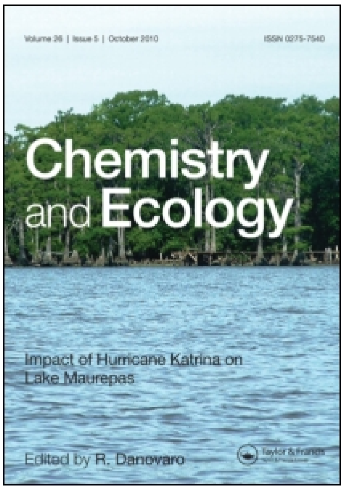

Chemistry and Ecology
Publication details, including instructions for authors and subscription information:
http://www.informaworld.com/smpp/title~content=t713455114

Trade-off between conservation and exploitation of the transitional water ecosystems of the northern Adriatic Sea

M. Abbiati[a]; M. Mistri[b]; M. Bartoli[c]; V. U. Ceccherelli[a]; M. A. Colangelo[a]; C. R. Ferrari[d]; G. Giordani[c]; C. Munari[b]; D. Nizzoli[c]; M. Ponti[a]; R. Rossi[b]; P. Viaroli[c]
[a] Centro Interdipartimentale di Ricerca per le Scienze Ambientali, Università di Bologna in Ravenna, Ravenna, Italy [b] Dipartimento di Biologia, Università di Ferrara, Ferrara, Italy [c] Dipartimento di Scienze Ambientali, Università di Parma, Parma, Italy [d] ARPA Emilia Romagna, Struttura Oceanografica Daphne, Cesenatico

Online publication date: 20 May 2010

To cite this Article Abbiati, M. , Mistri, M. , Bartoli, M. , Ceccherelli, V. U. , Colangelo, M. A. , Ferrari, C. R. , Giordani, G. , Munari, C. , Nizzoli, D. , Ponti, M. , Rossi, R. and Viaroli, P.(2010) 'Trade-off between conservation and exploitation of the transitional water ecosystems of the northern Adriatic Sea', Chemistry and Ecology, 26: 1, 105 — 119

To link to this Article: DOI: 10.1080/02757541003693193

URL: http://dx.doi.org/10.1080/02757541003693193

Chemistry and Ecology
Vol. 26, Supplement, June 2010, 105–119

Trade-off between conservation and exploitation of the transitional water ecosystems of the northern Adriatic Sea

M. Abbiati[a]*, M. Mistri[b], M. Bartoli[c], V.U. Ceccherelli[a], M.A. Colangelo[a], C.R. Ferrari[d], G. Giordani[c], C. Munari[b], D. Nizzoli[c], M. Ponti[a], R. Rossi[b] and P. Viaroli[c]

[a]*Centro Interdipartimentale di Ricerca per le Scienze Ambientali, Università di Bologna in Ravenna, Ravenna, Italy;* [b]*Dipartimento di Biologia, Università di Ferrara, Ferrara, Italy;* [c]*Dipartimento di Scienze Ambientali, Università di Parma, Parma, Italy;* [d]*ARPA Emilia Romagna, Struttura Oceanografica Daphne, Cesenatico*

(*Received 15 July 2009; final version received 19 January 2010*)

Transitional waters (TWs) provide ecosystem goods and services that are essential for the well-being of human populations. These unpredictable aquatic systems, characterised by large environmental fluctuations, are under severe stress due to human activities. Increasing pressures (e.g. over-harvesting, eutrophication, habitat loss) inevitably lead to the degradation of these ecosystems. Analysis of the complexity of species distribution patterns within and among TW habitats is relevant to understanding the underlying processes and promoting appropriate management strategies. Assessment of the trophic status is one of the most critical aspects of TWs. Untangling the relevance of anthropogenic nutrient inputs from internal biogeochemical processes is of primary importance in defining appropriate restoration strategies. Biotic indices have been suggested as an operational tool to assess environmental quality in TWs. However, the application in TWs of indices developed for coastal waters can give distorted results (e.g. low species diversity and high abundance are natural features). The BITS approach provides a rapid assessment of ecological quality, although its sensitivity in reflecting field conditions remains to be assessed. The major challenge to TWs management is to couple long-term conservation with productive activities. This goal can be achieved using an integrated approach, forecasting conservation of TW ecosystem functioning together with sustainable economic development. North-western Adriatic TW habitats have been exploited for centuries and major shifts in ecological processes have occurred. In this study, knowledge of the ecological features of these habitats is summarised and analysed using recent ecological tools. Based on these findings, possible strategies for conservative management have been discussed.

