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Spatial models to support the management of coastal marine ecosystems: a short review of best practices in Liguria, Italy

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Abstract

Spatial modelling is an emerging approach to the management of coastal marine habitats, as it helps understanding and predicting the results of global change. This paper reviews critically two recent examples developed in Liguria, an administrative region of NW Italy. The first example, aiming at predicting habitat status depending on human pressures (i.e. anthropogenic activities capable of producing impact), provides managers with the opportunity of envisaging different scenarios for the consequences of coastal development choices. The second example defines the status of an important Mediterranean coastal marine habitat (*Posidonia oceanica* meadows) under natural conditions, allowing quantification of human impact on regressed meadows. Both modelling approaches are useful to define the targets of coastal management and, combined with information on cost of conservation (or management), actions can provide guidance to decision-making. Well-planned and sustained monitoring is essential for model validation and improvement.

Keywords: marine ecosystem, environmental modelling, coastal management, *Posidonia oceanica*, Liguria, Mediterranean Sea.

Introduction

Human use of nature resources and services is increasing worldwide, leading to alterations in ecosystem structure and functioning (Kubiszewski & Costanza, 2012). Unexpected ecosystem changes are becoming frequent, and the complex ways through which multiple human pressures (i.e. anthropogenic activities capable of producing impact) may interact, meaning that conservation practitioners and natural resource managers are faced with high uncertainty (Regan *et al*., 2005). As the management of ecosystems needs to account for the natural functional principles of ecosystems (Jørgensen & Nielsen, 2012), ecologists are facing the challenge of proposing methods to understand and predict these changes (Elith & Leathwick, 2009). Models are major examples of such methods. While models of population dynamics or food-web interactions are today well established practices in ecology (Thomas *et al*., 2005; Allesina *et al*., 2008), models at ecosystem or landscape level are recent achievements with promising prospects (Briske *et al*., 2005). Spatial distribution modelling, in particular, is a growing industry (Franklin, 2010); mapping habitat suitability and predicting species distribution are required for many aspects of environmental research, resource management and conservation planning (Wätzold *et al*., 2006). The need for similar approaches to coastal marine environments is evident (Issaris *et al*., 2012), but applications to the sea are still scarce (Galparsoro *et al*., 2012; Lyons *et al*., 2013).

Marine coastal ecosystems are particularly sensitive to global change. Climate warming combined with land-based sources of pollution, sedimentation, habitat destruction and overfishing (all of which are expected to increase in severity in the next decades) modify the status of coastal marine ecosystems. While climate change requires global actions to be tackled, good management practices may help reducing local human impacts (Cash & Moser, 2000).

This paper reviews two recent spatially explicit approaches applied in Liguria (an administrative region of NW Italy) for the management of marine coastal ecosystems under global change. The first is aimed at understanding the complex relationships between multiple human pressures and the status of coastal marine ecosystems, in order to predict future change. The second uses physical parameters to predict the reference conditions of one of the most important marine coastal ecosystems, namely seagrass meadows.

Coastal ecosystem management: a worldwide problem

Recent global assessments of marine ecosystem status revealed the diffuse impact of human activities (Halpern *et al.*, 2008). Environmental protection is necessary especially along coastal zones, for both ecological and socio-economic reasons (Borja *et al.*, 2008; Knowlton & Jackson, 2008).

Marine Protected Areas (MPAs) have often been considered a key tool for the conservation of coastal ecosystems but, at the same time, have been shown to be insufficient as a single measure to protect costal marine ecosystems (Allison *et al.*, 1998). Thus, several complementary measures have been developed worldwide to implement management schemes at the scale at which ecosystem processes occur (Bianchi *et al.*, 2012), leading to approaches that have been called Ecosystem Based Management (EBM) (Katsanevakis *et al.*, 2011) and Integrated Coastal Zone Management (ICZM). Major examples of such management approaches include the networks of areas of special protection in Europe (Fenberg *et al.*, 2012), the marine sanctuaries in the USA (Lester *et al.*, 2010) and the marine parks established on the Great Barrier Reef in Australia (Brodie & Waterhouse, 2012).

In the same direction, initiatives have recently been undertaken worldwide from a legislative point of view (Ricketts & Harrison, 2007; Barnes & McFadden, 2008). In Europe, the Marine Strategy Framework Directive (MSFD, 2008/56/EEC) imposes on all Member States the maintenance of seafloor integrity, which has to be assessed by comparison to reference conditions (Duarte *et al.*, 2008). Reference conditions can be retrieved using information from three sources: i) pristine situations or, alternatively, MPAs; ii) historical information; iii) predictive modelling. Pristine situations are hardly found (Jackson & Sala, 2001; Stachowitsch, 2003). Even if they are exponentially increasing in number, MPAs only protect a limited portion of the worldwide marine realm and are often insufficiently enforced (Edgar *et al.*, 2007; Montefalcone *et al.*, 2009). When available (which is not always the case), historical data are seldom reliable, because of insufficient standardisation, changes in technology and observer effects (Leriche *et al.*, 2004; Montefalcone *et al.*, 2013). Thus, despite their current limitations, such as the high uncertainty, models remain a little explored approach with interesting potential.

Due to binding European Union Directives, efforts have recently been made in most European countries to manage coastal marine ecosystems on the basis of a solid ecological background and recent scientific modelling. In Liguria, intense urban and industrial coastal development has led to massive decline of coastal and marine habitats and, in particular, of the meadows of the endemic seagrass *Posidonia oceanica* (Peirano & Bianchi, 1997). This decline has been particularly dramatic close to major coastal cities, such as the chief town of Genoa (Montefalcone *et al.*, 2007). Since the beginning of the 1960s, the proliferation of urban and industrial structures, coupled with intense use of the coastal area by the tourism industry, resulted in a dramatic loss of biodiversity (Peirano *et al.*, 2005; Montefalcone *et al.*, 2010a); since the mid 1980s, climate warming has increased the stress exerted on coastal marine ecosystems (Morri & Bianchi, 2001; Cattaneo Vietti *et al.*, 2010). At present, all *P. oceanica* meadows have been included within the so called Sites of Community Importance (SCIs), which require special conservation plans. In addition, four MPAs have been established recently: Portovenere Archipelago, Cinque Terre, Portofino, and Bergeggi Island; two others, Mortola and Gallinara Island, are planned (Guidetti & Sala, 2007; Montefalcone *et al.*, 2009; Rovere *et al.*, 2010; Fig. 1). In 2005, the Beigua Natural Park joined the Global and European Geoparks Network supported by Unesco. Since then, important measures have been adopted in order to preserve the coastal and marine portion of the park (extending for about 30 km westwards to Genoa, Fig. 1) with particular focus on the protection and valorisation of its underwater geological heritage (Burlando *et al.*, 2011).

