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Application and reassessment of the reduced species list index for macroalgae to assess the ecological status under the Water Framework Directive in the Atlantic coast of Southern Spain

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ABSTRACT

According to the Water Framework Directive (WFD) macroalgae are one of the Biological Quality Elements proposed to assess the ecological status of coastal water bodies. In the case of the North East Atlantic coastal shores, two methodologies have been implemented (RSL – reduced species list – in the U.K.; CFR – quality of rocky bottoms – in the Spanish Cantabric Sea). However, the ecological differences between these shores and the Atlantic coasts of Southern Spain imply a reassessment of these indices when applied to this water body. In this study, the RSL index has been reassessed for the rocky shores of the Atlantic coast of Andalusia (south-western Spain). In addition, an ecological and a morphological approximation to this index have been compared. After successive field sampling in the period 2006–2010, a reduced species list was developed for this shore. Based on anthropogenic pressures (water turbidity, nutrients, metal concentration and the distance to sources of stress), 19 sites along the coast were classified in five quality status (high, good, moderate, poor and bad) as proposed in the WFD. According to this classification the RSL index was calibrated. Finally, the results of the reassessed RSL-index were compared with the water quality. Overall, most of the elements yielded a significant relationship with the water quality and showed significant differences among the ecological quality classification. The less significant boundary among ecological status is the one lying between good and high. The results showed that both approximations of the RSL index were suitable to assess the ecological status, being the ecological approximation more suitable. Furthermore, the data analysis pointed out the existence of two coastal fringes with a different intertidal composition of algal species: Atlantic Cádiz and the Gibraltar Strait.

1. Introduction

Multiple activities producing very different stressors concur in coastal areas. Most of the national and international institutions have identified population density, urbanization, agriculture, tourism, industry, fisheries and marine transport as the main pressures on the coastal zone (Casazza et al., 2002; EEA, 1999; UNEP, 1996). These pressures can change the aquatic conditions producing different forms of pollution (e.g. acidification, eutrophication, heavy metals, invasions by alien species, pollution by organic compounds and by organic matter) and degrade the environment. In this sense, one of the main reasons that explain the regression of marine nearshore ecosystems is the organic and nutrient enrichment as a consequence of domestic wastes (Flechter, 1996). Hering et al. (2010) identified the eutrophication as the most important pressure in European marine ecosystems, being the reduction of nutrient loads the main restoration measure. This pressure can change the underwater light regime and substrate type (Nielsen et al., 2002; Schubert and Forster, 1997) implying a simplification of the architectural complexity of the communities (Arévalo et al., 2007). In addition, the increase of heavy metals introduced via polluted rivers, marine outfalls and through the deliberate dumping of wastes in coastal waters contributes to the overall deterioration of coastal ecosystems. In fact, anthropogenic releases of some heavy metals in aquatic ecosystems have been estimated to be up to three orders of magnitude greater than the natural inputs (Chase et al., 2001; Gheggour et al., 2002; Schindler, 1991). For these reasons, nutrients, turbidity and heavy metals are usually used to define the water quality. In this sense, a considerable effort has been made by the international community to monitor the distribution of nutrients, turbidity and heavy metals in the sea and to determine their effects on marine ecosystems. For instance, in the case of Andalusia (southern Spain) this monitoring has been carried out since 1988 onwards. However, analyses of

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water samples may give accurate, but local and transient information. Furthermore, this approach cannot determine the long-term effect of these pollutants on benthic communities (Licata et al., 2004).

Bioindicators have several noteworthy advantages compared to physico-chemical indicators. The most important is that bioindicators are a direct measurement of the pollution effects in organisms, which is often the main goal when assessing the effect of a pollutant. Secondly, bioindicators may indicate the long-term effects of pollutants in benthic communities when they cannot be measured or have disappeared from the environment (Licata et al., 2004). In addition, the use of bioindicators avoids drawbacks associated with a direct survey of contaminants in water samples, such as the need to periodically repeat numerous water drawings because of continuous movement of the waters and the fluctuation in contaminant levels (Ostapczuk et al., 1997). Therefore, the use of bioindicators can yield a more integrated response than physico-chemical indicators do.

Thus, the Water Framework Directive (WFD, 2000/60/EC) supports the use of biological indicators to assess water quality. Furthermore, to prevent further deterioration of marine habitats, WFD requires the assessment of the ecological status of surface waters to implement management plans. In the case of coastal water bodies, the ecological status has to be evaluated using different biological quality elements (BQEs; phytoplankton, macroalgae, marine angiosperms and benthic invertebrates), and supported by hydromorphological and physico-chemical quality elements. For the purposes of the WFD, the European coastal waters have been divided in relevant eco-regions (Mediterranean, Baltic, Black Sea and Atlantic) that include different biogeographic regions and subregions. In these regions, the coastal waters have been classified according to environmental characteristics to delimit different types (IES, 2009). In spite of this practical classification, it is of little value to develop a single indicator, even based on the same BQE, to assess the ecological status of all coastal waters of the same eco-region. There are biogeographical differences in these large eco-regions, which may not be acknowledged by indices that are developed in particular areas. For instance, in the case of the BQE macroalgae, Guinda et al. (2008) pointed out that although the WFD considers the Northeast Atlantic as an entire eco-region, the large marine ecosystems project (LME), initiated to support the global objectives of Agenda 21, clearly distinguishes the Iberian coastal marine ecosystem from northern coastal areas (EEA, 2006). Accordingly, when a biological indicator designed in one sub-region is used in others, this indicator may be reassessed.

