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Setting the maximum ecological potential of benthic communities, to assess ecological status, in heavily morphologically-modified estuarine water bodies

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ABSTRACT

Investigations on setting benthic macroinvertebrates reference conditions in natural waters have increased recently. Under the European Water Framework Directive, importance is given to research in morphological heavily-modified water bodies (HMWBs), which are very common in countries with high human pressure. However, research has not been undertaken on setting the maximum ecological potential (MEP), as a reference in HMWB. The objective of the present investigation is to set the MEP of two metrics (diversity and richness), used in assessing the ecological status in different benthic indices. The Oiartzun estuary (Basque Country) is used as a case study, which changed morphologically in the 19th Century, following harbour construction. Data obtained from 1874 and the present were used to model changes in currents, water residence time, salinity, volume, and intertidal area. Benthic macroinvertebrate data, from 1995 to 2011, were used to predict 19th Century and present MEP. Changes in the estuary were described: loss of all of the intertidal areas; doubling of the volume; residence time, changing from 2 to 95 days; current velocity reduced by 50%; salinity increase. All these factors have led to changes in the benthic communities and the structural variables. Predicted richness and diversity, for 1874, were lower (48–76%) than those at present. Taking into account the differences between natural and modified waters, it is proposed to utilize 75% of the natural reference conditions, as the MEP values for Basque HMWB.

1. Introduction

The European Water Framework Directive (WFD; 2000/60/EC) has several objectives, for example: to prevent water ecosystems deterioration and to protect and enhance the status of water resources. However, the most important is to achieve a 'good ecological status' (GES) for all waters, by 2015.

Ecological quality is based upon the status of the biological (phytoplankton, macroalgae, macrophytes, benthos and fishes), hydromorphological and physico-chemical quality elements. Consequently, many methodologies have been proposed in recent years to assess the ecological status of water bodies, within the WFD (Birk et al., 2012).

Most of these methodologies were developed for its use within natural water bodies, in which the ecological status is a perceived, or measured, deviation from a reference condition (Hering et al., 2010). However, the WFD also defines and considers 'Artificial Water Bodies', i.e. those created by human activity, such as an artificial lake, and 'Heavily Modified Water Bodies' (HMWBs), a water body resulting from physical alterations by human activity, which substantially change its hydrogeomorphological character, e.g. a harbour. In both cases of definition, Member States may designate a body of surface water as being 'artificial' or 'heavily modified' when there are significant adverse effects to the hydromorphological characteristics of that body, which otherwise would be necessary for achieving GES. Such effects include: (i) the wider environment; (ii) navigation, including port facilities, or recreation; (iii) activities for the purposes for which water is stored, such as drinking-water supply, power generation or irrigation; (iv) water regulation, flood protection, land drainage; or (v) other equally-important sustainable human development activities.

In terms of implementing the WFD, environmental managers are required to assess the status of HMWB, in relation to achieving at least 'Good Ecological Potential' (GEP). A water body shows a GEP when there are slight changes in the values of the relevant abovementioned biological quality elements, compared to the values found at the maximum ecological potential (MEP). The MEP is considered as the reference condition for HMWB; it is intended to describe the best approximation to a natural aquatic ecosystem, which could be achieved given the hydromorphological characteristics that cannot be changed without significant adverse effects on the specified use, or the wider environment (CIS, 2003a, 2003b; Borja and Elliott, 2007). The meaning of potential is something that can only be achieved if something else happens e.g. the

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hydromorphological causes of change are removed. Accordingly, the MEP biological conditions should reflect, as far as possible, those associated with the closest comparable natural water body type, at reference conditions; in accordance with the established hydromorphological and associated physico-chemical conditions.

Although the WFD implementation process for natural waters has experienced important advances since approval of the Directive (Hering et al., 2010), at present only minimal advancement has taken place, in terms of understanding the meaning of GEP, especially within an ecological context; likewise, how to define MEP (CIS, 2006; Wetzel et al., 2012). However, Borja and Elliott (2007) have proposed an approach to establish the MEP, through several steps, as outlined below:

(i) Select the appropriate quality elements for MEP, identifying the closest comparable natural surface water category.
(ii) Establish the hydromorphological conditions required for the MEP, which determine the values for the biological and general physico-chemical quality elements.
(iii) Establish the MEP physico-chemical conditions, identifying the closest comparable surface water body type, taking into account the MEP hydromorphological conditions.
(iv) Finally, establish the MEP biological conditions that reflect, as far as possible, those associated with the closest comparable water body type.

