Università
degli Studi
di Ferrara

PH.D. Course
in
Evolutionary Biology and Ecology

In cooperation with:
Università degli Studi di Parma
Università degli Studi di Firenze

CYCLE _XXXII_

COORDINATOR Prof. Guido Barbujani

Environmental changes and biodiversity governance in Po river delta area: a contribution of nature-based criteria

Scientific/Disciplinary Sector (SDS) BIO/07

Candidate
Dott. NOBILI Giovanni

Supervisor
Prof. FANO Elisa Anna

Year 2016/2019

PRESENTATION

The work originates from the professional experience of the author, Giovanni Nobili who has the role of manager of the State Natural Reserves in the study area, as Commander of the Reparto Carabinieri for Biodiversity, Punta Marina – Ravenna.

As a rule, the approach to the conservation of species and habitats follows a repetitive pattern (Ludwig, 2001). *Initially, specialists and researchers highlight the crisis phase. Legal regulations follows, with a general and local application. The subsequent management phase must take into account the limits of applicability (for example technical, economic).*

In highly dynamic environments, management interventions do not always achieve the set objectives, in particular in a long-term perspective.

Historically, the cultural and socio-economic context produces the initial intentions of protection and management. These management perspectives change over time together with environmental conditions. Human society make governance choices in an adaptive way, by selecting some natural processes (Ecosystem services) which produce benefits for man in a given historical phase (see Smail, 2017).

The ongoing environmental changes impose the challenge of redesigning the Po Delta landscape to promoting biodiversity also in light of past management decisions.

Author is primarily responsible for the choice of the research topic of this PhD thesis and for the management aspects contained in the scientific publications. The final discussion and the proposals for territorial governance are completely original and under the responsibility of the author.

Special thanks to the tutor, Prof. Elisa Anna Fano and to dr. Mattias Gaglio. Without their support this work could not have been completed. Thanks to all the other authors of the scientific publications that allowed me to participate in their works.

Bibliography

Ludwig D. (2001). The Era of Management Is Over. Ecosystems 4: 758–764.

Smail D.L. (2017). Storia profonda. Il cervello dell'uomo e l'origine della storia. Bollati Boringhieri Editore, Torino.

INTRODUCTION

Anthropogenic activities and related impacts deeply altered the ecosystems and the ecological functions they provide (Millennium Ecosystem Assessment 2005; Rocha et al. 2015). At the same time, humans strictly depend on the conservation of the environment for the support of their well-being. Moreover, such impacts act synergistically with natural environmental variations and climate change, resulting in cumulative effects that harm biodiversity conservation and economic activities (Halpern et al. 2015; Colombani et al. 2017; Pecl et al. 2017).

This is particularly evident in deltaic environments, which host a peculiar diversity of ecosystems and species, while being among the most productive areas worldwide (Yu et al. 2018; Ahmed et al. 2019). Despite their importance in terms of biological conservation and their intrinsic multifunctionality (Do & Bennet 2009; Cui et al. 2009), river deltas are typically subjected to very relevant anthropogenic alterations aimed to maximize one or few specific functions (e.g. food production).

Since deltaic soils are often highly productive, humans reclaimed significant parts of deltaic landscapes with the aim to obtain agricultural land (Gaglio et al. 2017a; Xu et al. 2017). The consequent alteration of hydraulic balances combined with increasing temperatures, altered precipitation patterns and the increasing demand for irrigation led to relevant depletion of freshwater resources (Piao et al. 2010; Elliott et al. 2014; Iglesias & Garrote 2015).

Deltaic transitional environments are exploited for fishing and intensive aquaculture. These activities produce important environmental impacts (Tamburini et al. 2019) and often require important ecological alterations. On the other hand, they generate very relevant economic incomes that support the local communities that need to face climate change and to be harmonized with conservation and other uses (Truong & Do 2018; Ahmed et al. 2019).

Additionally, deltas are naturally changing environments. Coastal habitats continuously evolve as result of sediment loads carried by rivers and sand erosion performed by the sea (Mikhailov & Mikhailova 2003; Anthony et al. 2015). The raising of sea levels is harming several coastal zones that are also subjected to natural subsidence because of soil compaction (Dang et al. 2018; Voosen 2019).

Overall, the challenge of conserving and managing deltaic areas within the complex framework of human impacts, natural environmental and climatic changes calls for proper environmental governance. Since passive protection resulted not effective in conserving both biodiversity and human well-being (Gaglio et al. 2017a, b; 2019; García Márquez et al. 2017), the adoption of a governance which includes active environmental measures is urgently needed.

In this sense, the concept of "Ecosystem Services" (ES) gained an increased popularity among scientists and environmental managers, as suitable argument to link biodiversity conservation and human well-being (De Groot et al. 2002; Maes et al. 2012). ES was defined by the Millennium Ecosystem Assessment (MEA 2005) as "the benefits people obtain from ecosystems". Anyway, ES definition and concept undergone an import evolution together with the attempts to implement them into decision making processes. A clear role of "ecological functionality" in the ES framework was introduced by the definition provided by Kremen (2005): "*Ecosystem services are the set of ecosystem functions that are useful to humans*" and after more widely implemented by Haines-Young & Potschin (2010).

The implementation of the ES approach in the governance of river deltas may represent a solution for sustainable development of these areas. However, ES operationalization urgently need investigations and data concerning ecological functions, effects of human uses and climate change on deltaic ecosystems.

In work's first part, two cases of study show effects of environmental changes on freshwater communities caused of salinity enhance. Study area is a strictly protected State Natural Reserve: salinity increase as the result of a series of causes like subsidence, arise of sea level, reclamation of freshwater wetlands, decrease in rainfall and stiffening of coastline. Then, changes in local Biodiversity is the result of many indirect causes link with human activities rather than result from management interventions. Which are effects of salinity increase? Is it possible to asses future effects of salinity changes on local Biodiversity? There is a threshold value before irreversible habitat deterioration?

In work's second part, two other cases of study concern large scales changes in habitat and water table salinity directly linked with human activities. Aquaculture in a brackish lagoon needed to increase seawater circulation leading to habitat simplification and Ecosystem Services degradation, with loss of regulating, supporting and cultural services. A particular case of study show salinity under dunes water table decrease after a pinewood fire. Sand dune capacity to maintain freshwater water table is an ES never locally recognized before. Burnt pinewood behaved like a "rigid" structure, in ecological and cultural sense. Therefore, in both study cases, restoration of ESs function need a cultural process through recognition of intrinsic natural value of pristine habitats, respectively vegetated lagoon and sand dunes. ESs approach is sustainable for governance? Are ESs stable over time? Changes in ESs over time have a governance interest? Cultural ESs could have a key role to force governance choices?

Bibliography

Ahmed, N., Thompson, S., & Glaser, M. (2019). Global aquaculture productivity, environmental sustainability, and climate change adaptability. Environmental management, 63(2), 159-172.

Anthony, E. J., Brunier, G., Besset, M., Goichot, M., Dussouillez, P., & Nguyen, V. L. (2015). Linking rapid erosion of the Mekong River delta to human activities. Scientific reports, 5, 14745.

Colombani, N., Giambastiani, B., & Mastrocicco, M. (2017). Impact of climate variability on the salinization of the coastal wetland-aquifer system of the Po Delta, Italy. Journal of Water Supply: Research and Technology-Aqua, 66(7), 430-441.

Cui, B., Yang, Q., Yang, Z., & Zhang, K. (2009). Evaluating the ecological performance of wetland restoration in the Yellow River Delta, China. Ecological Engineering, 35(7), 1090-1103.

Dang, T. D., Cochrane, T. A., & Arias, M. E. (2018). Future hydrological alterations in the Mekong Delta under the impact of water resources development, land subsidence and sea level rise. Journal of Hydrology: Regional Studies, 15, 119-133.

de Groot R. S., Wilson M. A., Boumans R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological economics, 41(3), 393-408

Do, T. N., & Bennett, J. (2009). Estimating wetland biodiversity values: a choice modelling application in Vietnam's Mekong River Delta. Environment and Development Economics, 14(2), 163-186.

Elliott, J., Deryng, D., Müller, C., Frieler, K., Konzmann, M., Gerten, D., ... & Eisner, S. (2014). Constraints and potentials of future irrigation water availability on agricultural production under climate change. Proceedings of the National Academy of Sciences, 111(9), 3239-3244.

Gaglio, M., Aschonitis, V. G., Gissi, E., Castaldelli, G., & Fano, E. A. (2017a). Land use change effects on ecosystem services of river deltas and coastal wetlands: case study in Volano–Mesola–Goro in Po river delta (Italy). Wetlands ecology and management, 25(1), 67-86.

Gaglio, M., Aschonitis, V. G., Mancuso, M. M., Reyes Puig, J. P., Moscoso, F., Castaldelli, G., & Fano, E. A. (2017b). Changes in land use and ecosystem services in tropical forest areas: a case study in Andes mountains of Ecuador. International Journal of Biodiversity Science, Ecosystem Services & Management, 13(1), 264-279.

Gaglio, M., Aschonitis, V., Pieretti, L., Santos, L., Gissi, E., Castaldelli, G., & Fano, E. A. (2019). Modelling past, present and future Ecosystem Services supply in a protected floodplain under land use and climate changes. Ecological Modelling, 403, 23-34.

García Márquez, J. R., Krueger, T., Páez, C. A., Ruiz-Agudelo, C. A., Bejarano, P., Muto, T., & Arjona, F. (2017). Effectiveness of conservation areas for protecting biodiversity and ecosystem services: a multi-criteria approach. International Journal of Biodiversity Science, Ecosystem Services & Management, 13(1), 1-13.

Halpern, B. S., Frazier, M., Potapenko, J., Casey, K. S., Koenig, K., Longo, C., ... & Walbridge, S. (2015). Spatial and temporal changes in cumulative human impacts on the world's ocean. Nature communications, 6, 7615.

Haines-Young & R, Potschin M. 2010. The links between biodiversity, ecosystem services and human well-being. Ecosyst Ecol A New Synth.110–139.

Iglesias, A., & Garrote, L. (2015). Adaptation strategies for agricultural water management under climate change in Europe. Agricultural water management, 155, 113-124.

Kremen C. (2005). Managing ecosystem services: what do we need to know about their ecology? Ecology letters, 8(5), 468-479

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J. P et al. (2012). Mapping ecosystem services for policy support and decision making in the European Union. Ecosystem Services, 1(1), 31-39

Mikhailov, V. N., & Mikhailova, M. V. (2003). Deltas as indicators of natural and human-induced changes in the regimes of rivers and seas. Water Resources, 30(6), 602-612.

MEA (Millennium Ecosystem Assessment). (2005). Ecosystems and human well-being: synthesis. Island Press, Washington,DC

Pecl, G. T., Araújo, M. B., Bell, J. D., Blanchard, J., Bonebrake, T. C., Chen, I. C., ... & Falconi, L. (2017). Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. Science, 355(6332), eaai9214.

Piao, S., Ciais, P., Huang, Y., Shen, Z., Peng, S., Li, J., ... & Friedlingstein, P. (2010). The impacts of climate change on water resources and agriculture in China. Nature, 467(7311), 43.

Rocha, J., Yletyinen, J., Biggs, R., Blenckner, T., & Peterson, G. (2015). Marine regime shifts: drivers and impacts on ecosystems services. Philosophical Transactions of the Royal Society B: Biological Sciences, 370(1659), 20130273.

Tamburini, E., Fano, E. A., Castaldelli, G., & Turolla, E. (2019). Life Cycle Assessment of Oyster Farming in the Po Delta, Northern Italy. Resources, 8(4), 170.

Truong, T. D., & Do, L. H. (2018). Mangrove forests and aquaculture in the Mekong river delta. Land use policy, 73, 20-28.

Voosen, P. (2019). Seas are rising faster than believed at many river deltas. Science. 363, (6426), 441

Xu, Y., Cai, Y., Sun, T., & Tan, Q. (2017). A multi-scale integrated modeling framework to measure comprehensive impact of coastal reclamation activities in Yellow River estuary, China. Marine pollution bulletin, 122(1-2), 27-37.

Yu, S., Cui, B., & Gibbons, P. (2018). A method for identifying suitable biodiversity offset sites and its application to reclamation of coastal wetlands in China. Biological conservation, 227, 284-291.

Part 1

Strictly protected State Nature Reserve “Bosco della Mesola” - faced on Sacca di Goro, a brackish lagoon at south Po river delta - host small water bodies with peculiar biodiversity.

Despite high level of protection, aquatic macroinvertebrate communities are threatened by indirect environmental changes such as saltwater intrusion.

Macroinvertebrate communities are resilient to moderate increases of salinity, but salinization increase to polihaline levels causes dramatic biodiversity losses and a drastic community simplification in terms of functional groups composition.

Since subsidence, climate change and anthropogenic activities are expected to exacerbate salinization, management measures are required for the conservation of aquatic biodiversity in coastal small wetlands.

Muresan A.N., Gaglio M., Aschonitis V., Nobili G., Castaldelli G., Fano E.A. (2019) Structural and functional responses of macroinvertebrate communities in small wetlands of the Po delta with different and variable salinity levels. Submitted to *Estuarine, Coastal and Shelf Science*

The wetland vegetation of the Nature Reserve is arranged according to a salinity gradient, from salt-free habitats to strongly saline habitats. The saline habitats have high nutrient levels, due to the influx of nitrate-rich saltwater from adjacent lagoon. Water-table depth and concentration of dissolved nutrients in the water were the main factors structuring waterplant vegetation.

The main driver of future changes in the wetland vegetation is the ongoing increase in salinity levels which may enhance expansion of halophilic species and communities, thus outcompeting locally rare freshwater species.

If nutrient, especially nitrate, load further increases in the next future, this may exert negative effects on wetland species and communities preferring nutrient-poor habitats.

Gerdol R., Brancaleoni L., Lastrucci L., Nobili G., Pellizzari M., Ravaglioli M., Viciani D. (2018) Salinization and eutrophication negatively affect waterplant diversity in a coastal Nature Reserve. Implications for nature conservation. *Estuaries and Coasts 41(3)* https://doi.org/10.1007/s12237-018-0396-5

Structural and functional responses of macroinvertebrate communities in small wetlands of the Po delta with different and variable salinity levels

Muresan A.N.[1], Gaglio M.*[1], Aschonitis V.[2], Nobili G.[1], Castaldelli G.[1], Fano E.A.[1,3]

[1]Department of Life Sciences and Biotechnology, University of Ferrara. Via L. Borsari 46, 44121 Ferrara, Italy
[2]Institute of Soil and Water Resources, Hellenic Agricultural Organization Demeter. Thermi, 57001, Thessaloniki, Greece
[3]National Interuniversity Consortium for Marine Sciences (CoNISMa). Piazzale Flaminio, 9, 00196 Roma, Italy
*corresponding author: gglmts@unife.it

Abstract

Coastal areas often host small water bodies described by high levels of biodiversity, which are threatened by environmental changes such as saltwater intrusion. This work evaluates the salinization effects on macroinvertebrate invertebrate communities of 16 permanent small wetlands (ponds) located in a coastal Mediterranean forest in Northern Italy, characterized by different salinity levels. From a preliminary multivariate analysis (CCA), salinity was detected as the only important parameter affecting taxa distribution. Thus, taxonomic diversity, biological and functional traits of macroinvertebrate communities were analyzed considering three salinity classes (freshwater, oligo-mesohaline and polihaline). The threshold indicator taxa analysis (TITAN) was used for detecting changes in taxa abundance distributions within the salinity range and for assessing synchrony among taxa abundance change points as evidence of community thresholds. Taxonomic diversity indices and functional/biological traits among the three salinity classes were also compared. The findings demonstrated that ponds' macroinvertebrate communities are resilient to moderate increases of salinity, but salinization increase to polihaline levels causes dramatic biodiversity losses and a drastic community simplification in terms of functional groups composition. Since climate change and anthropogenic activities are expected to exacerbate salinization, management measures are required for the conservation of aquatic biodiversity in coastal small wetlands.

Keywords:
Macrozoobenthos, salinization, polihaline, TITAN, taxonomic diversity indices, functional traits.

Introduction

Climate change is expected to generate significant ecosystems modifications worldwide, causing changes in environmental conditions and ecosystem processes (Scholze et al., 2006).

Transitional environments as deltaic areas are markedly vulnerable to climatic and environmental changes because of their sensitive hydrological balances and the increasing presence of human settlements and activities (Gu et al., 2011; House et al., 2016; Gaglio et al., 2019). Deltas host different typologies of aquatic environments, as river branches, wetlands, salt marshes and mudflats, that bestow them a high environmental diversity and provide several ecological functions (Gaglio et al., 2017a). Since the changes of environmental and climatic conditions are predicted to affect aquatic biotic and abiotic components underpinning deltas ecological functions, the investigation on and the management of consequences on river deltas are of paramount importance to guarantee biodiversity conservation and human well-being.

The deltaic areas of Mediterranean region are particularly vulnerable to climate change, as an increase of temperature and a decrease of precipitation has been already identified (Cramer et al., 2018). The combined effects of raising water demand and water scarcity will significantly affect future water availability in Mediterranean basins (Saadi et al., 2015). Raising temperatures will increase evapotranspiration rates, while the reduced amount of rainfall will exacerbate plant water stress, requiring higher amounts of water withdrawals for crop irrigation. This trend will significantly influence the wetlands biota by favoring species more tolerant to drought and variable conditions at the expense of more specialist ones (Johansen et al., 2018). At community level, the responses to such phenomena can be observed by investigating macroinvertebrate assemblages. In Mediterranean coastal systems, climate change was observed to cause losses of taxonomic and functional diversity in macroinvertebrate communities (Cardoso et al., 2008; Pitacco et al., 2018).

The resilience of aquatic ecosystems to environmental changes relies on the capacity of aquatic biota to reestablish living communities after perturbations (Downing and Leibold, 2010; Schaffner, 2010). Nonetheless, the newly established communities may present different levels of taxonomic and functional diversity, as a consequence of adaptation to the new environmental conditions (Macleod et al., 2008). This may lead to a general loss of both biodiversity and capacity to respond to additional perturbations, which further threaten these ecosystems.

Aquatic ecosystems are widely subjected to increasing pressures harming their ecological quality. The intensification of human activities, such as agriculture, aquaculture, water withdrawals, and the ongoing climatic changes can lead to detrimental effects in their ecological standards and capacity to support human well-being (Blann et al., 2009; Day et al., 2008; Gaglio et al., 2019; Xenopoulos et al., 2005).

Salinization is one of the main stressors affecting deltaic areas that occur as a synergic effect of climatic changes and anthropogenic effects (Colombani et al., 2016). These transitional environments are highly sensitive to changes of both terrestrial and marine components (Harley et al., 2006), as well as climatic factors (Nielsen and Brock, 2009; Scavia et al., 2002). Different natural and human-related factors concur to the increase of salinity levels of water bodies. Primary salinization is referred to natural salt accumulation from rainwater and leached from terrestrial sources unaffected by human activities. Contrary, secondary salinization is caused by human-induced mechanisms, such as vegetation clearance, intensive irrigation, river regulation and land reclamation (Gaglio et al., 2017a; Herbert et al., 2015). Unlike primary salinization, secondary salinization occurs on a time frame of decades or less, under the consequence of human alteration of hydrological cycles (Herbert et al., 2015).

Salinity affects the presence of species both directly, for example through osmoregulation physiology, and indirectly, by influencing abiotic factors and biotic interactions (Liancourt et al., 2005; Pinder et al., 2005; Withers, 1992). Freshwater invertebrates can withstand small salinity increases maintaining constant iso-osmotic conditions between haemolymph and external solutions. With the increase of external solute concentrations, many freshwater invertebrates suffer from dehydration, while salt-tolerant aquatic invertebrates respond to the increased salinity by adopting osmoregulation strategies for maintaining constant the osmotic concentration of body fluids (Evans, 2008).

Salinity also influences competition among species. The role of competition may vary with the level of abiotic stress, such as salinity, as a result of different tolerance and competitive response of species (La Peyre et al. 2001; Liancourt et al. 2005). Variation of salinity levels mediates trophic cascade by influencing predators' abundance, thus altering their top-down control on preys (Cañedo-Argüelles et al., 2016; Herbst, 2006; Herbst and Blinn, 1998). Moreover, salinity may interact with other environmental factors to influence species composition (Larson and Belovsky, 2013).

The effects of increasing salinity levels on macroinvertebrate communities were described in literature for coastal marine habitats (Zettler et al., 2014), estuarine (Little et al., 2017; Ritter et al., 2005) and lagoons (Como et al., 2018), but very few studies exist for pond systems (Boix et al., 2008).

Ponds are small and isolated ecosystems ranging from 1 m^2 to few hectares, which temporarily or permanently store water (De Meester et al., 2005). They are biodiversity hotspots both in terms of species composition and biological traits, and provide ecosystem services to support human well-being (Céréghino et al., 2014, 2012). Pond ecosystems host a large number of species and rare species, exceeding those of other aquatic ecosystems such as streams and lakes (Williams et al., 2004). Consequently, despite their limited dimensions, ponds are fundamental features for biodiversity conservation at the landscape scale (Céréghino et al., 2014; Coccia et al., 2016). In fact, in addition to the diversity of their own communities, ponds also play a role as stepping stones for aquatic mobile species, thus mitigating the negative effects of habitat fragmentation and increasing aquatic habitats' connectivity (Pereira et al., 2011; Rothermel, 2004).

Ponds are ideal sentinels and early warning systems of environmental changes due to their sensitivity to environmental changes, which is associated to their small size and the high interaction with the groundwater (De Meester et al., 2005). Particularly, pond macroinvertebrate communities can be sensitive indicators of how living communities respond to environmental variations in coastal systems.

However, the set of environmental variables governing pond community composition and species traits is specific for each climatic/biogeographic region (Céréghino et al., 2012; De Marco et al., 2014; Ruhí et al., 2013). Therefore, site-specific investigations are needed to assess the response of macroinvertebrate communities of pond systems.

The aim of this study is to investigate macroinvertebrate communities' changes within a salinity range in a coastal permanent system of ponds located in the Po river delta (Northern Italy), by means of taxonomic and functional (i.e. biological traits) analyses. Overall, the assessment of macroinvertebrate community responses to salinity can shed light on how biodiversity and ecological functions of aquatic ecosystems of deltaic areas are expected to change as consequences of future environmental and climatic modifications.

Material and Methods

Study area

This study was carried out at "Bosco della Mesola" (44°50′ 28′′ N, 12°15′ 12′′ E), a National Natural Reserve of 1058 ha located in the province of Ferrara (Northern Italy), with an altitude ranging from -1 to +3 m a.s.l (Fig. 1).

The Nature Reserve "Bosco della Mesola" hosts a ponds' system formed by small water bodies mainly located among the ancient dunes, where water stagnation is fed either by rainfall or by the coastal aquifer that is hydrologically connected to the sea or, in a lesser extent, by incoming water from the canal system. Their aquatic biota includes macroinvertebrate, amphibian and reptile communities. No fishes were observed in ponds. The ponds' system is characterized by different salinity levels. Water bodies near the shoreline are influenced by seawater rising from groundwater table and by occasional floods, while the ponds located northern receive freshwater from drainage canals.

The surrounding area was strongly altered by human interventions during the last century, e.g. wetland reclamations, that affected the local hydrological balance (Gaglio et al., 2017a). Additionally, the area is subjected to subsidence that causes the progressive intrusion of saltwater towards the inner part of the Reserve, due to the progressive difficulties in recharging water table with freshwater, affecting the communities of living organisms of local water bodies (Gerdol et al., 2018). Moreover, climatic changes could speed up the salinization level of ponds' system, thus exacerbating such impacts. Macroinvertebrate communities were studied as suitable indicators for detecting salinization effects on aquatic biota of small wetlands of deltaic areas, such as the case of the "Bosco della Mesola" ponds in the river Po delta.

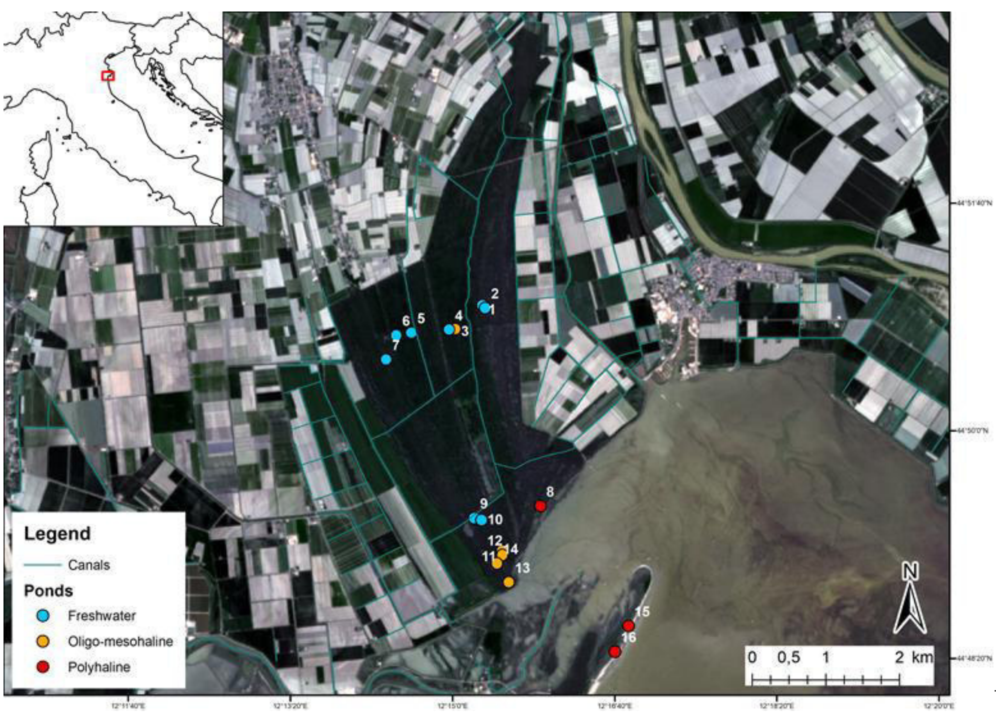


Fig.1 The study area of the 16 ponds where samplings were performed.

Sampling procedures

The macroinvertebrate communities of 16 permanent coastal ponds were sampled during May 2017. The 14 ponds are located inside the Nature Reserve of "Bosco della Mesola" in north-eastern Italy. Moreover, two additional ponds were sampled outside the Reserve, located southern in a coastal outlet (Fig. 1). No temporary ponds were considered in the analysis. The salinity levels ranged between 0.2 and 29.3 psu. The ponds were classified into three salinity classes according to the classification of Por (1972): freshwater (<0.5 psu), oligo-mesohaline (0.5-18 psu) and polyhaline (>18 psu). Benthic macrofauna was collected sweeping a 40 cm-wide D-frame hand net (mesh size = 500 μm) for an area of 1 m^2. Samplings were carried out in 3 different sites within each pond, one in central and two in the littoral parts, to capture intra-pond variability. Animals were preserved in 4% formalin solution and later identified in laboratory up to the genus level, and when it was no feasible, family level was reached. Hence, the respective biological/functional traits (feeding, mobility, adult life habitat, body size, life span, reproductive frequency, habitat choice) were attributed to each taxon by means of bibliographic information using the databases of Horton et al. (2017) and (MarLIN, 2006). When observed taxa were not covered by these sources, Thorp and Covich (2010) was used as alternative literature. Table 1 provides the 39 taxa observed in the sampling ponds.

Eight environmental factors were measured/assessed during samplings (Table 2). Water temperature, oxygen saturation and salinity were measured using a multi-parameter probe (YSI Model 85). Pond surface and water depth were also measured in situ. Euclidean nearest-neighbor distance (ENN) was computed using Fragstat 4.2 (McGarigal, 2014) as a measure of pond isolation. ENN can be defined as the shortest straight-line distance between the focal patch and its nearest neighbor of the same class. ENN was calculated for each pond, also considering the presence of other water bodies, such as channels and coastal lagoons. Aquatic vegetation and canopy coverage above the pond (%) were measured by visual assessment of photographs taken on site.

Table 1. Taxa observed in the 16 sampling ponds of Bosco della Mesola.

Phylum	**Order**	**Family**	**Family abbr.**	**Genus**
Anellida	Haplotaxidae	Naitidae	Nai	-
Anellida	Hirudinida	Hirudinidae	Hir	*Hirudo*
Anellida	Polychaeta	Nereidae	Ner	*Nereis*
Crustacea	Amphipoda	Corophiidae	Cor	*Corophium*
Crustacea	Amphipoda	Gammaridae	Gam	*Echinogammarus*
				Gammarus
Crustacea	Decapoda	Cambaridae	Cam	*Procambarus*
Crustacea	Decapoda	Portunidae	Por	*Carcinus*
Crustacea	Isopoda	Asellidae	Ase	*Asellus*
Crustacea	Isopoda	Sphaeromatidae	Sph	*Sphaeroma*
Insecta	Coleoptera	Dytiscidae	Dyt	-
Insecta	Coleoptera	Haliplidae	Hal	-
Insecta	Coleoptera	Hydrophilidae	Hyd	*Helochares*
Insecta	Diptera	Ceratopogonidae	Cer	-
Insecta	Diptera	Chaoboridae	Chi	*Chaoborus*
				Chironomus
				Cladopelma
				Cryptochironomus
				Parachironomus
				Polypedilum
Insecta	Diptera	Orthocladiinae	Ort	*Orthocladius*
Insecta	Diptera	Tanyponidae	Tan	*Procladius*
				Psectrotanypus
				Tanypus
Insecta	Ephemeroptera	Baetidae	Bae	*Baetis*
				Cloeon
Insecta	Heteroptera	Corixidae	Crx	*Cymatia*
				Micronecta
Insecta	Heteroptera	Nepidae	Nep	*Nepa*
Insecta	Lepidoptera	Crambidae	Cra	*Cataclysta*
				Paraponyx
Insecta	Odonata	Lestidae	Les	*Chalcolestes*
Insecta	Odonata	Libellulidae	Lib	*Libellula*
Mollusca	Bivalvia	Corbulidae	Crb	*Corbula*
Mollusca	Gasteropoda	Bithyniidae	Bit	*Bithynia*
Mollusca	Gasteropoda	Lymnaeidae	Lym	*Lymnae*
Mollusca	Gasteropoda	Physidae	Phy	*Physa*
Mollusca	Gasteropoda	Planorbidae	Pla	*Planorbis*
Nematoda	-	-	Nem	-

Table 2. Environmental parameters used in this study for analyzing taxa responses.

Parameter	Unit	Transformation	Abbrev.	Max	Min	Mean	SD
Temperature	°C	Log (x+1)	Temp	29.6	12.4	17.2	5.7
Oxygen saturation	%	arcsin $(x/100)^{0.5}$	O2	100.0	8.5	56.3	30.5
Surface area	m²	Log (x+1)	Area	6191.5	24.0	641.9	1514.0
Depth	cm	Log (x+1)	Dep	63.0	10.0	34.3	16.6
Salinity	psu	Log (x+1)	Salt	29.3	0.2	6.3	9.8
Vegetation	%	arcsin $(x/100)^{0.5}$	Veg	40.0	0.0	4.1	10.8
Canopy	%	arcsin $(x/100)^{0.5}$	Can	50.0	0.0	14.4	18.6
ENN	m	Log (x+1)	ENN	356.5	28.3	103.9	88.0

Methods of analysis

Detection and evaluation of salinity gradient effects on taxa abundance

All environmental parameters and taxa abundance were transformed to reduce normality departures. Environmental parameters, which are ratios/percentages, were transformed using arcsin$(x/100)^{0.5}$ while the rest environmental parameters and taxa abundance (ind. m^{-2}) using log(x+1) (Aschonitis et al., 2016). Spearman correlations were performed among environmental variables of Table 2.

Multiple gradient analysis was performed for assessing the effect of multiple descriptors (environmental parameters) on multiple target variables (taxa) using Canonical Correspondence Analysis (CCA) (Lepš and Šmilauer, 2003; ter Braak and Smilauer, 2002). CCA was performed using CANOCO 4.5, based on target variables correlations and their standardized scores (ter Braak and Smilauer, 2002). The method was applied following the same steps as those described in a similar case study (Aschonitis et al., 2016) and significant descriptors were identified using CANOCO's forward selection procedure and Monte Carlo permutation test (499 permutations) (a default option in the CANOCO software). Collinear variables with variance inflation factor VIF>8 (Zuur et al., 2007) or variables with statistical significance $p>0.5$ were excluded from multiple gradient analysis. The multivariate analysis was performed for macroinvertebrate communities at the family level, which is considered a sufficient level for invertebrate community analysis (Gayraud et al., 2003). This analysis detected that the only statistically significant descriptor variable at $p<0.05$ level is salinity. For this reason, more details regarding this analysis are not presented in this study because a method of single gradient analysis is more appropriate.

The threshold indicator taxa analysis TITAN (Baker and King 2010) is a single gradient analysis method that is used in ecological studies for detecting changes in taxa abundance distributions along a unique environmental gradient (i.e. salinity) and for assessing synchrony among taxa abundance change points as evidence of community thresholds (Baker and King 2010). TITAN uses bootstrapping for estimating purity and reliability (measures of statistical significance for this type of analysis) as well as uncertainty of change points related to individual taxa abundances along the salinity gradient. Usually, a cut off value of 95% is used in both purity and reliability criteria for identifying statistical robust responses of taxa abundance versus an environmental gradient (i.e. salinity). The purity cut off value defines what is considered a pure response direction. A purity value of 0.95 indicates that 95% of the results from bootstrap replicates agree with the observed response direction. The reliability cut off value defines what is considered a reliable response magnitude. A reliability value of 0.95 indicates that 95% of the results from bootstrap replicates have a IndVal *p*-value less than or equal to 0.05, indicating a response magnitude at a given change point location that is significantly different from what would expect from random permutation (Baker and King 2010) {for more explanations about IndVal *p*-value, see Dufrêne and Legendre (1997) and Baker and King (2010)}. In this study, the purity and reliability values were estimated and the response plots versus salinity gradient were developed for all taxa without considering cut off values. The TITAN analysis was performed with TITAN2 version 2.1 (Baker et al. 2015) in R language using 500 random permutations of taxa abundances and creating 1000 new bootstrap datasets created by resampling the observed data with replacement.

Effects of salinity on biological traits and diversity measures

The differences among population indices such as species richness (S), Shannon's diversity index (H´) and Pielou evenness index (J´) among the three salinity classes of ponds (i.e. Freshwater, Oligo-mesohaline, Polihaline) were evaluated.

Significant differences in biological traits among the three respective salinity classes of ponds were evaluated through the comparison of proportions with χ^2 test for P-value $\leq$ 0.01. For each trait, the comparison was performed between the proportion of each trait type of the three salinity classes versus the overall proportion of the remaining types in the respective salinity class (e.g. for the "adult life habitat" trait, the significance of difference of "aquatic" type in the three salinity classes was evaluated comparing "aquatic" vs. "aeric"). The null hypothesis was that the proportion of the two trait types (or of the proportion of the one trait type versus the remaining ones) did not differ over the three salinity classes. Afterward, an analysis of means (ANOM) plot with 99% confidence was applied to provide indications about the direction of the significant differences based on the deviation from the grand mean of the ANOM plots (Fedrigotti et al., 2016; Gaglio et al., 2017b).

Results

Effect of salinity gradient on taxa abundance

As it was mentioned in the Methods section, CCA detected that the most important parameter from Table 2 but also the only parameter with statistical significance at p<0.05 level describing the taxa variance was salinity. The Spearman correlations among the environmental variables (Table 3) showed that salinity is significantly positively correlated with temperature, oxygen saturation and pond area while it is significantly negatively correlated with canopy coverage. Thus, salinity can also be used as a general surrogate descriptor of the aforementioned environmental parameters.

Table 3. Spearman correlations among environmental variables of Table 2.

	Temp	O2	Area	Dep	Salt	Veg	Can	ENN
Temp	1							
O2	0.531**	1						
Area	0.639**	0.472*	1					
Dep	-0.309	-0.196	-0.149	1				
Salt	0.848**	0.732**	0.714**	-0.172	1			
Veg	-0.207	0.261	-0.146	-0.197	-0.197	1		
Can	-0.422	-0.555**	-0.255	0.490*	-0.478*	-0.292	1	
ENN	0.060	0.215	0.407	-0.074	0.319	0.021	0.155	1

**,* Statistical significance for $p<0.01$ and $p<0.05$ level, respectively.

The results of TITAN analysis that concern the effects of salinity gradient on taxa abundance are provided in Table 4 and Fig. 2. Table 4 provides the indicator change point (CP) along the salinity gradient (median of 1000 bootstrap replicates), the purity and reliability % of CP, and the response (positive + or negative -) of each taxon versus the increase of salinity gradient. Fig. 2 shows the declining taxa on the left axis and the increasing taxa on the right axis. The observed change point (median of bootstrap replicates) is indicated by the circular symbol, while the horizontal lines behind each circular symbol describe the 5-95% quantiles from the bootstrapped change point distribution. Taking into account Table 4 and Fig. 2, the following observations were made:

- 18 out of 28 taxa showed purity >95%, while 9 out of 28 taxa showed both purity and reliability >95% (other taxa showing only reliability >95% were not observed). From the nine highly pure (>95%) and highly reliable (>95%) taxa, four showed a positive and three a negative response versus the salinity gradient (in the range of salinity where they observed).
- The most sensitive taxa to salinity were found to be Cambaridae (Cam), Sphaeromatidae (Sph), Dytiscidae (Dyt), Haliplidae (Hal), Orthocladiinae (Ort), Bithyniidae (Bit) and Planorbidae (Pla), showing negative responses for indicator change points (CP) of salinity <0.5 psu.

- The taxa of highest tolerance to salinity were Naitidae (Nai), Nereidae (Ner), Chaoboridae (Chi), Corbulidae (Crb) and Nematoda (Nem) showing positive responses for indicator change points (CP) of salinity >10 psu.

Table 4. Indicator change point (CP) along the salinity gradient (median of bootstrap replicates), purity and reliability of CP, and response (positive + or negative -) of each taxon versus the increase of salinity gradient according to TITAN analysis.

Taxon	CP (psu)	Purity	Reliability	Response group	Taxon	CP (psu)	Purity	Reliability	Response group
Nai**	14.25	0.978	0.958	+	Ort	0.3	0.886	0.654	-
Hir**	8.15	0.998	0.985	+	Tan**	3.05	0.994	0.985	-
Ner**	10.2	1	1	+	Bae	4.8	0.82	0.871	-
Cor*	8.15	0.962	0.751	+	Crx*	8.15	0.96	0.939	-
Gam**	0.5	1	1	+	Nep	1.85	0.87	0.306	+
Cam*	0.3	0.967	0.76	-	Cra	1.85	0.665	0.668	-
Por*	8.15	0.962	0.719	+	Les	0.45	0.733	0.922	+
Ase**	2.6	1	1	-	Lib	1.85	0.87	0.317	+
Sph	0.25	0.862	0.807	-	Crb*	14.25	0.982	0.938	+
Dyt	0.25	0.921	0.857	-	Bit*	0.3	0.967	0.734	-
Hal*	0.3	0.967	0.734	-	Lym**	4.8	0.997	0.97	+
Hyd	3.05	0.771	0.841	-	Phy*	4.8	0.964	0.616	+
Cer*	0.8	0.982	0.936	+	Pla**	0.4	1	0.997	-
Chi**	10.2	1	1	+	Nem	22.5	0.754	0.694	+

*Statistically significant purity (>95%) or reliability (>95%)

**Statistically significant purity (>95%) and reliability (>95%)

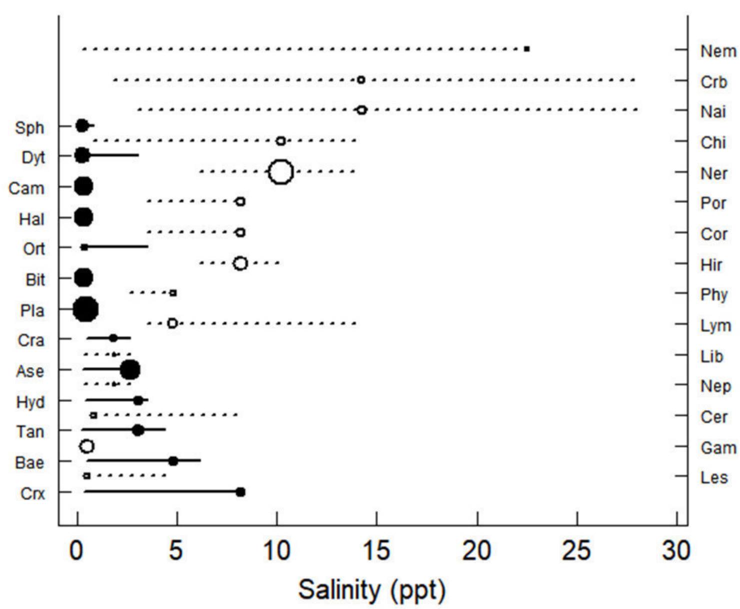


Fig.2 CP response plots of each taxon versus the salinity gradient according to TITAN analysis.

Effect of salinity on diversity indices and biological traits

The values of population indices such as species richness (S), the Shannon's diversity index H´ and the Pielou evenness index J´ of the three salinity classes of ponds (i.e. Freshwater, Oligo-mesohaline, Polihaline) are given in Table 5.

According to Table 5, it is observed that species richness of polihaline class is much lower from the other two pond salinity classes. A substantial difference in species richness among the freshwater and the oligo-mesohaline class was not observed. Similar results were also observed in the case of Shannon index. On the other hand, substantial differences in the values of evenness index were observed among freshwater and oligo-mesohaline classes, while differences were not observed between the oligo-mesohaline and the polyhaline. The above indicates that the evenness index was more sensitive to salinity than the species richness and Shannon diversity.

Considering the analysis of differences in biological traits among the three respective salinity classes, all the considered traits showed significant variations (Table 6), highlighting taxa sensitiveness to salinity in terms of functional and biological characteristics. Significant variations in the 18 out of the 22 analyzed trait types appeared in the transition from oligo-mesohaline to polyhaline class (Table 7).

Table 5. Average values of S, H' and J' observed in each salinity class.

Pond salinity classes	**Species richness (S)**	**Shannon's Diversity (H')**	**Pielou Evenness (J')**
Freshwater	6.375	0.932	0.572
Oligo-mesohaline	7.800	0.878	0.509
Polihaline	3.333	0.509	0.508

Table 6. Significant variations in functional and biological traits observed at increasing salinity

Trait	***Trait type increasing in %***
Feeding	Deposit Feeders
Mobility	Burrowers
Adult Life Habitat	Aeric
Body Size	Medium
Life Span	Medium
Reproductive frequency	Semelparous
Habitat Choice	Generalist

Table 7. Analysis of means (ANOM) for the different functional attributes in the three salinity classes. Statistical significances are highlighted in bold ($p<0.05$). The three codes a, b and c denote the respective location of the proportion values (above, inside and below the upper and lower 95% confidence limits) of the three salinity classes for each trait type.

Trait type	***χ^2 (df=2)***	***P value***	***Freshwater***	***Oligo-mesohaline***	***Polihaline***
FEEDING					
Predator	1035.52	**<0.001**	a	a	c
Grazer	63.86	**<0.001**	a	b	b
Shredder	6158.39	**<0.001**	a	a	c
Scraper	814.6	**<0.001**	a	a	c
Deposit feeder	6038.5	**<0.001**	c	c	a
Filter feeder	8.3	**0.0158**	a	b	b
MOBILITY					
sessile	8.3	**0.0158**	a	b	b
swimmer	8245.1	**<0.001**	a	a	c
borrower	7195.53	**<0.001**	c	c	a
walker	3242.25	**<0.001**	a	a	c
ADULT LIFE HABITAT					
aquatic	830.57	**<0.001**	a	a	c
aeric	830.57	**<0.001**	c	c	a
BODY SIZE (g AFDW)					
small (<0.01)	2.71	0.2575	b	b	b
medium (0.01-0.05)	1723.75	**<0.001**	c	c	a
large (>0,05)	134.61	**<0.001**	a	a	c
LIFE SPAN (years)					
short (< 1)	7709.88	**<0.001**	a	a	c
medium (1-5)	8024.5	**<0.001**	c	c	a
large (>5)	1518.22	**<0.001**	a	a	c
REPRODUCTIVE FREQUENCY					
Iteroparous	1905.59	**<0.001**	c	c	a
Semelparous	1905.59	**<0.001**	a	a	c
HABITAT CHOICE					
generalist	1679.05	**<0.001**	c	c	a
specialist	1679.05	**<0.001**	a	a	c

Discussion

Small wetlands are often non-investigated biodiversity hotspots that should be instead regarded with great attention under a conservation perspective (Viaroli et al., 2016). Particularly, the permanent ponds system of Bosco della Mesola hosts a high number of taxa (39) if compared with ponds surrounded by other land use types. Hill et al. (2016) investigated macroinvertebrate diversity in urban, arable and floodplain ponds, recording 22, 30 and 32 taxa, respectively. The positive effect of surrounding forest is further corroborated by the comparison of our data with those of Bazzanti (2015), who found a similarly high number of taxa in temporary and permanent ponds located in a Mediterranean Tyrrenian coastal forest ecosystem. However, biodiversity of coastal ponds is seriously threatened by the pressing environmental changes occurring in these areas. Our findings highlighted the key role of salinity in shaping macroinvertebrate community compositions in coastal permanent ponds and that these communities could be significantly affected by salt water intrusion, which causes a relevant simplification of taxonomic and functional diversity.

