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WETLAND RESTORATION

Vegetation Dynamics on a Restored salt Marsh Mosaic: a Re-Visitation Study in a Coastal Wetland in Central Italy

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Abstract

Coastal wetlands are biodiversity hotspots, highly threatened, and for which restoration actions have been widely implemented. Systematic monitoring of biodiversity after restoration actions on Mediterranean salt marshes vegetation needs further attention. We analyzed temporal changes in plant species composition and ecology in a restored brackish wetland on the Adriatic coast (Central Italy) by a re-visitation study of 33 historical plots (year 2010), newly collected after 10 years (2021), across a brackish mosaic composed by salt meadows, halophilous scrubs and salt steppes referable to three habitats of conservation concern in Europe (EU codes: 1410, 1420 and 1510*). Changes in species richness and cover, in the ecological characteristics of the mosaic and each habitat type were tested by comparing some ecological groups (e.g. diagnostic, alien and ruderal species) and Ellenberg bio-indicator values by a Mann-Whitney test. Similarity percentage procedure for identifying which species indicate temporal changes was also performed. After restoration, we observed a general improvement of the environmental quality of the brackish mosaic with the establishment of typical pauci-specific plant communities, a significant recovery of diagnostic species cover and a reduction of ruderal and alien ones. We also registered an increase in Ellenberg salinity and temperature values likely related also to coastal erosion and climatic change. The results of our study suggest that vegetation dynamics could be used to monitor coastal restoration trajectory in the Mid- and Long-Term local interventions.

Keywords Adriatic coast · Multitemporal analysis · Vascular plants · Ecological groups (diagnostic, ruderal, alien) · Brackish vegetation · Ellenberg bioindicators

Introduction

Coastal wetlands are complex and dynamic ecosystems widely distributed on the world's shorelines (Scott et al. 2014). Occupying transitional waters between freshwater and marine realms (Pérez-Ruzafa et al. 2011) they conform intricate mosaics (Holland 1988) following steep environmental gradients (e.g. oxygen, pH, salinity) which encompass a particularly specialized flora and fauna. Wetland mosaics are shaped by seasonally changing abiotic (e.g. salinity, soil aeration, frequency and duration of inundations and elevation of the marsh surface) (Cooper 1982; Snow and Vince 1984; Armstrong et al. 1985; Niedowski 2000; Lefeuvre et al. 2003) and biotic factors (e.g. interspecific competition for light and nutrients) (Levine et al. 1998; Ungar 1998).

In coastal salt marshes plant species are mainly stress tolerant and specialist, well adapted to highly variable and dynamic ecological conditions (Lefeuvre et al. 2003), however also some generalist species coming from the adjacent ecosystems could occur. Specifically, the Mediterranean coastal salt marshes vegetation consists of a mosaic of low-growing meadows with herbaceous plants able to dwell on wet and hydromorphic soils periodically flooded (Cutini et al. 2010; Gennai et al. 2022). Such meadows are composed by grasses, sedges, rushes and other herbaceous angiosperms distributed across an observable zonation, according to topographic and environmental variability as well as

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vegetation succession linked to the geo morphogenesis of salt marshes (Taramelli et al. 2021).

Coastal wetlands also play a key supporting role for animal biodiversity as they provide critical habitats for resident (e.g. arthropods) and migratory fauna (e.g. birds) (Perennou et al. 2018; Sala et al. 2000). Indeed, they ensure different stages of the life cycle to a great variety of species offering a suitable habitat for fish and invertebrate spawning as well as for the larval and juvenile stages. Many migratory birds use marshes as feeding (offering trophic resources as fishes, invertebrates, insects and plants) (Niedowski 2000), nesting and resting areas (Viciani and Lombardi 2001).

Coastal wetlands also provide essential benefits to society, some of which with a considerable socio-economic impact (Martínez-Megías and Rico 2021; Millennium Ecosystem Assessment 2005). They contribute more than 20% of the total value of global ecosystem services (Costanza et al. 2014), while covering only a small percentage (4–9%) of global land surface (Morganti et al. 2019; Zedler and Kercher 2005). Salt marshes provide a wide range of services as nutrient cycling, water remediation (Quin et al. 2015; Chalov et al. 2017), flood control (Acreman and Holden 2013; Quin and Destouni 2018), soil moisture regulation (Golden et al. 2017; Ameli and Creed 2019) and biodiversity conservation (Mitchell et al. 2008; Cohen et al. 2016). In addition, they play a major role on carbon sequestration (Herbert et al. 2015) and climate regulation (Camacho et al. 2017; Morant et al. 2020) with blue carbon (e.g. belowground carbon stocks and carbon burial rates) stocks reaching one of the highest values in the biosphere (Donato et al. 2011; Mcleod et al. 2011) and subsequently they represent an excellent training ground to explore global change dynamics (Lefeuvre et al. 2003).

Despite the high biodiversity value and the numerous benefits for the human wellbeing, coastal wetlands are among the most imperiled ecosystems both, globally (Golden et al. 2017; Chen 2019) and in the Mediterranean basin (Erwin 2009). Approximately 50% of the world's wetlands have been lost since 1900 and their loss rate during the 20th and early 21st centuries averaged $-1.085\%.y^1$, varying between regions (e.g. Asia has lost the 83.7%, Europe the 71.0% and North America the 36.5%) (Davidson 2014; Davidson et al. 2018). In the Mediterranean, brackish marshes have undergone a drastic reduction due to land reclamation and conversion to croplands, changes in water regimes, urbanization and invasive alien species (Lefeuvre et al. 2003; Destouni et al. 2013; Jaramillo and Destouni 2015; Adam 2019; Maneas et al. 2019) combined with climate change (Seneviratne et al. 2006; Orth and Destini 2018) and coastal erosion (Erwin 2009; Taramelli et al. 2021) which have caused a drastic reduction of ecosystem services (Ghajarnia et al. 2020) and a loss of biodiversity. Specifically, the alterations on marshes hydrology (depth and hydroperiod) along with the increasing temperatures and the reduction of water supply registered during the last decades (Root et al. 2003) consistently threat more than 35% of wetland species (Martínez-Megías and Rico 2021).

For such outstanding threatened biodiversity, wetlands are protected by the intergovernmental Convention of Ramsar (Bonells and Zavagli 2011) which provides the regulatory framework for defining national and international conservation sites (so called Ramsar sites) and dedicated actions for their conservation and management (Matthews 1993; de Klemm 1995; Bonells and Zavagli 2011). Furthermore, in Europe, most of the salt marsh plant communities have been of conservation concern and listed in the Habitats Directive (here after HD; European Directive 92/43/EEC) for which conservation and restoration actions are claimed. According to HD, member states are committed to monitoring and preserving habitats extension into the Union and implementing the necessary management measures to keep them in a good "conservation status".