In the case of natural coastal and transitional waters, much research has been undertaken in setting reference conditions and boundaries for quality classes, especially in terms of benthic macroinvertebrates (see Borja et al. (2009a), for a review of European methods, and Borja et al. (2012), for the problems associated with the setting of reference conditions). After the WFD, any macroinvertebrates assessment must be based upon richness, diversity and the ratio between opportunistic and sensitive species. One of the methods being used by many European countries, in determining this ratio, is the AZTI's Marine Biotic Index (AMBI), as described in Borja et al. (2000). More recently, amongst the methods used in assessing benthic status (see Borja et al., 2009a), the multivariate-AMBI (M-AMBI; Muxika et al., 2007) is being used in several countries; this includes the three single metrics mentioned above (diversity, richness and the AMBI). From these metrics, AMBI (as the ratio of opportunistic/sensitive species) values for MEP should not differ considerably from those required in natural waters, because most of the sensitive taxa of the type-specific communities should be present in the HMWB, under good conditions. However, richness and diversity will change, compared to natural conditions.

Hence, the objective of the present investigation is to set the reference conditions (i.e. MEP) of these two single metrics (diversity and richness), to assess the ecological potential of benthic macroinvertebrates in an HMWB, following the approach proposed previously by Borja and Elliott (2007).

2. Materials and methods

2.1. Study area and main human pressures

In order to investigate this particular topic, an HMWB in the Basque Country, southeastern Bay of Biscay, has been selected for investigation. This area is the Oiartzun estuary, which was modified considerably to create the Pasaia Harbour (Fig. 1). The morphology of this estuary has changed dramatically since the mid-19th Century, following the development of the harbour (Reizabal et al., 1987; Rivas and Cendrero, 1992). Dredging activities, to maintain the adequacy of the navigation channels, together with the building of the port infrastructures, have resulted in morphological changes of the natural environment (Uriarte et al., 2004; Tueros et al., 2009; Montero et al., 2013). Furthermore, the estuary has been highly contaminated by: industrial development on the area, e.g. with a thermal power station and a paper mill; and the settlement of several villages, with hydrometallurgical- and mining-related discharges (Cantón and Legorburu, 1991).

The Oiartzun estuary is 5.5 km in length, with a total surface of about 1 km^2 and an average water depth of around 10 m (Valencia et al., 2004). The Oiartzun river flows into this estuary, and drains an 87 km^2 area and has an annual mean flow of 4.8 $m^3 s^{-1}$ (Borja et al., 2006).

The estuary is sheltered and the main natural driving force is the tide, which is semidiurnal in the Bay of Biscay. The maximum spring tidal range exceeds 4.5 m. The region is defined as 'low meso-tidal' during neap tides, but 'high meso-tidal' during springs (González et al., 2004).

2.2. Approach followed in the study and the data used

The harbour is located within the North-East Atlantic (NEA)-1/26 typology of the WFD. Following the approach discussed in the Introduction (based upon Borja and Elliott, 2007), a further series of steps were completed (see below).

2.2.1. Benthic macroinvertebrates data

As macroinvertebrates are the biological quality element to be analysed for the MEP, some old benthic data extracted from Navaz (1948) and for the present (1995–2011), were obtained for the Oiartzun estuary and other 11 estuaries from the Basque Country (see sampling details in Borja et al., 2009b). The use of data from other Basque estuaries is related to the fact that the present MEP conditions, from the HMWB, should be associated with the closest comparable water body type.

2.2.2. Modification of the hydromorphological conditions

The study of this modification, required for the MEP, was based upon changes in the docks and seabed within the estuary; these produce distinct alterations in water residence times, tidal prisms, etc. In order to estimate such changes, existing bathymetric and morphological charts of the previous and present configuration of the estuary were used. The oldest nautical chart available for the Oiartzun estuary, published by the Spanish Directorate of Hydrography in 1874, was digitized. As it was neither georeferenced, nor scaled, a more recent and referenced nautical chart published by the French *Service Hydrographique et Océanographique de la Marine*, in 1992, was used for rectification purposes (6375 Chart, Port de Pasajes, at 1:7500). Hence, 54 common points were identified on both of the charts; these permitted a graphical rectification of the 1874 chart. Although 118 years separate the charts, some common locations were identified, i.e. a lighthouse, several castles, churches, the train rail and the non-modified coastline of Pasajes San Juan village. Additionally, some bathymetric information gaps, as detected for the inner part of the estuary, were completed using a bathymetric chart published in 1909. Fig. 2 shows the overlapping of the rectified 1874 nautical chart, with the coastline published in 1992. With respect to the present bathymetric and morphological characterization of the estuary, the bathymetric data from a multibeam survey undertaken on 28th July 2009 for the Port Authority of Pasajes, together with the coastline published in 1992, were used.

Intertidal and subtidal volumes and areas were calculated for the past and present configurations of the estuary. To achieve this objective, two 5 m resolution Digital Terrain Models (DTM) were constructed: (i) Pasajes 1874: 5 m grid DTM referred to the hydrographical zero at that time; and (ii) Pasajes 2009: 5 m grid DTM referred to the local hydrographical zero. It is of note that intertidal