	0.03
Leaf turnover time (day)	116	106
CymoSkew^{m}	1.183	1.161

Table 2. Shoot demography and population dynamics of *Cymodocea nodosa* at Boumehdi and Cala Iris of Al Hoceima National Park. The standard deviation of the mean (within parentheses) is presented for the shoot age. The number of shoots used for the analysis were 144 and 130, respectively.

Stations	**Boumehdi**	**Cala Iris**
Parameters		
Shoot mean age (year)	1.7 (1.1)	1.3 (0.7)
Maximum shoot age (year)	9.50	3.40
Gross shoot recruitment rate (year^{-1})	0.37	0.37
Shoot mortality rate (year^{-1})	0.58	0.81
Net shoot population growth (year^{-1})	-0.20	-0.44
State of the meadow	regression	regression

faster than they appeared, indicating weaker growth conditions. It is well documented that continuous light deprivation caused by excessive epiphyte overgrowth can lead to significant declines in leaf production, shoot density and standing biomass (Leoni *et al*., 2008; Bryars *et al*., 2011; Kelaher *et al*., 2013). Our findings are consistent with previous research that reported a high *C. nodosa* leaf turnover rate in localities under higher anthropogenic disturbance (Perez *et al*., 1994).

Vertical and horizontal rhizome growth values of *C. nodosa* at Al Hoceima National Park are smaller when compared to meadows from other localities at smaller depths (Table 3). In contrast to our results, Terrados *et al.* (2006) examined the vegetative development of *C. nodosa* along depth gradients and found that horizontal rhizome elongation of meadows growing at depths (8 - 11 m), comparable to those of Al Hoceima (9 - 15 m), were three times greater than in shallow waters.

High *C. nodosa* shoot mortality recorded in Cala Iris agreed well with findings in stands with a greater quantity of leaf-biofilm in Mar Menor lagoon (Spain, Marin-Guirao *et al*., 2005). However, mortality rate values estimated in Al Hoceima were smaller than elsewhere (0.58 – 0.81 year^{-1} compared to 0.02 - 12.3 year^{-1}, Table

Table 3: Literature data on annual range of *Cymodocea nodosa* density, morphological, biomass, growth rates, and population dynamics.

Parameter	Value	Location	Reference
Shoot density (shoot m^{-2})	849-1594	Alfacs Bay, Spain	Perez *et al.*, 1994
	597-1140	Canary Island, Alicante and Mallorca, Spain	Manez-Crespo, 2020
	573-822	Ria Formosa, Portugal	Cabaço *et al.*, 2010
	633-704	Monastir Bay, Tunisia	Sghaier *et al.*, 2017
	204-814	Gabes Gulf, Tunisa	Zakhama-Sraieb *et al.*, 2010
	978-1658	Grado Lagoon, Italy	Guidetti, 2002
	757-1520	Urbinu Lagoon, Corsica	Agostini *et al.*, 2003
	364-1936	N. Aegean Sea, Greece	Papathanasiou and Orfanidis, 2018
Leaf number (leaves shoot^{-1})	3-4	Alfacs Bay, Spain	Perez *et al.*, 1994
	3.1-3.3	Ria Formosa, Portugal	Cabaço *et al.*, 2010
	4-5	Monastir Bay, Tunisia	Sghaier *et al.*, 2017
	3-5	Gabes Gulf, Tunisa	Zakhama-Sraieb *et al.*, 2010
	1-4	Venice Lagoon, Italy	Rismondo *et al.*, 1997
	3.0-5.0	N. Aegean Sea, Greece	Papathanasiou et. al., 2016
	3.0-5.0	N. Aegean Sea, Greece	Papathanasiou and Orfanidis, 2018
Leaf length (cm)	20-43	Alfacs Bay, Spain	Perez *et al.*, 1994
	24-38	Ria Formosa, Portugal	Cabaço *et al.*, 2010
	5-20	Monastir Bay, Tunisia	Sghaier *et al.*, 2017
	11-44	Gabes Gul, Tunisa	Zakhama-Sraieb *et al.*, 2010
	12-48	Venice Lagoon, Italy	Rismondo *et al.*, 1997
	8.9- 44.08	N. Aegean Sea, Greece	Papathanasiou and Orfanidis, 2018
	3.0- 79.4	N. Aegean Sea, Greece	Papathanasiou et. al., 2016
	11.3- 36.7	N. Aegean Sea, Greece	Orfanidis *et al.*, 2010
Aboveground Biomass (g_{DB} m^{-2})	20-130	Alfacs Bay, Spain	Olesen *et al.*, 2002
	62-104	Ghar El Melh lagoon, Tunisia	Sghaier *et al.*, 2011; 2012
	44-810	Venice Lagoon, Italy	Rismondo *et al.*, 1997
	30-160	Grado, Italy	Guidetti, 2002
	48-98	Vulcano, Adamas and Peleochori seeps, Italy and Greece	Mishra *et al.*, 2021
	200-400	Urbinu Lagoon, Corsica	Agostini *et al.*, 2003
Bellowground Biomass (g_{DB} m^{-2})	123-192	Blanes Bay, Spain	Marba & Duarte, 2001
	186-116	Sant Pol beach, Spain	Terrados *et al.*, 2006
	10-539	Canary Island, Alicante and Mallorca, Spain	Manez-Crespo, 2020
	188-539	Ghar El Melh lagoon, Tunisia	Sghaier *et al.*, 2011; 2012