Amongst the possible conservation measures, the restoration of salt marshes, aimed at bringing back the brackish mosaic to its original condition faster than nature does on its own and at establishing a self-sustaining ecosystem status, has rapidly accelerated over the last decades with the great support of government agencies and conservation organizations (Adams et al. 2021). There is evidence that salt marshes vegetation recovery time under natural conditions is quite fast (e.g. around 10 or more years, depending on the perturbation and the maturity of the marsh) (Broome et al. 1988), so after the necessary hydraulic reconstruction works, soft restoration schemes promoting spontaneous recovery of natural key species are advisable (Wolters et al. 2005, 2008). The assessment of the effectiveness of saltmarsh restoration actions in terms of plant species composition in some European wetlands (Wolters et al. 2005; Billah et al. 2022) have evidenced a good recovery of native plant diversity over time (Curado et al. 2014). Despite the importance of the restoration of salt marshes and its widely implementation in several coasts in the world (Billah et al. 2022), updated research and systematic monitoring activities aiming to assess biodiversity changes after restoration actions on Mediterranean salt marsh areas should be improved (Moreno-Mateos et al. 2015; Billah et al. 2022).

In this context, the present work sets out to analyze vegetation dynamics on a restored salt marsh mosaic, through a multi-temporal analysis of vegetation plots collected before and after the implementation of restoration actions in the Central Adriatic coast in Italy. We hypothesized a good response of vegetation that after restoration will evolve towards improved of ecosystems, with a gain of diagnostic native species and a reduction of alien and ruderal ones. Specifically, by a re-visitation study (data collection carried out in the years 2010 and 2021) we explored plant species composition and ecology changes across the brackish habitats addressing the following questions: (i) Have the abundance and distribution of vascular plant species changed during the last decade?; (ii) which are the abundance trends in the main plant groups (diagnostic, ruderal and alien species) and in halophilous and thermophilous species over time in the brackish mosaic habitats?

By increasing the knowledge on vegetation dynamics and how it varies across the different habitats of the brackish mosaic after a restoration actions, we wish to contribute to improve the current scientific understanding on the effectiveness of implemented conservation strategies (Wolters et al. 2005; Billah et al. 2022) and give new insights for the adaptive management and the prioritization of the conservation actions in such highly vulnerable environment.

Materials and Methods

Study Area

The study area is located in the Adriatic coast of Central Italy (Molise Region; Fig. 1) characterized by Mediterranean climate (Blasi 2003) and composed by sandy dunes which alternates with alluvial plains and river mouths (e.g. Trigno, Biferno and Saccione) (Stanisci et al. 2007; Carranza et al. 2008). Salt marshes only occur at Biferno river mouth area and they represent a residual wetland which was larger one century ago (Forleo 2005). Salt marshes have not direct connections to the sea and are fed by salt water table and partially by artificial wetland drainages.

The target area has been exposed to high erosion risk with strong coastal erosion processes (Rosskopf et al. 2018). The period 1954–2014 registered an erosion rate of − 2.90 m/year in the study area and such trend is expected to proceed over time (Aucelli et al. 2018).

The climate in the analyzed coastal tract, as in the whole Mediterranean region, is changing rapidly (IPCC 2022). The

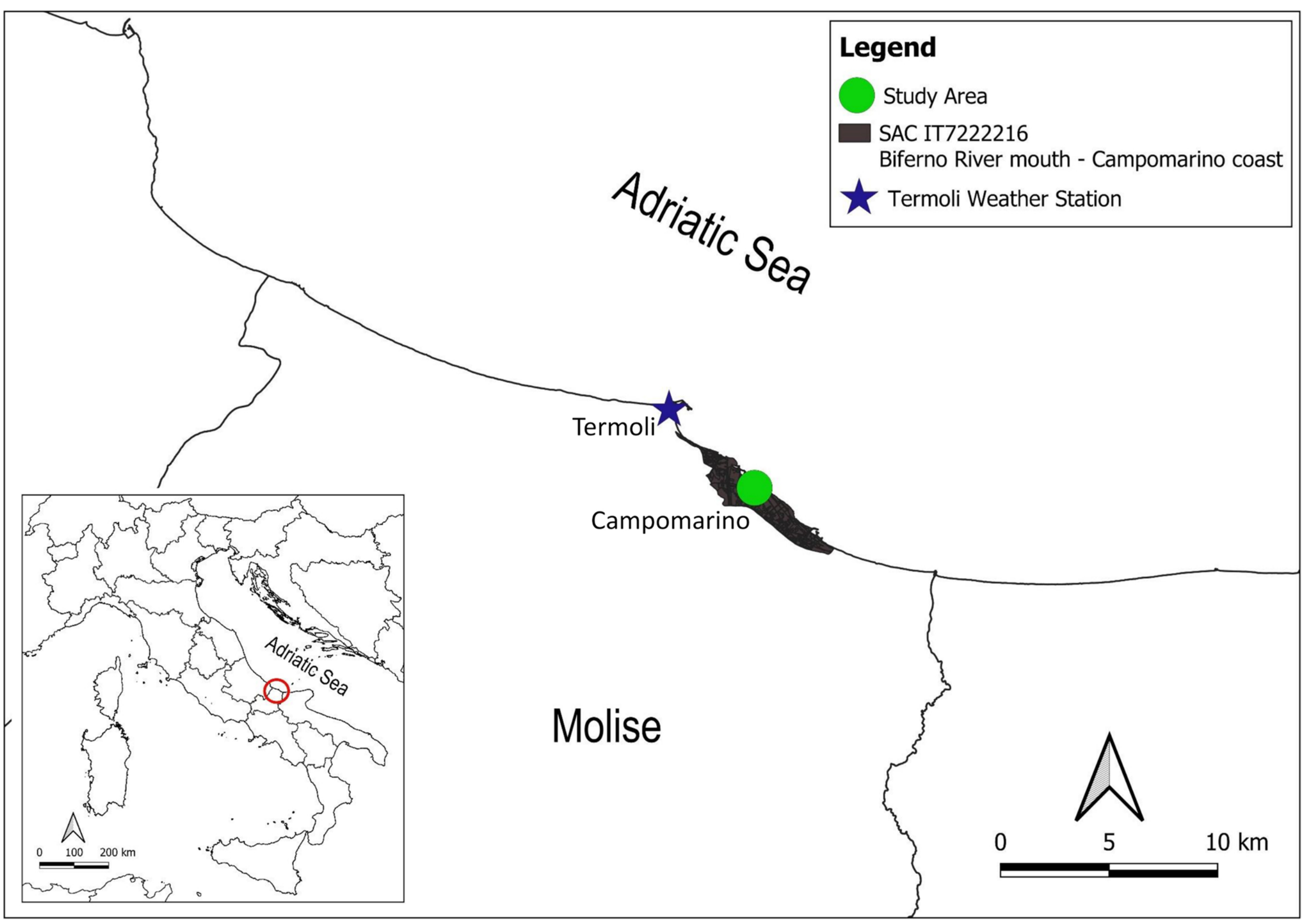


Fig. 1 Study area included in the Special Area of Conservation Biferno River mouth – Campomarino (SAC IT7222216)

