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CO_2 sequestration in two mediterranean dune areas subjected to a different level of anthropogenic disturbance

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ABSTRACT

Coastal sand dunes are among the most threatened habitats, especially in the Mediterranean Basin, where the high levels of human pressure impair the presence of plant species, putting at risk the maintenance of the ecosystem services, such as CO_2 sequestration provided by these habitats. The aim of this study was to analyze how disturbance-induced changes in plant species abundance patterns account for variations in annual CO_2 sequestration flow (CS) of Mediterranean sand dune areas. Two sites characterized by a high (site HAD) and a lower (site LAD) anthropogenic disturbance level were selected. At both sites, plant species number, cover, height and CS based on net photosynthesis measurements were sampled. At the plant species level, our results highlighted that *Ammophila arenaria* and *Pancratium maritimum*, had a key role in CS. Moreover, the results revealed a patchy species assemblage in both sites. In particular, HAD was characterized by a higher extension of the anthropogenic aphytoic zone (64% of the total transect length) than LAD. In spite of the observed differences in plant species composition, there were not significant differences between HAD and LAD in structural and functional traits, such as plant height and net photosynthesis. As a consequence, HAD and LAD had a similar CS (443 and 421 Mg CO_2 ha^{-1} y^{-1}, respectively). From a monetary point of view, our estimates based on the social costs of carbon revealed that the flow of sequestered CO_2 valued on an average \$ 3181 ± 114 ha^{-1} $year^{-1}$ (mean value for the two sites). However, considering also the value of the CO_2 negative flow related to loss of vegetated area, the annual net benefit arising from CO_2 sequestration amounted to \$ 1641 and \$ 1772 for HAD and LAD, respectively. Overall, the results highlighted the importance to maximize the efforts to preserve dune habitats by applying an effective management policy, which could allow maintaining also a regulatory ecosystem service such as CO_2 sequestration.

1. Introduction

Climate change mitigation requires a complex strategy involving multiple actions mainly focused on reducing CO_2 emissions to atmosphere. CO_2 is the most abundant greenhouse gas, much of which is not attributable to natural causes but rather to fossil fuel combustion and deforestation worldwide (Nowak and Crane, 2002). The abundant CO_2 emissions in human times, in conjunction with other greenhouse gases, have contributed to global temperature increasing over the 20th century. Moreover, current projections highlight that the further increase of the atmospheric greenhouse gases, including CO_2, will lead to an increase of the earth mean temperature by 0.3–0.7 °C for the period 2016–2035 compared to 1986–2005 (IPCC, 2014). Among the actions addressed to mitigate climate change, carbon sequestration by plants is an offset mechanism for CO_2 emissions (Knoke and Weber, 2006). Vegetation represents a sink of CO_2 because plants fix carbon via photosynthesis and store the carbon excess as biomass (Gratani et al., 2013; Nowak et al., 2013). Specifically, carbon sequestration refers to the flow of annual carbon uptake through photosynthesis, while carbon stock refers to the carbon excess stored as biomass in plant organs (Beaumont et al., 2014). Different plant species sequester CO_2 at different rates depending on growth rate, physiology, structure and coverage (Nowak et al., 2002; Gratani et al., 2013). As CO_2 sequestration is driven by biological processes, any factor that alters the plant species functioning may constrain their sequestration ability and ultimately the whole ecosystem functioning (Freudenberger et al., 2012). Changing in the ecosystem structure and processes compromises the maintenance of the related ecosystem services, which are defined as the benefits that humans derives from ecosystem functions (Costanza et al., 1997; MEA, 2005). Specifically, the ecosystem structure and

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processes and links to the abiotic environment constitutes the capital stocks from which a flow of ecosystem services is delivered (Turner et al., 2014). Among the different ecosystem service categories such as supporting, regulating, provisioning and cultural (MEA, 2005), CO_2 sequestration is considered as a regulating service because it benefits human well-being by its positive effect on climate regulation (Fisher and Turner, 2008). Yet, Fisher and Turner (2008) and Turner et al. (2014) suggest referring to CO_2 sequestration as an intermediate service because it produces benefits indirectly, differently from a final service that acts directly. The distinction between intermediate and final ecosystem services is useful in order to assess the ecosystem services from an economic point of view. This approach allows avoiding the problem of the 'double counting', which can arise when different aspects of the same service are quantified separately on a monetary base leading to overestimate that service (Costanza et al., 1997). Thus, in economic terms only the final ecosystem service supplies the real benefit that therefore can be monetized. In view of this, CO_2 sequestration is an intermediate service because it contributes to increase carbon stock, which is the final ecosystem service. In fact, the carbon stock provides the benefit in terms of climate change mitigation by locking away over long-term the CO_2 removed from the atmosphere by sequestration process.