Keywords: transitional waters; Adriatic Sea; heterogeneity; trophic status; biotic indices; environmental quality; management and conservation; Water Framework Directive

1. The northern Adriatic lagoons

The northern Adriatic coastal area mainly consists of lagoon–river delta systems, hosting a large number of transitional water (TW) bodies (Figure 1), which differ in their environmental features. These habitats range from the largest and most studied Lagoon of Venice, to wetlands, estuaries, embayments and ponds, differing in their extent and connection with the sea. This article reviews

*Corresponding author. Email: marco.abbiati@unibo.it

ISSN 0275-7540 print/ISSN 1029-0370 online

DOI: 10.1080/02757541003693193
http://www.informaworld.com

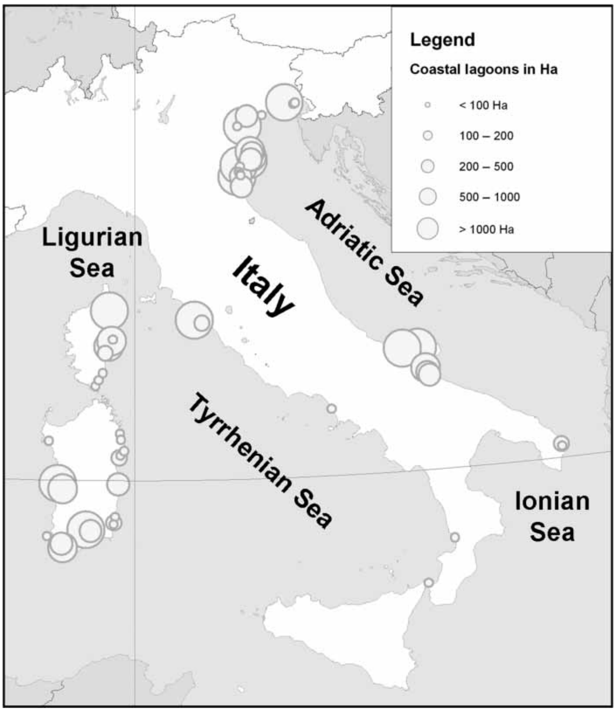

Figure 1. Geographical distribution and size of Italian coastal lagoons.

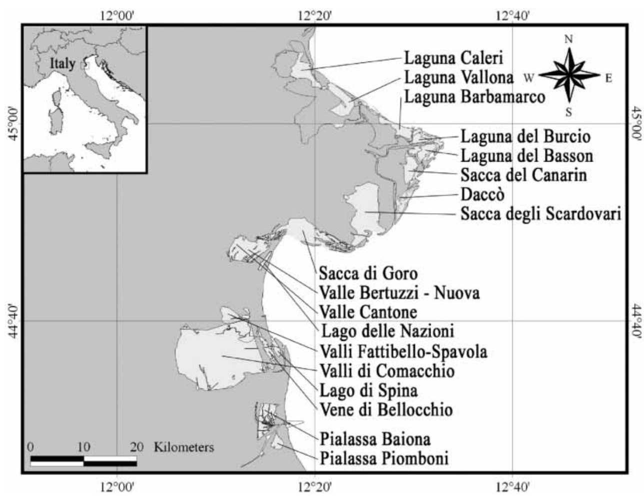

Figure 2. Geographical distribution of the transitional water ecosystems along the northern Adriatic coast of Italy, between the Po River Delta and Ravenna.

environmental conditions (quality or ecological status), focusing on the most critical features and on prospects for the conservation of TW ecosystems distributed between the Po River Delta and Ravenna harbour (Figure 2). Seventeen major TW ecosystems occur along this limited stretch of the Italian coast (∼100 km) (Table 1).

It is widely accepted that TWs are unpredictable aquatic systems characterised by large fluctuations in abiotic variables. Moreover, these habitats are very heterogeneous in terms of

Table 1. Transitional water ecosystems between the Po River Delta and Ravenna, major geographical and morphological features.

Name	Latitude	Longitude	Area (km^2)	Mean depth (m)	Salinity (psu)	Tide
Laguna Caleri	45.09° N	12.31° E	11.5	2.0	15–35	mt
Laguna Vallona	45.05° N	12.38° E	10.0	0.5–2	10–30	mt
Laguna Scannello – Barbamarco	45.00° N	12.46° E	8.0	0.8	15–30	mt
Laguna di Burcio	44.97° N	12.52° E	2.6	0.5	10–30	mt
Laguna di Basson	44.95° N	12.52° E	4.2	0.3–1	6–30	mt
Sacca del Canarin	44.92° N	12.49° E	10.0	0.5–1.8	14–32	mt
Daccò	44.88° N	12.48° E	2.7	na	na	mt
Sacca degli Scardovari	44.86° N	12.42° E	32.0	1.5	25–28	mt
Sacca di Goro	44.82° N	12.31° E	37.1	1.5	25–28	mt
Valle Bertuzzi – Nuova	44.78° N	12.21° E	14.1	0.6	15–30	nt
Lago delle Nazioni	44.77° N	12.24° E	0.9	2.5	15–30	nt
Valle Cantone	44.79° N	12.20° E	5.5	0.8	15–30	nt
Valli Fattibello – Spavola	44.68° N	12.19° E	5.5	1	12–40	mt
Valli di Comacchio	44.61° N	12.17° E	117.7	0.8–1.5	12–40	nt
Lago di Spina	44.63° N	12.26° E	1.3	na	na	nt
Vene di Bellocchio	44.62° N	12.26° E	1.6	na	na	nt
Pialassa Baiona	44.50° N	12.25° E	11.8	1.1	6–37	mt
Pialassa Piomboni	44.46° N	12.27° E	3.0	na	25–37	mt

Notes: na, Not available; mt, micro-tidal; nt, non-tidal.

physiography and hydrology (e.g. in extension, water depth, water exchange with the sea and salinity range), which are usually considered to be key features that explain a significant proportion of the environmental variability [1]. Habitat heterogeneity is a major constraint on any attempt to categorise these systems within strict typologies and no general consensus has been reached on their ecological classification [1–3]. Based on the recommendation of the European Union Water Framework Directive (WFD; 2000/60/EC), a classification scheme for the Mediterranean TWs has been suggested [1]. Following this scheme, which is still largely debated, TW habitats distributed between the Po River Delta and Ravenna fall into three major categories: small and large micro-tidal and non-tidal habitats (Table 1).

