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Mesozooplankton affinities in a recovering freshwater estuary

Sophie Chambord [a,b,c,*], Tom Maris [c], Fanny Colas [a], Tom Van Engeland [c,d], Akoko-C. Sossou [a], Frédéric Azémar [a], Maïwen Le Coz [a,b], Tom Cox [c,d], Laetitia Buisson [a], Sami Souissi [b], Patrick Meire [c], Michèle Tackx [a]

[a] ECOLAB, Université de Toulouse, CNRS, INPT, UPS, Toulouse, France
[b] Université Lille 1 – CNRS, Université du littoral Côte d'Opale, UMR 8187 LOG, Laboratoire d'Océanologie et de Géosciences, 62930 Wimereux, France
[c] University of Antwerp, Ecobe (Ecosystem Management Research Group), Universiteitsplein 1C B2610 Antwerpen, Belgium
[d] NIOZ Royal Netherlands Institute for Sea Research, Korringaweg 7, Yerseke, The Netherlands

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A B S T R A C T

Water quality of the Scheldt estuary (Belgium/The Netherlands) has considerably improved in recent years, especially in the upstream, freshwater reaches. Within the zooplankton community, the copepod *Eurytemora affinis*, typically abundant in brackish water and quasi-absent from freshwater before 2007, has since substantially developed in the latter, where it now represents 90% of the crustacean mesozooplankton community. Simultaneously, cyclopoid copepod abundance has greatly decreased, while cladoceran abundance did not change. The study aim was: 1) to verify if the zooplankton community described for the period 2007–2009 by Mialet et al. (2011) has stabilized until present, and 2) to look for the environmental conditions favouring *E. affinis* development and causing changes in the upstream freshwater zooplankton community. The 2002–2012 temporal evolution of the zooplankton distribution at three stations in the upstream freshwater Scheldt estuary was analyzed. Water quality remained better after 2007 than before, and some factors revealed continuous improvement in annual mean concentrations (e.g. increase in O_2, decrease in BOD5 and NH_4–N concentration). The increase in oxygen and the decrease in NH_4–N concentration, together with low discharge during summer were the main environmental factors explaining the development and timing of *E. affinis* in the upstream freshwater reach. In this reach, *E. affinis* maximal abundance is shifted to higher temperatures (summer) compared to its typical maximum spring abundance peak in the brackish zone of the Scheldt estuary and in most temperate estuaries. The changes in zooplankton community followed a temporal and spatial gradient induced by the spatio-temporal evolution of water quality improvement. The most downstream station (3) allowed *E. affinis* development (oxygen concentration > 4 mg L^{-1}; NH_4–N concentration < 2 mg L^{-1}, discharge (Q) < 50 m^3 s^{-1}) from 2007 onwards, and this station showed the highest *E. affinis* and the lowest cyclopoid abundance. At the more upstream stations *E. affinis* developed later and less strongly, and cyclopoids decreased less in abundance than at station 3. While there may be several explanations for the decrease in cyclopoid abundance (competitive grazing, high predation pressure, NH_3–N toxicity, sensitivity to oxygen, etc.), there is no clear cause for their decline. Water quality improvement in the freshwater Scheldt estuary has led to environmental, post-heavy polluted conditions, under which no data on zooplankton populations in this estuary were available. This has indicated a plasticity in the temperature tolerance of *E. affinis*.

1. Introduction

After decades of ecological degradation, many ecosystems benefit from restoration efforts. Also aquatic ecosystems in general and estuarine ecosystems in particular recovered due to better waste water treatment and habitat restoration. The initial objective of restoration to achieve an original ecological status has often been replaced by a more realistic goal. At present, an ecosystem is considered restored when it is able to sustain itself structurally and functionally and consequently to provide ecosystem services (Borja

* Corresponding author. ECOLAB, Université de Toulouse, CNRS, INPT, UPS, Toulouse, France.
E-mail address: sophie.chambord@univ-tlse3.fr (S. Chambord).

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et al., 2008; Borja and Dauer, 2008; Druschke and Hychka, 2015). Achievement of this goal for severely and long-term polluted systems is a slow and often fluctuating process, which necessitates long-term monitoring to allow adjustment of the restoration and management strategies when necessary (Borja et al., 2010). Indeed, long-term studies allow the detection of latency of biological responses, as it may take time to observe a recovery or a change of communities (Hawkins et al., 2002; Etcheber et al., 2011). However, long-term studies are quite rare because they are expensive and require a lot of material and human effort (Hawkins et al., 2002).

Monitoring of restoration outputs is also a new experience in science, in the sense that, until a decade or two ago degradation of ecosystems was witnessed, not recovery (e.g. Verity, 2002a,b; Kemp et al., 2005; Verity and Borkman, 2010; Langseth et al., 2014). In most cases ecological quantitative monitoring only started after the system was already substantially polluted and consequently little information is available on the pristine or slightly polluted state of system. Hence, it would not be surprising that presently recovering systems face situations that have not been described or quantified yet and reveal some unknown or unexpected ecological relationships. Also, as restoration efforts occur in parallel to ongoing global changes (Anneville et al., 2002, 2009; Verissimo et al., 2013), and estuaries continue to respond to the evolution of multiple user demands, precise effects of environmental factors are often difficult to disentangle.

Many studies on response of estuarine communities focus on large scale aspects, such as hydro-geomorphology, wetland restoration, recovery of top-predator populations (e.g. Orson and Howes, 1992; Ducrotoy and Dauvin, 2008; Beauchard et al., 2011; Maire et al., 2013; Teuchies et al., 2013; Hogg et al., 2014).