Improving our knowledge on the sequestration capacity of different plant species is thus a keystone in order to test the vegetation ability to limit the CO_2 atmospheric concentration. This is in agreement with the United Nations Framework Convention on Climate Change (UNFCCC, 1997), which promotes the implementation of national or regional carbon sink inventories. Accordingly, in the last decades much attention has been paid to forests, which are important components of the global carbon budget because of the large amount of biomass stored in these ecosystems (Pan et al., 2011; Fang et al., 2014). On the contrary, there has been a general lack of information for other habitats (Beaumont et al., 2014). Recently, it has been attempted to fill this gap and thus coastal habitats have gained attention. Many studies have highlighted the relevance of these habitats such as mangrove forests, saltmarsh, seagrass beds and sand dunes in local and global climate regulation by their carbon sequestration and/or storage capacity (e.g. Nellemen et al., 2009; Irving et al., 2011; Alongi, 2012; Beaumont et al., 2014; Hu et al., 2015). In particular, the carbon sequestered by coastal habitats through living organisms is referred to as 'blue carbon' (Luisetti et al., 2013; Sutton-Grier et al., 2014). Mangroves forests, saltmarsh and seagrass beds show a higher carbon sequestration rate per unit area than terrestrial forests (McLeod et al., 2011). The effectiveness of coastal habitats as a blue carbon sink is due to their capacity to store carbon not only in plant biomass but also in sediments, which represent large carbon stocks created over a timescale of thousand years (Duarte et al., 2005). This is due to a slow decomposition rate to which plant parts buried in the coastal habitat sediments are subjected because of the oxygen-poor conditions caused by tidal inundations (Sutton-Grier et al., 2014). Unlike mangroves forests, saltmarsh and seagrass beds, the carbon sequestration and storage of sand dunes is still few recognized and quantified (Beaumont et al., 2014; Drius et al., 2016). Mediterranean dune systems are usually characterized by a complex coast-to-inland environmental gradient, along which different plant communities coexist in a relatively small area (Acosta et al., 2007; Fenu et al., 2012, 2013; Ciccarelli, 2014). Sand dunes are stressful and intrinsically vulnerable habitats (Frondoni and Iberite, 2002; Fenu et al., 2013) being subject to hard environmental conditions such as drought, low nutrient content, high temperatures, salt spray and sand burial (Gratani et al., 2009). In addition, these habitats have been severely degraded over time because of coastline erosion (Feagin et al., 2005) and human activities, such as urbanization, development of infrastructures, tourism and excessive exploitation of natural resources (Muñoz Vallés et al., 2011; Fenu et al., 2012, 2013; Ciccarelli, 2014). According to Brown and McLachlan (2002), about 70% of the European coastal dunes disappeared due to increasing urbanization. Consequently, the high human pressure fragmented the extant dune vegetation leading to disappearance of the typical zonation of vegetation along the coast-inland gradient (Ciccarelli, 2014; Fenu et al., 2012; Pinna et al., 2015). In view of this, sand dune habitats deserve special management and conservation attention. Quantifying their CO_2 sequestration capacity may be an important focus to identify sustainable management practices for preserving their potential to sequester and store carbon as well as to trigger a positive feedback in mitigating the climate change impact (Sutton-Grier et al., 2014).

In this context, our objective was to estimate the CO_2 sequestration capacity of plant species from two sand dune habitats subject to different levels of human-related disturbance. In particular, we analyzed whether changes in plant species cover induced by disturbance accounted for changes in CO_2 sequestration. Finally, we evaluated the CO_2 sequestration service in economic terms. To achieve this goal, we also considered the carbon storage capacity. As CO_2 sequestration is an intermediate ecosystem service, its economic value is intrinsically captured when the carbon storage is captured (Morse-Jones et al., 2011). Assessing the monetary value of CO_2 sequestration, especially for threatened habitats, which suffer from area loss, could help to direct the decision-makers toward a more efficient management policy in order to protect or enhance the immediate and longer-term value of this ecosystem service.

2. Materials and methods

2.1. Study area

The study was carried out in two coastal dune areas at Ostia (41°40′N, 12°23′E) and Marina di Palidoro (41° 54′N; 12°08′E) along the Tyrrhenian coast near Rome. The study area in Ostia is exposed to a high level of direct anthropogenic disturbance due to the presence of a bathing establishment built right on the beach. This area suffers from an intense human trampling, especially in summer, because of the high number of tourists. Moreover, the beach is periodically subject to intensive mechanical cleaning operations. The beach appears flat up to approx. 100–105 m from the shoreline, followed by a mobile dune characterized by a moderate slope of about 6% extending for approx. 15 m, and by a fixed dune colonized by Mediterranean shrubs.

In Marina di Palidoro, the study area is characterized by a lower level of anthropogenic disturbance. In this site, there are no bathing establishments or mechanical cleaning operations and the trampling is less intensive due to the lower number of tourists. Accordingly, the drift line and the foredune extend from the shoreline toward the inland for about 40 m with a moderate slope (approx. 2%) followed by a mobile dune extending for approx. 20–25 m. Due to past agricultural activities, the shrubland vegetation of the fixed dunes was substituted by a grassland (Carboni et al., 2009).

Ostia is regarded here as the study site of high anthropogenic disturbance (HAD) and Marina di Palidoro as the site of lower anthropogenic disturbance (LAD). Both sites show a typical Mediterranean climate; the total annual rainfall at HAD is 589 mm, the mean minimum air temperature of the coldest months (January and February) is 4.8 °C and the mean maximum temperature of the warmest months (July and August) is 29.1 °C (data from the Meteorological Station of Pratica di Mare for the years 2000–2012). The total annual rainfall at LAD is 556 mm, the mean minimum air temperature of the coldest months (January and February) is 3.2 °C and the mean maximum air temperature of the warmest months (July and August) is 29.6 °C (data from the Meteorological Station of Fiumicino for the years 2000–2012).

2.2. Data collection

At each site, plant species were sampled in 10 evenly spaced square plots of 2 × 2 m along a transect perpendicular to shoreline, from the beach to the inland. The first plot was located in correspondence to the first plant species occurrences, while the last plot was located before the Mediterranean shrubland vegetation at HAD and a grassland at LAD. The number of plots was established on the base of the dune morphology, and based on field survey the number of 10 plots was enough to encompass the whole species variability within each site. Moreover, all sampled plants had the same ecological conditions. The taxonomic treatment followed Anzalone et al. (2010).

In each plot, the number of plant species (NS) and the plant height (H) and the percent ground cover (COV) of each plant species were recorded in spring-summer 2014. COV was visually estimated in the field. The total plant species cover of each plot (COV_p) was then calculated by summing the cover values of all plant species present in the plot. H was measured on six individuals per plant species. The mean plant height at each plot (H_p) was then calculated as the mean height of all plant species present in the plot weighted by their percent cover.