TWs provide essential ecosystem goods and services such as shoreline protection, fishery resources and improvements in water quality, and are also very productive ecosystems, providing habitats and food for migratory and resident animals. For millennia, northern Adriatic TW systems have been modified to fulfil human needs and exploited for fish farming. Nowadays, like many coastal ecosystems, they are under severe stress due to human activity and climate change [4]. The effects of human activities are mainly due to permanent and seasonal increases in population density, aquaculture, fisheries, agriculture and industry. These increasing pressures inevitably lead to environmental crises such as the overexploitation of resources, habitat destruction, contamination by pollutants from discharged urban and agricultural waste water, eutrophication, seasonal hypoxia/anoxia and the introduction of alien species [5].

All the TW habitats included in this study are heavily impacted by human activities and human alterations, but they differ markedly in use and the source of any threats. For example, the TW habitats of the Po River Delta are mainly open bodies receiving major nutrient inputs and undergoing intense harvesting of allochthonous clams (*Ruditapes philippinarum*), leading to major disturbances in the sedimentary environment. Comacchio lagoon is managed to maximise the productivity of fish farming. Pialassa Baiona and Pialassa Piomboni lagoons are mainly impacted by industry and harbour activities. Consequently, TWs along this stretch of the Italian coast provide a complex picture in terms of both habitat types and the impact of human disturbance.

The major aim of this article is to provide an overview of the status of TW habitats in the area, provide evidence of sources of disturbance and threats, comment on gaps in our knowledge and suggest an integrated approach to reconcile conflicting management goals.

2. Heterogeneity of species assemblages

TWs are very heterogeneous habitats, characterised by major variation in the abiotic and biotic features over a wide range of temporal and spatial scales [6]. Variability in physiographical and hydrological characteristics among lagoons, which is believed to drive the majority of the differences in patterns of species assemblages among habitats, has been largely addressed [1,7,8]. By contrast, variation in patterns of species distribution within lagoons has been attributed mainly to variation in salinity and confinement (i.e. isolation from the sea), which may be summarised as a seaward–landward gradient. However, it has been shown that patterns of variability among lagoons within a geographic region are often of the same magnitude as patterns of variability within single lagoons [2,9,10]. Drivers of the intra- and inter-lagoon habitat complexity may differ because small-scale variations in sediment grain size, organic matter content and river inflow, as well as different sources of anthropogenic disturbance affect the complexity of species distribution within habitats. Variability in connectivity patterns and species dispersal among and within habitats [11] may also deeply affect species distribution. Analysis of patterns of variability in the distribution of the biota within single lagoons is necessary to understand the underlying processes and promote appropriate management strategies; however, this aspect has received little attention.

The Pialassa Baiona lagoon is located between Ravenna harbour and the mouth of the Lamone River. Artificial embankments divide the lagoon into several shallow fresh- to salt-water ponds, connected by canals. Adjustable dams control the water exchange in some ponds. The inner canals converge in a main canal connected to the sea via the Candiano Canal (Ravenna harbour). The lagoon extends over an area of 11.8 km^2, including the embankments. Sediments are characterised by large areas of muddy bottoms with variable proportions of clay/silt and organic matter. Sandy sediments are abundant close to the relict dunes. Thanks to the relatively small size and complex patterns of natural and anthropogenic stressors, it provides a natural mesocosm to test theories of species distribution in TW habitats.

Pialassa Baiona, like most temperate coastal lagoons, is a heterogeneous environment, in which physical and biological variables change on different spatial and temporal scales. In several studies on the ecology and environmental assessment of the lagoon, patterns of variability in abiotic parameters and benthic assemblages have been investigated using multiscale sampling designs, covering scales from tens of metres to kilometres, and including hierarchical sampling within ponds and channels [9,12,13]. The underlying hypothesis of the studies was that major differences were expected between habitat types (ponds vs. channels), whereas within channels and ponds a major seaward–landward gradient would emerge. Most of the environmental parameters measured, however, significantly differentiated samples within habitats, at scales of metres. Analyses of the benthic assemblage showed high heterogeneity throughout the whole lagoon at spatial scales of hundreds of metres. Patterns of species distribution, species richness and evenness also showed the highest heterogeneity on a small spatial scale. Multivariate analysis of the overall benthic assemblages confirmed major heterogeneity on a small spatial scale [13,14], but also highlighted some differences between habitats (channels vs. ponds). The occurrence of weak pollution and confinement gradients emerged from the multivariate distribution pattern of macrobenthos, with a complex underlying pattern in species distribution.