statistical analysis of climatic data recorded in the last fifty years (1970–2020) in the nearby weather station of Termoli (SCIA climatic database; Desiato et al. 2006, 2007, 2011) evidenced a consistent rise of temperatures and a slight decrease of annual precipitations. The mean annual temperatures in the last half century has been of 16,74 °C with annual values that significantly increased from 15,5 °C to 18,6 °C (R^2 = 0,839, p-value < 0,001) (Fig. 3). Precipitations in summer (that is the period of greatest aridity stress for plants in the Mediterranean biome, Nardini et al. 2014) registered a mean value of 21,50 mm and a slight decline from ≈ 30 mm to ≈ 19 mm (R^2 = 0,053, p-value = 0,1081) (Fig. 2).

We analyzed the plant communities of the residual brackish wetlands occurring in the inter-dunal humid depressions of the Biferno river mouth area (Fig. 1), including a rich mosaic of ecosystems of Conservation Concern in Europe (included in Annex I of the Habitats Directive, hereafter HD) (EEC 1992; European Commission 2013; Stanisci et al.

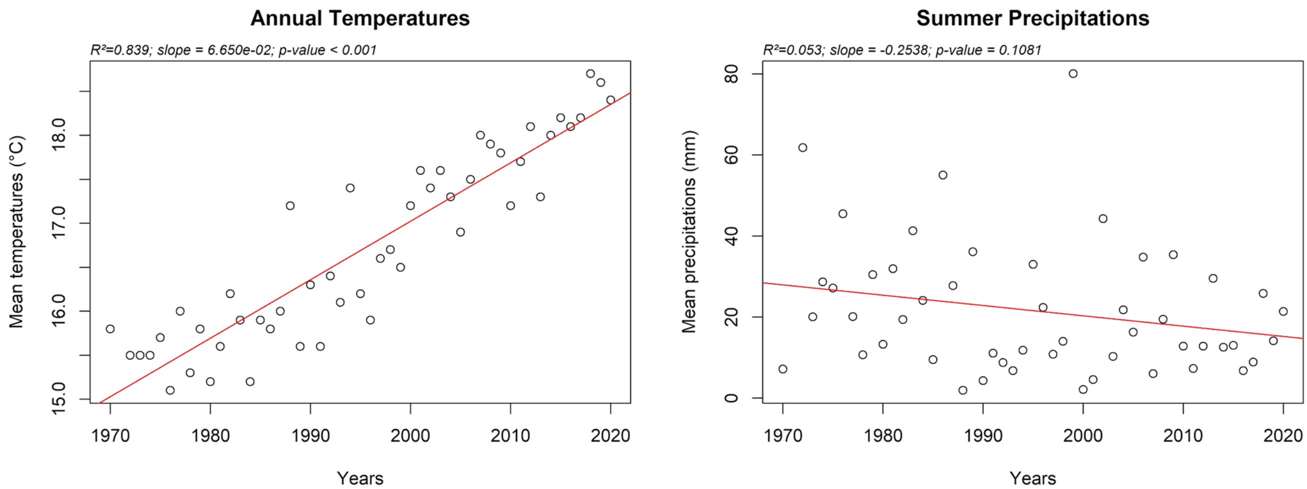


Fig. 2 Mean annual temperature and summer precipitation from 1970 to 2020 (Termoli weather station). Data were retrieved from SCIA climatic database (Desiato et al. 2006, 2007, 2011). Regression and graphs made with R statistical software (R Core Team 2020)

