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Measuring and managing changes in estuaries and lagoons: morphological and eco-toxicological aspects

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Just like any other ecosystem, coastal and estuarine ecosystems are complex and dynamic in terms of their species structure and functioning. The resilience of the different types of coastal ecosystem to change depends on the structural complexity of the different trophic relationships within the local food web, the connectivity between habitats and larger (eco)system units, and the ecological and biological strategies of the species they contain. There is evidence that changes in living systems may be irreversible and that the course of change is asymmetrical (Scheffer & Carpenter 2003). The path to structural complexity, maturity and climax for coastal systems is assumed to be long and slow, while the reverse is usually faster (Margalef, 1997). In this context, natural perturbations play a role in maintaining spatio-temporal heterogeneity in ecosystems, introducing more or less local and rapid fluctuations, which may either prevent the system from reaching steady state (Scheffer et al. 2003) or stimulate backward steps in the ecological succession, thus maintaining a certain level of youngness in the ecosystem. Transitional waters (TW), including part of the coastal waters, estuaries, coastal lagoons and brackish coastal inland waters, have the added difficulty of being naturally stressed – thanks to their strong spatio-temporal variability – and exposed to frequent environmental disturbances and fluctuations (Barnes, 1980; UNESCO, 1980, 1981; Kjerfve, 1994).

Measuring and managing changes in estuaries and lagoons is a complex task that must take into account the economic as well as the ecological system (Fig. 1). Relevant aspects of the latter are the environmental, geomorphological and biological heterogeneity of the transitional waters and the fact that the majority of the common biological indicators measure the degree of structure of a given assemblage but do not integrate structure and functions (de Jonge this issue). Furthermore, these indicators are unable to differentiate between human sources of stress and natural sources of variability and fluctuation.

The Water Framework Directive (WFD), recently introduced by the European Union, seeks to ensure the highest ecological status possible for water bodies, from rivers to seas, within EU borders (European Union, 2000). This should be achieved by optimising local habitat conditions and the chemical status of the different water bodies. The WFD stipulates that the development of water status should be monitored by Member States on a systematic and comparable basis throughout the EU, using standardized methods of monitoring, sampling and analysis. The WFD requires that scientifically sound biological criteria be used to establish the basis for the classification of coastal ecosystems and transitional waters (lagoons and estuaries). The ecological status of a water body must be evaluated using biological and or ecological (more integral) indicators. The latter requirement implies a good understanding of the mechanisms and processes involved in the functioning, development, and deterioration of ecosystems, the steps and timing of ecological succession, and the environmental conditions required for recovery of these ecosystems. This is, however, not an easy task since industrial, agricultural and urban wastes and system modifications by land reclamation and dredging have produced problems of pollution and eutrophication and have affected species structure and system functioning. The European Union directives require member states to follow the concept of sustainable use and sustainable development by application of an ecosystem approach. This approach is meant to optimise the biological quality of our ecosystems despite the exploitation of natural resources for economic reasons to increase welfare (Fig. 1).

The focus in this special issue is largely on morphological aspects and the effects of pollution of sediments.

External input of organic matter constitutes one of the most severe threats affecting coastal ecosystems. The presence of large amounts of organic matter in coastal sediments typically results in oxygen depletion, followed by the production of sulphide by sulphate-reducing bacteria. Detailed knowledge of the local functioning of the sulphur cycle in the sediment, sulphate reduction and associated fermentation reactions produced in the sediment by anaerobic metabolisms may help to prevent risks when polluted sediments are dredged, as they are in many coastal lagoons and estuaries (Zaggia et al. this issue).

Despite a ban on the use of chemical pollutants and notwithstanding the increasing efforts to outlaw, for example, TBT in antifouling paints, there is evidence of persistent contamination of the aquatic environment in some areas and of mobilisation from the sediments through the food web. The accumulation of TBT in sediments appears to be favoured by the accumulation of organic matter, while the significant release of DBT into the water phase could be a consequence of anthropogenic disturbance to the sediments resulting from a combination of dredging, salt marsh reclamation and local techniques in clam harvesting, although natural processes, such as tidal flushing, should not be excluded. Resuspension processes may explain the higher release of DBT than TBT from sediments into the water (Berto et al. this issue)

Not only direct human activities are to blame, since climate change associated with increasing torrential rains and runoff can stimulate the mobilization of ancient pollutants retained in sediments, while extreme river flood events in mining regions, despite their rarity, can make a considerable contribution to the input of particulate Hg into coastal areas and lagoon systems (Covelli et al. this issue).