Continued

Table 3 continued

Parameter	**Value**	**Location**	**Reference**
	108-358	Vulcano, Adamas and Peleochori seeps, Italy and Greece	Mishra *et al.*, 2021
	300-650	Urbinu Lagoon, Corsica	Agostini *et al.*,2003
Plastochrone interval (days)	28-45	Alfacs Bay, Spain	Olesen *et al.*, 2002
	38-40	Sant Pol beach, Spain	Terrados *et al.*, 2006
	12-43	Ghar El Melh lagoon, Tunisia	Sghaier *et al.*, 2011; 2012
	23	Ischia Island, Italy	Cancemi *et al.*, 2002
Leaf production (leaves shoot^{-1} year^{-1})	13-20	Alfacs Bay, Spain	Perez *et al.*, 1994
	16	Ghar El Melh lagoon, Tunisia	Sghaier *et al.*, 2011; 2012
	12	Monastir Bay, Tunisia	Sghaier *et al.*, 2017
	16	Ischia Island, Italy	Cancemi *et al.*, 2002
Vertical rhizome elongation (cm year^{-1})	1.50-3.19	Ria Formosa, Portugal	Cunha and Duarte, 2005
	2.44-5.79	Ria Formosa, Portugal	Cabaço *et al.*, 2010
	0.68-1.85	Vulcano, Adamas and Peleochori seeps, Italy and Greece	Mishra *et al.*, 2021
Horizontal rhizome elongation (cm year^{-1})	15-35	Alfacs Bay, Spain	Olesen *et al.*, 2002
	14-31	Ria Formosa, Portugal	Cunha and Duarte, 2005
	7-18	Ghar El Melh lagoon, Tunisia	Sghaier *et al.*, 2011; 2012
	4.6-25	Monastir Bay, Tunisia	Sghaier *et al.*, 2017
	3.7-22	Venice Lagoon, Italy	Rismondo *et al.*, 1997
Mean age (year)	0.80-0.94	Alfacs Bay, Spain	Perez *et al.*, 1994
	2.5-3.2	Sant Pol beach, Spain	Terrados *et al.*, 2006
	0.43-0.9	Ria Formosa, Portugal	Cabaço *et al.*, 2010
Recruitment (year^{-1})	0.6-2.3	Alfacs Bay, Spain	Olesen *et al.*, 2002
	0.82-2.36	Ria Formosa, Portugal	Cabaço *et al.*, 2010
	0.36-0.77	Vulcano, Adamas and Peleochori seeps, Italy and Greece	Mishra *et al.*, 2021
Mortality (year^{-1})	0.87-1.62	Alfacs Bay, Spain	Perez *et al.*, 1994
	0.7-1.2	Alfacs Bay, Spain	Olesen *et al.*, 2002
	3.15-12.4	Ria Formosa, Portugal	Cabaço *et al.*, 2010
	0.22-0.99	Vulcano, Adamas and Peleochori seeps, Italy and Greece	Mishra *et al.*, 2021
Net growth rate (year^{-1})	(-0.1)-1.1	Alfacs Bay, Spain	Olesen *et al.*, 2002
	0.4-0.68	Mar Menor Lagoon, Spain	Marın-Guirao *et al.*, 2005
	(-2.35)-0.29	Ria Formosa, Portugal	Cunha and Duarte, 2005

3), while the recruitment rates showed a severely slower pattern. The reduced recruitment of the shoots was consistent with the low elongation of rhizomes as described previously by several researchers (Cunha & Duarte, 2005; Cabaço *et al*., 2010; Tupan & Uneputty, 2017). Rhizome growth regulates the seagrass shoot development, which is the basis of meadow resilience and dispersion (Duarte & Sand-Jensen, 1990; Marba & Duarte, 1994). The negative net growth calculated with reference to the recruitment and mortality rates concluded that *C. nodosa* populations at Al Hoceima were declining, most probably due to epiphytic overload and limited rhizome growth.