Major sources of anthropogenic disturbance in this coastal lagoon are the input of waste water from urban and industrial sewage treatment plants and cooling waters from power plants, all located on the southern side of the lagoon. The set of environmental variables analysed provides an overall picture of the different physical and chemical factors that could affect and threaten the Pialassa Baiona [12–21]. Water surface temperature follows the dispersal of the thermal plume due to the two thermal power plants located on the southern side. Organic carbon content and the depth of the redox potential discontinuity (RPD), although influenced by grain size distribution, reflect the algal biomass accumulation related to eutrophication and dystrophic events. The intensity of the dystrophic crises, which often occur in the summer, as well as their effects on benthic macro-invertebrates, are greater in the southern areas of the lagoon [9].

Water flow and the distance from the confluence of the channels may influence water renewal, oxygenation, larval distribution, pollutant transport, re-suspension and the deposition of sediments. Southern areas are disturbed by thermal/chemical pollution and eutrophication, whereas reduced water flow allows the deposition of re-suspended sediments and pollutants in the northern areas.

A clear pattern of variability in species abundance was observed only for opportunistic species like the polychaetes *Streblospio shrubsolii* and *Capitella capitata*, which are more abundant in the southern polluted areas. By contrast, species sensitive to pollution and thermal stress [22], for example, the amphipod *Corophium insidiosum*, were more abundant in the northern part of the lagoon.

Although this coastal lagoon appeared to be a complex system, environmental variables and macro-benthic invertebrate assemblages showed high heterogeneity at the spatial scale, smaller than previously supposed, based on lagoon morphology and salinity gradients. Discrimination between two putative habitats, channels and ponds, is weakly supported by the patchy distribution patterns of invertebrate assemblages. Pollution affects mainly the abundance of sensitive species, as well as species richness [23]. In areas affected by intense anthropogenic disturbance, a reduction in the natural variability of the benthic assemblages has been observed, with increasing small spatial scale heterogeneity and strengthening of landward–seaward gradients.

3. Trophic status

3.1. *Eutrophication and changes in plant assemblages*

Coastal lagoons and TWs located along the northern Adriatic coast, south of Venice, are influenced by the Po River catchment area. Here, $\sim 5 \times 10^6$ pigs, 4×10^6 cattle and 16.5×10^6 humans account for $\sim 60 \times 10^6$ inhabitant equivalents. Overall, the Po plain contributes 35% of Italian agricultural gross production, 55% of animal production and 40% industrial production [24]. As a consequence, the Po River delivers 5–12,000 $t \cdot year^{-1}$ of total phosphorus and 75–170,000 $t \cdot year^{-1}$ of total nitrogen [25]. Total phosphorous levels peaked in the mid-1970s and decreased afterwards with the implementation of national standards for water protection and waste water treatment plants, although nitrogen has increased mainly in the last decade and is primarily from agricultural and zootechnical sources, which is a worldwide pattern [26].

Increasing nutrient loadings have been recognised as the main cause of eutrophication in coastal waters in the Emilia Romagna region [27], with severe impacts also in sheltered coastal lagoons and ponds where macroalgal blooms have often developed [28]. The resulting build-up of biomass is a major concern; with organic matter decomposition causing oxygen depletion and alterations in the biogeochemical cycles within lagoons [29]. Degenerative patterns in coastal lagoons result from a complex suite of interactions among systems geomorphology, hydrology, nutrient enrichment, internal biogeochemical processes, primary production and animal communities [30,31]. However, in shallow TWs, benthic communities and benthic biogeochemical processes are the major drivers of eutrophication processes. Here, submerged aquatic vegetation,

Table 2. Qualitative description of the evolution of benthic vegetation.

	Pristine conditions	1975–1997	Present
Sacca di Goro (restricted lagoon)	*Ruppia cirrhosa, Nanozostera noltii*	*Ulva* and *Gracilaria* blooms	Moderate growth of *Ulva* and *Gracilaria*
Valli di Comacchio (choked lagoon)	*Ruppia cirrhosa, Lamprothamnion papulosum*	Filamentous macroalgae blooms	Nanoplankton, cyanobacteria

Table 3. Main morphometric features, nutrient loadings and net ecosystem metabolism in three northern Adriatic lagoons with different benthic vegetation.

Lagoon	Surface area (km^2)	Mean depth (m)	Mean water retention time (days)	Dominant primary producer community	Dissolved inorganic phosphorus ($mmol \cdot m^{-2} \cdot year^{-1}$)	Dissolved inorganic nitrogen ($mmol \cdot m^{-2} \cdot year^{-1}$)	Net ecosystem metabolism ($mmol\,C \cdot m^{-2} \cdot year^{-1}$)
Sacca di Goro	26	1.00	3	Phytoplankton, macroalgae	45.2	6430	4198
Valli Comacchio	115	0.80	247	Phytoplankton	0.0	59	−40
Pialassa Baiona	11.8	0.75	3	Phytoplankton, macroalgae, rhyzophyte	291.3	6868	1117

namely phanerogams and macroalgae, and to lesser extent microphytobenthos, control overall primary production [32,33]. A combination of hydrological and geomorphic factors could greatly influence the primary production characteristics, leading to the dominance of phytoplankton and macroalgae in choked and restricted lagoons and of phanerogams and perennial macroalgae in leaky lagoons [34,35].