Ecotoxicological evaluation, based on international Sediment Quality Guidelines (SQGs) and mSQGq indices, has pointed to medium ecological risks for marine organisms, despite the extreme concentrations of PCBs found (Cardellicchio et al. this issue). Therefore, the total concentration of metals or other pollutants is generally not sufficient to assess their environmental impact; it becomes necessary to estimate the bioavailable fraction. The bioavailable fraction is defined as the amount of metal that can be exchanged with biological organisms and be incorporated into their structure, and depends on the association of the elements with particles, their binding strength and water properties such as pH, redox potential and salinity, as well as the dissolved metal species which are in contact with the solid phase (Marmolejo-Rodríguez et al. this issue).

The physiological status of organisms is another factor that can determine the toxicity effects of pollutants. In the case of trace metals (Cd, Cr, Cu, Fe, Mn, Pb, Zn), PAHs and PCBs, pollutant concentrations and biological responses showed strong seasonal variations in both molluscs and fish in the Venice lagoon, generally related to cyclic physiological changes linked to reproduction and food availability (Nesto et al. this issue) or to differences in flesh weight of clams and stress conditions caused by transplanting (Boscolo et al. this issue). Similar results were obtained by Matozzo and Marin (2007), who found that the induction of Vitellogenin (Vg)-like proteins (a biomarker of exposure to estrogenic compounds) is markedly influenced by the bivalve reproductive cycle, in which differing phases of protein production, transport and storage occur. These results stress the need for careful evaluation of the biological cycles of selected sentinel organisms before planning biomonitoring surveys or before assessing PAH bioavailability in long-term experiments, and when analysing data obtained from different periods of the year. In this framework, biological indicators must be tested and continuously reviewed (Pavoni et al. this issue, Szymczak-Żyła et al. this issue).

The spatial heterogeneity of TW can be high, masking the relationship between chemical levels and ecological impact. Temporal variability can show an overlap with temporal trends in contamination. Therefore, spatio-temporal scales of variability must be taken into account when designing sampling strategies to evaluate human impact (Pérez-Ruzafa et al. 2007). Furthermore, changes in lagoon or estuary morphology and hydraulic conditions can change the spatial effects of pollution. While finer particles drive the distribution of the majority of contaminants, the construction of navigation channels can act as a boundary, inhibiting the transport of pollutants by tidal currents from the industrial zones toward other parts of a lagoon, so that the main sources of contaminants may not be directly linked to bigger industrial areas, but to “new” point sources (Zonta et al this issue). This underlines the need for flexibility in designing monitoring systems and indeed, suggestions have been made to extend monitoring by surveillance and to change the focus from ‘station oriented’ to ‘area oriented’, without changing the operational aspects and with little effect on costs (de Jonge et al. 2006; de Jonge this issue).

Increasing urban development and the dedication of coastal areas to recreation activities are accompanied by such activities as land reclamation, channel dredging, the pumping of sediments and the construction of harbour facilities and marinas. Consequently, the environmental effects are continuously increasing.

Coastal engineering has repercussions on the sedimentological regime and on mudflat topography in estuaries (Boyes et al. this issue). Some systems, such as the Sulina mouth of the Danube River, have shown strong morphological modifications in recent decades, with coastal processes ranging from swift coastline accumulation and silting to erosion, with consequent changes in the sediment transport capacity of longshore currents (Stanica et al 2007). Knowledge of the physical factors, which are the main structuring processes influencing the development of macrofaunal communities (Mazik et al. this issue), must form the basis for recovering these degraded ecosystems. The main factors influencing colonisation and community development appear to be tidal inundation, particle size and organic matter content. Additionally, colonisation is restricted in areas of low or excessively high accretion.

Habitat Suitability (HS) models have been extensively used over the last decade by conservation planners to estimate the spatial distribution of species. These models may be a useful tool for exploring the relationship between habitat features and the distribution of threatened species. They can also be used in land management, to identify suitable sites for farming and the corresponding yield potential, or for recovering over-exploited species (Vincenzi et al. this issue).

All the efforts and recent developments in EU water quality legislation may be insufficient if public preferences are not considered. When policy intervention and associated management action imposes substantial costs on the local community then their support for such measures is imperative (Atkins et al. this issue).

