Environmental Pollution 191 (2014) 17–24

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

Phytoplankton community indicators of changes associated with dredging in the Tagus estuary (Portugal)

Maria Teresa Cabrita

Portuguese Institute of Sea and Atmosphere (IPMA), Av. de Brasília, 1449-006 Lisboa, Portugal

ARTICLE INFO

Article history:
Received 30 September 2013
Received in revised form
21 February 2014
Accepted 1 April 2014
Available online 5 May 2014

Keywords:
Phytoplankton community indicators
Environmental assessment
Dredging
Estuary

ABSTRACT

This work reports changes in suspended particulate matter, turbidity, dissolved Cr, Ni, Cu, Cd, Hg and Pb concentrations, and phytoplankton biomass and composition during a 5-month period dredging operation, in a trace element contaminated area of the Tagus estuary (Portugal). Phytoplankton biomass, diatom:other groups ratio, benthic:pelagic diatom ratio, Margalef's, Simpson's diversity, Shannon–Wiever's, and Warwick and Clarke's taxonomic diversity and distinctness indices, and individual taxa were investigated as indicators of dredging induced changes. Significant rise in sediment resuspension and trace element mobilisation caused by dredging influenced the community structure but not the overall biomass. Benthic diatom displacement into the water column maintained species diversity, and therefore, none of the indices highlighted community changes. Contrastingly, diatom:other groups ratio and benthic:pelagic diatom ratio were reliable indicators for the assessment of dredging induced changes. A shift in composition towards species less susceptible to trace elements was observed, disclosing some individual taxa as potential indicators.

1. Introduction

Dredging is commonly required to maintain navigation of ports and waterways in estuaries (Kenny and Rees, 1994). Impacts of dredging in the water column, resulting from excavation and removal of the bottom sediments, are the resuspension of sediments with an associated increase in turbidity (Newell et al., 1998). Additional impact may arise from dredging in polluted estuarine areas with the release of dredged anoxic sediments into oxygenated waters that triggers particle surface reactions, in particular oxidation of metal sulphides (Caetano et al., 2003). The result is the release of cations to the water column, commonly observed in several estuaries subject to dredging (Vale et al., 1998; Caetano et al., 2003), which eventually become biologically available (Monteiro et al., 1995; Cabrita et al., 2013). The impact of this mobilisation on estuaries is of particular concern due to the persistent toxicity of several trace elements (Pan and Wang, 2012).

The development of reliable indicators for detecting dredging induced changes in water quality and ecological condition in estuarine systems is essential because of increasing anthropogenic pressures and the need for more effective regulation (Rogers and Greenaway, 2005). A wide range of analytical tools for measuring physical, chemical and biological shifts in estuaries is available (Danilov and Ekelund, 2001). However, difficulties arise when trying to identify the most informative, sensitive and reliable indicators to employ in a particular area (Salas et al., 2006). Phytoplankton has been proposed as an efficient and easily detectable indicator of water quality and ecological changes because they are sensitive to various environmental stressors (Paerl et al., 2007) and respond rapidly to changes in the water column (Rainbow, 2006). The structure of the phytoplankton community has been widely used as indicator of ecological condition for the evaluation of a range of impacts, for instance, eutrophication (Tas et al., 2009) and climatic change (Paerl et al., 2010), but has seldom been applied in dredging assessments. Previous studies showed that phytoplankton indicators provided valid information on the level of eutrophication in a dredged lake (Wang et al., 2012; Xu and Pan, 2013). However, suitable phytoplankton indicators need to be identified for dredging monitoring in estuarine systems contaminated with metals, taking into account the local hydrologic conditions and the natural variability of the estuarine phytoplankton assemblages in a given area.

The present work investigates effective and reliable phytoplankton community indicators of changes in water quality triggered by dredging. It is hypothesized that species richness is maintained during dredging due to benthic contribution and that diatom:other groups ratio, benthic:pelagic diatom ratio and

E-mail addresses: maria.teresa.cabrita@gmail.com, tcabrita@ipma.pt.

http://dx.doi.org/10.1016/j.envpol.2014.04.001
0269-7491/

individual taxa may be more adequate and efficient indicators of changes associated with dredging than other indicators. The present work examines phytoplankton biomass, diatom:other groups ratio, benthic:pelagic diatom ratio, Margalef's, Simpson's diversity, Shannon–Wiever's, and Warwick and Clarke's taxonomic diversity and distinctness indices, and individual phytoplankton taxa as indicators of water quality changes induced by a 5-month dredging operation, in a trace element contaminated area of the Tagus estuary (Portugal).