Spatial methodologies for ecosystem management

Tricart & Kilian (1979) were among the first to recognize cartography as a key tool for the spatial investigation of natural environments. Human pressures and coastal ecosystems have, by definition, a spatial component and cartography is thus the natural approach to conservation and management (Bock *et al.*, 2005; Bianchi *et al.*, 2012). This conviction is even stronger nowadays and its application made easier by the availability of Geographical Information Systems (GISs). The predictive methodologies we are critically reviewing in this paper were both developed on cartographic data in a GIS environment, and represent proper tools to be adopted in the framework of an effective EBM.

Modelling the relationship between human pressure and coastal ecosystem status in a Ligurian MPA

The management of natural resources largely relies on the possibility to plan the spatial distribution of human pressure and modify the intensity. Management choices require an understanding of the relationship between the spatial distribution of human pressures, their intensity, and ecosystem status. This is crucial for developing spatial tools allowing the evaluation of the efficiency of alternative management strategies (Douvere, 2008).

In the marine realm, the dominant approaches are based on expert-judgment surveys or literature review, which are used to estimate the vulnerability of ecosystem

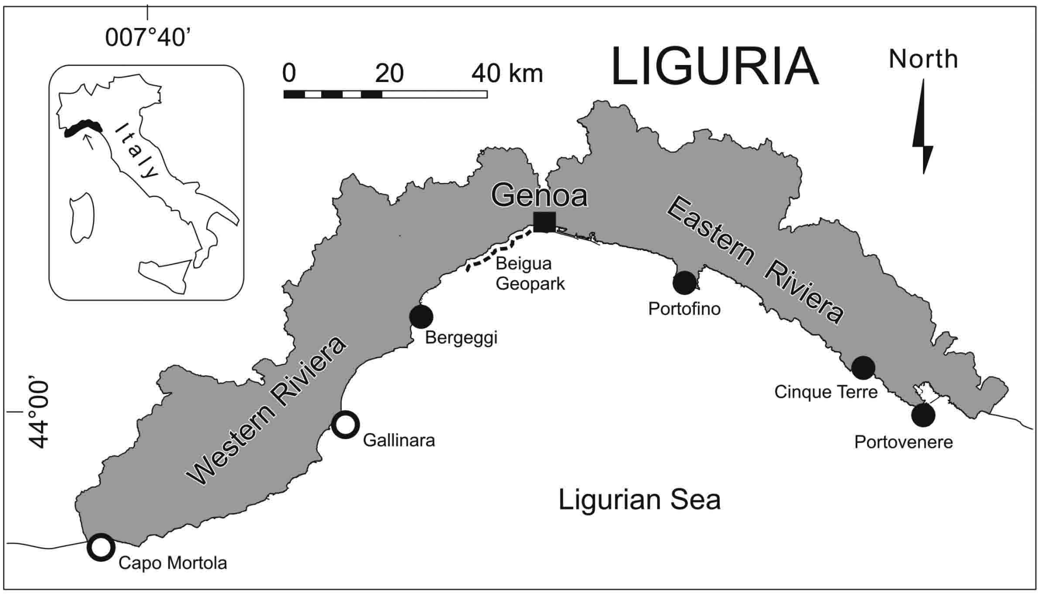

Fig. 1: Geographical setting of Liguria, together with location of Marine Protected Areas along the coastline. Solid circles: established; open circles: planned. The dashed line indicates the extent of the coastal and marine part of the Beigua Geopark.

or habitat types according to pressure identity and intensity (Muxika *et al.*, 2007). On this basis, if the spatial distribution of habitats and pressures is known, the potential risk of impact can be computed and represented on maps, thereby helping to identify an efficient management solution, i.e. the one capable of minimizing the risk of impact (Halpern *et al.*, 2009; Stelzenmuller *et al.*, 2010). When possible, such management solutions should be prioritized according to the coupled opinion of both experts in the field and local stakeholders (Giakoumi *et al.*, 2012). In addition, the risk of arbitrariness in the selection of the variables can significantly affect the assessment of habitat condition; in order to minimize this problem, Game *et al.* (2013) underlined the importance, wherever possible, of estimating variables of interest on natural scales.

This kind of coupled GIS and modelling approach has the invaluable advantage of making possible to support the creation of management plans at large spatial scales, which can then be implemented taking into account costs (Wilson *et al.*, 2007) or when data on ecosystem status are missing or scarce. However, they generally arbitrarily assume that multiple pressures interact additively (Halpern *et al.*, 2009). This is a limitation, especially when considering that most studies on the effects of multiple pressures unveiled significant non-additive interactions (Crain *et al.*, 2008; Darling & Côté, 2008). Without using data on ecosystem status, literature-based approaches can hardly detect the complex interactions that may exist among pressures (e.g. synergisms or antagonisms). In addition, such interactions are spatially variable and extremely site-specific, making it difficult, if not impossible, to extrapolate general rules to be used *a priori* over vast spatial scales (Crain *et al.*, 2008).

In many cases, regional monitoring programs are established for the assessment of ecosystem status. In Europe, for instance, *ad hoc* monitoring plans are applied for computing indices of ecosystem status according to the WFD. If the spatial distribution of pressures is available and the status of the ecosystems is known, spatial explicit modelling represents a good alternative to literature-based approaches. If modelling is employed, the effect of individual pressures and the type of interaction among multiple pressures (e.g. synergism or antagonism) can be predicted by the model under different management scenarios. An example of such an approach is given below.