The use of macroalgae as bioindicators to assess pollution in the marine environment has been proved successful in many ecological studies (e.g. Borowitzka, 1972; Díez et al., 1999; Gorostiaga and Díez, 1996). The sedentary condition of attached macroalgae integrates the effects of long-term exposure to nutrients and/or other pollutants resulting in a decrease or even disappearance of the most sensitive species and their replacement by highly resistant, nitrophilic or opportunists species (Díez et al., 1999; Murray and Littler, 1978; Tewari and Joshi, 1988). Therefore, macroalgal communities arise as a useful tool to analyze changes in water quality (Fairweather, 1990). Furthermore, as macroalgal communities provide habitat and canopy cover for a wide variety of intertidal organisms (e.g. Pavia et al., 1999), changes in these communities will have significant effects on shore ecosystems (e.g. Hereu, 2004). For these reasons, the WFD proposed, among others BQE (see above), the use of composition and abundance of macrophyte communities to develop bioindicators to assess ecological quality of European coastal waters.

So far, two indices, based on the study of macroalgal communities along intertidal rocky shores, have been proposed for Atlantic coastal waters: reduced species list (RSL; Wells, 2008; Wells et al., 2007) and quality of rocky bottoms (CFR; Guinda et al., 2008; Juanes et al., 2008).

The RSL index utilises five elements to describe ecological status: species richness of a reduced species list; proportion of red algae; proportion of green algae; ESG (ecological status group) ratio, and proportion of opportunist species (Wells et al., 2007). The RSL index is based on species occurrence while CFR index uses the relative abundance of species. This fact is very important when results are analyzed, because the sensibility and spatio-temporal scale depend on it. For this reason RSL is less sensitive but more robust and can be used in meso-scale studies (Bermejo, 2009). Furthermore, the RSL index is less subjective and more precise than the CFR index (Guinda et al., 2008). These characteristics suggest that the RSL index may be more suitable to assess the ecological status of coastal waters. In spite of this, the preliminary results obtained for this index in the northern coast of Spain were worse than the result obtained for CFR when a pollution gradient was assessed (Guinda et al., 2008). However, the same authors proposed that to achieve a good calibration and validation of both indices, further analyses should be carried out at a different geographical location and against different types of pollution sources.

Some elements used in the RSL-index have been previously discussed (Arévalo et al., 2007; Guinda et al., 2008). For instance, the classification of species in two ESG, based on the functional form groups of macroalgae proposed by Littler and Littler (1980) and Littler et al. (1983) may have some limitations, as functional forms were originally suggested to predict productivity and other ecological attributes (e.g. grazing resistance, competitive abilities, reproductive effort); however, the resistance to pollution cannot be directly deduced from morphological features of the species (Arévalo et al., 2007). This has sometimes led to assign a particular species to different ESG (e.g., *Corallina*, ESG-I by Orfanidis et al., 2001 and ESG-II by Ballesteros et al., 2007) or to give them an opportunistic character (e.g. *Ceramium,* considered opportunist by Guinda et al., 2008 and non-opportunist by Wells et al., 2007). On the other hand, the proportion of rhodophyta evidenced problems in adjusting due to the biogeographical and ecological differences between northern cold and southern temperate waters (Guinda et al., 2008).

Therefore, in this framework, this study pursues a double objective: (i) to apply and reassess the RSL index to the Atlantic coast of southern Spain and (ii) to compare the values of this index with the previous classification of the water quality based on concentration of nutrients, metals turbidity and distance to sources of stress at a spatial meso-scale.

2. Materials and methods

From March of 2008 to April of 2010, 19 sites located along the Atlantic coast of southern Spain were sampled (Fig. 1). The field samplings were carried out during spring and summer, coinciding with the peak growth of littoral communities (Ballesteros, 1992). In each site, a stratified sampling, registering all subhabitats, was carried out to obtain a macroalgal species list (Wells et al., 2007). Each sampling lasted approximately 1 h and was carried out during the low tide along 50–60 m width of the whole rocky intertidal shore. When identification of specimens *in situ* was impossible, they were taken to the laboratory for a detailed observation. The taxonomical nomenclature used followed AlgaeBase (Guiry and Guiry, 2010). At each sampling site, physical characterization of the shore was estimated according to Wells et al. (2007).

2.1. Reduced species list

A reduced species list for Atlantic coasts of Andalusia was elaborated from the full species list obtained at each site. According