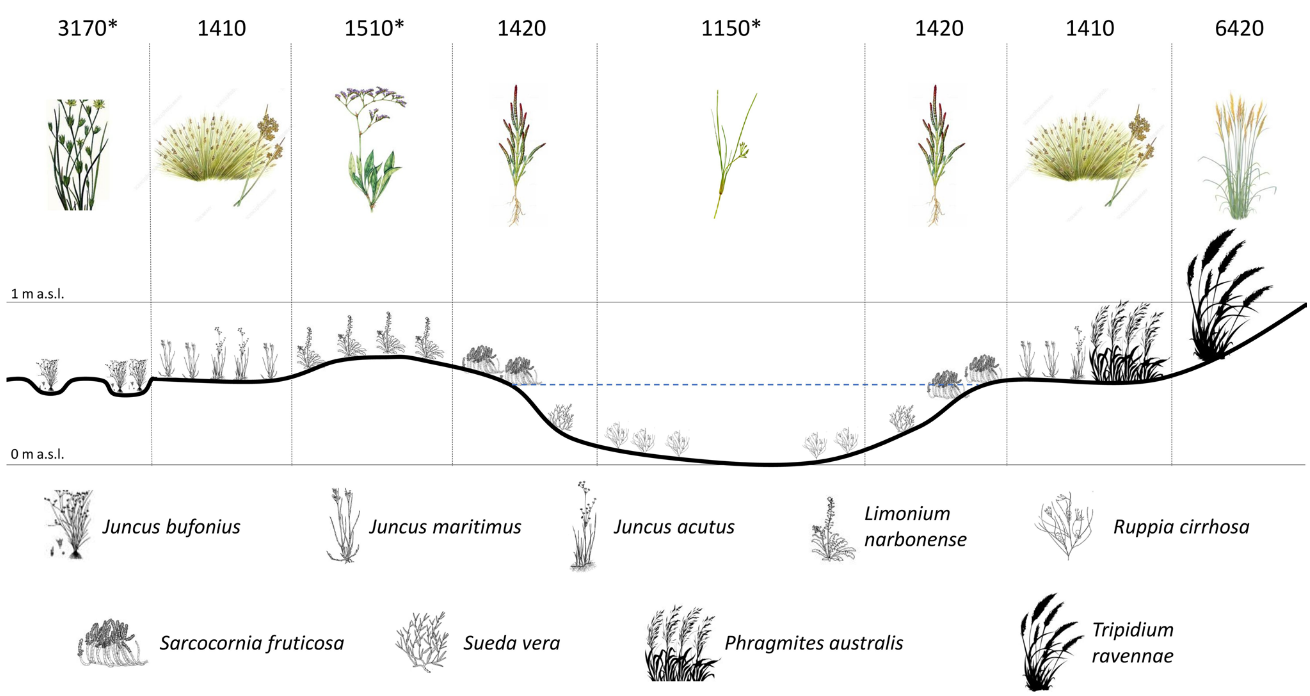


Fig. 3 A schematic profile describing the typical brackish vegetation zonation in the study area and the respective EU habitat types (EEC 1992) along with their codes. Asterisks indicate EU priority habitats. A description of the habitats is reported in Table 1

2014) (Fig. 2) and conforming a key site for the conservation of the fauna. For its great biodiversity value, the area is a node of the Natura 2 K network (Special Area of Conservation: IT7222216 Biferno river mouth-Campomarino coast) and is a pilot site for testing ecological monitoring tools, in situ and remotely sensed (Marzialetti et al. 2020).

Vegetation and Biodiversity of the Brackish Mosaic

Salt marshes in the Biferno mouth are composed by a mosaic of habitats of Conservation Concern (HD 92/43/CEE; http://vnr.unipg.it/habitat/index.jsp) whose spatial variability (e.g. zonation) is shaped by the interplay of several environmental factors as: water table level, local micro morphology, substrate salinity and seashore distance. Vegetation zonation in the Biferno mouth brackish mosaic is schematically reported in Fig. 2 and briefly described in Table 1, below.

Biferno mouth wetlands also host several species of fauna, such as the migratory and sedentary birds (e.g. *Ixobrychus minutus*, *Gallinula chloropus*, *Phalacrocorax carbo*, *Ciconia nigra*, *Himantopus himantopus* and *Botaurus stellaris*) (De Lisio et al. 2008), reptiles (i.e. *Hemys orbicularis, Testudo hermanni*) (Berardo et al. 2015), amphibians (e.g. *Epidalea viridis)* and bats (Prisco et al. 2017).

In 2016, the area was part of an environmental restoration program, funded by LIFE10 NAT/IT/00262 project which aimed at recovering the wetland ecosystem. The water flow pattern was re-established by opening the artificial wetland drainages and recovering the local hydrological regime (Prisco et al. 2017). Restoration included the demolition of artifacts, the reclamation of hazardous materials and the reconstruction of banks (Pellizzari et al. 2007; Prisco et al. 2017). Still, a boardwalk and a set of picket fences were put in place to protect salt marshes area from human trampling.

Vegetation Sampling

During the years 2020-21, we re-visited (hereafter T2), 33 vegetation plots collected in 2010 (hereafter T1) (Di Franco et al. 2012). Vegetation plots, collected within the Biferno brackish wetlands before and after the restoration of wetlands carried out in the year 2016 (Prisco et al. 2017) are representative of the brackish mosaic dominated by the following habitats of conservation concern (HD: 92/43/EEC): 1410: Mediterranean salt meadows - *Juncetalia maritime;* 1420: Mediterranean and thermo-Atlantic halophilous scrubs - *Sarcocornietea fruticose* – 1510*: Mediterranean salt steppes – *Limonietalia*). Phytosociological relevés of 16 m^2 (4 × 4 m) were carried out following a stratified random protocol that used a detailed land cover map (1: 5000 scale;

Table 1 EU habitat names (EEC 1992) along with their short name, brief description and the dominant species present in the Biferno mouth brackish mosaic

Habitat name	Short name	Description	Dominant species
Coastal lagoons (EU habitat 1150*)	Coastal lagoons	Aquatic vegetation growing on shallow brackish waters with strong temporal variations in salinity and water depth, responding to differences in water table inputs, rainfalls and temperatures.	*Ruppia cirrhosa*
Mediterranean salt meadows (EU habitat 1410)	Salt meadows	Subalophilic meadows of backdunal humid depressions with medium-high sandy substrates flooded by brackish water for medium-long period.	*Juncus acutus, J. maritimus*
Mediterranean and thermo-Atlantic halophilous scrubs (EU habitat 1420)	Halophilous scrubs	Pauci-specific communities consisting of perennial halophytes, mainly chamaephytes and succulent nanophanerophytes, growing on periodically flooded areas	*Sarcocornia fruticosa*
Mediterranean salt steppes (EU habitat 1510*)	Salt steppes	Halophilic perennial herbaceous species of the back side of the halophilous scrubs, on small dumps with salty soils (clayey, clayey-slimy or sandy), temporarily humid, but not submerged.	*Limonium narbonense*
Mediterranean temporary ponds (EU habitat 3170*)	Temporary ponds vegetation	Amphibious vegetation given by small therophytic and geophytic species with late-winter/spring phenology, growing in small temporary ponds.	*Isolepis cernua, Juncus bufonius*
Sub-pannonic steppic grasslands (EU habitat 6420)	Steppic grasslands	Reed vegetation growing on sandy-clay soils in contact with dune grasslands.	*Tripidium ravennae*

For the schematic description of brackish vegetation zonation see Fig. 2

AA.VV. 2008) and high-resolution color digital orthophotos (flight 2007, granted by the Civil Protection) for identifying the strata.

For re-visitation, we sampled the same T1 plots following the description of the location reported in the reference study (Di Franco et al. 2012). We carried out phytosociological relevés following the same sampling protocol (Chytrý et al. 2014) and in the same season (April-October) to remove the effects of phenological differences (Vymazalová et al. 2012). In addition, in order to limit the pseudo-turnover caused by observer bias (Klimeš et al. 2001; Vittoz and Guisan 2007), one of the researchers who had conducted the T1 sampling campaign was also involved in T2 field work activity. For each georeferenced vegetation plot we registered the complete list of vascular plants and their cover values in compliance with Braun-Blanquet scale (Westhoff and Van Der Maarel 1978; Pignatti 1995; Braun-Blanquet 2013) using the classical phytosociological approach. Species nomenclature follows the updated checklist of "Flora d'Italia" (Pignatti et al. 2017–2019).