In order to identify the best management strategy for the Bergeggi Island MPA, located downstream of the expanding commercial harbour of Vado Ligure (Gatti *et al.*, 2012), a spatial explicit tool was developed to allow for: i) modelling of the relationship between human pressures and ecosystem status; ii) prediction of the expected effect of alternative management scenarios on the status of ecosystems; iii) cartographic visualization of the effect of each management scenario (Parravicini *et al.*, 2012).

The conceptual framework behind this approach is comprised of four distinct steps (Fig. 2): 1) building a GIS database of human pressures and their intensities; 2) GIS mapping of the marine ecosystem status; 3) modelling the relationships between the distribution of human pressures and marine ecosystem status; 4) use of the model calibrated in step 3 to build maps of expected ecosystem status according to different management alternatives – i.e. expected or planned variations in human pressures distribution and intensities. Within the framework of this geospatial approach, once an efficient solution is found,

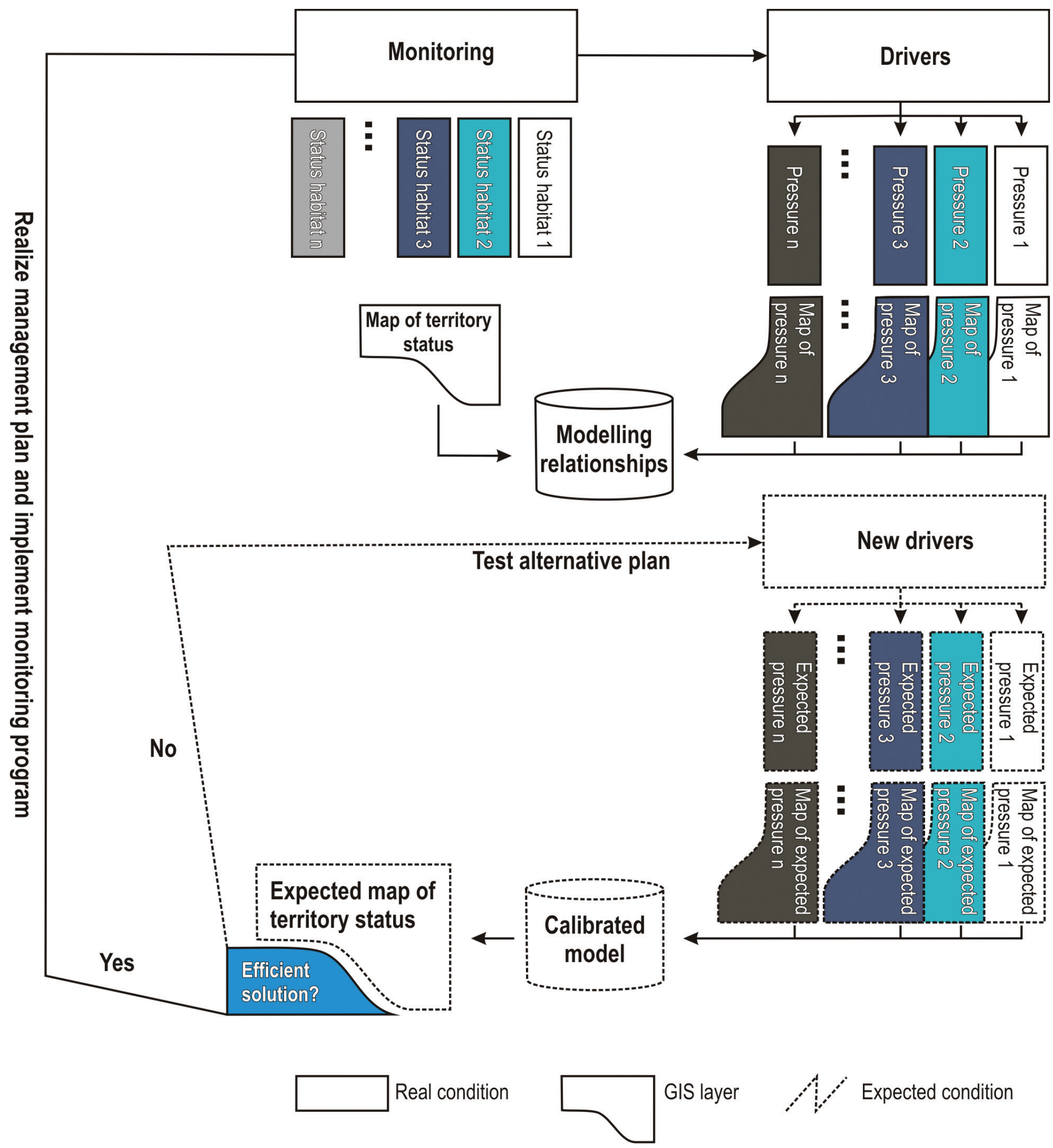

Fig. 2: Conceptual diagram of the steps used to develop the geospatial modelling approach of Parravicini *et al.* (2012). The approach includes four main steps: 1) mapping human pressures and their intensities; 2) mapping marine territory status; 3) modelling the relationships between human pressure distribution and marine territory status; 4) use of the model calibrated in step 3 to build maps of expected territory status according to different management alternatives – i.e. expected or planned variations in human pressure distribution and/or intensities.

appropriate monitoring plans must be implemented to allow for future more accurate calibration of the model.

In order to use such an approach it is mandatory to have quantitative information on both pressure intensities and the status of coastal ecosystems. Parravicini *et al.* (2012) adopted a Multi-Criteria Decision Aid (MCDA) procedure (Behzadian *et al.*, 2012, and references therein) and, in particular, the fuzzy extension of TOPSIS, the Technique for Order Preference by Similarity to Ideal Solution (Chen, 2000). The following activities produce the different pressures selected for the model: anchoring (i.e. seafloor abrasion), beach nourishment, commercial harbour influence (i.e. pollutants and traffic related pressures), influence of coastal outfalls (i.e. organic enrichment), angling, gillnet, fishing, trawling, influence of pipe outlets (i.e. organic enrichment and pollutants), intensity of SCUBA diving visitation, and urbanization (i.e. artificial coast). The intensity of each pressure at the source was estimated using expert elicitation; the uncertainty linked to the potential disagreement among experts was incorporated solving the MCDA problem under a fuzzy environment. The distance from the source was taken as a proxy of pressure intensity abatement. However, any metric (e.g. presence/absence of pressures or weighted distance from pressure sources) has to be employed according to the user needs.