2.3. CO_2 sequestration

Annual CO_2 sequestration flow (CS; Mg CO_2 ha^{-1} y^{-1}) per plant species was calculated according to Gratani et al. (2013) by multiplying total plant area per unit of covered area at the sand level (PA; m^2m^{-2}) by the mean annual net photosynthetic rate (P_N; μmol CO_2 $m^{-2}s^{-1}$) and the total annual photosynthetic activity time (in hours). PA was calculated from Leaf Area Index (LAI) measured by the “LAI 2000 Plant Canopy Analyzer” (LICOR Inc., Lincoln, USA).

P_N was measured periodically during the study by an open infrared CO_2 gas analyzer (ADC LCA4, UK) equipped with a leaf chamber (PLC, Parkinson Leaf Chamber). Measurements were made *in situ* on cloud-free days (PAR > 1000 μmol $m^{-2}s^{-1}$) in the morning (from 9 a.m. to 12 a.m.) to ensure that near-maximum daily photosynthetic rates were measured. In each sampling occasion, fully expanded leaves were used. All measurements were carried out on six plants per species. The mean photosynthetic rate of each plot (P_{Np}) was then calculated as the average P_N of all plant species present in the plot weighted by their percent cover, while the total annual CO_2 sequestration flow per each plot (CS_p) was calculated by summing the CS values of all plant species recorded in the plot.

In addition to CO_2 sequestration, carbon storage in above-ground biomass (CST; Mg C ha^{-1}) was calculated. CST was obtained by multiplying above-ground biomass per 0.5 according to McPherson (1998). Then, carbon storage values were converted to CO_2 by multiplying by 3.67, namely the CO_2 molecular weight. Data for above-ground biomass was taken by Gratani et al. (1983) and by Gratani and Crescente (2000) who carried out measurements of plant biomass in Mediterranean coastal dune areas comparable to HAD and LAD. Since carbon storage was calculated from literature data, it was considered the same for HAD and LAD.

2.4. Economic valuation of net CO_2 sequestration service

Net CO_2 sequestration capacity (NCS; Mg CO_2) of HAD and LAD during one year was valued. NCS was estimated according to Beaumont et al. (2014) through three steps.

First, total CO_2 sequestration of the study area (S_{CO2}) was calculated as:

$$S_{CO_2} = (CS_{Tot} * A)$$

where CS_{Tot} was the sum of all CS_p and A was the surface (ha) of the study area. A was 1.40 ha (114 m × 123 m) in HAD and 0.81 ha (61 m × 133 m) in LAD.

Second, taking in account the area characterized by the absence of the psammophylous vegetation (i.e. aphytoic anthropogenic zone), a value of the dis-service in terms of CO_2 loss was estimated. In fact, the loss of vegetated area, which reflects the anthropogenic impact, can be considered as a di-service because it is potentially a source of CO_2. Thus, total CO_2 release (R_{CO2}) accounts from both sequestration and storage and it was calculated as:

$$R_{CO_2} = (CS_{Tot} * A_{aaz}) + ((CST * A_{aaz}) * \text{number of years of sequestration})$$

where A_{aaz} was the surface (ha) of the aphytoic anthropogenic zone (0.90 and 0.24 ha in HAD and LAD, respectively). In the calculation of A_{aaz}, the length of the aphytoic anthropogenic zone has been calculated as difference between the total length of the unvegetated area and the length of the aphytoic natural zone (4 m both in HAD and LAD), which is the natural occurring zone where vegetation is missing because of the harsh environmental conditions. Moreover, concerning the years of sequestration calculations on the basis of the current year.

Third, NCS was calculated as:

$$NCS = S_{CO_2} - R_{CO_2}$$

The economic valuation was estimated by using a damage cost avoided - based method (Pearce, 2003). This method assesses the social cost of carbon (SCC), namely the marginal monetary value of the avoided carbon releases to the atmosphere due to the sequestration and storage (Luisetti et al., 2013). Specifically, we used a SCC value of $ 7.36 for each Mg of CO_2 avoided. This value was taken from Peper et al. (2007) and it was based on the average value found in Pearce (2003).

2.5. Data analysis

One way ANOVA was used to test for differences in NS, COV_p, H_p, P_{Np} and CS_p between HAD and LAD. Since the distance from the shoreline of the plots was the only factor that changed between the two sites because of the different extension of the vegetated area an analysis of Covariance (ANCOVA) was performed to account for the effects of the distance from the shoreline on the analyzed variables. Accordingly, ANCOVA was performed using HAD and LAD as factors and the distance from the shoreline as covariate. Once the homogeneity of the slopes was tested, the model analyzed also the differences in the intercepts.

A multiple regression analysis was carried out to evaluate the relationship among CS_p (dependent variable) and NS, COV_p, H_p and P_{Np} (independent variables). Kolmogorov–Smirnov and Levene tests were used to verify the assumptions of normality and homogeneity of variances, respectively.

3. Results

3.1. Plant species sampling and structural measurements

The first plant species occurred at 77 m from the shoreline in HAD. Starting from this point, the vegetated area extended for 37 m (total length of the transect from the shoreline to the fixed dunes = 114 m). Along the transect 14 plant species were recorded: *Elymus farctus* and *Echinophora. spinosa* were the most frequent species (from 70 to 90%) of analyzed plots, respectively, followed by *Anthemis maritima*, *Silene canescens* and *Cyperus capitatus* which were recorded in 60% of the plots (Table 1). Conversely, *Ammophila arenaria*, *Crucianella maritima*, *Vulpia fasciculata* and *Cakile maritima* subsp. *maritima* were the less frequent plant species. The lowest NS was recorded at 85 m from the shoreline (Fig. 1A) and the highest number in the plots further from the shoreline. COV_p ranged from 15% to 90% (Fig. 1B). *Ammophila arenaria* and *Crucianella maritima* showed the highest COV (mean ± standard error = 50 ± 0% and 30 ± 0%, respectively; Table 1), while *Vulpia fasciculata* and *Cakile maritima* subsp. *maritima* showed the lowest values (1 ± 0%). H_p ranged from 14 ± 2 cm to 44 ± 4 cm with a mean value of 24 ± 3 cm (Fig. 1C). *Ammophila arenaria* was the tallest species (117 ± 4 cm) and *Ononis variegata* the shortest (7.7 ± 1.0 cm; Table 1).

