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Capacity for recovery of rocky subtidal assemblages following pollution abatement in a scenario of global change

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A B S T R A C T

The successful protection and management of marine ecosystems depend on understanding the capability of biota for recovering after stressor mitigation actions are taken. Here we present long-term changes (1984–2012) in degraded subtidal assemblages following the implementation of the sewerage scheme for the metropolitan area of Bilbao (1 million inhabitants). Qualitative and quantitative species composition of disturbed vegetation shifted over time, making it more similar to that of the reference assemblages considered. Species density in the disturbed habitats increased, which is also a positive sign of recovery. However, eleven years after the clean-up was completed, canopy-forming macrophytes showed no signs of recovery. We argue that the ecological resilience of the ecosystem may have been eroded after a long-standing pollution perturbation and that underlying climate change could be influencing the recovery trajectory of the degraded assemblages. The implications of these conclusions for the implementation of European marine environmental legislation are discussed.

1. Introduction

Increasing cumulative impacts of human populations in coastal areas are leading to broad degradation of ecosystems all around the world (Halpern et al., 2008). Anthropogenic drivers threatening ecosystem functions and biodiversity include pollution, habitat destruction, over-exploitation, invasion by alien species and, more recently, climate change (Crain et al., 2008). Scientists and environmental policy-makers devote substantial efforts to protecting natural environments from further harm and repairing impacts already caused (Hobbs, 2007). Pollution removal is one of the imperative human-mediated actions required to restore coastal ecosystems. But to prevent deterioration and protect ecosystems, knowledge of how aquatic assemblages respond after stressors mitigation actions are taken is also essential (Verdonschot et al., 2013). Despite great efforts and significant investment to restore water quality, assemblage responses following pollution abatement are poorly understood (Tsiamis et al., 2013; Veríssimo et al., 2013).

In relation to macroalgal assemblages, several studies have documented that water quality improvement promotes increases in species richness and diversity (Bonk et al., 1996; Soltan et al., 2001; Díez et al., 2009), and leads to decreases in the abundance of ephemeral algae (Archambault et al., 2001; Pinedo et al., 2013; Tsiamis et al., 2013). What remains unclear, however, is whether a complete recovery of benthic communities can be attained (Díez et al., 2013; Pinedo et al., 2013). One major concern is the potential restoration of canopy-forming macroalgae that provide habitat, shelter and food for a plethora of organisms in temperate coastal rocky habitats (Schmidt and Scheibling, 2007). These habitat engineers are highly susceptible to pollution (Thibaut et al., 2005) and a decline in their populations, sometimes to the point of disappearance, as result of sewage effluent discharges has been a worldwide trend (Sfriso, 1987; Brown et al., 1990; Kautsky et al., 1992; Munda, 1993). By contrast, the restoration of canopy-forming macroalgae appears not to be easily achievable (Coleman et al., 2008; Díez et al., 2009). Suggested causes underlying delays in their recovery include pre-emptive competition, particularly with algal turfs (Bellgrove et al., 2010) and low dispersal potential (Sales et al., 2011). The recovery of macroalgal canopies is influenced by numerous factors, including the nature, magnitude and duration of environmental perturbation (Borja et al., 2010), the distance from sources of potential colonists (Bellgrove et al., 2010), the presence of concomitant stressors (Bertocci et al., 2010) and the reproduction, recruitment and growth characteristics of canopy species (Crowe et al., 2013). Furthermore, the feedback between biotic and abiotic factors that has developed in a degraded ecosystem during the perturbation may alter its capacity for recovery (Suding et al., 2004). It has been hypothesised that the

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loss of canopy-forming macroalgae may reduce the ecological resilience of assemblages, in such a way that degraded ecosystems may permanently shift to alternative stable states dominated by simpler algae, mussels or barrens (Perkol-Finkel and Airoldi, 2010).

Quantifying recovery success in natural ecosystems is a challenging task, because it requires the potential changes in the recovered community to be isolated from noise introduced by a background of natural variability (Veríssimo et al., 2013). Abiotic factors and biotic interactions operating simultaneously over different spatial and temporal scales generate complex variability in ecosystems (Smale et al., 2010). To assess the effectiveness of pollution mitigation measures, it is necessary to measure the natural change at multiple reference locations that represent what it is sought to achieve by restoration, and compare this natural change with that undergone by the putatively recovered system (Chapman, 1999). Long-term monitoring programmes are essential for achieving this purpose, since recovery is often a complex or nonlinear process that may require long periods of time (Hawkins et al., 2002; Borja et al., 2010) and natural communities exhibit great temporal variability (Sousa-Dias and Melo, 2008). However, long-term monitoring programmes that evaluate the recovery of degraded ecosystems following the removal of a stressor are relatively rare (Clements et al., 2010). Moreover, the ongoing global environmental change makes distinguishing biological recovery from underlying natural variability even more difficult (Veríssimo et al., 2013). Global-scale trends such as warming and ocean acidification, along with region-specific changes in storminess, upwelling patterns, terrigenous nutrient runoff and coastal salinity are profoundly influencing the survival, recruitment, growth and reproduction of seaweeds from kelp forests to coral reefs (Harley et al., 2012). In the particular case of the Cantabrian Sea, environmental changes are being documented by an increasing number of studies. In line with rising global temperatures, summer surface water temperatures off the coast of Asturias increased by 0.52 °C from 2002 to 2012 (Voerman et al., 2013). Moreover, it has been suggested that the Cantabrian Sea is becoming less productive as result of decreases in upwelling intensity and stronger thermal stratification (Llope et al., 2007; Valdés et al., 2007). These environmental changes could be responsible for the shifts reported in the distribution patterns of species along the northern coast of Spain (Fernández, 2011; Voerman et al., 2013). Declines in kelp populations (Fernández, 2011; Díez et al., 2012; Voerman et al., 2013) and increases in warm temperate flora are among the changes detected (Díez et al., 2012; Voerman et al., 2013).