A qualitative description of changes in benthic vegetation in Sacca di Goro and Comacchio lagoons in recent decades is reported in Table 2. Relationships between N and P loadings and benthic vegetation are also reported in Table 3. Before the massive nutrient inputs, under pristine conditions with clear waters, perennial phanerogams formed extensive meadows of few species, namely *Zostera marina* and *Nanozostera noltii*, often associated with *Cymodocea nodosa* [36]. In choked lagoons, *Ruppia cirrhosa* was the dominant species, often associated with *Lamprothamium paplulosum* [37]. In the mid-1970s, increasing nutrient loadings favoured the development of bloom-forming ephemeral seaweeds such as *Gracilaria* spp., *Ulva* spp. and *Cladophora* spp., mainly in Goro and Pialassa lagoons [38,39]. In the Comacchio lagoon, transition from phanerogams to nanoplankton and cyanobacteria occurred rapidly, likely because of heavy exploitation of the aquatic ecosystem along with the higher degree of confinement and the very long hydraulic retention time [40].

Ulva spp. may attain biomass densities $>$10–20 kg fresh weight$\cdot m^{-2}$, because of transport by currents and accumulation. Drifted foliose thalli can be also stranded or deposited at the sediment surface where they cause profound alterations in biogeochemical processes.

In shallow environments dominated by macroalgae, e.g. in Sacca di Goro lagoon, there are wide fluctuations in oxygen concentrations [41,42]. Abnormal O_2 production is usually accompanied by the retention of labile organic matter at the sediment surface, where, under calm weather and high temperatures conditions, decomposition processes take place, leading to persistent hypoxia and often anoxia (Figure 3). Anoxia quickly leads to the onset of sulphate bacterial reduction which releases toxic sulphides primarily into the pore water, but also into the bottom waters [29,30]. Concerns over these issues have arisen in the last two decades, when a wealth of studies have been carried out mainly in the Sacca di Goro lagoon, which has became an

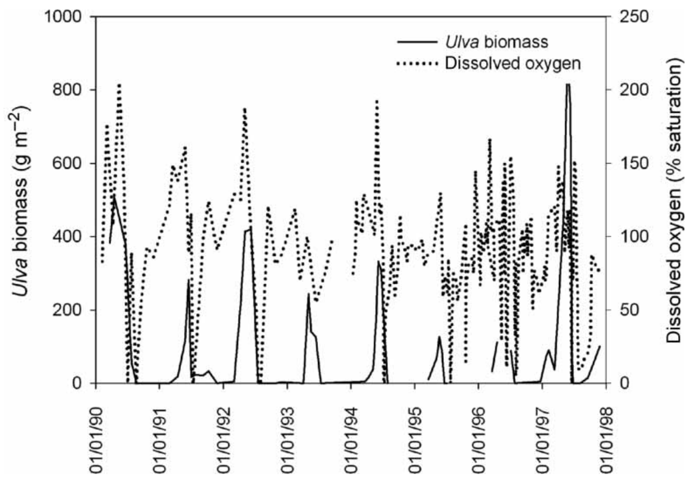

Figure 3. *Ulva* spp. biomass and dissolved oxygen in the water column from 1990 to 1997 in Sacca di Goro.

experimental field station for studies on eutrophication and its consequences for sediment biogeochemistry [28,29,43–49]. Nitrogen, phosphorus, iron and sulphur cycles have been studied in relation to different benthic vegetation communities. Key biogeochemical factors have been identified, namely the iron–sulphide–phosphorus and the carbonate–iron–phosphate buffer systems which have been proved to regulate sedimentary processes, benthic communities and the whole ecosystem functioning.

Processes of nitrogen cycling were also addressed, with awareness that nitrogen is a growing concern for marine coastal waters [26]. Within certain thresholds, healthy seagrass meadows are able to buffer nitrogen, and also regulate the whole ecosystem with balanced oxygen, low turbidity and low nutrient concentrations [49]. Increased nitrogen availability shifts the benthic community towards the dominance of bloom-forming nitrophilous species [28,36]. In phanerogam-dominated lagoons, oxygen transport to the rhizosphere can be viewed as the key function which controls meadow persistence, maintaining healthy conditions in highly reducing sediments. Floating macroalgae attenuate light penetration into the water column, thus depressing photosynthesis. In parallel, the oxic layer moves upwards, thus inducing bottom anoxia. The ultimate effect of such a perturbation is a shift from rhizophytes to pleustophytes, with the onset of anoxic and sulphidic conditions in the top sediment horizon, and even in the bottom water column. In both rhizophyte and floating macroalgae communities, nitrification and denitrification rates and their coupling are usually low, with potential nitrogen accumulation into the biomass bulk. Although phanerogam meadows are efficient traps for nitrogen through incorporation into refractory tissues and burial of plant detritus, macroalgae undergo rapid decomposition determining cyclic growth and decay phases which cause the whole ecosystem degeneration.

3.2. *Eutrophication and mollusc farming*

Paradoxically, eutrophication of lagoons in the Po River Delta provides a background for shellfish farming, which is the main revenue for the local economy. At present, the average crop is 15–18,000 t·year^{-1}, which is equivalent to a gross income of 50–100 million euros. Phytoplankton production supported by nutrients and suspended organic particulate matter inputs from rivers are the main trophic resources for bivalve filter-feeders. This way, bivalves can control the surrounding aquatic environment, modifying the particulate/dissolved nutrient ratio, changing sediment composition [50,51] and controlling benthic–pelagic coupling [52].