LAD was characterized by 16 plant species with NS ranging from 2 to 8 (Fig. 2A). The first plant species was at 22 m from the shoreline. Starting from this point, vegetated area extended for 39 m (total length of the transect = 61 m). The most frequent species were *Elymus farctus* and *Cutandia maritima*, found in 90% and 70% of the plots, respectively (Table 1). *Ammophila arenaria*, *Sporobolus virginicus* and *Vulpia fasciculata* were the species with the lowest frequency and were found in only one plot. COV_p ranged from 7% to 50% (Fig. 2B). The species with the highest COV were *Crucianella maritima* (20.5 ± 19.5%), *Pancratium arenarium* (12.5 ± 1.5%) and *Ammophila arenaria* (18.0 ± 0.0%) (Table 1). *Medicago marina*, *Vulpia fasciculata* and *Chamaesyce peplis* were the species with the lowest COV (1.0 ± 0.0%, 1.0 ± 0.0% and 0.8 ± 0.3%, respectively).

H_p ranged from 9.9 ± 1.7 cm to 30.8 ± 16.3 cm with a mean value of 18 ± 2 cm (Fig. 2C). Among the plant species present in LAD, *Ammophila arenaria* was the tallest (108 ± 3 cm) and *Chamaesyce peplis* the shortest (3.5 ± 1.8 cm) (Table 1).

ANOVA analysis revealed no significant differences in NS, COV_p, and H_p between HAD and LAD. However, at both sites, NS and COV_p significantly co-varied with the distance from the shoreline increasing linearly with distance away from the shoreline. In particular, the coefficients of determination from the ANCOVA models were $R^2 = 0.33$ and 0.59 for NS and COV_p, respectively (Table 2). Nevertheless, the effect of the covariate was the same at both sites as highlighted by the homogeneity of slopes allowing the comparison between the two sites independently of the distance from the shoreline. Significant differences were found out at intercept level both for NS and COV_p. With regard H_p, ANCOVA showed a not significant relationship with the distance from the shoreline ($R^2 = 0.15$, $p = 0.268$).

3.2. CO_2 sequestration and storage

P_{Np} did not significant differ between HAD and LAD (12.1 ± 0.3 and 12.6 ± 0.6 μmol CO_2 m^{-2} s^{-1}, respectively; Figs. 1D and 2D). CS_p at HAD was slightly higher than at LAD (Table 3). However, mean value did not show significant differences (44 ± 30 Mg CO_2 ha^{-1} y^{-1} and 42 ± 12 Mg CO_2 ha^{-1} y^{-1} at HAD and LAD, respectively). At both sites the highest contribution to CS was provided by *Ammophila arenaria* (309.4 ± 8.6 Mg CO_2 ha^{-1} y^{-1} at HAD and 111.4 ± 4.5 Mg CO_2 ha^{-1} y^{-1} at LAD, respectively; Table 4). Moreover, at LAD a high CS was estimated for *Pancratium arenarium* (53.6 ± 6.2 Mg CO_2 ha^{-1} y^{-1}). ANCOVA showed that distance from the shoreline was not a significant covariate for CS_p ($R^2 = 0.16$, $p = 0.2149$) (Table 2). Multiple regression analysis showed that CS_p depended on a linear combination of H_p, COV_p, NS and P_{Np} according to the following equation: $CS_p = -29.067 + (2.39NS) + (0.571H_p)$ - $(0.0000282\ COV_p) + (0.759\ P_{Np})$ ($R^2 = 0.50$; $F_{4,15} = 3.78$; $p \le 0.05$). H_p was the most significant variable ($p = 0.016$) accounting for the ability to predict CS_p.

The estimated CST was 0.554 Mg CO_2 ha^{-1} in both the sites (Table 5). However, NCS was higher in LAD (241 Mg CO_2) than in HAD (223 Mg CO_2).

3.3. Economic valuation of CO_2 sequestration and storage

From a monetary point of view the value for the flow of CO_2 sequestration service was \$ 3261 ha^{-1} y^{-1} and \$ 3100 ha^{-1} y^{-1} at HAD and LAD, respectively (Table 5) while the value of the CO_2 storage was \$ 4 ha^{-1} (Table 5). However, NCS was valued at \$ 1641 and \$ 1772 for HAD and LAD, respectively.

Table 1
Percentage of plant species presence, species-specific cover (COV) and plant height (H) at HAD (high anthropogenic disturbance) and LAD (lower anthropogenic disturbance) sites.
(np = not present).