This finding is consistent with other studies describing community variations in different aquatic environments (Castillo et al., 2018; Little et al., 2017; Piscart et al., 2005b; Zettler et al., 2014), including temporary wetlands (Waterkeyn et al., 2008), and with Kefford et al. (2016) who found a limited tolerance of aquatic insects to salinity increases in freshwater habitats.

Despite the fact that CCA detected salinity as the only statistically significant parameter of Table 2 for describing taxa variation, the Spearman correlations among the environmental variables (Table 3) showed that salinity is significantly positively correlated with temperature, oxygen saturation and pond area while it is significantly negatively correlated with canopy coverage. Thus, the observed salinity effect on macroinvertebrate communities partly includes some effects of these parameters. Water depth and pond isolation (described through ENN) were the least important parameters affecting the communities (according to preliminary CCA analysis) and the least associated parameters to salinity. The insignificant effect of water depth was probably the result of the generally shallow profile of all analyzed ponds. In the case of pond isolation, its insignificant effect can be attributed to the fact that the recruitment of flying insects may be guaranteed by the surrounding forested landscape rather than by other water bodies.

Taxonomic diversity indices were negatively influenced by salinity depicting an overall loss of taxa diversity due to saltwater intrusion. Although freshwater ponds show higher values of taxonomic diversity indices, oligo-mesohaline ones hosted higher taxa richness according to Table 5. This partially corroborates to the 'intermediate disturbance hypothesis' (Connell, 1978) of salinity gradient proposed by Piscart, Lecerf, et al. (2005), according to which an intermediate level of salinity promotes a higher level of biodiversity because of the co-occurrence of both halotolerant and freshwater species. Higher values of taxa richness detected in oligo-mesohaline ponds are due to the occurrence of more insect and crustacean taxa. However, this does not equal to higher values of taxonomic diversity indices because of the increase of individual densities along the salinity gradient. The increasing proportions of crustacean taxa and crustaceans/insects ratio along salinity levels observed by Boix et al. (2008) were confirmed by our findings only up to oligo-mesohaline levels, while opposite trends were observed at polihaline level.

As it was shown from Table 6, salinization of permanent ponds above the polihaline level leads to a drastic loss of functional diversity. The results highlighted that functional traits' analysis is sensitive to depict community responses mainly to high levels of salinity transitions. This also provides evidences that functional variables can be used as indicators of drastic environmental perturbations and should always be studied when assessing disturbance impacts on biota (Sandin and Solimini, 2009).

The salinization of the ponds system toward polihaline conditions caused the shift of macroinvertebrate communities towards assemblages dominated by r-strategist taxa (*sensu* Pianka 1970), which in general are generalist taxa with smaller body size, shorter life span and higher reproductive frequency. Polihaline ponds are dominated by borrowing deposit feeder taxa, mainly Chironomidae, Tubificinae and Nereidae. All other taxa with different feeding and mobility attributes were rarely found in polihaline conditions. Piscart et al. (2006) found an increase in filter-feeding at intermediate salinity levels and an increase in deposit-feeding thereafter, in accordance with an energy transfer from water column (i.e. suspended organic material) to sediment (deposited organic material) along a salinity gradient. However, in our case the low filter-feeding abundances observed in oligo-mesohaline ponds were insufficient to justify such food web modification.

The increase of ionic concentrations requires specific adaptations of macroinvertebrate communities. The faster metabolic rates induced by the elevated maintenance costs for osmoregulation are reflected by smaller body size (Woodward et al., 2005), supporting its use as an effective indicator for assessing community variations in transitional environments (Basset et al., 2012). Some flying insect taxa, as the case of Diptera, have aeric adult life stage in order to reduce the permanence in saltwater environments.

Contrarily to the results presented by Venâncio et al. (2019), who found changes in community structures and trophic relations even with small increments of salinity in laboratory experiment, the macroinvertebrate communities of the permanent ponds of Bosco della Mesola were found to be resilient to moderate salinization. This could be due to the functional redundancy phenomena and, in a lesser extent, to the tolerance of some taxa to moderate salinity levels. In fact, communities of oligo-mesohaline ponds maintain all the functional traits occurring in freshwater ones, except for a lower grazer abundance, which has found to be sensitive even at low salinity variations. Moreover, the observed results for filter feeders and sessile mobility attributes are due to abundance variations of a single species (*Corbula sp.*) that is the only observed organism with these functional traits, rather than to a general pattern. Since the relation between salinity and abundance of grazer organisms is mediated by the occurrence of microphytobenthos (De Jonge and Van Beuselom, 1992; Juneau et al., 2015), the relative decrease of such functional attribute may highlight the sensitivity of microphytobenthic assemblages to salinity variations (Waska and Kim, 2010). Therefore, changes in grazer relative abundances could be regarded as a sentinel of salinization effects on of microphytobenthic assemblages in the permanent ponds' systems.

Since the salinity levels observed in the study area significantly increased during last two decades (Gerdol et al., 2018), and this trend is expected to keep on in the next years, this investigation can provide evidence to be exported on other deltaic contexts on how biodiversity of the pond systems is expected to also respond to climate change. From an environmental conservation perspective, the ongoing salinization of coastal water bodies observed in our and other study cases (Ketabchi et al., 2016) is a serious threat to aquatic biodiversity. The described effects on macroinvertebrate communities in terms of taxonomic and functional diversity are expected also to have consequences on higher trophic levels and ecosystem stability, functioning and services (Landuyt et al., 2014; Pinto et al., 2014; Schratzberger and Ingels, 2018). Future management measures should be designed for mitigating the impact of salinization phenomena, which could be also the result of climate change, through the control of human activities in coastal areas and through targeted environmental restoration works.

For instance, in the case of Bosco della Mesola ponds' systems new management measures have been recently undertaken to decrease salinity levels in the near coastal lagoon (Gaglio et al., 2019). Such interventions are expected to have beneficial outcomes also in the coastal water bodies which are in contact with groundwaters. However, this does not represent a definitive solution to the conservation of biodiversity in a long period vision.

Acknowledgments

This research was carried out as part of the projects RITMARE (la Ricerca ITaliana per il MARE - Italian Research for the sea) and PRIN-NOACQUA: responses of communities and ecosystem processes in intermittent rivers (Prot. 201572HW8F) both funded by the Italian Ministry of Education, University and Research. The authors are grateful to the local offices of the Unità Territoriale Carabinieri per la Biodiversità Punta Marina which manage the study sites for their valuable support.

References

Aschonitis, V.G., Feld, C.K., Castaldelli, G., Turin, P., Visonà, E., Fano, E.A., 2016. Environmental stressor gradients hierarchically regulate macrozoobenthic community turnover in lotic systems of Northern Italy. Hydrobiologia 765, 131–147. https://doi.org/10.1007/s10750-015-2407-x

Baker, M.E., King, R.S., 2010. A new method for detecting and interpreting biodiversity and ecological community thresholds. Methods Ecol. Evol. 1, 25–43. https://doi.org/10.1111/j.2041-210X.2009.00007.x

Baker, M.E., King, R.S., Kahle, D., 2015. Glades.TITAN: An Introduction to Threshold Indicator Taxa Analysis with TITAN2 (v2.1). Vignette of R language [WWW Document]. URL https://cran.r-project.org/web/packages/TITAN2/vignettes/glades.TITAN.html

Basset, A., Barbone, E., Borja, A., Brucet, S., Pinna, M., Quintana, X.D., Reizopoulou, S., Rosati, I., Simboura, N., 2012. A benthic macroinvertebrate size spectra index for implementing the Water Framework Directive in coastal lagoons in Mediterranean and Black Sea ecoregions. Ecol. Indic. 12, 72–83. https://doi.org/10.1016/j.ecolind.2011.06.012

Bazzanti, M., 2015. Pond macroinvertebrates of the Presidential Estate of Castelporziano (Rome): a review of ecological aspects and selecting indicator taxa for conservation. Rend. Lincei 26, 337–343. https://doi.org/10.1007/s12210-015-0431-4

Blann, K.L., Anderson, J.L., Sands, G.R., Vondracek, B., 2009. Effects of agricultural drainage on aquatic ecosystems: A review. Crit. Rev. Environ. Sci. Technol. 39, 909–1001. https://doi.org/10.1080/10643380801977966

Boix, D., Gascón, S., Sala, J., Badosa, A., Brucet, S., López-Flores, R., Martinoy, M., Gifre, J., Quintana, X.D., 2008. Patterns of composition and species richness of crustaceans and aquatic insects along environmental gradients in Mediterranean water bodies. Hydrobiologia 35, 1154–1166. https://doi.org/10.1007/s10750-007-9221-z

Cañedo-Argüelles, M., Sala, M., Peixoto, G., Prat, N., Faria, M., Soares, A.M.V.M., Barata, C., Kefford, B., 2016. Can salinity trigger cascade effects on streams? A mesocosm approach. Sci. Total Environ. 540, 3–10. https://doi.org/10.1016/j.scitotenv.2015.03.039

Cardoso, P.G., Raffaelli, D., Lillebø, A.I., Verdelhos, T., Pardal, M.A., 2008. The impact of extreme flooding events and anthropogenic stressors on the macrobenthic communities' dynamics. Estuar. Coast. Shelf Sci. 76, 553–565. https://doi.org/10.1016/j.ecss.2007.07.026

Castillo, A.M., Sharpe, D.M.T., Ghalambor, C.K., De León, L.F., 2018. Exploring the effects of salinization on trophic diversity in freshwater ecosystems: a quantitative review. Hydrobiologia 807, 1–17. https://doi.org/10.1007/s10750-017-3403-0

Céréghino, R., Boix, D., Cauchie, H.M., Martens, K., Oertli, B., 2014. The ecological role of ponds in a changing world. Hydrobiologia 723, 1–6. https://doi.org/10.1007/s10750-013-1719-y

Céréghino, R., Oertli, B., Bazzanti, M., Coccia, C., Compin, A., Biggs, J., Bressi, N., Grillas, P., Hull, A., Kalettka, T., Scher, O., 2012. Biological traits of European pond macroinvertebrates. Hydrobiologia 689, 51–61. https://doi.org/10.1007/s10750-011-0744-y

Coccia, C., Vanschoenwinkel, B., Brendonck, L., Boyero, L., Green, A.J., 2016. Newly created ponds complement natural waterbodies for restoration of macroinvertebrate assemblages. Freshw. Biol. 61, 1640–1654. https://doi.org/10.1111/fwb.12804

Colombani, N., Osti, A., Volta, G., Mastrocicco, M., 2016. Impact of Climate Change on Salinization of Coastal Water Resources. Water Resour. Manag. 30, 2483–2496. https://doi.org/10.1007/s11269-016-1292-z

Como, S., van der Velde, G., Magni, P., 2018. Temporal Variation in the Trophic Levels of Secondary Consumers in a Mediterranean Coastal Lagoon (Cabras Lagoon, Italy). Estuaries and Coasts 41, 218–232. https://doi.org/10.1007/s12237-017-0265-7

Connell, J.H., 1978. Diversity in tropical rain forests and coral reefs. Science (80-.). 199, 1302–1310. https://doi.org/10.1126/science.199.4335.1302

Cramer, W., Guiot, J., Fader, M., Garrabou, J., Gattuso, J.P., Iglesias, A., Lange, M.A., Lionello, P., Llasat, M.C., Paz, S., Peñuelas, J., Snoussi, M., Toreti, A., Tsimplis, M.N., Xoplaki, E., 2018. Climate change and interconnected risks to sustainable development in the Mediterranean. Nat. Clim. Chang. 8, 972–980. https://doi.org/10.1038/s41558-018-0299-2

Day, J.W., Christian, R.R., Boesch, D.M., Yáñez-Arancibia, A., Morris, J., Twilley, R.R., Naylor, L., Schaffner, L., Stevenson, C., 2008. Consequences of climate change on the ecogeomorphology of coastal wetlands. Estuaries and Coasts 31, 477–491. https://doi.org/10.1007/s12237-008-9047-6

De Jonge, V.N., Van Beuselom, J.E.E., 1992. Contribution of resuspended microphytobenthos to total phytoplankton in the EMS estuary and its possible role for grazers. Netherlands J. Sea Res. 30, 91–105. https://doi.org/10.1016/0077-7579(92)90049-K

De Marco, P., Nogueira, D.S., Correa, C.C., Vieira, T.B., Silva, K.D., Pinto, N.S., Bichsel, D., Hirota, A.S.V., Vieira, R.R.S., Carneiro, F.M., de Oliveira, A.A.B., Carvalho, P., Bastos, R.P., Ilg, C., Oertli, B., 2014. Patterns in the organization of Cerrado pond biodiversity in Brazilian pasture landscapes. Hydrobiologia 723, 87–101. https://doi.org/10.1007/s10750-013-1695-2

De Meester, L., Declerck, S., Stoks, R., Louette, G., Van De Meutter, F., De Bie, T., Michels, E., Brendonck, L., 2005. Ponds and pools as model systems in conservation biology, ecology and evolutionary biology. Aquat. Conserv. Mar. Freshw. Ecosyst. 15, 15–725. https://doi.org/10.1002/aqc.748

Downing, A.L., Leibold, M.A., 2010. Species richness facilitates ecosystem resilience in aquatic food webs. Freshw. Biol. 55, 2123–2137. https://doi.org/10.1111/j.1365-2427.2010.02472.x

Dufrêne, M., Legendre, P., 1997. Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366. https://doi.org/10.2307/2963459

Evans, D.H., 2008. Osmotic and ionic regulation: Cells and animals, Osmotic and Ionic Regulation: Cells and Animals. CRC Press, Boca Raton. https://doi.org/10.1201/9780849380525

Fedrigotti, C., Aschonitis, V., Fano, E.A., 2016. Effects of forest expansion and land abandonment on ecosystem services of alpine environments: Case study in ledro valley (Italy) for the period 1859–2011. Glob. Nest J. 18, 875–884.

Gaglio, M., Aschonitis, V.G., Gissi, E., Castaldelli, G., Fano, E.A., 2017a. Land use change effects on ecosystem services of river deltas and coastal wetlands: case study in Volano–Mesola–Goro in Po river delta (Italy). Wetl. Ecol. Manag. 25, 67–86. https://doi.org/10.1007/s11273-016-9503-1

Gaglio, M., Aschonitis, V.G., Mancuso, M.M., Reyes Puig, J.P., Moscoso, F., Castaldelli, G., Fano, E.A., 2017b. Changes in land use and ecosystem services in tropical forest areas: a case study in Andes mountains of Ecuador. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 13, 264–279. https://doi.org/10.1080/21513732.2017.1345980

Gaglio, M., Lanzoni, M., Nobili, G., Viviani, D., Castaldelli, G., Fano, E.A., 2019. Ecosystem Services approach for sustainable governance in a brackish water lagoon used for aquaculture. J. Environ. Plan. Manag. in press. https://doi.org/10.1080/09640568.2019.1581602

Gayraud, S., Statzner, B., Bady, P., Haybachp, A., Schöll, F., Usseglio-Polatera, P., Bacchi, M., 2003. Invertebrate traits for the biomonitoring of large European rivers: An initial assessment of alternative metrics. Freshw. Biol. 48, 2045–2064. https://doi.org/10.1046/j.1365-2427.2003.01139.x

Gerdol, R., Brancaleoni, L., Lastrucci, L., Nobili, G., Pellizzari, M., Ravaglioli, M., Viciani, D., 2018. Wetland Plant Diversity in a Coastal Nature Reserve in Italy: Relationships with Salinization and Eutrophication and Implications for Nature Conservation. Estuaries and Coasts 41, 2079–2091. https://doi.org/10.1007/s12237-018-0396-5

Harley, C.D.G., Hughes, A.R., Hultgren, K.M., Miner, B.G., Sorte, C.J.B., Thornber, C.S., Rodriguez,

L.F., Tomanek, L., Williams, S.L., 2006. The impacts of climate change in coastal marine systems. Ecol. Lett. 9, 228–241. https://doi.org/10.1111/j.1461-0248.2005.00871.x
Herbert, E.R., Boon, P., Burgin, A.J., Neubauer, S.C., Franklin, R.B., Ardon, M., Hopfensperger, K.N., Lamers, L.P.M., Gell, P., Langley, J.A., 2015. A global perspective on wetland salinization: Ecological consequences of a growing threat to freshwater wetlands. Ecosphere 6, 206. https://doi.org/10.1890/ES14-00534.1
Herbst, D.B., 2006. Salinity controls on trophic interactions among invertebrates and algae of solar evaporation ponds in the Mojave Desert and relation to shorebird foraging and selenium risk. Wetlands 26, 475–485. https://doi.org/10.1672/0277-5212(2006)26[475:SCOTIA]2.0.CO;2
Herbst, D.B., Blinn, D.W., 1998. Experimental mesocosm studies of salinity effects on the benthic algal community of a saline lake. J. Phycol. 34, 772–778. https://doi.org/10.1046/j.1529-8817.1998.340772.x
Hill, M.J., Ryves, D.B., White, J.C., Wood, P.J., 2016. Macroinvertebrate diversity in urban and rural ponds: Implications for freshwater biodiversity conservation. Biol. Conserv. 201, 50–59. https://doi.org/10.1016/j.biocon.2016.06.027
Horton, T., Kroh, A., Ahyong, S., Al, E., 2017. World Register of Marine Species (WoRMS). Available from http//www.marinespecies.org. https://doi.org/10.14284/170
Johansen, P.O., Isaksen, T.E., Bye-Ingebrigtsen, E., Haave, M., Dahlgren, T.G., Kvalø, S.E., Greenacre, M., Durand, D., Rapp, H.T., 2018. Temporal changes in benthic macrofauna on the west coast of Norway resulting from human activities. Mar. Pollut. Bull. 128, 483–495. https://doi.org/10.1016/j.marpolbul.2018.01.063
Juneau, P., Barnett, A., Méléder, V., Dupuy, C., Lavaud, J., 2015. Combined effect of high light and high salinity on the regulation of photosynthesis in three diatom species belonging to the main growth forms of intertidal flat inhabiting microphytobenthos. J. Exp. Mar. Bio. Ecol. 463, 95–104. https://doi.org/10.1016/j.jembe.2014.11.003
Kefford, B.J., Davis, J., Duncan, R.P., Thompson, R., Buchwalter, D., Cañedo-Argüelles, M., Hoffmann, A., 2016. Salinized rivers: Degraded systems or new habitats for salt-tolerant faunas? Biol. Lett. 12, 20151072. https://doi.org/10.1098/rsbl.2015.1072
Ketabchi, H., Mahmoodzadeh, D., Ataie-Ashtiani, B., Simmons, C.T., 2016. Sea-level rise impacts on seawater intrusion in coastal aquifers: Review and integration. J. Hydrol. 535, 235–255. https://doi.org/10.1016/j.jhydrol.2016.01.083
La Peyre, M.K., Grace, J.B., Hahn, E., Mendelssohn, I.A., 2001. The importance of competition in regulating plant species abundance along a salinity gradient. Ecology 82, 62–69. https://doi.org/10.1890/0012-9658(2001)082[0062:TIOCIR]2.0.CO;2
Landuyt, D., Lemmens, P., D'hondt, R., Broekx, S., Liekens, I., De Bie, T., Declerck, S.A.J., De Meester, L., Goethals, P.L.M., 2014. An ecosystem service approach to support integrated pond management: A case study using Bayesian belief networks - Highlighting opportunities and risks. J. Environ. Manage. 535, 235–255. https://doi.org/10.1016/j.jenvman.2014.06.015
Larson, C.A., Belovsky, G.E., 2013. Salinity and nutrients influence species richness and evenness of phytoplankton communities in microcosm experiments from Great Salt Lake, Utah, USA. J. Plankton Res. 35, 1154–1166. https://doi.org/10.1093/plankt/fbt053
Lepš, J., Šmilauer, P., 2003. Multivariate analysis of ecological data using CANOCO. Cambridge University Press.
Liancourt, P., Callaway, R.M., Michalet, R., 2005. Stress tolerance and competitive-response ability determine the outcome of biotic interactions. Ecology 86, 1611–1618. https://doi.org/10.1890/04-1398
Little, S., Wood, P.J., Elliott, M., 2017. Quantifying salinity-induced changes on estuarine benthic fauna: The potential implications of climate change. Estuar. Coast. Shelf Sci. 198, 610–625.

https://doi.org/10.1016/j.ecss.2016.07.020
Macleod, C.K., Moltschaniwskyj, N.A., Crawford, C.M., 2008. Ecological and functional changes associated with long-term recovery from organic enrichment. Mar. Ecol. Prog. Ser. 365, 17–24. https://doi.org/10.3354/meps07534
MarLIN, 2006. BIOTIC - Biological Traits Information Catalogue [WWW Document]. Mar. Life Inf. Network. Plymouth Mar. Biol. Assoc. United Kingdom. https://doi.org/10.1002/aqc.965.
McGarigal, K., 2014. Fragstats v.4.2. Comput. Softw. Progr. Prod. by authors Univ. Massachusetts, Amherst, available Follow. web site http//www.umass.edu/landeco/research/fragstats/fragstats.html. https://doi.org/10.1093/ntr/nts298
Nielsen, D.L., Brock, M.A., 2009. Modified water regime and salinity as a consequence of climate change: Prospects for wetlands of Southern Australia. Clim. Change 95, 523–533. https://doi.org/10.1007/s10584-009-9564-8
Pereira, M., Segurado, P., Neves, N., 2011. Using spatial network structure in landscape management and planning: A case study with pond turtles. Landsc. Urban Plan. 100, 67–76. https://doi.org/10.1016/j.landurbplan.2010.11.009
Pianka, E.R., 1970. On r- and K-Selection. Am. Nat. 104, 592–597. https://doi.org/10.1086/282697
Pinder, A.M., Halse, S.A., McRae, J.M., Shiel, R.J., 2005. Occurrence of aquatic invertebrates of the wheatbelt region of Western Australia in relation to salinity. Hydrobiologia 543, 1–24. https://doi.org/10.1007/s10750-004-5712-3
Pinto, R., De Jonge, V.N., Marques, J.C., 2014. Linking biodiversity indicators, ecosystem functioning, provision of services and human well-being in estuarine systems: Application of a conceptual framework. Ecol. Indic. 36, 644–655. https://doi.org/10.1016/j.ecolind.2013.09.015
Piscart, C., Lecerf, A., Usseglio-Polatera, P., Moreteau, J.C., Beisel, J.N., 2005a. Biodiversity patterns along a salinity gradient: The case of net-spinning caddisflies. Biodivers. Conserv. 14, 2235–2249. https://doi.org/10.1007/s10531-004-4783-9
Piscart, C., Moreteau, J.C., Beisel, J.N., 2005b. Biodiversity and structure of macroinvertebrate communities along a small permanent salinity gradient (Meurthe River, France). Hydrobiologia 551, 227–236. https://doi.org/10.1007/s10750-005-4463-0
Piscart, C., Usseglio-Polatera, P., Moreteau, J.-C., Beisel, J.-N., 2006. The role of salinity in the selection of biological traits of freshwater invertebrates. Arch. für Hydrobiol. 166, 185–198. https://doi.org/10.1127/0003-9136/2006/0166-0185
Pitacco, V., Mistri, M., Munari, C., 2018. Long-term variability of macrobenthic community in a shallow coastal lagoon (Valli di Comacchio, northern Adriatic): Is community resistant to climate changes? Mar. Environ. Res. 137, 73–87. https://doi.org/10.1016/j.marenvres.2018.02.026
Por, F.D., 1972. Hydrobiological notes on the high-salinity waters of the Sinai Peninsula. Mar. Biol. 14, 111. https://doi.org/10.1007/BF00373210
Ritter, C., Montagna, P.A., Applebaum, S., 2005. Short-term succession dynamics of macrobenthos in a salinity-stressed estuary. J. Exp. Mar. Bio. Ecol. 323, 57–69. https://doi.org/10.1016/j.jembe.2005.02.018
Rothermel, B.B., 2004. Migratory success of juveniles: A potential constraint on connectivity for pond-breeding amphibians. Ecol. Appl. 14, 1535–1546. https://doi.org/10.1890/03-5206
Ruhí, A., Boix, D., Gascón, S., Sala, J., Batzer, D.P., 2013. Functional and phylogenetic relatedness in temporary wetland invertebrates: Current macroecological patterns and implications for future climatic change scenarios. PLoS One 8, e81739. https://doi.org/10.1371/journal.pone.0081739
Saadi, S., Todorovic, M., Tanasijevic, L., Pereira, L.S., Pizzigalli, C., Lionello, P., 2015. Climate change and Mediterranean agriculture: Impacts on winter wheat and tomato crop evapotranspiration, irrigation requirements and yield. Agric. Water Manag. 147, 103–115. https://doi.org/10.1016/j.agwat.2014.05.008

Sandin, L., Solimini, A.G., 2009. Freshwater ecosystem structure-function relationships: From theory to application. Freshw. Biol. 54, 2017–2024. https://doi.org/10.1111/j.1365-2427.2009.02313.x
Scavia, D., Field, J.C., Boesch, D.F., Buddemeier, R.W., Burkett, V., Cayan, D.R., Fogarty, M., Harwell, M.A., Howarth, R.W., Mason, C., Reed, D.J., Royer, T.C., Sallenger, A.H., Titus, J.G., 2002. Climate change impacts on U.S. coastal and marine ecosystems. Estuaries 25, 149–164. https://doi.org/10.1007/BF02691304
Schaffner, L.C., 2010. Patterns and Rates of Recovery of Macrobenthic Communities in a Polyhaline Temperate Estuary Following Sediment Disturbance: Effects of Disturbance Severity and Potential Importance of Non-local Processes. Estuaries and Coasts 33, 1300–1313. https://doi.org/10.1007/s12237-010-9301-6
Schratzberger, M., Ingels, J., 2018. Meiofauna matters: The roles of meiofauna in benthic ecosystems. J. Exp. Mar. Bio. Ecol. 502, 12–25. https://doi.org/10.1016/j.jembe.2017.01.007
ter Braak, C.J.F., Smilauer, P., 2002. CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (Version 4.5), Section on Permutation Methods. Microcomputer Power. Biometris-Plant Research International, Wageningen and České Budějovice. https://doi.org/citeulike-article-id:7231853
Thorp, J.H., Covich, A.P., 2010. Ecology and Classification of North American Freshwater Invertebrates, Ecology and Classification of North American Freshwater Invertebrates. Accademic Press. https://doi.org/10.1016/C2009-0-02669-5
Venâncio, C., Castro, B.B., Ribeiro, R., Antunes, S.C., Abrantes, N., Soares, A.M.V.M., Lopes, I., 2019. Sensitivity of freshwater species under single and multigenerational exposure to seawater intrusion. Philos. Trans. R. Soc. B Biol. Sci. 374, 20180252. https://doi.org/10.1098/rstb.2018.0252
Viaroli, P., Bartoli, M., Vymazal, J., 2016. Preface: Wetlands biodiversity and processes—tools for conservation and management. Hydrobiologia 774, 1–5. https://doi.org/10.1007/s10750-016-2793-8
Waska, H., Kim, G., 2010. Differences in microphytobenthos and macrofaunal abundances associated with groundwater discharge in the intertidal zone. Mar. Ecol. Prog. Ser. 407, 159–172. https://doi.org/10.3354/meps08568
Waterkeyn, A., Grillas, P., Vanschoenwinkel, B., Brendonck, L., 2008. Invertebrate community patterns in Mediterranean temporary wetlands along hydroperiod and salinity gradients. Freshw. Biol. 53, 1808–1822. https://doi.org/10.1111/j.1365-2427.2008.02005.x
Williams, P., Whitfield, M., Biggs, J., Bray, S., Fox, G., Nicolet, P., Sear, D., 2004. Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. Biol. Conserv. 115, 329–341. https://doi.org/10.1016/S0006-3207(03)00153-8
Withers, P.C., 1992. Comparative Animal Physiology. Saunders College Publishing, Forth Worth, TX.
Woodward, G., Ebenman, B., Emmerson, M., Montoya, J.M., Olesen, J.M., Valido, A., Warren, P.H., 2005. Body size in ecological networks. Trends Ecol. Evol. 20, 402–409. https://doi.org/10.1016/j.tree.2005.04.005
Xenopoulos, M.A., Lodge, D.M., Alcamo, J., Märker, M., Schulze, K., Van Vuuren, D.P., 2005. Scenarios of freshwater fish extinctions from climate change and water withdrawal. Glob. Chang. Biol. 11, 1557–1564. https://doi.org/10.1111/j.1365-2486.2005.001008.x
Zettler, M.L., Karlsson, A., Kontula, T., Gruszka, P., Laine, A.O., Herkül, K., Schiele, K.S., Maximov, A., Haldin, J., 2014. Biodiversity gradient in the Baltic Sea: A comprehensive inventory of macrozoobenthos data. Helgol. Mar. Res. 68, 49. https://doi.org/10.1007/s10152-013-0368-x
Zuur, A.F., Ieno, E.N., Smith, G.M., 2007. Analyzing Ecological Data, Methods. Springer New York. https://doi.org/10.1016/B978-0-12-387667-6.00013-0

Estuaries and Coasts (2018) 41:2079–2091
https://doi.org/10.1007/s12237-018-0396-5

Wetland Plant Diversity in a Coastal Nature Reserve in Italy: Relationships with Salinization and Eutrophication and Implications for Nature Conservation

Renato Gerdol[1] · **Lisa Brancaleoni**[2] · **Lorenzo Lastrucci**[3] · **Giovanni Nobili**[4] · **Mauro Pellizzari**[5] · **Michele Ravaglioli**[4] · **Daniele Viciani**[3]

Received: 13 July 2017 / Revised: 1 March 2018 / Accepted: 12 March 2018 / Published online: 2 April 2018

Abstract

Wetlands are important centers of biodiversity. Coastal wetlands are subject to anthropogenic threats that can lead to biodiversity loss and consequent negative effects on nature conservation. We investigated relationships between wetland vegetation and habitat conditions in a coastal Nature Reserve in Northern Italy that has undergone seawater intrusion and eutrophication for several decades. The wetland vegetation in the Nature Reserve consisted of nine communities of hygrophytic and helophytic vegetation and five communities of waterplant vegetation. The hygrophytic and helophytic communities were arranged according to a salinity gradient, from salt-free habitats to strongly saline habitats. The saline habitats had high nutrient levels, due to the influx of nitrate-rich saltwater from an adjacent lagoon. The waterplant communities were all typical of freshwater habitats. Water-table depth and concentration of dissolved nutrients in the water were the main factors structuring waterplant vegetation. The main driver of future changes in the wetland vegetation of the Nature Reserve is the ongoing increase in salinity levels which may enhance expansion of halophilic species and communities, thus outcompeting locally rare freshwater species. If nutrient, especially nitrate, load further increases in the next future, this may exert negative effects on wetland species and communities preferring nutrient-poor habitats.

Keywords Aquatic plants · Biodiversity · Nitrate · Nutrient load · Saline wedge · Vegetation

Introduction

Wetlands represent important centers of biodiversity, hosting specialized plant and animal species. Wetlands generally exhibit high level of functional diversity (Bedford et al. 2001) and thus assume a central role for nature conservation (García-Madrid et al. 2014; Angiolini et al. 2017). Wetlands are prone to a number of threats which mine their survival: over-exploitation, habitat fragmentation, water eutrophication and pollution, human alteration of natural water cycles and alien species invasion are only some of the main threats and pressures which affect wetlands worldwide (Bedford et al. 2001; Dudgeon et al. 2006). Coastal wetlands are not immune from anthropogenic threats. Natural events such as erosion, hydrological and hydrochemical alterations, sea-level rise and subsidence may specifically affect these wetland types (Turner 1990). In particular, the Mediterranean basin represents one of the regions most prone to loss of coastal wetlands in the future (Nicholls et al. 1999).

Wetland plant diversity is scale-dependent and varies across regions, showing a decreasing trend from more humid toward less humid climatic regions (see Hrivnák et al. 2014 and

Communicated by Just Cebrian

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s12237-018-0396-5) contains supplementary material, which is available to authorized users.

✉ Renato Gerdol
grn@unife.it

[1] Department of Life Sciences and Biotechnology, University of Ferrara, Corso Ercole I d'Este 32, 44121 Ferrara, Italy

[2] Botanical Garden, University of Ferrara, Corso Ercole I d'Este 32, 44121 Ferrara, Italy

[3] Department of Biology, University of Florence, Via La Pira 4, 50121 Florence, Italy

[4] Carabinieri per la Biodiversità di Punta Marina, Lungomare C. Colombo, 21, 48122 Punta Marina, Ravenna, Italy

[5] Istituto Comprensivo "Bentivoglio", Via Salvo D'Acquisto 5/7, 44028 Poggio Renatico, Ferrara, Italy

references therein). A number of environmental variables play an important role in driving plant species diversity in wetlands, such as size of the site, ground morphology, water-table depth and water chemistry (Edvardsen and Økland 2006; Lacoul and Freedman 2006; Hrivnák et al. 2014). Salinity is paramount for shaping the distribution of plant species in coastal wetlands (Watt et al. 2007). Salinity becomes a critical factor when sea-level rise and/or subsidence increase salinity levels, thus altering the composition and the distribution of wetland plant communities (Brock et al. 2005; Spalding and Hester 2007). Indeed, stress resulting from increased salinity has been reported as a cause of biodiversity loss in terrestrial to riparian and aquatic habitats (see Brock et al. 2005 and references therein). Ecological modifications due to increased salinity levels can also trigger dynamic processes which can eventually lead to dramatic changes in the vegetation of coastal wetlands (Donnelly and Bertness 2001). Water eutrophication is considered one of the most important causes of degradation of many types of wetlands, from inland to marine ecosystems. Zaldivar et al. (2008) focused on direct and indirect effects of eutrophication of transitional waters, showing a complex net of drivers and pressures brought about by eutrophication which in turn reverberate in complex socio-economic and ecological aspects. The consequences of nutrient enrichment include development of algal blooms which can lead to anoxic conditions with negative impact on waterfowl, fish, and invertebrates (Davis and Froend 1999). Water chemistry is also important in determining abundance, composition, and distribution of wetland communities. Consequently, several types of plant communities have been identified as indicators of water quality (Klosowski and Jablońska 2009; Ceschin et al. 2010; Sakurai et al. 2017). Anthropic activities such as urbanization and agriculture produce deep modifications in the chemical and physical features of the water ecosystems, inducing significant changes in the spatial pattern of wetland vegetation (Ceschin et al. 2010).

Vegetation dynamics, i.e., changes in vegetation composition over time, represents one of the most frequent causes of loss of habitats of conservation interest (Viciani et al. 2014). In Europe, plant and habitat conservation benefits from a powerful tool provided by the European Union, i.e., the Habitats Directive (Commission of the European Community 1992). The identification of habitat types and the selection of sites are supported by the implementation of the Interpretation Manual of European Union Habitats (Commission of the European Community 2013). Another important document is the European Red List of Habitats that provides an overview of the degree of endangerment of terrestrial and freshwater habitats in the European Union (Janssen et al. 2016). At the regional level, Biondi and Blasi (2009) produced the Italian Interpretation Manual of European Union Habitats and, more recently, Angelini et al. (2016) produced a handbook for monitoring habitats of community interest in Italy. This handbook is based on European guidelines (Evans and Arvela 2011) and provides tools for an effective monitoring activity of Annex I Habitats occurring in Italy (Gigante et al. 2016). These documents not only show that wetlands host habitats of conservation importance but also report that these habitats are often subject to serious threats and pressures. Several vegetation types strictly related to wetlands, often widespread in Central Europe but very rare in Southern Europe, do not fall in any habitat type of conservation interest listed in the Habitat Directive (Angiolini et al. 2017). Analyzing relationships among species, plant communities, and ecological drivers provides important and useful information that can be used for the protection and monitoring of plants and habitats of conservation interest either listed in environmental protection laws or not.

The Nature Reserve “Bosco della Mesola” represents a relict of ancient coastal forests. Most of these forested areas disappeared in more or less recent times, while others underwent deep changes. Consequently, very few coastal forests still occur in relatively pristine conditions in this region (Stampi 1966; Piccoli et al. 1983; Pellizzari and Piccoli 2001). The Nature Reserve is rich in wetlands that host different plant communities, including both freshwater and brackish vegetation types. This area has undergone subsidence and eutrophication for several decades, which can potentially trigger significant changes in wetland vegetation. In this study, we analyzed relationships between wetland vegetation and environmental conditions in the Nature Reserve. Our main objective was to analyze environmental factors affecting wetland vegetation, with special focus on habitats and species of conservation interest. We also aimed at forecasting future changes in wetland vegetation and defining appropriate guidelines directed to implementing and updating the Management Plan of the Nature Reserve.

Materials and Methods

Study Area The study was carried out at Bosco della Mesola, North-Eastern Italy (44°50′ N, 12°15′ E, 1088 ha, 0–2.8 m above sea level, Fig. 1), a National Nature Reserve hosting Special Areas of Conservation, Special Protection Areas and a strict Nature Reserve area in the Po Delta Park. The study area lies on a dune system originated during the eleventh to fifteenth centuries, consisting of sand dunes and dune slacks with parallel, approximately North-South orientation. This area is prevalently covered with woodlands that represent a relict of ancient coastal forests (Stampi 1966; Piccoli and Gerdol 1984; Piccoli et al. 1983). Most of these areas disappeared in more or less recent times, while others underwent deep changes (Stampi 1966). The woodlands in the Reserve present transitional features between Mediterranean and Central European forest types (Gerdol et al. 1985). A xerophilic *Quercus ilex* community is situated on high, more recent arid dunes. A mesophilic *Carpinus betulus–Quercus robur* community is situated on flat, more

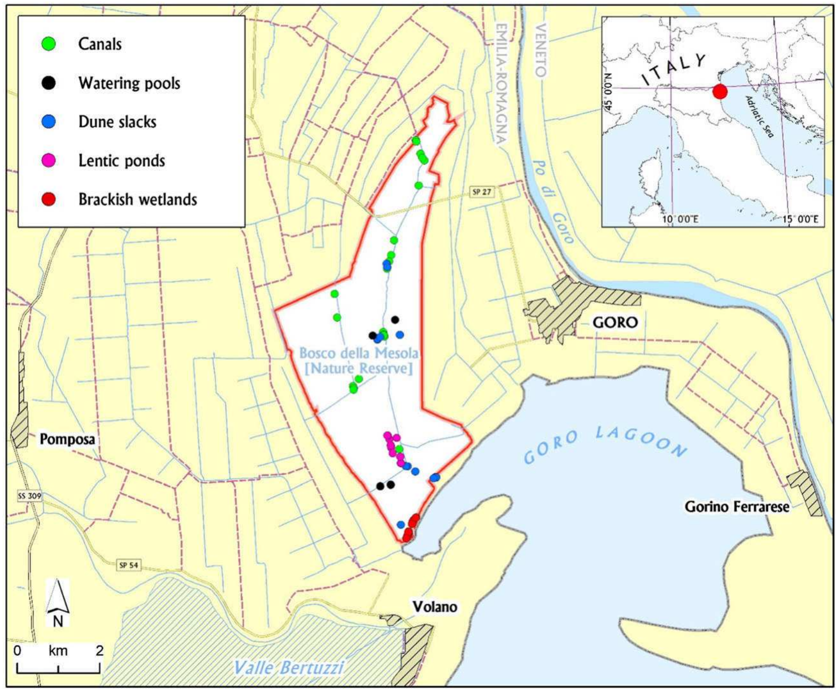


Fig. 1 Map of the study area with location of the sampling sites in the five wetland types

ancient and less arid dunes. The dune slacks are colonized by a *Populus alba–Fraxinus angustifolia* community typical of wet habitats (Gerdol et al. 1985). Hydrologically, this area is characterized by a coastal aquifer system in hydraulic continuity with the sea. This determines saltwater inflow which enables the development of halophilic vegetation in the marginal south-eastern part of the area (Piccoli et al. 1983; Fig. 1). Three main soil types occur in this area: typic xeropsamments, aquic xeropsamments and psammaquents, considerably differing from each other in depth and structure. However, all of these soils are rich in sand and have A-C profile and moderately alkaline pH (Gerdol et al. 1985). The soil pore-water is enriched in salt at the south-eastern border of the area (Piccoli et al. 1983).

Wetlands and their Management The study area hosts a rich variety of freshwater and brackish wetlands. Natural freshwater wetlands in the inland part of the area are mainly situated in temporarily flooded dune slacks not covered with forest vegetation. Therefore, dune slacks are prevalently fed by rainwater, and secondarily by water flowing from canals so that water-table depth in these wetlands is conditioned by precipitation and by the water level in the canal system. Natural brackish wetlands are concentrated in the south-eastern marginal part of the area and are fed with saltwater from an adjacent lagoon (Fig. 1). These brackish wetlands usually are flooded throughout the year, except during prolonged dry periods in summer. Three types of artificial wetlands occur in the study area: (1) a network of canals, (2) a number of pools used for watering wildlife, and (3) a lentic pond permanently fed by discharging water table. The watering pools have a more or less regular circular shape and usually are flooded, except during periods of prolonged drought. A dense network of canals represents the main source of water supply to the Nature Reserve. This is of fundamental importance for supporting basic ecological processes that eventually guarantee correct functioning of the forest ecosystem. The canals provide freshwater from the surrounding areas, whereas a number of locks limit the intrusion of saltwater from the sea (the so-called saline wedge). Indeed, the Nature Reserve is situated in a reclamation area so that meteoric water must be drained in winter, while oxygen-rich freshwater must be supplied to the forest in summer. Over time, these canals gradually turned from simple hydraulic systems to complex ecosystems, representing preferential habitats for aquatic plants, fish, and herpetological species. The management of the protected area follows the guidelines stated in the Reserve Management Plan, as well as a series of conservation actions defined in specific LIFE projects (e.g., LIFE00NAT/7147). Major objectives are the conservation of priority habitats and species of the Habitats Directive. Most of the relevant conservation actions are closely related to the hydraulic management of the water supply through the canal systems.

Sampling and Analysis Wetlands in the study area were surveyed several times during the growing season (May–September) of 2016. A total of 84 phytosociological relevés were carried out at the peak of the growing season in all of the wetland types occurring in the Nature Reserve (Table 1; Fig. 1). Alluvial forests of the *Populus alba–Fraxinus angustifolia* community, although inhabiting wet habitats, were not considered in our survey. The choice of the sampling sites was subjective, but the number of relevés approximately

Table 1 List of the wetland types, with cover area and number of vegetation relevés

Wetland type	Area (ha)	No. of relevés
Canals	10.0	22
Watering pools	1.2	7
Dune slacks	14.9	25
Lentic ponds	7.1	16
Brackish wetlands	9.2	14

mirrored frequency and area covered by each wetland type (Table 1), partially calibrated by visual inspection of vegetation diversity in each wetland type.

