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Ecosystem services, biodiversity and water quality in transitional ecosystems

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Ecosystem services, biodiversity and water quality in transitional ecosystems

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Table of contents

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Editorial: Ecosystem services, biodiversity, and water quality in transitional ecosystems

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Editorial on the Research Topic
Ecosystem services, biodiversity, and water quality in transitional ecosystems

Transitional ecosystems, including estuaries, lagoons, and coastal lakes, are complex human-environmental systems that offer a wide range of societal benefits, including both commercial and non-commercial values. This book sets the stage for significant advances in several aspects of transitional ecosystem research and management.

Climate change shifts in temperature, precipitation, storminess, and sea levels affect the evolution of transitional ecosystems, including the functioning of ecosystem processes, and alterations to species biodiversity. Long-term coastal observations and historical reconstructions can thus provide valuable insights into the consequences of global change on transitional ecosystems (Newton et al.).

The complexity of transitional ecosystems is further increased by the activities of economic sectors that take place in them, acting as drivers of change with different degrees of (un)sustainability. The most important sectors are tourism, fishing, aquaculture, salt mining, industrial activities, maritime shipping and ports, development of urban areas, and related activities, and agriculture. These complex, coastal zones are therefore subject to various social and environmental pressures, including changes in land use, hydrology, and sedimentology, as well as the extraction of mineral resources such as salt and sand. These pressures can lead to changes in the physical environment, including variations in geomorphology or salinity and dissolved oxygen levels, and can also result in the loss of biodiversity and the decline of important ecosystem services, such as coastal protection and seafood. The combination of past interventions, physical forcing functions and present activities is responsible for the numerous pressures and issues that threaten transitional ecosystems (Newton et al.). Understanding the causes and consequences of these anthropogenic pressures helps to identify effective management strategies that minimize negative consequences and promote the sustainable use of valuable resources. By understanding the drivers of change and the impacts of human activities, managers can work toward the sustainable use and conservation of these important systems. This may involve a range of measures, such as the regulation of freshwater inflows and connectivity, the implementation of best management practices and the designation of protected areas.

Environmental, economic and social issues call for new integrated management perspectives. The need for systematic conservation planning has further motivated the analysis of patterns and processes at regional scales. Social-environmental analysis is a useful tool to inform management of coastal lagoons. El Mahrad et al., shows this for 11 lagoons in North Africa by applying an adaptive management framework (Drivers-Activities-Pressures-State Change-Impact-Responses), to provide a structured approach for the social-ecological analysis of coastal lagoons systems to identify potential management measures as responses to address negative impacts.

Human activities and pressures, such as land use change and pollution, can also have significant consequences on the biodiversity and water quality of transitional ecosystems. The implementation of environmental policies, such as Natura 2000 and the Water Framework Directive (WFD), provide a legal framework for improving water quality, ecological status and biodiversity. Picone et al., using the polychaete worm *Hediste diversicolor,* show the biota itself can be used to monitor contaminant levels in the environment. In this example, *Hediste diversicolor* bioaccumulates polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs).

Biodiversity and water quality are important aspects of the functioning and value of transitional ecosystems, to both human and animal populations, such as birds. The distribution and abundance of water birds can also be used as indicators of ecosystem health and water quality, as many species are sensitive to changes in their habitat and can be affected by pollution and other forms of human disturbance (Maneas et al.).

Coastal lagoons and other transitional ecosystems thus provide a range of valuable goods and services that support human wellbeing and economic development. Ecosystem services, including both commercial and non-commercial values, can be important considerations in the management and conservation of transitional ecosystems. Coastal management approaches that consider the full range of ecosystem services can help to ensure sustainable use of these systems. These ecosystem services offer a wide range of societal benefits. Key ecosystem processes, such as nutrient cycling and primary production, can support the biological resources that underpin these industries. In addition, these ecosystems play a critical role in the regulation of environmental processes, such as the purification of water, the sequestration of carbon, and the protection against storms and erosion. Some are commercial and can be readily valued in monetary terms, for example shellfish and finfish harvested for human consumption, as well as timber and medicinal plants (Afonso et al.).

The ecosystem services approach values the ecosystem by a variety of means, including monetary, non-monetary and ecological. There is also a need for more research on the spatio-temporal evolution of the ecological services provided by mangroves and salt marshes in transitional ecosystems. Practical methodologies can be used to quantify the variability of limiting factors that impact the provision of these services, as shown by the literature review of Carrasco. Such studies can help to identify the key drivers of change and inform the development of effective conservation and management strategies to maintain the resilience of these ecosystems and the benefits they provide to humans. Furthermore, the use of modeling, to better understand the consequences of human activities combined with climate change, provide the knowledge-base necessary to identify potential methods to improve the functioning of coastal lagoons and restoration options.

Effective conservation and management of transitional ecosystems is essential to maintain and enhance the goods and services they provide. Coastal management strategies and decisions should also consider the social, cultural, and historical values of these ecosystems, as well as the interconnections between lagoons and other marine protected areas (De Wit et al.). Transitional ecosystems may also be connected to marine protected areas (MPAs), and the interplay between these systems can be important in terms of conservation and sustainable use. This may involve a range of approaches, such as the identification and protection of key habitats, the restoration of degraded areas, and the implementation of best management practices, such as nature-based solution, for example using reed-beds (Karstens et al.).

Transitional ecosystems also offer potential opportunities for sustainable use, such as the exploitation of macroalgal biomass as a tool for the recovery of these systems (Sfriso et al.). The use of green and blue infrastructures, such as mangroves and salt marshes and reed beds, can provide multiple benefits, including the enhancement of biodiversity, the protection of coastal communities from storms and erosion, and the provision of recreational and cultural services (Afonso et al.; Karstens et al.). In addition to the traditional goods and services provided by transitional ecosystems, there is also potential to produce energy and the development of blue growth and blue infrastructure. The storage of blue carbon in aquatic, riverine, and underwater ecosystems, such as seagrasses, can also have significant economic value.

Overall, transitional ecosystems provide a range of valuable goods and services, and the management and conservation of these systems should consider the full range of these values. For example, unsustainable fisheries and aquaculture activities can also affect transitional ecosystems, requiring strategies and methodologies for adopting sustainable practices in these economic sectors. Transitional ecosystems can also have impacts on human health and wellbeing, both directly and indirectly. For example, the provision of clean water, food, and recreation can all contribute to human health and wellbeing. Nevertheless, transitional ecosystems can also be affected by human activities, such as pollution and contamination, which can have negative impacts on human health. The full range of impacts on human health and wellbeing should be considered when managing and conserving transitional ecosystems. Effective governance and decision-making processes are critical for the sustainable management and conservation of transitional ecosystems. This may involve the participation of a range of stakeholders, including local communities, government agencies, and industries. Inclusive and transparent decision-making processes can help to ensure that the needs and values of all stakeholders are considered.

There are still many questions and challenges to be addressed in the study and management of transitional ecosystems. Future research may focus on improving our understanding of the impacts of human activities and climate change on these systems, as well as developing and testing new management and conservation approaches (Newton et al.). It will also be important to continue to engage with stakeholders and decision-makers to ensure that the needs and values of all interested parties are considered in the management of transitional ecosystems. The economic value of ecosystem services provided by transitional ecosystems, such as coastal lagoons, is often undervalued and not fully recognized. This can lead to the under investment in their conservation and management. More research is needed to better understand the economic impact of these ecosystems and to develop mechanisms for the valuation and payment of ecosystem services. This could include the use of economic instruments, such as the creation of markets for carbon credits or the development of compensation schemes for the provision of ecosystem services. The potential economic impact of blue carbon stocks, such as those found in mangroves, saltmarshes and seagrasses, is an area of growing interest and could provide a valuable source of funding for the conservation and management of these valuable ecosystems.

Author contributions

MM provided the outline. AN drafted and edited the text and submitted the editorial. AP-R, SR, and MM provided edits and suggestions. After feedback from the journal Editor, AN re-submitted a revised and improved text. All authors contributed to the article and approved the submitted version.

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Simple Assessment of Spatio-Temporal Evolution of Salt Marshes Ecological Services

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A number of previous research studies have addressed the enormous role played by biodiversity and ecosystems in human well-being and have placed particular emphasis on the consequences of the reduction or loss of these services. A handful of studies have implemented practical methodologies to quantify the variability of limiting factors leading to reductions in these ecological services. The aim of this article is to document the limited number of studies that have analyzed coastal ecosystem services and acknowledge the impacts of physical changes in habitat provision. In one example, it is clear that the maintenance of salt marshes depends on sedimentary supply and consequent morphological variability in spite of the fact that there is usually no recurrent integration of habitat time-space dynamics (sediment availability) during the quantification and monetization of marsh services (i.e., monetary valuation of salt marsh services). This means that one key challenge facing the analysis of salt marsh (or other ecosystem) services in a global climate context is to predict future value, based on past trends, while at the same time guaranteeing conservation. Research in this field has been very broad and so the use of long-term evolutionary datasets is proposed here to explain future habitat provision. An empirical approximation is also presented here that accounts for service provision and enables time-space analysis. Although improvements will be required, the equation presented here represents a key first step to enable managers to cope with the constraints of resource limitations and is also applicable to other habitats.

Keywords: ecosystem services, time, physical changes, conservation, salt marsh

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INTRODUCTION

Coastal and terrestrial ecosystems all provide a number of fundamental life-supporting services (i.e., natural capital, namely varied benefits that humans freely gain from the nature; Costanza et al., 2017) and societal goods (i.e., that benefits the largest number of people in the largest possible way; Daily, 1997; Costanza et al., 2017) on which human welfare is dependent (Millennium Ecosystem Assessment, 2005; Carpenter et al., 2006). de Groot et al. (2002) explains the assessment of ecosystem goods and services has involving the translation of the ecological complexity (structures and processes) into a number of ecosystem functions (i.e., the capacity of natural processes and components to provide goods and ecosystem services) that satisfy human needs, directly or indirectly (in agreement with Daily, 1997; Balmford et al., 2002).

Significant research has therefore been devoted to explaining declines in services and the survival rates of coastal ecosystems over the last two decades. A brief history of ecosystem services and natural capital can be found in Costanza et al. (2017). In previous research, Costanza et al. (1997) and others have presented interesting perspectives about how the world economy would come to an abrupt stop if such ecosystems and their services were to cease to exist (e.g., Turner et al., 2003; Palmer et al., 2005; Carpenter et al., 2006). Ecosystem services can be particularly important at a national and sub-national scale, if they play an effective role in making informed decisions about the use and management of the planet's resources, especially when trade-offs and synergies need to be taken into account (Balvanera et al., 2017). However, although recent scientific efforts have focused on quantifying current ecosystem service provisions in detail (e.g., Barbier et al., 2011; de Groot et al., 2012; Costanza et al., 2014), much less attention has been paid to the evolution of these variables in time and space, especially in light of future climatic adjustments (Greenberg et al., 2006). This lack of information can be filled with an increase in the number of conservation studies advising that ecosystem service provisions are limited in time, and they should be predicted based on prediction of spatial changes (Koch et al., 2009). Moreover, natural processes are characterized by thresholds and limiting functions (Koch et al., 2009), and ecosystem provision may sometimes not occur linearly.

The representation of the temporal dynamics of ecosystem services is a crucial research frontier in the field of ES modeling (e.g., Rova et al., 2019). Thus, it is clear that enhanced research on the spatio-temporal dynamics of ecosystems is likely to contribute to reductions in the cost of coastal conservation actions given the ongoing effects of climate change. A time-space-evolutionary analysis therefore purports to past information about physical changes through time (e.g., via morphology, morphodynamics, or morphometric analysis) within a given coastal ecosystem and can therefore help to predict the temporal functionality of such a system (scenarios).

The aim of this perspective is to express concerns about the need for a more compelling and integrative approach that triggers the analysis of variation in temporal (and spatial) coastal ecosystem services. A reminder is presented here that ecosystem service provision analysis should couple both current ecological state (i.e., flow or actual ecosystem service use) with morphological variability in order to enable more effective and sustainable ecosystem management. The morphological variability of a given ecosystem determines stock, or the potential provision of ecosystem services. Salt marshes are very commonly cited examples in ecosystem research, and so these habitats (i.e., natural environment supporting functional and structural functions) are discussed here in an attempt to express the degree of relationship between the various factors affecting given system variability as well as their expected impact on the services provided.

FREQUENT USE OF THE TERM "MORPHOLOGICAL VARIABILITY" IN COASTAL ECOSYSTEM SERVICES ARTICLES

The Thomson Reuters Web of ScienceTM database (WoS; web of science core collection) was mined in this analysis to compile articles in English purporting to relate coastal ecosystem services with morphological variability. This approach enabled an understanding of just how detached the two processes remain from one another in the recent scientific literature. The scientific terms used in each search have a cross-disciplinary relationship between them. This allows us to understand the evolution of a topic in a scientific domain; thus, strings and terms were defined on an author-by-author basis after reviewing articles about ecosystem services and the most relevant keywords. Papers were automatically identified by searching the terms throughout the published text. All output results were generated by WoS and utilized the same result exploitation search strategy as used in other studies (e.g., Chang et al., 2017; Walls, 2018; Wang et al., 2018; Zhao et al., 2018).

The Boolean search operators "ecosystem services" and "coast" were used for the first level of search in January, 2019 (Indexes=SCI-EXPANDED, SSCI, A&HCI, CPCI-S, CPCI-SSH, ESCI, CCR-EXPANDED, IC Timespan=All years); this approach led to the identification of 1,920 English articles from 108 different countries around the world published over the last 20 years to acknowledge coastal ecosystem services. This first level of search allowed the establishment of a starting sample universe and also limited the output of ecosystem services in relation to coastal domain context. The articles output from this search were distributed within five main WoS categories encompassing environmental sciences and ecology, marine and freshwater biology, oceanography, water resources, science technology, and other topics (**Figure 1**). Each of these subject categories contained at least 10% of the total number of search records; those referring to coastal ecosystem services (including the terms "ecosystem services" and "coast") can be shown to decrease in number from environmental and biological sciences into their physical counterparts, including oceanography and other topics with less outputs (**Figure 1**). This result suggests that most research to date has been concentrated within ecology and on conservation issues.

The second level of search undertaken here was aimed at validating the dependence between coastal ecosystem services studies and morphological variability. Thus, the terms "ecosystem services," "coast," and "morphology" (including "morphodynamics" or "morphometric") where searched in this case but led to the identification of just 35 published articles to acknowledge the morphological variability of coastal ecosystem services, albeit in increasing numbers over the last 4 years. As this comprises just 2% of total records output across the broad field of coastal ecosystem services (i.e., from a total of 1,920), it is clear that morphological variability analysis has not often been included in ecosystem services articles. A coupling between coastal ecosystem services and morphological variability is most frequent in marine biological and oceanography research categories (**Figure 1**).

The terms used in the first and second levels of search were defined based on the key concepts of the discussion herein presented, the topic- "ecosystem services," the domain—"coast,"

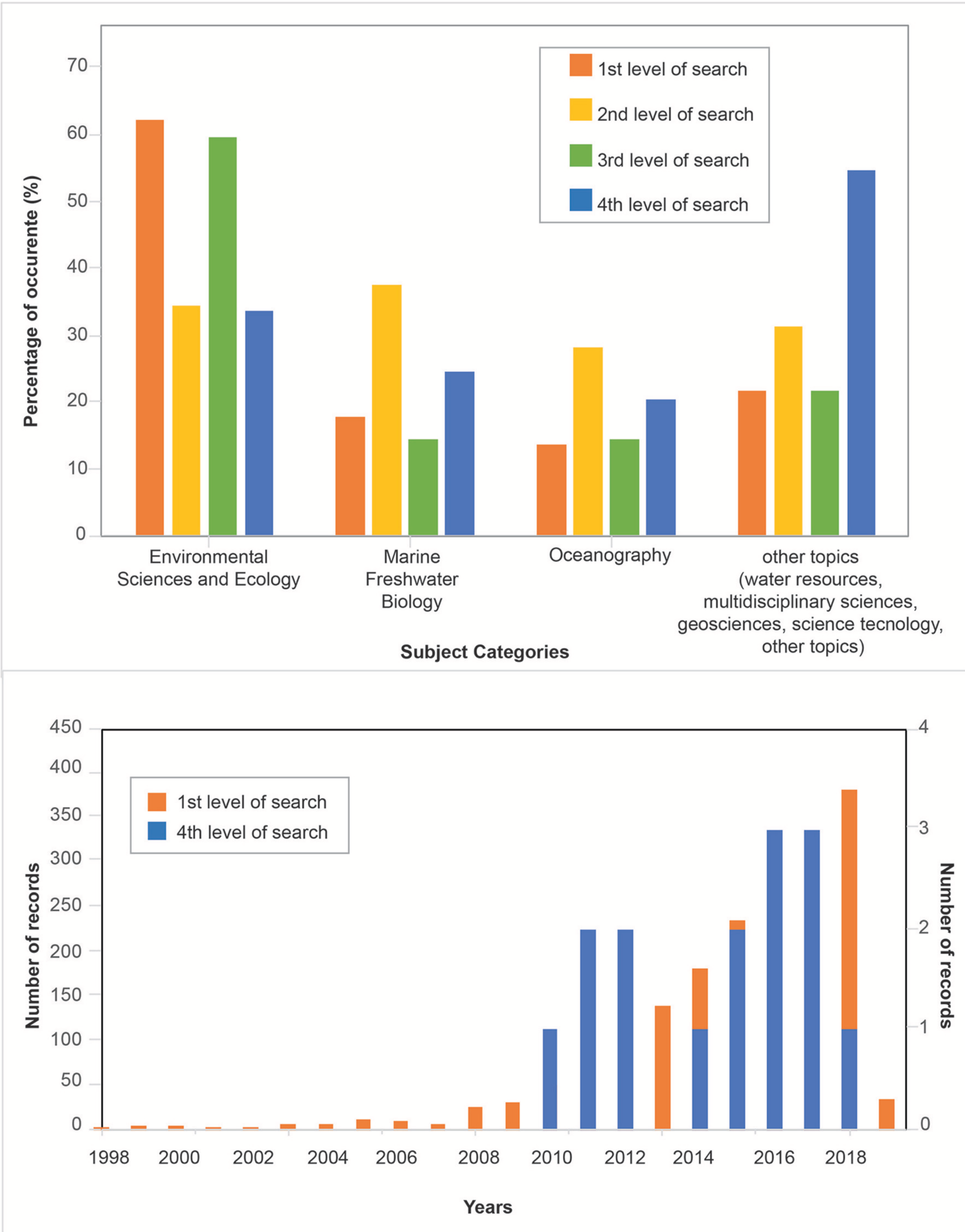

FIGURE 1 | Above, records reporting coastal ecosystem services, morphology analyses, and salt marsh strings obtained for the last 20 years, for the four levels of search, distributed within the five main WoS categories; below, comparison between the number of records obtained for the last 20 years in the first level of search and in the fourth level of search.

and the analysis—"morphology." There was no manipulation of the generated output results and, as initially preconized, they express the content of all publications in natural sciences, conducted in the last decades. The majority of authors of the different works used the same terms to express different conclusions on the same topic and/or domain.

FREQUENT USE OF THE TERM "MORPHOLOGICAL VARIABILITY" IN SALT MARSH ECOSYSTEM SERVICES ARTICLES

The WoSwas then searched at a third level to compile studies relating coastal ecosystem services and the morphological variability of salt marshes. With the introduction of the third level of search, it is possible to limit output results to a specific subject. The terms "ecosystem services," "coast," and "salt marsh (or saltmarsh)" were considered here and led to the identification of 247 articles that acknowledge the ecological value of salt marsh systems, mostly in environmental and biological sciences, and oceanography (**Figure 1**). The fourth search level also considered the terms "ecosystem services," "coast," "salt marsh" (or "saltmarsh"), and "morphology" (including "morphodynamics" and "morphometric") but led to the identification of just 15 articles (i.e., 37% of 247 acknowledging "salt marsh ecosystems") establishing a connection between salt marsh evolution and morphological variability (**Figure 1**). As the records output from this fourth level search are distributed between biological and physical categories, a recognition of the importance of morphological variability on salt marsh services assessment was also performed, but only for the last 10 years (**Figure 1**). Therefore, questions remain regarding whether current assessments of salt marsh ecosystem provision are incomplete and regarding how morphological variability, a factor not included very often in salt marsh (ecosystem) services, should be studied.

PHYSICAL PROCESSES LEADING TO CHANGES IN SALT MARSHES

Salt marshes are one of the most productive ecosystems on Earth (e.g., Caçador et al., 2015). Thus, if these ecosystems are left to migrate and adapt unimpeded (Kirwan et al., 2016b), such marshes may lessen the adverse impacts associated with sea level rises, including increases in coastal flooding, storm surges, and erosion (Bouma et al., 2014). Salt marshes also act like sponges, absorbing excess water, and reducing flooding, but persist as a delicate balance between their own growth and changing sea levels (Schile et al., 2014). Projecting future changes in salt marshes is a crucial step toward planning for and mitigating the impact of climate change on marsh biodiversity, as well as for detect early warning tipping points in these complex systems (Crosby et al., 2016).

Marsh coastal areas have been squeezed due to sea level rises and erosion, both primary threats to these ecosystems around the world (Torio and Chmura, 2013). Although sea level rise can pose a serious threat to the survival of salt marshes, there is growing evidence that as long as sediment supply is sufficient, vegetation-sedimentation feedback enables these ecosystems to vertically accrete in concert with sea level change (e.g., Redfield, 1972; Kirwan and Temmerman, 2009; IPCC, 2013; Carrasco et al., 2016; Kirwan et al., 2016a).

Sea level rise has been portrayed as creating accommodation space within which fine-grained sediments can settle (sediment supply rate); this increases in concert with the rate of sea level rise and theoretically can lead to concomitant changes in mineral sediment deposition (Carrasco et al., 2016); given high rates of sea level rise, however, alongside an insufficient supply of sediment and organic material, salt marsh inundation is likely to occur. In contrast, if the rate of sediment supply is much higher than that of sea level rise, silting-up will dominate, a marsh will shift toward a different environment and ecosystem, and areal losses should be expected (e.g., Schile et al., 2014; Carrasco et al., 2016). The overall morphological stability of salt marsh ecosystems in response to rising sea level depends on the present elevation of the marsh community in relation to its depth-response curve and the local mean sea level (e.g., Morris et al., 2002).

All of these outcomes suggest that the future of salt marshes and their services should be founded on an understanding of temporal and spatial morphological variability as well as via the identification of different target states. Although recent scientific efforts have been focused on quantifying salt marsh ecosystem provisions (Chmura, 2012; de Groot et al., 2012; Ouyang and Lee, 2014) as well as in determining the effect of sediment supply on the growth of marshes (e.g., Lovelock et al., 2011; Rizzetto and Tosi, 2012; Kirwan et al., 2016b; Müller-Navarra et al., 2019), the two approaches remained detached and so less attention has been paid to spatio-temporal service dynamics and will require adjustments. The result of this analysis therefore advances the conclusion that the maintenance of ecosystem services and the assessment of future salt marsh services provision in a global climate change context (as well as in other coastal ecosystems) should be based in the analysis of long-term physical datasets (e.g., Barbier et al., 2008). As it is essential that natural variability and cumulative effects be considered in the valuation of ecosystem services (Koch et al., 2009).

A number of previous studies, although not exclusively investigating ecosystem services, have nevertheless demonstrated the importance of past trends in coastal decision making and planning (e.g., Baily and Nowell, 1996; Murray et al., 2014; Chang et al., 2018; Wu et al., 2018). In terms of coastal planning efficiency and effectiveness variables considered by policy-makers over recent decades, emphasis has been placed on long-term trend analyses of hydro forcing mechanisms and their spatial distribution with a view to formulating climate change and sea level rise management plans. In previous work, Santana-Cordero et al. (2016) also demonstrated the key role of historical datasets to understanding socio-ecological dynamics and ecosystem services within a Mediterranean coastal landscape, as well as for prioritizing conservation tasks and formulating governance. Although previous researchers have illustrated these variables in practical terms, any relationship must encompass time-space dynamics and the amount of service provision. Just a handful of studies have so far been able to demonstrate the impact of long-term datasets on ecosystem-based approaches (e.g., Rieb et al., 2017), or in the modeling of the dynamics of multiple services (e.g., Rova et al., 2019), and so a framework is therefore proposed here (**Figure 2**) for the future analysis of ecosystem services. This framework suggests that the cumulative

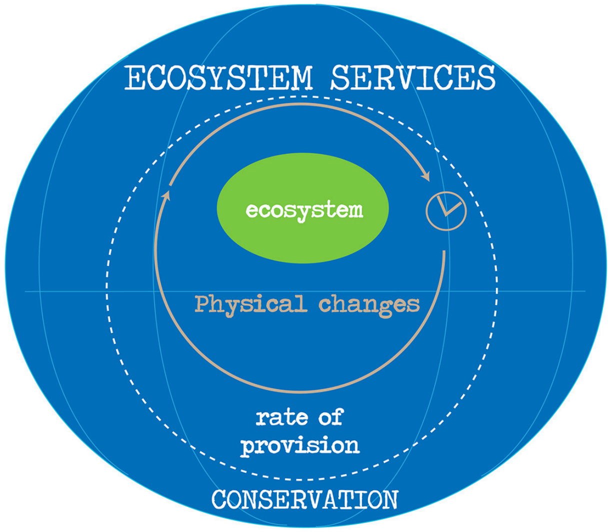

FIGURE 2 | A framework for ecosystem service analysis based on physical changes through time and space; the dashed white line in this figure limits the conservation dimension that should be taken into account alongside temporal variability in physical changes (brown line coupled with a clock symbol) for a certain ecosystem (green ellipse).

changes observed in the past (clock time in **Figure 2**) should be contained and serve as limitation to predictions of the provision rate; this rate encompasses an open boundary (dashed line) in the broad context of ecosystem service assessments, conservation, and the surrounding environment (**Figure 2**). In a first hand, the amount of physical variation in time (**Figure 2**) shown by a given ecosystem can be used to predict the rate of service provision; and in a second hand, the changes in time in the rates of provision can be used to schedule conservation measures distributed in time. Indeed, by including a time-space analysis (e.g., ecosystem growth or reduction tendency), it is possible to predict different scenarios for future services provision. Although modest, this is a reasonable assertion in coastal areas and in the framework of climate change. The framework proposed here is nevertheless sustainable and can be adapted via other ecosystem-based approaches.

IMPROVING ECOSYSTEM SERVICES CONSERVATION EFFECTIVENESS UNDER MORPHOLOGY ANALYSIS

A detailed time-space analysis includes lifetime features (e.g., the presence or absence of environments) and volumetric tendencies and is based on ecological dynamics (e.g., vegetation types and canopy) and their relationship with human alterations. It is therefore vital to understand both past and current ecosystem behaviors in this context; examples of morphological variables that be useful in predicting changes in marsh provision include cartographic variables (i.e., spatial length and width) as well as sediment accumulation rates (e.g., at different marsh zones), amongst others.

Analytical timeframes for time-space consideration should be established according to available datasets, while at the same time bearing in mind that longer-term analyses enable more adequate predictions for future evolution. This means that, depending on existing datasets, variables can be coupled, or separately applied, to predict service provision volumes. In one example, salt marsh past horizontal evolution (e.g., shape variability) and sediment accumulation rates can usefully be integrated, *marsh* morphological evolution, for a given time period (dt), and then multiplied by the current provision (*Pcurrent*) to predict service provision, *ESPmarsh*, for a given future time period (time = i; Equation 1), as follows:

$$\mathbf{ESPmarsh\ (i)\ = Pcurrent} \int_{\mathbf{t0}}^{\mathbf{t}} \mathbf{f\,(marsh)\ dt}$$

Thus, over a 100 year time period from present (I = 100), the marsh provision of a certain service *ESPmarsh* (100) can be determined by multiplying current provision, *Pcurrent*, with the rate of morphological change over recent years [f(*marsh*)]. The latter incorporates both time and space dimensions, i.e., as erosion, accretion, growth, reduction, increase, decrease, amongst others. *ESPmarsh* is thus a predictor of evolution that accounts for the ecosystem cumulative changes over time.

Equation (1) can be individually applied for each service as *Pcurrent* (and the rate of morphological change over recent years) is dependent on the type of positive service provided. The total value of each offered service (i.e., Provision services, Regulation services, Habitat services, and Cultural services; Costanza et al., 1997) per habitat in monetary units (de Groot et al., 2012) can also be subsequently determined. Thus, Equation (1) also enables different provision rates for services to be distinguished as well as across analytical periods. This approach also enables a time-provision curve to be completed for a given habitat, and assists coastal stewards to more accurately put habitat conservation in place. This also leads to enhanced plasticity during the construction and discussion of morphological change scenarios. The equation is of a simple application by a non-expert public (e.g., coastal decision making). The multifactorial interaction (or non-linearities in ecosystem) could be part of an improved formula version, but it would depend very much on the type of the ecosystem, besides having direct site-specific limitations.

SUMMARY

Although the dependence between ecology and geophysics for coastal and marine sciences has been well-described in the literature, the same relationship needs to be better described within the context of ecosystem services analysis. Significant research over the past two decades has therefore been devoted to explaining the importance of coastal ecosystem services, but just a few have described the available methodologies to access the factors limiting and affecting its evolution.

This work advances that the future maintenance of coastal ecosystem services, including increases and decreases in service provision, necessitates the study of time-space variability. A strong connection between the amount of services provided and the habitat morphological changes was underlined. This should be valid for other coastal systems, salt marsh being an example.

It is concluded that long-term spatial analyses should become one of the most important prerequisites when predicting the amount of services provided. The benefits of long-term predictions in the provision of services will enable the maximization of effective conservation, value maintenance and optimized opportunities vs. minimized risk, in the event that services are lost.

DATA AVAILABILITY

All datasets generated for this study are included in the manuscript and/or the supplementary files.

AUTHOR CONTRIBUTIONS

The author confirms being the sole contributor of this work and has approved it for publication.

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REFERENCES

Baily, B., and Nowell, D. (1996). Techniques for monitoring coastal change: a review and case study. *Ocean Coast. Manage.* 32, 85–95. doi: 10.1016/S0964-5691(96)00058-0

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., and Green, R. E. (2002). Economic reasons for conserving wild nature. *Science* 297, 950–953. doi: 10.1126/science.1073947

Balvanera, P., Quijas, S., Karp, D. S., Ash, N., Bennett, E. M., Boumans, R., et al. (2017). "Ecosystem Services," in *The GEO Handbook on Biodiversity Observation Networks*, eds. M. Walters and R. J. Scholes (Cham: SpringerOpen) 39–78. doi: 10.1007/978-3-319-27288-7_3

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., and Silliman, B. R. (2011). The Value of estuarine and coastal ecosystem services. *Ecol. Monogr.* 81, 169–183. doi: 10.1890/10-1510.1

Barbier, E. B., Koch, E. W., Silliman, B., Hacker, S. D., Wolansk, E., Primavera, J., et al. (2008). Coastal ecosystem-based management with non-linear ecological functions and values. *Science* 319, 321–323. doi: 10.1126/science.1150349

Bouma, T. J., van Belzen, J., Balke, T., Zhu, Z., Airoldi, L., Blight, A. J., et al. (2014). Identifying knowledge gaps hampering application of intertidal habitats in coastal protection: opportunities & steps to take. *Coast. Eng.* 87, 147–157. doi: 10.1016/j.coastaleng.2013.11.014

Caçador, I., Duarte, B., Marques, J. C., and Sleimi, N. (2015). "Carbon mitigation: a salt marsh ecosystem service in times of change," in *Halophytes for Food Security in Dry Lands*, eds M. A. Khan, M. Ozturk, B. Gul and M. Z. Ahmed (Elsevier), 83-110. doi: 10.1016/B978-0-12-801854-5.00006-6

Carpenter, S. R., Bennett, E. M., and Peterson, G. D. (2006). Scenarios for ecosystem services: an overview. *Ecol. Soc.* 11:29. doi: 10.5751/ES-01610-110129

Carrasco, A. R., Ferreira, Ó., and Roelvink, D. (2016). Coastal lagoons and rising sea level: a review. *Earth-Sci. Rev*. 154, 356–368. doi: 10.1016/j.earscirev.2015.11.007

Chang, H. T., Lin, M. H., Hwang, I. H., Chen, T. J., Lin, H. C., Hou, M. C., et al. (2017). Scientific publications in gastroenterology and hepatology in Taiwan: an analysis of web of science from 1993 to 2013. *J. Chin. Med. Assoc.* 80, 80–85. doi: 10.1016/j.jcma.2016.06.006

Chang, Y., Chu, K., and Chuang, L. K. (2018). Sustainable coastal zone planning based on historical coastline changes: a model from case study in Tainan, Taiwan. *Landsc.Urban Plan.* 174, 24–32. doi: 10.1016/j.landurbplan.2018.02.012

Chmura, G. L. (2012). What do we need to assess the sustainability of the tidal salt marsh carbon sink? *Ocean Coast. Manage.* 83, 25–31. doi: 10.1016/j.ocecoaman.2011.09.006

Costanza, R., D'Arge, R., de Groot, R. L., Farberk, S., Grasso, M., Hannon, B., et al. (1997). The value of the world's ecosystem services and natural capital. *Nature* 397, 253–260.

Costanza, R., de Groot, R., Braat, L., Kubisze- Wski, I., Fioramonti, L., Sutton, P., et al. (2017). Twenty years of ecosystem services: How far have we come and how far do we still need to go? *Ecosyst. Serv.* 28, 1–16. doi: 10.1016/j.ecoser.2017.09.008

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., et al. (2014). Changes in the global value of ecosystem services. *Glob. Environ. Chang.* 26, 152–158. doi: 10.1016/j.gloenvcha.2014.04.002

Crosby, S. C., Sax, D., Palmer, M. E., Booth, H. S., Deegan, L. A., Bertness, M. D., et al. (2016). Salt marsh persistence is threatened by predicted sea-level rise. *Estuar. Coast. Shelf Sci.* 181, 93–99. doi: 10.1016/j.ecss.2016.08.018

Daily, G. (1997). *Nature's Services: Societal Dependence on Natural Ecosystems.* Washington, DC: Island Press, 392.

de Groot, R., Brander, L., Ploeg, R., Costanza, R., Bernard, F., Braat, F., et al. (2012). Global Estimates of the value of ecosystems and their services in monetary units. *Ecosyst. Serv.* 1, 50–61. doi: 10.1016/j.ecoser.2012.07.005

de Groot, R. S., Wilson, M. A., and Boumans, R. M. J. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecol. Econ.* 41, 393–408. doi: 10.1016/S0921-8009(02)00089-7

Greenberg, R., Maldonado, J. E., Droege, S., and McDonald., M.V. (2006). Tidal marshes: a global perspective on the evolution and conservation of their terrestrial vertebrates. *BioScience* 56, 675–685. doi: 10.1641/0006-3568(2006)56[675:TMAGPO]2.0.CO;2

IPCC (2013). "Climate Change 2013: The Physical Science Basis," in *Contribution of Working Group I to the Fifth Assessment Report of the Intergovern- mental Panel on Climate Change,* eds. T. F. Stocker, D. Qin, G. K. Plattner, M. Tignor, S. K. Allen, J. Boschung, et al. (Cambridge; New York, NY: Cambridge University Press) 1535.

Kirwan, M., and Temmerman, S. (2009). Coastal marsh response to historical and future sea level acceleration. *Quatern. Sci. Rev.* 28, 1801–1808. doi: 10.1016/j.quascirev.2009.02.022

Kirwan, M. L., Temmerman, S., Skeehan, E. E., Guntenspergen, G. R., and Fagherazzi, S. (2016a). Overestimation of marsh vulnerability to sea level rise. *Nat. Clim. Chang.* 6, 253–260. doi: 10.1038/nclimate2909

Kirwan, M. L., Walters, D. C., Reay, W. G., and Carr, J. A. (2016b). Sea level driven marsh expansion in a coupled model of marsh erosion and migration. *Geophys. Res. Lett*. 43, 4366–4373. doi: 10.1002/2016GL068507

Koch, E. W., Barbier, E. B., Silliman, B. R., Reed, D. J., Perillo, G. M. E., Hacker, S. D., et al. (2009). Non-linearity in ecosystem services temporal and spatial variability in coastal protection. *Front. Ecol. Environ.* 7:126. doi: 10.1890/080126

Lovelock, C., Bennion, V., Grinham, A., and Cahoon, D. (2011). The role of surface and subsurface processes in keeping pace with sea level rise in intertidal wetlands of Moreton Bay, Queensland, Australia. *Ecosystems* 14, 745–757. doi: 10.1007/s10021-011-9443-9

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-being: Synthesis*. Washington, DC: Island Press. 138.

Morris, J. T., Sundareshwar, P. V., Nietch, C. T., Kjerfve, B., and Cahoon, D. R. (2002). Responses of coastal wetlands to rising sea level. *Ecology* 83, 2869–2877. doi: 10.1890/0012-9658(2002)083[2869:ROCWTR]2.0.CO;2

Müller-Navarra, K., Milker, Y., Bunzel, D., Lindhorst, S., Friedrich, J., Arz, H.W., et al. (2019). Evolution of a salt marsh in the southeastern North Sea region – anthropogenic and natural forcing. *Estuar. Coast. Shelf Sci.* 218, 268–277. doi: 10.1016/j.ecss.2018.12.022

Murray, N. J., Clemens, R. J., Phinn, S. R., Possingham, H. P., and Fuller, R. A. (2014). Tracking the rapid loss of tidal wetlands in the Yellow Sea. *Front. Ecol. Environ.* 12, 267–272. doi: 10.1890/130260

Ouyang, X., and Lee, S. (2014). Updated estimates of carbon accumulation rates in coastal marsh sediments. *Biogeosciences* 11, 5057–5071. doi: 10.5194/bg-11-5057-2014

Palmer, M. A., Bernhardt, E. S., Chornesky, E. A., Collins, S. L., Dobson, A. P., Duke, C. S., et al. (2005). Ecological science and sustainability for the 21st century. *Front. Eco. Environ.* 3, 4–11. doi: 10.1890/1540-9295(2005)003[0004:ESASFT]2.0.CO;2

Redfield, A. C. (1972). Development of a New England salt marsh. *Ecol. Monogr.* 42, 201–237. doi: 10.2307/1942263

Rieb, J. T., Chaplin-Kramer, R., Daily, G. C., Armsworth, P. R., Böhning-Gaese, K., Bonn, A., et al. (2017).When, where, and how nature matters for ecosystem services: challenges for the next generation of ecosystem service models. *Bioscience* 67, 820–833. doi: 10.1093/biosci/bix075

Rizzetto, R., and Tosi, L. (2012). Rapid response of tidal channel networks to sea-level variations (Venice Lagoon, Italy). *Glob. Planet. Chang.* 92–93, 191–197. doi: 10.1016/j.gloplacha.2012.05.022

Rova, S., Meire, P., Müller, F., Simeoni, M., and Pranovi, F. (2019). A Petri net modelling approach to explore the temporal dynamics of the provision of multiple ecosystem services. *Sci. Total Environ.* 655, 1047–1061. doi: 10.1016/j.scitotenv.2018.11.184

Santana-Cordero, A. M., Arizab, E., and Romagos, F. (2016). Studying the historical evolution of ecosystem services to inform management policies for developed shorelines. *Environ. Sci. Policy* 64, 18–29. doi: 10.1016/j.envsci.2016.06.002

Schile, L. M., Callaway, J. C., Morris, J. T., Stralberg, D., Parker, V. T., and Kelly, M., (2014). Modeling tidal marsh distribution with sea level rise: evaluating the role of vegetation, sediment, and upland habitat in marsh resiliency. *PLoS ONE* 9:e88760. doi: 10.1371/journal.pone.0088760

Torio, D. D., and Chmura, G. L. (2013). Assessing coastal squeeze of tidal wetlands. *J. Coast. Conserv.* 29, 1049–1061. doi: 10.2112/JCOASTRES-D-12-00162.1

Turner, N., Davidson-Hunt, I., and O'Flaherty, M. (2003). Living on the edge: ecological and cultural edges as sources of diversity for social-ecological resilience. *Hum. Ecol.* 31, 439–461. doi: 10.1023/A:102502 3906459

Walls, S. C. (2018). Coping with constraints: achieving effective conservation with limited resources. *Front. Ecol. Evol.* 6:24. doi: 10.3389/fevo.2018.00024

Wang, R., Liu, K., Ye, X., and Yan, S. (2018). Association between cerebral microbleeds and depression in the general elderly population: a meta-analysis. *Front. Psychiatry*. 9:94. doi: 10.3389/fpsyt.2018.00094

Wu, W., Yang, Z., Tian, B., Huang, Y., and Zhang, T. (2018). Impacts of coastal reclamation on wetlands: loss, resilience, and sustainable management. *Estuar. Coast. Shelf Sci.* 210, 153–161. doi: 10.1016/j.ecss.2018.06.013

Zhao, J., Tuo, Y., Luo, W., He, S., and Chen, Y. (2018). Prognostic and clinicopathological significance of SATB1 in colorectal cancer: a meta-analysis. *Front. Physiol.* 9:535. doi: 10.3389/fphys.2018.00535

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Expert-Based Evaluation of Ecosystem Service Provision in Coastal Reed Wetlands Under Different Management Regimes

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A characteristic feature of lagoons and estuaries along the Baltic Sea is the dominance of reed (*Phragmites australis*) along their coasts. Reed wetlands are ecologically valuable ecosystems and play an important role for nutrient and matter cycling as well as for biodiversity. They provide a broad spectrum of ecosystem services and have been utilized by humans already for centuries. We assess the ecosystem service provision of reed wetlands and analyze how this is affected by different management scenarios and how the results of an expert-based ecosystem service assessment can be used in practice. Because of strong internal gradients and interactions with the surrounding, coastal reed belts show a higher ecosystem service provision compared to homogeneous inland reed. The three different coastal management scenarios are (1) winter harvest of reed, (2) summer harvest of reed, and (3) grazing by livestock. According to the views of 18 involved experts from Lithuania, Poland, and Germany, winter harvest is regarded as the scenario with the lowest conflict potential between nature protection and reed utilization. Experts expect no changes or even slight increases for regulating and cultural services. However, experts see the need to establish a sustainable and regionally anchored winter harvest concept. Summer harvest and grazing entail the risk to change the ecosystem structure and could lead to a shift in vegetation pattern toward short salt marsh grassland. Experts expect a slight decrease in regulating services. In particular, erosion control, biodiversity, and nutrient sequestration are rated controversially. To our experience, these expert-based ecosystem service assessments can support policy implementation (e.g., NATURA 2000, European Water Framework Directive or Marine Strategy Framework Directive). It can serve as a tool that allows stakeholders to visualize trade-offs, analyze patterns and processes at regional scales, and hence facilitate decision-making.

Keywords: ***Phragmites australis*, CICES, transitional waters, ecotones, expert-based assessment, Baltic Sea**

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INTRODUCTION

Historically, wetlands along the Baltic Sea used to be very heterogeneous with a wide range of species due to strong gradients in salinity, climate, or water level fluctuations (Dijkema, 1990). The biodiversity also resulted partly from human interventions: Many Baltic coastal wetlands were traditionally grazed, mown for hay-making, or harvested for construction material. Since the decline of such activities due to economic reasons or nature protection goals, common reed [*Phragmites australis* (Cav). Trin. ex Steud]. has replaced other halophytes in many wetlands and expanded heavily (Dijkema, 1990; Jutila, 2001; Köbbing et al., 2013). *Phragmites* is a perennial grass (family Poaceae) that can grow up to 4 m and overtops most other emergent macrophytes in wetlands such as Typha, Scripus or Spartina (Cronk and Siobhan Fennessy, 2001). Although reed is principally a freshwater plant, it is well-adapted to brackish water conditions because it is able to cope with a wide range of salinities (Karsten et al., 2003; Meriste et al., 2012; Altartouri et al., 2014). Reed wetlands act as bio-engineers of their own environment: They can grow vertically and horizontally by litter accumulation and can trap sediments by buffering wind and wave energy. Reed has thus the potential to sequester nutrients or heavy metals, to stabilize soils, or to provide habitats in urban or industrial areas where many plants would not thrive otherwise (Kiviat, 2013; Karstens et al., 2016). However, reed also tends to form near-monocultures with only few accompanying species and thereby limits biodiversity at the landscape scale (Prach and Pyšek, 1994; Wanner, 2009; Sweers et al., 2013).

The benefits that reed systems deliver to human well-being can be regarded as ecosystem services (ESs). ESs are defined as the tangible and intangible goods from nature's processes and functions to humans (Millennium Ecosystem Assessment, 2005). The concept has been increasingly used as a holistic approach to support management and decision-making processes (Baker et al., 2013; Posner et al., 2016; Bouwma et al., 2018; Geneletti et al., 2018). ES analysis allows one to disentangle complex interdependencies in socio-ecologic systems (Bouwma et al., 2018) and brings a more sustainable perspective into decision-making and policy outputs (Geijzendorffer et al., 2017). To achieve both human well-being and nature conservation, it is important to understand the dynamics and relationships (trade-offs) of ESs (Raudsepp-Hearne et al., 2010; Daw et al., 2015; Renard et al., 2015; Geneletti et al., 2018). In particular, the analysis of trade-offs has gained attention in policy and decision-making processes (Bennett et al., 2015; Bennett and Chaplin-Kramer, 2016). To assess the impact of management options in ESs provision, expert-based matrix approaches (e.g., Burkhard et al., 2012; Schernewski et al., 2018) can be used for their simplicity. Such approaches can easily be integrated in a stakeholder meeting, and the results can be used as a starting point to extract valuable information that can eventually influence the implementation and design of policies and management approaches. While ES provision is fairly well-studied in seagrass meadows, mangroves, or freshwater wetlands (e.g., Bowden, 1987; Moore et al., 1994; Ewel et al., 1998; Reddy et al., 1999; Moberg and Rönnbäck, 2003; Holmer et al., 2006; Deborde et al., 2008; Delgard et al., 2013), very few studies have addressed ESs in coastal wetlands colonized by *Phragmites*, and to our knowledge, no studies so far investigated the impact of different management options (e.g., grazing, reed harvest) on ES provision.

Main research questions for this study are as follows: (1) How does ES provision differ in transitional and homogeneous reed systems? (2) How do different management scenarios impact the ES provision in reed wetlands along the Baltic Sea? In order to approach these questions, different methods were applied: In a first step, the ESs based on the CICES v5.1 were assessed for homogeneous reed wetlands around shallow inland waters and transitional reed belts along coastlines (e.g., Baltic Sea) by the authors. In a second step, an expert-based ES assessment was applied in order to evaluate changes in service provision due to three different management scenarios: (1) winter harvest, (2) summer harvest, and (3) grazing. Both steps were accompanied by an extensive literature study to allow a diverse discussion of the authors' and experts' assessments. This study shall test whether ES assessments can be applied in facilitating and visualizing management decisions in transitional systems like coastal reed belts.

MATERIALS AND METHODS

Study Site: Transitional Reed Wetlands Along the Southern Baltic Sea

Large areas of the southern Baltic coastline are dominated by *P. australis* (Cav). Trin. ex Steud (**Figure 1**). These coastal reed wetlands are transitional systems that possess stronger internal gradients than homogeneous reed areas around shallow inland waters (see **Figure 2**). Various abiotic stress factors impact ecological gradients and thus vegetation patterns, *inter alia* salinity, flooding, desiccation, erosion, ice scouring, nutrient availability, or human activities such as livestock grazing in wetlands (Wanner, 2009). Several studies showed that flooding seems to be the most controlling factor for species distribution and diversity (Gough et al., 1994; Sanchez et al., 1996; Grace and Jutila, 1999; Jutila, 2001). The interior zone of reed wetlands that borders the hinterland is rarely flooded, and *Phragmites* is often accompanied by plant species such as Calystegia, Urticaceae, *Trifolium fragiferum*, or *Crataegus monogyna*. In wetter and more saltwater influenced areas, *Aster tripolium*, *Carex* spp., or *Bolboschoenus maritimus* occurs besides Phragmites. In the fringe zone with permanent high water levels, submerged macrophytes such as *Stuckenia pectinata*, *Potamogeton* spp., and *Chara* sp. can grow alongside *Phragmites*. The zone in between interior and fringe is often characterized by dense monocultures of *Phragmites* (www.umweltkarten.mv-regierung.de; Paulson and Raskin, 1998; Berthold et al., 2018). Values for salt tolerance of *P. australis* vary in different studies, e.g., up to 6‰ (Raabe, 1981; Jeschke, 1987), 13‰ (Ranwell, 1972), 15–20‰ (Esselink et al., 2000), 5–25‰, and even up to 60‰ for individual clones (Lissner and Schierup, 1997). However, even if low salinities are not limiting *Phragmites* occurrence, it still affects productivity and plant performance (Hellings and Gallagher, 1992) and above salinities of 5‰ growth rates and leaf production decline (Lissner and Schierup, 1997). Besides salinity, limiting factors for reed along the southern Baltic coast are waves and other mechanical stressors such as ice scouring or wild boar grubbing and deer grazing (Krisch, 1989, 1992; Dijkema, 1990; Puurmann et al., 2002; Wanner, 2009).

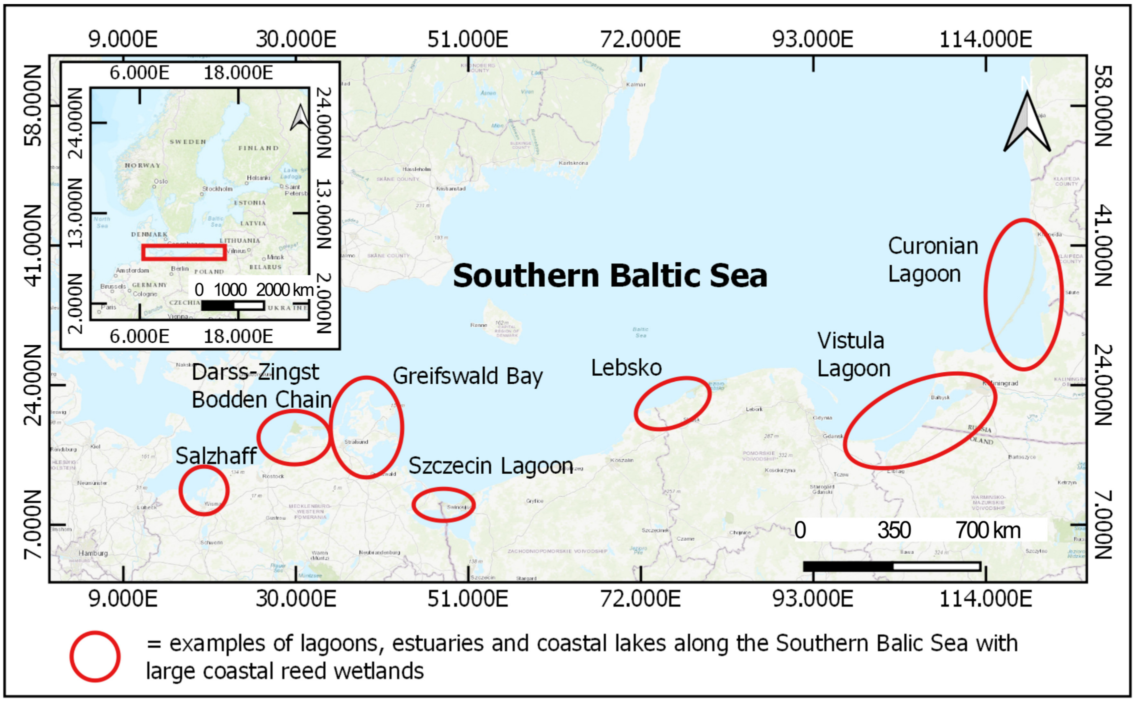

FIGURE 1 | Lagoons, estuaries, and coastal lakes along the southern Baltic coastline are dominated by *Phragmites australis* (Cav). Trin. ex Steud. (common reed). (Background data ESRI Topo Map).

Comparison of Ecosystem Service Provision in Different Reed Wetland Types

Most studies about ES provision in reed wetlands focus firstly only on inland reed wetlands, and secondly, they cover mainly regulating services such as erosion control or nutrient dynamics, but do not take into account cultural services such as the role of reed to coastal heritage, the landscape aesthetics, and values for tourism and recreation. As a consequence, we included not only all sections (regulating–provisioning–cultural) in our study but compared also the ESs potential and use for two different reed wetland types: homogeneous reed wetlands around shallow inland waters (e.g., Neusiedler See) vs. transitional reed belts along coastlines (e.g., Baltic Sea; **Figure 1**). An extensive literature research about ES provision in transitional reed wetlands along coastlines was conducted to allow a complex discussion of the results.

According to Burkhard et al. (2012), ES potential refers to the maximum potential yield of an ES in a spatial unit. ES use (generally known as flow) is the actual use of ES over a period of time. Our aim is to evaluate whether differences in ES potential exist between these two types of reed. We chose a qualitative matrix-like approach similar to Burkhard et al. (2012), common and widely used in the research field of ES, to assess potential and use of ES. To define which services to tackle, we screened through the new version of the Common International Classification on Ecosystem Services, CICES v5.1 (Haines-Young and Potschin, 2018) and discussed based on our background knowledge and the conducted literature study which ESs are relevant in reed wetlands. The CICES classification was chosen for its wide use in ES assessments and because it is the "official" classification used in EC. Services such as cultivated terrestrial plants grown for nutritional purposes were excluded, and the CICES list was narrowed down to 30 ESs relevant for reed wetland (see **Table 1**). Each service was then assessed by us regarding the potential (in percentage) of ES provision for the two reed types (transitional and homogeneous). We used six categories: 0% (no potential), 1–20% (slight potential), 21–40% (considerable potential), 41–60% (medium potential), 61–80% (high potential), and 81–100% (very high potential). The highest potential (100%) was defined having in mind an ecological system that could deliver the maximum provision of each service. The last step was then to assess the real use of each ES also for the two reed types. The use is defined as a percentage of the potential that is currently being exploited: 0% (no use), 1–20% (slight use), 21–40% (considerable use), 41–60% (medium use), 61–80% (high use), and 81–100% (very high use). We, the authors, belong to different institutions and have distinct academic backgrounds ranging from geography, marine ecology, marine biology and conservation, economics, and coastal and marine management. Working in different fields of research, we have different expertise in the topic of ES.

Expert-Based Ecosystem Services Assessment

To understand how different management scenarios could potentially influence the provision of ES, an expert-based approach was used, similar to Schernewski et al. (2018). During

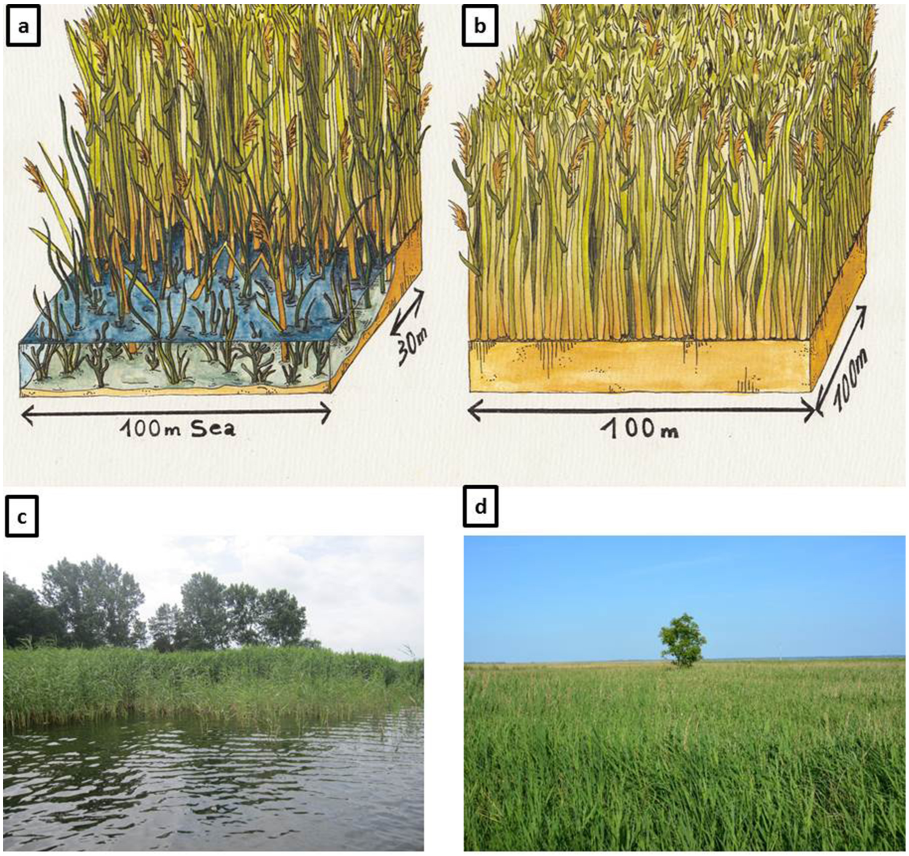

FIGURE 2 | Schematic representation and photos of transitional reed wetlands along coastlines **(a,c)** and homogeneous reed wetlands around shallow inland waters **(b,d)**. (Drawing by Ronja Trübger, with permission).

the cross-border workshop "Coastal biomass: Combining nutrient reduction in coastal waters with blue-growth opportunities" (14th of November 2018, Wieck, Germany) a total of 18 invited experts from the field of coastal management were asked to conduct an ES assessment. The Baltic Lagoons Network (BALLOON, www.balticlagoons.net) as well as a stakeholder mapping conducted within the Interreg South Baltic Project LiveLagoons helped to identify relevant stakeholders. Invited experts were representatives of science institutions (10), state authorities (3), and NGOs (5) and came mostly from Germany (7), Poland (6), or Lithuania (4). Three scenarios were presented to the expert audience: (1) winter harvest in coastal reed wetlands, (2) summer harvest, and (3) grazing by livestock. Reed is a wetland plant that has been utilized by man since ancient times. Harvested reed can be used for a variety of products, *inter alia* as insulation material for walls or as roofing material when harvested in winter, as energy source (combustion, biogas, biofuel), or as fodder and fertilizer when harvested in summer (Köbbing et al., 2013). However, harvest and grazing activities are declining nowadays due to economic reasons or nature protection (Wanner, 2009; Köbbing et al., 2013). In nature conservation, two diverging concepts exist: the "wilderness" concept, where no human intervention shall take place, vs. the "biodiversity" concept, where human management aims at reaching pre-fixed goals such as high species richness or maintaining target communities (Kiehl and Stock, 1994; Bakker et al., 1997; Wanner, 2009). We asked ourselves whether a conflict between reed utilization and nature protection exists *per se*.

The experts were asked to give their opinion on how the different management scenarios [(1) winter harvest, (2) summer harvest, and (3) grazing] impact the ES provision in reed wetlands along the Baltic Sea. Information regarding background knowledge on wetland functioning and nationality were collected from the experts and included in the results (see **Tables 4–6**). In order to reduce the duration of the assessment during the workshop to <2 h and thereby ensure the motivation of participants, the number of services was shortened to a total of 14 services (see **Table 4**). The services were described using a less technical and more user-friendly language, and indicators were used to give examples for each service. Each management scenario (winter harvest, summer harvest, and grazing) was presented with one PowerPoint slide describing the process and subsequent utilization of reed. Photos were shown additionally to visualize management scenarios (e.g., harvest machinery). The experts were asked to choose a category regarding the changes

TABLE 1 | Results of the authors' ecosystem service assessment for regulating services in transitional and homogeneous reed wetlands.

Section	Class	Potential MEAN (potential compared to possible maximum)		Use MEAN (% of potential)	
		Transitional reed (%)	Homogeneous reed (%)	Transitional reed (%)	Homogeneous reed (%)
R&M	Bio-remediation by micro-organisms, algae, plants, and animals	61–80	41–60	61–80	61–80
R&M	Filtration/sequestration/storage /accumulation by micro-organisms, algae, plants, and animals	61–80	41–60	61–80	61–80
R&M	Smell reduction	1–20	1–20	41–60	41–60
R&M	Noise attenuation	1–20	1–20	41–60	41–60
R&M	Visual screening	21–40	21–40	41–60	61–80
R&M	Control of erosion rates	61–80	21–40	61–80	81–100
R&M	Buffering and attenuation of mass movement	61–80	41–60	61–80	81–100
R&M	Hydrological cycle and water flow regulation (Including flood control, and coastal protection)	61–80	41–60	61–80	61–80
R&M	Wind protection	61–80	61–80	61–80	61–80
R&M	Seed dispersal	41–60	21–40	61–80	61–80
R&M	Maintaining nursery populations and habitats (Including gene pool protection)	61–80	21–40	61–80	81–100
R&M	Pest control (including invasive species) and disease control	41–60	21–40	61–80	61–80
R&M	Weathering processes and their effect on soil quality	41–60	41–60	61–80	61–80
R&M	Decomposition and fixing processes and their effect on soil quality	61–80	41–60	61–80	81–100
R&M	Regulation of the chemical condition of fresh- and salt -waters by living processes	61–80	41–60	61–80	61–80
R&M	Regulation of temperature and humidity, including ventilation and transpiration	41–60	41–60	81–100	81–100

that each management scenario might have on ES provision compared to an unmanaged coastal reed wetland. The scale ranges from −3 to 3 where −3 (high), −2 (medium), and −1 (low) represent a decrease in services provision, 0 no change in provision and +3 (high), +2 (medium), and +1 (low) represent an increase in services provision.

RESULTS AND DISCUSSION

Comparison of Ecosystem Services Provision by Transitional and Homogeneous Reed Wetlands

Regulating Services

The potential of most regulating services is considered to be higher in transitional reed wetlands than in homogeneous reeds (**Table 1**). Our views did not differ significantly and standard deviation was low.

The potential for the regulation of baseline flows and extreme events (e.g., erosion control) is regarded as high in transitional reed, while in homogeneous wetlands, it is only considerable to medium (**Table 1**). This is supported by the scientific literature that emphasizes the capability of coastal wetlands to reduce impact forces at the sea and land side (Möller et al., 2011; Duarte et al., 2013; Karstens et al., 2015a,b). Reed stems are flexible, and their "bending stiffness" (Ostendorp, 1995) enables the plant to cause high drag forces and attenuate waves (Möller et al., 2011). How the plants impact erosion regulation depends strongly on the location within the wetland: Dense *Phragmites* stands in the interior zone effectively suppress particle transport even during heavy winter storms. Wind attenuation profiles in coastal reed beds showed that wind speed at the sediment surface was <10% of that measured at 2-m height (Karstens et al., 2015b). In the fringe zone bordering the sea, waves and water flow are the dominant impact forces. Möller et al. (2011) compared wave height attenuation in a sheltered reed site at the southern Baltic Sea (attenuation of 2.6% at the transition from open water to reed vegetation) with an exposed site (attenuation of 11.8%) and showed that reed plant morphology and stem density are important. Vegetation density and stem width were also responsible for the reduction of turbulent kinetic energy from the sea toward the inner part of reed wetlands (Karstens et al., 2015a). Also, the large reed rhizome network supports shoreline stabilization (Ostendorp, 1993), but the ability to trap and accumulate sediment and thereby to

TABLE 2 | Results of the authors' ecosystem service assessment for provisioning services in transitional and homogeneous reed wetlands.

Section	Class	Potential MEAN (potential compared to possible maximum)		Use MEAN (% of potential)	
		Transitional reed (%)	Homogeneous reed (%)	Transitional reed (%)	Homogeneous reed (%)
P	Animals reared for nutritional purposes	1–20	21–40	1–20	1–20
P	Fibers and other materials from reared animals for direct use or processing	21–40	1–20	1–20	0
P	Wild plants used for nutrition	1–20	21–40	0	0
P	Fibers and other materials from wild plants for direct use or processing	41–60	61–80	21–40	21–40
P	Wild plants used as a source of energy	21–40	41–60	0	1–20
P	Wild animals used for nutritional purposes	41–60	41–60	21–40	21–40
P	Fibers and other materials from wild animals for direct use or processing	21–40	21–40	0	0

change the bathymetry is of higher importance for shoreline protection (Duarte et al., 2013).

Also, the potential for the mediation of wastes or toxic substances and the regulation of soil quality is assumed to be higher in transitional reed areas than in homogeneous areas (**Table 1**). Processes such as filtration, sequestration, storage, accumulation, decomposition, and fixing by plants and microorganisms in transitional reed wetlands are important ESs. Nutrient uptake in *Phragmites* is larger than in many other wetland plants due to the high biomass (Wanner, 2009; Berthold et al., 2018). During growth in spring and early summer, large amounts of nutrients are incorporated in the aboveground biomass (Schieferstein, 1999; Berthold et al., 2018). In autumn, the majority of nutrients is transported back into the rhizomes and stored belowground during winter (Ostendorp, 1993). Peat formation is an important contribution to nitrogen and phosphorus deposition, and for the coastal *Phragmites* peatland Karrendorfer Wiesen in Mecklenburg-Vorpommern, a nitrogen deposition of 80 kg N ha^{-1} $year^{-1}$ at a predicted peat growth of 1.5 mm $year^{-1}$ was calculated (Lampe and Wohlrab, 1996; Wanner, 2009). Also, carbon burial in peat is an important contribution to the reduction of atmospheric CO_2 (Succow and Joosten, 2001; Chmura et al., 2003; Choi and Wang, 2004; Andrews et al., 2006). Buczko et al. (under review) measured carbon stocks down to 1-m depth in two coastal *Phragmites* wetlands at the southern Baltic Sea, and values ranged from 10 to almost 60 kg C m^{-2}, with lowest carbon contents in the fringe zones due to lower biomass production. Averaged over all wetland zones, carbon stocks were 16 and 39 kg C m^{-2} at the two wetland sites and comparable to the worldwide average for salt marshes of 25 kg C m^{-2} (Pendleton et al., 2012). Lampe and Wohlrab (1996) calculated a carbon fixation of 5.1 t CO_2 ha^{-1} $year^{-1}$ for the de-embanked coastal peatland Karrendorfer Wiesen in Mecklenburg-Vorpommern, which is dominated by *Phragmites*. However, the authors did not include the possible emission of CH_4 in their net carbon sequestration estimations, which can occur under anaerobic conditions in waterlogged soils (Succow and Joosten, 2001; Wanner, 2009). While in many terrestrial wetlands, carbon sequestration is partially offset by methane emission from plant decomposition, methanogenesis can be inhibited by sulfates in coastal wetlands, thus reducing greenhouse gas emissions (Howe et al., 2009).

In our view, the potential to maintain habitats and nursery populations is high in transitional reed belts, whereas it is only considerable in homogeneous inland reed areas (**Table 1**). In homogeneous wetlands around shallow inland waters, reed tends to form near monocultures with only few accompanying species. In transitional systems, habitat gradients are more pronounced and *Phragmites* might be accompanied by Calystegia, Urticaceae, *T. fragiferum*, or *C. monogyna* in the interior zone or by submerged macrophytes such as *S. pectinata*, *Potamogeton* spp., and *Chara* sp. in the fringe zone. However, the zone in between is also often characterized by dense reed monocultures (Paulson and Raskin, 1998; Berthold et al., 2018). Coastal *Phragmites* wetlands are important (breeding) habitats and refugees for birds such as bittern (Botaurinae), red-necked grebe (*Podiceps*), reed warbler (*Acrocephalus scirpaceus*), or water rail (*Rallus aquaticus*); for insects such as the Flame Wainscot (*Senta flammea*), large copper (*Lycaena dispar*), or dragonflies (*Aeshna isosceles*); and for mammals such as water shrew (*Neomys fodiens*), otter (Lutrinae), raccoon dogs (Caninae), deer (*Dama dama*), or wild boars (*Sus scrofa*) [LUNG (Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern), 2003].

The actual use of the potential of the abovementioned regulating services was seen as mostly high in the homogeneous reed wetlands, with some even very high (**Table 1**). This shows that although reed wetlands offer a high potential for regulating services, the demand can exceed a sustainable supply.

Provisioning Services

The highest potential has the utilization of reed stems for direct use or processing (e.g., roof thatching, insulation material) in homogeneous reed wetlands. Also, the potential to use reed as an energy source (e.g., combustion, biofuel, biogas) is considered higher in homogeneous than in transitional reed wetlands (**Table 2**). In homogeneous areas, harvest with heavy machinery is easier than in transitional systems with stronger gradients regarding water level as well as species composition.

A medium potential exists for the use of wild animals for nutritional purposes (**Table 2**). Currently, mainly wild boars are hunted in reed wetlands along the Baltic Sea. Wild boars are omnivores and find plenty of food there, e.g., young reeds, insects, or small animals. During summertime, they benefit from the shading and cooling effects inside the dense reed stands. Hunters report that they often find the nests for the young boars, indicating that reed areas are also a popular place for birth (Task force "sustainable stock reduction wild boars Greifswald-Vorpommern", personal communication). In some regions, wild boars have become a nuisance, causing major destructions to agriculture and infrastructure. As a response, nature conservation authorities have revised the permit procedure and now allow the cutting of "hunting aisles" into reed wetlands to facilitate the hunt on wild boars (Merkblatt Schussschneisen StALU Vorpommern).

All provisioning services are currently only slightly or considerably used, some even not at all (**Table 2**). This was different in the past, where harvest of reed stems or grazing of cattle in wetlands was very common (Köbbing et al., 2013). Reed-thatched houses are still popular, but the majority of the reed for roof thatching is currently imported [LUNG (Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern), 2017]. The underutilization of the potential can be explained by the strict nature protection status of reed wetlands in Germany and also in other countries along the Baltic Sea. They are legally protected biotopes. Reed harvest, grazing activities, or other interventions in the ecosystem have to take into account biodiversity concerns and require specific approvals from the responsible federal nature conservation authority [LUNG (Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern), 2003].

Cultural Services

The potential of cultural services is considered to be higher in transitional than in homogeneous reed wetlands (**Table 3**). Reed has a great cultural importance in the Baltic Sea region and its utilization has a long tradition, explaining that the authors valued the importance for heritage as high (**Table 3**). Roofs thatched with reed are characteristic in the coastal regions. Locals appreciate the use of reed as construction material, and it forms part of their regional identity (Stoll-Kleemann, 2015).

However, not only the utilization of reed as a resource has a cultural value, but also the landscape itself. The recreation potential through passive or observational activities is regarded as high in transitional systems while only a medium potential exists in homogeneous reed wetlands (**Table 3**). Bird-watching and active interactions such as fishing and canoeing along coastal *Phragmites* wetland are popular recreational activities. However, reed wetlands are considered less aesthetic than salt meadows (Stoll-Kleemann, 2015). Semistructured interviews and group discussions in 2012/2013 with people living at the Darss-Zingst Bodden Chain showed that reed areas were only considered as "beautiful" when growing in moderation. If they expand and become dominant, e.g., due to mowing and grazing prohibitions, people start to perceive only the negative aspects such as hindering the view to the bay, reducing biodiversity, and increasing the abundance of wild boars (Stoll-Kleemann, 2015). However, perceiving something as aesthetically pleasant is very subjective and individualistic. This is also reflected in our assessment, where one author regards the aesthetic potential as very high, whereas the other two authors viewed it as only moderately aesthetically pleasant.

Expert-Based Ecosystem Service Assessment of Different Management Scenarios

In transitional reed wetlands along coastlines, the potential for regulating and cultural services is regarded as moderate to high while the potential for provisioning services remains between slight to medium. According to Burkhard et al. (2014) the provision of crops, bioenergy, or fibers is not relevant in marshes. The potential is low, as well as the current use, which is based on the fact that nature conservation agencies heavily restrict the utilization of coastal reed. For harvest or grazing activities, specific approvals are needed. This was different in the past when not only summer and winter harvest but also grazing by cattle was very common in Baltic wetlands (Wanner, 2009).

Scenario 1: Winter Harvest

For the winter harvest scenario in reed wetlands, experts expected the highest increases for biomass utilization (e.g., reed as construction material, insulation material, pulp, or paper), for bioenergy, and for culture and heritage (**Table 4**). Assessments in the section "regulation and maintenance" reflected the very contrasting views of different experts (ranging from −2 to +3) but were less negative than for the summer harvest scenario (**Annex I**). During the discussion, the experts pointed out that for the assessment of regulating services, it is important to have more detailed information about the winter harvest scenario, e.g., the exact time of harvest or the machinery used. Harvest in November before the winter storm season could lead to a decrease in erosion control and mass stabilization, while harvest in February would not impact service provision in their eyes.

Winter harvest has a long tradition along the Baltic Sea (Köbbing et al., 2013). The amount of harvested reed during winter time ranges between 3.6 and 15 t dry mass h year^{-1} (Rodewald-Rudescu, 1974; Knoll, 1986; Timmermann, 2009; Dahms et al., 2015). Most commonly, winter reed is used for roof thatching. First references for the use of reed for roof thatching along the coast of the North and Baltic Sea date back to the last ice age (Schaatke, 1992). Along the coast, reed and straw were often the only materials available for roofing until the late 1800s (Iital et al., 2012). With the yield from 1 ha reed wetland, approximately up to 100 m^2 of roof can be thatched (Schaatke, 1992; Haslam, 2009). Today, the annual reed demand for thatching often exceeds the supply (Köbbing et al., 2013) and 80% of the reed for roof thatching is currently imported [LUNG (Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern), 2017]. Reed can be used as an industrial material, such as for the construction of garden fences and indoor furnishings (such as blinds, floor, and wall coverings), as an insulation material, and for bio-based plastics or the cellulose for pulp and paper production (Köbbing et al., 2013). Some utilizations of harvested reed have become almost forgotten and less popular today, e.g., the manufacture of schnapps, coffee, and boats (Holzmann and Wangelin, 2009; Köbbing et al., 2013).

TABLE 3 | Results of the authors' ecosystem service assessment for cultural services in transitional and homogeneous reed wetlands.

Section	Class	Potential MEAN (potential compared to possible maximum)		Use MEAN (% of potential)	
		Transitional reed (%)	Homogeneous reed (%)	Transitional reed (%)	Homogeneous reed (%)
C	Characteristics of living systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions	41–60	21–40	21–40	1–20
C	Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions	61–80	41–60	61–80	61–80
C	Characteristics of living systems that enable scientific investigation or the creation of traditional ecological knowledge	61–80	41–60	41–60	41–60
C	Characteristics of living systems that enable education and training	41–60	41–60	21–40	21–40
C	Characteristics of living systems that are resonant in terms of culture or heritage	61–80	41–60	1–20	1–20
C	Characteristics of living systems that enable aesthetic experiences	41–60	41–60	61–80	61–80
C	Characteristics or features of living systems that have an existence value	41–60	41–60	61–80	61–80

Harvest during winter compared to summer harvest reduces conflicts with nature protection (e.g., bird breeding), and harvest costs are lower when the wetland soils are frozen (Köbbing et al., 2013). Winter cutting can increase culm density and overall aboveground biomass production of *Phragmites* in the following vegetation period (Ostendorp, 1999). Also, Hansson and Graneli (1984) and Huhta (2009) noted an increase in reed vitality after winter harvest. According to Günther et al. (2015), reed harvest has no negative effect on greenhouse gas balances on a timescale of a few years; however, the long-term effects are still under investigation, and once results are available, they should be incorporated into the sustainable harvest concept for coastal wetlands. Reed harvest diminishes insect and fungus populations and decreases oxygen consumption by decomposer organisms due to the biomass removal (Hansson and Graneli, 1984; Brix, 1988; Schäfer and Wichtmann, 1999; Hansson and Fredriksson, 2004; Kask et al., 2007; Köbbing et al., 2013). However, nutrient removal efficiency is minimal during winter harvest with phosphorus concentrations in the aboveground plant material with 1,100 mg P m^{-2} in November down to 100 mg P m^{-2} in March (Berthold et al., 2018). This is reflected in the experts' results, which show a higher increase regarding nutrient accumulation for the summer harvest scenario than for the winter harvest scenario (**Table 4** vs. **Table 5**). Reed harvest impacts cultural, social, and economic aspects. In particular, Lithuanian experts expect a high increase (+3) for culture and heritage (**Table 4**). Roofs thatched with reed are characteristic along the Baltic coast. Many of those houses are even under historic preservation underlining their cultural importance (FAZ, 2016).

Scenario 2: Summer Harvest

Reed harvested during summer has a higher nutrient content than winter biomass, and it is usually utilized as fodder or fertilizer or for biogas production with the advantage that the land of coastal reed wetlands seldom competes with food production (Köbbing et al., 2013). Productivity surveys showed that 6.5–23.8 t dry mass ha^{-1} $year^{-1}$ of reed could be harvested during summertime (Steffenhagen et al., 2008; Schulz et al., 2011). It is thus not surprising that the questioned experts of this study saw the highest provision increases for the following services: agriculture (e.g., harvested amount of reed as fodder, straw for stables, fertilizer, or compost) and filtration, sequestration, accumulation, and storage of nutrients (**Table 5**). The assessment of changes in the section "regulation and maintenance" was again very heterogeneous, ranging from −3 to +3 for services such as erosion control, maintaining nursery populations and habitats or local climate regulation (**Annex I**). On average, a low decrease of mass stabilization and local climate regulation is predicted by the experts. Regarding cultural services, on average, no or only low changes were expected for the summer harvest scenario. Experts with only a moderate knowledge on reed wetland functioning saw a higher increase of agricultural services for the summer harvest scenario compared to the assessment of experts with excellent or good knowledge on reed wetlands (**Table 5**).

During the discussion, the experts pointed out that a shift in vegetation patterns and thus ecosystem structure and functions can be introduced by continuous summer harvest over several years. In some areas along the Baltic coast, summer harvest is applied as a nature conservation measure, for example, for the promotion of ground-nesting birds (Köbbing et al., 2013). The

TABLE 4 | Mean values and standard deviation (SD) of expert assessments of the changes in ecosystem service provision for management scenario 1: Winter harvest.

Ecosystem Service	Indicator	Mean (all experts)	SD (all experts)	Mean (experts with excellent/good knowledge)	Mean (experts with moderate knowledge)	Mean (German experts)	Mean (Lithuanian experts)	Mean (Polish experts)
Livestock	e.g., number of animals (e.g., cattles of water buffalos, herd of sheep)	0	1	0	0	0	0	1
Agriculture	e.g., harvested amount of reed as fodder, straw for stables, fertilizer, or compost	1	1	0	1	1	0	1
Biomass utilization: Fibers and other material for direct use or processing	e.g., harvested amount of reed as construction material, insulation material, pulp, or paper	2	1	2	2	2	3	2
Bioenergy	e.g., harvested amount of reed as energy source (combustion, biogas, biofuel)	2	1	2	2	2	3	2
Wild animals and their output	e.g., number of hunted animals (e.g., wild boar, deer)	0	1	0	0	0	0	0
Filtration/sequestration/ storage/accumulation	e.g., nutrient removal efficiency or carbon storage	1	1	1	1	2	1	2
Mass stabilization and control of erosion rates	e.g., sediment accumulation rate and buffer for wind and water energy	0	2	1	0	0	0	1
Maintaining nursery populations and habitats	e.g., biodiversity (Wild plant and animal species richness)	0	1	1	0	0	1	0
Regulation of soil quality	e.g., decomposition rates	0	1	0	0	0	0	0
Local climate regulation	e.g., impacts on temperature and humidity, including ventilation, and transpiration	0	1	0	0	0	0	0
Health, recuperation or enjoyment	e.g., number of visitors looking for enjoyment provided by ecosystems (e.g., view, wildlife, activites)	0	1	0	0	0	0	0
Scientific and educational	e.g., scientific and educational publications, documentaries, exhibitions, nature trails	1	1	1	1	1	0	1
Culture and heritage	e.g., number of reed thatched houses	2	1	2	2	2	3	2
Existence and bequest	Non-use value, preservervation for future generations, protected areas	0	1	0	0	0	0	0

Results were further divided for different groups based on the experts' background knowledge on reed wetland functioning and nationality. Negative numbers represent a decrease in services provision, positive numbers represent an increase, and the number "0" represents no changes. For the individual results of the expert assessment, see ***Annex I****.*

TABLE 5 | Mean values and standard deviation (SD) of expert assessments of the changes in ecosystem service provision for management scenario 2: Summer harvest.

Ecosystem Service	Indicator	Mean (all experts)	SD (all experts)	Mean (experts with excellent/good knowledge)	Mean (experts with moderate knowledge)	Mean (German experts)	Mean (Lithuanian experts)	Mean (Polish experts)
Livestock	e.g., number of animals (e.g., cattles of water buffalos, herd of sheep)	0	1	1	0	0	0	0
Agriculture	e.g., harvested amount of reed as fodder, straw for stables, fertilizer, or compost	2	1	1	2	2	2	2
Biomass utilization: Fibers and other material for direct use or processing	e.g., harvested amount of reed as construction material, insulation material, pulp, or paper	1	1	0	1	1	1	1
Bioenergy	e.g., harvested amount of reed as energy source (combustion, biogas, biofuel)	1	1	1	2	2	1	1
Wild animals and their output	e.g., number of hunted animals (e.g., wild boar, deer)	–1	1	0	–1	–1	–1	0
Filtration/sequestration/ storage/accumulation	e.g., nutrient removal efficiency or carbon storage	2	1	2	2	2	1	2
Mass stabilization and control of erosion rates	e.g., sediment accumulation rate and buffer for wind and water energy	–1	2	0	–1	–1	0	–1
Maintaining nursery populations and habitats	e.g., biodiversity (Wild plant and animal species richness)	0	2	2	–1	0	0	1
Regulation of soil quality	e.g., decomposition rates	0	1	0	0	–1	0	0
Local climate regulation	e.g., impacts on temperature and humidity, including ventilation, and transpiration	–1	2	–1	–1	–1	–1	0
Health, recuperation or enjoyment	e.g., number of visitors looking for enjoyment provided by ecosystems (e.g., view, wildlife, activites)	0	1	1	0	1	2	0
Scientific and educational	e.g., scientific and educational publications, documentaries, exhibitions, nature trails	1	1	1	1	1	1	2
Culture and heritage	e.g., number of reed thatched houses	0	1	0	0	0	0	0
Existence and bequest	Non-use value, preservervation for future generations, protected areas	0	1	0	0	0	0	0

Results were further divided for different groups based on the experts' background knowledge on reed wetland functioning and nationality. Negative numbers represent a decrease in services provision, positive numbers represent an increase, and the number "0" represents no changes. For the individual results of the expert assessment, see ***Annex I****.*

TABLE 6 | Mean values and standard deviation (SD) of expert assessments of the changes in ecosystem service provision for management scenario 3: Grazing.

Ecosystem Service	Indicator	Mean (all experts)	SD (all experts)	Mean (experts with excellent/good knowledge)	Mean (experts with moderate knowledge)	Mean (German experts)	Mean (Lithuanian experts)	Mean (Polish experts)
Livestock	e.g., number of animals (e.g., cattles of water buffalos, herd of sheep)	2	1	2	3	3	2	3
Agriculture	e.g., harvested amount of reed as fodder, straw for stables, fertilizer, or compost	0	1	1	0	1	–1	0
Biomass utilization: Fibers and other material for direct use or processing	e.g., harvested amount of reed as construction material, insulation material, pulp, or paper	–1	1	–1	–1	0	–2	0
Bioenergy	e.g., harvested amount of reed as energy source (combustion, biogas, biofuel)	–1	1	0	–1	0	–2	0
Wild animals and their output	e.g., number of hunted animals (e.g., wild boar, deer)	0	1	0	0	0	–1	1
Filtration/sequestration/storage/accumulation	e.g., nutrient removal efficiency or carbon storage	0	2	0	1	1	1	0
Mass stabilization and control of erosion rates	e.g., sediment accumulation rate and buffer for wind and water energy	–1	1	–1	–1	–1	–2	0
Maintaining nursery populations and habitats	e.g., biodiversity (Wild plant and animal species richness)	1	2	2	0	1	0	1
Regulation of soil quality	e.g., decomposition rates	0	1	0	0	–1	0	1
Local climate regulation	e.g., impacts on temperature and humidity, including ventilation, and transpiration	–1	1	0	–1	–1	0	0
Health, recuperation or enjoyment	e.g., number of visitors looking for enjoyment provided by ecosystems (e.g., view, wildlife, activites)	1	1	1	1	1	2	2
Scientific and educational	e.g., scientific and educational publications, documentaries, exhibitions, nature trails	1	1	2	1	1	2	2
Culture and heritage	e.g., number of reed thatched houses	–1	1	–1	0	0	–2	–1
Existence and bequest	Non-use value, preservervation for future generations, protected areas	0	1	0	0	0	–1	0

Results were further divided for different groups based on the experts' background knowledge on reed wetland functioning and nationality. Negative numbers represent a decrease in services provision, positive numbers represent an increase, and the number "0" represents no changes. For the individual results of the expert assessment, see ***Annex I****.*

possible shift in ecosystem structure made it difficult for the experts to assess the expected increases or decreases in service provision. This is especially true with regard to maintaining nursery populations and habitats as it really depends on target species. Thus, expert ratings were ambiguous regarding habitats and biodiversity (**Annex I**).

Scenario 3: Grazing

For the grazing scenario, an increase in livestock and maintenance of nursery populations and habitats was expected by the experts, as well as an increase in health, recuperation, or enjoyment and in scientific and educational services (**Table 6**).

Grazing has a long tradition in the Baltic Sea region, and until the 1940s, coastal wetlands were usually used for livestock (Wanner, 2009). Continuous grazing in coastal reed wetlands can lead to a shift in vegetation pattern toward short salt marsh grassland, which is preferred by ground-nesting birds (Jeschke, 1987; Esselink et al., 2000; Jutila, 2001; Bernhardt and Koch, 2003; Rannap et al., 2004; Burnside et al., 2007; Wanner, 2009). Once grazing activities stop, reed will quickly re-dominate the area, which often results in a loss of biodiversity and habitats (Esselink et al., 2000; Rannap et al., 2004; Burnside et al., 2007; Wanner, 2009). However, the use of common cattle for reducing spread and growth of reed is only successful, when grazing pressure is kept high (Vulink et al., 2000). This contradicts the nature conservation goal to keep cattle stocking densities low. Further, high grazing intensities might also threaten the nesting success of waders (Müller et al., 2007). A moderate grazing pressure with mosaics of intensively and moderately grazed patches often provides the highest biodiversity benefit (Doody, 2008).

Regarding regulating services, the experts' views were again very contrasting; for example, for the service "maintaining nursery populations and habitats," the individual assessments ranged from −3 to +3 (**Annex I**). This is comparable to the results for the summer harvest scenario. The experts pointed out that more details about temporal and spatial scales are important to evaluate whether the provision of regulating services increases or decreases. Information about grazing pressure (length of grazing season, livestock unit per hectare) and the type of livestock (cattle, sheep, horses, water buffaloes) impact the reed wetland structure (Scherfose, 1993; Kiehl et al., 1996; Kiehl, 1997; Kleyer et al., 2003; Rannap et al., 2004; Doody, 2008; Wanner, 2009). In scientific literature, water buffaloes with their wetland-adopted hooves and grazing behaviors are described as most suitable for conservation purposes (Georgoudis et al., 1999; Wiegleb and Krawczynski, 2010; Wichtmann, 2011; Sweers et al., 2013). A grazing study in brackish coastal reed wetlands by Sweers et al. (2013) showed that grazing by water buffalos successfully reduced the reed dominance and led to a shift toward salt marsh grassland with higher species diversity. Water buffaloes carry out this transformation process already at lower livestock densities than common cattle (Sweers et al., 2013). This supports the observations by Georgoudis et al. (1999), Wiegleb and Krawczynski (2010), and Wichtmann (2011) that water buffaloes have a greater preference for wetland plants. Therefore, they are suitable animals for wetland management especially when it aims at shifting reed monocultures into diverse salt marsh grassland.

Similar to maintaining habitats and nursery populations, the expert assessment was very heterogeneous for the service "filtration/sequestration/storage/accumulation," ranging from −3 to +3 (**Annex I**). Also, the scientific literature offers no clear results whether nutrient retention and peat growth are enhanced or reduced by different grazing regimes (Wanner, 2009). On one hand, reed contributes to peat formation and nutrient accumulation (Schieferstein, 1997; Mitsch and Gosselink, 2000; Succow and Joosten, 2001; Meuleman et al., 2002). A shift from reed wetlands toward salt grasslands could potentially release accumulated nutrients (Huhta, 2007). Also, sedimentation rates are usually lower in intensively grazed salt marshes with shorter vegetation, and thus nutrient deposition would be lower in salt marshes than in reed wetlands (Andresen et al., 1990; Bakker et al., 1997; Kiehl, 1997; Stock et al., 1997; Esselink et al., 1998; Neuhaus et al., 1999). On the other hand, biomass and thus organic matter are directly removed by livestock, and some authors argue that grazing has the potential to increase carbon and nutrient sequestration (Jones and Donnelly, 2004). Furthermore, soil compaction as a result of grazing pressure may lead to more waterlogged soils, resulting in higher denitrification rates of grazed salt marshes (Jensen et al., 1990).

SYNTHESIS AND CONCLUSIONS

Coastal reed belts are transitional systems with pronounced gradients from land to sea. The resulting higher heterogeneity of abiotic factors, such as vegetation structure, salinity, or topography, and a higher spatial biodiversity lead to an increased provision of regulating and cultural ESs, compared to reed wetlands surrounding inland waters. This study deals with the impacts of three different habitat management scenarios on ES provision in coastal reed wetlands: (1) winter harvest, (2) summer harvest, and (3) grazing. If reed utilization—and thus an increase in provisioning services—conflicts with nature protection depends strongly on (a) spatial and temporal scales as well as on (b) the pre-defined set of nature protection goals. For the latter, Natura 2000 management plans with its prefixed target species and habitats are a good example, e.g., designated areas in the Curonian Spit National Park (Lithuania) as well as in the Western Pomerania Lagoon Area National Park (Germany) are supposed to serve as habitats for ground-nesting birds (see **Figure 1**). To restrict the reed dominance in these areas, management intervention that leads to a shift in vegetation toward salt marsh grasslands is necessary. This can be achieved by grazing or by summer harvest of reed. However, the temporal scale determines the success of the intervention: Only if grazing or harvest is carried out continuously every summer for several years can reed be restrained.

Our study contributes to an enhanced knowledge with respect to reed wetland ecosystem functioning. Further, the assessments allow the identification of trade-offs between ESs. These trade-offs serve as a basis to explore the impact of multiple management options. For example, grazing with livestock leads to a reduction of reed area. As a consequence, the provision of regulating services like erosion control and cultural services like heritage (e.g., loss of reed for roof thatching) would decrease. This is just one example of a trade-off that was identified by the ES assessments. The identification of trade-offs is considered as beneficial for decision-making processes (e.g., Seppelt et al., 2013; Howe et al., 2014; Bennett et al., 2015; King et al., 2015). Further, the communication of anticipated trade-offs resulting from different management options is an important prerequisite for successful ecological governance. An example is the Natura 2000 site management: Based on a social network analysis, Manolache et al. (2018) show that productive collaboration between various actors (e.g., law enforcement agencies, NGOs, enterprises) is still low, regardless by whom the protected areas are governed. Simply delegating administration of protected areas to NGOs in order to increase collaborations proved to be insufficient (Manolache et al., 2018). Our evaluation of ES provision under different management regimes can increase the information flow between different actors and thereby improve their cooperation. The inclusion of stakeholder views at an early state can help to identify conflicts and thus contributes to a better acceptance of the taken decisions (Hauck et al., 2013; Ruiz-Frau et al., 2018). Our assessment approach can be easily transferred to other situations, ranging from specific local management demands to conceptual management consideration within an international policy implementation context. An assessment not necessarily results in consensus on management decisions, but the tool highlights topics that are controversial and allows more focused discussions between stakeholders. An example is the need to reduce nutrient loads into coastal waters according to the European Water Framework Directive. Compared to winter harvest, our experts expect a higher nutrient removal efficiency for the summer harvest scenario, due to the higher nutrient concentration in reed biomass. However, if harvest is carried out in summer instead of winter, the experts also assume a decrease with respect to mass stabilization and erosion control. Stronger erosion could lead to a sediment transport into the coastal water and counteract nutrient removal. As our methodological framework relies on a tier-1 ES (qualitative) approach, the results reflect the expert views. For a more comprehensive understanding, specific analysis of regional patterns and processes would be beneficial. This would require the use of more sophisticated tier-2 or tier-3 ES (quantitative) approaches.

Cultural and regulating services are regarded as more important in coastal reed belts than provisioning services. This does not mean *per se* that reed utilization has to be in conflict with nature protection or diminish the other services. For the winter harvest scenario, experts expect no changes or even slight increases for regulating and cultural services. Roofs thatched with reed have a long tradition along the Baltic Sea and are part of the regional identity and heritage. However, most of the reed used for roof thatching has to be imported nowadays. The regional supply of winter reed for roof construction could enhance the regional bond and offer an income opportunity in economically weak regions. Winter harvest can be in line with nature protection goals and can be carried out in a sustainable way: A rotating system should be applied, where each year another area is harvested. A "greenbelt" between the terrestrial hinterland and the coastal wetland without harvest should always remain to maintain the erosion control also immediately after cutting in wintertime. Sensitive areas (e.g., steep topography and vulnerable to erosion) should remain untouched. Timing of harvest should take into account the regional climate (e.g., in February after winter storms). A transferability of this recommendation to other areas outside the Baltic Sea is difficult because reed-thatched houses are part of the regional identity and markets for harvested reed biomass might not exist in other regions. However, the tool itself—the assessment of ESs under changing management scenarios—is transferable and universally applicable.

DATA AVAILABILITY

All datasets generated for this study are included in the manuscript and the **Supplementary Material**.

AUTHOR CONTRIBUTIONS

SK developed the article concept, took care of the data analyses, and did most of the article writing. MI provided the assessment tool, took part in the assessment, supported the analysis, and commented on the paper. GS supported the article concept development, the writing, and the analysis and took part in the assessment.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2019.00063/full#supplementary-material

Annex I | Overview of the single expert assessments for all three scenarios as well as mean values and standard deviation (SD).

REFERENCES

Altartouri, A., Nurminen, L., and Jolma, A. (2014). Modeling the role of the close-range effect and environmental variables in the occurrence and spread of Phragmites australis in four sites on the Finnish coast of the Gulf of Finland and the Archipelago Sea. *Ecol. Evol.* 4, 987–1005. doi: 10.1002/ece3.986

Andresen, H., Bakker, J. P., Brongers, M., Heydemann, B., and Irmler, U. (1990). Long-term changes of salt marsh communities by cattle grazing. *Plant Ecol.* 89, 137–148. doi: 10.1007/BF00032166

Andrews, J. E., Burgess, D., Cave, R. R., Coombes, E. G., Jickells, T. D., Parkes, D. J., et al. (2006). Biogeochemical value of managed realignment, Humber estuary, UK. *Sci. Total Environ.* 371, 19–30. doi: 10.1016/j.scitotenv.2006.08.021

Baker, J., Sheate, W. R., Phillips, P., and Eales, R. (2013). Ecosystem services in environmental assessment — Help or hindrance? *Environ. Impact Assess. Rev.* 40, 3–13. doi: 10.1016/j.eiar.2012.11.004

Bakker, J. P., Esselink, P., Van der Wal, R., and Dijkema, K. S. (1997). "Options for restoration and management of coastal salt marshes in Europe," in *Restoration Ecology and Sustainable Development*. eds K. M. Urbanska, N. R. Webb, and P. J. Edwards (Cambridge: Cambridge University Press), 286–322.

Bennett, E. M., and Chaplin-Kramer, R. (2016). Science for the sustainable use of ecosystem services. *F1000Res.* 5:2622. doi: 10.12688/f1000research.9470.1

Bennett, E. M., Cramer, W., Begossi, A., Cundill, G., Díaz, S., Egoh, B. N., et al. (2015). Linking biodiversity, ecosystem services, and human well-being: three challenges for designing research for sustainability. *Curr. Opin. Environ. Sustain.* 14, 76–85. doi: 10.1016/j.cosust.2015.03.007

Bernhardt, K. G., and Koch, M. (2003). Restoration of a salt marsh system: temporal change of plant species diversity and composition. *Basic. Appl. Ecol.* 4, 441–451. doi: 10.1078/1439-1791-00180

Berthold, M., Karstens, S., Buczko, U., and Schumann, R. (2018). Potential export of soluble reactive phosphorus from a coastal wetland in a cold-temperate lagoon system: buffer capacities of macrophytes and impact on phytoplankton. *Sci. Total Environ.* 616-617, 46–54. doi: 10.1016/j.scitotenv.2017.10.244

Bouwma, I., Schleyer, C., Primmer, E., Winkler, K. J., Berry, P., Young, J., et al. (2018). Adoption of the ecosystem services concept in EU policies. *Ecosyst. Serv.* 29, 213–222. doi: 10.1016/j.ecoser.2017.02.014

Bowden, W. B. (1987). The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry* 4, 313–348. doi: 10.1007/BF02187373

Brix, H. (1988). Gas exchange through dead culms of reed, *Phragmites australis* (Cav). *Trin. Ex Steudel. Aquat. Bot.* 35, 81–98. doi: 10.1016/0304-3770(89)90069-7

Burkhard, B., Kandziora, M., Hou, Y., and Müller, F. (2014). Ecosystem service potentials, flows and demands ? Concepts for spatial localisation, indication and quantification. *Landsc. Online* 34, 1–32. doi: 10.3097/LO.201434

Burkhard, B., Kroll, F., Nedkov, S., and Müller, F. (2012). Mapping ecosystem service supply, demand and budgets. *Ecol. Indic.* 21, 17–29. doi: 10.1016/j.ecolind.2011.06.019

Burnside, N. G., Joyce, C. B., Puurmann, E., and Scott, D. M. (2007). Use of vegetation classification and plant indicators to assess grazing abandonment in coastal wetlands. *J. Veg. Sci.* 18, 645–654. doi: 10.1111/j.1654-1103.2007.tb02578.x

Chmura, D. G. L., Anisfeld, S. C., Cahoon, D. R., and Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Glob. Biogeochem. Cycles* 17:1111. doi: 10.1029/2002GB001917

Choi, Y., and Wang, Y. (2004). Dynamics of carbon sequestration in a coastal wetland using radiocarbon measurements. *Glob. Biogeochem. Cycles* 18:GB4016. doi: 10.1029/2004GB002261

Cronk, J. K., and Siobhan Fennessy, M. (2001). *Wetland Plants: Biology and Ecology, 1st Edn.* CRC Press, 492.

Dahms, T., Oehmke, C., Kowatsch, A., Abel, S., Wichmann, S., Wichtmann, W., et al. (2015). *Paludi-Pellets-Broschüre - Halmgutartige Festbrennstoffe aus nassen Mooren*. Greifswald: Universität Greifswald, Partner im Greifswald Moor Centrum, 73.

Daw, T. M., Coulthard, S., Cheung, W. W. L., Brown, K., Abunge, C., Galafassi, D., et al. (2015). Evaluating taboo trade-offs in ecosystems services and human well-being. *Proc. Natl. Acad. Sci. U.S.A.* 112, 6949–6954. doi: 10.1073/pnas.1414900112

Deborde, J., Abril, G., Mouret, A., Jézéquel, D., Thouzeau, G., Clavier, J., et al. (2008). Effects of seasonal dynamics of a *Zostera noltii* meadow on phosphorus and iron cycles in a tidal mudflat (Arcachon Bay, France). *Mar. Ecol. Prog. Ser.* 355, 59–71. doi: 10.3354/meps07254

Delgard, M. L., Deflandre, B., Deborde, J., Charbonnier, C., and Anschutz, P. (2013). Changes in nutrient biogeochemistry in response to the regression of Zostera noltii meadows in the Arcachon bay (France). *Aquat. Geochem.* 19, 241–259. doi: 10.1007/s10498-013-9192-9

Dijkema, K. S. (1990). Salt and brackish marshes around the Baltic Sea and adjacent parts of the North Sea: their vegetation and management. *Biol. Conserv.* 51, 3, 191–210. doi: 10.1016/0006-3207(90)90151-E

Doody, J. P. (2008). *Management of Natura 2000 Habitats. 1330 Atlantic Salt Meadows (Glauco-Puccinellietalia maritimae).* European Commission, 27.

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I., and Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nat. Clim. Chang.* 3, 961–968. doi: 10.1038/nclimate1970

Esselink, P., Dijkema, K.-S., Reents, S., and Hageman, G. (1998). Vertical accretion and profile changes in abandoned man-made tidal marshes in the Dollard Estuary, the Netherlands. *J. Coast. Res.* 14, 570–582.

Esselink, P., Zijlstra, W., Dijkema, K. S., and Van Diggelen, R. (2000). The effects of decreased management on plant species distribution patterns in a salt-marsh nature reserve in the Wadden Sea. *Biol. Conserv.* 93, 61–76. doi: 10.1016/S0006-3207(99)00095-6

Ewel, K., Twilley, R., and Ong, J. I. N. (1998). Different kinds of mangrove forests provide different goods and services. *Global Ecol. Biogeogr. Lett.* 7, 83–94. doi: 10.2307/2997700

FAZ (2016). *Reetdächer: Haus mit Strohhut*. Available online at: http://www.faz.net/aktuell/wirtschaft/wohnen/reetdaecher-haus-mit-strohhut-14343926.html (accessed August 8, 2018).

Geijzendorffer, I. R., Cohen-Shacham, E., Cord, A. F., Cramer, W., Guerra, C., and Martín-López, B. (2017). Ecosystem services in global sustainability policies. *Environ. Sci. Policy* 74, 40–48. doi: 10.1016/j.envsci.2017.04.017

Geneletti, D., Scolozzi, R., and Adem Esmail, B. (2018). Assessing ecosystem services and biodiversity tradeoffs across agricultural landscapes in a mountain region. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manag.* 14, 188–208. doi: 10.1080/21513732.2018.1526214

Georgoudis, A. G., Papanastasis, V. P., and Boyazoglu, J. G. (1999). Use of water buffalo for environmental conservation of waterland. *Asian Australas J. Anim. Sci.* 12, 1324–1331. doi: 10.5713/ajas.1999.1324

Gough, L., Grace, J. B., and Taylor, K. L. (1994). The relationship between species richness and community biomass: the importance of environmental variables. *Oikos* 70, 271–79. doi: 10.2307/3545638

Grace, J. B., and Jutila, H. (1999). The relationship between species density and community biomass in grazed and ungrazed coastal grasslands. *Oikos* 85, 398–408. doi: 10.2307/3546689

Günther, A., Huth, V., Jurasinski, G., and Glatzel, S. (2015). The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. *GCB Bioenergy* 7, 1092–1106. doi: 10.1111/gcbb.12214

Haines-Young, R., and Potschin, M. B. (2018). *Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure*. Available online at: www.cices.eu. doi: 10.3897/oneeco.3.e27108

Hansson, L. A., and Graneli, W. (1984). Effects of winter harvest on water and sediment chemistry in a stand of reed (*Phragmites australis*). *Hydrobiologia* 112, 131–136. doi: 10.1007/BF00006917

Hansson, P. A., and Fredriksson, H. (2004). Use of summer harvested common reed (*Phragmites australis*) as nutrient source for organic crop production in Sweden. *Agric. Ecosyst. Environ.* 102, 365–375. doi: 10.1016/j.agee.2003.08.005

Haslam, S. M. (2009). *The Reed*. Updated edition. Norwich: British Reed Growers Association, Brown & Co., 38.

Hauck, J., Görg, C., Varjopuro, R., Ratamäki, O., and Jax, K. (2013). Benefits and limitations of the ecosystem services concept in environmental policy and decision making: some stakeholder perspectives. *Environ. Sci. Policy* 25, 13–21. doi: 10.1016/j.envsci.2012.08.001

Hellings, S. E., and Gallagher, J. L. (1992). The effects of salinity and flooding on *Phragmites australis.*. *J. Appl. Ecol.* 29, 41–49. doi: 10.2307/2404345

Holmer, M., Carta, C., and Andersen, F. O. (2006). Biogeochemical implications for phosphorus cycling in sandy and muddy rhizosphere sediments of

Zostera marina meadows (Denmark). *Mar. Ecol. Prog. Ser.* 320, 141–151. doi: 10.3354/meps320141

Holzmann, G., and Wangelin, M. (2009). *Natürliche und pflanzliche Baustoffe: Rohstoff–Bauphysik–Konstruktion (Natural and Plant Construction Materials: Raw Materials–Building Physics–Construction)*. Wiesbaden: Vieweg and Teubner, 225.

Howe, A., Rodríguez, J., and Saco, P. (2009). Surface evolution and carbon sequestration in disturbed and undisturbed wetland soils of the Hunter estuary, southeast Australia. *Estuar. Coast. Shelf. Sci.* 84, 75–83. doi: 10.1016/j.ecss.2009.06.006

Howe, C., Suich, H., Vira, B., and Mace, G. M. (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. *Glob. Environ. Chang.* 28, 263–275. doi: 10.1016/j.gloenvcha.2014.07.005

Huhta, A. (2007). "To cut or not to cut? The relationship between Common Reed, mowing and water quality," in *Read up on Reed!*, eds I. Ikonen and E. Hagelberg (Turku: Southwest Finland Regional Environment Centre), 30–37.

Huhta, A. (2009). "Decorative or outrageous—The significance of the common reed (*Phragmites australis*) on water quality," in *Comments from Turku University of Applied Sciences*, *Vol. 48* (Turku),1–33.

Iital, A., Klõga, M., Kask, Ü., Voronova, V., and Cahill, B. (2012). "Reed harvesting," in *Compendium: An Assessment of Innovative and Sustainable Uses of Baltic Marine Resources*, eds A. Schultz-Zehden and M. Matczak (Gdansk: Maritime Institute in Gdansk), 103–124.

Jensen, A., Skovhus, K., and Svendsen, A. (1990). "Effects of grazing by domestic animals on salt marsh vegetation and soils, a mechanistic approach," in *Salt Marsh Management in the Wadden Sea Region*, ed C. H. Ovensen (Copenhagen: Ministry of the Environment, The national Forest and Nature Agency), 153–161.

Jeschke, L. (1987). Vegetationsdynamik des Salzgraslandes im Bereich der Ostseeküste der DDR unter dem Einfluß des Menschen. *Hercynia N.F. Leipzig* 24, 321–328.

Jones, M. B., and Donnelly, A. (2004). Carbon sequestration in temperate grassland ecosystems and the influence of management, climate and elevated CO_2. *New Phytol.* 164, 423–439. doi: 10.1111/j.1469-8137.2004.01201.x

Jutila, H. (2001). How does grazing by cattle modify the vegetation of coastal grassland along the Baltic Sea? *Ann. Bot. Fennici* 38, 181–200.

Karsten, U., Schumann, R., and Witte, K. (2003). Filter zwischen land und see: darß-zingster boddengewässer. *Biol. Zeit* 33, 46–55. doi: 10.1002/biuz.200390008

Karstens, S., Buczko, U., and Glatzel, S. (2015a). Phosphorus storage and mobilization in coastal *Phragmites* wetlands: Influence of local-scale hydrodynamics. *Estuar. Coast. Shelf. Sci.* 164, 124–133. doi: 10.1016/j.ecss.2015.07.014

Karstens, S., Buczko, U., Jurasinski, G., Peticzka, R., and Glatzel, S. (2016). Impact of adjacent land use on coastal wetland sediments. *Sci. Total Environ.* 550, 337–348. doi: 10.1016/j.scitotenv.2016.01.079

Karstens, S., Schwark, F., Forster, S., Glatzel, S., and Buczko, U. (2015b). Sediment tracer tests to explore patterns of sediment transport in coastal reed beds—A case study from the Darss-Zingst Bodden Chain. *Rostocker Meeresbiol. Beiträge* 25, 41–57.

Kask, Ü., Kask, L., Paist, A. (2007). "Reed as energy resource in Estonia," in *Read Up on Reed!*, eds I. Ikonen and E. Hagelberg (Turku: Southwest Finland Regional Environment Centre), 102–114.

Kiehl, K. (1997). "Vegetationsmuster in Vorlandsalzwiesen in Abhängigkeit von Beweidung und abiotischen Standortfaktoren," in *Mitteilungen AG Geobotanik Schleswig-Holstein Hamburg* Vol. 52 (Kiel: Arbeitsgemeinschaft Geobotanik in Schleswig-Holstein und Hamburg), 142.

Kiehl, K., Eischeid, I., Gettner, S., and Walter, J. (1996). Impact of different sheep grazing intensities on salt marsh vegetation in Northern Germany. *J. Veg. Sci.* 7, 99–106. doi: 10.2307/3236421

Kiehl, K., and Stock, M. (1994). "Natur- oder Kulturlandschaft? Wattenmeersalzwiesen zwischen den ansprüchen von naturschutz, küstenschutz und landwirtschaft," in *Warnsignale aus dem Wattenmeer,* eds J. Lozán, E. Rachor, K. Reise, H. V. Westernhagen, and W. Lenz (Berlin: Blackwell), 190–196.

King, E., Cavender-Bares, J., Balvanera, P., Mwampamba, T. H., and Polasky, S. (2015). Trade-offs in ecosystem services and varying stakeholder preferences: evaluating conflicts, obstacles, and opportunities. *Ecol. Soc.* 20:25. doi: 10.5751/ES-07822-200325

Kiviat, E. (2013). Ecosystem services of Phragmites in North America with emphasis on habitat functions. *AoB Plants* 5:plt008. doi: 10.1093/aobpla/plt008

Kleyer, M., Feddersen, H., and Bockholt, R. (2003). Secondary succession on a high salt marsh at different grazing intensities. *J. Coast. Conserv.* 9, 123–134. doi: 10.1652/1400-0350(2003)009[0123:SSOAHS]2.0.CO;2

Knoll, T. (1986). Der Schilfschnitt am Neusiedler See. Analyse einer Landschaftsnutzung für Landschaftsplanung. *Geographisches Jahrbuch Burgenland* 1987, 34–67.

Köbbing, J., Thevs, N., and Zerbe, S. (2013). The utilisation of reed (*Phragmites australis*): a review. *Mires and Peat* 13, 1–14.

Krisch, H. (1989). Die Brackwasserröhrichte des Greifswalder Boddens. *Meer Museum* 5, 14–24.

Krisch, H. (1992). Systematik und Ökologie der Bolboschoenus- und der *Phragmites*-Brackwasserröhrichte der vorpommerschen Boddenküste (Ostsee). *Drosera* 92, 89–116.

Lampe, R., and Wohlrab, B. (1996). Zum Belastungs- und Entsorgungspotential von Salzgrasland—Untersuchungen an Bodenproben unterschiedlich genutzter Standorte des Küsten-Überflutungsgebietes "Karrendorfer Wiesen" bei Greifswald. *Natur Naturschutz Mecklenburg-Vorpommern* 32, 6–69.

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

LUNG (Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern) (2003). "Gesetzlich geschützte Biotope und Geotope in Mecklenburg-Vorpommern," *Schriftenreihe des Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern,* Vol. 1 (Güstrow: Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern), 1–45.

LUNG (Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern) (2017). *Umsetzung von Paludikultur auf landwirtschaftlich genutzten Flächen in Mecklenburg-Vorpommern. Fachstrategie zur Umsetzung der nutzungsbezogenen Vorschläge des Moorschutzkonzeptes*. Schwerin: Ministerium für Landwirtschaft und Umwelt Mecklenburg-Vorpommern.

Manolache, S., Nita, A., Cinocanea, C. M., Popescua, V. D., and Rozylowicz, L. (2018). Power, influence and structure in Natura 2000 governance networks. A comparative analysis of two protected areas in Romania. *J. Environ. Manage.* 212, 54–64. doi: 10.1016/j.jenvman.2018.01.076

Meriste, M., Kirsimäe, K., and Freiberg, L. (2012). Relative sea-level changes at shallow coasts inferred from reed bed distribution over the last 50 years in Matsalu Bay, the Baltic Sea. *J. Coast. Res.* 28, 1–10. doi: 10.2112/JCOASTRES-D-10-00049.1

Meuleman, A. F. M., Beekman, J. P., and Verhoeven, J. T. A. (2002). Nutrient retention and nutrient-use efficiency in *Phragmites australis* stands after wastewater application. *Wetlands* 22, 712–721. doi: 10.1672/0277-5212(2002)022[0712:NRANUE]2.0.CO;2

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

Mitsch, W. J., and Gosselink, J. G. (2000). *Wetlands*, *3rd Edn.* New York, NY: Wiley & Sons, 920.

Moberg, F., and Rönnbäck, P. (2003). Ecosystem services of the tropical seascape: interactions, substitutions and restoration. *Ocean Coast. Manage.* 46, 27–46. doi: 10.1016/S0964-5691(02)00119-9

Möller, I., Mantilla-Contreras, J., Spencer, T., and Hayes, A. (2011). Micro-tidal coastal reed beds: hydro-morphological insights and observations on wave transformation from the southern Baltic Sea. *Estuar. Coast. Shelf. Sci.* 92, 424–436. doi: 10.1016/j.ecss.2011.01.016

Moore, B. C., Lafer, J. E., and Funk, W. H. (1994). Influence of aquatic macrophytes on phosphorus and sediment porewater chemistry in a freshwater wetland. *Aquat. Bot.* 49, 137–148. doi: 10.1016/0304-3770(94)90034-5

Müller, J., Kayser, M., and Isselstein, J. (2007). "The effect of stocking rate on nesting success of meadow birds," in *Permanent and Temporary Grassland. Plant, Environment and Economy*, eds A. De Vliegher and L. Carlier (Gent: European Grassland Federation, Gent), 72–73.

Neuhaus, R., Stelter, T., and Kiehl, K. (1999). Sedimentation in salt marshes affected by grazing regime, topographical patterns and regional differences. *Senckenbergiana Maritima* 29(Suppl.), 113–116. doi: 10.1007/BF03043134

Ostendorp, W. (1993). "Schilf als Lebensraum," in *Beihefte zu den Veröffentlichungen für Naturschutz und Landschaftspflege in Baden-Württemberg, Vol. 68* (Karslruhe: Landesanstalt für Umweltschutz Baden-Württemberg), 173–280.