The status of ecosystems may be quantified according to any index devised for such purpose (e.g. the indices developed within the Water Framework Directive, WFD). Parravicini *et al.* (2012) calculated the ecological

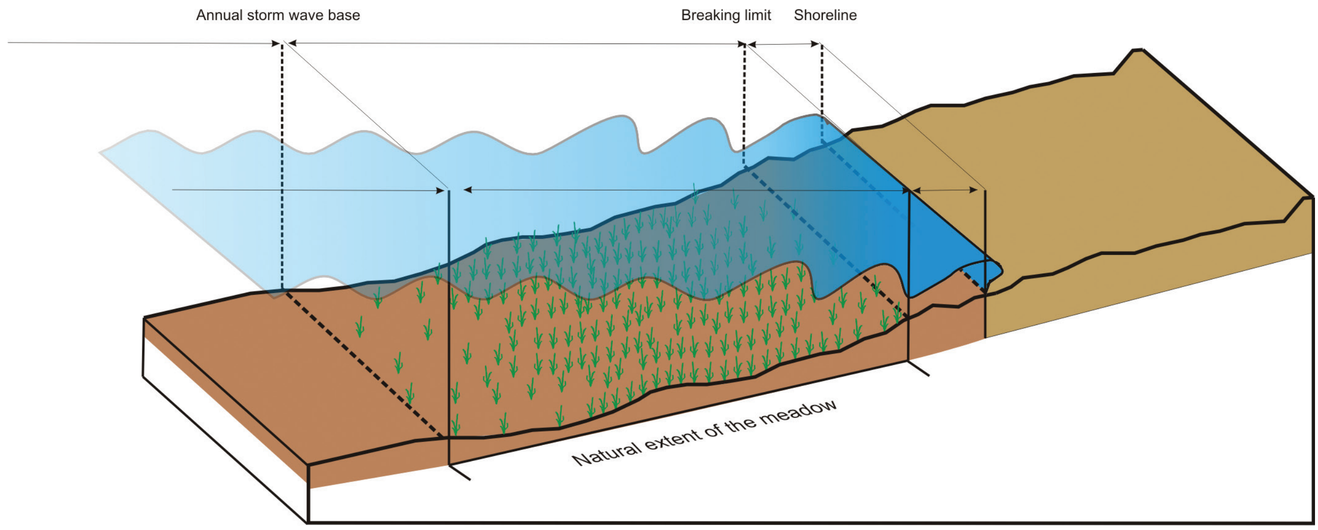

Fig. 3: Spatial extent of *Posidonia oceanica* meadows between two hydrodynamic boundaries (the wave break limit and the annual storm wave base), as expected to occur under natural conditions according to the models of Vacchi *et al.* (2012).

distance of benthic habitats from reference conditions historically described for Mediterranean marine coastal benthic habitats.

Once pressure intensities are assessed and the status of coastal ecosystem is defined, modelling their relationships is required for predictive purposes and the evaluation of alternative management scenarios. Parravicini *et al.* (2012) used a spatial explicit extension of Random Forests, which is a machine learning technique handling non-linearity and collinearity among predictor variables (see Dormann *et al.*, 2013).

Considering that the goal of spatial plans is the accurate prediction of the expected status of ecosystems, modelling techniques based on boosting or bagging algorithms may be seen as preferred. Among them, those based on classification and regression trees have the advantage of handling categorical variables, which is often the case when working on ecosystem status indices. Other techniques such as multivariate regression splines may be used when several response variables need to be modelled, while Geographically Weighted Regression may be employed in the case of spatial non-stationarity.

Whatever the type of index employed and the preferred modelling technique, any application of the approach shown in figure 2 represents a good solution for developing spatial management tools when field information is available. The calibrated model may be used to predict the environmental consequence of variation in the spatial distribution of human pressures and/or variation in their intensities, thereby allowing for the evaluation of several management scenarios. Importantly, when the tool is employed to evaluate alternative scenarios, estimates of uncertainty around model prediction may be crucial. These can be obtained easily by simple bootstrapping of the original dataset.

Application of this spatial modelling approach predicted that if the Vado Ligure Port Authority supports the Bergeggi Island MPA in reducing fishery, beach nourishment and anchoring within the MPA boundaries, it is not expected that the marine ecosystems of the MPA will further deteriorate as a consequence of harbour expansion. The visual representation of the expected effect of different management alternatives will facilitate the consultation process between the MPA and the Port Authority, which is needed to achieve the goal of a win-win strategy. This aspect is important for policy aimed at solving the dilemma of finding the appropriate balance between conservation and use of natural resources (Parravicini *et al.*, 2012). Further information regarding the cost of alternative management options should be incorporated in this approach. This will significantly improve its effectiveness.

Modelling the natural spatial boundaries of seagrass meadows along the Ligurian coastline

Seagrass meadows are among the most important and productive ecosystems on earth (McRoy & McMillan, 1977) and one of the most valuable habitats in coastal areas, shaping coastal seascapes and providing essential ecological and economic services (Green & Short, 2003). A global decline of seagrass meadows has been largely documented over several decades, with rates estimated at 2–5% per year (Waycott *et al.*, 2009; Short *et al.*, 2011).