Plant communities were sampled according to the phytosociological method (Braun-Blanquet 1932; Biondi 2011; Pott 2011). A phreatimeter was placed and the following data were recorded at each relevé site: GPS coordinates (UTM33, WGS84), relevé area (m^2), and vegetation cover (%). At the end of the growing season, water-table depth was measured and a 50-mL water sample was collected at each of the relevé sites. All water-table measurements and water collections were carried out on a single day (25 October 2016) in order to make the data comparable. The day after collection, electrical conductivity and salinity were measured with a Crison CM 35 conductivity-meter (Hach-Lange, L'Hospitalet de Llobregat, Spain), and pH was measured with a Hanna Jenway 3510 pH-meter (Hanna Instruments, Villafranca Padovana, Italy). The water samples were immediately filtered with Whatman 25-mm GD/X syringe filters (pore size 0.45 μm) and colorimetrically analyzed for ammonium (NH_4^+), nitrate (NO_3^-), and phosphate (PO_4^{3-}) concentrations using a continuous-flow autoanalyzer (Systea Flowsys, Anagni, Italy). Ammonium was determined by the salicylate method in the presence of hypochlorite, nitrate by the cadmium-reduction method through a column, and phosphate by the molybdenum-blue method. Alkalinity (HCO_3^-) was determined by the double indicator method with phenolphthalein and methyl orange.

The spatial distribution of electrical conductivity was graphically represented by a Geographical Information System (GIS), using the software ArcView GIS 3.2 for Windows and IDW interpolation. The resulting map was compared with a similar map obtained using data on electrical conductivity measured in 2003 at 60 sites, most of which coincided with the sampling sites of the present study.

Statistics The matrix of the vegetation data (i.e., the cover of each species in the 84 relevés) was statistically treated by a hierarchical numerical classification method after transforming the ordinal scale used for assessing species cover. Scale transformation was performed using the van der Maarel (1979) numerical scale. A cluster analysis (i.e., numerical classification) was performed by the Ward method based on the matrix of between-relevé Euclidean distances (Orlóci 1978). Relationships between species composition and environmental variables were statistically analyzed by a multivariate ordination method (detrended canonical correspondence analysis, DCCA). Significance level of the environmental variables was assessed by Monte Carlo permutations based on forward selection of the variables. Occasional species, i.e., those recorded in one or two relevés only, were excluded from the cluster analysis and the multivariate ordinations as well.

Species composition of the plant communities defined by the numerical classification was assessed in terms of species rarity. Species rarity was calculated based on rarity indices of all plant species, i.e., also including the occasional species not considered in the multivariate analyses. The rarity indices (Table 2) were derived from estimates of species frequency in the local plant species checklist (Piccoli et al. 2014). The cluster analysis and the DCCA were performed with the statistical software CANOCO 5.0 (ter Braak and Šmilauer 2012). Univariate statistics were computed with the statistical software STATISTICA 7.0 (StatSoft©; Version 7; StatSoft Inc., Tulsa, OK, USA).

Results

Vegetation Classification The vegetation relevés were clustered into two big groups (Fig. 2). The first group (rel. 1–56) included relevés of hygrophytic or helophytic vegetation, while the second group (rel. 57–84) included relevés of waterplant communities (sensu Den Hartog and Segal 1964). Fourteen clusters were recognized at a lower hierarchical level in the dendrogram, i.e., at a Euclidean distance of 25–30 (Fig. 2). These clusters represented the main wetland vegetation types in the Nature Reserve. A synthetic description of these communities, as well as their correspondence with the Natura 2000 habitat codes and the Red List Habitat EUNIS codes, is given in Table 3. Details on the full species composition can be found in Supplementary Table 1.

Table 2 List of the rarity indices derived from estimates of species abundance in the local plant species checklist

Species frequency in the checklist	Rarity index
Very rare	6
Rare	5
Uncommon	4
Widespread	3
Common	2
Very common	1

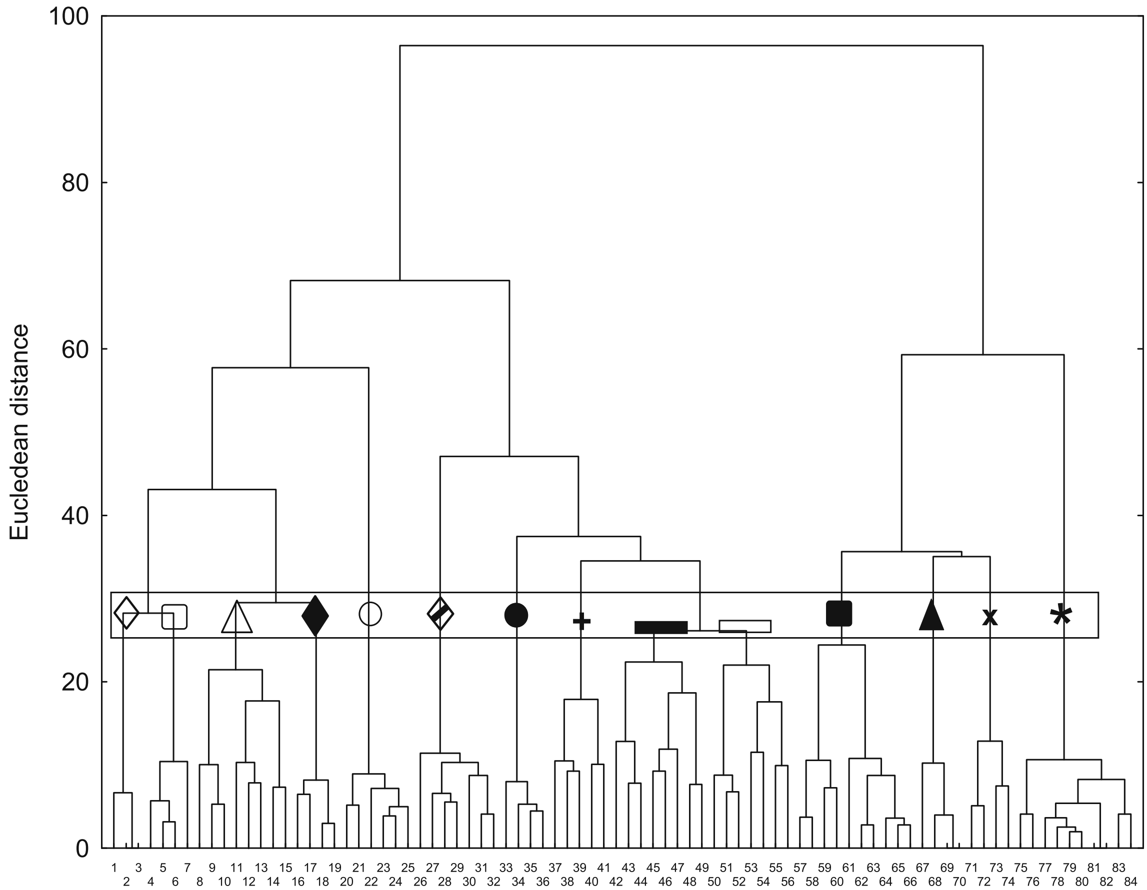


Fig. 2 Classification dendrogram of the 84 vegetation relevés. The rectangle indicates the range of Euclidean distance (ca. 25–30) in which the clusters corresponding to the 14 plant communities were recognized (symbols as in Fig. 3)

In brief, nine hygrophytic and helophytic communities were recognized based on the cluster analysis (Fig. 2; Table 3). Two of these communities (c. of *Spartina anglica* and c. of *Tripolium pannonicum*) were composed of few halophytes, exclusive of brackish wetlands. Three communities (c. of *Juncus acutus* and *J. littoralis*, c. of *J. subnodulosus* and *Phragmites australis*, and c. of *J. acutus* and *J. maritimus*) had different rush species and/or common reed as dominant species. These three communities were richer in species and were composed of more or less strongly salt-tolerating hygrophytes and helophytes. They were found in brackish wetlands, lentic ponds, and dune slacks. The community of *Cladium mariscus* was found only in dune slacks and had *Cladium mariscus* as the dominant species with scattered occurrence of a few other, moderately salt-tolerating species. Three further communities (c. of *Schoenus nigricans* and *Tripidium ravennae*, c. of *Juncus maritimus* and *Galium palustre*, and the wet meadows) were composed of several species generally possessing low levels of salt tolerance. These communities usually occurred in lentic ponds and dune slacks, and occasionally in watering pools with poor if any saltwater influx. The waterplant communities were overall poorer in species compared with the hygrophytic and helophytic communities. Five waterplant communities were recognized based on the cluster analysis (Fig. 2; Table 3). Four of them were characterized by dominance of an only waterplant species each. These dominating waterplant species were both pleustophytes (Den Hartog and Segal 1964), i.e., waterplants floating on the water surface (*Spirodela polyrrhiza* and *Lemna minuta*) or floating freely between the bottom and the surface (*Ceratophyllum demersum*) and rhizophytes (Den Hartog and Segal 1964), i.e., waterplants with their basal parts penetrating into the bottom (*Myriophyllum spicatum*). These four communities were usually found in canals. The fifth waterplant community (c. of *Chara* sp.) was prevalently comprised of rhyzophytes, with *Chara vulgaris* or *C. intermedia* (not separated in the field during the relevés) usually dominating. This community was found only in watering pools.

With respect to conservation value, the community of *Chara* sp. stood out among all other communities because of the presence of three locally very rare species: *Chara fragilis*, *C. intermedia*, and *Ranunculus peltatus* ssp. *baudotii* (Supplementary Table 1). Hence, this community presented the highest rarity index (Table 4). The community of *Chara* sp. presented considerable importance with respect to habitat type because this community is quite rare in the lowland

Table 3 Synthetic description of the vegetation types with reference to Habitat Natura 2000 codes and European Red List of Habitats—EUNIS codes

Community type	Species composition	Wetland type	Natura 2000 code	EUNIS code
C. of *Spartina anglica*	Almost pure *Spartina anglica* stands	Brackish wetlands	1320	A2.5d
C. of *Tripolium pannonicum*	Very species-poor community with *Tripolium pannonicum* as the dominant species and scattered occurrence of few other helophytes	Brackish wetlands	1410	A2.5d
C. of *Juncus acutus* and *J. littoralis*	Dominated by halophytes or salt-tolerant species often with high frequency of *Phragmites australis*	Brackish wetlands, dune slacks, lentic ponds	1410	A2.5d
C. of *Juncus subnodulosus* and *Phragmites australis*	Co-dominated by *Juncus subnodulosus* and *Phragmites australis* with scattered occurrence of salt-tolerant species	Lentic ponds	–	C5.1a
C. of *Juncus acutus* and *J. maritimus*	Co-dominated by *Juncus acutus* and *J. maritimus* with low frequency of *Phragmites australis* and some other salt-tolerant species	Dune slacks	1410	A2.5d
C. of *Cladium mariscus*	Species-poor community strongly dominated by *Cladium mariscus*	Dune slacks	7210	B1.8b (D4.1b)
C. of *Schoenus nigricans* and *Tripidium ravennae*	Species-rich community co-dominated by *Schoenus nigricans* and *Tripidium ravennae* together with several moderately salt-tolerating species	Lentic ponds	6420	E3.1a
C. of *Juncus maritimus* and *Galium palustre*	Species-rich community co-dominated by *Juncus maritimus* and *Galium palustre* with modest, if any, saltwater influx	Dune slacks	6420	B1.8b (E3.1a)
Wet meadows	Species-rich community having *Mentha aquatica*, *Molinia arundinacea*, and *Scirpoides holoschoenus* as the most abundant species associated with several poorly salt-tolerating species	Dune slacks, watering pools, lentic ponds	6420	E3.1a (B1.8b)
C. of *Spirodela polyrrhiza*	Species-poor community dominated by the small pleustophyte *Spirodela polyrrhiza*	Canals	3150	C1.2b
C. of *Ceratophyllum demersum*	Species-poor community dominated by the freely-floating pleustophyte *Ceratophyllum demersum*	Canals	3150	C1.2b
C. of *Lemna minuta*	Extremely species-poor community dominated by the small pleustophyte *Lemna minuta*	Canals, watering pools	3150	C1.2b
C. of *Chara* sp.	Species-poor, prevalently rhizophytic community with *Chara vulgaris* and/or *C. intermedia* as the dominant species and rather high frequency of *Potamogeton pectinatus* and *Ranunculus peltatus* ssp. *baudotii*	Watering pools	3140	C1.2a
C. of *Myriophyllum spicatum*	Species-poor community dominated by the rhyzophyte *Myriophyllum spicatum* usually associated with *Ceratophyllum demersum*	Canals	3150	C1.2b

regions of northern Italy. Other communities were important in terms of species rarity, although presenting lower rarity index (Table 4; Supplementary Table 1). Firstly, the community of *Juncus acutus* and *J. littoralis* hosted one of the very few populations of *Kosteletzkya pentacarpos* in Italy. This species is listed both in the 92/43/CEE Habitat Directive (Annex II) and in the Bern Convention (Annex I) and is considered threatened with extinction in Italy and in Europe, where *Kosteletzkya pentacarpos* is classified as "critically endangered" and "vulnerable," respectively (Bilz et al. 2011; Rossi et al. 2016). Secondly, the community of *Juncus maritimus* and *Galium palustre* hosted three locally very rare species, i.e., *Euphorbia lucida*, *Hydrocotyle vulgaris*, and *Teucrium scordium* (Table 4; Supplementary Table 1). The community of *Juncus subnodulosus* and *Phragmites australis* and the community of *Cladium mariscus* had quite high rarity indices for being characterized by high abundance of the locally very rare species *Juncus subnodulosus* and *Cladium mariscus*, respectively (Table 4; Supplementary Table 1). The species-poor community of *Lemna minuta* had a high rarity index owing to the high abundance of the locally uncommon, but invasive alien species *Lemna minuta*. Therefore, the rarity index seemed to overemphasize the real conservation value of this community. All of the other communities did not exhibit remarkable conservation value based on the rarity index (Table 4).

Ecological Gradients As the hygrophytic and helophytic vegetation on one side and the waterplant vegetation on the other side differed very strongly from each other in terms of species

Table 4 Mean (± SE) values of the rarity index and of hydrochemical variables in nine communities of hygrophytic and helophytic vegetation and five communities of waterplant vegetation. For each variable in each of the two vegetation groupings, the means followed by the same letter do not differ significantly ($P < 0.05$) based on Tukey's HSD post hoc tests (capital letters for hygrophytic and helophytic vegetation; small letters for waterplant vegetation)

Community	Rarity index	pH	Electrical conductivity (μS cm^{-1})	Salinity (mg L^{-1})	Water-table depth (cm)	Alkalinity (mg L^{-1})	NO_3^- (mg L^{-1})	PO_4^{3-} (mg L^{-1})
Hygrophytic and helophytic vegetation								
C. of *Spartina anglica*	2.67 ± 0.17 B	7.57 ± 0.03 A	38,967 ± 524 A	23,833 ± 384 A	8.0 ± 0.6 AB	186 ± 11 B	0.540 ± 0.087 A	0.053 ± 0.003 B
C. of *Tripolium pannonicum*	2.69 ± 0.19 B	7.58 ± 0.03 A	34,375 ± 2226 A	10,342 ± 6088 B	4.5 ± 1.9 ABC	204 ± 16 B	0.349 ± 0.077 A	0.042 ± 0.004 B
C. of *Juncus acutus* and *J. littoralis*	2.93 ± 0.11 B	7.63 ± 0.06 A	12,801 ± 2932 B	7268 ± 1739 B	4.9 ± 4.1 AB	363 ± 48 B	0.110 ± 0.049 B	0.175 ± 0.106 B
C. of *Juncus subnodulosus* and *Phragmites australis*	3.31 ± 0.12 AB	7.45 ± 0.06 A	1196 ± 149 C	576 ± 73 C	14.0 ± 3.0 A	206 ± 13 B	0.007 ± 0.001 B	0.004 ± 0.001 B
C. of *Juncus acutus* and *J. maritimus*	2.66 ± 0.12 B	7.36 ± 0.04 A	16,539 ± 493 B	9269 ± 297 B	−0.7 ± 3.5 ABC	412 ± 31 B	0.030 ± 0.004 B	0.305 ± 0.155 B
C. of *Cladium mariscus*	3.04 ± 0.1 AB	7.53 ± 0.17 A	6207 ± 2987 BC	3363 ± 1671 BC	−19.8 ± 4.2 BC	358 ± 202 B	0.054 ± 0.016 B	0.562 ± 0.555 AB
C. of *Schoenus nigricans* and *Tripidium ravennae*	2.68 ± 0.2 B	7.50 ± 0.00 A	2140 ± 759 C	1087 ± 403 C	−27.6 ± 12.6 C	188 ± 17 B	0.018 ± 0.009 B	0.003 ± 0.001 B
C. of *Juncus maritimus* and *Galium palustre*	3.54 ± 0.15 A	7.11 ± 0.07 B	2918 ± 532 C	1491 ± 282 C	−14.9 ± 5.7 BC	907 ± 141 A	0.102 ± 0.029 B	1.990 ± 0.701 A
Wet meadows	3.07 ± 0.25 AB	7.49 ± 0.12 A	1876 ± 775 C	969 ± 414 C	−13.0 ± 8.2 BC	349 ± 62 B	0.070 ± 0.017 B	0.011 ± 0.002 B
Waterplant vegetation								
C. of *Spirodela polyrrhiza*	2.53 ± 0.21 b	7.60 ± 0.12 a	881 ± 147 b	434 ± 74 b	15.8 ± 6.8 ab	240 ± 17 b	0.154 ± 0.085 a	0.013 ± 0.006 a
C. of *Ceratophyllum demersum*	2.24 ± 0.15 b	7.73 ± 0.17 a	497 ± 31 c	239 ± 15 c	33.0 ± 5.5 a	170 ± 14 c	0.168 ± 0.096 a	0.006 ± 0.002 a
C. of *Lemna minuta*	3.75 ± 0.25 a	7.20 ± 0.11 b	1322 ± 211 a	661 ± 112 a	12.8 ± 3.3 b	357 ± 34 a	0.550 ± 0.005 a	0.012 ± 0.002 a
C. of *Chara* sp.	4.38 ± 0.13 a	7.48 ± 0.02 a	428 ± 20 c	356 ± 10 bc	10.8 ± 1.2 b	306 ± 27 ab	0.012 ± 0.001 a	0.006 ± 0.001 a
C. of *Myriophyllum spicatum*	2.24 ± 0.11 b	7.52 ± 0.04 a	474 ± 7 c	227 ± 3 c	33.3 ± 3.5 a	190 ± 7 bc	0.046 ± 0.007 a	0.004 ± 0.001 a

composition, we run two separate DDCAs for each of the two groups (Fig. 3). All environmental factors accounted significantly for the observed variation in the species composition of the hygrophytic and helophytic vegetation (Table 5). Electrical conductivity accounted for the largest part of the variance in the hygrophytic and helophytic vegetation. Salinity was significant as well but accounted for a smaller portion of variance because of its high collinearity with electrical conductivity (Fig. 3a; Table 5). The nine hygrophytic and helophytic communities were thus arranged according to a gradient of increasing salt tolerance, parallel to the vectors of electrical conductivity and salinity, from the community of *Juncus subnodulosus* and *Phragmites australis*, the wet meadows, the community of *Schoenus nigricans* and *Tripidium ravennae*, and the community of *Juncus maritimus* and *Galium palustre*, all characterized by mean salinity levels of about 1000 mg L^{-1} or lower indicating no saltwater influx, to the community of *Juncus acutus* and *J. littoralis*, the community of *Juncus acutus* and *J. maritimus*, the community of *Tripolium pannonicum*, and the community of *Spartina anglica*, all settled in more or less strongly saline habitats. The community of *Cladium mariscus* had intermediate position across the gradient (Fig. 3a; Table 4). Similarly, the species were arranged according to increasing salt-tolerance levels from *Hedera helix*, *Brachypodium sylvaticum*, and *Vincetoxicum hirundinaria* at the left end of the gradient to *Kosteletzkya pentacatpos*, *Carex extensa*, *Atriplex prostrata*, *Spartina anglica*, and *Tripolium pannonicum* at the right end of the gradient (Fig. 3a). The vectors of water-table depth and pH on one side, and the vectors of alkalinity and phosphate on the other side presented similar trends but opposite orientation (Fig. 3a). Water-table depth was higher in the four communities having rushes (*Juncus subnodulosus*, *J. acutus*, and *J. littoralis*), tall helophytes (*Phragmites australis* and *Typha angustifolia*), or halophytes (*Tripolium pannonicum*, *Spartina anglica*, and *Atriplex prostrata*) as dominant or abundant species. Accordingly, the centroids of these four communities (c. of *Spartina anglica*, c. of *Tripolium pannonicum*, c. of *Juncus acutus* and *J. littoralis*, and c. of *Juncus subnodulosus* and *Phragmites australis*) were located in the lower-right part of the DCCA diagram (Fig. 3a). In all of these communities, mean water-table depth was positive indicating that the soil was submerged (Table 4). These communities also presented somewhat higher mean pH although there were very poor, if any, significant differences in terms of mean pH that ranged from 7.11 to 7.63 across all of the wetland communities (Table 4). Alkalinity and phosphate were both higher in less humid habitats, with highest values in the community of *J. maritimus* and *Galium palustre* (Fig. 3a; Table 4). The nitrate vector was oriented almost parallel to the first DCCA axis. This corresponded to highest nitrate concentrations in the community of *Tripolium pannonicum* and the community of *Spartina anglica*, both having halophytes as the dominant species (Fig. 3a; Table 4).

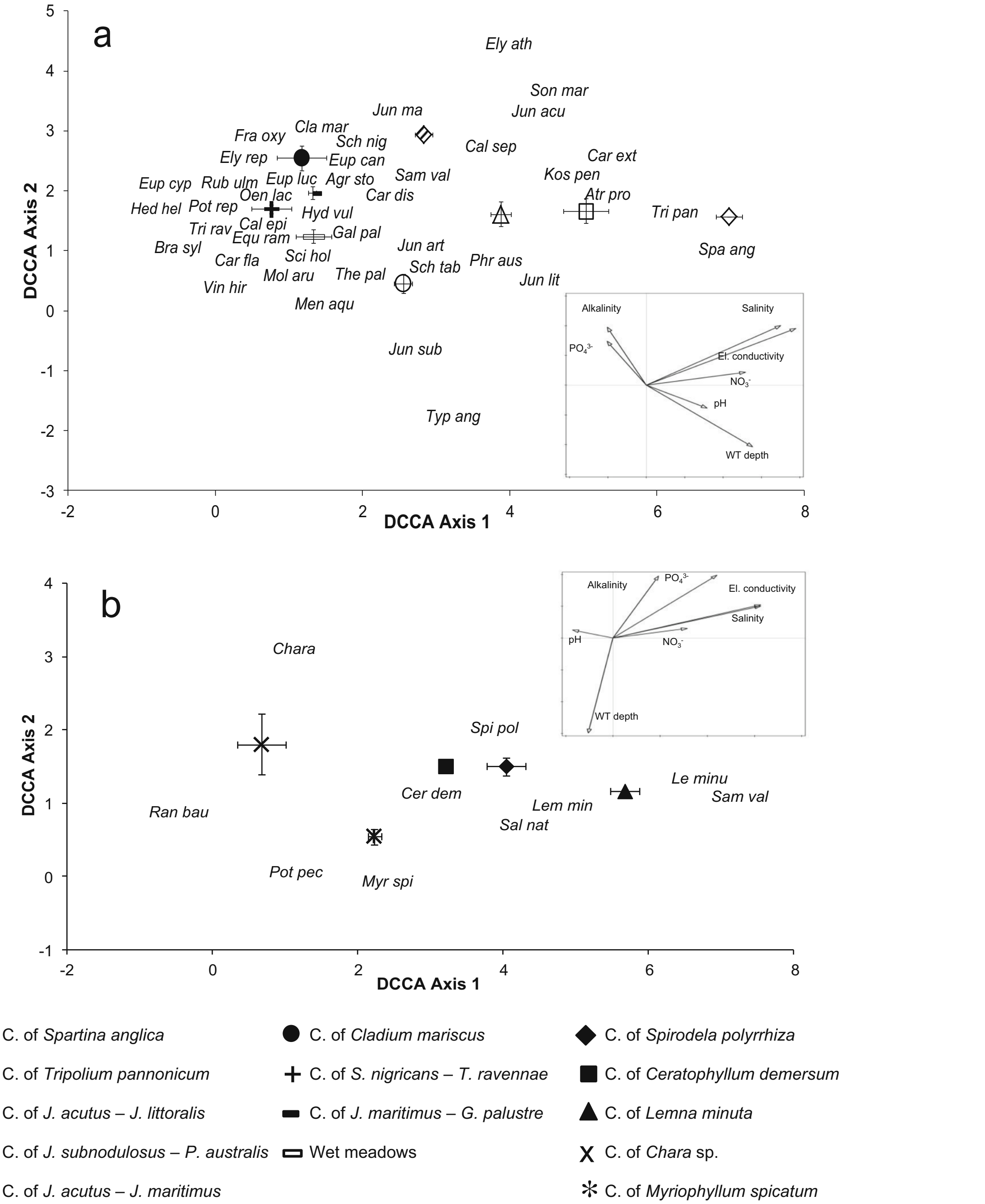


Fig. 3 Species scores and environmental variables scores on the first two DCCA axes for the hygrophytic and helophytic vegetation (**a**) and for the waterplant vegetation (**b**) Agr sto *Agrostis stolonifera*, Atr pro *Atriplex prostrata*, Bra syl *Brachypodium sylvaticum*, Cal epi *Calamagrostis epigejos*, Cal sep *Calystegia sepium*, Car dis *Carex distans*, Car ext *Carex extensa*, Car fla *Carex flacca*, Cer dem *Ceratophyllum demersum*, Chara *Chara* sp. (including *C. vulgaris* and *C. intermedia*), Cla mar *Cladium mariscus*, Ely ath *Elymus athericus*, Ely rep *Elymus repens*, Eup can *Eupatorium cannabinum*, Eup cyp *Euphorbia cyparissias*, Eup luc *Euphorbia lucida*, Gal pal *Galium palustre*, Hyd vul *Hydrocotyle vulgaris*, Jun acu *Juncus acutus*, Jun art *Juncus articulatus*, Jun lit *Juncus littoralis*, Jun mar *Juncus maritimus*, Jun sub *Juncus subnodulosus*, Kos pen *Kosteletzkya pentacarpos*, Lem min *Lemna minor*, Le minu *Lemna minuta*, Men aqu *Mentha aquatica*, Mol aru *Molinia arundinacea*, Myr spi *Myriophyllum spicatum*, Oen lac *Oenanthe lachenalii*, Phr aus *Phragmites australis*, Pot pec *Potamogeton pectinatus*, Ran bau *Ranunculus peltatus* ssp. *baudotii*, Rub ulm *Rubus ulmifolius*, Sal nat *Salvinia natans*, Sam val *Samolus valerandi*, Sch tab *Schoenoplectus tabernaemontani*, Sch nig *Schoenus nigricans*, Sci hol *Scirpoides holoschoenus*, Son mar *Sonchus maritimus*, Spa ang *Spartina anglica*, Spi pol *Spirodela polyrrhiza*, The pal *Thelypteris palustris*, Tri rav *Tripidium ravennae*, Tri pan *Tripolium pannonicum*, Typ ang *Typha angustifolia*, Vin hir *Vincetoxicum hirundinaria*. The symbols indicate the centroids (mean ± SE) of the relevé scores for the 14 plant communities

Table 5 Summary of Monte Carlo statistics for the hygrophytic and helophytic vegetation (upper part) and for the waterplant vegetation (lower part). Significant ($P < 0.05$) P adjusted values in bold character

	Contribution (%)	Pseudo-F	P	P adjusted
Hygrophytic and helophytic vegetation				
Electrical conductivity	32.1	6.8	0.002	**0.014**
Water-table depth	18.8	4.2	0.002	**0.014**
Alkalinity	13.2	3.1	0.002	**0.014**
Nitrate	12.7	3.1	0.002	**0.014**
Phosphate	8.7	2.2	0.004	**0.02**
pH	7.6	1.9	0.004	**0.02**
Salinity	6.9	1.8	0.046	**0.046**
Waterplant vegetation				
Electrical conductivity	36.5	6.6	0.002	**0.014**
Water-table depth	16.4	3.4	0.004	**0.02**
Alkalinity	14.6	3.4	0.004	**0.02**
Salinity	12.2	2.3	0.056	0.224
pH	8.1	1.9	0.076	0.228
Nitrate	6.0	1.5	0.22	0.416
Phosphate	6.1	1.5	0.208	0.416

Only three out of the seven environmental variables (electrical conductivity, water-table depth and alkalinity) did significantly account for the observed variation in waterplant vegetation (Table 5). The five waterplant communities were arranged from left to right according to increasing values of electrical conductivity from the communities of *Ceratophyllum demersum*, *Chara* sp., and *Myrophyllum spicatum* to the community of *Spirodela polyrrhiza* and, especially, the community of *Lemna minuta* (Fig. 3b; Table 4). The ordination of the five waterplant communities along the vector of water-table depth reflected a gradient of increasing water-table depth from the communities of *Chara* sp. and *Lemna minuta* to the communities of *Ceratophyllum demersum* and *Myriophyllum spicatum*. Similar to the hygrophytic and helophytic vegetation, the vector of alkalinity was oriented in opposite direction to that of water-table depth. Accordingly, mean alkalinity levels were somewhat higher in the communities of *Chara* sp. and *Lemna minuta* than in the communities of *Ceratophyllum demersum* and *Myriophyllum spicatum*. The community of *Spirodela polyrrhiza* had intermediate levels of both water-table depth and alkalinity (Fig. 3b; Table 4). Contrary to helophytic and hygrophytic vegetation, waterplant vegetation was not affected by pH, nitrate or phosphate concentrations, and salinity (Table 5). Mean salinity levels always were < 1000 mg L^{-1}, indicating no saltwater influx in any of the waterplant communities.

Discussion

Conservation Values and Ecological Gradients The rich variety of plant communities recorded in the Nature Reserve reflected strong environmental differences among wetlands that ranged from natural to artificial, lentic to lotic, permanent to temporary, and freshwater to saline wetland types. The species composition of the plant communities reflected such environmental heterogeneity. Indeed, species diversity is strongly affected by niche differentiation, in turn depending on relationships between environmental heterogeneity and species diversity at different scale levels (Lundholm 2009; Lastrucci et al. 2015). The wetland vegetation in the study area was rather heterogeneous in terms of structural diversity as well, because it consisted of plant communities dominated by hygrophyes, helophytes, pleustophytes, or rhizophytes. Most of the plant communities in the Nature Reserve had good correspondence with Natura 2000 habitat types. Several of these habitats presented high conservation value based on the species rarity index, although none of them falls within the Community priority habitats. Interestingly, the community of *Juncus subnodulosus* and *Phragmites australis* presented one of the highest rarity values even if it does not fall under any of the Natura 2000 habitat types. This supports the results of recent studies showing that, especially in Mediterranean wetlands, plant communities not corresponding to Natura 2000 habitat types may be extremely important from the conservation viewpoint because they host locally rare and threatened species (Angiolini et al. 2017; Viciani et al. 2017).

The multivariate ordination pointed to electrical conductivity, closely related to salinity, as the main ecological factor structuring the hygrophytic and helophytic vegetation in the Nature Reserve. The three communities at the high end of the salinity gradient, i.e., the community of *Juncus acutus* and *J. littoralis* and especially the communities of *Tripolium pannonicum* and *Spartina anglica*, only occurred in brackish wetlands close to the adjacent lagoon where salinity levels were highest. The community of *Juncus subnodulosus* and *Phragmites australis*, the community of *Schoenus nigricans* and *Tripidium ravennae*, and the wet meadows were all situated at greater distance from the lagoon in habitats with poor if any saltwater influx. However, some of the species most frequently occurring in these communities, especially *Phragmites australis* and to a lesser extent *Schoenus nigricans*, were also found in moderately saline habitats owing to their rather broad ecological amplitude with respect to salinity level (Lissner and Schierup 1997; Bernhardt and Kropf 2006). The community of *Cladium mariscus* was located in the mid part of the salinity gradient. *Cladium mariscus* also possesses high tolerance to salinity. Indeed, *Cladium mariscus*-dominated communities are quite frequent both in inland areas free from saltwater influx and in coastal habitats with quite high salt content (Géhu and Biondi 1988; Landucci et al. 2013). Nutrient levels also played a significant role in structuring the hygrophytic and helophytic vegetation. However, nitrate and phosphate appeared to affect differently the hygrophytic and helophytic communities in the Nature Reserve. The high nitrate levels detected in the saline habitats colonized by the communities of *Spartina anglica* and *Tripolium pannonicum* were determined by riverine N influx into the lagoon (Tappin 2002). The saltwater progressing landwards as an effect of saline wedge intrusion became progressively impoverished in dissolved N probably because the plant species in the communities of *Spartina anglica* and *Tripolium pannonicum* were effective in absorbing nitrate that was thus removed from the soil water (Sousa et al. 2008). Hence, the habitats with low to moderate salinity had lower nitrate concentrations and were characterized by species usually regarded as sensitive to high N load such as *Juncus subnodulosus* (Ceschin et al. 2010), *Cladium mariscus* (Landucci et al. 2013), and *Schoenus nigricans* (Bakker et al. 2005). Contrary to nitrate, phosphate was not associated with salinity but rather with alkalinity which resulted in highest phosphate concentration and highest alkalinity level in the salt-poor community of *Juncus maritimus* and *Galium palustre*. A possible explanation consists in a higher fraction of phosphate being incorporated on calcium phosphate in the soil of this community (Turner 2008).

The waterplant vegetation in the Nature Reserve was strongly conditioned by water-table depth. Water-table depth generally is a major driver in affecting species composition of waterplant communities (Edvardsen and Økland 2006; Klosowski and Jabłońska 2009; Hrivnák et al. 2014; Sakurai et al. 2017). The community of *Chara* sp. was located at the low end of the water-table depth gradient. In these habitats, the water table fluctuated strongly depending on precipitation, but the mean water-table depth was lower than in all other waterplant communities. The temporary character of this kind of wetlands represents a favorable condition for species such as *Ranunculus peltatus* subsp. *baudotii* or *Chara* sp. (see also Melendo et al. 2003; Florencio et al., 2014). Water bodies with relatively shallow water were colonized by small pleustophytes in the communities of *Spirodela polyrrhiza* and *Lemna minuta*. In contrast, bigger pleustophytes or rhyzophytes in the communities of *Ceratophyllum demersum* and *Myriophyllum spicatum* were found in deeper water in line with results of previous studies (Buchwald 1994; Lastrucci et al. 2014; Sakurai et al. 2017). An additional factor responsible for variation in the waterplant vegetation of the Nature Reserve was electrical conductivity. Unlike what we observed for the hygrophytic and helophytic vegetation, relationships between electrical conductivity and nitrate or phosphate concentrations were not straightforward for the waterplant vegetation. Neither was electrical conductivity related to salinity as all of the waterplant communities were situated in freshwater habitats. This suggests that electrical conductivity in the waterplant habitats mirrored the total load of nutrients with no close association with specific components such as nitrate or phosphate. The community of *Chara* sp. presented rather low values of electrical conductivity which indicates quite low levels of dissolved nutrients in the water. However, the species occurring in the community of *Chara* sp. have been found to tolerate quite high levels of dissolved nutrients in the water. For example, Ceschin et al. (2010) considered *Potamogeton pectinatus* as an indicator of moderate eutrophication. *Chara vulgaris* has rather wide ecological amplitude with respect to nutrient levels, ranging from mesotrophic to eutrophic waters (Lambert-Servien et al. 2006). *Ranunculus peltatus* ssp. *baudotii* has also been frequently observed in mesotrophic to eutrophic waters (Brullo et al. 2001; Melendo et al. 2003). The low nutrient levels recorded in the community of *Chara* sp. probably depended on the peculiar hydrological features of the watering pools where water supply prevalently comes from precipitation rather than from the watertable. The communities dominated by bigger plants, either pleustophytes (*Ceratophyllum demersum*) or rhyzophytes (*Myriophyllum spicatum*), had lower mean values of electrical conductivity compared with the communities dominated by small pleustophytes (*Lemna minuta* and *Spirodela polyrrhiza*). This may suggest that these two types of waterplant communities were associated with waterbodies having differing levels of dissolved nutrients. As all of these communities were found in canals receiving nutrient-rich water from the surrounding arable land, a more likely explanation consists in greater amounts of dissolved nutrients being removed by plant uptake in the communities where plant biomass was higher.

Environmental Changes and Vegetation Dynamics The present state of the wetlands in the Natural Reserve is the result of the history and the hydraulic management of the Reserve and of the surrounding areas. Reclamation of freshwater wetlands that once surrounded the Nature Reserve has altered the balance of the aquifer, enhancing the intrusion of the saline wedge. Furthermore, increasing subsidence rates hamper the hydrological

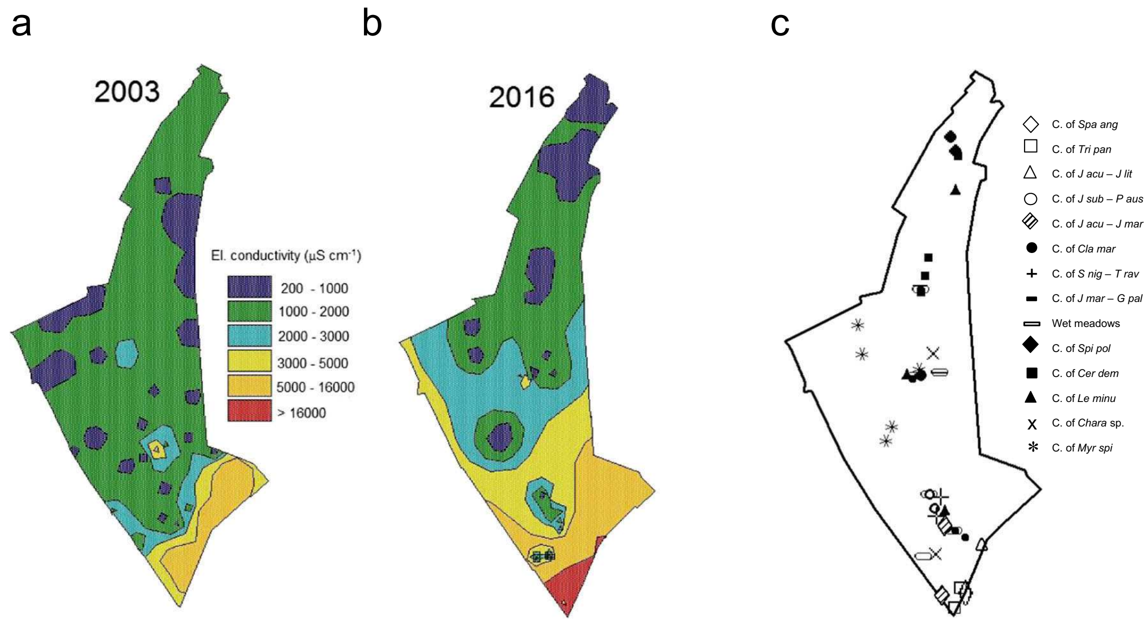


Fig. 4 Maps of the spatial distribution of electrical conductivity in 2003 (**a**) and in 2016 (**b**) and location of the relevés for the 14 plant communities (**c**) with abbreviated legend for the communities (full legend in Fig. 3)

turnover in the Natural Reserve (Caschetto et al. 2016), which is largely situated below the sea level. As a combined effect of these causes, electrical conductivity in the water is strongly increasing (2034 ± 343 µS cm^{-1} in 2003 vs. 7410 ± 1210 µS cm^{-1} in 2016; Fig. 4). Progressing saltwater intrusion may imply expansion of halophilic species and communities in the hygrophytic and helophytic vegetation. Hence, rare species occurring in poorly to moderately saline habitats, such as *Euphorbia lucida*, *Hydrocotyle vulgaris*, *Teucrium scordium* and, to a lesser extent, *Juncus subnodulosus* and *Cladium mariscus* may be out-competed by species preferring more strongly saline conditions. Saltwater may also impact waterplant species and communities if the intrusion of the saline wedge turns freshwater into brackish water. This may have negative effects on the locally rare species *Chara vulgaris*, *C. intermedia*, and *Ranunculus peltatus* ssp. *baudotii*.

Two sources of eutrophication may impact the wetlands in the Nature Reserve. The main source of eutrophication consists in influx of nutrient-rich saltwater by the intrusion of saline wedge from the adjacent lagoon. Our results show that the vegetation of brackish wetlands presently acts as an effective filter in removing nutrients, thus reducing the influx of nutrients to oligotrophic habitats. However, if the nutrient load in the lagoon increases, this may imply degradation of salt-tolerant vegetation (Deegan et al. 2012) which may hamper the filtering capacity of vegetation. An additional source of wetland eutrophication in the Nature Reserve is the network of canals conveying water from the surrounding arable land to the Nature Reserve. Agriculture is the main source of nutrient, especially nitrate, load with agricultural activities accounting for about 60% of nitrogen (N) compounds influx in the waters of arable lands in Northern Italy (Castaldelli et al. 2013). Eutrophication may increase in the near future because of a derogation from European rules (Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources) allowing farmers to raise the amount of manure applied to arable farms up to 250 kg N ha^{-1} year^{-1}. If nitrate load increases in the next future, this may have negative effects on freshwater species preferring nitrate-poor habitats such as *Cladium mariscus* or *Juncus subnodulosus*. Both of these species are rather sensitive to eutrophication and may thus be out-competed by more tolerant species, especially *Phragmites australis* (see Viciani et al. 2017). On the other hand, the endangered species *Kosteletzkya pentacarpos* may benefit from future increases in nitrate levels (Abeli et al. 2017).

In conclusion, the wetlands of the Reserve are subject to a series of threats deriving from a complex interplay of environmental changes and anthropogenic alterations. Improved hydraulic management including more accurate maintenance and higher flow rate in the canals, with consequently increased drainage capacity, could counter the saline wedge intrusion (Antonellini et al. 2015). Furthermore, hydraulic management should be oriented to create a system of freshwater habitats around the area in order to reduce the influx of nitrate-rich water from canals draining agricultural land.

Acknowledgements Prof. Giuseppe Castaldelli provided useful data and comments on nitrate loads. Dr. Roberta Marchesini did the chemical analyses. Mrs. Lorella Dell'Olmo and Dr. Roberta Marchesini assisted in drawing the maps. Mr. Mauro Menghini helped during the field work. All are kindly acknowledged. We wish to thank two anonymous reviewers for constructive criticism on a previous version of the paper. This study was partly financed by a Grant of Ferrara University to RG.

References

Abeli, T., L. Brancaleoni, R. Marchesini, S. Orsenigo, G. Rossi, and R. Gerdol. 2017. Fertilizer application positively affects plants performance but reduces seed viability in seashore mallow (*Kosteletzkya pentacarpos*): Implication for biomass production and species conservation. *Annals of Applied Biology* 170 (1): 263–272. https://doi.org/10.2179/11-022.

Angelini, P., L. Casella, A. Grignetti, and P. Genovesi. 2016. *Manuali per il monitoraggio di specie e habitat di interesse comunitario (Direttiva 92/43/CEE) in Italia: habitat. ISPRA, Serie Manuali e linee guida, 142/2016*. Rome: ISPRA.

Angiolini, C., D. Viciani, G. Bonari, and L. Lastrucci. 2017. Habitat conservation prioritization: A floristic approach applied to a Mediterranean wetland network. *Plant Biosystems* 151 (4): 598–612. https://doi.org/10.1080/11263504.2016.1187678.

Antonellini, M., D.M. Allen, P.N. Mollema, D. Capo, and N. Greggio. 2015. Groundwater freshening following coastal progradation and land reclamation of the Po plain, Italy. *Hydrogeology Journal* 23 (5): 1009–1026. https://doi.org/10.1007/s10040-015-1263-0.

Bakker, C., J. Rodenburg, and P.M. van Bodegom. 2005. Effects of Ca- and Fe-rich seepage on P availability and plant performance in calcareous dune soils. *Plant and Soil* 275 (1-2): 111–122. https://doi.org/10.1007/s11104-005-0438-1.

Bedford, B.L., D.J. Leopold, and J.P. Gibbs. 2001. Wetlands ecosystems. In *Encyclopedia of Biodiversity (4th vol)*, ed. S.A. Levin, 781–804. Orlando: Academic Press.

Bernhardt, K.G., and M. Kropf. 2006. *Schoenus nigricans* (Cyperaceae) xerophytic grasslands on the NE Adriatic islands Cres and Krk (Croatia). *Acta Botanica Croatica* 65: 127–136.

Bilz, M., S.P. Kell, N. Maxted, and R.V. Lansdown. 2011. *European red list of vascular plants*. Luxembourg: Publications Office of the European Union.