Ostendorp, W. (1995). Estimation of mechanical resistance of lakeside *Phragmites* stands. *Aquat. Bot.* 51, 87–101. doi: 10.1016/0304-3770(95)00470-K

Ostendorp, W. (1999). Management impacts on stand structure of lakeshore *Phragmites* reeds. *Int. Rev. Hydrobiol.* 84, 33–47.

Paulson, C., and Raskin, R. (1998). Die Vegetation des Großen Werder als Ausdruck von Küstendynamik und Landnutzung. *Natur Naturschutz Mecklenburg-Vorpommern* 34, 24–42.

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., et al. (2012). Estimating global blue carbon emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS ONE* 7:e43542. doi: 10.1371/journal.pone.0043542

Posner, S. M., McKenzie, E., and Ricketts, T. H. (2016). Policy impacts of ecosystem services knowledge. *Proc. Natl. Acad. Sci. U.S.A.* 113, 1760–1765. doi: 10.1073/pnas.1502452113

Prach, K., and Pyšek, P. (1994). Clonal plants—What is their role in succession? *Folia Geobot. Phytotaxonimica* 29, 307–320. doi: 10.1007/BF02803803

Puurmann, E., Ratas, U., and Rivis, R. (2002). "The change in plant diversity of seashore meadows on Estonian uplifting lowshores," in *Proceedings of the 1st Conference Salt Grasslands and Coastal Meadows in the Baltic Region*, eds T. Fock, K. Hergarden, and D. Repasi Reihe A (Neubrandenburg: Schriftenreihe der Fachhochschule Neubrandenburg), 292–296

Raabe, E.-W. (1981). Über das Vorland der östlichen Nordsee-Küste. *Mitteilungen AG Geobotanik Schleswig-Holstein Hamburg* 31, 1–118.

Rannap, R., Briggs, L., Lotman, K., Lepik, I., and Rannap, V. (2004). *Coastal Meadow Management. Best practice Guidelines. The Experience of LIFE-Nature Project "Boreal Coastal Meadow Preservation in Estonia"*. Tallinn: Ministry of the Environment of the Republic of Estonia, 95.

Ranwell, D. S. (1972). *Ecology of Salt Marshes and Sand Dunes*. London: Chapman and Hall, 258.

Raudsepp-Hearne, C., Peterson, G. D., and Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proc. Natl. Acad. Sci. U.S.A.* 107, 5242–5247. doi: 10.1073/pnas.0907284107

Reddy, K. R., Kadlec, R. H., Flaig, E., and Gale, P. M. (1999). Phosphorus retention in streams and wetlands: a review. *Crit. Rev. Environ. Sci. Technol.* 29, 83–146. doi: 10.1080/10643389991259182

Renard, D., Rhemtulla, J. M., and Bennett, E. M. (2015). Historical dynamics in ecosystem service bundles. *Proc. Natl. Acad. Sci. U.S.A.* 112, 13411–13416. doi: 10.1073/pnas.1502565112

Rodewald-Rudescu, L. (1974). "Das Schilfrohr," in *Die Binnengewässer, Vol. 27* (Stuttgart: Schweizerbart), 302.

Ruiz-Frau, A., Krause, T., and Marbà, N. (2018). The use of sociocultural valuation in sustainable environmental management. *Ecosyst. Serv.* 29, 158–167. doi: 10.1016/j.ecoser.2017.12.013

Sanchez, J. M., Izco, J., and Medrano, M. (1996). Relationships between vegetation zonation and altitude in a salt-marsh system in northwest Spain. *J. Veg. Sci.* 7, 695–702. doi: 10.2307/3236381

Schaatke, W. (1992). *Das Reetdach—Natürliches Wohnen unter sanftem Dach—Von der Uhrzeit bis heute (The Reed Roof—Natural Living Under a Soft Roof—From Primeval Times Until Today)*. Hamburg: Christians Verlag, 264.

Schäfer, A., and Wichtmann, W. (1999). Sanierte Niedermoore und weitergehende Abwasserreinigung (Restored fens and largely sewage treatment). *Archiv Naturschutz Landschaftsforschung* 38, 315–334.

Scherfose, V. (1993). Zum Einfluß der Beweidung auf das Gefäßpflanzen-Artengefüge von Salz- und Brackmarschen. *Zeitschrift Ökologie Naturschutz* 2, 201–212.

Schernewski, G., Inácio, M., and Nazemtseva, Y. (2018). Expert based ecosystem service assessment in coastal and marine planning and management: A Baltic Lagoon case study. *Front. Environ. Sci* 6, 19. doi: 10.3389/fenvs.2018.00019

Schieferstein, B. (1997). Ökologische und molekularbiologische Untersuchungen an Schilf (*Phragmites australis* (Cav). Trin. ex Steud). im Bereich der Bornhöveder Seen. *Beiträge Ökosystemforschung EcoSys Suppl.* 22, 1–143.

Schieferstein, B. (1999). Ökologische und molekularbiologische Untersuchungen an Schilf (*Phragmites australis* [Cav]. Trin. ex Steudel) von norddeutschen Seen—Ein Überblick. *Limnologica* 29, 28–35. doi: 10.1016/S0075-9511(99)80036-X

Schulz, K., Timmerma nn, T., Steffenhag en, P., Zerbe, S., and Succ ow, M. (2011). The effect of flooding on carbon and nutrient standing stocks of helophyte biomass in rewetted fens. *Hydrobiologia* 674, 25–40. doi: 10.1007/s10750-011-0782-5

Seppelt, R., Lautenbach, S., and Volk, M. (2013). Identifying trade-offs between ecosystem services, land use, and biodiversity: a plea for combining scenario analysis and optimization on different spatial scales. *Curr. Opin. Environ. Sustain.* 5, 458–463. doi: 10.1016/j.cosust.2013.05.002

Steffenhagen, P., Timmermann, T., Schulz, K., and Zerbe, S. (2008). "Biomassenreproduktion sowie Kohlenstoff- und Nährstoffspeicherung durch Sumpfpflanzen (Helophyten) und Wasserpflanzen (Hydrophyten)," *in Phosphorund Kohlenstoff-Dynamik und Vegetationsentwicklung in wiedervernässten Mooren des Peenetals in Mecklenburg-Vorpommern. Status, Steuergrößen und Handlungsmöglichkeiten, eds* J. von Gelbrecht, D. Zak, and J. Augustin, Berichte des IGB 26. S (Berlin: Leibniz-Institut für Gewässerökologie und Binnenfischerei (IGB) im Forschungsverbund Berlin e.V), 145–154.

Stock, M., Kiehl, K., and Reinke, H.-D. (1997). Salzwiesenschutz im schleswig-holsteinischen Wattenmeer. *Schriftenreihe Nationalpark Schleswig-Holsteinisches Wattenmeer* 7, 48.

Stoll-Kleemann, S. (ed.). (2015). *Local Perceptions and Preferences for Landscape and Land Use in the Fischland-Darß-Zingst Region, German Baltic Sea*. Greifswalder Geographische Arbeiten Bd. 51. Greifswald: Institut für Geographie und Geologie der Ernst-Moritz-Arndt Universität Greifswald.

Succow, M., and Joosten, H. (Eds.). (2001). "Landschaftsökologische Moorkunde. Schweizerbart'sche Verlagsbuchhandlung," in *Stuttgart, 2nd Edn,* eds M. Succow and H. Joosten (Stuttgart: Schweizerbart science publishers), 622.

Sweers, W., Horn, S., Grenzdörffer, G., and Müller, J. (2013). Regulation of reed (*Phragmites australis*) by water buffalo grazing: use in coastal conservation. *Mires Peat* 13, 1–10. Available online at: http://www.mires-and-peat.net/

Timmermann, T. (2009). *Biomasse- und Standortskatalog (Standortpotenzial). Bericht zum Forschungs- und Entwicklungsprojekt Energiebiomasse aus Niedermooren (ENIM)*, eds S. Wichmann and W. Wichtmann. Greifswald: Ernst-Moritz-Arndt Universität Greifswald. DUENE e.V, 37–52.

Vulink, J. T., Drost, H. J., and Jans, L. (2000). The influence of different grazing regimes on *Phragmites* and shrub vegetation in the well drained zone of a eutrophic wetland. *Appl. Veg. Sci.* 3, 73–80. doi: 10.2307/1478920

Wanner, A. (2009). *Management, biodiversity and restoration potential of salt grassland vegetation of the baltic sea: analyses along a complex ecological gradient*. Ph.D. Thesis, Hamburg, Germany: Universität Hamburg.

Wichtmann, W. (2011). "Land use options for rewetted peatlands—Biomass use for food and fodder," in *Carbon Credits From Peatland Rewetting*, eds F. Tanneberger and W. Wichtmann (Stuttgart: E. Schweizerbart), 110–113.

Wiegleb, G., and Krawczynski, R. (2010). Biodiversity management by water buffalos in restored wetlands. *Waldokol. Online* 10, 17–22.

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Management and Exploitation of Macroalgal Biomass as a Tool for the Recovery of Transitional Water Systems

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Aquatic angiosperms favor the development of ecosystems services, the welfare of marine organisms and people. Generally, the presence of angiosperms in transitional water systems (TWS) are indicators of good ecosystem status. Presently, these environments are densely populated and often are so highly degraded that angiosperms have almost disappeared, replaced by tionitrophilic macroalgae responsible of anoxic events that deteriorate the environment furtherly. Although this trend is hardly reversible because the anthropogenic impact is increasing and the restoring of damaged environments within a reasonable time is difficult, recent studies have shown that by managing the harvesting of the natural algal species of commercial interest a progressive environmental recovery is achievable. Biomass-harvesting can contribute both to the removal of high amounts of nutrients and the generation of economic revenues for a sustainable, self-financed environmental restoration. In fact, unlike clam-farming which destroys the seabed and re-suspends large amounts of sediments, the proper management of the macroalgal biomass, can favor the nutrient abatement and the recolonization of aquatic angiosperms which help restore the conditions necessary for the conservation of the benthic and fish fauna and birds, and produce valuable economic resources.

Keywords: macrophytes, *Agarophyton vermiculophyllum*, *Gracilariopsis longissima*, *Gracilaria gracilis*, nutrient removal, agar production, ecosystem services, transitional water systems

INTRODUCTION

Primary producers in transitional water systems (TWS) are mainly represented by macrophytes: aquatic angiosperms and macroalgae. The former usually prevail over the others, but that depends on the ecological status of the study area. Aquatic angiosperms are considered the most important producers under pristine conditions (Orfanidis et al., 2001; Sfriso et al., 2007; Viaroli et al., 2008).

They colonize environments with a low trophic status and provide a variety of ecosystem services (ES) whose most common meaning is:

- The benefits people obtain from ecosystems (Millenium Ecosystem Assessment, 2005);
- Natural processes and components that benefit human needs directly or indirectly (Nordlund et al., 2016).

A conceptual framework for ES classification was established in the studies of the Millennium Ecosystem Assessment and the Economics of Ecosystems and Biodiversity (Arico et al., 2005; Barker et al., 2010) that define four major classes, inclusive of numerous subclasses, based on the kind of service provided by ecosystems for human well-being (Liquete et al., 2013; Hattam et al., 2015; Haines-young and Potschin, 2018), i.e., Regulation services, Habitat services, Provisioning services, and Cultural services.

The ES supplied by the macrophytes, both aquatic angiosperms and macroalgae, may belong to all those categories. In fact, aquatic angiosperms reduce sediment resuspension, favor clear waters, contrast sediment erosion, and contribute to permanent CO_2 sequestration (Regulation services); provide shelter and nursery areas for benthic and fish fauna and pasture areas for birds (Habitat services); increase recreational activities (Cultural services); and sustain traditional fishing activities (Provisioning services) as a consequence of a general improvement of the environment. In contrast, macroalgae are an important source of biomass which is exploitable for the production of compost, fertilizer, human and animal food, pharmaceuticals, and cosmetics (Provisioning services). In pristine environments the calcareous species trap CO_2, whereas in degraded environments other taxa accumulate nutrients (Sfriso and Marcomini, 1994; Sfriso et al., 1994) and contaminants (Maroli et al., 1993) (Habitat services).

Recent studies have shown that macrophytes can be used as water quality indicators and bio-ecological sentinels. They play a key role in the indices of ecological status set up for the assessment of TWS (Ecological Evaluation Index continuous – EEI-c, Orfanidis et al., 2011; Macrophyte Quality Index – MaQI, Sfriso et al., 2014) and coastal areas (*Posidonia oceanica* Rapid Easy Index – PREI, Buia et al., 2003; EEI-c, Orfanidis et al., 2011; CARLIT, Ballesteros et al., 2007; Sfriso and Facca, 2011) according to the requirements of the Water Framework Directive (2000/60/EC) (European.Union [Eu], 1992; European Union [Eu], 2000).

Terrados and Borum (2004) estimated that environments colonized by aquatic angiosperms produced a minimum revenue of US$ 15,837 ha^{-1} y^{-1}, which is nearly twice the revenue from croplands. Costanza et al. (1997) have come to the same conclusions estimating for these plants a revenue of US$ 19,002 ha^{-1} y^{-1}. Generally, nutrient recycling and water quality were the most important ES considered by those authors. However, the economic value calculated per single ES is also very important. By considering the value of large-sized, commercially targeted, fish species Tuya et al. (2014) estimated the areas covered by *Cymodocea nodosa* in Canary islands to be worth €866 ha^{-1} y^{-1}. Other researchers as Guerry et al. (2012) reported that the value of multiple services resulting from seagrass beds reached €4228 ha^{-1} y^{-1}. If we consider the carbon storage and sequestration both in the aquatic plants and surface sediments, the ES is worth ca. US$ 30.5 ha^{-1} y^{-1} (Barbier et al., 2011); but even more important is the organic-rich materials which favors the rise of the seafloor at rates from 0.6 to 6 mm y^{-1} (Duarte, 2013) and can be preserved over millennia.

However, the increase of anthropogenic impacts worldwide has led to a progressive degradation of the marine coasts, especially transitional areas where human pressure is remarkable (Sfriso et al., 2017a) with the consequence that often aquatic angiosperms have been replaced by fast-growing macroalgae (Morand and Briand, 1996; Ménesguen, 2018). The same changes have been observed in the Italian TWS where in the worst conditions phytoplankton or cyanobacteria blooms have prevailed (Sfriso and Facca, 2007; Cecere et al., 2009; Munari and Mistri, 2012). Under eutrophic conditions, the benefits due to the presence of aquatic angiosperms are lost, but recent studies focused on the production of macroalgae for food, cosmetics, or pharmaceuticals (Francavilla et al., 2013, 2015; Sfriso and Sfriso, 2017; Sfriso et al., 2017b) showed that the management of the biomass of some macroalgae may be economically very interesting and provide more ES (in terms of nutrient abatement and commodities) than aquatic angiosperms. In addition, the harvesting of macroalgae, which in eutrophic areas accumulate high amounts of phosphorus (P) and nitrogen (N) (Sfriso and Marcomini, 1994; Sfriso et al., 1994), may contribute to the reduction of nutrients in the environment attenuating eutrophication. By the present paper the authors intend to show that highly degraded TWS dominated by macroalgae can be recovered by a good management, and the cause of degradation transformed into a valuable resource for both the environment and people welfare, notwithstanding the fact that it is not always possible to reduce the trophic status of a water body in a short time. The studied areas are the lagoons of the Po Delta, some chocked areas of the Venice Lagoon and Pialassa della Baiona where trophic conditions are high and suffer from intense clam-farming and/or a rich growth of macroalgal biomass.

MATERIALS AND METHODS

The data on macroalgal distribution come from researches carried out in various Italian TWS (**Figure 1**) over the last 10 years for different purposes: to apply the Water Framework Directive (2000/60/EC) (European.Union [Eu], 1992; European Union [Eu], 2000) in the lagoons of Venice (114 sites), Pialassa della Baiona (three sites) and the Po Delta (21 sites); and to study the macroalgal production of some Gracilariaceae and *Ulva rigida* C. Ag. over 1 year (Sfriso and Sfriso, 2017; Sfriso et al., 2017a) as part of the COST Action FA1406 (2015–2019) (Phycomorph, 2015).

The information obtained by those studies allows to assess the water surfaces colonized by aquatic angiosperms or

FIGURE 1 | Map of the studied lagoons in North-western Adriatic Sea.

macroalgae, and the areas which are potentially exploitable for macroalgal harvesting and nutrient abatement following the removal of the biomass.

In addition, a comparison between the potential revenue from macroalgal harvesting and the one from clam breeding/harvesting, which is a highly destructive activity for the environment (Pranovi and Giovanardi, 1994; Sfriso et al., 2005), highlights the benefits that might come from a sustainable management of the environment.

Study Areas

The Venice Lagoon is the largest TWS of the Mediterranean Sea. It has a surface of ca. 549 km^2, which totals 432 km^2, if islands, fishing ponds, and salt marshes are excluded. It is a very polyhedral shallow (mean depth ca. 1.2 m) environment with wide areas colonized by five different aquatic angiosperms and an almost equivalent surface colonized by a biomass of macroalgae among which Gracilariaceae, Ulvaceae, and Cladophoraceae prevail. For a more complete representation of the environmental variability of the Venice Lagoon, 114 sampling sites were selected among the 118 identified by the Regional Agency for Environmental Prevention and Protection of Veneto (ARPAV) for macrophytes monitoring according to WFD (2000/60/EC).

The Po Delta exhibits a high number of lagoons and ponds scattered on a surface of 204 km^2. The environment is eutrophic, macrophyte biodiversity is low and aquatic angiosperms are missing, whereas macroalgae, especially Gracilariaceae, Solieriaceae, and Ulvaceae are abundant. These rather uniform lagoons were studied by monitoring 20 sites. Pialassa della Baiona in the Emilia Romagna Region is a small eutrophic lagoon of ca. 11 km^2 which is mainly colonized by Gracilariaceae and aquatic angiosperms are absent. Its variability was tested by monitoring three sites. The sites selected in the Po Delta and Pialassa della Baiona Lagoons were also the same monitored by the Regional Agencies for Environmental Prevention and Protection of Veneto and Emilia Romagna.

On the whole, these TWS have a surface of 764 km^2 and represent ca. 76% of TWS of the Northern Adriatic Sea (1008 km^2) and ca. 55% of the total TWS in Italy (1398 km^2, Sfriso et al., 2017a).

Macroalgal Composition Analysis

Samples of three Gracilariaceae: *Agarophyton vermiculophyllum* (Ohmi) Gurgel, J. N. Norris et Fredericq; *Gracilaria gracilis* (Stackhouse) Steentoft, L. M. Irvine & Farnham; and *Gracilariopsis longissima* (S. G. Gmelin) Steentoft, L. M. Irvine & Farnham (12 samples per 4 species) were collected monthly for chemical analyzes from March 2014 to February 2015 in two stations of the Venice Lagoon choked area, one with clear waters (st. a) and the other with turbid waters (st. b). Macroalgae were wet weighed, lyophilized, and re-weighed to calculate the dry weight and to determine the dry/wet-weight ratio. Ashes were measured after 2 h combustion at 440°C. The organic matter content was calculated by weight difference. The mean and standard deviation values determined in the two stations were presented.

Nutrient Analysis in Algal Tissues

The concentration of P was determined by applying the phosphomolybdenum blue method according to Strickland and Parsons (1972) after a 2 h digestion of 0.3 g of dry biomass in teflon bombs at 130°C with 4 ml of Milli-Q, 3 ml of HNO_3, and 3 ml of $HClO_4$ (Kornfeldt, 1982). Analyzes were replicated in different days and the result uncertainty was <5%. The total N was determined by using a Flash 2000 CHNS Elemental Analyzer of Thermo Fisher Scientific (Monza, Italy). Measurements were replicated in different days and precision was >3%.

Polysaccharide Determination

The native sulfated agar was extracted from a 50 mg algal sample using 10 ml of Milli-Q water at 100°C for 2 h, vortexing for few seconds every 15 min for 1.5 h. The extracts were analyzed only after the solution became clear by the colorimetric method of Soedjak (1994), adding a methylene blue solution to a diluted hydrocolloid shifting absorbance to 559 nm. The calibration curves were calculated from the native sulfated agar of the respective algae after they had been purified using the method of Gonzalez-Leija et al. (2009). This native sulfated agar can be converted to a valuable commercial agar by means of pre-treatments that removes the sulfated groups and reduces the yield by approx. 55% (Wakibia et al., 2001) improving the chemical and physical properties of the gel. All reagents were purchased from Sigma–Aldrich, SRL, Milano (Italy). All the chemical analyzes were done in triplicate and repeated until the analytical reproducibility (coefficient of variation) was within 5%.

Biomass Harvesting and Production

The biomass level recorded during a sampling period must be taken into account before selecting the areas intended for production. The highest yield of Gracilariaceae is obtained from a biomass of approx. 3–4 kg fw m^{-2} (Friedlander and Levy, 1995; Sfriso and Sfriso, 2017). Accordingly, in the areas where Gracilariaceae prevail it is important to keep the biomass around that value by collecting only the exceeding macroalgae every 7–15 days, depending on the season and growth rate. In addition, the biomass should be hand-harvested, since bottoms are usually lower than 1 m on the mean tide level. Flat boats should be used to avoid sediment re-suspension and reduce impacts on the environment. A similar procedure was followed in the 1980s when up to 7000 tonnes y^{-1} of Gracilariaceae were harvested in the Venice Lagoon for agar production (Orlandini and Favretto, 1987; Sfriso et al., 1994).

In our production measurements 200 g of *A. vermiculophyllum*, *G. gracilis*, and *G. longissima* were inserted in green plastic-coated cubical wire-cages (25 × 25 × 25 cm with 1-cm net mesh), one species per cage. Two cages per species were placed on the bottom where, seasonally, other species could be present. On the whole, between March 2014 to February 2015, 39 sampling campaigns were carried out. The sampling interval ranged between 7 and 11 days, depending on the weather conditions. On each sampling day, in-cage biomass was collected, dripped with a salad spinner to remove excess water (ca. 10 s), and weighed (precision: 1 g). Then, the in-cage biomass was restored to 200 g in order to obtain productions suitable to be compared with the samples from the other campaigns. The biomass of 200 g per cage used to estimate algal production throughout 1 year was selected by comparison with the mean biomass present in the two areas during preliminary surveys (ca. 3 kg fw m^{-2}). This value was in the optimal range (3–4 kg fw m^{-2}) for intensive *Gracilaria* production (Friedlander and Levy, 1995).

The relative growth rate (RGR) between two consecutive sampling days was calculated using the formula reported by Lignell et al. (1987): $\%RGR = 100 \times [(B_t/B_o)^{1/t} - 1]$ where B_0 = initial biomass, B_t = final biomass at time t expressed in days.

RESULTS

Macrophyte Cover

The macrophyte distribution in the studied lagoons showed that their diversity was strongly related to the ecological conditions of the considered areas. By considering the Venice Lagoon, aquatic angiosperms colonized 23 stations out of 114 (20.2% of the total), accounting for ca. 87 km^2. Conversely, the number of stations with a mean macroalgal biomass $\geq$3 kg fw m^{-2} was 18 out of 114 (16% of the total), accounting for a surface of ca. 68 km^2. The lagoons of the Po Delta showed 3 stations out of 20 with a biomass $\geq$3 kg fw m^{-2}, accounting for ca. 30.6 km^2.

Pialassa della Baiona showed two stations covered with a high macroalgal biomass, but excluding salt marshes and canals only ca. 3 km^2 showed a biomass $\geq$3 kg fw m^{-2}. The dominant species was the non-indigenous species (NIS) *A. vermiculophyllum*.

Macroalgal Composition

The concentration of nutrients in the macroalgal tissues of the dominant taxa was studied in many areas of the Venice Lagoon (Sfriso and Marcomini, 1994; Sfriso et al., 1994). The attention was particularly focused on *U. rigida* and *G. gracilis* that represented a large portion of the lagoon biomass (Sfriso and Facca, 2007). The nutrient concentrations in tissues were strongly related to the sampling areas and seasons; the highest values were recorded in eutrophic areas where the trophic level was high and the biomass abundant (Sfriso et al., 1994). **Table 1A** shows the nutrients and carbon content of *G. gracilis* found in three very different areas in 1994, **Table 1B** the composition of different species of Gracilariaceae recorded in the choked area in 2014.

The dry weight/wet weight (dw/wt) ratio of Gracilariaceae was approximately 15%, ranging between 14.2 (**Table 1A**) and 15.5% (**Table 1B**). The ash percentage fluctuated around 30% whereas, on average, the organic matter was approximately 70%. The tissue concentrations of P and N were significantly affected by the harvesting area. At st. 3 (choked area, **Table 1A**) *Gracilaria* P and N concentrations were ca. 4.43 and 41.2 mg g^{-1}, respectively (high water renewal area), i.e., ca. 8 and 0.5 times higher than those recorded at st. 1, respectively. No significant differences were found by considering the three Gracilariaceae. The native agar concentration of these species recorded in 2014, on average, was 27.1% of the dry weight (**Table 1B**), with slightly higher, values in *G. longissima* (31.5%) accounting for a commercial (desulfurized agar) of ca.14–17% dw (Sfriso et al., 2017b).

The annual net biomass production of the three Gracilariaceae ranged from 9.0 to 12.8 kg fw m^{-2} y^{-1} in very turbid waters, accounting for 24.7–35.2 g fw m^{-2} d^{-1} and a percent growth rate of 0.77–1.10 d^{-1}. The highest net production was recorded in clear waters with 21.9–28.2 kg fw m^{-2} y^{-1} (**Table 2**).

The potential economical value of these biomasses is reported in **Table 3** together with the estimation for the ecosystems colonized by clams and aquatic angiosperms. The potential revenue of the raw macroalgal biomass, in accordance with the estimate made by Food Agriculture Organization of the United Nations Food and Agriculture Organization of the United Nations [FAO] (2012) in 35 countries (US$ 0.35 kg^{-1} of fresh biomass) could vary between US$ 31,500 and 98,000 ha^{-1} y^{-1}, depending on the biomass production. Specifically, if we consider the production of commercial agar and the price of US$ 11–17 kg^{-1}, the revenue can vary from US$ 45,100–124,100 ha^{-1} y^{-1}. This value is very similar to that obtained from clam production which ranged between US$ 40,000 and 120,000 ha^{-1} y^{-1}. Instead the revenue obtainable from ecosystems colonized by aquatic angiosperms ranged between US$ 15,837 and 19,002 ha^{-1} y^{-1} (Costanza et al., 1998; Terrados and Borum, 2004).

Table 4 shows an estimation of the amount of nutrient abatement due to Gracilariaceae harvesting in the considered

TABLE 1 | Composition of Gracilariaceae in different areas of the Venice Lagoon: **(A)** composition of *Gracilaria gracilis* found in 1994 in three very different areas: st. 1 (high water renewal area), st. 2 (watershed area), and st. 3 (choked area) (Sfriso et al., 1994); **(B)** mean composition of different Gracilariaceae recorded at sts. a and b in the choked area.

(A) *Gracilaria gracilis* composition									
Area	**Samples (*N*)**	**Dry weight (%)**	**Ashes (%)**	**Organic matter (%)**	**Total (mg g^{-1} dw)**			**Agar (% dw)**	
					Phosphorus	**Nitrogen**	**Carbon**	**Native**	**Desulfurized**
st. 1 (high water renewal area)	12	15.4	27.2	72.8	0.53	26.8	256	–	–
st. 2 (lagoon watershed area)	12	14.2	33.9	66.2	3.25	37.6	265	–	–
st. 3 (choked area)	12	14.3	33.4	66.6	4.43	41.2	315	–	–
Mean		14.6	31.5	68.5	2.74	35.2	279	–	–
Std		0.67	3.7	3.7	2.00	7.5	31.7	–	–
(B) Gracilariaceae composition									
Species	**Samples (*N*)**	**Dry weight (%)**	**Ashes (%)**	**Organic matter (%)**	**Total (mg g^{-1} dw)**			**Agar (% dw)**	
					Phosphorus	**Nitrogen**	**Carbon**	**Native**	**Desulfurized**
Agarophyton vermiculophyllum	12	15.5	31.0	69.0	4.21	27.8	296	24.6	13.5
Gracilaria gracilis	12	15.0	28.9	71.1	3.98	33.1	302	25.1	13.8
Gracilariopsis longissima	12	14.6	27.2	72.8	3.75	30.9	312	31.5	17.3
Mean		15.0	29.0	71.0	3.98	30.6	303	27.1	14.9
Std		0.45	1.89	1.89	0.23	2.69	7.79	3.85	2.11

lagoons. These amounts are between 1521–4737 tonnes y^{-1} for N and 198–615 tonnes y^{-1} for P.

Finally, the potential revenue obtainable from the production of agar in areas predominantly covered by Gracilariaceae is reported in **Table 5**. In a total lagoon surface of approx. 3682 ha prevalently colonized by Gracilariaceae, approx. 14,985–26,731 tonnes of commercial agar per year could be produced with a total potential revenue ranging from US$ 165 to 454 million. However, this potential profit should be evaluated taking into account the production costs that have been calculated from the literature and can vary greatly depending on various factors that must be taken into due consideration.

DISCUSSION

Pristine TWS are predominantly colonized by aquatic angiosperms which have a high ecological value and affect people's recreational activities and their life positively. Unfortunately, most TWS are strongly eutrophic and plants have been replaced by opportunistic fast-growing macroalgae such as Ulvaceae, Cladophoraceae, Gracilariaceae, and Solieriaceae.

In some cases, especially in wide TWS, pristine and eutrophic conditions can coexist in the same basin and its surface is year by year sensitive to changes related to the weather variations or different anthropogenic impacts. The efforts of local administrations, national environmental agencies, and the European Community (see LIFE projects) aim at the recovery and conservation of natural and semi-natural habitats, wild flora and fauna (Habitat Directive 92/43/EEC, water Directive 2000/60/EC). However, it is not always possible to bring back pristine conditions and/or the efforts to do that are judged economically unreasonable. In Italy, there are many deltaic systems or confined areas such as the Po Delta, some areas of the Venice Lagoon, Pialassa della Baiona, and the "Valli di Comacchio" in the Veneto and Emilia-Romagna Regions where the ecological conditions are strongly eutrophic and the abatement of nutrients has not been successful. Under such conditions aquatic angiosperms have disappeared, clam farming (breading or free harvesting of *Ruditapes philippinarum* Adams & Reeve) activities are intense and macroalgae grow massively, often triggering dystrophic conditions (Morand and Briand, 1996; Ménesguen, 2018).

Despite the situation an alternative to reduce the trophic level and at the same time have an economic return is possible. Although the estimation of ES provided by aquatic plants is high (Costanza et al., 1997; Terrados and Borum, 2004; Short et al., 2011) the value of eutrophic environments employed for clam-farming or macroalgal harvesting is considered higher especially in terms of provisioning services at the expense of regulation and habitat services. Additionally, the well-being produced from an environment of high ecological quality cannot always be easily estimated in economic terms.

In 2011 the production of macroalgae in 35 countries worldwide was estimated to be 21 million tonnes fresh weight, corresponding to US$ 7.35 billion revenue, that is US$ 0.35 per kg (Food and Agriculture Organization of the United Nations [FAO], 2012). In 2016 the production increased to 31.2 million tonnes (Food and Agriculture Organization of the United Nations [FAO], 2018) showing the importance of this resource. Recent studies on the natural productivity of the most common macroalgae carried out in the Venice Lagoon (Sfriso and Sfriso, 2017) showed that the annual net production of Gracilariaceae ranged between 9.0 and 28 kg fw m^{-2} y^{-1}, depending on water turbidity (**Table 2**). According to FAO quotation this biomass would account from US$ 31,500 to 98,000 ha^{-1} (**Table 3**).

Significant results may be also obtained by the production of commercial agar as it occurred in the Venice Lagoon in the 1980s where up to 7000 tonnes of Gracilariaceae were collected to produce this phytocolloid (Sfriso et al., 1994). In the following years the biomass of macroalgae declined in the whole lagoon (Sfriso and Facca, 2007) and the harvesting of Gracilariaceae was no longer economically feasible. Currently, Gracilariaceae biomass has increased significantly, also thanks to the introduction of the NIS *A. vermiculophyllum* which prefers degraded environments, and the harvesting of these species for agar production could be re-evaluated.

A study by Sfriso et al. (2017b) showed that the production of native agar by Gracilariaceae ranged between 7.4 and 13.2 tonnes ha^{-1} y^{-1}, accounting for 4.07–7.26 tonnes ha^{-1} y^{-1} of commercial product (desulfurized agar). By considering that the price of this product can vary between US$ 11 kg^{-1} (FAO) and US$ 15–19 kg^{-1} with a mean value of US$ 17 kg^{-1} (mean of 50 suppliers in "Alibaba Group Holding Limited" a Chinese multinational holding specialized in e-commerce[1]), the potential revenue from one ha of lagoon surface is similar to or even higher than the revenue obtainable from the raw biomass (**Table 3**).

In addition, taking into account the mean nutrient concentrations in the dry biomass recorded in Sfriso et al. (1994) (on average 30.6 and 3.98 g kg^{-1} dw for N and P, respectively) in the choked area (mean of sts. a and b) of the Venice Lagoon, the amount of nutrients that could be removed by macroalgal harvesting ranges between 413–1285 and 54–167 kg ha^{-1} y^{-1} for N and P, respectively (**Table 4**). These amounts account for a removal of 1521–4732 tonnes y^{-1} of N and 198–615 tonnes y^{-1} of P if related to the entire harvesting area of 3682 ha. In the case of the Venice Lagoon, the nutrients removed would be almost equal or even higher than the nutrients released into this basin on a yearly basis (Solidoro et al., 2010). However, the continuous supply of nutrients both from the rivers that flow into this area and from the urban centers of Mestre and the historic center of Venice would guarantee long-term production.

The potential revenue of macroalgal-harvesting is about the same as clam-farming by considering a clam yield of 2–6 kg m^{-2} during a 2-year period, as it occurs in the most productive environments such as some areas of the Venice Lagoon and the Po Delta[2] (Orel et al., 2000).

However, in both cases the economical assessment didn't consider the harvesting and processing costs therefore the final profit would be actually lower. But, while the activities of clam-farming and clam-harvesting are highly destructive and endangers the environment (Sfriso et al., 2005), the macroalgal harvesting can be managed without risks favoring a constant improvement of the environment through the removal of considerable amounts of nutrients.

[1] https://www.alibaba.com/showroom/prices-agar-agar.html. (last access March 10, 2019).

[2] http://www.gral.venezia.it/attachments/159_Piano_uso_sostenibile.pdf (last access March 10, 2019).

TABLE 2 | *In field* annual net production and growth rate of some macroalgae in the Venice Lagoon.

Species	Very turbid waters			Clear waters		
	g fw m^{-2} y^{-1}	g fw m^{-2} d^{-1}	% d^{-1}	g fw m^{-2} y^{-1}	g fw m^{-2} d^{-1}	% d^{-1}
Agarophyton vermiculophyllum	12,848	35.2	1.10	22,016	60.3	1.88
Gracilaria gracilis	9024	24.7	0.77	21,856	59.9	1.87
Gracilariopsis longissima	10,576	29.0	0.91	28,192	77.2	2.41
Mean	10,816	29.6	0.93	24,021	65.8	2.06

TABLE 3 | Economical value of ecosystems colonized by aquatic angiosperms, macroalgae and clams.

	Economical value			
Ecosystems colonized by aquatic angiosperms	Multiple services (maintenance of marine biodiversity, regulation of the quality of coastal waters, protection of the coast line, etc.)		US$ ha^{-1} y^{-1} 15,837	Terrados and Borum (2004)
			19,002	Costanza et al. (1997)
Eutrophic ecosystems colonized by macroalgae	Yield Biomass = $\frac{9.0 - 28.0 \text{ kg fw } m^{-2}\, y^{-1}}{\text{Mean values of 35 countries}}$	US$ kg^{-1} fw 0.35	US$ ha^{-1} y^{-1} 31,500–98,000	Sfriso and Sfriso (2017)
	Yield Agar = $\frac{0.41 - 0.73 \text{ kg fw } m^{-2}\, y^{-1}}{\text{Mean value}}$	US$ kg^{-1}		Sfriso et al. (2017b)
		11	45,100–80,300	Food and Agriculture Organization of the United Nations [FAO] (2012, 2018)
		17	69,700–124,100	www.alibaba.com
Eutrophic ecosystems colonized by clams	Yield Clams = $\frac{1 - 3 \text{ kg } m^{-2}\, y^{-1}}{\text{Mean value}}$	US$ kg^{-1} 4	40,000–120,000	Orel et al. (2000)

TABLE 4 | Estimate of the potential nutrient removal by macroalgal harvesting taking into account the mean nutrient concentrations.

		Nitrogen		Phosphorus	
	ha	kg ha^{-1} y^{-1}	Total tonnes y^{-1}	kg ha^{-1} y^{-1}	Total tonnes y^{-1}
Pialassa della Baiona	270	413–1285	112–347	54–167	15–45
Sacca di Goro	300		124–386		16–50
Veneto Po Delta	900		372–1157		48–150
Venice Lagoon	2212		914–2843		119–370
Total	3682		1521–4732		198–615

Pialassa della Baiona, with a surface of 11 km^2 out of whom ca. 3 km^2 are colonized by a dense biomass of Gracilariaceae, especially *A. vermiculophyllum,* is the smallest TWS, but its potential yield ranges between 1099 and 1960 tonnes of commercial agar accounting for US$ 12.1–33.3 million (**Table 5**), depending on nutrient availability and water transparency. Sacca di Goro, which is invaded by the same species, shows a similar revenue (US$ 13.4–37.0 million) whereas in the Veneto lagoons of the Po Delta the potential revenue reaches US$ 40.3–111.1 million. This value could be reached in only 7% (ca. 30 km^2) of the Venice Lagoon surface in the areas prevalently colonized by Gracilariaceae [especially: *A. vermiculophyllum, G. gracilis, Gracilaria bursa-pastoris* (S. G. Gmelin) P.C. Silva and *G. longissima*], where a minimum of 9002–16,058 tonnes of commercial agar could be produced achieving a potential revenue in the range of US$ 99–177 million. The four considered environments have altogether a surface of ca. 37 km^2 (3682 ha) suitable for algal exploitation where no aquatic angiosperms are present or clam-harvesting can occur. These areas have a potential yield of ca. 15–27 ktonnes y^{-1} of commercial agar, accounting for US$ 165–454 million, depending on the different market surveys.

As for clams, this analysis does not take into account the production costs, which can be very variable, and depend on the location, the size of the plant, the costs of the biomass production/harvesting and processing, and the country tax rates.

Herrera-Rodriguez et al. (2018) by a techno-economic analysis of industrial agar production from *Gracilaria* sp. in North Colombia pointed out the role of plant size for a profitable agar production. They found that for a plant with a processing capacity of 8,640 tonnes y^{-1} and life of 15 years, the break-even production capacity was around 4,200 tonnes of red algae per year.

Delgado et al. (2018) analyzed two different routes (freeze-thaw and evaporation) for agar extraction at industrial scale in order to select the process that offers greater profitability under defined criteria. The effect of raw material cost and plant location on profitability of both routes was also evaluated through sensibility analysis. It was found that evaporation route showed a higher profitability due to its lower fixed capital, operating and utilities costs. But they indicated that raw material cost as the most influential factor in seaweed profitability and this depended on plant location and total tax rate applied by the Government to the industrial sector.

On the other hand, an analysis of costs for clam production in the lagoon of Venice and the Po Delta showed that the revenue depended on the type of company, the means used: small boat, fishing gear with vibrating rake or manual rake. For example, a company that harvest 100 kg of clams per day with a small boat selling them at 3 euros per kg would reach the break-even production after 186 days of work, while a fishing gear with vibrating rake that collects 150 kg per day would reach a balance after 139 days. In the Po Delta the clam fishing with manual rake that harvests 25 kg of clam per day selling them at 4 euros per kg would reach the break-even production after 119 days (Mauracher et al., 2011).

Therefore, in both cases: macroalgal and clam exploitation, the net gain depends on many factors which are difficult to predict as they depend on the environment considered. However, these sources, if adequately managed, can guarantee a profitable exploitation of these eutrophic environments, especially the abundant macroalgal biomass naturally produced in the lagoons of the Northern Adriatic Sea, whose oculate exploitation can promote environmental recovery with nutrient abatement. Indeed, the management of Gracilariaceae (Sfriso and Sfriso, 2017; Sfriso et al., 2017b), but also of invasive allochthonous Laminariales such as *Undaria pinnatifida* (Harvey) Suringar and Fucales such as *Sargassum muticum* (Yendo) Fensholt (Armeli-Minicante et al., 2016; Sfriso et al., 2020) that have

TABLE 5 | Estimate of the potential revenue obtainable from the production of agar in areas with Gracilariaceae biomass >3 kg m^{-2}.

Lagoon	Surface				Agar production		Potential revenue	
							FAO	Alibaba
	Total lagoon (km^2)	With Gracilariaceae cover >3 kg m^{-2} (km^2)	Cover (%)	Effective cover (ha)	Tonnes ha^{-1} y^{-1}	Total (tonnes y^{-1})	US$ /kg	
							11 (million US$)	17 (million US$)
Pialassa della Baiona	11	3	90	270	4.07–7.26	1099–1960	12–22	19–33
Sacca di Goro	26	4	75	300		1221–2178	13–24	21–37
Veneto Po Delta	178	15	60	900		3663–6534	40–72	62–111
Venice Lagoon	549	30.3	73	2212		9002–16,058	99–177	153–273
Total				3682		14,985–26,731	165–294	255–454

colonized the lagoon of Thau (France, Verlaque, 2001) and the lagoon of Venice (Italy, Sfriso and Facca, 2013) could be a sustainable solution to restore the environment and at the same time complement traditional fishing or replace mollusk-farming which has severe environmental impacts. In addition, the biomass control could help avoid anoxic crises and their environmental and socio-economic repercussions (Morand and Briand, 1996; Ménesguen, 2018). Even though, harvest should be conducted under strict protocols to avoid environmental impacts as sediment disturbances and resuspension, as well as other disturbances on aquatic flora and fauna.

CONCLUSION

Transitional water systems have a significant impact on the quality of life, the well-being, and the economy of riparian populations. However, environmental quality is often in strong contrast with anthropogenic activities that affect it and trigger negative effects also of an economic nature. The high economic value provided by ES of poorly impacted basins colonized by aquatic angiosperms is well known. However, degraded environments characterized by a significant macroalgal biomass, if carefully managed, could provide and create complementary resources, even higher value than aquatic angiosperms do. They could integrate the revenues from other activities such as traditional fishing, a good cultural heritage to maintain, or progressively replace clam harvesting/farming activities which are negative for the environment, and at the same time contributing to the recovery of eutrophic areas by self-financing the removal of high amounts of nutrients.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/supplementary material.

AUTHOR CONTRIBUTIONS

All the authors have participated to the sampling efforts, the data analysis, and manuscript preparation.

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REFERENCES

Arico, S., Bridgewater, P., El-Beltagy, A., Harms, E., Program, S., Hepworth, R., et al. (2005). *Ecosystems and Human Well-Being*. Washington: Island Press.

Armeli-Minicante, S. A., Michelet, S., Bruno, F., Castelli, G., Vitale, F., Sfriso, A., et al. (2016). Bioactivity of phycocolloids against the mediterranean protozoan leishmania infantum: an inceptive study. *Sustainability* 8, 1–9.

Ballesteros, E., Torras, X., Pinedo, S., García, M., Mangialajo, L., and de Torres, M. (2007). A new methodology based on littoral community cartography dominated by macroalgae for the implementation of the European water framework directive. *Mar. Poll. Bull.* 55, 172–180. doi: 10.1016/j.marpolbul.2006.08.038

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., and Silliman, B. R. (2011). The value of estuarine and coastal ecosystem services. *Ecol. Monograph.* 81, 169–193.

Barker, T., Mortimer, M., and Perrings, C. (2010). Biodiversity, ecosystems and ecosystem services. *Econ. Ecosyst. Biodivers. Ecol. Econ. Found* 41–104.

Buia, M. C., Gambi, M. C., and Dappiano, M. (2003). Seagrass systems, in: mediterranean marine benthos: a manual of methods for its sampling and study, ed. *M.C. Gambi, M. Dappiano Biol. Mar. Mediterr.* 11, 133–183.

Cecere, E., Petrocelli, A., Izzo, G., and Sfriso, A. (2009). *Flora and Vegetation of the Italian Transitional Water Systems*. Venezia: CoRiLa, Stampa Multigraf Spinea.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., et al. (1997). The value of the world's ecosystem services and natural capital. *Nature* 387, 253–260.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., et al. (1998). The value of ecosystem services: putting the issues in perspective. *Ecol. Econ.* 25, 67–72. doi: 10.1016/s0921-8009(98)00019-6

Delgado, ÁG., León-Pulido, J., and Peralta-Ruíz, Y. Y. (2018). Economic evaluation and comparison of two topologies for agar production from red algae. *Contemp. Eng. Sci.* 11, 659–667. doi: 10.12988/ces.2018.7994

Duarte, C. M. (2013). "The role of Seagrass in Climate Change Mitigation and Adaptation," in *The University of Western Australia and Spanish National Research Council* (Bonn), 24–25.

European.Union [Eu]. (1992). Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Offi. J. L* 206, 0007–0050.

European.Union [Eu]. (2000). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. *Off. J. L* 327, 0001–0073.

Food and Agriculture Organization of the United Nations [FAO] (2012). *Fisheries, and Aquaculture Department. The State of the World's Fisheries and Aquaculture*. FAO: Rome, 209.

Food and Agriculture Organization of the United Nations [FAO] (2018). *The State of World Fisheries and Aquaculture 2018 – Meeting the Sustainable Development Goals*. Rome: Licence.

Francavilla, M., Franchi, M., Monteleone, M., and Caroppo, C. (2013). The red seaweed gracilaria gracilis as a multi products source. *Mar. Drugs* 11, 3754–3776. doi: 10.3390/md11103754

Francavilla, M., Manara, P., Kamaterou, P., Monteleone, M., and Zabaniotou, A. (2015). Cascade approach of red macroalgae Gracilaria gracilis sustainable valorization by extraction of phycobiliproteins and pyrolysis of residue. *Bioresour. Technol.* 184, 305–313. doi: 10.1016/j.biortech.2014.10.147

Friedlander, M., and Levy, I. (1995). Cultivation of gracilaria in outdoor tanks and ponds. *J. Appl. Phycol.* 7, 315–324. doi: 10.1007/bf00004005

González-Leija, J.A., Hernández-Garibay, E., Pacheco-Ruíz, I., Espinoza-Avalos, J., López-VivasJosé, J. M. Bautista-Alcantar, J., et al. (2009). Optimization of the yield and quality of agar from Gracilariopsis lemaneiformis (Gracilariales) from the Gulf of California using an alkaline treatment. *J. Appl. Phycol.* 21, 321–326. doi: 10.1007/s10811-008-9370-0

Guerry, A. D., Ruckelshaus, M. H., Arkema, K. K., Bernhardt, J. R., Guannel, G., Kim, C.-K., et al. (2012). Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. *Int. J. Biodivers. Sci.Ecosyst. Serv. Manag. First* 2012, 1–15.

Haines-young, R., and Potschin, M. (2018). *Common International Classification of Ecosystem Services Guidance on the Application of the Revised Structure*. Nottingham: Fabis Consulting Ltd.

Hattam, C., Atkins, J., Beaumont, N., Börger, T., Böhnke-Henrichs, A., Burdon, D., et al. (2015). Marine ecosystem services: Linking indicators to thei classification. *Ecol. Indic.* 49, 61–75. doi: 10.1016/j.ecolind.2014.09.026

Herrera-Rodriguez, T., Parejo-Palacio, V., and González-Delgado, ÁD. (2018). Technoeconomic sensibility analysis of industrial agar production from red Algae. *Chem. Eng. Transactions* 70, 2029–2034.

Kornfeldt, R. A. (1982). Relation between nitrogen and phosphorus content of macroalgae and the waters of Northern Oresund. *Bot. Mar.* 25, 197–201.

Lignell, A., Ekman, P., and Pedersèn, M. (1987). Cultivation technique for marine seaweeds allowing controlled and optimized conditions in the laboratory and on a pilotscale. *Bot. Mar.* 30, 417–424.

Liquete, C., Piroddi, C., Drakou, E. G., Gurney, L., Katsanevakis, S., Charef, A., et al. (2013). Current status and future prospects for the assessment of marine and coastal ecosystem services: a systematic review. *PLoS One* 8:e67737. doi: 10.1371/journal.pone.0067737

Maroli, L., Pavoni, B., Sfriso, A., and Raccanelli, S. (1993). Concentrations of polychlorinated biphenyls and pesticides in different species of macroalgae from the lagoon of Venice. *Mar. Pollut. Bull.* 26, 553–558. doi: 10.1016/0025-326x(93)90405-9

Mauracher, C., Pellizzato, M., and Trevisan, G. (2011). "Una Valutazione Tecnico-Economica Del Comparto Veneto Della Vongola," in *Le vongole dell'alto Adriatico tra Ambiente e mercato,* ed C. Trevisan (Milano: Franco Angeli), 102–136.

Ménesguen, A. (2018). *). Les Marées Vertes. 40 Clés Pour Comprendre. Édition Quae RD 10.* France: Versailles,.

Millenium Ecosystem Assessment, (2005). *Ecosystems and Human Well-Being: Current States and Trends.* Washington, D.C: Island Press.

Morand, P., and Briand, X. (1996). Excessive growth of macroalgae. A Symptom of environmental disturbance. *Bot. Mar.* 39, 491–516.

Munari, C., and Mistri, M. (2012). Ecological status assessment and response of benthic communities to environmental variability: the Valli di Comacchio (Italy) as a study case. *Mar. Environ. Res.* 81, 53–61. doi: 10.1016/j.marenvres.2012.08.008

Nordlund, L. M., Koch, E. W., Barbier, E. B., and Creed, J. C. (2016). Seagrass ecosystem services and their variability across genera and geographical regions. *PLoS One* 11:e0163091. doi: 10.1371/journal.pone.0163091

Orel, G., Boatto, V., Sfriso, A., and Pellizzato, M. (2000). *Piano Per la Gestione Delle Risorse Alieutiche Delle Lagune Della Provincia di Venezia.* Benevento: Sannioprint.

Orfanidis, S., Panayotidis, P., and Stamatis, N. (2001). Ecological evaluation of transitional and coastal waters: a marine benthic macrophytes-base model. *Mediterr. Mar. Sci.* 2/ 2, 45–65.

Orfanidis, S., Panayotidis, P., and Ugland, K. I. (2011). Ecological evaluation index continuous formula (EEI-c) application: a step forward for functional groups, the formula and reference condition values. *Mediterr. Mar. Sci.* 12, 199–231.

Orlandini, M., and Favretto, L. (1987). "Utilisation of macroalgae in Italy for pollution abatement and as source of energy and chemicals," in *COST-48, Symposium of Sub-Group III, Crema-L'Homeau* (France: CNRS-IFREMER), 25–28.

Phycomorph, (2015). "Advancing knowledge on seaweed growth and development," in *COST Action FA*1406 (2015–2019). Brussels: European Commission

Pranovi, F., and Giovanardi, O. (1994). The impact of hydraulic dredging for shortnecked clams, Tapes spp., on an infaunal community in the lagoon of Venice. *Sci. Mar.* 58, 345–353. doi: 10.1016/j.marenvres.2015.05.003

Sfriso, A., Buosi, A., Wolf, M. A., and Sfriso, A. A. (2020). Invasion of alien macroalgae in the Venice Lagoon, a pest or a resource? *Aquat. Invasion* 15, (in press).

Sfriso, A., Buosi, A., Facca, C., and Sfriso, A. A. (2017a). Role of environmental factors in affecting macrophyte dominance in transitional environments: the Italian Lagoons as a study case. *Mar. Ecol.* 38, e12414. doi: 10.1111/maec.12414

Sfriso, A., and Facca, C. (2007). Distribution and production of macrophytes in the lagoon of Venice. Comparison of actual and past abundance. *Hydrobiologia* 577, 71–85. doi: 10.1007/s10750-006-0418-3

Sfriso, A., and Facca, C. (2011). Macrophytes in the anthropic constructions of the Venice littorals and their ecological assessment by an integration of the "CARLIT" index. *Ecol. Indic.* 11, 772–781. doi: 10.1016/j.ecolind.2010.10.002

Sfriso, A., and Facca, C. (2013). Annual growth and environmental relationships of the invasive species Sargassum muticum and Undaria pinnatifida in the lagoon of Venice. *Est. Coast. Shelf Sci.* 129, 162–172. doi: 10.1016/j.ecss.2013.05.031

Sfriso, A., Facca, C., Bonometto, A., and Boscolo, R. (2014). Compliance of the macrophyte quality index (MaQI) with the WFD (2000/60/EC) and ecological status assessment in transitional areas: The Venice lagoon as study case. *Ecol. Indic.* 46, 536–547. doi: 10.1016/j.ecolind.2014.07.012

Sfriso, A., Facca, C., and Ghetti, P. F. (2007). Rapid quality index (R-MaQI), based mainly on macrophyte associations, to assess the ecological status of mediterranean transitional environments. *Chem. Ecol.* 23, 1–11.

Sfriso, A., Facca, C., and Marcomini, A. (2005). Sedimentation rates and erosion processes in the lagoon of Venice. *Environ. Int.* 31, 983–992. doi: 10.1016/j.envint.2005.05.008

Sfriso, A., and Marcomini, A. (1994). Gross primary production and nutrient behaviours in shallow lagoon waters. *Bioresour. Technol.* 47, 59–66. doi: 10.1016/0960-8524(94)90029-9

Sfriso, A., Marcomini, A., and Pavoni, B. (1994). Gracilaria distribution, production and composition in the lagoon of Venice. *Bioresour. Technol.* 50, 165–173. doi: 10.1016/0960-8524(94)90069-8

Sfriso, A. A., Gallo, M., and Baldi, F. (2017b). Seasonal variations and yields of sulfated polysaccharides in seaweeds from the Venice Lagoon. *Bot. Mar. Special Issue Phycomorph* 60, 339–349.

Sfriso, A. A., and Sfriso, A. (2017). In field gracilariaceae and Ulva biomass production: the venice lagoon as study case. *Bot. Mar. Special Issue Phycomorph* 60, 271–283.

Short, F. T., Polidoro, B., Livingstone, S. R., Carpenter, K. E., Bandeira, S., Bujang, et al. (2011). Extinction risk assessment of the world's seagrass species. *Biol. Conserv.* 144, 1961–1971.

Soedjak, H. S. (1994). Colorimetric determination of carragenans and other anionic hydrocolloids with methylene blue. *Anal. Chem.* 66, 4514–4518. doi: 10.1021/ac00096a018

Solidoro, C., Bandelj, V., Bernardi, F. A., Camatti, E., Ciavatta, S., Cossarini, G., et al. (2010). "Response of the Venice Lagoon Ecosystem to Natural and Anthropogenic Pressures over the last 50 years," in *Coastal Lagoons - Critical Habitats of Environmental Change*, eds M. J. Kennish, and H. W. Paerl (Boca Raton FL: CRC Press), 483–511. doi: 10.1201/ebk1420088304-c19

Strickland, J. D. G., and Parsons, T. R. (1972). *A Practical Handbook of Seawater Analyses.* Ottawa: Fish. Res. Board of Canada.

Terrados, J., and Borum, J. (2004). "Why are seagrasses important? Goods and services provided by seagrass meadows," in *European Seagrasses: An Introduction to Monitoring and Management. The M & MS Project*, eds J. Borum, C. M. Duarte, D. Krause-Jensen, and T. N. Greve (Brussels: European Community).

Tuya, F., Haroun, R., and Espino, F. (2014). Economic assessment of ecosystem services: Monetary value of seagrass meadows for coastal fisheries. *Ocean Coast.Manage* 96, 181–187. doi: 10.1016/j.ocecoaman.2014.04.032

Verlaque, M. (2001). Check-list of the macroalgae of Thau lagoon (Hérault, France): a hot spot of marine species introduction in Europe. *Oceanol. Acta* 24, 29–49. doi: 10.1016/s0399-1784(00)01127-0

Viaroli, P., Bartoli, M., Giordani, G., Naldi, M., Orfanidis, S., and Zaldívar, J. M. (2008). Community shifts, alternative stable states, biogeochemical controls and feed- backs in eutrophic coastal lagoons: a brief overview. *Aquat. Conserv.* 18, S105–S117.

Wakibia, J. G., Anderson, R. J., and Keats, D. W. (2001). Growth rates and agar properties of three gracilarioids in suspended open-water cultivation in St. Helena Bay, South Africa. *J. Appl. Phycol* 13, 195–207.

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Social-Environmental Analysis for the Management of Coastal Lagoons in North Africa

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This study provides an overview of 11 lagoons in North Africa, from the Atlantic to the Eastern Mediterranean. Lagoons are complex, transitional, coastal zones providing valuable ecosystem services that contribute to the welfare of the human population. The main economic sectors in the lagoons included fishing, shellfish harvesting, and salt and sand extraction, as well as maritime transport. Economic sectors in the areas around the lagoons and in the watershed included agriculture, tourism, recreation, industrial, and urban development. Changes were also identified in land use from reclamation, changes in hydrology, changes in sedimentology from damming, inlet modifications, and coastal engineering. The human activities in and around the lagoons exert multiple pressures on these ecosystems and result in changes in the environment, affecting salinity, dissolved oxygen, and erosion; changes in the ecology, such as loss of biodiversity; and changes in the delivery of valuable ecosystem services. Loss of ecosystem services such as coastal protection and seafood affect human populations that live around the lagoons and depend on them for their livelihood. Adaptive management frameworks for social–ecological systems provide options that support decision makers with science-based knowledge to deliver sustainable development for ecosystems. The framework used to support the decision makers for environmental management of these 11 lagoons is Drivers–Activities–Pressures–State Change–Impact (on Welfare)–Responses (as Measures).

Keywords: coastal lagoons, North Africa, water management, environmental assessment, adaptive management frameworks, social–ecological systems, DAPSI(W)R(M) a modified DPSIR, ecosystem services

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HIGHLIGHTS

- A social–environmental analysis of 11 North African coastal lagoons.
- The behavior of users is similar across North African lagoons.
- Responses (as measures) to problems are identified.
- Adaptive management of coastal lagoons can deliver more ecosystem services.

INTRODUCTION

Coastal lagoons are important zones for life between the land and the sea. They are one of the most productive environments and deliver ecosystem services (ES) that provide many ecological, cultural, and socioeconomic benefits, supporting a range of natural services that are highly valued by society (Gönenç and Wolflin, 2005; Newton et al., 2018). In the specific case of North Africa, coastal lagoons have a wide geographical distribution extending from the Atlantic Ocean along the western coastline of Morocco to the southern coastline of the Mediterranean Sea through Morocco, Algeria, Tunisia, Libya, and Egypt. In common with other regions of the world (Newton et al., 2014), the terminology used for North African lagoons varies from country to country and even among different regions in the same country that use different dialects and languages. These differences in terminology complicate research into historical knowledge about these lagoons systems. Nonetheless, it has been possible to identify most of the terms needed for this study (see section "Discussion").

Areas surrounding lagoons in North Africa have a long history of human occupation and utilization (Ramdani et al., 2001; Thompson and Flower, 2009; Trigui et al., 2012). These important social–ecological systems (SESs) continue to be heavily impacted by human activities, such as extraction of freshwater and urbanization, as well as the development of economic sectors, for example, fisheries, agriculture and tourism. Many of these SES are provided with some legal protection through international conventions, particularly for the protection of bird communities, in the case of Ramsar sites (Ramdani et al., 2009a; Sayoud et al., 2017). However, current levels of exploitation are unsustainable, and there is an increasing need for an integrated, basin-wide approach to the management of water resources and aquatic ecosystems in the region (Thompson and Flower, 2009). Indeed, there have been recent research projects to improve the understanding of these lagoon ecosystems and their surrounding resources (see section **Supplementary Material**).

The variety of issues in North African coastal lagoons (ecological functioning, biodiversity, productivity, human uses) requires conceptual models to organize, understand, and clarify issues and recapitulate information in a standard, logical, and hierarchical method (Patrício et al., 2016). The DAPSI(W)R(M) [Drivers-Activities-Pressures-State Changes–Impacts (on Welfare)–Responses (as Measures)] framework (Elliott et al., 2017) is the conceptual model that has been followed for the social environmental analysis of coastal lagoons in North Africa. In the context of this study, *State* refers to environmental state (e.g., water quality), ecological state (e.g., biodiversity), and consequently state of ES. This analysis enables local planners and state regulators to identify management solutions and measures for adaptation to changes in the lagoon ecosystems. An important aspect of these measures is the maintenance and improvement in the ES that these lagoons provide for human welfare.

MATERIALS AND METHODS

Study Sites: Distribution of North African Lagoons

North Africa is defined as the Southern Mediterranean Region or the North of the African Sahara (Desert). It is an area surrounded by the Atlantic Ocean to the west of Morocco, by the Mediterranean Sea to the north of Morocco, as well as the coast of Algeria, Tunisia, Libya, and Egypt, and by the Red Sea to the east of Egypt (**Figure 1**).

There are 22 lagoons in North Africa, 4 along the Atlantic shore of Morocco and 18 along the South Mediterranean Region. They are shallow systems, permanently linked to the sea and regulated by tidal exchanges and fluxes at the sediment interface.

Eleven lagoon sites were selected on the basis of available data and literature; the importance of their social ecological and economic roles in the chosen countries; and their different key environmental characteristics in terms of basin surface, land use distribution, typology, lagoon size, the adjacent sea (Atlantic, Mediterranean), dynamics, and climate. The 11 coastal lagoons include Khenifiss and Oualidia (Morocco) along the Atlantic shore, and Nador (Morocco), El Mellah (Algeria), Bizerte (Tunisia), Tunis (Tunisia) and Boughrara (Tunisia), Farwa (Libya), Marsa Matrouh (Egypt), Bardawil (Egypt), and Burullus (Egypt) along the Mediterranean shore (names in white on black background, **Figure 1**).

Data Sources and Collection

The data were found in web-based searches using Web of Knowledge (ISI Web of Science), Science Direct (Scopus), Google Scholar, and Google searches. Searches were constrained to the period from 2000 to 2018 to focus on the most recent publications (531 articles). This identified peer-reviewed articles in ISI journals (e.g., 89 for Nador), as well as a significant body of "gray" literature (106 reports), such as reports by Environmental agencies or Ministries (e.g., 15 for Nador). There was considerable variation in the number of sources available for the different systems (e.g., 104 for Nador, only 24 for Kenifiss).

The search words were applied in English, French, and Arabic. Most of the articles listed in the *References* are from peer-reviewed literature in English and French, published for an international audience. There is also a substantial literature in Arabic and French, which is used in the North African countries for reports, which was consulted for this review, but this is not listed in the *References*.

The search keywords that were used, both singly and in combination, were as follows: "name" of the lagoon, including local variations on their names, see section "Confusion About Nomenclature of Lagoons in the Context of North African Lagoons"; "lagoon, lake, lac, lagune"; "country," also in the different languages." The studies identified were selected in a systematic manner based on title and abstract eligibility according to keywords, and then the full texts were screened. Many studies were removed because of missing data (313 articles and reports).

We then extracted information about the lagoons in relation to the drivers, activities, pressures, state changes, impacts, humans, and welfare issues as identified by Elliott et al. (2017) (e.g., agriculture, pH change, and underwater noise). The reported presence and absence of various activities/pressures/state changes/impact (on welfare) in the studied lagoons were extracted and tabulated, with some quantitative and qualitative details mentioned in each result section, if data were available (e.g., number of catches, number of boats, input of pollutants).

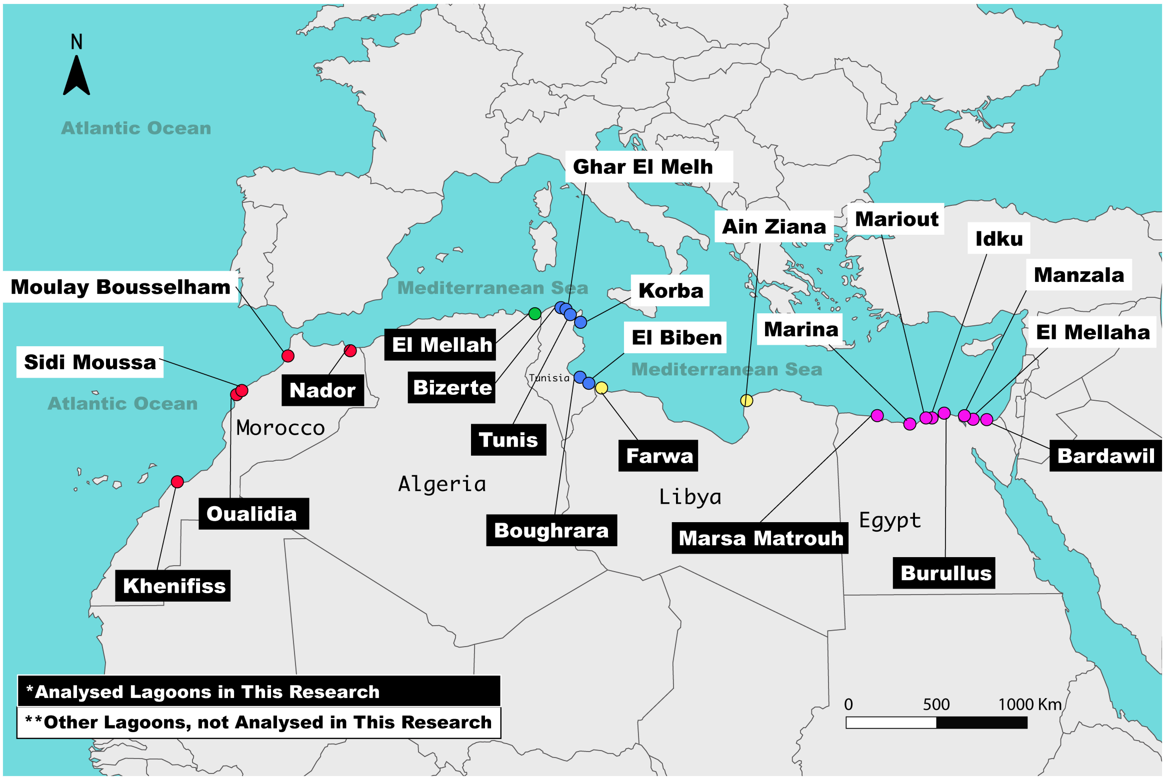

FIGURE 1 | Location of North African lagoons. The lagoon names in white on a black background* were analyzed in detail; those in black on white** were not analyzed. The dots show the lagoons in Morocco (red), Algeria (green), Tunisia (blue), Libya (yellow), and Egypt (purple).

Analytical Framework

DAPSI(W)R(M), a social–ecological framework for adaptive management from Elliott et al. (2017), was used for the analysis of the lagoons. This framework has evolved from earlier versions (Gari et al., 2015) starting with the PSR (Pressures–State–Response) framework proposed by Rapport and Friend (1979) and then developed and supported by the Organization for Economic Cooperation and Development (OECD) to organize its work on environmental policies and reporting (OECD, 1994). A later version is the DPSIR [Drivers–Pressures–State–Impacts (on Welfare)–Responses (as Measures)] used by, among others, the US Environmental Protection Agency (EPA, 1994), the United Nations Environment Programme (UNEP, 1994), the European Environment Agency (EEA, 1995), and the European Commission (EU, 2000). Patrício et al. (2016) and Elliott et al. (2017) highlight the anomalies and focus on the confusions of each component of DPSIR as a reason for improving the framework; for practicable management purposes, they advocate an extension of the framework to DAPSI(W)R(M). In this version, Drivers or driving forces indicate the basic needs of humans (Maslow, 1943), such as food, transport, and goods. These needs are addressed by society through socioeconomic **Activities**, such as fishing, aquaculture, and building infrastructure that have effects that produce **Pressures**, such as overextraction of resources, input of nutrients and/or heavy metals, pH changes, and so on. These effects lead to **State** changes on the ecology of the environment, as well as on the intermediate and final ES (Elliott et al., 2017). These changes have **Activities** (on human **Welfare**), such as provisioning services (e.g., fish food), regulating services (e.g., healthy climate), and cultural services (e.g., tourism). **Responses** (as **Measures**) to changes resulting from drivers, activities, and pressures require scientific knowledge to produce appropriate laws for governance, as well as a range of economic mechanisms and technologies to implement the appropriate management (Wolanski and Elliott, 2015; Elliott et al., 2017).

RESULTS

Drivers of Changes in North African Lagoons

Societal Drivers in relation to North African lagoon context are outlined below:

Stage 1_basic biological and physiological needs: these are fundamental necessities for all humans such as food, drink, shelter, and so on.

Stage 2_safety needs: these include protection from inclement natural phenomena, such as storms, droughts, and floods; law and order; allowing for the use of goods and services from lagoons; political stability; and allowing North African populations to live without fear.

Stage 3_love and belonging needs: these represent interpersonal relationships among North Africans; friendship, trust, and acceptance; receiving and giving affection and even love. At the scale of an individual, this involves belonging to a group, having friends, and creating a family. It also includes the behavior of stakeholders (e.g., fishers, farmers, workers), citizenship, and being part of society.

Stage 4_esteem needs: these are represented by an individual's achievements, self-respect, and status that are part of the common culture among North African people. For instance, men (e.g., fishers) are still the main wage earners of the household with their work providing them self-respect and the respect of others in the context of the local culture.

Stage 5_self-actualisation: the final stage is represented by realizing personal fulfillment. For example, the decision makers responsible for the Nador lagoon are trying to provide economic growth for the region based on sustainable management of the goods and services. By ensuring successful management of this type of ecosystem, they are setting an excellent case study not only for North Africa but also for the other countries in Africa and elsewhere.

Activities Associated With North African Lagoons

The assessment based on the human activities associated with North African lagoons is related mainly to general activities associated with major sectors rather than individual, specific activities. **Table 1** summarizes some of the more important activities (adapted from Smith et al., 2016) occurring in these lagoons that are presented in more detail in the following sections.

As far as possible, a common style and content has been used for the description of these activities, but there are big differences in the content and quantity of data that are available for the different lagoons.

Extraction of Living Resources and Aquaculture

Fishing

Artisanal fisheries have been identified as common activity in all lagoons, which has increased in the last decades due to growth in local demand linked to urban growth. The fisheries activities use small fishing boats because the lagoons are shallow waters (only a few meters). The boats are called "feluccas" in Morocco, Algeria, and Tunisia or "marceb" in Libya and Egypt.

Khenifiss lagoon (Morocco), an ecosystem protected from UNESCO World Heritage, has few fishing and aquaculture activities at present. Fisheries are more developed in Nador lagoon, where the tonnage of the fish catches increased from 480 tons in 2001 to 1,157 tons in 2012, which represents an increase of 241%. The monthly net profit per boat is approximately 222 Euros (2,465 Moroccan Dirhams). Overall, the main fish catches are eel (i.e., *Anguilla anguilla*), bream (i.e., *Sparus aurata*), and gray and red mullet (i.e., *Mullus barbatus*). Other catches include cuttlefish, octopus (i.e., *Octopus vulgaris*), and shrimp (i.e., *Penaeus japonicus)*. The 887 fishers of this lagoon are all male with an average age of 41 years (including sailors), while boat owners are of an average age of 48 years. For 46% of fishers, fishing is their only economic activity, whereas the others work seasonally in agriculture or small commerce (Najih et al., 2015).

The Mellah lagoon (Algeria) is also well known for its artisanal fishery activities with an annual average fish production of approximately 40 tons. The production comprises eels (58%), followed by mullet (31%) represented by *Mugil cephalus*, *Liza aurata*, *Liza saliens*, *Liza ramada*, and *Chelon labrosus*. There is also harvesting of mollusks including clams (i.e., *Ruditapes decussatus*) and cockles (i.e., *Cerastoderma glaucum*) (Chaoui et al., 2006). The only available data about fishers are from 1982, with 13 fishers supervised by the Enterprise Nationale Algérienne de Pêche-Unité Aquacole; they are all male with an average age of 44 years, supporting a total of 67 children (Food and Agriculture Organization, 1982).

The catch from Farwa lagoon (Libya) is 26 tons annually, mainly fish, shellfish, and octopus, but there are no available data about numbers of fishers (Banana and Mohamed, 2016).

Fishing production in Egypt has increased from an annual production of 140,400 tons in 1980 to 1,079,500 tons in 2009, with 934,000 tons coming from inland areas. This is an important sector for the country, representing 2 billion Euros in income. Egyptian lagoons represent 27.8% of total fisheries production in the country and are mostly for crab (El Nahas et al., 2017).

The reason for the large variation in the number of fishers and catches among North African lagoons depends on the size of the lagoon (both area and depth), as well as the population surrounding the lagoon.

Aquaculture

Moroccan aquaculture started in the 1950s with the launch of shellfish farming in the Oualidia lagoon, with a pilot project for breeding oysters. Currently, a dozen farms continue to operate at this lagoon. Oyster production reached 60 tons in 2011, with shell fishing representing 15% of total production in the region (Maanan et al., 2014).

Shellfish production in Tunisia started in 1964 in Bizerte lagoon (Ghribi et al., 2016) and included mussels, oysters, and clams (Fertouna-Bellakhal et al., 2014; Turki et al., 2014). Current production is more than 100 tons per year.

In Libya, mussel and oyster cultivation is ongoing at aquaculture farms on Farwa lagoon.

In Egypt, fish farms have been developed along the lagoon shores of Burullus. The annual production is approximately 52,000 tons (Eid and Shaltout, 2014). Bardawil lagoon (Egypt) is a rich environment for the extraction of living resources. The production of high-quality fish is mainly exported to Europe, enabling an increase in economic development (El-Kassas et al., 2016). This activity produces approximately 80% of the exported fish and crustacea from Egyptian lagoons (El-Kassas et al., 2016; Nassar et al., 2018).

Agriculture and Golf

Agricultural practices have a high impact on lagoons environments. In North Africa, many lagoons are surrounded by agriculture and, in some cases, have been modified by land reclamation, draining, and infilling to enable agriculture (e.g., Burullus in Egypt). In recent years, some areas around lagoons

TABLE 1 | The major economic sectors and activities in North African lagoons: "X" represents documented, and "?" undocumented* activities.

Country	Morocco			Algeria	Tunisia			Libya	Egypt		
Lagoon	**Khenifiss**	**Oualidia**	**Nador**	**El Mellah**	**Bizerte**	**Tunis**	**Boughrara**	**Farwa**	**Marsa Matrouh**	**Burullus**	**Bardawil**
Activity/economic sector											
Extraction of living resources	X	X	X	X	X	X	X	X	X	X	X
Aquaculture	X	X	X	X	X	X	X	X	X	X	
Agriculture		X	X	X	X			?	X	X	X
Golf			X						X		
Tourism and recreation		X	X	?		X		X	X	X	X
Extraction of non-living resources		X	X					X	X		X
Land-based industry		X	X	X	X	X	X	X	X	X	X
Land reclamation						X			X	X	X
Damming of streams		X	X		X					X	
Inlet consolidation (hard structures that fix the position of the inlet)			X		X		X		X		
Navigational dredging		X	X		?	X	X			X	X
Coastal infrastructure		X	X	X		X			X	X	X
Transport and shipping	X	X	X	X	X	X		X	X	X	X

*List of activities adapted from Smith et al. (2016). *Activity is mentioned in informal communications such as newspapers, websites, and so on.*

have been converted to golf courses, although the actual locations and environmental impacts have yet to be fully analyzed.

In Morocco, an area of 92 km^2 surrounding the Nador lagoon is agricultural, most of which is irrigated (El Yaouti et al., 2008; García-Ayllón, 2017). There is intensive cereal production to the south of the lagoon (Giuliani et al., 2015a), with barley occupying 56% of total area, olive trees nearly 22%, and other crops approximately 22%. Fishing activities, agriculture, and livestock production employ up to 46% of the working population in the region (Najih et al., 2015). There are intensive agricultural activities for vegetables and use of fertilizers (Maanan et al., 2014) in the area surrounding Oualidia lagoon (Morocco), which have expanded in area from 10.52% in 1946 to 40.15% in 2006.

Only 9% of the watershed for the El Mellah lagoon (Algeria) is agricultural with an area of 7.34 km^2, of which 1.30 km^2 is dedicated to livestock, mainly cattle and goat production (Melouah, 2013).

The Bizerte lagoon (Tunisia) has a catchment area of 480 km^2, of which agricultural activities cover 117 km^2 area, with 78, 34, and 5 km^2 dedicated to cereal culture, horticulture, and arboriculture, respectively (Garali et al., 2009).

The cultivated area around Burullus lagoon (Egypt) has increased by 68.7%, from 231.31 km^2 in 1984 to 336,60 km^2 in 2015 (Husain et al., 2016). This lagoon receives approximately 4 billion m^3 of water per year of drainage water from a catchment area of approximately 403 km^2 of the agricultural land in the Nile Delta (El-Amier et al., 2017; Orabi et al., 2017).

Tourism and Recreation

Lagoons ecosystems are attractive to tourism for both scenic and cultural characteristics. The most popular touristic lagoons in North Africa are Nador and Oualida in Morocco, Marina, and Marsa Matrouh in Egypt.

In Morocco, the Oualidia lagoon provides many touristic activities including sailing, bathing on beaches, bird watching, and nature watching. The seaside resort has a diversity of landscapes, and in summer, the population increases by 54,000[1] with a daily capacity of 30,000 people. Tourism activities have also been developed around the Nador lagoon, particularly in the northeastern part of the lagoon with new residential complexes, resorts, villas, marinas, restaurants, riads, and a golf academy. This ecosystem has 20 km of coastline comprising the Boqueronisa and Arkman beaches. The new Med Marchica project aims to provide 101,200 beds and seven marinas and to employ 80,000 people by 2020.

The Farwa lagoon in Libya is well known for kitesurfing, which attracts tourists during the windy season, from November to March.

The Marsa Matrouh is a coastal lagoon area in Egypt where tourism is expanding. There are 10 beaches that extend for a distance of 7 km, five of them are situated in the lagoon (Abdel, 2015). In summer, up to 450,000 visitors enjoy the attractive, white, sandy beaches and turquoise seawater (Khaled et al., 2014). Because of these ecosystems services, 25 hotels have been established surrounding the lagoon and employ 7,000 workers (Mulazzani et al., 2017).

Extraction of Non-living Resources and Land-Based Industries

Many lagoons in North Africa are used for extraction of non-living resources, such as mining activities, and are also associated with industrial areas.

[1] www.hcp.ma

Nador lagoon in Morocco has several industrial activities, including mining and metallurgy, such as iron, pyrite, pyrrhotite, and chalcopyrite. The industrial effluent from the Selouane industrial zone is discharged into the lagoon during the wet season (Piazza et al., 2016).

The Bizerte lagoon (Tunisia) has been the site of industrial development since the 1950s. There are industrial complexes of approximately 130 factories located in the towns of Bizerte, Zarzouna, Menzel Abderrahmn, Menzel Jemil, and Menzel Bourguiba. The first and biggest was a cement factory (1950), followed by the "El Fouled" steelworks (1967). Other factories are for metallurgy, electronic industries, textiles, chemical production, petrochemicals including oil refineries, agro-alimentary production, cement manufacturing, and fish processing.

The GCCI (General Company of Chemical Industries) industrial complex is located in the east part of the Farwa lagoon (Libya) at Abu-Kamash. This complex was opened in the 1970s, containing three units with an annual production of 104,000 tons of ethylene dichloride, 60,000 tons of polyvinyl chloride, 50,000 tons of caustic soda, and 45,000 tons of chlorine, as well as the production of sodium carbonate, sodium hypochlorite, and hydrochloric acid (Banana and Mohamed, 2016). These industrial processes also lead to the discharge of mercury into the lagoon ecosystems (Banana and Mohamed, 2016).

Urbanization, Land Reclamation, and Coastal Infrastructures

Intensive urbanization has occurred around North African lagoons. Essentially, these areas are attractive for both residents and visitors, with increasing land reclamation and construction of a coastal infrastructure, which enables the development of new urban areas. Thus, these regions are now highly populated, with numbers increasing annually.

Over the past 50 years, the population around the Oualidia lagoon (Morocco) has rapidly increased by 240% from 7,741 inhabitants in 1971 to 18,616 inhabitants in 2014. The expansion of the city has modified some sedimentary forms especially the dunes on the east side of the lagoon (Yamna et al., 2014). However, the most populated lagoon in Morocco is Nador, with an increase in inhabitants from 683,914 to 859,590 between 1994 and 2014[2]. Recently, the Moroccan government through the "Marchica agency" has invested 4 billion Euros in seven new projects: Cité Atalayoun, Cité des 2mers, Nador new city, fisherman's village, Baie des Flamants, Marchica Sport, and Les Vergers de Marchica, which are destined as houses, apartments, a harbor, and a research center and natural park for resident and touristic growth; some are constructed already, and some are still under construction.

The inlet located at the center of the Nador lagoon has been modified many times during the last decades (Raji et al., 2013). In 2011, there was an enlargement of the inlet to a width of 300 m and a depth of 6 m to improve circulation and shipping. This included the 1,450-m east breakwater and the 1,350-m west breakwater (Daghor et al., 2016).

Monastir lagoon (Tunisia) has been transformed by port structures, marinas, hotels, golf courses, and an airport. This lagoon has been completely drained and has not been included in the list of North African lagoons (**Figure 1**). Tunis lagoon (Tunisia) is the most urbanized lagoon in North Africa. The population had doubled from 887,803 (2004) to 1,507,000 inhabitants (2014)[3]. The largest airport in Tunisia (Tunis–Carthage Airport) and military airbase have also been constructed from land reclaimed from the lagoon. Furthermore, the lagoon has been split by the construction of a road "La Goulette," dividing the lagoon into a northerly and a southerly section.

Bizerte lagoon (Tunisia) has four harbors (Harbor of Bizerte, Menzel Abderrahmane harbor, Menzel Bourguiba, and The Carrier bay harbor parked) (Abidli et al., 2016).

Marsa Matrouh is an important city along the 500 km Mediterranean coast between Alexandria city and the Libyan border in the north part of Egypt. It has 193,000 inhabitants, which represent a density of 0.9 inhabitants km^{-2} (Khaled et al., 2014). The area has many five-star hotels.

Transport and Shipping

North Africans societies have used and are still using lagoons for transport and shipping activities by developing harbors and marinas for commercial purposes. The shipping includes oil tankers, passenger cruise liners, packet boats, container ships, and military vessels, as well as smaller boats for artisanal fishing.

In the Nador lagoon (Morocco), the number of "feluccas" for artisanal fishing increased by 30% from 300 to 390 between 2001 and 2012.

In the Tunisian lagoon of Bizerte, there were 997 boats in 2010, including 490 fishing boats, and 507 commercial vessels from passenger cruise liners, oil tankers, bulk carriers, container ships, and gas and oil tankers, as well as military vessels. The Menzel Bourguiba harbor receives approximately 542 boats annually including 480 fishing "feluccas", as well passenger cruise liners, commercial ships, ferries, and military vessels. There is also a shipyard that provides shipbuilding, maintenance and painting. The Menzel Abderrahman harbor has approximately 181 artisanal fishing boats (Abidli et al., 2016; Lahbib et al., 2018). The Tunis lagoon has a commercial harbor at Rades-La Goulatte used by 2,902 merchant ships, including "feluccas", passenger cruise liners, packet boats, and container ships (Lahbib et al., 2018). The Boughrara lagoon has 50 "feluccas" in the Boughrara harbor and approximately 100 in Adjim harbor (Lahbib et al., 2018).

The Marsa Matrouh lagoon (Egypt) has 66 "feluccas" with no motors, 6 "feluccas" with outboard motors, and 17 with inboard motors that are used for fishing with trammel nets and long lines. The small "feluccas" contribute 20% to the fishing activity production, whereas larger boats contribute 80% within an annual production of approximately 272 tons (Mulazzani et al., 2017). In the Bardawil lagoon, 1,235 "feluccas" of 4 to 6 m are used for fishing activity. The Burullus lagoon has 2,098 boats where 2,049 are small "feluccas"

[2] www.hcp.ma

[3] http://dataportal.ins.tn/fr/Map

TABLE 2 | The existing pressures on North African lagoons: "X" represents documented and "?" represents undocumented pressures.

Country	**Morocco**			**Algeria**	**Tunisia**			**Libya**	**Egypt**		
Lagoon	**Khenifiss**	**Oualidia**	**Nador**	**El Mellah**	**Bizerte**	**Tunis**	**Boughrara**	**Farwa**	**Marsa Matrouh**	**Burullus**	**Bardawil**
Pressure											
Smothering					X	X					X
Substratum loss						X					X
Changes in siltation		X	X			X			X	X	
Abrasion			X		X	X		X			X
Selective extraction of non-living resources		X	X							X	X
Presence of underwater noise	X	X	X	X	X	X	X	X	X	X	X
Presence of litter					X	X	X			X	
Thermal regime change		X			X	X	X		X	X	X
Salinity regime change			X		X	X	X		X	X	X
Introduction of synthetic compounds		X	X	X	X	X			X	X	X
Introduction of non-synthetic compounds		X	X	X	X	X	X	X	X	X	X
Introduction of radionuclides		X								X	X
Introduction of other substances		X	X	X	X	X	X	X		X	
Nitrogen and phosphorus enrichment		X	X	X	X	X	X	X	X	X	X
Input of organic matter		X	X	X	X	X	X	X	X	X	X
Introduction of microbial pathogens		X		X	X	X	X			X	X
Introduction of non-indigenous species and translocations		X	X				X				
Selective extraction of species				X	X	X	X				X
Death or injury by collision		X	X	X	?					X	
Barrier to species movement		X			X	X					
Water ?ow rate changes		X			?	X	X		X		X
pH changes		X				X	X		X	X	X
Change in wave exposure									X		

List of pressures adapted from Smith et al. (2016).

(4–6 m), 41 are medium boats (6–8 m), and eight are large boats (>8 m).

Pressures on North African Lagoons

In the context of North African lagoons, there are multiple pressures that have different forms originating from the variety of economic activities (**Tables 1**, **2**). The distribution of these pressures among the 11 selected lagoons is shown in **Table 2**, and more details about specific pressures are described below.

Pressures From Agricultural Effluents

Agriculture in the drainage basins along the coast of North Africa is responsible for contaminating lagoons with agrochemicals such as fertilizers, pesticides, and herbicides. Additionally, there are pressures from the manure derived from animal rearing.

The main pressure from the high nutrient inputs to the Oualidia lagoon in Morocco is from surface and groundwaters that drain from the cultivated areas that cover 78% of the catchment (Damsiri et al., 2015, 2017) and where nitrogen and phosphate fertilizers are applied (Maanan et al., 2014). These nutrients promote a significant increase in chlorophyll *a* within the lagoon ecosystem (El Asri et al., 2017a). Another Moroccan example of pressures from agrochemicals is an irrigated agricultural area of 92 km^2 around the Nador lagoon (Re and Sacchi, 2017).

In the case of Algeria, many studies describe the effect of agricultural pollutants and livestock manure (Nadira, 2008) from the two rivers "R'kibet and El Mellah" (Chaoui et al., 2006; Magni et al., 2015) on fish, mollusks, and crustaceans (Mebarki et al., 2015) in the El Mellah lagoon.

The pressure from runoff draining the agricultural areas around Bizerte lagoon (Tunisia) increases with rainfall in winter. Runoff contains various contaminants and pollutants including nutrients, pesticides and heavy metals (Kamel et al., 2014). The increase in nutrient concentrations, especially during the wet season, stimulates phytoplankton blooms (Béjaoui et al., 2017).

One of the most heavily polluted lagoons in North Africa is Burullus on the Nile delta in Egypt. Nutrient, pesticide, and metal inputs come from eight streams and a canal draining from the adjacent watershed (Orabi et al., 2017). In the southern area, there are high concentrations of tin (144 ppm) and of arsenic (44 ppm), far exceeding the World Health Organization guidelines for arsenic in soil (1.5 ppm) (El-Monsef et al., 2017).

Pressures From Extraction of Living Resources

There are multiple pressures from the extraction of living (e.g., fishing, shellfish harvest) and non-living resources (e.g., dredging and quarrying for sand). These result in selective extraction of species, death or injury of fauna, introduction and translocation of non-indigenous species, substratum loss, abrasion and resuspension of sediment, smothering, nutrient mobilization, and underwater noise (**Table 2**).

The Oualidia lagoon is an ecosystem with high pressure from the extraction of living and non-living resources such as traditional fisheries (fish and molluscs), oyster aquaculture (Bocci et al., 2016), seaweed harvesting, and sand extraction (Beryouni et al., 2012; Maanan et al., 2014; El Asri et al., 2015). These activities have led to pressures such as input of organic matter from aquaculture and blooms of phytoplankton (Damsiri et al., 2017). Nador lagoon is also known for its aquaculture activities and fishing. The use of small mesh sizes has exerted a pressure on the fish population.

There are pressures on El Mellah lagoon (Algeria) from aquaculture with organic matter enrichment of sediment (Embarek et al., 2017).

Pressures on the Boughrara lagoon (Tunisia) include organic matter from the introduction of non-indigenous species, such as the invasive species polychaete *Branchiomma bairdi* McIntosh (1885), which settles in high densities (up to 35 m^{-2}) on buoys and hulls of vessels (Khedhri et al., 2017b).

There is little information about Farwa lagoon (Libya). However, one resulting pressure of the fishing activity is underwater noise pollution (Essghaier et al., 2013).

The fishing, harvesting, and aquaculture extraction of living resources from Bardawil lagoon (Egypt) exert significant pressures on this ecosystem (i.e., overfishing).

Pressures From Urban and Industrial Effluents

There are three types of effluent affecting lagoons including (i) urban sewage untreated or through a wastewater treatment plants (WWTP); (ii) urban wet weather discharges including rainwater, runoff water, and discharges from separated stormwater system outfalls resulting from rainfall in an urbanized catchment; these flow into the lagoons without passing through a wastewater treatment system (Gooré et al., 2015); and (iii) the industrial effluents through a local or common WWTP. These pressures increase from both urban growth and climate change effects and can lead to a state change by contamination and pollution.

Some of the lagoons (Nador in Morocco, El Mellah in Algeria, Tunis in Tunisia, and Burullus in Egypt) are enriched with nutrients from agriculture (**Table 1**), but there are additional pressures from urban and industrial effluents (Oczkowski et al., 2008; Tlig-Zouari and Maamouri-Mokhtar, 2008; Nassar and Gharib, 2014; Alves Martins et al., 2015; Derradji et al., 2015; Bocci et al., 2016; Daghor et al., 2016; Hammani et al., 2016; El Asri et al., 2017b; El-Zeiny and El-Kafrawy, 2017; Khedhri et al., 2017a).

The increasing tourist development activities around lagoons in North Africa have also increased pressures from domestic effluents due to the seasonal increase of population, especially during the summer.

Oualidia lagoon (Morocco) is located between two cities "El Jadida" and "Safi," adjacent to one of the biggest phosphate mines in the world and fertilizer plants. Indeed, the surface water of the lagoon is phosphate enriched (Damsiri et al., 2017). Furthermore, this lagoon is known also for touristic activities and a population increase by a factor of four times during summer time. This overwhelms the capacity of the WWTP, increasing the nutrient pressures. The WWTP of Nador city was built in 1980 and expanded in 1990, but it has become inadequate due to urban growth (Ruiz et al., 2006). Two new WWTP were constructed in 2010; the Grand Nador WWTP is an activated sludge plant with a daily treatment of 14,000 m^3, and El Aroui WWTP is a natural lagoon with a treatment capacity of 2,500 m^3 per day. These treatment plants are undersized; thus, there are pressures from effluent into the lagoon, as well as anaerobic microbial degradation (Giuliani et al., 2015b). Furthermore, there is industrial effluent coming from the Selouane industrial area that is discharged into the lagoon during the wet season. The pressures from industrial effluent (Nador lagoon) include inputs of pollutants, such as polychlorinated biphenyls (PCB), polycyclic aromatic hydrocarbons (Giuliani et al., 2015b), and increasing inputs of polybrominated diphenyl ethers (Piazza et al., 2016).

The effluent from industries around Bizerte lagoon (Tunisia) has only primary wastewater treatment (Barhoumi et al., 2014; Hammani et al., 2016). SACEM (electrical transformers) uses 900 tons of PCB (Barhoumi et al., 2014). These activities have led to chemical contamination of mussels (Barhoumi et al., 2014).

The Tunis lagoon is vulnerable to pressures from industrial effluent and domestic sewage due to the doubling of the local resident population during the last 10 years.

Marsa Matrouh lagoon (Egypt) is contaminated by pressures from metal pollution (Abdel, 2015) and inadequate sewage treatment due to high tourist influx (Gharib et al., 2011).

Pressures From Civil Engineering Projects

Infrastructure for the management of the lagoons' ecosystems includes artificial channels for maritime navigation of harbors and ports. Examples include the modification of inlets at Moulay Bousselham and Nador in Morocco, El Mellah in Algeria, Boughara in Tunisia, and Bardawil in Egypt.

Nador lagoon (Morocco) has had a new inlet since 2011 that was made to increase water exchange with the sea (Bocci et al., 2016). Most of the modern centers of Tunis city have been built on land reclaimed from the Tunis lagoon (Thornton et al., 1980). A comparison between aerial photographs and topographic maps produced between 1902 and 2002 shows the pressures of coastal infrastructure and the expansion of the city at the northern and southern limits of the lagoon (Chouari, 2015). The eastern inlet (El Kantra Channel) of Burullus lagoon (Tunisia) has been enlarged from a narrow passage of 12.5 m across in 2004 to 160 m in 2007. This has increased the daily water exchange with the sea from 0.8 million m^3 to 6.9 million m^3 per day (DGPA, 2001). The choking of the inlets in Bardawil lagoon (Egypt) has led to pressures such as siltation of the ecosystem that has increased significantly. It has also changed the water exchange between the sea and the lagoon, causing sedimentation at the entrance to the inlets (Nassar et al., 2018).

Pressures From Transport and Shipping

Pressures from transport and shipping activities were found in all North African lagoons. These include (i) underwater noise from motorized ferries, vessels, and yachts; (ii) hydrocarbons from accidental oil spills from motorized boats; and (iii) antifouling agents that are based on organic diluters mixed with highly concentrated toxic metals such as copper and zinc (Guerra-García and García-Gómez, 2005) or tributyltin (TBT) compounds.

These multiple pressures often result from conflicting interests among stakeholders, as well as from previous development and management measures for the lagoons (Lillebø, 2015). The pressures (**Table 2**) result in a state change of the environment such as symptoms of eutrophication in the lagoons (e.g., Nador in Morocco, Bizerte and Tunis in Tunisia, Bardawil in Egypt). These changes affect ES (Newton et al., 2018) and can lead to an impact on human welfare.

State Change in North African Lagoons

Human activities and modifications to North African lagoon ecosystems and their catchment areas have resulted in increased pressures that have modified the state of these ecosystems. The change of state can be at the environmental, ecological, and/or ecosystem level. **Table 3** shows the state changes in lagoon ecosystems (components and processes), intermediate services (supporting and regulating services), and the final ES (provisioning, regulating, and cultural) of the lagoons. No state changes have yet been detected for the protected Khenifiss lagoon, an almost pristine lagoon.

The changes to the inlets of Nador lagoon (Morocco), especially the most recent ones in 2011, have altered the hydrological processes and nutrient condition of the ecosystem. Eutrophication has stimulated algal blooms of *Caulerpa prolifera*, *Gracilaria bursa-pastoris*, and *Colpomenia sinuosa*. These changes have modified the state of ES components, intermediate services, and regulating and supporting services, as well as final ES (El Asri et al., 2017c).

The environmental state of El Mellah lagoon (Algeria) has been degraded by the presence of contaminants and pollutants, such as heavy metals and neurotoxic pesticides (Benradia et al., 2016). The application of pesticides from agricultural activities and pressures from inputs of organophosphates affects the ecological state of the benthos, for example, the toxic effect of malathion on clams (Nadji et al., 2010). As this is a species of economic value, the delivery of provisioning ES has declined. Furthermore, the change in salinity, due to aquaculture activities, has caused a decline in bird species diversity (2013−2017) degrading cultural services, such as bird watching (Telailia et al., 2017).

The intensive use of fertilizers on agricultural activities in the catchment area of Bizerte Lagoon (Tunisia) has modified an intermediate service through increased pressures from high inputs of nitrogen and phosphorus that have altered the state of nutrient condition and the state of phytoplankton biomass and primary production (Béjaoui et al., 2017), especially in the inner part of the lagoon. Another state change is in the hydrography and water cycling (intermediate service) of the Bizerte lagoon with an estimated deficit of -7.5 Mm^3 of the lagoon's annual water flowing into the Mediterranean sea (Béjaoui et al., 2017).

The subsidence land rate around Tunis lagoon (Tunisia) is 22 mm $year^{-1}$, which may lead to state change through marine submersion and flooding hazards (Ennesser et al., 2011). The enlargement of the eastern inlet of Boughrara lagoon (Tunisia) has not improved the environmental state or ecological state. Harmful algal blooms (HAB) have increased organic matter accumulation and stimulated the growth of the polychaete population (Khedhri et al., 2017a), changing the state of regulating and supporting services.

Pressures from the industrial wastewater of a chemical company have affected the environmental state of Farwa lagoon (Libya) that has been highly contaminated by mercury, during the period January to August 2014 (Banana and Mohamed, 2016). Marine flora (fish, cuttlefish, and oyster) and fauna of the lagoon have been contaminated as high concentrations of mercury were found in different species (Banana and Mohamed, 2016), impacting the provisioning and regulating services.

The Egyptian lagoons are the most affected by human activities, and resulting pressures have led to significant changes in their state of environment. For instance, the large increase of water inflow from agricultural freshwater drainage (from 81.7% in 1971 to 98.18% in 2003) has changed the hydrological state of Burullus lagoon (El-Adawy et al., 2013). The decrease in the environmental state of salinity has led to a change in intermediate services and final ES. The provisioning services of total marine fish decreased from 15.99 to 1.81% by weight (El-Adawy et al., 2013). Inflow of contaminants changed the state of Burullus lagoon (Orabi et al., 2017), where symptoms

TABLE 3 | The state change in North African lagoons: "X" represents documented and "?" undocumented state change.

Countries		Morocco			Algeria	Tunisia			Libya	Egypt		
Lagoon		**Khenifiss**	**Oualidia**	**Nador**	**El Mellah**	**Bizerte**	**Tunis**	**Boughrara**	**Farwa**	**Marsa Matrouh**	**Burullus**	**Bardawil**
State change												
Lagoons ecosystems	Components		X	X	X	X	X	X	X	X	X	X
	Processes		X	X	X	X	X	X	?	X	X	X
Intermediate services	Supporting		X	X	X	X	X	X	?	X	X	X
	Regulating		X	X	X	X	X	X	?	X	X	X
Final ecosystem services	Provisioning		X	X	X	X	X	X	X	X	X	X
	Regulating		X	X	X	X	X	X	X	X	X	X
	Cultural		?	?		?	X	X	?	X	X	X

These are based on Elliott et al. (2017) ES classification.

of microbial anaerobic degradation have been observed from agricultural input (Giuliani et al., 2015a). A further, devastating pressure is the destruction of the dune barrier system to use the sediments for reclamation purposes (El-Asmar et al., 2013). Marsa Matrouh lagoon (Egypt) is contaminated by metals (Abdel, 2015) and domestic sewage (Gharib et al., 2011) that promote phytoplankton blooms.

Sedimentation in front of the inlets of Bardawil lagoon (Egypt) has changed the environmental state by restricting the exchange of water between the sea and the lagoon (Nassar et al., 2018), which could lead to changes in the provisioning of final ES such as fishing activities. Meanwhile, the ecological state has changed because of overfishing pressures during recent decades. For instance, the crab landings increased from 754.2 tons in 2000 to 2,053.1 tons in 2009 and then decreased to 518.7 tons in 2014, followed by an increase to 1,973.4 tons in 2015, which represents 42% of total production. There are also changes in ecological state due to a new invasive species of crab, *Callinectes sapidus* (Abdel Razek et al., 2016), which affects nets and netted fish. The armored dinoflagellate *Alexandrium* species, which is a toxic species, now accounts for 17% of total Dinophycae taxa (El-Kassas et al., 2016).

The environmental state of sediments in Marsa Matrouh lagoon (Egypt) has deteriorated with contamination and pollution from the trace metals vanadium, aluminum, tin, arsenic, and selenium (Abdel, 2015). This deterioration in sediment quality affects the regulating and provisioning final ES; for example, high concentrations of aluminum (100 $\mu g\ g^{-1}$; wet weight) have been observed in the fish *Pagellus erythrinus* and may damage its liver (Abdel, 2015).

Impact (on Human Welfare) in North African Lagoons

Human activities and resulting pressures that cause the degradation of the environmental state, the ecological state, and the delivery of ES of coastal lagoons may ultimately impact on human welfare (**Table 4**). For example, lagoons are nursery habitats for juveniles of commercial species; therefore, changes in the state of the environment and ecology lead to a loss of this nursery service that supports fisheries and thus provisioning ES. Impacts on human welfare are detected in most of the North African lagoons, except the Khenifiss lagoon (Morocco), which is not impacted at present. Other impacts result from toxins in HAB and the contamination of seafood by chemicals.

Civil engineering and infrastructures that modify the connectivity of most of the lagoons in the study can change the wave exposure on part of the shoreline and cause erosion. This can lead to negative impacts on the shoreline protection of the lagoon with an increased risk of flooding or storm damage to the human population due to the erosion or removal of sediment, which can aggravate conflicts among stakeholders (Conde et al., 2019; **Table 4**).

The Marsot aquaculture activities in Nador and Oualidia lagoons (Morocco) started in 2005 but ceased in 2010 due to environmental problems. This impacted local jobs and the availability of seafood. The presence of highly persistent pollutants, such as PCB and PAH in Nador lagoon (Morocco), has contaminated seafood and is a risk to human health (Giuliani et al., 2015a). The harvesting and selling of oysters from the breeding site of Oualidia lagoon were prohibited in March 2017 by the Department of Marine Fisheries of Morocco (Huffpostmaghreb, 2017) because of possible impacts on human health. The welfare of fishers of Nador lagoon was impacted by the loss of catches, leading to protests in 2018.

The degradation of environmental state due to organic overenrichment of sediments of El Mellah lagoon (Algeria) has decreased the biodiversity and the availability of seafood in some areas, especially during the summer of 2015 (Magni et al., 2015).

The pollution in Bizerte lagoon (Tunisia) affects not only the aquatic organisms but also impacts human health. For example, the mollusc disease "Marteiliosis" that is caused by the protozoan parasite *Marteilia* species has infected *Mytilus*

TABLE 4 | Summary of Impacts on human welfare in North African lagoons: "X" represents documented and "?" undocumented impacts on human welfare.

Countries	Morocco			Algeria	Tunisia			Libya	Egypt		
Lagoon	**Khenifiss**	**Oualidia**	**Nador**	**El Mellah**	**Bizerte**	**Tunis**	**Boughrara**	**Farwa**	**Marsa Matrouh**	**Burullus**	**Bardawil**
Impact on human welfare											
Loss of revenue or jobs from declining fish catches		X	X	X	X	X		?		X	X
Loss of revenue or jobs from Harmful Algal Blooms			?		X		X			X	X
Loss of revenue or jobs from pollutants in seafood		X	X	X	X		X	X		X	
Loss of seafood provision or risk to public health form contaminated seafood (toxins or chemicals)		X			X			?		X	X
Risk to public safety due to subsidence, flooding, or storm damage							?		?	X	

galloprovincialis, which has impacted provisioning services of a commercial species (Elgharsalli et al., 2016). Furthermore, the high concentrations of TBT in seafood, such as commercial species of bivalves (*R. decussatus*, *C. glaucum*, *P. nobilis*, and *M. galloprovincialis*), have impacted both revenues and seafood provision (Abidli et al., 2016).

The increased loading of organic matter and the presence of HAB in Boughrara lagoon (Tunisia) have caused high fish mortality and thereby an impact on food provisioning (Khedhri et al., 2017b).

The mercury contamination from the GCCI company detected in the fish, oysters, and cuttlefish in Farwa lagoon (Libya) may represent a health risk to people living in the area (Kim et al., 2016; Ha et al., 2017).

The high concentrations of metals detected in fish from Marsa Matrouh lagoon (Egypt) impact both the provision of seafood and are a risk to human health (Abdel, 2015).

The chemical pollutants and salinity changes in Burullus lagoon (Egypt) have had an effect on fish diversity that has declined from 32 to 25 species (Fadili et al., 2016). This impacts the provision of commercial fish species (El-Zeiny and El-Kafrawy, 2017) to people living in the region (Orabi et al., 2017). Furthermore, modifications to the inlet can increase the recession rates of the shoreline, which impacts the security of the inhabitants by decreasing the sea defenses (supporting service) and their livelihoods from tourism revenue (cultural service) (Nassar et al., 2018).

The revenues from fisheries in the Bardawil Lagoon (Egypt), food provision, and public health are threatened by HAB (Abdel Razek et al., 2016; El-Kassas et al., 2016).

DISCUSSION

North African lagoons provide many supporting, regulation, provisioning, and cultural services. Nevertheless, they are subject to numerous activities that represent sources of conflict among the different users (Newton et al., 2014; Dolbeth et al., 2016; Newton and Elliott, 2016; Lillebø et al., 2017) producing multiple pressures that represent a negative state change on the environment and impact human welfare.

Most drivers, activities, pressures, state change, and impacts (on welfare) in North African lagoons are similar to those identified in European lagoons (Newton et al., 2014; Dolbeth et al., 2016), because of similar activities and behavior of the stakeholders using lagoons ecosystems.

There have been several, past management responses in North Africa aimed at reducing nutrient and phosphorus enrichment, protection of the aquatic resources, and developing environmental regulation to control aquaculture industries in the lagoons. These include building sewage treatments plants, improving aquatic resources, providing technical support, and monitoring and scientific research.

Past Management Measures

The construction of domestic sewage treatment plants has been the main management measure in the past. Nevertheless, sewage treatment is still non-existent or inadequate for many North African lagoons.

Morocco launched a National Plan of Sanitation in 2005. As a result, Nador lagoon has two UWWTP, Moulay Bousselham and Oualidia have one, but Sidi Moussa has none. Morocco also created the National Agency for Aquaculture Development in 2011, which has developed environmental measures that should be followed by the aquaculture industries. The National Agency for Aquaculture Development is the leading actor for the aquaculture industry to promote the development of sustainable aquaculture along the Moroccan coast, including lagoons.

Algeria built a UWWTP behind El Mellah lagoon, but the water quality is still poor with respect to fecal microbes (Kherifi and Bousnoubra-Kherici, 2016). There is an unresolved conflict between the managers of the El Kala National Park and the inputters to this lagoon, mainly industries and farmers (Telailia et al., 2017).

Urban WWTP were built in the lagoons of Bizerte, Tunis, Boughrara, and Ghar El Melh (Tunisia) because of sewage contamination. Tunis lagoon was the worst affected by the contamination, but there was a restoration project in 1985 to stop pollution and eutrophication (Vandenbroeck and Rafik, 2001), and the conditions were improved by collecting waste and algae around the margin and dredging. However, the long-term monitoring plan has not been continuously maintained for nutrients and total suspended solids, but only for water temperature and salinity. Nevertheless, it highlighted sudden meteorological events (rainfalls) and/or unpredictable accidental pollution (Trabelsi et al., 2013).

No data about environmental management responses in Libya have been found in this study.

The Ministry of Environment of Egypt updated the National Biodiversity, Strategy and Action Plan for the years 2015—2030 (Temraz et al., 2016). This includes wetland habitats in Egypt. The targets are to reduce the rate of wetland loss by 50% by 2021, improve water efficiency in farming by 50%, and develop inland water ecosystems (Finkl, 2017).

Policy Instruments With Respect to North African Lagoons

North African lagoons are important at the local, national, and regional scale to achieve sustainable development. Regulatory authorities are responsible for monitoring programs to protect biodiversity and ES of the lagoons. This can be achieved by policy instruments to protect these ecosystems.

At the national level, Morocco implemented new legislation (Conseil Economique, Social et Environnemental, 2014) in 2014 for the protection of coastal areas, without a specific mention of lagoons. For instance, the law prohibits building on a strip of land 100 m wide, adjacent to the shoreline. The new law also prohibits the discharge of wastewater, waste, and any pollutants in the coastal area without specific authorization and respecting specific limits for discharges (Conseil Economique, Social et Environnemental, 2014). Another law that is under development calls for the conservation of fisheries ecosystems and the protection of the marine environment against pollution

(Secrétariat Général du Gouvernement Marocain, 2017). Morocco has created a special planning agency for the Nador Lagoon, the Agence d'Amenagement de la Marchica.

In 2002, Algeria passed legislation (Journal Officiel de la Republique Algerienne, 2002) for the protection and development of the coastal areas. The law includes lagoons as ecosystems, but uses the term "lido," which can be confused with coastal lakes.

Tunisia has created a special agency, named Agence de protection et d'aménagement du littoral[4], for the planning and the protection of coastal areas including lagoons.

Libyan authorities issued legislation (no. 14) for marine protected areas in 1992 that specifies sites that include lagoons, based on the biological, physical, and socioeconomic universal criteria (Haddoud and Rawag, 1995).

The Egyptian environmental affairs agency implemented Law 4 for the protection of the environment Amended by Law 9/2009 (Egyptian Environmental Affairs Agency, 2009) to ensure the environmental protection in coastal zones as an Integrated Environmental Management of Coastal Zones without defining lagoons specifically but including them within wetlands.

At the Mediterranean region scale, there are a number of policies relevant to lagoons in the context of the management of coastal areas. These include the Ramsar convention for wetlands; the Barcelona Convention (1975) aimed at the protection of the marine environment and the coastal region of the Mediterranean; and the Integrated Coastal Zone Management "ICZM" Protocol signed by most Mediterranean countries, except Egypt and Libya. The action plan for the ICZM protocol was implemented for the period 2011–2019, to strengthen capacities for the use of ICZM policies, instruments, tools, and processes, as well as promoting the protocol within the region and worldwide.

At the global level, the United Nations defined 17 Sustainable Development Goals (SDGs) and 244 indicators to be met individually and collectively by the signatory states, including North African countries (UN, 2015). Four SDGs are relevant to lagoon ecosystems both directly and indirectly. The most relevant to lagoons is SDG 14 "Life below Water" followed by SDG 6 "Clean Water and Sanitation," The SDG 15 "Life and Land," and the SDG 13 "Climate Action."

Options for Management Measures

Identifying the sensitive and vulnerable zones in the lagoons allows managers to prioritize immediate measures for water bodies at risk (Ferreira et al., 2006). This knowledge also allows managers to protect specific habitats, such as seagrass beds, and specific, keystone species that provide supporting and provisioning ES (e.g., food).

Once the management options have been identified, these should be checked to see whether they follow the 10-tenet approach (Elliott, 2013; Barnard and Elliott, 2015; Elliott et al., 2017), to be both successful and sustainable. The measures should include both short- and long-term plans that can deliver immediate, positive results, as well as mitigate and prevent future issues. One of the main short-term measures is the construction of UWWTP in some lagoons, for example, Farwa and Ain Zaina in Libya and Sidi Moussa in Morocco. The capacity of existing UWWTP is insufficient because of the seasonal increase of population due to visiting tourists. This occurs mainly summer when temperatures are high and oxygen solubility is low. Existing UWWTP can be increased in capacity or upgraded from primary to tertiary treatment, for example, Oualidia and Tunis lagoons, and the installation of nutrient removal ponds using plants such as reed beds. These measures will reduce the organic and nutrient loading into the lagoons and improve their oxygen conditions to mitigate future eutrophication. As the Mediterranean climate is hot and dry in summer, eutrophication increases during the summer and algal proliferation reaches its maximum. Effluent from UWWTP can be reutilized as a fertilizer for some crops, for example, fruit trees. The use of UWWTP effluent for irrigating lawns and golf courses can improve the sustainability in arid countries.

Another possible management measure is the construction of belt canals around the lagoons within small areas in order to intercept and divert rainwater and agricultural runoff. Moreover, increasing green areas in cities and villages and constructing uncemented cobblestone roads instead of concrete-asphalt streets can help increase the infiltration rate of rain into the soil (Rocheta et al., 2017).

Additional management measures can be implemented to restore water and sediment fluxes, especially for the lagoons with dams in the catchment and/or with inlet modifications, for example, Moulay Bousselham in Morocco, Bizerte and Boughrara in Tunis, and Marina in Egypt.

Another effective measure is monitoring and protecting surface water and groundwater from pollution sources surrounding the lagoon. This includes protecting water resources from excessive extraction, as well as contamination and pollution from runoff of agrochemicals, industrial effluents and emissions.

Management measures to reduce litter include conveniently located bins, environmental notices targeting the tourists, litter collection, and cleaning of lagoon beaches in summer, for example, Marsa Matrouh in Egypt and Oualidia and Nador in Morocco.

Land subsidence and increased risk of flooding in the area surrounding the lagoon can be avoided by an urbanization and land use plan to avoid overconstruction and establish setback lines, for example, Tunis lagoon.

Different types and intensity of monitoring programs, surveillance operational and investigative monitoring (EU, 2000), can be implemented as a response to different risks. The monitoring plans should consider seasonality and hydrological, physiochemical, biological, and ecological aspects of the lagoons, and not focus on only one aspect, such as the concentration of a pollutant.

Awareness raising campaigns, especially for the inhabitants of the lagoon area, can be very effective. Managers can work with the local schools to organize educational field trips around the lagoons. The research group for the protection of birds in Morocco previously organized guided tours for individuals, local associations, and local schools in the Oualidia lagoon to educate and raise awareness regarding the importance of lagoons ecosystems. During 2017, 1,863 students and

[4]http://www.apal.nat.tn/site_web/index.html

professors were introduced to the value of lagoons by this research group[5].

Conflict management can be improved by organizing a stakeholder forum with timely stakeholder consultation to reach agreement on how to manage these ecosystems (Newton and Elliott, 2016). In Nador lagoon (Morocco), civil society confronted the Marchica development project, which led to protesters questioning administration decisions regarding the prohibition of access to certain sites, the protection of local properties, and the inclusion of management processes.