Seagrasses are biological quality elements used to define the ecological status of transitional or coastal waters, because they are sensitive to human disturbance (Marbà *et al.*, 2013). *Posidonia oceanica*, the most important seagrass in the Mediterranean Sea, is regularly used as a bioindicator because of its sensitivity to pressures (Pergent-Martini *et al.*, 2005; Montefalcone, 2009). Its selection as biological quality element (BQE) in monitoring programs led to develop a number of indices that combine different parameters of both the plant and the meadow (Romero *et al*., 2007; Fernandez Torquemada *et al*., 2008; Gobert *et al*., 2009). Shoot density and meadow cover are generally considered as the most important descriptors of meadow health (Montefalcone, 2009), although the epiphyte community has been shown to provide early warning to environmental alterations before the whole meadow regresses (Giovannetti *et al.*, 2010). Depth and the position of the meadow limits have also been recognized as proper indicators of the state of health of *P. oceanica* meadows (Pergent-Martini *et al*., 2005) and are the easiest to model spatially in order to predict the modifications of meadow distribution in response to global change. Modelling suitable habitats for seagrasses, using explicit spatial criteria and ecological niche factor analysis, is an emerging approach (Valle *et al*., 2011; Downie *et al*., 2013). In Liguria, the physical parameters affecting the bathymetrical distribution of *P. oceanica* were investigated following recent studies that unveiled the role of nearshore hydrodynamics on both position and structure of seagrass meadows (Folkard, 2005; Infantes *et al*., 2009). The main goal was to understand whether, in pristine conditions, the seaward and landward boundaries of a meadow can be predicted on the basis of physical parameters alone, namely wave, climate and seafloor morphology (Montefalcone *et al*., 2010b; Vacchi *et al*., 2010, 2012). The challenging perspective was to create innovative and reliable models for identifying the extent of *P. oceanica* under natural conditions, thereby allowing estimation, by retrodiction, of the amount of habitat loss due to anthropogenic effects.

Vacchi *et al*. (2010, 2012) demonstrated that the hydrodynamic regime controls the bathymetrical distribution of *P. oceanica* meadows on the seafloor (Fig. 3). The landward or upper limit of the meadow is significantly controlled by the breaking depth, i.e. the depth where the wave breaks (Smith, 2002). The breaking depth (d_b) may be computed according to the formula:

$$d_b = H_b/\gamma_b,$$

where $H_b = H_0 \cdot K_{sh} \cdot \sqrt{\varphi_o/\varphi_b}$ (H_0 = offshore wave height, with return time 1 year; K_{sh} = shoaling coefficient; φ_o and φ_b = offshore and nearshore waves approach angle) and $\gamma_b = b\text{-}a \cdot (H_b/gT_0^2)$ (*a* and *b* being empirical coefficients depending on the slope of the beach).

Using the breaking depth, Vacchi *et al.* (2013) defined a model able to locate the region of the seafloor where the meadow upper limit should lie in natural conditions (i.e. those governed only by hydrodynamics, in the absence of significant anthropogenic impact). This model was validated at regional spatial scale and current investigations are evaluating its suitability at the Mediterranean scale.

The seaward or lower limit of a *P. oceanica* meadow is traditionally considered to be under the sole influence of light penetration (Duarte, 1991). According to Vacchi *et al*. (2012), however, an important role is also played by the storm wave base, i.e. the limit of interaction between waves and seafloor, corresponding to $L_0/2$, where L_0 is the offshore wavelength (Svendsen, 2006).

The following equation may be applied to estimate the natural position of the *P. oceanica* meadow lower limit, as determined by wave regime:

$$Z_c = 0.32 \cdot L_0 + 5.62,$$

where Z_c is the depth of meadow lower limit (in meters) and L_0 is the annual offshore wave length (in meters), computed as a climatological mean.

This equation has to be flanked to the one proposed by Duarte (1991), based on water transparency:

$$\ln Z_c = 0.26 - 1.07 \cdot \ln K,$$

where Z_c is the depth of meadow lower limit (in meters) and *K* is the coefficient of light attenuation underwater.

Knowing the depth where the lower limit of a meadow should occur under natural conditions, it is important to quantify any regression potentially caused by human impact; thus, the shallower of the two values resulting from the equations described above may be taken as the baseline depth of the meadow lower limit before regression occurred.

The hydrodynamic influence on the meadow lower limit is important in sheltered bays or in coastal areas not exposed to intense storm waves (Vacchi *et al*., 2012). In exposed areas, hydrodynamics probably play a minor role when compared to light penetration, and the position of the meadow lower limit is mainly related to water transparency (Montefalcone *et al*., 2009).

Final remarks

The Ligurian experiences reviewed in this paper represent two examples of spatial modelling for coastal management. The first example, predicting habitat status depending on pressures, provides managers with the opportunity of envisaging different scenarios for the consequences of coastal development choices. It is the first tangible application of the DPSIR (Driving forces, Pressures, States, Impacts, Responses) framework to an Italian sea (Bianchi & Morri, 2003). Parravicini *et al.* (2012) applied it at local scale in an MPA. Ongoing studies carried out on further Mediterranean MPAs suggest that extensions to larger spatial scales might be feasible, provided that pressures and habitat status are known with sufficient detail. The second example predicts (or, better, retrodicts) the status of a crucial coastal marine habitat, i.e. *Posidonia oceanica* meadows, under natural conditions; this information is basic for quantifying any suspected or observed meadow regression, and represents the first step needed to infer the effect of an already occurred impact. Both models are useful to define the targets of coastal management (Borja *et al*., 2012). With the model of Parravicini *et al*. (2012), accepting a specific scenario against other possible scenarios will depend on the degree of habitat alteration deriving from each potential solution; the choice should be made taking into account the value of the natural capital with respect to the economic capital (Wilson *et al*., 2007; van Teeffelen & Moilanen, 2008). The models of Vacchi *et al*. (2012, 2013) may guide the best management option for degraded *Posidonia oceanica* meadows. Due to the low resilience of this seagrass, highly regressed meadows have no real recovery potential (Marbà *et al.*, 1996; Montefalcone, 2009); attempts to re-establish *P. oceanica* in such areas might be a waste of time and money. On the contrary, meadows showing a limited to moderate regression could still fully recover thanks to removal of the major causes of regression and to specific restoration programs (Montefalcone *et al*., 2007).

No matter how efficient they are, spatial models alone are not sufficient for coastal management. Firstly, they should include information about cost within the framework of systematic conservation planning (Giakoumi *et al.*, 2012). Secondly, well-planned and sustained monitoring of the marine ecosystems to be managed play a key role in the validation of model outputs, allowing fitting of unmatched residuals with the field-constrained benchmarks of benthic habitats, reducing uncertainty around prediction or potentially providing basic information for the development of more complex dynamic models.

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References

Allesina, S., Alonso, D., Pascual, M., 2008. A general model for food web structure. *Science,* 320 (5876), 658-661.