2. Materials and methods

2.1. Environmental setting

Water sampling was performed in a sheltered bay (0.4 km^2, 3–7 m depth) located in the Tagus estuary (Portugal), where approximately 10^6 m^3 of sediments were intermittently dredged over a five-month period. Dredging was performed with a cutter suction dredger that dislodged bottom materials with a rotating cutter equipped with cutting teeth. The purpose of dredging was the maintenance of the depth and the deepening of existing navigable waterways and harbour basins in order to allow larger ship access. Non-contaminated dredged sediments were discarded at the mouth of the estuary and contaminated materials were disposed in open waters adjacent to the Tagus estuary coastal area. Relatively elevated levels of Cu, Zn, As, Hg and Pb have been detected in surface sediments of confined areas of the Tagus estuary (Canário et al., 2005). In particular, the dredged bay has been a historical repository of contaminants from industries situated in the southern margin (Bettencourt, 1988; Canário et al., 2005; Vale et al., 2008), and urban effluents (Hampel et al., 2009).

2.2. In situ observations and sampling

In situ observations and water sampling were performed in a site at a distance of 0.3–0.5 km to the dredging area in the following days: 1, 7 and 14 (before dredging, April 2010), 21, 44, 49, 75, 84, 89 and 96 (during the dredging period, May–July 2010), and 281 and 287 (after dredging, January and February 2011). Measurements and sampling were always performed at high tide to avoid any bias due to changes in phytoplankton community associated with the semi-diurnal tidal cycle. Temperature (Temp), salinity (S), pH, dissolved oxygen (DO) and turbidity of surface water (0.5 m depth) were determined *in situ* with a 650 MDS (YSI Incorporated). Surface water was sampled in triplicate to 2-L polypropylene bottles for the determination of dissolved Cr, Ni, Cu, Cd, Hg and Pb. Surface water was also sampled in triplicate to 5-L polyethylene bottles for suspended particulate matter (SPM), nutrient (NH_4^+, $NO_3^- + NO_2^-$, PO_4^{3-} and $Si(OH)_4$) and chlorophyll *a* (Chl *a*) determination. Additionally, 200 mL water samples were taken and preserved with Lugol's solution immediately after collection for phytoplankton identification and cell counts.

2.3. Analytical determinations

2.3.1. Suspended particulate matter (SPM) and nutrients

SPM was obtained by filtration in polycarbonate filters (0.45 μM) and determined gravimetrically. For the determination of nutrients (NH_4^+, NO_3^-, NO_2^-, PO_4^{3-} and $Si(OH)_4$), water samples were filtered through MSI Acetate Plus filters and frozen for later analysis on a autoanalyser SKALAR 6250, according to Grasshoff et al. (1983).

2.3.2. Trace element concentrations

At the lab, a diffusive gradient in thin-films (DGT) was suspended with a nylon thread inside each 2-L polypropylene bottle for determination of dissolved trace elements (Cr, Ni, Cu, Cd and Pb). All DGT holders, Chelex-100 resins and diffusive gels (type APA, 0.8 mm thickness, open pore) (Zhang and Davison, 1999) were purchased from DGT Research. After a 48 h period, the DGTs were carefully removed from water and resins were immersed in 5 mL of 1 M HNO_3. Element concentration were directly quantified in resin eluates by an Inductively Coupled Plasma Mass Spectrometer, ICP-MS (Thermo Elemental, X-Series), equipped with a Peltier impact bead spray chamber and a concentric Meinhard nebuliser. All eluates were analysed with reagent blanks to control eventual contaminations during the analytical procedure, and with an international standard of river water (SLRS-5, from the National Research Council of Canada) to control the accuracy of the procedure. Filtered and non-filtered water samples were analysed for Hg, in a cold-vapour atomic fluorescence spectrometry (PS Analytical). Water samples for dissolved Hg were filtered though 0.45 μm cellulose acetate membrane capsules (FP30 Whatman, 0.45 μm) and then preserved according to EPA 1631 (2002).