Measures to protect culturally important areas are essential to reduce conflict between the local community and visitors. Cross-cultural sensitivity is necessary to maintain respect for local cultures and populations (e.g., typical seafood such as "Mussel Tagine," and typical design of traditional boats). Cultural erosion can be mitigated through cultural adaptation, for example, using traditional boats for tourism around the lagoon. The *gondolas* of Venice lagoon are an example of this, as are the *moliceiro*s of Aveiro lagoon.

Economic instruments (fines, taxes, subsidies) can also be used in management. The "polluter pays" principle advocated by the OECD can be used so that economic sectors are held to account if they exceed emission standards, especially for pollutants. Aquaculture, textile, and oil industries are all examples of polluters within these lagoon systems.

A policy response focused on the lagoons ecosystems may result in new regulations to reduce the impact of activities and pressures from the surrounding areas.

Supporting sustainable investment by lowering taxes in order to balance economic growth, environmental awareness and actions, food security, and social development can lead to new blue economy activities such as bioagriculture, ecotourism, ecological shipping, and organic aquaculture. This could ensure an increase in blue growth related to North African lagoons.

Confusion About Nomenclature of Lagoons in the Context of North African Lagoons

The terminology used for coastal lagoons is extremely varied throughout the world, and it hampers the transfer of knowledge (Newton et al., 2014). The vocabulary also changes among North African countries even if they are using the same language (Arabic, Amazigh, and the two official foreign languages, French in Morocco, Tunisia and Algeria, and English in Libya and Egypt). "Lagune" is the most common French term used in Maghreb countries. Moroccan Arabic dialect includes "Merja" as in Merja Zerga Moulay Bousselham and sometimes the term "sebkha." Others names include "Mar Chica" for Nador lagoon, a remnant of Spanish colonization. In Algeria, "Lac," which means lake, is used, for example, "Lac El Mellah," and also "Ildo" for coastal lakes. Tunisia also uses the French term "lac" such as "lac de Tunis," "lac Bizerte." "Bouhaira" is the Arabic term. Farwa "Island" is the term used for "Farwa lagoon" in Libya or "Jazirat Farwa," which means an island. The term "Bouhaira" is used in most of the Egyptian lagoons, "Bihira" in the eastern part and sometimes using even "Bouhaira El Malha," which means "Salted lake," but in English, lagoons are still referred to as "lake" such as Burullus or Manzala lake. Moreover, Libya has used "lagoon" for four ecosystems, of which only two can be considered lagoons as Ain Ghazala and El Burdi are coastal inundations.

Knowledge Gaps in North African Lagoons

The analysis carried out in the present study revealed the lack of interdisciplinary studies on lagoons. Most studies are in specific areas, mostly in natural sciences such as biological, physicochemical, environmental, and hydrological studies with a few geological studies (Ayache et al., 2009; Flower et al., 2009; Ramdani et al., 2009b; Thompson et al., 2009; Raji et al., 2018).

North African countries need targeted but interdisciplinary studies about coastal lagoons to provide more information and knowledge in order to better understand the issues and provide more appropriate management measures. For instance, Libya has some literature reviews completed through international collaboration (Essghaier et al., 2013; Mahmoud Khamis and El-Sayed El-Sayed, 2015; Banana and Mohamed, 2016), but lacks a governmental platform to provide data about the lagoons.

More data are needed for all North African lagoons, especially on the relationship between the water catchment and the lagoons that can provide more knowledge about pressures. Social, economic, and even cultural studies about coastal lagoons are generally not available. These can provide more knowledge about the ES and the impact on human health and human welfare.

CONCLUSION

The information about North African lagoons is diffuse and heterogeneous, and the terminology is often confusing. North African coastal lagoons are complex, social–ecological environments; thus, an interdisciplinary approach is needed for their analysis and management. The DAPSI(W)R(M) approach used in the research (Elliott et al., 2017) has evolved from the previous applications of DPSIR approach for lagoons analysis (Newton et al., 2014; Dolbeth et al., 2016).

The research mapped the existing North African lagoons, and 11 of 21 coastal lagoons were analyzed. The most common economic sectors and human activities are the extraction of living resources (mostly fishing and aquaculture) followed by agriculture, transport and shipping, land-based industries, coastal infrastructure, and urbanization as secondary major activities in most of the North African lagoons.

The most common pressures in North African lagoons are underwater noise, introduction of synthetic (e.g., pesticides) and non-synthetic compounds (e.g., heavy metals), and input of organic matter (e.g., sewage), as well as nutrient and phosphorus enrichment (e.g., fertilizers).

The least affected of the lagoons is Khenifiss lagoon, an almost pristine system, because it is a protected area and listed as a world heritage site for UNESCO. However, in the remaining North African lagoons, there are many state changes on the environment. These affect lagoon hydrology, salinity, connectivity, erosion and accretion, nutrient and chemical

[5]http://www.grepom.org/wp-content/uploads/Rapport-annuel-_Activité_Centre-Walidia_2017.pdf

contamination, dissolved oxygen, and redox potential to varying degrees. There are also many state changes in the ecology of coastal lagoons both from the ecological components and ecosystem processes, which include intermediate services (supporting and regulating services) and the final ES (provisioning, regulating, and cultural). These changes have led to an impact on human welfare, especially in provisioning services (e.g., revenue from catches and provision of food) and other impacts on human welfare such as health (contaminated seafood).

Lack of knowledge about the value of lagoon ES and the impacts on human welfare hampers the sustainable management of North African lagoons. Some immediate responses as management measures, such as improving UWWTP, could be implemented by North African countries. The complexity of the lagoon systems also calls for the adoption of participatory methods that include all stakeholders in order to manage the multiple issues affecting the social and economic services and ES of these important environments. This can foster a more transdisciplinary approach and improve the implementation of policies aimed at coastal protection and improving ES. An investment in sustainable extraction of resources, bioagriculture, tourism, ecological transport and shipping, and raising awareness could ensure the blue growth of the lagoons in North Africa benefiting both the surrounding population and the ecology.

In summary, DAPSI(W)R(M) is an adaptive management framework for social–ecological systems that could provide options for supporting decision makers in North Africa with science-based knowledge to deliver sustainable development to the North African lagoons.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

All authors helped to shape the research, analysis, and manuscript and provided the critical feedback. All authors discussed the results and contributed to the final version of the manuscript.

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REFERENCES

Abdel, S. A. (2015). Trace metals in seawater, sediments and some fish species from Marsa Matrouh beaches in north-western Mediterranean coast. *Egypt. Egypt. J. Aquat. Res.* 41, 145–154. doi: 10.1016/j.ejar.2015.02.006

Abdel Razek, F. A., Ismaiel, M., and Ameran, M. A. A. (2016). Occurrence of the blue crab *Callinectes sapidus*, Rathbun, 1896, and its fisheries biology in Bardawil lagoon, Sinai peninsula, Egypt. *Egypt. J. Aquat. Res.* 42, 223–229. doi: 10.1016/j.ejar.2016.04.005

Abidli, S., Lahbib, Y., González, P. R., Alonso, J. I. G., Trigui, and El Menif, N. (2016). Butyltin compounds in sediment and biota from the lagoon of Bizerte (northern Tunisia): potential risk for consumers? *Hum. Ecol. Risk Assess.* 22, 337–349. doi: 10.1080/10807039.2015.1064761

Alves Martins, M. V., Zaaboub, N., Aleya, L., Frontalini, F., Pereira, E., Miranda, P., et al. (2015). Environmental quality assessment of Bizerte Lagoon (Tunisia) using living foraminifera assemblages and a multiproxy approach. *PLoS One* 10:e0137250. doi: 10.1371/journal.pone.0137250

Ayache, F., Thompson, J. R., Flower, R. J., Boujarra, A., Rouatbi, F., and Makina, H. (2009). Environmental characteristics, landscape history and pressures on three coastal lagoons in the southern Mediterranean region: Merja Zerga (Morocco), Ghar El Melh (Tunisia) and Lake Manzala (Egypt). *Hydrobiologia* 622, 15–43. doi: 10.1007/s10750-008-9676-6

Banana, A. A. S., and Mohamed, R. M. S. R. (2016). Mercury pollution for marine environment at Farwa Island. *Libya. J. Environ. Heal. Sci. Eng.* 14, 1–8. doi: 10.1186/s40201-016-0246-y

Barhoumi, B., Menach, K. L., Clérandeau, C., Ameur, W. B., Budzinski, H., and Driss, M. R. (2014). Assessment of pollution in the Bizerte lagoon (Tunisia) by the combined use of chemical and biochemical markers in mussels, *Mytilus galloprovincialis*. *Mar. Pollut. Bull.* 84, 379–390. doi: 10.1016/j.marpolbul.2014.05.002

Barnard, S., and Elliott, M. (2015). The 10-tenets of adaptive management and sustainability: an holistic framework for understanding and managing the socio-ecological system. *Environ. Sci. Policy* 51, 181–191. doi: 10.1016/J.ENVSCI.2015.04.008

Béjaoui, B., Solidoro, C., Harzallah, A., Chevalier, C., Chapelle, A., Zaaboub, N., et al. (2017). 3D modeling of phytoplankton seasonal variation and nutrient budget in a southern Mediterranean Lagoon. *Mar. Pollut. Bull.* 114, 962–976. doi: 10.1016/J.MARPOLBUL.2016.11.001

Benradia, H., Berghiche, H., and Soltani, N. (2016). Measure of environmental stress biomarkers in the shrimp *Palaemon adspersus* from the Mellah lagoon (Algeria): spatial and temporal variations. *Fresenius Environ. Bull.* 25, 2563–2566.

Beryouni, K., Méar, Y., Murat, A., Poizot, E., and Chaibi, M. (2012). Geographical variability of environmental parameters versus GPS precision: toward a better sampling strategy. *Mar. Pollut. Bull.* 64, 2507–2518. doi: 10.1016/j.marpolbul.2012.05.015

Bocci, M., Brigolin, D., Pranovi, F., Najih, M., Nachite, D., and Pastres, R. (2016). An ecosystem approach for understanding status and changes of Nador lagoon (Morocco): application for of food web models and ecosystem indices. *Estuar. Coast. Shelf Sci.* 171, 133–143. doi: 10.1016/j.ecss.2016.01.004

Chaoui, L., Kara, M. H., Faure, É, and Quignard, J. (2006). L'ichtyofaune de la lagune du Mellah (Algérie Nord-Est): diversité, production et analyse des captures commerciales. *Cybium* 30, 123–132.

Chouari, W. (2015). Apport de la cartographie au suivi de l'anthropisation des milieux humides littoraux. *Méditerranée* 125, 75–84. doi: 10.4000/mediterranee.8015

Conde, D., Solari, S., de Álava, D., Rodríguez-Gallego, L., Verrastro, N., Chreties, C., et al. (2019). Ecological and social basis for the development of a sand barrier breaching model in Laguna de Rocha. *Uruguay. Estuar. Coast. Shelf Sci.* 219, 300–316. doi: 10.1016/J.ECSS.2019.02.003

Conseil Economique, Social et Environnemental (2014). *Projet de loi n81-12 Relative au Littoral*. Available online at: http://www.ces.ma/Documents/PDF/Saisines/S-13-2014-Projet-de-loi-sur-le-littoral/Avis-S-13-2014-VF.pdf

Daghor, L., Hssaïda, T., Chakir, S., Slimani, H., Mouflih, M., Hamoumi, N., et al. (2016). Etude des kystes de dinoflagellés des sédiments de surface du système lagunaire atlantique marocain Oualidia - Sidi Moussa et de la lagune méditerranéenne de Nador. *Bull. l'Institut Sci. Rabat, Sect. Sci. la Terre* 38, 1–18.

Damsiri, Z., Elkalay, K., and Khalil, K. (2015). Modelling the biogeochemical cycle of the Oualidia lagoon (Atlantic. Morocco). *Int. J. Adv. Res.* 3, 18–30.

Damsiri, Z., Natij, L., Elkalay, K., Khalil, K., Loudiki, M., Richir, J., et al. (2017). Seasonal characterization of the nutrients state in Oualidia lagoon (Moroccan atlantic coast). *J. Mater. Environ. Sci.* 8, 67–77.

Derradji, F., Benmeziane, F., Benaabidate, L., Maoui, A., Bousnoubra, H., and Kherici, N. (2015). Heavy metals analysis in the water of Mellah, Oubeira and Tonga Lakes of El kala wetland complex, North east Algeria. *Int. J. Innov. Sci. Res.* 13, 107–113.

DGPA (2001). "Etude del'amélioration de la qualité de l'eau de la lagune de BouGhrara," in *Phase II –partie I: Etude approfondie de la solution retenue*. Direction Générale de la Pêche et de l'Aquaculture, 1–18.

Dolbeth, M., Stålnacke, P., Alves, F. L., Sousa, L. P., Gooch, G. D., Khokhlov, V., et al. (2016). An integrated Pan-European perspective on coastal Lagoons management through a mosaic-DPSIR approach. *Sci. Rep.* 6, 1–12. doi: 10.1038/srep19400

EEA (1995). *Europe's Environment: The Dobris Assessment. Report No 1/1995.* Copenhagen: European Environmental Agency, 8.

Egyptian Environmental Affairs Agency (2009). *Environmental Protection Law in Egypt*. Available online at: http://www.eeaa.gov.eg/en-us/laws/envlaw.aspx

Eid, E. M., and Shaltout, K. H. (2014). Monthly variations of trace elements accumulation and distribution in above- and below-ground biomass of *Phragmites australis* (Cav.) Trin. ex Steudel in Lake Burullus (Egypt): a biomonitoring application. *Ecol. Eng.* 73, 17–25. doi: 10.1016/j.ecoleng.2014.09.006

El Asri, F., Zidane, H., Errhif, A., Tamsouri, M.-N., Maanan, M., Malouli Idrissi, M., et al. (2017a). Polychaete diversity and assemblage structure in the Oualidia Lagoon, Moroccan Atlantic coast. *J. Mar. Biol. Assoc. United Kingdom* 98, 1337–1346. doi: 10.1017/S0025315417000388

El Asri, F., Zidane, H., Maanan, M., Tamsouri, M., and Errhif, A. (2015). Taxonomic diversity and structure of the molluscan fauna in Oualidia lagoon (Moroccan Atlantic coast). *Environ. Monit. Assess.* 187, 545. doi: 10.1007/s10661-015-4752-7

El Asri, O., Ramdani, M., Latrach, L., Haloui, B., Mohamed, R., and Afilal, M. (2017b). Energetic valorization of Nador lagoon algae and proposal to use it as a means of elimination of the eutrophication in this lagoon. *Ecol. Eng.* 103, 236–243. doi: 10.1016/j.ecoleng.2017.04.016

El Asri, O., Ramdani, M., Latrach, L., Haloui, B., Ramdani, M., and Afilal, M. E. (2017c). Comparison of energy recovery after anaerobic digestion of three Marchica lagoon algae (*Caulerpa prolifera*, *Colpomenia sinuosa*, *Gracilaria bursa-pastoris*). *Sustain. Mater. Technol.* 11, 47–52. doi: 10.1016/j.susmat.2016.12.002

El Nahas, A. F., Abdel-Razek, M. A. S., Helmy, N. M., Mahmoud, S., and Ghazy, H. A. (2017). Impaired antioxidant gene expression by pesticide residues and its relation with other cellular biomarkers in Nile Tilapia (*Oreochromis niloticus*) from Lake Burullus. *Ecotoxicol. Environ. Saf.* 137, 202–209. doi: 10.1016/j.ecoenv.2016.12.006

El Yaouti, F., El Mandour, A., Khattach, D., and Kaufmann, O. (2008). Modelling groundwater flow and advective contaminant transport in the Bou-Areg unconfined aquifer (NE Morocco). *J. Hydro- Environ.ment Res.* 2, 192–209. doi: 10.1016/j.jher.2008.08.003

El-Adawy, A., Negm, A. M., Elzeir, M. A., Saavedra, O. C., El-Shinnawy, I. A., and Nadaoka, K. (2013). Modeling the hydrodynamics and salinity of El-Burullus Lake (Nile Delta, Northern Egypt). *J. Clean Energy Technol.* 1, 157–163. doi: 10.7763/JOCET.2013.V1.37

El-Amier, Y. A., Elnaggar, A. A., and El-Alfy, M. A. (2017). Evaluation and mapping spatial distribution of bottom sediment heavy metal contamination in Burullus Lake. *Egypt. Egypt. J. Basic Appl. Sci.* 4, 55–66. doi: 10.1016/j.ejbas.2016.09.005

El-Asmar, H. M., Hereher, M. E., and El Kafrawy, S. B. (2013). Surface area change detection of the Burullus Lagoon, North of the Nile Delta, Egypt, using water indices: a remote sensing approach. Egypt. . *J. Remote Sens. Sp. Sci.* 16, 119–123. doi: 10.1016/j.ejrs.2013.04.004

Elgharsalli, R., Hili, H. A. E. L., Boubaker, H., Boulaares, K., and Ayari, W. (2016). First record of marteiliosis in *Mytilus galloprovincialis* in Bizerte lagoon. *Bull. Inst. Natn. Scien. Tech. Mer Salammbô* 43, 125–132.

El-Kassas, H. Y., Nassar, M. Z. A., and Gharib, S. M. (2016). Study of phytoplankton in a natural hypersaline lagoon in a desert area (Bardawil Lagoon in Northern Sinai. Egypt). *Rend. Lincei* 27, 483–493. doi: 10.1007/s12210-016-0506-x

Elliott, M. (2013). The 10-tenets for integrated, successful and sustainable marine management. *Mar. Pollut. Bull.* 74, 1–5. doi: 10.1016/j.marpolbul.2013.08.001

Elliott, M., Burdon, D., Atkins, J. P., Borja, A., Cormier, R., de Jonge, V. N., et al. (2017). "And DPSIR begat DAPSI(W)R(M)!" - A unifying framework for marine environmental management. *Mar. Pollut. Bull.* 118, 27–40. doi: 10.1016/j.marpolbul.2017.03.049

El-Monsef, A., El-Badry, A., and Khalifa, M. M. (2017). The occurrence and distribution of high-arsenic, selenium, tin and antimony in bottom sediments of Burullus lagoon and its effects on human health, Egypt. *J. African Earth Sci.* 136, 305–311. doi: 10.1016/j.jafrearsci.2017.08.001

El-Zeiny, A., and El-Kafrawy, S. (2017). Assessment of water pollution induced by human activities in Burullus Lake using Landsat 8 operational land imager and GIS. *Egypt. J. Remote Sens. Sp. Sci.* 20, S49–S56. doi: 10.1016/j.ejrs.2016.10.002

Embarek, R., Amara, R., and Kara, M. H. (2017). Fish assemblage structure in shallow waters of the Mellah lagoon (Algeria): seasonal and spatial distribution patterns and relation to environmental parameters. *Acta Ichthyol. Piscat.* 47, 133–144. doi: 10.3750/AIEP/02080

Ennesser, Y., Said, V., and Errier, M. (2011). Évaluation des risques en situation actuelle et à l'horizon 2030 pour la ville de Tunis, Rapport final de phase 1 du projet «Adaptation au changement climatique et aux désastres naturels des villes côtières de l'Afrique du Nord», Egis BCEOM/IAU-IDT/BRGM. *Banque Mondiale* 1, 1–14.

EPA (1994). "A conceptual framework to support the development and use of environmental information," in *Environmental Statistics and Information Division. Office of Policy, Planning and Evaluation. EPA* 230-R-94-012 (Washington DC: USEPA).

Essghaier, M. F. A., Etayeb, K. S., Bourass, E., and Joel, P. J. (2013). Status and distribution of coastal birds at Farwa Island. *Trav. . l'Institut Sci. Rabat,. Série Zool* 78–85.

EU (2000). Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for Community action in the field of water policy. *Off.icial J.ournal of the Eur.opean Commun.ities* 327, 1–73.

Fadili, A., Najib, S., Mehdi, K., Riss, J., Makan, A., Boutayeb, K., et al. (2016). Hydrochemical features and mineralization processes in coastal groundwater of Oualidia. *Morocco. J. African Earth Sci.* 116, 233–247. doi: 10.1016/j.jafrearsci.2016.01.014

Food and Agriculture Organization (1982). *Organisation de la pêche dans le Lac Mellah L'Unité Aquacole de El-Kala*. Available online at: http://www.fao.org/tempref/FI/CDrom/aquaculture/a0845t/volume1/docrep/field/007/af018f/AF018F09.htm.

Ferreira, J. G., Nobre, A. M., Simas, T. C., Silva, M. C., Newton, A., Bricker, S. B., et al. (2006). A methodology for defining homogeneous water bodies in estuaries - Application to the transitional systems of the EU Water Framework Directive. *Estuar. Coast. Shelf Sci.* 66, 468–482. doi: 10.1016/j.ecss.2005.09.016

Fertouna-Bellakhal, M., Dhib, A., Béjaoui, B., Turki, S., and Aleya, L. (2014). Driving factors behind the distribution of dinocyst composition and abundance in surface sediments in a western Mediterranean coastal lagoon: report from a high resolution mapping study. *Mar. Pollut. Bull.* 84, 347–362. doi: 10.1016/J.MARPOLBUL.2014.04.041

Finkl, C. W. (2017). *Coastal Wetlands: Alteration and Remediation, Coastal Research Library*. Cham. Springer International Publishing.

Flower, R. J., Appleby, P. G., Thompson, J. R., Ahmed, M. H., Ramdani, M., Chouba, L., et al. (2009). Sediment distribution and accumulation in lagoons of the Southern Mediterranean Region (the MELMARINA Project) with special reference to environmental change and aquatic ecosystems. *Hydrobiologia* 622, 85–112. doi: 10.1007/s10750-008-9677-5

Garali, A., Ben, Ouakad, M., and Gueddari, M. (2009). Bilans hydrologiques de la lagune de Bizerte (nord-est de la Tunisie). *Rev. des Sci. l'eau.* 22, :525. doi: 10.7202/038329ar

García-Ayllón, S. (2017). Integrated management in coastal lagoons of highly complexity environments: resilience comparative analysis for three case-studies. *Ocean Coast. Manag.* 143, 16–25. doi: 10.1016/j.ocecoaman.2016.10.007

Gari, S. R., Newton, A., and Icely, J. D. (2015). A review of the application and evolution of the DPSIR framework with an emphasis on coastal social-ecological systems. *Ocean Coast. Manag.* 103, 63–77. doi: 10.1016/J.OCECOAMAN.2014.11.013

Gharib, S. M., El-Sherif, Z. M., Abdel-Halim, A. M., and Radwan, A. A. (2011). Phytoplankton and environmental variables as a water quality indicator for the beaches at Matrouh, south-eastern Mediterranean Sea, Egypt: an assessment. *Oceanologia* 53, 819–836. doi: 10.5697/oc.53-3.819

Ghribi, F., Richir, J., Boussafa, D., El Cafsi, M., and Gobert, S. (2016). Trace metals in soft tissue of marine bivalve Noah's ark (Arca noao) from Bizerte lagoon (Northern Tunisia). *Rapp. Comm. Int. Mer Médit.* 41:241.

Giuliani, S., Piazza, R., El Moumni, B., Polo, F. P., Vecchiato, M., Romano, S., et al. (2015a). Recognizing different impacts of human and natural sources on the spatial distribution and temporal trends of PAHs and PCBs (including PCB-11) in sediments of the Nador lagoon (Morocco). *Sci. Total Environ.* 526, 346–357. doi: 10.1016/j.scitotenv.2015.04.057

Giuliani, S., Piazza, R., El Moumni, B., Polo, F. P., Vecchiato, M., Romano, S., et al. (2015b). Recognizing different impacts of human and natural sources on the spatial distribution and temporal trends of PAHs and PCBs (including PCB-11) in sediments of the Nador lagoon (Morocco). *Sci. Total Environ.* 526, 346–357. doi: 10.1016/j.scitotenv.2015.04.057

Gönenç, I. E., and Wolflin, J. P. (2005). *Coastal Lagoons Ecosystem Processes and Modeling for Sustainable Use and Development*. Boca Raton, FL: CRC Press.

Gooré, B. E., Monette, F., and Gasperi, J. (2015). Analysis of the influence of rainfall variables on urban effluents concentrations and fluxes in wet weather. *J. Hydrol.* 523, 320–332. doi: 10.1016/J.JHYDROL.2015.01.017

Guerra-García, J. M., and García-Gómez, J. C. (2005). Assessing pollution levels in sediments of a harbour with two opposing entrances. Environmental implications. *J. Environ. Manage.* 77, 1–11. doi: 10.1016/J.JENVMAN.2005.01.023

Ha, E., Basu, N., Bose-O'Reilly, S., Dórea, J. G., McSorley, E., Sakamoto, M., et al. (2017). Current progress on understanding the impact of mercury on human health. *Environ. Res.* 152, 419–433. doi: 10.1016/J.ENVRES.2016.06.042

Haddoud, D. A., and Rawag, A. A. (1995). *Marine Protected Areas along Libyan coast. D. A. Haddoud and A. A. Rawag*. Tajura: Marine Biology Research Centre, 23–31.

Hammani, J., Brahim, M., and Gueddari, M. (2016). Assessment of the water quality of Bizerte lagoon of Tunisia by use of statistical analyses. *Hydrol. Curr. Res.* 7, 1–8. doi: 10.4172/2157-7587.1000237

Huffpostmaghreb (2017). *Huffpostmaghreb*. Available online at: https://marocainun.wordpress.com/tag/maroc-the-huffington-post/page/327/

Husain, M., Hassan, H. A. E., Khater, M., and Ghoniem, A. (2016). Analysis of Reinforced Concrete D-Regions Using Strut-and-Tie Model. *Egypt. Int. J. Eng. Sci. Technol.* 20, 25–37.

Journal Officiel de la Republique Algerienne (2002). *Loi n 02-02 du 22 Dhou El Kaada 1422 correspondant au 5 février 2002 relative à la protection et à la valorisation du littoral*. Available online at: http://www.enssmal.dz/fr/images/Relementation_milieux_marin/Loi_littoral_02-02.pdf.

Kamel, N., Burgeot, T., Banni, M., Chalghaf, M., Devin, S., Minier, C., et al. (2014). Effects of increasing temperatures on biomarker responses and accumulation of hazardous substances in rope mussels (*Mytilus galloprovincialis*) from Bizerte lagoon. *Environ. Sci. Pollut. Res.* 21, 6108–6123. doi: 10.1007/s11356-014-2540-5

Khaled, A., Hessein, A., Abdel-Halim, A. M., and Morsy, F. M. (2014). Distribution of heavy metals in seaweeds collected along Marsa Matrouh beaches, Egyptian Mediterranean sea. *Egypt. J. Aquat. Res.* 40, 363–371. doi: 10.1016/j.ejar.2014.11.007

Khedhri, I., Afli, A., and Aleya, L. (2017a). Structuring factors of the spatio-temporal variability of macrozoobenthos assemblages in a southern Mediterranean lagoon: how useful for bioindication is a multi-biotic indices approach? *Mar. Pollut. Bull.* 114, 515–527. doi: 10.1016/j.marpolbul.2016.10.023

Khedhri, I., Afli, A., and Aleya, L. (2017b). Structuring factors of the spatio-temporal variability of macrozoobenthos assemblages in a southern Mediterranean lagoon: how useful for bioindication is a multi-biotic indices approach? *Mar. Pollut. Bull.* 114, 515–527. doi: 10.1016/J.MARPOLBUL.2016.10.023

Khedhri, I., Tovar-Hernández, M., Bonifácio, P., Ahmed, A., and Aleya, L. (2017c). First report of the invasive species *Branchiomma bairdi* McIntosh, 1885 (*Annelida*: *Sabellidae*) along the Tunisian coast (Mediterranean Sea). *BioInvasions Rec.* 6, 139–145. doi: 10.3391/bir.2017.6.2.09

Kherifi, W., and Bousnoubra-Kherici, H. (2016). Study of the influence of the physicochemical parameters on microbial abundance in various ambient conditions. *Water Resour.* 43, 546–558. doi: 10.1134/S0097807816030180

Kim, K.-H., Kabir, E., and Jahan, S. A. (2016). A review on the distribution of Hg in the environment and its human health impacts. *J. Hazard. Mater.* 306, 376–385. doi: 10.1016/J.JHAZMAT.2015.11.031

Lahbib, Y., Abidli, S., and Trigui-El Menif, N. (2018). First assessment of the effectiveness of the international convention on the control of harmful anti-fouling systems on ships in Tunisia using imposex in *Hexaplex trunculus* as biomarker. *Mar. Pollut. Bull.* 128, 17–23. doi: 10.1016/J.MARPOLBUL.2018.01.012

Lillebø, A. I. (2015). *Coastal Lagoons in Europe: Integrated Water Resource Strategies*. London: IWA Publishing, 14.

Lillebø, A. I., Stålnacke, P., Gooch, G. D., Krysanova, V., and Bielecka, M. (2017). Pan-European management of coastal lagoons: a science-policy-stakeholder interface perspective. *Estuar. Coast. Shelf Sci.* 198, 648–656. doi: 10.1016/J.ECSS.2016.03.008

Maanan, M., Ruiz-Fernández, A. C., Maanan, M., Fattal, P., Zourarah, B., and Sahabi, M. (2014). A long-term record of land use change impacts on sediments in Oualidia lagoon. *Morocco. Int. J. Sediment Res.* 29, 1–10. doi: 10.1016/S1001-6279(14)60017-2

Magni, P., Draredja, B., Melouah, K., and Como, S. (2015). Patterns of seasonal variation in lagoonal macrozoobenthic assemblages (Mellah lagoon. Algeria). *Mar. Environ. Res.* 109, 168–176. doi: 10.1016/j.marenvres.2015.07.005

Mahmoud Khamis, E., and El-Sayed El-Sayed, M. (2015). Feeding habits of the copper shark, *Carcharhinus brachyurus* (Günther, 1870) from Ain El-Ghazala Lagoon, Eastern Libya during the Period from February till June 2013. *J. Life Sci.* 9, 347–355. doi: 10.17265/1934-7391/2015.08.001

Maslow, A. H. (1943). A theory of human motivation. *Psychol. Rev.* 50, 370–396. doi: 10.1037/h0054346

Mebarki, R., El, M., Khebbeb, H., and Soltani, N. (2015). Biomonitoring of El mellah lagoon (Northeast, Algeria): seasonal variation of biomarkers in *Cerastoderma glaucum* (Mollusc, Bivalvia). *J. Entomol. Zool. Stud. JEZS* 3, 408–413.

Melouah, K. (2013). *Étude de la Faune Malacologique de la Lagune Mellah avec un intérêt Particulier pour le bivalve: Cerastoderma Glaucum*. PhD Thesis. Université Badji Mokhtar Algérie.

Mulazzani, L., Zanasi, C., Errico, A., Pugliese, P., Zuccaro, M., Zerrouki, R., et al. (2017). The comparative analysis of Mediterranean coastal communities: six case studies. New medit Mediterr. . *J. Econ. Agric. Environ. Rev. Méd. Econ. Agric. Environ.* 16, 27–37.

Nadira, S. (2008). *Evolution des Peuplements Phytoplanctoniques au Niveau du lac Oubéira Et la lagune El Mellah*. PhD Thesis, Annaba: University of Annaba, Annaba. PhD Thesis.

Nadji, S., Amrani, A., Mebarki, R., and Khebbeb, M. E. (2010). Acetylcholinesterase and catalase activities in several tissues of a bivalve mollusc (*Ruditapes decussatus*) fished from Mellah lagoon (North East of Algeria) after malathion exposure. *Ann. Biol. Res.* 1, 138–144.

Najih, M., Berday, N., Lamrini, A., Nachite, D., and Zahri, Y. (2015). Situation de la pêche aux petits métiers après l'ouverture du nouveau chenal dans la lagune de Nador. *Rev. Marocaine des Sci. Agron. Vét.érinaires* 3, 19–30.

Nassar, K., Mahmod, W. E., Masria, A., Fath, H., and Nadaoka, K. (2018). Numerical simulation of shoreline responses in the vicinity of the western artificial inlet of the Bardawil Lagoon. *Sinai Peninsula, Egypt. Appl. Ocean Res.* 74, 87–101. doi: 10.1016/j.apor.2018.02.015

Nassar, M. Z. A., and Gharib, S. M. (2014). Spatial and temporal patterns of phytoplankton composition in Burullus lagoon, southern Mediterranean coast, Egypt. *Egypt. J. Aquat. Res.* 40, 133–142. doi: 10.1016/j.ejar.2014.06.004

Newton, A., Brito, A. C., Icely, J. D., Derolez, V., Clara, I., Angus, S., et al. (2018). Assessing, quantifying and valuing the ecosystem services of coastal lagoons. *J. Nat. Conserv* 44, 50–65. doi: 10.1016/J.JNC.2018.02.009

Newton, A., and Elliott, M. (2016). A typology of stakeholders and guidelines for engagement in transdisciplinary, participatory processes. *Front. Mar. Sci.* 3:230. doi: 10.3389/fmars.2016.00230

Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A. C., Colijn, F., et al. (2014). An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. *Estuar. Coast. Shelf Sci.* 140, 95–122. doi: 10.1016/j.ecss.2013.05.023

Oczkowski, A., Nixon, S., Granger, S., El-Sayed, A. F. M., Altabet, M., and McKinney, R. (2008). A preliminary survey of the nitrogen and carbon isotope characteristics of fish from the lagoons of Egypt's Nile delta. *Estuaries and Coasts* 31, 1130–1142. doi: 10.1007/s12237-008-9102-3

OECD (1994). *Environmental Indicators- OECD Core Set. Organisation for Economic Co-operation and Development*. Paris: OECD, 37.

Orabi, O. H., El-Badry, A. A., and Badr-ElDin, A. M. (2017). Benthic foraminifera for heavy metal pollution monitoring: a case study from Burullus Lagoon of Egypt. *Mar. Pollut. Bull.* 121, 411–417. doi: 10.1016/j.marpolbul.2017.06.015

Patrício, J., Elliott, M., Mazik, K., Papadopoulou, K.-N., and Smith, C. J. (2016). DPSIR—Two decades of trying to develop a unifying framework for marine environmental management? *Front. Mar. Sci.* 3:177. doi: 10.3389/fmars.2016.00177

Piazza, R., Bellucci, L. G., Giuliani, S., Romano, S., Frignani, M., Pizzini, S., et al. (2016). Can PBDE natural formation and degradation processes interfere with the identification of anthropogenic trends and sources? Evidences from sediments of the Nador Lagoon (Morocco). *Mar. Pollut. Bull.* 108, 15–23. doi: 10.1016/j.marpolbul.2016.05.007

Raji, O., Dezileau, L., Tessier, B., Niazi, S., Snoussi, M., Von Grafenstein, U., et al. (2018). Climate and tectonic-driven sedimentary infill of a lagoon as revealed by high resolution seismic and core data (the Nador lagoon. NE Morocco). *Mar. Geol.* 398, 99–111. doi: 10.1016/J.MARGEO.2018.01.010

Raji, O., Niazi, S., Snoussi, M., Dezileau, L., and Khouakhi, A. (2013). Vulnerability assessment of a lagoon to sea level rise and storm events: Nador lagoon (NE Morocco). *J. Coast. Res.* 65, 802–807. doi: 10.2112/SI65-136.1

Ramdani, M., Elkhiati, N., and Flower, R. J. (2009a). *Lakes of Africa: North of Sahara, in: Encyclopedia of Inland Waters*. Amsterdam: Elsevier, 544–554. doi: 10.1016/B978-012370626-3.00035-1

Ramdani, M., Elkhiati, N., Flower, R. J., Thompson, J. R., Chouba, L., Kraiem, M. M., et al. (2009b). Environmental influences on the qualitative and quantitative composition of phytoplankton and zooplankton in North African coastal lagoons. *Hydrobiologia* 622, 113–131. doi: 10.1007/s10750-008-9678-4

Ramdani, M., Flower, R. J., Elkhiati, N., Kraïem, M. M., Fathi, A. A., Birks, H. H., et al. (2001). North African wetland lakes: characterization of nine sites included in the Cassarina Project. *Aquat. Ecol* 35, 281–302. doi: 10.1023/A:1011957324901

Rapport, D., and Friend, A. (1979). "Towards a comprehensive framework for environmental statistics: a stress-response approach," in *Statistics Canada Catalogue* (Ottawa: Minister of Supply and Services Canada), 11–510.

Re, V., and Sacchi, E. (2017). Tackling the salinity-pollution nexus in coastal aquifers from arid regions using nitrate and boron isotopes. *Environ. Sci. Pollut. Res.* 24, 13247–13261. doi: 10.1007/s11356-017-8384-z

Rocheta, V. L. S., Isidoro, J. M. G. P., and de Lima, J. L. M. P. (2017). Infiltration of Portuguese cobblestone pavements–An exploratory assessment using a double-ring infiltrometer. *Urban Water J.* 14, 291–297. doi: 10.1080/1573062X.2015.1111914

Ruiz, F., Abad, M., Olías, M., Galán, E., González, I., Aguilá, E., et al. (2006). The present environmental scenario of the Nador Lagoon (Morocco). *Environ. Res.* 102, 215–229. doi: 10.1016/j.envres.2006.03.001

Sayoud, M. S., Salhi, H., Chalabi, B., Allali, A., Dakki, M., Qninba, A., et al. (2017). The first coordinated trans-North African mid-winter waterbird census: the contribution of the International Waterbird Census to the conservation of waterbirds and wetlands at a biogeographical level. *Biol. Conserv.* 206, 11–20. doi: 10.1016/j.biocon.2016.12.005

Secrétariat Général du Gouvernement Marocain (2017). Projet de loi Relative à la Préservation des Écosystèmes Halieutiques et à Laprotection du Milieu Marin contre la Pollution. Avaliable at: http://www.sgg.gov.ma/portals/0/AvantProjet/63/Avp_loi_42-13_Fr.pdf

Smith, C. J., Papadopoulou, K.-N., Barnard, S., Mazik, K., Elliott, M., Patrício, J., et al. (2016). Managing the marine environment, conceptual models and assessment considerations for the European marine strategy framework directive. *Front. Mar. Sci.* 3:144. doi: 10.3389/fmars.2016.00144

Telailia, S., Boutabia, L., Khemis, M. D. E.-H., Elafri, A., and Djebbari, N. (2017). Multi-annual and seasonal patterns of waterbird assemblages in a Mediterranean coastal lagoon (El Mellah lagoon) of Northeastern Algeria. *Ekológia (Bratislava)* 36, 146–157. doi: 10.1515/eko-2017-0013

Temraz, T. A., Zedan, H., Fouda, M., Saber, M., Salama, W., and Harhash, K. A. (2016). Egyptian biodiversity strategy and action plan (2015-2030). *Minist. Envirnoment* 1–83.

Thompson, J. R., and Flower, R. J. (2009). Environmental science and management of coastal lagoons in the southern Mediterranean region: key issues revealed by the MELMARINA project. . *Hydrobiologia*

Thompson, J. R., Flower, R. J., Ramdani, M., Ayache, F., Ahmed, M. H., Rasmussen, E. K., et al. (2009). Hydrological characteristics of three North African coastal lagoons: Insights from the MELMARINA project. *Hydrobiologia* 622, 45–84. doi: 10.1007/s10750-008-9680-x

Thornton, S. E., Pilkey, O. H., Doyle, L. J., and Whawling, P. J. (1980). Holocene evolution of a coastal lagoon, Lake of Tunis, Tunisia. *Sedimentology* 27, 79–91. doi: 10.1111/j.1365-3091.1980.tb01159.x

Tlig-Zouari, S., and Maamouri-Mokhtar, F. (2008). Macrozoobenthic species composition and distribution in the Northern lagoon of Tunis. *Transitional Waters Bull.* 2, 1–15. doi: 10.1285/i18252273v2n2p1

Trabelsi, E. L. B., Armi, Z., Trabelsi-Annabi, N., Shili, A., Maiz, N., and Ben. (2013). Water quality variables as indicators in the restoration impact assessment of the north lagoon of Tunis, South Mediterranean. *J. Sea Res.* 79, 12–19. doi: 10.1016/j.seares.2013.01.003

Trigui, N., Menif, E., Kara, H., and Mahé, G. (2012). *Final Project Report for 2011 START Grants for Global Change Research in Africa Sensitivity of Coastal Lagoon Ecosystems to Climate and Related Global Changes: Developing a North African Lagoons Network*.

Turki, S., Dhib, A., Fertouna-Bellakhal, M., Frossard, V., Balti, N., Kharrat, R., et al. (2014). Harmful algal blooms (HABs) associated with phycotoxins in shellfish: what can be learned from five years of monitoring in Bizerte Lagoon (Southern Mediterranean Sea)? *Ecol. Eng.* 67, 39–47. doi: 10.1016/J.ECOLENG.2014.03.028

UN (2015). *Transforming our world: the 2030 agenda for sustainable development. In: Resolution Adopted by the General Assembly on 25 September 2015. Seventieth Session, Agenda Items 15 and 116. A/RES/70/1*. Available online at: https://daccess-ods.un.org/access.nsf/GetFile?OpenAgent&DS=A/RES/70/1&Lang=E&Type=DOC.

UNEP (1994). *World Environment Outlook: Brainstorming Session. ENEP/EAMR. 94-5. Environment Assessment Programe Nairobi*. United Nations Environment Programme.

Vandenbroeck, J., and Rafik, B. C. (2001). Restoration and development project of south lake of Tunis and its shores. *Terra Aqua* 11–20.

Wolanski, E., and Elliott, M. (2015). Estuarine ecohydrology: an introduction. doi: 10.1016/B978-008045405-4.00044-6

Yamna, T., Algouti, Ahmed, Algouti, Abdellah, Kheir Addin, A., et al. (2014). Apport de la télédétection à l'étude de la dynamique sédimentaire au sein de la lagune Oualidia (Maroc). *Eur. Sci. J.* 10, 175–185.

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A Multidisciplinary Approach for Restoration Ecology of Shallow Coastal Lagoons, a Case Study in South France

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By the end of the 20th century, many of the coastal lagoons along the French Mediterranean coast showed insufficient water quality and degraded ecosystem states due to anthropogenic impacts. Among these, nutrient over-enrichment, resulting in eutrophication, has been a major concern. The EU Water Framework Directive (WFD) has initiated public action to improve their water quality and ecosystem state using an approach rooted in restoration ecology. Here we analyze how this has been applied for the coastal lagoons in South France, considering eutrophication as an example of ecosystem degradation and oligotrophication as the corresponding trajectory for ecological restoration of the eutrophied coastal lagoons. Oligotrophication trajectories, initiated by the reduction of external nutrient loading, have resulted in a quick recovery (i.e., within 3 years) of integrative water column variables (Chlorophyll *a*, total N and P) and phytoplankton. Starting from hypertrophic systems, the oligotrophication trajectory is described by a sequence of three ecosystem states dominated respectively by (i) phytoplankton with bare non-vegetated sediments, (ii) opportunistic macroalgae, (iii) angiosperm and perennial macroalgae, punctuated by regime shifts between these ecosystem states. Nevertheless, the latter regime shift has not been observed for the most degraded ecosystems after 10-years oligotrophication. The N and P accumulated in sediments during eutrophication may also retard the ecological restoration. In shallow freshwater lakes, the phytoplankton-dominated and the angiosperm-dominated states are also characteristic for highly degraded and fully restored ecosystems states, respectively. In contrast, opportunistic macroalgae do not bloom in these systems. Hence, the alternative stable state model, used successfully for these lakes, cannot be applied straightforwardly for coastal lagoons. To be successful, ecological restoration should consider the legislative and societal questions as according the DPSIR framework it typically is a response of society. The conservation-oriented Habitats Directive systematically applies to coastal lagoons and the Birds Directive in some cases as well. The WFD approach is complementary to these more conventional nature conservation approaches. Collectively, local citizens and highly involved stakeholders strongly value the coastal lagoons and attribute very high importance to their regulating ecosystem services (ESs), while differences between stakeholder profiles are related to different perceptions and appreciations of cultural ESs. Hence, coastal lagoon ESs are very important for the different stakeholders and public policies aiming at the ecological restoration of these ecosystems may count on public support.

Keywords: restoration ecology, ecological restoration, water quality, WFD, ecosystem trajectories, DPSIR, conservation, ecological indices

INTRODUCTION

The EU Water Framework Directive 2000/60/EC (WFD) presents a highly integrated approach focused on water quality, which according Voulvoulis et al. (2017) is widely accepted as the most substantial and ambitious piece of European environmental legislation to date. Interestingly, the major aim of this directive is to improve the ecological status of water bodies, which calls for a pro-active approach rooted in the theory of restoration ecology. In general, a difference is made in the literature between restoration ecology and ecological restoration; the latter is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (Society for Ecological Restoration International Science and Policy Working Group, 2004), The former, restoration ecology, is a scientific discipline, a field within ecology, focused on studying the concepts, experiences and practices of ecological restoration (Clewell and Aronson, 2013). Both restoration ecology and the WFD advocate that the approaches for ecological restoration should be based on (i) the identification of a reference state, (ii) a description of a desired state that is quite close to the reference state, and (iii) the development of a roadmap, which specifies how the ecosystem trajectory should converge toward that desired state. For restoration ecology, the reference state is a historical reference state referring to the ecosystem state before it was degraded by human impacts (Clewell and Aronson, 2013). The reference state according the WFD is the ecosystem state in conditions of minimal anthropogenic impact. Hence, for a degraded aquatic system considered for ecological restoration, this implies that such conditions of minimal anthropogenic impact occurred in the past and the reference state can thus be considered as a historical reference state *sensu* restoration ecology.

The Ecosystem state of many coastal lagoons worldwide has been degraded as a result of anthropogenic impacts comprising nutrient over-enrichment leading to eutrophication (Zaldívar et al., 2008), habitat destruction (De Wit, 2011; Newton et al., 2018), contaminant inputs (Covelli, 2012; Munaron et al., 2012). Environmental awareness of citizens, the practice of Integrated Coastal Zone Management (ICZM), and specific legislation derived from the EU, i.e., the WFD mentioned above together with the EU Habitats Directive 1992 (Council Directive 92/43/EEC) and the EU Birds Directive (2009) (Directive 2009/147/EC), have triggered public policies in the EU member states that value these ecosystems and aim to protect and improve their water quality, protect public health issues, conserve their biodiversity and develop their ecological potential for the delivery of ecosystem services. The WFD considers the ecological functioning of the aquatic ecosystems, including coastal lagoons, and formulates its main objectives as conserving and achieving "good ecological status" of water bodies. Therefore, the WFD is particularly important for biodiversity conservation in the EU and it has been stated that:

> "*the legislative framework in place to achieve the Biodiversity Strategy in aquatic ecosystems* (in the EU, *sic*) *can be linked to a complex array of interlinked policies, of which the most far-reaching ones are the Birds and Habitats Directives, Water Framework Directive, and Marine Strategic Framework Directive*"
>
> (Rouillard et al., 2017).

The latter has fewer implications for coastal lagoons, for which the water quality regulations are derived from the WFD. Coastal lagoons are listed as a priority habitat in the Habitats Directive and many coastal lagoons are particularly important for waterfowl and concerned by the Birds Directive (Dolbeth et al., 2016). Article 6 of the WFD requests member states to establish a register of areas which have been designated as requiring special protection under specific Community legislation for the conservation of habitats and species directly depending on water and WFD Annex IV explicitly links this requirement to the Natura 2000 sites designated according the Habitats and Birds Directives.

In this article, we use eutrophication as an example of ecosystem degradation, and oligotrophication as the corresponding trajectory for the ecological restoration of eutrophied lagoons. The many actions employed in South France to combat eutrophication of coastal lagoons and induce their oligotrophication thus represents an interesting study case for assessing ecological restoration according the theoretical framework of restoration ecology. We believe that such an analysis should particularly address the issues to which coastal lagoon managers are confronted in their daily practice. Therefore, first we propose a conceptualization placing the ecological restoration in societal context and adapt the conceptual scheme in such a way to accommodate the specificities of the WFD. Secondly, our study includes an analysis of the technical and natural science aspects of the ecological restoration practice as well as important questions concerning the societal context. Therefore, we have formulated the following three main questions (**Figure 1**):

1. Do scientific concepts and theories provide guidance for the ecological restoration practice and are their theoretical predictions of ecosystem trajectories supported by empiric ecological knowledge?

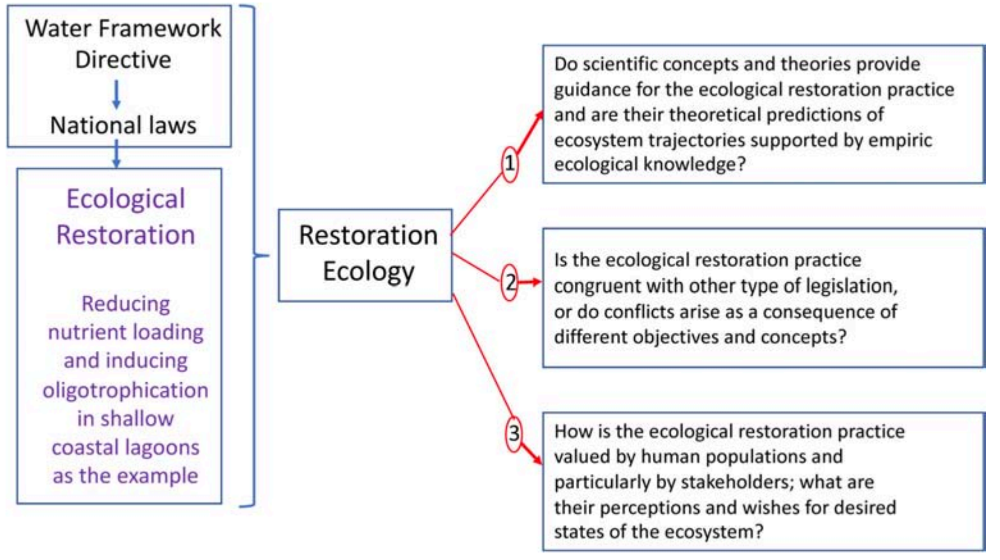

FIGURE 1 | The Water Framework Directive (2000/60/EC) through its transposition into national laws [in France: Loi n° 2006–1772 du 30 décembre 2006 sur l'eau et les milieux aquatiques (French law)] has resulted in South France in ecological restoration practice to improve the ecological status of coastal lagoons. A particular focus has been on reducing nutrient loading to combat eutrophication and thus induce oligotrophication of coastal lagoons. In this article this practice has been analyzed according the theoretical framework of restoration ecology and three main questions have been formulated for this assessment.

2. Is the ecological restoration practice congruent with other type of legislation, or do conflicts arise as a consequence of different objectives and concepts?
3. How is the ecological restoration practice valued by human populations and particularly by stakeholders; what are their perceptions and wishes for desired states of the ecosystem?

Ecological restoration of coastal lagoons has started around 2000 and is, therefore, quite novel in contrast to the ecological restoration of water quality in freshwater lakes initiated since the 1980s (Marsden, 1989; Gulati and Van Donk, 2002; Jeppesen et al., 2002). For providing scientific guidance for the technical aspects, we consider if the lessons learned from the ecological restoration of freshwater lakes can be applied to the brackish to saline coastal lagoons located at the interface between the land and the sea. A general model of alternative stable states was developed originally for shallow freshwater lakes (Scheffer et al., 1993), with a transparent water state dominated by submerged aquatic vegetation (SAV) and a turbid state dominated by phytoplankton. Hence, here we evaluate whether this model can also be applied to shallow coastal lagoons. In addition, we consider the sediment compartment as a possible internal source for nitrogen and phosphorus that could retard the oligotrophication process as has been observed in shallow freshwater lakes (Marsden, 1989; Søndergaard et al., 2003; Jeppesen et al., 2005). We report the different approaches for coastal lagoons in the WFD and the Habitats Directive and discuss the challenges faced by the managers of these ecosystems to cope simultaneously with the requirements of both directives. Concerning the social aspects, we study whether the normative approach for water quality imposed by the WFD is accepted by the local populations and stakeholders by studying their perceptions of water quality and their preferences for ecosystem services.

STUDY SITES AND METHODS

The study sites comprise the shallow coastal lagoons (average depth <2 m, surface >50 ha) along the Mediterranean coastlines of continental France and the island of Corsica, which are shown in **Figure 2**. Deeper coastal lagoons (average depth >2 m, in gray in **Figure 2**) have been excluded from this analysis as deeper water bodies show different ecological structure and functioning. In addition, deep lagoons are often complex systems comprising both shallow and deeper parts that may interact in a complex way. The terminology based on their salinities follows the Venice System (1958) and was described based on monitored salinity values in Le Fur et al. (2018). The shallow lagoons in **Figure 2** cover 396 km^2. Coastal lagoons occupy about 50% of the coastline along the Gulf of Lion, are numerous in the delta of the Rhône River and less common on the Côte d'Azur. In Corsica, coastal lagoons are located on the Eastern littoral facing the Tyrrhenian Sea.

Coastal lagoons are characterized by permanent or temporary connections with the adjacent sea, via one or several inlets (Kjerfve, 1994). Today, very few of the shallow lagoons shown in **Figure 2** have natural moving inlets (i.e., La Palme, Ayrolle, Biguglia), which only for La Palme lagoon close off completely every year (Larue and Rouquet, 2016). The natural inlet of Biguglia also tends to fill in, although it is regularly dredged to keep a permanent connection with the sea. In many cases, inlets have been modified by humans by construction of hard-substrate artificial inlet banks and are being dredged regularly.

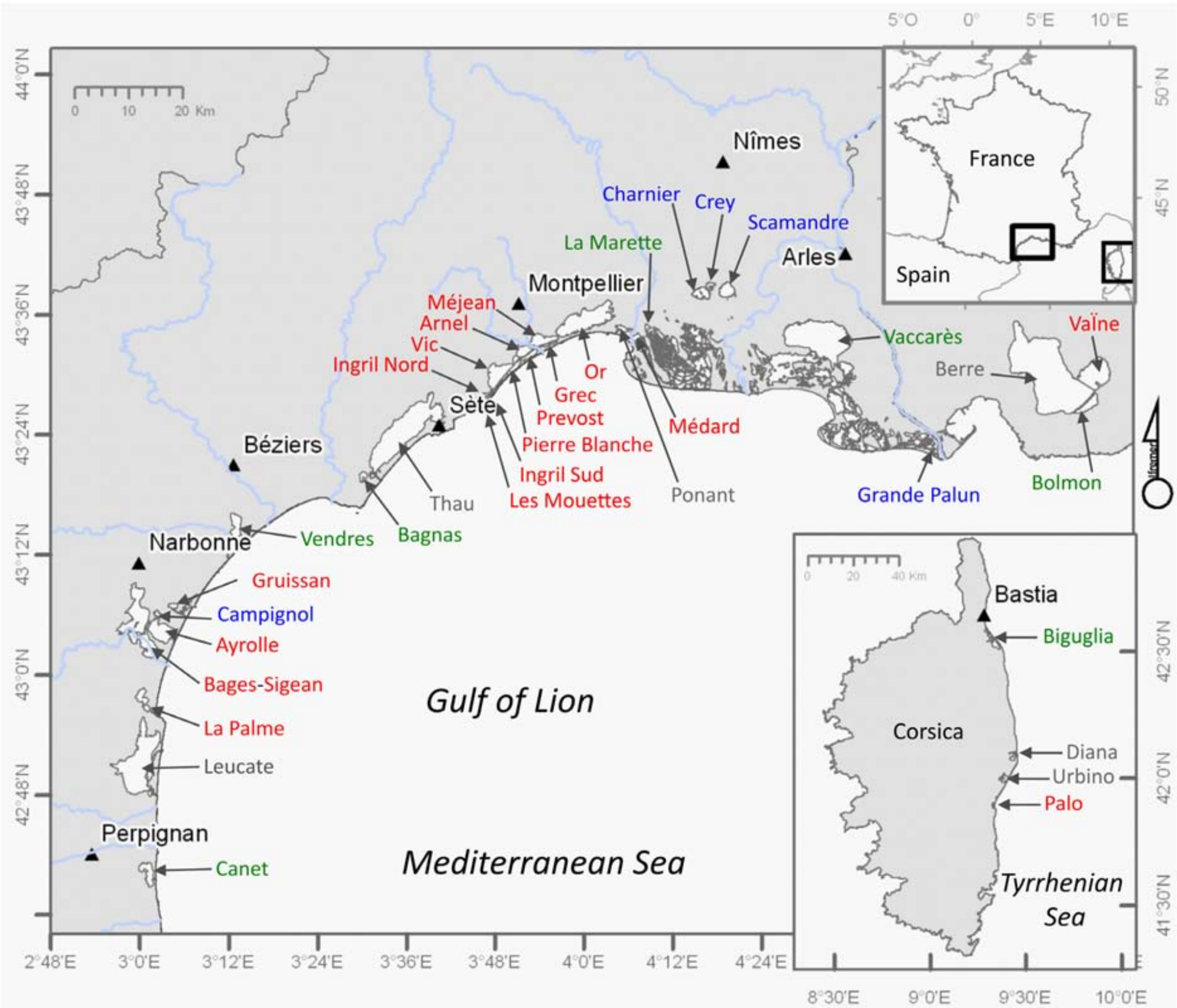

FIGURE 2 | Distribution of coastal lagoons, recognized for the WFD implementation in France, along the Mediterranean coastlines of continental France and the island of Corsica. In gray: deep lagoons (>2 m) not considered in this study. In blue: shallow (<2 m) oligohaline lagoons; in green: shallow (<2 m) mesohaline lagoons; in red: shallow (<2 m) poly- and euhaline lagoons. Triangles represent main urban centers (>40,000 inhabitants) in the proximity of the coastal lagoons.

This has fixed the locations of the inlets and resulted in a permanent connection of the lagoons with the sea. In some areas artificiality is even more striking. Some of the permanent lagoons in the deltaic setting have been separated from the sea several km's by the progression of the delta; the connections with the sea are maintained through artificial canals (Scamandre, Crey, Charnier) or through a wetland complex protected by a dike (Vaccarès). The Palavas lagoon complex (PLC) (in French: complexe lagunaire Palavasien) close to the city of Montpellier currently comprises eight lagoons (Ingril Nord, Ingril Sud, Vic, Pierre Blanche, Prévost, Arnel, Méjean, Grec) that have been created through the compartmentalization of the original "lagune de Mergueil." The creation of the Rhône-to-Sète canal since the 18th century has separated four permanent lagoons (Ingril Nord, Vic, Arnel, Méjean) from the coastline and exchanges of these lagoons with the sea occur through this canal.

To address the questions listed in **Figure 1** and described in section "Introduction" we use a combination of studies. The natural science information in this article is based on reviewing and synthesizing our previous studies (see **Table 1**), literature review including some gray literature that has been complemented with some additional unpublished data from three Ph.D. thesis (Leruste, 2016; Le Fur, 2018; Sy, 2019) and additional monitoring data from monitoring programs extracted from the French data bases "Banque Hydro[1]," Naïade[2], and "Surval," which among others hosts the data of the Réseau Suivi Lagunaire (RSL) monitoring program (Ifremer[3]). In the RSL, the total Nitrogen (TN) and Total Phosphorus (TP) data of the top 5 cm of the sediment (sampled at 6-year intervals) were expressed

[1] http://www.hydro.eaufrance.fr/

[2] http://www.naiades.eaufrance.fr/acces-donnees

[3] https://wwz.ifremer.fr/surval/

TABLE 1 | Scientific papers used to synthesize and review the information, which served to create the knowledge base about eutrophication gradients and oligotrophication trajectories in shallow Mediterranean coastal lagoons in South France.

Type of data used[1]	Period	Geographic area[2]	Subject	Biological (physico-chemical) compartment(s)	References
Monitoring (RSL)	1999–2001	G-Lion, Corsica	Eutrophication gradient	TN and TP as proxy for phytoplankton biomass, nutrient stoichiometry	Souchu et al., 2010
Monitoring (RSL)	1998–2002	G-Lion, Corsica	Eutrophication gradient	Phytoplankton taxa and biomass	Bec et al., 2011
Monitoring (RSL)	1998–2015	G-Lion, Rhône-d, C-azur, Corsica	Eutrophication gradient	Benthic macrophytes taxa	Le Fur et al., 2018
Historical observations	1970–2014	Biguglia (Corsica)	Eutrophication gradient and hydrological changes	Phytoplankton biomass and benthic macrophyte taxa	Pasqualini et al., 2017
Monitoring (RSL)	2000–2013	Palavas lagoon complex (G-Lion)	Oligotrophication	Phytoplankton biomass and taxa	Leruste et al., 2016
Monitoring (RSL)	2001–2014	G-Lion, Corsica	Oligotrophication	Phytoplankton biomass, water column nutrients	Derolez et al., 2019
Monitoring (RSL)	1998–2015	G-Lion, Corsica	Oligotrophication (poly and euhaline lagoons)	Benthic macrophytes	Le Fur et al., 2019
Experimental	2013–2014	Biguglia (Corsica)	Bioassay to detect nutrient limitation of phytoplankton	phytoplankton taxa and cell sizes, water column nutrients	Leruste et al., 2019a
Experimental	2014	Méjean, Ingril N, Ayrolle (G-Lion)	Bioassay to detect nutrient limitation of phytoplankton	Phytoplankton taxa and cell sizes, water column nutrients	Leruste et al., 2019b

[1] RSL, Réseau Suivi Lagunaire monitoring program. [2] G-Lion, Gulf of Lion; Rhône-d, Rhône delta; C-azur, Côte d'Azur.

as N and P contents in g/kg dry weight of sediments. Due to variable water contents and densities of the sediments it is not possible to convert these values to an aerial basis. These values were converted to mmol/kg for a stoichiometric analysis.

The question of the congruencies between WFD and Habitats directives has been based on analysis of legal texts and technical documents of the European Commission. In addition to the previous studies on citizens and stakeholder perceptions and opinions reviewed in this article, it also includes new results from citizen workshops organized in 2017 and 2018 at two different places in the eastern and western vicinity of the PLC. Forty-three randomly selected citizens working or recreating in both parts of the lagoon complex participated in the workshops (see **Supplementary Tables S1, S2** for details of the survey). The methodology was based on Sy et al. (2018) using Q-method to analyze consensus and diverging preferences of local citizens for ecosystem services provided by these lagoons. The results obtained for the citizens were compared with those obtained for the highly involved stakeholders (Sy et al., 2018).

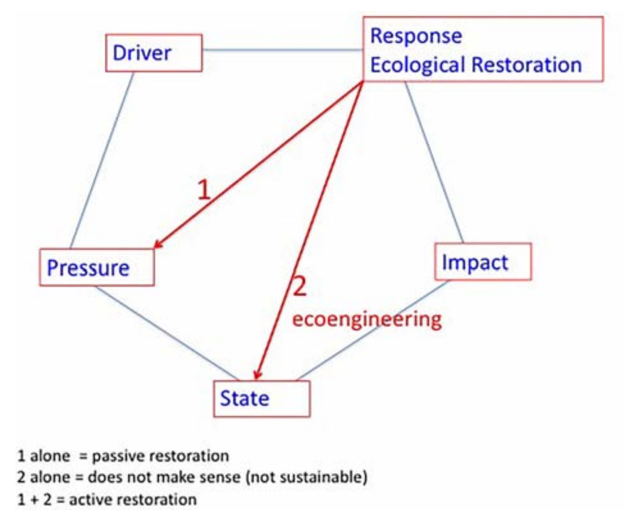

FIGURE 3 | Placing the Ecological Restoration practice in the D (drivers), P (pressures), S (state), I (impact), and R (response) framework. Accordingly, Ecological Restoration clearly is a societal response that targets 1) reducing the pressure on the ecosystem and 2) interfering directly with the ecosystem state through eco-engineering.

RESULTS AND DISCUSSION

Concept of Ecological Restoration in the Frame of the WFD

Ecological Restoration is a human action focused on degraded ecosystems that can thus be taken into account by the DPSIR framework, as this framework aims at analyzing the interactions between humans and ecosystems. Accordingly, ecological restoration should be considered as a Response (see **Figure 3**). In general, responses in DPSIR can potentially target D (drivers), P (pressures), S (state), and I (impact), although for ecological restoration it is clear that drivers and impacts are not really pertinent targets. Acting on major drivers is beyond the scope of action for ecological restoration *sensu stricto*; i.e., increasing urbanization and intensification of agriculture are drivers that could potentially be changed by spatial planning and agricultural policies, respectively, but not directly by ecological restoration. As ecological restoration has the ambition to repair degraded ecosystems it does not make sense to target the impacts; this would correspond to combating the symptoms and thus completely neglects the major objectives set for ecological restoration. Hence, ecological restoration could target the P (pressures) and the S (state).

For the problem of eutrophication of coastal lagoons, the pressure corresponds most often to nutrient over-enrichment mainly caused by nutrient loadings from their watershed. Hence, targeting the pressure thus corresponds to reducing the nutrient loading into the coastal lagoons. This action induces the reverse of eutrophication, i.e., oligotrophication. However, there is some debate about terminology, e.g., the term re-oligotrophication has been coined to describe the trajectories for lakes (Jeppesen et al., 2002) and coastal lagoons (Le Fur et al., 2019). This latter term seems to imply that the eutrophied systems were originally oligotrophic and that the trajectory should lead to full oligotrophic conditions. As not all coastal lagoons were originally oligotrophic (Nixon, 2009), it is preferable to use the more neutral term oligotrophication, which designates a process of moving toward more oligotrophic conditions. Moreover, in some cases the term de-eutrophication has been used.

An example of targeting the state (**Figure 3**) is seeding or planting marine angiosperms (Orth et al., 2012, MEPS thematic section, Van Katwijk et al., 2016), which are indicator species for good ecological conditions in coastal lagoons. Nevertheless, it is obvious that such eco-engineering activities alone without tackling the pressures on the system is not sustainable and would most likely result in failure in the short or mid-term (Van Katwijk et al., 2009; Cunha et al., 2012), a conclusion that has been confirmed in a recent study (Van Katwijk et al., 2016). We differentiated between passive and active ecological restoration (De Wit et al., 2017). Accordingly, passive ecological restoration is defined by action focused on reducing and combating the pressure (1 alone in **Figure 3**) and relies on the spontaneous ecological processes in the ecosystem for a trajectory toward improved conditions. Active ecological restoration implies a combination of reducing the pressure on the ecosystem in conjunction with eco-engineering *in situ* to improve the state of the ecosystem directly. A pertinent example for active ecological restoration in coastal lagoons is to combine a reduction of nutrient loading with seeding or planting marine angiosperms.

Figure 4 is the conceptual presentation derived from restoration ecology (Clewell and Aronson, 2013) that can be used as a roadmap and has been adapted to accommodate the specific requirements of the WFD. In the WFD, ecological status is measured using indicators and calculated as an ecological quality ratio (EQR; Zaldívar et al., 2008), with EQR being equal to one for the reference state. Hence, the EQR measures the deviation from reference conditions. The reference state is then taken as the ideal target for the ecological restoration, which is considered according WFD as high ecological quality status. Nevertheless,

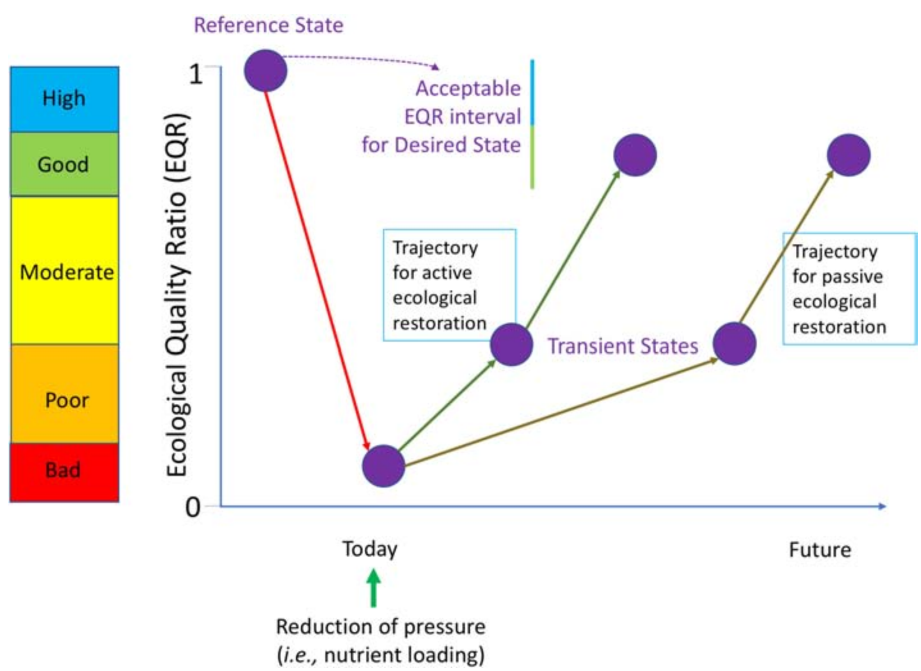

FIGURE 4 | Roadmap for ecological restoration applied in the frame of the WFD. The ecosystem states are characterized by a set of biological quality elements (BQE, e.g., phytoplankton, macrophytes) used as indicators and expressed as an Ecological Quality Ratio (EQR) represented on the *Y*-axis, with their qualifications for the corresponding ecological status (color coding). A historical ecosystem state not impacted by human pressures is taken as the reference condition for the ecological restoration. The WFD allows a small deviation from these reference conditions by targeting good or high ecological status for the Desired State of the coastal lagoons. After reduction of the pressure it is presumed that the degraded ecosystem (in the example currently in bad ecological status) will move gradually toward the Desired State, probably passing through one or multiple transient states (passive restoration). The recovery can be speed up by combining the reduction of the pressure with additional eco-engineering measures (active restoration, cf. **Figure 3**).

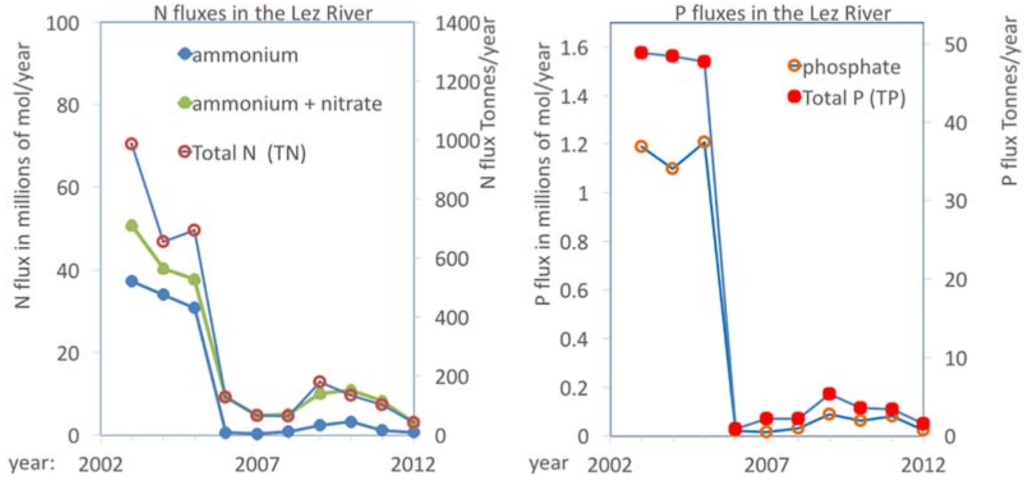

FIGURE 5 | Annual fluxes of N and P compounds in the Lez River. Values can be read on both scales (left in millions of mols/year; right in metric tonnes/year for N and P, respectively). Source: "Banque Hydro" and "Naïade" (réseau de données eaufrance, DCE).

the WFD is flexible and for achieving the target allows "*low levels of distortion resulting from human activity*," which "*deviate only slightly from those normally associated with the surface water body type under undisturbed conditions*" (WFD, Annex V) and ecological quality status are characterized as good for these conditions. Hence good or high ecological status correspond to the "Desired state" of the system that should be achieved in the future before a given deadline. After reduction of the pressure it is presumed that the system will move gradually toward this desired state, probably passing through one or multiple transient states. The recovery can be speed up by combining the reduction of the pressure with additional eco-engineering measures. The trajectories in the context of this study correspond to the different oligotrophication trajectories for the different scenarios. Hence, scientific knowledge of oligotrophication processes in coastal lagoons is of paramount importance for managers.

Can Scientific Knowledge of Eutrophication and Oligotrophication Trajectories in Mediterranean Coastal Lagoons Be Used to Provide Guidance for Ecological Restoration Actions?

Nutrient Loading Into Coastal Lagoons

In general, the demographic developments in the coastal zone since the 1950s and the resulting increased urbanization with insufficient sanitation and sewage treatments were the main drivers for the nutrient over-enrichment in the shallow coastal lagoons, particularly for those close to the cities of Montpellier, Narbonne, and Perpignan (cf. **Figure 2**). The urbanization to the south of Bastia accelerated later and particularly increased during the first decade of the 21st century (Pasqualini et al., 2017). Subsequently, action has been undertaken in these areas to reduce the nutrient loadings by improved sanitation (Leruste et al., 2016; De Wit et al., 2017; Pasqualini et al., 2017; Derolez et al., 2019; Le Fur et al., 2019). The case of the PLC is described in more detail. In the coastal lagoons of this complex, the bulk of the nutrient inputs originated from the urban waste-water treatment plant (WWTP) of the Montpellier agglomeration that discharged its effluents in the Lez River. As a result, most nutrients entered these lagoons via the Lez River and the Rhône-to-Sète canal. In 2005, a major investment was realized of 150 Million€ to upgrade the waste-water treatment facility in Montpellier, and create an off-shore outfall (Leruste et al., 2016; De Wit et al., 2017). All eight lagoons of the PLC benefited from this investment. In 2009, the WWTP of the city of Palavas (a smaller facility) was connected to the main WWTP in Montpellier, which resulted in an additional major reduction of nutrient loading into the Grec lagoon (Le Fur et al., 2019).

The annual fluxes of N and P carried by the Lez River are presented in **Figure 5**. This complete charge does not enter the lagoons as part flows through the river mouth directly into the Mediterranean Sea. Nevertheless, the Lez River represent the major tributary to the coastal lagoons of the PLC and this **Figure 5** gives therefore a good indication of the nutrient enrichment pressure on the system. From the Lez River, some water directly leaks into the Méjean and Arnel lagoons, but most enters the lagoons through the Rhône-to-Sète canal. As a result, the lagoon complex showed in the early 2000s an interesting gradient ranging from hypertrophic conditions in the lagoons close to the intersection of the Lez River and the canal to mesotrophic conditions in the two Ingril lagoons located farthest away (Bec et al., 2011; Souchu et al., 2010; Leruste et al., 2016; De Wit et al., 2017). Comparing the periods before (2003–2005) and after (2007–2012) showed decreases of 75 and 87% of total N and P, respectively (see **Figure 5**). Considering all other sources together with the major contribution from the Lez River it has been estimated that the intervention resulted in a reduction of the nutrient loadings from the watershed into the PLC of 83 and 73%, for total N and total P, respectively (Meinesz et al., 2013). Moreover, in the Lez River there was a strong shift in the relative proportions of NH_4^+ and NO_3^-, with NH_4^+ representing 79 and 20% of dissolved inorganic nitrogen (DIN), before and after, respectively. Before, organic nitrogen, i.e., the sum of dissolved and particulate organic nitrogen (DON and PON, respectively), represented 23% of total N and dropped to virtually 0 after the intervention (**Figure 5**).

What Can Be Learned From Shallow Lakes for Succession Patterns of Primary Producers in Mediterranean Coastal Lagoons?

Decades before tackling the eutrophication problems in coastal lagoons, the ecological restoration of aquatic systems started with the oligotrophication of shallow freshwater lakes. Therefore, it is inspiring to review freshwater lake oligotrophication and tempting to use it as a guideline for the ecological restoration of coastal lagoons. Scheffer (2001) stated "*Ponds and shallow lakes can be very clear with abundant submerged plants, or very turbid due to a high concentration of phytoplankton and suspended sediment particles*." This statement has been related to the alternative stable state theory (Scheffer et al., 1993). Accordingly, two attractors exist for these type of ecosystems, i.e., the SAV stable state attractor and the turbid stable state attractor, which dominate at very low and very high nutrient loadings, respectively. At intermediate nutrient loadings both attractors coexist and mathematically the ecosystem shows these two alternative stable states, which are each stabilized by a mixture of positive and negative feedback loops (Scheffer et al., 1993). Hence, a window of environmental conditions exists with alternative stable states in these aquatic ecosystems, where the actually occurring ecosystem state depends on the history of the system. Following the Scheffer model for shallow lakes, for environmental conditions within the window, the system can remain in the SAV state during increasing eutrophication until it reaches a critical turbidity, which is imposed by the minimum light requirements for growth and survival of the SAV. Above this threshold, the aquatic ecosystem shows a forward regime shift toward the turbid state. On the other hand, when nutrient loading is reduced in a turbid eutrophic lake, the system remains in the turbid state until the phytoplankton densities have decreased to such low values with a corresponding turbidity below the threshold. Hence, the model provides one possible explanation for hysteresis during eutrophication/oligotrophication trajectories. In addition, it can predict vulnerability of ecosystems states to perturbation, as for conditions within the window of two alternative stable states a perturbation may act to induce a regime shift (Beisner et al., 2003), e.g., a perturbation that causes a sudden die-back of SAV could result in a regime shift into the turbid state.

We may now ask the question, whether these two attractors and corresponding ecosystem states also exist for coastal lagoons? In general, the above-mentioned statement of Scheffer (2001) is pertinent for coastal lagoons as well. The clear water SAV dominated state in eu- and polyhaline lagoons is found under oligotrophic conditions and corresponds to the marine angiosperms, *Zostera noltei* Hornemann, 1832, *Ruppia cirrhosa* (Petagna) Grande, 1918, *Ruppia maritima* Linnaeus, 1753, *Cymodocea nodos*a (Ucria) Ascherson, 1870 and some slow-growing perennial macroalgae as e.g., *Acetabularia acetabulum* (Linnaeus) P. C. Silva, 1952, and *Valonia aegagropila* C. Agardh, 1823 (Le Fur et al., 2018, 2019). On the other hand, the highly turbid state is characterized by dense phytoplankton blooms often dominated by small phytoplankton (Bec et al., 2011; Leruste et al., 2016), belonging to picophytoplankton (<3 μm size; Bec et al., 2011) and ultraphytoplankton (>3 μm and <5 μm size; Li, 1995), with Chlorophyll *a* (Chl *a*) concentrations that may achieve several hundreds of mg/m^3. Clear examples of the latter are Or lagoon and before 2005 Méjean lagoon as well. Both ecosystem states have similar positive and negative feedback loops as those observed for their freshwater counterparts that stabilize these ecosystem states (Maxwell et al., 2017; Le Fur et al., 2019).

The succession patterns of primary producers with increasing eutrophication in Mediterranean coastal lagoons (Le Fur et al., 2018) are different from that of freshwater lakes. Accordingly, Le Fur et al. (2018) observed that poly- and euhaline lagoons follow the same general pattern as described by Schramm (1999) for the shallow coastal zone; i.e., with increasing eutrophication four stages have been observed, comprising successively (i) healthy marine angiosperms and perennial macroalgae, (ii) declining angiosperms with increasing loads of epiphytes, (iii) opportunistic macroalgae, and (iv) phytoplankton and bare non-vegetated sediment. In contrast, oligohaline coastal lagoons are dissimilar, because no blooms of macroalgae do develop. Hence, shallow oligohaline coastal lagoons are more similar to freshwater lakes. The dominant SAV in the Mediterranean oligohaline lagoons are charophytes and the angiosperm *Stuckenia pectinata* (Linnaeus) Börner, 1912 (formerly *Potamogeton pectinatus*). The latter species form long stems allowing an important proportion of its leaves to float at the surface of the water. This way, this species escapes from competition with planktonic algae for light. Mesohaline coastal lagoons are intermediate between the oligohaline on one side and the poly- and euhaline lagoons on the other side. As in poly- and euhaline lagoons, opportunistic macroalgae develop at intermediate eutrophication levels. On the other side, *Ruppia* species are characteristic angiosperms in mesohaline lagoons, which like *S. pectinata* form long stems with floating leaves (Le Fur et al., 2018).

In conclusion, in meso-, poly-, and euhaline lagoons, the Scheffer model for shallow freshwater lakes is not directly applicable as a third group, i.e., opportunistic macroalgae, can dominate primary producer communities and as a rather persistent community represent a third stable state in addition to the angiosperms and planktonic algae stable states. By comparison, the stage of declining angiosperms with increasing loads of epiphytes appears as a transient state, that is not stable because the developing epiphytes weaken their own support. It thus appears that the opportunistic macroalgae occupy a niche in the window where bistability of the two end members could occur. However, so far no clear mathematical analysis allows to determine whether the opportunistic macroalgal states completely overrules the theoretical window of bistability, or whether multiple stability domains exist in coastal lagoons. Increasing eutrophication in these coastal lagoons is correctly characterized by regime shifts according to the Schramm scheme (Le Fur et al., 2018). Nevertheless, threshold effects and regime shifts do not necessarily imply multiple stable states, as very steep but continuous shifts in equilibrium states are also possible (Petraitis and Hoffman, 2010). Viaroli et al. (2008) suggested two possible mechanisms for the regime shift from angiosperms to floating macroalgae, i.e., either as the conventional mechanism based on continuous shift or on a so-called dynamic shift implying a domain of bi-stability. One conclusion from their work, i.e., "*The alternative states which occur through the transition from pristine to modified primary producer communities can also be viewed as a sequence of stable states*......" (Viaroli et al., 2008) is a bit ambiguous in this respect. We think that modified primary producer communities along eutrophication gradients in coastal lagoons should indeed be viewed as a sequence of stable states, but that it remains so far, uncertain whether these represent continuous shifts in equilibrium states or whether it hides multiple stable state domains.

Oligotrophication Trajectories

Eight-year to seventeen-year time series from monitoring programs have been used to study the oligotrophication trajectories in the French Mediterranean coastal lagoons, which allowed to make comparisons before and after nutrient reduction measures (Leruste et al., 2016; Derolez et al., 2019; Le Fur et al., 2019). In addition, a longer time series of more heterogeneous observations was available for Biguglia lagoon (Pasqualini et al., 2017). The monitoring effort was concentrated during summer and based on three samplings in June, July, and August, successively. Water column variables, including nutrients (NO_3^-, NO_2^-, NH_4^+, PO_4^{3-}, silicates), TN, TP, Chl *a*, and the pico- and nanophytoplankton cell abundances in 16 lagoons, including shallow and deeper lagoons, have been analyzed by a between-station principal component analysis (PCA). The trajectories of 14-year time series for some selected stations in shallow lagoons (Ayrolle, Bages, Méjean) have been plotted in the PCA plane (Derolez et al., 2019). The first axis of this PCA, which explained 81% of the variation, correlated mainly with Chl *a*, TN, TP, and dissolved inorganic phosphorus (DIP, i.e., PO_4^{3-}), and was clearly interpreted as the main eutrophication axis. The second axis (11% of variance explained) was correlated to DIN. The station in Méjean lagoon, which was hypertrophic before 2005, moved toward oligotrophy, although it was still eutrophic in 2014. The station in Bages, which was eutrophic before 2004 also moved toward oligotrophy, albeit less abruptly than the station in Méjean. It was concluded that integrative water column variables (Chl *a*, TN, and TP) recover quickly, i.e., within 1–3 years after nutrient reduction measures (Derolez et al., 2019).

Changes in phytoplankton communities following nutrient reduction measures were studied for the coastal lagoons of the PLC (Leruste et al., 2016). The Chl *a* concentrations in the hypertrophic Méjean were $185 \pm 131\ \mu gL^{-1}$ before and dropped on average by one order of magnitude after the nutrient input reduction (cf. **Figure 5**), with a concomitant drop of autotrophic picoeukaryotes cell numbers. There was a shift from small diatoms that still dominated in 2006 to green algae. The other lagoons of the PLC also showed significant decreases in Chl *a* after nutrient input reduction and increases in the proportions of peridin-containing dinophytes (Leruste et al., 2016), probably related to their mixotrophic capacities (phagocytosis and osmotrophy). Oligotrophication in shallow freshwater lakes also results in a drop of Chl *a* concentrations and phytoplankton abundance with the species composition shifting toward diatoms, cryptophytes and chrysophytes (Jeppesen et al., 2005). It has been observed that summer phytoplankton communities in the Mediterranean coastal lagoons in South France are clearly limited by P under oligotrophic conditions and with increasing eutrophication level change through P/N co-limitation to N-limitation (Souchu et al., 2010). It appears that this tendency is maintained during the oligotrophication trajectories and perhaps even strengthened. Hence, during the summer of 2014, bioassay experiments showed that the phytoplankton community in Méjean lagoon was clearly N-limited. The ambient concentrations of DIN and DIP were around 1 and 2.5 μM, respectively, showing excess of DIP. In contrast, a co-limitation by N and P was observed in the eutrophic Biguglia, the mesotrophic Ingril and the oligotrophic Ayrolle lagoons (**Figure 2**); the latter was studied for comparisons. In the PLC, the phytoplankton communities of the Ingril and Méjean lagoons strongly responded to the experimental nutrient pulses, suggesting that despite their oligotrophication trajectories, these lagoons were still vulnerable to occasional eutrophication events (Leruste et al., 2019a,b).

The impact of the oligotrophication process on the macrophyte assemblages was studied in 21 poly- and euhaline lagoons, by comparing the taxonomic composition of the macrophytes sampled at different stations in the lagoons with variables characterizing the water column using 17 years of observations (Le Fur et al., 2019). The observations were again restricted to the summer period. The data comprised a series of paired data tables, i.e., for species and the other for the water column environmental conditions. Two axes were considered for the multivariate analysis (STATICO factor map); the first axis (88.5% of total variability explained) correlated with Chl *a* concentrations, turbidity, TN and TP, and was again interpreted as the main eutrophication axes. The second axis (9.5% of total variability explained) correlated with DIN and DIP concentrations. A general scheme was inferred for the changes in macrophyte assemblages during the oligotrophication process. Hence, when placing hypertrophic and oligotrophic conditions end to end, the oligotrophication trajectories were described by a sequence of three ecosystem states dominated by (i) phytoplankton with bare non-vegetated sediments, (ii) opportunistic macroalgae, (iii) seagrass and perennial macroalgae, punctuated by regime shifts between these ecosystem states. The regime shift from the phytoplankton-dominated state to opportunistic macroalgae was observed in Méjean lagoon, where *Ulva rigida* C. Agardh, 1823, dominated in 2009 after a strong decrease in Chl *a* (see above), followed by more diverse communities since 2012 with among others different *Gracilaria* spp. and *Chaetomorpha aerea* (Dillwyn) Kützing, 1849. However, during the 10-year period following the reduction of the nutrient loading, Méjean lagoon did not achieve the oligotrophic

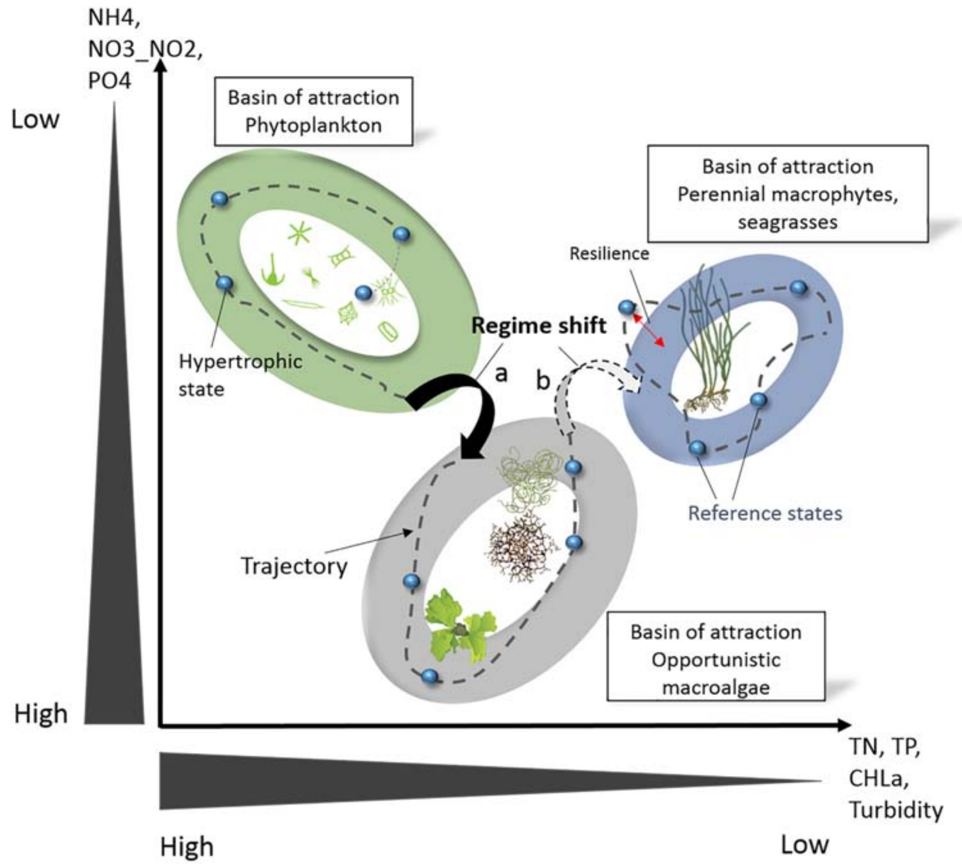

FIGURE 6 | Illustration of ecosystem states and regime shifts along oligotrophication trajectories in Mediterranean coastal lagoons. The ellipses represent the three basins of attraction that were identified, i.e., three ecosystem states dominated by (i) phytoplankton with bare non-vegetated sediments, (ii) opportunistic macroalgae, and (iii) seagrass and perennial macroalgae. The small blue spheres represent the different ecosystem states and the stippled lines their trajectory according their chronology. Small pictograms courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/symbols/). Creative Commons license, reproduced from Le Fur et al. (2019).

state dominated by seagrass and perennial macroalgae (Le Fur et al., 2019).

Interestingly, for primary producers, the oligotrophication trajectories in poly- and euhaline lagoons appear as the inverse of the eutrophication trajectories (cf. above, Schramm, 1999), with the exception of the transitional state occurring during eutrophication of angiosperms perishing under a high coverage of epiphytes. Again, during oligotrophication, opportunistic macroalgae, including *Ulva intestinalis* Linnaeus, 1753, *U. rigida*, *C. aerea*, and *Gracilaria* spp., occupy an intermediate domain between the two end members, i.e., ecosystem states dominated by (i) phytoplankton and (ii) seagrass and perennial macroalgae, at the hypertrophic and oligotrophic extremes, respectively. We have assumed that these three ecosystem states are the result of attraction basins reinforced by positive and negative feedback loops. Hence, these three attraction basins and the oligotrophication trajectories have been plotted graphically (**Figure 6**) and compared with the STATICO factor map (Le Fur et al., 2019). Despite representing the hypertrophic state, the attraction basin for phytoplankton-dominated systems is projected at low DIN and DIP levels, related to the high affinity uptake capacities of phytoplankton.

Sediment N and P

Figure 7 shows that the sediments of the oligotrophic lagoons La Palme and Ayrolle are characterized by low TP and low TN contents of approximately 7 and 50–80 mmol/kg, respectively. These values are roughly four times higher in the hypertrophic Méjean lagoon. The geometric mean regression shows for the poly- and euhaline lagoons a N/P molar ratio of 19.3, which is close to the Redfield ratio of 16. This suggests that during eutrophication, the sediments become enriched in N and P, probably largely due to the accumulation of organic matter. The regression line shows that on average 5.8 mmol/kg TP is predicted for TN = 0. Hence, even in the absence of organic matter, phosphorus is expected to be present, which can be explained by the presence of inorganic mineral forms like phosphoapatites and P sorbed on iron compounds. Søndergaard et al. (2003) stated that "*Internal P loading originates from a pool accumulated in the sediment at high external loading, and significant amounts of phosphorus in lake sediments may*

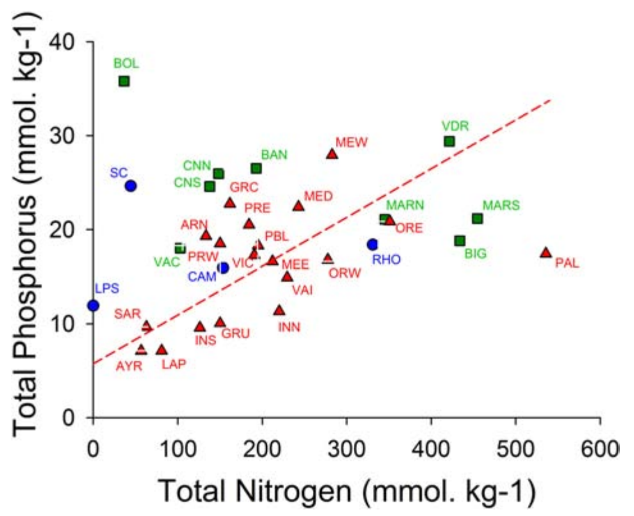

FIGURE 7 | Graph of total nitrogen (TN) and total phosphorus (TP) contents of the top 5 cm of the sediments expressed per dry weight (DW) of sediments. Oligohaline lagoons in blue circles, mesohaline lagoons in green squares and poly- and euhaline lagoons in red triangles (cf. **Figure 2**). A geometric mean regression was calculated for the poly- and euhaline lagoons and depicted as a broken red line (TP = 0.0519 × TN + 5.752 (mol · kg^{-1}), $r = 0.494$), corresponding to a N/P molar ratio of 19.27. Codes for sampling stations: AYR, Ayrolle; ARN, Arnel; BAN, Bagnas; PBL, Pierre Blanche; BIG, Biguglia; BOL, Bolmon; CAM, Campignol; CNN, CNS, Canet North and South stations, respectively; GRC, Grec; GRU, Gruissan; INN, Ingril Nord; INS, Ingril Sud; LAP, La Palme; LPS, La Palissade; MARN, La Marette; MED, Médard; MEE, MEW, Méjean, East and West stations, respectively; ORE, ORW, Or East and West stations, respectively; PAL, Palo; PRE, PRW, Prévost, East and West stations, respectively; RHO, Rhône de Saint Romans; SC, Scamandre; VAI, Vaïne; VDR, Vendres.

be bound to redox-sensitive iron compounds or fixed in more or less labile organic forms." In addition, the mineralization of the organic matter in the sediment provide inorganic N and P that could sustain a flux of DIN and DIP across the sediment-water interface, which represents thus another internal source for the water column.

Coastal Lagoons in the Habitats Directive and Links With WFD Approaches

The scientific literature about coastal lagoons in Europe comprises numerous studies on the implications of the WFD, but to our knowledge very few studies have assessed the consequences of its co-implementation with the Habitats and Birds Directives for these ecosystems. While the WFD explicitly links the WFD with the Habitats and Birds Directives (WFD, Annex VI) in common day practice, coastal lagoon managers in the EU are systematically confronted with co-implementation of WFD and Habitats Directive, in some cases even together with the Birds Directive.

Coastal lagoons are considered in the WFD as transitional water bodies (TWB) together with river mouth ecosystems (water bodies in estuaries and deltas). For the WFD application in the French Mediterranean water basin, the coastal lagoons are considered as a subcategory within TWB, i.e., Mediterranean lagoons, which are non-tidal or nano-tidal. Moreover, the "Comité de Bassin Rhône Méditerranée" decided in 2009 to exclude ephemeral coastal lagoons as well as permanent coastal lagoons smaller than 50 ha. The Habitats Directive recognizes coastal lagoon as a priority habitat 1150, which has been defined (European Commission DG Environment, 2013) as "*expanses of shallow coastal salt water, of varying salinity and water volume, wholly or partially separated from the sea by sand banks or shingle, or, less frequently, by rocks. Salinity may vary from brackish water to hypersalinity depending on rainfall, evaporation and through the addition of fresh seawater from storms, temporary flooding of the sea in winter or tidal exchange. With or without vegetation from Ruppietea maritimae, Potametea, Zosteretea, or Charetea (CORINE 91: 23.21 or 23.22).*" The latter are vegetation units defined by phytosociology. Hence, the coastal lagoons listed in **Figure 2** all belong to the priority habitat 1150 of the Habitats directive and have been included in Natura 2000 sites. In France, many temporal saline ponds along the Mediterranean coastline have also been included in the priority habitat 1150. In some cases, the temporal ponds are coastal lagoons according the definition of Kjerfve (1994), i.e., with a temporal inlet connecting with the sea. In such cases, during summer the inlet closes and the lagoon dries out completely. However, many of the temporal ponds that function as endorheic systems without direct connection to the sea, have also been included in habitat 1150. Nonetheless, such an approach is fully acceptable according the Interpretation Manual of European Union Habitats – EUR28 (European Commission DG Environment, 2013) as "*the salt basins and salt ponds may also be considered as lagoons, providing they had their origin on a transformed natural old lagoon or on a saltmarsh, and are characterized by a minor impact from exploitation.*" Many of the former salt-ponds in abandoned Salinas typically represent such a case (De Wit et al., 2019). In the PLC two Salinas have been abandoned, i.e., the Salins de Frontignan and Villeneuve, which have been included as peripheral wetlands in the Natura 2000 site of the "etangs Palavasiens" (FR9101410). In the Mediterranean climate, these temporal ponds are typically filled with water from rain and run-off after heavy rainfall in autumn and tend to dry out during late spring or the summer period. The salinity of the ephemeral water column, which originates in the endorheic systems from dissolving the salt in the soils, varies strongly during the year. The aquatic plants in these ephemeral ponds comprise species of the association of the angiosperm *Althenia filiformis* Petit, 1928 and the charophyte *Lamprothamnium papulosum* (K. Wallroth) J. Groves, 1916.

The Natura 2000 network is a network of protected areas designated according the Habitats directive [Site of Community Interest (SCI)] and the Birds Directive [Special Protection Areas (SPA)]. The member states have an obligation to report on the conservation status of habitats within Natura 2000 sites. It is a major challenge to combine the management and surveillance monitoring for the "ecological status" according the WFD with that for the "conservation status" of habitats and species according the Habitats Directive and extent these to the temporal lagoons.

Communication With Coastal Lagoon Stakeholders, Their Values, Perceptions, and Opinions

The normative approach of the WFD is based on the use of EQRs calculated for different biological quality elements (BQE, e.g., phytoplankton, macrophytes). Based on the EQR values classes are attributed using the qualifications Bad, Poor, Moderate, Good, and High, based on the deviation from the reference state corresponding to conditions of minimal anthropogenic impact. However, an aggregated qualification is attributed based on the principle "one out all out," which means that the BQE with the lowest EQR determines the overall EQR and quality class, thus overruling all the others.

These aggregated qualifications are often restituted to the managers, policy makers and general public by using a color code. Accordingly, **Figure 8** presents the color coding for the ecological status of the water column in the PLC restituted every year by the Lagoon monitoring network "RSL" (IFREMER, 2014).

Figure 8 shows that in general the water quality has improved from 2000 to 2013 for the lagoons of the PLC, although in 2013 only two (Ingril Nord and Ingril Sud) of eight lagoons had achieved "good" ecological status for the water column, while the rest was reported as poor or moderate. Social representation of water quality by the local populations was identified and compared to the water quality assessment carried out according to the French surveillance monitoring system (Audouit et al., 2019). More than half of the interviewees in the PLC considered water quality as moderate (40%) or good (26%). In 2013, only Ingril Nord and Ingril Sud lagoons were in a good status. Hence, social representation gave a higher score than the surveillance monitoring. Partly this difference could be attributed to the conservative scoring used in the WFD based on the principle "one out all out" (Audouit et al., 2019). Nevertheless, both the surveillance monitoring and the social perception of water quality explain a perception in society of the failure of achieving WFD goals for a large number of coastal lagoons. Interestingly, the social representation of the biodiversity of the coastal lagoons was much more positive than that of water quality, with 40% valuing as high and 40% valuing as good for the lagoons of the PLC. In general, interviewees judged that the current situation was better

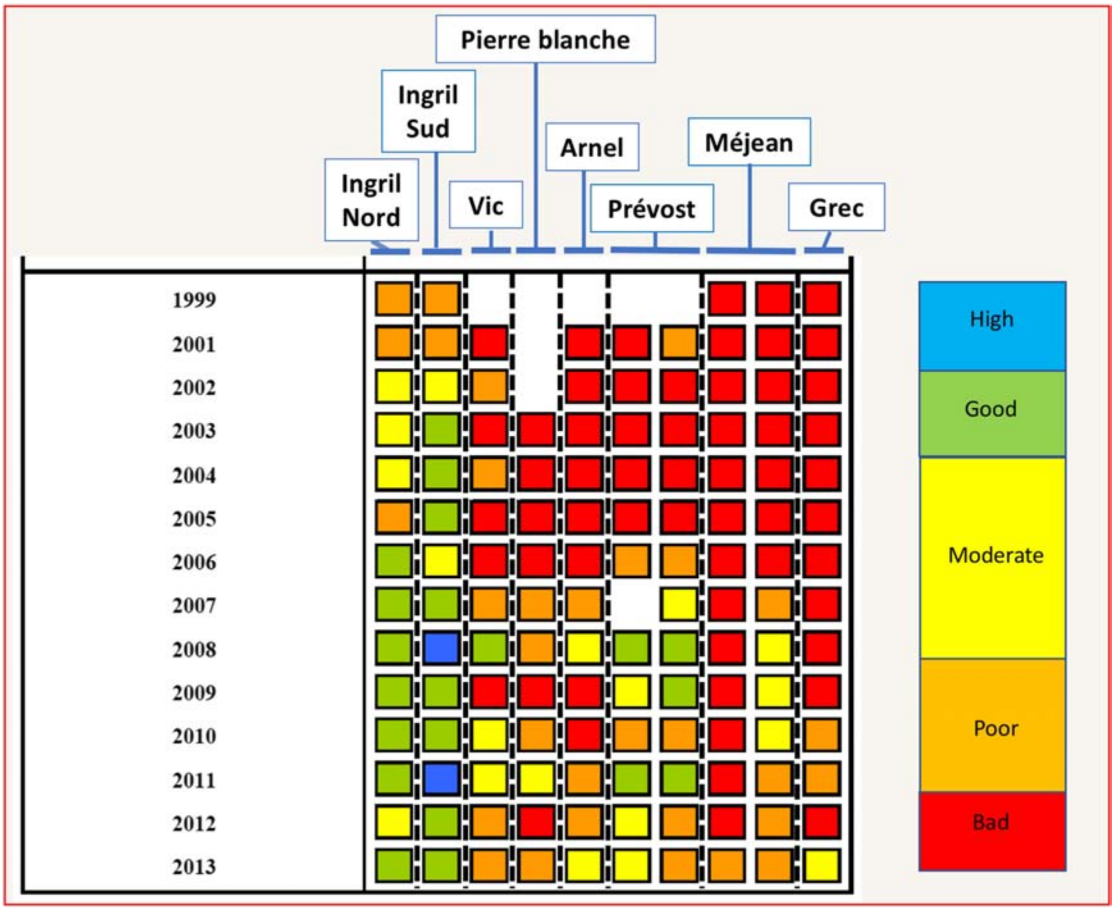

FIGURE 8 | Color coding restitution of water quality by the Lagoon Monitoring Network for the lagoons of the PLC from 1999–2013 (Figure designed using Tableau 11.2, page 175 of IFREMER, 2014, Courtesy Ifremer).

than in the past (Audouit et al., 2019), meaning that some positive impact of public policy and management has been perceived by the general public.

The lack of full restoration of water quality and of other aspects for qualifying as good ecosystem state (e.g., benthic macrophytes) raised the question whether the reduction of the nutrient loading should not be combined with additional eco-engineering measures (cf. **Figures 3, 4**, active restoration). Therefore, the hypertrophic Méjean and the mesotrophic Ingril Sud lagoons were compared. While it was considered that Ingril Sud lagoon was already on a trajectory for spontaneous recovery after the nutrient reduction (passive restoration), additional measures for active restoration were considered for Méjean lagoon. These measures comprise a first phase based on collection of macroalgae and their exportation from the lagoon followed by a second phase based on planting marine angiosperms. Based on these suggestions, four scenarios were defined for Méjean lagoon, ranging from passive to a high level of active ecological restoration and questionnaires were designed for residents and tourists and face-to-face interviews were performed. More than 85% of both residents and tourists expressed that they have a positive perception of the ecological and societal importance of coastal lagoons and of their ecological restoration. A willingness to pay for active restoration (i.e., additional costs for the eco-engineering measures on top of the 150 Million€ engaged for the improvement of the water quality, see above) was expressed by 70 and 60% of the resident and tourists' respondents, respectively. The residents were willing to pay each about 25€ a year for the highest level of ecological restoration and willing to allocate a much smaller amount of about 5€ for the improvement of the infrastructure for visiting the site (De Wit et al., 2017).

The study of how stakeholders perceive ecosystem services of coastal lagoons and how they value them it is important to understand the benefits people obtain from these ecosystems and get insights on their expectations for the future. Hence, such an assessment of ecosystem services is very instrumental for ecological restoration projects to assess whether the project corresponds to the expectations and perceptions of the stakeholders. Monetary valuations of ecosystem services may present an obvious advantage for ecological restoration projects as it allows a straightforward cost-benefit analysis (De Groot et al., 2013; De Wit et al., 2017). Nevertheless, monetary approaches can be problematic when working with stakeholders as most of the ecosystem services provided by the coastal lagoons areas are non-market services and, stakeholders in general, tend to reject the monetary approaches. Therefore, we turned to non-monetary approaches and used the Q-method (Sy et al., 2018), which is based on a serious card game that allows to capture levels of consensus and divergence among participants. Thirty-one ecosystem services provided by the lagoons were selected by a focus group (see **Table 2**) and highly involved stakeholders were asked to rank these ecosystem services. The highly involved stakeholders were characterized by their strong involvement in following the management process and many of them were members of the Natura 2000 committee. They were grouped in seven categories, i.e., local government, private sector, NGOs, scientists, public and para-public sectors, managers, and local residents. Statistical analysis was improved by bootstrapping in order to obtain additional and more detailed measures of variability and thus help to better understand the data and the outcomes. Accordingly, three groups of these stakeholders, each sharing similar point of views the group, were extracted (Sy et al., 2018). Here, we complement this study with new data obtained through citizen workshops for local citizens (see **Supplementary Table S2** for their sociodemographic composition). These local citizens can thus be considered as less-involved stakeholders (e.g., García-Nieto et al., 2015).

Forty-three citizens actively participated in these citizen workshops and completed the Q-method ranking of the same 31 ecosystem services (**Table 2**), ranked earlier by the highly involved stakeholders (Sy et al., 2018). The analyses again extracted three different groups which are listed in **Table 3** with their most salient results (more detail in **Supplementary Figures S1, S2**). All three groups attributed high level of priority to regulation and maintenance services. Consensus among these citizens was particularly striking for providing protection against flooding and other extreme natural events. Biodiversity and nursery, water purification capacity and microclimate regulation were among the top of priorities for two or one of the groups, but the overlapping variability of the bootstrapped estimates indicated still a certain degree of consensus of their importance. There was also consensus on the most unimportant side, which was most likely based on mere rejection, i.e., for camping. Two of the three groups also seemed to reject waterfowl hunting. Group #1, which gave an absolute priority to regulation and maintenance services, was quite similar to the preferences of group #2 of the highly involved stakeholders of the PLC, characterized as "environmental and territorial approach" (Sy et al., 2018). Group #2 had a clearly naturalist profile, giving among the regulating services the highest priority to biodiversity and nursery services with strong preferences for cultural services as bird watching and esthetic value of species and habitats. This result was rather similar to group #1's preferences ("environmental and hedonic vision" identified for the highly involved stakeholders (Sy et al., 2018). The hedonist aspect is also reflected by the importance group #2 attached to the sentiment of relaxation. A very small group #3 comprising only four persons, among which three from the eastern part of the PLC, was atypical, because it valued two provisioning services very high, i.e., (i) shellfish farming, developed in the Prévost lagoon and (ii) commercial inland navigation, related to traffic on the Rhône-to-Sète canal running through the lagoon complex. This group was hardly interested in naturalistic aspects, although it valued the esthetic value of the landscape very high. Perhaps this is related to a sensitivity for heritage aspects.

GENERAL DISCUSSION AND SYNTHESIS

Restoration ecology of coastal lagoons should integrate solid knowledge of the biodiversity of the communities and ecosystem functioning. But, as it studies a human activity, i.e., ecological restoration, we argue that it should also take into account

TABLE 2 | Modified from Sy et al. (2018).

ES category	ES subcategory	Ecosystem service	General definition
Provisioning services	Food provision	Shellfish resources	The provision of biomass for human consumption and the conditions to grow it. It mostly relates to cropping, animal husbandry, and fisheries.
		Biomass for grazing	
		Crops	
		Shellfish farming	
		Fish resources	
		Fish farming	
	Water provision	Commercial inland navigation	The provision of water for human consumption and for other uses.
	Biotic materials and biofuels	Non-food products	The provision of biomass or biotic elements for non-food purposes.
Regulation and maintenance services	Water purification	Purification capacity	Biochemical and physicochemical processes involved in the removal of wastes and pollutants from the aquatic environment.
		Wastes decomposition	
	Coastal protection	Flooding and other extreme events regulation and protection	Protection against floods, droughts, hurricanes, and other extreme events. Also, erosion prevention in the coast.
		Banks reinforcement	
	Climate regulation	Microclimate regulation	Regulation of greenhouse and climate active gases. The most common proxies are the uptake, storage, and sequestration of carbon dioxide.
	Life cycle maintenance	Nursery and biodiversity maintenance	Biological and physical support to facilitate the healthy and diverse reproduction of species.
Cultural services	Symbolic and esthetic values	Esthetic value of landscapes	Exaltation of senses and emotions by landscapes, habitats, or species.
		Local identity	
		Esthetic value of habitats or species	
		Historical sites	
	Recreation and tourism	Recreational boat navigation	Opportunities that the natural environment provide for relaxation and amusement.
		Non-motorized water sports	
		Bird watching	
		Cycling	
		Horse riding	
		Waterfowl hunting	
		Sentiment of relaxation	
		Camping	
		Recreational hiking and walking	
		Recreational fishing	
	Cognitive effects	Artistic inspiration	Trigger of mental processes like knowing, developing, perceiving, or being aware resulting from natural landscapes or living organisms.
		Research opportunity	
		Environmental education	

The Ecosystem Services (ESs) supplied by the coastal lagoons of the Palavas lagoon complex have been categorized according to the classification designed for coastal and marine ESs by Liquete et al. (2013), which has recently been included in CICES version 5.1 (Haines-Young and Potschin, 2018). These 31 ESs (Q-set) were ranked according Q methodology by both the highly-involved stakeholders (Sy et al., 2018) and the local citizens in citizens' workshops. The Ecosystem service in italics, i.e., Fish farming, is currently not exploited in these lagoons.

the pertinent aspects of legislation, as well as socio-economic aspects concerning perceptions by local stakeholders and their expectations for the future desired state of the ecosystems. Hence, restoration ecology, in addition to sound ecological knowledge, thus has to establish multidisciplinary collaborations with social sciences. In this study, we analyzed by such an integrated multidisciplinary approach, the practice of ecological restoration to combat eutrophication, which has been achieved by reducing the external nutrient loading to shallow Mediterranean coastal lagoons in South France. This action has resulted in oligotrophication of formerly eutrophied coastal lagoons.

When the reduction of nutrient loadings to shallow coastal lagoons was initiated in the early 2000s through the WFD, little scientific knowledge was available for predicting the ecosystem trajectories of shallow coastal lagoons during oligotrophication and ideas were inspired by the experience obtained for the oligotrophication of shallow freshwater lakes that had been initiated two decades earlier. The experience in South France was followed by intensive surveillance monitoring (RSL) and

TABLE 3 | Most salient results from analyzing the Q-sorts of the 43 participants in citizen workshops.