Data Preparation

We investigated brackish plant communities' ecology over time (T1: 2010, T2: 2020/21) exploiting the bio indication value of some plant groups (e.g. diagnostic, ruderal and alien species) (Santoro et al. 2012; Del Vecchio et al. 2016) and the Ellenberg's ecological indicator scores for salinity and temperature (Ellenberg 1974).

We considered three main ecological groups which provide key information on habitat health (e.g. conservation status, disturbance, threat degree) (Cardinale et al. 2012; Keith et al. 2013). Diagnostic species, playing a major role in determining the structure and functioning of the EU habitats, are a reliable indicator of conservation status (Chytrý and Tichý 2003). We defined the diagnostic species for each habitat type according to the Italian Interpretation Manual of Habitats Directive (Biondi et al. 2009) and accounting of updated information reported on the "Italian Vegetation Prodrome" (Biondi et al. 2014; European Commission 2013). Ruderal native species, having an opportunist ecological strategy and being well adapted to disturbed habitats (Malavasi et al. 2016), are excellent indicators of ecosystem alterations (Del Vecchio et al. 2015a). Ruderals were here identified based on previous phytosociological studies of the Italian Adriatic coast (Bini et al. 2002; Di Franco et al. 2012; Pirone et al. 2014; Sciandrello and Tomaselli 2014; Tomaselli et al. 2020). Alien plant species (IAPs) that are species growing outside their natural range (Richardson et al. 2000) which could severely alter ecosystem functioning (Pyšek et al. 2020), point out a consistent threat to biodiversity. IAPs were identified following the inventory of the non-native flora of Italy (Viciani and Lombardi 2001; Celesti-Grapow et al. 2009; Galasso et al. 2018).

Temporal changes in the brackish communities' ecology were also explored by Ellenberg's salinity and temperature indicator values. To each plant we assigned the Ellenberg's Bioindicator Value (Ellenberg 1974), which is an ordinal number (1–9) describing species preference along ecological gradients assigned according to Pignatti et al. (2005).

Statistical Analysis

After a brief comparison of the number and cover of species of the different groups (e.g. diagnostic, ruderal and alien) over time for the entire mosaic and each habitat type, we explored temporal changes in the ecology of the analyzed vegetation, by comparing Ellenberg bioindicator values. For each relevé, we calculated the mean Ellenberg bioindicator values weighted according to species cover as follows:

where *rji* is the cover of the species *i* in the relevé *j*, and *xi* is the Ellenberg bioindicator value *x* for the species *i* (Diekmann 2003; Evangelista et al. 2016; Calabrese et al. 2018). For each habitat and temporal step we calculated the *WA* for salinity and temperatures depicting environmental conditions (Pignatti et al. 2005; Jantsch et al. 2013; Del Vecchio et al. 2015b).

Furthermore, we analyzed the temporal variation of ecological groups and for the Ellenberg values by a Mann-Whitney post hoc test on ranked data (cover and richness). The two-tailed (Wilcoxon) Mann-Whitney U test was used to test whether the medians of the two time steps are different.

Afterward, we identified the species that contribute most consistently to the differences between the two temporal groups (T1 and T2) using a similarity percentage procedure (SIMPER) (Clarke 1993).

Statistical analyses were performed in the R statistical computing program (R statistical software, R Core Team 2020) using the Vegan package (Oksanen et al. 2020) and using PAST (paleontological statistics software for education and data analysis) (Hammer et al. 2001).

Results

In the whole brackish mosaic, we recorded 92 vascular plant species and subspecies of which 29 were diagnostic of at least one EU habitat type (31,5%), 25 were ruderal (27,2%) and 6 were alien(6,5%).

We observed a general decrease in the total number of species (from 71 to 54; Table 1) as well as in the number of diagnostic (from 24 to 22), ruderal (21 to 10) and alien species (from 5 to 4).

Concerning the single EU habitats, we observed a decline on the total number of species, however diagnostic species slightly increased in salt steppes, whereas IAPs remained stable in the salt meadows and the halophilous scrubs (Table 2).

The analysis of ecological groups and Ellenberg bio-indicator values over time revealed important changes in the entire brackish mosaic (from 2010 to 2020/21) and such changes varied across the different habitat types.

We registered in the whole brackish mosaic a significant increase in the cover of diagnostic species (P_{same} = 0,031) and a significant decrease in the cover and richness of ruderals (respectively P_{same} < 0,001 and P_{same} = 0,014) (Fig. 4).

As observed at mosaic level, we registered significant gains of diagnostic species cover in the salts meadows (P_{same} = 0,046) and steppes (P_{same} = 0,011) and a significant decrease of their richness per plot in the halophilous scrubs (P_{same} = 0,007) (Fig. 4). As regards the ruderal species, we observed a significant decrease in cover and richness in halophilous scrubs (respectively P_{same} < 0,001 and P_{same} < 0,001) and salt steppes (respectively P_{same} = 0,025 and P_{same} = 0,025). Concerning alien species, we found a significant decrease in cover in salt steppes (P_{same} = 0,038) (Fig. 4).

Concerning the Ellenberg salinity value, the analysis showed a significant increase over time in brackish mosaic (P_{same} < 0,001) and in halophilous scrubs and salt steppes (P_{same} < 0,001 and P_{same} =0,012 respectively) (Fig. 5).

The species that, according to SIMPER analysis (similarity percentage) (Table 3), contributed 50% of floristic changes in the salt meadows habitat are diagnostics and thermophilous (T ≥ 7) with low-medium Ellenberg Salinity values. The cover of these species increased over time, except for *Juncus maritimus* that decreased. In halophilous scrubs and salt steppes habitats the temporal changes are given by the increase of some halophilous species (e.g. *Sarcocornia fruticosa and Limonium narbonense* with Ellenberg Salinity value of 8–9) and by the reduction of species with low Ellenberg Salinity value (e.g. *Juncus maritimus* and *Plantago crassifolia* with 6 and 1 indicator value).

Discussion

The analysis of vegetation dynamics on restored salt marshes in the Adriatic coast in Central Italy (Biferno brackish area) revealed consistent changes on floristic composition and an improved conservation status.