Species	% of plant species presence	HAD		LAD		
		COV (%)	H (cm)	% of plant species presence	COV (%)	H (cm)
Ammophila arenaria	10	50.0 ± 0.0	117.0 ± 4.0	10	18.0 ± 0.0	111.3 ± 4.0
Anthemis maritima	60	9.3 ± 0.1	20.3 ± 2.1	50	9.0 ± 4	15.2 ± 1.1
Cakile maritima subsp.*maritima*	10	1.0 ± 0.0	12.5 ± 0.5	20	2.0 ± 1.0	10.8 ± 1.0
Calystegia soldanella	np	–	–	30	2.7 ± 1.2	4.3 ± 0.9
Chamaesyce peplis	np	–	–	20	0.8 ± 0.3	3.5 ± 1.8
Crucianella maritima	10	30.0 ± 0.0	28.3 ± 1.9	20	20.5 ± 19.5	25.1 ± 4.4
Cutandia maritima	np	–	–	70	6.1 ± 1.7	12.5 ± 2.4
Cyperus capitatus	60	4.5 ± 1.1	11.9 ± 0.7	np	–	–
Echinophora spinosa	70	7.9 ± 3.8	19.9 ± 3.0	20	1.5 ± 0.5	9.8 ± 1.8
Elymus farctus	90	10.0 ± 4.0	39.9 ± 2.4	90	4.4 ± 1.3	23.0 ± 1.8
Eryngium maritimum	30	2.0 ± 0.0	17.1 ± 7.3	40	3.3 ± 1.1	21.1 ± 4.0
Medicago marina	40	26.8 ± 12.2	10.6 ± 0.9	20	1.0 ± 0.0	7.1 ± 0.4
Ononis variegata	30	2.7 ± 0.7	7.7 ± 1.0	30	12.0 ± 9.1	7.4 ± 1.2
Pancratium maritimum	np	–	–	40	12.5 ± 1.5	34.9 ± 2.0
Phleum arenarium subsp.*caesium*	40	11.5 ± 6.5	23.7 ± 2.1	np	–	–
Salsola kali	np	–	–	30	1.7 ± 0.7	5.6 ± 1.7
Silene canescens	60	3.8 ± 1.5	17.3 ± 1.5	np	–	–
Sporobolus virginicus	30	1.7 ± 0.7	15.0 ± 2.3	10	5.0 ± 0.0	8.8 ± 1.5
Vulpia fasciculata	10	1.0 ± 0.0	20.5 ± 1.7	10	1.0 ± 0.0	15.5 ± 0.3

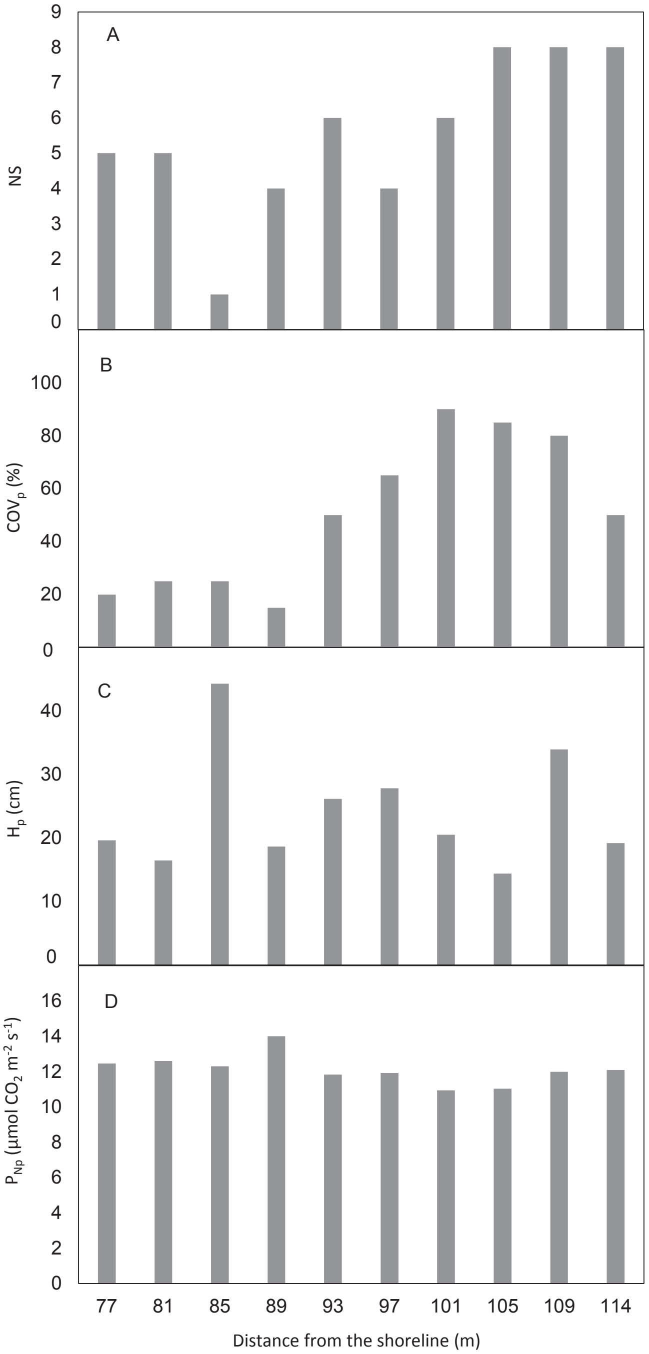

Fig. 1. A) Number of plant species per plot (NS), B) total plant species cover per plot (COV_p), C) mean plant height per plot (H_p) and D) mean photosynthetic rate per plot (P_{Np}) at site HAD characterized by a high anthropogenic disturbance.