		Group #1	Group #2	Group #3	Bootstrap results
		Environmental and territorial approach	**Naturalist**	**Environmental utilitarian – local identity**	**Groups with overlapping variability, *all three groups different $p < 0.05$**
Most important (score type: + + = 4 or + = 3)					
Regulating and maintenance	Protection against flooding	++	+	+	1, 2, 3
	Biodiversity + nursery	+	++		1, 2, 3
	Purification capacity	++		++	1, 2, 3
	Waste decomposition	+			2, 3
	Microclimate regulation	+			1, 2, 3
Cultural services	Sentiment of relaxation		++		1, 2, 3
	Bird watching		+		1, 3
	Esthetic value species and habitats		+		*
	Esthetic value of landscape			++	*
Provisioning	Commercial inland navigation			+	*
	Shellfish farming			+	*
Most unimportant (score type: −− = −4)					
Cultural services	Camping	−−	−−	−−	1, 2, 3
	Waterfowl hunting	−−	−−		1, 2, 3
	Esthetic value species and habitats			−−	*
Composition of the groups					
Home of resident	Number of respondents				
Eastern part of PLC	10	4	3	3	
Western part of PLC	17	11	6	0	
Montpellier	11	6	5	0	
Others	5	3	1	1	
Total – number	43	24	15	4	
Total		56%	35%	9%	

*Statistical analysis was improved by bootstrapping and groups with overlapping variability for a given ES are indicated. The analysis meaningfully extracted three factors allowing to identify three corresponding groups. The representative Q-sorts for the different groups are provided in **Supplementary Figure S2**. This table resumes the services selected for most important and less important and provides the geographic position of the home address of the respondents.*
PLC, Palavas lagoon complex; eastern part PLC, municipalities of Pérols, Palavas, Lattes, Villeneuve-les-Maguelone; western part of PLC, municipalities of Mireval, Vic-la-Gardiole, Frontignan; Others, other municipalities close to the lagoon complex.

scientific studies (cf. **Table 1**) focused on improving the understanding of the eutrophication (Souchu et al., 2010, Bec et al., 2011; Le Fur et al., 2018) and of oligotrophication trajectories (Leruste et al., 2016; Pasqualini et al., 2017; Derolez et al., 2019; Le Fur et al., 2019). From this we have obtained a large body of understanding and empirical knowledge to guide the ecological restoration practice. We hereby summarize the main findings. Coastal lagoons show similar attractors at both extremes along the eutrophication gradient as described for freshwater lakes (cf. Scheffer, 2001). At the oligotrophic side, shallow poly- and euhaline lagoons show transparent water and are home to meadows of the marine angiosperms, *Z. noltei*, *Ruppia* spp. and *C. nodosa* together with some slow-growing perennial macroalgae as e.g., *A. acetabulum* (Le Fur et al., 2018). At the hypertrophic extreme, shallow poly- and euhaline lagoons show turbid water with dense phytoplankton (Bec et al., 2011; Leruste et al., 2016). However, in contrast to freshwater lakes, massive blooming of opportunistic macroalgae represents a third ecosystem state in shallow coastal lagoons (Valiela et al., 1997; De Wit et al., 2001; Viaroli et al., 2008; Le Fur et al., 2018, 2019) in between both extremes. The importance of this third attractor, implies that the attractive bistability scheme proposed by Scheffer for freshwater lakes does not apply straightforwardly for coastal lagoons. Nevertheless, for the poly- and euhaline lagoons, oligotrophication trajectories from hypertrophy to oligotrophy have now been described as a sequence from (i) phytoplankton with bare non-vegetated sediments, (ii) opportunistic macroalgae, and (iii) seagrass and perennial macroalgae, punctuated by regime shifts between the ecosystem states (Le Fur et al., 2019). At present, in these systems, it remains unclear whether domains of multiple stable states occur for the eutrophication – oligotrophication pathways or whether all the regime-shifts represent steep but continuous shifts in equilibrium states. The shift from the phytoplankton-dominated state to the macroalgal-dominated state occurs very quickly upon reduction of the external nutrient loading, i.e., within one or a couple of years (Le Fur et al., 2019), and it has been argued that water-column showed little hysteresis (Derolez et al., 2019). In contrast, ten years of oligotrophication of the hypertrophic lagoons did not yet result in the second regime shift from a macroalgal-dominated state into the SAV state (Le Fur et al., 2019), suggesting strong hysteresis for this final part of the trajectory. Such hysteresis may in part be explained by a time lag to re-establish the positive feedbacks (Maxwell et al., 2017) and mutualistic networks (Van der Geest et al., 2020) in the seagrass systems. Another source of hysteresis during oligotrophication may be due to time lags caused by internal loading from the biogeochemical sinks for nutrients accumulated during the eutrophication phase. Indeed, the poly- and euhaline lagoons showed concomitant increase of N and P with increasing eutrophication (see **Figure 7**) most likely related to the increase of the organic matter content (average N/P molar ratio of 19.3, which is close to the Redfield ratio). Benthic fluxes of N and P need to be measured (Ouisse et al., 2013) to provide more information on the role of internal loading in delaying oligotrophication. Inertia caused by internal loadings have also been described for the oligotrophication of shallow freshwater lakes, although the focus has been on P alone (Jeppesen et al., 2002), while for the lagoons we show that both N and P play a role.

We can conclude that scientific knowledge and empirical observations are nowadays operational for providing guidance for ecological restoration of the poly- and euhaline Mediterranean coastal lagoons. Nevertheless, insufficient knowledge is still available for the oligo- and mesohaline lagoons. The specific behavior of the aquatic angiosperms forming floating leaves (*S. pectinata* in oligohaline and *Ruppia* sp. in mesohaline lagoons) has to be taken into account. Another source of problem is the fact that for hypertrophic poly- and euhaline lagoons it is still not possible to predict whether a complete oligotrophication trajectory will occur and how long it may take. Many of these formerly hypertrophic lagoons appear to remain stuck in the macroalgal stage, even after 10 years of oligotrophication. This raises the question whether in such cases, the ecological restoration should become an active approach by combining the nutrient reduction with the active planting and seeding of the angiosperms (Orth et al., 2012; De Wit et al., 2017). Currently, such an approach is undertaken in Venice lagoon (Sfriso et al., 2019).

Ecological restoration is a human activity that is clearly identified as a Response in the DPSIR framework (see **Figure 3**). The link between DPSIR and ecological restoration has been invoked in studies focusing on the ecological restoration of rivers (Song and Frostell, 2012; Lalande et al., 2014) although their focus has been more upstream underscoring the importance of identifying drivers and pressures for designing ecological restoration rather than analyzing what is actually targeted by the different practices of ecological restoration. Collectively, these studies together with our study highlight the paramount importance of clearly identifying the pressures on aquatic ecosystems and that ecological restoration should target a reduction of these pressures in the first place. Hence improved sanitation and appropriate management of non-point sources of nutrient in watersheds is paramount for the ecological restoration of eutrophied lagoons. Ecological restoration measures only targeting the state of the ecosystems are not efficient without acting on the pressures.

Surveillance monitoring is a key aspect to follow the state of the ecosystem (S in the DPSIR). The principle "one out all out" was designed for surveillance monitoring of the ecological status of aquatic ecosystems in the WFD, particularly to check if a system is and remains in better than moderate, i.e., good or high conditions. The application of this principle is probably a reason why (i) the perception of local populations is more optimistic than that of WFD, (ii) the assessment systems have been judged by 8% of the interviewed managers as "*overly strict to define success*" (Carvalho et al., 2019), and (iii) it appears as particularly problematic for assessing the impact of specific actions as ecological restoration. For example, it does not identify correctly if variables move in the right direction (the aggregated score only moves until the worst valued has also moved in the positive direction). Thus, the surveillance monitoring that has been used in the frame of the WFD is very conservative and does not give a sufficient indication of minor to major advances in ecological status. Hence, it is important to introduce specific action monitoring for following the impact of ecological restoration that should be designed to detect more responsively (i) whether the ecosystem state moves in the right direction, (ii) identify possible transient states, and (iii) assess whether the different targets for the ecological restoration project have been achieved.

For the conservation and management of coastal lagoons, there is a clear need to study the juxtaposition particularly of the WFD and Habitats Directives, as well as the Birds Directive in some cases. Hence, complementarities and possible incongruences between these Directives should be identified and solutions proposed. For protected areas, the WFD approach has clear advantages with respect to more conventional nature conservation approaches, because it adopts the concept of the aquatic continuum, underscores importance of land-use in the watershed for ecological processes in the lagoon, explicitly considers chemical pollutants and provides a framework that accommodates the concepts of restoration ecology. This approach works well for addressing water quality and the aquatic biota. However, the WFD does not consider the terrestrial habitats as the coastal barrier with beaches and dune systems, the salt marshes and wetlands in its immediate surroundings, as part of the lagoon ecosystem. Moreover, in addition to nutrient reduction targeting oligotrophication, ecological restoration in coastal lagoons includes an array of different actions, e.g., including restoring the freshwater-salt water ecoclines and ecotones through hydrological measures

(Yáñez-Arancibia et al., 2013; Feola et al., 2018), restoration of dune vegetation on the coastal barrier (Buisson et al., 2014) and the restoration of meadows of submerged marine angiosperms (Sfriso et al., 2019). In these cases, the Natura 2000 network offers workable sites comprising different habitats to pursue such management actions and the Life program a funding mechanism for ecological restoration demonstration projects. Hence, the ecological restoration projects that are being developed in coastal lagoon territories have to adapt to the Habitats Directive and for the aquatic compartment also to the WFD. This may represent a challenge as in the Habitats Directive, the assessment of the conservation status of priority habitats is very much rooted in phytosociological approaches, and the possibilities offered by more modern ecological approaches like restoration ecology need to be included more explicitly. While, the WFD promotes a highly integrated vision and is explicitly linked to the Habitats Directive, the differences of approaches and implementation by different public bodies in member states can result in rather sectorial approaches. Hence, WFD is not fully congruent with the Habitats Directive as a consequence of different objectives and concepts. Both have their advantages and inconveniences and an intelligent application of both should be based on integrative non-sectorial application of national legislation and pragmatic management. The active restoration of angiosperm meadows and ecological restoration of aquatic connectivity are examples where co-implementation of both Directives is particularly beneficial. We also recommend the integration of some of the valuable concepts of the WFD into the conservation management of temporal and smaller (i.e., in France <50 ha) lagoons requested by the Habitats Directive.

Knowledge of social representation of the ecosystem and perceptions of management actions are of paramount importance for managers engaged in ecological restoration projects. In fact, public understanding and appreciation lead to more legitimate and accepted public policies such as coastal lagoon ecological restoration projects. In contrast, a low level of congruency will imply a high risk of conflict for implementing such policies. Nevertheless, the results of our studies are encouraging as the ecological restoration practice is highly valued by human populations and stakeholders. Stakeholders have a positive perception of the ecological and societal importance of coastal lagoons and of the need of their ecological restoration, i.e., more than 85% of both residents and tourists close to Montpellier expressed this point of view (De Wit et al., 2017). However, the social representation of the lagoons is more focused on biodiversity aspects than on water quality *sensu stricto* (Audouit et al., 2019). There is consensus among both the highly involved stakeholders (Sy et al., 2018) and among the local residents, who participated in our citizen workshops, on giving highest importance to the regulation and maintenance ecosystem services. While primarily characterized by the consensus on the role of regulating services, stakeholder profile groups' preferences mainly varied based on cultural services and more rarely on provisioning services. Many stakeholders have a sort of collective appreciation of the importance of these regulation and maintenance services for the local surrounding of the lagoons (territorial approach). Other enjoyments as a personal experience provided by contemplating nature, are also important wishes as e.g., the coastal lagoons should contribute to the sentiment of relaxation, the esthetics of species and landscapes, and possibilities for birdwatching. Some groups showing a more naturalist type of profile (citizens) compared to others with a more environmentalist profile (highly involved stakeholders). These studies allow to conclude that coastal lagoons ecosystem services are very important for the stakeholders and that public policies aiming at the ecological restoration of these fragile ecosystems may count on public support.

NOMENCLATURE

Taxonomic nomenclature followed AlgaeBase (Guiry and Guiry, 2016) and World Register of Marine Species (WoRMS Editorial Board, 2016).

DATA AVAILABILITY STATEMENT

This article was based on review of the ecological publications listed in **Table 1** and social science data (publications cited in the text). The ecological monitoring data are available in the French data bases "Banque Hydro" (http://www.hydro.eaufrance.fr/), Naïade (http://www.naiades.eaufrance.fr/acces-donnees), and "Surval," which among others hosts the data of the Réseau Suivi Lagunaire (RSL) monitoring program (Ifremer, https://wwz.ifremer.fr/surval/). Data of the citizens' workshop are available in **Supplementary Material**.

ETHICS STATEMENT

The citizen workshop was based on voluntary participation and the data have been treated anonymously in compliance with the EU General Data Protection Regulation (GDPR).

AUTHOR CONTRIBUTIONS

AL, IL, MS, BB, VO, VD, and RD contributed to the field and laboratory work, and ecological data compilation and interpretation. VD hold responsibilities for the Lagoon Monitoring Network "RSL," and Water Framework Directive (WFD) monitoring programs. MS, HR-V, and RD organized the citizen workshops. MS analyzed the data. BB, VO, HR-V, and RD contributed to supervising the Ph.D. thesis of AL, IL, and MS. RD wrote this publication with the help of all co-authors.

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SUPPLEMENTARY MATERIAL

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REFERENCES

Audouit, Ch, Pasqualini, V., De Wit, R., Flanquart, H., Deboudt, Ph, and Rufin-Soler, C. (2019). Comparing social representation of water quality in coastal lagoons with normative use of ecological indicators. *Mar. Polic.* 101, 137–146. doi: 10.1016/j.marpol.2017.08.023

Bec, B., Collos, Y., Souchu, P., Vaquer, A., Lautier, J., Fiandrino, A., et al. (2011). Distribution of picophytoplankton and nanophytoplankton along an anthropogenic eutrophication gradient in French Mediterranean coastal lagoons. *Aqua. Microb. Ecol.* 63, 29–45. doi: 10.3354/ame01480

Beisner, B. E., Haydon, D. T., and Cuddington, K. (2003). Alternative stable states in ecology. *Front. Ecol. Environ.* 1:376–382.

Buisson, E., Bravet, P., Mieusset, J., Baillot, N., Dubreucq, C., Sadonès, L., et al. (2014). Plant communities of a coastal lagoon foredune: definition of the reference and restoration after compaction. *Acta Bot. Gall.* 161, 277–286. doi: 10.1080/12538078.2014.919875

Carvalho, L., Mackay, E. B., Cardoso, A. C., Baattrup-Pedersen, A., Birk, S., Blackstock, K. L., et al. (2019). Protecting and restoring Europe's waters: an analysis of the future development needs of the water framework directive. *Sci. Total Environ.* 658, 1228–1238. doi: 10.1016/j.scitotenv.2018.12.255

Clewell, A. F., and Aronson, J. (2013). *Ecological Restoration, Principles, Values, and Structure of an Emerging Profession*. 2nd Edn. Washington, DC: Island Press.

Covelli, S. (2012). The MIRACLE Project: an integrated approach to understanding biogeochemical cycling of mercury and its relationship with lagoon clam farming. *Estuar. Coast. Shelf Sci.* 113, 1–6. doi: 10.1016/j.ecss.2012.08.011

Cunha, A. H., Marbá, N., Van Katwijk, M., Pickerell, C., Henriques, M., Bernard, G., et al. (2012). Changing paradigms in seagrass restoration. *Restor. Ecol.* 20, 427–430. doi: 10.1111/j.1526-100x.2012.00878.x

De Groot, R., Blignaut, J., Van der Ploeg, S., Aronson, J., Elmqvist, T., and Farley, J. (2013). Benefits of investing in ecosystem restoration. *Conserv. Biol.* 27, 1286–1293. doi: 10.111/cobi.12158

De Wit, R. (2011). "Biodiversity of coastal lagoon ecosystems and their vulnerability to global change," in *Ecosystems Biodiversity*, eds O. Grillo and G. Venore (London: InTech Open Access Publisher), 29–40.

De Wit, R., Rey-Valette, H., Balavoine, J., Ouisse, V., and Lifran, R. (2017). Ecological restoration of coastal lagoons; prediction of ecological trajectories and economic valuation. *Aqua. Conserv.* 27, 137–157. doi: 10.1002/aqc.2601

De Wit, R., Stal, L. J., Lomstein, B. A., Herbert, R. A., Van Gemerden, H., Viaroli, P., et al. (2001). ROBUST: The ROle of BUffering capacities in STabilising coastal lagoon ecosystems. *Cont. Shelf Res.* 21, 2021–2041.

De Wit, R., Vincent, A., Foulc, L., Klesczewski, M., Scher, O., Loste, C., et al. (2019). Seventy-year chronology of Salinas in southern France: coastal surfaces managed for salt production and conservation issues for abandoned sites. *J. Nat. Conserv.* 49, 95–107. doi: 10.1016/j.jnc.2019.03.003

Derolez, V., Bec, B., Munaron, D., Fiandrino, A., Pete, R., Simier, M., et al. (2019). Recovery trajectories following the reduction of urban nutrient inputs along the eutrophication gradient in French Mediterranean lagoons. *Ocean Coast. Manag.* 171, 1–10. doi: 10.1016/j.ocecoaman.2019.01.012

Dolbeth, M., Stålnacke, P., Alves, F. L., Sousa, L. P., Gooch, G. D., Khokhlov, V., et al. (2016). An integrated Pan-European perspective on coastal Lagoons management through a mosaic-DPSIR approach. *Sci. Rep.* 6:19400. doi: 10.1038/srep19400

EU Birds Directive (2009). *Directive 2009/147/EC of the European Parliament and of the Council on the conservation of wild birds (the first version of this Directive was adopted in 1979 as Directive 79/409/EEC that was amended in 2009)*. Brussels: EU.

EU Habitats Directive (1992). *Directive 92/43/EEC of the European Parliament and of the Council on the Conservation of Natural Habitats and of Wild Fauna and Flora*. Available online at: http://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm (accessed February 26, 2020).

EU Water Framework Directive (2000). *Directive 2000/60/EC of the European Parliament and of the Council Establishing a Framework for the Community Action in the Field of Water Policy*. Available online at: http://ec.europa.eu/environment/water/water-framework/ (accessed February 26, 2020).

European Commission DG Environment (2013). *The Interpretation Manual of European Union Habitats - EUR28*. Available online at: http://ec.europa.eu/environment/nature/legislation/habitatsdirective/docs/Int_Manual_EU28.pdf (accessed February 26, 2020).

European Environment Agency (1999). *Technical Report No 25, Environmental Indicators: Typology and Overview*. Copenhague: European Environment Agency.

Feola, A., Bonometto, A., Ponis, E., Cacciatore, F., Oselladore, F., Matticchio, B., et al. (2018). *LIFE LAGOON REFRESH. Ecological restoration in Venice Lagoon (Italy): Concrete Actions Supported by Numerical Modeling and Stakeholder Involvement*. Available online at: http://www.lifelagoonrefresh.eu/file/pubblicazioni/COWM_2018_Extendedabstract.pdf (accessed February 26, 2020).

García-Nieto, A. P., Quintas-Soriano, C., García-Llorente, M., Paloma, I., Montes, C., and Martín-López, B. (2015). Collaborative mapping of ecosystem services : the role of stakeholders' profiles. *Ecosyst. Serv.* 13, 141–152. doi: 10.1016/j.ecoser.2014.11.006

Guiry, M. D., and Guiry, G. M. (2016). *AlgaeBase. World-wide electronic publication*. Galway: National University of Ireland.

Gulati, R. D., and Van Donk, E. (2002). Lakes in the Netherlands, their origin, eutrophication and restoration: state-of-the-art review. *Hydrobiologia* 478, 73–106. doi: 10.1007/978-94-017-1335-1_5

Haines-Young, R., and Potschin, M. B. (2018). *Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure*. Brussel: European Environment Agency.

IFREMER (2014). *Réseau de Suivi Lagunaire du Languedoc-Roussillon. Bilan des résultats 2013*. Issy-les-Moulineaux: IFREMER.

Jeppesen, E., Jensen, J. P., and Søndergaard, M. (2002). Response of phytoplankton, zooplankton, and fish to re-oligotrophication: an 11 year study of 23 Danish lakes 31-43. *Ecosyst. Health Manag.* 5, 31–43. doi: 10.1080/14634980260199945

Jeppesen, E., Søndergaard, M., Jensen, J. P., Havens, K. E., Anneville, O., Carvalho, L., et al. (2005). Lake responses to reduced nutrient loading—an analysis of contemporary long-term data from 35 case studies. *Freshw. Biol.* 50, 1747–1771. doi: 10.1111/j.1365-2427.2005.01415.x

Kjerfve, B. (1994). "Coastal lagoons," in *Coastal Lagoon Processes*, ed. B. Kjerfve (Amsterdam: Elsevier Oceanographic Series), 1–8. doi: 10.1016/s0422-9894(08)70006-0

Lalande, N., Cernesson, F., Decherf, A., and Tournoud, M.-G. (2014). Implementing the DPSIR framework to link water quality of rivers to land use: methodological issues and preliminary field test. *Int. J. River Basin Manag.* 12, 201–217.

Larue, J. P., and Rouquet, J. (2016). La lagune de La Palme (Aude, France) face au comblement et à l'eutrophisation. *Physiol. Géol.* 10, 45–60. doi: 10.4000/physio-geo.4761

Le Fur, I. (2018). *Rôle Des Macrophytes Dans la Restauration Des Milieux Lagunaires : Successions Écologiques*. Montpellier: Université de Montpellier, 210.

Le Fur, I., De Wit, R., Plus, M., Oheix, J., Derolez, V., Simier, M., et al. (2019). Re-oligotrophication trajectories of macrophyte assemblages in Mediterranean coastal lagoons based on 17-year time-series. *Mar. Ecol. Progr. Ser.* 608, 13–32. doi: 10.3354/meps12814

Le Fur, I., De Wit, R., Plus, M., Oheix, J., Simier, M., and Ouisse, V. (2018). Submerged benthic macrophytes in Mediterranean lagoons: distribution patterns in relation to water chemistry and depth. *Hydrobiologia* 808, 175–200. doi: 10.1007/s10750-017-3421-y

Leruste, A. (2016). *Trajectoires D'évolution des Communautés Phytoplanctoniques au Cours du Processus de Restauration Écologique des Milieux Lagunaires Méditerranéens*. Montpellier: Université de Montpellier, 314.

Leruste, A., Guilhaumon, F., De Wit, R., Malet, N., Collos, Y., and Bec, B. (2019a). Phytoplankton strategies to exploit nutrients in coastal lagoons with different eutrophication status and involved in a re-oligotrophication process. *Aqua. Microb. Ecol.* 83, 131–146. doi: 10.3354/ame01906

Leruste, A., Pasqualini, V., Garrido, M., Malet, N., De Wit, R., and Bec, B. (2019b). Physiological and behavioral responses of phytoplankton communities to nutrient availability in a disturbed Mediterranean coastal lagoon. *Estuar. Coast. Shelf Sci.* 219, 176–188. doi: 10.1016/j.ecss.2019.02.014

Leruste, A., Malet, N., Munaron, D., Derolez, V., Hatey, E., Collos, Y., et al. (2016). First steps of ecological restoration in Mediterranean lagoons: shifts in phytoplankton communities. *Estuar. Coast. Shelf Sci.* 180, 190–203. doi: 10.1016/j.ecss.2016.06.029

Li, W. K. W. (1995). Composition of ultraphytoplankton in the central North Atlantic. *Mar. Ecol. Progr. Ser.* 122, 1–8. doi: 10.3354/meps122001

Life programme (2020). Available online at: https://ec.europa.eu/easme/en/life (accessed February 28, 2020).

Liquete, C., Piroddi, C., Drakou, E. G., Gurney, L., Katsanevakis, S., Charef, A., et al. (2013). Current status and future prospects for the assessment of marine and coastal ecosystem services: a systematic review. *PLoS One* 8:e67737. doi: 10.1371/journal.pone.0067737

Marsden, M. (1989). Lake restoration by reducing external phosphorus loading: the influence of sediment phosphorus release. *Freshw. Biol.* 21, 139–162.

Maxwell, P. S., Eklof, J. S., Van Katwijk, M. M., O'brien, K. R., De La Torre-Castro, M., Bostrom, C., et al. (2017). The fundamental role of ecological feedback mechanisms for the adaptive management of seagrass ecosystems - a review. *Biol. Rev.* 92, 1521–1538. doi: 10.1111/brv.12294

Meinesz, C., Derolez, V., and Bouchoucha, M. (2013). *Base de Données "Pressions Sur Les Lagunes Méditérranéennes" - Analyse des liens état - pression. Report in French*. Montpellier: Agence de l'Eau Rhône-Méditeranée-Corse.

Munaron, D., Tapie, N., Budzinski, H., Andral, B., and Gonzalez, J. L. (2012). Pharmaceuticals, alkylphenols and pesticides in Mediterranean coastal waters: results from a pilot survey using passive samplers. *Estuar. Coast. Shelf Sci.* 114, 82–92. doi: 10.1016/j.ecss.2011.09.009

Newton, A., Brito, A. C., Icely, J. D., Derolez, V., Clara, I., Angus, S., et al. (2018). Assessing, quantifying and valuing the ecosystem services of coastal lagoons. *J. Nat. Conserv.* 44, 50–65. doi: 10.1016/j.jnc.2018.02.009

Nixon, S. W. (2009). Eutrophication and the macroscope. *Hydrobiologia* 629, 5–19. doi: 10.1007/978-90-481-3385-7_2

Orth, R. J., McGlathery, K. J., and Heck, K. L. Jr. (eds) (2012). Theme section: eelgrass recovery induces state changes in a coastal bay system. *Mar. Ecol. Progr. Ser.* 434, 171–301.

Ouisse, V., Fiandrino, A., De Wit, R., and Malet, N. (2013). *Restauration des écosystèmes lagunaires : évaluation du rôle du sédiment et des herbiers à phanérogames. Rapport Final du Contrat n 2012 1835*. Report in French. France: Agence de l'Eau Rhône-Méditeranée-Corse.

Pasqualini, V., Derolez, V., Garrido, M., Orsoni, V., Baldi, Y., Etourneau, S., et al. (2017). Spatiotemporal dynamics of submerged macrophyte status and watershed exploitation in a Mediterranean coastal lagoon: understanding critical factors in ecosystem degradation and restoration. *Ecol. Eng.* 102, 1–14. doi: 10.1016/j.ecoleng.2017.01.027

Petraitis, P. S., and Hoffman, C. (2010). Multiple stable states and relationship between thresholds in processes and states. *Mar. Ecol. Progr. Ser.* 413, 189–200. doi: 10.3354/meps08691

Rouillard, J. J., Lago, M., Roeschel, L., Abhold, K., Kafyeke, T., Klimmek, H., et al. (2017). Protecting and restoring aquatic biodiversity: is the existing EU policy framework fit for purpose? *Environ. Policy Gov.* 28, 114–128. doi: 10.1002/eet.1793

Scheffer, M. (2001). Alternative attractors of shallow lakes. *Sci. World* 1, 254–263. doi: 10.1100/tsw.2001.62

Scheffer, M., Hosper, S. H., Meijer, M.-L., Moss, B., and Jeppesen, E. (1993). Alternative equilibria in shallow lakes. *Trends Ecol. Evol.* 8, 275–279. doi: 10.1016/0169-5347(93)90254-m

Schramm, W. (1999). Factors influencing seaweed responses to eutrophication: some results from EU-project EUMAC. *J. Appl. Phycol.* 11, 69–78.

Sfriso, A., Buosi, A., Tomio, Y., Juhmani, A.-S., Facca, C., Sfriso, A. A., et al. (2019). Aquatic angiosperm transplantation: a tool for environmental management and restoring in transitional water systems. *Water* 11:2135. doi: 10.3390/w11102135

Society for Ecological Restoration International Science and Policy Working Group (2004). *The SER International Primer on Ecological Restoration*. Available online at: https://www.ctahr.hawaii.edu/littonc/PDFs/682_SERPrimer.pdf (accessed July 11, 2019).

Søndergaard, M., Jensen, J. P., and Jeppesen, E. (2003). Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia* 50, 135–145. doi: 10.1023/b:hydr.0000008611.12704.dd

Song, X., and Frostell, B. (2012). The DPSIR framework and a pressure-oriented water quality monitoring approach to ecological river restoration. *Water* 4, 670–682. doi: 10.3390/w4030670

Souchu, P., Bec, B., Smith, V. H., Laugier, T., Fiandrino, A., Benau, L., et al. (2010). Patterns in nutrient limitation and chlorophyll a along an anthropogenic eutrophication gradient in French Mediterranean coastal lagoons. *Can. J. Fish. Aqua. Sci.* 67, 743–753. doi: 10.1139/F10-018

Sy, M. M. (2019). *Evaluation Des Services Écosystémiques Fournis Par les Complexes Lagunaires Dans un Processus de Restauration Écologique*. Montpellier: Université de Montpellier, 175.

Sy, M. M., Rey-Valette, H., Simier, M., Pasqualini, V., Figuières, Ch, and De Wit, R. (2018). Identifying consensus on coastal lagoons ecosystem services and conservation priorities for an effective decision making: A Q approach. *Ecol. Econ.* 154, 1–13. doi: 10.1016/j.ecolecon.2018.07.018

Valiela, I., McLelland, J., Hauxwell, J., Behr, P. J., Hersh, D., and Foreman, K. (1997). Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. *Limnol. Oceanogr.* 42, 1105–1118. doi: 10.4319/lo.1997.42.5_part_2.1105

Van der Geest, M., Van der Heide, T., Holmer, M., and De Wit, R. (2020). First field-based evidence that the seagrass-lucinid mutualism can mitigate sulfide stress in seagrasses. *Front. Mar. Sci.* 7:11. doi: 10.3389/fmars.2020.00011

Van Katwijk, M. M., Bos, A. R., De Jonge, V. N., Hanssen, L. S. A., Hermus, D. C. R., and De Jong, D. J. (2009). Guidelines for seagrass restoration: importance of habitat selection and donor population, spreading of risks, and ecosystem engineering effects. *Mar. Pollut. Bull.* 58, 179–188. doi: 10.1016/j.marpolbul.2008.09.028

Van Katwijk, M. M., Thorhaug, A., Marba, N., Orth, R. J., Duarte, C. M., Kendrick, G. A., et al. (2016). Global analysis of seagrass restoration: the importance of large-scale planting. *J. Appl. Ecol.* 53, 567–578. doi: 10.1111/1365-2664.12562

Venice System. (1958). The Venice system for the classification of marine waters according to salinity. *Limnol. Oceanogr.* 3, 346–347. doi: 10.4319/lo.1958.3.3.0346

Viaroli, P., Bartoli, M., Giordano, G., Naldi, M., Orfanidis, S., and Zaldivar, J. M. (2008). Community shifts, alternative stable states, biogeochemical controls and feedbacks in eutrophic coastal lagoons: a brief overview. *Aqua. Conserv.* 18, 105–117. doi: 10.1002/aqc.956

Voulvoulis, N., Arpon, K. D., and Giakoumis, T. (2017). The EU Water Framework Directive: from great expectations to problems with implementation. *Sci. Total Environ.* 575, 358–366. doi: 10.1016/j.scitotenv.2016.09.228

WoRMS Editorial Board (2016). *World Register of Marine Species*. doi: 10.14284/170 Available online at: http://www.marinespecies.org at VLIZ (accessed July 23, 2019).

Yáñez-Arancibia, A., Day, J. W., Sánchez-Gil, P., Day, J. N., Lane, R. R., Zárate-Lomelí, D., et al. (2013). Ecosystem functioning: the basis for restoration and management of a tropical coastal lagoon, Pacific coast of Mexico. *Ecol. Eng.* 65, 88–100. doi: 10.1016/j.ecoleng.2013.03.007

Zaldívar, J.-M., Cardoso, A. C., Viaroli, P., Newton, A., De Wit, R., Ibañez, C., et al. (2008). Eutrophication in transitional waters: an overview. *Trans. Waters Monogr.* 2, 1–78.

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Anthropogenic, Direct Pressures on Coastal Wetlands

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Coastal wetlands, such as saltmarshes and mangroves that fringe transitional waters, deliver important ecosystem services that support human development. Coastal wetlands are complex social-ecological systems that occur at all latitudes, from polar regions to the tropics. This overview covers wetlands in five continents. The wetlands are of varying size, catchment size, human population and stages of economic development. Economic sectors and activities in and around the coastal wetlands and their catchments exert multiple, direct pressures. These pressures affect the state of the wetland environment, ecology and valuable ecosystem services. All the coastal wetlands were found to be affected in some ways, irrespective of the conservation status. The main economic sectors were agriculture, animal rearing including aquaculture, fisheries, tourism, urbanization, shipping, industrial development and mining. Specific human activities include land reclamation, damming, draining and water extraction, construction of ponds for aquaculture and salt extraction, construction of ports and marinas, dredging, discharge of effluents from urban and industrial areas and logging, in the case of mangroves, subsistence hunting and oil and gas extraction. The main pressures were loss of wetland habitat, changes in connectivity affecting hydrology and sedimentology, as well as contamination and pollution. These pressures lead to changes in environmental state, such as erosion, subsidence and hypoxia that threaten the sustainability of the wetlands. There are also changes in the state of the ecology, such as loss of saltmarsh plants and seagrasses, and mangrove trees, in tropical wetlands. Changes in the structure and function of the wetland ecosystems affect ecosystem services that are often underestimated. The loss of ecosystem services impacts human welfare as well as the regulation of climate change by coastal wetlands. These cumulative impacts and multi-stressors are further aggravated by indirect pressures, such as sea-level rise.

Keywords: coastal wetland, salt marsh, mangrove, seagrass, pressure, state and impact on human welfare, sustainability, climate change

INTRODUCTION

Transitional waters are naturally fringed by many types of coastal wetlands, such as tidal flats, seagrass meadows, saltmarshes and mangroves, that are an integral part of the transitional ecosystems (Perillo et al., 2019). The coastal wetlands covered in this article fringe river mouth systems, estuaries, deltas and other transitional waters, such as coastal lagoons and shallow, semi-enclosed, coastal systems, such as bays (Newton et al., 2013). They include a variety of habitats: mudflats, seagrass meadows, saltmarshes, and mangrove swamps. The article covers wetlands from the tropics, temperate zones and polar regions as examples to illustrate the main direct pressures (**Figure 1** and **Table 1**). The wetlands have been chosen to give a broad range and variety, including very large wetlands (e.g., the Mekong Delta) and much smaller ones (e.g., Malanza).

The range of population and human pressures also varies from sparsely populated, (e.g., Mackenzie-Beaufort), to very large populations, (e.g., Yangtze-Changjiang). Nevertheless, the resident population numbers are not always the best indicator of human pressure, for example, the Venice municipality has a low resident population (about 85,000), but a very large number of visitors (20 million y^{-1}).

The examples chosen include some very well-known systems, such as the Chesapeake Bay, but an effort has been made to also include some less well-known systems, such as the mangroves of Gulf of Papua and Malanza. Many other systems could have been chosen, but there were limitations to the length of the article, so information on some additional examples is included only in **Supplementary Tables S1–S5**. There is also a recent review of coastal wetlands in North Africa by El Mahrad et al. (2020).

The human pressures on wetlands are both direct and indirect. An example of a direct pressure is the loss of connectivity and interruptions to sediment supply as a result of constructing a dam. Indirect, human pressures are mostly related to climate change and these include rising temperature, acidification, changes in precipitation/runoff, as well as sea level rise. The aim of this article is to give an overview of the multiple, anthropogenic, direct pressures on coastal wetlands resulting from human activities, the effect that these pressures have on the state of the wetland and ecosystem services, as well as the eventual impacts on human welfare.

There are many possible frameworks for analyzing Social-Ecological Systems such as coastal wetlands (Binder et al., 2013). The present analysis follows a 'Activity-Pressure-State-Impact on Human Welfare' approach, modified from the 'Driver Pressure State Impact Response' framework (DPSIR), (Gari et al., 2015; Patrício et al., 2016; Elliott et al., 2017). This approach has been selected because it is widely used in international studies by international organizations, (e.g., the Organization for Economic Cooperation and Development, the European Environment Agency, the United Nations Environment Programme, and the United Nations World Ocean Assessment). It was also recently applied to eleven coastal wetlands in North Africa, El Mahrad et al. (2020). The focus is on direct pressures that result from human activities (Elliott and Whitfield, 2011) and follows the typology proposed by Elliott et al. (2014).

The article focuses on Activities-Pressures-State-Impact and these are bolded in the text rather than sub-dividing the sections. The article also considers the effect of the change in state of the environment and the wetland ecosystem on the delivery of ecosystem services (ES). Ecosystem services provide an important link between the state of the natural system and human welfare. The are the answer to the questions '*so what*'? '*why is this important*'? In this review, we used the Millennium Ecosystem Assessment [MEA] (2005) classification of ecosystem services. There are more recent classifications of ecosystem services but the Millennium Ecosystem Assessment [MEA] (2005) is still widely used by international organizations, and there remain challenges to the valuation of ecosystem services in coastal wetlands (Newton et al., 2018).

At the end of each section, we have also included some brief comments about indirect pressures. Indirect pressures, such as climate induced sea-level rise, are uncontestably important to coastal wetlands. However, they are not the focus of the current article and thus not discussed at length. They are especially important in the wetlands where local population and the direct pressures are low, such as the Mackenzie-Beaufort polar wetland and the Gulf of Papua.

HUMAN ACTIVITIES AND ANTHROPOGENIC PRESSURES ON COASTAL WETLANDS

Polar Coastal Wetlands

Arctic and Antarctic coasts have low human population densities. The development, structure, and land use of northern Polar coastal wetlands has been reviewed by Martini et al. (2019). The inhabitants still carry out some traditional activities but there is an increase in economic activities. A large proportion of small settlements on the polar coasts of Alaska, northern Canada, Greenland and parts of Russia have mixed cash and

FIGURE 1 | Map showing location of the coastal wetlands in the text. (Credit: Dylan Taille).

subsistence economies, in contrast to northern Norway, Iceland, or large urban centers such as Murmansk (Jungsberg et al., 2019). Regions where economic activities are more developed have more extensive industrial infrastructure, such as the Prudhoe Bay oilfield on the Alaskan North Slope (Walker and Peirce, 2015). Past activities, such as dumping of mine tailings or encasement of drilling muds in permafrost, may still exert pressures on coastal wetlands, decades after the activity has stopped (Thienpont et al., 2013).

The low populations result in relatively low or localized anthropogenic, direct pressures on coastal wetlands. The loss of sea ice and resulting rapid increase in navigation along Arctic coasts will increase pressures from shipping, tourism, commercial fishing, hydrocarbon exploration and extraction, and mining activities. Economic investment from globalization has stimulated human activities and pressures that are increasing environmental state changes, with progressive reduction of sea ice, degradation of permafrost (Forbes, 2011). Many polar coastal wetlands exhibit relatively rapid state changes attributed to global and regional climate warming (Forbes and Hansom, 2011).

Where traditional activities remain important, e.g., subsistence harvesting, loss of ecosystem services may translate to impacts on welfare, food insecurity and affect cultural integrity, health, and wellbeing.

The negative effects may be magnified in the future because of indirect pressures that cause changes in key environmental variables, e.g., temperature, pH, and redox potential (Kokelj et al., 2010; Barkay et al., 2011; Grosse et al., 2016).

Mackenzie-Beaufort Coastal Wetland, Canada

The Mackenzie Delta covers an area of 13,000 km^2 (~60 km × ~200 km) in the southeastern Beaufort Sea, on the western Arctic coast of Canada (Forbes, 2019). The Arctic tree-line bisects the delta, the southern part of which is forested. Slightly over 50% of the delta plain is occupied by sedge wetlands, channels, and over 49,000 lakes (Emmerton et al., 2007). Permafrost extends to about 60 m depth in Holocene delta sediments, with many thaw taliks beneath lakes or channels that do not freeze completely in winter (Todd and Dallimore, 1998). There is higher ground to the east and west of the outer delta, with permafrost extending to a depth of 500–700 m. Erosional breaching of thaw lake basins has created a highly indented coast with eroding bluff headlands, local dunes, low spits and barrier islands, and shallow estuaries with supratidal *Puccinellia phryganodes* flats and accumulations of driftwood from the Mackenzie River (Ruz et al., 1992; Forbes and Hansom, 2011; Forbes et al., 2014). The delta is micro-tidal (range <0.5 m), but storm surges of up to 2.5 m cause extensive flooding (Manson and Solomon, 2007; Lamoureux et al., 2015).

The human population in the region is small (Kokelj et al., 2012), consisting of the regional administrative, industrial, and transportation hub of Inuvik (2015 pop. 3265, 68% indigenous), the smaller delta settlement of Aklavik (2015 pop. 668, 94% indigenous), and the outer-coast community of Tuktoyaktuk (2015 pop. 965, 89% indigenous).

Some human activities are seasonal and intermittent, such as the extensive occupation of hunting and fishing camps throughout the delta and at seasonal traditional settlements along the outer coast (Lamoureux et al., 2015). Human activities of the Inuvialuit (western Arctic Inuit) and Gwich'in people include subsistence hunting and fishing as well as administration and public services. Land and resource uses are co-managed with federal and territorial government agencies under the Inuvialuit Final Agreement 1984, the Gwich'in Land Claim Settlement Act 1992, and fisheries and game co-management bodies. Subsistence food harvesting and hunting activities in the Inuvialuit Settlement Region includes berries, medicinal plants, fish including cisco (*Coregonus* spp.) and char (*Salvelinus* spp.), birds (especially geese and ducks), and mammals such as polar bear, caribou, moose, muskrat, beluga whale, bearded and ringed seal (Stephenson, 2004).

TABLE 1 | List of coastal wetlands used as examples in this article, with some of the descriptors.

Name of coastal wetland	Adjacent sea/Ocean	Continent and country (ies)	Latitudinal	Geomorphological type	Habitats	Protection/conservation	Area km^2	Human population
Bahia Blanca	South West Atlantic	South America, Argentina	Temperate	Coastal plain estuary	Tidalflats, Saltmarsh	Provincial and municipal reserves, [1]WHSB Reserve	3,000	440,000
Chesapeake Bay	Atlantic Ocean	North America	Temperate	Estuary	Saltmarsh, Tidal wetlands	[2]CBP, Alliance for the Bay, CBF, River keepers	1,141	17,700,000
Bizerte Lagoon	Mediterranean Sea	North Africa, Tunisia	Semi-arid climate	Coastal lagoon	Seagrass meadow Micro-marshes Sandy coasts Mudflats		128	568,000 (all around lagoon)
Ichkeul Lake	Mediterranean Sea	North Africa, Tunisia	Semi-arid climate	Coastal lake	Marshes Reed beds	Biosphere reserve, [3]UNESCO World Heritage Site, [4]Ramsar Site, National Park	120	350 (in biosphere reserve)
Mekong Delta wetlands	South China Sea, Vietnamese East Sea	Asia, Vietnam	Subtropical	Coastal alluvial river delta	Saltmarsh, Mangroves, Mudflats, Grassy wetlands, Controlled rice paddies	Several nature reserves	40,600	18,000,000
Mississippi delta	North West Atlantic, Gulf of Mexico	North America, United States	Temperate	Delta and deltaic plain	Tidal flats, Fresh to Saltmarsh	State of Louisiana	17,417	2,000,000
Mackenzie-Beaufort	Arctic Ocean	North America, Canada	Low-Arctic (68–70°N)	Delta, Coastal embayments	Delta plain, Supratidal marsh, Inundated tundra, Estuary	Migratory bird sanctuary (federal), co-management regime	13,000	4,900
Malanza coastal lagoon	South East Atlantic	Africa, São Tomé and Principe	Tropical	Coastal lagoon	Mangroves Mudflats	Obô Natural park	0.7	1,000
Gulf of Papua mangroves	Coral Sea, Pacific	Australasia, Papua New Guinea	Tropical	Delta complex	Mangroves		3,780	107,000
Ria Formosa	North East Atlantic	Europe, (SW), Portugal	Temperate	Coastal lagoon	Seagrass, Saltmarsh, Dunes, Mudflats	Ramsar Site, Natura (EU), National Park	80	135,500 (3main towns)
Sundarbans mangroves	Indian Ocean, Bay of Bengal	Asia (S) Bangladesh, India	Sub-Tropical	Delta	Mangroves	Ramsar Site, UNESCO World Heritage Site	10,000	3,500,000
Vembanad Lake	Arabian Sea, Indian Ocean	Asia (S), India (W)	Tropical	Coastal lagoon	Mangroves, Marsh, Backwater, Estuary	Ramsar Site, Bird sanctuary under [5]WLPA	1,521	1,676,560
Venice	Adriatic, Mediterranean Sea	Europe (S), Italy	Temperate	Coastal lagoon	Seagrass, Saltmarsh, Subtidal and intertidal Mudflats	Ramsar Site (Valle Averto)	550	85,000 (Venice municipality only)
Watamu, Mida Creek	Indian Ocean	Africa (E), Kenya	Tropical	Coastal creek	Mangroves, Mudflats, Coral reef	UNESCO Biosphere reserve; National Reserve	32	57,200 (3main towns)
Yangtze (Changjiang) Delta	East China Sea	Asia (E), China (E)	Temperate	Delta	Saltmarsh, Mudflats	Ramsar Site, 5 nature reserves	350	1,000,000,000
Yellow River (Huang He) Delta	Bohai Sea	Asia (E), China (NE)	Temperate	Delta	Saltmarsh, Mudflats, Seagrass, Dunes	Ramsar Site, Shandong Province reserve, National Reserve	18,000	5,200,000

[1]WHSBR, Western Hemisphere Shore Bird Reserve; [2]CBP, Chesapeake Bay Programme; [3]UNESCO, United Nations Educational, Scientific and Cultural Organization; [4]Ramsar, Convention on Wetlands; [5]WPLA, Wildlife Protection Area.

Human activities related to economic development in the wetland region include infrastructure development, such as construction of Distant Early Warning (DEW) line sites in the 1950s; establishment of the planned community of Inuvik in 1953–1960 to replace flood-prone Aklavik; roads, airports, and port facilities; fuel storage, electric generation and distribution infrastructure; freshwater and waste management systems; hydrocarbon exploration and production installations (well sites, mud sumps, other waste dumps, and camps); seismic cut-lines and degraded artificial islands on the inner shelf (Lamoureux et al., 2015). Road access to Aklavik (and until 2017 to Tuktoyaktuk) has been by winter ice road on delta channels, but the ice season is shortened by climate warming (Lesack et al., 2014; Lamoureux et al., 2015). Tuktoyaktuk is a regional shipping hub for remote community sea-lift operations and offshore hydrocarbon exploration activities. Artificial shore protection at Tuktoyaktuk has been partially successful at limiting erosion. There has been dredging in harbour approaches and to build artificial islands for drilling. Future activities such as infrastructure development, new travel routes, and growing tourism opportunities may impact cultural sites (Irrgang et al., 2019).

Direct pressures related to activities such as oil and gas exploration have ended with increased supply of low-cost natural gas. Oil exploration on the shelf and offshore supply activities in Tuktoyaktuk have also been suspended (Byers, 2016). These past activities still exert pressures on the environment. Contaminated hydrocarbon exploration waste, such as drilling fluids, was placed in sumps to be contained in permafrost. However, about 50% of legacy drilling mud and camp sumps in the delta have collapsed due to warmer temperatures (Kokelj and GeoNorth Limited, 2002).

These pressures affect the state of the environment. Dredging for artificial island construction in the past disturbed the state of the benthic habitat, but ubiquitous and perennial seabed scouring by ice causes more disturbance (Blasco et al., 2013). The changes in the state of the environment result in a loss of ecosystem services. The Mackenzie Delta and Kendall Island Bird Sanctuary are of international importance for the large numbers of migratory birds – particularly shorebirds, cranes, swans, geese, and ducks – that depend on the extensive staging and breeding habitat. The wetlands also provide critical habitat for fish and other wildlife. Regional (glacial-isostatic) and local (compaction) subsidence combined with climate-driven rising sea level causes the gradual inundation of outer-delta avian breeding habitat and low-lying tundra along the coast (Lamoureux et al., 2015; Forbes, 2019), and accelerating erosional retreat of the delta front and adjacent coast, increasing carbon inputs to the Arctic Ocean (Fritz et al., 2017; Couture et al., 2018; Tanski et al., 2019). The Tarium Niryutait Marine Protected Area was established in 2010 in three areas totaling 1,800 km^2 of the outer Mackenzie Delta and estuary. One of the objectives was the conservation of one of the largest summering stocks of beluga whale (*Delphinapterus leucas*), anadromous fish, waterfowl, and their habitat. This preserves provisioning and cultural services as well as indigenous harvesting traditions[1].

Loss of ecosystem services and localized contamination have negatively impacted the welfare of subsistence-dependent residents, with implications for community and household health and food security (Wesche and Chan, 2010). The leaching of contaminants, especially hydrocarbons, into the surrounding soil and groundwater has caused localized contamination of the wetlands, raising health concerns due to extensive fish consumption and impacts on human welfare.

Indirect pressures resulting from global emissions and climate change have much greater consequences in the Mackenzie-Beaufort region than direct pressures. These indirect pressures include: accelerated sea-level rise; reduced sea ice, increased open water in the Beaufort Sea; increased wave energy with expanded open-water fetch; increased frequency and reach of storm surges; and accelerated coastal erosion; reduced snowfall; earlier and more rapid ice breakup in the Mackenzie delta; lower peak water levels and reduced off-channel flooding in the delta; rising temperatures contributing to thaw subsidence; accelerated coastal erosion, and ecological change (Manson and Solomon, 2007; Barber et al., 2008; Burn and Kokelj, 2009; Lesack et al., 2013; Vermaire et al., 2013; Obu et al., 2016; Forbes, 2019).

Coastal Wetlands in North America (Excluding Arctic)

Chesapeake Bay Wetlands, United States

The Chesapeake Bay is a 320 km long, shallow, north-south estuary surrounded by the states of Maryland and Virginia on the east coast of the United States. Major tributaries supply freshwater to the bay, including the Susquehanna that supplies more than 50% of the freshwater, the Patuxent, Potomac, York, and James rivers. There were approximately 114,100 ha of tidal wetlands, consisting of saltmarsh and tidal fresh and saltwater wetlands in 2010, almost 37,200 ha of Submerged Aquatic Vegetation (SAV) in 2012, primarily eelgrass (*Zostera marina*) and widgeon grass (*Ruppia maritima*) (Chesapeake Bay Program [CBP], 2012).

The watershed is large, approximately 165,760 km^2, including both heavily urbanized areas in and around the major cities, as well as large portions of distinctly rural, agricultural lands. It includes almost all the state of Maryland, and parts of Virginia, West Virginia, Delaware, Pennsylvania, and New York. Major cities include Washington DC, Baltimore, Richmond, Norfolk, and Harrisburg. The total watershed population is approximately 17.7 million people, (2012 census), expected to grow to 20 million by 2030. The increased spread of urban areas is expected to continue with population growth.

Human activities in the watershed that exert pressures on the Chesapeake wetlands include agriculture, land-use changes, shoreline modification and introduction of non-indigenous species. The watershed includes many thriving economic sectors such as sales, services, construction, manufacturing, government, agriculture and use of natural resources (McKendry, 2009). Activities such as hardening of shorelines are now strictly

[1] https://laws-lois.justice.gc.ca/PDF/SOR-2010-190.pdf

regulated. Nevertheless, existing shoreline modification and intense development along the shoreline can lead to wetland loss or prevent new wetlands from forming, blocking the migration of wetlands to upland areas as sea levels rise (Torio and Chmura, 2013; Raposa et al., 2016), a synergy of direct and indirect pressures.

A complex mixture of direct pressures result from intensive land-use and the introduction of invasive species. Nutria (*Myocastor coypus*, a South American rodent) was introduced in the late 1960s and displaced native species such as beaver, muskrat and otter. Nutria thrived on native marsh grasses, damaging thousands of hectares of wetland through destructive feeding (National Fish and Wildlife Service, 2016). In addition, the invasive common reed, *Phragmites australis*, grows rapidly to outcompete other wetland vegetation, such as native grasses (Chesapeake Bay Program [CBP], 2012). However, the main direct pressures to the overall condition of Chesapeake wetlands are now nutrient and sediments from intensive land-use, urban wastewater and atmospheric deposition of nitrogen (Zhang et al., 2015).

Pressures from intense land use activities directly and indirectly affects the state of Chesapeake wetlands. Excess nutrients result in areas of low dissolved oxygen and reduced water clarity (Kemp et al., 2005). This has affected the state of the submerged aquatic vegetation (SAV), and habitat for commercial species, including oysters (*Crassostrea virginica*), crabs (*Callinectes sapidus*), and bass (*Morone saxatilis*). Fishing pressure has also contributed to the decline in the state of these important species and may mask the signal from eutrophication effect (Kemp et al., 2005). Sediment inputs affect the benthic state, cover hard-bottom habitat, smother oyster reefs, increase turbidity, reducing light available for SAV (Lefcheck et al., 2017). Lefcheck et al. (2017) document a change of state, a 29% decline in eelgrass area from 1991 to 2016, primarily from decreasing water clarity increases and water temperature. Despite this degradation of the state, management measures to reduce nutrient pressures can increase SAV growth and area (Lefcheck et al., 2018). This is a positive sign that the state of ecosystems may recover after humans alleviate pressures put on sensitive habitats like SAV (Lefcheck et al., 2018).

Loss of ecosystem services include a decrease of Chesapeake wetland habitat for economically valuable species of fish and crabs, both of which are cultural keystones in the Chesapeake region (Paolisso, 2008). There is also a loss of storm surge mitigation, filtration of sediments and nutrients from urban and agricultural stormwater runoff, sequestration of organic carbon and opportunities for recreation (Costanza et al., 1989; Millennium Ecosystem Assessment [MEA], 2005). Tidal marshes are carbon, nutrient and sediment sinks, so the degradation of their state increases pressures from nutrient and sediment fluxes to Chesapeake Bay (Kemp et al., 2005).

The main indirect pressure on the Chesapeake wetland is from sea-level rise. Wetlands and marshes can increase elevation in response to indirect pressures, such as sea level rise, under favorable conditions of sediment supply, root growth, and litterfall (Becket et al., 2016). Mean sea level rise in Chesapeake Bay ranged from 2.7–4.5 mm y^{-1} from 1950 to 2000, faster than the global mean of 1.8 mm y^{-1}, probably due to land subsidence. Predicted future rates of sea-level rise (about 6 mm y^{-1}) will increase marsh loss rates and area (Cahoon, 2007), as sea levels are expected to rise by 0.4–1.6 m by 2100 (Rybicki and Landwehr, 2007; Najjar et al., 2010). In a conservative scenario of 60 cm sea-level rise by 2100, 65,000 ha of coastal marsh would be converted to brackish marsh and 11,735 ha of tidal swamp to saltmarsh or open water, which is less diverse. Chesapeake wetlands at risk from sea-level rise include Blackwater Wildlife Refuge, Tangier Sound, and Virginia's Eastern Shore. Shifts in salinity will make the current marshes more favorable for certain species and less for others (Najjar et al., 2010).

Mississippi River Delta Coastal Wetlands, United States

The Mississippi River, the largest in North America, meanders across the coastline every 700–1,200 years, seeking the hydrologic path of least resistance to the sea. These cycles of delta growth and decay (Frazier, 1967) created 17,417 km^2 of fresh to salt marshes in a micro-tidal regime, which represent 41% of the United States coastal wetlands. The soils alongside the main channel wetlands are mostly comprised of inorganic matter, and hurricanes deposit a considerable return flow of inorganics to the fringing coast (Tweel and Turner, 2014). In contrast, the wetland soils between the distributary levees are organic-rich. The rates of land loss rose in the 1930s, peaked in the between 1965 and 1986 (126 km^2 y^{-1}; 0.07% y^{-1}) and the current rates are near zero (Couvillion et al., 2017). The cumulative losses in the last 100 years are about 25% of the original amount, but are expected to increase with the rate of sea-level rise (Strauss, 2013; Church et al., 2014). These losses are well-publicized through media campaigns to acquire restoration funding.

New Orleans is the main urban area and population center, (population 393,000 in 2017). New Orleans is vulnerable to natural disasters because it is built on land of low elevation, in an area of relatively frequent hurricanes as well as storm surges, and it has a weak evacuation strategy (Turner, 2007). There are multiple human activities by the approximately 2 million people who live in the twelve coastal parishes of the Mississippi River delta, such as: fishing, shellfish (e.g., oysters) and aquaculture (e.g., catfish); poultry rearing; agriculture (cotton, sugar, rice, corn, and soybean); oil and gas; shipping and ports; recreation (hunting, fishing, and tourism).

Anthropogenic pressures are well-documented. Sediment fluxes to the coastal zone have varied in the last 200 years. River-mouth wetlands expanded in the 1800s, when trees in the watershed were cut and soils were plowed, enhancing sediment delivery to the coastal zone (Turner and Rabalais, 2003; Tweel and Turner, 2012). Implementation of soil conservation measures trapped sediments behind dams, which decreased sediment loads after the 1930s. River-mouth wetland areas receded dramatically and are now close to the 1700s area (Turner, 2017). The dredging of canals for navigation and drilling for oil-gas exploration have left spoil banks of dredged material as a continuous, 20,000 km long levee that reaches 1–2 m in wetlands where the tidal range is only 20–30 cm. This presents a barrier to tidal water flowing in and out of wetlands, so canal density is directly related to land loss rates from 1900s to present throughout the deltaic plain (Turner and McClenachan, 2018). The Deepwater Horizon oil spill in 2010 released 5 million barrels of oil in 2010 with a surface slick covering 149,000 km^2 of the Gulf of Mexico and oiling coastal marshes for decades (Turner et al., 2019). The nutrient pressure from the Mississippi River watershed has increased nitrogen and phosphorous loads in the last 100 years (Turner and Rabalais, 2003). This is a much greater nutrient pressure than the sewage and local runoff.

These pressures affect the state of the Mississippi wetlands. The contaminated sediments from the Deepwater Horizon oil spill (Rabalais and Turner, 2016) degraded the salt marsh vegetation, decreasing the cover of living marsh vegetation, covering and killing marsh fauna. Marsh shoreline erosion was 2.5 times higher after oiling and the marsh will not recover (Turner et al., 2016). Diversions of the Mississippi water to deltaic wetlands for restoration (a point-source) or cultural eutrophication (non-point source) increase the availability of nitrogen and phosphorous to the wetlands, contributing to general eutrophication of the delta, which also affects the state of the soils and causes carbon losses (Darby and Turner, 2008; Kearney et al., 2011; Deegan et al., 2012; Bulseco et al., 2019).

The delivery of ecosystem services is affected by the degradation of the wetland. This includes regulating services, such as protection from hurricanes, by increasing resistance and lowering storm surge height. The reduction in storm-surge height is ∼0.3 m per 1–2 km of wetland and the presence of vegetation is especially important (Barbier et al., 2008; Costanza et al., 2008; Gedan et al., 2011; Shepard et al., 2011). This contributes to the protection and restoration of coastal areas, which costs $50 billion in Louisiana (Coastal Protection Restoration Authority [CPRA], 2017). Furthermore, the wetland provides an important fish and shrimp habitat for juveniles (Turner, 1977) that become 25–35% of the United States commercial fish landings value and tonnage.

The loss of these ecosystem services can impact human welfare, including loss of life, public infrastructure and private property. A third (36%) of the land in the United States that is less than 3.5 m elevation is in coastal Louisiana and the height of a Category 3 storm-surge is at least 3 m. The slope (elevation gradient) is gradual and sediment supply has been reduced by flood protection levees. New Orleans is a 'city below sea level' in this vulnerable zone, *seaward* of the edge of the upland. Water does not drain out naturally when it rains but must be pumped out uphill. Urban expansion and construction consolidate and compact the fine-grained and highly organic sediments, squeezing out the water and resulting in soil subsidence. The structural integrity of foundations, levees, gas lines and roads are affected by the slow changes in soil oxidation, drainage, volume, settling and subsidence.

Future indirect pressures from sea level rise (SLR) rates will affect the vulnerability and sustainability of river deltas. The Mississippi delta formation began when SLR slowed about 8,000 years ago to around 5 mm y^{-1} or less (Turner and Rabalais, 2018; Turner et al., 2018). The rates were 5.6 mm y^{-1} from 1996 to 2016, which is at the limit of vertical accretion (Morris et al., 2016). Future estimates of SLR vary from an intermediate low rate of 5.6 mm y^{-1} to a high rate of 16 mm y^{-1} (National Research Council [NRC], 2010), reaching 8–16 mm y^{-1} by 2080–2100 (Church et al., 2014). The Mississippi River delta has been sustained and protected by substantial, compensating infrastructure and by multiple, short-term and energy-intensive strategies. This has lowered its vulnerability to risks and raised its sustainability relative to other deltas (Tessler et al., 2015). However, such energy-intensive interventions will become more expensive and less sustainable in a more energy-constrained future scenario. Sustaining the delta in the future will require long-term policies to develop, sustainability solutions (Tessler et al., 2015).

Coastal Wetlands in South America

Bahia Blanca Wetlands, Argentina

The Bahia Blanca is a temperate and mesotidal coastal wetland located in southwest of Buenos Aires Province, Argentina. Despite its size, most of the estuary is marine dominated (Perillo, 1995), with very little river input (Piccolo and Perillo, 1990; Perillo and Piccolo, 2020). The extensive coastal wetland is roughly triangular, about 90 km in length and 53 km wide at the mouth, about 3,000 km^2, cut by NW-SE tidal channels and islands (Perillo and Piccolo, 1999, 2020). Large areas of tidal flats, as well as *Spartina* and *Salicornia* marshes, have developed, providing conditions for a rich benthic infauna that supports significant artisanal fisheries. This wetland includes both a provincial natural reserve that includes the whole estuary south of the Canal Principal and a municipal reserve of only 3 km^2, SW of the city of Bahia Blanca (Perillo and Iribarne, 2003a,b). The wetland is part of the Western Hemisphere Shorebird Reserve Network since March 2016. To the north of Canal Principal, there are two major cities, Bahía Blanca, population 350,000, and Punta Alta, population 70,000, as well as several small towns, such as Villa del Mar, Ingeniero White and General Cerri. The total population around the wetland was estimated at 440,000 in 2016. There is no direct access or settlements on other parts of the wetland.

There are many important human activities in and around the wetland. The main economic sector is maritime transport and shipping, with construction and maintenance of port facilities providing one of the major activities. The largest and deepest harbor system in Argentina is on the coast of the Canal Principal, that exports industrial products and grain. There are oil transfer buoys in the harbors, a trading dock at the port of Rosales and a navy base at Puerto Belgrano. There are two ports further in the estuary, Ingeniero White that has an area devoted to grain and another for general merchandise (i.e., oil, containers, and petrochemical products), and Puerto Galván for oil and petrochemical products (Zilio et al., 2013).

Dredging is a significant human activity that keeps the port facilities accessible to large ships. The navigation channel was dredged to deepen it from 10 to 13.5 m during 1989–1991, which facilitated the development of one of the most important petrochemical complexes in Argentina (Zilio et al., 2013). The navigation channel was further deepened to 15 m depth at low tide during 2012–2014. Additionally, there is dredging of the piers for maintenance purposes. The port authorities are planning to further develop harbor facilities and the installation of new industries planned for 2020. Furthermore, the wetland is used for sewage disposal (Speake et al., 2020).

These activities exert anthropogenic pressures on the wetland along the northern coast of the Canal Principal. Pressures include pollution, harbor, and industrial development. The major Urban Waste-Water Treatment is not working properly and there are four untreated discharges of sewage into the wetland, two from Bahía Blanca City, one for Punta Alta City, and one for the navy base. Pollution is mainly due to sewage discharges, with minor industrial pollution from the industrial flares and deposition of airborne contaminants. These include *Salmonella* and *Escherichia* from sewage discharge and metals from industrial discharges (Simonetti et al., 2017; Severini et al., 2018). The material from the dredging has covered many tidal flats and marshes along the Canal Principal, in some cases forming new islands where the port authorities are planning to develop future harbor facilities.

These pressures have affected the state of the wetland environment and ecosystem. The estuary is eutrophic from the pressures of nutrients from untreated discharges (Freije et al., 2008) that may result in widespread eutrophication, if the flushing capacity of the estuary is exceeded. Nevertheless, pollution is low at present because of the strong currents and rapid mixing of the water in the Canal Principal (Botté et al., 2007). The disposal of dredged materials has affected the most important nesting area of the Olrog's Gull (*Larus atlanticus*), which is an endangered species native of Argentina, Uruguay, and Brazil, of which 2/3 of the total population lives in the Bahia Blanca wetland.

This degradation of the state of the wetland environment and ecosystem leads to a loss of ecosystem services. The long-term effects of sewage pollution could be damaging to the regulating services of the wetland such as denitrification. Meanwhile, there is a decrease in the provisioning services of the wetland, i.e., the catch of prawns, shrimp and local fish. Nevertheless, there is little evidence at present that this reduction can be directly associated to pollution rather than large scale effects, such as Southern Oscillation events, especially during dry *La Niña* periods. However, artisanal fishers complain of the deformations of fish caught in the vicinity of effluent discharges and there maybe impacts on human welfare (Speake et al., 2020).

There are also indirect pressures from climate change. The islands in the wetland ecosystem are only 50 cm above spring high water levels. Mean sea level rise for Argentina is on the order of 1.6 mm y^{-1} (Lanfredi et al., 1988). However, the estimated global increase of about 83 cm by year 2,100 (Oppenheimer et al., 2019) would completely flood the estuary (Perillo and Piccolo, 2020), and overwhelm the wetland and islands. There is no practical way to mitigate this situation. There also appears to be changes in the typical wind regime. These are strong winds from the N-NW blowing about 40% of the time with average speeds of the order of 20 km h^{-1}. These winds blow parallel to major channels affecting tidal prediction, navigation programming and the generation of large wind-tide interaction waves (Perillo and Sequeira, 1989; Perillo and Piccolo, 1991). However, in the last 5 years the number of calm days has increased significantly, which reduced the number of storm surge phenomena on the town of Ingeniero White from once a year to once every 2 or 2.5 years. That also changed the conditions in beaches just outside the estuary where there is a reduction of the appearance of jelly fishes (always associated to upwelling forced by N winds) during the summer months (Brendel et al., 2017).

Coastal Wetlands in Europe

Only two European case studies are presented in the text in order to reduce the length of the article. This is to provide examples of less well-known wetland systems, outside Europe. Nevertheless, some details of 12 further coastal wetlands in Europe are included in the supporting **Supplementary Tables S1–S5**. These European wetlands that are not represented in the text are shown in italics where they appear in a supporting table.

Ria Formosa Wetland, Portugal

The Ria Formosa wetland is part of a temperate, mesotidal, 80 km^2 coastal lagoon, on the south coast of Portugal (Aníbal et al., 2019). The Gilão is the only permanent river flowing into the wetland that is connected to the Atlantic by several inlets. The wetland includes extensive areas of salt marsh (Arnaud-Fassetta et al., 2006) and seagrass meadows (Cabaço et al., 2009). There are three main towns in the catchment: Faro, the district capital (∼64,500); Olhão (∼45,000); and Tavira (∼26,000), as well as several smaller urban areas (population data from 2011 Census, Instituto Nacional de Estatistica[2]). The influx of visitors in the summer increases the population, in some cases several-fold, overwhelming infrastructure designed for the resident population, such as Urban Waste-Water Treatment facilities (Newton et al., 2013; Cravo et al., 2015; Veríssimo et al., 2019).

Multiple human activities cause pressures on the Ria Formosa wetlands. These include change of land-use, such as conversion of saltmarsh into salt-extraction pans (salinas), a historical activity. Some of these ponds have now been converted for aquaculture of fish (Cunha et al., 2013), and others have been converted into Urban Waste-Water Treatment ponds, e.g., near the airport (Veríssimo et al., 2019). Construction in the three towns (Faro, Olhão, and Tavira) has encroached on the surrounding wetlands as well as on the sand dunes of the barrier islands that separate it from the Atlantic. Construction includes groin to retain sand along beaches to the west of the lagoon, the creation of consolidated inlets, such as Barra do Farol (Carrasco and Matias, 2019). An airport, a port (Faro) and several marinas (Faro, Olhão, and Tavira) have been constructed from the wetland. The wetland channels are regularly dredged for sediment extraction and to keep the access to the ports and marinas navigable, since the mean depth is only 1.5 m. Several small, torrential streams (e.g., the Ribeira de São Lourenço and Gondra) were dammed with dykes to retain freshwater in the 19th Century, and some land was reclaimed from the wetland marshes for agriculture (e.g., Ludo). An artificial inlet, Nova Barra do Ancão, was opened in the western part of the lagoon in June 1997. Artisanal fishing and shellfish harvesting are important activities in the Ria Formosa and shellfish concessions are also in the wetland. Olhão is a

[2] https://www.ine.pt

large fishing port, but most of the catch comes from the adjacent coastal waters rather than the lagoon. The Campina de Faro in the watershed is the richest agricultural region of the Algarve. Intensive farming includes water cress and salads (Vitacress), fruit and vegetables in greenhouses, as well as orange groves. Some agricultural land has been converted to golf (e.g., Quinta do Lago). There are also intensive poultry farms (Ludo) and pig-farms in the catchment of the wetland. The area around the Ria Formosa is not industrial, although there are some food industries, such as fish processing in Olhão.

The various human activities in the catchment of the Ria Formosa exert multiple anthropogenic pressures on the wetlands. Foremost of these is the loss of habitat and associated biodiversity, especially the loss of seagrass meadows and saltmarsh, as well as dunes. The groin, impoundments, ponds, obstructed channels and dykes, represent a loss of connectivity of the wetlands that disrupt both the hydrology and the sedimentology. This has resulted in a saline intrusion and salinization of the western part of the wetland, ever since the 19th century. Overfishing, shellfish harvesting, illegal harvesting of sea-cucumbers (*Holothuria arguinensis*) and the attempted introduction of non-indigenous species (Manila clam, *Ruditapes philippinarum*; Japanese prawn, *Penaeus japonicus*; Pacific oyster, *Crassostrea gigas*) are further pressures on the ecosystem. The use of fertilizers in the agricultural zone and golf-courses around the Ria Formosa, manure from animal-rearing, and inadequate sewage treatment are all pressures that can lead to eutrophication (Newton et al., 2003) and microbial contamination (Cravo et al., 2015). Several studies have documented contamination and pollution of the Ria Formosa by metals and organic chemicals (Bebianno, 1997; Mudge and Duce, 2005; Bebianno et al., 2019; Moreira da Silva et al., 2019), despite the low industrial development around the wetland.

The deteriorating state of the environment and ecosystem has consequences on the ecosystem services provided by the Ria Formosa wetland (Newton et al., 2018). This has been most notable in the declining clam harvest (provisioning service). Decline in the coastal fisheries has also been attributed to the decline in seagrass (García-Marín et al., 2013) and nursery supporting services (Abecasis et al., 2009). The Ria Formosa is a popular destination for ‘beach and sand’ tourism (Zacarias et al., 2011; Semeoshenkova and Newton, 2015) as well as eco-tourism, thus contributing revenues to the local economy through cultural services. Perturbation of sediment fluxes and supply by construction on the dunes of the barrier islands, as well as the retention of sediment by dams in the catchment and groin to the west, make the wetland vulnerable to erosion (loss of regulating services) (Ceia et al., 2010) and storm surges. This threatens including important infrastructure, such as the international airport, thus potentially affecting tourism (cultural services).

The decline of ecosystem services has impacts on human welfare (Bebianno et al., 2019). Continued microbial contamination from sewage (Cravo et al., 2015) as well as algal toxins from HAB (Lage et al., 2014) and other public health issues (Bebianno et al., 2019) are related to the degradation of the wetland. The declining incomes of shellfish harvesters and artisanal fishers eventually make these culturally unique lifestyles unsustainable and fishers seek other sources of income and ultimately employment.

Like all coastal wetlands, the Ria Formosa is also threatened by indirect pressures such as sea-level rise and increased temperature (Brito et al., 2012), and storm surges. Violent Atlantic storms in recent years have frequently washed over the barrier islands (Almeida et al., 2012). So far, there has not been any loss of life, although infrastructure (airport) and housing has been affected by a tornado on 2011. These indirect pressures ultimately threaten the whole Ria Formosa wetland, which is constrained by urban development at its landward boundary.

Venice Lagoon Wetlands, Italy

The lagoon of Venice (Italy) is in the NW part of the Adriatic Sea and includes one of the largest Mediterranean wetlands. The lagoon is shallow (mean depth 1.5 m) and covers about 550 km^2, of which 432 km^2 are tidal. The mean tidal amplitude is about 0.6 m, with about 60% of water being renewed through three inlets every 12 h (Facca et al., 2014). Freshwater inputs are about 30 m^3 s^{-1}, delivered by small rivers and a network of artificial channels. The main rivers, Brenta and Sile, where diverted long ago in order to avoid the siltation of the lagoon (Solidoro et al., 2010).

Due to its complex morphology, the wetland includes subtidal flats, intertidal mudflats, saltmarshes, ‘*valli da pesca*’ (marginal areas close to tidal exchanges) and a network of relatively deep channels (Rova et al., 2015). Macroalgae and seagrasses account for most of the primary production. Besides the city of Venice, there are several small islands, such as Murano, Burano, Lido, S. Erasmo, which collectively host about 85,000 residents, according to Venice Municipality data. Venice and its lagoon are a highly attractive touristic venue and the numbers of visitors exceed the resident population.

Shipping and navigation, from small boat to large cruise ships, have always been important human activities for the islands of Venice lagoon, and tourism also increases the internal navigation. Shipping has entailed several modifications including rivers diversions and the construction of jetties at the inlets in the 19th and early 20th centuries. More recently, from 1927 to 1970, the navigation channels were dredged and enlarged for access to the main port facilities at Porto Marghera, which stimulated industrial development and activities around Mestre, affecting the wetlands. Clam dredging activities are now mostly illegal because of contamination. The catchment of the lagoon (Veneto) includes areas of market gardening, fruit-growing and vine-growing reclaimed from the wetland. Groundwater extraction in the 1960’s and 1970’s was excessive and contributed to the subsidence of the wetland.

Resulting direct pressures include disruption of hydrology, re-suspension of large amounts of sediment that exacerbate the erosion of the wetland, increased nutrient and organic matter supply, as well as inputs of industrial effluents and contaminants to the wetland. Nutrient run-off from agriculture represents the main source of Nitrogen and nutrient pressure to the wetland. Poorly controlled shellfish harvesting, Manila clam

(*Ruditapes philippinarum*), has increased sediment disturbance and resuspension in the wetland. Resuspension is exacerbated by increased internal navigation, mainly linked to tourism. Sediment resuspension can release contaminants that were trapped in the sediment as a legacy of industrial pollution in the 20th century, especially in the central part of the wetland (O'Higgins et al., 2014; Giubilato et al., 2016).

The state of the water and wetland sediment quality have been degraded by industrial activities that heavily contributed to the pollution of the lagoon in the past. Both sediment budget and sediment transport processes have been heavily affected by anthropogenic interventions. The disruption of hydrology and sediment supply has resulted in subsidence. Eutrophication can cause macrophyte blooms (*Ulva*), localized anoxia in confined areas of the lagoon wetlands and loss of seagrass, as in 2013. However, concentrations of inorganic nitrogen and phosphorus is decreasing in the water and at the sediment surface (Facca et al., 2014). The loss of wetland habitat, especially the reduction of seagrass meadows, saltmarshes and associated biodiversity, as well as the introduction of exotic, invasive algal species are severe. The area of saltmarshes has halved from 68 to 32 km^2 between 1927 and 2002, while the area of subtidal flats has increased from 88 to 206 km^2. Overall, the average depth of the lagoon has increased from 0.62 m in 1927 to 0.88 m in 2002 (Sarretta et al., 2009).

Ecosystem services provided by the Lagoon of Venice are still being assessed (Rova et al., 2015; Newton et al., 2018). Overfishing has led to sharp decrease in clam yield, from about 40,000 tons in the 1990s to 2,000–5000 tons in 2016. Seagrass meadows have recently recolonized large areas of southern sub-basin, contributing to sediment bio-stabilization. Recolonization in the northern part is being stimulated by an EU funded LIFE project[3].

Indirect pressures such as sea level rise are expected to worsen due to climate change. Thus, the conservation of the wetland islands and protection from *acqua alta* flooding is one of the main current issues, with disastrous floods in November 2019. This has led to the major infrastructural changes to the inlets, e.g., the MOSES flood barrier.

Coastal Wetlands in Africa

A DPSIR analysis of coastal wetlands in North Africa includes 11 systems from south west Morocco to Egypt (El Mahrad et al., 2020). Thus, only one North African example is included in the present review, because of its unique characteristics as a 'double' system, the Bizerte–Ichkeul wetlands.

Bizerte Lagoon and Ichkeul Lake Wetlands, Tunisia

Bizerte lagoon and Ichkeul lake are linked, coastal wetlands in Tunisia, where the climate is Mediterranean, semi-arid, with a hot, dry summer and mild, wet winter (MAERH, 2003). Bizerte lagoon, also known as 'Lac de Bizerte' or 'Mezaouka', has an area of 128 km^2 and a mean depth of 7 m. A 6 km long inlet connects to the Mediterranean Sea and a 5 km long Tinja channel connects to Ichkeul lake (Béjaoui et al., 2008). Ichkeul is a brackish-freshwater lake with an annual depth variation of 1–3 m, low winter salinity (S = 3) and high summer salinity (S = 30–50), (Savoure, 1977; Hollis, 1986; Ben Rejeb-Jenhani et al., 1991; Tamisier et al., 2000). Rainfall during autumn and winter flows in through 6 wadis, seasonal rivers, some overflowing through the Tinja channel into Bizerte lagoon. The reverse happens in summer, when high evaporation lowers the water level in Ichkeul lake and water flows in from Bizerte lagoon through the Tinja channel (Casagranda et al., 2006). The exceptional salinity gradient (3–45) is a unique feature that gives Ichkeul lake and wetlands a rich biodiversity. The area comprises three units: the 89 km^2 brackish water lake, 27 km^2 marshes, and a 13 km^2 isolated and wooded massif; Djebel Ichkeul (Trabelsi et al., 2012).

Few people (about 350 in 1999) work in the quarries and live in the Ichkeul reserve. Activities in the Ichkeul wetland include quarrying fishing, grazing of cattle, sheep and goats. Over-grazing is causing serious degradation and causing erosion, as well as affecting the water buffalo (Bousquet, 1988). Poaching, illegal hunting and overfishing are further activities in the Ichkeul wetland ecosystem. However, there is intensive agriculture, plowed land, orchards and pasture in both densely populated Ichkeul and Bizerte catchments, so the wadis have been dammed for freshwater (Koundouri et al., 2006).

The area around Bizerte lagoon is much more developed and densely populated. Economic sectors and activities include industry (petrochemical, metallurgical, textile, agro-alimentary, and cement), farming, animal rearing, aquaculture, fisheries, shipping and port, tourism.

These activities exert various direct, anthropogenic pressures on the lagoon and lake wetlands. Bizerte wetlands are affected by pressures from domestic sewage, industrial waste, atmospheric pollution, farmland runoff, effluents from fisheries and bivalve aquaculture (Béjaoui et al., 2010). The urban and industrial effluents are mainly discharged in the southeast, southwest and the north sectors of the lagoon wetlands. The southeastern sector is affected by the domestic and industrial effluent through the Guenniche River, the southwestern sector by the steel factory discharge, and the northern sector by urban wastes and sewage effluent. The mismanagement of the fisheries sector has resulted in an increase in the numbers of illegal fishing boats. The fishing pressure has decreased the stock to such a low level that the reproductive capacity is in peril (DGPA, 2012).

The main, direct, anthropogenic pressure on the Ichkeul wetland comes from the dams that have changed the hydrology and sediment supply to the lake. The dams have modified the connectivity and hydrology of both the Ichkeul and Bizerte wetlands. The changes in hydrology and freshwater supply in the lake and lagoon have altered the water equilibrium between the wetlands, rivers, lake, lagoon, and the sea (Smart, 2004; Saied and Elloumi, 2007). A lock on the Tinja channel was constructed in 1996 to control salinization and the decline of the high biodiversity (IUCN, 1994). Drainage canals have lowered water in the Ichkeul wetland, so livestock now graze formerly inaccessible reedbeds. Open cast quarrying threatens the wetland ecosystem and pollutes the lake water (Bousquet, 1988).

Direct, anthropogenic pressures affect the state of the environment and ecology of both wetlands. The dominant flora

[3] http://www.lifeseresto.eu/

of Ichkeul wetlands was extensive meadows of *Potamogeton pectinatus* L., the major food source for wintering waterfowl (Bousquet, 1988), and *Ruppia cirrhosa* (Petagna Grande), (Hollis, 1986; Bureau Central d'Etudes pour les Equipements d'Outre-Mer [BCEOM], FresinusConsult, CESalzgitter, and STUDI, 1995; Tamisier et al., 2000) until the early 1990s. The area occupied by *P. pectinatus* varied both annually and seasonally, with a summer-autumn maximum and a late winter minimum (Hollis et al., 1977; Hollis, 1986). The annual yield of *P. pectinatus* was cropped largely by wintering birds. However, rising salinity now restricts the growth of this species (Zairi, 1997). The wetland marshes were famous for their variety of hydrophilic vegetation, especially *Scirpus maritimus*, important for geese (*Anser anser*), (Hollis et al., 1977; Hollis, 1986). Poaching, illegal hunting and overfishing have caused a decrease in biodiversity. The damming of the water supply to Ichkeul lake has led to evaporation of the *Scirpus maritimus* marshes, which have been replaced by invasive, annual weeds. Salt-loving plants have replaced fresh-water species in the wetland marshes. Industrial development in the catchment has contributed to the contamination and pollution of Ichkeul lake by organo-metals and organic chemicals (Zairi, 1997).

The pressures have also affected the state of Bizerte lagoon. Run-off of fertilizers led to eutrophication and increases in water temperature have stimulated microbial activity, hypoxia and eutrophication of Bizerte lagoon (Essid et al., 2008; Ben Omrane et al., 2010; Fertouna-Bellakhal et al., 2014). The relatively high levels of organic matter recorded in the wetland sediment caused a decrease in dissolved oxygen causing repeated fish kills (Béjaoui et al., 2008). Industrial contamination is the source of metal (Pb and Zn) in the sediments of Bizerte (Zaaboub et al., 2014), and accumulation of Zn may contribute to sediment toxicity. Increase of marine navigation contaminates the lagoon and wetlands with Polycyclic Aromatic hydrocarbons and Polychlorinated biphenyls. There are occurrences of harmful algal blooms (HAB) of species belonging to the genus *Alexandrium* in Bizerte lagoon (Turki et al., 2007, 2014; Sahraoui et al., 2009), possibly related to the increase in internal marine navigation and introduction of non-indigenous dinoflagellates from ballast water.

The degradation of the state of the environment has led to a loss of ecosystem services. The provisioning services of fishing and aquaculture of Bizerte lagoon and wetland are threatened by changes in salinity, pollution, harmful algal blooms and overfishing. The Tunisian government created Ichkeul National Park in 1977 as an internationally important wintering ground for waterfowl (Scott, 1980; Hollis, 1986). The lake and wetlands of Ichkeul are the most important stopping-over point in the Western Mediterranean basin for migrating birds (150,000–250,000), as recognized by the Ramsar Convention List, the MAB Biosphere Reserve, and the UNESCO World Heritage List (Hollis et al., 1977; Tamisier and Boudouresque, 1994; Casagranda et al., 2006). The deteriorating state of the environment and ecosystem has consequences on the supporting and regulating ecosystem services provided by the Ichkeul wetland, especially a decrease population of migratory birds such as ducks, geese, storks, and pink flamingoes. Important species are *Anse ranser*, *Anas penelope*, *Aythya farina*, and *Fulica atra* (Tamisier and Boudouresque, 1994; Tamisier et al., 2000). The degradation meant that Ichkeul National Park was placed on the list of World Heritage in Danger in 1996, while in 1998 data from IUCN showed that the increase in salinity threatened the value of the World Heritage site.

There are also impacts on human welfare such as food safety, public health, decreased fisheries and aquaculture revenues. Mussel production in Bizerte lagoon has decreased in recent decades due to harmful microalgae and mortality (Sahraoui et al., 2009; Turki et al., 2014) with considerable economic losses (Sahraoui et al., 2009; Turki et al., 2014) and this is an obstacle to the development of shellfish aquaculture industry. This has also been associated with increased bacterial activity (Sakka Hlaili et al., 2006, 2007; Essid et al., 2008; Ben Omrane et al., 2010; Fertouna-Bellakhal et al., 2014). The decrease of fish stock and diversity has also caused a decline in jobs of fishers and aquaculture farmers. Ichkeul Park is a popular destination for eco-tourism and cultural heritage, thus contributing revenues to the local economy through cultural services, although eco-tourism has decreased. Numbers of visitors were 23,000 in 1987 and 2,500 in a 6-week period of 1988 (IUCN, 1994). However, visitor numbers decreased by 2016, partly because of security issues in the tourism sector.

Indirect pressures on both wetlands are expected to increase with climate change. In particular, the hydrological imbalance between Lake Ichkeul and Bizerte lagoon (Tamisier and Boudouresque, 1994; Tamisier et al., 2000), initiated by the reduction in freshwater supply to the lake, will be aggravated by sea-level rise in the lagoon, (Casagranda et al., 2006; Koundouri et al., 2006; Béjaoui et al., 2008; Béjaoui et al., 2010).

Watamu, Mangrove Wetland of Mida Creek, Kenya

Mida Creek covers an area of 31.6 km^2 at Watamu, on the Indian Ocean coast of Kenya (Dahdouh-Guebas et al., 2000). The climate is tropical with temperatures of 26–32°C. The monsoons bring two, distinct rainy seasons, in April/May and in October/November (Gang and Agatsiva, 1992). Mida Creek has habitats that include mangrove forest, rock outcrops, sandflats, seagrass beds, and coral reefs adjacent to deeper waters. The mangroves dominate the wetland occupying 1,746 ha, (Dahdouh-Guebas et al., 2000; Kairo et al., 2002). Seven of the nine mangrove species found in Kenya occur in Mida Creek with abundant *Rhizophora mucronata*, *Avicennia marina*, and *Ceriops tagal* and more sparsely distributed *Sonneratia alba*, *Xylocarpus granatum*, *Bruguiera gymnorhiza*, and *Lumnitzera racemosa*, (Kairo et al., 2002).

Mida Creek is part of the Watamu-Malindi marine protected area established in 1968 and was designated in 1979 as a Biosphere Reserve by UNESCO (Dahdouh-Guebas et al., 2000). Nevertheless, human activities continue in the national marine reserve, because traditional harvesting of resources, as well as research and tourism, are allowed. The traditional communities use the mangrove forest as a source of food (honey, crab, and fish), construction material, firewood, and medicinal purposes (Gang and Agatsiva, 1992; Dahdouh-Guebas et al., 2000). The population live in traditional villages around the administrative sub-locations Gede (~29,600 people) and

Matsangoni (~16,000 people) and tourist developments around Watamu of ~26,000 people (2009 census in KNBS, 2010). Associated activities include nature watching, recreational fishing and boating, scientific research and education (Frank et al., 2017; Owuor et al., 2017, 2019a). Farming activities include cultivation of coconuts, cashew nuts and mangoes. Artisanal fishing using traditional methods is also important activity (Gang and Agatsiva, 1992). Cultural shrines associated with the forest are important to the traditional communities in Mida Creek (McClanahanan et al., 2005).

The human activities exert direct pressures by the selective removal of specific species and the increasing encroachment of settlements, within the reserve. Other pressures are land derived pollutants from agricultural runoff, untreated sewage, litter and oil spills due the substantial increase in maritime activity (Lang'at and Kairo, 2008).

These pressures result in a degradation of state of the mangrove. This includes a decline in the mangrove cover and a change to the mangrove community structure (Owuor et al., 2017), contributing to the overall rate of decline in mangrove cover for Kenya of between 0.76% per year between 1985 and 2015 (Kirui et al., 2013). High sediment loads, due to poor agricultural practices in the hinterland, have been particularly destructive for mangroves, causing siltation of aerial roots during periods of flooding, such as during the El Nino years of 1997 and 1998 (Lang'at and Kairo, 2008). Oil spills have also contributed to damage of mangrove aerial roots.

Mida Creek provides important ecosystem services (ES) for *provisioning* timber and firewood, medicinal uses, and food products; whilst its *cultural* services are spiritual shrines, educational, recreational, and tourism (Owuor et al., 2017; Frank et al., 2017; Newton et al., 2018). The s*upporting* and *regulation* services of the creeks are critical for providing shore protection, water filtration, pollution regulation, and habitat to many organisms (Frank et al., 2017; Owuor et al., 2017, 2019a,b). The mangroves provide an important habitat for fauna, including commercially important crabs and fish, as well as providing a breeding ground and nursery (Nagelkerken et al., 2000; Granek et al., 2009). Mangroves have a high capacity to sequester and store carbon (Alongi, 2014), which is an important ecosystem service contributing to mitigation measures such as Reduced Emissions from Deforestation and Degradation (REDD), (Webber et al., 2016).

Indirect pressures related to climate change and sea level rise (SLR) are not well studied for Africa (Ward et al., 2016). However, two studies in Kenya provide some idea what could happen to the mangroves of Mida Creek in the future, with Kebede et al. (2010) predicting an SLR of 1 mm.y^{-1}, although Lang'at et al. (2014) observed an increase of 4.2 mm.y^{-1} in surface elevation in mangrove stands from Gazi Bay, approximately 150 km south of the Creek.

Malanza Wetlands, São Tomé and Príncipe

São Tomé is an equatorial island, west of Gabon with two main seasons, rainy September–May and dry June–September. The Malanza mangrove is only 0.7 km^2, however, it is the largest in São Tomé and located in Obô Natural Park (PNOST), created in 2006. The wetland is constituted by a complex network of shallow, narrow, channels flowing to a central basin that is connected to the Atlantic through a constricted opening. The fringing vegetation is dominated by the black mangrove *Avicennia germinans* and the red mangrove *Rhizophora racemosa*.

There are two village communities, Malanza and Porto Alegre. Each village has a population of approximately 500 inhabitants who rely on coastal fisheries and agricultural activities. Fisheries inside Malanza lagoon include a high percentage of juveniles (Félix et al., 2017). The communities of Malanza and Porto Alegre organize boat tours that complement the income of local families.

The human activities exert several anthropogenic pressures on the wetland. A challenging pressure is the introduction of non-indigenous species, such as the tilapia *Oreochromis mossambicus*, which tends to be an aggressive invader (Canonico et al., 2005) in the wetland. The construction of a road-bridge that crosses the lagoon to give access to the village of Porto Alegre, severely restricts the flow of water and sediments (Félix et al., 2017). Agricultural intensification and oil palm monoculture threaten the natural wetland habitats and biodiversity.

The current state of the environment includes water column stratification and hypoxia/anoxia at the bottom and wetland sediments. Oxygen depletion can reduce vegetation and biodiversity, as well as affect fish and shellfish stocks. Inputs of nutrients from fertilizer run-off may also promote future eutrophication.

The state changes affect the provision of wetland ecosystem services. The local population is poor and very dependent on the natural resources available in the wetland. Provision of food (e.g., fish and shellfish protein) is affected by the current oxygen depletion at the bottom, with reports of declining fish populations and impacts on human welfare. The wetland also provides an opportunistic feeding ground for marine species and a nursery for the juveniles (Félix et al., 2017). Moreover, palm tree plantations have also reduced the land area available for local agriculture. On the other hand, recreational and cultural activities related to tourism have been increasing, providing a valuable contribution to local subsistence.

Indirect pressures from climate change and sea level rise will affect the future of this mangrove, since the whole system is only just above present sea level. In recent years, the ocean has been claiming a sand bar to the west of the system, near Praia Jalé. If this sand bar disappears, it may affect the entire dynamic of the region, effectively turning the southern tip of the island of São Tomé into an islet, with consequences for the ecology of the mangroves.

Coastal Wetlands in Asia

Vembanad Wetlands, India

Vembanad Lake is a RAMSAR site that spans three districts Alappuzha (Alleppey), Kottayam and Ernakulam, and comprises 4% of the state of Kerala, SW India. It is narrow (0.8–6.9 km), but the longest (96 km) and largest (area 1521.5 km^2) wetland in India, running parallel to the coast and broadening in the south. Barrier spits separate the wetland from the Arabian Sea, to which it is connected by two tidal inlets, at Kochi (400 m) and further north at Azhikode (250 m). The wetland includes marshes, mangrove forests, a network of canals and backwaters, small islands and reclaimed land. The wetland has a freshwater southern zone and a saltwater northern zone. The total drainage area is 40% of the area of the State, and there are 10 main rivers draining into the wetland. The annual surface runoff is 2.19×10^{10} m^3, nearly 30% of the surface water resource of Kerala (Anon, 2016).

The Vembanad wetland supports a diverse flora and fauna, including small, isolated patches of mangrove (e.g., at Kumarakom, Vypeen, Kannamali, and Chettuva), (Anon, 2016). The Vembanad wetland and the Kumarakom Bird Sanctuary on the east coast support waterfowl migrating along the Central Asian Flyway and a large wintering population (Narayanan et al., 2011). Fauna also includes a large variety of reptiles and commercial fish (mainly oligohaline), valuable crustaceans, (e.g., prawns), and molluscs, (e.g., clams).

The population density (1,103 km^{-2}) in the 2011 census was almost three times the mean for India, (382 km^{-2}). There are many human activities in settlements along the wetland, including shipping, agriculture, fisheries, aquaculture, shell mining for lime and houseboat tourism on the famous backwaters. The town of Alappuzha is sandwiched between the wetland, the lagoon and the Arabian Sea and famed for its large network of canals that meander through the town. The major shipping port of Kochi (Cochin) is located at one of the inlets to the Arabian Sea and recently, the new container terminal at Vallarpadam has been added[4]. The 1,252 m long Thanneermukkom salt-water barrier, the largest mud-regulator in India, is located where the lagoon is narrowest and has changed the hydrology of the wetland. Thanneermukkom barrage was built in 1976 to prevent saline ingression during the dry season in the low lying Kuttanad region, known as the 'Rice bowl of Kerala.' Shallow parts have been bounded for punja cultivation, Padasekharam/low lying rice fields. Rice cultivation is the main crop, with extensive plantations of coconut, banana, tapioca, cereals such as millet, pigeon peas, peanut, and pawpaw grown as intercrops (Planning Commission Government of India, 2008). Constructed infrastructure includes roads across drainage canals, as well as a network of roads and bridges in the Kuttanad region, and commercial developments. Fermentation of coconut husk for coir manufacture is carried out extensively on the surface waters of the lagoon (Kumar, 2018). The black clam *Villorita cyprinoides* is harvested by hand picking or using a canoe for its shell rather than as shellfish (Laxmilatha and Appukuttan, 2002).

Changes in land-use from reclamation is a major and growing direct, anthropogenic pressure. There are extensive encroachments changing the land-use of the wetland for agriculture, especially rice cultivation (Planning Commission Government of India, 2008). Excess fertilizers and pesticides used in agriculture drain into the wetland system. Another major pressure results from changes in hydrology for irrigation and hydroelectric schemes affecting the connectivity of wetlands and surface waters, tidal and inundation regimes, circulation and mixing patterns (Haldar et al., 2019). Untreated sewage is discharged directly into the lake from houses in the wetland, as well as from houseboats used by the tourism industry. Also, many rivers in the catchment area flow through industrial areas carrying polluted effluents to the wetland (Varkey et al., 2016). Indiscriminate exploitation of small-sized clams, closure of the Thaneermukkom Barrage, frequent dredging, pollution due to retting, effluents from shrimp processing plants/factories and extensive weed growth are major pressures on the wetland (Laxmilatha and Appukuttan, 2002).

The state of the environment and ecosystem of the wetland is affected by these pressures, especially south of the barrage at Thaneermukkom, which affects the distribution and ecology of the wetland. The area of Vembanad wetland has decreased by 465 ha from 2002 to 2014 (Varkey et al., 2016). Reduced connectivity and flushing has resulted in eutrophication (Varkey et al., 2016). It has also led to the proliferation of non-indigenous, invader species (*Eichhornia crassipes* and *Salvinia molesta*) from Latin America that are widely distributed and a threat to the wetland (Kumar and Rajan, 2012). The 2008–2011 fish count by the Ashoka Trust for Research in Ecology and the Environment (ATREE), recorded 67 species of fin fishes and 14 species of shellfish. Earlier studies indicated that the fish fauna of Vembanad was dominated by marine migratory species (56%), but the 2011 fish-count indicated that salinity-tolerant, freshwater species dominated (69%) and the marine migrants were reduced to 31% (Kumar and Rajan, 2012). Plastic bags, bottles and other wastes dumped in the wetland and lake settle to the bottom and adversely affect bottom feeders, e.g., gobids, and the fishes that attach their eggs to the benthic substratum, (Kumar and Rajan, 2012). A decline in carnivores and a dominance of omnivores (58%) followed by 6% herbivores, 6% larvivores, and 2% detritivores also indicates organic pollution (Kumar and Rajan, 2012). The discharges of industrial effluents contaminate and pollute the wetland. The pollution load index indicates that the sediment is heavily polluted in the north and moderately polluted in the extreme south port region (Selvam et al., 2012). For example, there is severe and moderately severe enrichment of Cd and Zn with minor enrichment of Pb and Cr in the north. Mercury contamination is also higher in the north (Mohan et al., 2014). The presence of high mercury concentration in the subsurface sediment indicates historic, industrial mercury deposition.

The degradation results in the loss of ecosystem services in the wetland. The Thaneermukkom barrage has affected the production of fish and shrimp from the lagoon but has been successful in keeping the Kuttanad water fresh for agriculture (Varkey et al., 2016). The estimated annual mean fishery catch (provisioning ecosystem service) is 4774.46 t, only 10.1% in the south and 89.9% in the north (Asha et al., 2015). The main commercial species are marine penaeid prawns, clams such as *Villorita cyprinoides* that represents 70% of the production, cyprinoid fish and *Meretrix meretrix* (Kumar and Rajan, 2012).

[4]https://ernakulam.nic.in/vallarpadam-terminal/

Furthermore, indirect pressures from climate change and rising sea levels are of concern as the entire wetland is low lying. Estimates indicate that a 1m rise in sea level will inundate about 169 km^2 of the coastal region around Kochi, especially areas adjacent to the tidal creeks, backwaters and lakes reaching inland areas far from the coast (INCCA, 2010).

Sundarbans Wetlands, Bangladesh and India

The Sundarbans coastal wetland on the south coast of Bangladesh and northeastern India is fed by the river Ganges and hundreds of tributaries. It includes the largest, single tract, mangrove forest in the world with a total area of 10,000 km^2, 62% in Bangladesh and 38% in India (Islam and Gnauck, 2009). This mangrove forest constitutes about 51% of the forest area of Bangladesh and about 50% of the revenue for the forestry sector (Islam and Gnauck, 2009). The Sundarbans wetland is famous for its diverse biodiversity, which includes about 334 species of plants, 282 bird species, 49 mammal species, 210 fish species, 63 reptile species, and 10 each of amphibians and molluscs (Rashid et al., 1994; Biswas et al., 2007). Several rare species such as the tiger (*Panthera tigris*), the dolphins (*Platanista gangetica*), and the crocodile (*Gavialis gangeticus*) are found in Sundarbans wetland (Gopal and Chauhan, 2006). Endangered species such as the tiger and dolphin epitomize the conflict between the loss of habitat (quantity and quality) and human activities in the Sundarbans wetland. In 1992, about 601,700 ha of forest reserve of the Sundarbans in Bangladesh, and in 2019, 423,000 hectares of the Indian Sundarbans were declared Ramsar sites[5]. In 1999, 32,400 hectares of the Sundarbans in Bangladesh part were declared a UNESCO World Heritage Site (Islam and Gnauck, 2009).

Human activities in the Sundarbans mangrove forest ecosystem exert multiple pressures on this coastal wetland and the over-exploitation of natural resources. Sundarbans has a population of over 3.5 million inhabitants, mostly living below the poverty level and heavily dependent on forest natural resources (e.g., fire wood, livestock fodder, timber, honey, and fishes) for their food and livelihood (Hoq, 2007; Ahmad et al., 2009). Households that are dependent on Sundarbans obtain important monetary benefits annually from capture fishery (U$ 976 ha^{-1}), fuel energy (U$ 80 ha^{-1}), honey (U$ 53 ha^{-1}), and fodder (U$ 26 ha^{-1}), (Rahman et al., 2018). Unplanned developments threaten this World Natural Heritage (Islam et al., 2018). Land-reclamation and polders in the 1960's increased agricultural land. Aquaculture farms cleared large areas of mangrove for shrimp ponds, without considering the externalities such as the chemicals from the effluents. Poor people, especially women and children, use illegal, fine-meshed push nets to catch of wild post larvae (PL) of *Penaeus monodon* shrimp. Barrages in the catchment (e.g., the Farakka barrage in the river Ganges) have decreased the freshwater flux to the wetland, especially in dry season February–June (Hoque and Alam, 1995; Wahid et al., 2007; Islam and Gnauck, 2009; Bahar and Reza, 2010). Dykes to protect from coastal flooding have also been constructed in the SW of Bangladesh, further disrupting the natural hydrology. The construction of the Rampal power station (Bangladesh), a 1,300 MW, coal-fired power plant close to Sundarbans territory (Tamim et al., 2013; Islam and Al-Amin, 2019), has resulted in protest from environmentalist groups anxious about pollution of the Sundarbans wetlands. Consequences include atmospheric emissions and land storage of fly ash from the power plant (Tamim et al., 2013).

Human activities in the Sundarbans exert multiple pressures on this coastal wetland and the over-exploitation of natural resources. Pressures include modifications of hydrology, changes in land-use and encroachment into the forest, for example to construct aquaculture ponds, unsustainable harvesting of shrimp post-larvae, over-fishing and illegal logging. The balance of freshwater and seawater, to the mangrove wetlands (Islam, 2006) has been modified.

These pressures have affected the state of the environment and ecology of the Sundarbans wetlands (Islam and Gnauck, 2009; Rahman et al., 2010). Pressures such as oil pollution, metals and industrial waste, untreated sewage effluents degrade the environmental state (Iftekhar, 2004; Rahman et al., 2009; Islam et al., 2017; Ranjan et al., 2018). The water and soil salinity in Sundarbans area has increased due to low freshwater influx from upstream, which is the main cause of top-dying and die-back diseases of trees, especially the Sundari (*Heritiera fomes*) and Goran (*Ceriops decandra*), (Khan et al., 1994; Islam and Gnauck, 2009). The destruction of 9,500 ha of mangrove forest for shrimp cultivation has caused a dramatic decline in dominant mangrove species such as *Excoecaria agallocha* and *Xylocarpus mekongensis* (Iftekhar and Saenger, 2007) and widespread loss of biodiversity.

There is a loss of ecosystem services because of the degradation of the state of the wetland. The provisioning services that supply natural resources (mainly forest products) from the Sundarbans have declined since 1996 (Hossain et al., 2016). The reduction of fresh water flow from the river Ganges, the over-capture of wild shrimp post-larvae, the simultaneous bycatch of juvenile fish and crustaceans, salinity intrusion, uncontrolled tourism, inadequate planning and lack of ecosystem based management are effecting the biodiversity of Sundarbans ecosystems, hence affecting livelihood and food security of coastal community of Sundarbans region (Ahmed and Troell, 2010; Islam and Bhuiyan, 2018; Rahman et al., 2018; Islam, 2019). The collection of *Penaeus monodon* post-larvae destroys the larvae and juveniles of other shrimps, finfish, and other macro- zooplankton (Hoq et al., 2001). Therefore, the collection of shrimp PL from natural sources for aquaculture operations is directly influencing in reduction of wild fisheries (Primavera, 2006). The wild aquatic biodiversity (specially the benthic community) might be also in risk due to the toxicological effect of various chemicals used in shrimp ponds (Flaherty et al., 2000). The Sundarbans provide a regulating ecosystem service as a protective, natural boundary during cyclones, and in the last couple of decades it has protected over 10 million human lives from cyclonic storms (CEGIS, 2007). Clearing the mangrove for agriculture or aquaculture has a deleterious effect on this regulating ecosystem service.

With respect to indirect pressures, the population density and the susbsidence low-lying lands of the Sundarbans make them especially vulnerable to flood risk, whether from sea-level rise or cyclone storm surges (Brown et al., 2018). Like all coastal

[5]https://rsis.ramsar.org/ris/2370

wetlands, the Sundarbans regions is at greatest risk due to global climate change, which will influence the distribution and productivity of Sundarbans resources and aquatic biodiversity (Loucks et al., 2010). An additional problem is the increased salinity of the wetlands that affects the distribution and survival of the mangroves (Dasgupta et al., 2017). The loss of the coastal protection ecosystem service of mangroves would only increase the vulnerability of the whole system (Barbier, 2016).

Huang He (Yellow River) Delta Wetland, People's Republic of China

The coastal wetland of the delta of the Huang He (Yellow River) is near the city of Dongying, in the Shandong Province, China. The area of the wetland is 18,000 km^2 with a population of 5.2 million. The continental monsoon climate is warm and temperate (Li et al., 2014) with a mean annual temperature range from 11.7 to 12.6°C and ~200 frost-free days. The mean annual precipitation is 530–630 mm, with 70% of the rainfall during the wet, early summer from May to July (Li et al., 2014). The wetlands include various aquatic habitats such as rivers and channels, estuarine waters, coastal and salt lakes, ponds and reservoirs, shrimp and crab cultivation ponds (Cui et al., 2009). Migrating birds from northeast Asia and from the western Pacific rim use the wetland as an overwintering and breeding site.

The delta wetland is affected by huge pressures from several human activities especially agriculture, aquaculture of fish and petro-chemical industries (Li et al., 2014). Land-use changes have converted grassland and wilderness to cultivate salt-tolerant crops and grasses, clover, maize, cotton, and Chinese date. Shrimp ponds and saltpans have been constructed in large areas of the wetland (Zhang et al., 2011). There are also thousands of oil wells of the Shengli oilfield in the wetland.

These human activities exert direct pressures on the wetland. Intensive agriculture and the growth of urban areas have almost halved the area of natural wetlands from 1976 to 2008 (Chen et al., 2011). The construction of roads between the river and wetland has decreased the water supply to the wetland. The implementation of flow-sediment regulation has reduced connectivity and modified the hydrological conditions, hence the wetland landscape (Bai et al., 2012). River diversions for irrigation have altered the hydrology and decreased the river fluxes to the wetland (Ottinger et al., 2013). Over-extraction of deep groundwater causes land subsidence, which could accelerate local relative sea level rise, increasing the vulnerability to storm surge, flooding, saltwater intrusion and coastline erosion (Liu and Huang, 2013). Saline aquaculture may cause secondary salinization in the future. The oil exploration and flow-sediment regulation could cause contamination by petroleum hydrocarbon, polycyclic aromatic hydrocarbons, arsenic, and cadmium (Wang et al., 2009; Bai et al., 2012).

The state of the ecosystems in the wetland was seriously affected by the depleting water resources, degradation of soil quality, pollution, agricultural, and industrial activities (Zhang et al., 2016). Economic development has resulted in a decrease of the habitat for the red-crowned crane habitat by 5,935 ha from 1992 to 2008 (Wang et al., 2017). The coastal blue carbon storage in the delta has decreased by 10.2% during 1970–2010 due to land use change (Ma et al., 2019). Since 2000s, the implementation of a wetland restoration project and a water regulation scheme (WSRS) has significantly improved the structure and functions of the wetland ecosystem. However, the altered flux and grain-size composition of sediments due to the implementation of the WSRS may in turn aggravate elemental imbalance, impacting estuarine and coastal geochemical processes and the wider ecosystem (Li et al., 2019).

There are impacts on human welfare from the loss of wetland ecosystem services. The lost wetland area and the decreased water supply has degraded bird habitats. Construction of dams in the catchment perturbs sediment supply, making the wetland vulnerable to erosion. Houses and oil fields on the coast have been partly destroyed by storm surges and coastal erosion (Liu and Huang, 2013). The increased salinity level in groundwater and soils caused by seawater intrusion has decreased agricultural production. Potential heavy metals and oil contamination from agro-activities and oil exploration taint seafood and are public health issues.

Indirect pressures from sea-level rise also affect the Yellow River Delta wetlands. The topography is such that the river delta has changed position many times in the past but it is now constrained by engineering projects. The combined effects of these changes, subsidence and sea-level rise threaten the wetland habitat, highlighting the need for a social-ecological adaptive management plan (Zhang et al., 2016). Nevertheless, the Yellow River Delta is one of the few large deltas in the world that can offset Sea Level Rise by 2100 due to the huge sediment input from the Yellow River (Giosan et al., 2014).

Changjiang (Yangtze) Coastal Wetlands, People's Republic of China

The Changjiang (Yangtze) River, the longest river in China, discharges into the East China Sea in a temperate climate zone near Shanghai. The Changjiang coastal wetlands are estuarine and meso-tidal. The wetland, including the intertidal land and the areas shallower than 6 m water depth, occupies an area of some 3,050 km^2 at the beginning of the 21st century. There are saltmarshes, mudflats, and estuarine shoals (Li et al., 2009). The deltaic areas are extensive, with a total area of around 25,000 km^2, and they are densely populated, with more than 1,000 million inhabitants. Shanghai had a population of 22.3 million (Statutory City = SC) to 34 million (Functional Urban Area = FUA) in 2010 (OECD, 2015). In addition, there are other large cities in the Changjiang River delta, such as Nanjing (7.2 million SC to 11.7 million FUA) as well as Suzhou, Wuxi, Nantong, Changzhou, Yangzhou, Taizhou, Jiaxing, and Huzhou. The area is experiencing rapid changes in response to both natural processes (e.g., shoreline hydrodynamics and sediment dynamics) and human activities (e.g., land reclamation). The shoreline is often hit by typhoon induced storm surges, which has a trend of enhancing intensity in response to sea level rise and the warming climate (Gao et al., 2019).

Intensive human activities exert direct pressures on the Changjiang coastal wetlands. The most important ones include: dam construction that causes modifications to hydrology and sediment dynamics; rapid urbanization in the catchment and increased pollutant discharges; land reclamation along the estuarine shoreline; artificial introduction of the *Spartina alterniflora* species into the region; and projects for water resource utilization. More than 50,000 dams have been built in the Changjiang catchment, of which the Three Gorges Dam is the largest. As a result, the sediment discharge has been reduced by up to 70% since the 1980s (Yang et al., 2005) and the suspended sediment concentration of the river water in the downstream section is significantly reduced (Yang et al., 2018). Land-reclamation for the urban development of Shanghai and Pudong has been an important activity up to the 1990s.

These activities have resulted in pressures on the wetland. Foremost are pressures on the hydrological system and delivery of sediments. Although the total water discharge remains almost the same, the timing of the delivery has changed with an increase in the dry seasons but a decrease in flood seasons. The Changjiang River delta reached the limits to its expansion after the construction of the Three Gorges Dam (Gao, 2007), and shoreline recession is now occurring in some places. Water supply to Shanghai is a critical issue. The treatment of the water from local small rivers is costly, and the main body of the estuary is affected by saline water intrusion. To solve this problem, reservoirs to store Changjiang freshwater have been built on deltaic islands such as Chongming (the largest) and Jiuduansha (newly formed). The largest reservoir, Qingcaosha, is on Changxing Island. It was completed in 2010 and provides some 7.2 million m^3 of water per day, which is sufficient for 11 million people. Land reclamation has declined because the shoreline progradation has slowed down in response to the reduction of river sediment input. The rapid industrial, agricultural, and urban development has also resulted in increased pollutant discharges, including nutrients, heavy metals, pesticides and micro- plastic particles (Zhao et al., 2014). A non-indigenous species from North America, *Spartina alterniflora*, was artificially introduced into the wetland in the 1980s. It has expanded rapidly (Gao et al., 2014) since the 1990s and is now a dominant species in the Changjiang wetland (Li et al., 2009). The local nematodes and macrobenthic invertebrates have adapted to the *S. alterniflora* marsh, but the native marsh plants such *Scirpus mariqueter* and *P. australis* are affected (Chen et al., 2004, 2007; Zhao et al., 2009). *S. alterniflora* occupies a lower position in the tidal flat compared with the native species, so land reclamation is primarily of *Scirpus mariqueter* and *P. australis* marshes and these species have very limited space to expand outside the sea dykes. Ecological restoration has become a recent focus for regional development. Five Nature Reserves were established by the State Council in 2005 to protect the coastal wetlands, including the Chongming Dongtan National Nature Reserve. The Ramsar Convention listed it as a 'wetland of international importance' in 2002. New techniques and measures have been adopted to create wetland along the shorelines that are artificially modified. For example, underwater dykes are built parallel with the sea dykes to enhance the accretion rate and the area of intertidal zone, so that new marshes can be formed in front of the sea dykes.

The anthropogenic pressures from human activities have changed the state of the environment and ecosystem. Land reclamation, the reduction of riverine sediment input and decreasing deltaic shoreline growth have led to a large-scale decrease of coastal wetland area and loss of biodiversity. The wetland area in 2016 was less than half that of 15–20 years prior. The native saltmarshes are disappearing because of lack of space for expansion. Although some are conserved in the nature reserves, future morphological changes due to sediment accretion may eventually destroy these last refuges. The *S. alterniflora* marsh initially expanded, but now its area is also decreasing because of reclamation. If the wetland loss from reclamation is not compensated by the various ecological restoration schemes, future land use requirements for urban development will further reduce the area. Contamination and pollution of the estuary affect the ecosystem health. Water quality becomes a critical problem due to the enhanced input of heavy metals and organic chemicals (Cao et al., 2012). The contaminant concentrations are high in the wetland sediment of the estuary. The pressure from excessive loading of nitrogen from fertilizer, manure, sewage and aquaculture stimulate eutrophication, hence Harmful Algal Blooms and hypoxia events occur with an increased frequency of occurrence and spatial extent (Bianchi and Allison, 2009).

The present changes taking place in the wetlands have consequences on the ecosystem services. The loss of area decreases all the ecosystem services of the wetland such as coastal protection, denitrification and carbon sequestration. The wetland provided fish spawning grounds and a nursery feeding place for young fish. Thus, the life cycle of fish species in the 3 traditional fishing grounds near the Changjiang River mouth is affected. The loss of 71% loss of wetlands from 1990 to 2000 has caused an estimated loss of 62% of the total value of ecosystem services in the Chongming Dongtan Nature Reserve (Zhao et al., 2004). Although *S. alterniflora* marshes increased the primary productivity, some sea birds (e.g., *Charadriidae* and *Scolopacidae*) that need mudflats are affected, and the natives have insufficient space to grow (Ma et al., 2007, 2009; Li et al., 2009). How the ecosystem functioning is influenced by *S. alterniflora* needs further investigation.

Efforts have been made by the Shanghai government to reduce the negative effects of wetland changes on human welfare. One of the responses has been the establishment of nature reserves and areas for ecological restoration. The fishery catch is decreasing, partly because of the insufficient wetland space (Jin et al., 2007). The management response would be to maintain the stability of the native saltmarshes and simultaneously to use the *S. alterniflora* marsh as an ecological restoration tool. Soft engineering could be adopted to increase the space for wetland development, and pollutant input should be controlled to improve the water quality of the estuary. For these purposes, efficient management is required, in the context of the long-term climate and sea level factors and the short- term anthropogenic influences (Cheng and Chen, 2016; Ding, 2016).

Indirect pressures also affect the Changjiang (Yangtze), where a megacity is built on a low-lying, sinking deltaic wetland, in a zone prone to supertyphoons that is also very vulnerable to flooding and storm surges (Cui et al., 2015).