Biondi, E. 2011. Phytosociology today: Methodological and conceptual evolution. *Plant Biosystems* 145 (Suppl): 19–29. https://doi.org/10.1080/11263504.2011.602748.

Biondi, E., and C. Blasi. 2009. *Manuale italiano di interpretazione degli Habitat della Direttiva 92/43/CEE*. http://vnr.unipg.it/habitat. Accessed 18 May 2017.

Braun-Blanquet, J. 1932. *Plant sociology: The study of plant communities*. New York: McGraw-Hill.

Brock, M.A., D.L. Nielsen, and K. Crosslé. 2005. Changes in biotic communities developing from freshwater wetland sediments under experimental salinity and water regimes. *Freshwater Biology* 50 (8): 1376–1390. https://doi.org/10.1111/j.1365-2427.2005.01408.x.

Brullo, S., F. Scelsi, and G. Spampinato. 2001. *La vegetazione dell'Aspromonte. Studio fitosociologico*. Laruffa: Reggio Calabria.

Buchwald, R. 1994. Vegetazione e odonatofauna negli ambienti acquatici dell'Italia Centrale. *Braun-Blanquetia* 11: 3–77.

Caschetto, M., N. Colombani, M. Mastrocicco, M. Petitta, and R. Aravena. 2016. Estimating groundwater residence time and recharge patterns in a saline coastal aquifer. *Hydrological Processes* 30 (22): 4202–4213. https://doi.org/10.1002/hyp.10942.

Castaldelli, G., E. Soana, E. Racchetti, E. Pierobon, M. Mastrocicco, E. Tesini, E.A. Fano, and M. Bartoli. 2013. Nitrogen budget in a lowland coastal area within the Po River basin (Northern Italy): Multiple evidences of equilibrium between sources and internal sinks. *Environmental Management* 52 (3): 567–580. https://doi.org/10.1007/s00267-013-0052-6.

Ceschin, S., V. Zuccarello, and G. Caneva. 2010. Role of macrophyte communities as bioindicators of water quality: Application on the Tiber River basin (Italy). *Plant Biosystems* 144 (3): 528–536. https://doi.org/10.1080/11263500903429221.

Commission of the European Community. 1992. Council directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Official Journal* L206. 22 July 1992 (Consolidated version 1 January 2007).

Commission of the European Community. 2013. *Interpretation manual of European Union habitats, vers. EUR28*. Brussel: European Commission, DG Environment.

Davis, J.A., and R. Froend. 1999. Loss and degradation of wetlands in southwestern Australia: Underlying causes, consequences and solutions. *Wetlands Ecology and Management* 7 (1): 13–23. https://doi.org/10.1023/A:1008400404021.

Deegan, L.A., D.S. Johnson, R.S. Warren, B.J. Peterson, J.W. Fleeger, S. Fagherazzi, and W.M. Wollheim. 2012. Coastal eutrophication as a driver of salt marsh loss. *Nature* 490: 388–392. https://doi.org/10.1038/nature11533.

Den Hartog, C., and S. Segal. 1964. A new classification of the water-plant communities. *Acta Botanica Neerlandica* 13 (3): 367–393. https://doi.org/10.1111/j.1438-8677.1964.tb00163.x.

Donnelly, J.P., and M.D. Bertness. 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. *Proceedings of the National Academy of Sciences of the United States of America* 98 (25): 14218–14223. https://doi.org/10.1073/pnas.251209298.

Dudgeon, D., A.H. Arthington, M.O. Gessner, Z. Kawabata, D.J. Knowler, C. Lévêque, and C.A. Sullivan. 2006. Freshwater biodiversity: Importance, threats, status and conservation challenges. *Biological Reviews of the Cambridge Philosophical Society* 81 (02): 16–182. https://doi.org/10.1017/S1464793105006950.

Edvardsen, A., and R.H. Økland. 2006. Variation in plant species richness in and adjacent to 64 ponds in SE Norwegian agricultural landscapes. *Aquatic Botany* 85 (2): 79–91. https://doi.org/10.1016/j.aquabot.2006.01.015.

Evans, D., and M. Arvela. 2011. Assessment and reporting under article 17 of the habitats directive. Explanatory notes & guidelines for the period 2007–2012. Final version. July 2011. ETC-BD. European Commission, Brussel.

Florencio, M., L. Serrano, P. Siljestrom, R. Fernandez-Zamudio, P. Garcia-Murillo, and C. Díaz-Paniagua. 2014. The influence of geomorphology on the composition of aquatic flora and fauna within a temporary pond network. *Limnetica* 33: 327–340.

García-Madrid, A.S., J.A. Molina, and P. Cantó. 2014. Classification of habitats highlights priorities for conservation policies: The case of Spanish Mediterranean tall humid herb grasslands. *Journal for Nature Conservation* 22 (2): 142–156. https://doi.org/10.1016/j.jnc.2013.10.002.

Géhu, J.M., and E. Biondi. 1988. Données sur la végétation des ceintures d'atterrissement des lacs Alimini (Salento, Italie). *Documents Phytosociologiques* 11: 353–380.

Gerdol, R., C. Ferrari, and F. Piccoli. 1985. Correlation between soil characters and forest types: A study in multiple discriminant analysis. *Vegetatio* 60 (1): 49–56. https://doi.org/10.1007/BF00053911.

Gigante, D., F. Attorre, R. Venanzoni, A.T.R. Acosta, E. Agrillo, M. Aleffi, and S. Zitti. 2016. A methodological protocol for annex I habitat monitoring: The contribution of vegetation science. *Plant Sociology* 53: 77–87. https://doi.org/10.7338/pls2016532/06.

Hrivnák, R., J. Kochjarová, H. Otáhelóvá, P. Palóve-Balang, M. Slezák, and P. Slezák. 2014. Environmental drivers of macrophyte species richness in artificial and natural aquatic water bodies—Comparative approach from two central European regions. *Annales de Limnologie – International Journal of Limnology* 50 (4): 269–278. https://doi.org/10.1051/limn/2014020.

Janssen, J.A.M. et al. 2016. European red list of habitats. Part 2. Terrestrial and freshwater habitats. England: European Union, pp. 44. https://doi.org/10.2779/091372. http://ec.europa.eu/environment/nature/knowledge/pdf/terrestrial_EU_red_list_report.pdf.

Klosowski, S., and E. Jabłońska. 2009. Aquatic and swamp plant communities as indicators of habitat properties of astatic water bodies in North-Eastern Poland. *Limnologica* 39 (2): 115–127. https://doi.org/10.1016/j.limno.2008.01.003.

Lacoul, P., and B. Freedman. 2006. Relationships between aquatic plants and environmental factors along a steep Himalayan altitudinal gradient. *Aquatic Botany* 84 (1): 3–16. https://doi.org/10.1016/j.aquabot.2005.06.011.

Lambert-Servien, E., G. Clemenceau, O. Gabory, E. Douillard, and J. Haury. 2006. Stoneworts (*Characeae*) and associated macrophyte species as indicators of water quality and human activities in the Pays-de-la-Loire region, France. *Hydrobiologia* 570 (1): 107–115. https://doi.org/10.1007/s10750-006-0169-1.

Landucci, F., D. Gigante, R. Venanzoni, and M. Chytrý. 2013. Wetland vegetation of the class *Phragmito-Magno-Caricetea* in Central Italy. *Phytocoenologia* 43 (1): 67–100. https://doi.org/10.1127/0340-269X/2013/0043-0545.

Lastrucci, L., G. Bonari, C. Angiolini, F. Casini, T. Giallonardo, D. Gigante, M. Landi, F. Landucci, R. Venanzoni, and D. Viciani. 2014. Vegetation of lakes Chiusi and Montepulciano (Siena, Central Italy): Updated knowledge and new discoveries. *Plant Sociology* 51: 29–55. https://doi.org/10.7338/pls2014512/03.

Lastrucci, L.L., T. Guidi Lazzaro, V. Gonnelli, P. Giordani, and R. Benesperi. 2015. Different components of plant diversity suggest the protection of a large area for the conservation of a riparian ecosystem. *Biologia* 70 (8): 1033–1041. https://doi.org/10.1515/biolog-2015-0115.

Lissner, J., and H.H. Schierup. 1997. Effects of salinity on the growth of *Phragmites australis*. *Aquatic Botany* 55 (4): 247–260. https://doi.org/10.1016/S0304-3770(96)01085-6.

Lundholm, J.T. 2009. Plant species diversity and environmental heterogeneity: Spatial scale and competing hypotheses. *Journal of Vegetation Science* 20 (3): 377–391. https://doi.org/10.1111/j.1654-1103.2009.05577.x.

Melendo, M., E. Cano, and F. Valle. 2003. Synopsis of aquatic plant-communities of the class *Potametea* in the southern Iberian Peninsula. *Acta Botanica Gallica* 150 (4): 429–444.

Nicholls, R.J., F.M.J. Hoozemans, and M. Marchand. 1999. Increasing flood risk and wetland losses due to global sea-level rise: Regional and global analyses. *Global Environmental Change* 9: 69–87. https://doi.org/10.1016/S0959-3780(99)00019-9.

Orlóci, L. 1978. *Multivariate analysis in vegetation research*. 2nd ed. The Hague: Junk.

Pellizzari, M., and F. Piccoli. 2001. La vegetazione dei corpi idrici del Bosco della Mesola (Delta del Po). *Quaderni della Stazione Ecologica del Civico Museo di Storia Naturale di Ferrara* 13: 7–24.

Piccoli, F., and R. Gerdol. 1984. Typology and dynamics of a wood in the Po plane (N-Italy): The “Bosco della Mesola”. *Colloques Phytosociologiques* 9: 161–170.

Piccoli, F., R. Gerdol, and C. Ferrari. 1983. Carta della vegetazione del Bosco della Mesola (Ferrara). *Atti dell’Istituto Botanico e Laboratorio Crittogamico dell’Università di Pavia* 7 (2): 3–23.

Piccoli, F., M. Pellizzari, and A. Alessandrini. 2014. *Flora del Ferrarese. Istituto per i Beni Artistici e Culturali. Regione Emilia-Romagna*. Bologna: Longo.

Pott, R. 2011. Phytosociology: A modern geobotanical method. *Plant Biosystems* 145 (Suppl): 9–18. https://doi.org/10.1080/11263504.2011.602740.

Rossi, G., S. Orsenigo, C. Montagnani, G. Fenu, D. Gargano, L. Peruzzi, R.P. Wagensommer, B. Foggi, G. Bacchetta, G. Domina, F. Conti, F. Bartolucci, M. Gennai, S. Ravera, A. Cogoni, S. Magrini, R. Gentili, M. Castello, C. Blasi, and T. Abeli. 2016. Is legal protection sufficient to ensure plant conservation? The Italian Red List of policy species as a case study. *Oryx* 50 (03): 431–436. https://doi.org/10.1017/S003060531500006X.

Sakurai, Y., K. Yabe, and K. Katagiri. 2017. Factors controlling changes in the aquatic macrophyte communities from 1984 to 2009 in a pond in the cool-temperate zone of Japan. *Limnology* 18 (2): 153–166. https://doi.org/10.1007/s10201-016-0498-3.

Sousa, A., A.I. Lillebø, I. Caçador, and M. Pardal. 2008. Contribution of *Spartina maritima* to the reduction of eutrophication in estuarine systems. *Environmental Pollution* 156 (3): 628–635. https://doi.org/10.1016/j.envpol.2008.06.022.

Spalding, E.A., and M.W. Hester. 2007. Interactive effects of hydrology and salinity on oligohaline plant species productivity: Implications of relative sea-level rise. *Estuaries and Coasts* 30 (2): 214–225.

Stampi, P. 1966. Il Gran bosco della Mesola (Ferrara): notizie storiche, floristiche e geobotaniche. *Annali di Botanica (Roma)* 28: 599–612.

Tappin, A.D. 2002. An examination of the fluxes of nitrogen and phosphorus in temperate and tropical estuaries: Current estimates and uncertainties. *Estuarine Coastal and Shelf Science* 55 (6): 885–901. https://doi.org/10.1006/ecss.2002.1034.

ter Braak, C.J.F., and P. Šmilauer. 2012. *Canoco reference manual and User’s guide: Software for ordination (version 5.0)*. Ithaca: Microcomputer Power.

Turner, R.E. 1990. Landscape development and coastal wetland losses in the Northern Gulf of Mexico. *American Zoologist* 30 (1): 89–105.

Turner, B.L. 2008. Resource partitioning for soil phosphorus: A hypothesis. *Journal of Ecology* 96 (4): 698–702. https://doi.org/10.1111/j.1365-2745.2008.01384.x.

Van der Maarel, E. 1979. Transformation of cover-abundance values in phytosociology and its effects on community similarity. *Vegetatio* 39: 97–144.

Viciani, D., L. Lastrucci, L. Dell’Olmo, G. Ferretti, and B. Foggi. 2014. Natura 2000 habitats in Tuscany (Central Italy): Synthesis of main conservation features based on a comprehensive database. *Biodiversity and Conservation* 23 (6): 1551–1576. https://doi.org/10.1007/s10531-014-0686-6.

Viciani, D., L. Dell’Olmo, C. Vicenti, and L. Lastrucci. 2017. Natura 2000 protected habitats, Massaciuccoli Lake (northern Tuscany, Italy). *Journal of Maps* 13: 219–226.

Watt, S.C.L., E.G. Berthou, and L. Vilar. 2007. The influence of water level and salinity on plant assemblages of a seasonally flooded Mediterranean wetland. *Plant Ecology* 189 (1): 71–85. https://doi.org/10.1007/s11258-006-9167-7.

Zaldivar, J.M., A.C. Cardoso, P. Viaroli, A. Newton, R. de Wit, C. Ibanez, S. Reizopoulou, F. Somma, A. Razinkovas, A. Basset, M. Holmer, and N. Murray. 2008. Eutrophication in transitional waters: an overview. *Transitional Waters Monographies* 1: 1–78.

Part 2

Human impacts on deltas and coastal wetlands degrading Ecosystem Services (ES). Threats deriving from a complex interplay of environmental changes and anthropogenic alterations causing dramatic losses of ecological functions.

Coastal brackish lagoon named Sacca di Goro is one of the few remaining in the upper Adriatic that were widespread before land reclamations.

Clam production produces fast spatio-temporal changes of aquatic vegetated habitats. The analyses showed that 87% of emerging vegetation was lost during the period 1954-2008, while aquaculture production increased rapidly.

The results demonstrated that sectorial management was ineffective to maintain ESs by promoting the exploitation of few provisioning services while decreasing many others.

Restoring aquatic vegetation to offset anthropic impacts have great potential for the future sustainable governance of deltas.

Gaglio M., Lanzoni M., Nobili G., Viviani D., Castaldelli G., Fano E. A. (2019) Ecosystem services approach for sustainable governance in a brackish water lagoon used for aquaculture. *Journal of Environmental Planning and Management, Vol. 62, No. 9, 1501–1524*, https://doi.org/10.1080/09640568.2019.1581602

As a particular study case, the fire-induced changes in groundwater recharge rate was examined in the Ravenna coastal area.

Pine forest grow on coastal dune belts, overlying a sandy unconfined aquifer, which is strongly affected by marine ingression.

The estimated recharge rates increased in the partially and completely burnt areas compared with the pristine pine forest area. A post-fire decrease in salinity was observed across the burnt forest.

The fire highlighted a new ES and provided an opportunity to evaluate a new forest management approach.

Giambastiani B., Greggio N., Nobili G., Dinelli E., Antonellini M. (2018) Forest fire effects on groundwater in a coastal aquifer (Ravenna, Italy). *Hydrological Processes:1–13* https://doi.org/10.1002/hyp.13165

Journal of Environmental Planning and Management, 2019
Vol. 62, No. 9, 1501–1524, https://doi.org/10.1080/09640568.2019.1581602

Ecosystem services approach for sustainable governance in a brackish water lagoon used for aquaculture

Mattias Gaglio[a]*, Mattia Lanzoni[a], Giovanni Nobili[a], Diego Viviani[b], Giuseppe Castaldelli[a] and Elisa Anna Fano[a]

[a]*Department of Life Sciences and Biotechnology, University of Ferrara, Ferrara, Italy;* [b]*Municipality of Goro, Goro, Italy*

(*Received 26 April 2018; final version received 8 February 2019*)

Human impacts on deltas often involve reclamation of coastal wetlands, causing a dramatic loss of ecological functions. We propose an Ecosystem Services (ES) approach to promote coordinated governance of aquaculture and environmental conservation in a brackish lagoon of the Po River delta (Italy). Spatiotemporal changes of aquatic vegetated habitats and clam production were evaluated, and experimentally related to ESs: climate regulation, habitat provision for birdlife, and potential for birdwatching. Almost all emergent vegetation was lost during past decades, while aquaculture production increased rapidly. Vegetated habitats sequestered significant amounts of carbon and supported more diverse bird communities than non-vegetated wetlands, including protected species of interest for birdwatching. We demonstrated that sectoral management was ineffective in maintaining ESs, promoting the exploitation of few provisioning services while decreasing many others. We propose an innovative, integrated management that focuses on restoring aquatic vegetation to offset anthropic impacts for the future sustainable governance of deltas.

Keywords: Ecosystem Services; Po River delta; Clam aquaculture; Life AGREE; *Phragmites australis*; habitat restoration

1. Introduction

River deltas host a high degree of habitat diversity and provide several ecological functions to support human well-being (Ojeda, Mayer, and Solomon 2008; Schlüter, Leslie, and Levin 2009). At the same time, their high heterogeneity and productivity have attracted different human activities with relevant environmental impacts (Syvitski *et al*. 2009; Mikhailov and Mikhailova 2003). The conflicts arising from different potential uses, conservation, and legislative issues call for new solutions and views in the governance of deltaic areas. Strategic delta planning has been proposed as a promising planning approach based on a public sector-led process with a sustainable long-term vision to be implemented through actions and means that cover multiple domains (Seijger *et al*. 2017). However, its translation from theory to practice requires innovative solutions compelling public sector and stakeholders to a change of paradigm, from a sectoral to a coordinated long-term vision.

The concept of Ecosystem Services (ESs), defined as the benefit people obtain from ecosystems (Millennium Ecosystem Assessment 2005), has gained great

*Corresponding author. E-mail: gglmts@unife.it

popularity among researchers and decision-makers during the last two decades as a link between human well-being and nature conservation (Costanza *et al*. 1997; Daily 1997; Millennium Ecosystem Assessment 2005; de Groot *et al*. 2012; Maes *et al*. 2012). Recently, different studies demonstrated that this concept has a great potential to make informative decisions and address planning processes toward more sustainable choices in various domains, including renewable energies (Gissi *et al*. 2018; Gissi and Garramone 2018), river basin (Terrado *et al*. 2014), forest (Mori, Lertzman, and Gustafsson 2017), urban (Maragno *et al*. 2018), and maritime planning (Drakou *et al*. 2017).

Braat and de Groot (2012) shaped the ES approach in three main steps: (i) identification and assessment of ESs through quantification and mapping, (ii) value estimations in physical and monetary terms, and (iii) capturing and managing values. Haines-Young and Potschin (2010) conceptualized the so-called "cascade model" based on consequential steps to describe human–nature relationships. The model states that ESs cannot exist in isolation from human needs. The framework states that biophysical structures and processes deliver ecological functions that, in turn, can be considered as ecosystem services only when representing a benefit for human well-being. Hence, the ES theory assumes that people depend on ecosystems and their related functions. This implies that the maintenance of natural capital as the provider of ESs and its rational use through time are the basis for sustainable development.

The ES approach has also been proposed as an instrument for management decisions in protected areas (Gaglio, Aschonitis, Mancuso *et al*. 2017; García *et al*. 2017; Sallustio *et al*. 2017) and coastal planning (Drakou *et al*. 2017), since it contemplates multiple benefits deriving from ecosystems that are typically evaluated separately (Arkema *et al*. 2015) without explicitly considering neither natural capital nor ES values (Rall, Kabisch, and Hansen 2015; McKinley, Ballinger, and Beaumont 2018). However, the sectoral regulation framework has contributed to unmanaged conflicts between the different uses of natural resources, which has often resulted in the missing of sustainable development goals (Howes *et al*. 2017). Furthermore, adoption of nature-based solutions was recently proposed as an effective strategy to enhance the sustainability of socio-environmental systems by promoting multiple ESs (Keesstra *et al*. 2018; Saleh and Weinstein 2016). Specifically, restoration of degraded or previously converted ecosystems could recover several of the lost ecological functions (Cohen-Shacham *et al*. 2016).

River deltas are hot spots for biodiversity and deliver a wide range of ESs with relevant socio-economic value (Danovaro and Pusceddu 2007). In particular, the aquatic vegetation of coastal wetlands are an important asset of deltas' natural capital, since they (i) provide habitat for numerous species (Dealteris *et al*. 2004), (ii) play a fundamental role in nutrient and carbon cycling (Duke, Francoeur, and Judd 2015; Welsh *et al*. 2000), and (iii) increase aesthetic and recreational landscape values (Kiviat 2013).

As vegetated coastal ecosystems sequester and store significant amounts of carbon from the atmosphere (Duarte *et al*. 2013), restoring degraded coastal wetlands is a cost-effective approach to mitigate climate change, thus supplying a climate regulation service (Greiner *et al*. 2013). On the other hand, conversion or degradation of these ecosystems causes the release of the stored "blue carbon" to the atmosphere, thus exacerbating the ongoing climatic changes (Pendleton *et al*. 2012).

Coastal wetlands are also important hotspots for species conservation. In particular, various aquatic bird species are protected under the EU Birds Directive 2009/147/CE through designation of Special Protection Areas (SPA) that require the Member States to prohibit deterioration or destruction of breeding sites or resting places.

Various recreational activities in coastal areas can support the sustainable development of related zones and cater to increasing ecotourism demand in protected areas (Balmford *et al*. 2015). Among these, birdwatching is a growing sector of nature-based tourism (Puhakka, Salo, and Sääksjärvi 2011). The suitability of areas for birdwatching depends on the good ecological status of wetlands, the presence of rare taxa and bird abundance (Booth *et al*. 2011), which in turn, are strictly dependent on the presence of aquatic vegetation.

The aim of the current analytical study is to evaluate whether an ES approach relying on nature-based solutions (i.e. aquatic vegetation restoration) could assist the governance of deltaic areas towards more sustainable strategies to overcome the limitations of a sectoral and fragmented regulatory framework. We assessed the ESs in the Sacca di Goro lagoon, a coastal lagoon of the river Po delta (Italy) selected as a case study for the analysis and corresponds to the first step of the ES approach (Braat and de Groot 2012). Thereby, we assessed (i) a set of ESs provided by aquatic emergent vegetation (i.e. *Phragmites australis*-dominated ecosystems), including carbon storage, habitat provision for birdlife and potential for ecotourism (i.e. birdwatching), (ii) the spatiotemporal changes of aquatic vegetated habitats in the lagoon, and (iii) temporal evolution of aquaculture activities (a provisioning service), as the main driver of change in the lagoon. Additionally, we explored opportunities to operationalize subsequent steps of the ES approach by restoring aquatic vegetated habitats.

2. Methodology

2.1. Study area

2.1.1. The Po river Delta

The Po river delta is the largest deltaic area in Italy and one of the largest in Europe. Over the centuries, humans have significantly altered the sensitive natural and hydrological balances of the delta system through urbanizing watercourses and converting wetlands to croplands with the aim of gaining arable land for human development and improving hygienic conditions (Bondesan 1990; Cencini 1998; Gaglio, Aschonitis, Gissi, *et al*. 2017).

The environmental protection and management of the Po delta depends on two independent parks located in two different administrative Regions: the Regional Park of the Po delta – Emilia Romagna, established in 1988, corresponding to the “ancient” delta and the Regional Park of the Po delta – Veneto, established in 1997, which corresponds to the so-called “active” delta (i.e. part of the delta subjected to coastline evolution because of sediment delivery).

This administrative framework resulted in a lack of coordinated management of the area. To overcome such difficulties, in 2015, the two parks and the administrative bodies involved obtained the recognition of UNESCO Man and the Biosphere (MAB) of the whole delta.

2.1.2. The VMG district and the Sacca di Goro lagoon

The environmental management of the Regional Park of the Po delta-Emilia Romagna is organized into six districts (named stations, *stazioni* in Italian) (Figure 1). The

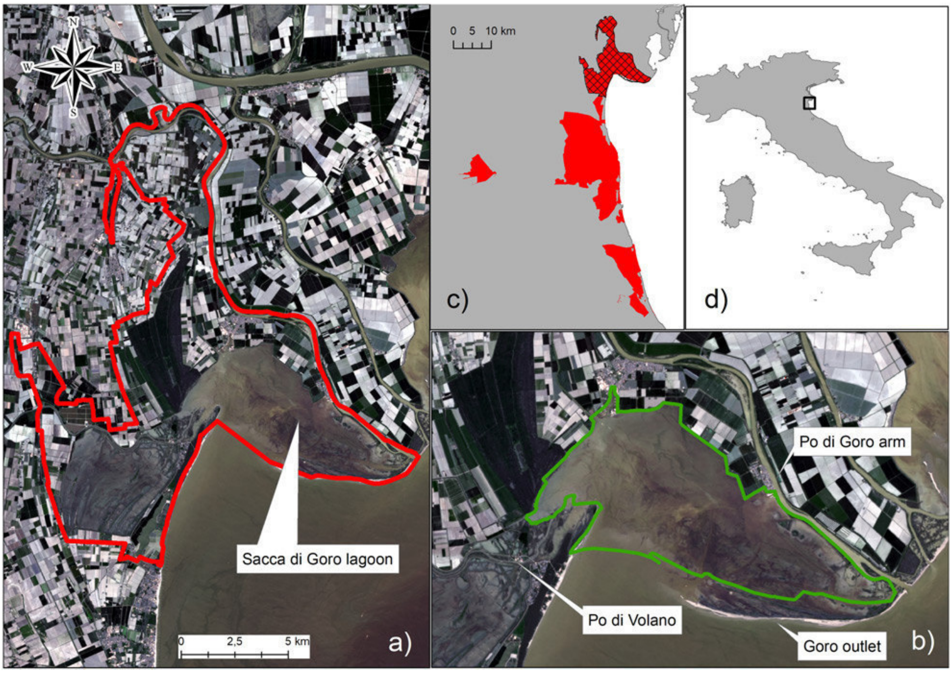


Figure 1. (a) The boundaries of VMG district (red line); (b) Sacca di Goro lagoon (green line); (c) Regional Park of Po Delta – Emilia Romagna (red areas) and VMG district (dashed line), and (d) localization of the Park in Italy.

Volano-Mesola-Goro (VMG) is the northernmost station and is the only one belonging to the active Po Delta, while the other five belong to the ancient delta. The VMG station, together with ecosystems typical of the ancient delta, is semi-closed lagoons managed for extensive aquaculture, coastal forests and reclaimed land. Sacca di Goro, a nascent lagoon in terms of formation, belongs to the present Po Delta and sediment delivery at this site is a very active process, occurring from the southernmost branch of the Po (i.e. Po di Goro). For this reason, the VMG district has been selected as a case study representing the Po delta in its entirety.

VMG covers 13,730 ha and includes different areas with high conservation values (Figure 1(a)). Among these, the Sacca di Goro lagoon is a coastal brackish lagoon of about 2,000 ha with an average depth of 1.5 m and a watershed of 860 km^2 whose hydrographic network is mainly artificially regulated.

It is one of the few remaining coastal lagoons in the upper Adriatic that were widespread before land reclamations. The lagoon receives saltwater from the sea and freshwater from the Po di Goro arm in the east and from Po di Volano River and Canal Bianco channel in the west (Figure 1(b)). The high heterogeneity of its environment results in three main zones: (i) the eastern part (named "*Valle di Gorino*"), with higher conservation value, (ii) the Goro outlet, a sandbank that divides the lagoon from the open sea, and (iii) the open part of the lagoon. Since the lagoon is prone to summer anoxic crises due to excessive algal blooms (*Ulva* spp.) that harm habitats and human activities (Viaroli *et al*. 1995), several hydraulic interventions were performed during recent decades.

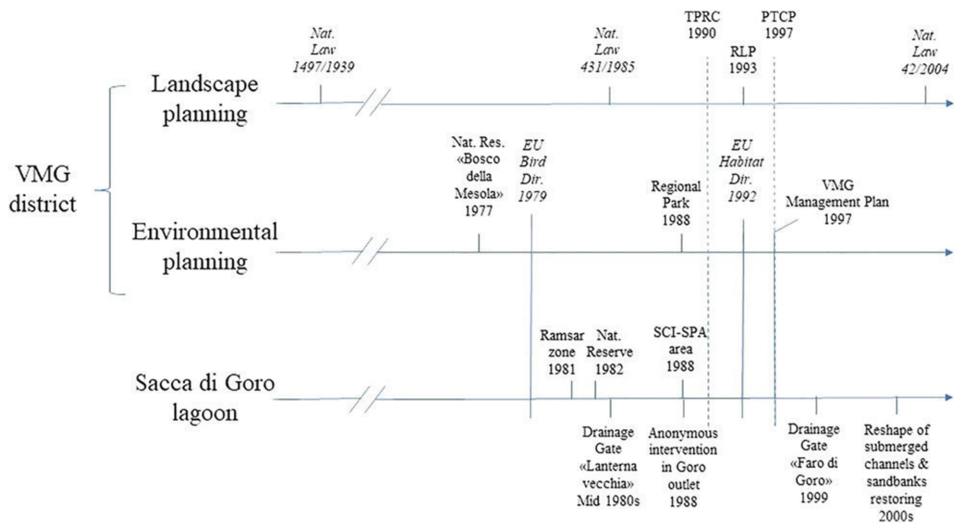


Figure 2. Evolution of landscape and environmental planning in the VMG district and Sacca di Goro lagoon. The main interventions performed in the Sacca di Goro lagoon are also provided.

The Sacca di Goro lagoon has been traditionally exploited for its high aquaculture productivity. Since 1987, farming of indigenous clam (*Tapes decussatus*) was abandoned in favor of the non-native Manila clam (*Ruditapes philippinarum*), since the latter exhibited higher growth rates and better environmental adaptability. Manila clam introduction led to the intensification of aquaculture practices in the Sacca di Goro lagoon.

Clam farming is managed based on geographically defined licensed areas. Since 1992, the former Fishers Consortium of Goro was split into several minor cooperatives resulting in an increase in total exploited area. Currently, the aquaculture in Sacca di Goro supports the local economy with high revenues and provides about 1,500 direct jobs and significant indirect employment (Bartoli *et al*. 2016).

2.1.3 Landscape and environmental planning evolution in VMG district

The evolution of different landscape and environmental planning instruments affecting the study area occurred together with the general increase in environmental awareness (Figure 2).

At European level, the Birds Directive (European Parliament 2009) and Habitat Directive (European Commission 1992) are milestones in EU environmental legislation.

At national scale, Law n.1497 of 29 June 1939, encompassing “natural beauty protection” is intended to safeguard aesthetic and cultural values derived from landscape and nature. It was the first attempt to deal with environmental conservation from an aesthetic perspective (Settis 2010). The rising awareness concerning environmental issues led to the institution of the Nation Law n.431 of 8 August 1985 (known as “Galasso Law”), the first organic law to protect landscape and environmental assets in Italy. Galasso law further regulates the free civic usage rights that allow members of the local community to benefit from an area under landscape classification. Subsequently, the law on cultural and landscape heritage 42/2004 (known as “Codice

Table 1. The four ESs mapped in the Sacca di Goro lagoon and respective indicators and categories.

Category	Ecosystem service	Indicator	Notes
Provisioning	Aquaculture	Quantitative: annual claim production (t yr^{-1}) Spatial: distribution of licensed areas	The temporal evolution of clam production was also provided as main driver of environmental change
Regulating	Carbon storage	Carbon accumulation within the lagoon	As a function of reed NPP and decomposition (Equation (1))
Supporting	Habitat provision	Bird abundance and diversity	
Cultural	Ecotourism	Number of protected bird species	(potential for ES provision)

Urbani") was passed to regulate the management and exploitation of landscapes and environmental assets.

At regional and provincial level, the Territorial Plan for Regional Coordination (TPRC) was issued in 1990 and was committed to spatial planning at regional level and the Provincial Territorial Coordination Plan of Province of Ferrara (PTCP), issued in 1997, attempted to coordinate the different planning levels. PTCP constitutes guidelines and regulations aiming to encourage environmental conservation, and protection at Provincial level. Additionally, the Regional Landscape Plan of Emilia Romagna (RLP), approved in 1993 under the National Law 431/1985, organizes and coordinates the analysis, objectives, and actions related to environmental and landscape protection and conservation.

Finally, at local level, designation of the Nature Reserve of "Bosco della Mesola" was the first measure aimed towards environmental conservation purview in VMG district (1977). The Regional Park of river Po delta-ER was established in 1988, identifying six districts, regulated by dedicated Management Plans. The VMG district is regulated by the dedicated plan adopted in 1997.

The eastern part of the Goro lagoon was designated as a Ramsar zone in 1981. Subsequently in 1982, a zone including the outlet of the Goro was declared a National Nature Reserve. These regions were declared a Special Protection Area (SPA) and Site of Community Importance (SCI), according to Bird and Habitat EU Directives respectively in 1988.

2.2 Materials and methods

First, as a representative case study of the whole river Po delta, spatiotemporal changes of VMG wetlands were evaluated to detect the extension of wetland reclamation phenomena. The land cover maps were derived from a previous analysis of the changes occurring in the VMG area (Gaglio, Aschonitis, Gissi *et al*. 2017).

Second, in order to demonstrate the value of aquatic vegetation habitats as a nature-based solution to offset the wetland reclamations that occurred in the river Po delta area, we investigated the ESs focusing on common reed vegetation, *Phragmites australis*, the most abundant emergent macrophyte in the river Po delta. We selected three ESs that included regulating, supporting and cultural classes (Table 1), as a comprehensive set representative of the different ES categories (Millennium Ecosystem

Assessment 2005). These services were subjected to trade-offs with a provisioning service, namely clam production, that caused the loss of such habitats.

The changes in space and time of aquatic vegetation habitats in the Sacca di Goro lagoon were mapped to detect the magnitude of vegetation loss phenomena. Clam production was also assessed as a provisioning service as the main driver of environmental change.

2.2.1. Ecosystem services provided by aquatic vegetation

Aquatic vegetation supports a wide range of ecological functions (Kiviat 2013). Its disappearance results in transition from aquatic vegetation to bare sediment dominated habitats, leading to the loss of ecological functions. In order to measure the capacity of *P. australis* to provide ESs in the Sacca di Goro lagoon, two ecological functions, namely (i) habitat provision for birdlife and (ii) the capacity to store carbon in the ecosystem, were quantified through *in situ* measures and observations. These functions deliver three different ESs: habitat provision, potential for ecotourism and climate regulation (Table 1).

The capacity of aquatic vegetation ecosystems to provide habitat for birdlife was assessed through comparative monthly field surveys in two contrasting sites: a site vegetated with a monospecific common reed stand and a non-vegetated site. Visual identification of bird species was performed using a Leica Vector 10×50 binoculars during the nesting period (from March to June), which corresponded to the greatest abundance of bird specimens. The similarities between avian communities associated with the two sites were expressed by the Jaccard coefficient (Jaccard 1901). The number of taxa included in the EU Birds Directive (European Parliament 2009) was used as a proxy for birdwatching support capacity.

The carbon storage capacity was calculated as a function of net primary aboveground production and annual mass loss (%) (Duke, Francoeur, and Judd 2015), considering a stems-leaves mass ratio of 3:1 (Bellavance and Brisson 2010) and a 0.47 carbon content in biomass (IPCC 2006):

$$Cstorage = 0.47 \times \left(NPP - (\Delta Stem_{\%} \times NPP \times 0.75) \times (\Delta Leaf_{\%} \times NPP \times 0.25)\right) \quad (1)$$

where NPP is the aboveground Net Primary Production of *P. australis* and $\Delta\text{Stem}_{\%}$ and $\Delta\text{Leaf}_{\%}$ are the estimated percentage annual mass loss in terms of stems and leaves, respectively. To estimate decomposition rates of *Phragmites*, mass loss from senesced aboveground biomass was measured by means of an *in situ* litterbag experiment (Petersen and Cummins 1974). Standing dead stems and leaves of *P. australis* were collected in November from Goro lagoon, corresponding to the end of the plant life cycle, in order to simulate the real decomposition process. The material was dried in an oven at 50 °C for three days and afterwards in a muffle furnace at 375 °C for three hours to obtain the Ash Free Dry Weight (AFDW). AFDW is then packed in twelve 5 g litterbags of two plant organ types (leaves and stems) and immersed in water for an incubation time of 365 d. Six replicates of two different mesh sizes (10 and 1 mm) were used to better capture the environmental variability and test the possible influence of aquatic macro-invertebrates on the decomposition process. Since no statistically significant differences between replicates of different mesh sizes were observed, all replicates were pooled. After 365 d, the litterbags were recovered and the

AFDW measured again using the same procedure to estimate the annual mass loss of each tissue type.

The annual aboveground net primary production was measured as a sum of monthly values obtained by collecting samples (five replicates per month) of the aboveground vegetation biomass on an area of 1 m^2 over one year. The AFDW of collected monthly samplings were measured using the abovementioned procedure.

2.2.2. *Aquaculture*

Aquaculture activities in Goro lagoon are regulated by leasing licensed areas to private cooperatives of farmers at a concession. Aquaculture service was mapped by means of current spatial distribution of licensed areas and historical data on clam productivity. The spatial distribution of these areas is relevant for the assessment related to environmental impacts, whereas the production levels of Manila clam, representing the larger fraction of the total aquaculture production in the Sacca di Goro lagoon, is a good indicator for the service. The map of aquaculture concessions was obtained from the Regional Environmental Protection Agency (Agenzia Regionale per la Prevenzione, l'Ambiente e l'energia dell'Emilia-Romagna – ARPAE). The data on clam productivity from 1987 to 2007 were derived from Turolla (2008).

2.2.3. *Aquatic vegetation loss in the Sacca di Goro lagoon*

The spatiotemporal change of aquatic vegetation in Goro lagoon was assessed by comparing vegetated habitat maps for four dates: 1954, 1976, 1993, and 2005 and processing them with ArcGis 10.3 (ESRI). Maps for 1954 and 1976 were obtained from visual interpretation of aerial photograms, as reported in Gaglio, Aschonitis, Gissi *et al*. (2017). Maps for 1993 (Piccoli, Dell'Aquila, and Pellizzari 1999) and 2005 (Noferini, Passerella, and Pellizzari 2006) were obtained in vector format from the Regional Park of Po delta – Emilia Romagna. Habitats with emerging vegetation were mapped for all four dates. Submerged vegetation dominated habitats were mapped only for 1993 and 2005, since no spatial data were available for previous dates.

3. Results

3.1 Wetland reclamation in VMG district

The VMG district was subject to major wetland reclamations in the past that have significantly affected its natural capital (Figure 3). Gaglio, Aschonitis, Gissi *et al*. (2017) classified its recent history into two periods: (i) the land reclamation period (1954–1976), characterized by extensive conversion from wetland to croplands, and (ii) the environmental protection period (1976–2005), when significant efforts for applying environmental protection measures were undertaken (Figure A1). The authors also found that, during the land reclamation period, a slight increase in provisioning services was observed at the cost of heavy losses to regulating, supporting and cultural services. During the second period, despite the adoption of conservation measures that reduced direct human interventions, the decrease of natural capital value continued as a consequence of the loss of aquatic vegetated habitats in the Sacca di Goro lagoon.

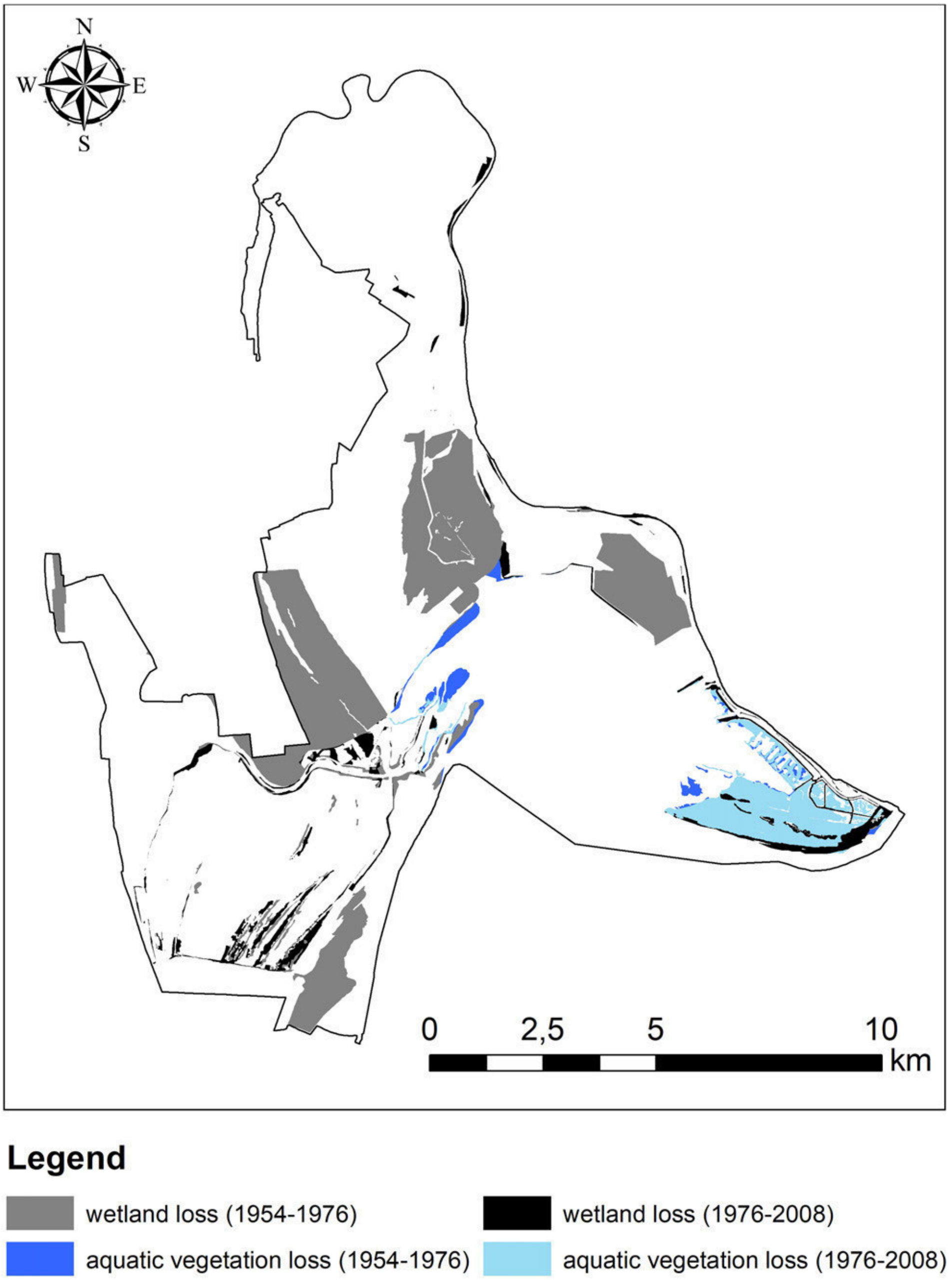


Figure 3. Map of wetland reclamations performed in VMG district for the periods 1954–1976 and 1976–2008. Land cover types to which wetlands were converted are specified. Conversion to "non-vegetated coastal wetlands" depicts the loss of aquatic vegetation.

The lagoon has been subjected to several interventions during past decades. These interventions were mainly to limit the inflow of freshwater, resulting in reduced nutrient loads, and to facilitate the entrance of seawater, seeking to increase tidal currents, hydrodynamics, and oxygenation (Figure 2). In the mid-1980s, the first interventions were affected by building water pumps and initial reshaping of submerged channels, to regulate bi-directional water exchanges between the lagoon and the sea. Unfortunately, these plans did not materialize due to unsustainable management costs. However, these initial interventions did not affect hydraulic circulation of the lagoon. The ineffectiveness of institutional action led later on, in 1988, to an anonymous intervention. This involved the removal of the western part of the outer sand bank of the lagoon to favor the flow of seawater into the lagoon. However, the benefits were limited and restricted to the open part of the lagoon. Other minor interventions were carried out in 1987 to further reduce the nutrient loads from Po di Goro and Po di Volano (Spinelli *et al*. 1996) and, in 1997, to ameliorate hydrodynamics (Carafa *et al*. 2006). In 1999, a new unidirectional water gate was built in the east corner of the lagoon, to direct water fluxes from the mouth of the lagoon to Po di Goro. Since 2000, other actions subsequently reshaped submerged channels and restored sandbanks with the aim of ameliorating hydrodynamics and preserving habitat and species.