In the present study we use robust data series (from 1984 to 2012) on subtidal macrophytes to track recovering trajectories of benthic assemblages following the implementation of pollution mitigation actions (particularly, the sewerage scheme for the metropolitan area of Bilbao, which has 1 million inhabitants) over a period of climate variability. We examine changes in macrophyte assemblages at 4 disturbed locations, and contrast them with those undergone by assemblages at 8 unpolluted reference locations. We test the following specific hypotheses: (1) degraded assemblages are more dissimilar to those of reference sites at the start than at the end of the study period in terms of their multivariate structure (i.e., quali-quantitative species composition), diversity, percentage cover of functional groups and percentage cover of life forms and (2) biological recovery depends on the community measure examined. We also discuss the following questions: (1) can the system recover on its own or is some degree of active restoration required? and (2) could the climate variability be affecting the recovery trajectory and "end-point" of disturbed assemblages? these issues have important implications for conservation and for the achievement of protection goals established by EU environmental legislation.

2. Methods

2.1. Study area

The study area extends approximately 100 km along the western Basque coast, in the southernmost part of the Bay of Biscay (Fig. 1). It is mostly erosional with extensive cliffs (Cearreta et al., 2004). Mean sea surface temperature (SST) off the Basque coast used to range between 12 °C in February and 22 °C in August (Valencia et al., 2004). However, an increase has been detected in the last few decades, particularly in summer, when SST is found to have risen by as much as 1 °C from 1980 to 2008 (Díez et al., 2012). The average summer temperatures above the 75th percentile increased significantly from 22.4 °C to 23.2 °C during this period. The maximum temperatures reached in summer also increased, and exceeded 25 °C in the summers of 1997, 2003 and 2006 (Díez et al., 2012). The coast is also exposed to strong swell, (coming mainly from the NW) which has intensified during the last ten years (Borja et al., 2013). These authors found that the number of waves >5 m height had increased significantly between 1993 and 2012 in the Basque coast, with <60 waves being recorded before 2003 and 100–130 after that time (with a maximum of 221).

The Nervión river flows into its westernmost part. This river runs through the city of Bilbao, located 15 km upstream from its mouth. In the second half of the nineteenth century and throughout the twentieth century, Bilbao underwent heavy industrialization accompanied by a sharp rise in population. By 2000 almost 80% of the population of Biscay (around 1 million people) lived in the metropolitan area of Bilbao (Díez et al., 2000). Before 1990 there was no wastewater treatment in the area and the Nervión estuary received some 250,000 $m^3\ d^{-1}$ of urban wastewater and 67,000 $m^3\ d^{-1}$ of industrial water (coming mainly from chemical, iron, steel and paper-manufacturing firms) highly charged with toxic products (Borja et al., 2006). This waste dumping led to gross environmental deterioration with harmful consequences for the estuarine (Bustamante et al., 2007) and marine biota (Díez et al., 1999; Pagola-Carte and Saiz-Salinas, 2001a). Water oxygen deficiency led to the virtual elimination of all heterotrophic life in the estuarine sediments (Saiz-Salinas and González-Oreja, 2000) and subtidal macrophytes almost disappeared close to the mouth of the Nervión river (Gorostiaga and Díez, 1996). To reverse this situation, a general sewerage scheme for the metropolitan area of Bilbao was progressively applied by the local water authority Consorcio de Aguas Bilbao-Bizkaia. In 1990 physico-chemical water treatment was introduced, and in 2001 sewage treatment was upgraded with the addition of biological treatment. Moreover, some of the most polluting factories and mineral washeries shut down during the 80s and 90s. As a consequence, the dissolved oxygen content of the Nervión estuary progressively increased (Villate et al., 2013), water (Garcia-Barcina et al., 2006) and sediment quality improved (Bartolomé et al., 2006; Fernández-Ortiz de Vallejuelo et al., 2010) and benthic (Pagola-Carte and Saiz-Salinas, 2001b; Borja et al., 2006; Díez et al., 2009), plankton (Villate et al., 2004) and fish communities (Uriarte and Borja, 2009) gradually recovered.

2.2. Sampling locations and data collection

Polluted locations

Prevailing winds are largely from the northwest, so the pollution plume from the Nervión river is carried eastwards. Four impacted locations were chosen following this pollution gradient: Arrigunaga, La Galea, Meñakoz and Matxilando (Fig. 1). At each location eleven surveys were carried out over the period 1984–2012. Samplings were conducted in the summer months.