Mekong Delta Wetlands, Vietnam

The Mekong delta in the south of Vietnam includes 13 provinces occupying 40,600 km^2 and is home to about 18 million inhabitants. Here, the Mekong drains via nine branches into the South China Sea. The subtropical delta undergoes distinct dry and rainy seasons. The delta includes temporarily inundated sandbars and mudflats, as well as several types of wetlands. These are: salt marshes; coastal mangrove ecosystems, such as Ca Mau National Park, Soc Trang, and Bac Lieu; reed wetlands, such as Ha Thien Plain and Tram Chim National Park (Plain of Reeds); and peat swamp wetlands, such as U Minh Thuong National Park. Several National Parks and nature reserves have been established in the Mekong Delta wetland. The area of mangrove in Mekong delta coast is 100,000 Ha. These mangrove forests are especially biodiverse with 98 species of trees, 36 species of mammal, 182 species of birds, 34 species of reptile, 6 species of amphibians, and 260 species of fishes. The main cities in the delta are Can Tho, Ho Chi Minh, Ca Mau, Bac Lieu, My Tho, Tra Vinh, Soc Trang cities and these are growing rapidly in population and area.

The main human activity is rice crop cultivation in rice paddy fields that are human controlled wetlands. Some are protected by complex diking systems and are intensively farmed, yielding 2, 3, and recently up to 4 harvests per year. Some paddies are under less intensive use and used as fishing grounds during the flooding season (Duy et al., 2015). Illegal clear cutting of mangroves continues in strictly managed and protected regions, such as the famous Can Gio Mangrove Biosphere reserve (Kuenzer and Vo, 2013). These illegal activities are hard to monitor and law enforcement is still weak. The overall socio-economic transformation, urbanization (e.g., the Delta Cities of Ca Mau, Soc Trang, Ben Tre, and especially Can Tho, just to name a few), and industrial expansion (petrochemical and coal/fuel) ongoing in the delta (e.g., sea-food processing and agro-industry especially around Can Tho and Long Anh) lead to a general overexploitation of resources, including overfishing (*Pangasius*, sting ray, catfish, and elephant fish to name only a few), unsustainable sand dredging, and the sale of topsoil. Economic activities are rice cultivation, animal husbandry (chicken, ducks, pigs, and cows) and aquaculture (especially shrimp and *Pangasius*). Extensive mangrove areas have been destroyed by the rapid expansion of coastal aquaculture (Kuenzer and Vo, 2013; Vo et al., 2015).

Many human activities exert direct pressures on these fragile wetlands. The construction of dykes, dams, urban, and industrial areas affect the hydrology and connectivity of the wetland. Coastal engineering activities are often driven by the hope for economic benefits and are executed for ministerial line agencies and related companies. These result in the ongoing closure of Mekong River main stem and tributary channels, decreasing connectivity. Wetland connectivity is decreased as the network of river branches and canals in the delta are increasingly obstructed by dykes, levees and sluice gates. The same applies to terrestrial ecosystem connectivity, which is decreasing due to fractionation by highways and roads. These are aggravated by the decrease in sediment loads from the upstream areas (Kuenzer et al., 2013).

These pressures affect the state of the wetlands, so that flora and fauna biodiversity is declining. The state of the water quality affected by fertilizer, pesticides and endocrine disruptors from rice farming, vegetable cultivation and animal husbandry. Furthermore, chemicals and antibiotics used in aquaculture ponds contaminate groundwater, soils, and the food chain (Ottinger et al., 2016). Elevated levels of pesticide have resulted in water pollution with traces detectable in ground, surface and even locally filled bottled drinking water (Toan et al., 2013). The area of area of mangrove in Mekong delta coast is decreasing with rate of 5% per year induced by human activities.

The deteriorating state of the wetland affects the delivery of ecosystem services. Healthy mangroves provide: provisioning services, e.g., fire wood, construction wood, medicinal products, honey, and fish; supporting, e.g., breeding grounds; protecting services, e.g., erosion and storm control; regulating services, e.g., water filtration and buffering capacities; cultural and recreational services, e.g., resting, eco-tourism, and spiritual activities (Vo et al., 2012, 2015; Kuenzer et al., 2013). The conversion of natural reed wetlands decreases their delivery of ecosystem services. These include a loss of provisioning ecosystem services, e.g., roofing and weaving material, fish supply; ecosystem services, e.g., water filtration and cleansing; and cultural- recreational ecosystem services (Vo et al., 2012, 2015; Kuenzer et al., 2013). Natural mangrove belts acted as barriers against high tides, storm surges and general sea level rise. The loss of these mangroves results in increased salt-water intrusion into the aquifers and soils of the delta.

The activities and related pressures exerted on the Mekong wetland have led to partially irreversible impacts on human welfare, especially affecting the rural poor who depend on natural resources for their livelihood. Artisanal fishers report a loss of fish species and decreased catch landings. Salt-water intrusion affects small-scale rice farmers. The rural population often drink untreated ground and surface water and suffer from intestinal problems and other health issues. Boat-dependent mobility is impaired by an increased regulation of water ways. Many typical, traditional lifestyles are likely to disappear in the next years, including floating markets, artisanal clam collection and fisheries, small-scale water transportation and complex, sustainable farming approaches, such as the multispecies duck-rice-vegetable agriculture. This will entail a loss of cultural values and an overall decrease of the unique character of the Mekong delta wetland.

Climate change and related sea level rise are the main causes of indirect pressures in the wetland, including coastal erosion and salt-water intrusion. Managing flood risks in the Mekong Delta is a serious challenge in the context of climate change and socioeconomic developments (Hoang et al., 2018).

Coastal Wetlands of Oceania

Mangrove Forests of the Gulf of Papua, Papua New Guinea

The Central Highlands of Papua New Guinea are drained by the rivers Purari, Kikori, and Fly that pass through regions of extremely high rainfall and into a complex of tide- and river-dominated estuaries, delivering (384 Mt a^{-1} of sediment and 470–690 km^3 water into the Gulf of Papua) (Aller et al., 2008). The wetlands are dynamic, with accretion and erosion occurring simultaneously indifferent parts of the deltas, with delta-front erosion of up to 43 m per year over the period 1973–2002 (Shearman, 2010). There are 3,780 km^2 of wetland mangrove on the Gulf of Papua (Bryan et al., 2015), the largest, virtually pristine mangrove forests in the Asia-Pacific Region (Cragg, 1983), and particularly diverse (Duke et al., 1998). Pioneering species, e.g., *Sonneratia lanceolata* dominate accreting shorelines. Mature forests with canopies of up to 30 m are dominated by *Rhizophora* and *Bruguiera*. Transitional forests occur at the boundary with freshwater habitats (Cragg, 1983). There are large areas of pure *Nypa* palm stands in low salinity parts of the deltas. The mangrove wetlands are nursery grounds for fish, crabs, and prawns, (Frusher, 1983). Offshore of the mangroves, there are feeding and breeding grounds for commercially harvested prawns (Gwyther, 1983), and rock lobster (Dennis et al., 2001).

Human activities in the immediate vicinity of this huge mangrove are remarkably small. Four to five hundred years ago, traditional trading Hiri voyages traversed the Gulf (Barker et al., 2015; David et al., 2015). There are no urban areas but some villages on sand bars at the seaward edge or just inland of the mangrove rather than within the mangrove (Cragg, 1983). The population is low, approximately 107,000, and relies on subsistence utilization of a range of mangrove resources: mangrove wood for poles for house construction, fuel and tools; edible mangrove fruits and palm hearts; medicinal uses and dyes (Liebezeit and Rau, 2006); thatching and basket weaving using *Nypa fruticans* fronds. Offshore there is one of the most commercially important fisheries in Papua New Guinea – the demersal prawn trawl fishery that gains an income of (US$3M per year through a local fishing fleet) (Kompas and Kuk, 2008; White et al., 2019).

The low population and subsistence utilization of mangrove resources represent only a low, local direct pressure at present. The estimated use of fuel wood is only 6.4 kg $person^{-1}$ d^{-1} and construction wood at 0.6 m^{-3} $household^{-1}$ a^{-1} (Page et al., 2016). The main pressures are from human activities elsewhere: the large, commercial logging concessions inland of the wetland (Bryan et al., 2015); the oil pipelines from the highlands at Kutubu through the mangrove forests to the Kumul Marine Terminal offshore; and the huge Ok Tedi open-pit gold and copper mine is in the watershed of the river Fly. Mine tailings have increased the sediment load at peak rainfall (Shearman et al., 2013), transporting copper in dissolved and particulate form (Bolton, 2009).

Direct pressures may increase as the area is further developed in the future. Logging inland of the mangroves is likely to increase sediment input to the wetland and may require construction of coastal log-loading facilities. The potential commercial exploitation of ethanol production from tapping *Nypa* syrup has been explored but not exploited. There have been evaluations of potential for pulp wood and wood chips, but the soft sediments make harvesting problematic. Bark from *Rhizophora* trees was used to produce tannin extract (cutch) in a factory that operated between 1954 and 1957, but this activity ceased due to competition from alternatives to cutch (Percival and Womersley, 1975). The construction of a hydroelectric dam on the Purari River (Wabo) has not yet taken place. A multidisciplinary environmental impact study of plans predicted a reduction in sediment load and increased predation on prawn larvae, but no significant effects on delta fertility (Cragg, 1983). A project to exploit natural gas envisages an additional pipeline from the highlands into the Gulf.

The effect of these pressures may affect the state of the wetland and will entail a loss of ecosystem services such as supporting services, e.g., the high primary production and the nursery function for fish and crustaceans. Unusually high bacterial decomposition rates of sediment organic carbon leads to close benthic-pelagic coupling supporting the Gulf of Papua prawn fishery (Alongi, 1995). The regulating services of the mangrove forests help to stabilize the sedimentation and erosion of the delta. Furthermore, the sequestration of carbon by the Papua New Guinea mangroves is a globally significant ecosystem service: 2.9% of global carbon stocks in mangroves is estimated to be in Papua New Guinea mangrove forests (Hamilton and Friess, 2018). This is the 5th most important mangrove ecosystem in the world, in terms of carbon stocks once the additional offshore sequestration is combined. Dead plant matter is rich in the lignocellulose complex, which is more resistant to break down than marine-generated particulate organic matter, particularly under anoxic conditions in sediments (Cragg et al., 2020). Part of the forest detritus and almost all subterranean roots are retained, accounting for the exceptionally high carbon sequestration capacity of mangrove wetlands, particularly in New Guinea (Sasmito et al., 2020). An estimated 1.7 × 109 kg a^{-1} of the organic carbon, derived from the vascular plants and plankton of the mangroves, is exported through the Fly delta to the Gulf of Papua and the Coral Sea Robertson and (Alongi, 1995). The residence-time of soil organic matter is longer in marine sediments (Goni et al., 2006), so 13–27% of the organic matter accumulates long-term offshore (Aller et al., 2008).

Indirect pressures from climate change may alter rainfall patterns, hydrology, intensity and frequency of storms affecting the delta region. Sea level rise is interacting with subsidence of the delta sediments in affecting mangrove distribution (Shearman, 2010).

SUMMARY OF THE MAIN DIRECT PRESSURES ON COASTAL WETLANDS AND CONSEQUENCES

The Millennium Ecosystem Assessment [MEA] (2005) identified that human population growth and economic development lead to the degradation and loss of coastal wetlands. Anthropogenic degradation and loss of wetland was mainly due to land-use changes, construction of infrastructure, water extraction, eutrophication and pollution, overharvesting of commercial species and overexploitation of natural resources, as well as the introduction of non-indigenous species (Millennium Ecosystem Assessment [MEA], 2005).

We identified that the most important human activities exerting anthropogenic pressures on coastal wetlands can be placed in 2 groups: (i) land reclamation and conversion to agriculture, aquaculture, ports, and urban areas; and, (ii) construction of dams, dykes, polders, drainage channels, dredging that modify the natural hydrology, connectivity, and sedimentology. Thus, the main anthropogenic pressures are land-use change and changes in hydrology, as well as connectivity. The main anthropogenic pressures on the coastal wetlands included in this overview are summarized in **Supplementary Table S1**. The relationship between these human activities in or around coastal wetlands and resulting anthropogenic pressures on the environment and ecosystem are summarized in **Supplementary Table S2**.

The anthropogenic pressures (direct and indirect) alter the dynamics of ecosystem function of the coastal wetlands. This affects the natural resilience of the systems and increases their vulnerability. This overview is summarized in **Figure 2**. Damming in the catchment disrupts the sediment fluxes to coastal wetlands. This leads to erosion and subsidence, (Syvitski and Saito, 2007; Syvitski et al., 2009). Wetlands are dynamic systems, illustrated by the 'wandering mouths,' of deltas, such as the Huang He (Yellow River) and Mississippi. In the case of dynamic, coastal lagoons such as the Ria Formosa, inlets open and close. Humans try to stabilize the position of river mouths and inlets for shipping, and channels are dredged to accommodate ships of increasing size. These modifications in the catchment and at the mouth cause changes to the sedimentology and hydrology. Over-extraction of water for agriculture is another common feature that changes the freshwater-saline water balance, so that wetlands that naturally vary from freshwater to saline become increasingly saline. The linked coastal wetlands of Bizerte lagoon and Ichkeul lake illustrate the dramatic effects of loss in connectivity. Changes in hydrology, flow rates and timing of freshwater inputs to estuarine wetlands affects the flora of the wetlands, whether it is seagrass, saltmarsh or mangrove. The vegetation is fundamental to the ecosystem services of the coastal wetlands and have dramatic impacts on human welfare, for example, provisioning services such as fish catch (Kennedy and Barbier, 2016). A holistic understanding of the interlinked hydrology, sedimentology and biology is fundamental for the management of estuarine systems such as coastal wetlands (Alber, 2002). This is particularly important for the assessment of diffuse sources, for example nutrient pressures from farming activities such as fertilizer application.

Another important and widespread anthropogenic pressure is the point source discharge of effluents by industry, farms and domestic sewage shown in **Supplementary Table S3**. Effluents, especially from industries, can contaminate and pollute wetlands, while eutrophication may result from the discharge of effluents that are rich in organic matter and nutrients shown in **Supplementary Table S4**.

Most of the anthropogenic pressures identified in this overview correspond to the types I to N of "anthropogenic hazards" from the typology of Elliott et al. (2014) and are summarized in **Supplementary Table S5**. The value of ecosystem services of a wetland, e.g., the mangroves of Papua Guinea can be globally significant, but the lack of understanding about the value of coastal wetland ecosystem services (Newton et al., 2018) continues to contribute to their degradation. Externalities and the long-term value of ecosystem services are often ignored, for short term, economic gain and/or development. Even when the knowledge exists, it may be disregarded, as in the case of the dredge spoil dumping onto the Abbot Point and Caley Valley wetlands in Australia, which is a habitat of more than 40,000 water-birds (Australian Marine Conservation Society, 2014), despite lessons learned in the Mississippi and other coastal wetlands.

Cumulative pressures change both the state of the environment (e.g., anoxia) and the state of the ecosystem (e.g., loss of saltmarsh, mangrove, and seagrasses). Coastal wetlands provide ecosystem services, such as regulatory services for water quality, coastal protection, carbon sequestration and denitrification, as well as supporting services such as habitat for a wide variety of fauna. The loss of vegetation of the coastal wetlands, represents a loss of a wide range of ecosystem services. There is a decrease in provisioning ecosystem services, e.g., timber from mangroves, seafood yield from the wetland, and also from the adjacent sea; supporting ecosystem services, e.g., oxygenation from seagrasses, nursery services, denitrification, carbon sequestration, and habitat for migrating wildfowl; regulating ecosystem services, e.g., water purification and natural hazard protection, e.g., floods, storm surge, and tsunamis cultural ecosystem services, such as loss of traditional lifestyles and artisanal fisheries.

However, there are some 'bright-spots.' Awareness and recognition of the value and importance of coastal wetlands is improving, and some nations are now developing ambitious management plans. In particular, the protecting ecosystem service of coastal wetlands gained recognition after the 2004 Banda Aceh tsunami (Spalding et al., 2014). Some of these efforts are transdisciplinary and bring together scientists, policy makers and environmental managers, as in the case of the Huang He (Yellow River) delta and coastal wetlands management plan (Wohlfart et al., 2016).

This is also the case for mangrove wetlands. Papua New Guinea is implementing the UN-REDD+ mechanism (the United Nations Reducing Emissions from Deforestation and forest Degradation plus Sustainable Forest Management, Conservation of Forests, Enhancement of carbon sinks), (Grussu et al., 2014). This aims to decrease emissions (e.g., from slash and burn deforestation) and enhance forest carbon-sequestration. The remote-sensing methodology detailed by Hamilton and Friess (2018), could be used for monitoring to support payment for ecosystem services such as REDD+.

The emphasis of this article has been direct pressures rather indirect, but both will affect the future of wetlands. Coastal

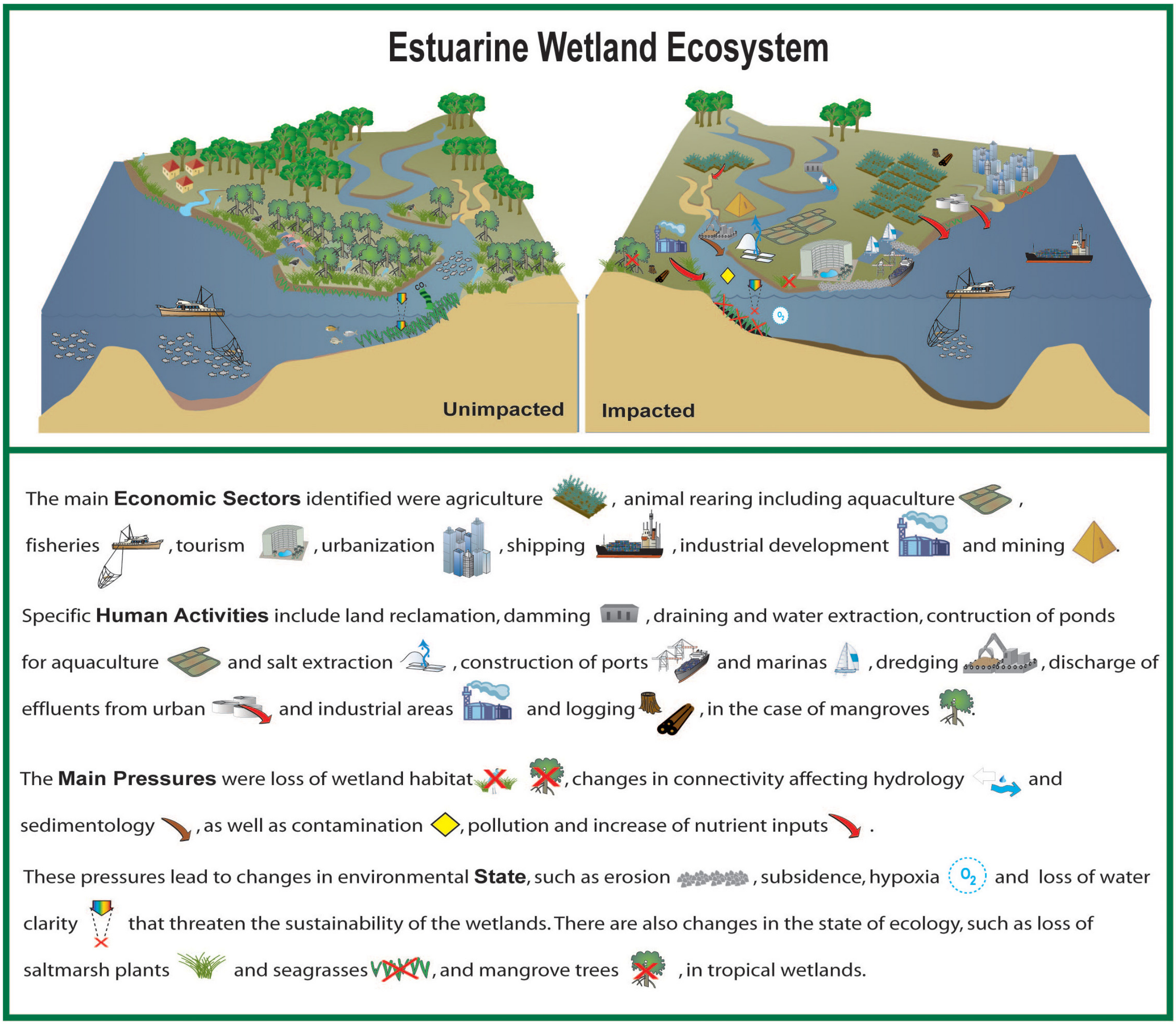

FIGURE 2 | Conceptual diagram summarising the findings, showing an estuarine wetland ecosystem in an unimpacted and impacted state. Symbols are courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (https://www.umces.edu/ian/search). (Credit: Sonia Cristina).

wetlands, mangrove forests, saltmarshes and seagrasses are particularly effective at sequestering carbon, as illustrated by the Papua New Guinea mangroves. It is important to consider both historical and contemporary environmental factors to better understand this regulating ecosystem service of coastal wetlands (Witman, 2017). Marshes, seagrass meadows, and mangroves are some of the ecosystems that sequester carbon most efficiently and store large amounts of carbon dioxide as "blue" carbon (Lovelock and Duarte, 2019; Cragg et al., 2020). Conservation of coastal wetlands can therefore contribute to the mitigation and adaptation to climate change (Neubauer and Verhoeven, 2019; Serrano et al., 2019). However, the Blue Carbon potential of coastal wetland restoration varies with inundation and rainfall (Negandhi et al., 2019) as well as management approaches (O'Connor et al., 2020). Degraded coastal wetlands could potentially be re-naturalized to promote an increase in carbon sequestration, as well as additional and valuable ecosystem services, such as denitrification and halophyte cultivation, for example *Salicornia.*

We conclude this overview by highlighting the importance of coastal wetlands both ecologically and economically. They provide an important buffer between land and sea in the context of sea-level rise and climate change. They are important cogs in the biogeochemical cycles, providing exceptional services such as denitrification and carbon sequestration. The ecosystem services that they provide are consistently underestimated leading to irreversible land-use changes that ultimately destroy the wetlands and impact human welfare. The conservation and ecosystem-based management of coastal wetlands will contribute to several of the United Nations Sustainable Development Goals, SDG, in particular, 2, 3, 6, 12, 13, 14, 15 but also 1, 4, 8, 11.

AUTHOR CONTRIBUTIONS

AN wrote several sections and edited the entire manuscript. JI wrote Mida Creek, mangroves and edited the entire manuscript for submission. SoC wrote Ria Formosa lagoon and produced the conceptual diagram. GP wrote the Bahia Blanca estuary. RT wrote the Mississippi River. DA wrote the Sunderbans mangroves. SiC wrote the Gulf of Papua mangroves. YLi, CT, YLu, and HZ wrote the Yellow River delta wetland. RR wrote Vembanad Lake. DF wrote Polar coastal wetlands and Mackenzie-Beaufort coastal wetland. CS and BB wrote Bizerte lagoon and Ichkeul Lake. SG wrote Yangtze coastal wetlands. RP wrote Venice lagoon. HK and DT wrote Chesapeake Bay. NN and CK wrote Mekong delta wetlands. AB and RL wrote Malanza coastal lagoon.

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SUPPLEMENTARY MATERIAL

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REFERENCES

Abecasis, D., Bentes, L., and Erzini, K. (2009). Home range, residency and movements of *Diplodus sargus* and *Diplodus vulgaris* in a coastal lagoon: connectivity between nursery and adult habitats. *Estuar. Coast. Shelf S.* 85, 525–529. doi: 10.1016/j.ecss.2009.09.001

Ahmad, I. U., Greenwood, C. J., Barlow, A. C. D., Islam, M. A., Hossain, A. N. M., Khan, M. M. H., et al. (2009). *Bangladesh Tiger Plan 2009–2017*. Dhaka: Bangladesh Forest Department, 46.

Ahmed, N., and Troell, M. (2010). Fishing for prawn larvae in Bangladesh: an important coastal livelihood causing negative effects on the environment. *Ambio* 39, 20–29. doi: 10.1007/s13280-009-0002-y

Alber, M. A. (2002). Conceptual model of estuarine freshwater inflow management. *Estuaries* 25, 1246–1261. doi: 10.1007/bf02692222

Aller, R. C., Blair, N. E., and Brunskill, G. J. (2008). Early diagenetic cycling, incineration, and burial of sedimentary organic carbon in the central Gulf of Papua (Papua New Guinea). *J. Geophys. Res-Earth.* 113:F01S09.

Almeida, C., and Soares, F. (2012). Microbiological monitoring of bivalves from the Ria Formosa Lagoon (south coast of Portugal): a 20 years of sanitary survey. *Mar. Pollut. Bull.* 64, 252–262. doi: 10.1016/j.marpolbul.2011.11.025

Almeida, L. P., Vousdoukas, M. V., Ferreira, Ó, Rodrigues, B. A., and Matias, A. (2012). Thresholds for storm impacts on an exposed sandy coastal area in southern Portugal. *Geomorphology* 143–144, 3–12. doi: 10.1016/j.geomorph.2011.04.047

Alongi, D. M. (1995). Decomposition and recycling of organic matter in muds of the Gulf of Papua, northern Coral Sea. *Cont. Shelf Res.* 15, 1319–1337. doi: 10.1016/0278-4343(94)00087-4

Alongi, D. M. (2014). Carbon cycling and storage in mangrove forests. *Annu. Ver. Mar. Sci.* 6, 195–219. doi: 10.1146/annurev-marine-010213-135020

Aníbal, J., Gomes, A., Mendes, I., and Moura, D. (eds) (2019). *Ria Formosa: Challenges of a Coastal Lagoon in a Changing Environment*, 1st Edn. Faro: University of Algarve.

Anon (2016). *Ramsar Sites in Kerala*. Available online at: https://www.researchgate.net/file.PostFileLoader.html?id=57879561ed99e1c4f36f3857&assetKey=AS%3A383745195560961%401468503392919 (accessed August 8, 2016).

Arnaud-Fassetta, G., Bertrand, F., Costa, S., and Davidson, R. (2006). The western lagoon marshes of the Ria Formosa (Southern Portugal): sediment-vegetation dynamics, long-term to short-term changes and perspective. *Cont. Shelf Res.* 26, 363–384. doi: 10.1016/j.csr.2005.12.008

Asha, C. V., Cleetus, R., Suson, P. S., and Bijoy Nandan, S. (2015). Environmental factors structuring the fish assemblage distribution and production potential in Vembanad estuarine system. *India. Int. J. Mar. Sci.* 5, 1–13.

Australian Marine Conservation Society (2014). *The Values of the Caley Valley Wetlands*. South Brisbane QLD: Australian Marine Conservation Society.

Bahar, M. M., and Reza, S. M. (2010). Hydrochemical characteristics and quality assessment of shallow groundwater in a coastal area of Southwest Bangladesh. *Environ. Earth Sci.* 61, 1065–1073. doi: 10.1007/s12665-009-0427-4

Bai, J. H., Xiao, R., Zhang, K. J., and Gao, H. F. (2012). Arsenic and heavy metal pollution in wetland soils from tidal freshwater and Saltmarshes before and after the flow-sediment regulation regime in the Yellow River Delta. China. *J. Hydrol.* 450, 244–253. doi: 10.1016/j.jhydrol.2012.05.006

Barber, D. G., Lukovich, J. V., Keogak, J., Baryluk, S., Fortier, L., and Henry, G. H. R. (2008). The changing climate of the Arctic. *Arctic* 61, 7–26.

Barbier, E. B. (2016). The protective service of mangrove ecosystems: a review of valuation methods. *Mar. Pollut. Bull.* 109, 676–681. doi: 10.1016/j.marpolbul.2016.01.033

Barbier, E. B., Koch, E. W., Silliman, B. R., Hacker, S. D., Wolanski, E., Primavera, J., et al. (2008). Coastal ecosystem-based management with nonlinear ecological functions and values. *Science* 319, 321–323.

Barkay, T., Kroer, N., and Pulain, A. J. (2011). Some like it cold: microbial transformations of mercury in polar regions. *Polar Res.* 30:15469. doi: 10.3402/polar.v30i0.15469

Barker, B., Lamb, L., David, B., Skelly, R., and Korokai, K. (2015). Dating of *in situ* longhouse (dubu daima) posts in the Kikori River delta: refining chronologies of island village occupation in the lower Kikori River delta, Papua New Guinea. *Quatern. Int.* 385, 27–38. doi: 10.1016/j.quaint.2015.04.003

Bebianno, M. J. (1997). Effect of pollutants in the Ria Formosa lagoon, Portugal. *Sci. Total Environ.* 171, 107–115. doi: 10.1016/0048-9697(95)04672-9

Bebianno, M. J., Patrícia, P., Ângela, S., Belisandra, L., and Alice, N. (2019). *Ria Formosa: Challenges of a Coastal Lagoon in a Changing Environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Faro: University of Algarve), 109–123.

Becket, L. H., Andrew, H. B., and Michael, S. K. (2016). Tidal marshes are not keeping up with sea-level rise. *PLoS One* 11:e0159753. doi: 10.1371/journal.pone.0159753

Béjaoui, B., Ferjani, D., Zaaboub, N., and Chapelle, A. (2010). Seasonal hydrobiological characterization of the Bizerte Lagoon (Tunisia). *J. Water Sci.* 23, 215–232.

Béjaoui, B., Harzallah, A., Moussa, M., Chapelle, A., and Solidoro, C. (2008). Analysis of hydrobiological pattern in the Bizerte lagoon (Tunisia). *Estuar. Coast Shelf. Sci.* 80, 121–129. doi: 10.1016/j.ecss.2008.07.011

Ben Omrane, B., Elbour, I., Belhassen, M., Mraouna, M., Mejri, R., Béjaoui, S., et al. (2010). Distribution spatiale des Vibrionaceae dans les sédiments lagunaires (la lagune de Bizerte, Nord Tunisie). *Rev. Microbiol. Indus. San. Environ.* 4, 83–94.

Ben Rejeb-Jenhani, A., Kartas, F., Le moalle, J., and Amblard, C. (1991). Phytoplancton du lac Ichkeul (Tunisie): composition, structure et activité photosynthétique. *Int. Revue Ges. Hydrobiol.* 75, 111–127.

Bianchi, T. S., and Allison, M. A. (2009). Large-river delta-front estuaries as natural "recorders" of global environmental change. *Proc. Natl. Acad. Sci. U.S.A.* 106, 8085–8092. doi: 10.1073/pnas.0812878106

Binder, C. R., Hinkel, J., Bots, P. W. G., and Pahl-Wostl, C. (2013). Comparison of frameworks for analyzing social- ecological systems. *Ecol. Soc.* 18, 26.

Biswas, S. R., Choudhury, J. K., Nishat, A., and Rahman, M. (2007). Do invasive plants threaten the Sundarbans mangrove forest of Bangladesh? *J. Forest. Ecol. Manag.* 245, 1–9. doi: 10.1016/j.foreco.2007.02.011

Blasco, S., Bennett, R., Brent, T., Burton, M., Campbell, P., Carr, E., et al. (2013). *2010 State of Knowledge: Beaufort Sea Seabed Geohazards Associated with Offshore Hydrocarbon Development*. Ottawa: Natural Resources Canada, 340. doi: 10.4095/292616

Bolton, B. R. (2009). "The Fly River, Papua New Guinea: environmental studies in an impacted tropical river system," in *Developments in Environmental Science*, Vol. 9, ed. B. R. Bolton (Amsterdam: Elsevier), 617.

Botté, S. E., Freije, R. H., and Marcovecchio, J. E. (2007). Dissolved heavy metal (Cd. Pb. Cr. Ni) concentrations in surface water and porewater from Bahía Blanca Estuary Tidal Flats. *B Environ. Contam. Tox.* 79, 415–421. doi: 10.1007/s00128-007-9231-6

Bousquet, B. (1988). *Rapport de Mission du Parc National de l'Ichkeul et autres aires protegees de Tunisie septentrionale et centrale*. Paris: MAB-UNESCO et Fonds du Patrimoine Mondial.

Brendel, A. S., Dutto, M. S., Menéndez, M. C., Huamantinco Cisneros, M. A., and Piccolo, M. C. (2017). Wind pattern change along a period of coastal occurrence variation of a stinging medusa on a SW Atlantic beach. *Anu. Inst. Geociencias* 40, 303–315. doi: 10.11137/2017_3_303_315

Briant, N., Bancon-Montigny, C., Elbaz-Poulichet, F., Freydier, R., Delpoux, S., and Cossa, D. (2013). Trace elements in the sediments of a large Mediterranean marina (Port Camargue, France): levels and contamination history. *Mar. Pollut. Bull.* 73, 78–85. doi: 10.1016/j.marpolbul.2013.05.038

Brito, A. C., Newton, A., Tett, P., and Fernandes, T. F. (2012). How will shallow coastal lagoons respond to climate change? A modelling investigation. *Estuar. Coast. Shelf Sci.* 112, 98–104. doi: 10.1016/j.ecss.2011.09.002

Brown, S., Nicholls, R. J., Lázár, A. N., Hornby, D. D., Hill, C., Harza, S.,et al. (2018). What are the implications of sea-level rise for a 1. 5, 2, and 3°C rise in global mean temperatures in the Ganges-Brahmaputra-Meghna, and other vulnerable deltas? *Reg. Environ. Change* 18, 1829–1842. doi: 10.1007/s10113-018-1311-0

Bryan, J., Shearman, P., Aoro, G., Wavine, F., and Zetty, J. (2015). "The current state of PNG's forests and changes between 2002 and 2014," in *The State of the Forests of Papua New Guinea 2014: Measuring change over the period 2002-2014*, eds J. E. Bryan and P. L. Shearman (Port Moresby: University of Papua New Guinea), 7–42.

Bulseco, A. N., Giblin, A. E., Tucker, J., Murphy, A. E., Sanderman, J., Hiller-Bittrolff, K., et al. (2019). Nitrate addition stimulates microbial decomposition of organic matter in salt marsh sediments. *Glob. Change Biol.* 25, 3224–3241. doi: 10.1111/gcb.14726

Bureau Central d'Etudes pour les Equipements d'Outre-Mer [BCEOM], FresinusConsult, CESalzgitter, and STUDI (1995). *(Etude)pour la sauvegarde du Parc National de l'Ichkeul. Rapport de 3èmepartie: Mesures et études spécifiques/Ministère de l'Environnement et de l'Aménagement duTerritoire*. Tunis: ANPE, 421.

Burn, C. R., and Kokelj, S. V. (2009). The environment and permafrost of the Mackenzie Delta area. *Permafrost Periglac.* 20, 83–105. doi: 10.1002/ppp.655

Byers, M. (2016). *Arctic Oil: Canada's Chance to Get it Right.* Social Sciences and Humanities Research Council, Canada. Knowledge Synthesis Report, 57. Available online at: https://byers.typepad.com/files/michaelbyers_knowledgesynthesisreport_arcticoil_18may2016.pdf (accessed May 12, 2020).

Cabaço, S., Machás, R., and Santos, R. (2009). Individual and population plasticity of the seagrass *Zostera noltii* along a vertical intertidal gradient. *Estuar. Coast. Shelf S.* 82, 301–308. doi: 10.1016/j.ecss.2009.01.020

Cahoon, D. R. (2007). *Factors Affecting Coastal Wetland Loss and Restoration Synthesis of U.S. Geological Survey Circular 1316*, ed. S. W. Phillips (Reston, VA: U.S. Geological Survey), 50–54.

Canonico, G. C., Arthington, A., McCrary, J. K., and Thieme, M. L. (2005). The effects of introduced tilapias on native biodiversity. *Aquatic Conserv.* 15, 463–483. doi: 10.1002/aqc.699

Cao, Z., Liu, X., Zhang, X., Chen, L., Liu, S., and Hu, Y. (2012). Short-term effects of diesel fuel on rhizosphere microbial community structure of native plants in Yangtze estuarine wetland. *Environ. Sci. Pollut. R.* 19, 2179–2185. doi: 10.1007/s11356-011-0720-0

Carrasco, A. R., and Matias, A. (2019). "Backbarrier shores along the Ria Formosa Lagoon," in *Ria Formosa: Challenges of a Coastal Lagoon in a Changing Environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Faro: University of Algarve), 17–28.

Casagranda, C., Dridi, M. S., and Francois Boudouresque, C. (2006). Abundance, population structure and production of macro-invertebrate shredders in a Mediterranean brackish lagoon, Lake Ichkeul, Tunisia. *Estuar. Coast. Shelf S.* 66, 437–446. doi: 10.1016/j.ecss.2005.10.005

CEGIS (2007). *Effect of Cyclone Sidr on the Sundarbans a Preliminary Assessment*. Dhaka: Center for Environmental and Geographic Information Services CEGIS, 1–8.

Ceia, F. R., Patrício, J., Marques, J. C., and Dias, J. M. (2010). Coastal vulnerability in barrier islands: the high risk areas of the Ria Formosa (Portugal) system. *Ocean Coast. Manage.* 53, 478–486. doi: 10.1016/j.ocecoaman.2010.06.004

Chen, H., Li, B., Hu, J., Chen, J., and Wu, J. (2007). Effects of *Spartina alterniflora* invasion on benthic nematode communities in the Yangtze Estuary. *Mar. Ecol. Prog. Ser.* 336, 99–110. doi: 10.3354/meps336099

Chen, J., Wang, S. Y., and Mao, Z. P. (2011). Monitoring wetland changes in Yellow River Delta by remote sensing during 1976-2008. *Prog. Geog.* 30, 585–592.

Chen, Z., Li, B., Zhong, Y., and Chen, J. (2004). Local competitive effects of introduced *Spartina alterniflora* on *Scirpus mariqueter* at Dongtan of Chongming Island, the Yangtze River estuary and their potential ecological consequences. *Hydrobiologia* 528, 99–106. doi: 10.1007/s10750-004-1888-9

Cheng, H. Q., and Chen, J. Y. (2016). *Influences of Sea Level Rise on the Changjiang Estuary*. Beijing: China. Science Press, 414.

Chesapeake Bay Program [CBP] (2012). *Wetlands*. Available online at: http://chesapeakebay.net/issues/issue/wetlands (accessed August 28, 2016).

Church, J. A., Clark, P. U., Cazenave, A., Gregory, J. M., Jevrejeva, S., Levermann, A., et al. (2014). Sea-level rise by 2100. *Science* 342:1445.

Coastal Protection Restoration Authority [CPRA] (2017). *Louisiana's Comprehensive Master Plan for a Sustainable Coast 2017*. Baton Rouge: CPRA.

Costanza, R., Farber, S. C., and Maxwell, J. (1989). Valuation and management of wetland ecosystems. *Ecol. Econ.* 1, 335–361. doi: 10.1016/0921-8009(89)90014-1

Costanza, R., Perez-Maqueo, O., Martinez, M. L., Sutton, P., Anderson, S. J., and Mulder, K. (2008). The value of coastal wetlands for hurricane protection. *Ambio* 37, 241–248.

Couture, N. J., Irrgang, A., Pollard, W., Lantuit, H., and Fritz, M. (2018). Coastal erosion of permafrost soils along the Yukon Coastal Plain and fluxes of organic carbon to the Canadian Beaufort Sea. *J. Geophys. Res. Biogeosci.* 123, 406–422. doi: 10.1002/2017JG004166

Couvillion, B. R., Beck, H., Schoolmaster, D., and Fischer, M. (2017). *Land area change in coastal Louisiana 1932 to 2016: US Geological Survey Scientific Investigations Map 3381*. Reston, VA: US Geological Survey, 16.

Cragg, S. M. (1983). "The mangrove ecosystem of the Purari Delta," in *The Purari - Tropical Environment of a High Rainfall River Basin*, ed. T. Petr (Dordrecht: Springer Netherlands), 295–324. doi: 10.1007/978-94-009-7263-6_16

Cragg, S. M., Friess, D. A., Gillis, L. G., Trevathan-Tackett, S. M., Terrett, O. M., Watts, J. E., et al. (2020). Vascular plants are globally significant contributors to marine carbon fluxes and sinks. *Ann. Rev. Mar. Sci.* 12, 469–497. doi: 10.1146/annurev-marine-010318-095333

Cravo, A., Fernandes, D., Damião, T., Pereira, C., and Reis, M. P. (2015). Determining the footprint of sewage discharges in a coastal lagoon in South-Western Europe. *Mar. Pollut. Bull.* 96, 197–209. doi: 10.1016/j.marpolbul.2015.05.029

Cui, B. S., Yang, Q. C., Yang, Z. F., and Zhang, K. J. (2009). Evaluating the ecological performance of wetland restoration in the Yellow River Delta. China. *Ecol. Eng.* 35, 1090–1103. doi: 10.1016/j.ecoleng.2009.03.022

Cui, L., Ge, Z., Yuan, L., and Zhang, L. (2015). Vulnerability assessment of the coastal wetlands in the Yangtze Estuary, China to sea-level rise. *Estuar. Coast. Shelf Sci.* 156, 42–51. doi: 10.1016/j.ecss.2014.06.015

Cunha, A. H., Assis, J. F., and Serrão, E. A. (2013). Seagrasses in Portugal: a most endangered marine habitat. *Aquat. Bot.* 104, 193–203. doi: 10.1016/j.aquabot.2011.08.007

Dahdouh-Guebas, F., Kairo, J. G., Koedam, N., and Mathenge, C. (2000). Utilization of mangrove wood products around Mida Creek (Kenya) amongst subsistence and commercial users. *Econ. Bot.* 54, 513–527. doi: 10.1007/bf02866549

Darby, F. A., and Turner, R. E. (2008). Below- and aboveground biomass of Spartina alterniflora: response to nutrient addition in a Louisiana salt marsh. *Estuar. Coasts* 31, 326–334. doi: 10.1007/s12237-008-9037-8

Dasgupta, S., Sobhan, I., and Wheeler, D. (2017). The impact of climate change and aquatic salinization on mangrove species in the Bangladesh Sundarbans. *Ambio* 46, 680–694. doi: 10.1007/s13280-017-0911-0

David, B., Aplin, K., Petchey, F., Skelly, R., Mialanes, J., Jones-Amin, H., et al. (2015). Kumukumu 1, a hilltop site in the Aird Hills: Implications for occupational trends and dynamics in the Kikori River delta, south coast of Papua New Guinea. *Quatern. Int.* 385, 7–26. doi: 10.1016/j.quaint.2014.06.058

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

Dennis, D. M., Pitcher, C. R., and Skewes, T. D. (2001). Distribution and transport pathways of *Panulirus ornatus* (Fabricius, 1776) and *Panulirus* spp. larvae in the Coral Sea, Australia. *Mar. Fresh. Res.* 52, 1175–1185. doi: 10.1071/MF01186

DGPA (2012). *Annuaire Statistique de la Direction Générale de la Pêche et de l'Aquaculture*. Tunis: DGPA.

Ding, P. X. (2016). *Evolution Trend and Vulnerability Assessment of Typical Coastal zones in China Under the Influence of Climate Change*. Beijing: China Science Press.

Dionisio, L. P. C., Rheinheimer, G., and Borrego, J. J. (2000). Microbiological pollution of Ria Formosa (south of Portugal). *Mar. Pollut. Bull.* 40, 186–193.

Doody, J. P. (2004). 'Coastal squeeze'—an historical perspective. *J. Coast. Conser.* 10, 129–138.

Duke, N. C., Ball, M. C., and Ellison, J. C. (1998). Factors influencing biodiversity and distributional gradients in mangroves. *Global Ecol. Biogeogr.* 7, 27–47. doi: 10.2307/2997695

Duy, N. B., Clauss, K., Cao, S., Naeimi, V., Kuenzer, C., and Wagner, W. (2015). Mapping rice seasonality in the mekong Delta with Multi-Year Envisat ASAR WSM Data. *Remote Sens.* 7, 15868–15893. doi: 10.3390/rs71215808

El Mahrad, B., Abalansa, S., Newton, A., Icely, J. D., Snoussi, M., and Kacimi, I. (2020). Social-environmental analysis for the management of coastal lagoons in North Africa. *Front. Environ. Sci.* 8:37. doi: 10.3389/fenvs.2020.00037

Elliott, M., and Whitfield, A. K. (2011). Challenging paradigms in estuarine ecology and management. *Estuar. Coast. Shelf S.* 94, 306–314. doi: 10.1016/j.ecss.2011.06.016

Elliott, M., Burdon, D., Atkins, J. P., Borja, A., Cormier, R., De Jonge, V. N., et al. (2017). "And DPSIR begat DAPSI (W) R (M)!"-A unifying framework for marine environmental management. *Mar. Pollut. Bull.* 118, 27–40. doi: 10.1016/j.marpolbul.2017.03.049

Elliott, M., Cutts, N. D., and Trono, A. (2014). A typology of marine and estuarine hazards and risks as vectors of change: a review for vulnerable coasts and their management. *Ocean Coast. Manag.* 93, 88–99. doi: 10.1016/j.ocecoaman.2014.03.014

Emmerton, C. A., Lesack, L. F. W., and Marsh, P. (2007). Lake abundance, potential water storage, and habitat distribution in the Mackenzie River delta, western Canadian Arctic. *Water Resour. Res.* 43:W05419. doi: 10.1029/2006WR005139

Essid, N., Mahmoudi, E., Boufahja, F., Dellali, M., Beyrem, H., and Aissa, P. (2008). Impact des pseudo-féces de moules sur la densité des bactéries hétérotrophes dans le secteur mytilicole de la lagune de Bizerte (Tunisie). *Rev. Sci. Equ.* 20, 383–392. doi: 10.7202/016912ar

Facca, C., Ceoldo, S., Pellegrino, N., and Sfriso, A. (2014). Natural recovery and planned intervention in coastal wetlands: Venice Lagoon (Northern Adriatic Sea. Italy) as a case study. *Sci. World J.* 968618, 1–15. doi: 10.1155/2014/968618

Félix, P. M., Chainho, P., Lima, R. F., Costa, J. L., Almeida, A. J., Domingos, I., et al. (2017). Mangrove fish of São Tomé Island (Gulf of Guinea): new occurrences and habitat usage. *Mar. Freshw. Res.* 68, 123–130.

Fertouna-Bellakhal, M., Dhib, A., Béjaoui, B., Turki, S., and Aleya, L. (2014). Driving factors behind the distribution of dinocyst composition and abundance in surface sediments in a western Mediterranean coastal lagoon: report from a high resolution mapping study. *Mar. Pollut. Bull.* 84, 1–2.

Flaherty, M., Szuster, B., and Miller, P. (2000). Low salinity inland shrimp farming in Thailand. *Ambio* 29, 174–179. doi: 10.1579/0044-7447-29.3.174

Forbes, D. L. (ed.). (2011). *State of the Arctic Coast 2010: Scientific Review and Outlook. International Arctic Science Committee, Land-Ocean Interactions in the Coastal Zone, Arctic Monitoring and Assessment Programme, International Permafrost Association*. Geesthacht: Helmholtz-Zentrum Geesthacht. Available online at: http://library.arcticportal.org/1277/1/state_of_the_arctic_coast_2010.pdf

Forbes, D. L. (2019). "Arctic deltas and estuaries: a Canadian perspective," in *Coasts and Estuaries: the Future*, eds E. Wolanski, J. W. Day, M. Elliott, and R. Ramachandran (Burlington: Elsevier), 123–147. doi: 10.1016/B978-0-12-814003-1.00008-3

Forbes, D. L., and Hansom, J. D. (2011). "Polar coasts," in *Treatise on Estuarine and Coastal Science*, Vol. 3, eds E. Wolanski and D. S. McLusky (Waltham, MA: Academic Press/Elsevier), 245–283. doi: 10.1016/b978-0-12-374711-2.00312-0

Forbes, D. L., Manson, G. K., Whalen, D. J. R., Couture, N. J., and Hill, P. R. (2014). "Coastal products of marine transgression in cold-temperate and high-latitude coastal plain settings: Gulf of St Lawrence and Beaufort Sea," in *Sedimentary Coastal Zones from High to Low Latitudes: Similarities and Differences*, Special Publication 388, eds I. P. Martini and H. R. Wanless (London: Geological Society), 131–163. doi: 10.1144/SP388.18

Frank, C., Kairo, J. G., Bosire, J. O., Mohamed, M. O., Dahdouh-Guebas, F., and Koedam, N. (2017). Involvement, knowledge and perception in a natural reserve under participatory management: Mida Creek, Kenya. *Ocean Coast. Manag.* 142, 28–36. doi: 10.1016/j.ocecoaman.2017.03.009

Frazier, D. E. (1967). Recent deltaic deposits of the Mississippi River: their development and chronology. *Trans. Gulf Coast Soc. Geol. Trans.* 17, 287–315.

Freije, R. H., Spetter, C. V., Marcovecchio, J. E., Popovich, C. A., Botté, S. E., Negrín, V., et al. (2008). "Water chemistry and nutrients in the Bahía Blanca Estuary," in *Perspectives on Integrated Coastal Zone Management in South America*, eds R. Neves, J. Baretta, and M. Mateus (Lisboa: IST Press), 243–256.

Fritz, M., Vonk, J. E., and Lantuit, H. (2017). Commentary: collapsing Arctic coastlines. *Nat. Clim. Change* 7, 6–7. doi: 10.1038/nclimate3188

Frusher, S. D. (1983). "The ecology of juvenile penaeid prawns, mangrove crab (*Scylla serrata*) and the giant freshwater prawn (*Macrobrachium rosenbergii*) in the Purari Delta," in *The Purari - Tropical Environment of a High Rainfall River Basin*, ed. T. Petr (The Hague: Dr W. Junk Publishers), 341–353. doi: 10.1007/978-94-009-7263-6_18

Gang, P. O., and Agatsiva, J. L. (1992). The current status of mangroves along the Kenyan Coast: a case study of Mida Creek mangroves based on remote sensing. The Ecology of Mangrove and Related Ecosystems. *Hydrobiologia* 247, 29–36. doi: 10.1007/978-94-017-3288-8_4

Gao, S. (2007). Modeling the growth limit of the Changjiang Delta. *Geomorphology* 85, 225–236. doi: 10.1016/j.geomorph.2006.03.021

Gao, S., Du, Y. F., Xie, W. J., Gao, W. H., Wang, D. D., and Wu, X. D. (2014). Environment-ecosystem dynamic processes of *Spartina alterniflora* saltmarshes along the eastern China coastlines. *Sci. China Earth Sci.* 57, 2567–2586. doi: 10.1007/s11430-014-4954-9

Gao, S., Jia, J. J., Yang, Y., Zhou, L., Wei, W., Mei, Y. J., et al. (2019). Obtaining typhoon information from sedimentary records in coastal-shelf waters. *Haiyang Xuebao* 41, 141–160.

García-Marín, P., Cabaço, S., Hernández, I., Vergara, J. J., Silva, J., and Santos, R. (2013). Multi-metric index based on the seagrass *Zostera noltii* (ZoNI) for ecological quality assessment of coastal and estuarine systems in SW Iberian Peninsula. *Mar. Pollut. Bull.* 68, 46–54. doi: 10.1016/j.marpolbul.2012.12.025

Gari, S. R., Newton, A., and Icely, J. D. (2015). A review of the application and evolution of the DPSIR framework with an emphasis on coastal social-ecological systems. *Ocean Coast. Manag.* 103, 63–67.

Gedan, K. B., Kirwan, M. L., Wolanski, E., Barbier, E. B., and Silliman, B. R. (2011). The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. *Clim. Change* 106, 7–29. doi: 10.1007/s10584-010-0003-7

Giosan, L., Syvitski, J., Constantinescu, S. D., and Day, J. (2014). Protect the world's deltas. *Nature* 516, 31–33. doi: 10.1038/516031a

Giubilato, E., Radomyski, A., Critto, A., Ciffroy, P., Brochot, C., Pizzol, L., et al. (2016). Modelling ecological and human exposure to POPs in Venice lagoon. Part I — Application of MERLIN-Expo tool for integrated exposure assessment. *Sci. Total Environ.* 565, 961–976. doi: 10.1016/j.scitotenv.2016.04.146

Glibert, P. M., Magnien, R., Lomas, M. W., Alexander, J., Tan, C., Haramoto, E., et al. (2001). Harmful algal blooms in the Chesapeake and coastal bays of Maryland. USA: comparison of 1997, 1998, and 1999 events. *Estuaries* 24, 875–883.

Goni, M. A., Monacci, N., Gisewhite, R., Ogston, A., Crockett, J., and Nittrouer, C. (2006). Distribution and sources of particulate organic matter in the water column and sediments of the Fly River Delta, Gulf of Papua (Papua New Guinea). *Estuar. Coast. Shelf Sci.* 69, 225–245. doi: 10.1016/j.ecss.2006.04.012

Gopal, B., and Chauhan, M. (2006). Biodiversity and its conservation in the Sundarban mangrove ecosystem. *Aquat. Sci.* 68, 338–354. doi: 10.1007/s00027-006-0868-8

Granek, E. F., Compton, J. E., and Phillips, D. L. (2009). Mangrove-exported nutrient incorporation by sessile coral reef invertebrates. *Ecosystems* 12, 462–472. doi: 10.1007/s10021-009-9235-7

Grosse, G., Goetz, S., McGuire, A. D., Romanovsky, V. E., and Schuur, A. A. G. (2016). Changing permafrost in a warming world and feedbacks to the Earth system. *Environ. Res. Lett.* 11:040201. doi: 10.1088/1748-9326/11/4/040201

Grussu, G., Attorre, F., Mollicone, D., Dargusch, P., Guillet, A., and Marchetti, M. (2014). Implementing REDD plus in Papua New Guinea: can biodiversity indicators be effectively integrated in PNG's National Forest Inventory? *Plant Biosyst.* 148, 519–528. doi: 10.1080/11263504.2014.900131

Guillet, F., and Mermet, L. (2017). How to make biodiversity knowledge compelling? The case of mosquito control implementation in the Camargue (France). *Environ. Sci. Policy* (in press). doi: 10.1016/j.envsci.2017.05.004

Gwyther, D. (1983). "The importance of the Purari River Delta to the prawn trawl fishery of the Gulf of Papua," in *The Purari - Tropical Environment of a High Rainfall River Basin*, ed. T. Petr (The Hague: Dr W. Junk Publishers), 355–365. doi: 10.1007/978-94-009-7263-6_19

Haldar, R., Khosa, R., and Gosain, A. K. (2019). Impact of anthropogenic interventions on the vembanad lake system. *Water Res. Environ. Eng.* 1, 9–29.

Hamilton, S. E., and Friess, D. A. (2018). Global carbon stocks and potential emissions due to mangrove deforestation from 2000 to 2012. *Nat. Clim. Change* 8, 240–244. doi: 10.1038/s41558-018-0090-4

Hoang, L. P., Biesbroek, R., Tri, V. P. D., Kummu, M., van Vliet, M. T. H., Leemans, R., et al. (2018). Managing flood risks in the Mekong Delta: How to address emerging challenges under climate change, and socioeconomic developments. *Ambio* 47, 635–649. doi: 10.1007/s13280-017-1009-4

Hollis, G. E. (1986). The modelling and management of the internationally important wetland at Ek Garaet El Ichkeul, Tunisia. International Waterfowl Research Bureau. *Spec. Publ.* 7:121.

Hollis, G. E., Agnew, C. T., Battarbee, R. W., Chrismall, N., Fisher, R. C., Flower, R., et al. (1977). *A Management Plan for the Proposed Park National de l'Ichkeul, Tunisia. Conserv. Rep. Ser.* 11:240.

Hoq, M. E. (2007). An analysis of fisheries exploitation and management practices in Sundarbans mangrove ecosystem, Bangladesh. *Ocean Coast. Manag.* 50, 411–427. doi: 10.1016/j.ocecoaman.2006.11.001

Hoq, M. E., Islam, M. N., Kamal, M., and Wahab, M. A. (2001). Abundance and seasonal distribution of *Penaeus monodon* postlarvae in the Sundarbans mangrove, Bangladesh. *Hydrobiologia* 457, 97–104.

Hoque, M. A., and Alam, S. M. K. (1995). "Post Farakka dry season surface and ground water conditions in the Ganges and vicinity," in *Women for Water Sharing*, ed. J. M. Hasna (Dhaka: Academic Publishers), 48–64.

Hossain, M. S., Dearing, J. A., Rahman, M. M., and Salehin, M. (2016). Recent changes in ecosystem services and human well-being in the Bangladesh coastal zone. *Reg. Environ. Change* 16, 429–443. doi: 10.1007/s10113-014-0748-z

Iftekhar, M. S. (2004). Environmental conscequences of oil pollution on the Bangladesh Sundarban: a brief review. *ISME* 3:11.

Iftekhar, M. S., and Saenger, P. (2007). Vegetation dynamics in the Bangladesh Sundarbans mangroves: a review of forest inventories. *Wetl. Ecol. Manag.* 16, 291–312. doi: 10.1007/s11273-007-9063-5

INCCA (2010). *Climate Change and India: A 4 X 4 Assessment, a Sectoral and Regional Analysis for 2030s*, Vol. 2. New Delhi: Ministry of Environment & Forests, Government of India.

Inuvialuit Harvest Study Working Group (2003). *Inuvialuit Harvest Study: Data and Methods Report 1988-1997*. Inuvik: Joint Secretariat.

Irrgang, A. M., Lantuit, H., Gordon, R. R., Piskor, A., and Manson, G. K. (2019). Impacts of past and future coastal changes on the Yukon coast – threats for cultural sites, infrastructure, and travel routes. *Arctic Sci.* 5, 107–126. doi: 10.1139/as-2017-0041

Islam, D. S. M., and Bhuiyan, M. A. H. (2018). Sundarbans mangrove forest of Bangladesh: causes of degradation and sustainable management options. *Environ. Sustain.* 1, 113–131. doi: 10.1007/s42398-018-0018-y

Islam, M. A., Mamun, A. Al, Hossain, F., Quraishi, S. B., Naher, K., Khan, R., et al. (2017). Contamination and ecological risk assessment of trace elements in sediments of the rivers of Sundarban mangrove forest, Bangladesh. *Mar. Pollut. Bull.* 124, 356–366. doi: 10.1016/j.marpolbul.2017.07.059

Islam, M. N., and Al-Amin, M. (2019). The Rampal power plant, ecological disasters and environmental resistance in Bangladesh. *Int. J. Environ. Stud.* 76, 922–939. doi: 10.1080/00207233.2019.1662183

Islam, M. R. (2006). "Managing diverse land uses in coastal Bangladesh: institutional approaches," in *Environment and Livelihoods in Tropical Coastal Zones*, eds C. T. Hoanh, T. P. Tuong, J. W. Gowing, and B. Hardy (Manila: CAB International and International Rice Research Institute (IRRI)), 237–248. doi: 10.1079/9781845931070.0237

Islam, S. N. (2019). "Sundarbans a dynamic ecosystem: an overview of opportunities, threats and tasks," in *The Sundarbans: A Disaster-Prone Eco-Region. Coastal Research Library*, Vol. 30, ed. H. Sen (Cham: Springer), doi: 10.1007/978-3-030-00680-8_2

Islam, S. N., and Gnauck, A. (2009). Threats to the Sundarbans mangrove wetland ecosystems from transboundary water allocation in the Ganges Basin: a preliminary problem analysis. *Int. J. Ecol. Econ. Stat.* 13, 64–78.

Islam, S. N., Reinstädtler, S., and Gnauck, A. (2018). "Vulnerability of mangrove forests and wetland ecosystems in the Sundarbans natural world heritage site (Bangladesh)," in *Threats to Mangrove Forests. Coastal Research Library*, Vol. 25, eds C. Makowski and C. Finkl (Cham: Springer).

IUCN (1994). *1993 United Nations List of National Parks and Protected Areas*. Gland: IUCN.

Jin, B., Fu, C., Zhong, J., Li, B., Chen, J., and Wu, J. (2007). Fish utilization of a salt marsh intertidal creek in the Yangtze River estuary, China. *Estuar. Coast. Shelf S.* 73, 844–852. doi: 10.1016/j.ecss.2007.03.025

Jungsberg, L., Turunen, E., Heleniak, T., Wang, S., Ramage, J., and Roto, J. (2019). *Atlas of Population, Society and Economy in the Arctic. Nordregio Working Paper 2019:3*. Stockholm: Nordregio, doi: 10.30689/WP2019:3.1403-2511

Kairo, J. G., Dahdouh-Guebas, F., Gwada, P. O., Ochieng, C., and Koedam, N. (2002). Regeneration status of mangrove forests in Mida Creek, Kenya: a compromised or secured future? *Ambio* 31, 562–568. doi: 10.1579/0044-7447-31.7.562

Kearney, M. S., Riter, C. A., and Turner, R. E. (2011). Freshwater diversions for marsh restoration in Louisiana: twenty-six years of changing vegetative cover and marsh area. *Geophys. Res. Lett.* 38:L16405. doi: 10.1029/2011GL047847

Kebede, A., Nicholls, R., Hanson, S., and Mokrech, M. (2010). Impacts of climate change and sea-level rise: a preliminary case study of Mombasa. Kenya. *J. Coast. Res.* 28, 8–19. doi: 10.2112/jcoastres-d-10-00069.1

Kemp, W. M., Boynton, W. R., Adolf, J. E., Boesch, D. F., Boicourt, W. C., Brush, G., et al. (2005). Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Mar. Ecol. Prog. Ser.* 303, 1–29. doi: 10.3354/meps303001

Kennedy, C. J., and Barbier, E. B. (2016). The economic value of freshwater inputs to an estuarine fishery. *Water Res. Econ.* 13, 46–59. doi: 10.1016/j.wre.2015.11.003

Khan, M. S., Haq, E., Huq, S., Rahman, A. A., Rashid, S. M. A., and Ahmed, H. (1994). *Wetlands of Bangladesh*. Dhaka: BCAS.

Kirui, K. B., Kairo, J. G., Bosire, J., Viergever, K. M., Rudra, S., Huxham, M., et al. (2013). Mapping of mangrove forest land cover change along the Kenya coastline using Landsat imagery. *Ocean Coast. Manag.* 83, 19–24. doi: 10.1016/j.ocecoaman.2011.12.004

KNBS (2010). *The 2009 Kenya Population and Housing Census Population Distribution by Administrative Unit*, Vol. IA. Kenya: Kenya National Bureau of Statistics, 260.

Kokelj, S. V., and GeoNorth Limited (2002). *Drilling Mud Sumps in the Mackenzie Delta Region: Construction, Abandonment and Past Performance*. Yellowknife: Contract report to Department of Indian Affairs and Northern Development.

Kokelj, S. V., Lantz, T. C., Solomon, S., Pisaric, M. F. J., Keith, D., Morse, P., et al. (2012). Using multiple sources of knowledge to investigate northern environmental change: regional ecological impacts of a storm surge in the outer Mackenzie Delta, NWT. *Arctic* 65, 257–272.

Kokelj, S. V., Riseborough, D., Coutts, R., and Kanigan, J. C. N. (2010). Permafrost and terrain conditions at northern drilling-mud sumps: impacts of vegetation and climate change and the management implications. *Cold Reg. Sci. Technol.* 64, 46–56. doi: 10.1016/j.coldregions.2010.04.009

Kompas, T., and Kuk, R. (2008). Managing the Gulf of Papua prawn fishery: sustainability, maximum returns and cooperation between commercial fishing and indigenous fishing communities. *Pac. Econ. Bull.* 23, 29–38.

Koundouri, P., Karousakis, K., Assimacopoulos, D., Jeffrey, P., and Lange, M. A. (2006). *Water Management in Arid and Semi-Arid Regions: Interdisciplinary Perspectives*. Cheltenham: Edward Elgar Publishing.

Kuenzer, C., and Vo, Q. T. (2013). Assessing the ecosystem services value of Can Gio Mangrove Biosphere Reserve: combining earth-observation- and household-survey-based analyses. *Appl. Geogr.* 45:16.

Kuenzer, C., Campbell, I., Roch, M., Leinenkugel, L., Vo Quoc, T., and Dech, S. (2013). Understanding the impacts of hydropower developments in the context of upstream-downstream relations in the Mekong River Basin. *Sustain. Sci.* 8, 565–584. doi: 10.1007/s11625-012-0195-z

Kumar, K. K., and Rajan, P. D. (2012). *Fish and Fisheries in Vembanad Lake. Consolidated report of Vembanad fish count 2008- 2011*. New Delhi: CERC, 42.

Kumar, S. P. (2018). On the effect of hydrogen sulphide on primary productivity from retting area in Vembanad Lake, India. *Indian J. Mar. Sci*. 47, 378–380.

Lage, S., Costa, P. R., Moita, T., Eriksson, J., Rasmussen, U., and Rydberg, S. J. (2014). BMAA in shellfish from two Portuguese transitional water bodies suggests the marine dinoflagellate *Gymnodinium catenatum* as a potential BMAA source. *Aquat. toxicol.* 152, 131–138. doi: 10.1016/j.aquatox.2014.03.029

Lamoureux, S., Forbes, D. L., Bell, T., and Manson, G. K. (2015). "The impact of climate change on infrastructure in the western and central Canadian Arctic," in *From Science to Policy in the Western and Central Canadian Arctic: an Integrated Regional Impact Study (IRIS) of Climate Change and Modernization*, Chap. 7, eds G. A. Stern and A. Gaden (Québec: ArcticNet), 300–341.

Lanfredi, N. W., D'Onofrio, E. E., and Mazio, C. C. (1988). Variations of the mean sea level in the Southwest Atlantic Ocean. *Cont Shelf Res* 8, 1211–1220. doi: 10.1016/0278-4343(88)90002-7

Lang'at, J. K., and Kairo, J. G. (2008). *Conservation and Management of Mangrove Forests in Kenya*. Mombasa: Mangrove Reforestation Program, Kenya Marine and Fisheries Research Institute.

Lang'at, J., Kairo, J., Mencuccini, M., Bouillon, S., Skov, M., Waldron, S., et al. (2014). Rapid losses of surface elevation following tree girdling and cutting in tropical mangroves. *PLoS One* 10:e0118334. doi: 10.1371/journal.pone.0107868

Laxmilatha, P., and Appukuttan, K. K. (2002). A review of the black clam (*Villorita cyprinoides*) fishery of the Vembanad Lake. *Indian J. Fish.* 49, 85–91.

Lefcheck, J. S., David, J. W., Rebecca, R. M., Scott, R. M., and Robert, J. O. (2017). Multiple stressors threaten the imperiled coastal foundation species eelgrass (Zostera marina) in Chesapeake Bay. USA. *Glob. Change Biol.* 23, 3474–3483. doi: 10.1111/gcb.13623

Lefcheck, J. S., Orth, R. J., Dennison, W. C., Wilcox, D. J., Murphy, R. R., Keisman, J., et al. (2018). Long-term nutrient reductions lead to the unprecedented recovery of a temperate coastal region. *Proc. Natl. Acad. Sci. U.S.A.* 115, 3658–3662. doi: 10.1073/pnas.1715798115

Lesack, L. F. W., Marsh, P., Hicks, F. E., and Forbes, D. L. (2013). Timing, duration, and magnitude of peak annual water levels during ice breakup in the Mackenzie Delta and the role of river discharge. *Water Resour. Res.* 39, 8234–8249. doi: 10.1002/2012WR013198

Lesack, L. F. W., Marsh, P., Hicks, F. E., and Forbes, D. L. (2014). Local spring warming drives earlier river-ice breakup in a large Arctic delta. *Geophys. Res. Lett.* 41, 1560–1566. doi: 10.1002/2013GL058761

Li, B., Liao, C. H., Zhang, X. D., Chen, H. L., Wang, Q., Chen, Z. Y., et al. (2009). *Spartina alterniflora* invasions in the Yangtze River estuary. China: an overview of current status and ecosystem effects. *Ecol. Eng.* 35, 511–520. doi: 10.1016/j.ecoleng.2008.05.013

Li, Y., Zhang, H. B., Chen, X. B., Tu, C., Luo, Y. M., and Christie, P. (2014). Distribution of heavy metals in soils of the Yellow River Delta: concentrations in different soil horizons and source identification. *J. Soils Sediments* 14, 1158–1168. doi: 10.1007/s11368-014-0861-0

Li, Y., Zhang, H. B., Fu, C. C., Tu, C., Luo, Y. M., and Christie, P. (2019). A red clay layer in soils of the Yellow River Delta: occurrence, properties and implications for elemental budgets and biogeochemical cycles. *Catena* 172, 469–479. doi: 10.1016/j.catena.2018.09.015

Li, Y., Zhang, H. B., Li, Q. B., Zhou, Q., Chen, X. B., Tu, C., et al. (2016). Characteristics of residual organochlorine pesticides in soils undue different land-use types on a coastal plain of the Yellow River Delta: concentrations in different soil horizons and source identification. *Environ. Geochem. Health* 38, 535–547. doi: 10.1007/s10653-015-9738-4

Liebezeit, G., and Rau, M. T. (2006). New Guinea mangroves - traditional usage and chemistry of natural products. *Senck. Marit.* 36, 1–10. doi: 10.1007/bf03043698

Liu, Y., and Huang, H. J. (2013). Characterization and mechanism of regional land subsidence in the Yellow River Delta. China. *Nat. Hazards* 68, 687–709. doi: 10.1007/s11069-013-0648-4

Loucks, C., Barber-Meyer, S., Hossain, M. A. A., Barlow, A., and Chowdhury, R. M. (2010). Sea level rise and tigers: predicted impacts to Bangladesh's Sundarbans mangroves. *Clim. Change* 98:291. doi: 10.1007/s10584-009-9761-5

Lovelock, C. E., and Duarte, C. M. (2019). Dimensions of Blue Carbon and emerging perspectives. *Biol. Lett.* 15:20180781. doi: 10.1098/rsbl.2018.0781

Luo, Y. M., Li, Y., Zhang, H. B., and Tu, C. (eds) (2017). *Soil and environment of the Yellow River delta*. Beijing: China Science Press, 002–038.

Ma, T., Li, X., Bai, J., Ding, S., Zhou, F., and Cui, B. (2019). Four decades' dynamics of coastal blue carbon storage driven by land use/land cover transformation under natural and anthropogenic processes in the Yellow River Delta, China. *Sci. Total Environ.* 655, 741–750. doi: 10.1016/j.scitotenv.2018.11.287

Ma, Z., Gan, X., Choi, C., Jing, K., Tang, S., Li, B., et al. (2007). Wintering bird communities in newly- formed wetland in the Yangtze River estuary. *Ecol. Res.* 22, 115–124. doi: 10.1007/s11284-006-0193-7

Ma, Z., Wang, Y., Gan, X., Li, B., Cai, Y., and Chen, J. (2009). Waterbird population changes in the wetlands at Chongming Dongtan in the Yangtze River estuary. China. *Environ. Manag.* 43, 1187–1200. doi: 10.1007/s00267-008-9247-7

MAERH (2003). *Etude sur la dépollution industrielle dans le bassin versant du lac de Bizerte. Rapport*. Tunis: Ministère de l'Agriculture, de l'Environnement et des Ressources Hydrauliques, 182.

Manson, G. K., and Solomon, S. M. (2007). Past and future forcing of Beaufort Sea coastal change. *Atmos. Ocean.* 45, 107–122. doi: 10.3137/ao.450204

Martini, I. P., Morrison, R. I. G., Abraham, K. F., Sergienko, L. A., and Jefferies, R. L. (2019). "Northern polar coastal wetlands: development, structure, and land use," in *Coastal Wetlands: An Integrated Ecosystem Approach* 2nd Edn, eds G. M. E. Perillo, E. Wolanski, D. R. Cahoon, and C. S. Hopkinson (Amsterdam: Elsevier), 153–186. doi: 10.1016/B978-0-444-63893-9.00004-6

Mateo, R., Green, A. J., Lefranc, H., Baos, R., and Figuerola, J. (2007). Lead poisoning in wild birds from southern Spain: a comparative study of wetland areas and species affected, and trends over time. *Ecotox. Environ. Safe*. 66, 119–126.

McClanahanan, T. R., Mwagunib, S., and Muthiga, N. A. (2005). Management of the Kenyan coast. *Ocean Coast. Manag.* 48, 901–931. doi: 10.1016/j.ocecoaman.2005.03.005

McKendry, J. E. (2009). *A Socioeconomic Atlas for the Chesapeake Bay Watershed and Its Region*. Available online at: https://www.chesapeakebay.net/content/publications/cbp_46698.pdf

Mei, X. F., Dai, Z. J., Stephen, D., Gao, S., Wang, J., and Jiang, W. G. (2018). Modulation of extreme flood levels by impoundment significantly offset by floodplain loss downstream of the Three Gorges Dam. *Geophys. Res. Lett.* 45, 3147–3155. doi: 10.1002/2017GL076935

Millennium Ecosystem Assessment [MEA] (2005). *Ecosystems and Human Well-Being: Wetlands and Water Synthesis*. Washington, DC: World Resources Institute, 80.

Miralles, J., Radakovitch, O., Cochran, J. K., Veron, A., and Masque, P. (2004). Multitracer study of anthropogenic contamination records in the Camargue, Southern France. *Sci. Total Environ.* 320, 63–72.

Mohan, M., Shylesh Chandran, M. S., Jayasooryan, K. K., and Ramasamy, E. V. (2014). Mercury in the sediments of Vembanad Lake, western coast of India. *Environ. Monit. Assess.* 186, 3321–3336. doi: 10.1007/s10661-014-3620-1

Moreira da Silva, M., Duarte, D., and Luís, C. (2019). "Metal contamination in Ria Formosa saltmarsh, sediments and halophyte vegetation," in *Ria Formosa: Challenges of a Coastal Lagoon in a Changing Environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Faro: University of Algarve), 99–107.

Morris, J. T., Barber, D. C., Callaway, J. C., Chambers, R., Hagen, S. C., Hopkinson, C. S., et al. (2016). Contributions of organic and inorganic matter to sediment volume and accretion in tidal wetlands at steady state. *Earth's Future* 4, 110–121. doi: 10.1002/2015EF000334

Mudge, S. M., and Duce, C. E. (2005). Identifying the source, transport path and sinks of sewage derived organic matter. *Environ. Pollut.* 136, 209–220. doi: 10.1016/j.envpol.2005.01.015

Nagelkerken, I., Van der Velde, G., Gorissen, M. W., Meijer, G. J., Van't Hof, T., and Den Hartog, C. (2000). Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. *Estuar. Coast. Shelf Sci.* 51, 31–44. doi: 10.1006/ecss.2000.0617

Najjar, R. G., Christopher, R. P., Mary, B. A., Denise, B., Carl, H., Michael, K., et al. (2010). Potential climate-change impacts on Chesapeake Bay. *Estuar. Coast. Shelf S.* 86, 1–20.

Narayanan, S. P., Thomas, A. P., and Sreekumar, B. (2011). Ornithofauna and its conservation in the Kuttanad wetlands, southern portion of Vembanad-Kole Ramsar site, India. *J. Threat. Taxa* 3, 1663–1676. doi: 10.11609/JoTT.o1870.1663-76

National Fish and Wildlife Service (2016). *Nutria Eradication Project. Nutria Biology and Identification*. Available online at: https://www.fws.gov/chesapeakenutriaproject/index.html (accessed August 28, 2016).

National Research Council [NRC]. (2010). *Sea Level Rise and the Coastal Environment. Advancing the science of climate change*. Washington, DC: NRC Press. 243–247.

Negandhi, K., Edwards, G., Kelleway, J. J., Howard, D., Safari, D., Saintilan, N., et al. (2019). Blue carbon potential of coastal wetland restoration varies with inundation, and rainfall. *Sci. Rep*. 9:4368. doi: 10.1038/s41598-019-40763-8

Neubauer, S. C., and Verhoeven, J. T. A. (2019). "Wetland effects on global climate: mechanisms, impacts, and management recommendations," in *Wetlands: Ecosystem Services, Restoration and Wise Use*, eds S. An, and J. T. A. Verhoeven (Cham: Springer), 39–62. doi: 10.1007/978-3-030-14861-4_3

Newton, A., Brito, A. C., Icely, J. D., Derolez, V., Clara, I., Angus, S., et al. (2018). Assessing, quantifying and valuing the ecosystem services of coastal lagoons. *J. Nat. Conserv.* 44, 50–65. doi: 10.1016/j.jnc.2018.02.009

Newton, A., Icely, J. D., Falcão, M., Nobre, A., Nunes, J. P., Ferreira, J. G., et al. (2003). Evaluation of eutrophication in the Ria Formosa coastal lagoon. Portugal. *Cont. Shelf Res.* 23, 1945–1961. doi: 10.1016/j.csr.2003.06.008

Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A. C., Colijn, F., et al. (2013). An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. *Estuar. Coast. Shelf Sci.* 140, 95–122. doi: 10.1016/j.ecss.2013.05.023

O'Connor, J. J., Fest, B. J., Sievers, M., and Swearer, S. E. (2020). Impacts of land management practices on blue carbon stocks, and greenhouse gas fluxes in coastal ecosystems—A meta-analysis. *Glob. Change Biol.* 26, 1354–1366. doi: 10.1111/gcb.14946

Obu, J., Lantuit, H., Fritz, M., Pollard, W. H., Sachs, T., and Gunther, F. (2016). Relation between planimetric and volumetric measurements of permafrost coast erosion: a case study from Herschel Island, western Canadian Arctic. *Polar Res.* 35:303013. doi: 10.3402/polar.v35.303013

OECD (2015). *OECD Urban Policy Reviews: China 2015*. Paris: OECD Publishing, doi: 10.1787/9789264230040-en

O'Higgins, T., Cooper, P., Roth, E., Newton, A., Farmer, A., Goulding, I., et al. (2014). Temporal constraints on ecosystem management: definitions and examples from Europe's regional seas. *Ecol. Soc.* 19:46.

Oppenheimer, M., Glavovic, B. C., Hinkel, J., van de Wal, R., Magnan, A. K., Abd-Elgawad, A., et al. (2019). "Sea level rise and implications for low-lying islands, coasts and communities," in *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate*, eds H.-O. Pörtner, et al. (Geneva: IPCC).

Ottinger, M., Clauss, K., and Kuenzer, C. (2016). Aquaculture: relevance, distribution, impacts and spatial assessments. *Ocean Coast. Manag.* 119, 244–266. doi: 10.1016/j.ocecoaman.2015.10.015

Ottinger, M., Kuenzer, C., Liu, G. H., Wang, S. Q., and Dech, S. (2013). Monitoring land cover dynamics in the Yellow River Delta from 1995 to 2010 based on Landsat 5 TM. *Appl. Geogr.* 44, 53–68. doi: 10.1016/j.apgeog.2013.07.003

Owuor, M. A., Icely, J., and Newton, A. (2019a). Community perceptions of the status and threats facing mangroves of Mida Creek, Kenya: implications for community based management. *Ocean Coast. Manag.* 175, 172–179. doi: 10.1016/j.ocecoaman.2019.03.027

Owuor, M. A., Icely, J., Newton, A., Nyunja, J., Otieno, P., Tuda, A. O., et al. (2017). Assessment of the flow of ecosystem service in an East African marine reserve. *Ocean Coast. Manag.* 140, 11–12.

Owuor, M. A., Mulwa, R., Otieno, P., Icely, J., and Newton, A. (2019b). Valuing mangrove biodiversity and ecosystem services: a deliberative choice experiment in Mida Creek. Kenya. *Ecosyst. Serv.* 40:101040. doi: 10.1016/j.ecoser.2019.101040

Page, T., Murphy, M. E., Mizrahi, M., Cornelius, J. P., and Venter, M. (2016). Sustainability of wood-use in remote forest-dependent communities of Papua New Guinea. *Forest Ecol. Manag.* 382, 88–99. doi: 10.1016/j.foreco.2016.09.043

Paolisso, M. (2008). Cultural models and cultural consensus of Chesapeake Bay blue crab and oyster fisheries. *NAPA Bull.* 28, 123–135. doi: 10.1525/napa.2007.28.1.123

Parris, A., Bromirski, P., Burkett, V., Cayan, D., Culver, M., Hall, J., et al. (2012). *Global Sea Level Rise Scenarios for the US National Climate Assessment. NOAA Tech Memo OAR CPO*-1. Silver Spring, MA: NOAA.

Patrício, J., Elliott, M., Mazik, K., Papadopoulou, K. N., and Smith, C. J. (2016). DPSIR—two decades of trying to develop a unifying framework for marine environmental management? *Front. Mar. Sci.* 3:177. doi: 10.3389/fmars.2016.00177

Percival, M., and Womersley, J. S. (1975). *Floristics and ecology of the mangrove vegetation of Papua New Guinea, in: Botany Bulletin Lae*. Papua New Guinea: Department of Forests, Division of Botany, 1–96.

Perillo, G. M. E. (1995). "Definition and geomorphologic classifications of estuaries," in *Geomorphology and Sedimentology of Estuaries, Development in Sedimentology*, Vol. 53, ed. G. M. E. Perillo (Amsterdam: Elsevier Science BV), 17–47. doi: 10.1016/s0070-4571(05)80022-6

Perillo, G. M. E., and Iribarne, O. O. (2003a). New mechanisms studied for creek formation in tidal flats: from crabs to tidal channels. *EOS Am. Geophys. Union Trans.* 84, 1–5.

Perillo, G. M. E., and Iribarne, O. O. (2003b). Processes of tidal channels develop in salt and freshwater marshes. *Earth Surf. Proc. Land.* 28, 1473–1482. doi: 10.1002/esp.1018

Perillo, G. M. E., and Piccolo, M. C. (1991). Tidal response in the Bahía Blanca Estuary. *J. Coast. Res.* 7, 437–449.

Perillo, G. M. E., and Piccolo, M. C. (1999). *Geomorphological and physical Characteristics of the Bahia Blanca Estuary, Argentina*, eds G. M. E. Perillo, M. C. Piccolo, and M. Pino Quivira (Berlin: Springer Verlag), 195–216.

Perillo, G. M. E., and Piccolo, M. C., (2020). Physical oceanography of the Bahía Blanca Estuary. *Estuar. Coast. Shelf Sci.* 31, 303–317.

Perillo, G. M. E., and Sequeira, M. E. (1989). Geomorphologic and sediment transport characteristics of the middle reach of the Bahía Blanca Estuary. Argentina. *J. Geophys. Res. Oceans* 94, 14351–14362.

Perillo, G., Wolanski, E., Cahoon, D., and Hopkinson, C. (eds) (2019). *Coastal Wetlands: An Integrated Ecosystem Approach*, 2nd Edn. Amsterdam: Elsevier, 1105.

Piccolo, M. C., and Perillo, G. M. E. (1990). Physical characteristics of the Bahía Blanca Estuary (Argentina). *Estuar. Coast. Shelf S.* 31, 303–317. doi: 10.1016/0272-7714(90)90106-2

Planning Commission Government of India (2008). *Report on Visit to Vembanad Kol, Kerala, a Wetland Included Under the National Wetland Conservation and Management Programme of the Ministry of Environment and Forests, 29 June-1 July, 2008*. New Delhi: Planning Commission, 22.

Pranovi, F., Franceschini, G., Casale, M., Zucchetta, M., Torricelli, P., and Giovanardi, O. (2006). An ecological imbalance induced by a non-native species: the Manila clam in the Venice Lagoon. *Biol. Invasions* 8, 595–609. doi: 10.1007/s10530-005-1602-5

Primavera, J. H. (2006). Overcoming the impacts of aquaculture on the coastal zone. *Ocean Coast. Manag.* 49, 531–545. doi: 10.1016/j.ocecoaman.2006.06.018

Rabalais, N. N., and Turner, R. E. (2016). Effects of the Deepwater Horizon oil spill on coastal habitats and fauna. *Oceanography* 29, 150–159. doi: 10.5670/oceanog.2016.79

Rahman, M. M., Chongling, Y., Islam, K. S., and Haoliang, L. (2009). A brief review on pollution and ecotoxicologic effect on Sundarbans mangrove ecosystem in Bangladesh. *Int. J. Environ. Eng.* 1, 369–382.

Rahman, M. M., Jiang, Y., and Irvine, K. (2018). Assessing wetland services for improved development decision- making: a case study of mangroves in coastal Bangladesh. *Wetlands Ecol. Manag.* 26, 563–580. doi: 10.1007/s11273-018-9592-0

Rahman, M. M., Rahman, M. M., and Islam, K. S. (2010). The causes of deterioration of Sundarban mangrove forest ecosystem of Bangladesh: conservation and sustainable management issues. *AACL Bioflux* 3, 77–90.

Ranjan, P., Ramanathan, A. L., Kumar, A., Singhal, R. K., Datta, D., and Venkatesh, M. (2018). Trace metal distribution, assessment and enrichment in the surface sediments of Sundarban mangrove ecosystem in India and Bangladesh. *Mar. Pollut. Bull.* 127, 541–547. doi: 10.1016/j.marpolbul.2017.11.047

Raposa, K. B., Wasson, K., Smith, E., Crooks, J. A., Delgado, P., Fernald, S. H., et al. (2016). Assessing tidal marsh resilience to sea-level rise at broad geographic scales with multi-metric indices. *Biol. Conserv.* 204, 263–275. doi: 10.1016/j.biocon.2016.10.015

Rashid, S. M. A., Khan, A., and Akonda, A. W. (1994). "Fauna," in *Mangrove of the Sundarbans*, eds Z. Hussain, and G. Acharya (Bangkok: IUCN), 125–132.

Riba, I., Garcia-Luque, E., Blasco, J., and DelValls, T. A. (2003). Bioavailability of heavy metals bound to estuarine sediments as a function of pH and salinity values. *Chem. Speciation Bioavailability* 15, 101–114.

Robertson, A. I, and Alongi, D. M. (1995). Role of riverine mangrove forests in organic carbon export to the tropical coastal ocean: a preliminary mass balance for the Fly Delta (Papua New Guinea). *Geo-Mar. Lett.* 15, 134–139. doi: 10.1007/bf01204454

Rova, S., Pranovi, F., and Muller, F. (2015). Provision of ecosystem services in the lagoon of Venice (Italy): an initial spatial assessment. *Ecohydrol. Hydrobiol.* 15, 13–25. doi: 10.1016/j.ecohyd.2014.12.001

Ruz, M. -H., Héquette, A., and Hill, P. R. (1992). A model of coastal evolution in a transgressed thermokarst topography, Canadian Beaufort Sea. *Mar. Geol.* 106, 25–278.

Rybicki, N. B., and Landwehr, J. M. (2007). "Submerged aquatic vegetation," in *Synthesis of U.S. Geological Survey Science for the Chesapeake Bay Ecosystem and Implications for Environmental Management: U.S. Geological Survey Circular* 1316, Ed. S.W. Phillips (Reston, VA: U.S. Geological Survey), 46–49.

Sahraoui, I., Sakka, H. A., Hadj, M. H., Leger, C., and Bates, S. S. (2009). Blooms of the diatoms genus *Pseudo-nitzschia* H. Peragallo in Bizerte Lagoon (Tunisia, SW Mediterranean). *Diatom Res.* 24, 175–190. doi: 10.1080/0269249x.2009.9705789

Saied, M., and Elloumi, M. J. (2007). *Prise en Compte des Besoins Écologiques de l'Ichkeul dans la gestion de l'eau en Tunisie. Gestion de la demande en eau en Méditerranée*. Available online at: http://planbleu.org/sites/default/files/upload/files/Eco_5_TN_47_Saied_final_FR.pdf

Sakka Hlaili, A., Chikhaoui, M. A., El Grami, B., and Hadj Mabrouk, H. (2006). Effect of N and P supply on phytoplankton in Bizerte lagoon (Western Mediterranean). *J. Exp. Mar. Biol.* 333, 79–96. doi: 10.1016/j.jembe.2005.12.049

Sakka Hlaili, A., El Grami, B., Hadj Mabrouk, H., Gosselin, M., and Hamel, D. (2007). Phytoplankton growth and microzooplanton grazing rates in a restricted Mediterranean lagoon (Bizerte lagoon, Tunisia). *Mar. Biol.* 151, 767–783. doi: 10.1007/s00227-006-0522-y

Sarretta, A., Pillon, S., Molinaroli, E., Guerzoni, S., and Fontolan, G. (2009). Sediment budget in the Lagoon of Venice, Italy. *Cont. Shelf Res.* 30, 934–949. doi: 10.1016/j.csr.2009.07.002

Sasmito, S. D., Sillanpää, M., Hayes, M. A., Bachri, S., Saragi-Sasmito, M. F., Sidik, F., et al. (2020). Mangrove blue carbon stocks and dynamics are controlled by hydrogeomorphic settings and land-use change. *Glob. Change Biol.* 2020, 1–12. doi: 10.1111/gcb.15056

Savoure, B. (1977). Etude hydrobiologique des lagunes du Nord de la Tunisie. *Hydrobiologia* 56, 209–224. doi: 10.1007/bf00017507

Scott, D. A. (1980). *A Preliminary Inventory of Wetlands of International Importance for Waterfowl in Western Europe and North-West Africa*. Slimbridge: IWRB.

Selvam, A. P., Priya, S. L., Banerjee, K., Hariharan, G., Purvaja, R., and Ramesh, R. (2012). Heavy metal assessment using geochemical and statistical tools in the surface sediments of Vembanad Lake, Southwest Coast of India. *Environ. Monit. Assess.* 184, 5899–5915. doi: 10.1007/s10661-011-2389-2388

Semeoshenkova, V., and Newton, A. (2015). Overview of erosion and beach quality issues in three Southern European countries: Portugal. Spain and Italy. *Ocean Coast. Manag.* 118, 12–21. doi: 10.1016/j.ocecoaman.2015.08.013

Serrano, O., Kelleway, J. J., Lovelock, C., and Lavery, P. S. (2019). "Conservation of blue carbon ecosystems for climate change mitigation and adaptation," in *Coastal Wetlands*, eds G. Perillo, D. R. Cahoon, G. M. E. Perillo, and E. Wolanski (Amsterdam: Elsevier), 965–996. doi: 10.1016/b978-0-444-63893-9.00028-9

Severini, M. D. F., Carbone, M. E., Villagran, D. M., and Marcovecchio, J. E. (2018). Toxic metals in a highly urbanized industry-impacted estuary (Bahia Blanca Estuary, Argentina): spatio-temporal analysis based on GIS. *Environ. Earth Sci.* 77, 393.

Shearman, P. L. (2010). Recent change in the extent of mangroves in the northern Gulf of Papua, Papua New Guinea. *Ambio* 39, 181–189. doi: 10.1007/s13280-010-0025-4

Shearman, P., Bryan, J., and Walsh, J. P. (2013). Trends in deltaic change over three decades in the Asia-Pacific region. *J. Coastal Res.* 29, 1169–1183.

Shepard, C. C., Crain, C. M., and Beck, M. W. (2011). The protective role of coastal marshes: a systematic review and meta-analysis. *PLoS One* 6:e27374. doi: 10.1371/journal.pone.0027374

Simonetti, P., Botté, S. E., and Marcovecchio, J. E. (2017). Occurrence and spatial distribution of metals in intertidal sediments of a temperate estuarine system (Bahía Blanca, Argentina). *Environ. Earth Sci.* 76:636.

Smart, M. (2004). *River Flow Regulation and Wetland Conservation in a Dry Country: Ichkeul, Tunisia. Assessment and Provision of Environmental flows in Mediterranean Watercourses. Mediterranean case study. IUCN Report document*. Available online at: http://cmsdata.iucn.org/downloads/tunisia.pdf (accessed June 17, 2014).

Solidoro, C., Bandelj, V., Aubry Bernardi, F., Camatti, E., Ciavatta, S., Cossarini, G., et al. (2010). "Response of the Venice Lagoon ecosystem to natural and anthropogenic pressures over the Last 50 Years," in *Coastal Lagoons: Critical Habitats of Environmental Change*, eds M. J. Kennish and H. W. Paerl (Boca Raton, FL: CRC Press), 483–511. doi: 10.1201/ebk1420088304-c19

Spalding, M., McIvor, A., Tonneijck, F. H., Tol, S., and van Eijk, P. (2014). *Mangroves for Coastal Defence. Guidelines for Coastal Managers & Policy Makers*. Wageningen: Wetlands International, 42.

Speake, M. A., Carbone, M. E., and Spetter, C. V. (2020). *Análisis del sistema socio-ecológico del estuario Bahía Blanca (Argentina) y su impacto en los servicios ecosistémicos y el Bienestar Humano*. Buenos Aires: National Council for Scientific and Technical Research.

Stephenson, S. A. (2004). *Harvest Studies in the Inuvialuit Settlement Region, Northwest Territories, Canada, 1999 and 2001–2003*. Winnipeg, MB: Resource Management and Aboriginal Affairs, Central and Arctic Region, Fisheries and Oceans Canada.

Strauss, B. H. (2013). Rapid accumulation of committed sea-level rise from global warming. *Proc. Natl. Acad. Sci. U.S.A.* 110, 13699–13700. doi: 10.1073/pnas.1312464110

Steckler, M. S., Nooner, S. L., Akhter, S. H., Chowdhury, S. K., Bettadpur, S., Seeber, L. et al. (2010). Modeling Earth deformation from monsoonal flooding in Bangladesh using hydrographic, GPS, and Gravity Recovery and Climate Experiment (GRACE) data. *J. Geophys. Res. Solid Earth* 115:B08407. doi: 10.1029/2009JB007018

Syvitski, J. P. M., and Saito, Y. (2007). Morphodynamics of deltas under the influence of humans. *Glob. Planet. Change* 57, 261–282. doi: 10.1016/j.gloplacha.2006.12.001

Syvitski, J. P., Kettner, A. J., Overeem, I., Hutton, E. W., Hannon, M. T., Brakenridge, G. R., et al. (2009). Sinking deltas due to human activities. *Nat. Geosci.* 2, 1–6.

Tamim, M. M., Dhar, A., and Hossain, M. S. (2013). Fly ash in Bangladesh- an overview. *Int. J. Sci. Eng. Res.* 4, 1–4.

Tamisier, A., and Boudouresque, C. F. (1994). "Aquatic bird populations as possible indicators of seasonal nutrient flow at Ichkeul lake, Tunisia," in *Aquatic Birds in the Trophic Web of Lakes. Hydrobiologia*, Vol. 279/280, ed. J. J. Kerekes (Dordrecht: Kluwer), 149–156. doi: 10.1007/978-94-011-1128-7_14

Tamisier, A., Dehorter, O., Defosse, A., Poydenot, F., Gravez, V., and Boudouresque, C. F. (2000). "Benefits, costs and risks for a field biologist. Ichkeul Lake Tunisia, a case atudy," in *Limnology and Aquatic Birds Modelling Aquatic Ecosystem in Monitoring, Modelling and Management*, eds F. A. Comin, J. A. Herrera, and J. Ramfrez (Mérida: Universidat Autonoma de Yucatan), 185–203.

Tanski, G., Wagner, D., Knoblauch, C., Fritz, M., Sachs, T., and Lantuit, H. (2019). Rapid CO2 release from eroding permafrost in seawater. *Geophys. Res. Lett.* 46, 11244–11252. doi: 10.1029/2019GL084303

Tavecchia, G., Pradel, R., Lebreton, J. D., Johnson, A. R., and Mondain-Monval, J. Y. (2001). The effect of lead exposure on survival of adult mallards in the Camargue, southern France. *J. Appl. Ecol.* 38, 1197–1207.

Tessler, Z. D., Vörösmarty, C. J., Grossberg, M., Gladkova, I., Aizenman, H., Syvitski, J. P. M., et al. (2015). Profiling risk and sustainability in coastal deltas of the world. *Science* 349, 638–639.

Thienpont, J. R., Kokelj, S. V., Korosi, J. B., Cheng, E. S., Desjardins, C., Kimpe, L. E., et al. (2013). Exploratory hydrocarbon drilling impacts to Arctic lake ecosystems. *PLoS One* 8:e78875. doi: 10.1371/journal.pone.0078875

Toan, P. V., Sebesvari, S., Bläsing, M., and Renaud, F. (2013). Pesticide management and their residues in sediments and surface and drinking water in the Mekong Delta, Vietnam. *Sci. Total Environ.* 45, 28–39. doi: 10.1016/j.scitotenv.2013.02.026

Todd, B. J., and Dallimore, S. R. (1998). Electromagnetic and geological transect across permafrost terrain, Mackenzie River delta, Canada. *Geophysics* 63, 1914–1924. doi: 10.1190/1.1444484

Torio, D. D., and Chmura, G. L. (2013). Assessing coastal squeeze of tidal wetlands. *J. Coastal Res.* 29, 1049–1061. doi: 10.2112/jcoastres-d-12-00162.1

Trabelsi, Y., Gharbi, F., El Ghali, A., Oueslati, M., Samaali, M., Abdelli, W., et al. (2012). Recent sedimentation rates in Garaet El Ichkeul Lake, NW Tunisia, as affected by the construction of dams and a regulatory sluice. *J. Soils Sediment* 12, 784–796. doi: 10.1007/s11368-012-0496-y

Turki, S., Balti, N., and Ben Janet, H. (2007). First bloom of dinoflagellate *Alexandrium catenella* in Bizerte Lagoon (northern Tunisia). *Harmful Algae News* 35, 7–9.

Turki, S., Dhib, A., Belakhal, M. F., Frossard, V., Balti, N., Kharrat, R., et al. (2014). Harmful algal blooms (HABs) associated with phycotoxins in shellfish: what can be learned from five years of monitoring in Bizerte Lagoon (Southern Mediterranean Sea)? *Ecol. Eng.* 67, 39–47. doi: 10.1016/j.ecoleng.2014.03.028

Turner, R. E. (1977). Intertidal vegetation and commercial yields of penaeid shrimp. *Trans. Am. Fish. Soc.* 106, 411–416. doi: 10.1577/1548-8659(1977)106<411:ivacyo>2.0.co;2

Turner, R. E. (2007). Geomorphology and perception in New Orleans after Iberville and Bienville. *Technol. Soc.* 29, 227–238. doi: 10.1016/j.techsoc.2007.01.013

Turner, R. E. (2017). The mineral sediment loading of the modern Mississippi River Delta: What is the restoration baseline? *J. Coast. Conserv.* 21, 867–872. doi: 10.1007/s11852-017-0547-z

Turner, R. E., and McClenachan, G. (2018). Reversing wetland death from 35,000 cuts: opportunities to restore Louisiana's dredged canals. *PLoS One* 13:e0207717. doi: 10.1371/journal.pone.0207717

Turner, R. E., and Rabalais, N. N. (2003). Linking landscape and water quality in the Mississippi River basin for 200 years. *Bioscience* 53, 563–572.

Turner, R. E., and Rabalais, N. N. (2018). *World Seas: An Environmental Evaluation, Volume I: Europe, The Americas and West Africa*, Vol. The Gulf of Mexico, Chap. 18, ed. C. Shepard (Cambridge: Academic Press), 445–464.

Turner, R. E., Kearney, M. S., and Parkinson, R. W. (2018). Sea level rise tipping point of delta survival. *J. Coast. Res.* 34, 470–474. doi: 10.2112/JCOASTRES-D-17-00068.1

Turner, R. E., McClenachan, G., and Tweel, A. W. (2016). Islands in the oil: quantifying salt marsh shoreline erosion after the Deepwater Horizon oiling. *Mar. Pollut. Bull.* 110, 316–323. doi: 10.1016/j.marpolbul.2016.06.046

Turner, R. E., Rabalais, N. N., Overton, E. B., Meyer, B. M., McClenachan, G., Swenson, E. M., et al. (2019). Oiling the continental shelf and coastal wetlands over eight years after the 2010 Deepwater Horizon oil spill. *Environ. Pollut.* 252, 1367–1376. doi: 10.1016/j.envpol.2019.05.134

Tweel, A. W., and Turner, R. E. (2012). Watershed land use and river engineering drive wetland formation and loss in the Mississippi River birdfoot delta. *Limnol. Oceanogr.* 57, 18–28. doi: 10.4319/lo.2012.57.1.0018

Tweel, A. W., and Turner, R. E. (2014). Contribution of tropical cyclones to the sediment budget for coastal wetlands in Louisiana, USA. *Landscape Ecol.* 29, 1083–1094. doi: 10.1007/s10980-014-0047-6

Ubertini, M., Lagarde, F., Mortreux, S., Le Gall, P., Chiantella, C., Fiandrino, A., et al. (2017). Gametogenesis, spawning behavior and larval abundance of the Pacific oyster *Crassostrea gigas* in the Thau lagoon: #vidence of an environment-dependent strategy. *Aquaculture* 473, 51–61. doi: 10.1016/j.aquaculture.2017.01.025

Varkey, L. M., Kumar, P., Mridha, N., Sekhar, I., and Sahoo, R. N. (2016). Ecosystem services and fishery production dynamics of wetland ecosystem: an appraisal of Alappuzha District of Kerala. India. *Fish. Technol.* 53:2.

Veríssimo, F., Flávio, M., and João, J. (2019). "The Role of Ria Formosa as a Waste Water Receiver," in *Ria Formosa: Challenges of a Coastal Lagoon in a Changing Environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Faro: University of Algarve), 47–65.

Vermaire, J. C., Pisaric, M. F. J., Thienpont, J. R., Courtney Mustaphi, C. J., Kokelj, S. V., and Smol, J. P. (2013). Arctic climate warming and sea ice declines lead to increased storm surge activity. *Geophy. Res. Lett.* 40, 1386–1390. doi: 10.1002/grl50191

Vo, Q. T., Kuenzer, C., and Oppelt, N. (2015). How remote sensing supports mangrove ecosystem service valuation: a case study in Ca Mau Province, Vietnam. *Ecosyst. Ser.* 14, 67–75. doi: 10.1016/j.ecoser.2015.04.007

Vo, Q. T., Kuenzer, C., Quang, M., Moder, F., and Oppelt, N. (2012). Review of valuation methods for mangrove ecosystem services. *J. Ecol. Indicat.* 23, 431–446. doi: 10.1016/j.ecolind.2012.04.022

Wahid, S. M., Babel, M. S., and Bhuiyan, A. R. (2007). Hydrologic monitoring and analysis in the Sundarbans mangrove ecosystem, Bangladesh. *J. Hydrol.* 332, 381–395. doi: 10.1016/j.jhydrol.2006.07.016

Walker, D. A., and Peirce, J. L. (2015). *Rapid Arctic Transitions due to Infrastructure and Climate (RATIC): A Contribution to ICARP III. Alaska Geobotany Center Publication AGC* 15-02. Fairbanks: University of Alaska Fairbanks.

Wang, H. J., Wu, X., Bi, N. S., Li, S., Yuan, P., Wang, A. M., et al. (2017). Impacts of the dam-oriented water-sediment regulation system on the lower reaches of the Yellow River (Huanghe): a review. *Glob. Planet Change* 157, 93–113. doi: 10.1016/j.gloplacha.2017.08.005

Wang, L. L., Yang, Z. F., Niu, J. F., and Wang, J. Y. (2009). Characterization, ecological risk assessment and source diagnostics of polycyclic aromatic hydrocarbons in water column of the Yellow River Delta, one of the most plenty biodiversity zones in the world. *J. Hazard Mater.* 169, 460–465. doi: 10.1016/j.jhazmat.2009.03.125

Ward, R. D., Friess, D. A., Day, R. H., and Mackenzie, R. A. (2016). Impacts of climate change on mangrove ecosystems: a region by region overview. *Ecosyst. Health Sustain.* 2:e01211. doi: 10.1002/ehs2.1211

Webber, M., Calumpong, H., Ferreira, F., Granek, E., Green, S., Ruwa, R., et al. (2016). *Mangroves. Chapter 48, Section BII Marine Ecosystems and Habitats. United Nations World Ocean Assessment*. Available online at: http://www.un.org/Depts/los/global_reporting/WOA_RPROC/Chapter_48.pdf (accessed August 13, 2016).

Wesche, S. D., and Chan, H. M. (2010). Adapting to the impacts of climate change on food security among Inuit in the Western Canadian Arctic. *Ecohealth* 7, 361–337.

White, W. T., Baje, L., Simpfendorfer, C. A., Appleyard, S. A., Chin, A., Sabub, B., et al. (2019). Elasmobranch bycatch in the demersal prawn trawl fishery in the Gulf of Papua, Papua New Guinea. *Sci. Rep.* 9:9254. doi: 10.1038/s41598-019-45715-w

Witman, S. (2017). Coastal wetlands effectively sequester "blue carbon". *Eos* 98, doi: 10.1029/2017EO079397

Wohlfart, C., Kuenzer, C., Cui, C., and Liu, G. (2016). Socio-ecological challenges in the Yellow River basin (China): a review. *Environ. Earth Sci.* 75:1066.

Yang, H. F., Yang, S. L., Meng, Y., Xu, K. H., Luo, X. X., Wu, C. S., et al. (2018). Recent coarsening of sediments on the southern Yangtze subaqueous delta front: a response to river damming. *Cont. Shelf Res.* 155, 45–51. doi: 10.1016/j.csr.2018.01.012

Yang, S. L., Zhang, J., Zhu, J., Smith, J. P., Dai, S. B., Gao, A., et al. (2005). Impact of dams on Yangtze River sediment supply to the sea and delta intertidal wetland response. *J. Geophys. Res- Earth* 110: F03006.

Zaaboub, N., Ounis, A., Helali, M. A., Béjaoui, B., Lillebø, A. I., Ferreira da Silva, E., et al. (2014). Phosphorus speciation in sediments and assessment of nutrient exchange at the water-sediment interface in a Mediterranean lagoon: implications for management and restoration. *Ecol. Eng.* 73, 115–125. doi: 10.1016/j.ecoleng.2014.09.017

Zacarias, D. A., Williams, A. T., and Newton, A. (2011). Recreation carrying capacity estimations to support beach management at Praia de Faro. *Portugal. Appl. Geogr.* 31, 1075–1081. doi: 10.1016/j.apgeog.2011.01.020

Zairi, M. (1997). *Impact de la gestion des eaux du Nord et de l'activité industrielle correspondante sue le biotope du Lac Ichkeul*. Paris: UNESCO.

Zhang, H., Chen, X. B., and Luo, Y. M. (2016). An overview of ecohydrology of the Yellow River delta wetland. *Ecohydrol. Hydrobiol.* 16, 39–44. doi: 10.1016/j.ecohyd.2015.10.001

Zhang, Q., Damian, C., Brady, W. R., Boynton, A., and William, P. B. (2015). Long-term trends if nutrients and sediment from the non-tidal Chesapeake Bay watershed: an assessment of progress by river and season. *J. Am. Water Res. Ass.* 51, 1534–1555. doi: 10.1111/1752-1688.12327

Zhang, T. T., Zeng, S. L., Gao, Y., Ouyang, Z. T., Li, B., Fang, C. M., et al. (2011). Assessing impact of land uses on land salinization in the Yellow River Delta, China using an integrated and spatial statistical model. *Land Use Policy* 28, 857–866. doi: 10.1016/j.landusepol.2011.03.002

Zhao, B., Kreuter, U., Li, B., Ma, Z., Chen, J., and Nakagoshi, N. (2004). An ecosystem service value assessment of land-use change on Chongming Island, China. *Land Use Policy* 21, 139–148. doi: 10.1016/j.landusepol.2003.10.003

Zhao, B., Yan, Y., Guo, H., He, M., Gu, Y., and Li, B. (2009). Monitoring rapid vegetation succession in estuarine wetland using time series MODIS-based indicators: an application in the Yangtze River Delta area. *Ecol Indic.* 9, 346–356. doi: 10.1016/j.ecolind.2008.05.009

Zhao, S., Zhu, L., Wang, T., and Li, D. (2014). Suspended microplastics in the surface water of the Yangtze Estuary System, China: first observations on occurrence, distribution. *Mar. Pollut. Bull.* 86, 562–568. doi: 10.1016/j.marpolbul.2014.06.032

Zilio, M. I., London, S., Perillo, G. M. E., and Piccolo, M. C. (2013). The social cost of dredging: the Bahía Blanca Estuary case. *Ocean Coast. Manag.* 71, 195–120.

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Bioaccumulation of Polychlorinated Dibenzo-p-Dioxins (PCDDs) and Dibenzofurans (PCDFs) in *Hediste diversicolor* (Polychaeta: Nereididae)

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The effectiveness and reliability of the polychaete *Hediste diversicolor* (O.F. Müller, 1776) to bioaccumulate polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) was assessed in an *in-situ* passive biomonitoring study. Field collected specimens were sampled in five sites within the Venice Lagoon (Italy), selected along a PCDD/F contamination gradient. The homolog profiles in the tissues of the common ragworm were considerably different from those observed in the sediments, independent of sediment contamination. Moreover, *H. diversicolor* accumulated preferentially the less chlorinated 2,3,7,8-TCDD, 2,3,7,8-TCDF and 2,3,4,7,8-PeCDF compared to the more chlorinated and hydrophobic hexa-, hepta- and octa-substituted congeners, as evidenced by the significant and linearly decreasing trend of the Biota-to-Sediment Accumulation Factor (BSAF) with the increasing lipophilicity of the congeners, expressed as the logarithmic form of the octanol/water partition coefficient ($\log K_{OW}$). The BSAFs for dioxins and furans were generally low compared to other organochlorine compounds such as polychlorinated biphenyls and organochlorine pesticides, suggesting that *H. diversicolor* may eliminate both dioxins and furans efficiently.

Keywords: Venice Lagoon, TCDD, TCDF, benthos, bioaccumulators

INTRODUCTION

Polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) are highly persistent, bioaccumulative and toxic contaminants not intentionally produced, but occurring in the environment as byproducts of chemical processes involving the use of chlorine (i.e., wood pulp and magnesium industries), as a result of combustion processes from both natural (forest fires) and anthropogenic sources (smoke from incinerators, car and boat exhaust fumes) or as impurities in chemicals (i.e., pesticides and herbicides) (Swerev and Ballschmiter, 1989).

Once released, PCDD/F enter the marine and coastal environments via atmospheric deposition, riverine inputs from the inland, and direct discharges of urban and industrial wastewaters into the coastal waters (Armitage et al., 2009).

In the marine environment PCDD/F are quickly adsorbed onto suspended matter, due to their high hydrophobicity, and then deposited onto the sediment where they accumulate over time, due to their persistence. As a consequence, sediment may act both as a secondary source of PCDD/F pollution for the water column (Khairy et al., 2016) and as a primary source of contamination for the benthic species living and/or feeding on sediment-bound contaminants (Pruell et al., 1993, 2000).

Polychaetes are a major component of the coastal and estuarine systems, where they may represent up to 40% of the taxa constituting the communities (Amiard-Triquet et al., 2013). Many species living in soft-bottom habitats are endobenthic and occupy several ecological niches, serving as relevant vectors for the recycling of the detritus, as primary consumers of benthic algae and predators of other invertebrates, and as food for the organisms at the top of the benthic and aquatic trophic webs (Scaps, 2002; Dean, 2008; Amiard-Triquet et al., 2013).

Close contact with the sediment as well as detritivorous habits of most species make polychaetes vulnerable to the uptake of contaminants through both contact with dissolved chemicals and ingestion of pollutants bound to sediment particles or adsorbed onto detritus particles (Dean, 2008). Consequently, contaminants accumulated into the tissues of the polychaetes may then be transferred to higher levels of the trophic web, leading to possible biomagnification phenomena (Ruus et al., 2012; Sizmur et al., 2013). For these reasons, polychaetes are often used in ecotoxicological studies to assess both early warning signals of chemical stress, by using relevant biomarkers, and the bioavailability of the contaminants (Durou et al., 2007a,b; Nesto et al., 2010; Ruus et al., 2012; Amiard-Triquet et al., 2013).

A commonly used polychaete species in ecotoxicological research is the common ragworm *Hediste diversicolor* (O.F. Müller, 1776). This burrowing polychaete is omnivorous, but also behaves as a deposit feeder, by collecting detritus near the opening of its burrows, and is widespread in the shallow marine and estuarine waters of the European coasts of the Atlantic, the Mediterranean Sea, the Black Sea and the Caspian Sea (Scaps, 2002). Its large dispersion into brackish waters has favored its frequent use as a biological indicator for assessing exposure and effects of sediment-bound contaminants in estuaries and coastal lagoons affected by pollution due to anthropogenic sources, including metals (Volpi Ghirardini et al., 1999; Berthet et al., 2003; Frangipane et al., 2005) and organic contaminants, such as polynuclear aromatic hydrocarbons (PAHs) – although the ability of polychaetes to biotransform PAHs was demonstrated (Jørgensen et al., 2008) – and polychlorinated biphenyls (PCBs) (Gunnarsson et al., 1999; Cornelissen et al., 2006; Langston et al., 2012). Nevertheless, the ability of *H. diversicolor* to also accumulate other persistent and bioaccumulative toxicants, as PCDD/F, has been rarely explored, although these polychlorinated compounds may accumulate in fish – and also humans (Raccanelli et al., 2007) – through the marine and estuarine trophic web, where *H. diversicolor* represents a relevant trophic link (Nunes et al., 2011).

The present study examined the effectiveness and reliability of *H. diversicolor* as an indicator of exposure for PCDD/F in an *in-situ* passive biomonitoring study, using field-collected polychaetes. The Venice Lagoon was chosen as a case study since sources, inputs and sediment concentration of PCDD/F have been extensively studied in the past decades (Fattore et al., 1997; Marcomini et al., 1997; Jimenez et al., 1998; Bellucci et al., 2000; Frignani et al., 2001a). The aims of the study were the assessment of possible bioaccumulation in tissues of *H. diversicolor*, the identification of the congeners with higher ability to accumulate into polychaete tissues, and the analysis of the possible relationships between accumulation and the lipophilicity (expressed as $\log K_{OW}$) of the 2,3,7,8 chlorinated congeners.

The results of this study, performed in 1998, are presented only now because we found that in the last 20 years no significant progress has been made in the field of bioaccumulation of organochlorine compounds in estuarine environments. Although they play a relevant role in the trophic webs of marine and estuarine environments, benthic infaunal organisms such as polychaetes, have been too often disregarded as bioindicators for bioavailability/bioaccessibility of PCDD/Fs, in favor of other species of greater commercial interest (fish and molluscs). The focal purpose of this paper is thus to underline how these organisms are suitable for studying bioaccumulation of organic contaminants (other than PAHs), as well as to promote a more widespread exploitation of polychaetes in monitoring programs.