Overall, while recent interventions were somehow beneficial for human activities, they were not sufficient to guarantee good ecological status of the lagoon. Specifically, the increase in salinity caused by the reduction of freshwater inflow and the contemporary increase of marine water fluxes, namely marinization, were likely responsible for reed stands dieback (Fogli, Marchesini, and Gerdol 2002). Despite the effort to mitigate nutrient inputs during this time, intense macro-algal blooms took place, thereby inhibiting the recovery of submerged macrophytes (Viaroli *et al*. 2006).

3.2. ESs provided by Phragmites australis-dominated habitats

The field surveys showed that the two sites differed in both richness and abundance of taxa (Table 2). Divergent taxa were classified at species level, except for Passeriformes. Overall, 15 taxa, of which nine were included in the 2009/147/EC Directive, and 90 individuals were identified during the four months of analysis. The Jaccard coefficient was 0.125, highlighting a low similarity between sites. The vegetated site hosted 63 individuals belonging to 11 taxa, while 27 individuals belonging to five taxa were observed in the non-vegetated site. Grey heron (*Ardea cinerea*) was the only species observed in both sites.

When considering species included in the 2009/147/EC Directive, eight taxa out of nine were recorded in the vegetated site (five in Annex I and three in Annex II). In the non-vegetated site, Common pheasant (*Phasianus colchicus*) was the only observed species included in the 2009/147/EC Directive (Annex II).

Monthly aboveground NPP (Figure 4) showed the expected seasonal trends, except for December when the average value reached 141.2 (±38.5) g AFDW m^{-2}. The total annual aboveground NPP was equal to 1056.02 (±32.9) g AFDW m^{-2}.

The results of the litterbag experiment highlighted different decomposition dynamics of *P. australis* stems and leaves (Table 3). The estimated decay rate (k) (Olson 1963) for leaf and stems was 0.00856 and 0.00228 per day, respectively. This led to different fractions of mass loss and, together with the stems to leaves ratio, to different contributions in terms of carbon accumulation. According to Equation (1), the overall carbon accumulation capacity was estimated to 194.39 g C m^{-2}.

Table 2. Observed abundance of bird taxa in vegetated and non-vegetated sites. Species included in 2009/147/EC Directive are marked with (*).

Taxa	Common name	Vegetated site	Non-vegetated site	Directive 2009/147/EC
Anas platyrhyncos	Mallard	9	0	
Ardea cinerea	Grey heron	2	2	
**Ardea purpurea*	Purple heron	4	0	Annex I
Cuculus canorus	Common cuckoo	0	1	
**Fulica atra*	Eurasian coot	4	0	Annex II
Garrulus glandarius	Eurasian jay	0	1	
**Larus argentatus*	European herring gull	2	0	Annex II
**Larus genei*	Slender-billed gull	1	0	Annex I
**Larus melanocephalus*	Mediterranean gull	1	0	Annex I
**Larus ridibundus*	Black-headed gull	25	0	Annex II
Passeriformes	passerine	0	22	
Phalacrocorax carbo	Great cormorant	2	0	
**Phasianus colchicus*	Common pheasant	0	1	Annex II
**Sterna albifrons*	Little tern	1	0	Annex I
**Sterna hirundo*	Common tern	12	0	Annex I
Abundance		63	27	
Taxa richness		11	5	

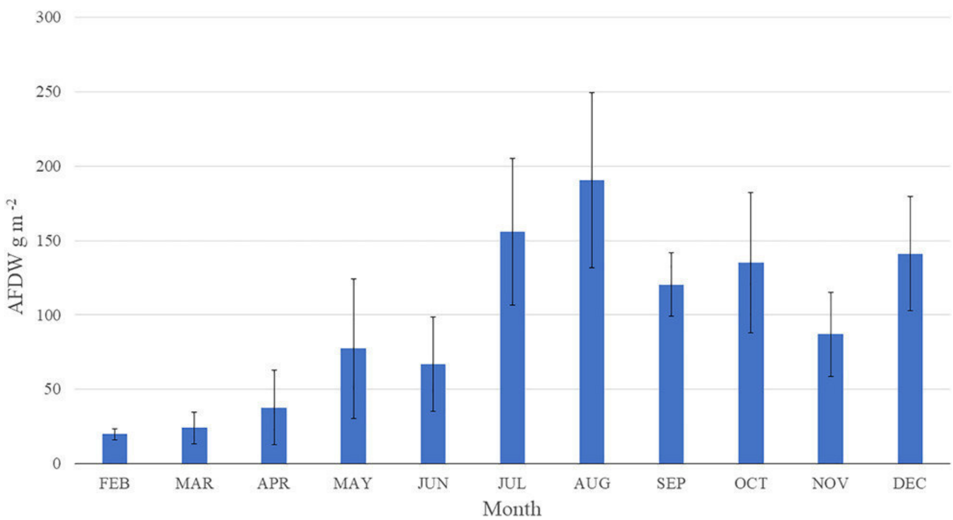


Figure 4. Measured mean monthly aboveground NPP. Values are expressed as g AFDW m^{-2}. Error bars show standard deviation.

3.3 Aquaculture

The spatial distribution of licensed areas for aquaculture activities in Goro lagoon is showed in Figure 5. The 75 different licensed areas cover a large part of the lagoon, accounting for a total area of 1,416 ha. The clam production in the lagoon rapidly increased between 1987 and 1991, reaching an annual yield of 15,500 tons, only to decline in the 1990s. Subsequently, anoxic crisis was recorded during summer 2002

Table 3. Decay rate (k), annual mass loss (%) and carbon storage estimated for leaves and stems.

	Decay rate (k) (day^{-1})	Annual mass loss (%)	Carbon storage[a] ($g\ C\ m^{-2}\ year^{-1}$)
Leaves	0.00856	95.60	7.28
Stems	0.00228	43.45	187.10
Total			194.39

Note: [a]Carbon storage was calculated according to Equation (1).

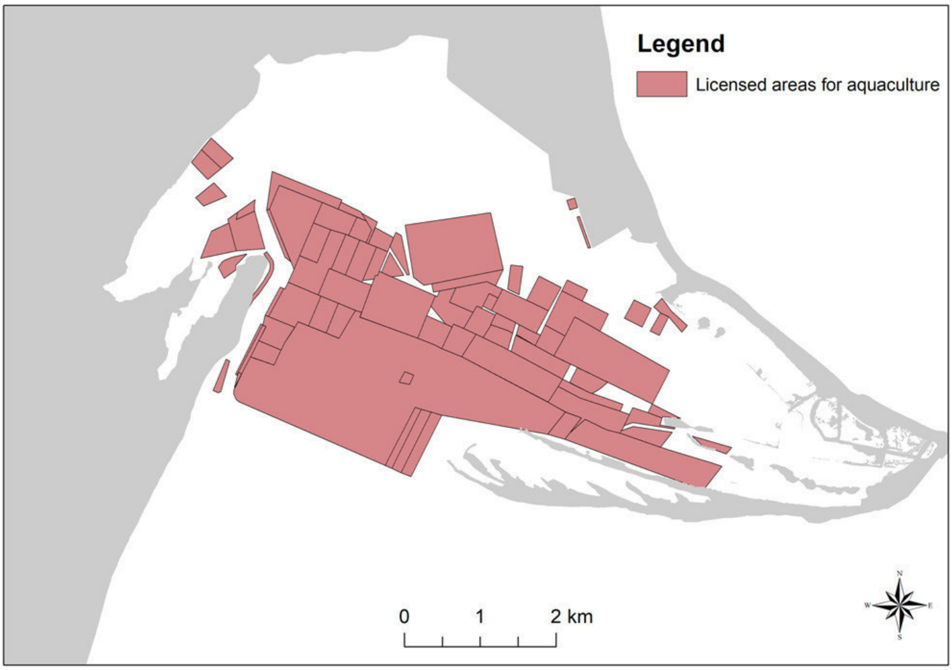


Figure 5. Spatial distribution of licensed areas for aquaculture in the Sacca di Goro lagoon.

(Figure 6). Total clam production increased again between 2001 and 2002 and reached the previous maximal levels in 2005, with an annual yield of 14,657 tons, as a consequence of excavation of submerged channels in the eastern part of the lagoon.

3.4. Aquatic vegetation loss in the Sacca di Goro lagoon

Spatiotemporal changes of emerging and submerged aquatic vegetation in the eastern part of Goro lagoon are reported in Figure 7. The estimated extension of *Phragmites australis* dramatically decreased from 386 ha in 1954 to only 50 ha in 2005 (−87%). Two hundred thirty-three out of 336 ha were lost during the 1993–2005 period, when hydraulic balances of the lagoon were strongly altered. The area covered by submerged vegetation suffered even larger losses during 1993–2005. In fact, areas covered by *Ruppia cyrrosa* almost disappeared, decreasing from 270 to 4 ha (−98%).

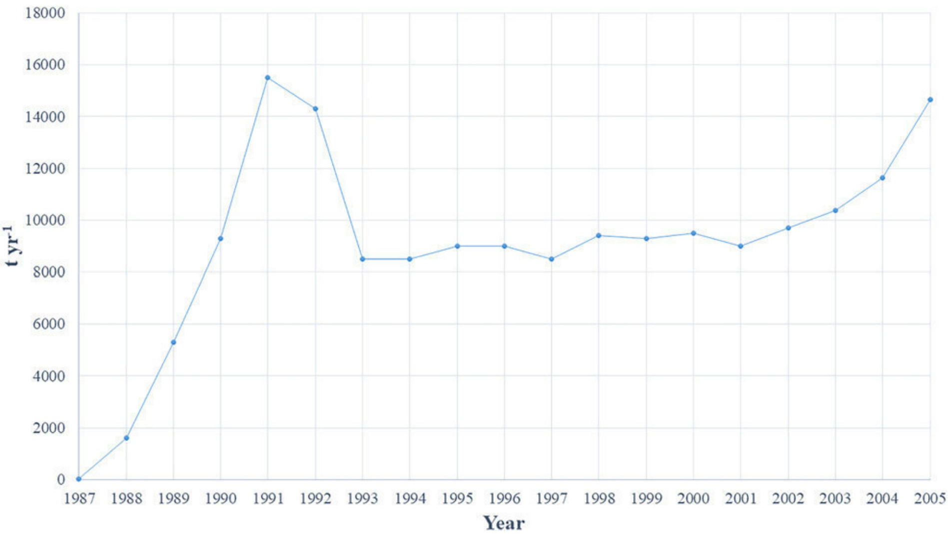


Figure 6. Trend of clam production in Sacca di Goro lagoon.

4. Discussion

4.1 Trade-off between clam farming and ES provided by aquatic vegetation

The experimental analysis demonstrates that aquatic vegetation significantly contributes to increasing landscape multi-functionality, particularly in the context of agriculture-dominated surroundings. In fact, the remaining wetland vegetation of the Sacca di Goro lagoon provides a comprehensive set of ESs (Table 1).

The NPP of *P. australis* coupled with the low decay rate of their tissues allows the net accumulation of carbon in aquatic systems, thus contributing to climate regulation. The productivity dynamics of coastal vegetated habitats are strongly influenced by nutrient availability (Sundareshwar *et al*. 2003), particularly those dominated by species with high nutrient-use efficiency such as *P. australis* (Tylova-Munzarova *et al*. 2005). The mean monthly aboveground NPP (Figure 4) follows the typical seasonal pattern, except for December observations. The anomalous high NPP value measured in December can be explained with the higher availability of nutrients deriving from the surrounding agricultural land during this particular period.

The estimated carbon accumulation capacity of *P. australis*-dominated habitats is concordant with the range of values reported by McLeod *et al*. (2011), who found an average burial rate of 218 g C m^{-2} in saltmarsh ecosystems. Moreover, this study underestimates the overall NPP by not considering the carbon stored in the belowground biomass, as well as carbon amounts in trapped sediments (Duarte *et al*. 2013).

The comparison between vegetated and non-vegetated sites revealed important differences in terms of habitat provision and potential for ecotourism. The higher abundance and taxa richness observed in vegetated habitats are reflected by the low levels of similarity between sites. The positive effects of aquatic vegetation on water bird abundance and diversity are justified by the larger amount of aquatic meio and macrofauna (i.e. food availability) (Angradi, Hagan, and Able 2001) and suitable nesting habitat (Parsons 2003). On the contrary, bird taxa observed in non-vegetated sites are mainly associated with surrounding terrestrial environment. Thus, aquatic vegetation

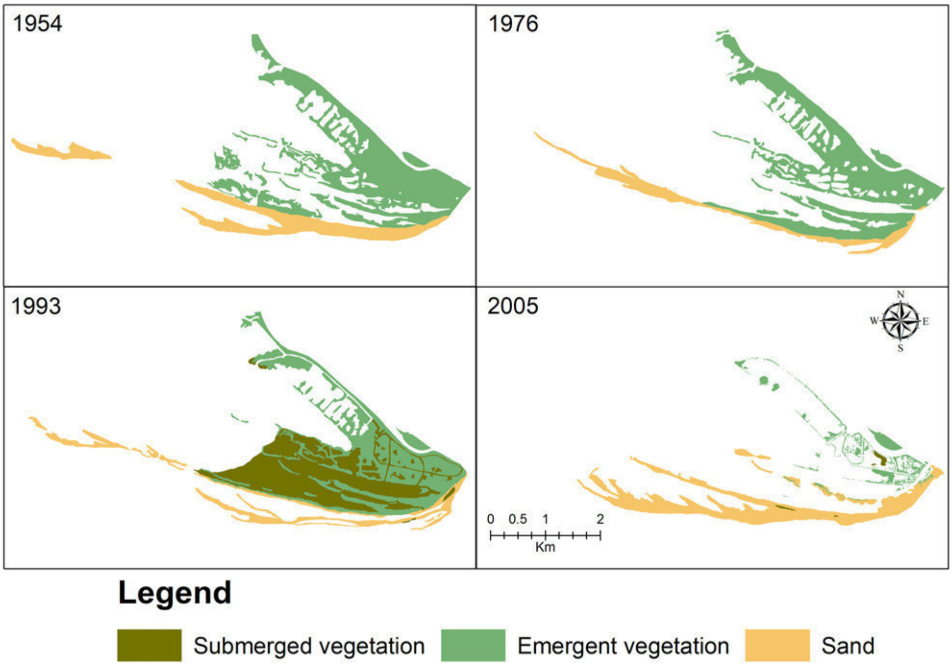


Figure 7. Spatial distribution of sandy and aquatic vegetated habitat with submerged and emerging vegetation for the years 1954, 1976, 1993, and 2005 in the eastern part of Sacca di Goro lagoon. Habitats with submerged vegetation are mapped just for 1993 and 2005.

plays a fundamental role in wetland biodiversity conservation by providing habitat for aquatic fauna.

Aquatic vegetated habitats were also found to host a higher number of protected bird taxa than non-vegetated wetlands, thus having a larger potential for ecotourism. These results can be explained by the fact that the presence of aquatic vegetation can mitigate most of the impacts threatening many protected bird species, such as habitat fragmentation (Schuh and Guadagnin 2018), predatory pressure (Schüttler *et al*. 2009), and environmental quality degradation (Tavares *et al*. 2015).

Despite their relevant ecological functions, coastal vegetated wetlands have suffered heavy losses worldwide (Millennium Ecosystem Assessment 2005). In the case of the Sacca di Goro lagoon, the interventions planned with the aim of supporting aquaculture activities appear to have been the main driver of change. Vegetation loss occurred together with the increase in clam production and expansion of relative licensed areas, which, however, were not overlapped spatially, indicating that habitat protection and human activities may coexist. Plausibly, the loss of ecological functions underpinning regulating, supporting, and cultural services were due to management choices that were adopted with the best intent, but the specific environmental needs of the area were not fully addressed. Thanks to the EU Life Project AGREE (LIFE13 NAT/IT/000115; http://lifeagree.eu/), a more comprehensive evaluation, based on previous experience, was performed to better address the specific needs of the different sub areas of the lagoon, thereby, reinforcing a new paradigm of management, i.e. monitoring and research as the basis for any management choice, which has to be dimensioned on the specificity of each single sub-area.

Trade-offs among provisioning and other services as a consequence of unsustainable human exploitation of natural resources have been extensively described in the literature (Howe *et al*. 2014). Similar trade-offs involving intensive aquaculture activities have been observed worldwide. For example, in tropical coastal areas, extended coverage of mangroves was converted to shrimp ponds (Arifanti *et al*. 2019; Kauffman *et al*. 2018). In temperate zones, salmon aquaculture was found to reduce regulating and cultural services (Outeiro and Villasante 2013), shellfish aquaculture is responsible for eelgrass deterioration in nearshore ecosystems (Ferriss *et al*. 2019) which is the provider of several ESs (Mtwana Nordlund *et al*. 2016).

In this study, aquaculture activities did not cause a direct trade-off for ESs and biodiversity, as demonstrated by previous studies carried out in the lagoon (Castaldelli *et al*. 2003; Mantovani *et al*. 2006), but rather the evidenced loss of ESs can be related to management choices and not to the aquaculture itself. This spurs a debate on the importance of integrated evaluation as a basis for the decision support system.

4.2 ES loss under a fragmented sectoral management

The present work has highlighted the partial success of rigorous legislation, such as the Nature 2000 Directive, that incorporate fragmented sectoral management in promoting human well-being and multiple use of the landscape. Therefore, there is an immediate need to adapt the ES approach in a site-specific manner and to temporal evolution that a certain area has undergone. This means that in deltaic areas, based on long-term monitoring studies, ES governance must integrate a wide set of services with provisioning ones, as in the case of aquaculture in Sacca di Goro lagoon.

A unified vision is needed to coordinate instruments that consider multiple values of ecosystems beyond conservation scopes. It is common knowledge that socio-ecological systems suffer from scale mismatches (Guerrero *et al*. 2013). Since ecological systems are regulated by processes occurring at different scales, sectoral measures operating at other spatial scales than those of the impacted ecological processes, often fail to effectively manage ecosystems (Cumming, Cumming, and Redman 2006). Additionally, the rate of action implementation does not reflect the rate of change of ecological systems, thus also introducing temporal scale mismatches (Guerrero *et al*. 2013).

Suitable arguments for adoption of an ES approach are provided fragmentally, as there is no specific EU policy for governing ESs (Bouwma *et al*. 2018). For example, the EU Biodiversity strategy to 2020 provides measures to halt the loss of biodiversity and ecosystem services in Europe. In particular, Target 2 aims to restore at least 15% of degraded ecosystems by 2020 and to map and assess the state and economic value of ecosystems and their services.

In the early 1980s, the initial conservation approach of the Ramsar Convention was superseded by the principles of "wise use" of wetlands (Hettiarachchi, Morrison, and McAlpine 2015), defined as "the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development," suggesting the importance of guaranteeing multiple ESs.

Finding solutions for local implementation of these principles is a future key challenge to include the ES approach in the deltaic governance. However, there is a need for local evidence to be exported to other coastal contexts concerning: (i) trade-off analysis based on measures rather than estimates (Mach, Martone, and Chan 2015) and (ii) effective ES restoring strategies, leading to win-win solutions (Howe *et al*. 2014).

4.3 Restoring ESs in the river Po Delta and the Sacca di Goro lagoon

The VMG district has been subjected to widespread reclamations of its coastal wetlands and loss of aquatic vegetation in the past. Restoration of the lost ecological functions to offset past reclamations is needed to achieve conservation targets and guarantee sustainable governance for the area. However, the original ecological conditions after restoration of some degraded areas and wetland loss might be challenging to achieve (Yu *et al*. 2017). For these reasons, the restoring hierarchy in VMG should be aimed at prioritizing the eastern part of the lagoon (the so-called Valle di Gorino), where extended reed stands were present in the past. Complete conversion of reclaimed coastal wetlands to their original status would be critical under a socio-economic perspective. In fact, these areas currently have high agricultural yields and provide income and valuable traditional products. Although restoring pristine habitats would be beneficial in terms of natural capital assets, reconversion of large areas would not lead to proportional gains of ESs, because of the non-linear relationship between habitat size and ecological functions (Barbier *et al*. 2008; Koch *et al*. 2009).

Because of their capacity to deliver a comprehensive set of ESs, restoring aquatic vegetated habitats can represent a nature-based solution to increase the multi-functionality of coastal areas and achieve the EU biodiversity strategy targets to 2020. Many studies have focused on modeling or describing single regulation functions (Koch *et al*. 2009; Temmerman *et al*. 2013; Li *et al*. 2016; Arkema *et al*. 2017) rather than considering multiple values. This study provides evidence that restoring vegetation in coastal wetlands in the river Po delta might significantly increase multiple ESs to overcome the past fragmented and sectoral management, while avoiding socio-economic impacts due to agricultural land loss. These findings could be exported to other deltaic areas and, particularly, for those that have undergone extended wetland reclamation.

Recently, within the EU Life Project AGREE, a substantial change in the management of the Sacca di Goro has been adopted on the basis of monitoring results (Castaldelli *et al*. 2016). Two water gates between the Po di Goro and the Valle di Gorino have been opened permanently with the aim of increasing the freshwater inflow and favoring restoration of reed stands in the eastern part of the lagoon. At the same time, negotiations have started with clam farmers to move some of the licensed areas for clam farming from internal zones of the lagoon to offshore. Involvement of stakeholders and information on multiple benefits derived from restoring degraded habitats might play a fundamental role in the mediation process. Higher costs involved in longer travelling distances to reach offshore areas may be offset by the higher potential yields that seem to characterize these zones (Vincenzi *et al*. 2006), while avoiding further risks of *Ulva* spp. blooms in the lagoon (Viaroli *et al*. 1996). This solution may harmonize the different needs and uses, such as habitat conservation and aquaculture activities, while restoring regulating, supporting, and cultural ESs that contribute to achieving sustainable goals. Specifically, the more rational use of the lagoon is expected to support the local economy under a blue growth strategy, by maintaining clam farming productivity and promoting ecotourism initiatives. The greenhouse gas emissions generated by human activities can be offset by the sequestration of higher amounts of blue carbon while achieving habitat and species conservation targets.

Moreover, the improved hydrodynamic conditions of the lagoon and, to a lesser extent, the benefits in terms of phytodepuration of restoring reed stands, are expected to reduce the risks of anoxic events and algal blooms, thus favoring the possible recovery of submerged vegetation in the process. Presently, the most important disturbance to these habitats is due to agricultural activities, both at the scale of the whole Po basin and locally. In particular, small basins that carry drainage water and discharge directly into the lagoon (Castaldelli *et al*. 2013; Viaroli *et al*. 2018) are areas prone to nutrient leaching (Aschonitis *et al*. 2013). For this reason, the reduction of nutrient leaching from agricultural fields may represent the most effective measure to restore submerged macrophyte habitats. The reduction of fertilizer applications may be achieved by promoting more extensive practices, such as those related to precision agriculture (Cilia *et al*. 2014) or organic farming (Reganold and Wachter 2016).

5. Conclusions

This study provides evidence that a rational use of natural resources and the promotion of multifunctional systems are needed to obtain sustainable goals. The results demonstrate the importance of aquatic vegetated habitats as a provider of regulating, supporting, and cultural ESs. In deltaic areas subjected to past widespread reclamation, the loss of vegetated coastal wetlands has led to concurrent loss of relevant amounts of carbon that could be sequestered and stored in these ecosystems and habitats for bird communities, resulting in a decrease in conservation values and potential for bird-watching-related tourism.

As observed in other socio-environmental contexts, trade-offs between aquaculture and other ESs, as well as environmental conservation, occurred during past decades in a Mediterranean delta, despite various environmental protection measures and planning instruments adopted at European, national and local levels. Sectoral regulation and fragmented measures failed in guaranteeing the sustainable use of natural resources and human well-being over time.

By means of ES assessment carried out in a brackish lagoon of the Po river delta, the findings demonstrate the potential for restoring lost habitats as a nature-based solution that could be adopted in delta governance. The restoration of such degraded habitats, together with relocation of licensed areas for clam aquaculture in the Sacca di Goro lagoon, led to maximization of win-win strategies in the context of an ES approach. Nonetheless, such solutions require increased scientific efforts in investigating ecosystems functioning, as well as a better understanding of stakeholders' involvement mechanisms.

However, even though the investigated services could be categorized into all the four ES categories, a wider set of ESs is needed to assess and better capture the value of aquatic ecosystems and relative trade-offs. Moreover, when evaluating cultural services, people perception and real fruition should be taken into account.

Overall, given the high environmental and socio-economic relevance of the area, the Po delta might be an ideal case study to test the contribution of nature-based solutions in sustainable management of deltas. Monitoring the outcomes of ongoing restoration attempts and management strategies would provide suitable information to be exported to other deltaic areas.

Disclosure statement

No potential conflict of interest was reported by the authors.

Funding

The work was partially supported by EU LIFE Project AGREE (LIFE13 NAT/IT/000115) and the project PRIN-NOACQUA: responses of communities and ecosystem processes in intermittent rivers (Prot. 201572HW8F) funded by the Italian Ministry of Education, University and Research.

ORCID

Mattias Gaglio http://orcid.org/0000-0002-2027-0732
Elisa Anna Fano http://orcid.org/0000-0001-8552-9621

References

Angradi, T. R., S. M. Hagan, and K. W. Able. 2001. "Vegetation Type and the Intertidal Macroinvertebrate Fauna of a Brackish Marsh: Phragmites vs. Spartina." *Wetlands* 21 (1): 75–92. doi:10.1672/0277-5212(2001)021.

Arifanti, V. B., J. Boone Kauffman, D. Hadriyanto, D. Murdiyarso, and R. Diana. 2019. "Carbon Dynamics and Land Use Carbon Footprints in Mangrove-Converted Aquaculture: The Case of the Mahakam Delta, Indonesia." *Forest Ecology and Management* 432: 17–29. doi:10.1016/j.foreco.2018.08.047.

Arkema, K. K., G. M. Verutes, S. A. Wood, C. Clarke-Samuels, S. Rosado, M. Canto, A. Rosenthal, et al. 2015. "Embedding Ecosystem Services in Coastal Planning Leads to Better Outcomes for People and Nature." *Proceedings of the National Academy of Sciences United States of America* 112 (24): 7390–7395. doi:10.1073/pnas.1406483112.

Arkema, K. K., R. Griffin, S. Maldonado, J. Silver, J. Suckale, and A. D. Guerry. 2017. "Linking Social, Ecological, and Physical Science to Advance Natural and Nature-Based Protection for Coastal Communities." Annals of the New York Academy of Sciences 1399 (1): 5–26. doi:10.1111/nyas.13322.

Aschonitis, V. G., E. Salemi, N. Colombani, G. Castaldelli, and M. Mastrocicco. 2013. "Formulation of Indices to Describe Intrinsic Nitrogen Transformation Rates for the Implementation of Best Management Practices in Agricultural Lands." *Water, Air, and Soil Pollution* 224 (3): 149. doi:10.1007/s11270-013-1489-1.

Balmford, A., J. M. H. Green, M. Anderson, J. Beresford, C. Huang, R. Naidoo, M. Walpole, and A. Manica. 2015. "Walk on the Wild Side: Estimating the Global Magnitude of Visits to Protected Areas." *PLoS Biology* 13 (2): e1002074. doi:10.1371/journal.pbio.1002074.

Barbier, E., E. W. Koch, B. R. Silliman, S. D. Hacker, E. Wolanski, J. Primavera, E. F. Granek, et al. 2008. "Coastal Ecosystem-Based Management with Nonlinear Ecological Functions and Values: Supporting Material." *Science* 319 (5861): 321–323. doi:10.1126/science.1150349.

Bartoli, M., G. Castaldelli, D. Nizzoli, E. A. Fano, and P. Viaroli. 2016. "Manila Clam Introduction in the Sacca Di Goro Lagoon (Northern Italy): Ecological Implications." *Bulletin of Japan Fisheries Research and Education Agency* 42: 43–52.

Bellavance, M. E., and J. Brisson. 2010. "Spatial Dynamics and Morphological Plasticity of Common Reed (Phragmites Australis) and Cattails (Typha Sp.) in Freshwater Marshes and Roadside Ditches." *Aquatic Botany* 93 (2): 129–134. doi:10.1016/j.aquabot.2010.04.003.

Bondesan, M. 1990. "L'area Deltizia Padana: Caratteri Geografici e Geormorfologici." In *Il Parco Del Delta Del Po. Studi e Immagini. L'ambiente Come Risorsa*, edited by Spazio Libri, 10–48. Ferrara, Italy: Universita di Ferrara http://www.marcobondesan.it/wp-content/uploads/2015/01/IL-PARCO-DEL-DELTA.pdf.

Booth, J., E. Gaston, K. J. K. L. Evans, and P. R. Armsworth. 2011. “The Value of Species Rarity in Biodiversity Recreation: A Birdwatching Example.” *Biological Conservation* 144 (11): 2728–2732. doi:10.1016/j.biocon.2011.02.018.

Bouwma, I., C. Schleyer, E. Primmer, K. J. Winkler, P. Berry, J. Young, E. Carmen, et al. 2018. “Adoption of the Ecosystem Services Concept in EU Policies.” *Ecosystem Services* 29: 213–222. doi:10.1016/j.ecoser.2017.02.014.

Braat, L. C., and R. S. de Groot. 2012. “The Ecosystem Services Agenda: Bridging the Worlds of Natural Science and Economics, Conservation and Development, and Public and Private Policy.” *Ecosystem Services* 1 (1): 4–15. doi:10.1016/j.ecoser.2012.07.011.

Carafa, R., D. Marinov, S. Dueri, J. Wollgast, J. Ligthart, E. Canuti, P. Viaroli, and J. M. Zaldívar. 2006. “A 3D Hydrodynamic Fate and Transport Model for Herbicides in Sacca Di Goro Coastal Lagoon (Northern Adriatic).” *Marine Pollution Bulletin* 52 (10): 1231–1248. doi:10.1016/j.marpolbul.2006.02.025.

Castaldelli, G., S. Mantovani, D. T. Welsh, R. Rossi, M. Mistri, and E. A. Fano. 2003. “Impact of Commercial Clam Harvesting on Water Column and Sediment Physicochemical Characteristics and Macrobenthic Community Structure in a Lagoon (Sacca Di Goro) of the Po River Delta.” *Chemistry and Ecology* 19 (2–3): 161–171. doi:10.1080/0275754031000119915.

Castaldelli, G., E. Soana, E. Racchetti, E. Pierobon, M. Mastrocicco, E. Tesini, E. A. Fano, and M. Bartoli. 2013. “Nitrogen Budget in a Lowland Coastal Area Within the Po River Basin (Northern Italy): Multiple Evidences of Equilibrium Between Sources and Internal Sinks.” *Environmental Management* 52 (3): 567–580. doi:10.1007/s00267-013-0052-6.

Castaldelli, G., F. Vincenzi, M. Lanzoni, and E. A. Fano. 2016. LIFE AGREE – CoAstal LaGoon Long TeRm Management. *Action A.6: Monitoraggio Ex-Ante Delle Acque*. http://lifeagree.eu/wp-content/uploads/2017/10/A6_2_Monitoraggio_acque-ex-ante.pdf.

Cencini, C. 1998. “Physical Processes and Human Activities in the Evolution of the Po Delta, Italy.” *Journal of Coastal Research* 14 (3): 774–793. doi:10.2307/4298834.

Cilia, C., C. Panigada, M. Rossini, M. Meroni, L. Busetto, S. Amaducci, M. Boschetti, V. Picchi, and R. Colombo. 2014. “Nitrogen Status Assessment for Variable Rate Fertilization in Maize Through Hyperspectral Imagery.” *Remote Sensing* 6 (7): 6549–6565. doi:10.3390/rs6076549.

Cohen-Shacham, E., G. Walters, C. Janzen, and S. Maginnis. 2016. *Nature-Based Solutions to Address Global Societal Challenges.* Gland, Switzerland: IUCN. doi:http://dx.doi.org/10.2305/IUCN.CH.2016.13.en.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, et al. 1997. “The Value of the World’s Ecosystem Services and Natural Capital.” *Nature* 387 (6630): 253–260. doi:10.1038/387253a0.

Cumming, G. S., D. H. M. Cumming, and C. L. Redman. 2006. “Scale Mismatches in Social-Ecological Systems: Causes, Consequences, and Solutions.” *Ecology and Society* 11 (1): 14. doi:10.5751/ES-01569-110114.

Daily, G. C. 1997. *Nature’s Services: Societal Dependence on Natural Ecosystems*. Washington, DC: Island Press.

Danovaro, R., and A. Pusceddu. 2007. “Biodiversity and Ecosystem Functioning in Coastal Lagoons: Does Microbial Diversity Play Any Role?” *Estuarine, Coastal and Shelf Science* 75 (1–2): 4–12. doi:10.1016/j.ecss.2007.02.030.

de Groot, R., L. Brander, S. van der Ploeg, R. Costanza, F. Bernard, L. Braat, M. Christie, et al. 2012. “Global Estimates of the Value of Ecosystems and Their Services in Monetary Units.” *Ecosystem Services* 1 (1): 50–61. doi:10.1016/j.ecoser.2012.07.005.

Dealteris, J. T., B. D. Kilpatrick, R. B. Rheault, R. B. Dealteris, J. T. Kilpatrick, and B. D. Rheault. 2004. “A Comparative Evaluation of the Habitat Value of Shellfish Aquaculture Gear, Submerged Aquatic Vegetation and Non-Vegetated Seabed.” *Journal of Shellfish Research* 23 (3): 867–874. http://www.coonamessettfarm.com/sitebuildercontent/sitebuilderfiles/gear-vs-sav-habitat.pdf.

Drakou, E. G., C. Kermagoret, C. Liquete, A. Ruiz-Frau, K. Burkhard, A. I. Lillebø, A. P. E. V. Oudenhoven, et al. 2017. “Marine and Coastal Ecosystem Services on the Science-Policy-Practice Nexus: Challenges and Opportunities from 11 European Case Studies.” *International Journal of Biodiversity Science, Ecosystem Services and Management* 13 (3): i–iv. doi:10.1080/21513732.2017.1417330.

Duarte, C. M., I. J. Losada, I. E. Hendriks, I. Mazarrasa, and N. Marbà. 2013. "The Role of Coastal Plant Communities for Climate Change Mitigation and Adaptation." *Nature Climate Change* 3 (11): 961–968. doi:10.1038/nclimate3062.

Duke, S. T., S. N. Francoeur, and K. E. Judd. 2015. "Effects of Phragmites Australis Invasion on Carbon Dynamics in a Freshwater Marsh." *Wetlands* 35 (2): 311–321. doi:10.1007/s13157-014-0619-x.

European Commission. 1992. "Council Directive 92/43/EEC of 21 May 1992 on the Conservation of Natural Habitats and of Wild Fauna and Flora. Council of the European Communities (CEC)." *Official Journal of the European Communities* 206: 7–50.

European Parliament. 2009. "Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009." *Official Journal of the European Union* 140 (16): 16–62. doi:10.3000/17252555.L_2009.140.

Ferriss, B. E., L. L. Conway-Cranos, B. L. Sanderson, and L. Hoberecht. 2019. "Bivalve Aquaculture and Eelgrass: A Global Meta-Analysis." *Aquaculture* 498: 254–262. doi:10.1016/j.aquaculture.2018.08.046.

Fogli, S., R. Marchesini, and R. Gerdol. 2002. "Reed (Phragmites Australis) Decline in a Brackish Wetland in Italy." *Marine Environmental Research* 53 (5): 465–479. doi:10.1016/S0141-1136(02)00091-0.

Gaglio, M., V. G. Aschonitis, E. Gissi, G. Castaldelli, and E. A. Fano. 2017. "Land Use Change Effects on Ecosystem Services of River Deltas and Coastal Wetlands: Case Study in Volano–Mesola–Goro in Po River Delta (Italy)." *Wetlands Ecology and Management* 25 (1): 67–86. doi:10.1007/s11273-016-9503-1.

Gaglio, M., V. G. Aschonitis, M. M. Mancuso, J. P. Reyes Puig, F. Moscoso, G. Castaldelli, and E. A. Fano. 2017. "Changes in Land Use and Ecosystem Services in Tropical Forest Areas: A Case Study in Andes Mountains of Ecuador." *International Journal of Biodiversity Science, Ecosystem Services and Management* 13 (1): 264–279. doi:10.1080/21513732.2017.1345980.

García, M., J. R. Krueger, T. C. A. Páez, C. A. Ruiz-Agudelo, P. Bejarano, T. Muto, and F. Arjona. 2017. "Effectiveness of Conservation Areas for Protecting Biodiversity and Ecosystem Services: A Multi-Criteria Approach." *International Journal of Biodiversity Science, Ecosystem Services and Management* 13 (1): 1–13. doi:10.1080/21513732.2016.1200672.

Gissi, E., M. Gaglio, V. G. Aschonitis, E. A. Fano, and M. Reho. 2018. "Soil-Related Ecosystem Services Trade-off Analysis for Sustainable Biodiesel Production." *Biomass and Bioenergy* 114: 83–99. doi:10.1016/j.biombioe.2017.08.028.

Gissi, E., and V. Garramone. 2018. "Learning on Ecosystem Services Co-Production in Decision-Making from Role-Playing Simulation: Comparative Analysis from Southeast Europe." *Ecosystem Services* 34: 228–253. doi:10.1016/j.ecoser.2018.03.025.

Greiner, J. T., K. J. McGlathery, J. Gunnell, and B. A. McKee. 2013. "Seagrass Restoration Enhances 'Blue Carbon' Sequestration in Coastal Waters." *PLoS One* 8 (8): e72469. doi:10.1371/journal.pone.0072469.

Guerrero, A. M., R. R. J. McAllister, J. Corcoran, and K. A. Wilson. 2013. "Scale Mismatches, Conservation Planning, and the Value of Social-Network Analyses." *Conservation Biology* 27 (1): 35–44. doi:10.1111/j.1523-1739.2012.01964.x.

Haines-Young, R., and M. Potschin. 2010. "The Links Between Biodiversity, Ecosystem Services and Human Well-Being." In *Ecosystem Ecology: A New Synthesis. BES Ecological Reviews Series*, edited by D. Raffaelli, and C. Frid. Cambridge, UK: Cambridge University Press. doi:10.1017/CBO9780511750458.007

Hettiarachchi, M., T. H. Morrison, and C. McAlpine. 2015. "Forty-Three Years of Ramsar and Urban Wetlands." *Global Environmental Change* 32: 57–66. doi:10.1016/j.gloenvcha.2015.02.009.

Howe, C., H. Suich, B. Vira, and G. M. Mace. 2014. "Creating Win-Wins from Trade-Offs? Ecosystem Services for Human Well-Being: A Meta-Analysis of Ecosystem Service Trade-Offs and Synergies in the Real World." *Global Environmental Change* 28: 263–275. doi:10.1016/j.gloenvcha.2014.07.005.

Howes, M., L. Wortley, R. Potts, A. Dedekorkut-Howes, S. Serrao-Neumann, J. Davidson, T. Smith, and P. Nunn. 2017. "Environmental Sustainability: A Case of Policy Implementation Failure?" *Sustainability (Switzerland)* 9 (2): 165. doi:10.3390/su9020165.

IPCC. 2006. "2006 IPCC Guidelines for National Greenhouse Gas Inventories: Mineral Industry Emissions." Chapter 2 in *2006 IPCC Guidelines for National Greenhouse Gas Inventories Volume 3*, 1–40. Geneva: IPCC.

Jaccard, P. 1901. "Étude Comparative de La Distribution Florale Dans Une Portion Des Alpes et Des Jura." *Bulletin Del La Société Vaudoise Des Sciences Naturelles* 37: 547–579. doi: http://dx.doi.org/10.5169/seals-266450.

Kauffman, J. B., A. F. Bernardino, T. O. Ferreira, N. W. Bolton, L. E. de, O. Gomes, and G. Nuto Nobrega. 2018. "Shrimp Ponds Lead to Massive Loss of Soil Carbon and Greenhouse Gas Emissions in Northeastern Brazilian Mangroves." *Ecology and Evolution* 8 (11): 5530–5540. doi:10.1002/ece3.4079.

Keesstra, S., J. Nunes, A. Novara, D. Finger, D. Avelar, Z. Kalantari, and A. Cerdà. 2018. "The Superior Effect of Nature Based Solutions in Land Management for Enhancing Ecosystem Services." *Science of the Total Environment* 610–611: 997–1009. doi:10.1016/j.scitotenv.2017.08.077.

Kiviat, E. 2013. "Ecosystem Services of Phragmites in North America with Emphasis on Habitat Functions." *AoB PLANTS* 5: plt008. doi:10.1093/aobpla/plt008.

Koch, E. W., E. B. Barbier, B. R. Silliman, D. J. Reed, G. M. E. Perillo, S. D. Hacker, E. F. Granek, et al. 2009. "Non-Linearity in Ecosystem Services: Temporal and Spatial Variability in Coastal Protection." *Frontiers in Ecology and the Environment* 7 (1): 29–37. doi:10.1890/080126.

Li, Y., X. Zhang, X. Zhao, S. Ma, H. Cao, and J. Cao. 2016. "Assessing Spatial Vulnerability from Rapid Urbanization to Inform Coastal Urban Regional Planning." *Ocean and Coastal Management* 123: 53–65. doi:10.1016/j.ocecoaman.2016.01.010.

Mach, M. E., R. G. Martone, and K. M. A. Chan. 2015. "Human Impacts and Ecosystem Services: Insufficient Research for Trade-off Evaluation." *Ecosystem Services* 16: 112–120. doi:10.1016/j.ecoser.2015.10.018.

Maes, J., B. Egoh, L. Willemen, C. Liquete, P. Vihervaara, J. P. Schägner, B. Grizzetti, et al. 2012. "Mapping Ecosystem Services for Policy Support and Decision Making in the European Union." *Ecosystem Services* 1 (1): 31–39. doi:10.1016/j.ecoser.2012.06.004.

Mantovani, S., G. Castaldelli, R. Rossi, and E. A. Fano. 2006. "The Infaunal Community in Experimentally Seeded Low and High Density Manila Clam (Tapes Philippinarum) Beds in a Po River Delta Lagoon (Italy)." *ICES Journal of Marine Science* 63 (5): 860–866. doi:10.1016/j.icesjms.2006.02.004.

Maragno, D., M. Gaglio, M. Robbi, F. Appiotti, E. A. Fano, and E. Gissi. 2018. "Fine-Scale Analysis of Urban Flooding Reduction from Green Infrastructure: An Ecosystem Services Approach for the Management of Water Flows." *Ecological Modelling* 386:1–10. doi:10.1016/j.ecolmodel.2018.08.002.

McKinley, E., R. C. Ballinger, and N. J. Beaumont. 2018. "Saltmarshes, Ecosystem Services, and an Evolving Policy Landscape: A Case Study of Wales, UK." *Marine Policy* 91: 1–10. doi:10.1016/j.marpol.2018.01.021.

McLeod, E., G. L. Chmura, S. Bouillon, R. Salm, M. Björk, C. M. Duarte, C. E. Lovelock, W. H. Schlesinger, and B. R. Silliman. 2011. "A Blueprint for Blue Carbon: Toward an Improved Understanding of the Role of Vegetated Coastal Habitats in Sequestering CO2." *Frontiers in Ecology and the Environment* 9 (10): 552–560. doi:10.1890/110004.

Mikhailov, V. N., and M. V. Mikhailova. 2003. "Deltas as Indicators of Natural and Human-Induced Changes in the Regimes of Rivers and Seas." *Water Resources* 30 (6): 602–612. doi:10.1023/B:WARE.0000007585.01852.ee.

Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-Being: Synthesis*. Washington, DC: Island Press.

Mori, A. S., K. P. Lertzman, and L. Gustafsson. 2017. "Biodiversity and Ecosystem Services in Forest Ecosystems: A Research Agenda for Applied Forest Ecology." *Journal of Applied Ecology* 54 (1): 12–27. doi:10.1111/1365-2664.12669.

Mtwana Nordlund, L., E. W. Koch, E. B. Barbier, and J. C. Creed. 2016. "Seagrass Ecosystem Services and Their Variability Across Genera and Geographical Regions." *PLoS One* 11 (10): e0163091. doi:10.1371/journal.pone.0163091.