Long term monitoring of water quality in estuarine systems are generally focusing on nutrient loads and their management in a eutrophication context. In general, these studies show that system responses to decreased nutrient loading can differ substantially between systems and between subsystems of an estuary, mainly according to hydrography. (e.g. Verity, 2002a,b; Kemp et al., 2005; Boynton et al., 2014; Romero et al., 2016).

However, most papers dealing with estuarine water quality and its effect on pelagic biota report ongoing degeneration of the system rather than restoration, and suggest restoration as a future perspective. For example, Smit et al. (1997) report the evolution in water quality and pelagic community of the Rhine-Meuse Delta after its enclosure in 1970 and give some recommendations (i.e. restoring the estuarine character) for future management. Flaherty et al. (2013) describe the dependence of the nekton community to disturbances of the natural patterns of freshwater delivery to the Florida Bay estuary (USA) by flood-control and water-supply projects and highlight the importance of nekton community monitoring prior to hydrologic manipulations.

Within the pelagic system, phytoplankton and bacteria, as major producers and recyclers, are classically included in biogeochemical studies. Estuarine zooplankton, in spite of being the main trophic link between the primary estuarine resources (i.e. phytoplankton, detritus) and the higher trophic levels (i.e. hyperbenthos, juvenile fishes and some adult fish species) has received little attention. Falcao et al. (2012) report consequences of restoration measures in the Mondego estuary (Portugal) for the zooplankton community. After re-establishment of water circulation between the two branches of this estuary, eutrophication symptoms decreased and higher mesozooplankton density, mainly of estuarine species was observed.

The lack of information on zooplankton response to estuarine restoration is probably due to the minor importance of zooplankton in quantitative energy flow budgets. In addition, contrary to fishes, birds and macrobenthos, the microscopic zooplankton organisms are not readily considered by various stakeholders as a proof of successful management. Also, while phytoplankton can in part be studied by indirect methods (i.e. pigment concentrations, automatic fluorescence monitoring), there are no automated methods used routinely for the evaluation of community composition or activity of zooplankton in estuarine systems. Methods such as Zooscan (Grosjean et al., 2004; Gorsky et al., 2010), applicable in open marine systems or lakes (Schultes and Lopes, 2009; Lelièvre et al., 2012; Marcolin et al., 2013) are not of use in estuarine systems because of the high suspended matter concentration. Analysing estuarine zooplankton samples thus remains a painstaking task, demanding expertise and patience. Yet, having relatively short lifespan, zooplankton organisms can react rapidly to changing environments (Falcao et al., 2012; Cardoso et al., 2013) and can therefore be considered worthwhile monitoring.

This paper presents the results of a long-term (11 years) monitoring of the Scheldt estuary, after restoration of water quality from a heavily polluted status since the 1960–1990s to a less polluted one in the last decades (Heip, 1988; Baeyens et al., 1998; Van Damme et al., 2005; Cox et al., 2009). The Scheldt estuary is a macrotidal estuary covering a marine, brackish and freshwater gradient under tidal influence (Meire et al., 2005; Van Damme et al., 2005) (Fig. 1). The Scheldt has its source in the North of France and runs through Belgium to join the North Sea at Vlissingen in the Netherlands. Its estuary is situated from the mouth at Vlissingen until the city of Ghent, where the tide is stopped by sluices. The tidal reach between the Dutch – Belgian boarder and the city of Ghent, called the Sea Scheldt, covers 110 km of brackish water and 80 km of one of the few remaining freshwater tidal habitats in Europe (Meire et al., 2005; Van Damme et al., 2005). The main tributaries entering the Sea Scheldt are the Dender, Durme and Rupel. In contrast to most temperate estuaries, the Scheldt estuary is characterized by vertically well-mixed water flows (Baeyens et al., 1998), generally showing no salinity or current stratification (Heip, 1988).

The Scheldt estuary has historically been one of the most polluted in Europe (Heip, 1988; Meire et al., 2005). Since two decades, European directives, and specifically the 2000 European Water Framework Directive (WFD), have incited important efforts to increase wastewater treatment capacity on the Scheldt basin and reduce pollutant loads, including organic matter, entering the estuary (Brion et al., 2015). In relation to the upstream reach treated in this paper, the Boven Scheldt watershed wastewater treatment capacity was increased from $2.6 \cdot 10^6$ inhabitant equivalents (IE) in 1986 to $5.0 \cdot 10^6$ IE in 2014. The capacity on the basin of the Dender, which joins the Scheldt just upstream station 3, increased, during the same period, from $31.5 \cdot 10^3$ to $343 \cdot 10^3$ IE (Brion, pers. comm., 2016). As a consequence, oxygen concentrations increased while nutrient concentrations (i.e. NH_4, PO_4) decreased concomitantly. Since 2009, the morphology of the estuary has also changed due to its deepening between the mouth and Antwerp harbour, leading to increased salinity of the estuary and an increase of the tidal pumping.

In response to the improved water quality, the zooplankton community experienced important changes (Appeltans et al., 2003; Tackx et al., 2004; Mialet et al., 2010, 2011). Between 1996 and 2006, the brackish water community was dominated by calanoid copepods, with the calanoid copepod *Eurytemora affinis* being the most abundant species, especially during spring. The freshwater community, (i.e: upstream of Antwerp) was more diverse, dominated by rotifers, cyclopoid copepods and cladocerans. In 2007, a community shift occurred in the freshwater tidal part with *E. affinis* becoming dominant and reaching higher densities than in the brackish part. Concomitantly, cyclopoid copepods decreased to very low abundances. The abundance of cladocerans in the