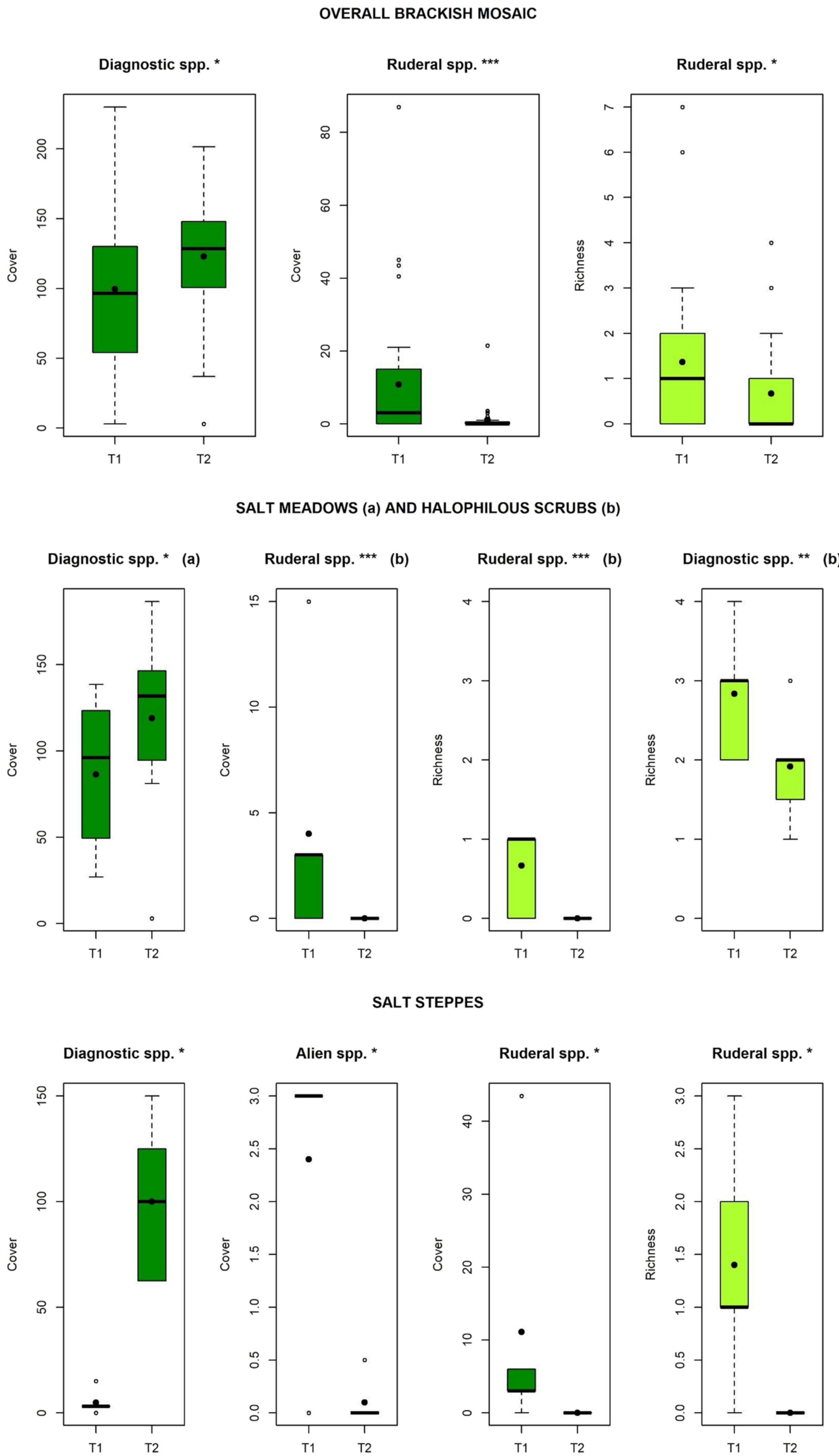
The significant increment of diagnostic species along with the significant decrease of ruderal and alien plants, are likely related to the improvement of the environmental conditions after the restoration actions carried out in 2016 by the project LIFE + MAESTRALE (NAT/IT/000262) (Prisco et al. 2017). As observed in other wetland ecosystems after naturalization interventions in America (e.g. Roman et al. 2002; Gratton and Denno 2005; Buchsbaum et al. 2006; Spieles et al. 2006; Matthews et al. 2009) or Europe (e.g. Curado et al. 2014), also in the Adriatic coast the native plant diversity tends to recover. The observed recolonization can suggest the incipient establishment of a self-sustaining ecosystem status (Zedler and Kercher 2005; Rey Benayas et al. 2009).

We observed a reduction in the number of species in the salt marsh mosaic which is probably linked to the successional process that led the salt marsh mosaic towards more natural conditions characterized by paucispecific plant communities with average richness ranging from 4 to 13 species (Géhu et al. 1984). The richness decline could respond to the interplay of different processes favored by the restoration of wetlands and the construction of boardwalks (Prisco et al. 2017), as: (a) the expansion and gain in cover of the salt tolerant native species (*Artemisia caerulescens*, *Halimione portulacoides*, *Limonium narbonense* and *Sarcocornia fruticosa*) that have morphological and physiological adaptations to live on saline environments (Moreno-Mateos et al. 2015) aided by the reconstruction of ponds and wetland (Prisco et al. 2017), (b) the reduction and loss of ruderal species (e.g. *Arundo plinii*, *Melilotus albus* and *Vicia sativa*) partially due to the decrease of human trampling disturbance prevented by dedicated paths for tourists and visitors of the area (Prisco et al. 2017), (c) the low number of alien species is likely

Table 2 Total number of species over time (T1: 2010 and T2: 2020/21) for the entire mosaic and for each EU Habitat type

	Mosaic		Salt meadows		Halophilous scrubs		Salt steppes	
	T1	T2	T1	T2	T1	T2	T1	T2
Total number of species	71	54	57	38	23	18	21	12
Number of diagnostic species	24	22	14	13	6	4	1	2
Number of ruderal species	21	10	15	9	2	0	5	0
Number of alien species	5	4	3	3	1	1	3	1

Salt meadows (EU habitat 1410); Halophilous scrubs (EU habitat 1420) and Salt steppes (EU habitat 1510*)


OVERALL BRACKISH MOSAIC
Diagnostic spp. *
Ruderal spp. ***
Ruderal spp. *
Cover
Richness
T1
T2
SALT MEADOWS (a) AND HALOPHILOUS SCRUBS (b)
Diagnostic spp. * (a)
Ruderal spp. *** (b)
Ruderal spp. *** (b)
Diagnostic spp. ** (b)
SALT STEPPES
Diagnostic spp. *
Alien spp. *
Ruderal spp. *
Ruderal spp. *

◂**Fig. 4** Boxplot comparing cover (dark green) and richness (light green) for the ecological groups (diagnostic, ruderal and alien species) in the two time steps (T1: 2010 and T2: 2020/21) for the entire brackish mosaic, Salt meadows (EU habitat 1410); Halophilous scrubs (EU habitat 1420) and Salt steppes (EU habitat 1510*). Asterisks indicate significant differences according to the Mann-Whitney post hoc test (*$p<.05$, **$p<.01$, ***$p<.001$)

due to the competition with the native halophilic species that increased their cover over time. But the persistence of some exotic species could be due to the fact that they also grow in croplands and farms in the neighboring fields (Joyce et al. 2016) and (d) the loss of plant species from other ecosystems of the close coastal dune mosaic, likely due to wetland restoration interventions. Indeed 2021 species composition of each habitat type appeared closer to mature conditions (e.g. increase of diagnostic species) and the vegetation zonation along the brackish ecological gradient seemed less fragmented. Moreover, the different EU habitats of the brackish mosaic are currently quite distinguishable in the field and their species composition seemed closer to typical halophytic species assemblage (Biondi et al. 2009; Bonari et al. 2021).