Allison, G.W., Lubchenco, J., Carr, MH., 1998. Marine reserves are necessary but not sufficient for marine conservation. *Ecological Applications,* 8 (1), 79-92.

Barnes, C., McFadden, K.W., 2008. Marine ecosystem approaches to management: challenges and lessons in the United States. *Marine Policy,* 32 (3), 387-392.

Behzadian, M., Khanmohammadi Otaghsara, S., Yazdani, M., Ignatius, J., 2012. A state-of-the-art survey of TOPSIS applications. *Expert Systems with Applications,* 39, 13051-13069.

Bianchi, C.N., Morri, C., 2003. Indicatori biologici ed ecologici nell'ambiente marino. p. 111-120. In: *Studi per la creazione di strumenti di gestione costiera: Golfo del Tigullio*. Ferretti, O. (Ed.). ENEA Centro Ricerche Ambiente Marino, La Spezia, Italy.

Bianchi, C.N, Parravicini, V., Montefalcone, M., Rovere, A., Morri, C., 2012. The challenge of managing marine biodiversity: a practical toolkit for a cartographic, territorial approach. *Diversity,* 4 (4), 419-452.

Bock, M., Rossner, G., Wissen, M., Remm, K., Langanke, T. *et al*. 2005. Spatial indicators for nature conservation from European to local scale. *Ecological Indicators,* 5, 322-338.

Borja, Á., Bricker, S.B., Dauer, D.M., Demetriades, N.T., Ferreira, J.T. *et al*., 2008. Overview of integrative tools and methods in assessing ecological integrative in estuarine and coastal systems worldwide. *Marine Pollution Bulletin,* 56, 1519-1537.

Borja, Á., Dauer, D.M., Grémare, A., 2012. The importance of setting targets and reference conditions in assessing marine ecosystem quality. *Ecological Indicators*, 12 (1), 1-7.

Briske, D.D., Fuhlendorf, S.D., Smeins, F.E., 2005. State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. *Rangeland Ecology and Management,* 58, 1-10.

Brodie, J, Waterhouse, J., 2012. A critical review of environmental management of the "not so Great" Barrier Reef. *Estuarine, Coastal and Shelf Science,* 104-105, 1-22.

Burlando, M., Firpo, M., Queirolo, C., Rovere, A., Vacchi, M., 2011. From geoheritage to sustainable development: strategies and perspectives in the Beigua Geopark (Italy). *Geoheritage*, 3 (2), 63-72.

Cattaneo Vietti, R., Albertelli, G., Aliani, S., Bava, S., Bavestrello, G. *et al.,* 2010. The Ligurian Sea: present status, problems and perspectives. *Chemistry and Ecology,* 26 (S1), 319-340.

Cash, D.W., Moser, S.C., 2000. Linking global and local scales: designing dynamic assessment and management processes. *Global Environmental Change*, 10 (2), 109-120.

Chen, C.T., 2000. Extensions of the TOPSIS for group decision-making under fuzzy environment. *Fuzzy Sets and Systems,* 114, 1-9.

Crain, C.M., Kroeker, K., Halpern, B.S., 2008. Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters,* 11, 1304-1315.

Darling, E.S., Côté, I.M., 2008. Quantifying the evidence for ecological synergism. *Ecology Letters,* 11, 1278-1286.

Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G. *et al.* 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography,* 36 (1), 27-46.

Douvere, F., 2008. The importance of marine spatial planning in advancing ecosystem-based sea use management. *Marine Policy,* 32, 762-771.

Downie, A.L., von Numers, M., Boström, C., 2013. Influence of model selection on the predicted distribution of the seagrass *Zostera marina*. *Estuarine, Coastal and Shelf Science* 121-122, 8-19.

Duarte, C.M., 1991. Seagrass depth limits. *Aquatic Botany,* 40(4), 363-377.

Duarte, C.M., Conley, D.J., Carstensen, J., Sánchez-Camacho, M., 2008. Return to Neverland: shifting baselines affect eutrophication restoration targets. *Estuaries and Coasts,* 32 (1), 29-36.

Edgar, G.J., Russ, G.R., Babcock, R.C., 2007. Marine protected areas. p. 533-555. In: *Marine Ecology*. Oxford University Press, South Melbourne, VIC, Australia.

Elith, J., Leathwick, J.R., 2009. Species distribution models: ecological explanation and prediction across space and time. *Annual Review of Ecology, Evolution, and Systematics,* 40, 677-697.

Fenberg, P.B., Caselle, J.E., Claudet, J., Clemence, M., Gaines, S.D. *et al.* 2012. The science of European marine reserves: status, efficacy, and future needs. *Marine Policy,* 36, 1012-1021.

Fernandez Torquemada, Y., Diaz Valdes, M., Luna, B., Colilla, F., Luna, B. *et al*., 2008. Descriptors from *Posidonia oceanica* (L.) Delile meadows in coastal waters of Valencia, Spain, in the context of the EU Water Framework Directive. *ICES Journal of Marine Science,* 65, 1492-1497.

Folkard, A.M., 2005. Hydrodynamics of model *Posidonia oceanica* patches in shallow water. *Limnology and Oceanography,* 50 (5), 1592-1600.

Franklin, J., 2010. *Mapping species distributions: spatial inference and prediction*. Cambridge University Press, Cambridge, UK, 320 pp.

Galparsoro, I., Liria, P., Legorburu I., Bald, J., Chust, G. *et al*. 2012. A marine spatial planning approach to select suitable areas for installing Wave Energy Converters (WECs), on the Basque continental shelf (Bay of Biscay). *Coastal Management,* 40, 1-19.

Game, E.T., Kareiva, P., Possingham, H. P. 2013. Six common mistakes in conservation priority setting. *Conservation Biology*, 27: 480–485.

Gatti, G., Montefalcone, M., Rovere, A., Parravicini, V., Morri, C. *et al*. 2012. Seafloor integrity down the harbor waterfront: the coralligenous shoals off Vado Ligure (NW Mediterranean). *Advances in Oceanography and Limnology*, 3 (1), 51-67.