Noferini, A., M. Passerella, and M. Pellizzari. 2006. "La Sacca Di Goro□: Quadro Conoscitivo e Proposte Preliminari Di Gestione Della Laguna Compresa Nel SIC 'Sacca Di Goro, Po Di Goro, Valle Dindona, Foce Del Po Di Volano." In *Linee Guida e Casi Di Studio per La*

Gestione Dei Siti Della Rete Natura 2000 in Ambiente Di Transizione: Atti Del Workshop Tenutosi a Grado, Palazzo Dei Congressi, 7-8 Giugno 2006, 309–318. EUT Edizioni Università di Trieste.

Ojeda, M. I., A. S. Mayer, and B. D. Solomon. 2008. "Economic Valuation of Environmental Services Sustained by Water Flows in the Yaqui River Delta." *Ecological Economics* 65 (1): 155–166. doi:10.1016/j.ecolecon.2007.06.006.

Olson, J. S. 1963. "Energy Storage and the Balance of Producers and Decomposers in Ecological Systems." *Ecology* 44 (2): 322–331. doi:10.2307/1932179.

Outeiro, L., and S. Villasante. 2013. "Linking Salmon Aquaculture Synergies and Trade-Offs on Ecosystem Services to Human Wellbeing Constituents." *Ambio* 42 (8): 1022–1036. doi:10.1007/s13280-013-0457-8.

Parsons, K. C. 2003. "Reproductive Success of Wading Birds Using Phragmites Marsh and Upland Nesting Habitats." *Estuaries* 26 (2): 596–601. doi:10.1007/BF02823734.

Pendleton, L., D. C. Donato, B. C. Murray, S. Crooks, W. A. Jenkins, S. Sifleet, C. Craft, et al. 2012. "Estimating Global 'Blue Carbon' Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems." *PLoS One* 7 (9): e43542. doi:10.1371/journal.pone.0043542.

Petersen, R. C., and K. W. Cummins. 1974. "Leaf Processing in a Woodland Stream." *Freshwater Biology* 4 (4): 343–368. doi:10.1111/j.1365-2427.1974.tb00103.x.

Piccoli, F., L. Dell'Aquila, and M. Pellizzari. 1999. "Carta Della Vegetazione Del Parco Regionale Del Delta Del Po, Scala 1:35.000. 11.1. Stazione Volano – Mesola – Goro." Regione Emilia-Romagna, Servizio Cartografico e Geologico. Firenze: Selca. http://geoportale.regione.emilia-romagna.it/it/catalogo/dati-cartografici/biologia/vegetazione/carta-della-vegetazione-parco-regionale-del-delta-del-po-stazione-volano-mesola-goro-digitale-edizione-1999

Puhakka, L., M. Salo, and I. Sääksjärvi. 2011. "Bird Diversity, Birdwatching Tourism and Conservation in Peru: A Geographic Analysis." *PLoS One* 6 (11): e26786. doi:10.1371/journal.pone.0026786.

Rall, E. L., N. Kabisch, and R. Hansen. 2015. "A Comparative Exploration of Uptake and Potential Application of Ecosystem Services in Urban Planning." *Ecosystem Services* 16: 230–242. doi:10.1016/j.ecoser.2015.10.005.

Reganold, J. P., and J. M. Wachter. 2016. "Organic Agriculture in the Twenty-First Century." *Nature Plants* 2 (March): 1–8. doi:10.1038/NPLANTS.2015.221.

Saleh, F., and M. P. Weinstein. 2016. "The Role of Nature-Based Infrastructure (NBI) in Coastal Resiliency Planning: A Literature Review." *Journal of Environmental Management* 183 (3): 1088–1098. doi:10.1016/j.jenvman.2016.09.077.

Sallustio, L., A. De Toni, A. Strollo, M. Di Febbraro, E. Gissi, L. Casella, D. Geneletti, M. Munafò, M. Vizzarri, and M. Marchetti. 2017. "Assessing Habitat Quality in Relation to the Spatial Distribution of Protected Areas in Italy." *Journal of Environmental Management* 201: 129–137. doi:10.1016/j.jenvman.2017.06.031.

Schlüter, M., H. Leslie, and S. Levin. 2009. "Managing Water-Use Trade-Offs in a Semi-Arid River Delta to Sustain Multiple Ecosystem Services: A Modeling Approach." *Ecological Research* 24 (3): 491–503. doi:10.1007/s11284-008-0576-z.

Schuh, M. H., and D. L. Guadagnin. 2018. "Habitat and Landscape Factors Associated with the Nestedness of Waterbird Assemblages and Wetland Habitats in South Brazil." *Austral Ecology* 43 (8): 989–999. doi:10.1111/aec.12648.

Schüttler, E., R. Klenke, S. McGehee, R. Rozzi, and K. Jax. 2009. "Vulnerability of Ground-Nesting Waterbirds to Predation by Invasive American Mink in the Cape Horn Biosphere Reserve, Chile." *Biological Conservation* 142 (7): 1450–1460. doi:10.1016/j.biocon.2009.02.013.

Seijger, C., W. Douven, G. van Halsema, L. Hermans, J. Evers, H. L. Phi, M. F. Khan, et al. 2017. "An Analytical Framework for Strategic Delta Planning: Negotiating Consent for Long-Term Sustainable Delta Development." *Journal of Environmental Planning and Management* 60 (8): 1485–1509. doi:10.1080/09640568.2016.1231667.

Settis, S. 2010. *Paesaggio Costituzione Cemento: La Battaglia per L'ambiente Contro Il Degrado*. Torino, Italy: Einaudi.

Spinelli, E., R. Bisceglia, G. Castaldelli, and G. Colombo. 1996. "Funzionamento Di Un Bacino Idrico Costiero: I Deflussi D'acqua Dolce e Gli Apporti Di Fosforo e Di Azoto Del Po Di Volano Nella Sacca Di Goro." *Ingeneria Ambientale* XXV (1–2): 25–33.

Sundareshwar, P. V., J. T. Morris, E. K. Koepfler, and B. Fornwalt. 2003. "Phosphorus Limitation of Coastal Ecosystem Processes." *Science* 299 (5606): 563–565. doi:10.1126/science.1079100.

Syvitski, J. P. M., A. J. Kettner, I. Overeem, E. W. H. Hutton, M. T. Hannon, G. R. Brakenridge, J. Day, et al. 2009. "Sinking Deltas Due to Human Activities." *Nature Geoscience* 2 (10): 681–686. doi:10.1038/ngeo629.

Tavares, D. C., D. L. Guadagnin, J. F. de Moura, S. Siciliano, and A. Merico. 2015. "Environmental and Anthropogenic Factors Structuring Waterbird Habitats of Tropical Coastal Lagoons: Implications for Management." *Biological Conservation* 186: 12–21. doi:10.1016/j.biocon.2015.02.027.

Temmerman, S., P. Meire, T. J. Bouma, P. M. J. Herman, T. Ysebaert, and H. J. De Vriend. 2013. "Ecosystem-Based Coastal Defence in the Face of Global Change." *Nature* 504 (7478): 79–83. doi:10.1038/nature12859.

Terrado, M., V. Acuña, D. Ennaanay, H. Tallis, and S. Sabater. 2014. "Impact of Climate Extremes on Hydrological Ecosystem Services in a Heavily Humanized Mediterranean Basin." *Ecological Indicators* 37: 199–209. doi:10.1016/j.ecolind.2013.01.016.

Turolla, E. 2008. *L'allevamento Della Vongola Verace Nel Delta Del Po*. Taglio di Po: Grafiche Adriatica.

Tylova-Munzarova, E., B. Lorenzen, H. Brix, and O. Votrubova. 2005. "The Effects of NH4 + and NO3-on Growth, Resource Allocation and Nitrogen Uptake Kinetics of Phragmites Australis and Glyceria Maxima." *Aquatic Botany* 81 (4): 326–342. doi:10.1016/j.aquabot.2005.01.006.

Viaroli, P., M. Bartoli, C. Bondavalli, and M. Naldi. 1995. "Oxygen Fluxes and Dystrophy in a Coastal Lagoon Colonized by Ulva Rigida (Sacca Di Goro, Po River Delta, Northern Italy)." *Fresenius Environmental Bulletin* 4 (6): 381–386.

Viaroli, P., G. Giordani, M. Bartoli, M. Naldi, R. Azzoni, D. Nizzoli, I. Ferrari, et al. 2006. "The Sacca Di Goro Lagoon and an Arm of the Po River." *Handbook of Environmental Chemistry*, Vol.5*: Water Pollution* 5 (PART H): 197–232. doi:10.1007/698-5-030.

Viaroli, P., M. Naldi, C. Bondavalli, and S. Bencivelli. 1996. "Growth of the Seaweed Ulva Rigida C. Agardh in Relation to Biomass Densities, Internal Nutrient Pools and External Nutrient Supply in the Sacca Di Goro Lagoon (Northern Italy)." *Hydrobiologia* 329 (1–3): 93–103. doi:10.1007/BF00034550.

Viaroli, P., E. Soana, S. Pecora, A. Laini, M. Naldi, E. A. Fano, and D. Nizzoli. 2018. "Space and Time Variations of Watershed N and P Budgets and Their Relationships with Reactive N and P Loadings in a Heavily Impacted River Basin (Po River, Northern Italy)." *Science of the Total Environment* 639: 1574–1587. doi:10.1016/j.scitotenv.2018.05.233.

Vincenzi, S., G. Caramori, R. Rossi, and G. A. De Leo. 2006. "Estimating Clam Yield Potential in the Sacca Di Goro Lagoon (Italy) by Using a Two-Part Conditional Model." *Aquaculture* 261 (4): 1281–1291. doi:10.1016/j.aquaculture.2006.08.014.

Welsh, D. T., M. Bartoli, D. Nizzoli, G. Castaldelli, S. A. Riou, and P. Viaroli. 2000. "Denitrification, Nitrogen Fixation, Community Primary Productivity and Inorganic-N and Oxygen Fluxes in an Intertidal Zostera Noltii Meadow." *Marine Ecology Progress Series* 208: 65–77. doi:10.3354/meps208065.

Yu, S., B. Cui, P. Gibbons, J. Yan, X. Ma, T. Xie, G. Song, Y. Zou, and X. Shao. 2017. "Towards a Biodiversity Offsetting Approach for Coastal Land Reclamation: Coastal Management Implications." *Biological Conservation* 214: 35–45. doi:10.1016/j.biocon.2017.07.016.

Appendix

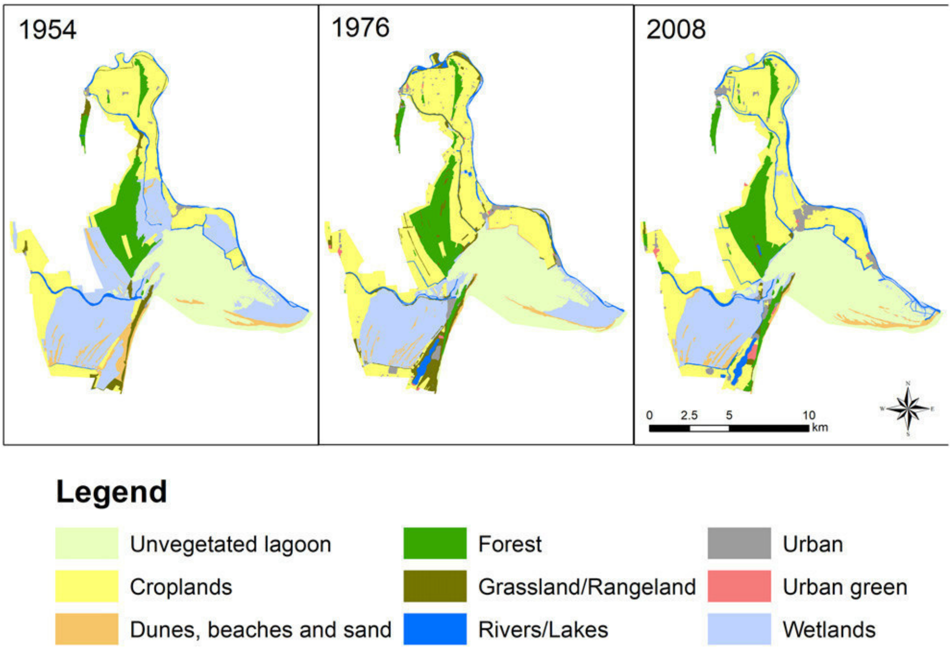


Figure A1. Land use maps of VMG district for the years 1954, 1976, and 2008. Elaboration from Gaglio, Aschonitis, Gissi, *et al*. (2017).

Received: 13 September 2017 | Accepted: 14 May 2018
DOI: 10.1002/hyp.13165

RESEARCH ARTICLE

WILEY

Forest fire effects on groundwater in a coastal aquifer (Ravenna, Italy)

Beatrice M. S. Giambastiani[1] | Nicolas Greggio[2] | Giovanni Nobili[3] | Enrico Dinelli[1] | Marco Antonellini[1]

[1] BiGeA—Department of Biological, Geological and Environmental Sciences, University of Bologna, Ravenna, Italy
[2] CIRI EA—Interdepartmental Centre for Industrial Research on Energy and Environment, University of Bologna, Ravenna, Italy
[3] Carabinieri for Biodiversity, Punta Marina, Ravenna, Italy

Correspondence
Nicolas Greggio, CIRI EA—Interdepartmental Centre for Industrial Research on Energy and Environment, University of Bologna, 48123 Ravenna, Italy.
Email: nicolas.greggio2@unibo.it

Funding information
Flaminia Foundation

Abstract

We examined the fire-induced changes in groundwater recharge rate. This aspect is particularly important in the case of large forested areas growing over a coastal aquifer affected by saltwater intrusion. In the Ravenna coastal area (Italy), pine forests grow on coastal dune belts, overlying a sandy unconfined aquifer, which is strongly affected by marine ingression. Three groundwater profiles across the forest and perpendicular to the coastline were monitored for groundwater level, physical, and chemical parameters. The aims were to define groundwater quality, recharge rate, freshwater volume, and highlight change, which occurred after a forest fire with reference to pre-fire conditions. Analytical solutions based on Darcy Law and the Dupuit Equation were applied to calculate unconfined flow and compare recharge rates among the profiles. The estimated recharge rates increased in the partially and completely burnt areas (219 and 511 mm year^{-1}, respectively) compared with the pristine pine forest area (73 mm year^{-1}). Although pre-fire conditions were similar in all monitored profiles, a post-fire decrease in salinity was observed across the burnt forest, along with an increase in infiltration and freshwater lens thickness. This was attributed to decrease canopy interception and evapotranspiration caused by vegetation absence after the fire. This research provided an example of positive forest fire feedback on the quantity and quality of fresh groundwater resources in a lowland coastal aquifer affected by saltwater intrusion, with limited availability of freshwater resources. The fire provided an opportunity to evaluate a new forest management approach and consider the restoration and promotion of native dune herbaceous vegetation.

KEYWORDS
aquifer recharge, coastal aquifer, forest fire, groundwater, saltwater intrusion

1 | INTRODUCTION

Coastal areas around the world are facing serious freshwater supply problems because of an increase in human activities, especially agriculture and tourism, and climate change, which affects the temperature and rainfall regime.

Many coastal regions have adopted or will adopt management methods to improve fresh groundwater supply to compensate and control, for instance, the salinization of coastal aquifers (Oude Essink, 2001). In past decades, several techniques that might lead to an increase in freshwater availability have been investigated globally, such as Managed Aquifer Recharge (Bekele et al., 2015; Dillon, 2005; Dillon, Pavelic, Page, Beringen, & Ward, 2009; Page, Bekele, Vanderzalm, & Sidhu, 2018; Sprenger et al., 2017), Aquifer Storage and Recovery (Dillon et al., 1999; Page, Peeters, Vanderzalm, Barry, & Gonzalez, 2017), Soil Aquifer Treatment (Barry, Vanderzalm, Miotlinski, & Dillon, 2017; Fox et al., 2001; Sharma & Kennedy, 2017), and water desalination plants (El Saliby, Okour, Shon, Kandasamy, & Kim, 2009; March, Saurí, & Rico-Amorós, 2014).

Controlled and prescribed burning are often used in ecosystems and forests to control vegetation community structure and growth (Obrist, DeLucia, & Arnone, 2003) and for managing forest resources to improve wildlife habitat, manage competing vegetation, prepare sites for seeding and planting, and control insects and disease (Fernandes et al., 2013; Penman, Binns, Shiels, Allen, & Penman, 2011; Vermeire, Mitchell, Fuhlendorf, & Wester, 2004). In Italy, unfavourable land characteristics, conflicting management goals, a hostile socio-cultural environment, and an inadequate regulatory framework have limited the application of prescribed burning until the 2000 (Ascoli & Bovio, 2013). From 2005 to 2012, prescribed burning programmes for fire hazard reduction, forest management, and biodiversity conservation were implemented throughout Italy (Ascoli et al., 2012).

Fire is not normally considered as a tool for enhancing water resources. The hydrological effects of forest fire for water catchment management purposes have been studied for several reasons. Some important reasons include the definition of fire impacts on soil properties (Shakesby & Doerr, 2006; Stoof et al., 2012), changes in erosion hazard and run-off generation due to decreased canopy interception and vegetation removal (Cawson, Nyman, Smith, Lane, & Sheridan, 2016; Fernandez, Vega, Fonturbel, Jimenez, & Perez, 2008; Singh, Schoonover, Monroe, Williard, & Ruffner, 2017), and influence on streamflow regime (Bart, 2016; Feikema, Sherwin, & Lane, P.N:J., 2013; Stoof et al., 2014; Zhou, Zhang, Vaze, Lane, & Xu, 2015). As a consequence, groundwater recharge rates, net infiltration, and water balance can change after a forest fire (Eben & Moody, 2016; Lane, Croke, & Dignan, 2004; Silberstein, Dawes, Bastow, Byrne, & Smart, 2013; Yesertener, 2005). Some recharge simulations and groundwater models (Vogwill, McHugh, O'Boy, & Yu, 2008) have demonstrated that increasing the burn frequency and removal of pine plantations can be the only viable option for a significant increase in recharge to groundwater system. Vegetation, in fact, plays an important role in the water cycle. Forests have contrasting feedbacks on groundwater resources (Callahan, Amatya, & Stone, 2016) depending on the climate system and the morphology of the territory. A positive feedback is the shading of the canopy that prevents soil evaporation. According to the canopy geometry, the stemflow (precipitation which flows down the tree stem) can increase, allowing for more efficient infiltration at the base of the trees (Tanaka, Taniguchi, & Tsujimura, 1996; Taniguchi, Tsujimura, & Tanaka, 1996). Negative feedbacks on groundwater consist of root uptake, which lead to an increase in evapotranspiration (ET) processes, causing a reduction of freshwater recharge into groundwater. High evaporation rates can be significantly important in the case of large forested areas growing over coastal aquifers affected by saltwater intrusion and in low-lying Mediterranean coastal zones where freshwater availability is limited and it has to be preserved. Coastal areas are further threatened by climate change and sea-level rise (IPCC, 2013), which will cause a landward ingression of the sea and modifications in the coastal aquifer recharge dynamics (Yechieli, Shalev, Wollman, Kiro, & Kafri, 2010). Sea level rise in the study area and Mediterranean coastal zone is predicted to increase 0.23–0.57 m from 2081 to 2100 (IPCC 2013; Perini et al., 2017). This will cause severe adverse impacts on saltwater intrusion processes in coastal aquifers and on sea water encroachment along river mouths (Giambastiani, Antonellini, Oude Essink, & Stuurman, 2007).

In the Emilia-Romagna region, northeast of Italy, temperature is predicted to increase, with higher minimum temperatures than average the whole year round. In winter, maximum temperature will be higher than current average, whereas summer will remain mostly unchanged (Mollema et al., 2012; Tomozeiu, Cacciamani, Pavan, Morgillo, & Busuioc, 2007). The annual rainfall regime is predicted to decrease by 2.5–6.1%, with less rain in summer and relatively more rain in winter (Mollema et al., 2012). The economic damage to agriculture, particularly in the Po River Valley with its extensive irrigation crops, may become considerable (Villani, Tomei, Tomozeiu, & Marletto, 2011). Altered precipitation regimes as well as temperature and ET increases may cause a decrease in aquifer recharge. Along the coast, this will result in a landward shift of the brackish-saltwater wedge (Mollema & Antonellini, 2013), an increase in the salinity of coastal wetlands (Antonellini & Mollema, 2010), and a progressive thinning of freshwater lenses below coastal dunes (Cozzolino, Greggio, Antonellini, & Giambastiani, 2017).

The objective of this study was to evaluate post-fire effects on aquifer recharge and alteration of groundwater salinity and freshwater availability in the coastal aquifer of Ravenna, a low-lying coastal area in the Po River Plain (NE Italy). Pre-fire and post-fire monitoring data related to groundwater level, physical, and chemical parameters were analysed to highlight changes occurring after a forest fire that, in July 2012, devastated a large coastal pine forest (about 118 ha). Post-fire increases in aquifer recharge and freshwater availability were evaluated by calculating flow and infiltration rates along profiles located in different portions of the forest (burnt, partially burnt, and pristine forest). Groundwater levels pre-fire and post-fire were compared, and the recharge rates were extrapolated with the objective of matching modelled and measured hydraulic head.

1.1 | Site description

In the Ravenna coastal area (Figure 1a), several dense pine forests grow on dune belts, bordering beaches, and active dunes. The Ravenna coastal pinewood, dominated by *Pinus pinaster*, were planted starting from the end of the 19th century, constituting an alien forest, providing a social and historical landmark for the local population. Originally, the pinewoods were planted to protect agriculture and reclamation land from marine spray (ASFD, 1960; Benini, 1931) and then they were also exploited for pine nuts and timber, providing employment to a large number of locals (ASFD, 1950). With the tourism boom of the 1970s, the coastal pinewoods increased their importance and functions as recreational areas (picnic, excursions, bird watching, bike tracks, etc.), and a new naturalistic and conservation role was approved by local management plans (Longhi, 1969; Naccarato, 1971). In order to guarantee the protection and preservation of the pinewood habitat, in 1977, the *Riserva Naturale dello Stato* (State Nature Reserve)—Ravenna Pine Forest was established (Cantiani, Ferretti, Pignatti, Andreatta & Nobili, 2008).

The study area is situated within the *Ramazzotti* coastal forest, located south-east of Ravenna (Figure 1). This pine forest extends

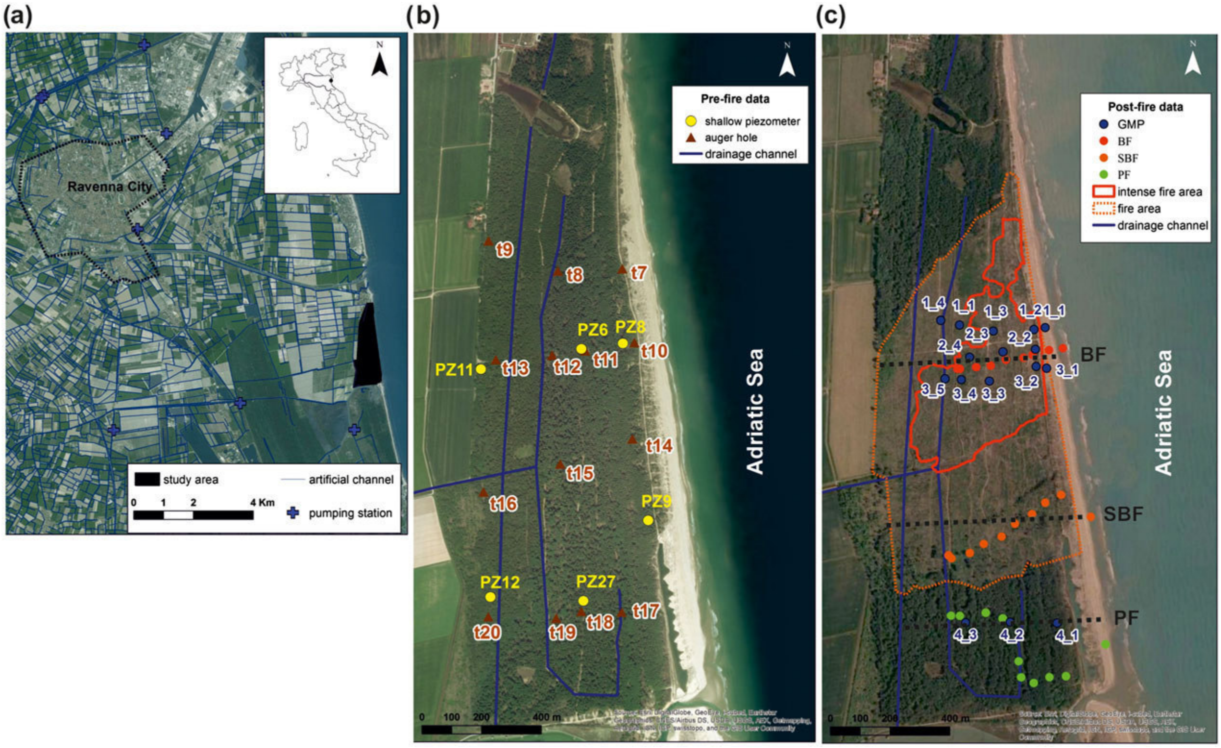


FIGURE 1 Location of the study area (black polygon) within the Ramazzotti coastal forest (a) and distribution of the pre-fire (b; 2015 base map) and post-fire (c; 2017 base map) monitoring points and profiles (black dashed lines). GMP is for Ground Monitoring Point as in Cozzolino et al. (2017), BF for burnt forest, SBF for partially burnt forest, and PF for pristine forest in order to distinguish the different impacts of the fire

parallel to the coast for 2.5 km and inland for 0.6 km where transition to low-laying farmland occurs.

The elevation of the Ravenna coastal area varies from 4 m a.s.l. of coastal dunes (where still present) to about −2 m a.s.l. in inland areas. The forest average elevation is 0.3 m a.s.l. The coastal plain is undergoing subsidence, mostly due to anthropogenic factors, such as groundwater overexploitation for industry and irrigation purposes, natural gas extraction, and drainage (Teatini, Ferronato, Gambolati, Bertoni, & Gonella, 2005). The high land subsidence rate presents a major threat for this coastal environment. Under natural condition, the subsidence is a few millimetres per year, whereas anthropogenic subsidence rate is about 6–21 mm year^{-1} in the monitored area (Taramelli, Di Matteo, Ciavola, Guadagnano, & Tolomei, 2015). Because of low topography, high rate of natural and anthropic subsidence and an extensive drainage system (Antonellini, Allen, Mollema, Capo, & Greggio, 2015), the coastal aquifer is subjected to marine ingression with groundwater that remains brackish to saline. Freshwater lenses are limited in areal extent, thickness, and time and are related to dune heights in coastal systems (Cozzolino et al., 2017). The thin freshwater lenses in the shallow Ravenna coastal aquifer are limited in space and have limited use for irrigation or human activity. Where present, however, they are important to reduce water and soil salinization, which in turn control the health of the whole coastal ecosystem (Antonellini & Mollema, 2010; Greggio, Giambastiani, & Antonellini, 2017; Mollema et al., 2012).

The low topography of the pine forest generates shallow water table, which reaches the surface during wet winter months. In order to prevent damages to pine roots (Antonellini & Mollema, 2010), three drainage ditches were excavated across the forest (blue lines in Figure 1a and 1b). They equally drain all portions of the forest and keep the topographically lowest portions dry by collecting and routing water towards water pumping stations located several kilometres inland (blue symbols in Figure 1a). The water level in the drains within the forest is almost constant throughout the year (around about −0.7 to −0.75 m a.s.l.).

The pine forest overlies a sandy coastal unconfined aquifer, which extends for about 6–7 km inland. The forest represents the only recharge area where rainfall may deeply percolate to the groundwater. The coastal aquifer has a thickness ranging from 12 to 25 m and consists of two sandy units (0–7 m and 23–25 m a.s.l.) separated by a prodelta deposit with alternations of silt, clay, and fine sand layers (Amorosi, Colalongo, Pasini, & Preti, 1999; Marchesini et al., 2000). The bottom-confining layer (aquifer basement) is an impermeable clay at −25 to −30 m a.s.l. The hydraulic conductivity decreases from 10^{-4} (sandy units) to 10^{-6} m s^{-1} (prodelta sediment), from the top of the aquifer to its bottom until the impermeable basement (10^{-9} m s^{-1}; Giambastiani, Colombani, & Mastrocicco, 2012). The saturated thickness of the aquifer investigated in this study was approximately 6–7 m, to the top of the prodelta unit, which acts as an aquitard unit especially in proximity of the sea.

1.2 | July 2012: Forest fire

In July 2012, a large fire devastated more than 60 ha of the pine forest with 20 ha that were completely destroyed and left with bare soil and ashes (Figure 2). A portion of forest was completely burnt (burnt forest—BF), a portion was partially burnt (semi-burnt forest—SBF), and the

(a) (b)

(c)

FIGURE 2 Photos of the coastal pine forest taken during fire in July 2012 (a) and following days (b; credits to Silvano Foschini, Archive of Carabinieri for Biodiversity, Punta Marina Office); aerial photo taken in February 2013 showing the burnt portion of the forest, which was left with bare soil and ashes (c, credits to Dr. G. Nobili)

southern portion of the forest was not reached by the fire (pristine forest—PF; Figure 1c).

2 | MATERIAL AND METHODS

2.1 | Data collection

2.1.1 | Pre-fire data

The pre-fire dataset is based on monitoring by Marconi, Antonellini, Balugani, and Dinelli (2011). Chemical and physical parameters from groundwater samples were collected via auger holes and shallow piezometers. The auger hole campaign included 14 locations within the pine forest (brown triangles in Figure 1b) where hydraulic head (m a.s.l.), temperature (°C), pH, redox potential (mV), and electrical conductivity (mS cm^{-1}) of the groundwater were measured in March 2008. In addition to field measurements, all groundwater samples were analysed for major ions (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, Cl^{-}, HCO_3^{-}, SO_4^{2-}, NO_3^{-}) using an isocratic dual pump ion chromatography ICS-1000 Dionex. Ionic balance was computed to evaluate analysis reliability, and data were discarded when exceeding + − 5%.

In order to investigate the deepest part of the aquifer, a network of six fully screened shallow piezometers (5 m deep) were installed along two transects perpendicular to coastline (yellow dots in Figure 1b). Two sampling campaigns were conducted in August 2009 and March 2010 using a straddle packers system with a 0.2-m-sampling-window connected to a submersible centrifuge pump (Mastrocicco, Giambastiani, Severi, & Colombani, 2012). In this case, only chemical–physical parameters were measured in the field, and no chemical analyses were performed in the lab.

2.1.2 | Post-fire data

Eight months after the forest fire, on March 2013, an auger hole campaign was carried out along the three profiles (BF, SBF, PF in red, orange, and green dots in Figure 1c, respectively).

Eight auger holes were drilled for each profile to monitor water level and groundwater chemical–physical parameters by a multiparameter probe. Samples of seawater and the two drainage channels in the forest were also collected. All water samples were analysed for major cations and anions (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, Cl^{-}, HCO_3^{-}, SO_4^{2-}, NO_3^{-}) to highlight any hydrochemical change in pre-fire and post-fire conditions.

To highlight changes occurring in the aquifer during the post-fire forest recovery, a network of 16 Groundwater Monitoring Points (GMPs) were put in place on July 2014. This groundwater monitoring network was based on multilevel sampler techniques (Balugani & Antonellini, 2011; Davis & Barber, 1994) consisting of boreholes with multiple filters at different depths. These filters provided samples from several isolated depths and achieved a high-resolution characterization of the aquifer, avoiding mixing and cross-flow problems. GMPs (blue dots in Figure 1c) were organized in four profiles located at 70, 100, 230, 350, and 400 m from the coastline (three profiles in BF and three in PF) and their elevation was determined by Real Time Kinematic-GPS surveys. The first GMP was located on the active foredune, whereas the innermost was situated between the two drainage channels. Each GMP consisted of eight minifilters (the first five were 0.5-m-apart and the last three were 1-m-apart) and a 0.06 m diameter piezometer, open at the bottom, for measuring water level. Groundwater samples from each minifilter were collected by low-flow technique through an inertial pump (about 0.5 L min^{-1}). Hydraulic head (m a.s.l.), electrical conductivity (mS cm^{-1}), and temperature (°C) were measured by water level logger and a multiparametric probe. (See Table 1 for monitoring campaign details.)

Climate data were derived from Ravenna weather stations included in the Hydro-Meteo-Climate Service of the Emilia-Romagna Region (Dexter system, http://www.arpae.it/). A 40-year time series was used to set the climate period considered in the study (2008–2016) into a wider time scale.

2.2 | Data analysis

The relative homogeneity of the aquifer properties and the simple 2D flow conditions in the coastal aquifer (Giambastiani et al., 2007, 2012; Mastrocicco et al., 2012) allowed the use of an analytic solution approach. The analytical solutions of Darcy Law and Dupuit equation (Fetter, 2001) were applied to calculate the unconfined flow and hydraulic head (h), based on the following:

$$h=\sqrt{h_1^2-\frac{\left(h_1^2-h_2^2\right)x}{L}+\frac{w}{K}(L-x)x}, \qquad (1)$$

where x is the distance from coastline (m); h_1 and h_2 are the hydraulic heads at the origin and at a distance L, respectively (m); K is the hydraulic conductivity (m day^{-1}); and w is the recharge rate (m day^{-1}; Figure 3). In these calculations, hydraulic heads were measured during the monitoring campaigns, and hydraulic conductivity value was derived by slug tests (Bouwer, 1989; Bouwer & Rice, 1976) carried out in several piezometers located in the forest. Distances and flow lengths were based on GPS topographic profiles and potentiometric surfaces. Along all profiles reported in Figure 1, groundwater levels, as calculated by using Equation (1), and measured levels were compared, whereas the recharge rates (w in Equation (1)) were inverted to match the measured hydraulic heads.

Rainfall was presented as Cumulative Rainfall Departure (Weber & Steward, 2004), and average daily temperature was presented in an annual curve (Figure 4).

Groundwater level and salinity data pre-fire and post-fire were compared and plotted along the monitored profiles (Figures 5–7). Major cation and anion concentrations were presented by using Langelier–Ludwig graph (Langelier & Ludwig, 1942; Figure 8).

3 | RESULTS AND DISCUSSION

The mean annual rainfall is 637 mm $year^{-1}$ in the period 1976–2016 (Figure 4). The Cumulative Rainfall Departure graph highlights a long dry period in the decade from 1982 to 1992, when the precipitation is under the historical average; the following period (1993–2005) is marked by above average precipitation. During recent years, a short

TABLE 1 List of monitoring campaigns and applied method for groundwater sample collection

	Date	Auger hole	Straddle packers	Multilevel sampler
Pre-fire	March 2008	Chemical–physical parameters. Cations and anions		
	August 2009; March 2010		Chemical–physical parameters.	
Post-fire	March 2013	Chemical–physical parameters. Cations and anions		
	November 2014; February and July 2015; May 2016			Chemical–physical parameters.

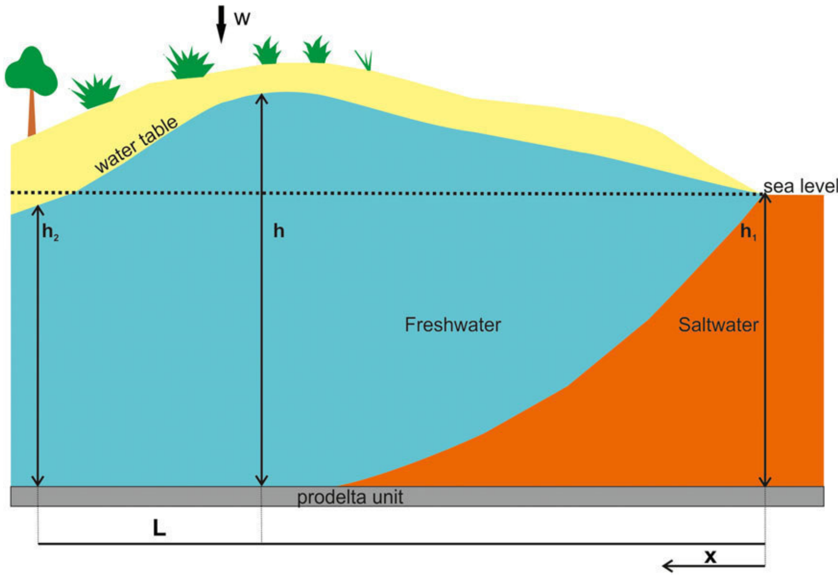


FIGURE 3 Scheme of the unconfined flow based on the analytical solutions of Darcy Law and Dupuit equation (Fetter, 2001)

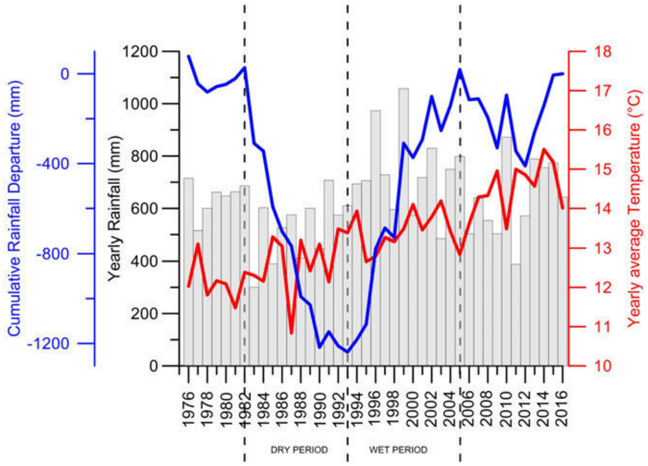


FIGURE 4 Bar graph showing the average annual rainfall. Red line indicates the average annual temperature, whereas blue line indicates the Cumulative Rainfall Departure for 40 years time period (1976–2016). Original data are from the Dexter database (http://www.arpae.it/)

dry phase is evident from 2006 until 2012; this phase is transitory, and in the last 4 years, excluding 2016, constant annual rainfall above 700 mm year^{-1} is recorded. The average annual rainfall for the period 2005–2016 corresponds to the annual average of the whole period 1976–2016, indicating stable amount of water input from precipitation. During the years monitored in this study, the mean annual rainfall does not show any particular trend.

The annual water surplus available for infiltration is minimal (60–150 mm year^{-1}) due to the high ET rate of the pine forest. The ET rate, in fact, ranges from about 300 mm year^{-1} to about 1,000 mm year^{-1} (Arpae, 2017; Mollema et al., 2012; Mollema, Antonellini, Gabbianelli, & Galloni, 2013a). Based on water budget developed for the forests of Ravenna by Mollema and Antonellini (2013) and Mollema, Antonellini, Gabbianelli, and Galloni (2013a), there is a general deficit over the whole year, during which ET exceeds precipitation and a limited surplus occurs only in winter (from November to March). The average air temperature constantly increases during the whole time series (1976–2016); in particular, in the last 6 years, 5 years have an average temperature at or above 15°C (Figure 4).

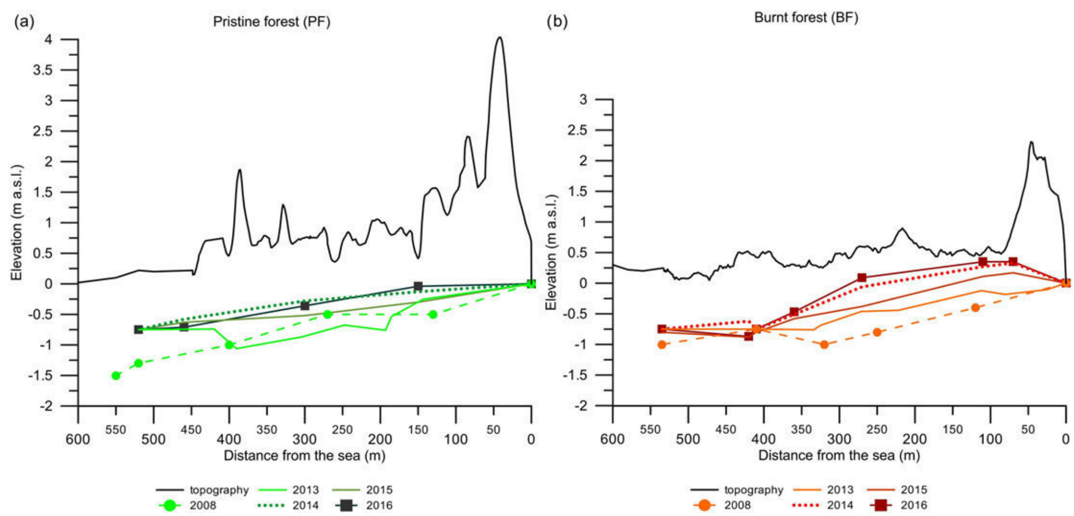


FIGURE 5 Water table level comparison pre-fire (2008) and post-fire (2013–2016) in the unburnt (PF, a) and burnt (BF, b) forest. Black lines show the topography profiles (refer to Figure 1 for locations)

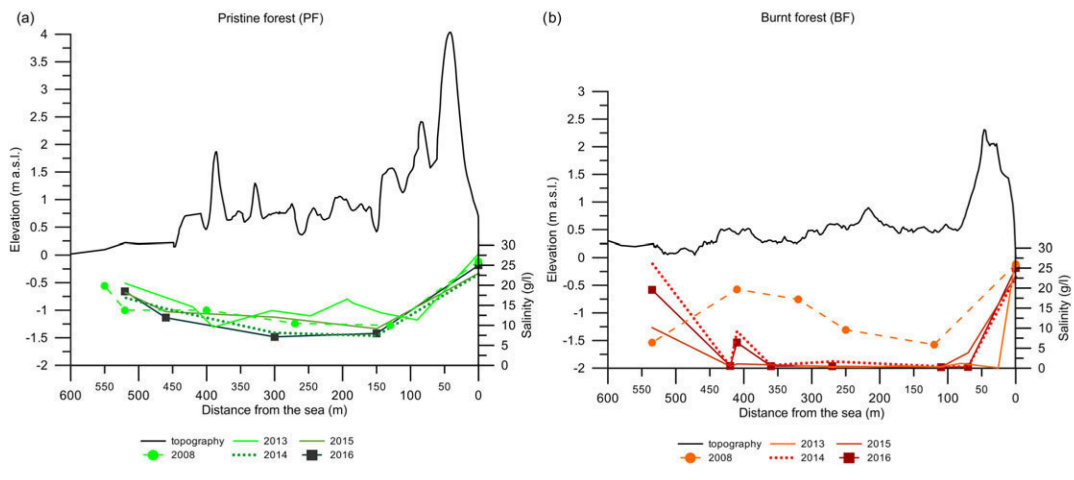


FIGURE 6 Groundwater salinity comparison pre-fire (2008) and post-fire (2013–2016) in the unburnt (PF, a) and burnt (BF, b) forest (refer to Figure 1 for locations)

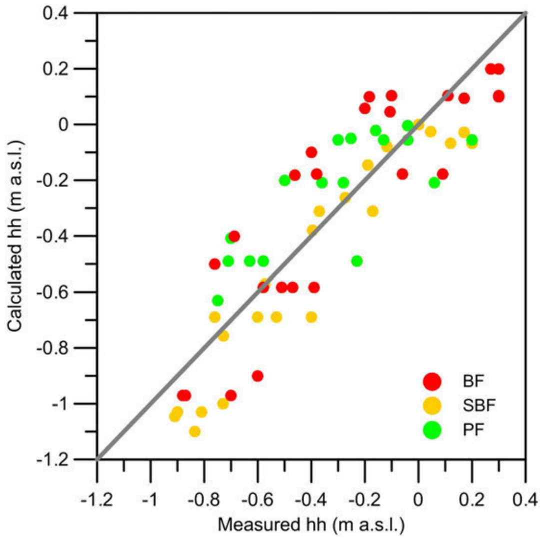


FIGURE 7 Comparison between measured and calculated hydraulic head in the three forest areas (BF, SBF, and PF) for the post-fire period

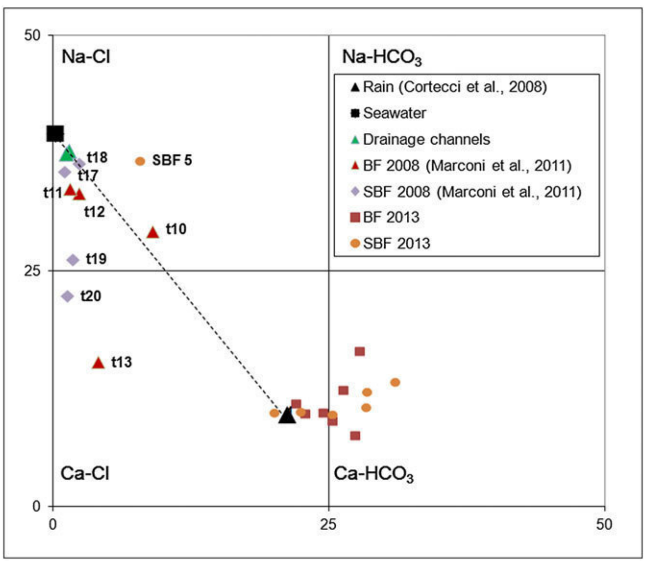


FIGURE 8 Langelier–Ludwig diagram of the groundwater samples collected via auger hole technique in 2008 (pre-fire) and 2013 (post-fire). The dashed line corresponds to the mixing line between seawater (black square) and rainwater (black triangle) end-members. The pre-fire data are from Marconi et al. (2011)

The water table level (Figure 5), in both pre-fire and post-fire conditions, illustrates a forced gradient from the sea towards the inland drainage ditches (about 400 and 500 m from the coast). Generally, the water table ranges from 0 m a.s.l. in proximity of the sea to −0.7 to −1.0 m a.s.l. nearby the drainage ditches. This agrees with past studies in this coastal area, indicating a general inland-directed hydraulic gradient created by the drainage ditches at the back of the beach-dune system (Antonellini et al., 2008; Cozzolino et al., 2017; Vandenbohede, Mollema, Greggio, & Antonellini, 2014). Before the fire (2008), monitoring data indicate similar water table levels in all portions of the pine forest, with average values ranging from −0.5 m a.s.l. below the dune to −1.2 m a.s.l. in the inner part of the forest.