Specifically, in halophilous scrubs (EU Habitat 1420) the perennial diagnostic *Sarcocornia fruticosa* increased its cover and, as observed in previous studies (Biondi and Casavecchia 2010; Moreno-Mateos et al. 2015), this expansion may be favored by an increase of soil salinity contents. Indeed, *S. fruticosa* is well adapted to saline soils and its seeds easily germinate in a wide range of salinities (up to 1 M NaCl) (Redondo et al. 2004; Muñoz-Rodríguez et al. 2017). Higher soil salt concentrations may be also behind the significant growth of some halo-tolerant species as *Halimione portulacoides* (Álvarez-Rogel et al. 2001) in halophilous scrubs (EU 1420) and steppes (EU 1510*). As *Halimione portulacoides* is quite rare in Italian wetlands (Géhu and Biondi 1996; Corbetta and Pirone 1999; Cutini et al. 2010), its observed increment in the central Adriatic coast is of great conservation interest.

Similarly, in salt steppes (EU habitat 1510*) an increased cover of the diagnostic species *Limonium narbonense* was registered. These salt steppes occur at intermediate saline gradient values (Álvarez-Rogel et al. 2001; Baumberger et al. 2012; González-Alcaraz et al. 2014) between the hypersaline *S. fruticosa* (EU habitat 1420) and the less tolerant *Juncus* spp. meadows (EU habitat 1410).

Furthermore, on salt meadows (EU Habitat 1410), we observed an increase in the cover of the diagnostic species with medium Ellenberg salinity indicator values (*Schoenus nigricans, Juncus acutus, J. littoralis);* such species prefer lower salinity (Molina et al. 2003), and are also tolerant to summer aridity (Angelini et al. 2016), that is increasing in

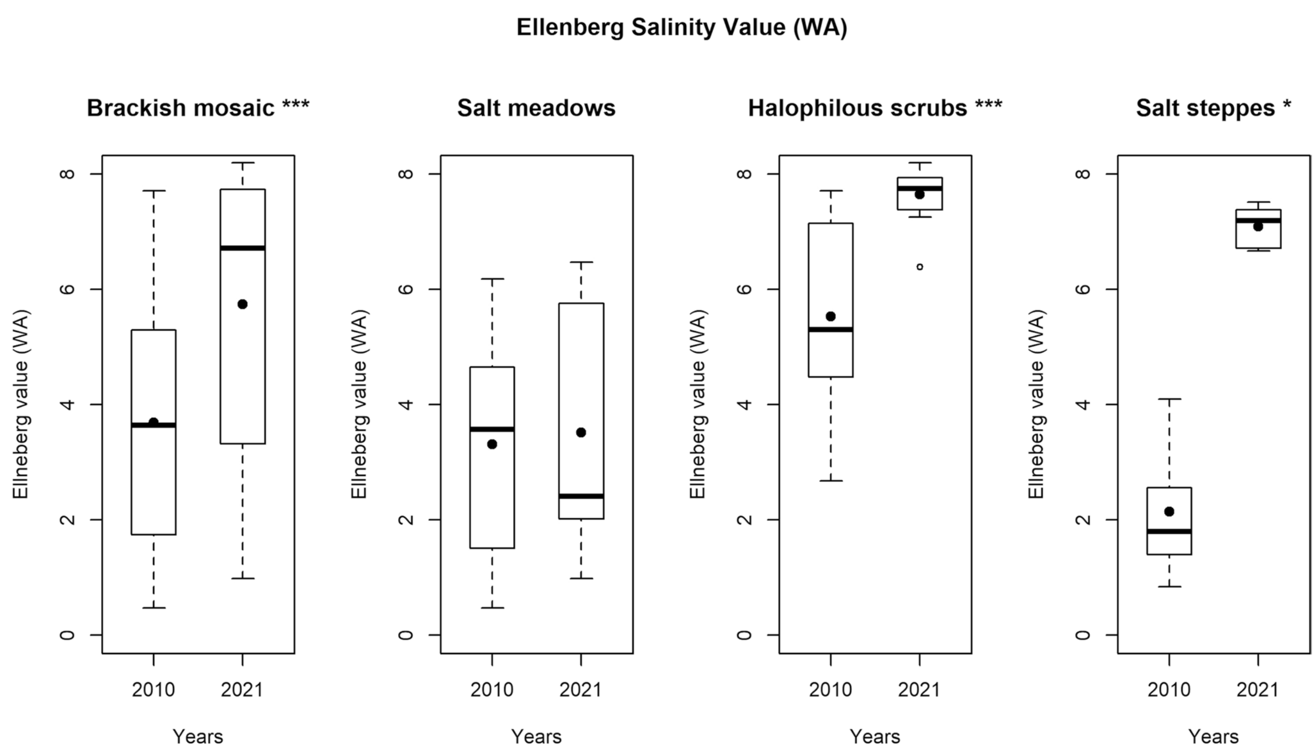


Fig. 5 Boxplot comparing Ellenberg Salinity weighted values (WA) in two time steps (T1: 2010 and T2: 2020/21) for brackish mosaic and for Salt meadows (EU habitat 1410); halophilous scrubs (EU habitat 1420) and Salt steppes (EU habitat 1510*). Asterisks indicate significant differences according to the Mann-Whitney post hoc test (*$p<.05$, **$p<.01$, ***$p<.001$)

Table 3 Plant species contribution to the temporal floristic changes and species mean cover (from 2010 to 2020) in the different EU Habitats assessed by the similarity percentage procedure (SIMPER; Clarke 1993)