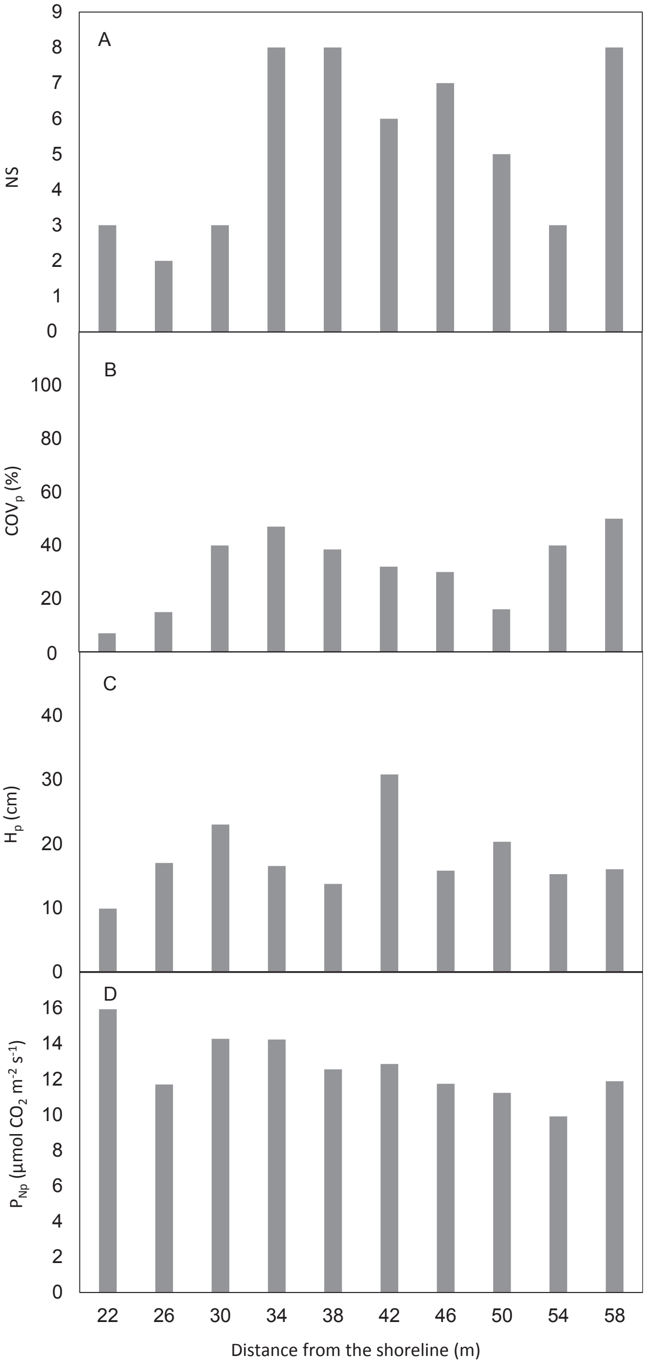

Fig. 2. A) Number of plant species per plot (NS), B) total plant species cover per plot (COV_p), C) mean plant height per plot (H_p) and D) mean photosynthetic rate per plot (P_{Np}) at site LAD characterized by a lower anthropogenic disturbance.

4. Discussion

The increased human pressure contributes to modify, mainly through land cover and use changes, the pattern of plant species presence and abundance and, consequently, the community structure of the coastal dune systems. Dune landscape changes are particularly evident along the Mediterranean coast where dunes often confine with urban areas, roadways, and tourist activities (Gratani et al., 2013; Bertacchi and Lombardi, 2014; Pinna et al., 2015). Our results did not show differences in plant richness between HAD and LAD (14 and 16 total plant species, respectively). In both sites, plant assemblages consisted of few characteristic or diagnostic species, which were arranged at different distance from the shoreline resulting in a patchy assemblage that did not follow the typical vegetation zonation according to previous studies (Gratani et al., 1983; Acosta et al., 2005; Carboni et al., 2010; Fenu et al., 2012). COV, which can be considered an indirect measure of the vegetation density, showed changes between HAD and LAD. The main differences were at the expense of annual community of the upper zone of the beach (i.e. drift line) that under well preserved conditions is characterized by *Cakile maritima* subsp. *maritima*, *Salsola kali* and *Chamaesyce peplis* (Carboni et al., 2010). In

Table 2
Results of ANCOVA analysis using the distance from the shoreline (DS) as covariate and number of plant species per plot (NS), total plant species cover per plot (COV_p), mean plant height per plot (H_p) and total yearly CO_2 sequestration flow per plot (CS_p) as dependent variables. In bold the parameters statistically significant.

Covariate	DS			
Dependent Variables	NS	COV_P	H_P	CS_P
R^2 (model)	0.33	0.59	0.15	0.16
p (model)	*0.03325*	*0.0004953*	*0.2685*	*0.2149*
F (model)	4.186	12.31	1.429	1.686
p (slope)	*0.698*	*0.379*	*0.8335*	*0.3355*
Intercept (site HAD)	−8.69	−135.91	10.43	−27.26
p (intercept site HAD)	*0.015*	*0.0013*	*0.595*	*0.098*
Intercept (site LAD)	0.01495	−33.4645	13.37	−7.58
p (intercept site LAD)	*0.012*	*0.043*	*0.413*	*0.279*

HAD, *Salsola kali* and *Chamaesyce peplis* were missing, while *Cakile maritima* subsp. *maritima* only was recorded in one plot with a low COV (1%). In LAD, residuals of the drift line vegetation were conserved although the plant species presence resulted rather fragmented. However, *Salsola kali* and *Cakile maritima* subsp. *maritima* were found in the plot nearest the shoreline (i.e. at 22 m). The decline of the drift line vegetation in HAD was mainly due to the high level of direct anthropogenic disturbance, resulting from an intensive beach cleaning by mechanical means as well as from human trampling linked to the presence of bathing establishments. Conversely, in LAD mechanical cleaning was not carried out, and human trampling was less intensive. Mechanical cleaning produced a direct effect on plants, reducing or destroying them and destructuring the foredune (Ciccarelli et al., 2017). This practice in HAD contributed to increase the extension of the aphytoic anthropogenic zone that was 64% of the total transect length. A further evidence of the intense trampling in HAD was highlighted by the presence of *Pheleum arenarium*, subsp. *caesium* a typical ruderal species. In HAD, this taxon was found in 40% of the analyzed plots with a mean COV of 11.5 ± 6.5% while it was absent in LAD. A lower direct anthropogenic impact in LAD was highlighted also by a smaller extension of the aphytoic anthropogenic zone, which was 30% of the total transect length.