In farmed areas, shellfish densities are up to two orders of magnitude greater than in natural populations, which enhances deposition rates of organic matter by 10- to 100-fold [53]. Recent field studies have analysed the influence of clam and mussel farming in the Sacca di Goro lagoon [54–57]. Mussels are usually farmed in suspended cultures, which influence benthic fluxes only through biodeposition on the sediment. The mussel bulk on the ropes consumes oxygen and releases ammonium at very high rates, directly into the water column. The Manila clam (*Ruditapes philippinarum*) is an infaunal species which is buried completely in the sediment, where it causes particle reworking and sediment mixing and supports sediment oxygen consumption and recycling of ammonium, phosphorus and silica [54]. Overall, both mussels and clams stimulate benthic metabolism, but mussels tend to have a major impact on oxygen availability, microbial processes and benthic fluxes [58].

Perturbations induced by clam harvesting, have major impacts compared with mussel harvesting, because the buried clams can be harvested only by sediment dredging. Dredging causes the release of both dissolved and particulate matter into the water column with possible heavy impacts on water quality, namely increases in turbidity, sulphides and other reduced compounds, ammonium and dissolved inorganic phosphorus [58]. By contrast, harvesting contributes to sediment reoxidation and further stabilisation, which is beneficial to clam farming itself making the sediment healthier [59].

The sustainability of shellfish farming in the Sacca di Goro lagoon has been studied using different model tools which integrate hydrodynamics, ecological components and economic issues [60–62]. Sustainable standing crops do not have fixed boundaries, but rather depend on many variables: farmed biomass and the sensitivity of clam farming to *Ulva* blooms and anoxia are key issues in such evaluations [63,64]. In other words, in the Sacca di Goro and similar lagoons of the Po River Delta, aquaculture productivity cannot be maximised but can only be optimised, achieving a compromise between water quality and farming productivity, which are also interlinked in a feedback loop.

4. Ecological quality assessment

Although TWs can be exploited for economic gain, the European Council Habitats Directive (92/43/EEC) lists them as a ‘priority habitat type’. The recent WFD (2000/60/EC) requires the analysis of biological quality elements (BQEs) for an ecological classification of aquatic systems. Benthic invertebrates are one BQE, the others being phytoplankton, aquatic flora and ichthyofauna (for TWs only). Benthic invertebrates have been largely used as bio-indicators for marine monitoring because they respond rapidly to anthropogenic and natural stress. 2000/60/EC also recommends the development of biotic indices, which represent an extreme synthesis in information reduction. Indices are most straightforward and easy to present to potential end users. However, their responses may be seriously distorted by the lack of knowledge on species biology and ecology [65]. Although the absolute values of an individual biotic index may be specific to a given ecoregion, theories and methods used in the development should be applicable across systems. To date, most indices proposed for the implementation of 2000/60/EC are based on the Pearson–Rosenberg’s paradigm [23], which relies on the predictability of changes in the benthic community in relation to environmental disturbance (e.g. organic enrichment). However, the main problem with those indices, aimed at determining anthropogenic stress, is that they rely on an abundance of stress-tolerant species. However, their abundance may be also affected by natural environmental stressors such as those typically encountered in TWs. For example, many indices refer to anthropogenically organic-rich systems, whereas coastal lagoons and other transitional environments are naturally organically enriched systems [66].

The ecological quality of four lagoons of the Po River Delta was compared by applying several indices proposed within the framework of the 2000/60/EC. The AMBI [67] and M-AMBI [68] were calculated using the freeware program available at www.azti.es. The factorial analysis (M-AMBI) combines AMBI with the Shannon diversity (H) and species richness (S). For the M-AMBI calculation, the reference values for a ‘High’ status were set as: H = 3.93, S = 39, AMBI = 1.54 (sandy sites), and H = 3.0, S = 37, AMBI = 1.67 (muddy sites); reference values for a ‘Bad’ status were: H = 0, S = 0, AMBI = 6. The Benthic Index based on Taxonomic Sufficiency (BITS; specifically developed to assess environmental quality status in Adriatic coastal lagoons) was calculated following Mistri and Munari [69], using the freeware program available at www.bits.unife.it. To calculate the BENTIX index [70], the methodology available at http://www.hcmr.gr/listview3.php?id=1195 was used. The lagoons investigated were Sacca di Goro, Lago delle Nazioni, Valli di Comacchio and Sacca di Scardovari.

The Sacca di Goro (26 km^2) is a lentic, micro-tidal lagoon that receives nutrient-rich freshwater from the Po di Goro and the Po di Volano. Nowadays, the Sacca is the most important site for rearing the Manila clam (*Ruditapes philippinarum*) in Italy. The Manila clam was imported at the beginning of the 1980s, and local authorities set up aquaculture regimes which gave extremely good socio-economic returns (the Goro model) [71]. The Sacca di Goro is subjected to anthropogenic eutrophication, which causes extensive growth of opportunistic green macroalgae leading to summer anoxia and dystrophy [72].