MATERIALS AND METHODS

Study Area

The Venice Lagoon is one of the largest and relevant Coastal Transitional Ecosystems of the Mediterranean (Tagliapietra et al., 2009); it is about 50 km long and 10 km wide, accounting for a surface of about 550 km^2. Out of them, 36 km^2 are salt marshes, 30 km^2 islands (excluding the barrier islands) and the rest is covered by water. The mean depth of the water column is about 1.2 m, with only 5% of the lagoon deeper than 5 m (Molinaroli et al., 2009). The volume of water contained in the lagoon is about 628 million m^3, and according to Kjerfve (1994), the Venice Lagoon can be defined as a "restricted" coastal lagoon. The basin is connected to the Adriatic Sea through three inlets (Lido, Malamocco, and Chioggia) which allow tidal flushing twice a day (microtidal and predominantly semidiurnal tides) (Tagliapietra and Volpi Ghirardini, 2006). Every day the Venice Lagoon exchanges with the Adriatic Sea about 400 million m^3 of water while the inflow from the inland through the rivers and subsoil averages 3.7 million m^3 (Bernstein and Montobbio, 2011). The drainage basin is about 1850 km^2, 40% of which is reclaimed land lying under the sea level.

The primary source of pollution is the Porto Marghera industrial district (Bellucci et al., 2000; Frignani et al., 2001a,b; Zonta et al., 2007). Other point and non-point sources of pollution flowing into the lagoon include also treated and untreated municipal wastewaters, streams, agricultural runoff, boat traffic and atmospheric deposition (Guerzoni et al., 2004; Secco et al., 2005; Volpi Ghirardini et al., 2005; Gambaro et al., 2009). The recorded pattern of pollution follows the urban and industrial development: evidence exists that contamination of waters and sediments started in 1920 with the development of the first part of the industrial area, then accelerated after 1933 and again after World War II (Frignani et al., 2001a,b). As a consequence, the contamination gradient was largely superimposed on the inland-sea transect (Picone et al., 2016, 2018).

The atmospheric and riverine inputs of PCDD/F into the Venice Lagoon has been estimated in the order of 2.2–195.7 g y^{-1} (Guerzoni et al., 2007) and 6.9 g y^{-1} (Bettiol et al., 2005), respectively; the contribution of treated and untreated discharges from the industrial district of Porto Marghera has been estimated in 0.10–0.26 g y^{-1} as toxicity equivalents (MAG. ACQUE-Uta, 2011).

Sediment and Polychaete Sampling

Sediment and polychaete samples were collected during summer 1998 from 5 sub-tidal shallows, selected along a gradient of chemical contamination and affected by different sources of pollution (**Figure 1**). Palude della Centrega (CE) is an inter-tidal mudflat in the northern basin of the Lagoon, chosen as possible reference site due to the low concentrations of contaminants characterizing the area (Volpi Ghirardini et al., 2005; Picone et al., 2008). Dese river (DE) and Osellino canal (OS) sites are two estuaries influenced by agricultural runoff; OS is also characterized by multi-sources pollution being affected by both urban and industrial discharges, due to the proximity to the urban area of Mestre and the illegal landfill of Campalto (Critto et al., 2003). Canale Industriale Sud (SA) and Canale Lusore-Brentelle (BR) are two industrial canals located within the Porto Marghera district; in particular, site BR is a polluted site both as concern metals and organic micropollutants (Volpi Ghirardini et al., 2005; Bellucci et al., 2009; Picone et al., 2018) catching also freshwater from the Naviglio di Brenta canal.

Sediment sampling was performed following the integrated design and Quality Assurance/Quality Control (QA/QC) procedures reported in Volpi Ghirardini et al. (2005). Briefly, in each site, the sampling area was defined as a circle with a diameter of 30 m. Within this area, eight sediment cores (depth 0–20 cm, diameter 5 cm) were collected with a Plexiglas® corer and then pooled to obtain an integrated sample. This procedure was performed in triplicate in each sampling area. Sediment samples were stored in 2 l glass containers and kept refrigerated until their arrival in the laboratory. Samples were processed within 2 weeks, according to ASTM guidelines (ASTM, 2014).

Polychaetes were collected within sediment sampling areas using a box corer (14×14×16 cm). Sediments collected with the box corer were washed into polyethylene tanks and then the sediment slurry was poured into a 1 mm mesh size sieve and thoroughly washed *in situ* with seawater to separate the worms from the substrate. Once collected, the polychaetes were placed in 1 liter polyethylene containers filled with a layer of clean quartz sand (grain size approx. 60–100 μm) and natural seawater, in order to minimize the stress during the transport in the laboratory. A variable number of polychaetes was retrieved in each site, from 30 up to 80 depending on the population density in the area. Only for sites BR and OS it was possible to obtain three replicates with sufficient biological material for analysis; as a consequence, it was necessary to pool organisms collected in each site in order to have enough biological material for performing chemical analysis with an adequate limit of detection (0.25 pg g^{-1} dw).

Sediment Analysis

Total organic carbon (TOC) analyses were performed using a CHNS-O analyzer on aliquots of 10–20 mg of dry sediment acidified with 20-μL of 1N HCl solution and dried at 105°C for 15 min. Sediment grain-size was determined following a gravimetric procedure and was subsequently classified according to Shepard (1954). PCDD/F in the sediments were analyzed according to the procedure reported in Bellucci et al. (2000), based on the United States. EPA method 1613/94 for the determination of 17 congeners 2,3,7,8 substituted of dioxins and furans using gas chromatography-mass spectrometry (GC-MS). One composite sample (n = 1) per site was analyzed.

Polychaete Holding and Analysis

Once in the laboratory, the polychaetes were depurated for 4 days in small glass aquaria (20 × 20 × 15 cm) containing acid-washed quartz sand and artificial seawater (1:1 v/v sand/water ratio), to allow the elimination of the gut's content. Then, they were transferred for 1 day in aquaria containing only artificial seawater, to remove also the quartz sand accumulated during depuration phase (Lobel et al., 1991; Volpi Ghirardini et al., 1999). Artificial seawater was prepared by dissolving an artificial sea salt mixture (Ocean Fish®, PRODAC International, Cittadella, Italy) in Milli-Q® purified water.

Depuration was carried out under constant aeration, at a temperature of 18°C and salinity of 20 psu; overlying water was renewed every 48-h and the fecal pellets were removed daily, to avoid the ingestion by the worms and the consequent failure of the gut cleaning. Temperature, salinity and ammonia were checked daily.

During the last day of depuration in seawater, glass tubes were introduced into the aquaria to allow polychaetes to individually settle in there, to avoid aggressive behavior, cannibalism and also damages due to the lack of substrate (e.g., autotomy). At the end of the depuration phase, animals were anaesthetized using a 10% ethanol solution in seawater 20% and individually checked for species confirmation under a dissecting microscope. After taxonomic confirmation and determination of biometric parameters (in order to select polychaetes of comparable age), pooled samples of 20–30 specimens per sampling site were frozen until the analysis. Taxonomic identification was performed in order to avoid mixing with specimens belonging to other species, such as *Alitta succinea* (Leuckhart, 1847) and *Perinereis cultrifera* (Grube, 1840), that may share the habitat with *H. diversicolor* (Volpi Ghirardini et al., 1999). PCDD/F in the tissues were analyzed using the same method reported for sediments; prior to analysis, the frozen polychaete samples were freeze-dried. One composite sample (n = 1) per site was analyzed.

Lipids were extracted by Accelerated Solvent Extraction (ASE) with 50 ml of n-hexane/dichloromethane 50/05 at 150°C, 1500 psi 7-min heat-up and two cycles of 5 min static time. The extracts

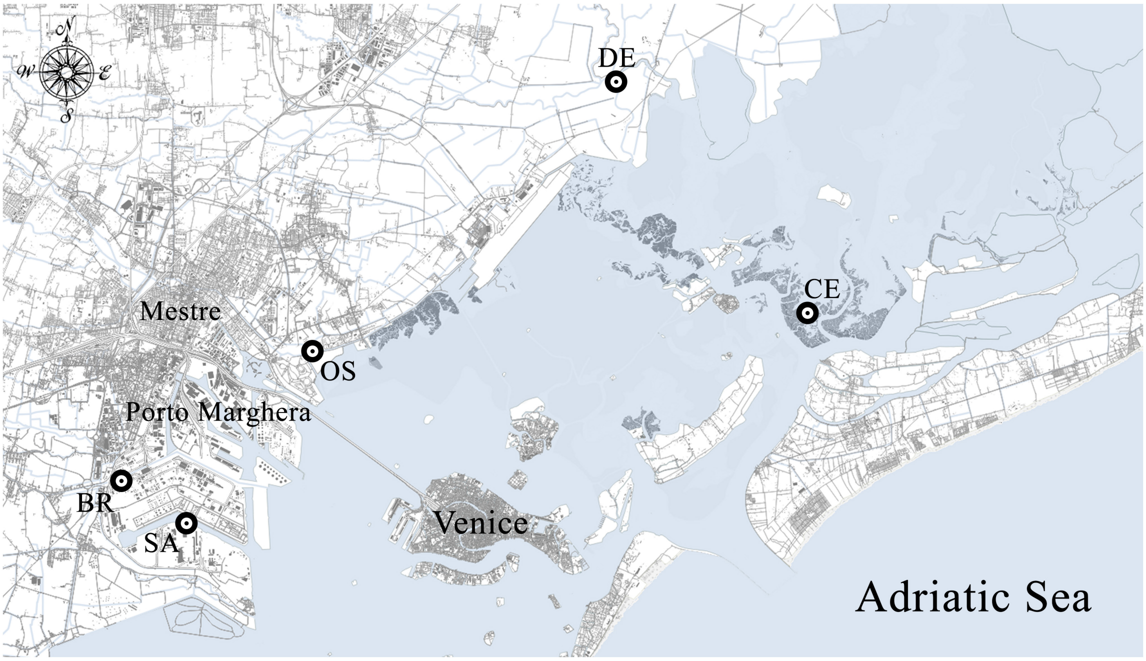

FIGURE 1 | Map of the sampling sites within the Venice Lagoon. CE = Palude della Centrega; DE = Dese river; OS = Osellino canal; SA = Canale Industriale Sud; BR = Canale Lusore-Brentelle.

were desiccated using an evaporator equipped with a vacuum controller. The lipid content was then determined gravimetrically using an analytical balance.

Data Elaboration and Analysis

Concentrations of PCDD/F in sediments and tissues were reported on dry weight (dw) basis. The dry weight concentrations were then also converted in 2,3,7,8-TCDD equivalents by multiplying the sediment concentrations by the toxic equivalency factors (TEFs) proposed by the World Health Organization (Van den Berg et al., 2006). Cluster analysis was used to categorize the homolog profile of PCDD/F in sediments and tissues; Euclidean distances and complete linkage were used as distance metric and joining rule, respectively. Before cluster analysis, sediment and tissue concentrations were transformed to per mil 2,3,7,8 chlorosubstituted homologs, by normalizing the sum of the isomers with the same degree of chlorination (i.e., PeCDD or HxCDD) to the sum of all dioxins and furans congeners with a concentration above the detection limit (Marcomini et al., 1997).

Spearman's non-parametric correlation was applied to test for significant correlations between sediment and tissue concentrations of the 17 congeners of PCDD/F, whilst linear regression was used to verify the linkage between BSAFs and the octanol-water partition coefficients (K_{OW}), after logarithmic transformation. LogK_{OW} values were taken from Chen et al. (2001).

Biota-Sediment Accumulation Factors (BSAFs) were calculated for each sampling site as the ratio of contaminant concentration in tissues (pg g^{-1} dw) on the contaminant concentration in sediments (pg g^{-1} dw), normalized to the lipid content and to the organic carbon content, respectively (Burkhard, 2009). For the calculation of BSAFs, concentrations below detection limits were assumed to equal a value half of the detection limit used. Welch's ANOVA and Games-Howell test were applied to the pooled BSAF data in order to check for the overall significant difference in the accumulation of the congeners of PCDD/F. Welch's ANOVA was chosen as a parametric method for the analysis of variance since homogeneity of variances condition was not met also after logarithmic and square root transformations.

All the statistical analysis were performed using the StatSoft Statistica v8.0 (Spearman's non-parametric correlation, linear regression) and IBM SPSS Statistics v.25 software (Welch's ANOVA and Games-Howell *post hoc* test).

RESULTS

Sediment Chemistry

Total organic carbon in the sediments ranged from 1.88% (CE) up to 4.67% (OS); grain size was characterized by a prevailing fraction of silt, ranging from 52.9% (BR) up to 69.6% (SA). According to the ternary classification of Shepard (1954) samples CE, DE, OS, and SA were classified as clayey silts, whilst sample BR was classified sandy silt.

Sediment PCDD/F concentrations at the five study sites ranged between 73 and 6,621 pg g^{-1} dw. The sediments collected at site SA (6,621 pg g^{-1} dw) were the most contaminated being characterized by total concentrations (ΣPCDD/F) of 1 or 2 order of magnitude higher than the other samples (**Table 1**). As concern dioxins, the total sediment concentrations ranged from 24 pg g^{-1} dw (CE) to 677 pg g^{-1} dw (OS), with OCDD accounting for more than 75% of total dioxins in all samples; 2,3,7,8-TCDD occurred at concentrations above the detection limit (0.5 pg g^{-1} dw) only in sample SA (1 pg g^{-1} dw). With regard to furans, the highest concentration was measured in sample SA (6061 pg g^{-1} dw) whilst the reference sample collected in site CE was, as expected, the least contaminated one (47 pg g^{-1} dw); the congeners detected at highest concentrations were 1,2,3,4,6,7,8-HpCDF and 1,2,3,4,6,7,8,9-OCDDF.

The equivalent toxicity ΣTE (WHO-TEQ) in the sediment samples ranged from 2 pg g^{-1} dw (CE) to 60 pg g^{-1} dw (SA) and showed the same gradient as the total concentrations: CE < DE < BR < < OS < SA.

Homolog profiles in sediment showed different patterns among sites, especially as concern the contribution of OCDD and OCDF (**Figure 2**). In sample SA, the profile was dominated by OCDF and characterized by negligible contributions of the other homologs except for OCDD and HpCDF; in contrast, in samples BR and OS the profiles showed a dominance of OCDD over the more chlorinated furans. In DE and CE the profile was characterized by a gradient of increasing concentrations of PCDD/F with increasing degree of chlorination, but with a dominance of OCDF over OCDD and HpCDF. Cluster analysis confirmed this categorization of samples into three groups (**Figure 3**).

Tissue Analysis

Polychaete tissues were characterized by a variable lipid concentration, ranging from 0.10 to 0.14 g g^{-1} of tissue, whilst organic carbon content averaged 0.53 g g^{-1} of tissue (range: 0.52–0.54 g g^{-1}).

Real total concentration (ΣPCDD/F) in polychaete tissues ranged from 33 pg g^{-1} dw (DE) up to 300 pg g^{-1} dw (SA) (**Table 1**). The gradient of real total concentration was as follows: DE < CE < OS < BR < SA. The concentrations of dioxins were quite homogeneous among the samples, ranging from 13 pg g^{-1} dw (DE) to 30 pg g^{-1} dw (BR). In samples CE and DE only 1,2,3,7,8-PCDD, 1,2,3,4,6,7,8-HpCDD and OCDD occurred at concentrations above the detection limits (0.25 pg g^{-1} dw); this latter was also the congener exhibiting the highest concentration in all the samples. As observed in the sediments, 2,3,7,8-TCDD was detectable only in sample SA. Site DE also showed the lowest concentration of furans (21 pg g^{-1} dw), whilst the highest were detected in the tissues of the polychaetes sampled in the industrial canals BR (111 pg g^{-1} dw of furans) and SA (281 pg g^{-1} dw of furans). The congeners occurring at higher concentrations were 2,3,7,8-TCDF and 1,2,3,4,6,7,8,9-OCDF. Concentrations normalized to lipid content ranged from 268 pg g^{-1} lipids (DE) to 2111 pg g^{-1} lipids (SA).

The equivalent toxicity in the tissues ranged from 3 pg g^{-1} dw (DE) to 26 pg g^{-1} dw (SA); the gradient (DE < CE < BR < OS < <SA) was slightly different from the one for total concentrations, since OS showed higher equivalent toxicity with respect to BR (10 pg g^{-1} dw vs. 5 pg g^{-1} dw), despite lower total concentration (104 pg g^{-1} dw vs. 141 pg g^{-1} dw).

Cluster analysis on homolog profiles evidenced the occurrence of distinct patterns of accumulation in the tissue (**Figure 3**). At site BR the polychaetes accumulated mostly OCDF and secondarily OCDD and TCDF, whereas the polychaetes collected at sites SA and OS accumulated primarily furans (contributing up to 87% and 94% to ΣPCDD/F, respectively), with a prevalence of HxCDF, TCDF and PeCDF, over the other furans and OCDD. The samples with less accumulation, DE and CE, showed a more heterogeneous pattern, with an accumulation of OCDD and less chlorinated PCDF (DE) or OCDD and more chlorinated PCDF (CE) (**Figure 4**). In any case, differences in homolog concentrations in these samples are very low.

Comparison Between Sediments and Tissues

No correlation was observed between PCDD congeners in the sediment and in the polychaete tissues ($p > 0.05$); in the case of furans, significant correlations (Spearman's $R > 0.9$; $p < 0.05$) were found for the PeCDFs, HpCDFs and two congeners of HxCDFs (namely, 1,2,3,4,7,8-HxCDF and 1,2,3,6,7,8-HxCDF). Spearman's correlation was not calculated for 2,3,7,8-TCDD and 1,2,3,7,8,9 HxCDD due to the large number of data below detection limits.

Sediment Accumulation Factors value ranged between 0.003 (OCDF in site SA) and 11.13 (TCDF in site BR). Hepta- and octachlorinated congeners showed a BSAF about one order of magnitude lower than the less chlorinated congeners (**Table 1**).

Linear regression revealed a significant relationship with a decreasing trend between logBSAF and logK_{Ow} ($F_{1,83} = 43.878$; $r^2 = 0.289$; $r = -0.539$; $p < 0.0001$), indicating a tendency to accumulate preferentially the congeners with lower logK_{OW} (**Figure 5**).

Welch's ANOVA highlighted significant differences among the BSAFs calculated for the congeners ($F_{16,25} = 6.108$, $p < 0.001$); Games-Howell test detected significant differences ($p < 0.05$) between 2,3,7,8-TCDF and several congeners (**Figure 6**).

DISCUSSION

The lack of replication did not allow to verify whether among-site differences are significant (especially for the less contaminated CE and DE samples) and may raise issues concerning the analytical variability of the data. However, specific contamination patterns emerged both in sediments and in polychaete tissues that are consistent with the underlying contamination gradient and other studies (Jimenez et al., 1998; MAG. ACQUE-Thetis, 2006, 2007; Picone, 2006), suggesting that analytical variability is of secondary relevance as compared with environmental variability of PCDD/F contamination in the Venice Lagoon.

PCDD/F in the Sediments

Sediment total concentrations and toxicity equivalents corroborated expectations for the underlying contamination gradient. The lowest concentration of PCDD/F were measured at the reference site CE and at the estuarine site DE located far from the industrial district, whilst the highest were observed within

TABLE 1 | PCDDs and PCDFs concentrations in composite sediment samples, composite polychaetes samples and Biota-to-Sediment Accumulation Factors (BSAFs) calculated for the 17 congeners of 2,3,7,8 chlorosubstituted dioxins and furans.

Congener	Sediment real concentration (pg g^{-1} dw)					Tissue real concentration (pg g^{-1} dw)					Sediment concentration normalized to organic carbon (pg g^{-1} C)				
	BR	CE	DE	OS	SA	BR	CE	DE	OS	SA	BR	CE	DE	OS	SA
2,3,7,8 TCDD	<0.5	<0.5	<0.5	<0.5	1	<0.25	<0.25	<0.25	<0.25	1	<19	<27	<16	11	29
1,2,3,7,8 PCDD	<0.5	<0.5	<0.5	2	4	1	2	1	2	2	<19	<27	<16	51	95
1,2,3,4,7,8 HxCDD	<0.5	<0.5	<0.5	2	13	1	1	1	1	1	<19	<27	<16	51	280
1,2,3,6,7,8 HxCDD	<0.5	<0.5	1	4	7	1	2	1	1	2	<19	<27	39	84	150
1,2,3,7,8,9 HxCDD	<0.5	<0.5	4	<0.5	<0.5	1	1	1	1	1	<19	<27	136	11	11
1,2,3,4,6,7,8 HpCDD	35	3	21	96	99	5	3	2	2	4	1,364	133	686	2,058	2,194
1,2,3,4,6,7,8,9 OCDD	194	20	87	571	435	22	10	9	8	9	7,519	1,059	2,816	12,227	9,603
2,3,7,8 TCDF	<0.5	<0.5	2	9	16	14	2	5	21	69	<19	<27	68	193	362
1,2,3,7,8 PCDF	6	2	3	19	66	3	2	3	13	28	225	96	107	400	1,464
2,3,4,7,8 PCDF	3	1	1	9	15	4	2	2	9	22	101	<27	36	197	336
1,2,3,4,7,8 HxCDF	12	2	5	32	153	4	3	2	12	50	473	96	159	681	3,377
1,2,3,6,7,8 HxCDF	4	1	2	15	65	2	2	1	5	17	151	59	78	323	1,426
1,2,3,7,8,9 HxCDF	30	<0.5	<0.5	115	10	1	2	1	1	3	1,163	<27	<16	2,463	214
2,3,4,6,7,8 HxCDF	1	<0.5	2	16	42	2	2	1	4	10	54	<27	68	334	927
1,2,3,4,6,7,8 HpCDF	61	13	55	295	1,305	10	5	2	12	36	2,345	676	1,780	6,317	28,808
1,2,3,4,7,8,9 HpCDF	9	3	5	17	192	2	2	1	3	11	357	133	175	373	4,238
1,2,3,4,6,7,8,9 OCDF	143	26	141	198	4,197	68	15	3	9	34	5,543	1,399	4,563	4,240	92,649
ΣPCDD	232	25	115	677	560	31	19	13	14	19	8,981	1,324	3,725	14,493	12,362
ΣPCDF	269	48	218	725	6,061	111	37	21	90	281	10,430	2,564	7,049	15,520	133,801
Σ PCDD/F	501	73	333	1,402	6,621	142	56	33	104	300	19,411	3,888	10,773	30,013	146,163
Σ TE (WHO-TEQ)	8	2	4	30	60	5	4	3	10	26	27	28	29	30	31

(Continued)

TABLE 1 | Continued

Congener	Tissue concentration normalized to lipid content (pg g^{-1} lipid)					BSAF					Spearman's correlation	
	BR	**CE**	**DE**	**OS**	**SA**	**BR**	**CE**	**DE**	**OS**	**SA**	**R**	**p**
2,3,7,8 TCDD	1	1	1	1	8	>0.10	>0.09	>0.12	>0.20	0.27	n.c.	n.c.
1,2,3,7,8 PCDD	7	19	7	13	14	>0.73	>1.41	>0.89	0.25	0.15	0.69	0.199
1,2,3,4,7,8 HxCDD	6	14	4	4	8	> 0.65	> 1.04	> 0.50	0.08	0.03	0.00	1.000
1,2,3,6,7,8 HxCDD	4	15	4	4	11	> 0.41	> 1.12	0.10	0.05	0.07	0.15	0.812
1,2,3,7,8,9 HxCDD	6	12	4	4	7	> 0.65	> 0.89	0.03	> 0.41	> 0.64	n.c.	n.c.
1,2,3,4,6,7,8 HpCDD	39	29	13	21	25	0.03	0.22	0.02	0.01	0.01	0.30	0.623
1,2,3,4,6,7,8,9 OCDD	176	95	70	72	62	0.02	0.09	0.02	0.01	0.01	−0.50	0.391
2,3,7,8 TCDF	108	22	43	181	487	> 11.13	> 1.64	0.63	0.94	1.35	0.82	0.089
1,2,3,7,8 PCDF	24	23	20	116	197	0.11	0.24	0.19	0.29	0.13	**0.99**	<**0.001**
2,3,4,7,8 PCDF	29	18	15	78	155	0.29	> 0.67	0.43	0.40	0.46	**0.99**	<**0.001**
1,2,3,4,7,8 HxCDF	34	26	15	108	354	0.07	0.27	0.10	0.16	0.10	**0.90**	**0.037**
1,2,3,6,7,8 HxCDF	16	19	8	46	117	0.10	0.32	0.10	0.14	0.08	**0.90**	**0.037**
1,2,3,7,8,9 HxCDF	9	19	4	10	19	0.01	> 1.41	> 0.50	0.00	0.09	0.00	1.000
2,3,4,6,7,8 HxCDF	17	24	10	38	72	0.32	> 1.79	0.15	0.11	0.08	0.60	0.285
1,2,3,4,6,7,8 HpCDF	80	45	15	102	256	0.03	0.07	0.01	0.02	0.01	**0.90**	**0.037**
1,2,3,4,7,8,9 HpCDF	18	22	8	24	80	0.05	0.16	0.05	0.07	0.02	**0.90**	**0.037**
1,2,3,4,6,7,8,9 OCDF	539	150	26	78	239	0.10	0.11	0.01	0.02	0.00	0.30	0.624
ΣPCDD	241	186	104	122	134							
ΣPCDF	874	366	165	782	1,977							
Σ PCDD/F	1,115	552	269	904	2,111							
Σ TE (WHO-TEQ)	32	33	34	35	36							

Concentration data are reported as real and normalized to organic carbon and lipid concentrations. For BSAFs calculation, data below detection limits were assigned a value equal to half detection limit. Results of correlation analysis performed on sediment and tissue dry weight data are also reported. Data highlighted in bold indicate significant correlations. n.c. = not calculated due to occurrence of too data under detection limit.

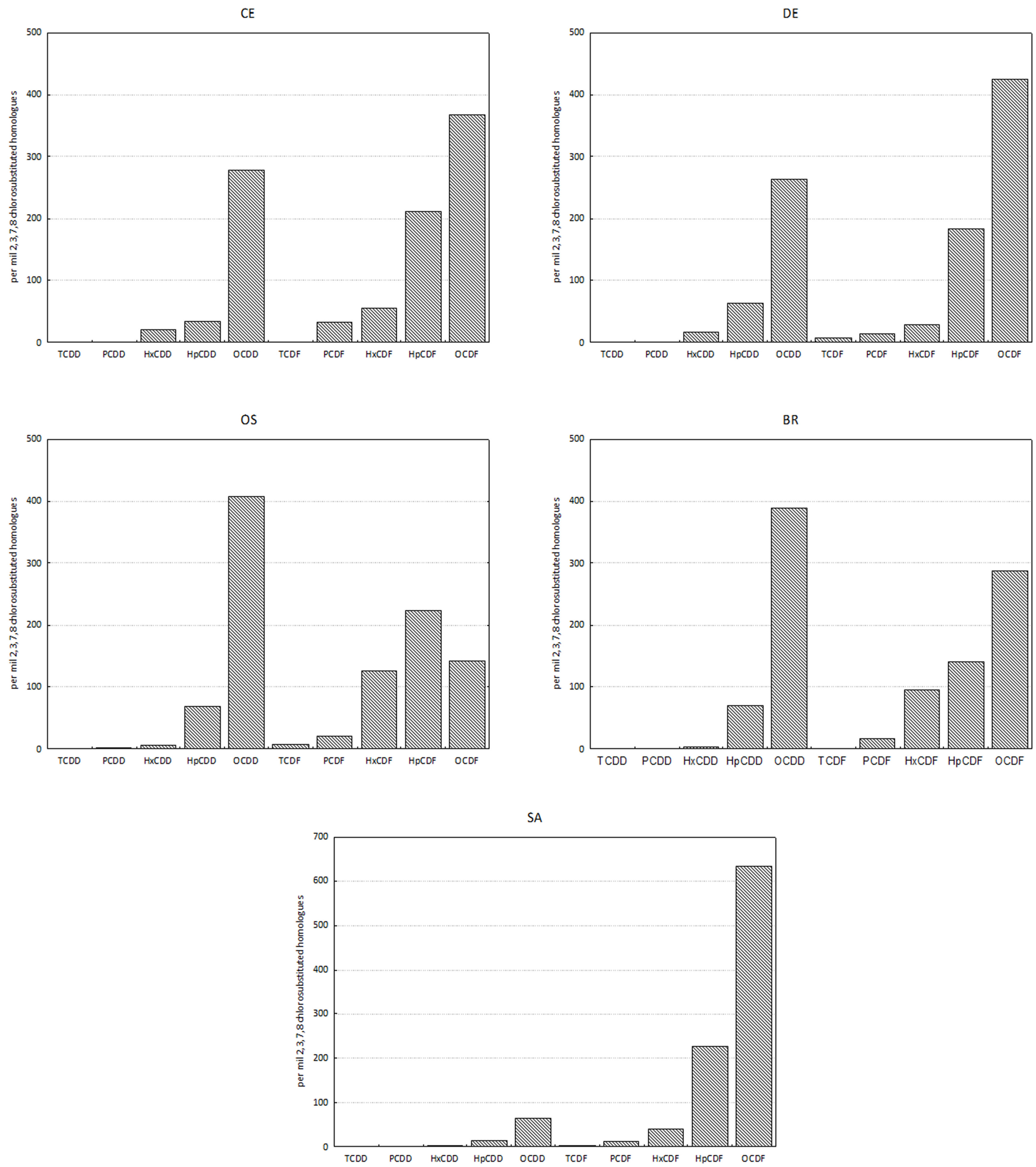

FIGURE 2 | Homolog profiles of PCDDs and PCDFs in sediments. Data are reported as per mil 2,3,7,8 chlorosubstituted homologs referred to the sum of all the isomers with concentration above detection limits.

the industrial area (SA) and at the estuarine site OS located in the proximity of the illegal landfill and also receiving urban and industrial discharges.

A review of PCDD/F concentration in the sediments of the Venice Lagoon reported a mean value of 14,000 pg g^{-1} dw of ΣPCDD/F for the industrial channels and 1,000 pg g^{-1} dw of ΣPCDD/F for the inner lagoon (Guerzoni et al., 2007). These data are higher than those observed in the present study, both for the industrial district and the reference and estuarine samples; nevertheless, the comparison with literature data is often complicated by the different depth of the sediment core analyzed in the various studies. For the industrial district, many data refer to the first 2 cm of sediments, where concentrations in the range 67–13,642 pg g^{-1} for dioxins and 592–126,561 pg g^{-1} for furans were reported (Bellucci et al., 2000; Frignani et al., 2001a); other projects investigated the vertical profile of PCDD/F

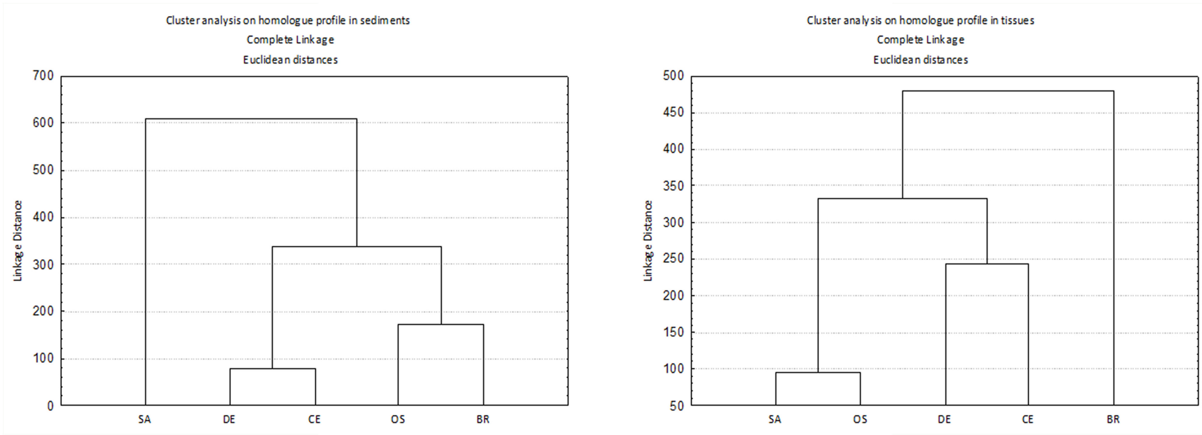

FIGURE 3 | Cluster analysis on homolog profile in sediments (left) and tissues (right). Before analysis, data were transformed to per mil 2,3,7,8 chlorosubstituted homologs referred to the sum of all the isomers with concentration above detection limits.

in industrial channels and reported concentrations up to 36,590 pg g^{-1} dw for dioxins in the layer 35–38 cm depth and up to 19,241 pg g^{-1} dw for furans in the layer 8–10 cm depth (Frignani et al., 2001a).

Later studies conducted by the Venice Water Authority investigated the top layer to a depth of 25 cm (MAG. ACQUE-Thetis, 2007) in the shallows of the inner lagoon facing the industrial district and reported concentrations in the range 13–8,312 pg g^{-1} dw of ΣPCDD/F and equivalent toxicity (WHO-TEQ) up to 112 pg g^{-1} dw. These data are in agreement with the concentrations observed in the present study, both for the industrial district (SA, BR) as well as the estuarine sample OS. At this latter sample, the measured concentrations (1,402 pg g^{-1} dw as ΣPCDD/F) were also higher than those reported in the literature (depth 0–20 cm) for the inner lagoon shallows opposite to the estuary and in the proximity of the illegal landfill of Campalto (120 pg g^{-1} dw as ΣPCDD/F) (Picone, 2006). These data suggest that the pollution of the area could be due mainly to the urban and industrial discharges upstream, rather than to leakages from the landfill.

Fewer data are available for the sites DE and CE. In surface sediment (0–5 cm) collected in the estuarine site DE, Jimenez et al. (1998) measured a total ΣPCDD/F concentration of 79 pg g^{-1} dw.

The profiles of the homologs and the cluster analysis evidenced three distinct patterns of PCDD/F at the five study sites, that may be related to different sources of contamination. In the sediment of the industrial area Bellucci et al. (2000) already identified three distinct type of homolog profiles that fit with the main local sources: (type 1) profile dominated almost entirely by OCDD, produced by combustion processes, untreated domestic sewage, urban wastes and boat engine exhausts; (type 2) profile with the dominance of OCDF (90%) and OCDD (10%) that could be ascribed both to the stripping of vinyl chloride monomer (Stringer et al., 1995) and to the accumulation of atmospheric contaminants caused by both the runoff from the industrial district and the wastes of a factory for the production of liquid gases (Frignani et al., 2001a); and (type 3) profile containing mainly furans, with a prevailing component of OCDF, probably due to the production of metals, coke and related chemicals in the industrial district.

Nevertheless, in the present study only the sample collected from Canale Industriale Sud (SA), showed a profile that could be categorized into one of the three types outlined by Bellucci et al. (2000) since the homologs provided a specific pattern of furans with a dominance of OCDF, attributable to industrial production of coke, metals and related manufactories (type 3), whilst contribution of dioxins in negligible. In the remaining four samples, the profiles did not allow for the identification of a prevailing source, but indicate the co-occurrence of multiple sources of contamination. Samples CE and DE were characterized by a pattern of PCDD/F, consistent with atmospheric deposition from industrial sources. However, their profiles showed also a noteworthy contribution of OCDD, suggesting a not-negligible contribution of combustion processes unrelated to industrial pollution and boat traffic. In contrast, in samples OS and BR the dominance of OCDD over OCDF and HpCDF suggested a major contribution of deposition due to combustion combined with untreated domestic sewage over the deposition of furans from industrial sources.

PCDD/F in Tissues

In general, very few data are available in the literature concerning the field bioaccumulation of PCDD/F in polychaetes, although ragworms and lugworms are widely used as a bioindicator for biomarker and bioaccumulation studies (Ruus et al., 2005; Durou et al., 2007a,b) and are a key component of the coastal and estuarine food webs. Studies on *H. diversicolor* were performed by Nunes et al. (2011), that reported a concentration in tissues of 1.38 pg g^{-1} ww of ΣPCDD/F (81.16 pg g^{-1} lipid) for the Mondego estuary, Portugal. These data were lower than the concentration normalized to lipids observed both in the industrial district and in the estuarine sites of the Venice Lagoon

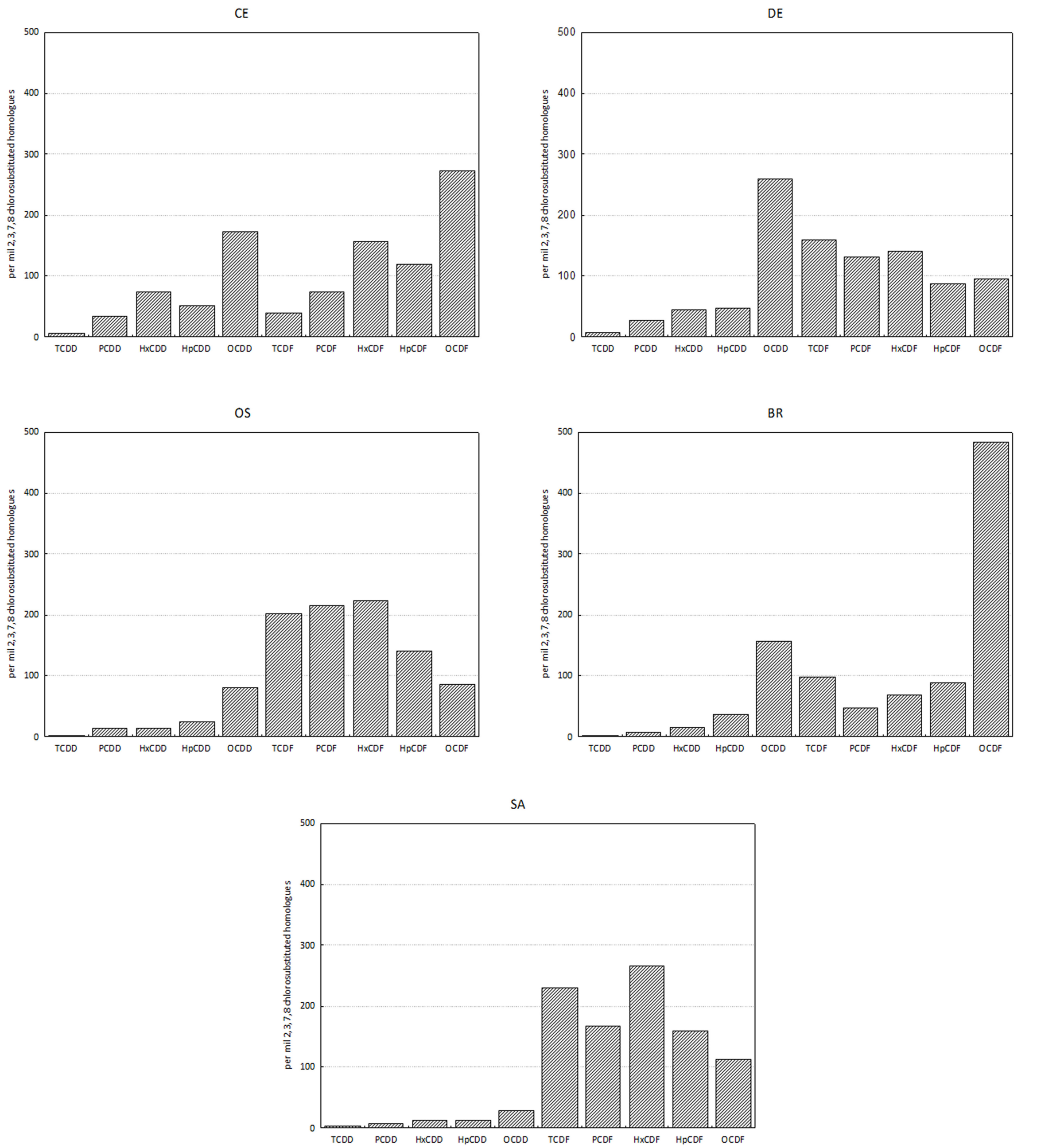

FIGURE 4 | Homolog profiles of PCDDs and PCDFs in tissues of *H. diversicolor*. Data are reported as per mil 2,3,7,8 chlorosubstituted homologs referred to the sum of all the isomers with concentration above detection limits.

in the present study (269–2,111 pg g^{-1} lipid), and may be related to the different levels and kind of pollution sources affecting the two estuaries.

In the Venice Lagoon the bioaccumulation of PCDD/F from the sediments has been most often assessed by measuring tissue concentration in clams and fishes, and never in polychaetes. Various studies have focused on the concentrations of PCDD/F in the Manila clam (*Ruditapes philippinarum*) collected both in the industrial district and in the inner lagoon: Sfriso et al. (2014) measured concentrations in the range of 40–110 pg g^{-1} dw for the inner lagoon, corresponding to an equivalent toxicity (WHO-TEQ) of 0.49–1.46 pg g^{-1} ww; Raccanelli et al. (2004, 2008) reported for the industrial area an equivalent toxicity of 2–9 pg g^{-1} ww; MAG. ACQUE-Thetis (2006) reported concentrations in the range 40–717 pg g^{-1} lipid and 9–58 pg g^{-1} lipid for industrial area and inner lagoon, respectively. In this latter study also the levels of PCDD/F in the muscle of the grass goby *Zosterisessor ophiocephalus* were

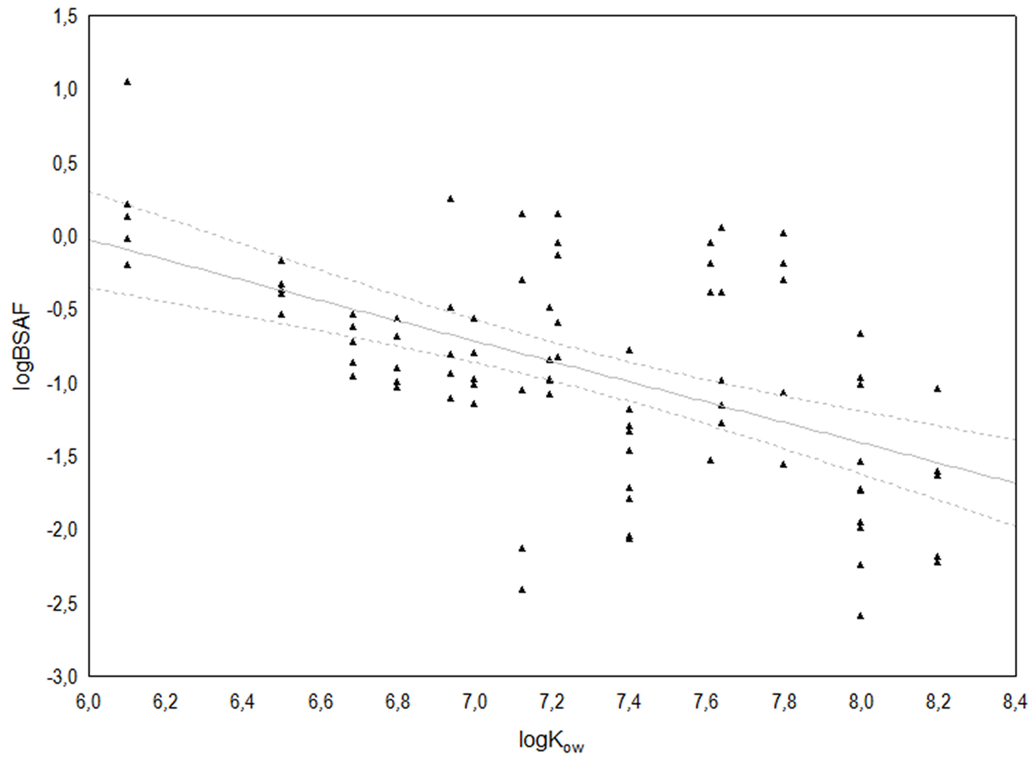

FIGURE 5 | Relationship between log K_{OW} and log BSAF for the 17 congeners of 2,3,7,8 chlorosubstituted dioxins and furans. The solid line represents the linear regression log(BSAF) = 4.12–0.69 log(K_{OW}) (r^2 = 0.289), whilst broken lines indicate the 95% confidence interval.

measured, ranging between 14 and 435 pg g^{-1} lipid for the inner lagoon and 78–640 pg g^{-1} lipid for the industrial area (MAG. ACQUE-Thetis, 2006). Tissue concentrations in gobies are in agreement with the observed concentrations measured in polychaete tissues in the present study. Other authors also reported data for the thicklip gray mullet *Chelon labrosus* (1.6–2.6 pg g^{-1} ww of Σ PCDD/F) and the crab *Carcinus aestuarii*, both in the industrial district (36 pg g^{-1} ww of Σ PCDD/F) and in the inner lagoon (1.5–5 pg g^{-1} ww of Σ PCDD/F) (Jimenez et al., 1998; MAG. ACQUE, 1999).

Homolog profiles indicated that the polychaetes accumulated OCDD preferentially as concern dioxins, whilst two different patterns are evident for furans: at sites CE and BR the dominant congener was the more chlorinated OCDF, whilst in samples SA, OS and also DE there was a relevant contribution of the less chlorinated congeners. In particular, there was a notable contribution provided by 2,3,7,8-TCDF, a typical marker of pollution due to chloralkali plants (Jimenez et al., 1998).

Comparison Between Sediments and Tissues

Sediment concentrations were higher than tissue concentrations for all the analyzed PCDD/F, apart from the less chlorinated furan 2,3,7,8-TCDF and 2,3,4,7,8-PeCDF, whose concentrations were higher in the tissues of *H. diversicolor* than in the sediments at the same stations. The differences were up to one order of magnitude for the hepta- and octa-chlorinated congeners, especially at sites DE, OS and SA, whilst the trend was less evident for the hexa-chlorinated congeners.

Homolog profiles and Spearman's correlation showed a generally low association between congener concentrations in sediment and polychaete tissues, resulting in large variability among the corresponding BSAF values. On the other hand, the significant linear relation observed between logBSAF and logK_{OW}, despite having a low coefficient of determination (r^2 = 0.289), outlined polychaete's tendency to take up preferentially the less chlorinated and less lipophilic congeners, characterized by lower logK_{OW}, as also observed by Kono et al. (2010) in a laboratory study aiming to model the dioxin transfer from sediments to *Perinereis nuntia*. These authors also evidenced that bioaccumulation of PCDD/F in polychaetes follow similar characteristics to those observed in fishes (Sijm et al., 1993).

The congeners 2,3,7,8-TCDD and 2,3,7,8-TCDF are characterized by high bioaccumulation and low elimination; thus their BSAF tends to be higher than those of the more chlorinated congeners (Sijm et al., 1993; Kono et al., 2010). In the case of *H. diversicolor* this is evident for 2,3,7,8-TCDF, whose accumulation factors were the highest in all the analyzed sample (except CE) despite the low sediment concentrations;

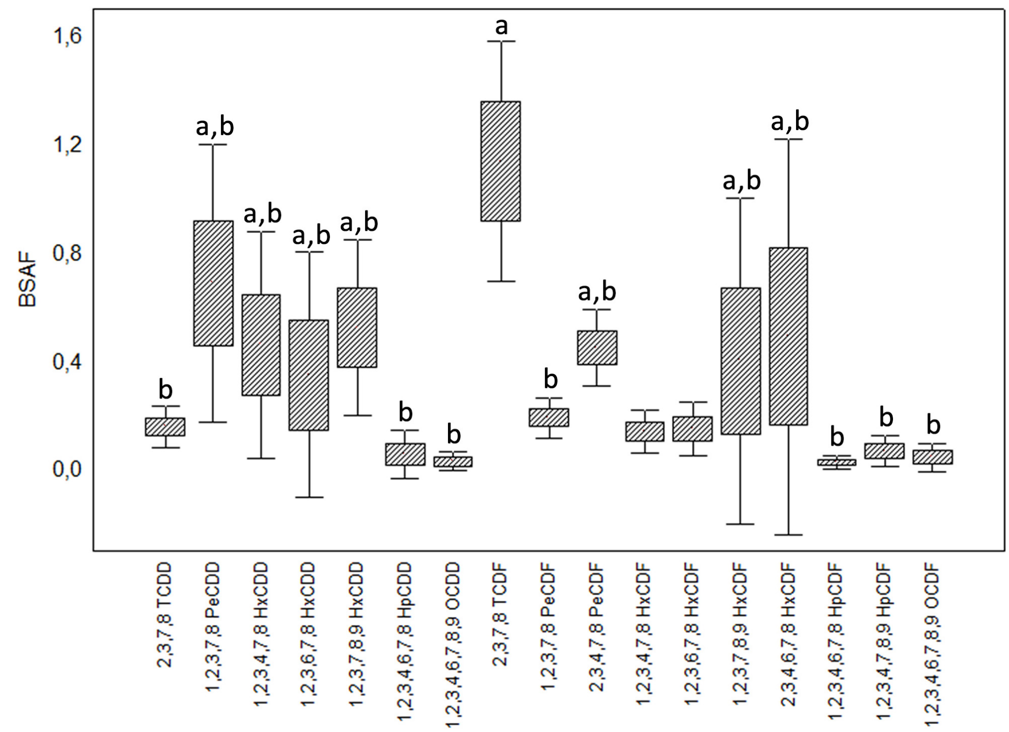

FIGURE 6 | Biota-to-Sediment Accumulation Factors (BSAF) calculated for the 17 congeners of 2,3,7,8 chlorinated dioxins and furans. Boxes indicate mean ± standard error, whiskers mean ± standard deviation. Letter "a" identifies congeners with mean BSAF statistically different from 2,3,7,8-TCDF (after Kruskal-Wallis ANOVA and multiple comparison test).

for 2,3,7,8-TCDD this trend is not tangible, probably due to the sediment concentrations below detection limits that impeded any further analysis. Preferential accumulation of tetrachlorinated congeners has also been observed by Pruell et al. (2000) for the Nereididae *Alitta virens* following 70-d laboratory exposure to contaminated sediments: in this case, polychaetes accumulated only 2,3,7,8-TCDD and 2,3,7,8-TCDF, although the sediments were characterized by a range of PCDD/F and also high concentrations of highly chlorinated congeners.

On the other hand, 1,2,3,4,6,7,8-HpCDD, HpCDFs, 1,2,3,4,6,7,8,9-OCDD and 1,2,3,4,6,7,8,9-OCDF are congeners characterized by low bioaccumulation, as applies to all the 1,4-substituted and/or 6,9-substituted PCDD/F (Sijm et al., 1993; Kono et al., 2010). Both low bioavailability and elimination may contribute to the low accumulation (and also BSAF) of highly chlorinated PCDD/F. Lipophilicity is a major contributor for determining the bioavailability of sediment-bound PCDD/F, as confirmed by the negative relationship of BSAF versus K_{OW} in *H. diversicolor* (this study) and *P. nuntia*. However, contaminant molecular size and conformation (specifically planarity/nonplanarity), sediment characteristics and feeding habits may also play a significant role, as observed in oligochaetes (Lyytikäinen et al., 2003). Elimination of highly chlorinated PCDD/F in caddisfly larvae and oligochaetes fits a biphasic model (Loonen et al., 1997; Pastershank et al., 1999). A first, rapid depuration phase was attributed to the elimination of the gut content, facilitated by a low assimilation efficiency for highly chlorinated compounds, whilst the slower phase of the depuration was attributed to elimination from other body compartments. The elimination process in polychaetes has not yet been elucidated, but a significant correlation between the elimination constant (k_2) and log K_{OW} has been observed by Kono et al. (2010), supporting the hypothesis that more chlorinated and hydrophobic congeners do not bioaccumulate since they are quickly eliminated. In fish and oligochaetes the more rapid elimination of super-hydrophobic organics as OCDD compared to lower chlorinated PCBs, TCDD and TCDF – also observed in human blood (Flesch-Janys, 1996) – also suggests that OCDD in invertebrates is mainly associated with fast-clearing compartments as cell membranes or blood serum (Kono et al., 2010).

The BSAFs for PCDD/F observed in the present study with *H. diversicolor* (0.02–1.41 for dioxins; 0.01 – 11.13 for furans) are at least one order of magnitude higher than those measured in *P. nuntia* (0.00036–0.22 for dioxins and 0.0002–0.36 for furans) (Kono et al., 2010); moreover, the data are in agreement with the BSAFs reported for other polychaetes such as *Nephtys* sp. (McFarland et al., 1994) and *A. virens* (Pruell et al., 1993, 2000; Schrock et al., 1997). In all cases, the BSAF obtained with *H. diversicolor* for 2,3,7,8-TCDF is higher than those reported for other polychaete species, suggesting a specific ability of *H. diversicolor* to accumulate this congener. Since bioavailability is a crucial factor driving bioaccumulation, and digestive fluids are the primary factors driving solubilization of organic contaminants in polychaetes (Voparil and Mayer, 2000; Ahrens et al., 2001), differences in BSAFs for 2,3,7,8-TCDF among *H. diversicolor* and other polychaete species may be due to differences in gut digestive chemistry. Concentrations and properties of digestive fluids vary broadly among species and it may affect the assimilation efficiency for some substances (Ahrens et al., 2001; Mayer et al., 2001). When compared to other invertebrates, the BSAFs calculated for *H. diversicolor* are in agreement with data reported for the freshwater oligochaete *Lumbriculus variegatus* (0.024–2.54), the crustaceans *Palaemonetes pugio* and *Callinectes sapidus* (0.089–0.73 in the hepatopancreas), the bivalves *Macoma nasuta* and *Corbicula japonica* (0.004–0.22 for PCDD/F) (Pruell et al., 1993; Schell et al., 1993; Kang et al., 2002; Pickard and Clarke, 2008; Iannuzzi et al., 2011). Also in this case, similarities in BSAFs may be due to comparable efficiency of the digestive fluids in the solubilization of the contaminants, as shown by Mayer et al. (2001) for benzo[a]pyrene.

Sediment Accumulation Factor data suggest that *H. diversicolor* may accumulate PCDD/F significantly from the sediments and may serve as an indicator for the bioavailability of dioxins and furans from sediments to the polychaete community, especially as concerns the less chlorinated 2,3,7,8-TCDF, 2,3,4,7,8-PeCDF and 2,3,4,7,8-PeCDD. Nevertheless, the BSAFs for PCDD/F in polychaetes are generally low (BSAF < 1 or less) when compared to the BSAFs measured for other organochlorine contaminants (i.e., PCBs and organochlorine pesticides) in a number of species including *A. virens, N. incisa, Glycera* sp., *Perinereis nuntia* and also *H. diversicolor* (Lake et al., 1990; Brannon et al., 1993; Volpi Ghirardini et al., 2004; Kono et al., 2010; Nesto et al., 2010); this trend was also observed in fishes where the apparent lower bioaccumulation of PCDD/F was attributed to lower solubility in the lipids, reduced membrane transport and also biotransformation mediated by cytochrome P450 monooxygenase isozymes (Opperhuizen and Sijm, 1990; Sijm et al., 1993; van der Oost et al., 2003). Biotransformation of PCDD/F in polychaetes has been rarely studied, and no data are available concerning the possible mechanisms involved. Nevertheless, since cytochrome P450 isozymes are active also in polychaetes (Zheng et al., 2013), it cannot be excluded that the lower bioaccumulation of PCDD/F in ragworms could be due to the same causes already identified for fishes. Further studies are needed, however, to elucidate these possible mechanisms of biotransformation and elimination.

CONCLUSION

Total PCDD/F concentrations in sediments in Venice Lagoon showed a clear pollution gradient along the investigated sites, with higher levels of contamination in the samples collected in the industrial district (SA) and in the estuarine site receiving urban and industrial discharges from the area of Mestre and the illegal landfill of San Giuliano. Distinctly different homolog profiles discriminated between sites influenced by heterogeneous pollution sources (BR, OS, DE and CE) and the industrial site (SA), where a specific pattern of furans with a dominance of OCDF attributable to industrial production of coke, metals and related manufactories was observed.

In the ragworm tissues, the concentration pattern of the congeners is different from those observed in the sediments, especially with regard to furans: in sample CE and BR the polychaetes accumulated mostly 1,2,3,4,6,7,8,9-OCDF whilst in samples SA, OS and DE the major contribution was due to the less substituted congeners and principally 2,3,7,8-TCDF, recognized as a marker of possible contamination deriving from chloralkali industrial plants.

In general, *H. diversicolor* was a good indicator for assessing the transfer of PCDD/F from sediment to biota, but accumulated more efficiently the less chlorinated congeners with lower $\log K_{OW}$ resulting in a negative correlation between BSAF and K_{OW}, after logarithmic transformation.

Despite *H. diversicolor*'s ability to accumulate 2,3,7,8-TCDD, 2,3,7,8-TCDF and 2,3,4,7,8-PeCDF, the BSAFs calculated for the PCDD/F are low as compared with those reported in the literature for other organochlorine compounds (i.e., PCBs and organochlorine pesticides) in a number of species including *A. virens, N. incisa, Glycera* sp., *Perinereis nuntia* and also *H. diversicolor*. Lower BSAFs for PCDD/F may be due to different uptake kinetics, or the presence in polychaetes of a more efficient elimination pathway for PCDD/F than for PCBs or pesticides, possibly involving also cyt-P450-mediated biotransformations; further research is needed to identify both biotransformation and elimination processes driving accumulation in polychaetes. Due to the paucity of information on this species, this research not only represents a baseline but also an invitation for further ecotoxicological studies on *H. diversicolor*.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/supplementary material.

AUTHOR CONTRIBUTIONS

MP developed the manuscript concept, wrote the manuscript, and took care of the statistical analyses. ED contributed to the development of the study design, supported sampling, took care of data analysis, and commented on the manuscript. DT participated in the development of study design, supported sampling and handling, and commented on the manuscript. IG supported the analysis and commented on the manuscript. AV developed the study design, participated in the manuscript concept development, supported writing and analysis. All authors contributed to the article and approved the submitted version.

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REFERENCES

Ahrens, M., Hertz, J., Lamoureux, E., Lopez, G., McElroy, A., and Brownawell, B. (2001). The role of digestive surfactants in determining bioavailability of sediment-bound hydrophobic organic contaminants to 2 deposit-feeding polychaetes. *Mar. Ecol. Prog. Ser.* 212, 145–157. doi: 10.3354/meps212145

Amiard-Triquet, C., Mouneyrac, C., and Berthet, B. (2013). "Polychaetes in Ecotoxicology," in *Encyclopedia of Aquatic Ecotoxicology*, eds J.-F. Férard and C. Blaise (Dordrecht: Springer Science+Business Media), 893–908. doi: 10.1007/978-94-007-5704-2_82

Armitage, J. M., McLachlan, M. S., Wiberg, K., and Jonsson, P. (2009). A model assessment of polychlorinated dibenzo-p-dioxin and dibenzofuran sources and fate in the Baltic Sea. *Sci. Total Environ.* 407, 3784–3792. doi: 10.1016/J.SCITOTENV.2009.03.001

ASTM (2014). *Standard Guide for Collection, Storage, Characterization, and Manipulation of Sediments for Toxicological Testing and for Selection of Samplers Used to Collect Benthic Invertebrates (E1391 - 03)*. West Conshohocken, PA: American Society for Testing Materials.

Bellucci, L. G., Frignani, M., Raccanelli, S., and Carraro, C. (2000). Polychlorinated dibenzo-p-dioxins and dibenzofurans in surficial sediments of the venice lagoon (Italy). *Mar. Pollut. Bull.* 40, 65–76. doi: 10.1016/S0025-326X(99)00171-X

Bellucci, L. G., Giuliani, S., Mugnai, C., Frignani, M., Paolucci, D., Albertazzi, S., et al. (2009). Anthropogenic metal delivery in sediments of porto marghera and venice lagoon (Italy). *Soil Sediment Contam. An Int. J.* 19, 42–57. doi: 10.1080/15320380903390562

Bernstein, A. G., and Montobbio, L. (2011). La gestione dei sedimenti in laguna di Venezia. Sintesi dei principali risultati acquisiti da studi e indagini di base. *Quad. Trimest. del Consorzio Venezia Nuova* 1.11, 41–59.

Berthet, B., Mouneyrac, C., Amiard, J. C., Amiard-Triquet, C., Berthelot, Y., Le Hen, A., et al. (2003). Accumulation and soluble binding of cadmium, copper, and zinc in the polychaete hediste diversicolor from coastal sites with different trace metal bioavailabilities. *Arch. Environ. Contam. Toxicol.* 45, 468–478. doi: 10.1007/s00244-003-0135-0

Bettiol, C., Collavini, F., Guerzoni, S., Molinaroli, E., Rossini, P., Zaggia, L., et al. (2005). Atmospheric and riverine inputs of metals, nutrients and persistent organic pollutants into the lagoon of Venice. *Hydrobiologia* 550, 151–165. doi: 10.1007/s10750-005-4372-2

Brannon, J. M., Price, C. B., Reilly, F. J., Pennington, J. C., and Mcfarland, V. A. (1993). Effects of sediment organic carbon on distribution of radiolabeled fluoranthene and PCBs among sediment, interstitial water, and biota. *Bull. Environ. Contam. Toxicol.* 51, 873–880. doi: 10.1007/bf00198284

Burkhard, L. (2009). *Estimation of Biota Sediment Accumulation Factor (BSAF) From Paired Observations of Chemical Concentrations in Biota and Sediment*. Cincinnati, OH: U.S. Environmental Protection Agency.

Chen, J., Quan, X., Yazhi, Z., Yan, Y., and Yang, F. (2001). Quantitative structure–property relationship studies on n-octanol/water partitioning coefficients of PCDD/Fs. *Chemosphere* 44, 1369–1374. doi: 10.1016/S0045-6535(00)00347-7

Cornelissen, G., Breedveld, G. D., Naes, K., Oen, A. M. P., and Ruus, A. (2006). Bioaccumulation of native polycyclic aromatic hydrocarbons from sediment by a polychaete and a gastropod: freely dissolved concentrations and activated carbon amendment. *Environ. Toxicol. Chem.* 25, 2349–2355.

Critto, A., Carlon, C., and Marcomini, A. (2003). Characterization of contaminated soil and groundwater surrounding an illegal landfill (*S. Giuliano*, Venice, Italy) by principal component analysis and kriging. *Environ. Pollut.* 122, 235–244. doi: 10.1016/S0269-7491(02)00296-8

Dean, H. K. (2008). The use of polychaetes (Annelida) as indicator species of marine pollution: a review. *Rev. Biol. Trop.* 56, 11–38.

Durou, C., Poirier, L., Amiard, J. C., Budzinski, H., Gnassia-Barelli, M., Lemenach, K., et al. (2007a). Biomonitoring in a clean and a multi-contaminated estuary based on biomarkers and chemical analyses in the endobenthic worm *Nereis diversicolor*. *Environ. Pollut.* 148, 445–458. doi: 10.1016/j.envpol.2006.12.022

Durou, C., Smith, B. D., Roméo, M., Rainbow, P. S., Mouneyrac, C., Mouloud, M., et al. (2007b). From biomarkers to population responses in *Nereis diversicolor*: assessment of stress in estuarine ecosystems. *Ecotoxicol. Environ. Saf.* 66, 402–411. doi: 10.1016/j.ecoenv.2006.02.016

Fattore, E., Benfenati, E., Mariani, G., Fanelli, R., and Evers, E. H. G. (1997). Patterns and sources of polychlorinated dibenzo-p-dioxins and dibenzofurans in sediments from the venice lagoon, Italy. *Environ. Sci. Technol.* 31, 1777–1784. doi: 10.1021/ES9608860

Flesch-Janys, D. (1996). Elimination of polychlorinated dibenzo-p-dioxins and dibenzofurans in occupationally exposed persons. *J. Toxicol. Environ. Health* 47, 363–378. doi: 10.1080/009841096161708

Frangipane, G., Volpi Ghirardini, A., Collavini, F., Zaggia, L., Pesce, A., and Tagliapietra, D. (2005). Heavy metals in *Hediste diversicolor* (polychaeta: nereididae) and salt marsh sediments from the lagoon of Venice (Italy). *Chem. Ecol.* 21, 441–454. doi: 10.1080/02757540500438649

Frignani, M., Bellucci, L. G., Carraro, C., and Favotto, M. (2001a). Accumulation of polychlorinated dibenzo-p-dioxins and dibenzofurans in sediments of the venice lagoon and the industrial area of porto marghera. *Mar. Pollut. Bull.* 42, 544–553. doi: 10.1016/S0025-326X(00)00197-1

Frignani, M., Bellucci, L. G., Carraro, C., and Raccanelli, S. (2001b). Polychlorinated biphenyls in sediments of the Venice Lagoon. *Chemosphere* 43, 567–575. doi: 10.1016/S0045-6535(00)00408-2

Gambaro, A., Radaelli, M., Piazza, R., Stortini, A. M., Contini, D., Belosi, F., et al. (2009). Organic micropollutants in wet and dry depositions in the Venice Lagoon. *Chemosphere* 76, 1017–1022. doi: 10.1016/J.CHEMOSPHERE.2009.04.063

Guerzoni, S., Rossini, P., Molinaroli, E., Rampazzo, G., and Raccanelli, S. (2004). Measurement of atmospheric deposition of polychlorinated dibenzo-p-dioxins and dibenzofurans in the Lagoon of Venice, Italy. *Chemosphere* 54, 1309–1317. doi: 10.1016/S0045-6535(03)00249-2

Guerzoni, S., Rossini, P., Sarretta, A., Raccanelli, S., Ferrari, G., and Molinaroli, E. (2007). POPs in the Lagoon of Venice: budgets and pathways. *Chemosphere* 67, 1776–1785. doi: 10.1016/j.chemosphere.2006.05.085

Gunnarsson, J. S., Hollertz, K., and Rosenberg, R. (1999). Effects of organic enrichment and burrowing activity of the polychaete *Neries diversicolor* on the fate of tetrachlorobiphenyl in marine sediments. *Environ. Toxicol. Chem.* 18, 1149–1156. doi: 10.1002/etc.5620180612

Iannuzzi, J., Butcher, M., and Iannuzzi, T. (2011). Evaluation of potential relationships between chemical contaminants in sediments and aquatic organisms from the lower Passaic River, New Jersey, USA. *Environ. Toxicol. Chem.* 30, 1721–1728. doi: 10.1002/etc.550

Jimenez, B., Hernandez, L. M., Gonzalez, M. J., Eljarrat, E., Rivera, J., and Fossi, M. C. (1998). Congener specific analysis of polychlorinated dibenzo-p-dioxins and dibenzofurans in crabs and sediments from the venice and orbetello lagoons, Italy. *Environ. Sci. Technol.* 32, 3853–3861. doi: 10.1021/ES9801994

Jørgensen, A., Giessing, A. M. B., Rasmussen, L. J., and Andersen, O. (2008). Biotransformation of polycyclic aromatic hydrocarbons in marine polychaetes. *Mar. Environ. Res.* 65, 171–186. doi: 10.1016/J.MARENVRES.2007.10.001

Kang, Y.-S., Yamamuro, M., Masunaga, S., and Nakanishi, J. (2002). Specific biomagnification of polychlorinated dibenzo-p-dioxins and dibenzofurans in tufted ducks (*Aythya fuligula*), common cormorants (*Phalacrocorax carbo*) and their prey from Lake Shinji, Japan. *Chemosphere* 46, 1373–1382. doi: 10.1016/S0045-6535(01)00259-4

Khairy, M., Barrett, K., and Lohmann, R. (2016). Changing sources of polychlorinated dibenzo- p -dioxins and furans in sediments and ecological risk for nekton in the lower Passaic River and Newark Bay, New Jersey, USA. *Environ. Toxicol. Chem.* 35, 550–562. doi: 10.1002/etc.3223

Kjerfve, B. (1994). *Coastal Lagoon Processes*. Amsterdam: Elsevier.

Kono, K., Tanaka, H., and Koyama, J. (2010). Dioxin transfer from sediment to the infaunal surface deposit-feeding polychaete *Perinereis nuntia* in a laboratory-rearing experiment. *Environ. Toxicol. Chem.* 29, 1512–1519. doi: 10.1002/etc.177

Lake, J. L., Rubinstein, N. I., Lee, H., Lake, C. A., Heltshe, J., and Pavignano, S. (1990). Equilibrium partitioning and bioaccumulation of sediment-associated contaminants by infaunal organisms. *Environ. Toxicol. Chem.* 9, 1095–1106. doi: 10.1002/etc.5620090816

Langston, W. J., O'Hara, S., Pope, N. D., Davey, M., Shortridge, E., Imamura, M., et al. (2012). Bioaccumulation surveillance in milford haven waterway. *Environ. Monit. Assess.* 184, 289–311. doi: 10.1007/s10661-011-1968-z

Lobel, P. B., Belkhode, S. P., Jackson, S. E., and Longerich, H. P. (1991). Sediment in the intestinal tract: a potentially serious source of error in aquatic biological monitoring programs. *Mar. Environ. Res.* 31, 163–174. doi: 10.1016/0141-1136(91)90009-W

Loonen, H., Muir, D. C. G., Parsons, J. R., and Govers, H. A. J. (1997). Bioaccumulation of polychlorinated dibenzo- p -dioxins in sediment by oligochaetes: influence of exposure pathway and contact time. *Environ. Toxicol. Chem.* 16, 1518–1525. doi: 10.1002/etc.5620160728

Lyytikäinen, M., Hirva, P., Minkkinen, P., Hämäläinen, H., Rantalainen, A.-L., Mikkelson, P., et al. (2003). Bioavailability of sediment-associated PCDD/Fs and PCDEs: relative importance of contaminant and sediment characteristics and biological factors. *Environ. Sci. Technol.* 37, 3926–3934. doi: 10.1021/ES034151O

MAG. ACQUE (1999). *Mappatura dell'inquinamento Dei Fondali Lagunari. Studi ed indagini. Rapporto finale*. Venice: MAG-ACQUE.

MAG. ACQUE-Thetis (2006). *Studio ICSEL. Rapporto finale sull'attività svolta nel triennio 2003 – 2006*. Venice: MAG-ACQUE.

MAG. ACQUE-Thetis (2007). *Indagini e Monitoraggi Nelle Aree Lagunari Tra Venezia e Porto Marghera. Studio MAPVE1 – OP/399. Iˆ fase. Attività A - Risultati Della Caratterizzazione Dello Stato Delle Matrici Ambientali*. Venice: MAG-ACQUE.

MAG. ACQUE-Uta (2011). *Gli scarichi idrici e la qualità delle acque dell'area industriale di Porto Marghera*. Venice: MAG-ACQUE.

Marcomini, A., Zanette, M., D'Andrea, F., and Della Sala, S. (1997). *Diossine, Salute E Ambiente*. Venezia: Arsenale Editrice.

Mayer, L. M., Weston, D. P., and Bock, M. J. (2001). Benzo[a]pyrene and zinc solubilization by digestive fluids of benthic invertebrates-a cross-phyletic study. *Environ. Toxicol. Chem.* 20, 1890–1900. doi: 10.1002/etc.5620200906

McFarland, V. A., Lutz, C. H., and Reilly, F. J. (1994). *Bioaccumulation Data and Analysis for Selected Contaminants in Sediments and Biota of the New York Bight Apex Mud dump Reference Site*. New York, NY: U.S Army Engineer District.

Molinaroli, E., Guerzoni, S., De Falco, G., Sarretta, A., Cucco, A., Como, S., et al. (2009). Relationships between hydrodynamic parameters and grain size in two contrasting transitional environments: the Lagoons of Venice and Cabras, Italy. *Sediment. Geol.* 219, 196–207. doi: 10.1016/J.SEDGEO.2009.05.013

Nesto, N., Cassin, D., and Da Ros, L. (2010). Is the polychaete, *Perinereis rullieri* (Pilato 1974), a reliable indicator of PCB and PAH contaminants in coastal sediments? *Ecotoxicol. Environ. Saf.* 73, 143–151. doi: 10.1016/J.ECOENV.2009.09.004

Nunes, M., Marchand, P., Vernisseau, A., Bizec, B., Le Ramos, F., and Pardal, M. A. (2011). PCDD/Fs and dioxin-like PCBs in sediment and biota from the Mondego estuary (Portugal). *Chemosphere* 83, 1345–1352. doi: 10.1016/J.CHEMOSPHERE.2011.02.081

Opperhuizen, A., and Sijm, D. T. H. M. (1990). Bioaccumulation and biotransformation of polychlorinated dibenzo- p -dioxins and dibenzofurans in fish. *Environ. Toxicol. Chem.* 9, 175–186. doi: 10.1002/etc.5620090207

Pastershank, G. M., Muir, D. C. G., and Fairchild, W. L. (1999). Accumulation and depuration of 2,3,7,8-tetrachlorodibenzofuran and octachlorodibenzo- *p* -dioxin by caddisfly larvae (*Hydropsyche bidens* (ross)) in miniature laboratory streams. *Environ. Toxicol. Chem.* 18, 2352–2360. doi: 10.1002/etc.5620181032

Pickard, S. W., and Clarke, J. U. (2008). Benthic bioaccumulation and bioavailability of polychlorinated dibenzo-p-dioxins/dibenzofurans from surficial lake ontario sediments. *J. Great Lakes Res.* 34, 418–433. doi: 10.3394/0380-1330(2008)34[418:bbabop]2.0.co;2

Picone, M. (2006). *Impiego di una batteria di saggi ecotossicologici nel monitoraggio della qualità dei sedimenti della Laguna di Venezia*. Venezia: Università Ca' Foscari Venezia.

Picone, M., Bergamin, M., Arizzi Novelli, A., Noventa, S., Delaney, E., Barbanti, A., et al. (2008). Evaluation of *Corophium orientale* as bioindicator for Venice Lagoon: sensitivity assessment and toxicity-score proposal. *Ecotoxicol. Environ. Saf.* 70, 174–184. doi: 10.1016/j.ecoenv.2006.06.005

Picone, M., Bergamin, M., Delaney, E., Volpi Ghirardini, A., and Kusk, K. O. (2018). Testing lagoonal sediments with early life stages of the copepod *Acartia tonsa* (Dana): an approach to assess sediment toxicity in the Venice Lagoon. *Ecotoxicol. Environ. Saf.* 147, 217–227. doi: 10.1016/j.ecoenv.2017.08.042

Picone, M., Bergamin, M., Losso, C., Delaney, E., Arizzi Novelli, A., and Volpi Ghirardini, A. (2016). Assessment of sediment toxicity in the Lagoon of Venice (Italy) using a multi-species set of bioassays. *Ecotoxicol. Environ. Saf.* 123, 32–44. doi: 10.1016/j.ecoenv.2015.09.002

Pruell, R. J., Rubinstein, N. I., Taplin, B. K., LiVolsi, J. A., and Bowen, R. D. (1993). Accumulation of polychlorinated organic contaminants from sediment by three benthic marine species. *Arch. Environ. Contam. Toxicol.* 24, 290–297. doi: 10.1007/BF01128727

Pruell, R. J., Taplin, B. K., McGovern, D. G., McKinney, R., and Norton, S. B. (2000). Organic contaminant distributions in sediments, polychaetes (*Nereis virens*) and American lobster (*Homarus americanus*) from a laboratory food chain experiment. *Mar. Environ. Res.* 49, 19–36. doi: 10.1016/S0141-1136(99)00046-X

Raccanelli, S., Frangipane, G., and Libralato, S. (2007). Serum levels of PCDDs, PCDFs and dioxin-like PCBs in relation to different exposures in Italian adult men. *Organohalogen Compd.* 69, 1934–1937.

Raccanelli, S., Libralato, S., Favotto, M. (2008). On the detoxification of benthic bivalves contaminated by POPs: insights from experimental and modelling approaches. *Environ. Chem. Lett.* 6, 251–258. doi: 10.1007/s10311-007-0127-9

Raccanelli, S., Pastres, R., Favotto, M., and Vio, P. (2004). Correlation between POPs in sediment and edible bivalve in the lagoon of Venice and estimation of the daily intake. *Organohalogen Compd.* 66, 1823–1828.

Ruus, A., Daae, I. A., and Hylland, K. (2012). Accumulation of polychlorinated biphenyls from contaminated sediment by Atlantic cod (*Gadus morhua*): direct accumulation from resuspended sediment and dietary accumulation via the polychaete Nereis virens. *Environ. Toxicol. Chem.* 31, 2472–2481. doi: 10.1002/etc.1973

Ruus, A., Schaanning, M., Øxnevad, S., and Hylland, K. (2005). Experimental results on bioaccumulation of metals and organic contaminants from marine sediments. *Aquat. Toxicol.* 72, 273–292. doi: 10.1016/j.aquatox.2005.01.004

Scaps, P. (2002). A review of the biology, ecology and potential use of the common ragworm *Hediste diversicolor* (O.F. Müller) (Annelida: Polychaeta). *Hydrobiologia* 470, 203–218. doi: 10.1023/A:1015681605656

Schell, J. D., Campbell, D. M., and Lowe, E. (1993). Bioaccumulation of 2,3,7,8-tetrachlorodibenzo- *P* -dioxin in feral fish collected from a bleach-kraft paper mill receiving stream. *Environ. Toxicol. Chem.* 12, 2077–2082. doi: 10.1002/etc.5620121115

Schrock, M. E., Barrows, E. S., and Rosman, L. B. (1997). Biota-to-sediment accumulation factors for TCDD and TCDF in worms from 28-day bioaccumulation tests. *Chemosphere* 34, 1333–1339. doi: 10.1016/S0045-6535(97)00430-X

Secco, T., Pellizzato, F., Sfriso, A., and Pavoni, B. (2005). The changing state of contamination in the Lagoon of Venice. Part 1: organic pollutants. *Chemosphere* 58, 279–290. doi: 10.1016/J.CHEMOSPHERE.2004.06.030

Sfriso, A., Facca, C., and Raccanelli, S. (2014). PCDD/F and dioxin-like PCB bioaccumulation by Manila clam from polluted areas of Venice lagoon (Italy). *Environ. Pollut.* 184, 290–297. doi: 10.1016/J.ENVPOL.2013.08.026

Shepard, F. P. (1954). Nomenclature based on sand–silt–clay ratios. *J. Sediment. Res.* 24, 151–158.

Sijm, D. T. H. M., Opperhuizen, A., and Wever, H. (1993). Congener-specific biotransformation and bioaccumulation of PCDDs and PCDFs from fly ash in fish. *Environ. Toxicol. Chem.* 12, 1895–1907. doi: 10.1002/etc.5620121016

Sizmur, T., Canário, J., Gerwing, T. G., Mallory, M. L., and O'Driscoll, N. J. (2013). Mercury and methylmercury bioaccumulation by polychaete worms is governed by both feeding ecology and mercury bioavailability in coastal mudflats. *Environ. Pollut.* 176, 18–25. doi: 10.1016/J.ENVPOL.2013.01.008

Stringer, R. L., Costner, P., and Johnston, P. A. (1995). PVC manifacture as a source of PCDD/Fs. *Organohalogen Compd.* 24, 119–123.

Swerev, M., and Ballschmiter, K. (1989). Pattern analysis of PCDDs and PCDFs in environmental samples as an approach to an occurrence source correlation. *Chemosphere* 18, 609–616. doi: 10.1016/0045-6535(89)90173-2

Tagliapietra, D., Sigovini, M., and Volpi Ghirardini, A. (2009). A review of terms and definitions to categorise estuaries, lagoons and associated environments. *Mar. Freshw. Res.* 60, 497–509. doi: 10.1071/MF08088

Tagliapietra, D., and Volpi Ghirardini, A. (2006). Notes on coastal lagoon typology in the light of the EU water framework directive: Italy as a case study. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 16, 457–467. doi: 10.1002/aqc.768

Van den Berg, M., Birnbaum, L. S., Denison, M., De Vito, M., Farland, W., Feeley, M., et al. (2006). The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. *Toxicol. Sci.* 93, 223–241. doi: 10.1093/toxsci/kfl055

van der Oost, R., Beyer, J., and Vermeulen, N. P. (2003). Fish bioaccumulation and biomarkers in environmental risk assessment: a review. *Environ. Toxicol. Pharmacol.* 13, 57–149. doi: 10.1016/S1382-6689(02)00126-6

Volpi Ghirardini, A., Arizzi Novelli, A., and Tagliapietra, D. (2005). Sediment toxicity assessment in the Lagoon of Venice (Italy) using *Paracentrotus lividus* (Echinodermata: Echinoidea) fertilization and embryo bioassays. *Environ. Int.* 31, 1065–1077. doi: 10.1016/j.envint.2005.05.017

Volpi Ghirardini, A., Cavallini, L., Delaney, E., Tagliapietra, D., Ghetti, P. F., Bettiol, C., et al. (1999). *H. diversicolor*, *N. succinea* and *P. cultrifera* (Polychaeta: Nereididae) as bioaccumulators of cadmium and zinc from sediments: preliminary results in the Venetian lagoon (Italy). *Toxicol. Environ. Chem.* 71, 457–474. doi: 10.1080/02772249909358815

Volpi Ghirardini, A., Delaney, E., Tagliapietra, D., Arizzi Novelli, A., Frangipane, G., Marchetto, D., et al. (2004). "*Hediste diversicolor* (Polychaeta: Nereididae) as effective biomonitor of POPs (PCBs and organochloride pesticides) for lagoonal sediments," in *Proceedings of the ECSA-ERF 38th Symposium*, Rouen.

Voparil, I. M., and Mayer, L. M. (2000). Dissolution of sedimentary polycyclic aromatic hydrocarbons into the lugworm's (*Arenicola marina*) digestive fluids. *Environ. Sci. Technol.* 34, 1221–1228. doi: 10.1021/es990885i

Zheng, S., Chen, B., Qiu, X., Lin, K., and Yu, X. (2013). Three novel cytochrome P450 genes identified in the marine polychaete *Perinereis nuntia* and their transcriptional response to xenobiotics. *Aquat. Toxicol.* 134–135, 11–22. doi: 10.1016/J.AQUATOX.2013.02.009

Zonta, R., Botter, M., Cassin, D., Pini, R., Scattolin, M., and Zaggia, L. (2007). Sediment chemical contamination of a shallow water area close to the industrial zone of Porto Marghera (Venice Lagoon, Italy). *Mar. Pollut. Bull.* 55, 529–542. doi: 10.1016/j.marpolbul.2007.09.024

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Status and Distribution of Waterbirds in a Natura 2000 Area: The Case of Gialova Lagoon, Messinia, Greece

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Located at the south-western most part of the Balkan peninsula, along an important migration route (the Mediterranean/Black Sea Flyway), the Gialova Lagoon wetland is one of the few remaining Important Bird Areas (IBAs) along the south-west coast of Greece, also designated as a Special Protection Area. The wetland serves as the first suitable stopover for many spring migrants who have flown non-stop over the Mediterranean Sea, and the last before their journey back to Africa in the autumn. In this study, we conducted monthly field visits during the period October 2016 to January 2019 with the aim to complement existing information about the site, to evaluate the current status and distribution of waterbirds, to provide insights for the management of the area and to re-assess the IBA/Ramsar criteria. A total of 149 bird species representing 43 families and 15 orders were recorded, including 36 threatened species at an International, European or/and national level, and 40 species listed in the Annex I of the EUs Birds Directive (21 species were listed as both threatened and under Annex I). 81 species were identified as wetland related species, of which 66 species were identified as waterbirds (7 orders, 11 families). Waterbirds richness and abundance were higher during the Wet season and corresponding periods (Wintering and Spring migration). All parts of the wetland supported waterbirds and threatened species, with the S. Wetland sub-area being the most diverse during the Breeding/ Nesting, and both migration periods. The abundance of most waterbirds and IBA species have declined over the last 20 years, but this does not necessarily mean that the area no longer fulfills Ramsar criterion 6 (and equivalent IBA criterion A4i). However, this outcome should not be overlooked by the site managers and conservation actions, such as the restoration of fresh water inflows which could improve habitats and water conditions for IUCN and IBA species, should be implemented with high priority. In addition, our results indicate that the area meets Ramsar criterion 4 and criterion 2, and thus we suggest that it should be further investigated and evaluated to potentially become the eleventh Greek Ramsar site.

Keywords: waterbirds, coastal wetland, Natura 2000, management suggestions, conservation needs

INTRODUCTION

Waterbirds, are defined as "species of bird that are ecologically dependent on wetlands" (Ramsar convention, 1994, art. 1.2) including all waterfowl, seabirds and waders. Wetlands are used both as wintering areas—to stay in for longer periods—but also as important "stopover" areas where migrating birds make briefer stops when they are migrating to or from their breeding grounds (Warnock, 2010). Waterbirds migrate along broadly similar, well-established routes known, as flyways. The Mediterranean/Black Sea Flyway is one of the eight major pathways around the globe and one of the three connecting Europe with Africa (BirdLife International, 2017). When wetlands along a bird's predetermined migration route disappear, the likelihood of the birds completing their migration is strongly impoverished (Moore et al., 2005).

To that end, more than 2,400 sites with international importance are protected by the Ramsar Convention (Ramsar convention, 1994; Ramsar Sites Information Service [RSIS], 2020). Nonetheless, the Ramsar site characterization criteria for waterbirds (criterion 5: *A wetland should be considered internationally important if it regularly supports 20,000 or more waterbirds,* and criterion 6: *A wetland should be considered internationally important if it regularly supports 1% of the individuals in a population of one species or subspecies of waterbirds*), cannot be easily met by all wetlands on the migration route of birds (Ramsar Sites Criteria, 2020). For example, Greece has about 400 wetlands (Greek Biotope Wetland Centre [GBWC], 2020), but only ten are protected under the Ramsar Convention (Greek Ramsar Sites, 2020).

To further enlarge the protection of bird species, BirdLife International, in collaboration with national NGOs working with birds, have established a network of Important Bird Areas (IBAs), including sites critical for the conservation of birds worldwide (BirdLife International, 2001). Depending on different numerical thresholds, the international importance of a site for a species may be categorized at three distinct geographical levels, from global (criteria A), to European (criteria B) and to European Union (criteria C) (IBA Criteria, 2001). Up to present, more than 13,000 sites at a global scale have been included in the IBA network (BirdLife International, 2020). The identification of IBAs in Europe is based on a site's international importance for: (a) threatened species, (b) congregatory bird species, (c) assemblages of restricted-range bird species, and (d) assemblages of biome-restricted bird species (IBA Criteria, 2001). The criteria build upon existing legal instruments, like the EUs Birds Directive which was adopted in 1979, and it took its current form in 2009 (Birds Directive 2009/147/EC, 2009). The European Union Birds Directive obliges all member countries to protect habitats supporting birds listed in the Annex I of the Directive, also regularly occurring migratory species not listed in Annex I, and to designate Special Protection Areas (SPA). The SPAs, together with the Sites of Community Importance (SCI) and the Special Areas of Conservation (SAC), which are based on the Habitats Directive (Habitats Directive 92/43/EEC, 1992), form the Natura 2000 network of protected areas in Europe (European Commission, 2016).

For birds crossing into Africa, the Mediterranean Sea constitutes a significant obstacle and migration over the sea is concentrated at a number of narrow straits and "land bridges" like those formed by the Italian and the Greek peninsula. The high mountainous morphology encountered in Greece, separates the peninsula into a western and eastern route. With the aim of creating a network of sites to ensure that migratory species find suitable breeding, stop-over and wintering places along their respective flyways, Greece has identified 196 IBAs (Hellenic Ornithological Society [HOS], 2019a). The eastern route, hosts most of the Greek Ramsar sites and marine IBAs (Important Bird Areas for Seabirds) and many IBAs are distributed across the islands of the Aegean Sea and the east coastline of mainland Greece on different latitudes (Fric et al., 2012; Ramsar, 2017; Hellenic Ornithological Society [HOS], 2019a). Most IBAs along the west coast of Greece are mainly located at higher latitudes. In the northern part, the most important wetlands exist in the form of large lagoon and delta complexes and they attract high numbers of wintering and staging birds during migration (Maragou and Mantziou, 2000; Liordos et al., 2014).

Along the western route, at lower latitudes, wetlands are scarcer, and the Gialova Lagoon wetland (GLw) is one of the few remaining IBA wetlands (Gialova, GR119) along the south-west coast of Greece (Heath et al., 2000). While few waterbirds use the wetland as a nesting ground, the area is important as a wintering and a stopover bird area (Kardakari, 2000; Bousbouras et al., 2011), and for some waterbirds the area meets IBA criteria A4i (*The site is known or thought to hold, on a regular basis, ≥ 1% of a biogeographic population of a congregatory waterbird species*) and B1i (*The site is known or thought to hold ≥ 1% of a flyway or other distinct population of a waterbird species*) (**Table 1**). To that end, it has been already classified as fulfilling the Ramsar criterion 4 (*A wetland should be considered internationally important if it supports plant and/or animal species at a critical stage in their life cycles, or provides refuge during adverse conditions*) and criterion 6 (BirdLife International, 2001), but is it not designated as a Ramsar site (Greek Ramsar Sites, 2020). The GLw is part of a wider Natura 2000 area (SPA site: GR2550008, 2001; Birds Directive 2009/147/EC, 2009, and SCI/SAC site: Habitats Directive 92/43/EEC, 1992; GR2550004, 1995). In addition to birds, the lagoon supports several fish species with commercial value (Koutsoubas et al., 2000; Zoulias et al., 2017), and it has a rich benthic diversity (Arvanitidis et al., 1999; Koutsoubas et al., 2000; McArthur et al., 2000). Moreover, the surrounding coastal area is the basic nesting habitat for the only European population of the critically endangered African Chameleon (*Chamaeleo africanus*) (Legakis and Maragou, 2009).

Despite its rich biodiversity, the area has been under unstable management for several years, and due to the lack of a clear management scheme[1] the functions of the wetland continue to

[1] The environmental management has been recently (January, 2019) assigned to the Management Body of Protected Areas of South Peloponnese and Kythira island (Maneas et al., 2019), but this scheme is about to change following changes in the national policy. The management and exploitation of the lagoon fish stocks is assigned by the local sub-region to local fishers for fixed terms (usually for 5 years), and the local archeological ephorate is also engaged in the management of the area, since part of the area is an archeological site.

degrade (Maneas et al., 2019). Over the years, the combined effects of increased salinity and limitation in water circulation, due to anthropogenic interventions, have led to extensive mortality of emergent aquatic macrophytes (reed and cattail), which are typical bird habitats (ibid). Even today, several parts of the wetland are gradually transformed to agricultural land, or parking lots reducing the size of the natural habitats (Maneas et al., 2019). Another major threat for aquatic species and habitats, is the problem of salinity which has already been linked to waterbirds conservation in the area (Kardakari, 2000). Recent studies have shown that at present the lagoon is characterized as saline with hypersaline conditions for nearly 30% of the year, a percentage which is expected to increase under future warmer and drier climatic conditions (Manzoni et al., 2020).

An improved water management, which will ensure increased fresh water inputs to the wetland, is already demanded by the local fishers for improving fish stocks in the lagoon (personal communication with local fishers), but such management plan should also aim to create favorable conditions for birds. However, the available knowledge on the current GLw birds' status and distribution (Standard Data Forms for site GR2550008, 2001) is based on old data reported in Kardakari (2000). Given the high salinity values, the degradation of habitats and the anthropogenic pressures mentioned above, there is a high probability that there have been changes in species numbers and the overall importance and use of the site by birds, which will need to be considered in future management strategies for the area.

Greece is currently validating the existing Environmental Assessments for all the Natura 2000 sites, including the GLw site, but in the absence of updated information, it is very likely that any management plan for the GLw will be based on previous data, which are quite detailed and important, but outdated. To that end, under this study we have contacted monthly field visits during the period October 2016 to January 2019, with the aim to complement existing information about the site and to evaluate the current status and distribution of waterbirds in the GLw, and to also provide a basis for comparison with previous studies. Since the wetland is divided in several sub-areas (Maneas et al., 2019), a sub aim of the study was to understand which parts of the wetland are mostly used by birds, identify conservation needs and potential management implications. An additional and explicit aim of our study was to re-assess the IBA/Ramsar criteria and the status of the site, as the last evaluation was in 2001 (BirdLife International, 2001). Our open data, results and suggestions could be used as insights for the sustainable management of the area and relevant decision making at local/national level, but also as updates for the waterbirds' status in a Greek IBA at a European/International level.

MATERIALS AND METHODS

Study Area

The GLw is a coastal wetland located in south-west Messinia, Greece (latitude: 36°58′, longitude: 21°39′), along the Mediterranean/Black Sea Flyway migration route (BirdLife International, 2017), at the most southern-western side of the Balkan peninsula (**Figure 1**). It is the first wetland on the west side of the Balkan peninsula that migratory birds come across when they return from Africa in spring or the last before crossing the Mediterranean during the autumn migration (Hellenic Ornithological Society [HOS], 2019b). The boundaries of our study area, are defined at the north and east by two canals constructed during the 1960s (**Figure 1**). At the west, a rocky

TABLE 1 | Waterbirds in GLw-Natura2000 that meet the IBA criteria A4i, B1i, C2, C3 (Heath et al., 2000; BirdLife International, 2001), and species of global conservation concern that do not meet IBA criteria (Hellenic Ornithological Society [HOS], 2019b).

Waterbirds that meet IBA criteria						Habitats
Species	**Season**	**Year**	**Pop. min**	**Pop. max**	**Criterion**	**Priority**
Phalacrocorax carbo	W	1996	1000	1000	B1i, C3	**Coastal lagoons (1140)**
Egretta garzetta	W	1995			B1i	**Coastal dunes with *Juniperus* spp. (2250)**
	P	1996	1500	2000	A4i, B1i, C2	
Ardea alba	W	1996	280	360	A4i, B1i, C2	**Pseudo-steppe with grasses and annuals of the Thero-Brachypodietea (6220)**
Plegadis falcinellus	P	1996	1600	1800	A4i, B1i, C2	
Glareola pratincola	P	1996	117	117	B1i, C2	
Tringa stagnatilis	P	1996	300	500	A4i, B1i, C3	
Tringa glareola	P	1996	7000	11000	A4i, B1i, C2	
Gelochelidon nilotica	P	1996	210	220	B1i, C2	
Bird species of global conservation concern that do not meet IBA criteria						**Other waterbird habitats**
Pelecanus crispus	P					Mudflats and sandflats not covered by seawater at low tide
Aythya nyroca	P, 10–20 individuals					Mediterranean and thermo-Atlantic halophilous scrubs (Sarcocornetea fruticosi)
Falco naumani	P, 10–20 individuals					Mediterranean salt meadows (Juncetalia maritimi)
Gallinago media	P, 10–20 individuals					

The listed IBA species are also the SPA species of concern (GR2550008, 2001). Priority habitats (highlighted with bold), and additional habitats linked to waterbirds that are targeted by the Habitats Directive are listed in the last column (GR2550004, 1995). W for Wintering and P (passage) for Migration period.

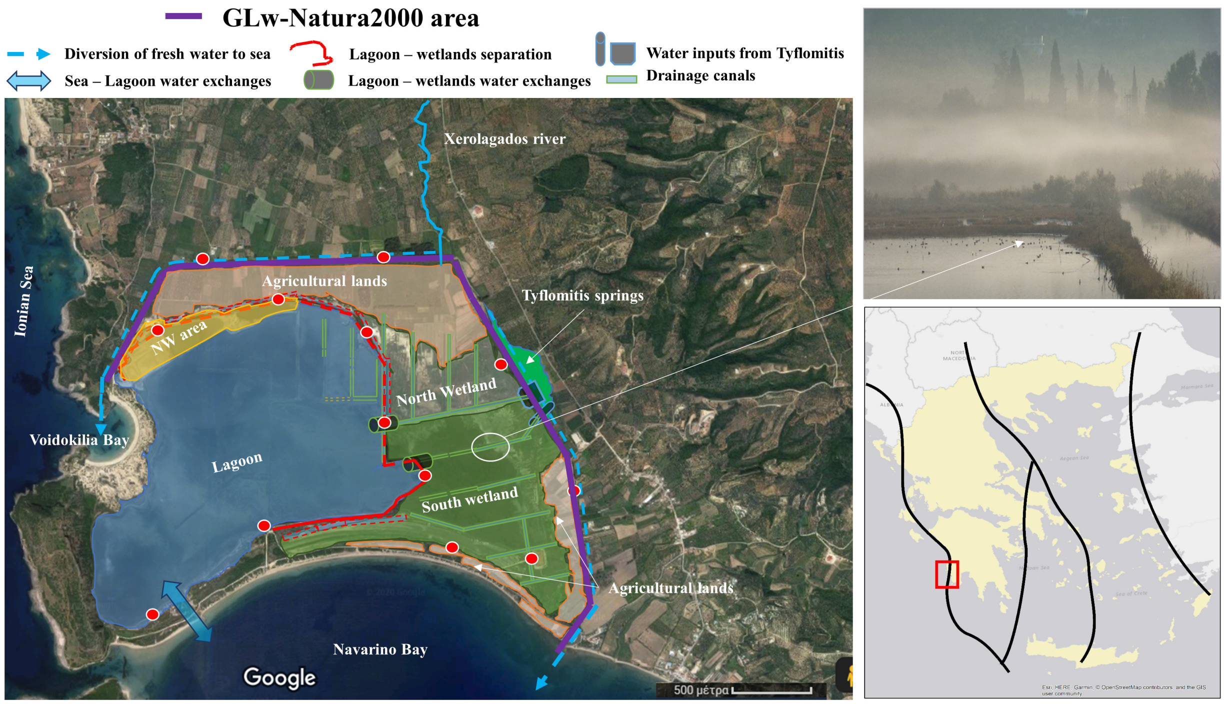

FIGURE 1 | Left side: The map of the study area located in SW Greece (36° 58′ N, 21° 40′ E), showing the different observation points (red dots), and the major sub-areas. It also contains information about the water connections. **Right side:** Birds' migration "national" routes in Greece (Hellenic forestry, 2020) along the Mediterranean/Black Sea Flyway (BirdLife International, 2017), and a picture of the site which captures one of the drainage canals in the form that it looks today.

hill (also known as the Palaiokastro) and the semi-enclosed Voidokilia Bay separate the area from the Ionian Sea. A 3.8 km long and about 150m wide natural sand formation, separates the area from the semi-enclosed Navarino Bay to the south. In this study, we will refer to this area as the GLw-Natura2000 area (purple line in **Figure 1**).

The selected case study is part of a wider area which is characterized as an IBA, a Wild Life Refuge, and it is included in the Natura 2000 network as a Special Protection Area (SPA), under the Birds Directive (site: GR2550008, 2001; Birds Directive 2009/147/EC, 2009), and as a Site of Community Importance (SCI) and as a Special Area of Conservation (SAC), under the Habitats Directive (site: Habitats Directive 92/43/EEC, 1992; GR2550004, 1995; **Table 1**).

Due to man-made constructions over a period of seventy years, the GLw-Natura2000 area has been divided into sub-areas, with different characteristics (**Table 2**). At the east of GLw-Natura2000 area, the Tyflomitis artesian springs (a 0.072 Km^2 aquatic habitat covered with reeds) provide freshwater inputs in this area (Manzoni et al., 2020). The Tyflomitis area was once connected to the wetland, but at present it is separated from it via a dike which diverts most of the up-welling groundwater to the sea (Maneas et al., 2019).

Overall, the area is characterized by Mediterranean climate, with mild wet winters and dry summers (**Figure 2**). Evaporation and temperature trends, exhibit a maximum during the summer months, when precipitation is at its minimum (Maneas et al., 2019). The mean annual temperature is 18°C and the mean annual precipitation is approximately 695 mm/y (measured from 1956 to 2011 at the Helenic National Meteorological Service's station of Methoni, 15.6 km South of Gialova).

Field Data Collection

Bird monitoring was based on monthly field visits during the period October 2016 to January 2019. During the first year (October 2016–October 2017), field monitoring was not performed in December 2016 and July 2017. In March 2017, the censuses were more frequent (every second day from March 9 to 24) to better cover the spring migration. The second year of monitoring, started in January 2018 and was completed in January 2019. During the second year, field monitoring was not performed in December 2018.

Bird counts were based on a protocol proposed by the Hellenic Ornithological Society (Bousbouras et al., 2011) following a methodology with predetermined observation points with good view, to extensively survey the GLw-Natura2000 area. The protocol consisted of 13 points covering all the sub-areas of the wetland (**Figure 1**). The observations started with the first light spending maximum 30 minutes at each point. During the breeding period, extra observation time was spent in specific points to thoroughly look for nests and juveniles. The field equipment consisted of binoculars (10x magnification), a scope (20-60x85) and a digital camera. During fieldwork, the Collins birds guide (Svensson et al., 2010) was used when help in bird identification was needed. Apart from the intense March

TABLE 2 | Description of the sub-areas inside the GLw-Natura2000 area, based on Maneas et al. (2019).

Sub-areas	Size	Characteristics
Lagoon (L)	2.483	Open shallow water area. The average depth of the main water body is approximately 0.6 m (Arvanitidis et al., 1999). Salt meadows at the fringes, and few embankments (creating small parallel islands) covered by halophytes are found at the north. High salinity fluctuations on an annual basis (Manzoni et al., 2020).
North-West area (NW area)	0.243	Shallow fish ponds and terrestrial area with limited vegetation in connection to the lagoon. It is flooded during the wet period and dried during the dry period.
North wetland (Nw)	0.444	Salt meadows which neighbor cultivations at the north, separated by relative deep drainage channels (with a maximum depth of 1.4 m in winter) within which some reeds remain today. High salinity fluctuations on an annual basis (Manzoni et al., 2020).
South wetland (Sw)	0.946	Shallow waters (maximum depth of 0.6 m in winter), salt meadows, and old embankments (creating small parallel islands) covered by halophytes. Limited reed vegetation. High salinity fluctuations on an annual basis (Manzoni et al., 2020).
Agriculture lands (A)	1.107	Managed areas mainly covered with olive trees (23%), horticulture and grain crops (36%).

Compared to the agricultural lands, the Lagoon, the north-western area, the North wetland and the South wetland are natural aquatic habitats. All sub-areas are given in Km^2. The total size of the GLw-Natura2000 is 5.295 Km^2, including the Tyflomitis area.

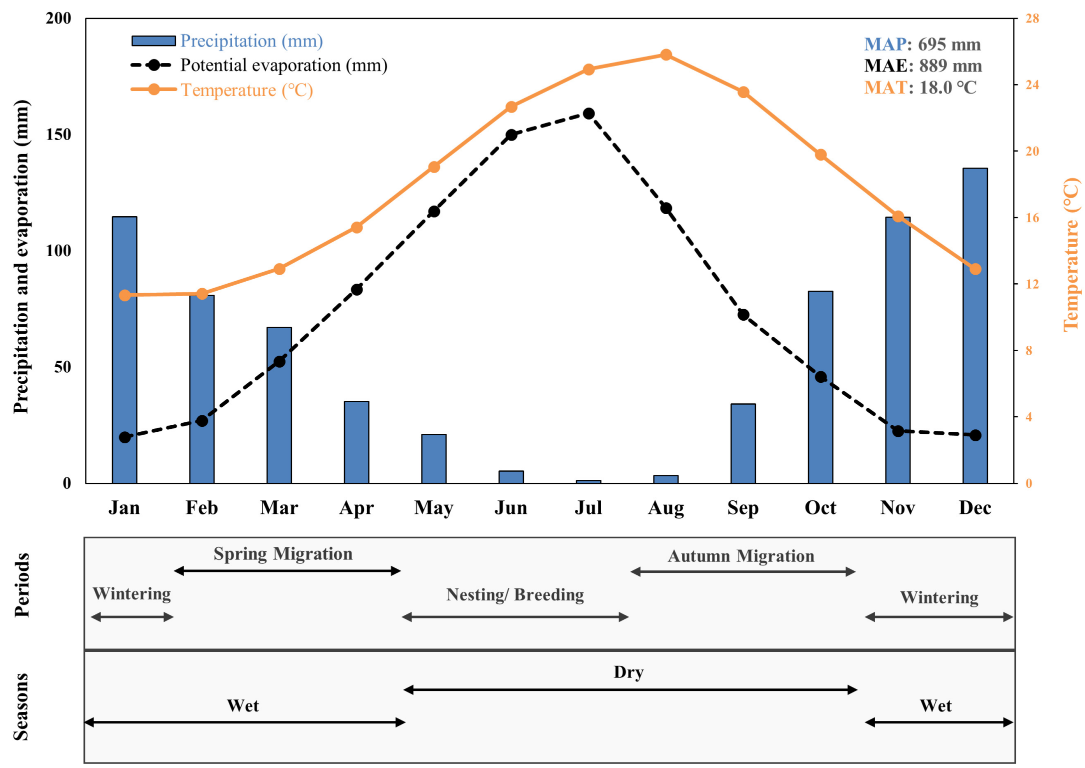

FIGURE 2 | Graph by Maneas et al. (2019), showing monthly mean precipitation (MAP), evaporation rate (MAE) and temperature (MAT) at the study area for the period 1956–2011. At the bottom, a graph showing the birds' periods and seasons used for the analyses.

2017 counts, the rest of the counts were conducted by the same two observers.

During the censuses, all observed birds (seen and heard) were recorded, i.e., a complete census (Gregory et al., 2004), and they were listed according to BirdLife taxonomic basis (BirdLife International, 2019). The primary focus of the monitoring was to record waterbirds richness and abundance (*Anatidae, Charadriidae, Laridae, Phalacrocoracidae, Phoenicopteridae, Podicipedidae, Rallidae, Recurvirostridae, Scolopacidae, Sternidae,* and *Threskiornithidae*). Waterbirds were noted and counted with high precision (richness and abundance at exact location) into the different sub-areas (**Table 2**).

The rest of the species, were only noted as extra species, but their abundance in the wetland was not counted. They were grouped as wetland-dependent species and as non-wetland species. Under wetland-dependent species, birds of prey which use the wetland as a feeding area (e.g., *Circus aeruginosus*, *Circus cyaneus,* and *Pandion haliaetus*), the *Alcedo atthis*, and species from the order of Passeriformes (*Acrocephalus arundinaceus, Acrocephalus melanopogon, Acrocephalus scirpaceus, Acrocephalus schoenobaenus, Riparia riparia, Emberiza schoeniclus, Cettia cetti, Cisticola juncidis, Remiz pendulinus, Panurus biarmicus*) which also depend on wetland habitats, were added.

Data Analysis

Diversity Indices

Species richness (S), as the number of observed species per month, was estimated for all the observed species, and for the different categories (waterbirds, wetland-dependent, non-wetland, all observed species).

Waterbirds species abundance (N), as the number of observed waterbirds during each count, was estimated for the whole GLw-Natura2000 area, and for each of the different sub-areas (as described in **Table 2**).

Waterbirds relative abundance (RN), as the number of observed waterbirds during each count per unit area (km^2), was estimated for each sub-area to allow comparisons during seasons and during periods.

Waterbirds Shannon-Weiner Index (H) was calculated based on species abundance using the Shannon and Weaver (1949) formula (Bibi and Ali, 2013; Issa, 2019):

$$H = -\left[\sum Pi * \ln(Pi)\right]$$

where (H) is the Shannon Index, Pi is the proportion of each species in the sample, and ln(Pi) is the natural logarithm of this proportion. The index was estimated for each sub-area to allow comparisons during seasons and during periods.

Temporal and Spatial Variations

Our analysis, was divided into two seasons following weather patterns as described in section "Study Area" (Wet: November–April and Dry: May–October), and four periods based on birds' patterns (Wintering, Breeding/ Nesting, Spring and Autumn migration) (Kardakari, 2000; **Figure 2**). Since the four periods overlap with each other (Kardakari, 2000), to avoid double counts in our statistical analysis, we followed the below month selection:

Wintering period (W):	November to January
Spring migration (sM):	February to April
Breeding/ Nesting (B):	May to July
Autumn migration (aM):	August to October

To assess the temporal distribution of wetland-dependent, non-wetland and of all the observed birds, the index of species richness was estimated for the two seasons and the four periods.

To assess the temporal distribution of waterbirds, apart from species richness we also used the index of species abundance. Both indices were estimated for the two seasons and the four periods. Since the different sub-areas are of different size, to assess the spatial distribution of waterbirds per season and per period, we used the indices (RN) and (H). As described above, relative abundance is the number of birds per unit area, thus it is independent of the size of each area and suitable for site comparisons. The Shannon-Weaver index is commonly used for site comparisons, and it was calculated in order to interpret differences in the species diversity (e.g., the index increases as both the richness and the evenness of the community increase) (Bibi and Ali, 2013; Issa, 2019). Nonetheless, we also present our (S) and (N) results, to provide a broader view of how waterbirds use the aquatic habitats of the GLw_Natura2000 area.

Statistical Analysis

The count data were analyzed with GLMs (Generalized Linear Models) assuming Poisson or Negative Binomial (for overdispersed data) distribution of the dependent variable, using a log link function, as it is suggested for count data analysis (Seavy et al., 2005; O'Hara and Kotze, 2010; Warton et al., 2016).

To test if there were statistically significant differences in species richness and abundance among the seasons, and among the periods, our interpretation was based on pairwise comparisons of the EM Means (Estimated Marginal Means) produced by the GLMs, applying the Bonferroni adjustment for multiple comparisons. The analysis of (S) was conducted with Poisson GLMs considering the individual effects of season and period. The corresponding analysis of (N) was conducted with Negative Binomial GLMs since its distribution was found to be overdispersed.

To identify statistically significant differences in relative abundance across the different sub-areas per season and per period, our interpretation was based on pairwise comparisons of the EM Means produced by the GLMs with Bonferroni adjustment for multiple comparisons. The analysis of (RN) for each season and each period was conducted with Negative Binomial GLMs, using sub-areas as the factor with the main effects. For comparing the Shannon diversity indices, we relied on the Hutcheson *t*-test, which is developed as a method to compare the diversity of two community samples using the Shannon diversity index (Hutcheson, 1970).

In all the analysis the statistical significance was set at $\alpha \leq 0.05$. The analysis and processing of the results was conducted in Microsoft Excel, and the statistical analysis was conducted in IBM SPSS Statistics Data Editor.

Species With Higher Conservation Value

The conservation status of each species at International level was retrieved from the databases of the "International Union for the Conservation of Nature" (IUCN, 2020) and Wetlands International (Wetlands International, 2020). The conservation status of each species at European level was retrieved from the "European Red List of birds" (BirdLife International, 2015) and the Annex I of EU Birds Directive (Birds Directive 2009/147/EC, 2009). The national conservation status was retrieved from the "The Red Book of Endangered Species in Greece" (Legakis and Maragou, 2009). For the observed endangered species, data of their regional population (1% threshold) were retrieved from the database of Wetlands International (Wetlands International, 2020). For these species, the estimates of their Greek population were retrieved from several sources (Legakis and Maragou, 2009; Handrinos et al., 2015; IUCN, 2020).

RESULTS

Birds Richness and Seasonality

A total of 149 bird species representing 43 families and 15 orders were recorded during the period October 2016–January 2019, in the GLw-Natura2000 (**Supplementary Table A1**). Out of the total

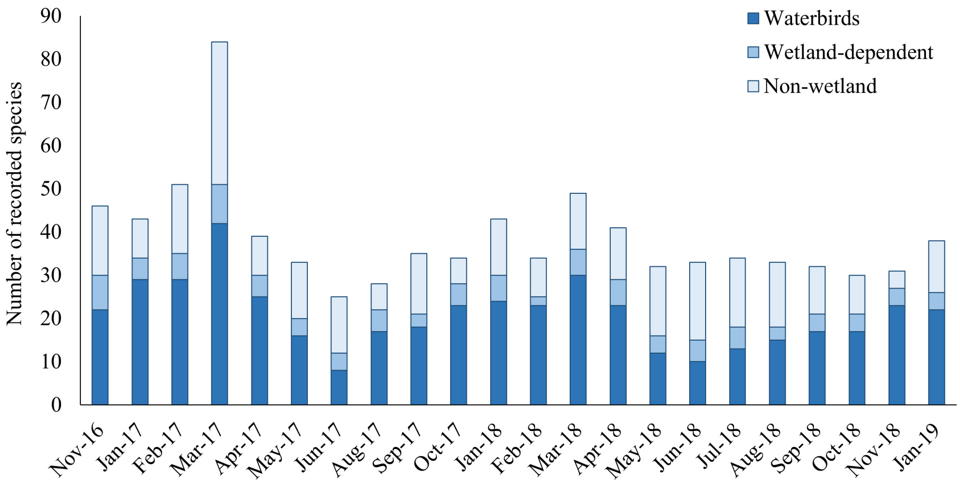

FIGURE 3 | Number of monthly recorded species in the GLw-Natura2000 based on counts during the period November 2016—January 2019.

TABLE 3 | Estimated marginal means (Mean) and standard errors (SE) for waterbirds' richness (S) and abundance (N) during the different seasons and periods, based on GLM analysis (GLM analysis for S was based on a Poisson distribution, and GLM analysis for N was based on a Negative Binomial distribution, using log as a link function in both analyses).

Indices		S			N		
		Max	Mean	SE	Max	Mean	SE
Periods	Wintering	29	24^{ab}	2.2	2912	1781^{a}	419.5
	Spring Migration	38	29^{b}	2.2	1937	1133^{a}	243.9
	Breeding/Nesting	16	12^{c}	1.5	204	135^{b}	32.2
	Autumn Migration	23	18^{a}	1.7	858	368^{c}	79.4
Seasons	Wet	38	27^{m}	1.6	2912	1428^{m}	273.5
	Dry	23	15^{n}	1.2	858	262^{n}	50.4

The maximum observed values during the field visits are also reported in the table (Max). Numbers in the same column of periods and of seasons that do not share the same superscript letter are statistically significant different (α < 0.05).

number of species, 81 (54 %) species were identified as wetland related species, of which 66 (44%) species were waterbirds representing 7 orders and 11 families (**Supplementary Table A1**). On a monthly basis, an average of 38 species were detected at the GLw-Natura2000 area (min: 25, max: 83) (**Figure 3**). The highest number of bird species was recorded during the Spring migration and the lowest during the Breeding/ Nesting period (**Figure 3** and **Supplementary Table A2**).

For all the observed birds (waterbirds, wetland-dependent and non-wetland species), species richness during the Wet season (45 ± 2.03) (mean $\pm$ standard error of mean) was higher ($p < 0.001$) than the Dry season (32 ± 1.7). When comparing between periods, Spring migration period was the most diverse. Species richness during the Spring migration (50 ± 2.8) was higher ($p < 0.001$) when compared to the Breeding/ Nesting (31 ± 2.5) and the Autumn migration periods (33 ± 2.3), but not when compared to the Wintering period (40 ± 2.8). For wetland-dependent and non-wetland species, the estimates of the GLM showed no statistically significant difference of species richness per season or per period, indicating that the wetland is used by birds all year around.