Finally, greater efforts will have to be made in the coordination of research and the discussions on how ecological processes work across the entire range of water bodies and the environmental gradients superimposed on them. There is an urgent need for coordination between policy-makers, governmental services and scientists in preparing sampling designs which will enable monitoring of the ecological status of water bodies, data interpretation and the development of concepts suitable for ecosystem modelling. Tools must be designed to judge the quality of ecosystems and test modern ecological theories and hypotheses. At the same time, the extended data sets derived from the 'area-oriented' monitoring approach will allow proper calibration and validation of dynamic ecosystem models – something that has not been possible on a routine basis until now (de Jonge, this issue).

The papers within this issue are derived from the 41st Annual International Conference of the Estuarine & Coastal Sciences Association (ECSA) entitled 'Measuring and Managing Changes in Estuaries and Lagoons' which was held from 15-20 October 2006 at the Scuola Grande San Giovanni Evangelista, Venice, Italy. It was organised by the Institute of Marine Sciences (ISMAR) and the Institute for the Dynamics of Environmental Processes (IDPA), the National Research Council (CNR), Venice , the Department of Environmental Sciences (DSA), University Ca' Foscari of Venice and the Consortium for Coordination of Research Activities concerning the Venice Lagoon System (CO.RI.LA), Venice on behalf of ECSA. The symposium involved about 200 participants, 92 papers and 99 posters. As with all international ECSA symposia (see www.ecsa-coast.org), while they provide a forum for work to be presented from many countries, they also act as a showcase for researchers from the host country. As shown in the present volume, and in a companion volume of Estuarine Coastal and Shelf Sciences (*Biodiversity and ecosystem functioning in coastal and transitional waters*), the ECSA41 symposium and these proceedings follow this tradition.

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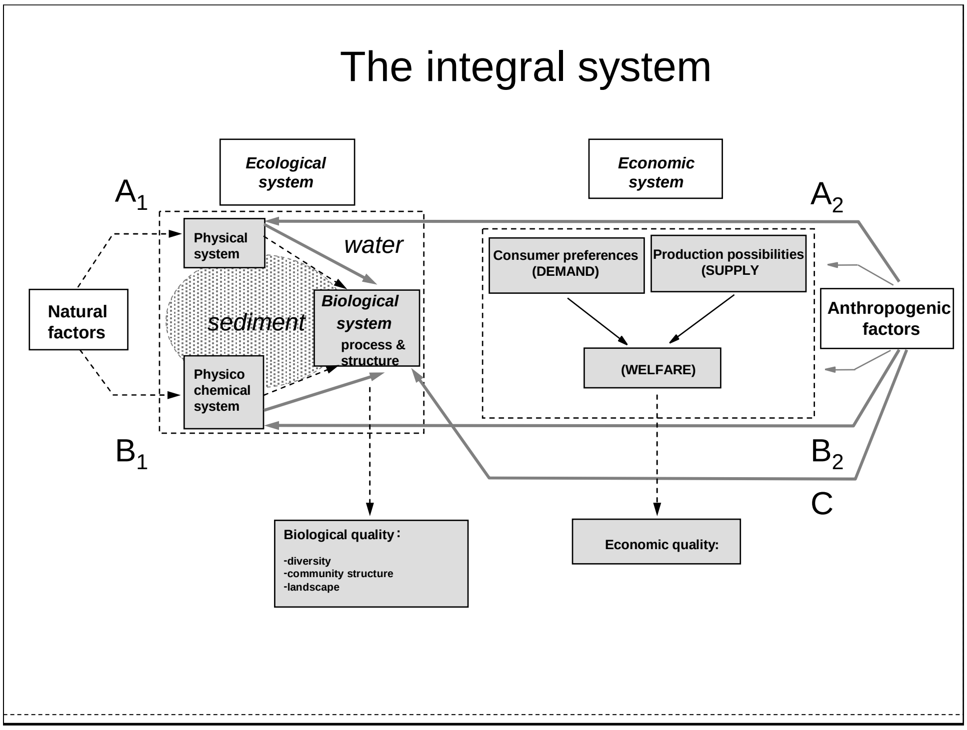

Fig. 1. Diagram representing the structure of the 'integral system' without explicit incorporation of the human/social component (modified from de Jonge et al., 2003). Natural influences are indicated by dashed lines and anthropogenic influences by solid lines. A_1 and A_2 refer to the natural or anthropogenic influence of the physical system, B_1 and B_2 to that of the physico-chemical system, and C refers to direct human effects on the biological system. (modified from de Jonge et al., 2003)