Both water table level and salinity in PF (Figures 5a and 6a) remain quite constant during the years following the fire. Compared with the 2008 values (Figure 5a), the water table rises 0.40 m below the dune, 0.14 m at about 250 m from the coastline, and 0.29 m in the central area (about 400 m from the coastline). On the contrary, a significant increase in water table level and a decrease in salinity are recorded in the SBF (profile not shown) and BF (Figures 5b and 6b) during 2013–2015. In 2016, the water table in BF rose 0.75 m below the dunes, 0.90 m at 250 m from the coastline, and 0.50 m in the central area (about 350 m from the coastline), compared with the 2008 values (Figure 5b). From 2013 to 2016, water table values rose on average 0.2 m year^{-1} in BF.

Figure 6b shows the abrupt decrease of top aquifer salinity recorded in the BF transect compared with the pre-fire situation. Apart from the sea and drainage ditches, which have high salinity values (25–30 g L^{-1}), all measurements display salinity ranging from 0.1 to 3 g L^{-1}. A post-fire freshwater lens forms below the crest of the dune, and it extends in the backdune and through the forest until 350 m from the sea (Figure 6b), whereas the pre-fire conditions shows higher salinity values, between 6 and 17 g L^{-1}, in all transects.

Water level and salinity time series for the same three profiles are not available. These results, however, indicate hydraulic conditions and salinity trends, which are typical of all Ravenna's coastal pine forests (Antonellini et al., 2008; Giambastiani et al., 2007; Marconi et al., 2011): a shallow brackish–freshwater interface with limited seasonal variations, and small freshwater lenses only below the highest dunes.

The comparison between hydraulic head measured in the field during monitoring campaigns and calculated hydraulic head (Figure 7) shows a general good coefficient of determination (R^2 = 0.87), ranging from the minimum value of 0.78 to the maximum value of 0.95 in PF and SBF, respectively. The overall mean root square error is 0.18 m, ranging from 0.16 to 0.20 m in the SBF and BF, respectively.

The recharge rates (w) are obtained by inversion from Equation (1) to obtain the best fit between modelled and measured hydraulic heads. The best fit data for the analytical modelling, necessary for recharge rate calculation, are reported in Table 2. The hydraulic head values are computed on average values. The total flow length L is constrained by the position of the dune and the drainage ditch on each profile. The hydraulic conductivity (K) is the average of the values obtained by slug tests in the study area. The results of these slug tests

TABLE 2 Best fit data used in the recharge rate calculation for the three forest areas (BF, SBF; and PF)

Parameters	BF	SBF	PF
h_1 (m)	7.00	7.00	7.00
h_2 (m)	6.03	5.97	6.51
K (m day^{-1})	10	10	10
L (m)	420	420	460
w (mm year^{-1})	511	256	73

Note. Refer to Equation (1) and Figure 3 for variable explanations: h_1 and h_2 are hydraulic heads at the origin and at a distance L; K is the hydraulic conductivity; and w is the recharge rate.

are representative of well-sorted sand, as expected in the case of beach-dune system, ranging from a minimum of 1 to a maximum of 10 m day^{-1}.

Previous water budget studies conducted by Mollema and Antonellini (2013) and Antonellini et al. (2008) in the same coastal area indicate limited water surplus (60–150 mm year^{-1}) available for aquifer recharge and infiltration. The estimated recharge rates, obtained here by analytical modelling, indicate that recharge increased in the burnt area (256 mm year^{-1} in SBF and 511 mm year^{-1} in BF), compared with PF (73 mm year^{-1}). These recharge values are within the average range calculated in the previous works (Mollema et al., 2012; Mollema & Antonellini, 2013; Mollema, Antonellini, Gabbianelli, & Galloni, 2013a). A recharge rate like the one estimated for BF (511 mm year^{-1}, corresponding to 1.4 mm day^{-1}), which is greater than 50% of the annual precipitation, is justified by complete vegetation removal and high decrease in ET due to land use change right after the fire: from dense arboreal cover to coarse bare soil and patchy grass. Infiltration rates on exposed unvegetated sand dunes appear to be quiet high (Dingman, 2008). This recharge rate will tend to decrease along with sand stabilization and the establishment of vegetation, which causes the increase in ET (Hugenholtz & Koenig, 2014), as supported by the smaller value of recharge estimated in SBF.

In the forested area, aquifer recharge is limited to autumn and winter seasons, whereas during spring and summer water budget is negative as high ET rate exceeds precipitation. For example, interception, transpiration, and soil evaporation for pine forests in Belgium and the Netherlands range, in total, between 25% and 58% of annual precipitation (Batelaan & de Smedt, 2007), whereas for pine forests in Ravenna, Mollema, Antonellini, Gabbianelli, and Galloni (2013a) obtain a maximum value of 25%. This latter study demonstrates that pine tree transpiration has an important role in the water budget. In fact, considering the estimated actual pine tree transpiration rates of 10–30 L per tree day^{-1} (Mollema, Antonellini, Gabbianelli, & Galloni, 2013a) and the density of trees in the PF (10–20 trees per 100 m^2; Antonellini & Mollema, 2010; Teobaldelli, Mencucci, & Piussi, 2004), it is possible to estimate a transpiration component of ET that can be up to 3 mm day^{-1} in the warm season. As a result of this, infiltration in the forest with high pine tree density is lower than in the portion left with bare soil after the fire. Moreover, during winter season, water infiltrating through the aquifer is quickly intercepted by the drainage system, which keeps the forest dry and preserves the pine trees. The high infiltration rates recorded in BF are likely due to reduced forest cover and associated decrease in ET.

Silberstein et al. (2013) demonstrated that forest sites affected by fire tend to show significant increase in soil moisture storage due to reduced leaf area relative to the unburnt sites and the associated reduction in ET from vegetation. Another study (Mullen, Springer, & Kolb, 2006) demonstrates that climate variations, such as very wet or dry periods after the fire, complicate the use of fire as a tool for enhancing groundwater recharge. Precipitation in the post-fire period (2013–2016) is 16% larger than the average annual value (1976–2016). We do believe, however, that rainfall increase cannot be the only main driver of the measured water table trends across the forest after the fire, given that the same increase does not occur in PF, which has experienced identical climatic conditions.

In some cases, the positive effects on infiltration, ET, and groundwater level can last only for a few seasons after the fire. Sometimes, after an initial period of enhanced infiltration that recharges the sandy aquifer, the rapidly regenerating vegetation may consume most of the infiltrating water, terminating the positive effect of the fire (Silberstein et al., 2013). In our case, however, the positive effects last longer and increased water table levels and high recharge rates, along with a decrease in shallow salinity, are also recorded in 2015 and 2016, 4 years after the fire.

In addition to the contribution of decreased canopy interception and plant uptake, increased preferential flow (finger flow) could be a potential mechanism that enhances deep infiltration of rain and causes more rapid groundwater recharge (Ritsema & Dekker, 1995; Stoof et al., 2014). The authors conclude that, by reducing topsoil moisture and increasing soil moisture variability, fire increases the propensity for preferential flow in burned soil, promoting deep infiltration. However, this is more important in clay-rich soils rather than in sandy soil like in our study area where infiltration is fairly rapid.

A diminished moisture content in topsoil after fire is common (Silva, Rego, & Mazzoleni, 2006; Stoof, Moore, Ritsema, & Dekker, 2011) and can be explained by increased albedo and lack of shading on exposed and blackened burned soil. These conditions lead to high post-fire soil temperatures and increased soil evaporation. Stoof et al. (2014), however, show that the drying effects of fire on soil can be often restricted to the first top centimetres (5–6 cm). Below the topsoil, on the other hand, burnt soils can be as wet as or even wetter than unburnt soils. Higher soil evaporation in burnt soil seems to be balanced by lower ET at the roots depth (Silva et al., 2006) and by an increased infiltration, in terms of finger flow development that enhances deep infiltration of rain and causes more rapid groundwater recharge (Ritsema & Dekker, 1995).

The other aspect to be taken into consideration is the soil water repellency, which is caused by hydrophobic, long-chained organic molecules released after vegetation fire from burning plant litter (Doerr, Shakesby, & Walsh, 2000). Some plants (e.g., pines) containing resin, wax, or aromatic oil, are commonly associated with soil water repellency (DeBano, 1981, 2000; Doerr et al., 2000). Several studies, however, show that an increase in soil water repellency is not always observed in post-fire conditions. The lack of fire-induced soil water repellency is common for soils that have “natural background” or “pre-fire” soil water repellency, as demonstrated, for instance, in soils under *P. pinaster* forest in Spain (Rodríguez-Alleres, Varela, & Benito, 2012) and Portugal (Doerr, Shakesby, & Walsh, 1996, 1998).

Fire behaviour, fire severity, and temperature gradients developing in the soil during a fire, all affect the formation of a water repellent layer (DeBano, 2000). Water repellency, however, is destroyed when soils are heated above 270°C (Doerr et al., 2000; Giovannini & Lucchesi, 1997; Savage, 1974). This temperature can be easily reached in the case of forest fire (Dennison, Charoensiri, Roberts, Peterson, & Green, 2006; Marcelli, Santoni, Simeoni, Leoni, & Porterie, 2004; Wotton, Gould, McCaw, Cheney, & Stephen, 2011) like the uncontrolled wildfire that occurred in 2012 within the study area.

Wildfires are often followed by a significant increase in run-off and erosion due to direct damage of vegetation and alteration of physical and chemical soil properties, which in turn affect infiltration, run-off, sediment erosion, and transport (Moody, Martin, Haire, & Kinner, 2008; Rulli, Offeddu, & Santini, 2013). Rainfall intensity after the fire, the time interval between the fire episodes (Rulli, Spada, Bozzi, Bocchiola, & Rosso, 2006), topography, and slope morphology could also influence the post-fire changes in run-off and soil erosion (Andreu, Imeson, & Rubio, 2001). In the study area, however, we estimate an increase in groundwater recharge, as demonstrated by the water table rise, and neither increased run-off nor soil erosion. It is possible that the presence of a large amount of ash and pine-needles mitigated soil erosion immediately after the fire (Cerdà & Doerr 2008). Nevertheless, in the study area, the post fire rainfall regime was not characterized by heavy rainfall (summer period) that could exacerbate soil erosion. Moreover, the area affected by the fire presents a rather flat topography (with the exception of the crest dune) without significant slope that can facilitate run-off and intensify water erosion processes after vegetation removal.

The geochemical results related to the two auger hole campaigns (2008 and 2013) are presented in Figure 8 (refer to Figure 1b for location), along with regional average values of rainwater measured by Cortecci, Dinelli, and Mussi (2008) and seawater and drainage water collected in the study area and used as end-members.

Water major element composition from pre-fire auger holes (BF 2008 and SBF 2008) belongs to the Na–Cl group, close to both seawater and drainage water end-members. In the pre-fire condition, the majority of shallow groundwater samples have a composition falling on the seawater mixing line (from brackish to salt waters). Among them, three samples collected in the inner part of the forest (t13, t19, and t20 in Figure 1b) deviate downward to Ca–Cl composition. This downward trend indicates the intrusion of saline water into freshwater (salinization process) as the dominant process in the system (Stuyfzand, 2008). Base-exchange reactions occurred in the inner portion of the pine forest, where dominant marine cations (mainly Na) could be absorbed in the matrix and fresh cations (such as Ca) could move to solution, as discussed in Capaccioni, Didero, Paletta, and Didero (2005). The cation exchange process, which is a common process during saltwater intrusion in coastal aquifers in the presence of exchangers (Andersen, Nyvang, Jakobsen, & Postma, 2005) explains the relative Ca^{2+} excess and Na^{+} depletion in these water samples (Giambastiani, Colombani, Mastrocicco, & Fidelibus, 2013). Mollema et al. (2013b) observed the same salinization trend along a wide portion of the Ravenna coastal aquifer, where freshwater types ($Ca–HCO_3$) were recognized only below the active coastal dune. Only auger hole "t10," located at the edge between pine forest and active dunes, shows sea and rainwater mixed composition, falling on conservative mixing line (dotted line in Figure 8).

Almost all samples from the post-fire auger hole campaign are situated near the rainwater end-member, with dominant $Ca–HCO_3$ composition, indicating important surface infiltration and mixing of rainwater with shallow groundwater. High concentration of Ca^{2+}, in fact, is typical of freshwater indicating infiltration of rainfall through the vadose zone enriched in carbonate, which are abundant in this aquifer (Cozzolino et al., 2017; Giambastiani et al., 2013; Mollema, Antonellini, Dinelli, et al., 2013b). The change in forest cover, as consequence of fire, left wide portions of palaeo-coastal dune covered only by bare sand or patchy grass and directly exposed to rainfall infiltration. In a recent study conducted on the Adriatic coastal dunes, Cozzolino et al. (2017) indicated land cover type and distance from the drainage ditches as two of the main factors controlling freshwater availability and thickness of freshwater lenses in this coastal aquifer. In Cozzolino et al.'s (2017) work, thin $Ca–HCO_3$ freshwater lenses are related to dunes with bare land cover.

Among post-fire water samples, no sample shows mixing composition of sea and rainwater, and only one sample, from a depression in SBF (auger hole SBF5), reveals Na–Cl composition.

3.1 | Pine forest management

Despite their anthropogenic origin, the coastal pine forests in the Italian NE Adriatic regions assume a relevant role as both biodiversity hot spots and ecological corridors. For these reasons, the current management of the *Ramazzotti* pine forest tries to achieve a multitude of goals: to improve the mechanical stability and ecological forest conditions, restore relevant coastal habitats, protect native fauna, and defend forest from marine ingression and salinization in a contest of high tourist relevance.

Management strategies can be divided in two categories:

1. Keep natural forest-dynamism—do not operate (DNO);
2. Interventions for renaturalization of pine forests encouraging gradual introduction of natural renewal of native species through the decrease of pine density and thinning interventions to reduce pine canopy cover.

The DNO strategy can be effective where the environmental conditions are stable. If changes due to marine ingression and soil and groundwater salinization are limited, pine forest has a long period of natural unconstrained evolution. On the contrary, an intense pinewood-thinning intervention can be conducted to encourage native species and limit pine growth. As reported by Yesertener (2005), water table level rise can result from bush fire but also clearing and thinning of trees, which induces positive effects on groundwater recharge. It has been modelled and proven (Vogwill et al., 2008) that thinning/clear felling of pines and conversion to low density native vegetation result into a locally increased groundwater recharge and storage, even in the case of medium-term dry climate scenario (IPCC, 2013). This could be particularly important with reference to future climate changes, and foreseen increase in temperature (IPCC, 2013), and with reference to the significant extent of coastal pine forests (about 2300 ha) along the entire Ravenna coastline.

In the study forest, the DNO strategy was widely applied before the 2012 fire. After the fire, a deep change in management strategy took place, and an intense pinewood-thinning intervention was conducted. The renaturalization interventions have shown positive effects both on groundwater quality and quantity, thus representing adaptive management strategy during fast changing phases.

Local stakeholders are now accepting that the position of this coastal forest needs to be reconsidered due to the high rate of coastal erosion and subsidence in the area.

In a medium-long period management perspective (>10 years), increasing soil and groundwater salinity levels will lead to decline or disappearance of pine forest and many species typical of freshwater habitats. In this new vision, native dune herbaceous species and bush vegetation need to be preserved along the remaining coastal dune belts, which are the only physical barriers to protect the inland from marine ingression and washover events. In our case, the forest fire and the current erosion rates affecting the area (Scarelli et al., 2017) have generated the conditions for an inland retreat of the forest leaving more space for coastal dunes. The forest renaturalization, by encouraging gradual natural renewal of native species, will create dynamic conditions and formation of different transition habitat types, counteracting the loss of biodiversity occurred by the fire. In this new management perspective, the inner forest will have highest topographic areas (palaeo-dunes) covered by groups of arboreal (*Pinus* spp., *Quercus ilex*) and shrub species (*Hippophae rhamnoides*, *Juniperus communis*, *Phyllirea angustifolia*), whereas the lowest topographic areas (flooded dune slacks) will be characterized by shallow water table and halophilic vegetation, able to adapt to high salt concentration during winter storm and washover events. Vegetation community in lentic ponds and occasionally in dune slacks probably will evolve in brackish wetlands mostly with halophytes or salt-tolerant species (community of *Juncus acutus* and *Juncus littoralis* and especially the communities of *Tripolium pannonicum* and *Spartina anglica*, occurring in saline habitats).

Most of the dunes along the Ravenna coast are confined and bordered by pine forests that limit their natural dynamic (Cozzolino et al., 2017). The perspective of leaving more space for dune natural evolution will permit to develop a wide buffer zone able to counteract and limit storm negative effects (soft engineering intervention; Nordstrom, Armaroli, Jacksin, & Ciavola, 2015). Moreover, wide coastal dune belt covered by native herbaceous and shrubby vegetation succession (Sburlino, Buffa, Filesi, Gamper, & Ghirelli, 2013) will also guarantee wide basin for infiltration and aquifer recharge (Antonellini et al., 2008). In a long-term perspective (around 50 years), when the last remaining dune system will be eroded, forest management will depend on the new position of the coastline and erosion rate.

Other possible management solutions that aim to improve groundwater quality and quantity in the study coastal aquifer have been recently investigated, such as coastal dune recovery and restoration (Cozzolino et al., 2017), infiltration ditch (Vandenbohede et al., 2014; Greggio et al., 2017 under review), but none of them take into consideration interventions of renaturalization of pine coastal forests and the effects on groundwater.

4 | CONCLUSIONS

This work provides an example of how forest fire can positively affect groundwater resources quantity and quality in lowland coastal aquifers where forests grow on top of unconfined aquifers. Pre-fire and post-fire groundwater quality and level were monitored in the coastal pine forest of Ravenna (NE Italy) to highlight fire effects on groundwater salinization and recharge rates in a sandy aquifer affected by saltwater intrusion. In the study area, the complete removal of forest cover, due to a fire in 2012, caused a decrease in shallow groundwater salinity along with an increase in water table level, recharge rates, and freshwater lens thickness for the 4 years following the fire, compared with the pre-fire situation.

Although the fire had a positive effect in our case, each situation is specific and needs to be monitored and evaluated in terms of post-fire climate conditions, as well as soil type, plant species, soil moisture, etc. In order to define the hydrological effects of forest fire for water catchment purposes, it is necessary to monitor and follow the complete recovery of the vegetation in the years following the event, because regenerating vegetation may cause a bigger water uptake decreasing or nullifying the initial positive effects on freshwater availability (increase in freshwater lens thickness) and groundwater salinization.

A positive coastal zone management development is that stakeholders now accept that the forest position needs to be reconsidered and the native dune herbaceous vegetation should be promoted, a paradigm shift from mere conservation to active restoration.

ACKNOWLEDGMENTS

This research was partially included in the RIGED-RA project—"Restoration and management of coastal dunes in the Northern Adriatic Coast – Ravenna" (Scientific Coordinator, Prof. Giovanni Gabbianelli), funded by the Flaminia Foundation with the collaboration of the Ravenna Municipality (Italy) and the IGRG lab (Integrated Geosciences Research Group) of the University of Bologna. The authors would like to thank the Carabinieri for Biodiversity, Punta Marina Office, which allowed the access to the protected natural areas, objective of this study.

ORCID

Nicolas Greggio http://orcid.org/0000-0002-7323-134X

REFERENCES

Amorosi, A., Colalongo, M. L., Pasini, G., & Preti, D. (1999). Sedimentary response to late quaternary sea-level changes in the Romagna Coastal Plain (northern Italy). *Sedimentology*, *46*, 99–121.

Andersen, M. S., Nyvang, V., Jakobsen, R., & Postma, D. (2005). Geochemical processes and solute transport at the seawater/freshwater interface of a sandy aquifer. *Geochimica et Cosmochimica Acta*, *69*(16), 3979–3994.

Andreu, V., Imeson, A. C., & Rubio, J. L. (2001). Temporal changes in soil aggregates and water erosion after a wildfire in a Mediterranean pine forest. *Catena*, *44*, 69–84.

Antonellini, M., Allen, D. M., Mollema, P. N., Capo, D., & Greggio, N. (2015). Groundwater freshening following coastal progradation and land reclamation of the Po Plain, Italy. *Hydrogeology Journal*, *23*(5), 1009–1026.

Antonellini, M., Mollema, P., Giambastiani, B. M. S., Bishop, K., Caruso, L., Minchio, A., … Gabbianelli, G. (2008). Salt water intrusion in the coastal aquifer of the southern Po Plain, Italy. *Hydrogeology Journal*, *16*, 1541–1556.

Antonellini, M., & Mollema, P. N. (2010). Impact of groundwater salinity on vegetation species richness in the coastal pine forests and wetlands of Ravenna, Italy. *Ecological Engineering*, *36*(9), 1201–1211.

Arpae (2017). Atlante climatico dell'Emilia-Romagna 1961-2015. Regional Agency for Prevention, Environment and Energy of Emilia-Romagna, Italy. Retrieved from https://www.arpae.it/cms3/documenti/_cerca_doc/meteo/clima/Atlante_climatico_1961-2015.pdf

Ascoli, D., & Bovio, G. (2013). Prescribed burning in Italy: Issue, advances and challenges. *iForest*, *6*, 79–89.

Ascoli, D., Catalanotti, A., Valese, E., Cabiddu, S., Delogu, G., Driussi, M., … Bovio, G. (2012). Prescribed burning experiences in Italy: An integrated approach to prevent forest fires. *Forest@*, *9*(1), 20–38.

ASFD - Azienda di Stato per le Foreste Demaniali (1950). Piano di assestamento per il decennio 1950-1969.

ASFD - Azienda di Stato per le Foreste Demaniali (1960). *L'Azienda di Stato per le Foreste Demaniali. Volume I*. Roma: Soc. A.B.E.T.E.

Balugani, E., & Antonellini, M. (2011). Barometric pressure influence on water table fluctuations in coastal aquifers of partially enclosed seas: An example from the Adriatic coast, Italy. *Journal of Hydrology*, *400*(1), 176–186.

Barry, K. E., Vanderzalm, J. L., Miotlinski, K., & Dillon, P. J. (2017). Assessing the impact of recycled water quality and clogging on infiltration rates at a pioneering Soil Aquifer Treatment (SAT) site in Alice Springs, Northern Territory (NT), Australia. *Water*, *9*(3), 179.

Bart, R. R. (2016). A regional estimate of post-fire streamflow change in California. *Water Resources Research*, *52*, 1465–1478.

Batelaan, O., & De Smedt, F. (2007). GIS-based recharge estimation by coupling surface-subsurface water balances. *Journal of Hydrology*, *337*, 337–355.

Bekele, E. B., Donn, M. J., Barry, K. E., Vanderzalm, J. L., Kaksonen, A. H., Puzone, G. J., … Dillon, P. J. (2015). *Managed Aquifer Recharge and recycling options (MARRO): Understanding clogging processes and water quality impacts*. (p. 162). Australia: Australian Water Recycling Centre of Excellence. CSIRO Land and Water.

Benini, L. (1931). L'Ufficio di Amministrazione delle Foreste Demaniali di Ravenna e la sua attività dal 1922 al 1931. In *Il Comune di Ravenna* (pp. 9–11). Fasc. III-IV:.

Bouwer, H. (1989). The Bouwer and Rice slug test—An update. *Ground Water*, *27*(3), 304–309.

Bouwer, H., & Rice, R. C. (1976). A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. *Water Resources Research*, *12*, 423–428.

Callahan, T. J., Amatya, D. M., & Stone, P. A. (2016). Coastal forests and groundwater: Using case studies to understand the effects of drivers and stressors for resources management. *Sustainability*, *9*, 447.

Cantiani, P., Ferretti, F., Pignatti, G., Andreatta, G., & Nobili, G. (2008). Le pinete demaniali costiere di Ravenna e Ferrara: Analisi dei tipi forestali e strutturali per la scelta del trattamento selvicolturale nel processo di pianificazione. *Annuali C.R.A – Centro di Ricerca per la Selvicoltura*, *35*, 61–68.

Capaccioni, B., Didero, M., Paletta, C., & Didero, L. (2005). Saline intrusion and refreshening in a multilayer coastal aquifer in the Catania Plain (Sicily, Southern Italy): dynamics of degradation processes according to the hydrochemical characteristics of groundwater. *Journal of Hydrology*, *307*, 1–16.

Cawson, J. G., Nyman, P., Smith, H. G., Lane, P. N. J., & Sheridan, G. J. (2016). How soil temperatures during prescribed burning affect soil water repellency, infiltration and erosion. *Geoderma*, *278*, 16–22.

Cerdà, A., & Doerr, S. H. (2008). The effect of ash and needle cover on surface runoff and erosion in the immediate post-fire period. *Catena*, *74*(3), 256–263.

Cortecci, G., Dinelli, E., & Mussi, M. (2008). Isotopic composition and secondary evaporation effects on precipitation from the urban centre of Bologna, Italy. *Periodico di Mineralogia*, *77*, 55–63.

Cozzolino, D., Greggio, N., Antonellini, M., & Giambastiani, B. M. S. (2017). Natural and anthropogenic factors affecting freshwater lenses in coastal dunes of the Adriatic coast. *Journal of Hydrology*, *551*, 804–818. https://doi.org/10.1016/j.jhydrol.2017.04.039

Davis, G. B., & Barber, C. (1994). Monitoring and sampling methods. In *CSIRO groundwater school notes*, Chapter 19, 12th Groundwater School, July 1993. Adelaide (Australia).

DeBano, L. F. (1981). *Water repellent soils: A state-of-art. General Technical Report PSW-46*. Berkeley: Pacific Southwest Forest and Range Experiment Station, United States Department of Agriculture, Forest Service.

DeBano, L. F. (2000). The role of fire and soil heating on water repellency in wildland environments: A review. *Journal of Hydrology*, *231-232*, 195–206.

Dennison, P. E., Charoensiri, K., Roberts, D. A., Peterson, S. H., & Green, R. O. (2006). Wildfire temperature and land cover modelling using hyperspectral data. *Remote Sensing of Environment*, *100*, 212–222.

Dillon, P. (2005). Future management of aquifer recharge. *Hydrogeology Journal*, *13*, 313–316.

Dillon, P., Pavelic, P., Page, D., Beringen, H., & Ward, J. (2009). *Managed aquifer recharge: An introduction* (p. 77). National Water Commission, CSIRO.

Dillon, P., Pavelic, P., Toze, S., Ragusa, S., Wright, M., Peter, P., … Rinck-Pfeiffer, S. (1999). Storing recycled water in an aquifer: Benefits and risks. *Water - Australian Water & Wasterwater Association Journal*, *26*(5), 21–29.

Dingman, L. (2008). *Physical hydrology* (Second ed.). (p. 656). Waveland Pr. Inc.

Doerr, S. H., Shakesby, R. A., & Walsh, R. P. D. (1996). Soil hydrophobicity variations with depth and particle size fraction in burned and unburned *Eucalyptus globulus* and *Pinus pinaster* forest terrain in the Águeda Basin, Portugal. *Catena*, *27*, 25–47.

Doerr, S. H., Shakesby, R. A., & Walsh, R. P. D. (1998). Spatial variability of soil hydrophobicity in fire-prone Eucalyptus and Pine forests, Portugal. *Soil Science*, *163*(4), 313–324.

Doerr, S. H., Shakesby, R. A., & Walsh, R. P. D. (2000). Soil water repellency: Its causes, characteristics and hydro-geomorphological significance. *Earth-Science Reviews*, *51*, 33–65.

Eben, B. A., & Moody, J. A. (2016). Synthesis of soil-hydraulic properties and infiltration timescales in wildfire-affected soils. *Hydrological Processes*, *31*, 324–340.

El Saliby, I., Okour, Y., Shon, H. K., Kandasamy, J., & Kim, I. S. (2009). Desalination plants in Australia, review and facts. *Desalination*, *247*(1–3), 1–14.

Feikema, P. M., Sherwin, C. B., & Lane, P.N:J. (2013). Influence of climate, fire severity and forest mortality on predictions of long term streamflow: potential effct of the 2009 wildfire on Melbourne's water supply catchments. *Journal of Hydrology*, *488*, 1–16.

Fernandes, P. M., Davies, G. M., Ascoli, D., Fernàndez, C., Moreira, F., Rigolot, E., … Molina, D. (2013). Prescribed burning in southern Europe: Developing fire management in a dynamic landscape. *Frontiers in Ecology and the Environment*, *11*, e4–e14. https://doi.org/10.1890/120298

Fernandez, C., Vega, J. A., Fonturbel, T., Jimenez, E., & Perez, J. R. (2008). Immediate effects of prescribed burning, chopping and clearing on runoff, infiltration and erosion in a shrublands area in Galicia (NW Spain). *Land Degradation & Development*, *19*(5), 502–515.

Fetter, C. W. (2001). *Applied hydrogeology* (Fourth ed.). (p. 598). Prentice Hall.

Fox, P., Houston, S., Westerhoff, P., Drewes, J. E., Nellor, M., Yanko, W., … Reinhard, M. (2001). *Soil Aquifer Treatment for sustainable water reuse*. Denver, CO: American Water Works Association Research Foundation.

Giambastiani, B. M. S., Antonellini, M., Oude Essink, G. H. P., & Stuurman, R. J. (2007). Saltwater intrusion in the unconfined costal aquifer of Ravenna (Italy): A numerical model. *Journal of Hydrology*, *340*, 91–104.

Giambastiani, B. M. S., Colombani, N., & Mastrocicco, M. (2012). Multilevel characterization of vertical hydraulic gradients, permeability, temperature and salinity in shallow coastal aquifers with low pressure packers. In *Proc. 22nd SWIM 2012*. Armação dos Búzios: Brasil. 17–22 June 2012; pp. 140–143.

Giambastiani, B. M. S., Colombani, N., Mastrocicco, M., & Fidelibus, M. D. (2013). Characterization of the lowland coastal aquifer of Comacchio (Ferrara, Italy): Hydrology, hydrochemistry and evolution of the system. *Journal of Hydrology*, *501*, 35–44.

Giovannini, G., & Lucchesi, S. (1997). Modifications induced in the soil physio-chemical parameters by experimental fires at different intensities. *Soil Science*, *162*, 479–486.

Greggio, N., Giambastiani, B. M. S., & Antonellini, M. (2017). Infiltration/irrigation trench for sustainable coastal drainage management: Emilia-Romagna (Italy). *Environmental Engineering and Management Journal*. under Review

Hugenholtz, C. H., & Koenig, D. (2014). Sand dune stabilization reduces infiltration and soil moisture: A case study from the northern Great Plains. *Ecohydrology*, *7*, 1135–1146.

IPCC (2013). Summary for policymakers. In T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, et al. (Eds.), *Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Lane, P. N. J., Croke, J., & Dignan, P. (2004). Runoff generation from logged and burnt convergent hillslopes: Rainfall simulation and modelling. *Hydrological Processes*, *18*, 879–892.

Langelier, W. F., & Ludwig, H. F. (1942). Graphical methods for indicating the mineral character of natural waters. *Journal American Water Works Association*, *34*(3), 335–352.

Longhi, G. (1969). Foresta "Pineta di Ravenna". Piano di Assestamento esecutivo per il decennio 1970–1979. Azienda di Stato per le Foreste Demaniali.

Marcelli, T., Santoni, P. A., Simeoni, A., Leoni, E., & Porterie, B. (2004). Fire spread across pine need fuel beds: Characterization of temperature and velocity distributions within the fire plume. *International Journal of Wildland Fire*, *13*(1), 37–48.

March, H., Saurí, D., & Rico-Amorós, A. M. (2014). The end of scarcity? Water desalination as the new cornucopia for Mediterranean Spain. *Journal of Hydrology*, *519*(Part C), 2642–2651.

Marchesini, L., Amorosi, A., Cibin, U., Zuffa, G., Spadafora, E., & Preti, D. (2000). Sand composition and sedimentary evolution of a late quaternary depositional sequence, Northwestern Adriatic Coast, Italy. *Journal of Sedimentary Research*, *70*(4), 829–838.

Marconi, V., Antonellini, M., Balugani, E., & Dinelli, E. (2011). Hydrogeochemical characterization of small coastal wetlands and forests in the Southern Po plain (Northern Italy). *Ecohydrology*, *4*(4), 597–607.

Mastrocicco, M., Giambastiani, B. M. S., Severi, P., & Colombani, N. (2012). The importance of data acquisition techniques in saltwater intrusion monitoring. *Water Resources Management*, *26*(10), 2851–2866.

Mollema, P., & Antonellini, M. (2013). Seasonal variation in natural recharge of coastal aquifers. *Hydrogeology Journal*, *21*, 787–797.

Mollema, P., Antonellini, M., Gabbianelli, G., Laghi, M., Marconi, V., & Minchio, A. (2012). Climate and water budget change of a Mediterranean coastal watershed, Ravenna, Italy. *Environmental Earth Sciences*, *65*(1), 257–276.

Mollema, P. N., Antonellini, M., Dinelli, E., Gabbianelli, G., Greggio, N., & Stuyfzand, P. J. (2013b). Hydrochemical and physical processes influencing salinization and freshening in Mediterranean low-lying coastal environments. *Applied Geochemistry*, *34*, 207–221.

Mollema, P. N., Antonellini, M., Gabbianelli, G., & Galloni, E. (2013a). Water budget management of a coastal pine forest in a Mediterranean catchment (Marina Romea, Ravenna, Italy). *Environmental Earth Sciences*, *68*(6), 1707–1721.

Moody, J. A., Martin, D. A., Haire, S. L., & Kinner, D. A. (2008). Linking runoff response to burn severity after a wildfire. *Hydrological Processes*, *22*, 2063–2074.

Mullen, R. M., Springer, A. E., & Kolb, T. E. (2006). Complex effects of prescribed fire on restoring the soil water content in a high-elevation riparian meadow, Arizona. *Restoration Ecology*, *14*(2), 242–250.

Naccarato, F. (1971). La pineta demaniale ravennate. *Bollettino Economico Della C.C.I.A.A. di Ravenna*, *6*, 3–6.

Nordstrom, K. F., Armaroli, C., Jacksin, N. L., & Ciavola, P. (2015). Opportunities and constraints for managed retreat on exposed sandy shores: Examples from Emilia-Romagna, Italy. *Ocean and Coastal Management*, *104*, 11–21.

Obrist, D., DeLucia, E. H., & Arnone, J. A. (2003). Consequences of wildfire on ecosystem CO_2 and water vapour fluxes in the Great Basin. *Global Change Biology*, *9*(4), 563–574.

Oude Essink, G. H. P. (2001). Improving fresh groundwater supply—Problems and solutions. *Ocean and Coastal Management*, *44*, 429–449.

Page, D., Bekele, E., Vanderzalm, J., & Sidhu, J. (2018). Managed Aquifer Recharge (MAR) in sustainable urban water management. *Watermark*, *10*, 239.

Page, D. W., Peeters, L., Vanderzalm, J., Barry, K., & Gonzalez, D. (2017). Effect of aquifer storage and recovery (ASR) on recovered stormwater quality variability. *Water Research*, *117*, 1–8.

Penman, T. D., Binns, D. L., Shiels, R. J., Allen, R. M., & Penman, S. H. (2011). Hidden effects of forest management practises: Responses of a soil stored seed bank to logging and repeated prescribed fire. *Austral Ecology*, *36*, 571–580.

Perini, L., Calabrese, L., Luciani, P., Olivieri, M., Galassi, G., & Spada, G. (2017). Sea-level rise along the Emilia-Romagna coast (Northern Italy) in 2100: Scenarios and impacts. *Natural Hazards and Earth System Sciences*, *17*, 2271–2287.

Ritsema, C. J., & Dekker, L. W. (1995). Distribution flow: A general process in the top layer of water repellent soils. *Water Resources Research*, *31*(5), 1187–1200.

Rodríguez-Alleres, M., Varela, M. E., & Benito, E. (2012). Natural severity of water repellency in pine forest soils from NW Spain and influence of wildfire severity on its persistence. *Geoderma*, *191*, 125–131.

Rulli, M. C., Offeddu, L., & Santini, M. (2013). Modeling post-fire water erosion mitigation strategies. *Hydrology and Earth System Sciences*, *17*, 2323–2337.

Rulli, M. C., Spada, M., Bozzi, S., Bocchiola, D., & Rosso, R. (2006). Rainfall simulations on a fire disturbed Mediterranean area. *Journal of Hydrology*, *372*(3–4), 323–338.

Savage, S. M. (1974). Mechanism of fire-induced water repellency in soil. *Soil Science Society of America Proceedings*, *38*, 652–657.

Sburlino, G., Buffa, G., Filesi, L., Gamper, U., & Ghirelli, L. (2013). Phytocoenotic diversity of the N-Adriatic coastal sand dunes—The herbaceous communities of the fixed dunes and the vegetation of the interdunal wetlands. *Plant Sociology*, *50*(2), 57–77.

Scarelli, F., Sistilli, F., Fabbri, S., Cantelli, L., Barboza, E. G., & Gabbianelli, G. (2017). Seasonal dune and beach monitoring using photogrammetry from UAV surveys to apply in the ICZM on the Ravenna coast (Emilia-Romagna, Italy). *Remote Sensing Applications: Society and Environment*, *7*, 27–39.

Shakesby, R. A., & Doerr, S. H. (2006). Wildfire as a hydrological and geomorphological agent. *Earth-Science Reviewers*, *74*, 269–307.

Sharma, S. K., & Kennedy, M. D. (2017). Soil aquifer treatment for wastewater treatment and reuse. *International Biodeterioration & Biodegradation*, *119*, 671–677.

Silberstein, R. P., Dawes, W. R., Bastow, T. P., Byrne, J., & Smart, N. F. (2013). Evaluation of changes in post-fire recharge under native woodland using hydrological measurements, modelling and remote sensing. *Journal of Hydrology*, *489*, 1–15.

Silva, J. S., Rego, F. C., & Mazzoleni, S. (2006). Soil water dynamics after fire in a Portuguese shrubland. *International Journal of Wildland Fire*, *15*, 99–111.

Singh, G., Schoonover, J. E., Monroe, K. S., Williard, K. W. J., & Ruffner, C. M. (2017). Prescribed burning and erosion potential in mixed hardwood forests of Southern Illinois. *Forests*, *8*(4), 112. https://doi.org/10.3390/f8040112

Sprenger, C., Hartog, N., Hernàndez, M., Vilanova, E., Grützmacher, G., Scheibler, F., & Hannappel, S. (2017). Inventory of managed aquifer recharge sites in Europe: Historical development, current situation and perspectives. *Hydrogeology Journal*, *25*, 1–14. https://doi.org/10.1007/s10040-017-1554-8

Stoof, C. R., Moore, D., Ritsema, C. J., & Dekker, L. W. (2011). Natural and fire-induced soil water repellency in a Portuguese shrubland. *Soil Science Society of America Journal*, *75*(6), 2283–2295.

Stoof, C. R., Slingerland, E. C., Mol, W., van der Berg, J., Vermeulen, P. J., Ferreira, A. J. D., … Steenhuis, T. S. (2014). Preferential flow as a potential mechanism for fire-induced increase in streamflow. *Water Resources Research*, *50*(2), 1840–1845.

Stoof, C. R., Vervoort, R. W., Iwema, J., van der Elsen, E., Ferreira, A. J. D., & Ritsema, C. J. (2012). Hydrological response of a small catchment burned by experimental fire. *Hydrology and Earth System Sciences*, *16*, 267–285.

Stuyfzand, P. J. (2008). Base exchange indices as indicators of salinization or freshening of (coastal) aquifers. In Proceedings: 20th salt water intrusion meeting, Naples, FL, USA, 23–27 June 2008.

Tanaka, T., Taniguchi, M., & Tsujimura, M. (1996). Significance of stemflow in groundwater recharge. 2: A cylindrical infiltration model for evaluating the stemflow contribution to groundwater recharge. *Hydrological Processes*, *10*, 81–88.

Taniguchi, M., Tsujimura, M., & Tanaka, T. (1996). Significance of stemflow in groundwater recharge. 1: Evaluation of the stemflow contribution to recharge using a mass balance approach. *Hydrological Processes*, *10*, 71–80.

Taramelli, A., Di Matteo, L., Ciavola, P., Guadagnano, F., & Tolomei, C. (2015). Temporal evolution of patterns and processes related to subsidence of the coastal area surrounding the Bevano River mouth (Northern Adriatic)–Italy. *Ocean and Coastal Management*, *108*, 74–88.

Teatini, P., Ferronato, M., Gambolati, G., Bertoni, W., & Gonella, M. (2005). A century of land subsidence in Ravenna, Italy. *Environmental Geology*, *47*(6), 831–846.

Teobaldelli, M., Mencucci, M., & Piussi, P. (2004). Water table salinity, rainfall and water use by umbrella pine trees (Pinus Pinea L.). *Plant Ecology*, *171*(1–2), 23–33.

Tomozeiu, R., Cacciamani, C., Pavan, V., Morgillo, A., & Busuioc, A. (2007). Climate change scenarios for surface temperature in Emilia-Romagna (Italy) obtained using statistical downscaling models. *Theoretical and Applied Climatology*, *90*, 25–47.

Vandenbohede, A., Mollema, P. M., Greggio, N., & Antonellini, M. (2014). Seasonal dynamic of a shallow freshwater lens due to irrigation in the coastal plain of Ravenna, Italy. *Hydrogeology Journal*, *22*, 893–909.

Vermeire, L. T., Mitchell, R. B., Fuhlendorf, S. D., & Wester, D. B. (2004). Selective control of rangeland grasshoppers with prescribed fire. *Journal of Range Management*, *57*(1), 29–33.

Villani, G., Tomei, F., Tomozeiu, R., & Marletto, V. (2011). Climatic scenarios and their impacts on irrigated agriculture in Emilia-Romagna, Italy. *Italian Journal of Agrometeorology*, *1*, 5–16.

Vogwill, R. I. J., McHugh, S. L., O'Boy, C. A., & Yu, X. (2008). PRAMS scenario modelling for water management of the Gnangara Groundwater Mound. HG21, Western Australia Department of Water.

Weber, K., & Steward, M. (2004). A critical analysis of the Cumulative Rainfall Departure concept. *Ground Water*, *42*(6), 935–938.

Wotton, B. M., Gould, J. S., McCaw, W. L., Cheney, N. P., & Stephen, W. T. (2011). Flame temperature and residence time of firesin dry eucalypt forest. *International Journal of Wildland Fire*, *21*(3), 270–281.