Waterbirds

Temporal Variations

Waterbirds richness during the Wet season was around 75% higher when compared to the Dry season ($p < 0.001$) (**Table 3**). When comparing between periods, the Spring migration period was the most diverse. The GLM showed that species richness during the Spring migration period was higher than the Breeding/ Nesting period by almost 140% ($p < 0.001$), and by almost 60% ($p = 0.001$) when compared to the Autumn migration period. Species richness during the Spring migration was also higher when compared to the Wintering period, but this difference was not statistically significant (**Table 3**). The Wintering and the Autumn migration periods were also more diverse when compared to the Breeding/ Nesting period, but not when compared to each other (**Table 3**). According to the model, during the Wintering period the species richness was almost twice as high ($p < 0.001$), and during the Autumn migration around 50% higher.

Waterbirds abundance per season and per period were analyzed using a GLM with Negative Binomial distribution choosing the log link function. The GLM showed that waterbirds

TABLE 4 | Maximum recorded value (Max), average and standard deviation (Mean and SD) of waterbirds' richness (S) and abundance (N) across the aquatic habitats of the GLw_Natura2000 area per season and per period.

Index			S				N			
	Sub-areas		Lagoon	NW area	N. Wetland	S. Wetland	Lagoon	NW area	N. Wetland	S. Wetland
Period	Wintering	Max	15	15	9	21	1,586	278	137	1,115
		Mean	11	12	5	17	864	174	46	686
		SD	3.6	2.4	2.9	2.8	597.4	60.8	53.6	370.3
	Spring migration	Max	22	18	10	26	1,026	224	67	790
		Mean	15	10	5	18	574	102	18	433
		SD	3.9	5.3	3.7	4.4	442.4	72.8	24.6	265.5
	Breeding/ Nesting	Max	6	1	4	13	44	7	18	160
		Mean	4	1	2	10	21	3	10	86
		SD	1.8	0.5	1.5	2.7	14.6	2.8	8.7	48.7
	Autumn migration	Max	11	9	4	14	238	74	18	641
		Mean	7	6	3	12	107	41	6	214
		SD	2.6	2.5	1.4	1.2	75.08	23.4	6.4	211.6
Season	Wet	Max	22	18	10	26	1,026	224	67	790
		Mean	14	11	5	17	706	134	31	548
		SD	4.2	4.1	3.2	3.6	513.4	74.5	40.7	328.02
	Dry	Max	11	9	4	14	238	74	18	641
		Mean	6	4	2	11	68	24	8	156
		SD	2.7	3.3	1.4	2.3	69.9	25.7	7.5	166.8

abundance between the two seasons was statistically significant different ($p < 0.001$), with estimated abundance during the Wet season being more than 400% higher (**Table 3**). The Wintering period had almost 57% more birds compared to the Spring migration period, but the difference was not statistically different. Waterbirds abundance during the Wintering period was higher when compared to the Breeding/ Nesting and Autumn migration periods, by around 1,200% ($p = 0.001$), and 380% ($p = 0.006$), respectively. Similar to the Wintering period, waterbirds abundance during the Spring migration period, was higher when compared to the Breeding/ Nesting and Autumn migration periods by around 740% ($p < 0.001$) and 200% ($p = 0.017$), respectively. In fact, waterbirds abundance during the Breeding/ Nesting period was statistically significant lower compared to the Autumn migration period as well, with the abundance being 65% lower ($p = 0.040$) (**Table 3**).

Taken together, the above results suggest that waterbirds' diversity varied significantly between seasons, with higher richness and abundance during the wet season compared to the dry season. In particular, during the Nesting/ Breeding period the wetland had the lowest species richness and abundance. On the contrary, during the Wintering and the Spring migration periods, the wetland held the highest abundance. Our results also suggest that compared to the Autumn migration, the Spring migration period was more diverse in terms of both species richness and abundance.

Spatial Variations

Waterbirds variations across the aquatic habitats of the GLw-Natura2000 area varied within the different seasons and periods. Variations in waterbirds richness and total abundance provided a qualitative approach of how each sub-area was used by waterbirds per season and per period (**Table 4**). The S. Wetland sub-area held the highest number of species per season and per period. During the Wet season, the lagoon had the highest abundance during both periods. On the contrary, during the Dry season, waterbirds abundance was higher at the S. Wetland sub-area. Waterbirds richness at the NW area was high during the Wintering and both migration periods, but low during the Breeding/ Nesting period. The N. Wetland sub-area held the lowest numbers of waterbirds richness and abundance during most of the periods, except for the Breeding/ Nesting period.

Our quantitative approach was based on the variations of (RN) and (H) indices across the four sub-areas per season and per period (**Table 5**).

During the Wet season, waterbirds relative abundance at the N. Wetland was lower from all the other sub-areas, and in particular by 88% ($p = 0.005$), 75% ($p = 0.032$) and 87% ($p = 0.006$) when compared to the S. Wetland, the Lagoon and the NW area, respectively. The RN in the Lagoon was 48 and 51% less when compared to the NW area and the S. Wetland, but these differences were not statistically different (**Table 5**). The Hutcheson t-tests, showed that the estimated value of the Shannon-Weaver index at the S. Wetland area, was statistically significant higher ($p < 0.001$) compared to all the corresponding (H) values in the other areas. The N. Wetland, was less diverse when compared to the Lagoon ($p = 0.003$) and the NW area ($p = 0.030$), which were the two sub-areas with similar values of H and thus similar species diversity during the Wet season (**Table 5**).

During the Dry season, waterbirds relative abundance at the S. Wetland was approximately five ($p = 0.048$) and eight ($p = 0.025$)

TABLE 5 | Estimated marginal means (Mean) and standard errors (SE) for waterbirds' relative abundance (RN), and corresponding mean values of the Shannon-Weaver index in the different sub-areas per season and per period.

Index	Sub-areas	Periods								Seasons			
		Wintering		Spring migration		Breeding/Nesting		Autumn migration		Wet		Dry	
		Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
RN	Lagoon	348^{a}	112.5	231^{a}	85.5	9^{a}	3.6	43^{a}	12.4	284^{a}	74.8	27^{a}	8.6
	NW area	714^{a}	230.5	417^{a}	154.5	13^{a}	5.1	167^{b}	47.5	553^{a}	145.3	97^{ab}	30.2
	N. Wetland	103^{a}	33.5	42^{a}	15.6	24^{a}	9.1	13^{a}	3.9	70^{b}	18.4	18^{a}	5.7
	S. Wetland	725^{a}	233.9	458^{a}	169.3	91^{a}	34.3	226^{b}	64.1	579^{a}	152.3	165^{b}	50.9
H	Lagoon	1.49^{ab}	0.23	1.82^{a}	0.27	1.02^{a}	0.39	1.21^{a}	0.59	1.67^{a}	0.3	1.12^{a}	0.49
	NW area	1.69^{ac}	0.31	1.43^{b}	0.66	0	0	1.18^{a}	0.51	1.56^{a}	0.52	0.65^{a}	0.71
	N. Wetland	1.16^{b}	0.72	1.13^{b}	0.71	0.34^{b}	0.35	0.68^{a}	0.56	1.14^{b}	0.68	0.53^{a}	0.49
	S. Wetland	1.85^{c}	0.33	2.17^{c}	0.25	1.76^{c}	0.38	1.88^{b}	0.22	2.02^{c}	0.32	1.83^{b}	0.29

Numbers in the same column of periods and of seasons that do not share the same superscript letter are statistically significant different ($\alpha < 0.05$).

times higher when compared to the Lagoon and the N. Wetland, respectively (**Table 5**). The (RN) at the NW area was also higher by approximately 250 and 430% when compared to the Lagoon and the N. Wetland, respectively, but these differences were not statistically different. During that time, the Shannon-Weaver index was the highest at the S. Wetland, and it was statistically different from all the other sub-areas ($p \leq 0.001$). However, there were no other significant differences across the other sub-areas.

During the Wintering period, most of the waterbirds were observed at the Lagoon (**Table 4**), but in terms of relative abundance there were no statistically significant differences with the other sub-areas (**Table 5**). The total abundance of the S. Wetland was higher than of the NW area, but the estimated means of the relative abundance between these two areas were not statistically different, indicating that they were both important for waterbirds. The (RN) at the S. Wetland and the NW area was approximately double when compared to the Lagoon and six times higher when compared to the N. Wetland, but these differences were statistically insignificant. The S. Wetland not only held the highest species richness, but it was also the most diverse area as indicated by the value of the Shannon-Weaver index. The difference was statistically significant higher when compared to the Lagoon and the N. Wetland ($p < 0.001$), but not when compared to the NW area, complementing the above results.

During the Breeding/Nesting period, the estimated mean of (RN) at the S. Wetland was about six, nine and almost three times higher when compared to the NW area, the Lagoon and the N. Wetland, respectively, but the differences were not statistically different (**Table 5**). The Shannon-Weaver index at the S. Wetland was significant higher from both the Lagoon and the N. Wetland ($p < 0.001$), which suggests that the S. Wetland was the sub-area with the higher waterbirds diversity during the Breeding/ Nesting period. The NW area was the sub-area with the lowest observations in terms of waterbirds richness and abundance, and thus the (H) was estimated at zero.

During the two migration periods, waterbirds patterns were different. During the Spring migration period, similar to the Wintering period, most of the waterbirds were observed at the Lagoon, and in high numbers at the S. Wetland (**Table 4**), but in terms of relative abundance there were no statistically significant differences with the other sub-areas (**Table 5**). On the other hand, during the Autumn migration period, the relative abundance at the Lagoon, the N. Wetland and the NW area were 80% (p = 0.030), 92% (p = 0.007), and 94% (p = 0.005) less when compared to the corresponding value at the S. Wetland (**Table 5**). The Shannon-Weaver index at the S. Wetland sub-area, was the highest in both migration periods, and different from all the other sub-areas ($p \leq 0.009$). During the Spring migration, the (H) values at the Lagoon were higher when compared to the NW area (p = 0.002) and to the N. Wetland (p = 0.004), a difference which was not evident during the Autumn migration (**Table 5**).

Summarizing the above, our results suggest that all the sub-areas supported waterbirds, but their occupation by waterbirds varied between seasons and periods. During the Wet season, waterbirds were more evenly distributed around the different sub-areas of the GLW-Natura2000 wetland, indicating that the whole area was used as a feeding or/and resting area. This pattern gradually started to change during the Dry season and became evident during autumn.

The S. Wetland was the sub-area with the highest species diversity during all year, and the highest relative abundance during the Dry season and corresponding periods (Breeding/ Nesting and Autumn migration). During our field visits, into this area we recorded most of the nests, and taken together these results suggest that the S. Wetland sub-area apart for suitable area for feeding and resting, it is also used as a nesting ground. Similar to the S. Wetland, the Lagoon was also used by waterbirds all year around, but mainly during the Wet season and the relevant periods (Wintering and Spring migration). The high numbers of (RN) and (H) at the NW area, during the Wintering and both migration periods, suggest that this area was important as a feeding and resting area. On the other hand, the limited observations during the Breeding/ Nesting period, and the absence of recorded nests suggest that waterbirds avoid to use this area as a nesting ground. The N. Wetland, was the sub-area which had the lowest (RN) and (H) during both seasons

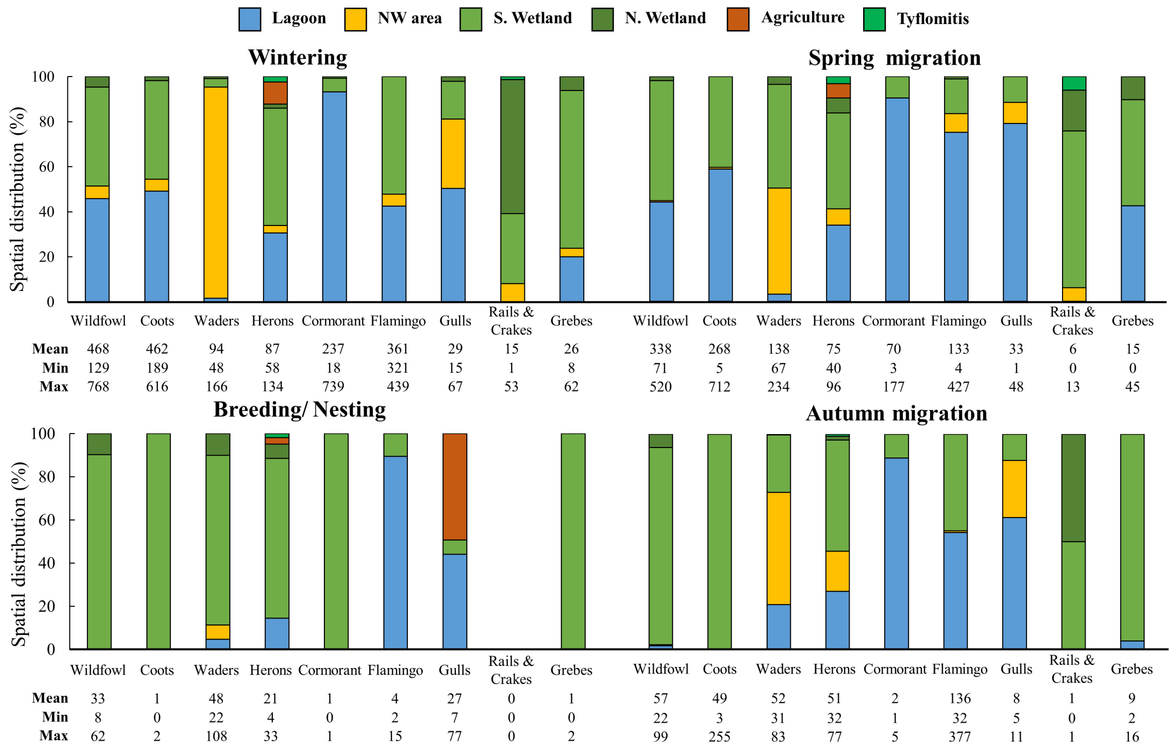

FIGURE 4 | Stacked column graph showing the spatial distribution (percentage) of waterbirds' categories in the different sub-areas of the GLw-Natura2000 area, based on their average population distribution during the different periods. The abundance (mean, minimum and maximum values) of each waterbirds category during the same period is presented in the table at the bottom of each graph.

and most periods. However, during the Breeding/ Nesting period, both indices were higher than those at the Lagoon and at the NW area, which highlights the importance of that area during that period. This finding is further supported by the fact that we recorded the N. Wetland as the basic nesting habitat of the *Cygnus olor* during both field visits (summer 2017 and summer 2018).

Categories in Space and Time

Wildfowl

Wildfowl observed at the wetland were dabbling and diving ducks, geese and a swan (**Supplementary Table A1**). Out of the 13 recorded wildfowl, 7 are listed in the IUCN lists (GR: one CR, four VU; Europe: two VU, International: one VU, one NT). Two species (*Anas platyrynchos* and *Cygnus olor*) were recorded as resident and breeding species, 6 as wintering species (*Anas acuta*, *Anas crecca*, *Anas clypeata*, *Anas penelope*, *Aythya nyroca,* and *Tadorna tadorna*) while the rest 5 Anatidae (*Anas strepera*, *Anser anser*, *Aythya ferina*, *Aythya marila,* and *Spatula querquedula*) were recorded during the Spring migration and marked as migrating species. During the Wintering and Spring Migration periods, wildfowl were observed in all the aquatic habitats of the wetland, but almost 90% of their population was observed at the Lagoon and S. Wetland sub-areas (**Figure 4**). During the dry season, when their numbers in the area were low, they were mainly observed at the east part of the wetland and in particular at the S. Wetland sub-area (**Figure 4**).

Coots

Coots were found in higher numbers during the wet season, and similar to wildfowl more than 90% of their population was observed at the Lagoon and S. Wetland sub-areas (**Figure 4**). During the dry season, the few coots were recorded at the S. Wetland side. One pair of coots was observed nesting in the area in 2018.

Waders

In total 25 different species of waders, of which 6 are threatened (GR: three VU, Europe: four VU, International: four NT), belonging to 3 different families were recorded (**Supplementary Table A1**). *Charadrius alexandrinus* was the only wader recorded in almost all counts and also nesting. Apart from the Kentish plover, *Himantopus himantopus* was also observed nesting in the area. However, the Black-winged stilt arrived in the area in spring and departed in autumn. Compared to waterfowl, during the Wintering period, waders were mainly observed at the NW-area side of the GLw-Natura2000 (**Figure 4**). During the Spring migration, their population was divided between the NW-area and the S. Wetland sub-areas (**Figure 4**). On the contrary, during the Nesting/ Breeding period, almost 90% of all waders were recorded at the east side of the wetland, in the S. Wetland sub-area (**Figure 4**). During the Autumn migration, their population was more spread, but still more than 75% of the observations were made in the NW-area and S. Wetland sub-areas (**Figure 4**).

Herons and allies

In total 8 different species of herons were counted in GLw-Natura2000 (**Supplementary Table A1**). *Egretta garzetta* and *Ardea cinerea* were recorded in all counts and marked as resident species. However even been present during the Breeding/ Nesting period no sign of nesting was recorded. The highest numbers of *Ardea alba* were recorded during the Wintering period, and they were present during the Spring and the Autumn migration as well. Small numbers of the other heron species were recorded during migration: *Ardea purpurea* and *Ardeola ralloides* in spring and *Botaurus stellaris, Ixobrychus minutus,* and *Nycticorax nycticorax* in autumn. During most of the periods, herons were observed all around the wetland with most frequent observations at the Lagoon and the S. Wetland sub-areas (**Figure 4**). Herons were among the waterbirds which were observed in the agricultural lands around the wetland, mainly during the wet season when the farmlands were wet or flooded (**Figure 4**). During the Breeding/ Nesting period most of their population was recorded at the S. Wetland sub-area (**Figure 4**). *Plegadis falcinellus* and *Platalea leucorodia* were observed in low numbers at the N. Wetland area during the spring migration.

Cormorants

Phalacrocorax carbo was the only cormorant observed in the area. The species was recorded wintering in the area and was present during the Spring migration period. The majority of their population, was recorded at the Lagoon side of the wetland (**Figure 4**).

Flamingos

Flamingos (*Phoenicopterus roseus*), arrived in high numbers during the Autumn migration and departed in February. They spent most of their time at the Lagoon and the S. Wetland sub-areas of the GLw-Natura2000, and they were occasionally observed at the NW-area (**Figure 4**).

Gulls

Larus michahellis was recorded during most counts and was the most common gull in the area. *Larus ridibundus* was recorded during the Wintering and both migration periods, and *Larus genei* only during the migration periods. Out of the 5 observed tern species, *Thalasseus sandvicensis* was present during the Wintering and the Spring migration periods. The other four terns (*Chlidonias hybrida, Chlidonias leucopterus, Hydroprogne caspia,* and *Sterna hirundo*), were only observed during the migration periods. During the wet season, gulls, and terns were observed in all the aquatic habitats of the wetland, but more than 50% at the Lagoon sub-area (**Figure 4**). During the Breeding/Nesting period, gulls were also observed at the agricultural lands (**Figure 4**). Their population during the Autumn migration was more spread, but still more than 50% of the observations were made at the Lagoon sub-area of the wetland (**Figure 4**).

Rails and crakes

Rallus aquaticus and *Gallinula chloropus* were recorded in few numbers in the S. Wetland and N. Wetland sub-areas, mainly during the wet period (**Figure 4**).

Grebes

Podiceps cristatus and *Podiceps nigricollis* were observed only during the Wintering and the Spring migration periods. On the other hand, two families with juveniles of *Tachybaptus ruficollis* were observed in August 2017, indicating breeding success of the species. Similar to coots and wildfowl, during the wet season more than 90% of their population was observed at the Lagoon and S. Wetland sub-areas (**Figure 4**). During the dry season, the majority of the observations were made at the S. Wetland sub-area (**Figure 4**).

Other wetland-dependent species

Three species of raptors (*Circus aeruginosus, Circus cyaneus,* and *Pandion haliaetus*) were observed hunting in the area during the Wintering period (**Supplementary Table A1**). Small numbers of *Grus grus* were recorded at the agricultural lands surrounding the wetland during Wintering and both migration periods. Kingfisher was also common in the area during the Wintering (at least 10) and both migration periods (at least 20) and was usually observed all around the wetland. The rest of the wetland-dependent species, belong to the order of Passeriformes, and were mainly recorded at the reeds zone of Tyflomitis artesian springs (**Supplementary Table A1**). *Acrocephalus arundinaceus*, *Acrocephalus scirpaceus,* and *Cisticola juncidi*s, were recorded nesting at Tyflomitis sub-area, while *Cettia cetti* in areas surrounding the wetland.

Species With Higher Conservation Value

Almost a quarter (40 species) of the total observed species are listed in the Annex I of the EU's Birds Directive (**Tables 6**, **7**).

As evident in **Table 6**, our case study, did not support big wintering populations (all counts were less than 1% of the regional population), but it supported many Nearly Threatened (NT), Vulnerable (VU), Endangered (EN) or Critically Endangered (CR) species, especially during the Spring migration. At an International level, two species are listed as VU and six as NT. At a European level, eight species are listed as VU, three as NT, and all the rest (139) as Least Concern (LC). At a national level, two species (*Anser anser* and *Plegadis falcinellus*) are listed as CR, six as EN, fourteen as VU, four as NT, and fourteen as LC (**Tables 3**, **4**). The rest 107 observed species have not been evaluated yet under the national IUCN criteria (**Supplementary Table A1**). From the species listed in **Table 6**, only the population of *Phalacrocorax carbo* was similar to the one reported in Kardakari (2000), and our recordings suggest declining numbers for most waterbirds including all the species with higher conservation value (**Table 3** and **Supplementary Table A1**).

DISCUSSION

Several of the threatened bird species with an IUCN status, and those listed in the Annex I of the EU Birds Directive are connected to wetland loss and degradation due to intensification of agriculture, drainage of wetlands, disturbance from tourism activities, illegal hunting and climate change

TABLE 6 | Observed waterbirds in GLw-Natura2000, which are threatened at International (Int), European (EU) and Greek (GR) level, and additional IBA/SPA species of concern (gray part of the table).

Observed waterbirds with a higher conservation status		IUCN status			Populations		Current study (2017–2019)			Kardakari, 2000 (1995–1999)		
Scientific name	**Common name**	**Int**	**Eur**	**GR**	**1%**	**GR**	**W**	**sM**	**aM**	**W**	**sM**	**aM**
Mareca strepera	Gadwall	LC	LC	**VU**	1,900	1,960^	9	21		65		
Anser anser	Greylag Goose	LC	LC	**CR**	350	360^		2		1		
Aythya ferina	Pochard	**VU**	**VU**	LC	6,000	32,800^		9		2,220		
Aythya marila	Scaup	LC	**VU**	NE	1,400	<1%^	2	1		no obs		
Aythya nyroca	**Ferruginous Duck**	**NT**	LC	**VU**	630	27^	2	12	4	26	15	
Spatula querquedula	Garganey	LC	LC	**VU**	13,400	<1%^		70			700	
Tadorna tadorna	Shelduck	LC	LC	**VU**	2,600	4,150^	7	9		43		
Fulica atra	Common coot	LC	**NT**	NE	25,000	89,000^	712			9,000		
Vanellus spinosus	**Spur-winged Lapwing**	LC	LC	**VU**	1,000	rare		1		No obs		
Vanellus vanellus	Northern Lapwing	**NT**	**VU**	**VU**	72,300	7,500	16			45		
Recurvirostra avosetta	**Avocet**	LC	LC	**VU**	390	3,500	2			4	35	
Calidris ferruginea	Culrew Sandpiper	**NT**	**VU**	NE	4,000	n.d.		28			700	
Limosa limosa	Black-tailed Godwit	**NT**	**VU**	NE	960	n.d.		8			255	
Numenius arquata	Eurasian Curlew	**NT**	**VU**	LC	7,600	1,800^^		21		20	225	
Larus genei	**Slender-billed Gull**	LC	LC	**VU**	1,700	3,000		2	1	20		
Larus melanocephalus	**Mediterranean Gull**	LC	LC	**EN**	2,400	500		38		No obs		
Chlidonias hybrida	**Whiskered Tern**	LC	LC	**EN**	2,000	<1%		1			39	
Thalasseus sandvicensis	**Sandwich Tern**	LC	LC	**VU**	1,100	<1,000	6	10		12	15	
Ardea purpurea	**Purple heron**	LC	LC	**EN**	350	175		4			79	
Ardeola ralloides	**Squacco Heron**	LC	LC	**VU**	390	85		2			45	
Botaurus stellaris	**Bittern**	LC	LC	**EN**	1,200	<1%		2		5	10	
Ardea alba*	**Great White Egret**	LC	LC	**VU**	780	1,500	83			360		
Nycticorax nycticorax	**Black-crowned Night Heron**	LC	LC	**NT**	1,600	n.d.			20			2,500
Plegadis falcinellus*	**Glossy Ibis**	LC	LC	**CR**	800	440		11			380	
Platalea leucorodia	**Eurasian Spoonbill**	LC	LC	**VU**	170	320		2		5	10	
Phalacrocorax carbo	Great cormorant	LC	LC	NE	5,000	39,600^^	739			1,000		
Egretta garzetta	Little egret	LC	LC	LC	730	2,600^^	54	32	21	200	2,000	
Tringa stagnatilis	Marsh sandpiper	LC	LC	NE		n.d.		11			100	
Tringa glareola	Wood sandpiper	LC	LC	NE	20,000	29		9			1,150	

*In the first column, bold text indicates species which are listed in the Annex I of EU's Birds Directive. In the IUCN status column, bold text is used to highlight the IUCN status. Species marked with the symbol * are both threatened and IBA/SPA species. The 1% threshold of the regional population, and the Greek population are also reported in the table. Numbers under Greek population are estimates reported in Legakis and Maragou, 2009. In cases where there were no estimates, the < 1% is used to show that the national population is less than the 1% of the reginal population (Legakis and Maragou, 2009). In the same column, symbol ^ and symbol ^^, are estimates reported at Handrinos et al. (2015) (period 1997–226) and IUCN (2020) (period 2007–2013), respectively. Under the column "current study" we provide the max observed abundance recorded in our study. Under the column "Kardakari, 2000" we provide the max observed abundance reported in that study (aM for Nycticorax nycticorax, and sM for Egretta garzetta are reported estimates of the total migration population of these species). Abbreviations: W for wintering, sM and aM for spring and autumn migration, respectively, n.d., no data available; no obs., no observations; CR, Critically Endangered; EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concerned; NE, Not Evaluated.*

(European Commission, 2016; BirdLife International, 2018). In this context, the Glw-Natura2000 area is a representative IBA/SPA wetland (Heath et al., 2000; Birds Directive 2009/147/EC, 2009), which has been altered by past drainage efforts and agriculture expansion (Maneas et al., 2019). Furthermore, it is located in an area which is fast becoming a tourist destination, an economic activity which also poses a threat for coastal wetlands in Greece (Maragou and Mantziou, 2000; Maneas et al., 2019). At present, the area suffers from a lack of management, and conservation actions for achieving a Favorable Conservation Status (FCS) for the avian communities is urgently needed. However, before drafting any management plan it is important to understand the current status and distribution of waterbirds, also in relation to previous studies in this area (Kardakari, 2000; Bousbouras et al., 2011).

Waterbirds' Status and Distribution

During the current study 149 species (average: 38 species per month) were recorded in the area, including 36 threatened species at an International, European or/and national level, and 40 species listed in the Annex I of the EU Birds Directive (21 species are listed as both threatened and under Annex I). 54% (81 species) of the total observed species have been identified as wetland-dependent birds, out of which 66 species (44%) were waterbirds. The results show that the wetland is mostly used by ducks, coots, herons, waders, flamingos and cormorants.

TABLE 7 | Observed birds in GLw-Natura2000 which are threatened at International, European and Greek level.

Observed birds with an IUCN status			IUCN status			Presence in GLw
Bird category	***Scientific name***	**Common name**	**International**	**Europe**	**Greece**	**Period**
Wetland dependent species	***Circus aeruginosus***	**Marsh Harrier**	LC	LC	**VU**	W, aM-sM
	Circus cyaneus	**Hen Harrier**	LC	**NT**	NE	W, aM-sM
	Alcedo atthis	**Kingfisher**	LC	**VU**	DD	W, aM-sM
	Acrocephalus melanopogon	**Mostached Warbler**	LC	LC	**VU**	aM-**sM**
Non-wetland Species	***Hieraaetus pennatus***	**Booted Eagle**	LC	LC	**EN**	sM
	Aquila pomarina	**Lesser Spotted Eagle**	LC	LC	**EN**	sM
	Circaetus gallicus	**Short-toed Snake Eagle**	LC	LC	**NT**	aM-sM
	Streptopelia turtur	Turtle Dove	**VU**	**VU**	NE	aM
	Anthus pratensis	Meadow Pipit	**NT**	**NT**	NE	aM-**sM**
	Lanius minor	Lesser Grey Shrike	LC	LC	**NT**	aM
	Alauda arvensis	Eurasian Skylark	LC	LC	**NT**	**aM**-sM

In the first column, bold text indicates species which are listed in the Annex I of EU's Birds Directive. In the IUCN status column, bold text is used to highlight the IUCN status. W for wintering, sM and aM for spring and autumn migration, respectively. EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concerned; NE, Not Evaluated; DD, Data Deficient.

Gulls, grebes, rails, and crakes were found in relatively low numbers. The higher waterbirds abundance was recorded during the Wintering period, and the higher waterbirds richness during the Spring migration period.

As reported in previous studies (Kardakari, 2000; Bousbouras et al., 2011), our results suggest the species richness and abundance were higher during the Spring migration compared to the Autumn migration. The location of the area along the Balkan peninsula, combined with better habitat during spring, should be the main reasons why the wetland has higher species richness in spring. Indeed, located at the most south-western part of the Balkan peninsula, the wetland is used as a suitable stopover for many exhausted spring migrants who have flown non-stop over the Mediterranean Sea, or even non-stop from south of Sahara (Bortels et al., 2011). The situation in the autumn is very different. When migrating from north to south over the Balkan peninsula birds have the possibility of staying in the larger wetlands at the northern part of Greece, which are very suitable stopovers for waterbirds (Hellenic Ornithological Society [HOS], 2019a). This may make a stop at GLw not as important as it may be in spring, when the wetland may be the first wetland that the birds encounter. In addition, during spring, increased volumes of freshwater due to precipitation and inflows from the catchment, result in lower salinity values and wider expansion of aquatic habitats compared to autumn conditions (Manzoni et al., 2020), making the whole area a more suitable habitat for resting and feeding.

The results of this study also confirm the importance of the site as a Wintering area (Kardakari, 2000). Each winter the GLw-Natura2000 held a high number of waterbirds; mainly wildfowl, coots, herons, cormorants and flamingos. The fluctuations of wintering populations from year to year (2017, 2018, and 2019) were expected and could be linked to reasons outside the site's condition. Adam et al. (2015) found that very cold winters in northern and central Europe forced the birds to move further south to find open water, making concentrations of birds to appear higher than usual in places such as our case study. In the same way, a warmer winter makes it possible to avoid flying further south than necessary, lowering the numbers in southern wetlands, as in 2018 in the GLw-Natura2000.

During the Nesting/ Breeding period the wetland had the lowest species richness and abundance, but some of the observed species (e.g., *Rallus aquaticus*, *Gallinula chloropus,* and *Tachybaptus ruficollis*) were cryptic species, and their abundance in the area might be higher. Nevertheless, some species nest in the area (e.g., *Anas platyrhynchos*, *Cygnus olor*, *Himantopus himantopus, Charadrius alexandrinus*), and it is important to keep their habitats untouched. Even though, large areas are needed by many species to maintain their populations, it is important to recognize the complementary value of smaller remnants such as smaller wetlands to their successful conservation (Fischer and Lindenmayer, 2002).

When compared to the other sub-areas, the higher values of the Shannon-Weaver index at the S. Wetland indicated that this part of the wetland had the highest species diversity all year around, and regularly supported high numbers of birds as shown by the comparisons of the relative abundance. Counter wise, the N. Wetland sub-area, held the lowest species diversity, abundance and relative abundance. Nevertheless, this sub-area was the basic habitat for the critically endangered (CR) Glossy Ibis. During the wet season, the Lagoon and the NW area had high species richness and abundance. The S. Wetland and the NW area were identified to be the most important sub-areas for waders during the Wintering and both migration periods. On the contrary, during the Dry season, and especially during the Breeding/ Nesting period, bird diversity was much lower in most of the aquatic sub-areas. In fact, our results suggest that during the Dry season, the majority of species and individuals (including most breeding species) were summoned at the S. Wetland sub-area.

Our observations in the agricultural lands, and the Tyflomitis artesian springs sub-areas suggest that these areas are less used by waterbirds. Nevertheless, during the Wet season, herons and in particular the *Ardea alba* (a species which is listed as CR in the Greek IUCN red list) were frequently observed in the agricultural areas. At the Tyflomitis artesian springs sub-area, the observations were also few, but our recordings suggest that this sub-area is important as a nesting area for small wetland-dependent species.

Comparisons With Previous Studies

Out of the eight IBA/SPA species of concern, *Phalacrocorax carbo* has been recorded in numbers similar to Kardakari, 2000, but for the other two wintering IBA waterbirds our results suggest that their abundance has been dramatically decreased since the last IBA evaluation in 2001 (BirdLife International, 2001). The maximum observed wintering abundance for *Egretta garzetta* (54 individuals) and *Ardea alba* (83 individuals) during this study (based on three winter counts), was about 75 and 80%, respectively, less when compared to corresponding values given in Kardakari (2000) (**Table 6**). Regarding the migrant species, although our observations during the spring migration period gave a picture of the species abundance on the day of the census, and the total number that use the wetland is multiple, they indicate that migrant species abundance has been largely decreased since the last IBA evaluation in 2001 (BirdLife International, 2001). *Plegadis falcinellus* was also recorded in low numbers during the Spring migration period (max: 11 individuals). This species has fluctuating numbers, but in the past, it was recorded in higher numbers (391 in 1996, and 366 in 1999) (Kardakari, 2000). *Tringa glareola* was referenced as the second most frequent wader in the wetland with maximum counts up to 1,150 individuals (Kardakari, 2000), but under this study we never encountered so many individuals (spring max: 8 individuals). In the same study, *Tringa stagnatilis* was recorded as a common wader during the Spring migration (Spring max: 100 in 1996), but under this study we recorded in total five individuals. Finally, *Glareola pratincola* and *Gelochelidon nilotica* were not observed during the study period, even though expected, since they were reported as “regular passage migrant often in large numbers” and “fairly common passage migrant,” respectively (Kardakari, 2000).

During the monitoring period of this study we have recorded one additional wildfowl species (*Aythya marila*—single observation) during the Spring migration period. On the contrary, our list contains 36 species less. The absence of twenty-six species from our list, which are reported as rare and very rare species in Kardakari, 2000, could be explained by the frequency and the type of our monitoring efforts. *Porzana porzana, Porzana parva,* and *Lymnocryptes minimus* are cryptic species and difficult to monitor (Stanley et al., 1977–1994; Kardakari, 2000), and *Scolopax rusticola* uses the wetland during the night (Kardakari, 2000), which could explain why we have not recorded these species. The absence of other migratory species from our list (*Ciconia Ciconia*, *Ciconia negra, Gelochelidon nilotica*, *Glareola pratincole, Arenaria interpres*), could be partly explained by the low frequency of our observations. Even though, we implemented frequent observations during March 2017, this only occurred for 2 weeks, and it is very likely that we have missed species which passed by the wetland during migration and stayed for only a short period of time. However, the absence of *Hydrocoloeus minutus* and the decline in the abundance of the six species of concern cannot be explained by the frequency of our observations.

A noticeable decline in the wintering population of several wildfowl species (except *Anas platyrhynchos*), herons, and the coot, as well as in the migrating populations of waders was also evident when compared to Kardakari (2000). Another noticeable difference was the status of flamingos. In Kardakari (2000), the species is referenced as “regular passage spring migrant and winter visitor in small numbers,” but our observations show that at present it is a common winter visitor in moderate numbers. Possible explanations such as declines and shifts due to climate change (Lehikoinen et al., 2013; Adam et al., 2015; Ramírez et al., 2018) or/and habitat degradation (Wang et al., 2011; Tavares et al., 2015; Brandis et al., 2018) which have been observed in other areas worldwide, should not be ruled out and further investigation will be needed to better understand these differences, which is out of the scope of this study.

Conservation Needs and Management Suggestions

The characterization of the area as an IBA and SPA (under the EUs Birds Directive)—also as an SCI and SAC (under the EUs Habitats Directive)—in the early 2000s, provided an adequate framework for site managers to take concrete actions for nature conservation and waterbirds protection, but few actions have been taken so far (Maneas et al., 2019). Instead, the area has been under unstable management for several years (Maneas et al., 2019), and the lack of conservation actions has added pressure to waterbirds survival at a national, European and International level. As analyzed above, compared to previous IBA evaluation (BirdLife International, 2001), our results suggest a dramatic decline in species abundance for 66.6% of the wintering IBA species, and for 100% of the migrant IBA species in Gialova Lagoon wetland, an outcome which should not be overlooked by the site managers.

Despite being an IBA and part of the Natura 2000 network, the protection status of the GLw could be further enhanced. Similar to previous evaluation (BirdLife International, 2001), our results indicate that the area cannot meet Ramsar criterion 5, which is to “regularly support 20,000 or more waterbirds” (Ramsar Sites Criteria, 2020), as the maximum abundance of waterbirds observed under this study was 2,912 in January 2017. The observed decrease of waterbirds abundance during the period of this study could imply that the area may no longer fulfill criterion 6, which is to “regularly support 1% of the individuals in a population of one species or subspecies of waterbirds” (Ramsar Sites Criteria, 2020), and equivalent IBA criterion A4i (BirdLife International, 2001). However, as mentioned above, our observations during the migration periods gave a picture of the species abundance on the day of the census, and the total abundance is multiple. Thus, we cannot make estimates based on the type of our monitoring approach, and further investigation is needed.

On the other hand, our results have confirmed the importance of the GLw-Natura2000 as an important stopover area for many waterbirds (including several threatened species), and thus we suggest that the area continue to meet Ramsar criterion 4, which is to "support animal species at a critical stage in their life cycle, or provide refuge during adverse conditions" (Ramsar Sites Criteria, 2020). Apart from an important stopover area, the GLw_Natura2000 site most likely provides a refuge during adverse conditions. As previously discussed, very cold winters in higher latitudes could force birds to move further south to find open water (Adam et al., 2015), and there was a strong indication that the area was increasingly used by waterbirds in January 2017, when exceptionally cold and snowy conditions occurred in Eastern Europe and the Balkan peninsula (European Cold Wave, 2017). Furthermore, our monitoring revealed that during the Spring and Autumn migration periods, the GLw-Natura2000 area supported several bird species which are threatened at an International (2 VU, 6 NT), European (8 VU, 3 NT) and national (2 CR, 6 EN, 14 VU, 4 NT) level, and it is already known of supporting the only European population of the CR African Chameleon (Legakis and Maragou, 2009). To that end, the area meets Ramsar criterion 2, which is to "support vulnerable, endangered or critically endangered species" (Ramsar Sites Criteria, 2020), and we suggest that it should be re-evaluated as to become a "*Site of international importance for conserving biological diversity*," under the Ramsar Convention.

Moreover, placing it in a larger regional context, it can be stated to be one of the few remaining wetland IBA's in south-western Greece (Hellenic Ornithological Society [HOS], 2019a), along the Mediterranean/Black Sea Flyway, an internationally important migration route (BirdLife International, 2017). Recent studies reveal that due to climate change, the distances that long-distance migrants will need to travel between suitable breeding and non-breeding habitats will significantly increase, and this increase in distance will require increase in refueling stopovers (Howard et al., 2018). It should be noted that the total abundance of waterbirds that use the wetland during migration is not known as their arrivals and departures are frequent and irregular. Our observations during the migration periods gave a picture of the number of individuals that have stopped in the wetland on the day of the census, but the total abundance during each migration period is multiple. If this wetland is to be further degraded or completely dried, the next closest wetland along this route is 110 km further north (Hellenic Ornithological Society [HOS], 2019a), making the GLw-Natura2000 area a very important node in the connectivity of wetlands (Smith and Chow-Fraser, 2010).

During the last 70 years, an annual water deficit of 200 mm per year and limited fresh water inputs, due to man-made constructions and increased irrigation needs, have led to increased salinity values and extensive vegetation mortality of habitats suitable for waterbirds especially at the S. Wetland side (Maneas et al., 2019). Apart from limitations in vegetation expansion, salinity values are critical for benthic communities (Newton et al., 2014) and fish (Zoulias et al., 2017), which are vital in the food chain for several waterbirds. At present, fresh water availability is affected by inland human activities, and the wetland is currently lacking fresh water inputs (Maneas et al., 2019; Manzoni et al., 2020). Waste waters from the olive-oil industry pollute surface water bodies (which could flow into the wetland, but currently flow into the sea), while increased irrigation needs in agriculture, and water demand for domestic use limit the amount of available groundwater resources during summer (Maneas et al., 2019). Unless freshwater inputs are enhanced by restoring hydrologic connectivity between the lagoon and the surrounding freshwater bodies, under future drier and warmer conditions, salinity in the lagoon is expected to increase (Manzoni et al., 2020).

The restoration of fresh water flows, could improve the status of nesting and feeding habitats enhancing FCS for several species (Stanley et al., 1977–1994; Kardakari, 2000; Bousbouras et al., 2011). For instance, if the water level at the S. Wetland sub-area could be kept at high levels until April/May (followed by level decreasing trends to avoid flooding of the nests), this could improve nesting habitats of existing nesting species such as the Mallard (*Anas platyrynchos*), and the Black winged stilt (*Himantopus himantopus*). Such management efforts could re-establish favorable nesting conditions for threatened species such as the vulnerable Garganey (*Spatula querquedula*), which used to nest at fresh water habitats in the past (Kardakari, 2000), and create favorable conditions for several waders including the Wood sandpiper (*Tringa glareola*) and the Marsh sandpiper (*Tringa stagnatilis*), two out of eight IBA characterization species. Less saline water conditions could lead to reed expansion (Álvarez-Rogel et al., 2007), which is the basic nesting habitat for other threatened species such as the nationally EN Purple Heron (*Ardea purpurea*) and the Ferruginous Duck (*Aythya nyroca*), which is considered VU at a European and NT at an International level (Legakis and Maragou, 2009; BirdLife International, 2015; IUCN, 2020). Increased water level during migration, could help to preserve important habitats for Collared pratincole (*Glareola pratincola*), Wood and Marsh sandpipers, and the critically endangered Glossy ibis (*Plegadis falcinellus*).

Our field observations revealed that the areas located at the wetland's fringe support several endangered waterbirds, but at present these areas are the ones most threatened by human activities. In the existing agricultural areas, plots with horticulture—which were used during the Wintering period by the nationally VU *Ardea alba*—are gradually replaced by olive cultivations (Maneas et al., 2019). In the past, management efforts have led to conflicts between farmers and Natura 2000 managers (Hellenic Ornithological Society, 2000), and due to poor enforcement of guidelines inside the GLw-Natura2000 area, parts of the N. Wetland sub-area—which hosts the nationally CR Glossy ibis (*Plegadis falcinellus*)—have been gradually transformed to cultivated land (Maneas et al., 2019). Unless the degradation of these sub-areas is halted, by engaging the local farmers to strategic decisions about the management of the GLW-Natura2000 area, it is very likely that the numbers of these species will continue to decline.

Human disturbance has been found to be negatively linked to waders' conservation as well (Tavares et al., 2015). In our case study, during late spring and summer we noticed that human disturbance from uncontrolled parking and motorcycle activity at the NW-area, the basic nesting habitat for the Kentish plover (Kardakari, 2000), has forced the species to search for other nesting areas, which were not ideal breeding areas. In fact, the species tried to nest in a similar soil habitat at the S. Wetland sub-area, but the nests were destroyed when the area was flooded after a rain event. Information signs and fencing for protecting the NW-area habitat at the north-western side of the wetland, could improve the conditions for Kentish plover, create favorable conditions for other waders to nest, and at the same time increase visitors' awareness.

Another critical issue for the conservation of waterbirds is the delineation of the protected zone (Hellenic Ornithological Society, 2000). The perimetrical ditches (man-made constructions from the 60s) act well as physical boarders for the delineation of the protected wetland area. However, the Tyflomitis artesian springs area is outside the boarders of the protected area zone. Since this area, is at present the only provider of surface freshwater to the wetland (Maneas et al., 2019), our suggestion is that it should be added as a protected wetland area.

Apart from a hot-spot for birds (Kardakari, 2000), the GLw-Natura 2000 area has an important fisheries value (Koutsoubas et al., 2000; Zoulias et al., 2017), and almost 20% of the area is used for cultivations (Maneas et al., 2019). Water related issues are perceived in different ways by the different stakeholders (Maniatakou, 2020), and water management should consider these needs as well. Agriculture is a basic economic activity at the surrounding areas, and tourism is growing fast (Collaborative Land-Sea Integration Platform [COASTAL], 2019).

The development of alternative forms of tourism on site and around it (e.g., eco-tourism) could enhance the bonds of the diverse socio-ecological system and support conservation actions. For instance, in the surroundings, farms which are under organic cultivation support many bird species which could complement the bird-watching activity inside the wetland (Myers et al., 2019). Bird-monitoring projects have been among the most successful at integrating citizens in collecting data (McCaffrey, 2005), leading to successful long-term monitoring projects in Greece (Hellenic Ornithological Society [HOS], 2019c) and worldwide (Chandler et al., 2017). However, these activities could also disturb wildlife (Cardoni et al., 2008; McFadden et al., 2017), and they need to be organized carefully (for example not during Breeding/Nesting period). The example from a similar wetland area in Israel, Collins-Kreiner et al. (2013) has shown that as the number of visitors increased, the number of birds decreased. To that end, buffer zones need to be carefully designed and be appropriate both in terms of social and ecological perspectives (Glover et al., 2011).

The GLw-Natura2000 area has good accessibility, a road taking visitors around and into the lagoon, and the relatively small sized area makes it possible to see many habitats and a large diversity of birds at close distance, without too much effort. Such outdoor activities can increase support for wildlife conservation, and enhance awareness among locals and visitors. Spatial and temporal data from this study could be used as a basis for organizing sustainable bird-watching activities and organized school visits on site. The fact that species richness and abundance is higher from October to April, could prolong the touristic season and attract visitors outside the high touristic season (May–September), adding to the local economy. Income from eco-tourism could both be an income for the local community creating a positive attitude to conservation, and also for funding some of the conservation efforts.

A sustainable management strategy for the GLw area should aim to improve and enlarge waterbirds' habitats, considering nonetheless the existing human activities and social needs at a broader scale (Habitats Directive 92/43/EEC, 1992). Such management requires not only interdisciplinary research, but also engagement of stakeholders at a broader scale (Tsianou et al., 2013). An Ecosystem Services approach could provide the links between nature and people (Díaz et al., 2015), which in turn could reduce conflicts and enhance the benefits for the local economy and ecology, by improving the mutual understanding between the stakeholders. Bringing together farmers, fishermen, tourism operators, Natura 2000 managers and policy makers could lead to improved land-sea interactions and widely accepted management strategies (Collaborative Land-Sea Integration Platform [COASTAL], 2019).

CONCLUSION

The Gialova Lagoon wetland supports many different bird species (149 species including 66 waterbirds and 15 wetland-dependent species), especially during the Wintering and both migration periods. The distribution of species richness and abundance varied significantly in the different sub-areas (formed after past anthropogenic interventions). The S. Wetland sub-area was the area mostly used by waterbirds, followed by the Lagoon and the NW area. Our results indicate that the populations of most IBA species have declined over the last 20 years. The current lack of management makes it difficult to implement efficient and inclusive conservation strategies. The restoration of fresh water inflows, could improve habitats and water conditions for the IUCN and the IBA/SPA species and gradually enhance their conservation status. With careful steps and management decisions the area could become a good example for sustainable management of multifunctional coastal wetlands, favoring both nature conservation and societal well-being. The area cannot meet Ramsar criterion 5 linked to waterbirds, and the decline in abundance of several species may imply that no longer fulfills criterion 6 (and equivalent IBA criterion A4i). However, our results indicate that the area meets criterion 4 and criterion 2, and thus we suggest that it should be further investigated and evaluated to potentially become the eleventh Greek Ramsar site.

DATA AVAILABILITY STATEMENT

The primary dataset (monthly observations of waterbirds) is archived in the open-access database of the Bolin Centre for Climate Research. Available at: https://bolin.su.se/data/maneas-2020.

AUTHOR CONTRIBUTIONS

GM, DB, and HB: conceptualization, design of the study, database, data processing, and methodology. GM, DB, and VN: fieldwork and investigation. HB: supervision. GM: writing—original draft preparation. All authors wrote sections of the manuscript, contributed to manuscript revision, read and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.501548/full#supplementary-material

REFERENCES

Adam, M., Musilová, Z., Musil, P., Zouhar, J., and Romportl, D. (2015). Long-term changes in habitat selection of wintering waterbirds: high importance of cold weather refuge sites. *Acta Ornithol.* 50, 127–138. doi: 10.3161/00016454ao2015.50.2.001

Álvarez-Rogel, J., Jiménez-Cárceles, F. J., Roca, M. J., and Ortiz, R. (2007). Changes in soils and vegetation in a Mediterranean coastal salt marsh impacted by human activities. *Estuar. Coast. Shelf Sci.* 73, 510–526. doi: 10.1016/j.ecss.2007.02.018

Arvanitidis, C., Koutsoubas, D., Dounas, C., and Eleftheriou, A (1999). Annelid fauna of a Mediterranean lagoon (Gialova Lagoon, south-west Greece): community structure in a severely fluctuating environment. *J. Mar. Biol. Assoc. U. K.* 79, 849–856. doi: 10.1017/S0025315499001010

Bibi, F., and Ali, Z. (2013). Measurement of diversity indices of avian communities at Taunsa Barrage Wildlife Sanctuary, Pakistan. *J. Anim. Plant Sci.* 23, 469–474.

BirdLife International (2001). *Important Bird Areas and Potential Ramsar Sites in Europe*. Wageningen: BirdLife International.

BirdLife International (2015). *European Red List of Birds*. Luxembourg: Publications Office of the European Union. Available online at: https://ec.europa.eu/environment/nature/conservation/species/redlist/downloads/European_birds.pdf (accessed September 10, 2019).

BirdLife International (2017). *Mediterranean/Black Sea Flyway*. Available online at: http://datazone.birdlife.org/userfiles/file/sowb/flyways/5_Mediterranean_Black_Sea_Factsheet.pdf (accessed 04 November, 2017).

BirdLife International (2018). *The Top Five Threats to Birds may Surprise You*. Available online at: https://www.birdlife.org/worldwide/news/top-five-threats-birds-may-surprise-you (accessed 22 July, 2019).

BirdLife International (2019). *BirdLife Taxonomic Basis*. Available online at: http://datazone.birdlife.org/species/taxonomy (accessed 27 September, 2019).

BirdLife International (2020). Available online at: http://datazone.birdlife.org/site/ibacriteria (accessed October, 2020).

Birds Directive 2009/147/EC (2009). *The European Union Birds Directive*. Available online at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32009L0147 (accessed November 2, 2020).

Bortels, L., Chan, J. C. W., Merken, R., and Koedam, N. (2011). Long-term monitoring of wetlands along the Western-Greek bird migration route using Landsat and ASTER satellite images: Amvrakikos Gulf (Greece). *J. Nat. Conserv.* 19, 215–223. doi: 10.1016/j.jnc.2011.01.004

Bousbouras, D., Maneas, G., Bilionis, S., and Euaggelopoulos, A. (2011). *Gialova Lagoon Hydrological Study. Determining the Necessary Management Measures for Achieving a Favourable Conservation Status for Birds*, Athens: Hellenic Ornithological Society, 212. (Available only in Greek).

Brandis, K. J., Bino, G., Spencer, J. A., Ramp, D., and Kingsford, R. T. (2018). Decline in colonial waterbird breeding highlights loss of Ramsar wetland function. *Biol. Conserv.* 225, 22–30. doi: 10.1016/J.BIOCON.2018.06.022

Cardoni, D. A., Favero, M., and Isacch, J. P. (2008). Recreational activities affecting the habitat use by birds in Pampa's wetlands, Argentina: implications for waterbird conservation. *Biol. Conserv.* 141, 797–806. doi: 10.1016/j.biocon.2007.12.024

Chandler, M., See, L., Copas, K., Bonde, A. M. Z., López, B. C., Danielsen, F., et al. (2017). Contribution of citizen science towards international biodiversity monitoring. *Biol. Conserv.* 213(Pt B), 280–294. doi: 10.1016/j.biocon.2016.09.004

Collaborative Land-Sea Integration Platform [COASTAL], 2019 (2019). *European Union's H2020 Research and Innovation Programme Under Grant Agreement* No. 773782. Available online at: https://h2020-coastal.eu/ (accessed 03 February, 2019).

Collins-Kreiner, N., Malkinson, D., Labinger, Z., and Shtainvarz, R. (2013). Are birders good for birds? Bird conservation through tourism management in the Hula Valley, Israel. *Tour. Manag.* 38, 31–42. doi: 10.1016/j.tourman.2013.01.009

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., et al. (2015). The IPBES conceptual framework – connecting nature and people. *Curr. Opin. Environ. Sustain.* 14, 1–16. doi: 10.1016/j.cosust.2014.11.002

European Cold Wave (2017). *Wikipedia Results on the European Cold Wave Occurred in 2017*. Available online at: https://en.wikipedia.org/wiki/January_2017_European_cold_wave (accessed 24 November, 2020).

European Commission (2016). *Wild Birds: Threatened Bird Species in Annex I. Environment, European Commission*. Available online at: http://ec.europa.eu/environment/nature/conservation/wildbirds/threatened/index_en.htm (accessed 19 June, 2019).

Fischer, J., and Lindenmayer, D. B. (2002). Small patches can be valuable for biodiversity conservation: two case studies on birds in southeastern Australia. *Biol. Conserv.* 106, 129–136. doi: 10.1016/S0006-3207(01)00241-5

Fric, J., Portolou, D., Manolopoulos, A., and Kastritis, T. (2012). *Important Areas for Seabirds in Greece.* Technical Report No. LIFE07 NAT/GR000285. Athens: Hellenic Ornithological Society (HOS / BirdLife Greece).

Glover, H. K., Weston, M. A., Maguire, G. S., Miller, K. K., and Christie, B. A. (2011). Towards ecologically meaningful and socially acceptable buffers: response distances of shorebirds in Victoria, Australia, to human disturbance. *Landsc. Urban Plan.* 103, 326–334. doi: 10.1016/j.landurbplan.2011.08.006

GR2550004 (1995). *NATURA 2000 – Standard Data Form for Special Protection Areas (SPA), Proposed Sites for Community Importance (pSCI), Sites of Community Importance (SCI) and for Special Areas of Conservation (SAC).* Available online at: https://natura2000.eea.europa.eu/natura2000/SDF.aspx?site=GR2550004 (accessed November 2, 2020).

GR2550008 (2001). *NATURA 2000 – Standard Data Form for Special Protection Areas (SPA), Proposed Sites for Community Importance (pSCI), Sites of*

Community Importance (SCI) and for Special Areas of Conservation (SAC). Available online at: https://natura2000.eea.europa.eu/Natura2000/SDF.aspx?site=GR2550008 (accessed November 22, 2018).

Greek Biotope Wetland Centre (GBWC) (2020). Available online at: http://www.ekby.gr/ekby/en/EKBY_Greek_Wetlands_en.html (accessed November 2, 2020).

Greek Ramsar Sites (2020). Available online at: https://www.ramsar.org/wetland/greece (accessed November 2, 2020).

Gregory, R. D., Gibbons, D. W., and Donald, P. F. (2004). "Bird census and survey techniques," in *Bird Ecology and Conservation: A Handbook of Techniques*, Vol. 6, eds W. J. Sutherland, I. Newton, and R. Green (Oxford: Oxford University Press), 17–55. doi: 10.1093/acprof:oso/9780198520863.001.0001

Habitats Directive 92/43/EEC (1992). *The European Union Habitats Directive*. Available online at: https://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm (accessed September 2, 2019).

Handrinos, G., Kazantzidis, S., Alivizatos, C., Akriotis, T., and Portolou, D. (2015). *International Waterbird Census in Greece (1968 – 2006). Analysis of the Populations of Wildfowl (Anseriformes) and the Coot (Fulica atra)*. Athens: Hellenic Ornithological Society – Hellenic Bird Ringing Centre.

Heath, M. F., Evans, M. I., Hoccom, D. G., Payne, A. J., and Peet, N. B. (eds) (2000). *Important Bird Areas in Europe: Priority Sites for Conservation. Volume 1: Northern Europe, Volume 2: Southern Europe*. Cambridge: BirdLife International.

Hellenic forestry (2020). *Birds' Migratory Routes in Greece*. Available online at: https://dasarxeio.com/2013/11/24/9770/ (accessed March 02, 2020).

Hellenic Ornithological Society (2000). Special Environmental Study of Gialova Lagoon and Sfaktiria Island, Implemented Under by EU Project (B4-3200/97/244): LIFE-Nature Implementation of Management Plans for Pylos Lagoon and Evrotas Delta, Natura 2000 Sites, Greece. Athens: Hellenic Ornithological Society. (Available in Greek).

Hellenic Ornithological Society (HOS) (2019b). *GR119 Divari Pilou Lagoon (Gialova)*. Available online at: http://www.ornithologiki.gr/page_iba.php?aID=119&loc=en (accessed June 16, 2019).

Hellenic Ornithological Society (HOS) (2019c). *Voluntary Projects*. Available online at: http://www.ornithologiki.gr/volunteer.php (accessed June 16, 2019).

Hellenic Ornithological Society (HOS) (2019a). *List of the Important Bird Areas of Greece*. Available online at: http://www.ornithologiki.gr/page_cn.php?tID=1866&aID=898 (accessed June 16, 2019).

Howard, C., Stephens, P. A., Tobias, J. A., Sheard, C., Butchart, S. H. M., and Willis, S. G. (2018). Flight range, fuel load and the impact of climate change on the journeys of migrant birds. *Proc. R. Soc. B Biol. Sci.* 285:20172329. doi: 10.1098/rspb.2017.2329

Hutcheson, K. (1970). A test for comparing diversities based on the Shannon formula. *J. Theor. Biol.* 29, 151–154. doi: 10.1016/0022-5193(70)90124-4

IBA Criteria (2001). *Identifying Important Bird Areas*. Available online at: http://datazone.birdlife.org/userfiles/file/IBAs/EuCntryPDFs/IBA2000Vol1pp011-020_IdentifyingIBAs.pdf (accessed November 23, 2020).

Issa, M. A. A. (2019). Diversity and abundance of wild birds species' in two different habitats at Sharkia Governorate, Egypt. *J. Basic Appl. Zool.* 80:34. doi: 10.1186/s41936-019-0103-5

IUCN (2020). *The IUCN Red List of Threatened Species*. Available online at: https://www.iucnredlist.org/ (accessed November 2, 2020).

Kardakari, N. (2000). *The Birds of Gialova Lagoon: A Report on the Avifauna of the Important Bird Area: Divari Pilou Lagoon (Gialova)*. Athens: Hellenic Ornithological Society. (Available only in Greek).

Koutsoubas, D., Dounas, C., Arvanitidis, C., Kornilios, S., Petihakis, G., Triantafyllou, G., et al. (2000). Macrobenthic community structure and disturbance assessment in Gialova Lagoon, Ionian Sea. *ICES J. Mar. Sci.* 57, 1472–1480. doi: 10.1006/jmsc.2000.0905

Legakis, A., and Maragou, P. (2009). *The Red Book of Endangered Species in Greece (in Greek)*. Athens: Hellenic Zoological Society, 1–528. Available in Greek.

Lehikoinen, A., Jaatinen, K., Vähätalo, A. V., Clausen, P., Crowe, O., Deceuninck, B., et al. (2013). Rapid climate driven shifts in wintering distributions of three common waterbird species. *Glob. Chang. Biol.* 19, 2071–2081. doi: 10.1111/gcb.12200

Liordos, V., Pergantis, F., Perganti, I., and Roussopoulos, Y. (2014). Long-term population trends reveal increasing importance of a Mediterranean wetland complex (Messolonghi lagoons, Greece) for wintering waterbirds. *Zool. Stud.* 53:12. doi: 10.1186/1810-522X-53-12

Maneas, G., Makopoulou, E., Bousbouras, D., Berg, H., and Manzoni, S. (2019). Anthropogenic changes in a Mediterranean coastal wetland during the last century-the case of Gialova Lagoon, Messinia, Greece. *Water* 11:350. doi: 10.3390/w11020350

Maniatakou, S., Berg, H., Maneas, G., and Daw, T. M. (2020). Unravelling diverse values of ecosystem services: a socio-cultural valuation using Q methodology in Messenia, Greece. *Sustainability* 12:10320. doi: 10.3390/su122410320

Manzoni, S., Maneas, G., Scaini, A., Psiloglou, B. E., Destouni, G., and Lyon, S. W. (2020). Understanding coastal wetland conditions and futures by closing their hydrologic balance: the case of Gialova Lagoon, Greece. *Hydrol. Earth Syst. Sci.* 24, 3557–3571. doi: 10.5194/hess-24-3557-2020

Maragou, P., and Mantziou, D. (2000). *Assessment of the Greek Ramsar Wetlands*. Lempesi: WWF-Greece, 59 + Answered questionnaires pp 118.

McArthur, V. E., Koutsoubas, D., Lampadariou, N., and Dounas, C. (2000). The meiofaunal community structure of a Mediterranean lagoon (Gialova lagoon, Ionian Sea). *Helgoland Mar. Res.* 54, 7–17. doi: 10.1007/s101520050031

McCaffrey, R. (2005). Using citizen science in urban bird studies. *Urban Habitat.* 3, 70–86.

McFadden, T. N., Alejandro, G. H., and Navedo, J. G. (2017). Waterbird responses to regular passage of a birdwatching tour boat: implications for wetland management. *J. Nat. Conserv.* 40, 42–48. doi: 10.1016/j.jnc.2017.09.004

Moore, F. R., Woodrey, M. S., Buler, J. J., Woltmann, S., and Simons, T. R. (2005). "Understanding the stopover of migratory birds: a scale dependent approach," in *Bird Conservation Implementation and Integration in the Americas: Proceedings of the 3rd International Partners in Flight Conference: USDA Forest Service Gen. Tech. Rep. No.*, Vol. 2013, eds C. J. Ralph and T. Rich (Asilomar, CA: USDA Forest Service), 684–689.

Myers, D., Berg, H., and Maneas, G. (2019). Comparing the soundscapes of organic and conventional olive groves: a potential method for bird diversity monitoring. *Ecol. Indic.* 103, 642–649. doi: 10.1016/j.ecolind.2019.04.030

Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A. C., Colijn, F., et al. (2014). An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. *Estuar. Coast. Shelf Sci.* 140, 95–122. doi: 10.1016/j.ecss.2013.05.023

O'Hara, R. B., and Kotze, D. J. (2010). Do not log-transform count data. *Method. Ecol. Evol.* 1, 118–122. doi: 10.1111/j.2041-210x.2010.00021.x

Ramírez, F., Rodríguez, C., Seoane, J., Figuerola, J., and Bustamante, J. (2018). How will climate change affect endangered Mediterranean waterbirds? *PLoS One* 13:e0192702. doi: 10.1371/journal.pone.0192702

Ramsar (2017). *Ramsar Sites Information Service*. Available online at: https://rsis.ramsar.org (accessed November 16, 2017).

Ramsar convention (1994). *Convention on Wetlands of International Importance Especially as Waterfowl Habitat 1971*. Available online at: http://portal.unesco.org/en/ev.php-URL_ID=15398&URL_DO=DO_TOPIC&URL_SECTION=201.html (accessed June 19, 2019).

Ramsar Sites Criteria (2020). *The Ramsar Sites Criteria: The Nine Criteria for Identifying Wetlands of International Importance*. Available online at: https://www.ramsar.org/sites/default/files/documents/library/ramsarsites_criteria_eng.pdf (accessed November 2, 2020).

Ramsar Sites Information Service (RSIS) (2020). Available online at: https://rsis.ramsar.org/?pagetab=2 (accessed November 2, 2020).

Seavy, N. E., Quader, S., Alexander, J. D., and Ralph, C. J. (2005). "Generalized linear models and point count data: statistical considerations for the design and analysis of monitoring studies," in *Proceedings of the 3rd International Partners in Flight Conference. 2002 March 20-24; Asilomar, California: Bird Conservation Implementation and Integration in the Americas. USDA For. Ser. Gen. Tech. Rep. PSW-GTR-191*, Vol. 2, eds C. J. Ralph and T. D. Rich (Albany, CA: United States Department of Agriculture), 744–753.

Smith, L. A., and Chow-Fraser, P. (2010). Implications of the species-area relationship on sampling effort for marsh birds in southern Ontario. *Wetlands* 30, 553–563. doi: 10.1007/s13157-010-0048-4

Stanley, C., Simmons, K. E. L., and Perrins, C. M. (1977–1994). *Handbook of the Birds of Europe the Middle East and North Africa, The Birds of the Western Palearctic*, Vols. I–IX. Oxford: Oxford University Press.

Svensson, L., Mullarney, K., and Zetterström, D. (2010). *Collins Bird Guide*, 2nd Edn. London: HarperCollins Publishers Ltd.

Tavares, D. C., Guadagnin, D. L., de Moura, J. F., Siciliano, S., and Merico, A. (2015). Environmental and anthropogenic factors structuring waterbird habitats of tropical coastal lagoons: implications for management. *Biol. Conserv.* 186: 12–21. doi: 10.1016/j.biocon.2015.02.027

Tsianou, M. A., Mazaris, A. D., Kallimanis, A. S., Deligioridi, P. S. K., Apostolopoulou, E., and Pantis, J. D. (2013). identifying the criteria underlying the political decision for the prioritization of the Greek Natura 2000 conservation network. *Biol. Conserv.* 166, 103–110. doi: 10.1016/j.biocon.2013.06.021

Wang, Z. Q., Wang, F., and Baiyan, W. (2011). Changes of the wetland landscape and the consequent impacts on the waterbirds in Western Songnen Plain. *Procedia Environ. Sci.* 10(Pt C), 2608–2613. doi: 10.1016/j.proenv.2011.09.405

Warnock, N. (2010). Stopping vs. Staging: the difference between a hop and a jump. *J. Avian Biol.* 41, 621–626. doi: 10.1111/j.1600-048X.2010.05155.x

Warton, D. I, Lyons, M., Stoklosa, J., and Ives, A. R. (2016). Three points to consider when choosing a LM or GLM test for count data. *Method. Ecol. Evol.* 7, 882–890. doi: 10.1111/2041-210X.12552

Wetlands International (2020). *Waterbird Population Estimates*. Available online at: wpe.wetlands.org (accessed November 08, 2020).

Zoulias, T., Papadopoulos, A., and Conides, A. (2017). An ecological evaluation using fisheries landings time-series data: a case study of two coastal lagoons in the Ionian Sea (E. Mediterranean, Greece). *Estuar. Coast. Shelf Sci.* 216, 229–239. doi: 10.1016/j.ecss.2017.09.011

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Assessing Ecosystem Services in Mangroves: Insights from São Tomé Island (Central Africa)

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Mangroves are some of the most productive coastal systems on the planet and provide valuable ecosystem services (ES). They are especially important in threatened ecosystems and developing countries, where they are likely to have direct impacts on local communities. An approach based on ES allows assessing ecosystems across the domains of ecology, sociology and economy. This study focused on the evaluation of ES in mangroves and started by creating a comprehensive global list of mangrove ES based on the Millennium Ecosystem Assessment. These services were then quantified using the best available indicators for mangrove systems. The mangroves of Diogo Nunes, São João dos Angolares and Malanza, located in the São Tomé Island, were used to illustrate the challenges in applying ES indicators in this type of ecosystems. The obtained results confirmed that mangroves can provide important and diverse services. However, the high variability among mangrove systems affects their ability to deliver ES, requiring caution for the extrapolation across regions. This assessment emphasizes how the ES framework can be used as a tool to develop management plans that integrate conservation goals and human wellbeing.

Keywords: environmental conservation, ecosystem services indicator, quantification of ecosystem services, transitional systems, gulf of guinea

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INTRODUCTION

The concept of ecosystem services (ES) appeared in the 1960s, intending to link ecological and economic research (Martin-Ortega et al., 2015). Since then, the concept of ES has been greatly expanded and in the Millennium Ecosystem Assessment (MEA) it was defined as 'the functions and products of ecosystems that benefit humans, or yield welfare to society' (MEA, 2005). The definition of ES remains elusive, often varying according to a stakeholder or specific context (Fisher et al., 2009), even though understanding the connection between human society and ecosystems is crucial to integrate the domains of ecology, economy and sociology (MEA, 2005). Later in 2009, the Common International Classification of Ecosystem Services (CICES) emerged and defined ES as the 'contributions that ecosystems make to human well-being' (Haines-Young and Potschin, 2018), adapting the MEA methodology to obtain a more detailed hierarchical method to classify ES (Czúcz et al., 2018). More recently, the Intergovernmental Platform on Biodiversity and Ecosystem Services

(IPBES) system was proposed. It differs substantially from previous ES assessment systems for being anchored on the concept of Nature's Contributions to People, defining ES as 'all the positive contributions, losses or detriments, that people obtain from nature' (Brauman et al., 2019).

The assessment of ES involves identification, mapping and quantification, the latter of which can be measured in three domains: biophysical, social, and economic (Haines-Young et al., 2018). In this study, only the identification and quantification of ES were considered. Together, these two steps provide stakeholders with tools to raise awareness and to manage the landscape effectively (Vihervaara et al., 2017). Mapping ES involves methodologies from all the domains previously mentioned and provides a spatial representation of the capacity of a system to deliver ES (Vihervaara et al., 2018). During the last 10 years, major changes and advances have been made in ES mapping (Englund et al., 2017), and several countries have been incorporating ES assessment methodologies in decision-making. In the European Union efforts are being made to develop integrated methodologies for ES mapping, valuation, accounting and assessment (e.g. ESMERALDA Project; www.esmeralda-project.eu) and to promote the use of ES in decision making (Burkhard et al., 2018) and implementation of the EU Biodiversity Directive. Nonetheless, many ES are difficult to identify, especially in under-studied systems, and quantification relies on indicators, which are often non-existent, inadequate or hard to measure (Müller and Burkhard, 2012). The economic valuation provides a monetary justification for the allocation of financial resources toward ecosystem preservation (Gómez-Baggethun et al., 2010) and it is based on the measure of the economic value of ES (Brander et al., 2018). However, they require strong safeguards since many ES hard to convert into a marketable value are undervalued (Castro et al., 2014).

The MEA has become the classical system of ES classification, recognizing four categories: 1) provisioning (e.g. food, fiber, and other resources); 2) regulating (e.g. climate regulation, protection against soil erosion, flood protection, water purification), 3) cultural (e.g. recreation, spiritual values, aesthetics, education and research); and 4) supporting (e.g. habitat diversity and nutrient cycling). The relevance of each of these categories is strongly context-dependent. For instance, in developing countries provisioning services have a more direct association with poverty alleviation and food security, and their impact is often felt almost instantly by human populations (MEA, 2005). The supporting, regulating and cultural services tend to be overlooked since their impacts on human well-being are less direct and they are harder to measure (Alcamo et al., 2003b). However, this does not mean that they are less relevant to human wellbeing (TEEB, 2010).

The *tragedy of the commons* is often evident in marine fisheries of developing countries, due to the difficulties in determining and enforcing property rights, while populations are often over-reliant on fisheries that depend on ecosystem integrity (Alcamo et al., 2003a; Ostrom and Ostrom, 2015). Our unawareness of ecosystem functioning and ES delivery also undermines our ability to manage resources (Alcamo et al., 2003b), making it crucial to find objective means of quantification. Furthermore, in many developing countries, the voices of impoverished local communities and conservation interests are ignored by political and economic interests, contributing to an undervaluation of ES (Samarakoon, 2004). Sustainable ecosystem management is key to preserve the long-term delivery of ES, but requires practices that promote ecological functioning (Agbenyega et al., 2009).

Mangrove forests are considered some of the most productive systems on Earth (Walters et al., 2008) and provide important ES, often related to the daily activities of rural communities (Spalding et al., 2010). These intertidal forests are known for their capacity to provide coastal protection against natural hazards, such as storm waves, and erosion (Badola and Hussain, 2005). Wetland areas, like mangroves, are also known to store carbon, which is an appealing contribution to climate change mitigation (Donato et al., 2011). They are very important nursery areas for a large variety of fish and invertebrates, providing refuge and food for many of these species during the first development stages (Mumby et al., 2004). However, they are frequently under strong anthropogenic pressure (Spalding et al., 2010) and are among the most threatened marine ecosystems (Duke et al., 2007). Over the last 20 years, 35% of the global area of mangrove forests was lost (Valiela et al., 2006). One of the most common drivers of biodiversity loss is habitat transformation at the expense of land conversion to agriculture, although this is a type of ES trade-off less documented in coastal ecosystems (MEA, 2005). The failure to implement adequate policies and the persistence of ill-defined property rights are some of the underlying causes of this loss (Sathirithai, 1998), making it essential to recognize the value of these ecosystems whilst developing efficient evidence-based conservation strategies. Also, there is a demand to understand the path of distribution of services costs and benefits, moreover to perceive the impact of trade-offs between ES to avoid corruption (MEA, 2005). The Ecosystem Service Framework (ESF) is a benefit-oriented approach and a valuable tool to engage managers and regulators, since it focuses on social and economic benefits, setting the basis for policy changes (Alcamo et al., 2003b). Recent studies have proposed methodologies to assess mangroves ES, namely through mapping (e.g. Kuenzer and Tuan, 2013), economic valuation (e.g. Barbier et al., 2011) and applied social evaluations (e.g. Satyanarayana et al., 2012).

This study aims to highlight the ecological and socio-economic importance of mangroves by evaluating the ES they deliver at a global, regional and local scale, using São Tomé Island (Central Africa) as a case study. On a global scale, mangrove ES will be identified and compared to other estuarine and terrestrial ecosystems. At the regional scale, mangrove ES in Africa will be quantified using previously identified adequate indicators and values available in the literature, and again comparing with regional estuarine and terrestrial ecosystems. Finally, São Tomé mangrove ES will be quantified using indicators based on field assessments or expert-based knowledge. A comparison across scales will showcase the challenges of ES assessment, which are known to be highly variable across regions and at multiple geographical scales. Even though the main focus of the study will be the regional and local assessments, it is essential to develop a global list of ES provided by mangroves. This is the first attempt to use the ESF in São Tomé mangroves and provides important clues to promote biodiversity conservation and the sustainable use of resources.

METHODOLOGY

Mangrove ES were identified (*Identification*) at a global scale, and quantified (*Quantification*) at a regional scale for tropical Africa, based on an extensive literature review. Subsequently, three São Tomé mangroves were chosen to evaluate ES, using the best available information (*Assessment of Ecosystem Services in São Tomé Mangroves*).

Assessment of Mangrove Ecosystem Services

Identification

An existing general list of ES (Layke et al., 2012) was adapted, focusing on mangroves ES. The list was revised to include all ES that were found in the literature, by performing a search for the keywords "ecosystem services" and "mangroves", on Google Scholar and Web of Knowledge, between January and June of 2018.

Mangroves can be classified as terrestrial, aquatic, or both (Friess et al., 2016). Therefore, the relative importance of mangrove ES was assessed by identifying and comparing ES delivery in terrestrial, estuarine (excluding mangroves) and mangrove ecosystems. To do so, the keywords "terrestrial" or "estuarine" and "ecosystem services" were used. This study followed the MEA ES classification scheme (MEA, 2005). Although other classification methodologies were considered, such as CICES and IPBES, the final decision was in favor of MEA, due to its well established and recognized methods (Caputo et al., 2019) and the thorough list of specific indicators provided by MEA assessments.

Quantification

Quantification requires the use of ES indicators. Several general lists of ES indicators have been published, even though there are no operational practices or guidelines to develop or select ES indicators (Broszeit et al., 2017). This study was based on an existing list of ES indicators (Layke et al., 2012), which was improved by adding and replacing indicators following information found in the literature (**Table 1**). Indicators were selected based on a confidence level assessment, using a scoring system based on two elements: 1) the ability to convey information: intuitive; sensitive; accepted; and 2) data availability: gathered at sufficient temporal and spatial scales; processed and available; normalized and disaggregated. Each element had three underlying criteria, classified from one (low) to three (high) and the value of each element was obtained as the arithmetic mean of the criteria scores (see Layke et al. (2012) for further details). The indicator with the highest score was selected (i.e. when the sum of each element value resulted in a low or medium score, the decision fell on the selection of another indicator to replace it).

Indicators can be measured directly, for instance when a state or process is quantified during field observations, or indirectly, for instance when based on proxy indicators, expert-based knowledge, or when the data requires interpretation or adjustments (Vihervaara et al., 2017). Most indicators were selected based on data availability. The most common adjustments were the addition of a temporal dimension to express ES flow (Vihervaara et al., 2017) and the conversion to International System units. Some cases required special adjustments, such as ES *biomass fuel*. This ES is most commonly assessed based on the consumption of fuel *per* capita but an estimate of fuel consumption of the overall population in the vicinity of the study area was used in this study, to provide a value representative of the mangrove system.

Several scientific research papers and reports were consulted to quantify each indicator at the regional level (Tropical Africa), separately for each of the three ecosystems considered (mangroves, estuaries and land). The search was performed on the web-search engines previously mentioned (*Identification*), it was established that the limit was the first 30 publications of the results, since it was intended to obtain as much information as possible without reaching the point of data repetition and that all the information acquired was specific to the selected indicators and study region.

Assessment of Ecosystem Services in São Tomé Mangroves

The assessment approach developed in *Assessment of Mangrove Ecosystem Services*, was then applied in the context of São Tomé Island. Firstly, each mangrove was mapped, using GPS locations and satellite images (Google Images, 2018. São Tomé. Digital Globe) on QuantumGIS (QGIS 2.18.13). The satellite image analysis was essential to identify areas with mangrove trees and watercourses designated as the "mangrove area". Then, to better comprehend the surrounding areas of the study site as well as to characterize the type of stakeholders present, the main land-use types (Burkhard et al., 2009) around each mangrove were mapped. This step is essential for well-developed decision-making. Since there was no standardized value for the definition of buffer area, we opted for the lower value found in the literature, 100 m, because of the small scale of the case study (Macintosh and Ashton, 2002; Atkinson et al., 2016).

Then, the improved global list of ES for mangrove systems (**Table 2**) was used to identify ES in São Tomé Island, using site-specific literature, complemented by expert-knowledge and a field assessment conducted in August 2017. The ES quantification was considered only for services with suitable data, which were *wild foods*, *water regulation,* and *nursery area* services. The indicators used were obtained from *Quantification* and the estimates were preferably based on field assessments. The *wild foods* service quantification was based exclusively on literature available for the study area (Pisoni et al., 2015), where it was possible to quantify the number of species used as food source. While *water regulation* and *nursery area* services were calculated based on the field assessments. The first was calculated by measuring the concentration of nitrogen in the water, while the second was calculated using different fishing techniques to quantify the proportion of juveniles in the local populations. The assessment only considered the mangrove area defined in the mapping.

The ES assessment took place in the mangroves of Diogo Nunes, Angolares and Malanza (**Figure 1**), in São Tomé Island (0°25′N - 0°01′S, 6°28′E - 6°45′E). These systems were chosen to represent the diversity of mangroves on the island, considering spatial distribution, mangrove size and anthropogenic pressure.

TABLE 1 | Indicators for ecosystem services quantification based on the reference article (Layke et al., 2012). Data availability (none *, little **, plenty ***) and necessary modifications. Services with no indicators provided in the reference article are represented as × and services absent in the reference article are shaded in gray.

	Ecosystem Services	Indicators based on reference article	Data availability	Selected indicator
Provisioning	Capture fisheries	Value of marine production	*	Yearly rate of seafood extraction (kg km^{-2} year^{-1}; Hattam *et al.*, 2013)
	Crops cultivation	Crops production	*	Land prices near the ecosystem (US$ km^{-2}; Liquete *et al.*, 2016)
	Aquaculture	Aquaculture production	***	Yearly aquaculture production (kg year^{-1})
	Wild foods	Number of wild species used as food	***	-
	Timber	Forest biomass production	***	Yearly forest biomass production (kg km^{-2} year^{-1})
	Fibers and ornamental resources	Value of forest products	***	Yearly value of forest products (US$ km^{-2} year^{-1})
	Biomass fuel	Fuelwood consumption	***	Yearly consumption of fuelwood (kg km^{-2} year^{-1})
	Genetic resources	Value of genetic resources	**	Yearly value of genetic resources (US$ km^{-2} year^{-1})
	Medicines and pharmaceuticals	Value of pharmaceutical products developed in natural systems	**	Yearly value of medical resources (US$ km^{-2} year^{-1}; de Groot *et al.*, 2012)
	Water for non-drinking purposes			Yearly freshwater runoff (m^3 year^{-1}; Egoh *et al.*, 2012)
Regulating	Air quality regulation	Flux of atmospheric gases	***	Yearly flux of atmospheric gases (g km^{-2} year^{-1})
	Global climate regulation	Capacity of Carbon sequestration	***	Yearly rate of carbon sequestration (kg km^{-2} year^{-1})
	Regional climate regulation	Evapotranspiration	***	Evapotranspiration rate (cm day^{-1})
	Water regulation	Soil water infiltration	*	Nitrogen concentration (mg N L^{-1}; EEA, 2018)
	Coastal Erosion regulation	Landslide frequency	*	Percentage of wave attenuated (for 100 m of ecosystem length; Atkinson *et al.*, 2016)
	Groundwater recharge			Groundwater recharge rate (mm km^{-2} year^{-1}; Burkhard *et al.*, 2009)
	Wastewater treatment	Amount of waste processed by ecosystems	*	Adaptation of the indicator **Nutrient retention** (Egoh *et al.*, 2012) to **Percentage of nutrients absorbed during wastewater discharge** (%)
	Disease regulation	Disease vectors predator populations	**	-
	Soil quality regulation	×		Index of soil quality (BISQ; Feld *et al.*, 2010)
	Pests regulation	×		Presence/absence/frequency of pests (Hattam *et al.*, 2013)
	Pollination	×		Costs of bees (US$ pollination period^{-1}; Egoh *et al.*, 2012)
	Natural hazards regulation	Mortality losses from natural disasters	***	Mortality losses during natural hazard with or without mangroves
	Nutrient cycle	Value of nutrient cycle for terrestrial ecosystems	*	Adaptation of the indicator **Turnover rate** (Burkhard *et al.*, 2009) to **Yearly rate of nitrogen storage** (N; kg km^{-2} year^{-1})
Cultural	Aesthetic/ethical values	Number of nature/rural visitors	**	Yearly number of visitors for sightseeing (visitors year^{-1})
	Recreation and ecotourism	Visitors to natural areas	***	Yearly number of visitors for recreation (visitors year^{-1})
	Spiritual and religious values	×		-
	Cultural heritage	×		Number of households which consider an area or aspects of an area as cultural heritage (Böhnke-Henrichs et al., 2013)
	Scientific/education	×		-
Supporting	Primary production	NPP	***	-
	Nutrient flow			Yearly nitrogen flow (N; kg km^{-2} year^{-1} - Burkhard *et al.*, 2009)
	Water cycling	×		Transpiration by total evapotranspiration (Burkhard *et al.*, 2009)
	Habitat heterogeneity			Habitat diversity index (Burkhard *et al.*, 2009)
	Nursery area			Number of species with juveniles by the total amount of species (Vasconcelos *et al.*, 2011)

TABLE 2 | Ecosystem services identified globally in mangrove, estuary and terrestrial systems. Black circles and white circles represent the presence and absence, respectively, of each ecosystem service (adapted from Layke et al., 2012 to mangrove ecosystems).

	Ecosystem Services	Mangrove	Estuary	Terrestrial
Provisioning	Capture fisheries	●	●	○
	Crops cultivation	●	●	●
	Aquaculture	●	●	●
	Wild foods	●	●	●
	Timber	●	●	●
	Fibers and ornamental resources	●	○	●
	Biomass fuel	●	○	●
	Genetic resources	●	●	●
	Medicines and pharmaceuticals	●	●	●
	Water for non-drinking purposes	●	●	○
Regulating	Air quality regulation	●	●	●
	Global climate regulation	●	●	●
	Regional climate regulation	●	●	●
	Water regulation	●	●	○
	Coastal erosion regulation	●	●	○
	Groundwater recharge	●	●	●
	Waste treatment	●	●	○
	Diseases regulation	●	●	●
	Soil quality regulation	●	●	●
	Pests regulation	●	●	●
	Pollination	●	●	●
	Natural hazards regulation	●	●	●
	Nutrient cycle	●	●	●
Cultural	Aesthetic/ethical values	●	●	●
	Recreation and ecotourism	●	●	●
	Spiritual and religious values	●	●	●
	Cultural heritage	●	●	●
	Scientific/education	●	●	●
Supporting	Primary production	●	●	●
	Nutrient flow	●	●	●
	Water cycling	●	●	●
	Habitat heterogeneity	●	●	●
	Nursery area	●	●	○

Black mangrove *Avicennia germinans* is present in all study mangroves, and true mangroves *Rhizophora* sp. are only absent from Diogo Nunes. The Diogo Nunes mangrove, on the northeast coast, is the smallest study system (0.01 km^2, Afonso, 2019). It is an intertidal mangrove system with low vegetation coverage, surrounded by agricultural fields (47.5% of the study site, **Figure 2A**; Afonso, 2019) and a community of 392 people (INE São Tomé e Príncipe, 2014). Located on the east coast, the Angolares mangrove has 0.13 km^2 (**Figure 2B**; Afonso, 2019). It is formed by two branches that are only connected to the sea during periods of high runoff or spring tides. The vicinities are occupied mostly by agroforests (59.4%) (Afonso, 2019). The nearest community, São João dos Angolares, has 2037 inhabitants (INE São Tomé e Príncipe, 2014). The Angolares and Diogo Nunes watersheds have both been seriously modified by human activities, and are mostly covered by agroforest (73% and 70%, respectively - based on Soares, 2016). Malanza, on the southern coast of the island, is the largest mangrove in the country (Brito et al., 2017), covering 1.52 km^2 (Afonso, 2019). It is dominated by mangrove and agroforests (53.6% and 36.7%, respectively, **Figure 2C**; Afonso, 2019). This is an open system, but its connection to the sea is heavily constricted by a bridge, which affects water, sediment and ecological dynamics (Félix et al., 2017). This bridge connects the two nearby communities of Porto Alegre

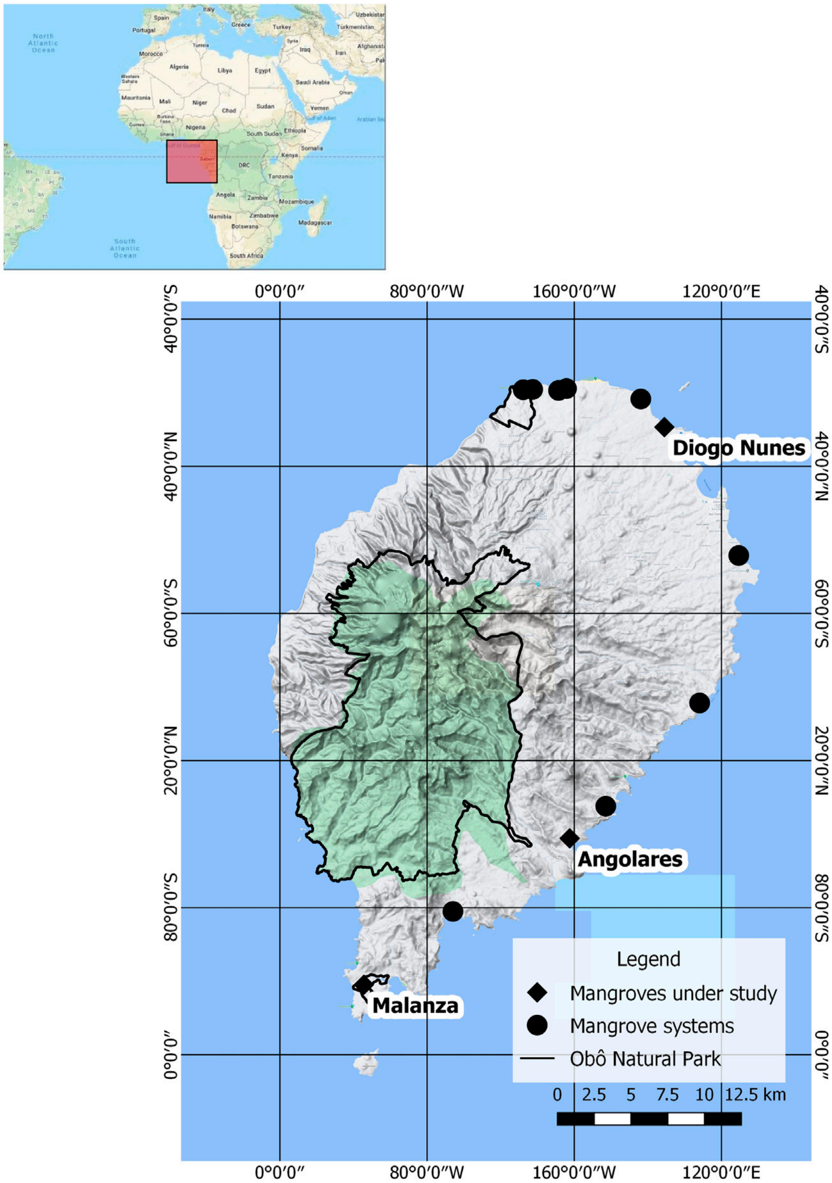

FIGURE 1 | São Tomé island and mangrove systems identified in the island.

and Vila Malanza, which have respectively 795 and 550 inhabitants (INE São Tomé e Príncipe, 2014).

RESULTS

Global Identification of Mangrove Ecosystem Services

A total of 33 ES were identified in mangroves globally (**Figure 3**). Most of these were regulating (13) or provisioning services (10), while cultural and supporting services were less represented (5 each). The original ES list (Layke et al., 2012) was extended to include water for non-drinking purposes, groundwater recharge, nutrient flow, habitat heterogeneity, and nursery area. Some ES, such as livestock, freshwater, and soil formation were excluded since they were not indicated for mangroves. Livestock and freshwater services were never mentioned for mangroves in the literature, while the role of mangroves for soil formation remains a topic of debate (Lee et al., 2014). *Nutrient cycling* is sometimes considered a supporting service (Burkhard et al., 2009), here it was classified as a regulating ES, while *nutrient flow* was classified as a supporting ES.

Only 31 and 27 of the 33 mangrove ES listed were assigned to other estuarine systems and terrestrial ecosystems, respectively

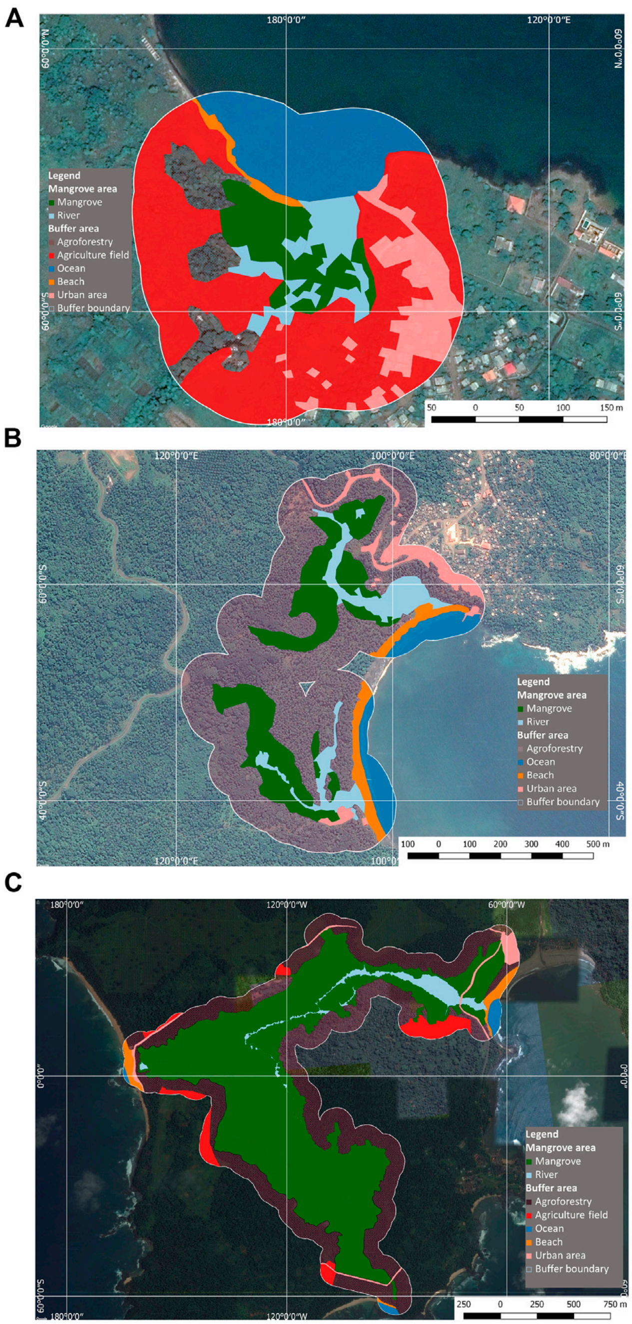

FIGURE 2 | Map of studied mangroves and 100m buffers: **(A)** Diogo Nunes, **(B)** Angolares, and **(C)** Malanza (image sources: Google Images, 2018; Digital Globe).

(**Table 2**). No ES was exclusive to mangroves, even though none of the systems used for comparison delivered as many ES as mangroves. Most differences between mangroves and estuaries were related to provisioning services since estuaries do not provide forest products, such as *fibers and ornamental resources* and *biomass fuel.* Regulating services were less represented in terrestrial systems since many of these are associated with water (**Table 2**).

Regional Quantification of Mangrove Ecosystem Services

Mangrove ES were quantified using existing indicators (**Table 1**) and data from Africa, obtained between 1964 and 2019 (**Table 3, Figure 4**). A thorough literature review provided values for most indicators (~43%), especially those relating to provisioning and supporting services. Regarding provisioning services indicators, only the one associated with *wild foods* was used with minor adaptations, the rest had to be adjusted to include a temporal/spatial scale. All indicators of regulating services were adjusted, except for those that were not used due to a low confidence level. Only two indicators were found for cultural services, of which only *recreation and ecotourism* was quantified. Concerning supporting services, the *primary production* indicator was used without modifications, but all others were adapted. Only 52% of the mangrove ES were quantified (excluding indicators with low or medium classifications) (**Table 3**), including 60% of both provisioning and supporting services (**Figures 4A,C**), 31% of regulating services (**Figure 4B**), and 20% of cultural services (**Figure 4C**).

The quantification of ES varied between ecosystems (**Table 3**). Seven ES indicators presented the most benefits in mangroves, namely capture fisheries, global climate regulation, regional climate regulation, water regulation, groundwater recharge, habitat heterogeneity and nursery area (**Table 3**). In this study, most benefits do not necessarily correspond to the quantification of the highest value, as it depends on the indicator being used. For instance, in this study, it corresponds to a high value in *aquaculture* but to a low value in *air quality regulation*. Only terrestrial systems presented more benefits than mangroves in some ES, more specifically four, that includes three provisioning services and one cultural service.

São Tomé Mangroves Ecosystem Services

Identification

Five provisioning, four regulating, two cultural and one supporting service were identified in São Tomé mangroves based on a literature review (**Table 4**). This survey was complemented by fieldwork assessments, which provided information to identify 15 additional ES. Literature information was less representative for regulating and supporting services and all ES listed for São Tomé in the literature were identified during field assessments.

Quantification

Of the 27 ES identified in São Tomé mangroves, only wild foods, water regulation, and nursery area were quantified locally (**Table 5**). Wild foods, which consisted mostly of seafood, had higher values than those found in the literature, while water regulation had lower values. The quantification of the supporting service *nursery area* was slightly lower in São Tomé that in other mangroves in tropical Africa.

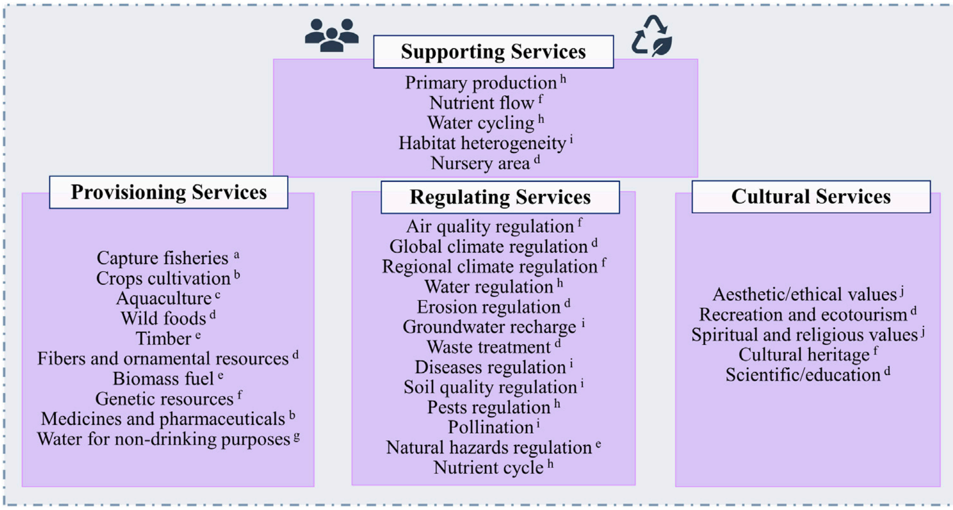

FIGURE 3 | Ecosystem Services identified in mangrove ecosystems from around the world. and represent the relationship between all ES and human well-being and biodiversity, respectively. Sources: [a]Conchedda et al., 2011, [b]Bandaranayake, 1998, [c]Palacios and Cantera, 2017, [d]Barbier et al., 2011; [e]Barbier, 2007; [f]Samoilys, et al., 2015; [g]Grizzetti et al., 2016; [h]Zedler and Kercher, 2005; [i]Macintosh and Ashton, 2002; [j]Atkinson et al., 2016.

DISCUSSION

The current study emphasizes the importance of mangroves as providers of a high variety of ES to local populations, particularly important in developing countries. ES provided by the mangrove biome were identified, quantified for tropical Africa and, for the first time, assessed for São Tomé Island. This exercise highlighted the challenges in obtaining local data and reliable indicators to quantify mangrove ES in developing countries. This study also provides an example of the relevance of ES approaches to support the implementation of conservation policies.

Mangrove as a Source of Ecosystem Services

Previous studies identifying global mangrove ES listed only 17 ES (Barbier et al., 2011; Vo et al., 2012; Drakou et al., 2017), while this study identified 33. This difference is most likely due to the fact that previous studies focused on selected ES, for instance, provisioning and regulating services (Liquete et al., 2013), while the current study assessed all mangroves ES, providing a comprehensive list of services and indicators.

The differences between the number of ES identified in the literature 12) and those identified, in the present study, for São Tomé (an additional 15) seem to be mainly due to two related factors: 1) spatial scale and 2) information availability. All mangroves in São Tomé are extremely small, with an area ranging from 0.01 km^2 (Diogo Nunes) to 2 km^2 (Malanza). The effect of scale in mangrove ES delivery is related to the minimum area required for the development of particular activities or ecological processes. For instance, ES delivered as *aquaculture* are highly dependent on the available production area. Although the regular pond size for shrimp production in Ecuador can be as big as 0.5 km^2 (Hamilton, 2011), most mangroves in São Tomé are too small to support economically sustainable aquaculture (Martín-López et al., 2019). On the other hand, the resolution and extent of the ES assessments may also affect the obtained results and ES estimates differ substantially between the fine and coarse resolution analyses (Grêt-Regamey et al., 2014). Our results suggest that finer resolution assessments conducted at the community level (i.e. mangrove specific information) capture ES spatially explicit information that would be lost at a coarser resolution. That is also related with the second factor concerning the difficulty in assessing some ES without in-depth analyses (*Challenges in Assessing Ecosystem Services*). Very few studies focused on ES identification in mangroves for the tropical African region (e.g. Owuor et al., 2019b). The additional ES identified for STP mangroves highlighted the importance of conducting local surveys to create comprehensive ES inventories (Afonso, 2019).

Mangroves across the globe provide a diverse set of ES. Differences were found in terms of the services provided by estuarine and terrestrial ecosystems, with the lowest number of ES identified in terrestrial systems, which was expected, as other aquatic environments share more similarities with mangrove ecosystems. When comparing the results of ES quantification, mangroves presented more benefits in seven out of the 14 quantified ES. This may be represented by the highest (e.g. net primary production indicator) or the lowest (e.g. nitrogen concentration on water indicator) quantities of a certain service suggesting that mangroves could have an overall positive impact on human societies, namely when compared to estuarine and terrestrial ecosystems. Regarding São Tomé mangroves, two out of three ES quantified at this local scale presented more benefits than those identified for tropical Africa, namely *wild foods,* and *water regulation*, underpinning the relative importance of small mangroves (Curnick et al., 2019).

The potential and the effective delivery of mangrove ES (ES flow) is another relevant question to be considered since it can be

TABLE 3 | Ecosystem Services identified and quantified (mean values and range) in mangrove, estuary and land systems from Tropical Africa (adapted from Layke et al., 2012). The number of estimates used to calculate the mean is indicated in parentheses.

	Ecosystem services	Indicators	Mangrove		Ecosystem of reference			
					Estuary		Terrestrial	
			Mean value	Range	Mean value	Range	Mean value	Range
Provisioning	Capture fisheries	Yearly rate of seafood extraction (kg km^{-2} year^{-1})	4.19×10^{5} (6) [1]	Min: 352.53 Max: 1.09×10^{6}	4.27×10^{3} (5) [2]	Min: 300 Max: 1.02×10^{4}		
	Crops cultivation	Land prices						
	Aquaculture	Yearly aquaculture production (kg year^{-1})	2.00×10^{7} (1) [3]	-	8.38×10^{5} (1) [4]	-	2.09×10^{7} (3) [5]	Min: 2.32×10^{6} Max: 5.74×10^{7}
	Wild foods	Number of wild species used as food	13.5* (2) [6]	Min: 10 Max: 17			122 (3) [7]	Min: 35 Max: 272
	Timber	Yearly forest biomass production (kg km^{-2} year^{-1})	2.49×10^{7} (1) [8]	-				
	Fibers and ornamental resources	Yearly value of forest products (US$ km^{-2} year^{-1})					294.88 (6) [9]	Min: 4.17×10^{-3} Max: 1.06×10^{3}
	Biomass fuel	Yearly consumption of fuelwood (kg km^{-2} year^{-1})	1.67 (1) [10]	-			3.82×10^{4} (5) [11]	Min: 9.11 Max: 1.14×10^{5}
	Genetic resources	Yearly value of genetic resources						
	Medicines and pharmaceuticals	Yearly value of medical resources						
	Water for non-drinking proposes	Yearly freshwater runoff (m^{3} year^{-1})	4.7×10^{9} (1) [12]	-				
Regulating	Air quality regulation	Yearly flux in atmospheric CH_4 (g km^{-2} year^{-1})					2.00×10^{-3} (1) [13]	-
	Global climate regulation	Yearly rate of carbon sequestration (kg km^{-2} year^{-1})	5.36×10^{7} (3) [14]	Min: 79.3 Max: 1.6×10^{8}	4.8×10^{5} (1) [15]	-	5.52×10^{4} (2) [16]	Min: 2.80×10^{4} Max: 8.24×10^{4}
	Regional climate regulation	Evapotranspiration rate (cm day^{-1})	0.59 (2) [17]	Min: 0.47 Max: 0.70	0.1 (1) [18]	-	0.23 (3) [19]	Min: 0.19 Max: 0.28
	Water regulation	Nitrogen concentration (mg L^{-1})	0.3 (1) [20]	-	0.39 (1) [21]	-		
	Coastal erosion regulation	Percentage of wave attenuated						
	Groundwater recharge	Groundwater recharge rate (mm km^{-2} year^{-1})	429.57 (1) [22]	-			74.17 (3) [23]	Min: 0.25 Max: 2.03×10^{3}
	Wastewater treatment	Percentage of nutrients absorbed during wastewater discharge						
	Disease regulation	Disease vectors predators						
	Soil quality regulation	Index of soil quality						
	Pest regulation	Presence of pests						
	Pollination	Cost of bees						
	Natural hazards regulation	Mortality losses during cyclone						
	Nutrient cycle	Yearly rate of nitrogen storage (kg km^{-2} year^{-1})					1.69×10^{5} (1) [24]	-
Cultural	Aesthetic/ethical values	Yearly no. visitors for sightseeing						
	Recreation and ecotourism	Yearly no. visitors for recreation	2 619.50 (6) [25]	Min: 84 Max: 1×10^{4}	2 736 (1) [26]	-	22 659 (6) [27]	Min: 10 Max: 9.58×10^{4}
	Spiritual and religious values							
	Cultural heritage	No. households considering area heritage						
	Scientific/ education							
Supporting	Primary production	NPP (kg km^{-2} year^{-1})	1.23×10^{6} (1) [28]	-	3.3×10^{5} (2) [29]	Min: 2.40×10^{5} Max: 4.20×10^{5}	1.07×10^{16} (15) [30]	Min: 1.05×10^{6} Max: 5.37×10^{16}
	Nutrient flow	Yearly nitrogen flow (kg km^{-2} year^{-1})					6.67×10^{3} (1) [31]	-
	Water cycling	Transpiration/total evapotranspiration						
	Habitat heterogeneity	Habitat diversity index	0.46 (4) [32]	Min: 0.12 Max: 0.84	0.12 (4) [32]	Min: 0.04 Max: 0.16	0.38 (4) [32]	Min: 0.06 Max: 1.14^{7}
	Nursery area	No. species with juveniles by total of species	0.68 (3) [33]	Min:0.56 Max: 0.77	0.54 (1) [34]	-		

** only data available quantified big groups, not species. In green indicators with no data available; orange indicators whit low or medium classification; blue ecosystem services without indicators; gray ecosystem services not relevant in the reference ecosystem. [1] (Blaber, 2002; Feka and Manzano, 2008; UN-REDD, 2014), [2] (Baran, 2000; Blaber, 2002; Ecoutin et al., 2010; Lamptey and Ofori-Danson, 2014), [3] (Feka and Ajonina, 2011), [4] (Lalèyè, 2000), [5] (FAO fisheries and aquaculture department, 2017a; FAO fisheries and aquaculture department, 2017b; FAO fisheries and aquaculture department, 2017c), [6] (Mmom and Arokoyu, 2010; Satyanarayana et al., 2012), [7] (Asibey, 1974; Falconer and Koppell, 1990; Jamnadass et al., 2015) [8] (Komiyama et al., 2008), [9] (Heubach et al., 2011; Heubes et al., 2012; INE são tomé e príncipe, 2014; Ndoye and Tieguhong, 2004; Nkem et al., 2010; Schaafsma, 2012) [10] (Feka and Ajonina, 2011), [11] (Brocard et al., 1998; de Faria et al., 2014; Kersten et al., 1998), [12] (Payet and Obura, 2004), [13] (Joint Research Center and NCEA, 2016), [14] (Ajonina et al., 2014b; Njana et al., 2018; UN-REDD, 2014), [15] (Mitsch et al., 2010), [16] (Lal, 2005; Sanogo et al., 2016), [17] (Khiteka and Kitheka, 1998; Roger et al., 2011), [18] (Li et al., 2005), [19] (Anayah et al., 2013; Chapas and Rees, 1964; Nizinski et al., 1994), [20] (Kamau, 1998), [21] (Bah et al., 2019; Healey et al., 1988), [22] (Roger et al., 2011), [23] (Anayah et al., 2013; Edmunds, 1990; Takounjou et al., 2011), [24] (Vitousek and Sanford, 1986), [25] (Ajonina et al., 2014a; Mallon et al., 2015; UN-REDD, 2014; UNEP, 2011), [26] (Mallon et al., 2015), [27] (Agyeman et al., 2019; Mallon et al., 2015) [28] (Alongi and Mukhopadhyay, 2015), [29] (Knoppers, 1994), [30] (Clark et al., 2001; Folega et al., 2015; Lo Seen Chong et al., 1993), [31] (Adepetu and Corey, 1977), [32] (Joint Research Center, 2018), [33] (Adite et al., 2013; Gajdzik et al., 2014; Kimani et al., 1996; Wright, 1986) [34] (Louca et al., 2009)* associated with aspects such as conservation status. *Recreation and ecotourism* is an important ES provided by the largest and best preserved mangrove of the island of São Tomé (Malanza), representing an important source of income to different stakeholders at the community level (Afonso, 2019). ES provided by mangroves are likely to vary from site to site. For example, mangroves in Thailand are well known for providing coastal protection, with an estimated economic value of nearly $6.4 US m^{-1} $year^{-1}$ (Sathirithai, 1998). In São Tomé, most mangroves are in inner basins, not directly exposed to the coastal dynamics, and occupy a very small percentage of the coastline. Thus, mangrove ES flow will be strongly context dependent.

Challenges in Assessing Ecosystem Services

The selection of an adequate classification system is an important step for the assessment of ES, and at the same time a challenge, since the quality of classification systems is inherently subjective (Caputo et al., 2019). The MEA classification system (MEA, 2005) was selected because it is a widely cited and well-known approach. It is the most used in global and regional assessments, and it is widely used by the scientific community, facilitating comparisons between studies. However, it has some limitations, namely those related to the simplification of extremely complex interactions. Recent studies ponder the use of only 'final services' to avoid considering processes as services because the value of end-products already includes the processes and components of the ecosystem needed for its production (Haines-Young et al., 2018). Therefore, the description of ES must integrate multiple concepts, such as ecosystem structure and composition, to facilitate the conceptualization of ES (Wallace, 2007) and the assessment of benefits to people (Raudsepp-Hearne et al., 2010). Like most ES classification systems, MEA does not consider the particularities of the marine systems (Liquete et al., 2013). This may add an additional layer of difficulty because ES lists often need to be adapted regionally.

Natural ecosystems provide many services that can be associated with ecological functions, and at least potentially, with an important revenue stream, even if this might not be recognized by the local community. Major challenges in assessing ES are related to the identification of regulating and supporting services. Their identification is seldom straightforward and, generally, they do not provide direct products. For instance, *air quality regulation*, delivered by many ecosystems, is a well-known service among the scientific community and its functional relevance, in terms of pollutants removal, has been clearly proven, especially in areas of high urbanization and increased population (MEA, 2005). The assessment of ES allows the recognition of some services that are difficult to identify, for example evaluating air pollutant concentrations in areas with or without specific ecosystems. Furthermore, it can change the ecosystem's valuation since it pinpoints different qualities relevant to effective ecosystem functioning that are essential to humans.

The quantification of ES is a complex process and a challenge already recognized in the scientific literature (Martinez-Harms et al., 2015; Newton et al., 2018). Preferentially, the indicators should represent a realistic value of ES flow, rather than service capacity, which considers the total value of the service that cannot be regularly used in its fullness. Several difficulties are associated with the selection of the most complete and appropriate indicators. In general, many adaptations are needed to capture the indicator flow. In many cases, only simple adaptations are needed, such as adjusting spatial and temporal units are needed since the temporal and spatial inadequacy were the most listed unresolved issues in the literature (Rivero and Villasante, 2016). Although the aim was to biophysical quantify every ES, it was not possible since some indicators were only expressed in economic value, implying the need to continue to find indicators that can capture the biophysical value of the service, before measuring the economic value to society. Many hundreds of indicators are available, as a consequence of the development of global and regional biodiversity targets (Vihervaara et al., 2017). Biophysical assessments can identify sources of benefit that can be helpful for local communities, thus a biophysical assessment should always be complemented with socio-economic information.

No data was available to quantify most mangrove and estuarine systems ES in tropical Africa (13 out of 27 and 16 out 25 ES, respectively), while terrestrial ES are more commonly quantified (9 out of 22 ES). São Tomé mangroves are reported in many global lists, but they are generally excluded from analyses due to lack of data (Hamilton and Casey, 2016). As already identified in this study, limitations of data availability are an obvious issue for tropical African coastal ecosystems (Adekola et al., 2015; Rivero and Villasante, 2016), hampering our ability to quantify ES in this region. Furthermore, most of the indicators were developed to assess status and trends in biodiversity and ecosystem integrity, but not directly to evaluate ES (Feld et al., 2010). Indicators cannot quantify the ES reliably if they are unsuitable to evaluate the quality and quantity of ES benefits (Layke et al., 2012). Establishing reliable and useful indicators is key to evolve in the quantification of ES since the lack of available data can lead to a biased overvaluation of better-known services, ecosystems, and regions, and that no data is assumed as no benefit. The *disease regulation* service is an example of a less studied service (Liquete et al., 2016), and thus, it has fewer available indicators. Additionally, many cultural services indicators are considered generic or underdeveloped (Hattam et al., 2013). This strongly limits the quantification of the number of people benefiting from the service (Hein et al., 2006).