Lago delle Nazioni (0.9 km^2) is a shallow, non-tidal coastal lagoon located in the south-eastern area of the Po River Delta. It is a remnant of the ancient Valle di Volano, one of a series of brackish embayment created in the late Middle Ages by repeated ingressions of marine waters into the ancient Po di Volano deltaic marshes. Subsequent reclamations deeply modified the lagoonal system. In the 1960s, following development of the tourist industry in the area, the lake was given over to recreational use. A tourist resort and facilities for summer water sports were built along the shore, and a two-lane motorway interrupted residual communication with the adjacent Valle Bertuzzi.

The non-tidal Valli di Comacchio are the largest lagoonal complex (∼100 km^2) of the Po River deltaic area and are located ∼40 km south of the city of Ferrara. Over the last 50 years this shallow-water ecosystem has suffered major anthropogenic impacts, from land reclamation to the contamination of the remaining areas. The Valli di Comacchio has always been an area of intense economic activity with extensive aquaculture exploitation and, by the early 1970s, eel aquaculture. Intensive aquaculture plants utilised one of the basins (Valle Magnavacca) as a receiver and self-purification basin for waste waters. By the mid-1980s, however, the productive activities and fisheries had collapsed [73]. During that period, the ecosystem of the Valli di Comacchio changed significantly at various levels, and a state of hypereutrophication of the lagoon in the early 1990s was documented by Sorokin et al. [74]. Unfortunately, only limited documentation on the changes is available, which makes it difficult to quantify them and evaluate their impact. In 1990, the utilisation of Valle Magnavacca as a receiver for waste waters from aquaculture was suspended.

The Sacca di Scardovari is a micro-tidal, large embayment (32 km^2) located between two branches of the Po River Delta. The lagoon is connected to the Adriatic Sea through a wide mouth that is partly obstructed by sand banks. It varies in depth from 0.5 to 2.8 m. Its northern area receives nutrient-rich agricultural run-offs, while the southern area hosts extensive bivalve cultures of mostly clams and mussels.

Despite the five-category water quality classification system requested by the WFD, for management purposes, the most important threshold is between Moderate (requiring investments to improve the status to Good) and Good (no investment needed). Use of the different indices gave a different pattern of overall ecological quality status (EcoQ) for the investigated sites. These results are summarised in Figure 4 which shows the percentage of stations for each class of ecological quality according to the benthic indices. In our study, taking into account the threshold between

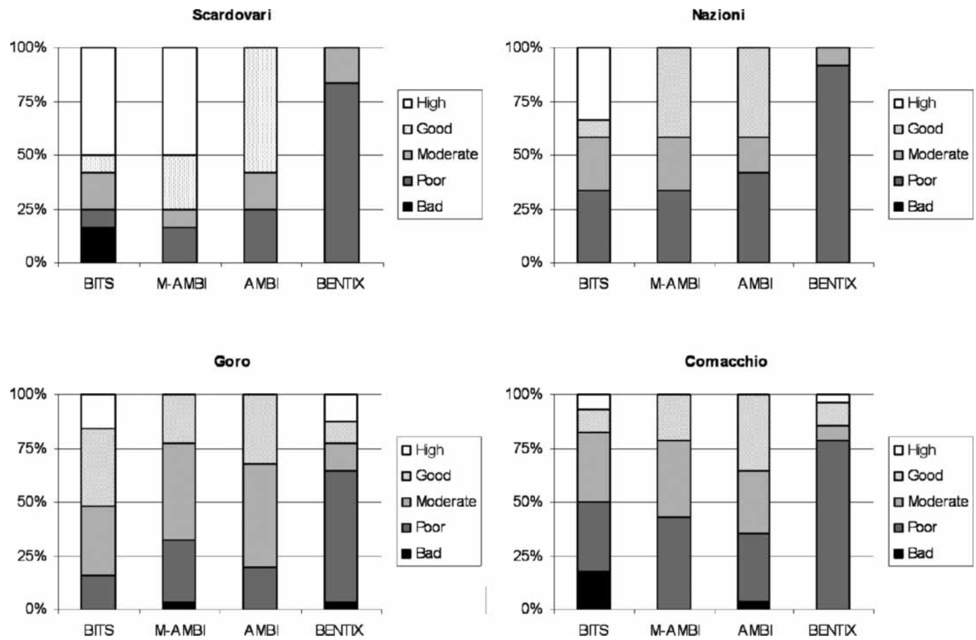

Figure 4. Ecological status of the lagoons Sacca di Goro, Lido delle Nazioni, Valli di Comacchio and Sacca degli Scardovari.

the classes Moderate and Good, it appears that the percentage of stations classed as 'Undegraded' EcoQ (High + Good) are, at Scardovari for example, 75% for M-AMBI, 58% for BITS and AMBI, and 0% for BENTIX. In the other lagoons different patterns were sometimes found (e.g. Goro, where the BITS gave 52% of undegraded sites, whereas AMBI gave 32% and M-AMBI and BENTIX 23%). By contrast, the classification of Lago delle Nazioni was identical using BITS, M-AMBI and AMBI, whereas BENTIX classified all stations as degraded. At Comacchio, BENTIX, M-AMBI and BITS classification was in accordance, although AMBI gave a higher percentage of undegraded sites (36%). As a general pattern, the classification of stations by the BENTIX index was always more severe with respect to the other three indices.