Differences between HAD and LAD also concerned the plant species belonging to embryo dunes with *Elymus farctus* and to mobile dunes with *Ammophila arenaria*. In HAD, these species had a higher COV than in LAD. This may be explained given that these species are characterized by rhizome or deep roots that confer them a high resistance to mechanical disturbance factors. At LAD, a lower COV of embryo dune species may be related to coastal erosion phenomena endanger the site (Caputo et al., 1983). Many studies carried out on Mediterranean coastal dunes correlate the

Table 3
Total yearly CO_2 sequestration flow per plot (CSp) at HAD (high anthropogenic disturbance) and LAD (lower anthropogenic disturbance) sites.

HAD			LAD		
Plot	Distance from the shoreline (m)	CS_p (Mg CO_2 ha^{-1} y^{-1})	Plot	Distance from the shoreline (m)	CS_p (Mg CO_2 ha^{-1} y^{-1})
1	77	5.67	1	22	1.42
2	81	6.06	2	26	4.19
3	85	10.36	3	30	13.20
4	89	5.38	4	34	71.90
5	93	9.38	5	38	69.87
6	97	17.84	6	42	116.09
7	101	19.06	7	46	45.07
8	105	13.71	8	50	44.32
9	109	316.58	9	54	5.54
10	114	38.99	10	58	49.62
Total		443.03			421.22

Table 4
Yearly CO_2 sequestration flow (CS) of the plant species present at HAD (high anthropogenic disturbance) and LAD (lower anthropogenic disturbance) sites.
(np = not present)

Plant species	HAD	LAD
	CS (Mg CO_2 ha^{-1} y^{-1})	CS (Mg CO_2 ha^{-1} y^{-1})
Ammophila arenaria	309.4 ± 8.7	111.4 ± 4.5
Anthemis maritima	2.8 ± 0.3	2.73 ± 1.22
Cakile maritima subsp. *maritima*	0.3 ± 0.1	0.65 ± 0.32
Calystegia soldanella	np	0.92 ± 0.50
Chamaesyce peplis	np	1.49 ± 0.50
Crucianella maritima	34.1 ± 4.2	23.2 ± 22.2
Cutandia maritima	np	0.14 ± 0.04
Cyperus capitatus	$7.3 \cdot 10^{-3} \pm 1.8 \cdot 10^{-3}$	np
Echinophora spinosa	1.3 ± 0.6	0.25 ± 0.08
Elymus farctus	4.1 ± 1.6	1.92 ± 0.56
Eryngium maritimum	0.6 ± 0.1	0.94 ± 0.30
Medicago marina	3.8 ± 1.7	0.14 ± 0.02
Ononis variegata	0.4 ± 0.1	1.96 ± 1.5
Pancratium maritimum	np	53.6 ± 6.2
Phleum arenarium subsp. *caesium*	4.2 ± 2.4	np
Salsola kali	np	0.02 ± 0.01
Silene canescens	0.10 ± 0.04	np
Sporobolus virginicus	$1.6 \cdot 10^{-2} \pm 0.65 \cdot 10^{-2}$	$5 \cdot 10^{-3} \pm 0.23 \cdot 10^{-3}$
Vulpia fasciculata	$1.8 \cdot 10^{-4} \pm 0.3 \cdot 10^{-4}$	$1.7 \cdot 10^{-5} \pm 0.03 \cdot 10^{-5}$

Table 5
Estimation on CO_2 amount (Mg CO_2) and on monetary ($) base of the yearly CO_2 sequestration flow (CS), CO_2 storage (CST), total CO_2 sequestration (SCO_2), total CO_2 release (RCO_2) and net CO_2 sequestration capacity (NCS) of the considered sites. Monetary estimation was based on social cost of carbon (SSC) that was considered to be $ 7.36/Mg CO_2.
Details on calculation method were given in section 2.4. HAD = site with a higher anthropogenic disturbance. LAD = site with a lower anthropogenic disturbance.

	HAD	LAD
CS (Mg CO_2 ha^{-1} y^{-1})	443.03	421.22
CST (Mg CO_2 ha^{-1})	0.554	0.554
S_{CO2} (Mg CO_2)	621.22	341.74
R_{CO2} (Mg CO_2)	398.29	100.97
NCS (Mg CO_2)	222.92	240.76
CS ($ ha^{-1} y^{-1})	3261	3100
CST ($ ha^{-1})	4.08	4.08
S_{CO2} ($)	4572	2515
R_{CO2} ($)	2931	743
NCS ($)	1641	1772

embryo dunes simplification or absence with erosion phenomena (Lomba et al., 2008; Carboni et al., 2009; Gratani et al., 2013; Ciccarelli, 2014). Moreover, coastal erosion negatively affects also COV of mobile dunes with *Ammophila arenaria* (Ciccarelli et al., 2012, 2014). At LAD, the disappearance of the embryo dunes exposes the mobile dunes to seashore salt spray and to a more intense wind action, decreasing the sand accumulation, and consequently the survival of *Ammophila arenaria* a sand-fixing species. In fact, *Ammophila arenaria* colonizes dunes with a heavy sand deposition (Lubke and Hertling, 2001), providing positive effects on sand stability (Rodríguez-Echeverría and Freitas, 2006; Fenu et al., 2012). Moreover, this species is in symbiosis with mycorrhizas, which play an important role on its growth (Gratani, 1987) contributing to enhance its sand-fixing capacity.