Giakoumi, S., Katsanevakis, S., Vassilopoulou, V., Panayotidis, P., Kavadas, S. *et al*. 2012. Could European marine conservation policy benefit from systematic conservation planning? *Aquatic Conservation: Marine and Freshwater Ecosystems*, 22 (6), 762-775.

Giovannetti, E., Montefalcone, M., Morri, C., Bianchi, C.N., Albertelli, G., 2010. Early warning response of *Posidonia oceanica* epiphyte community to environmental alterations (Ligurian Sea, NW Mediterranean). *Marine Pollution Bulletin,* 60 (7), 1031-1039.

Gobert, S., Sartoretto, S., Rico-Raimondino, V., Andral, B., Chery, A. *et al*., 2009. Assessment of the ecological status of Mediterranean French coastal waters as required by the Water Framework Directive using the *Posidonia oceanica* Rapid Easy Index: PREI. *Marine Pollution Bulletin,* 58 (11), 1727-1733.

Green, E.P., Short, F.T., 2003. *World atlas of seagrasses.* University of California Press, Los Angeles, USA, 298 pp.

Guidetti, P., Sala, E., 2007. Community-wide effects of marine reserves in the Mediterranean Sea. *Marine Ecology Progress Series*, 335, 43-56.

Halpern, B.S., Kappel, C.V., Selkoe, K.A., Micheli, F., Ebert, C.M. *et al*., 2009. Mapping cumulative human impacts to California current marine ecosystems. *Conservation Letters,* 2, 138-148.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F. *et al.,* 2008. A global map of human impact on marine ecosystems. *Science,* 319 (5865), 948-952.

Infantes, E., Terrados, J., Orfila, A., Cañellas, B., Álvarez-Ellacuria, A., 2009. Wave energy and the upper depth limit distribution of *Posidonia oceanica*. *Botanica Marina,* 52, 419-427.

Issaris, Y., Katsanevakis, S., Pantazi, M., Vassilopoulou, V., Panayotidis, P. *et al.,* 2012. Ecological mapping and data quality assessment for the needs of ecosystem-based marine spatial management: case study Greek Ionian Sea and the adjacent gulfs. *Mediterranean Marine Science,* 13 (2), 297-311.

Jackson, J.B.C., Sala, E., 2001. Unnatural oceans. *Scientia Marina,* 65, 273-281.

Jørgensen, S.E., Nielsen, S.N., 2012. Tool boxes for an integrated ecological and environmental management. *Ecological Indicators,* 21, 104-109.

Katsanevakis, S., Stelzenmüller, V., South, A., Sørensen, T.K., Jones, P.J.S. *et al.,* 2011. Ecosystem-based marine spatial management: review of concepts, policies, tools, and critical issues. *Ocean and Coastal Management,* 54, 807-820.

Knowlton, N., Jackson, J.B.C., 2008. Shifting baselines, local impacts and global change on coral reefs. *PLoS Biology,* 6 (2), e54.

Kubiszewski, I., Costanza, R., 2012. Ecosystem services for sustainable prosperity. p. 177-182. In: *State of the World 2012: moving toward sustainable prosperity*. Starke, L. (Ed.). Island Press, The Worldwatch Institute, Washington, USA.

Leriche, A., Boudouresque, C.F., Bernard, G., Bonhomme, P., Denis, J., 2004. A one-century suite of seagrass bed maps: can we trust ancient maps? *Estuarine, Coastal and Shelf Science,* 59 (2), 353-362.

Lester, S.E., McLeod, K.L., Tallis, H., Ruckelshaus, M., Halpern, B.S. *et al.,* 2010. Science in support of ecosystem-based management for the US West Coast and beyond. *Biological Conservation*, 143, 576-587.

Lyons, M.B., Roelfsema, C.M., Phinn, S.R., 2013. Towards understanding temporal and spatial dynamics of seagrass landscapes using time-series remote sensing. *Estuarine, Coastal and Shelf Science*, 120, 42-53.

Marbà, N., Duarte, C. M., Cebrián, J., Gallegos, M. E., Olesen, B. *et al*., 1996. Growth and population dynamics of *Posidonia oceanica* on the Spanish Mediterranean coast: elucidating seagrass decline. *Marine Ecology Progress Series*, 137(1), 203-213.

Marbà, N., Krause-Jensen, D., Alcoverro, T., Birk, S., Pedersen, A. *et al.,* 2013. Diversity of European seagrass indicators: patterns within and across regions. *Hydrobiologia,* 704, 265-278.

McRoy, C.P., McMillan, C., 1977. Production ecology and physiology of seagrasses. p. 53-87. In: *Seagrass ecosystems: a* *scientific Prospective*. McRoy, P.C., Helfferich, C. (Eds). M. Dekker Inc, New York, USA.

Montefalcone, M., 2009. Ecosystem health assessment using the seagrass *Posidonia oceanica*: a review. *Ecological Indicators,* 9, 595-604.

Montefalcone, M., Albertelli, G., Morri, C., Bianchi, C.N., 2007. Urban seagrass: status of *Posidonia oceanica* off Genoa city waterfront (Italy). *Marine Pollution Bulletin*, 54, 206-213.

Montefalcone, M., Albertelli, G., Morri, C., Bianchi, C.N., 2010a. Pattern of wide-scale substitution within *Posidonia oceanica* meadows of NW Mediterranean Sea: invaders are stronger than natives. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 20, 507-515.

Montefalcone, M., Albertelli, G., Morri, C., Parravicini, V., Bianchi, C.N., 2009. Legal protection is not enough: *Posidonia oceanica* meadows in marine protected areas are not healthier than those in unprotected areas of the northwest Mediterranean Sea. *Marine Pollution Bulletin,* 58, 515-519.

Montefalcone, M., Parravicini, V., Vacchi, M., Albertelli, G., Ferrari, M. *et al*., 2010b. Human influence on seagrass habitat fragmentation in NW Mediterranean Sea. *Estuarine, Coastal and Shelf Science*, 86, 292-298.

Montefalcone, M., Rovere, A., Parravicini, V., Albertelli, G., Morri, C. *et al*., 2013. Evaluating change in seagrass meadows: a time-framed comparison of Side Scan Sonar maps. *Aquatic Botany,* 104, 204-212.