Ponti et al. [7], comparing several biotic indices in Mediterranean and Black Sea lagoons, found that the quality assessment provided by indices was not consistent within each investigated site. Simboura and Reizopoulou [75] demonstrated that in the Hellenic ecoregion, biotic indices performed differently on the water category scale (coastal, TWs), and those combining diversity measures were habitat-type specific. As a consequence, non-consistent responses by the different indices might lead to doubt in managers' minds regarding the value of the methods, and can produce confusion regarding whether remediation measures are needed. Finally, it should be remembered that the application of all those indices requires a high level of taxonomic expertise, and problems of species identification for research consultancies conducting environmental impact assessments do exist [76]. In the lagoons of the Po River Delta, organic enrichment, and thus oxygen depletion, can be intrinsic characteristics of the system, because of its physiography, for example, water circulation and geomorphology. In contrast to the sea, in lagoon environments disturbance events may also be represented by huge variations in salinity, because of, for example, evaporation or abrupt river discharges, and may be so common that benthic assemblages are constantly re-establishing [77]. Such frequent disturbances may cause periodic unselective reductions in benthic populations, resulting in taxonomic and functional spread [78], and allowing sensitive, indifferent and tolerant species to coexist with first- and second-order opportunists [79,80]. Superimposed on this diffuse disturbance, are acute episodes that allow only opportunists to occupy the habitat [77]. To complicate this pattern, each lagoon has its own biological features, which are the result of latitudinal variations in the extent of regional influence on local species richness [81]. In summary, hydrologic and geomorphic attributes, biogeographic constraints, diffuse and acute disturbances act synergistically to shape lagoon benthic communities. As disturbance increases, the adaptability of first the individual, and then species, genus, family and so on is exceeded: disturbance can be evidenced using higher level taxa, thus saving taxonomic effort. The relevance of higher taxonomic levels is thought to be a consequence of the hierarchical structure of biological responses to disturbance. This approach of higher taxa analysis could also overcome the problem–paradox of the natural occurrence of opportunistic species that the biotic indices face in TW, suggesting that a genus or family level of identification for the WFD implementation in the benthic compartment might be sufficient for evaluating the status of the transitional water bodies.

5. Perspectives

The WFD requires assessment of the ecological quality and vulnerability of TW ecosystems with the aim of implementing processes to achieve the conservation, management and recovery of ecosystem health. Biological quality elements (BQEs) represent a suitable set of basic variables, indicative of ecosystem properties and functions, which may be useful in evaluating the quality of the habitats within and among TW ecosystems. For BQEs to be effective, two conditions are required: suitable reference control sites and a good knowledge of the spatial and temporal variability patterns of BQEs. Criticalities in the identification of the reference conditions is a widely recognised issue [1,3,10,82], and the relevance of variability patterns is not fully understood. Planning monitoring and quality assessment activities without considering this aspect may lead to the collection of useless data or, conversely, to a waste of resources because of over-replication and over-sampling. Monitoring activities required by the implementation of the WFD have to be based on a correct sampling and measurements design. A framework for the experimental design can be provided by clear formulation of the hypothesis to be tested (regarding the spatial and temporal patterns and the process that are causing them). Often the sampling effort (in terms of number of spatial and/or temporal replications) is established on the basis of economical constrains or practical convenience. But this does not represent a scientifically sound approach to the problem.

Several biotic indices have been proposed for use in the assessment of environmental quality in TWs. The biotic indices aim to condense and synthesise wide ecological datasets into a single quality value, but this may be a strong limitation of the method. Most of the indices used for TWs were originally developed for coastal waters, considering their typical species diversity and density. However, TWs are characterised by lower species diversity and higher abundances compared with coastal habitats, therefore these indices may give a distorted indication of environmental quality. Moreover, the reliability of the biotic indices depends strongly on knowledge of the sensitivity of the single species. Development and validation of biotic indices require better understanding of the population distribution patterns and biotic assemblage dynamics in response to different natural or anthropogenic disturbances. Field and laboratory experiments, if adequately designed, could provide a relevant contribution to our knowledge of species sensitivity.

A disturbance common to most TW habitats is represented by increased nutrient loadings, causing eutrophication and hypoxic/anoxic crises. Anthropogenic nutrient loads, deriving primarily from agriculture and waste water treatment plants, cause a shift in primary producer assemblages. There have been major efforts in recent years to divert nutrient loads from TW to coastal water bodies to reverse the shifts that have occurred in recent decades. However, improving the environmental quality conflicts with the reduction in filter-feeder production and with shellfish farming. It should also be considered that, when anoxic crises occurs, they cause a serious economical loss to shellfish farmers. The challenge is to couple long-term conservation with exploitation activities carried out in TWs ecosystems. Management of TW habitats cannot focus only on improving the environmental (trophic) conditions, but should also consider the need to preserve economic activities that rely on the high productivity of TW systems. In this context, strong support is also required from ecological research. In parallel, sustainable management/exploitation can be achieved by involving local social and cultural communities to mitigate conflict.

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