Improving Mangrove Management and Empowerment of Local Communities in Developing Countries

Mangroves are highly threatened ecosystems (Gilman et al., 2008), especially due to urban expansion in coastal areas (Nfotabong-Atheull et al., 2013). São Tomé is a developing country where 32,3% of the population is living below the international poverty line ($1.9 in purchasing power parity term - Conceição et al., 2019), and population density is high, pushing people toward unsustainable

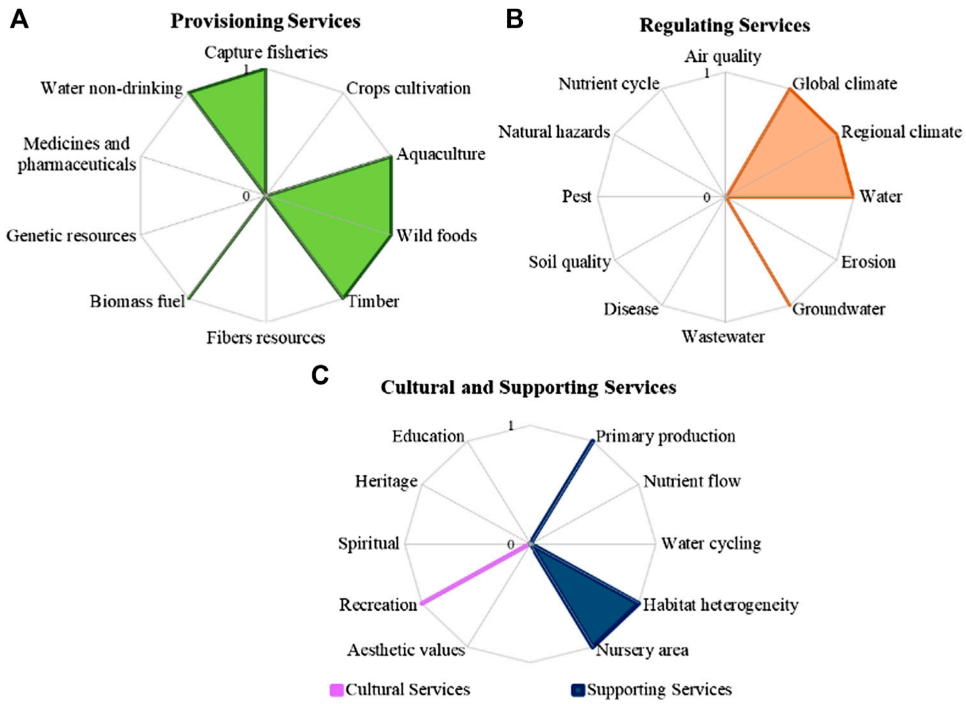

FIGURE 4 | Ecosystem Services quantified in African mangrove ecosystems (based on results from **Table 3**): **(A)** provisioning, **(B)** regulating, **(C)** cultural and supporting (0–not quantified; 1–quantified).

use of resources (Samarakoon, 2004). To make matters worse, ecosystem deterioration can reduce the delivery of ES in the long-term, further increasing the risk of poverty traps (Uchida et al., 2019). This is particularly important when people are not aware of all the threats that may affect mangroves, which is the case of São Tomé (Afonso, 2019). Developing countries typically have inadequate institutional "safety nets", forcing communities to choose between satisfying their short-term needs or long-term sustainability (Dawson et al., 2010; Uchida et al., 2019). This combination of factors creates a positive feedback loop, that can trap human societies and ecosystems in a downward spiral of ever-worsening conditions. The importance of protecting mangrove areas has been discussed with the inhabitants of communities in the vicinity of mangroves included in the Obô Natural Park (Pisoni et al., 2015). However, most locals were less aware of its boundaries and claimed to be reliant on resources taken from the park.

The ESF can be a tool to counteract this loop since it can be used to satisfy human needs and environmental sustainability (Poppy et al., 2014). By identifying and quantifying existing ES, as done herein for the mangroves of São Tomé, the ESF is also a useful and innovative tool to guide management decisions and to weigh alternatives, when compared to more traditional management tools (Martinez-Harms et al., 2015). Furthermore, it has the potential to contribute to conservation goals, developing informed decision-making, adding value to protected areas, and creating opportunities to sustainably manage ecosystems. However, some caution is needed, since incomplete assessments can undervalue endemic or threatened species with functional roles that are harder to evaluate or merely understudied (Ingram et al., 2012).

The ESF facilitates the assessment of ecosystems at different scales and contributes to the evolving knowledge of mangrove ES, especially in small mangroves where local surrounding communities demonstrate difficulties in recognizing the importance of maintaining natural ecosystems. This study provides an important contribution to identifying specific conservation measures in these ecosystems. Evaluating and comparing ES provided by mangroves at different scales also contributes to standardizing the process of ES identification and quantification for easier comparison between mangroves at similar scales. In places where assessment and conservation measures are already being implemented for mangrove use, is it now recognized that a balance between local needs and sustainable use of resources is key to achieve mangrove conservation (Satyanarayana et al., 2013). This is the case of Sri Lanka. Most importantly, despite the many challenges for effective implementation, it can help bring awareness about our reliance on ecosystems and the fragility of ecosystem functioning, facilitating wider societal engagement in environmental conservation. ESF is even more important at locations where ES are of most immediate benefit to local (and often poor) people and might be strongly affected by global changes such as climate change, sea-level rise and the consequent effects of ocean acidification, warming, salinity, and hypoxia on fisheries.

TABLE 4 | Ecosystem services identified in mangroves of São Tomé. Source of information: literature review (adapted from Pisoni et al., 2015) and field work. Black circles and white circles represent, respectively, the presence and absence of each ecosystem service.

	Ecosystem Services	São Tomé Mangroves	
		Literature review	Field work assessment
Provisioning	Capture fisheries	●	●
	Crops cultivation	○	○
	Aquaculture	○	○
	Wild foods	●	●
	Timber	●	●
	Fibers and ornamental resources	○	○
	Biomass fuel	●	●
	Genetic resources	○	○
	Medicines and pharmaceuticals	●	●
	Water for non-drinking purposes	○	●
Regulating	Air quality	○	●
	Global climate regulation	●	●
	Regional climate regulation	○	●
	Water regulation	○	●
	Erosion regulation	●	●
	Groundwater recharge	○	●
	Wastewater treatment	●	●
	Disease regulation	○	●
	Soil quality	○	●
	Pest regulation	○	●
	Pollination	○	●
	Natural hazards regulation	●	●
	Nutrient cycle	○	●
Cultural	Aesthetic/ethical values	●	●
	Recreation and ecotourism	●	●
	Spiritual values	○	○
	Cultural heritage	○	○
	Scientific	○	●
Supporting	Primary Production	○	●
	Nutrient flow	○	●
	Water cycling	○	●
	Habitat heterogeneity	○	●
	Nursery area	●	●

TABLE 5 | Ecosystem Services quantified in São Tomé Mangroves and values for comparison with mangroves in the tropical African region (indicators source: **Table 1**).

Ecosystem Service		Indicator	Mangroves	
			São Tomé	Tropical African region
Provisioning	Wild foods	Number of wild species used as food	27.00*	13.50
Regulating	Water regulation	Nitrogen concentration (mg N L^{-1})	Min: 0.01	0.30
			Max: 0.02	
Supporting	Nursery area	Number of species with juveniles by total of species	Min: 0.44 Max: 0.79	0.68 Min: 0.56 Max: 0.77

* Value from all mangroves in São Tomé (Pisoni *et al.*, 2015)

**value from all mangroves in são tomé (Pisoni et al., 2015)*

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/supplementary files.

AUTHOR CONTRIBUTIONS

Conceptualization and design of study, AB, FA, PF, and RL; Data Collection, AB, FA, FR, JH, PC, PF, and RL; Data analysis and interpretation, AB, FA, PF, and RL; Writing and preparation of original draft, FA; Funding acquisition, PF. All authors contributed to manuscript revision, read and approved the submitted version.

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REFERENCES

Adekola, O., Mitchell, G., and Grainger, A. (2015). Inequality and ecosystem services: the value and social distribution of Niger Delta wetland services. *Ecosystem Services* 12, 42–54. doi:10.1016/j.ecoser.2015.01.005

Adepetu, J. A., and Corey, R. B. (1977). Changes in N and P availability and P fractions in Iwo soil from Nigeria under intensive cultivation. *Plant Soil* 46, 309–316. doi:10.1007/BF00010087

Adite, A., ImorouToko, I., and Gbankoto, A. (2013). Fish assemblages in the degraded mangrove ecosystems of the coastal zone, Benin, west africa: implications for ecosystem restoration and resources conservation. *J. Environ. Psychol.* 4, 1461–1475. doi:10.4236/jep.2013.412168

Afonso, F. M. G. (2019). A importância dos mangais de São Tomé : perceções e serviços ecossistémicos. Master thesis, Lisbon (Portugal): University of Lisbon. Available at: https://repositorio.ul.pt/handle/10451/20?locale=en.

Agbenyega, O., Burgess, P. J., and Cook, M. (2009). Application of an ecossystem function framework to perceptions of community woodlands. *Land Use Policy*, 26(3), 551–557. doi:10.1016/j.landusepol.2008.08.011

Agyeman, Y. B., Aboagye, O. K., and Ashie, E. (2019). Visitor satisfaction at kakum national park in Ghana. *Tour. Recreat. Res.* 44, 178–189. doi:10.1080/02508281.2019.1566048

Ajonina, G. N., Agardy, T., Lau, W., Agbogah, K., and Gormey, B. (2014a). *Mangrove conditions as indicator for potential payment for ecosystem services in some estuaries of western region of Ghana. West Africa*, Berlin, Germany: Springer, 151–166. doi:10.1007/978-3-319-06388-1_13

Ajonina, G. N., Kairo, J., Grimsditch, G., Sembres, T., Chuyong, G., and Diyouke, E. (2014b). *Assessment of mangrove carbon stocks in Cameroon, Gabon, the republic of Congo (RoC) and the democratic republic of Congo (DRC) including their potential for reducing emissions from deforestation and forest degradation (REDD+)*, Berlin, Germany: Springer, 177–189. doi:10.1007/978-3-319-06388-1_15

Alcamo, J., Ash, N. J., Butler, C. D., Callicott, J. B., Capistrano, D., Carpenter, S. R., et al. (2003a). "Ecosystem and human well-being," in *Ecosystems and Human Well-being: a framework for assessment*. (Washington DC: Island Press, 71–83.

Alcamo, J., Ash, N. J., Butler, C. D., Callicott, J. B., Capistrano, D., Carpenter, S. R., et al. (2003b). "Ecosystem and their service," in *Ecosystems and Human Well-being: a framework for assessment*. Washington DC: Island Press, 49–62.

Alongi, D. M., and Mukhopadhyay, S. K. (2015). Contribution of mangroves to coastal carbon cycling in low latitude seas. *Agric. For. Meteorol.* 213, 266–272. doi:10.1016/j.agrformet.2014.10.005

Anayah, F. M., Kaluarachchi, J. J., Pavelic, P., and Smakhtin, V. (2013). Predicting groundwater recharge in Ghana by estimating evapotranspiration. *Water Int.* 38, 408–432. doi:10.1080/02508060.2013.821642

Atkinson, S. C., Jupiter, S. D., Adams, V. M., Ingram, J. C., Narayan, S., Klein, C. J., et al. (2016). Prioritising mangrove ecosystem services results in spatially variable management priorities. *PloS One* 11, e0151992–21. doi:10.1371/journal.pone.0151992

Asibey, E. O. A. (1974). Wildlife as a source of protein in africa south of the sahara. *Biol. Conserv.* 6, 32–39. doi:10.1016/0006-3207(74)900039-110.1016/0006-3207(74)90039-1

Badola, R., and Hussain, S. A. (2005). Valuing ecosystem functions: an empirical study on the storm protection function of Bhitarkanika mangrove ecosystem, India. *Environ. Conserv.* 32, 85–92. doi:10.1017/s0376892905001967

Bah, O. A., Kone, T., Yaffa, S., Ndiaye, M. L., and Sane, S. (2019). Water quality parameters and fisheries in central river region of the Gambia: differences between wet and dry seasons. *Int. J. Fish. Aquat. Stud.* 7, 285–295.

Bandaranayake, W. M. (1998). Traditional and medicinal uses of banana. *Mangroves Salt Marshes* 2, 133–148. doi:10.1023/A:1009988607044

Baran, E. (2000). Biodiversity of estuarine fish faunas in West Africa. *Naga. The ICLARM Quartely.* 23, 4–9.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., and Silliman, B. R. (2011). The value of estuarine and coastal ecosystem services. *Ecol. Monogr.* 81, 169–193. doi:10.1890/10-1510.1

Barbier, E. B. (2007). Valuing ecosystem services as productive inputs. *Econ. Pol.* 22, 178–229. doi:10.1111/j.1468-0327.2007.00174.x

Blaber, S. (2002). ?Fish in hot water?: the challenges facing fish and fisheries research in tropical estuaries. *J. Fish. Biol.* 61, 1–20. doi:10.1006/jfbi.2002.2063

Böhnke-Henrichs, A., Baulcomb, C., Koss, R., Hussain, S., and de Groot, R. S. (2013). Typology and indicators of ecosystem services for marine spatial planning and management. *J. Environ. Manag.* 130, 135–145. doi:10.1016/j.jenvman.2013.08.027

Brander, L. M., van Beukering, P., Balzan, M., Broekx, S., Liekens, I., Marta-Pedroso, C., et al. (2018). Report on economic mapping and assessment methods for ecosystem services, Technical Paper 642007.

Brauman, K. A., Garibaldi, L. A., Polasky, S., and Zayas, C. (2019). "Chapter 2.3 status and trends - nature's contributions to people," in *IPBES. 2019. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. S. Díaz, J. Settele, E. S. Brondizio, H. T. Ngo, M. Guèze, J. Agard, et al. (Bonn, Germany: IPBES secretariat).

Brito, A. C., Silva, T., Beltrán, C., Chainho, P., and de Lima, R. F. (2017). Phytoplankton in two tropical mangroves of São Tomé Island (Gulf of Guinea): a contribution towards sustainable management strategies. *Regional Studies in Marine Science.* 9, 89–96. doi:10.1016/j.rsma.2016.11.005

Brocard, D., Lacaux, J.-P., and Eva, H. (1998). Domestic biomass combustion and associated atmospheric emissions in West Africa. *Global Biogeochem. Cycles.* 12, 127–139. doi:10.1029/97GB02269

Broszeit, S., Beaumont, N. J., Uyarra, M. C., Heiskanen, A.-S., Frost, M., Somerfield, P. J., et al. (2017). What can indicators of good environmental status tell us about ecosystem services?: reducing efforts and increasing cost-effectiveness by reapplying biodiversity indicator data. *Ecol. Indicat.* 81, 409–442. doi:10.1016/j.ecolind.2017.05.057

Burkhard, B., Kroll, F., Müller, F., and Windhorst, W. (2009). Landscapes' capacities to provide ecosystem services - a concept for land-cover based assessments. *Lo.* 15, 1–22. doi:10.3097/LO.200915

Burkhard, B., Maes, J., Potschin-Young, M. B., Santos-Martín, F., Geneletti, D., Stoev, P., et al. (2018). Mapping and assessing ecosystem services in the EU - lessons learned from the ESMERALDA approach of integration. *One Ecosyst.* 3. doi:10.3897/oneeco.3.e29153

Caputo, J., D'Amato, D., and Matthies, B. D. (2019). "Ecosystem services in the service-dominant logic framework," in *Services in family forestry*. Editors T. Hujala, A. Toppinen, and B. J. Butler (Finland: Springer), 31–47. doi:10.1007/978-3-030-28999-7

Castro, A. J., Verburg, P. H., Martín-López, B., Garcia-Llorente, M., Cabello, J., Vaughn, C. C., et al. (2014). Ecosystem service trade-offs from supply to social demand: a landscape-scale spatial analysis. *Landsc. Urban Plann.* 132, 102–110. doi:10.1016/j.landurbplan.2014.08.009

Chapas, L. C., and Rees, A. R. (1964). Evaporation and evapotranspiration in southern Nigeria. *Q.J Royal Met. Soc.* 90, 313–319. doi:10.1002/qj.49709038510

Clark, D. A., Brown, S., Kicklighter, D. W., Chambers, J. Q., Thomlinson, J. R., Ni, J., et al. (2001). Net primary production in tropical forests: an evaluation and synthesis of existing field data. *Ecol. Appl.* 11, 371–384. doi:10.1890/1051-0761(2001)011[0371:nppitf]2.0.co;2

Conchedda, G., Lambin, E. F., and Mayaux, P. (2011). Between land and sea: livelihoods and environmental changes in mangrove ecosystems of Senegal. *Ann. Assoc. Am. Geogr.* 101, 1259–1284. doi:10.1080/00045608.2011.579534

Conceição, P., Assa, J., Calderon, C., Gray, G. R., Gulasan, N., Hsu, Y.-C., et al. (2019). *Human Development Report 2019-Beyond income, beyond averages, beyond today: inequalities in human development in the 21st century*. New York, USA: United Nations Development Programme.

Curnick, D. J., Pettorelli, N., Amir, A. A., Balke, T., Barbier, E. B., Crooks, S., et al. (2019). The value of small mangrove patches. *Science* 363, 239. doi:10.1126/science.aaw0809

Dahdouh-Guebas, B., Arany, I., Potschin-Young, M., Bereczki, K., Kertész, M., Kiss, M., et al. (2018). Where concepts meet the real world: a systematic review of ecosystem service indicators and their classification using CICES. *Ecosystem Services* 29, 145–157. doi:10.1016/j.ecoser.2017.11.018

Dawson, T. P., Rounsevell, M. D. A., Kluvánková-Oravská, T., Chobotová, V., and Stirling, A. (2010). Dynamic properties of complex adaptive ecosystems: implications for the sustainability of service provision. *Biodivers. Conserv.* 19, 2843–2853. doi:10.1007/s10531-010-9892-z

de Faria, M., Ferreira, P., Melo, J., and Vasconcelos, M. (2014). A social assessment of forest degradation in the "cacheu mangroves natural park", Guinea-bissau. *Forests* 5, 3327–3343. doi:10.3390/f5123327

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., et al. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services* 1, 50–61. doi:10.1016/j.ecoser.2012.07.005

Drakou, E. G., Liquete, C., Beaumont, N., Boon, A., Vitasalo, M., and Agostini, V. (2017). "Mapping marine and coastal ecosystem services," in *Mapping ecosystem services* Editors B. Burkhard and J. Maes (Sofia, Bulgaria: Pensoft Publishers), 250–255.

Donato, D. C., Kauffman, J. B., Murdiyarso, D., Kurnianto, S., Stidham, M., and Kanninen, M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nat. Geosci.* 4, 293–297. doi:10.1038/ngeo1123

Duke, N. C., Meynecke, J. O., Dittmann, S., Ellison, A. M., Anger, K., Berger, U., et al. (2007). A world without mangroves?. *Science* 317, 41–42. doi:10.1126/science.317.5834.41b

Ecoutin, J. M., Simier, M., Albaret, J. J., Laë, R., and Tito de Morais, L. (2010). Changes over a decade in fish assemblages exposed to both environmental and fishing constraints in the Sine Saloum estuary (Senegal). *Estuar. Coast Shelf Sci.* 87, 284–292. doi:10.1016/j.ecss.2010.01.009

Edmunds, W. M. (1990). Groundwater recharge in Senegal. Wallingford, UK. Report WD/90/49R.

Egoh, B., Drakou, E. G., Dunbar, M. B., Maes, J., and Willemen, L. (2012). Indicators for mapping ecosystem services: a review. doi:10.13140/2.1.3420.2565

Englund, O., Berndes, G., and Cederberg, C. (2017). How to analyse ecosystem services in landscapes-A systematic review. *Ecol. Indicat.* 73, 492–504. doi:10.1016/j.ecolind.2016.10.009

European Environment Agency (2018). Oxygen consuming substances in rivers, Denmark. Available at: http://www.eea.europa.eu (Accessed August 20, 2007).

FAO Fisheries and Aquaculture Department (2017a). National aquaculture sector overview: Cameroon. Available at: http://www.fao.org/fishery/countrysector/naso_cameroun/en (Accessed February 24, 2020).

FAO Fisheries and Aquaculture Department (2017b). National aquaculture sector overview: democratic republic of Congo. Available at: http://www.fao.org/fishery/countrysector/naso_congoRep/en (Accessed February 24, 2020).

FAO Fisheries and Aquaculture Department (2017c). National aquaculture sector overview: Ghana. Available at: http://www.fao.org/fishery/countrysector/naso_ghana/en (Accessed February 24, 2020).

Falconer, J., and Koppell, C. R. S. (1990). The Major Significance of "Minor" Forest Products: the local use and value of forests in the West African humid forest zone. Community for. Note 6, UN FAO Available at: http://www.fao.org/docrep/t9450e/t9450e00.htm#Contents.

Feka, N. Z., and Ajonina, G. N. (2011). Drivers causing decline of mangrove in West-Central Africa: a review. *International Journal of Biodiversity Science, Ecosystem Services & Management* 7, 217–230. doi:10.1080/21513732.2011.634436

Feka, N. Z., and Manzano, M. G. (2008). The implications of wood exploitation for fish smoking on mangrove ecosystem conservation in the south west province, Cameroon. *Tropical Conservation Science* 1, 222–241. doi:10.1177/194008290800100305

Feld, C. K., Sousa, J. P., da Silva, P. M., and Dawson, T. P. (2010). Indicators for biodiversity and ecosystem services: towards an improved framework for ecosystems assessment. *Biodivers. Conserv.* 19, 2895–2919. doi:10.1007/s10531-010-9875-0

Félix, P. M., Chainho, P., Lima, R. F., Costa, J. L., Almeida, A. J., Domingos, I., et al. (2017). Mangrove fish of São Tomé Island (Gulf of Guinea): new occurrences and habitat usage. *Mar. Freshw. Res.* 68, 123–130. doi:10.1071/MF15392

Fisher, B., Turner, R. K., and Morling, P. (2009). Defining and classifying ecosystem services for decision making. *Ecol. Econ.* 68, 643–653. doi:10.1016/j.ecolecon.2008.09.014

Friess, D. A., Thompson, B. S., Brown, B., Amir, A. A., Cameron, C., Koldewey, H. J., et al. (2016). Policy challenges and approaches for the conservation of mangrove forests in Southeast Asia. *Conserv. Biol.* 30, 933–949. doi:10.1111/cobi.12784

Folega, F., Woegan, Y. A., Marra, D., Wala, K., Batawila, K., Seburanga, J. L., et al. (2015). Long term evaluation of green vegetation cover dynamic in the Atacora Mountain chain (Togo) and its relation to carbon sequestration in West Africa. *J. Mt. Sci.* 12, 921–934. doi:10.1007/s11629-013-2973-1

Gajdzik, L., Vanreusel, A., Koedam, N., Reubens, J., and Muthumbi, A. W. N. (2014). The mangrove forests as nursery habitats for the ichthyofauna of Mida Creek (Kenya, East Africa). *J. Mar. Biol. Ass.* 94, 865–877. doi:10.1017/S0025315414000290

Gilman, E. L., Ellison, J., Duke, N. C., and Field, C. (2008). Threats to mangroves from climate change and adaptation options: a review. *Aquat. Bot.* 89, 237–250. doi:10.1016/j.aquabot.2007.12.009

Gómez-Baggethun, E., de Groot, R., Lomas, P. L., and 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. doi:10.1016/j.ecolecon.2009.11.007

Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A., and Cardoso, A. C. (2016). Assessing water ecosystem services for water resource management. *Environ. Sci. Pol.* 61, 194–203. doi:10.1016/j.envsci.2016.04.008

Grêt-Regamey, A., Weibel, B., Bagstad, K. J., Ferrari, M., Geneletti, D., Klug, H., et al. (2014). On the effects of scale for ecosystem services mapping. *PLoS One* 9, e112601. doi:10.1371/journal.pone.0112601

Haines-Young, R., and Potschin, M. (2018). *Common international classification of ecosystem services (CICES) V5. 1. Guidance on the application of the revised structure*.

Haines-Young, R., Potschin-Young, M., and Czúcz, B. (2018). Report on the use of CICES to identify and characterise the biophysical, social and monetary dimensions of ES assessments. Deliverable D4.2. EU Horizon 2020 ESMERALDA Project Grant agreement No. 642007.

Hamilton, S. E., and Casey, D. (2016). Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). *Global Ecol. Biogeogr.* 25, 729–738. doi:10.1111/geb.12449

Hamilton, S. (2011). The impact of shrimp farming on mangrove ecosystems and local livelihoods along the pacific coast of Ecuador 656. Dissertations. Ann Arbor, MI: ProQuest Dissertations.

Hattam, C., Atkins, J. P., Beaumont, N., Börger, T., Böhnke-Henrichs, A., Burdon, D., et al. (2015). Marine ecosystem services: linking indicators to their classification. *Ecol. Indicat.* 49, 61–75. doi:10.1016/j.ecolind.2014.09.026

Healey, M. J., Moll, R. A., and Diallo, C. O. (1988). Abundance and distribution of bacterioplankton in the Gambia river, west africa. *Microb. Ecol.* 16, 291–310. doi:10.1007/BF02011701

Hein, L., van Koppen, K., de Groot, R. S., and van Ierland, E. C. (2006). Spatial scales, stakeholders and the valuation of ecosystem services. *Ecol. Econ.* 57, 209–228. doi:10.1016/j.ecolecon.2005.04.005

Heubach, K., Wittig, R., Nuppenau, E.-A., and Hahn, K. (2011). The economic importance of non-timber forest products (NTFPs) for livelihood maintenance of rural west African communities: a case study from northern Benin. *Ecol. Econ.* 70, 1991–2001. doi:10.1016/j.ecolecon.2011.05.015

Heubes, J., Heubach, K., Schmidt, M., Wittig, R., Zizka, G., Nuppenau, E.-A., et al. (2012). Impact of future climate and land use change on non-timber forest product provision in Benin, west africa: linking niche-based modeling with ecosystem service values. *Econ. Bot.* 66, 383–397. doi:10.1007/s12231-012-9216-1

INE São Tomé e Príncipe (2014). IV Recenseamento geral da população e habitação: estado e estrutura da população. São Tomé e Príncipe.

Ingram, J. C., Redford, K. H., and Watson, J. E. M. (2012). Applying ecosystem services approaches for biodiversity conservation: benefits and challenges. *SAPIENS.* 5.

Jamnadass, R., McMullin, S., Iiyama, M., Dawson, I. K., Powell, B., Termote, C., et al. (2015). "Understanding the roles of forests and tree-based systems in food provision," in *Forests and Food: Addressing Hunger and Nutrition Across Sustainable Landscapes*, Editors: B. Vira, C. Wildburger, and S. Mansourian, (Cambridge, United Kingdom: Open Book Publishers). 27–70. doi:10.11647/obp.0085.02

Joint Research Centre (2018). The digital observatory for protected areas (DOPA) explorer 2.0 [online], ispra, Italy. Available at: http://dopa-explorer.jrc.ec.europe.eu (Accessed August 20, 2006).

Joint Research Centreand NCEA (2016). *Emissions database for global atmospheric research, release EDGAR v4 3.2 (1970–2012) of March 2016.* Available at: http://edgar.jrc.ec.europa.ec.

Kamau, F. K. (1998). Mangrove forests along the tidal flats and lagoons of Ngomeni, Ungwana bay. *Coast. Ecol. Serv.* 4, 111–133.

Kersten, I., Baumbach, G., Oluwole, A. F., Obioh, I. B., and Ogunsola, O. J. (1998). Urban and rural fuelwood situation in the tropical rain-forest area of south-west Nigeria. *Energy* 23, 887–898. doi:10.1016/S0360-5442(98)00025-5

Khiteka, J. U., and Kitheka, J. U. (1998). Groundwater outflow and its linkages to coastal circulation in a mangrove-fringed creek in Kenya. *Estuar. Coast Shelf Sci.* 41, 63–75. doi:10.1006/ecss.1998.0325

Kimani, E., Mwatha, G., Wakwabi, E., Ntiba, J., and Okoth, B. (1996). Fishes of a shallow tropical mangrove estuary, Gazi, Kenya. *Mar. Freshw. Res.* 47, 857–868. doi:10.1071/MF9960857

Knoppers, B. (1994). Chapter 9 aquatic primary production in coastal lagoons. *Elsevier Oceanogr. Ser.* 60, 243–286. doi:10.1016/S0422-9894(08)70014-X

Komiyama, A., Ong, J. E., and Poungparn, S. (2008). Allometry, biomass, and productivity of mangrove forests: a review. *Aquat. Bot.* 89, 128–137. doi:10.1016/j.aquabot.2007.12.006

Kuenzer, C., and Tuan, V. Q. (2013). Assessing the ecosystem services value of Can Gio mangrove biosphere reserve: combining earth-observation- and household-survey-based analyses. *Appl. Geogr.* 45, 167–184. doi:10.1016/j.apgeog.2013.08.012

Lal, R. (2005). Soil carbon sequestration in natural and managed tropical forest ecosystems. *J. Sustain. For.* 21, 1–30. doi:10.1300/J091v21n01_01

Lalèyè, P. (2000). "Acadja fisheries enhancement systems in Benin: their productivity and environmental impacts," in *Biodiversity and sustainable use of fish in the coastal zone, ICLARM Conference Proceedings*. Editors E. K. Abban, C. M. V. Casal, T. M. Falk, and R. S. V. Pulin, (Achimota, Ghana: International Center for Living Aquatic Resources Management - The World Fish Center). 51–52.

Lamptey, M. A., and Ofori-Danson, P. K. (2014). The status of fish diversity and fisheries of the Keta Lagoon, Ghana, West Africa. *Ghana J. Sci.* 54, 3–18.

Layke, C., Mapendembe, A., Brown, C., Walpole, M., and Winn, J. (2012). Indicators from the global and sub-global Millennium Ecosystem Assessments: an analysis and next steps. *Ecol. Indicat.* 17, 77–87. doi:10.1016/j.ecolind.2011.04.025

Lee, S. Y., Primavera, J. H., Dahdouh-Guebas, F., Mckee, K., Bosire, J. O., Cannicci, S., et al. (2014). Ecological role and services of tropical mangrove ecosystems: a reassessment. *Global Ecol. Biogeogr.* 23, 726–743. doi:10.1111/geb.12155

Li, K. Y., Coe, M. T., and Ramankutty, N. (2005). Investigation of hydrological variability in West Africa using land surface models. *J. Clim.* 18, 3173–3188. doi:10.1175/JCLI3452.1

Liquete, C., Piroddi, C., Drakou, E. G., Gurney, L., Katsanevakis, S., Charef, A., et al. (2013). Current status and future prospects for the assessment of marine and coastal ecosystem services: a systematic review. *PloS One* 8, e67737. doi:10.1371/journal.pone.0067737

Liquete, C., Cid, N., Lanzanova, D., Grizzetti, B., and Reynaud, A. (2016). Perspectives on the link between ecosystem services and biodiversity: the assessment of the nursery function. *Ecol. Indicat.* 63, 249–257. doi:10.1016/j.ecolind.2015.11.058

Lo Seen Chong, D., Mougin, E., and Gastellu-Etchegorry, J. P. (1993). Relating the global vegetation index to net primary productivity and actual evapotranspiration over africa. *Int. J. Rem. Sens.* 14, 1517–1546. doi:10.1080/01431169308953984

Louca, V., Lindsay, S. W., Majambere, S., and Lucas, M. C. (2009). Fish community characteristics of the lower Gambia River floodplains: a study in the last major undisturbed West African river. *Freshw. Biol.* 54, 254–271. doi:10.1111/j.1365-2427.2008.02105.x

Macintosh, D. J., and Ashton, E. C. (2002). *A review of mangrove biodiversity conservation and management*. Aarhus, Denmark: Centre for Tropical Ecosystems Research, University of Aarhus.

Mallon, D. P., Hoffmann, M., and McGowan, P. J. K. (2015). *An IUCN situation analysis of terrestrial and freshwater fauna in West and Central Africa*. doi:10.2305/iucn.ch.2015.ssc-op.54

Martín-López, B., Felipe-Lucia, M. R., Bennett, E. M., Norström, A., Peterson, G., Plieninger, T., et al. (2019). A novel telecoupling framework to assess social relations across spatial scales for ecosystem services research. *J. Environ. Manag.* 241, 251–263. doi:10.1016/j.jenvman.2019.04.029

Martinez-Harms, M. J., Bryan, B. A., Balvanera, P., Law, E. A., Rhodes, J. R., Possingham, H. P., et al. (2015). Making decisions for managing ecosystem services. *Biol. Conserv.* 184, 229–238. doi:10.1016/j.biocon.2015.01.024

Martin-Ortega, J., Jorda-Capdevila, D., Glenk, K., and Holstead, K. L. (2015). "What defines ecosystem services-based approaches," in *water ecosystem services: a global perspective*. Editors J. Martin-Ortega, R. C. Ferrier, L. J. Gordon, and S. Khan (Paris, France: UNESCO Publishing), 3–13.

Millenium Ecosystem Assessment (2005). *Ecosystems and human well-being: biodiversity synthesis*. Washington, DC: World Institute.

Mitsch, W. J., Nahlik, A., Wolski, P., Bernal, B., Zhang, L., and Ramberg, L. (2010). Tropical wetlands: seasonal hydrologic pulsing, carbon sequestration, and methane emissions. *Wetl. Ecol. Manag.* 18, 573–586. doi:10.1007/s11273-009-9164-4

Mmom, P. C., and Arokoyu, S. B. (2010). Mangrove forest depletion, biodiversity loss and traditional resources management practices in the Niger Delta, Nigeria. *Res. J. Appl. Sci. Eng. Technol.* 2, 28–34.

Müller, F., and Burkhard, B. (2012). The indicator side of ecosystem services. *Ecosystem Services* 1, 26–30. doi:10.1016/j.ecoser.2012.06.001

Mumby, P. J., Edwards, A. J., Arias-González, J. E., Lindeman, K. C., Blackwell, P. G., Gall, A., et al. (2004). Mangroves enhance the biomass of coral reef fish communities in the Caribbean. *Nature* 427, 533–536. doi:10.1038/nature02286

Ndoye, O., and Tieguhong, J. C. (2004). Forest resources and rural livelihoods: the conflict between timber and non-timber forest products in the Congo Basin. *Scand. J. For. Res.* 19, 36–44. doi:10.1080/14004080410034047

Newton, A., Brito, A. C., Icely, J. D., Derolez, V., Clara, I., and Angus, S. (2018). Assessing, quantifying and valuing the ecosystem services of coastal lagoons. *J. Nat. Conserv.* 44(3), 50–65. doi:10.1016/j.jnc.2018.02.009

Nfotabong-Atheull, A., Din, N., and Dahdouh-Guebas, F. (2013). Qualitative and quantitative characterization of mangrove vegetation structure and dynamics in a peri-urban setting of douala (Cameroon): an approach using air-borne imagery. *Estuar. Coast.* 36, 1181–1192. doi:10.1007/s12237-013-9638-8

Nizinski, J., Morand, D., and Fournier, C. (1994). Actual evapotranspiration of a thorn scrub with Acacia tortilis and Balanites aegyptiaca (North Senegal). *Agric. For. Meteorol.* 72, 93–111. doi:10.1016/0168-1923(94)90093-0

Njana, M. A., Zahabu, E., and Malimbwi, R. E. (2018). Carbon stocks and productivity of mangrove forests in Tanzania. *South. For. a J. For. Sci.* 80, 217–232. doi:10.2989/20702620.2017.1334314

Nkem, J., Kalame, F. B., Idinoba, M., Somorin, O. A., Ndoye, O., and Awono, A. (2010). Shaping forest safety nets with markets: adaptation to climate change under changing roles of tropical forests in Congo Basin. *Environ. Sci. Pol.* 13, 498–508. doi:10.1016/j.envsci.2010.06.004

Ostrom, V., and Ostrom, E. (2015). "Public goods and public choices," in *elinor Ostrom and the bloomington School of political economy: resource governance*. Editors D. H. Cole and M. D. McGinnis (Lanham MD: Lexington Books), 3–35. doi:10.1007/978-3-662-44143-5_5

Owuor, M. A., Mulwa, R., Otieno, P., Icely, J., and Newton, A. (2019). Valuing mangrove biodiversity and ecosystem services: a deliberative choice experiment in Mida Creek, Kenya. *Ecosystem Services* 40, 101040. doi:10.1016/j.ecoser.2019.101040

Palacios, M. L., and Cantera, J. R. (2017). Mangrove timber use as an ecosystem service in the Colombian Pacific. *Hydrobiologia* 803, 345–358. doi:10.1007/s10750-017-3309-x

Payet, R., and Obura, D. (2004). The negative impacts of human activities in the eastern African region: an international waters perspective. *Ambio.* 33, 24–33. doi:10.1579/0044-7447-33.1.24

Pisoni, T., de Lima, R. F., Brito, A. C., Chainho, P., Félix, P. M., Caçador, I., et al. (2015). Abordagem ecossistémica integrada para a conservação e gestão da biodiversidade na zona tampão dos parques naturais Obô de São Tomé e Príncipe: planos de gestão participativa para dois sítios de mangal na Ilha de S. Tomé (Praia das Conchas e Malanza). São Tomé.

Poppy, G. M., Chiotha, S., Eigenbrod, F., Harvey, C. A., Honzák, M., Hudson, M. D., et al. (2014). Food security in a perfect storm: using the ecosystem services framework to increase understanding. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.* 369, 20120288. doi:10.1098/rstb.2012.0288

Rivero, S., and Villasante, S. (2016). What are the research priorities for marine ecosystem services?. *Mar. Pol.* 66, 104–113. doi:10.1016/j.marpol.2016.01.020

Roger, F., Benjamin, N. N., Ghislain, T. Y. J., and Emmanuel, E. G. (2011). Relationship between climate and groundwater recharge in the besseke watershed (douala - Cameroon). *Jwarp.* 03, 607–619. doi:10.4236/jwarp.2011.38070

Raudsepp-Hearne, C., Peterson, G. D., and Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proc. Natl. Acad. Sci. U.S.A.* 107, 5242–5247. doi:10.1073/pnas.0907284107

Samarakoon, J. (2004). Issues of livelihood, sustainable development, and governance: bay of Bengal. *Ambio.* 33, 34–44. doi:10.1579/0044-7447-33.1.34

Samoilys, M., Pabari, M., Andrew, T., Maina, G. W., Church, J., Momanyi, A., et al. (2015). *Resilience of coastal systems and their human partners*. Nairobi, Kenya: IUCN ESARO, WIOMSA, CORDIO and UNEP Nairobi Convention. Available at: www.iucn.org.

Sanogo, K., Gebrekirstos, A., Bayala, J., Villamor, G. B., Kalinganire, A., and Dodiomon, S. (2016). Potential of dendrochronology in assessing carbon sequestration rates of Vitellaria paradoxa in southern Mali, West Africa. *Dendrochronologia* 40, 26–35. doi:10.1016/j.dendro.2016.05.004

Sathirithai, S. (1998). Economic valuation of mangroves and the roles of local communities in the conservation of natural Resources: case study of surat thani, south of Thailand. *EEPSEA Res. Rep. Ser.* doi:10.3390/su6063484

Satyanarayana, B., Bhanderi, P., Debry, M., Maniatis, D., Foré, F., Badgie, D., et al. (2012). A socio-ecological assessment aiming at improved forest resource management and sustainable ecotourism development in the mangroves of Tanbi Wetland National Park, the Gambia, West Africa. *Ambio.* 41, 513–526. doi:10.1007/s13280-012-0248-7

Satyanarayana, B., Mulder, S., Jayatissa, L. P., and Dahdouh-Guebas, F. (2013). Are the mangroves in the Galle-Unawatuna area (Sri Lanka) at risk? A social-ecological approach involving local stakeholders for a better conservation policy. *Ocean Coast Manag.* 71, 225–237. doi:10.1016/j.ocecoaman.2012.10.008

Schaafsma, M. (2012). Mapping NTFP collection in Tanzania: a comparison of surveys. CSERGE working paper, No. 2012-0505.

Soares, F. M. C. O. (2016). Modelling the distribution of São Tomé bird species: ecological determinants and conservation priorization. Master thesis, Lisbon, Portugal: Faculty of Sciences of the University of Lisbon. Available at: https://repositorio.ul.pt/handle/10451/30776.

Spalding, M. D., Kainuma, M., and Collins, L. (2010). *World atlas of mangroves*. London, UK: Earthscan.

Takounjou, A. F., Ngoupayou, J. R. N., Riotte, J., Takem, G. E., Mafany, G., Maréchal, J. C., et al. (2011). Estimation of groundwater recharge of shallow aquifer on humid environment in Yaounde, Cameroon using hybrid water-fluctuation and hydrochemistry methods. *Environ. Earth Sci.* 64, 107–118. doi:10.1007/s12665-010-0822-x

TEEB (2010). The Economics of Ecosystems and Biodiversity. Mainstreaming the economics of nature: a Synthesis of the approach, conclusions and recommendations of TEEB. Available at: www.dieaktivisten.de.

Uchida, E., Rivera-Monroy, V. H., Ates, S. A., Castañeda-Moya, E., Gold, A. J., Guilfoos, T., et al. (2019). "Collaborative research across boundaries: mangrove ecosystem services and poverty traps as ca coupled natural-human systems," in *collaboration across Boundaries for social-ecological systems science*. Cham, Switzerland: Palgrave Macmillan, 115–152.

UN-REDD (2014). *Carbon pools and muliple benefits of mangroves in central Africa*. Nairobi, Kenya: UNEP.

UNEP (2011). *Economic analysis of mangrove Forests : a case study in gazi bay*. Nairobi, Kenya: UNEP.

Valiela, I., Bowen, J. L., and York, J. K. (2006). Mangrove forests: one of the world's threatened major tropical environments. *Bioscience* 51, 807. doi:10.1641/0006-3568(2001)051

Vasconcelos, R. P., Reis-Santos, P., Costa, M. J., and Cabral, H. N. (2011). Connectivity between estuaries and marine environment: integrating metrics to assess estuarine nursery function. *Ecol. Indicat.* 11, 1123–1133. doi:10.1016/j.ecolind.2010.12.012

Vihervaara, P., Mononen, L., Nedkov, S., and Viinikka, A. (2018). Biophysical mapping and assessment methods for ecosystem services: deliverable D3.3 EU Horizon 2020 ESMERALDA Project, Grant agreement. No. 642007.

Vihervaara, P., Mononen, L., Santos, F., Adamescu, M., Cazacu, C., Luque, S., et al. (2017). "Biophysical quantification," in *Mapping ecosystem services*. Editors B. Burkhard and J. Maes (Sofia, Bulgaria: Pensoft Publishers), 93–101.

Vitousek, P. M., and Sanford, R. L. (1986). Nutrient cycling in moist tropical forest. *Annu. Rev. Ecol. Systemat.* 17, 137–167. doi:10.1146/annurev.es.17.110186.001033

Vo, Q. T., Kuenzer, C., Vo, Q. M., Moder, F., and Oppelt, N. (2012). Review of valuation methods for mangrove ecosystem services. *Ecol. Indicat.* 23, 431–446. doi:10.1016/j.ecolind.2012.04.022

Wallace, K. J. (2007). Classification of ecosystem services: problems and solutions. *Biol. Conserv.* 139, 235–246. doi:10.1016/j.biocon.2007.07.015

Walters, B. B., Rönnbäck, P., Kovacs, J. M., Crona, B., Hussain, S. A., Badola, R., et al. (2008). Ethnobiology, socio-economics and management of mangrove forests: a review. *Aquat. Bot.* 89, 220–236. doi:10.1016/j.aquabot.2008.02.009

Wright, J. M. (1986). The ecology of fish occurring in shallow water creeks of a Nigerian mangrove swamp. *J. Fish. Biol.* 29, 431–441. doi:10.1111/j.1095-8649.1986.tb04958.x

Zedler, J. B., and Kercher, S. (2005). Wetland resources: status, trends, ecosystem services, and restorability. *Annu. Rev. Environ. Resour.* 30, 39–74. doi:10.1146/annurev.energy.30.050504.144248

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Assessing the effectiveness of management measures in the Ria Formosa coastal lagoon, Portugal

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The Ria Formosa is an important transitional and coastal lagoon on the south coast of Portugal that provides valuable ecosystem services. The lagoon is a protected area under national and international conventions. There is a great potential for Blue Growth sectors, such as aquaculture and coastal tourism, but these rely on good water quality. European environmental legislation, such as the Water Framework Directive, requires member states, such as Portugal to implement management measures if a surface water body is not of good ecological status. This work addresses the effectiveness of management measures, such as wastewater treatment plant implementation and dredging, on the water quality of the Ria Formosa coastal lagoon system. This is an important social-ecological issue, since management measures can be very expensive. The ecological status of Ria Formosa was evaluated, according to the physico-chemical and biological quality elements of the Water Framework Directive. The main indicators were the physico-chemical quality elements of nutrient and oxygen condition, and the biological quality element chlorophyll *a*, as a proxy for phytoplankton biomass, under the Water Framework Directive. The data for these quality elements from the Ria Formosa were analyzed for consistency with the classification for the Water Framework Directive water bodies. The data after the implementation of management measures was compared with historical data to evaluate if these measures had been effective. The relation between nutrient pressures, meteorological and hydrological conditions was addressed, especially rainfall and runoff. Results showed a decrease in nutrient concentration after the management interventions, despite the increase of population and intensifying agriculture in the catchment. The Ecological

Status is spatially variable with an overall moderate status, indicating the need for further management measures. There is a significant reduction in nutrient pressure on the lagoon during drought years. This indicates that climate change may alter the structure and function of the lagoon in the future.

KEYWORDS

Ria Formosa, coastal lagoon, assessment of management, Water Framework Directive, intercalibration sites

Introduction

An estimated 2.4 billion people live within 100km of the coast with 40% percent of the world's population living on only 4% the land area (United Nations [UN], 2017). This is partly because coastal ecosystems are among the most productive ecosystems in the world (Levin et al., 2001), providing 38% of the total value for ecosystem services (Costanza et al., 1997). For example, in Catalonia, 40.3% of the total annual flow of ecosystem service value is provided by the coastal zone, which accounts for only 22.2% of the total surface area (Brenner et al., 2010). Globally, fisheries and fish products provided 203 ± 34 million full-time equivalent jobs in 2011 (Teh and Sumaila, 2013). Thus, the long-term sustainability of coastal populations relies on ecosystems and the services they provide (Barbier et al., 2008), including the Ria Formosa lagoon in Portugal (Newton et al., 2018).

The growth of coastal populations is higher than the global average, with multiple human activities also increasing, such as fisheries, aquaculture, energy production, shipping, tourism and recreation. There are also increasing resulting pressures, such as inputs of effluents and runoff from agriculture, that become multi-stressors in the context of climate change (Guerry et al., 2012). An analysis of 12 temperate estuarine and coastal ecosystems in Europe, North America, and Australia (Lotze et al., 2006) showed that these have already been depleted of 90% of formerly important species, 65% of seagrass and wetland habitat has been destroyed, water quality is degraded, and there are accelerated species invasions. This is a result of the pressures on wetlands, such as coastal lagoons, at the global scale (Pérez-Ruzafa et al., 2019; Newton et al., 2020). Thus, successfully designing and implementing management measures in coastal lagoons is vital to secure the valuable ecosystem services that they provide. This is a challenging task given the strong and wide variety of human activities and resulting pressures on coastal lagoons.

The Water Framework Directive (WFD) is the EU legal framework for integrated river basin management, including transitional and coastal waters (European Community [EC], 2000). It requires member states, such as Portugal, to implement management measures, when a surface water body is not of good ecological status. This is an important social-ecological issue, since management measures, such as building and upgrading sewage treatment plants, can be very expensive. When such management decisions are taken, it is assumed that they will result in improved water quality. However, this may not be the case if inputs increase due to population increase or because of diffuse inputs from runoff. The hypothesis underlying this study is that management measures should improve the water quality of a water body. Although the study is not intended as an environmental report on WFD compliance, it is the legislative and regulatory context of the study.

The study site is the Ria Formosa coastal lagoon that extends 55km along the coast of southern Portugal (Newton et al., 2013). The catchment is approximately 864 km^2 (Duarte et al., 2008). The lagoon has local, national regional and international significance (Newton and Mudge, 2003; Newton et al., 2018). It is a Portuguese Natural Park, since 1987; a Special Protection Area of the EU Birds Directive, a site of Natura 2000 network (site code PTZPE0017); and the international Ramsar convention (site 212, since 1980). The Ria Formosa protects 121 species of the Nature Directives and protects 19 habitat types of the Habitats Directive, according to the European Nature Information System of the European Environment Agency.

The Ria Formosa is a well-studied system (Newton et al., 2013) with numerous publications detailing aspects of its hydrology (e.g., Roselli et al., 2013) and human pressures (Newton et al., 2020). Historical pressures on the Ria Formosa include damming of water courses, e.g., in the west Ancao Basin; consolidation of inlets, e.g., the main Faro channel inlet at Farol; saltmarsh reclamation for salt-extraction and aquaculture ponds as well as major infrastructures, such as the international airport of Faro and sewage treatment plants; and construction on the dunes (Newton et al., 2020). The human pressures are increasing with population increase in the Algarve, from ~395000 in the 2001 census to 467000 in the 2021 census (Intituto National de Estatistica portal consulted on 30/08/2022[1]). The main threats now include erosion, storm

1 https://www.ine.pt/xportal/xmain?xpgid=ine_main&xpid=INE

surges and overtopping (Carrasco et al., 2021; Domingues et al., 2021); climate change (Brito et al., 2012b; Rodrigues et al., 2021); flooding and sea level rise (Lopes et al., 2022); sewage (Cravo et al., 2022a); eutrophication and hypoxia (Cravo et al., 2020); pharmacological contamination (Cravo et al., 2022b) and chemical pollution (Silva et al., 2021).

The illegal harvesting of high-value species such as seahorses and sea-cucumber, destined for foreign markets such as Asia, relies on local knowledge. Macrobenthos and fish are Biological quality elements of the WFD, so the systematic and illegal removal of species, such as sea-cucumbers, and sea-horses degrades the ecological status.

Good water quality for the Ria Formosa is important as the ecosystem services of the lagoon (Newton et al., 2018) support valuable economic activities such as shellfish harvesting, fishing, water sports, beaches and eco-tourism. Several environmental management measures have been implemented over three decades to improve the water quality of the western part of the lagoon. These include the installation (1989) and subsequent upgrade (2009) of Urban Wastewater Treatment plants (UWWT) and dredging (June 1997) of the Ancão Inlet.

Physico-chemical quality elements (e.g., nutrients, dissolved oxygen) and phytoplankton are commonly used for classification of water bodies in coastal waters (SWD, 2019). However, the Ria Formosa does not strictly fit the WFD definition of a coastal or transitional water body because of its salinity regime. Ferreira et al. (2005, 2006) defined the typology for the Ria Formosa as "sheltered coastal water" and identified 5 distinct waters bodies within the lagoon. Studies indicate that the tidal water exchange at the inlets result in both import and export of nutrients to and from adjacent coastal waters (Newton and Mudge, 2005; Cravo et al., 2013, 2014, 2015; Cravo and Jacob, 2019; Rosa et al., 2019).

The EU does not have generic WFD protocols for ecological status assessment of surface water bodies. It is left to the EU member states, according to the subsidiarity principle in the EU, to develop their own WFD surveillance monitoring protocols for coastal and transitional waters, provided that these protocols comply with the overall WFD and are intercalibrated within the eco-regions.

The WFD monitoring strategy for the surface waters in Portugal was developed during 4 national projects (SWD, 2015). The criteria for the selection of the monitoring stations that were used to select the stations in this study were designed to detect temporal changes in ecological status due to natural or anthropogenic factors and implemented according to Decree Law n° 77/2006. An intercalibration exercise, led by the European Union Joint Research Centre (JRC), was reported by Portugal to the European Environment Agency (EEA) by the Instituto da Água (INAG; Newton et al., 2007). Furthermore, the results were subsequently published in scientific journals (Nobre et al., 2005; Ferreira et al., 2006, 2007; Loureiro et al., 2006; Goela et al., 2009; Brito et al., 2010).

The philosophy and hypothesis underlying management measures is that these management measures are effective for the improvement of water quality. This paper assesses the effectiveness of different management measures in terms of the improvements to the ecological status of the Ria Formosa coastal lagoon within the context of the WFD, accounting for the spatial and temporal variability of abiotic factors and based on 5 research questions:

1. Are the WFD intercalibration stations (Ponte and Ramalhete) and the reference station (Praia) significantly different with respect to physico-chemical variables?
2. Has the water quality improved in the western section of the Ria Formosa?
3. What is the Ecological Status of the Ria Formosa in the context of the WFD, based on the physico-chemical supporting quality elements nutrients and oxygen condition, and the biological quality element phytoplankton biomass, using chlorophyll *a* as a proxy?
4. What is the relationship between rainfall, runoff and nutrient pressures, as this maybe relevant in a changing climate?
5. Are there significant differences in water quality between the lagoon and ocean indicating whether there is an overall export or import of nutrients?

Materials and methods

The protocols for analyses of coastal and transitional waters were based on the Portuguese implementation of the Water Framework Directive, led by the Portuguese water institute INAG (Bettencourt et al., 2003; Ferreira et al., 2003, 2005).

Sampling and sample treatment

The location of the various sampling sites in the Ria Formosa are shown in **Figure 1**, with the most intensive campaigns occurring at the sites shown within the inset rectangle in the figure. The numbering and names of the stations have varied in different projects, but the locations did not vary over time. Some stations were sampled throughout the period and some more sporadically. The GIS maps and data pooling were based on the coordinates of the stations.

The sampling and sample treatment are summarized below, with much more detailed descriptions provided in the relevant papers (see Introduction). Sampling and sample treatment was generally consistent over the various sampling campaigns between 1984 and 2012. Field *in situ* determinations included water temperature and salinity that were recorded in the field using a calibrated WTW® salinity/conductivity probe.

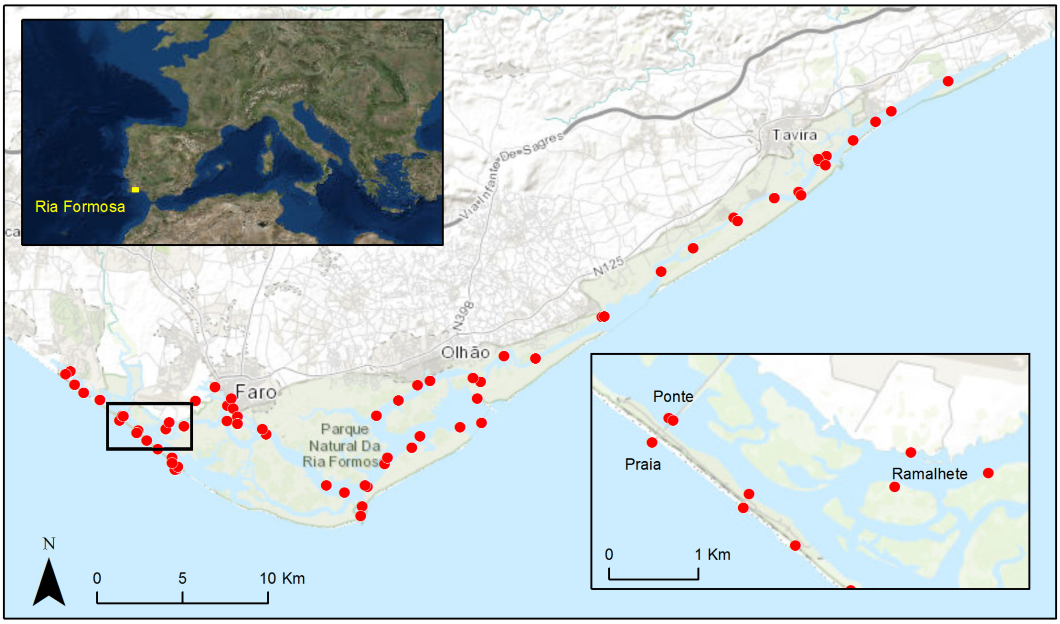

FIGURE 1
Map of the Ria Formosa lagoon, Portugal showing the main sampling stations. Inset _top left: geographical location of the lagoon; inset_bottom right: expansion of the rectangle at the western lagoon showing more intensively sampled stations, particularly Praia, Ponte and Ramalhete for the Water Framework Directive intercalibration.

Oxygen samples were fixed *in situ* for subsequent Winkler titration, modified by Grasshoff et al. (1999). Water samples of approximately 200 ml were collected for the analysis of nutrients (ammonium, nitrites, nitrates, phosphates, silicates); samples were stored in cool, dark boxes and subsequently frozen at −18°C in the laboratory until further analysis. Samples of 1 dm^3 of water were filtered through GF/F (47mm) filters and were kept frozen (−18°C) until further analysis for Chl*a*.

Laboratory determination

Nutrients were determined by molecular absorption spectrophotometry, following Grasshoff et al. (1999). For the determination of Chl*a*, extraction was carried out in 90% acetone for a period of 10–12h, at −18°C. The spectrophotometric determination was carried out at 665 nm, following Lorenzen (1967) equations. Some equipment (e.g., spectrophotomer) was substituted over the period.

Database

Data collected between 1984 and 2012 in the Ria Formosa lagoon were uploaded into a water quality management database, BarcaWin2000 (see: http://www.barcawin.com). This database includes the results of the various research projects conducted in the Ria Formosa coastal lagoon over the last decades, in a joint effort to enable long term water quality data assessment, including nutrients, oxygen and chlorophyll *a*. Depending on the objective of the past projects, samples were collected at several stations, in various tidal conditions and with different periodicities, especially up until 2006. From this period onward, samples were mostly collected at low water, twice per month until May 2010 and once per month until August 2012. The subsets of data that were analyzed to address each research question are specified in the corresponding result sections. All the data used in this study were obtained from BarcaWin2000 database, except for nitrates by the CIMA laboratory from 2006 (see **Figures 2–5** and **Tables 1–3**).

Statistical analyses

All the statistical analyses were performed using R (R Core Team, 2020). All the variables were standardized and redundant information was filtered through the elimination of linearly correlated variables. A Pearson's correlation test was performed, and when two variables had a correlation coefficient higher than 0.95 only one of them was selected. A principal components analysis (PCA) was performed using the "prcomp" function in R (R Core Team, 2020) to characterize the inter-annual variability of the water chemistry and to analyze which of the analyzed variables were explaining a higher percentage of total variance. ANOVA tests were used to evaluate differences in the mean of single variables

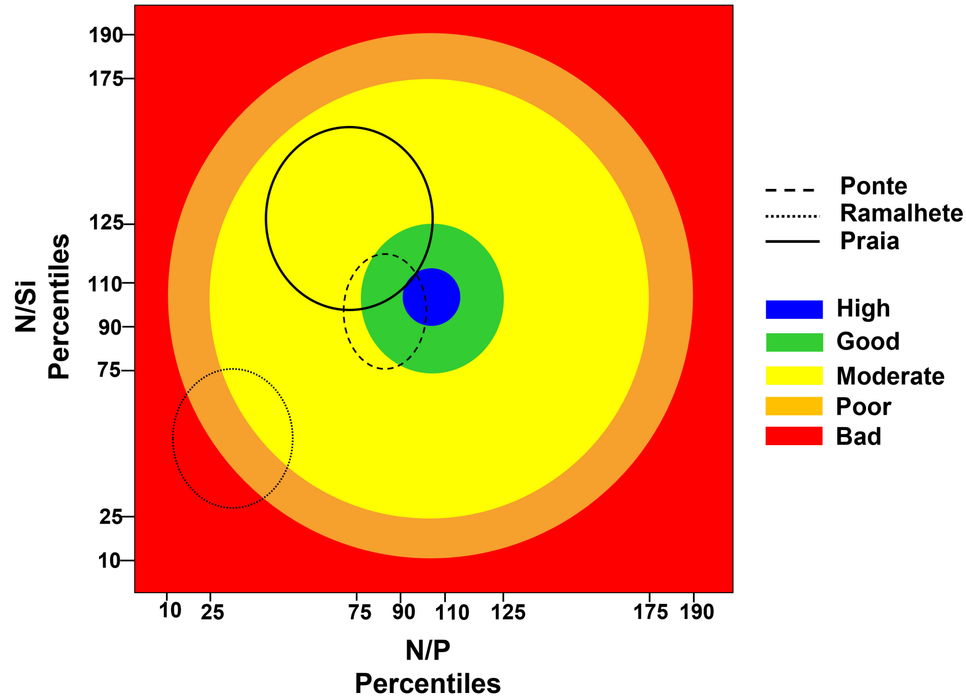

FIGURE 2
N:P and N:Si calculated from the data for the three stations Praia, Ponte and Ramalhete that were used in the boundaries intercalibration exercise for the Water Framework Directive.

(e.g., phosphate). Tukey's pairwise tests were performed to compare differences in the mean of single variables simultaneously with the set of all pairwise comparisons, i.e., groups of sites. PERMANOVA (Anderson, 2001) was used to evaluate differences in the multivariate data set (i.e., distance matrices including all water chemistry variables) between different groups of sites using the "adonis" function in the vegan package (Oksanen et al., 2020) and the "pairwise.adonis" function in the pairwise.adonis package (Arbizu, 2017).

Spatial modeling

The data were aggregated throughout the study period and the mean concentration value was calculated for each parameter and station. An inverse distance weighted (IDW) interpolation was used to map the concentration of each parameter in the study area, as described in Nobre et al. (2005). Concentrations were then reclassified into the five Water Framework Directive classes establishing quality boundaries according to data distribution as follows: High/Good = 0.9 percentile; Good/Moderate = 0.75 percentile; Moderate/Poor = 0.25 percentile; Poor/Bad = 0.1 percentile. In the case of dissolved oxygen, the optimum value is 100% and deviations below that value are considered as signals of deterioration in water quality. However, supersaturation (values much higher than 100%) may also be attributed to eutrophication during daylight hours, with the data split into concentrations below 100% and those above 100%: the percentiles to set quality boundaries were calculated for both data sets. For example, the High/Good boundary was set as the 0.9 percentile of both values below 100% and values above 100%. The same applies for the N:P ratio, which has an optimum value of 16, and the N:Si ratio, which has an optimum value of ∼1 (Redfield, 1958; Tett and Lee, 2005).

All data were re-projected using the statistical mapping system ETRS89-LAEA and then analysed using the R software (R Core Team, 2020) together with the gstat package (Pebesma, 2004).

Boundary setting

All the data were used to set the boundaries by calculating the deviations above and below the reference N:P and N:Si ratios (16:1 and 1:1, respectively), separating the data into four sets (N:P < 16; N:P > 16; N:Si > 1; N:Si < 1); the percentiles 0–10,10–25, 25–75, 75–90, 90–100 were used to set the H-High, G-Good, M-Moderate, P-Poor and B-Bad

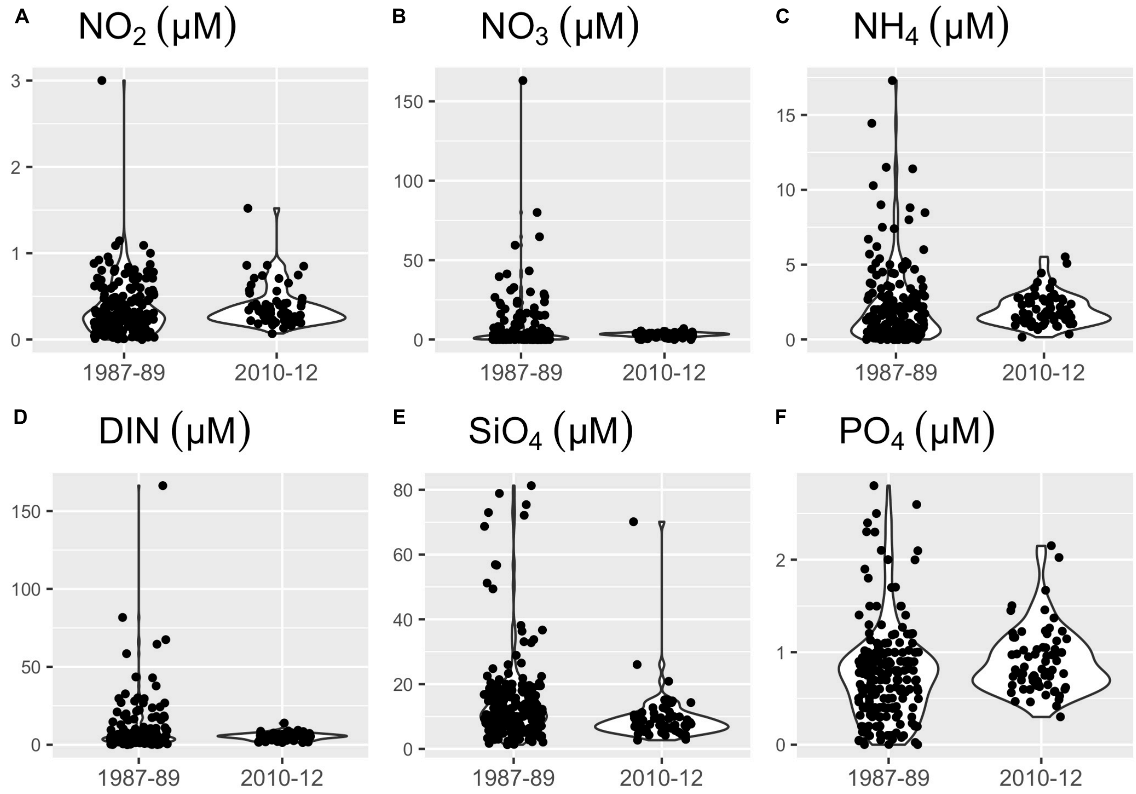

FIGURE 3
Violin plots of nutrient concentrations in micromolar (μM) comparing the two subsets of data: 1987–1989 prior to the interventions, and 2010–2012 after the latest UWWT plant upgrade. **(A)** Nitrite (NO_2); **(B)** Nitrate (NO_3); **(C)** Ammonia (NH_4); **(D)** Dissolved Inorganic Nitrogen (DIN); **(E)** Silicate (SiO_4); **(F)** Orthophosphate (PO_4) from Ramalhete and Ponte in the western lagoon. Raw data are shown as black dots.

boundaries (Table 1). For example, all N:Si values below the 10th percentile of the N:Si < 1 data set and above the 90th of the N:Si > 1 data set corresponded to a bad status for the N:Si ratio. Once the boundaries were set, two of the Ria Formosa stations were used for 'sheltered, coastal waters'. The stations with the longest data sets had been selected as sites for the EU-wide intercalibration exercise (between Member States) that was part of the WFD implementation. Two of the Ria Formosa stations (Ponte and Ramalhete) were used to set the boundaries between High – Good and Good – Moderate conditions. A station outside the lagoon (Praia) was selected for the type specific reference conditions (High). The relevant subset of 948 samples from the database was analyzed.

The mean N:P and N:Si and the mean standard deviation for each group of stations was calculated together with their corresponding percentile, i.e., their position in the percentile ranges calculated earlier using all the data, to assess their nutrient status. Percentile ranges were calculated using the "quantile" function and the correspondence between N:P, N:Si and percentiles calculated using an "empirical cumulative distribution" function.

Results

Question 1: Are the Water Framework Directive intercalibration stations (Ponte and Ramalhete) and the reference station (Praia) significantly different with respect to physico-chemical variables?

A PERMANOVA routine was performed to test the simultaneous response of the variables to the factor Ecological Status of the three stations "Ponte, Ramalhete and Praia," given as: High (H) = Praia; Good (G) = Ponte at Low Water; Moderate (M) = Ramalhete at Low Water. The results of the PERMANOVA show that there are significant differences between the Ecological Status of the 3 stations, ($F = 9.42$; $p = 0.001$). This is consistent with the proposed classifications of High (Praia), Good (Ponte) and Moderate (Ramalhete) status. However, the pairwise comparisons that the differences between categories were significant when comparing High (Praia) with Good (Ponte) and

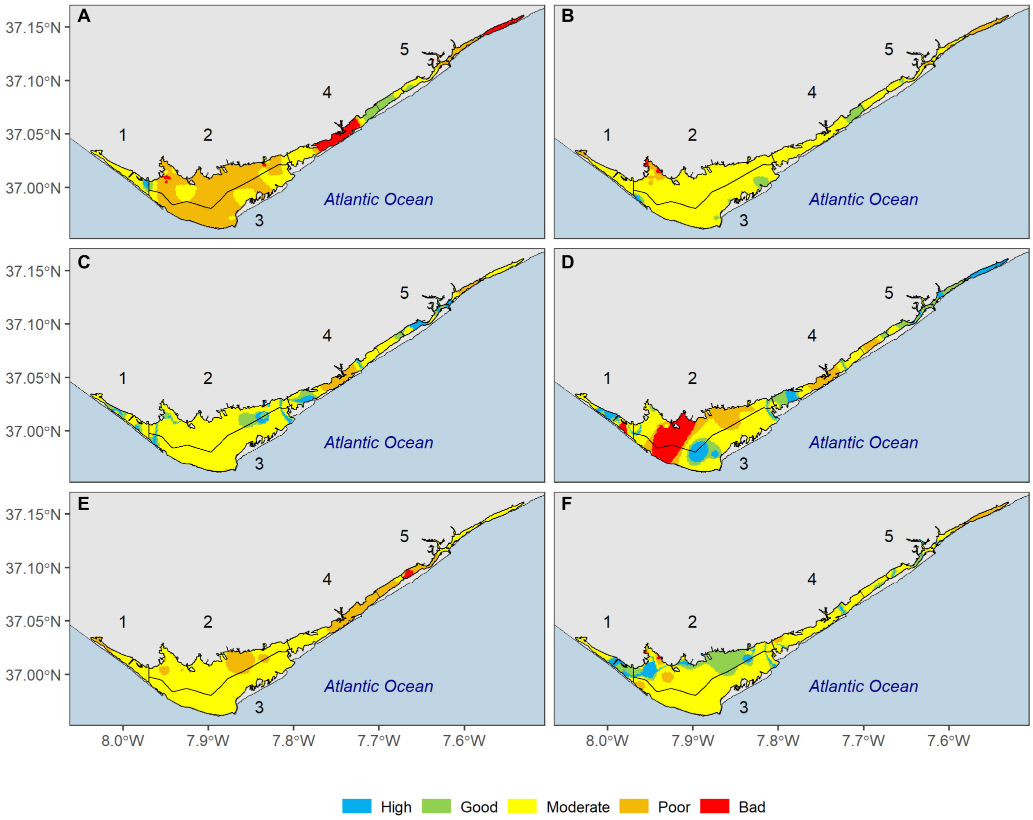

FIGURE 4
Spatial distribution of Ecological Status categories: **(A)** DIN-Dissolved Inorganic Nitrogen (Ammonium, Nitrite and Nitrate); **(B)** DIP-Dissolved Orthophosphate; **(C)** N:P; **(D)** N:Si; **(E)** Chlorophyll *a*; **(F)** Dissolved Oxygen (% saturation). The numbered polygons are the proposed water bodies from Ferreira et al. (2006).

(Moderate) conditions (p-adjusted values = 0.03), but not when comparing Good and Moderate conditions (p-adjusted value = 0.228).

The distribution of the N:P and N:Si data for these stations is shown in Figure 2 to determine "Nutrient Condition," a a Physico-Chemical Quality Element of the WFD. The size of the circles (line, dashed line, and dotted line) for the 3 stations represents 1 mean absolute deviation from the mean, which lies at the center of the respective circle. The distance of the center of each circle from the (blue) target that represents the reference condition indicates the deviation from the reference condition. This deviation is lowest for Ponte (dashed-line) and highest for Ramalhete (dotted line). The center of the circle for Ponte (dashed-line) lies on the N:Si reference condition axis and is displaced with respect to the N:P reference indicating low N:P ratios. The Ramalhete (dotted line) deviates to the lower left quadrant of the figure, consistent with lower N:P and lower N:Si than the reference condition. The Praia circle (line) deviates to the upper left quadrant, indicating higher N:Si and lower N:P. The circle for Ponte (dashed line) is the smallest, therefore indicating the least variation in the data, whereas the circle for Praia de Faro (line) is the largest indicating the most variation in the data. The results of this analysis are not consistent with the proposed classifications of High (Praia), Good (Ponte) and Moderate (Ramalhete) status.

Question 2: Has the water quality improved in the western section of the Ria Formosa since the 1980's and therefore have the management measures (urban wastewater treatment plants, new inlet, dredging) been effective for reducing nutrients?

The relevant subset of 225 samples for 1987–1989 and 69 samples for 2010-2012 was analyzed to address this question. Stations in the western part of the lagoon with long-term data sets (Ponte and Ramalhete) were considered. The two subsets of data were 1987–1989 prior to the interventions and 2010–2012, after the latest UWWT plant upgrade. The result of the comparison is shown in Figures 3A–F. A comparison of the

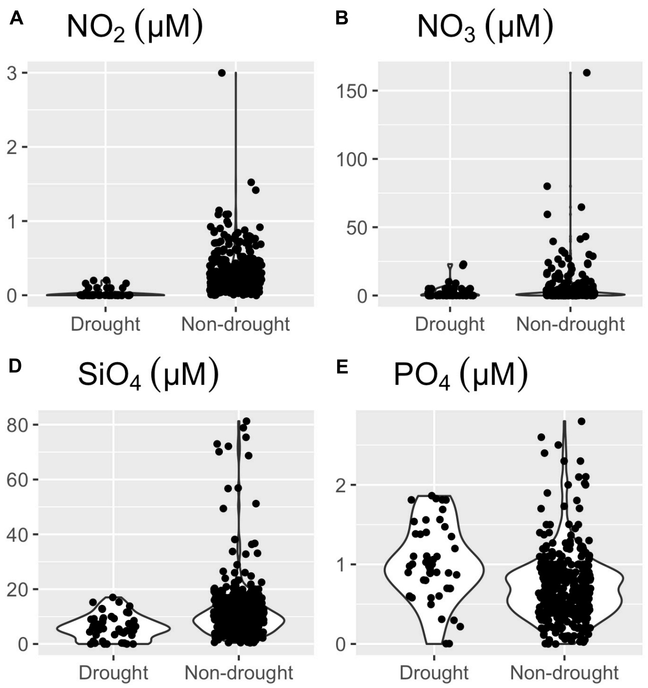

FIGURE 5
Violin plots of nutrient concentrations in micromolar (μM) comparing drought conditions (October 2004 – May 2005) to non-drought conditions (remaining data 1987–2012), showing the median, 95, 75, 25, and 5 percentiles for: **(A)** nitrite (NO_2); **(B)** nitrate (NO_3); **(C)** ammonia (NH_4); **(D)** silicate (SiO_4); and **(E)** orthophosphate (PO_4). Raw data are shown as black dots.

two data subsets shows a decreasing trend for the nutrients monitored, although it was only significant for dissolved inorganic nitrogen ($F = 4.97$, $p = 0.027$) and least for phosphate ($F = 4.28$, $p = 0.039$). Overall, the results show that the combined effect of the management interventions has decreased nutrient pressures to the lagoon, despite increasing resident population and intensification of agriculture in the catchment over the same period.

Question 3: What is the ecological status of the Ria Formosa in the context of the Water Framework Directive, based on the physico-chemical supporting quality elements nutrients and oxygen condition, and the biological quality element phytoplankton biomass using chlorophyll *a* as a proxy?

The results of the boundary setting calculations (High to Bad) are shown in Table 1 and in Figures 4A–F. The maps show the data analysis using the calculated boundary settings, for DIN- Dissolved Inorganic Nitrogen (Figure 4A), DIP-Dissolved Inorganic Phosphate (Figure 4B), N:P (Figure 4C), N:Si (Figure 4D), Chlorophyll *a* (Figure 4E), and Dissolved Oxygen% saturation (Figure 4F).

Figure 4A shows that the concentrations of DIN are elevated throughout the lagoon (moderate to bad), especially elevated in the channel between Olhão and Tavira (bad) and poor in the Faro-Ramalhete area. The phosphate concentrations (Figure 4B) were elevated throughout most of the lagoon (moderate) and the effect of the inlets (good) is clearly seen in green.

Figures 4C,D show the distribution with respect to the stoichiometry of the nutrients (N:P:Si). The colors are based on the Redfield ratio as a reference condition (high), which should be 16:1 for N:P and ~1:1 for N: Si (Tett and Lee, 2005). Deviation from these ratios (both above and below) are represented by the other colors (good, moderate, poor, bad). The N:P distribution (Figure 4C) shows that the N:P ratio is only close to balanced (green) in the Ancão basin and near Armona. The channel between Olhão and Tavira is poor, consistent with the elevated DIN concentrations. The N:Si distribution (Figure 4D) shows that the N:Si ratio is balanced or close to balanced (green and blue) in the Ancão basin and near the main Faro channel. This most closely represents the water quality of the Ria Formosa. The Ancão and Cacela channel are high, as is the area around the main inlet. The well flushed part of the Faro channel is good, while there are problematic areas around Faro and between Olhão-Tavira.

TABLE 1 Showing the calculated High to Bad boundaries for the Water Framework Directive from all the available data.

	High	Good	Moderate	Poor	Bad
DIN (µM)	x < 0.71	0.71 < x < 1.61	1.61 < x < 7.81	7.81 < x < 16.54	x > 16.54
DIP (µM)	x < 0.11	0.11 < x < 0.25	0.25 < x < 0.93	0.93 < x < 1.44	x > 1.44
N:P ref con = 16	11.85 < x < 17.42	8.26 < x < 11.85; 17.42 < x < 19.87	2.35 < x < 8.26; 19.87 < x < 48.33	0.95 < x < 2.35; 48.33 < x < 98.46	x < 0.95; x > 98.46
N:Si ref con = 1	0.80 < x < 1.12	0.58 < x < 0.80; 1.12 < x < 1.36	0.17 < x < 0.58; 1.36 < x < 3.28	0.08 < x < 0.17; 3.28 < x < 6.36	x < 0.08; x > 6.36
Chl *a*	0.27 > x	0.27 < x < 0.53	0.53 < x < 1.86	1.86 < x < 2.99	x > 2.99
D.0.% sat.	98 < x < 104	94 < x 98; 104 < x < 109	70 < x < 94; 109 < x < 127	45 < x < 70; 127 < x < 140	x < 45; x > 140

The distribution of Chl*a* (**Figure 4E**) is moderate-poor. Higher concentrations were found in the inner parts of the lagoon between Faro and Olhão and in the Marim channel. The percentage saturation of oxygen (**Figure 4F**) is highly variable and mainly moderate. Areas around some of the inlets, although not the Farol inlet, are clearly visible as High-Good. Some of the inner parts of the lagoon have small hotspots that are bad but areas around Faro and Olhao are good and even high.

Question 4: What is the relationship between rainfall, runoff and the nutrient pressures?

The year 2004 was characterized by the lowest rainfall since 1931 (Instituto de Meteorologia [IM], 2004). Thus, the dataset for nutrient concentrations at Ponte from October 2004–May 2005 with a total number of 48 samples was compared to the rest of the data (1987–2012) to determine whether drought conditions were characterized by significantly lower nutrient concentrations (**Figures 5A–E**).

Drought conditions were associated with a significant decrease in silicate ($F = 12.56$; $p = 4.3 \times 10^{-4}$), nitrite ($F = 42.69$; $p = 1.7 \times 10^{-10}$) and ammonium ($F = 30.48$; $p = 5 \times 10^{-8}$) concentrations and a significant increase in phosphate. There was no significant change in nitrate.

Question 5: Are there significant differences in water quality between the lagoon and ocean indicating whether there is an overall export or import of nutrients?

In order to assess this question, data was split into three different groups: Ria Formosa (including the data from all the sampling stations within the lagoon with 2049 samples), inlets (including all the sampling stations at the inlets of the lagoon with a total of 237 samples) and offshore (including all the sampling stations located in the ocean with a total of 315 samples). Then Tukey's test was implemented to assess the differences in nutrient concentrations between all the possible combinations (**Tables 2, 3**).

The nutrient concentrations in the lagoon were always significantly different from the concentrations in the inlets (**Table 2**), being significantly higher in the lagoon (**Table 3**). When comparing the lagoon to the offshore waters, all the nutrient concentrations, except nitrite, were significantly higher in the lagoon. Finally, the inlets and the offshore waters only differed significantly in terms of phosphorus, which was significantly higher offshore. Thus, our results show that the lagoon is generally exporting nutrients into the coastal waters.

Discussion

Monitoring and assessment are necessary for compliance with national and international assessments. Although the WFD has been a major achievement for the protection of European water bodies and their associated biodiversity and ecosystem services, the management and monitoring of some water bodies will most likely require site-specific prediction systems (Hering et al., 2010). This is especially true for highly dynamic (in time and space) ecosystems like the Ria Formosa. The trophic state of the lagoon varies widely in space due to hydro-morphological conditions (e.g., water renewal), but also in time, due to a diversity of factors (e.g., climate). For example, the inner part of the lagoon, where water renewal tends to be low (Mudge et al., 2008), were the most eutrophic. This is in alignment with previous studies showing the importance of water renewal for the eutrophication of coastal lagoons (Tett et al., 2003; Roselli et al., 2013) and the high spatial heterogeneity of these ecosystems (Pérez-Ruzafa et al., 2007). It also supports the findings of Rosa et al. (2022).

Even the water bodies proposed by Ferreira et al. (2006) had heterogeneous chemical conditions, especially in the middle part of the lagoon. There are some mismatches between the

TABLE 2 Tukey's pairwise comparisons of the nutrients' concentrations in three different areas: the inside of the lagoon (Ria Formosa), the inlets that connect it with the sea (inlets) and the offshore waters (Offshore).

	NH_4^+		NO_2^-		NO_3^{2-}		PO_4^{-3}		SiO_4^{2-}	
	Diff	p	Diff	p	Diff	p	Diff	p	Diff	p
Ria Formosa – Inlets	1.31	***	0.07	**	1.64	0.19	0.25	***	7.95	*
Ria Formosa – Offshore	1.93	***	0.06	0.06	2.92	**	−0.15	**	10.6	***
Inlets – Offshore	−0.62	0.89	0.02	0.89	−1.28	0.58	0.40	***	−2.65	0.76

* = $0.01 < p < 0.05$; ** = $0.001 < p < 0.01$; *** = $p < 0.001$.

distribution of the data and the proposed water body boundaries (**Figure 12** in Ferreira et al., 2006). This could lead to the management implications in the context of the WFD and make it very difficult to reach the good ecological status. Thus, long-term chemical and biological data covering wide spatial gradients, including at least those sites with distinct hydro-morphological features, are needed to guarantee the health of dynamic ecosystems, especially in the context of climate change. This is extremely important given the great natural, economic and social capital of this lagoon (Cravo et al., 2022a).

The study defined site-specific water quality based on dissolved oxygen, chlorophyll *a* and nutrient categories according to available long-term data and using percentiles for setting boundaries. According to the results, most of the Ria Formosa is in a moderate state. This is not consistent with our perception of the lagoon, which is heavily influenced by seawater, with high water renewal rates and complex hydrodynamics (Cravo et al., 2013, 2014, 2015, 2022a; Cravo and Jacob, 2019; Verissimo et al., 2019). As an analogy, imagine a small class ($n = 15$) of very able students. If these students are graded using percentage, they may all score over 90%, clearly demonstrating that they are all HIGH scoring and excellent students. However, if they are graded using percentiles with A–E categories, the students achieving 90% or just above will receive an E grade. Thus, it is probable that the boundary setting was inappropriate because of the lack of a strong eutrophication gradient, i.e., most of the studied sites had low chlorophyll *a* and nutrient concentration. The importance of mixing and complex hydrodynamics make the assessment of treated wastewater disposal in the Ria Formosa coastal lagoon particularly challenging (Cravo et al., 2022a). Further studies on site-specific water quality for the Ria Formosa should explore new approaches to redefine the boundaries between water quality categories, e.g., by plotting nutrient data against different pressure variables (Cravo et al., 2015). Nevertheless, the results of this study support the findings of Rosa et al. (2022) of an overall improvement of water quality. The decrease trend in nitrogen concentrations is probably due to a decrease in diffuse pollution coming from agriculture. The surprising small increase trend in phosphate is probably related to lower suspended particles in the water, since phosphate tends to adsorb to particles (Xu et al., 2015), and benthic phosphate release (Murray et al., 2006).

The Ria Formosa is an important area for the culture of bivalves and these filter feeders rely on plankton productivity. In addition, it is important as a nursery for juvenile fish. Thus, the nutrient condition is important not only for compliance with the WFD but also for the dynamics of phytoplankton communities in the lagoon (Brito et al., 2012a), while chlorophyll *a* and dissolved oxygen are important for the bivalves and other consumers. Understanding the link between hydrodynamics, water quality and the delivery of valuable ecosystem services is key to this management approach (Newton et al., 2018; Pérez-Ruzafa et al., 2019; Cravo et al., 2022a).

A system approach to the management of the Ria Formosa, is necessary to preserve this valuable social-ecological system (Newton, 2012; Gari et al., 2014). Several environmental management measures have been implemented over three decades to improve the water quality of the western part of the lagoon. These include the installation (1989) and subsequent upgrade (2009) of UWWT plants, dredging (June 1997) and relocation of Ancão inlet. The investment in such high-cost infrastructure has probably contributed to the improvement in the water quality that is indicated by our data analysis.

TABLE 3 Mean values and standard deviation of nutrient concentrations in n three different areas: the inside of the lagoon (Ria Formosa), the inlets that connect it with the sea (Inlets) and the offshore waters (Offshore).

	NH_4^+		NO_2^-		NO_3^{2-}		PO_4^{-3}		SiO_4^{2-}	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Ria Formosa	2.50	3.55	0.29	0.37	5.33	14.45	0.74	0.67	13.40	47.21
Inlets	1.18	1.26	0.22	0.36	3.69	6.36	0.48	0.44	5.44	8.88
Offshore	0.56	0.55	0.23	0.30	2.40	2.13	0.89	1.40	2.79	3.62

Nevertheless, based on all the available data and on the "one out all out principle", most of the Ria is classified as "moderate" and less flushed areas around Faro, Olhão and the Olhão-Tavira channels show up as "poor" or "bad" nearest to Tavira. Again, this result is surprising, and probably an artifact of applying an inappropriate boundary setting.

Management measures such as damming, dredging and inlet modifications can alter the physiography and hydrology of coastal lagoons (Roselli et al., 2013) thereby also modifying their functioning, such as residence time and flushing. The opening of an artificial inlet (Dias et al., 2009) and its subsequent closure may have modified aspects of the lagoon dynamics and the water bodies.

Furthermore, lagoons are sentinel systems for climate change (Eisenreich, 2005), so the response of the lagoon to climate change (Brito et al., 2012b; Rodrigues et al., 2021) and climate variability (drought-wet years) is important. This knowledge is necessary to understand how climate change will influence important fish and shellfish resources in the Ria Formosa and the economic activities depending on them (Newton et al., 2018; Pérez-Ruzafa et al., 2019). Management can use this knowledge to develop tools and strategies that help fisheries, aquaculture sectors and local authorities to prepare for the adverse changes or future benefits of climate change (e.g., CERES[2]). Changes in the productivity and ecology of the lagoon would have important economic consequences for these aquatic industries and knowledge is necessary to assess exposure, vulnerability and adaptive capacity within the fisheries and aquaculture sectors in order to suggest viable solutions for aquatic food production industries. These economic sectors will have to handle risks and expected benefits from climate change. In particular, it may be possible to support the ecological intensification of aquaculture in the lagoon, with the dual objectives of increasing production and competitiveness of the industry, while ensuring sustainability and compliance with EU regulations on food safety and environment. However, eco-intensification of aquaculture is a transdisciplinary challenge that requires the integration of scientific and technical innovations, new policies and economic instruments, as well as the mitigation of social constraints (e.g., GAIN[3]).

Conclusion

This study assessed the effectiveness of management measures in the Ria Formosa by answering 5 research questions that are important for the adaptive management of this valuable coastal lagoon for Blue Growth in the context of climate change:

1. The WFD intercalibration stations (Ponte and Ramalhete) and the reference station (Praia) are significantly different with respect to physico-chemical variables. This environmental variability is typical of complex systems such as lagoons (Pérez-Ruzafa et al., 2019). This may be attributed to the classification of the Ria Formosa as a "sheltered coastal water" rather than a "transitional" water. Although the Ria Formosa does not have the salinity gradient required to be classified as a transitional water under the WFD, nevertheless it is a region of restricted exchange that is behaves more as a transitional than coastal water (Tett et al., 2003; Newton and Icely, 2006; Zaldívar et al., 2008).
2. The water quality in the western section of the Ria Formosa has improved, indicating that the management measures have been partially effective, with respect to nutrients and oxygen saturation. However, there have been parallel studies, with shorter time series, of microbial contamination, metals and pesticides that are also relevant to water quality (Bebianno et al., 2019; Moreira da Silva et al., 2019). Microbiological contamination is still a concern (Cravo et al., 2015, 2022a; Galvão et al., 2019) is further evidence of metal, pesticide and pharmaceutical contamination and pollution (Silva et al., 2021; Cravo et al., 2022b).
3. The Ecological Status of the Ria Formosa in the context of the WFD is very spatially variable, based both on the physico- chemical supporting quality elements nutrients and oxygen condition and the biological quality element phytoplankton biomass with chlorophyll *a* as a proxy. The assessment results, following the "one out, all out" principle classify most of the water as "moderate" indicating that further management measures may be necessary. However, we believe this maybe a consequence of boundary setting forcing 5 categories in a system that has excellent tidal flushing with high-good quality coastal water.
4. The spatial distribution of the data is not consistent with the WFD water bodies derived from Ferreira et al. (2006) methodology. This may be due to changes in spatial complexity, particularly, in the western part of the lagoon, where inlets have been relocated (Pérez-Ruzafa et al., 2019).
5. Rainfall and runoff significantly affect the nutrient pressures in the Ria Formosa lagoon so that there are significant decreases in nutrient pressure in drought years. Results from winter 2021–2022, that was a particularly severe drought are not yet available, but it is probable that nutrient pressure also decreased due to decreased runoff. This is relevant in the context of climate change, whether rainfall increases or decreases. Increased temperatures will significantly affect oxygen solubility and the probability of hypoxia will increase (Brito et al., 2012a).

2 www.ceresproject.eu
3 https://cordis.europa.eu/project/rcn/216474/factsheet/en

6. There are significant differences in water quality between the lagoon and ocean indicating that the lagoon both exports and imports nutrients at different times of the year (Newton and Mudge, 2005; Cravo et al., 2013, 2014, 2015; Cravo and Jacob, 2019; Rosa et al., 2019).
7. Ocean-color is increasingly used for chlorophyll *a* estimations. Indeed, some of the data sets included in the study were obtained for sea-truthing of ESA missions. There are also new opportunities for monitoring using new remote sensing products from ESA's Sentinel missions (Cristina et al., 2019; El Mahrad et al., 2020). However, the reliability of ocean-color products depends on sea-truthing. Establishing long-term baselines, especially for climate change studies.
8. Despite the considerable efforts with respect to the installation of sewers and sewage treatment, there are still reports of sewage contamination in urban areas such as Faro and Olhao affecting the water quality. This has been investigated using robotic cameras and is mainly due to 'illegal' linking of sewage to rainwater-drains, in some cases a long time ago and prior to records.
9. The baseline for Ecological Status assessments of the WFD are type-specific reference conditions. These are most useful to assess whether the management measures are resulting in improvements in the water quality and ecological status of a system. They are less useful to make comparisons with other systems within the country or other EU Member States even from the same eco-region because of differing protocols. Intercalibration is therefore especially important in trans-boundary waters. For example, it would not be coherent for a shared water body to be classed as 'moderate' on one side of an international border and 'high' on the other side. Using the same agreed methods for assessment is also important since, different assessment methods applied to the same data set can give very different results (Newton et al., 2003).

Finally, management measures have had a positive effect on the water quality of the lagoon. The management of the water quality and ecological status are now legally binding and non-compliance is subject to fines from the European Union under the WFD. The 2019 EU "*fitness Check*" of the cluster of Water Directives concluded that they are "*overall fit for purpose, with some room for enhanced effectiveness*". The main shortcomings were attributed to insufficient implementation by Member States and Economic Sectors, such as agriculture. The findings of the present study indicate that some adjustments to the implementation may be necessary (conclusions 1, 3, 4, 8, 9). The EU "*fitness check*" also concluded that the WFD was "*future-proof*" and sufficiently flexible to accommodate new threats (e.g., pharmaceuticals, microplastics and climate change). The findings of the current study do support that there has been an overall improvement, but this may not continue with climate change, especially if temperatures increase (conclusion 5). Nonetheless, the continued threat from chemicals recognized in the EU "*fitness check*" is also true in the case of the Ria Formosa (Conclusion 2).

Data availability statement

The datasets generated for this study are available on request to the corresponding author.

Author contributions

AN: integrated her expertise on chemical oceanography and eutrophication and the implementation of the Water Framework Directive to contribute to all aspects of the article. MC-A: statistical analysis of the data and production of **Figures 2–5**, and writing and editing. DM: statistical analysis and production of the maps in **Figures 1**, **4**. PG: data collection and analysis, database management, and writing and editing. SC: data collection and analysis, database management, and draft writing and editing. MZ: data collection and analysis. JI: data collection, writing, editing, and preparation of the article for final submission. All authors contributed to the article and approved the submitted version.

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Conflict of interest

Author JI was a senior researcher with the research centre CIMA, at the University of Algarve, Portugal; he is also the co-owner of the Portuguese company Sagremarisco-Viveiros de Marisco Lda.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

Anderson, M. J. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecol.* 26, 32–46. doi: 10.1111/j.1442-9993.2001.01070.pp.x

Arbizu, P. (2017). Pairwise multilevel comparison using adonis. R package version 0.4.

Barbier, E. B., Koch, E. W., Silliman, B. R., Hacker, S. D., Wolanski, E., Primavera, J., et al. (2008). Coastal ecosystem-based management with nonlinear ecological functions and values. *Science* 319, 321–323. doi: 10.1126/science.1150349

Bebianno, M. J., Pedro, P., Serafim, A., Lopes, B., and Newton, A. (2019). "Human impact in the Ria Formosa," in *Ria Formosa: Challenges of a coastal lagoon in a changing environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Faro: University of Algarve), 109–123.

Bettencourt, A., Bricker, S. B., Ferreira, J. G., Franco, A., Marques, J. C., Melo, J. J., et al. (2003). *Typology and reference conditions for Portuguese transitional and coastal waters. Development of guidelines for the application of the European Union Water Framework Directive. INAG/IMAR, 2003.* Lisbon: INAG and IMAR.

Brenner, J., Jiménez, J. A., Sardá, R., and Garola, A. (2010). An assessment of the non-market value of the ecosystem services provided by the Catalan coastal zone Spain. *Ocean Coast. Manag.* 53, 27–38. doi: 10.1016/j.ocecoaman.2009.10.008

Brito, A. C., Newton, A., Tett, P., and Fernandes, T. F. (2012b). How will shallow coastal lagoons respond to climate change? A modelling investigation. *Estuar. Coast. Shelf Sci.* 112, 98–104. doi: 10.1016/j.ecss.2011.09.002

Brito, A. C., Quental, T., Coutinho, T. P., Branco, M. A., Falcão, M., Newton, A., et al. (2012a). Phytoplankton dynamics in southern Portuguese coastal lagoons during a discontinuous period of 40 years. *Estuar. Coast. Shelf Sci.* 110, 147–156. doi: 10.1016/j.ecss.2012.04.014

Brito, A., Newton, A., Tett, P., and Fernandes, T. F. (2010). Sediment and water nutrients and microalgae in a coastal shallow lagoon, Ria Formosa (Portugal): Implications for the Water Framework Directive. *J. Environ. Monitor.* 12, 318–328. doi: 10.1039/B909429F

Carrasco, A. R., Kombiadou, K., Amado, M., and Matias, A. (2021). Past and future marsh adaptation: Lessons learned from the Ria Formosa lagoon. *Sci. Total Environ.* 790:148082. doi: 10.1016/j.scitotenv.2021.148082

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., et al. (1997). The value of the world's ecosystem services and natural capital. *Nature* 387, 253–260. doi: 10.1038/387253a0

Cravo, A., and Jacob, J. (2019). "Role of the Ria Formosa inlets on the physical, chemical and biological exchanges with the adjoining ocean," in *Ria Formosa: Challenges of a coastal lagoon in a changing environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Lake Mary, FL: University of Algarve. Faro), 29–46.

Cravo, A., Barbosa, A., Correia, C., Matos, M., Caetano, S., Lima, M., et al. (2022a). Unravelling the effects of treated wastewater discharges on the water quality in a coastal lagoon system (Ria Formosa, South Portugal): Relevance of hydrodynamic conditions. *Mar. Pollut. Bull.* 174:113296. doi: 10.1016/j.marpolbul.2021.113296

Cravo, A., Cardeira, S., Pereira, C., Rosa, M., Alcântara, P., Madureira, M., et al. (2014). Exchanges of nutrients and chlorophyll a through two inlets of Ria Formosa, South of Portugal, during coastal upwelling events. *J. Sea Res.* 93, 63–74. doi: 10.1016/j.seares.2014.04.004

Cravo, A., Cardeira, S., Pereira, C., Rosa, M., Madureira, M., Rita, F., et al. (2013). Nutrients and particulate matter exchanges through the Ria Formosa coastal lagoon. *J. Coast. Res.* 65, 1999–2004. doi: 10.2112/SI65-338.1

Cravo, A., Fernandes, D., Damião, T., Pereira, C., and Reis, M. P. (2015). Determining the footprint of sewage discharges in a coastal lagoon in South-Western Europe. *Mar. Pollut. Bull.* 96, 197–209. doi: 10.1016/j.marpolbul.2015.05.029

Cravo, A., Rosa, A., Jacob, J., and Correia, C. (2020). Dissolved oxygen dynamics in Ria Formosa Lagoon (South Portugal)-A real time monitoring station observatory. *Mar. Chem.* 223:103806. doi: 10.1016/j.marchem.2020.103806

Cravo, A., Silva, S., Rodrigues, J., Cardoso, V. V., Benoliel, M. J., Correia, C., et al. (2022b). Understanding the bioaccumulation of pharmaceutical active compounds by clams *Ruditapes decussatus* exposed to a UWWTP discharge. *Environ. Res.* 208:112632. doi: 10.1016/j.envres.2021.112632

Cristina, S., Icely, J., and Newton, A. (2019). "The application of remote sensing for monitoring the Ria Formosa: the sentinel missions," in *Ria Formosa: Challenges of a coastal lagoon in a changing environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Lake Mary, FL: University of Algarve. Faro), 139–152.

Dias, J. M., Sousa, M. C., Bertin, X., Fortunato, A. B., and Oliveira, A. (2009). Numerical modeling of the impact of the Ancão Inlet relocation (Ria Formosa. Portugal). Environ. *Model Softw.* 24, 711–725. doi: 10.1016/j.envsoft.2008.10.017

Domingues, R. B., de Jesus, S. N., and Ferreira, Ó (2021). Place attachment, risk perception, and preparedness in a population exposed to coastal hazards: A case study in Faro Beach, southern Portugal. *Int J. Disast. Risk Reduct.* 60:102288. doi: 10.1016/j.ijdrr.2021.102288

Duarte, P., Azevedo, B., Guerreiro, M., Ribeiro, C., Bandeira, R., Pereira, A., et al. (2008). Biogeochemical modelling of Ria Formosa (South Portugal). *Hydrobiol.* 611, 115–132.

Eisenreich, S. J (ed.) (2005). *Climate change and the European water dimension - A report to the European Water Directors.* Ispra: Institute for Environment and Sustainability.

El Mahrad, B., Newton, A., Icely, J. D., Kacimi, I., Abalansa, S., and Snoussi, M. (2020). Contribution of remote sensing to holistic coastal and marine environmental management framework: Best practices and future potential. *Remote Sens* 12:2313. doi: 10.3390/rs12142313

European Community [EC] (2000). *Directive of the european parliament and of the council 2000/60/EC, Establishing a framework for community action in the field of water policy*. Maastricht: European Community , </UB>62.

Ferreira, J. G., Bettencourt, A., Bricker, S. B., Marques, J. C., Melo, J. J., Newton, A., et al. (2005). "Monitoring plan for water quality and ecology of Portuguese transitional and coastal waters," in *Development of guidelines for the application of the European Union Water Framework Directive*, 1.ed Edn, eds INAG –Water Institute and IMAR – Institute of Marine Research. Available online at: http://www.monae.org

Ferreira, J. G., Nobre, A. M., Simas, T. C., Silva, M. C., Newton, A., Bricker, S. B., et al. (2006). A methodology for defining homogeneous water bodies in estuaries-Application to the transitional systems of the EU Water Framework Directive. *Estuar. Coast. Shelf Sci.* 66, 468–482. doi: 10.1016/j.ecss.2005.09.016

Ferreira, J. G., Simas, T., Nobre, A., Silva, M. C., Schifferegger, K., and Lencart-Silva, J. (2003). "Identification of sensitive areas and vulnerable zones in transitional and coastal portuguese systems," in *Application of the united states national estuarine eutrophication assessment to the Minho, Lima, Douro, Ria de Aveiro, Mondego, Tagus, Sado, Mira, Ria Formosa and Guadiana systems*, (Lisbon: INAG/IMAR).

Ferreira, J. G., Vale, C., Soares, C. V., Salas, F., Stacey, P. E., Bricker, S. B., et al. (2007). Monitoring of coastal and transitional waters under the EU Water framework directive. *Env. Monit. Assess.* 135, 195–216. doi: 10.1007/s10661-007-9643-0

Galvão, H. M., Mendes, P. J., Caetano, S. M., Icely, J. D., and Newton, A. (2019). "Chapter 5 – Role of microbes in the Ria Formosa lagoon," in *Ria Formosa: Challenges of a coastal lagoon in a changing environment. 1st edition. University of Algarve*, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Lake Mary, FL: Faro), 67–81.

Gari, S. R., Newton, A., Icely, J., and Lowe, C. D. (2014). Testing the application of the Systems Approach Framework (SAF) for the management of eutrophication in the Ria Formosa. *Mar. Policy* 43, 40–45. doi: 10.1016/j.marpol.2013.03.017

Goela, P. C., Newton, A., Cristina, S., and Fragoso, B. (2009). Water Framework Directive implementation: Intercalibration exercise for biological quality elements – a case study for the South Coast of Portugal. *J. Coast. Res* SI, 1214–1218.

Grasshoff, K., Ehrhardt, M., and Kremling, K. (1999). *Methods of Seawater Analysis (3rd received and extended edition)*. Weinheim: Wiley-VCH. doi: 10.1002/9783527613984

Guerry, A. D., Ruckelshaus, M. H., Arkema, K. K., Bernhardt, J. R., Guannel, G., Kim, C. K., et al. (2012). Modeling benefits from nature: Using ecosystem services to inform coastal and marine spatial planning. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manag.* 8, 107–121. doi: 10.1073/pnas.1406483112

Hering, D., Borja, A., Carstensen, J., Carvalho, L., Elliott, M., Feld, C. K., et al. (2010). The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future. *Sci. Total Environ.* 408, 4007–4019. doi: 10.1016/j.scitotenv.2010.05.031

Instituto de Meteorologia [IM]. (2004). *Caracterizaçao climàtica 2004. Ministério da ciência, tecnologia, e ensino superior*. Lisbon: Instituto de Meteorologia.

Levin, L. A., Boesch, D. F., Covich, A., Dahm, C., Erséus, C., Ewel, K. C., et al. (2001). The function of marine critical transition zones and the importance of sediment biodiversity. *Ecosystems* 4, 430–451. doi: 10.1007/s10021-001-0021-4

Lopes, C. L., Sousa, M. C., Ribeiro, A., Pereira, H., Pinheiro, J. P., Vaz, L., et al. (2022). Evaluation of future estuarine floods in a sea level rise context. *Sci. Rep.* 12, 1–15. doi: 10.1038/s41598-022-12122-7

Lorenzen, C. J. (1967). Determination of chlorophyll and pheo-pigments: Spectrophotometric equations. *Limnol. Oceanogr.* 12, 343–346. doi: 10.4319/lo.1967.12.2.0343

Lotze, H. K., Lenihan, H. S., Bourque, B. J., Bradbury, R. H., Cooke, R. G., Kay, M. C., et al. (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312, 1806–1809. doi: 10.1126/science.1128035

Loureiro, S., Newton, A., and Icely, J. D. (2006). Boundary conditions for the European Water Framework Directive in the Ria Formosa lagoon, Portugal (physico-chemical and phytoplankton quality elements). *Estuar. Coast. Shelf Sci.* 67, 382–398. doi: 10.1016/j.ecss.2005.11.029

Moreira da Silva, M., Pedro, P., Duarte, D., and Chicharo, L. (2019). "Metal contamination in Ria Formosa saltmarsh, sediments and halophyte vegetation," in *Ria Formosa: Challenges of a coastal lagoon in a changing environment*, 1st Edn, eds J. Aníbal, A. Gomes, I. Mendes, and D. Moura (Faro: University of Algarve), 99–107.

Mudge, S. M., Icely, J. D., and Newton, A. (2008). Residence times in a hypersaline lagoon: Using salinity as a tracer. *Estuar. Coast. Shelf Sci.* 77, 278–284. doi: 10.1016/j.ecss.2007.09.032

Murray, L. G., Mudge, S. M., Newton, A., and Icely, J. D. (2006). The effect of benthic sediments on the dissolved nutrient concentrations and fluxes. *Biogeochem* 81, 159–178. doi: 10.1007/s10533-006-9034-6

Newton, A. (2012). A systems approach for sustainable development in coastal zones. *Ecol. Soc.* 17:41. doi: 10.5751/ES-04711-170341

Newton, A., and Icely, J. D. (2006). Oceanographic applications to eutrophication in tidal, coastal lagoons: The Ria Formosa, Portugal. *J. of Coast. Res.* 3, 1346–1350.

Newton, A., and Mudge, S. M. (2003). Temperature and salinity regimes in a shallow, mesotidal lagoon, the Ria Formosa, Portugal. *Estuar. Coast. Shelf Sci.* 57, 73–85. doi: 10.1016/S0272-7714(02)00332-3

Newton, A., and Mudge, S. M. (2005). Lagoon-sea exchanges, nutrient dynamics and water quality management of the Ria Formosa (Portugal). *Estuar. Coast. Shelf Sci.* 62, 405–414. doi: 10.1016/j.ecss.2004.09.005

Newton, A., Brito, A. C., Icely, J. D., Derolez, V., Clara, I., Angus, S., et al. (2018). Assessing, quantifying and valuing the ecosystem services of coastal lagoons. *J. Nat. Conserv.* 44, 50–65. doi: 10.1016/j.jnc.2018.02.009

Newton, A., Cristina, S., Guiomar, J., Goela, P. C., Fragoso, B., Marques, J. C., et al. (2007). "Intercalibration exercise of guidelines for the applications of the european union water framework Directive," in *Reference conditions and intercalibration of portuguese coastal waters*, eds IMAR – Institute of Marine Research and INAG (Lisbon: Water Institute).

Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A. C., Colijn, F., et al. (2013). An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. *Estuar. Coast. Shelf Sci.* 140, 95–122. doi: 10.1016/j.ecss.2013.05.023

Newton, A., Icely, J., Cristina, S., Perillo, G. M. E., Turner, R. E., Ashan, D., et al. (2020). Anthropogenic, direct pressures on coastal wetlands. *Front. Ecol. Evol.* 8:144. doi: 10.3389/fevo.2020.00144

Newton, A., Icely, J., Falcao, M., Nobre, A., Nunes, J., Ferreira, J. G., et al. (2003). Evaluation of eutrophication in the Ria Formosa coastal lagoon. Portugal. *Cont. Shelf Res.* 23, 1945–1961. doi: 10.1039/b909429f

Nobre, A. M., Ferreira, J. G., Newton, A., Simas, T., Icely, J. D., and Neves, R. (2005). Management of coastal eutrophication: Integration of field data, ecosystem-scale simulations and screening models. *J. Mar. Syst.* 56, 375–390. doi: 10.1016/j.jmarsys.2005.03.003

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., et al. (2020). *vegan: Community ecology package. R package version 2.5-7*. Available online at: https://CRAN.R-project.org/package=vegan

Pebesma, E. J. (2004). Multivariable geostatistics in S: The gstat package. *Comput. Geosci.* 30, 683–691. doi: 10.1016/j.cageo.2004.03.012

Pérez-Ruzafa, A., Pérez-Ruzafa, I.M., Newton, A., Marcos, C. (2019). Chapter 15. Coastal Lagoons: Environmental variability, ecosystem complexity, and goods and services uniformity in coast and estuaries, the future. Eds E. Wolanski, J. Day, M. Elliott, R. Ramachandran, 730. Amsterdam: Elsevier. doi: 10.1016/B978-0-12-814003-1.00015-0

Pérez-Ruzafa, A., Marcos, C., Pérez-Ruzafa, I. M., Barcala, E., Hegazi, M. I., and Quispe, J. (2007). Detecting changes resulting from human pressure in a naturally quick-changing and heterogeneous environment: Spatial and temporal scales of variability in coastal lagoons. *Estuar. Coast. Shelf Sci.* 75, 175–188. doi: 10.1016/j.ecss.2007.04.030

R Core Team. (2020). *R: A language and environment for statistical computing. R Foundation for Statistical Computing*. Vienna: R Core Team.

Redfield, A. C. (1958). The biological control of chemical factors in the environment. *Amer. Sci.* 46, 230A–221A.

Rodrigues, M., Rosa, A., Cravo, A., Jacob, J., and Fortunato, A. B. (2021). Effects of climate change and anthropogenic pressures in the water quality of a coastal lagoon (Ria Formosa. Portugal). *Sci. Total Env.* 780:146311. doi: 10.1016/j.scitotenv.2021.146311

Rosa, A., Cardeira, S., Pereira, C., Rosa, M., Madureira, M., Rita, F., et al. (2019). Temporal variability of the mass exchanges between the main inlet of Ria Formosa lagoon (southwestern Iberia) and the Atlantic Ocean. *Estuar. Coast. Shelf Sci.* 22:106349. doi: 10.1016/j.ecss.2019.106349

Rosa, A., Cravo, A., Jacob, J., and Correia, C. (2022). Water quality of a southwest Iberian coastal lagoon: Spatial and temporal variability. *Cont. Shelf Res.* 245:104804. doi: 10.1016/j.csr.2022.104804

Roselli, L., Cañedo-Arguelles, M., Goela, P. C., Cristina, S., Rieradevall, M., D'Adamo, R., et al. (2013). Do physiography and hydrology determine the physico-chemical properties and trophic status of coastal lagoons? A comparative approach. *Estuar. Coast. Shelf Sci.* 117, 29–36. doi: 10.1016/j.ecss.2012.09.014

Silva, M., Aníbal, J., Duarte, D. N., Veloso, N., Patrício, F., and Chicharo, L. (2021). Metals from human activities in a coastal Lagoon Saltmarsh-Sediment toxicity and phytoremediation by *Sarcocornia fruticosa*. *J. Environ. Protect. Ecol.* 22, 1441–1449.

SWD (2015). *Commission staff working document-report on the implementation of the water framework directive river basin management plans-member state: Portugal*. Brussels: European Commission, 82.

SWD (2019). *Commission Staff Working Document- Second River Basin Management Plans-Member State: Portugal*. Brussels: European Commission, 296

Teh, L. C., and Sumaila, U. R. (2013). Contribution of marine fisheries to worldwide employment. *Fish Fisher* 14, 77–88. doi: 10.1111/j.1467-2979.2011.00450.x

Tett, P., and Lee, J. Y. (2005). N: Si ratios and the 'balance of organisms': PROWQM simulations of the northern North Sea. *J. Sea Res.* 54, 70–91. doi: 10.1016/j.seares.2005.02.012

Tett, P., Gilpin, L., Svendsen, H., Erlandsson, C. P., Larsson, U., Kratzer, S., et al. (2003). Eutrophication and some European waters of restricted exchange. *Cont. Shelf Res.* 23, 1635–1671. doi: 10.1016/j.csr.2003.06.013

United Nations [UN] (2017). *People and oceans, the ocean conference - factsheet, 5-9 June*. New York, NY: United Nations.

Verissimo, F., Martins, F., and Janeiro, J. (2019). "Chapter 10:The role of Ria Formosa as a waste water receiver." In: Aníbal, J., Gomes, A., Mendes, I., and Moura, D. (eds) *Ria Formosa: challenges of a coastal lagoon in a changing environment* 1st edition. University of Algarve. Faro. 47–65.

Xu, H., Newton, A., Wolanski, E., and Chen, Z. (2015). The fate of phosphorus in Yangtze Estuary (Changjiang), China, under multi-stressors: Hindsight and forecast. Estuar. *Coast. Shelf Sci.* 163, 1–6. doi: 10.1016/j.ecss.2015.05.032

Zaldívar, J.-M., Cardoso, A. C., Viaroli, P., Newton, A., Wit, R., Ibañez, C., et al. (2008). Eutrophication in transitional waters: An overview (JRC-EU). *Transit. Waters Monograph.* 1, 1–78. doi: 10.1285/i18252273v2n1p1

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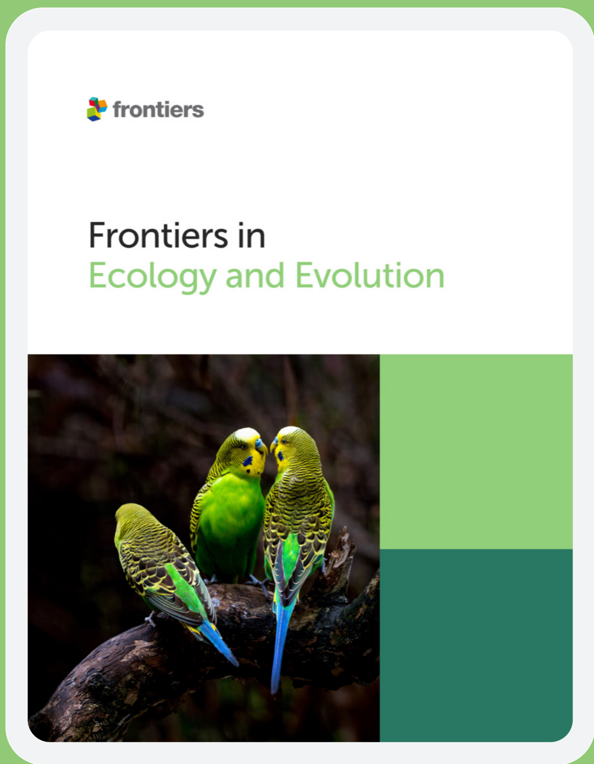

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