High epiphytic production may be related to the nutrient-rich Atlantic Jet via the Strait of Gibraltar that accumulates nutrients in the Alboran Sea and thus induces a permanent fertilisation of this area (Lorente *et al*., 2019). Freshwater runoff conducted from watersheds during the rainy months may also enhance nutrient enrichment. Concerning clonal growth regulation, previous studies underlined that it exhibits intraspecific variability in response to meadow genetic diversity (Manez-Crespo *et al*., 2020) and local scale environmental conditions (hydrodynamics, sediment dynamics, light, salinity, temperature, nutrients, and heavy metals, among others; Cunha & Duarte, 2005; Ambo-Rappe 2011; Sghaier *et al*., 2017; Tuya *et al*., 2019; Manez-Crespo *et al*., 2020). Azizi *et al.* (2020) recorded an average monthly salinity of the park water column oscillating between 36 and 40. The optimum *C. nodosa* meadows productivity is at around oceanic salinity (33 - 37), while extreme or suboptimal values negatively affect their photosynthesis, metabolism, and growth, determining their biomass, productivity and survival (Vermaat *et al*., 2000; Pagès *et al*., 2010; Fernandez-Torquemada & Sánchez-Lizaso, 2011; Sghaier *et al*., 2017; Tsioli *et al.*, 2019). Illegal fishing activities are another severe issue that can threaten the survival of Al Hoceima meadows. Many crossing scars by trawling gears have been observed in the park too close to the coast, although the law prohibits trawling activity at depths less than 80 m. This physical disturbance is one of the major causes of direct damage to seagrasses, injuring roots and rhizomes, reducing shoot density, and leading to fragmentation and permanent loss of habitat (Short & Wyllie-Echeverria, 1996; Ardizzone *et al*., 2000; Neckles *et al*., 2005). Fishing using highly toxic chemicals such as copper sulphate, which causes metal pollution, is also very common in the park. Overall, the risk of degradation of the AHNP seagrass meadows by fishing activities, evaluated by combining the sensitivity of this habitat and the amplitude of the pressures, was qualified as strong (RAC/SPA-UNEP/MAP, 2020). Other potential threats are related to port activities, urban discharges and/or terrigenous inputs after heavy rainfall that washes away all the dumped waste. The presence of eight invasive macroalgal species in the park is a supplementary biological stress to *C. nodosa* meadows (RAC/SPA-UNEP/MAP, 2020). Seven of the recorded species are included in the list of the worst invasive phytobenthos in the Mediterranean Sea (Streftaris & Zenetos, 2006; Verlaque *et al*., 2015) while *Rugulopteryx okamurae* (Dawson, E.Y., Hwang, I.K., Lee, W.J., Kim, H.S., 2009) is showing an overflowing expansion capacity to the detriment of native species around the Strait of Gibraltar (Garcia-Gomez *et al*., 2021).

Despite the lack of quantitative water quality parameters, seagrasses are recognised worldwide as an effective "biological quality element" for monitoring coastal ecosystem status thanks to the fast response of their metrics to environmental changes and anthropogenic stress (Duarte *et al*., 2017). This allowed the development of many angiosperm indices that provided satisfactory coastal water quality assessment (Krause-Jensen *et al.,* 2005; Foden & Brazier, 2007; Romero *et al.,* 2007; Gobert *et al.,* 2009; Oliva *et al*., 2012; Orlando-Bonaca *et al*., 2015; Karamfilov *et al.,* 2019). The application of CymoSkew^{m} index, the quantitative expression of *C. nodosa*'s leaf length (Orfanidis *et al*., 2020), to Al Hoceima meadows, reflected their degraded trend and assigned a slightly polluted ecological status.

The declining growth of *C. nodosa* meadows of Al Hoceima National Park and their sensitive ecological status due to the different encountered pressures are incompatible with the primary aim of SPAMIs, which is to maintain marine ecosystem functionality and health. The site suffers from a lack of coastal planning and an effective governance system. A management and development plan was proposed in 1993, resumed in 2002, and revised in 2011, but has never been implemented. There is also a major lack of legislation and regulations to ensure compliance with good environmental practices. According to Soriani *et al.* (2015), most of the problems and conflicts are related to non-coordination of sectoral actions, the rigidity of procedures, absence of prospective vision, and poor law enforcement. They also reported that the inclusion of research outcomes in coastal policy decisions rarely exceeds a statement of interest.

Conclusion

This study is the first report on the structure and dynamics of *Cymodocea nodosa* populations in Al Hoceima National Park. Our results provide support to the alarms being raised about the increasing worldwide decline of seagrass meadows. If these ecosystems have to be conserved, efforts to identify local scale factors driving the negative development pattern of these meadows and address the surrounding anthropogenic stressors must be built upon, and coordinated; this is what is urgently required to implement effective management actions to promote the recovery and restoration of the seagrass meadows.

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