	Species	Ellenberg Values		Species Contribution (%)	Cumulative Contribution (%)	Mean Cover	
		S	T			T1	T2
Salt meadows	*Plantago crassifolia*	1	8	13,24	13,24	11,3	30,5
	Juncus maritimus	6	7	10,78	24,03	22,1	13,5
	Schoenus nigricans	1	7	8,647	32,67	7,71	19,8
	Artemisia caerulescens	9	7	6,662	39,33	5,63	17,2
	Juncus littoralis	5	8	4,757	44,09	5,46	7,58
	Juncus acutus	5	8	4,755	48,85	1,5	11,1
	Elymus acutus	3	7	4,129	52,98	3,63	8,83
Halophilous scrubs	*Sarcocornia fruticosa*	8	9	18,85	18,85	34,5	58,3
	Halimione portulacoides	8	9	14,78	33,63	12	26,5
	Juncus maritimus	6	7	12,61	46,24	22,1	4,88
	Artemisia caerulescens	9	7	8,336	54,58	1,5	14,3
Salt steppes	*Limonium narbonense*	8	7	23,94	23,94	4,8	67,5
	Plantago crassifolia	1	8	15,18	39,12	40,8	0
	Halimione portulacoides	8	9	12,39	51,51	0	32,5

For each taxon, the Ellenberg ecological indicator values for salinity (S) and temperature (T) are also reported. Salt meadows (EU habitat 1410); halophilous scrubs (EU habitat 1420) and Salt steppes (EU habitat 1510*)

the Adriatic coasts (IPCC 2022) as on other coastal wetlands in the world (Osland et al. 2016).

Besides the floristic and ecological changes depicting an ongoing successional recover of the brackish communities following the restoration actions (reconstruction of water ponds and wetlands and boardwalk construction) (Prisco et al. 2017), the observed vegetation dynamics could be also linked to a variety of environmental processes (Balzan et al. 2020) affecting the Central Adriatic coast (e.g. coastal erosion, climate change) (Aucelli et al. 2018; IPCC 2022).

For instance, the observed reduction in cover of *Juncus maritimus* is likely due to its weak tolerance of aridity stress (Boscaiu et al. 2011), which has become more pronounced in the last decade. As observed on South-Eastern Europe salt rich grasslands (Eliaš et al. 2013), even in the Central Adriatic brackish area, the trajectories towards more halophytic status in some habitats of the mosaic (EU habitats 1420 and 1510*) and the increase of dry tolerant species in the salt meadows of the back-dunes (EU habitat 1410) are probably related to higher temperatures and increased summer aridity in the study area. Similar changes in species composition and structure were observed in other coastal habitats as sandy dunes (Fenu et al. 2013; Del Vecchio et al. 2015b; Prisco et al. 2016) and such variation was explained as a vegetation response to the rise of local temperatures and the reduction of summer rain-water availability.

The significant increase on Ellenberg Salinity values denotes an increase in the salt concentration in the Biferno wetland area, which is likely connected with the global warming and further favored by the ongoing coastal erosion processes affecting this section of the coast (Rosskopf et al. 2018) during the last 50 years, with an average erosion rate of −2.90 m/year. Indeed, with coastal erosion, the shoreline has come closer to the brackish grasslands, exposing them to a greater influence of salt aerosol coming from the sea. Coastal erosion seemed to be a crucial factor related to the substantial reduction in coastal dune plant cover not followed by a re-colonization of the typical species and when the amount of erosion is significant, in terms of the retraction speed of the coast line, all the habitats tends to vanish (Feagin et al. 2005; Schlacher et al. 2008; Attorre et al. 2012; Doody 2013; Ciccarelli 2014; Bertacchi et al. 2016). In particular, Prisco et al. (2016) found that in the sites of Molise shoreline affected by coastal erosion, there is a clear reduction in species richness of dune grasslands as well as a loss of the integrity of coastal vegetation zonation. The alteration of coastal dune morpho-ecological integrity which ensure inland protection (Acosta et al. 2003; Drius et al. 2019) promotes the development of more halophilic and selective environmental conditions in the back-dune wetlands. In addition, sea level rise linked to climate change may have caused the intrusion of salt-water into wetland aquifers as it was assessed in other regions (Erwin 2009).

Unfortunately, monitoring studies after restoration actions, based on ecological groups and key species abundance pattern, in salt marshes and coastal wetlands are very few so a comparative analysis between different geographical regions is not possible and as found by Moreno-Mateos et al. (2012) the recovery of wetlands following restoration as currently practiced is often slow, incomplete and in the Long term does not restore all ecosystem functions.

However, similar gain in diagnostic and native species was observed after restoration and conservation actions (the construction of boardwalk and the installation of picket fences to protect dune ecosystems from human trampling) on other coastal ecosystems (e.g. sand dunes) in the Adriatic coast (Santoro et al. 2012; Šilc et al. 2017; Prisco et al. 2021).

Conclusion

As we hypothesized, the vegetation dynamics in the analyzed wetland reflected a clear improvement in ecosystem quality after restoration, with a gain of diagnostic native species and a reduction of ruderal and alien ones. Moreover, we observed the cover increase of halophilous and thermophilous species over time.

We then found variations on ecological features and species occurrence and abundance pattern across the different EU habitats conforming the brackish mosaic: salt meadows, halophilous scrubs and salt steppes (respectively 1410, 1420 and 1510*). Such changes are most likely related to an intertwining of environmental changes (restoration actions, climate change and coastal erosion).

We observed, after the restoration action, a general improvement of the naturalness of the Biferno mouth with a successional process that led the salt marsh mosaic towards typical paucispecific plant communities. Moreover, the results demonstrated that, after adequate hydraulic work and reduced human pressure, these fragile ecosystems could be able to recover typical vegetation in the Mid and Long Term. In addition, the observed expansion of hypersaline communities may be also related to other environmental drivers as climate change (e.g. rise of local temperatures, the decline of summer precipitation) and coastal erosion that affected this section of Adriatic coast. These environmental changes likely exposed Biferno wetland to an increase on water salt concentration and to a greater influence of salt aerosol which favored the expansion of halophilous diagnostic species and the rarefaction and loss of ruderal and alien plant taxa.

The applied re-visitation approach, based on historical plots represents a cost-effective monitoring procedure that matches the need of periodical reporting requested by the European HD. We hope re-visitation monitoring studies by vegetation plots will be implemented over increasingly larger scales, in order to increase the current knowledge on vegetation dynamics after wetland restoration actions and identify the most effective approaches so as to manage and recover these fragile ecosystems.

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Declarations

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