Morri, C., Bianchi, C.N., 2001. Recent changes in biodiversity in the Ligurian Sea (NW Mediterranean): is there a climatic forcing? p. 375-384. In: *Structure and processes in the Mediterranean ecosystems*. Faranda, F.M., Guglielmo, L., Spezie, G. (Eds). Springer Verlag, Milano, Italy.

Muxika, I., Borja, A., Bald, J., 2007. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. *Marine Pollution Bulletin*, 55 (1), 16-29.

Parravicini, V., Rovere, A., Vassallo, P., Micheli, F., Montefalcone, M. *et al*., 2012. Understanding relationships between conflicting human uses and ecosystem status: a geospatial modeling approach. *Ecological Indicators,* 19, 253-263.

Peirano, A, Bianchi, C.N., 1997. Decline of the seagrass *Posidonia oceanica* in response to environmental disturbance: a simulation-like approach off Liguria (NW Mediterranean Sea). p. 87-95. In: *The response of marine organisms to their environments*, Hawkins, L.E., Hutchinson, S. (Eds). University of Southampton, Southampton, UK.

Peirano, A., Damasso, V., Montefalcone, M., Morri, C., Bianchi, C.N., 2005. Effects of climate, invasive species and anthropogenic impacts on the growth of the seagrass *Posidonia oceanica* (L.) Delile in Liguria (NW Mediterranean Sea). *Marine Pollution Bulletin,* 50, 817-822.

Pergent-Martini, C., Leoni, V., Pasqualini, V., Ardizzone, G.D., Balestri, E. *et al.,* 2005. Descriptors of *Posidonia oceanica* meadows: use and application. *Ecological Indicators,* 5, 213-230.

Regan, H.M., Ben-Haim, Y., Langford, B., Wilson, W.G., Lundberg, P. *et al*., 2005. Robust decision-making under severe uncertainty for conservation management. *Ecological Applications,* 15 (4), 1471-1477.

Ricketts, P., Harrison, P., 2007. Coastal and ocean management in Canada: moving into the 21st century. *Coastal Management,* 35 (1), 5-22.

Romero, J., Martinez-Crego, B., Alcoverro, T., Perez, M., 2007. A multivariate index based on the seagrass *Posidonia oceanica* (POMI) to assess ecological status of coastal waters under the Water Framework Directive (WFD). *Marine Pollution Bulletin,* 55, 196-204.

Rovere, A., Parravicini, V., Vacchi, M., Montefalcone, M., Morri, C. *et al*., 2010. Geo-environmental cartography of the marine protected area “Isola di Bergeggi” (Liguria, NW Mediterranean Sea). *Journal of Maps*, 6 (1), 505-519.

Short, F.T., Polidoro, B., Livingstone, S.R., Carpenter, K.E., Bandeira, S. *et al*., 2011. Extinction risk assessment of the world’s seagrass species. *Biological Conservation,* 144, 1961-1971.

Smith, J.M., 2002. Surf zone hydrodynamics. In: *Coastal engineering manual,* Part II*, Coastal hydrodynamics,* Chapter II-4*, Engineer manual* 1110-2-1100. Demirbilek, Z. (Ed.). US Army Corps of Engineers, Washington DC, USA.

Stachowitsch, M., 2003. Research on intact marine ecosystems: a lost era. *Marine Pollution Bulletin,* 46, 801-805.

Stelzenmuller, V., Lee, J., South, A., Rogers, S.I., 2010. Quantifying cumulative impacts of human pressures on the marine environment: a geospatial modeling framework. *Marine Ecology Progress Series,* 398, 19-32.

Svendsen, I.A., 2006. *Introduction to nearshore hydrodynamics. Advanced Series On Ocean Engineering 24.* World Scientific, Singapore.

Thomas, L., Buckland, S.T., Newman, K.B., Harwood, J., 2005. A unified framework for modelling wildlife population dynamics. *Australian and New Zealand Journal of Statistics,* 47 (1), 19-34.

Tricart, J., Kilian, J., 1979. *L’éco-géographie et l’aménagement du milieu naturel*. Maspero, Paris, France, 325 pp.

Vacchi, M., Montefalcone, M., Bianchi, C.N., Morri, C., Ferrari, M., 2010. The influence of coastal dynamics on the upper limit of the *Posidonia oceanica* meadow. *Marine Ecology,* 31, 546-554.

Vacchi, M., Montefalcone, M., Bianchi, C.N., Morri, C., Ferrari, M., 2012. Hydrodynamic constraints to the seaward development of *Posidonia oceanica* meadows. *Estuarine, Coastal and Shelf Science,* 97, 58-65.

Vacchi, M., Montefalcone, M., Schiaffino, C.F., Parravicini, V., Bianchi, C.N. *et al*., 2013. Towards a predictive model to assess the natural position of the *Posidonia oceanica* seagrass meadows upper limit. *Marine Pollution Bullettin*, in press, http://dx.doi.org/10.1016/j.marpolbul.2013.09.038

Valle, M., Borja, Á., Chust. G, Galparsoro, I., Garmendia, J.M., 2011. Modelling suitable estuarine habitats for *Zostera noltii*, using ecological niche factor analysis and bathymetric LiDAR. *Estuarine, Coastal and Shelf Science,* 94, 144-154.

van Teeffelen, A., Moilanen, A., 2008. Where and how to manage: Optimal selection of conservation actions for multiple species. *Biodiversity Informatics*, 5, 1-13.

Wätzold, F., Drechsler, M., Armstrong, C.W., Baumgärtner, S., Grimm, V. *et al.,* 2006. Ecological-economic modeling for biodiversity management: potential, pitfalls, and prospects. *Conservation Biology,* 20 (4), 1034-1041.

Waycott, M., Duarte, C.M., Carruthers, T.J., Orth, R.J., Dennison, W.C. *et al.,* 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences,* 106 (30), 12377-12381.

Wilson, K. A., Underwood, E. C., Morrison, S. A., Klausmeyer, K. R., Murdoch, W. W. *et al*., 2007. Conserving biodiversity efficiently: what to do, where, and when. *PLOS biology*, 5(9), e223.