Despite the patchy plant species distribution, it was possible to recognize in both the sites the typical sea-inland ecological gradient, with a significant increasing trend of NS and COV_p with the distance from the shoreline. This increase occurred at the same magnitude since the slopes of regression line were not significantly different while there were differences in terms of intercepts. On the contrary, the distance from the shoreline did not co-varied with H_p and CS_p. Indeed, the different level of direct disturbance altered only the plant species richness and abundance pattern without affect plant traits related to functioning such as H_p, P_{Np} and CS_p. This is confirmed by not significant differences found between HAD and LAD in mean value of CS_p, H_p and P_{Np}. Moreover, HAD and LAD shared most plant species (78% of the species were in common). These species, as discussed above, differed in COV but they did not significantly differ in H and P_N and as consequences in CS. Accounting for CS requires adequate knowledge of plant structural traits as they are strictly related to plant productivity (Saxe et al., 2001; Pan et al., 2011). In particular, H is a key trait correlated allometrically to above-ground biomass (Cornelissen et al., 2003; Puglielli et al., 2016). In our study, the relationship between plant height and CO_2 sequestration was confirmed by multiple regression analysis showing that H_p was the most significant variable accounting CS_p variations. Moreover, the species that more contributed to CS_p were those taller such as *Ammophila arenaria* and *Pancratium maritimum*. In fact, mean CS_p decreased by 70% and 78%, in HAD and LAD respectively if it was calculated without taking into account the fraction of CO_2 sequestered by *Ammophila arenaria* and *Pancratium maritimum*.

In addition to CO_2 sequestration and storage, sand dunes provide other ecosystem services among which soil formation, erosion, storm and flood coastal protection, biodiversity, recreation and tourism (Everard et al., 2010). However, estimates of the economic value of such ecosystem services are still poor (Barbier et al., 2011). Pérez-Maqueo et al. (2013) by comparing 18 studies carried out in different countries between 1992 and 2011, found that the economic value of the co-benefits of beaches and dunes based on the willingness to pay method, ranged from $ 3 ha^{-1} y^{-1} for the carbon sequestration service to $ 36,202 ha^{-1} y^{-1} for the coastal protection service. Quantify ecosystem services in monetary terms is important to increase the awareness of the policy maker (de Groot et al., 2012), encouraging new perspective in the management of these areas. Moreover, monetary estimates based on economic value such as SSC are more appropriate in order to make more balanced decisions on trade-offs between different possible development choices (Beaumont et al., 2014; Costanza et al., 2014).

Our estimates based on SSC revealed that the flow of sequestered CO_2 valued on an average $ 3181 ± 114 ha^{-1} $year^{-1}$ while CO_2 storage amounted to $ 4 ha^{-1}. However, considering also the value of the CO_2 negative flow related to loss of vegetated area, the annual net benefit arising from CO_2 sequestration amounted to $ 1707 ± 93 (mean value for the two sites). Extending this value to the Italian dune systems, which cover an area roughly 9000 ha (Feola et al., 2011) a nationwide estimated annual benefit in the order of $ 15.363 million was obtained. Currently, the main threat for coastal dune habitats worldwide is the decline in surface extension. The habitat loss is due to climate change effects, which are increasing coastal erosion, and to anthropogenic pressure mirrored by the expansion of urban and agricultural areas to detriment of dune ones. The habitat loss decreases the effectiveness of the service of CO_2 sequestration increasing on the contrary the negative flow of this service (i.e. release of CO_2 into the atmosphere) also with consequences in economic terms. In our case, the economic value of the net CO_2 sequestration service provided by HAD was $ 131 lower than the service supplied by LAD. This difference was related to a higher anthropogenic disturbance level in HAD mirrored by a greater extension of the un-vegetated zone (0.90 and 0.24 ha in HAD and LAD, respectively). In fact, R_{CO2} was valued at $ 2931 and $ 743 HAD and LAD, respectively.

In the Mediterranean Basin, including Italy, the loss of dune habitats is also particularly worrying considering their geomorphology. In fact, Mediterranean dunes are generally small and often characterized by the presence of a single dune belt (Feola et al., 2011). Along the Italian coasts about 36,000 ha of dunes were lost from the beginning to the end of the twentieth century (Feola et al., 2011). Considering the economic benefit that we calculated for the CO_2 sequestration service supplied by dunes, we can conclude that in Italy the loss of these habitats during the last century cost $ 61.452 million in terms of no provided service.

Moreover, in evaluating the economic losses due to dune degradation, it must be considered that coastal dunes are of paramount importance in protecting other blue carbon sinks such as seagrass beds. In fact, the beach erosion due to the dune destruction can affect also the submarine sand colonized by seagrasses causing their loss (Medina et al., 2001). Along the European coasts, and especially in the Mediterranean Basin seagrass species such as *Posidonia oceanica* are the most able to sequester and store CO_2 (Kennedy and Björk, 2009). Nevertheless, these habitats are in strong decline in the Mediterranean Basin because of climate change effects (Luisetti et al., 2013). These authors estimated that if the decline rate continues at the present extent, the economic loss of carbon storage service, based on SSC value, could be about $ 70,650 in the next 50 years. Thus, the degradation of dune habitats could accelerate the destruction of the seagrass beds and therefore compromise seriously the Mediterranean blue carbon sinks.

5. Conclusions

Our findings showed that a different level of a direct anthropogenic disturbance did not affect plant species structure and function as highlighted by no significant differences between the two considered site in CO_2 sequestration capacity. This result could lead to the hypothesis of an ecological and functional convergence allowing plant species to effect compensatory mechanisms to acclimate to disturbance factors. However, it must consider two aspects: (i) the narrow range of dune plant species adaptability, and (ii) the intrinsically vulnerability of dune habitats. In view of this, human impact could act as the main selective driver leading to change rapidly the physiological tolerance threshold of these plant species. Nevertheless, further studies are necessary to confirm this hypothesis. Despite this, even if the patchy plant species distribution due to anthropogenic disturbance did not influence the flow of the sequestered CO_2, human impact via habitat loss seriously compromised the capacity of these habitats to sequester and store CO_2, with a severe economic benefit loss. In addition, considering that the species having a key role in CO_2 sequestration such as *Ammophila arenaria* and *Pancratium maritimum* take a longer time to re-colonize, the restoration efforts of these habitats could be more expensive. In the light of this, it is necessary to maximize the efforts to preserve dune habitats by applying an effective management policy based on the biodiversity conservation, which could also allow maintaining a regulatory ecosystem service such as CO_2 sequestration.

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