Yechieli, Y., Shalev, E., Wollman, S., Kiro, Y., & Kafri, U. (2010). Response of the Mediterranean and Dead Sea coastal aquifers to sea level variations. *Water Resources Research*, *46*(12). https://doi.org/10.1029/2009WR008708

Yesertener, C. (2005). Impacts of climate, land and water use on declining groundwater levels in the Gnangara groundwater Mound, Perth, Australia. *Australian Journal of Water Resources*, *8*(2), 143–152.

Zhou, Y., Zhang, Y., Vaze, J., Lane, P., & Xu, S. (2015). Impact of bushfire and climate variability on streamflow from forested catchments in southern Australia. *Hydrological Sciences Journal*, *60*(7–8), 1340–1360.

How to cite this article: Giambastiani BMS, Greggio N, Nobili G, Dinelli E, Antonellini M. Forest fire effects on groundwater in a coastal aquifer (Ravenna, Italy). *Hydrological Processes*. 2018;1–13. https://doi.org/10.1002/hyp.13165

DISCUSSION

In Italy, first nature's protection legislation concerned natural landscapes and monuments. At first, establish of protected areas was linked to satisfy beautiful landscapes conservation needs and hunting estates (L. n. 778/1922; L. n. 1497/1939). Subsequently, were introduced constrains regarding hydrogeological protection to safeguard unstable mountain slopes by excessive forest cuts by large part of national population engaged in agricultural activity (L. n. 3267/1923). Only with the advent of environmentalism feel, regulation concerned landscape protection as a whole was introduced (L. 431/1985) together the established of a first system of protected areas (L. 394/1991). More recently, local concept was exceeded by a web protection level (Nature 2000 sites, Dir. 1992/43/CEE and 2009/147/CEE).

Thus, historic dynamic of national reference legislation has been strictly linked to national historic, socio-economic and cultural contest in which has been issued and developed. Similarly, planning and management choices in protected areas express a cultural process which aims to keep in good conservation conditions habitats and species respect changes in environmental matrix. The drafting of management documents develops over time, based on current legislation and available techniques to achieve management goals of that time.

Starting from the 70s of the last century the common feeling it was believed that it was sufficient to subtract some portions of the territory in the presence of man so that they would be preserved indefinitely.

In general, the conservation of Biodiversity and the environment was not considered necessary at the time. After a phase during which only some individual species have been identified as worthy of legal protection, today biodiversity as a whole is considered at risk and therefore in need of protection interventions.

Like other human activities, nowadays conservation activities are considered necessary. Questions about how to manage the world's ecosystems remain urgent, especially in light of growing concerns about human land-use pressures and global climate change (Batavia and Nelson, 2016; Berkes and Folke, 1998; Ludwig, 2001). In fact, the magnitude and rate of the changes taking place impose the need to manage the protected natural areas in parallel with the escalation of the management problems and the negative interferences that must be highlighted on the protected environment.

Management process is finalized: natural resource management aspires to achieve ecosystem health. If ecosystems carry out their task efficiently, environment is able to support human well-being with a number of goods and services. These goods and services are named as ecosystem services (hereafter ESs) (MA, 2005). Environmental processes get their value by the benefit people obtain from it (Lautenbach et al., 2011; Diaz et al., 2006).

During historic time, changes due for economic, social, demographic and technological reasons are occurred. Since always, anthropization have altered Delta Po species and habitats, ecosystems and landscapes, for hunting, fishing, reclamation and agriculture. In general, land use changes are seen to be the most important driving force with serious impacts on biodiversity and ecosystem services (MA, 2005). ESs are strictly linked with landscape history. In dynamic landscapes ESs too evolve over time by myriad processes which link nature to society.

Generally human activities follow forward coasts in progress phases (see Schwartz, 2018). When human society recognized SEs like a benefit, this one became an opportunity to seize like establishment of protected areas, implant of pinewoods, shellfish farming and so on.

Ecosystems are subject to natural fluctuations and trends, which can alter their functionality and their capacity to support ecosystem services. As well as the environment changes, SEs too change over

time. The consequences in the changes of key ESs have not been adequately considered so far (Bürgi et. al., 2015). Both environmental and ESs modification rates create opportunities for the human socio-economic context and the rapidity of the planning process do not follow a linear and constant trend over time. Temporal dynamics of ESs must be considered with interest at management level too. Anticipation of ESs changes is an important tool to guide governance respect environmental changes (Geneletti, 2013; Price et al., 2012).
For inference on future ESs is valuable an analysis on past management instruments. An historical analysis on past management criteria in the study area is reported in the tabs below, based on respective management plans (AA.VV., 1984; A.S.F.D., 1960; Benini, 1931; Cantiani et al., 2008; Longhi, 1969; Patrone, 1948; Zangheri, 1936).

Tab. 1a - Historic dynamic of management and ESs

Beaches and coastal dunes	1900	1930	1950	1970	1980	2000 ....
Socio-economic contest and functions expectations	Marine relics, unproductive and uncultivated areas, forestation for agricolture protection and provide work opportunity	Gradual increase of aesthetic functions, first tourist interest in front of country, bathers well-being		Enhance tourist attractions, fast increase of touristic value and related structures (balnear establishments, campings, hotels), build new housing settlement, natural protected areas establishment		High economic interest for tourism, Biodiversity conservation in protected areas, ecological corridor, recognized erosion protection and protection respect salinity enhance in water table, increase in environmental sensitivity, conscious tourism
ESs realized	regulation	regulation		regulation,		regulation, cultural
ESs recognized	-	cultural (aesthetics)		provision, cultural, (recreational tourism),		regulating, provision, cultural (recreational and educational tourism)
Management interventions	Progessive use			Strong enhance in uses linked on tourism		Clear distinctions between touristic and protected areas, protection and monitoring biodiversity in protected areas, sand dunes restoration, nourishments, regulating access to protect nesting period

Normative			
	1905 "Rava's law"	L. 431/1985	EU "Habitat" Dir.92/43 CE
	L. n. 1497/1939 beautiful landscapes conservation	L. 394/1991	

Tab. 1b

Coastal pinewoods	1900	1930	1950	1960	1970	2000 ....
Socio-economic contest and functions expectations	Protect agriculture and behind reclamation lands from marine spray, soil preparative for forestation, provide work opportunity for locals, replace historic Ravenna's pinewoods	Increase in products provisioning (pine nuts and timber), landscaping and aesthetics, pinewood is a collective good with economic value which provide work			Enhance tourist attractions, natural protected areas establishment, decrease production expectations and increase in environmental sensitivity, pinewood indirectly increase tourism attractiveness as well-being for people	Biodiversity conservation, place of touristic attractiveness, ecological corridor increase in sensitivity for biodiversity conservation
ESs realized	Cultural (aesthetic), regulating	Cultural (aesthetic), regulating, provision (products)			Cultural (aesthetic), regulating, provision (indirect)	cultural (aesthetic and educational), regulating, provision (indirest)
ESs recognized	Cultural (aesthetic)	Provision (products), cultural (aesthetic)			Provision (indirect), cultural (aesthetic and educational), regulating	cultural (aesthetic and educational), regulating
Management interventions	Planting, treatments, thinnings		Thinnings, decidous implant			Release dead wood, monitoring biodiversity

Tab. 1c

Lagoons	1900	1930	1950	1970	1980	2000 ……………
Socio-economic contest and functions expectations	Subsistence economy (hunting, fishing); reclamation in favor of agriculture Areas of expansion of the tides used for extensive fish farming			Increase in the functions of tourist attractiveness, establishment of protected areas in natural areas; clame farming increase; Increase in production activity; transition between traditional economy and intensive production activities; attention is paid to aspects of environmental conservation		Conservation of biodiversity in protected areas; strong tourist vocation; shellfish-intensive culture; high economic interest (shellfish farming) and increase in environmental sensitivity and for the conservation of biodiversity; conscious tourism
ESs realized	Provisioning, regulating			Provisioning, regulating		Provisioning, regulating, cultural
ESs ricognized	Provisioning			Provisioning, cultural (birdwatching)		Provisioning, regulating, cultural (tourism, educational, habitat conservation, birdwatching)
Management interventions	Reclamations			Hydraulic circulation is favored in favor of intensive shellfish farming		Recognition of importance for territorial protection; protection and monitoring of biodiversity in protected areas; environmental restoration; access regulation

Normative			
1905 "Rava's law"	L. n. 1497/1939 beautiful landscapes conservation	L. 431/1985 L. 394/1991	EU "Habitat" Dir.92/43 CE

Tab. 1d

BOSCO DELLA MESOLA	1900	1930	1950	1970	1980	2000 ……….
Socio-economic contest and functions expectations	Productive functions (hunting, timber, pasture, mushrooms)			Increase in the functions of tourist attractiveness; decrease productive functions		Biodiversity conservation; tourism attraction; local loss of centrality of the forest; sensitivity of a few for the conservation of biodiversity; attention to habitat conservation
	provide work opportunity for locals; the forest is a collective good that produces goods and labor				environmental sensitivity born, increase in the functions of tourist attractiveness, establishment of protected areas; expectations in tourism attraction	
ESs realized	Cultural, provisioning (hunting, timber, pasture, mushrooms funghi), regulating				Cultural, tourism, provisioning (timber), aesthetic, regulating	Cultural, conscious tourism, provisioning (timber), aesthetic, educational, regulating
ESs recognized	Provisioning (hunting, timber, pasture)				Provisioning (timber), aesthetic, tourism,	Provisioning (timber), aesthetic, conscious tourism
Management interventions	coppicing			High forest start		Release dead wood, monitoring biodiversity; contrast to the alien species (fallow deer); microhabitat creation; envisage again coppice

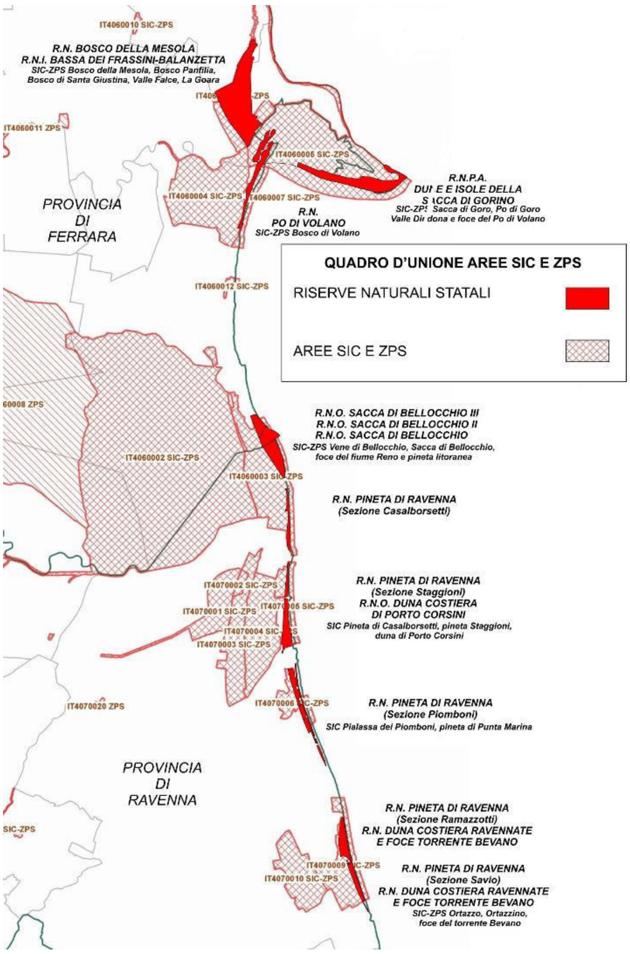
Millennium Ecosystem Assessment (MA, 2005) make prominent the idea that human well-being depends on ecosystems, and that such linkages can be tracked and framed through the notion of ecosystem services. The Common International Classification for Ecosystem Services (CICES), proposed by the European Environment Agency, is based on and influenced by the cascade framework proposed by Haines-Young & Potschinin 2010 (Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2016). The cascade framework links natural systems to elements of human well-being, following a pattern similar to a production chain: from ecological structures and processes generated by ecosystems, to the services and benefits eventually derived by humans. The advantage of this framework is to effectively communicate societal dependence on ecosystems (La Notte et al., 2017).
Clearly is an anthropocentric concept: ESs contains an utilitarian approach since late 1970 (Gómez-Baggethun et al., 2010): without a benefit there is no service. The concept of ecosystem services put

in relation ecology and economy. In some cases, benefits are tangible natural resources derived from provisioning or some regulating services. Benefits, however, can also be intangible (opportunities) (La Notte et al., 2017). Over time, ESs produce for society occasions and opportunities offered by nature in terms of provisioning, regulating, supporting and cultural goods.
While regulating ESs operate independently of being recognized by people, provisioning and cultural services need a specific demand to exist (Bürgi et. al., 2015).
At present whole Emilia-Romagna cost line except only three places is subjected to marine erosion (Aguzzi et al., 2016).
To know if the ESs carried out by coastal habitats with respect to the problems of erosion and subsidence are now recognized (and therefore required), an interview was addressed to technicians and public administrators of environmental management, as well as freelancers, voluntary ecological guards, tour operators, teachers with an interest in the management of the area (N = 49).
Among the key macro-environments considered (beaches and dunes, lagoons and brackish wetlands, wooded areas) the most relative importance value for the functions performed was attributed to the beach and dune systems in all residual protected natural areas along the coast of Emilia-Romagna: 1) Sacca di Goro and Bosco della Mesola; 2) Sacca di Bellocchio and the Valleys of Comacchio behind; 3) coastal pine forests and Pialasse of Ravenna; 4) mouth of the Bevano river, Ortazzo-Ortazino and Pineta di Classe. In the Ravenna area, where the beach environment has been distinguished from the dune, the latter has been identified as the most important. Likewise, coastal brackish wetlands were considered priority except in the Ravenna area. The lowest values of importance have been attributed to productive environments or more distant from the coast (Sacca di Goro, private fishing valleys in the area of Bellocchio, Pialasse ravennati). The innermost areas obtained an intermediate score only when very representative (Bosco della Mesola, Valli di Comacchio).
At the level of territorial management, tab. 2 shows the relative importance values attributed to each macro-habitat relative to the capacity to: 1) conservation of Biodiversity; 2) more at risk for the effects of climate changes; 3) support traditional economic activities; 4) to transmit cultural value; 5) have greater need for investments to be managed/conserved and 6) resist environmental changes (erosion and subsidence.

Tab. 2 Values attributed to macro-habitats considered respect some aspects of management importance [N = 49]

	Forests-pinewoods	Lagoons-wetlands	Beaches-dunes
Conservation of coastal Biodiversity	4	22	22
More threatened by climate changes	5	9	31
Support to traditional economy	8	29	12
Cultural value	11	26	9
It needs more funds to be managed	8	17	23
Resistance to modifications	32	13	2

Tab. 3 Adaptation to environmental changes in the 4 coastal natural areas considered in the E-R Region.

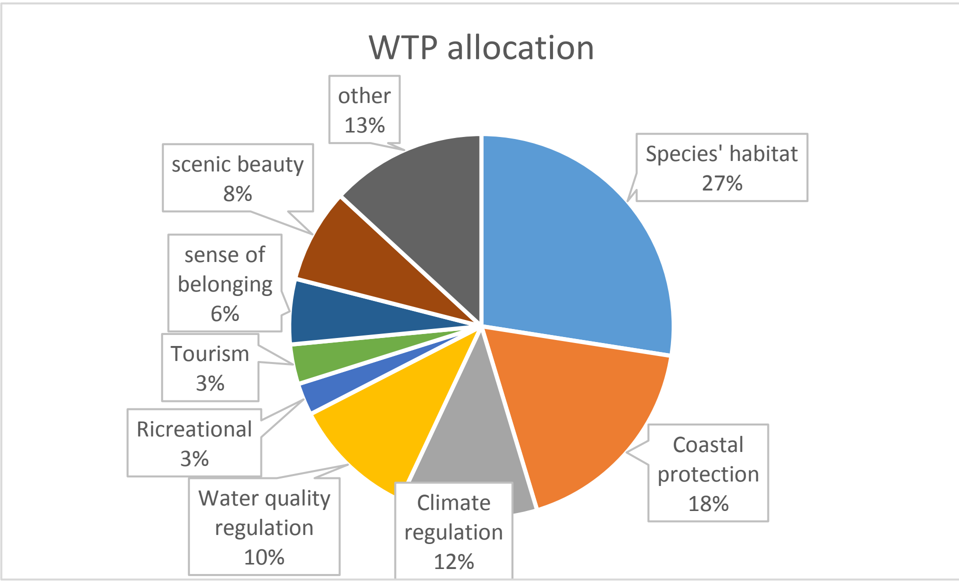


Area	Key macro-habitat	protection and adaptation interventions
Goro lagoon	Scanno/dunes;	brushes in wooden poles to intercept sediments and trigger the formation of new benches, dredging of the seat-minds to keep the main lagoon mouth open, cutting of the bench
	productive lagoon	Improvement of hydraulic circulation and nourishment of fishing supplies
	Reeds/salt marshes	Habitat restoration
	Mesola Forest	Conservation, vegetation adaptation
Bellocchio	Beach/dunes;	experimental restoration
	Salicornia bed	-
	Private fish farm	hydraulic maintenance, embankment strengthening
	Comacchio valley/infrastructure	Improvement of hydraulic circulation, embankment construction
Ravenna	Beach	nourishments
	dunes	Restoration and renaturation
	Coastal pinewood	Adaptation/conservation
	Pialassa (lagoon)	Creation of new sandy embankments
Bevano	Beach/dunes;	Nourishments, experimental restoration
	Coastal pinewood	Adaptation/conservation
	Brackish wetland	-
	Historic pineforest	Adaptation/conservation

Defense and adaptation interventions (Aguzzi et al., 2016) were carried out in all the areas considered (tab. 3), with methods increasingly respectful of the coastal dynamics of the sediments (nourishment, realization of filtering brushes in wooden poles, naturalistic/experimental reconstitution of the dunes). Therefore, the good news is that adaptation has already begun. Among technicians, the awareness of the changes taking place, the need to protect or reconstitute certain environments already exists, for the functions they perform.

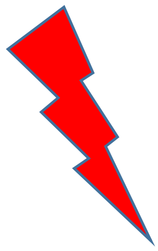
Like any management action, adaptation interventions have direct and indirect effects on biodiversity. The theme is therefore central: is it Biodiversity considered a value in itself to protect? Is the recognition of the importance of Biodiversity in itself capable of initiating and/or promoting activities for its long-term conservation?

Exactly like ESs, biodiversity is now widely used among scientists and policy makers to highlight its importance in sustaining human livelihoods (Convention on Biological Diversity, 2010; Maes et al., 2016).
In nature conservation, economic valuation of biodiversity and ecosystem services is an essential element for making conservation efforts financially sustainable, as it stimulates the perceived need for investing in conservation, be it through the establishment and management of protected areas (Costanza et al., 1997; Mertz et al., 2007).
Moreover, it is now widely recognized that nature conservation and related management strategies do not necessarily pose a trade-off between the "environment" and "development", but rather that investments in conservation, restoration and sustainable ecosystem use generate substantial ecological, social and economic benefits (Gantioler et al., 2010; Spangenberg and Settele, 2010).
A useful tool for the calculation of the economic effects (expenditures) of landscape management is economic valuation (Sejak et al., 2010; Spangenberg and Settele, 2010), i.e., the amount that society is ready to pay for the maintenance and restoration of valuable ecosystems and species (the "willingness-to-pay" approach).
For example, among those interviewed by us, those who think that the main goal of protected areas should be conservation is willing to pay more than those who think that the main use should be tourism and/or other functions. Moreover, the willingness to pay is directed more towards the conservation of habitats for the species compared to other Services.



Another good news is therefore that there seems to be a widespread sensitivity towards the importance of biodiversity conservation.
Indeed, MA (2005) emphasizes the linkages between ecosystems and human well-being, it recognizes the intrinsic value of species and ecosystems. Intrinsic value is the value of something in and for itself, irrespective of its utility for someone else.
On the one hand, biodiversity is an ecosystem property, and is a prerequisite for various ecosystem services. On the other hand, biodiversity can be the result of ecosystem services. Species diversity influences ecosystem properties and functioning, and *vice versa*.
In the first vision, biodiversity plays its regulatory role and it can be classified as a regulation service (Bastian, 2013), even without an actual request from the society. The existing biodiversity can also provide material (provisioning) or immaterial (cultural) services. Management activities are finalized to preserve existing biodiversity.

With the latter, an optimal environmental management maintain the capacity or the service to have suitable living conditions for plants and animals. Biodiversity is the goal to be achieved, as a benefit to be preserved and as a value to be obtained in perspective. There is a request for biodiversity from the current human society. With this new approach, a recognized service is pursued similarly like provisioning and cultural services: the request arises to conserve or encourage biodiversity because in this way there is still a material (provisioning or regulating services) or immaterial (cultural) advantage for society. When human society recognizes the importance of biodiversity, another vision acquire interest at a management level: if there is no biodiversity there is no benefit.
In fact, genetic resources represent an exception among provisioning services, since we consider them as information. While regulating services are based on interactions among biotic and abiotic elements of the ecosystems, cultural services derive from information. This information might influence humans, for example triggering nature conservation (La Notte et al., 2017).
In the cascade framework that constitutes the theoretical background of CICES, the value of the ecological system is intrinsic, and the approach is holistic, bio-centric and positivist. The ecosystem services narrative is part of the human system whose value is utilitarian, and its approach reductionist and human-centered (La Notte et al., 2017).
On a management level, the difference lies in the duration of the time of application of the measures or temporal scope and effectiveness of the management interventions: to conserve or favor in the short term, but if the environmental matrix is subject to profound and rapid modifications, conservation loses effectiveness and it is necessary to look more distant. Both management opportunities must be maintained but only the second vision allows you to project ourself sufficiently ahead, thinking, planning and finally creating the opportunity for future biodiversity.

Biodiversity:	ES	Management level	Short period management scope	Habitats radical change	Long-period governance
Ecosystem property/pre-requisite (Species diversity influences ecosystem properties and functioning)	- regulatory role (regulation service); - Provision - Cultural	Direct management (conservation) Indirect management (improve habitats)	Species conservation habitat improvement		failure to reach goal
Goal of environment management (maintains capacity or the service…to have suitable living conditions for plants and animals)	Cultural	Indirect management	Think, plan and create opportunity creating new habitats		

In reality, the conservation of endangered species is a problem that does not have a definitive solution, on the contrary, there will likely never be a final resolution (Ludwig, 2001).
The approaches to the conservation of the individual threatened species can be different and take into account the contribution that the species provide to the functioning of an ecosystem, to the genetic heritage they represent - which can guarantee adaptation to the changes taking place - or try to protect the hotspots of biodiversity so that as many species as possible can be safeguarded (see Nijhuis, 2102).
All species would have equal dignity to be preserved, but management costs spent to maintain an ecosystem or a population of wild plants or animals cannot be equated with the value they represent, or with the value of ecosystem services they provide. Moreover it is not possible to predict the effects that the loss of a single species or a population and/or ecosystem service might have.
Obviously, the precautionary approach is required, and ecosystems should be maintained intact as far as possible, to ensure continued service provision in the face of changing environmental conditions and biotic interactions, even if there is presently insufficient supporting scientific evidence (Bastian, 2013; Cooney and Dickson, 2005).
In Natura 2000 sites, management organisms are obliged to establish conservation measures necessary to maintain the habitat types and species in a favorable conservation status. In order to be and represent a success, management must be financed. These measures are often expensive therefore, in parallel, the PAF (priority activities framework) must be drawn up to prioritize the financial resources that are necessarily "finished".
The allocation of financial resources to be allocated to management programs is therefore a critical aspect when making priority choices (Joseph et al., 2009). This assessment has in fact technical/scientific, normative but also cultural bases.
In the PAF of the Emilia-Romagna Region period 2104 - 2020 for the management of the Natura 2000 network, for example, the term "subsidence" is never mentioned. There seems to be a gap between the different disciplines involved (biological, geological, engineering) and the future scenarios available which expect a drastic modification for 2100 (Antonioli et al., 2017; Perini et al., 2017).
“Historically, a pattern of co-evolutionary adaptations between social systems and natural systems must have been the norm, with the adaptations in many cases driven by crises, learning and redesign.. . . For this reason, ancient cultural practices of resource use are more than anthropological curiosities; they are part of humanity’s wealth of adaptations that can serve the contemporary world as well.” (Holling et al., 1998; Ludwig, 2001).
In species and habitats conservation, the approach follows a similar and repetitive pattern. Initially, specialists and researchers highlight the problem [crisis recognition]. Legal regulations follows, with a general and local application. The subsequent management phase must take into account the limits of applicability (for example technical, economic) [learning phase].
However, in highly dynamic environments, management interventions do not always achieve the set objectives, in particular in a long-term perspective [necessity of redesign].
Historically, the cultural and socio-economic context produces the initial intentions of protection and management. These management perspectives change over time together with environmental conditions. Human society make governance choices in an adaptive way, by selecting some natural processes (Ecosystem services) which produce benefits for man in a given historical phase (see Smail, 2017).
Some examples can be made referring to management activities aimed at the conservation of habitats and species. The case of the dunes has already been illustrated in tab. 1a. After an initial phase in which they were considered unproductive "wrecks" abandoned by the sea, in the early twentieth century they were leveled to make room for the coastal pine forest in front of Ravenna coast. As the

sea erosion and tourism development of the area progressed, the dunes were protected by laws and regulations starting from the first 80s. The current experimental restoration phase is however not effective to guarantee a healthy dune system if the main conditions, availability of sediment and sufficient distance from the sea are lacking.
Some other similar cases are shown in the table below.

	Crisis	*Learning*		*Redesign*	
		Regulation by law	*Management*		
dead wood	*Conservation of saproxilic insect species*	*"Habitat" Dir.92/43; Establishment of Integral Nature Reserve (1971)*	*LIFE "MIPP"; high availability of dead wood*	*Cerambix cerdo appears locally extinct: due for inadequate forestry practices or for forest habitat degradation/isolation?*	*Extinction*
Fratino (Charadrius alexandrinus)	*Conservation of endangered species nesting on beaches*	*"Habitat" Dir.92/43*	*LIFE "AGREE"; Creation and protection of nesting sites; regulation*	*Decline of the nesting population at regional level*	
Mesola red deer (Cervus Elaphus italicus)	*Conservation of Mesola forest indigenous population*	*Programma nazionale di conservazione del cervo della Mesola (2010)*	*fallow deer numerical control; support foraging; …………………*	*Plan relocation to different areas*	*Traslocation*

The previous cases show that there is no overall vision that leads to an effective redesign phase except for Mesola Deer.
Even some of the interventions carried out in the study area in the last twenty years as LIFE projects represent the attempt to adapt to a crisis situation. The crisis is already recognized when the project is applied. The actions carried out represent the subsequent experimentation and learning phase. Only in the case of LIFE "AGREE" – on long period lagoon management – an early redesign stage can be seen.

Governance

In habitats ongoing in rapid compromise, initially it is necessary to accept the management costs to continuous adaptation to changing conditions. However, long period conservation is not possible (technically, economically) only by relying on limited band-aid solutions.
For this reason, actions such as the incorporation of nonmarket values in ecosystem management process is a particularly important option (Costanza et al., 2014).
Cultural recognition of intrinsic value of Biodiversity like a benefit for human society is the first step to force long-term governance decisions address towards regulating and cultural services respect short-term provisioning services.
Therefore, economic valuation is not the adequate method for determining the goals or priorities of conservation policies (Bastian, 2013).
Where socio-economic background is active, like in Sacca di Goro, society seize the opportunities with a fast adaptive process finalized in a short term period advantage.
The excavation activity to promote hydraulic circulation in the Sacca di Goro for the benefit of intensive economic activities carried out locally (shellfish farming), is an opportunity to have

adequate quantities of sediment to be used to rebuild transition habitats (salt marshes, reeds) lost due to the progress of the effects of marine erosion and subsidence (LIFE "AGREE"). This is in the event of future availability of sediments, assuming that it will be able to dispose of substantial contributions of fresh water from the nearby delta branches.
Where fast marine erosion is active, like along the rest of the regional coast, and provisioning services are scarce, a long period vision delay coming. Cultural and regulating services, provide from transitional habitats biodiversity in the form of benefits are difficult to be recognized by society without a clear vision of immediate advantage.
The environmental area in front of the city of Ravenna will probably be protected with rigid defenses and conspicuous continuous nourishment. In this situation the defense of the dune will be considered as a priority compared to the defense of the beach, at the expense of the coastal pine forest.
Bellocchio and Bevano areas, on the other hand, are both rapidly eroding, also due to the particularly high subsidence rate and for the stiffening of the coastlines located immediately south of these areas. The rigid defense must be positioned further inland and the opportunity to create brackish lagoons to contain weather and sea phenomena will prevail at the expense of agricultural land in its time deriving from various reclamation phases.
It is necessary to fully integrate geosites into the planning documents of protected areas as a part of an ecosystem approach. The ecosystem approach recognizes the integrity of abiotic and biotic elements in nature conservation policies (Peña et al., 2017).
It is therefore to be considered desirable to start a governance phase based on a broader approach (temporal, spatial and cultural) which tends to create the conditions within which biodiversity can express itself in all its forms (genetics, specific, habitat) outside the boundaries of current protected areas affecting large territories that would still be invaded by the waters (see Nagendra, 2008).

CONCLUSIONS

According MA (2005) "Some climate change is inevitable, and ecosystems and human societies will need to adapt to new conditions. Human populations will face the risk of damage from climate change, some of which may be countered with current coping systems; others may need radically new behaviors. Climate change needs to be factored into current development plans.[…] Most of the direct drivers of change in ecosystems and biodiversity currently remain constant or are growing in intensity in most ecosystems. […] By the end of the century, climate change and its impacts may be the dominant direct drivers of biodiversity loss and the change in ecosystem services globally." And also "An effective set of responses to ensure the sustainable management of ecosystems requires substantial changes in institutions and governance, economic policies and incentives, social and behavior factors, technology, and knowledge".
As already known by technicians, it does not exist definitive solutions to long period management, and at Po river delta level a deep modification is expected.
Already for some years, management policies area direct to adaptation strategies rather than trying to counter marinization of the area (Aguzzi et al., 2016).
In the short period, management can be persecutes protecting and restoring habitats using traditional techniques. Beach nourishment, dune reconstitution, fresh water supply, reconstitution of transitional environments, fight against invasive species are all possibilities that can be used to favor adaptation to changed conditions.
Important signs of a change in approach must be highlighted in recent years. In Sacca di Goro, the shellfish farmers have started to support privately the costs for the maintenance of the lagoon,

dredging the sediments in the inlets and reusing them where it is needed for environmental restoration (LIFE "AGREE"). However, these are always interventions aimed at ensuring the clam farming activity. The possibility that these interventions are useful for the conservation of biodiversity is residual. The engine of intervention remains the economic one.
It is therefore necessary to operate at a higher level of governance, able to identify future natural and socio-economic scenarios.
I propose a framework in which ESs do not have a hierarchy of values.
In the course of history and in different places, the SEs assume the relative value that society attributes them. Provisioning services prevail in the historical places and moments of stability or progression of the coastline.
During periods of marine ingression it is necessary to favor regulating services and evaluate the historical dynamics of management SEs to target realistic management goals. In long-term governance, cultural services, understood as recognition of the need to safeguard the value of the protected good in themselves, are the only ones capable of directing and forcing planning policies.
The radical change in biodiversity must be addressed by promoting habitat translocations and creating new natural environments, in particular large brackish transition wetlands, with the purpose of conservation, production and tourist attractiveness, properly designed and expertly managed, with a selective approach to the crisis that allows transfer current and traditional knowledge to tomorrow (see Diamond, 2019).
This aspect of cultural attribution is initially the heritage of a few visionaries, subsequently it will be extended to the entire society. The perception of the advantages and opportunities offered by a new shared educational service should spread in society in a similar way to a cultural process. Therefore, ethic need to be institutionalized in the routine practice of natural resource management (Batavia and Nelson, 2016): create the ideal conditions for biodiversity to express itself as a priority goal.

Bibliography

AA. VV. (a cura di B. Minerbi), (1984). Riserva Naturale Gran bosco della Mesola. Piano di Gestione Naturalistica per il decennio 1980-1989. Ministero Agricoltura e Foreste.

ASFD - Azienda di Stato per le Foreste Demaniali (1960). *L'Azienda di Stato per le Foreste Demaniali. Volume I*. Roma: Soc. A.B.E.T.E.

Aguzzi M., Bonsignore F., De Nigris N., Morelli M., Paccagnella T., Romagnoli C., Unguendoli S. (2016). Stato del litorale emiliano-romagnolo al 2012. Erosione e interventi di difesa. I Quaderni di ARPAE, 227 pp.

Antonioli F., Anzidei M., Amorosi A., Lo Presti V., Mastronuzzi G., Deiana G., De Falco G., Fontana A., Fontolan G., Lisco S., Marsico A., Moretti M., Orrù P.E., Sannino G.M., Serpelloni E., Vecchio A. (2017). Sea-level rise and potential drowning of the Italian coastal plains: flooding risk scenarios for 2100. Quaternary Science Reviews 158: 29-43.

Bastian O. (2013). The role of biodiversity in supporting ecosystem services in Natura 2000 sites. Ecological Indicators 24: 12–22.

Batavia C. and Nelson M. P. (2016). Conceptual Ambiguities and Practical Challenges of Ecological Forestry: A Critical Review. *J. For.* 114 http://dx.doi.org/10.5849/jof.15-103

Benini, L. (1931). L'Ufficio di Amministrazione delle Foreste Demaniali di Ravenna e la sua attività dal 1922 al 1931. In Il Comune di Ravenna (pp. 9–11). Fasc. III-IV:.

Berkes F, Folke C. (editors. 1998). Linking social and ecological systems: management practices and social mechanisms for building resilience. Cambridge (England): Cambridge University Press.

Bürgi M., Silbernagel J., Wu J., Kienst F. (2015). Linking ecosystem services with landscape history. Landscape Ecol. 30: 11 – 20.

Cantiani, P., Ferretti, F., Pignatti, G., Andreatta, G., & Nobili, G. (2008). Le pinete demaniali costiere di Ravenna e Ferrara: Analisi dei tipi forestali e strutturali per la scelta del trattamento selvicolturale nel processo di pianificazione. Annuali C.R.A – Centro di Ricerca per la Selvicoltura, 35, 61–68.

Convention on Biological Diversity, 2010. Convention on Biological Diversity. COP Decision X/2.

Cooney, R., Dickson, B. (2005). Precautionary principle, precautionary practice: lessons and insights. In: Cooney, R., Dickson, B. (Eds.), Biodiversity and the Precautionary Principle: Risk and Uncertainty in Conservation and Sustainable Use. Earthscan, London, pp. 287–298.

Costanza, R., d'Arge, R., de Groot, R., Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neill R.V., Paruelo J., Raskin R.G., Sutton P. & van den Belt M. (1997). The value of the world's ecosystem services and natural capital. *Nature* 387, 253–260. https://doi.org/10.1038/387253a0

Costanza, R., Kubiszewski, I., Giovannini, E., Lovins, H., McGlade, J., Pickett, K.E., Ragnarsdottir, K.V., Roberts, D., De Vogli, R., Wilkinson, R. (2014). Time to leave GDP behind. Nature 505, 283–285.

Diamond J. (2019). Crisi. Einaudi.

Diaz S.,Fargione J., Chapin F.S., Tilman D. (2006). Biodiversity loss threatens human well-being. PLoSBiol. 4 (8): 1300 – 1305.

Gantioler, S., Rayment, M., Bassi, S., Kettunen, M., McConville, A., Landgrebe, R., Gerdes, H., ten Brink, P. (2010). Costs and socio-economic benefits associated with the Natura 2000 network. Final Report to the European Commission, DG Environment. Institute for European Environmental Policy/GHK/Ecologic, Brussels.

Geneletti D. (2013). Ecosystem services in environmental impact assessment and strategic environmental assessment. Environ. Impact Assess. Rev. 40: 1 – 2.

Gómez-Baggethun E., Mingorria S., Reyes-Garcia V., Valvet L., Montes C. (2010). The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecol. Econ. 69: 1209 – 1218.

Haines-Young, R., Potschin, M. (2010). The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D., Frid, C. (Eds.), Ecosystem Ecology: A New Synthesis, BES Ecological Reviews Series. CUP, Cambridge.

Holling C.S., Berkes F., Folke C. (1998). Science, sustainability and resource management. In: Berkes F, Folke C, editors. Linking social and ecological systems: management practices and social mechanisms for building resilience. Cambridge (England): Cambridge University Press. p 342–362.

Joseph L.N. Et al. (2009). Optimal allocation of resources among threatened species: a project prioritization protocol. Conservation Biology, vol. 23, n. 2: 328 – 338.

La Notte A., D'Amato D., Mäkinenc H., Paracchinia M.L., Liquetea C., Egohd B., Geneletti D., Crossman N.D. (2107). Ecosystem services classification: A systems ecology perspective of the cascade framework. Ecological Indicators 74 (2017) 392–402. http://dx.doi.org/10.1016/j.ecolind.2016.11.030

Lautenbach S., Kugel C., Laush A., Seppelt R. (2011). Analysis of historic changes in regional ecosystem service provisioning using land use data. Ecological Indicators 11: 676 – 687.

Longhi, G. (1969). Foresta “Pineta di Ravenna”. Piano di Assestamento esecutivo per il decennio 1970–1979. Azienda di Stato per le Foreste Demaniali.

Ludwig D. (2001). The Era of Management Is Over. Ecosystems 4: 758–764.

MA (2005). Ecosystems and Human well-being: Synthesis. Island Press, Washington, DC.

Maes, J., Liquete, C., Teller, A., Erhard, M., Paracchini, M.L., Barredo, J.I., Grizzetti, B.,Cardoso, A., Somma, F., Petersen, J. (2016). An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. Ecosyst. Serv. 17, 14–23.

Mertz O., Ravnborg H.M., Lovei G.L., Nielsen I., Konijnendijk C.C. (2007). Ecosystem services and biodiversity in developing countries. Biodivers. Conserv. 16, 2729–2737.

Nagendra H. (2008). Do Parks Work? Impact of Protected Areas on Land Cover Clearing. AMBIO: A Journal of the Human Environment 37(5), 330-337. https://doi.org/10.1579/06-R-184.1

Nijhuis M. (2012). Quali specie vivranno. Le Scienze, n. 530: 88 – 93.

Patrone G. (1948). “Piano di assestamento del Gran Bosco della Mesola” per il trentennio 1948 – 1977 – Documento dattiloscritto.

Peña L., Monge-Ganuzas M., Onaindia M., De Manuel B.F., Mendia M. (2017). A Holistic Approach Including Biological and Geological Criteria for Integrative Management in Protected Areas. Environmental Management 59:325–337 DOI 10.1007/s00267-016-0781-4

Perini L., Calabrese L., Luciani P., Olivieri M., Galassi G., Spada G. (2017). Sea-level rise along the Emilia-Romagna coast (Northern Italy) in 2100: scenarios and impacts. Nat. Hazards Earth Syst. Sci., 17, 2271–2287. https://doi.org/10.5194/nhess-17-2271-2017

Potschin, M., Haines-Young, R. (2016). Defining and measuring ecosystem services. In: Potschin, M., Haines-Young, R., Fish, R., Turner, R.K. (Eds.), Routledge Handbook of Ecosystem Services. Routledge, Taylor & Francis Group, London; New York, p. 2016.
Price J., Silbernagel J., Miller N., Swaty R., White M., Nixon K. (2012). Eliciting expert knowledge to inform landscape modelling of conservation scenarios. Ecol. Mod. 229: 76 – 87.

Schwartz J. (2018). Sott’acqua. Le Scienze 604: 76 – 87.

Sejak J., Dejmal I. et al. (2010). Method of monetary valuation of territorial ecological functions. Mscr., J.E. Purkyne University, Usti nad Labem, Czech Republic.

Smail D.L. (2017). Storia profonda. Il cervello dell’uomo e l’origine della storia. Bollati Boringhieri Editore, Torino.

Spangenberg, J.H., Settele, J. (2010). Precisely incorrect? Monetising the value of ecosystem services. Ecol. Complex. 7: 327–337.

Zangheri P. (1936). Romagna fitogeografica (1°) Flora e vegeta-zione delle Pinete di Ravenna e dei territori limitrofi fra queste e il mare, pp. 422, Valbonesi, Forlì.

Università degli Studi di Ferrara

Sezioni

Dottorati di ricerca

Il tuo indirizzo e-mail

giovanni.nobili@student.unife.it

Oggetto:

Dichiarazione di conformità della tesi di Dottorato

Io sottoscritto Dott. (Cognome e Nome)

NOBILI Giovanni

Nato a:

ROMA

Provincia:

ROMA

Il giorno:

04 luglio 1964

Avendo frequentato il Dottorato di Ricerca in:

Biologia evoluzionistica e Ecologia

Ciclo di Dottorato

32

Titolo della tesi:

Environmental changes and biodiversity governance in Po river delta area: a contribution of nature-based criteria

Titolo della tesi (traduzione):

Modificazioni ambientali e gestione della biodiversità nell'area del Delta del Po in una prospettiva di lungo periodo: un contributo sui criteri di sostenibilità

Tutore: Prof. (Cognome e Nome)

FANO Elisa Anna

Settore Scientifico Disciplinare (S.S.D.)

BIO/07

Parole chiave della tesi (max 10):

Servizi ecosistemici, gestione aree protette, Delta del Po, modificazioni ambientali, Ecosystem Services, governance, protected areas, environmental and climatic changes

Consapevole, dichiara

CONSAPEVOLE: (1) del fatto che in caso di dichiarazioni mendaci, oltre alle sanzioni previste dal codice penale e dalle Leggi speciali per l'ipotesi di falsità in atti ed uso di atti falsi, decade fin dall'inizio e senza necessità di alcuna formalità dai benefici conseguenti al provvedimento emanato sulla base di tali dichiarazioni; (2) dell'obbligo per l'Università di provvedere al deposito di legge delle tesi di dottorato al fine di assicurarne la conservazione e la consultabilità da parte di terzi; (3) della procedura adottata dall'Università di Ferrara ove si richiede che la tesi sia consegnata dal dottorando in 2 copie di cui una in formato cartaceo e una in formato pdf non modificabile su idonei supporti (CD-ROM, DVD) secondo le istruzioni pubblicate sul sito: http://www.unife.it/studenti/dottorato alla voce ESAME FINALE – disposizioni e modulistica; (4) del fatto che l'Università, sulla base dei dati forniti, archivierà e renderà consultabile in rete il testo completo della tesi di dottorato di cui alla presente dichiarazione attraverso l'Archivio istituzionale ad accesso aperto "EPRINTS.unife.it" oltre che attraverso i Cataloghi delle Biblioteche Nazionali Centrali di Roma e Firenze; DICHIARO SOTTO LA MIA RESPONSABILITA': (1) che la copia della tesi depositata presso l'Università di Ferrara in formato cartaceo è del tutto identica a quella presentata in formato elettronico (CD-ROM, DVD), a quelle da inviare ai Commissari di esame finale e alla copia che produrrò in seduta d'esame finale. Di conseguenza va esclusa qualsiasi responsabilità dell'Ateneo stesso per quanto riguarda eventuali errori, imprecisioni o omissioni nei contenuti della tesi; (2) di prendere atto che la tesi in formato cartaceo è l'unica alla quale farà riferimento l'Università per rilasciare, a mia richiesta, la dichiarazione di conformità di eventuali copie; (3) che il contenuto e l'organizzazione della tesi è opera originale da me realizzata e non compromette in alcun modo i diritti di terzi, ivi compresi quelli relativi alla sicurezza dei dati personali; che pertanto l'Università è in ogni caso esente da responsabilità di qualsivoglia natura civile, amministrativa o penale e sarà da me tenuta indenne da qualsiasi richiesta o rivendicazione da parte di terzi; (4) che la tesi di dottorato non è il risultato di attività rientranti nella normativa sulla proprietà industriale, non è stata prodotta nell'ambito di progetti finanziati da soggetti pubblici o privati con vincoli alla divulgazione dei risultati, non è oggetto di eventuali registrazioni di tipo brevettale o di tutela. PER ACCETTAZIONE DI QUANTO SOPRA RIPORTATO

Firma del dottorando

Ferrara, li _____14 febbraio 2020_____ (data) Firma del Dottorando ________________________________

Firma del Tutore

Visto: Il Tutore Si approva Firma del Tutore Elisa Anna Fano ________________________________