2.4. Phytoplankton

Phytoplankton biomass was measured as chlorophyll *a* (Chl *a*) concentration. Water samples were filtered through Whatman GF/F filters which were immediately frozen at −20 °C. Chlorophyll *a* was determined spectrophotometrically, using the method of Lorenzen (1967). Phytoplankton identification and cell counts were performed following the sedimentation method of Utermöhl (1958). In water samples with high sediment particles content, overlapping of cells and sediment particles is likely to occur, due to their similar sizes. To minimize the underestimation of small phytoplankton species, 10-mL water samples were used in counting chambers. The phytoplankton quantitative and qualitative analyses were carried out under an inverted microscope AXIOVERT 135 (Zeiss, Germany).

The species richness index used was the Margalef's index (d) (Margalef, 1958). The diversity indices employed were the Simpson's diversity, the Shannon–Wiever's, the Warwick and Clarke's taxonomic diversity and distinctness indices. All indices were calculated using absolute number data with PRIMER (Plymouth Routines In Multivariate Ecological Research) Version 6.0 software package. The Simpson diversity index ($1-D$) (Simpson, 1949) ranges between 0 and 1, so that the larger the value, the greater the sample diversity. The Shannon–Wiever index (H') (Shannon and Weaver, 1964) increases with rising richness community, values of this index are usually between 1.5 and 3.5 and rarely surpass 4.5. The Warwick and Clarke's taxonomic diversity (δ) and distinctness (δ^*) indices (Warwick and Clarke, 1995) may range from 0 (only one species present) to 100 (highest taxonomical distance between all species).

2.5. Statistical analysis

Differences in environmental data between no dredging and dredging conditions were compared through Kruskal–Wallis nonparametric test, performed with Statistica 6.1 Software (StatSoft, Inc.). Results yielding $p \leq 0.05$ were considered statistically significant. Principal Component Analysis (PCA) was applied to disclose association patterns in the individual species abundance data related to water

Table 1
Values of temperature (T, °C), salinity (S), pH, dissolved oxygen (DO, %), turbidity (NTU), suspended particulate matter (SPM, mg L^{-1}), dissolved inorganic nitrogen (DIN, μM), phosphate (HPO_4^{2-}, μM), silicate ($Si(OH)_4$, μM), DIN:P, Si:DIN, and Si:P, and Cr, Ni, Cu, Cd, Hg, Pb concentration (μg L^{-1}), at surface water of the sampling area (Tagus estuary), before, during and after dredging periods. Average, and minimum and maximum intervals are presented.

	Before dredging	During dredging	After dredging
T (°C)	17 (17–19)	21 (17–24)	11 (11–12)
S	26 (22–27)	30 (25–32)	22 (20–24)
pH	7.7 (7.4–8.0)	7.9 (6.6–8.4)	8.1 (8.0–8.2)
DO (%)	100 (100–112)	102 (94–120)	101 (100–101)
Turbidity (NTU)	25 (21–25)	30 (26–31)	23 (21–25)
SPM (mg L^{-1})	5.0 (3.9–11)	8.0 (4.4–13)	6.9 (5.2–11)
DIN (μM)	21 (13–37)	21 (17–25)	23 (17–29)
HPO_4^{2-} (μM)	1.1 (0.90–1.7)	1.0 (0.60–1.2)	1.2 (1.1–1.3)
$Si(OH)_4$ (μM)	30 (14–45)	12 (6.6–18)	32 (30–35)
DIN:P	14 (13–33)	22 (14–39)	18 (15–22)
Si:DIN	2.1 (0.39–2.3)	0.59 (0.28–1.1)	1.5 (1.2–1.8)
Si:P	27 (13–33)	13 (10–15)	26 (26)
Cr (μg L^{-1})	0.20 (0.044–0.30)	0.38 (0.13–0.97)	0.023 (0.020–0.026)
Ni (μg L^{-1})	0.62 (0.15–0.92)	1.1 (0.44–1.8)	0.27 (0.21–0.32)
Cu (μg L^{-1})	0.41 (0.37–0.41)	1.3 (0.25–2.7)	0.20 (0.14–0.27)
Cd (μg L^{-1})	0.020 (0.023–0.21)	0.030 (0.014–0.038)	0.012 (0.0092–0.016)
Hg (μg L^{-1})	0.0050 (0.0010–0.014)	0.018 (0.0034–0.086)	0.0039 (0.0027–0.051)
Pb (μg L^{-1})	0.021 (0.069–0.060)	0.14 (0.025–0.36)	0.021 (0.017–0.025)